SPATIO-TEMPORAL PATTERNS OF MACROINVERTEBRATE FUNCTIONAL FEEDING GROUPS IN THE SOSIANI-KIPKAREN RIVER, KENYA

BY

AUGUSTINE SITATI

A THESIS SUBMITTED IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE AWARD OF THE DEGREE OF MASTER OF SCIENCE IN FISHERIES AND AQUATIC SCIENCES (AQUATIC SCIENCE OPTION) IN THE SCHOOL OF NATURAL RESOURCE MANAGEMENT, UNIVERSITY OF ELDORET, KENYA

APRIL, 2021

DECLARATION

DECLARATION BY THE STUDENT

This thesis is my original work and has not been presented for award of any certificate or degree in any other institution or university. No part of this work should be reproduced without prior permission from the candidate and / or University of Eldoret.

AUGUSTINE SITATI

______________________________ ________________________

SNAT/FAS/M/010/17 **DATE**

DECLARATION BY THE SUPERVISORS

This thesis has been presented with our approval as University supervisors.

______________________________ ________________________

Dr. Frank O. Masese **DATE**
Senior Lecturer and HoD
Department of Fisheries and Aquatic Sciences
University of Eldoret, Kenya

______________________________ ________________________

Prof. Phillip O. Raburu **DATE**
Deputy Vice Chancellor
Planning, Research and Extension Division
University of Eldoret, Kenya

DEDICATION

I dedicate this work to my lovely father and mother; Mr. Joseph and Mrs. Gertrude Sitati for always believing in me and encouraging me throughout my life and study career and to my siblings for their continuous support and motivation.

ABSTRACT

Different land-use practices in river basins affect the hydrological characteristics, water quality and alter the complex biotic and abiotic processes that govern the functioning macroinvertebrate communities. Studies utilizing macroinvertebrate functional feeding groups to assess the ecological condition of streams in Kenya are scarce. This study set out to investigate changes in total abundance, taxon richness and biomass of macroinvertebrate functional feeding groups (FFGs) in response to land-use change and assess the suitability of abundance- versus biomass-based metrics as surrogates of ecosystems attributes and ecological integrity in the Sosiani-Kipkaren River in western Kenya. A total of 21 sites were sampled during the wet season (July-August 2018) and 14 of the same sites during the dry season (February-March 2019) along a land-use gradient. Four land-use categories; Forest (n = 5), Mixed (n = 6), Agriculture (n = 6) and Urban (n = 4) were sampled. Macroinvertebrates were collected seasonally, identified, assigned to functional feeding groups and used to derive the five metrics utilized as surrogates of ecosystem attributes. Water and habitat quality variables were also measured seasonally and their data used to correlate with the macroinvertebrates assemblages. There were significant ($p < 0.05$) spatial variation in habitat quality, organic matter standing stocks, electrical conductivity, temperature, sodium, potassium and nutrient concentrations across land-uses, with forested sites recording lowest values. Forest land-use sites had good habitat quality (QHEI Score of 56) while the rest were marginal. Macroinvertebrates total abundance was significantly higher (R-statistic = 0.30, $p < 0.007$) during the wet season (35,827 individuals) than dry season (7,652 individuals). Responses in macroinvertebrates differed among functional feeding groups, with biomass-based metrics responding more strongly to land-use change while richness-based metrics being the least predictive, indicating replacement of taxa within functional feeding groups across land-uses. Higher shredder abundance, biomass and richness were recorded in forested streams and lowest in urban streams during both seasons. Collector-gatherers dominated agricultural streams during both the wet and dry seasons, while predators dominated urban streams. Scrapers responded positively to increased nutrient levels and open canopy in mixed and agricultural streams. Abundance-based metrics were better predictors of ecosystem attributes, and displayed greater response to changes in stream size than biomass-based metrics. There was incongruence between abundance- and biomass-based indicators for Production/Respiration and coarse particulate organic matter to the fine one. Catchment land-use did not influence metric performance, suggesting that reach scale influences played a predominant role in structuring communities and determining ecosystem functioning. Even though there is need for more studies to refine the metrics used and establish thresholds for the various attributes, this study established that there is indeed potency of functional feeding groups approach to be used as a means of assessing both the ecological integrity and functioning of Afrotropical streams.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF PLATES

LIST OF APPENDICES

LIST OF ABBREVIATIONS AND ACRONYMS

ANOVA - Analysis of variance

BFPOM -Benthic fine particulate organic matter

CPOM - Coarse particulate organic matter

CA- Correspondence analysis

CCA - Canonical correspondence analysis

DO - Dissolved oxygen

EC - Electrical conductivity

EMCA – Environmental Management and Coordination Act

EPT - Ephemeroptera, Plecoptera and Trichoptera

FAO - Food and Agriculture Organization

FFGs - Functional Feeding Groups

FPOM - Fine particulate organic matter

GPS - Global positioning system

LVB - Lake Victoria Basin

NMDS - Non-Metric multidimensional scaling

NRB -Nzoia River Basin

PCA - Principle component analysis

POM - Particulate organic matter

PPT - Parts per thousand

RCC - River continuum concept

RDA - Redundancy analysis

SD - Standard deviation

SDGs - Sustainable development goals

SE - Standard error

SRP - Soluble reactive phosphates

TDS - Total dissolved solids

TFPOM - Transport fine particulate organic matter

TSS - Total suspended solids

UoE - University of Eldoret

YSI - Yellow Spring Instruments

ACKNOWLEDGEMENT

I am pleased that the journey through my master programme that started in September 2017 has finally come to an end. This journey has been successful by support of many individuals to whom I am so grateful. First, I would like to thank the Almighty God for the guidance, good health and care He granted me throughout this journey. Secondly, I would like to express my sincere gratitude and selfless appreciation to my supervisors Dr. Frank Masese and Prof. Phillip Raburu, for their guidance and close involvement in the whole process of designing and implementation of my research project and writing of my thesis and manuscripts. Their selfless nature and time they took off their busy schedules to guide me is commendable, it was an honour working with these great minds.

I also appreciate the assistance of Mr. Alfred A. Otieno, Dr. Simon W. Agembe and Dr. James E. Barasa (University of Eldoret) who also helped with designing the study and execution of fieldwork and Mr. Saeed Hassan (Egerton University) for a map of the study area. I am also grateful to my field and laboratory assistants who stood with me during the rainy and extremely sunny sampling days and in the laboratory processing of samples; Ms. Yegon Mourine, Mr. Lubanga Henry, Ms. Wanderi Elizabeth (fisheries department), Dr. Wanjohi Benard (Wildlife department) and Mr. Odero, Mr. Sang, Mrs. Margret and Mr. Kirwa (Chemistry department). Our work together will forever be impinged on my mind. I also want to extend my gratitude to the entire Kenya Invertebrate Scoring System (KISS) team and the National Research Fund (NRF) Kenya, as this work was supported by a multidisciplinary project (KISS) funded by NRF-Kenya and coordinated by the University of Eldoret.

Last but not least, I would like to extend my sincere gratitude to my parents; Mr. and Mrs. Sitati and to my siblings for always being emotionally and mentally supportive, praying for me and encouraging me throughout this journey. Finally, I would like to thank all my fellow colleagues (class of 2017-2020) for all the support and great memories we shared together.

CHAPTER ONE

INTRODUCTION

1.1 Background information

Despite the fact that tropical rivers and riparian zones are among the most vulnerable habitat types on the planet and constitute >50% of Earth's runoff (Schlunz & Schneider, 2000), little is known about how these ecosystems function (Dudgeon *et al*., 2006). Agricultural expansion, urbanization and industrialization, overgrazing, mining, water abstraction, hydro-morphological alterations, and untreated wastewater discharge are among the common factors that have a negative impact on habitat quality, food availability, and the ecological integrity and function of streams and rivers (Dudgeon *et al*., 2006; Ramirez *et al*., 2008; Carpenter *et al*., 2011). In order to more fully understand the influence of human activities on riverine ecosystem structure and functioning globally, and safeguard the myriad ecosystem services deriving from streams and rivers, there is a need to develop and validate regional decision-support tools for bioindication (Masese *et al*., 2017; Ko *et al*., 2020).

Aquatic macroinvertebrate communities exhibit spatial and temporal dynamics in response to changes in abiotic and biotic conditions due to their diverse behavioral and structural adaptations for food acquisition and habitats across different taxa (Eady *et al.*, 2014). The structural and functional composition of aquatic macroinvertebrates have been used as bioindicators of ecological integrity and ecosystem functioning (Palmer *et al*., 2005; O'Brien *et al*., 2016). However, assessment of ecological condition of streams and rivers has mainly relied on the structural composition (Stone *et al*., 2005; Jiang *et al*., 2010), whereas fewer studies have addressed the relationships between functional diversity and environmental variables (Cummins *et al*., 2005; Savić *et al*., 2018).

Benthic macroinvertebrates are organisms which lack a backbone, are visible to the naked eye and are captured in the mesh sizes of 200-1000 micrometres (Rajele, 2004). They exist in different habitats ranging from fast flowing mountainous streams to slow-flowing muddy waters (Dallas, 2007). These organisms utilize rocks and stones, logs, vegetation and soft sediments in aquatic systems as their habitats (Barbour *et al*., 1999). They are valuable indicators of the physical wellbeing of aquatic bodies because they show a broad range of pollution and disturbance sensitivities (Kazanci & Dugel, 2010; Freitas *et al*., 2012). Natural and human-caused changes in stream and river characteristics such as width, depth, substrate type, water velocity or discharge, and physico-chemical variables such as dissolved oxygen, water temperature, conductivity, salinity, and pH easily alter macroinvertebrates composition and distribution (Baptista *et al*., 2007; Arimoro, 2009; Arimoro *et al*., 2011; Masese *et al*., 2014a).

Other than the macroinvertebrates ability to indicate both long term and short term changes in the system, they are often used for biomonitoring because of their short life spans and ease of capture (Rosenberg *et al*., 2008). Macroinvertebrate communities also play an important role in the food webs of riverine ecosystems, as a source of food for higher trophic levels such as fish, and are fundamental to stream health in that, when classified into functional feeding groups (FFGs) they play vital roles in biofilm assimilation, breakdown of leaf litter and other associated materials and also as predators and prey (Cummins *et al.,* 2005; Taylor, 2005; Rosenberg *et al*., 2008). In general, they have a negative response to physical channel and riparian alterations and disturbances (Resh & Rosenberg, 1993; Taylor, 2005; Rosenberg *et al*., 2008).

Macroinvertebrate FFGs is a classification system based on behavioral processes of food acquisition rather than a taxonomic group (Cummins & Klug, 1979; Merritt & Cummins, 1996). Instead of studying hundreds of different taxa, use of FFGs allows a small number of individuals from the groups to be studied together based on how they function and absorb energy in the stream habitat (Merritt & Cummins, 1996). The major FFGs are; Shredders, that utilize on leaf litter or other coarse particulate organic matter (CPOM); the grazers, commonly known as scrapers, that eat algae and associated material; collector-gatherers (henceforth gatherers), which feed on fine particulate organic matter (FPOM) from the stream bottom; collector-filterers (henceforth filterers), that consume fine particulate organic matter from the water column using various filters; and predators, that consume the other invertebrates (scrapers, collectors and shredders) (Cummins & Klug, 1979).

In streams and rivers, the functional composition of macroinvertebrate assemblages is strongly influenced by both catchment-scale and riparian or reach-scale land-use and other factors that influence habitat conditions, food quality and quantity (Allan, 2004; Jiang *et al*., 2011; Masese *et al*., 2014a). Changes in land-use from natural forests to agriculture or settlement are a major source of worry around the world because they are linked to soil erosion, sedimentation, nutrient enrichment and hazardous chemical input into aquatic ecosystems and changes in biological communities (Dudgeon *et al*., 2006; Reid *et al*., 2019). As opposed to previous decades when land transformation was mainly for industrial and infrastructural development, land-use change is increasingly being driven by cropland farming and livestock grazing, with about 6 million km^2 of forests and grasslands being converted yearly (FAO, 2013). Agricultural activities can degrade streams by causing nutrient enrichment through fertilizer or manure use (McDowell & Sharpley, 2001), increase sediment input from farmlands (Burdon *et al*., 2013), increase mean water temperature (Benstead & Pringle, 2004), alter hydrologic regimes (O'Brien *et al*., 2018), and increase pesticide inputs (Osano *et al*., 2003) into streams. Similarly, urban development and its associated impervious surfaces can degrade streams through the release of excess nutrients, sediments and toxicants (Beasley & Kneale 2002; Walsh *et al*., 2005), cause shifts in temperature regimes and significantly alter basal resources for food webs (Walsh *et al*., 2005; Imberger *et al*., 2011).

Seasonality, which is defined in the tropics by rainfall availability and amount, also influences habitat conditions, water quality, energetic connectivity between terrestrial and aquatic environments, and the abundance and diversity of food resources, all of which influence the structural and functional composition of macroinvertebrate communities (Bunn & Arthington, 2002; Leigh, 2013). Studies have indicated that abundance of macroinvertebrate FFGs increase during the wet/rainy season (Camacho *et al*., 2009, Masese *et al*., 2014a), while others have shown more abundance and richness during the low flows (Jiang *et al*., 2010, Makaka *et al*., 2018). During the rainy season, increased turbidity is a limiting factor for riverine primary productivity and as a result, aquatic consumers ability to absorb algal resources is greatly diminished (Junk *et al*., 1989). During the dry season, however, algal quantity and contribution to aquatic food webs are higher, but terrestrial organic matter is more essential during the wet season (Zeug & Winemiller, 2008; Roach & Winemiller, 2015).

Although earlier studies captured in the River Continuum Concept (RCC) (Vannote *et al*., 1980) recognized the utility of macroinvertebrate FFGs as proxies for ecosystem attributes and functioning, the past three decades have seen an upsurge in the use of species or community functional metrics or traits as surrogates of ecosystem attributes and functioning in relation to disturbance gradients (Petchey *et al*., 2004; Verberk *et al*., 2013; Merritt *et al*., 2017), such as changes in land-use and loss of riparian vegetation (Verberk *et al*., 2013; Dolédec *et al*., 2011; Fierro *et al*., 2017), increased amounts of fine sediments (Wagenhoff *et al*., 2012; Mathers *et al*., 2017), and as indicators of ecosystem condition (Merritt *et al*., 2017). Thus, without having to quantify these attributes independently, the functional composition of macroinvertebrates can be utilized as a measure of trophic dynamics and ecological status of aquatic ecosystems (Cummins *et al*., 2005; Makaka *et al*., 2018; Abdul & Rawi, 2019). Macroinvertebrate FFGs are significant both environmentally and economically since they reduce the cost of measuring ecosystem attributes, which require are usually expensive in terms of time, personnel and equipment while providing a limited and short-term perspective of ecosystem functioning.

Studies that have utilized ratios of FFGs as surrogates of ecosystems attributes in tropical streams (e.g., Cummins *et al*., 2005) have relied on metrics developed for temperate streams (Merritt *et al*., 2017). However, because macroinvertebrate FFGs have intra- and inter-regional differences in composition (Boyero *et al.,* 2011), these metrics must be tested and validated before being used as indicators of ecosystem attributes in zones other than the temperate zone where they were developed. For instance, several studies in the tropics have reported a limited number of shredder taxa in comparison to temperate streams (Yule, 1996; Tumwesigye *et al.,* 2000; Dobson *et* *al.,* 2002; Makaka *et al.,* 2018), even though a number of other studies have documented a diverse shredder guild in headwater streams (Cheshire *et al.,* 2005; Yule *et al.,* 2009; Masese *et al.,* 2014a).

Furthermore, due to the abundance of macroconsumers in tropical streams, such as freshwater crabs and shrimps, which also play important roles in organic matter processing in detrital food webs (Wantzen *et al.,* 2008; Boyero *et al.,* 2020), there is likely to be a disparity in metrics of ecosystem functioning based on macroinvertebrate FFGs abundance and biomass between temperate and tropical streams. This study therefore did set out to investigate changes in taxon richness, numerical abundance and biomass of macroinvertebrate FFGs in response to land-use change and assess the suitability of abundance- vs. biomass- based metrics as surrogates of ecosystems attributes and ecological integrity in the Sosiani-Kipkaren River in western Kenya.

1.2 Statement of the problem and justification of the study

Changes in land-use, primarily at the catchment- and reach-scales, have a negative impact on the growth rates, abundance, diversity, and the trophic structure of macroinvertebrate communities (Raburu *et al.,* 2009; Aura *et al.,* 2011; Masese *et al.,* 2014a; Oruta *et al.,* 2017). The expansion of agricultural and urban areas, overgrazing, mining, abstraction of water for domestic use, deforestation, hydro-modification, storm sewer and discharge of untreated water as is the case along the Nzoia River Basin (NRB) headwater streams are some impacts that affect the riparian and water quality and integrity of biotic stream communities (Sangale *et al.,* 2005; GEF, 2004; Njiru *et al.,* 2008; Masese *et al.,* 2009).

Assessment of water quality and overall degradation has relied for long on the measurement of physico-chemical parameters, which is an expensive method and also lacks the integrative capacity to inform about the effects of pollutants on biodiversity and the ecological integrity of aquatic resources. There is therefore a need to adopt an effective and economical (both in terms of money and time) monitoring tool. Earlier studies by Benstead & Pringle, (2004), Dudgeon, (2010), Boyero *et al*. (2011) and Ferreira *et al*. (2012) indicated that the knowledge of macroinvertebrate FFGs in tropical streams and rivers is important as it aids in the understanding of the functioning of these streams and provides insights on the type of management that is required to reduce and/or prevent degradation of these streams. Thus, macroinvertebrate FFGs can and have been used for biomonitoring.

However, there is paucity of information on FFGs classification and application in tropical, as opposed to temperate freshwater systems (Masese *et al.,* 2014a). This makes it hard to properly implement this approach in many tropical streams. These limited studies have utilized temperate models and have not tested the temperate ratios on tropical rivers and streams on wider scales and have not established whether they work and respond on temporal and spatial dimensions of Afrotropical Rivers**.** Information on the factors that influence macroinvertebrate community structure is important not only for basic ecological understanding and biodiversity conservation, but also as a model for monitoring, repairing, and sustaining stream ecosystem health (Masese *et al.,* 2014a, b).

Given that the Sosiani-Kipkaren River has undergone gradual physical modifications over time, and the resultant habitat and water quality changes may be playing a role in influencing its biotic attributes, this study is important to help provide information on

FFGs of the Sosiani-Kipkaren on spatial and temporal scales, assess land-use impacts on the functional organization of macroinvertebrate communities and employ the abundance vs. biomass based ratios as surrogates of ecosystem attributes. The study is also important as it is aimed at providing information that will be utilized by the Kenyan biomonitoring protocol and aid in bridging the gap in FFGs knowledge around East Africa and their potential use in assessing ecological conditions in streams and rivers; in line with the attainment of Vision 2030 and Sustainable Development Goals (SDGs) on conservation of water resources and protection of the environment.

1.3 Study objectives

1.3.1 Overall objective

To assess the spatial and temporal changes in macroinvertebrate functional feeding groups along the Sosiani-Kipkaren River and their use as indicators of ecosystem attributes.

1.3.2 Specific objectives

The specific objectives of this study were to:

i. Determine spatial and temporal variation in water quality physico-chemical variables and habitat quality in the Sosiani-Kipkaren River.

ii. Determine spatial and temporal patterns of macroinvertebrates diversity, composition and biomass in the Sosiani-Kipkaren River.

iii. Evaluate the influence of water and habitat quality on macroinvertebrates diversity, biomass and composition in the Sosiani-Kipkaren River.

iv. Investigate whether abundance- vs. biomass-based ratios of macroinvertebrate FFGs can be utilized as proxies of ecosystem attributes in Sosiani-Kipkaren River.

1.4 Hypotheses

H_{A1}: Physico-chemical water quality variables display higher levels in agricultural and urban stream categories than in forested streams.

H_{A2}: The composition of macroinvertebrate FFGs would vary as a result of changes in land-use and seasonal variations in flow, water quality and habitat characteristics.

H_{A3}: The ratios of macroinvertebrate FFGs can be utilized as proxies of ecosystem attributes in Sosiani-Kipkaren River.

H_{A4}: There are differences between abundance- vs. biomass-based ratios of FFGs as surrogates of ecosystem attributes in the Sosiani-Kipkaren River because of the presence of macroconsumers.

CHAPTER TWO

LITERATURE REVIEW

2.1 Variation in water and habitat quality in streams and rivers

While fresh water is about 0.01% of the water on Earth, and covers less than 1% of the earth's surface, it is a habitat to a substantial 40% of recognized fish species and 25% of the known vertebrates (Dudgeon *et al*., 2006). Some of the world's most endangered species are included in the biota connected to these habitats (Amezaga *et al*., 2002). Freshwater habitats have been and remain changed at a higher rate than any other ecosystem by anthropogenic land-use change. Regardless of the fact that freshwater habitats are very critical, human actions have adversely affected at least 80 percent of streams (Dudgeon *et al*., 2006).

Water quality refers to particular water's physical, chemical and biological characteristics for intended use (Bouwer, 2000). Water quality is affected by changes in physical and chemical parameters which include: water temperature, dissolved oxygen (DO), biological oxygen demand (BOD), salinity, turbidity, phosphates, nitrites, chemical oxygen demand (COD), nitrates, electrical conductivity, pH and heavy metals (Wilhm & Dorris, 1968). The Environmental Management and Coordination Act (EMCA) of the Environmental Act of 1999 in Kenya specifies the maximum values of water quality criteria that can be used for domestic use as; Nitrates 10 mg/L, Ammonia 0.5 mg/L, Nitrites 3 mg/L, Total dissolved solids (TDS) 1200 mg/L, Total suspended solids (TSS) 30 mg/L, Phosphates 30 mg/L and Aluminium 5 mg/L (Kenya Gazette, 2006; NEMA, 2006; KEBS, 2015).

As is the case in Kenya, most parts of Africa and other developing countries, people have continued to live near streams and rivers in order to obtain water for their everyday needs, leading to contamination of these water bodies (Minaya *et al*., 2013; Masese *et al*., 2009, 2014a), so as is the case along the Sosiani-Kipkaren River. The discharge of organic and inorganic materials or contaminants into a water body frequently affects water quality, causing changes in the biotic community (Arimoro *et al*., 2007; Raburu *et al*., 2009; Aura *et al*., 2011; Arimoro *et al*., 2011; Masese *et al*., 2014a).

Habitat quality is important in that clearing of forests and riparian vegetation within stream catchments has led to changes in; resource base, flow and channel characteristics, sediment regime and producing homogeneity in habitats and instream characteristics (Allan, 2004). As a result of the increased surface runoff and river sediment loads, ecological changes such as smothering of littoral habitats, clogging of river bed, and floodplain aggravation have occurred (Dudgeon *et al*., 2006). Riparian areas are important for aquatic environment maintenance and regulation (Naiman *et al*., 2005). Availability of riparian vegetation acts as sediment barrier in streams, assisting in the maintenance of water quality and the provision of allochthonous resources (leaves from forest cover) which is essential for the maintenance of energy flow and stream balance (Mathooko & Kariuki, 2000; Dobson *et al*., 2002).

Because they may be found in all but the most severely contaminated or disturbed habitats and have a wide range of contamination tolerances across various taxa, benthic macroinvertebrates are effective indicators of water quality (Allan, 2004). Even though natural alterations are an important aspect of freshwater ecosystems, in the recent past, there is growing scientific evidence of their increasing severe effects on stream macroinvertebrate communities (Hawkins *et al*., 2015). According to studies on headwater streams, biological communities in most habitats can be described as creating a temporal sequence of synchronized species replacement (Hepp *et al*., 2016).

River ecosystem stability can be viewed as a tendency to reduce fluctuations in the flow of energy while keeping community structure and function in the face of environmental changes. This implicitly combines stability of the community (Masese *et al*., 2014a) with the physical system's instability. In ecosystems with a highly stable physical structure, biotic diversity may be low, but total flux ecosystem stability may still be retained, while on the other hand, systems with a lot of physical variability may have a lot of species variety or at least a lot of species complexity, which helps to maintain stability (Vannote *et al*., 1980).

The quality of water resources is intensely deteriorating on a daily basis in many places and this is one of the biggest problems people are facing along the Nzoia River headwaters mainly caused by deforestation and utilization of catchment and riparian areas for agricultural (Sugar cane, maize, livestock and vegetables) activities and urbanization (Eldoret, Turbo and Kipkaren) (GEF, 2004; Masese *et al*., 2009; Aura *et al*., 2011). The Nzoia River Basin has been reported to have high amounts of phosphates, nitrates, and prohibited substances such as aldrin, dieldrin, and DDT in its water, which has been linked primarily to agricultural activities (Osano *et al*., 2003; Twesigye *et al*., 2011). Moreover, nutrients from surface runoff, such as nitrates, phosphates, and total organic carbon, are a primary cause of eutrophication in Lake Victoria, resulting in large algal blooms, water hyacinth infestation, and oxygen depletion in the water (Okungu & Opango, 2005). Therefore, this study was set to look at physical, chemical and biological parameters dependent on the water and habitat quality and how the variability in water and habitat quality in turn influences macroinvertebrates structural and functional composition, diversity and biomass along the Sosiani-Kipkaren River.

2.2 Macroinvertebrates structural and functional composition

A river's ecological integrity is defined as its ability to support and maintain a balanced, integrated, and adaptive composition of physico-chemical characteristics with a biological community on a spatio-temporal scale comparable to those of a natural aquatic ecosystem in the region (Taylor *et al*., 2005)**.** The fundamental benefit of using a biological approach (such as macroinvertebrates) is that it examines organisms that are constantly exposed to pollution. As a result, species found in riverine ecosystems reflect both the current and previous history of the system's water quality, enabling for the detection of perturbations that would otherwise go unnoticed (Taylor *et al*., 2005).

Macroinvertebrates are mainly used in biomonitoring because of their advantageous characteristics which include; their diversity (taxonomically and trophically), abundance, low mobility (good indicators of localized conditions), suitable lifespans, ability to show responses to many pollution types and pollution levels, ability to provide information about cumulative and synergistic pollution effects, ease and low-cost of collection, ease of identification to family-level, importance in food webs, and the considerable amount of background information that is available (Resh *et al*., 1995, Pavluk *et al*., 2000, Wenger *et al*., 2009). In terms of their functional importance, benthic macroinvertebrates aid in the breakdown of CPOM, FPOM, microbes, diatoms, macrophytes, and other invertebrates, as well as constitute a major food source for other invertebrates, fishes, and waterfowl (Vannote *et al*., 1980, Pavluk *et al*., 2000).

Functional classification of macroinvertebrates (FFGs) is a classification centered on morpho-behavioral mechanisms employed by macroinvertebrates to obtain food stuff (Cummins & Klug, 1979). Macroinvertebrate FFGs classification offers the advantage of combining both morphological characteristics such as mouth part specialization and behavioral mechanisms such as food acquisition when consuming resources (Cummins *et al.,* 2005). This method of classifying organisms enriches the understanding of trophic dynamics in aquatic systems by simplifying the benthic community into trophic guilds (Merritt & Cummins, 1996, 2006). When assigning macroinvertebrates to functional feeding groups, knowledge of macroinvertebrates numerical abundance and biomass along the rivers and streams is important for determining the effects of land-use change and riparian alterations (Baptista *et al*., 2007; Masese *et al*., 2014a). Masese *et al*. (2014a) noted that the duty to allocate macroinvertebrates to FFGs is not clear and is generally problematic in some cases, especially when the assignment isn't backed up by information on feeding patterns and mouthparts morphology.

Assignment of macroinvertebrates into FFGs is mainly through gut content analysis and mouth part morphology, though, with a few exceptions, most studies around the tropics assign FFGs using available literature (Merritt & Cummins, 1996; Graca *et al*., 2001; Dobson *et al*., 2002; Polegatto & Froehlich, 2003; Molina, 2004; Cheshire *et al*., 2005, Masese *et al*., 2014a). There is little or no information of macroinvertebrate FFGs in many tropical streams (Boyero *et al*., 2009, Masese *et al*., 2014a). Other than relying on the already existing literature, most studies and research in these tropical streams and rivers also do use temperate models while assigning FFGs bringing a major question of whether ecological models created for temperate streams are applicable to their tropical counterparts (Masese *et al*., 2014a). This therefore necessitates the need for proper assignment of macroinvertebrates into FFGs.

For some taxa, this method has worked, but not all, because some closely related species in different regions have distinct diets. Furthermore, some species exhibit food ontogenic shifts while some have a variety of diets to be ascribed to a single FFG (Cheshire *et al*., 2005, Masese *et al*., 2014a). However, several efforts to advance the knowledge of the feeding ecology of tropical macroinvertebrates have been made; including: (Dobson *et al*., 2002, Tomanova *et al*., 2006, Boyero *et al*., 2009; Chará-Serna *et al*., 2010, Masese *et al*., 2014a). These studies used gut content analysis in conjunction with mouth part morphology to look at diet composition, trophic stages, and FFGs.

Macroinvertebrate community abundance, biomass and diversity can be used to assess ecological changes and impacts that might occur due to the change in land-use. Several studies have found a link between land-use and macroinvertebrate assemblages, with the total percentage of taxa and proportions of groups such as Ephemeroptera, Plecoptera, and Trichoptera (EPT) decreasing while Oligochaeta and Diptera increasing as pollution and variations in river quality increase (Raburu, 2003; Dudgeon, 2006; Raburu *et al*., 2009; Masese *et al*., 2009; Masese *et al*., 2014a). Therefore this study was set to investigate structural and functional composition, diversity and distribution of macroinvertebrates across different land-use sites from the less disturbed upper reaches of the river at Kaptagat forest to the more impacted mid (towns) and lower reaches of agricultural (large scale maize and sugar cane plantation) land-uses so as to investigate how the community structure changes with change in human activities along the riparian and catchment scales as potential candidates for bioindication.

2.3 Water and habitat quality on macroinvertebrates diversity, biomass and functional composition

Macroinvertebrates are the most common bioindicators in aquatic ecosystems because of their valuable characteristics such as; broad diversity of species with varying levels of tolerance to disturbance, high abundance, low mobility (good indicators of localized conditions), long lifespans which offers the ability to provide information about cumulative and synergistic effects of pollution, ease of collection and identification to family level, and the considerable amount of background information that is available (Barbour *et al*., 1999; Resh, 2008). To ease their use in bioindication, macroinvertebrates have been divided into functional feeding groups (FFGs) based on behavioural and physiological mechanisms for food acquisition and habitat use (Cummins & Klug, 1979; Merritt & Cummins, 1996).

Because of the varied environmental and food requirements, macroinvertebrate FFGs have been used as both indicators of ecological status and ecosystem functioning (Vannote *et al*., 1980; Merritt & Cummins, 1996, 2006). However, the use of FFGs for biomonitoring aquatic ecosystems in the tropics has received a number of challenges, notably lack of keys and schema for classifications (Masese *et al*., 2014a; Buss *et al*., 2015). As a result, most studies have used temperate keys and guides to identify and classify macroinvertebrates into FFGs, sometimes with misleading outcomes (Camacho *et al*., 2009).

Despite the challenges of using macroinvertebrates as indicators of ecological condition and ecosystem functioning in tropical streams, many studies have used structural responses of communities to develop indices for bioindication (e.g., Dickens & Graham, 2002; Aschalew & Moog, 2015; Kaaya *et al.*, 2015). Some of the factors that influence macroinvertebrate community composition in streams and rivers include seasonality in flow conditions and land-use change (which inturn affects the water and habitat quality) or human-induced disturbances (Collier & Quinn, 2003; Cooper *et al.*, 2013). The diversity, abundance and biomass of FFGs reflect a combination of seasonally varying factors including water quantity (flow velocity and discharge) and quality, habitat or biotope suitability and availability for different taxa, and the relative abundance of autochthonous and allochthonous food resources (Junker & Cross, 2014; Entrekin *et al.*, 2020).

Seasonal variation in taxon richness, abundance and biomass largely rely on biological traits of the organisms (Beche *et al.*, 2006). Collector-gatherers (generalist feeders) tend to be abundant both during the wet and dry seasons (Masese *et al.*, 2014a), whereas more specialised feeders such as shredders are dependent upon seasonal availability of coarse particulate organic matter (litter fall), and thus may exhibit strong seasonality in abundance (Bogan & Lytle, 2007). Similarly, collector-filterers distribution can vary with season (Bogan *et al.*, 2013), as they require specific flow conditions to acquire food from the water column, both of which can vary seasonally. Because of its impact on basal food resources through disruption of allochthonous resource subsidies, decreased stream shading and increased sedimentation and water temperature, changes in water and habitat quality have a significant impact on macroinvertebrate FFGs diversity (Lorion & Kennedy, 2009; García *et al.*, 2017; Masese *et al.*, 2018; Mwaijengo *et al.*, 2020).

Studies have shown that some shredder taxa are restricted to cooler and shaded forested streams, while FFGs that are tolerant to poorer water quality and habitat degradation (Yule *et al*., 2009; Masese *et al*., 2014a), such as collectors, can be more widespread (Masese *et al*., 2009a; Buss *et al*., 2015). Comparatively, forested streams have a higher taxon richness of macroinvertebrates compared to adjacent streams under other uses, such as agriculture or grazing (Minaya *et al*., 2013; García *et al*., 2017; Fugère *et al*., 2018). Watershed urbanization also negatively affects macroinvertebrate biomass and community structure, through changes in water quality and basal food resources (Lawrence & Gresens, 2004; Sterling *et al*., 2016; Alberts *et al*., 2018).

Understanding the effects of water and habitat quality on streams and rivers is an overarching objective for the management and conservation of riverine ecosystems. Because freshwater ecosystems are important not only to conservation, but to economics and culture as well, the effects of their alteration on their structure and functioning are of great interest (Dudgeon *et al*., 2006; Vörösmarty *et al*., 2010). In concert with the increasing rate of land-use and land cover changes and human population growth on tropical catchments (López-Carr & Burgdorfer, 2013), there is a need for a concomitant increase in knowledge on the impacts of these developments on biodiversity, water resources and ecosystem functioning (Dudgeon *et al*., 2006; Ramírez *et al*., 2008).

Data on macroinvertebrate functional organization in Afrotropical streams and rivers will aid in deeper understanding of organic-matter processing, trophic interactions, and management actions required to prevent ecosystem degradation (Dudgeon *et al*., 2010; Boyero *et al*., 2011; Ferreira *et al*., 2012; Fugère *et al*., 2018). This study therefore set out to investigate how land-use change from forestry to rural settlements, urbanization and farmlands influence water and habitat quality in streams and rivers and the ensuing responses in the functional organization of macroinvertebrate assemblages.

2.4 Spatio-temporal trends in macroinvertebrate FFGs as surrogates of ecosystem attributes

Food availability has a significant impact on the distribution of FFGs along streams and rivers, as well as seasonal variations in the structure of macroinvertebrates in habitats (Allan, 1995; Towsend *et al*., 1997). Both living and detrital food bases are continually processed in natural stream systems, but there is usually a seasonal shift in the relative importance of autotrophic production vs. detritus loading and processing. Trophic measures are proxies for complicated processes including trophic interaction, production, and the availability of food sources (Merritt *et al*., 2002). Numerous studies, like Vannote *et al*. (1980), have indicated that the patterns of FFG distribution in aquatic systems are connected to the environmental gradient, and this is being utilized in several water quality systems (Pavluk *et al*., 2000).

The nature of macroinvertebrate distribution provides information on the types of ecological processes that regulate assemblage structure (Tumwesigye *et al*., 2000; Raburu *et al*., 2009). Riparian vegetation, which inhibits autotrophic production via shading of the stream and generates substantial volumes of allochthonous detritus, has a significant impact on the functioning of many headwater streams. As stream size increases, terrestrial organic supply becomes less important, while autochthonous primary production and organic transport from upstream become more important

(Merritt & Cummins 2006). This shift from headwaters (which rely on terrestrial inputs) to medium-sized rivers (which rely on algal or rooted vascular plant production) is hypothesized to be indicated in a shift in the gross primary productivity to community respiration ratio and is usually dictated by the degree of shading (Minshall, 1978).

The river continuum concept (RCC), Vannote *et al*. (1980), hypothesized that food resources would shift with stream size, and that the FFGs would reflect changes in the types and locations of food resources as stream size increased (Plate 1). This concept was developed in the context of undisturbed, natural stream ecosystems in the temperate region and its application to tropical systems has shown mixed results, some negative (Yule, 1996; Tumwesigye *et al*., 2000), and others more positive (Dudgeon, 1984; Marchant *et al*., 1985; Greathouse & Pringle, 2006; Tomanova *et al*., 2006; Jiang *et al*., 2011).

River morphology, hydrologic aspects and the importance of riparian vegetation change along the upstream-downstream gradient usually shape the biological communities. The upstream reaches of the river are dominated by the shredders which break large leaves from coarse particulate organic matter (CPOM) to fine particulate organic matter (FPOM). In the middle river reach the functional feeding group is predominated by the grazers/ scrapers which tend to feed on the available algal material and diatom and the functional composition changes from shredders to collectors as the energy input changes downstream. The collectors are further categorized into collector-filters and collector-gatherers. The downstream river reach is dominated by the collectors and predators (Plate 1).

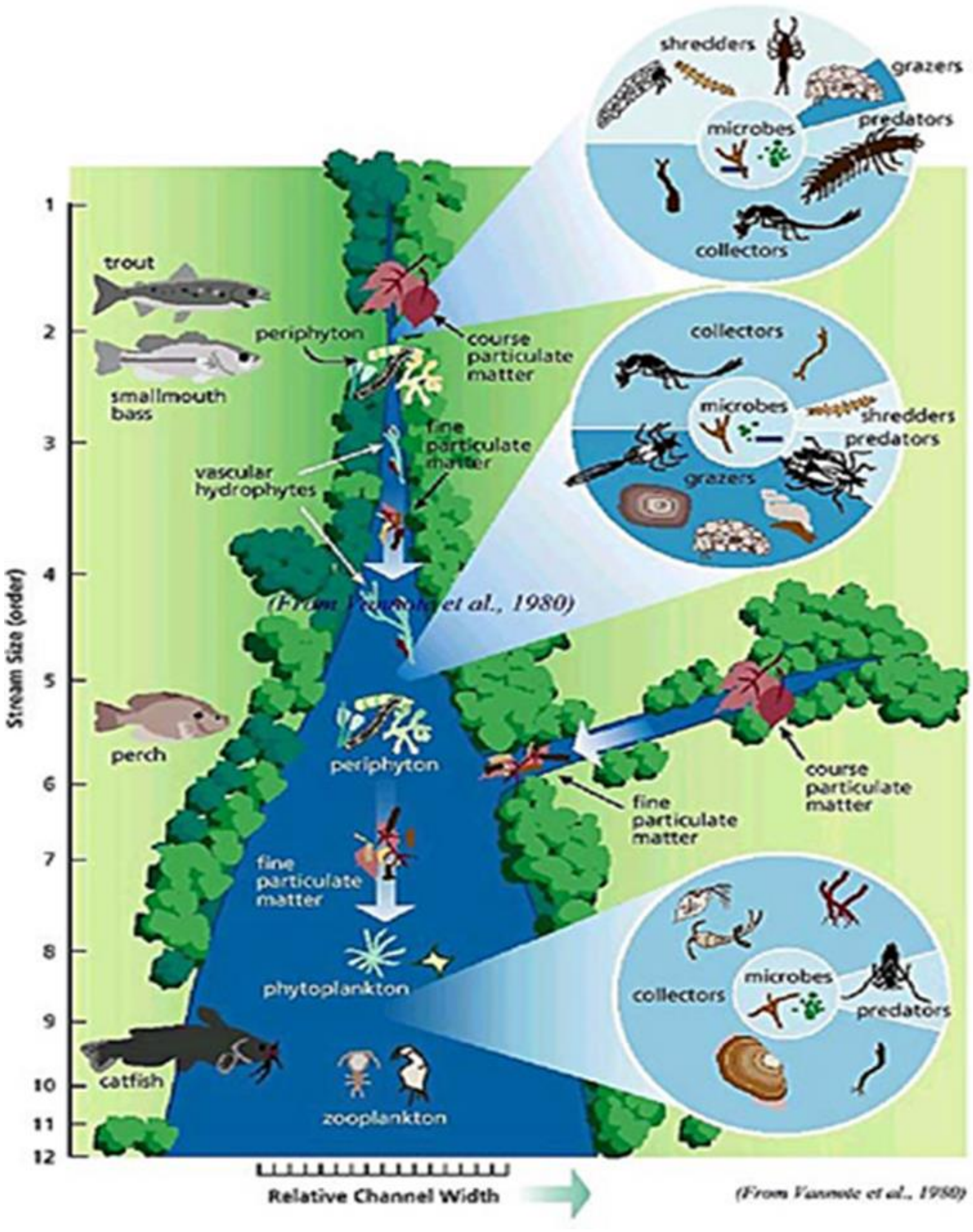


Plate 1: The river continuum concept (RCC) by Vannote *et al.*, 1980

Proportions of the various macroinvertebrate FFGs can be utilized as indicators of stream ecosystem attributes (Vannote *et al*., 1980; Merritt *et al*., 2002). This makes FFGs ecologically and economically important as they save on measurements of the ecosystem characteristics that need huge input in terms of time and equipment and provide only a small view of ecosystem functioning (Merritt & Cummins, 1996, 2006; Cummins *et al.*, 2005; Table 1).

Table 1: Ratios of macroinvertebrate FFGs as indicators of ecosystem attributes (Cummins *et al.*, 2005)

Ecosystem Parameter/Attribute	Symbols	Functional Feeding Group Ratios	Threshold Ratio
Autotrophy to Heterotrophy	P/R	Scrapers to (Shredders + Total Collectors)	Autotrophic>0.75
Coarse particulate organic matter (CPOM) to Fine particulate organic matter (FPOM)	CPOM/FPOM	Shredders to Total Collectors	Normal shredder linkage with functioning riparian >0.25
FPOM in transport (Suspended) to FPOM in storage (Deposited in Benthos)	TFPOM/BFPOM	Filtering collectors to Gathering collectors	FPOM Transport greater than normal FPOM loading in suspension >0.50
Substrate (Channel) Stability	Stable Channel	(Scrapers + Filterers) to (Shredders + Gatherers)	Stable Substrate Plentiful >0.50
Top-Down Predator Control	Top-Down Control	Predators to Total of all other groups	Normal predator Prey Balance 0.10-0.20

NB: P =Primary Production, R = Respiration

Despite the fact that macroinvertebrates are effective surrogates for ecosystem health and functioning, many tropical streams have inadequate information on functional composition, making this technique challenging to use (Boyero *et al*., 2009; Masese *et al*., 2014a). There is still paucity of information about macroinvertebrate FFGs of Kenyan streams for them to be fully used as surrogates of ecosystem attributes. Therefore, this study examined macroinvertebrate FFGs of the Sosiani-Kipkaren River a moderately disturbed river with diverse land-uses along temporal (wet and dry seasons) and spatial (land-use and longitudinal) scales and employed the FFG ratios as surrogates of ecosystem attributes.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study area

This study was conducted in the Sosiani-Kipkaren River, a tributary of the Nzoia River in Lake Victoria basin (LVB), Kenya. The river originates in the Kaptagat Forest, which is part of the larger Mau Forest Complex in the Kenyan Rift Valley. Geographically, the basin which is situated between latitudes 1^0 30’N and 0^0 05’S and between longitudes 34^0 and 36^0 45’E has a catchment area of 12,709 km^2 and is 334 km long from the source to the mouth in Lake Victoria (Engineers, 2002). It is located at an altitude of 1134m asl at Lake Victoria to the highest level at the peak of Mt. Elgon at 4,316 m asl. The climate of the area is mainly tropical humid, with mean annual rainfall ranging from 900 to 2200 mm and temperature ranging from 13 °C to 25 °C which varies strongly with elevation. The annual rainfall pattern is bimodal, with long rains between March and June, and short rains from September to November (Nyadawa & Mwangi, 2011).

The Nzoia River catchment serves approximately 3.5 million people with an average population density of 190-persons/ km^2 (GEF, 2004). The catchment area is characterized by forest cover at its upper reaches which transits to small and large-scale agriculture, settlements (rural and urban), grazing, urbanization and industrial activities. Agriculture both commercial and subsistence is the source of livelihood for a large proportion of the basin’s population. On a commercial scale the communities practice large scale farming of maize which are the grain baskets of the country and sugar cane farming though production has dramatically reduced in the recent years owing to the instability of the sugar mills within the region. Others commercial activities include tea farming and livestock keeping that ranges from small to large scale. On a subsistence scale the communities plant cabbages, potatoes, beans, kales, as well as keeping of donkeys and bees. Communities also abstract water from the river for irrigation and domestic chores as well utilizing the river as animals watering points and motor washing points. These land-use activities within the catchment potentially contribute to water quality deterioration and modifications of in-stream habitats.

3.2 Study design

Study sites were grouped into four categories characterized by catchment or riparian land-use, and reach-scale human influences: forested, agricultural, mixed and urban. Based on the Digital Elevation Model of Kenya (90 m by 90 m) produced using data from the Shuttle Radar Topography Mission, catchments were delineated and the area of each land-use category upstream of each sampling site was calculated. Forested sites, were sites that had a riparian zone that was > 60% forest and the catchment area upstream of the site had > 60% forest, shrublands or grasslands cover. Agricultural sites, had a riparian zone with > 60% agriculture and the catchment area upstream of the site with > 60% crop cover. Urban sites were located in urban areas within Eldoret City and its outskirts, with > 60% human settlements and other developments along the riparian zone. Mixed sites were those with varied percentages of the two primary land-uses; forest and agricultural, with none exceeding 60% areal coverage in the riparian zone and catchment areas (Masese *et al*., 2014b).

A total of 21 sites [Forested (n = 5), Mixed (n = 6), Agricultural (n = 6) and Urban (n = 4)] were sampled during the wet season. Due to logistical constraints, only 14 of the same sites [Forested (n = 3), Mixed (n = 5), Agriculture (n = 2) and Urban (n = 4)] were sampled during the dry season (Figure 1).

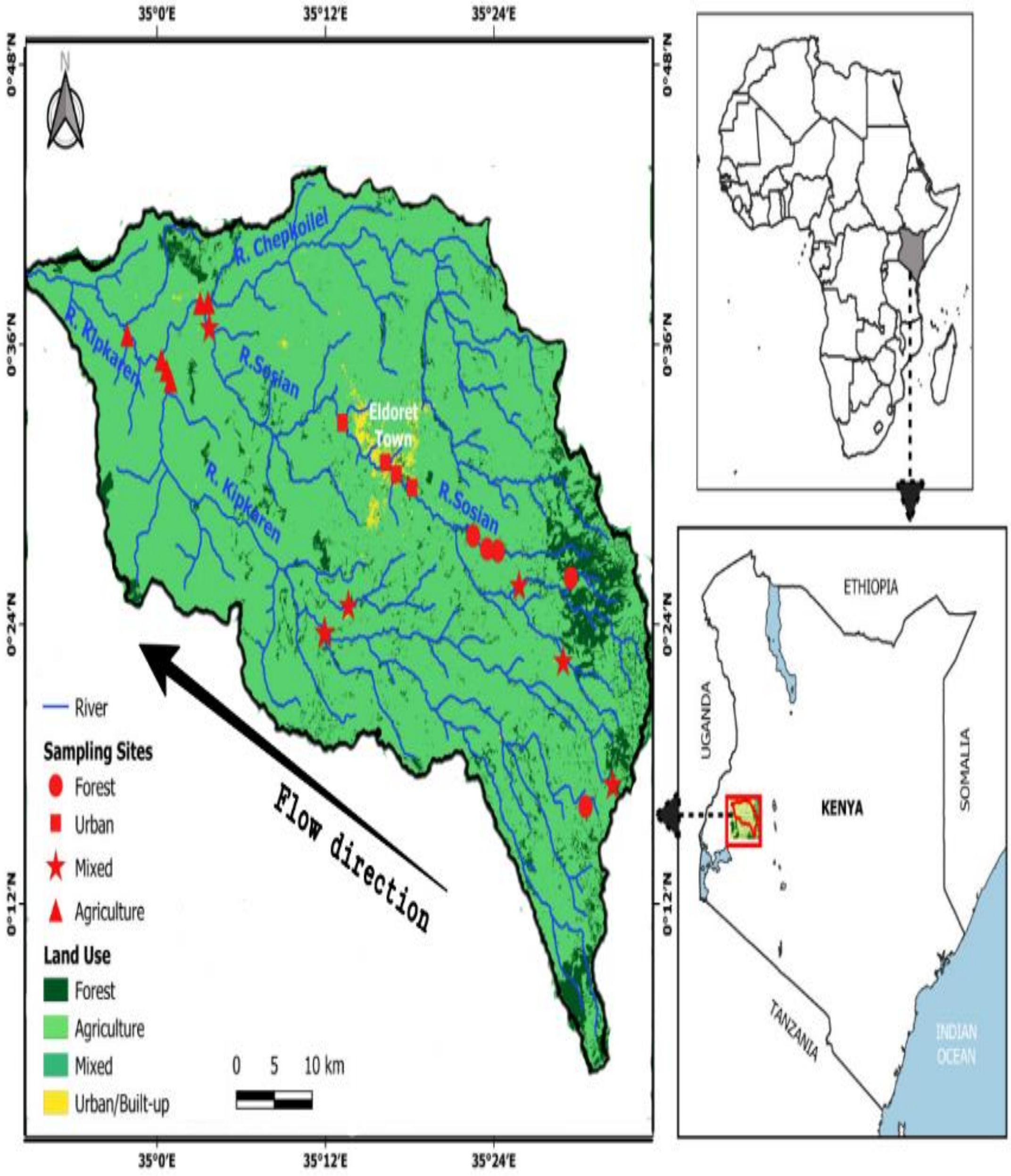


Figure 1: Location of sampling sites along the Sosiani-Kipkaren River, Kenya

The sampled sites traverse stream-size and land-use gradient from the forested upper-reaches, through the mixed, agricultural and urban middle reaches to the lower reaches that were largely agricultural (Table 2; Plate 2).

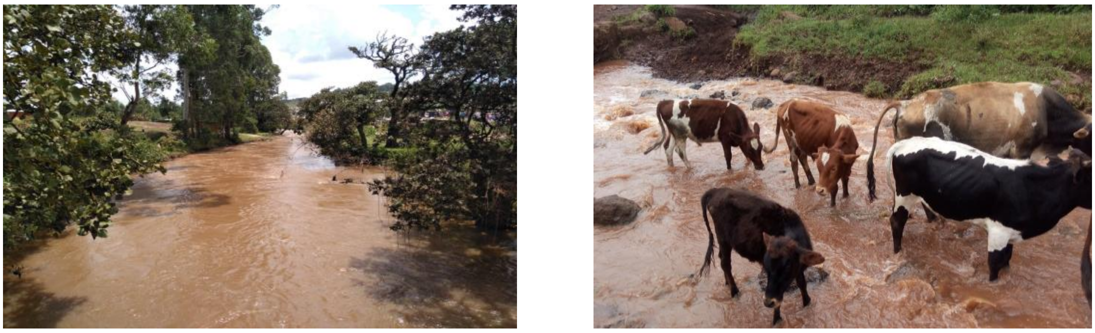
Table 2: Site groups, sites, elevation, GPS coordinates, and the dominant characteristics of the sites at the various site categories

Site group	Sites	Stream Order	Elevation (m)	Location (Latitude, Longitude)	Characteristics
Forested	Elseti upstream	1	2493	0° 16’ 10’’N, 35° 30’ 32’’E	-Forested catchment with > 60% under forest -Riparian zone intact with >70% instream canopy cover -Water clear with stony substrate -No major human activity
	Chepkoilel River upstream	2	2447	0° 25’ 59’’N, 35° 29’ 29’’E	
	Sabor at Chebolei Bridge	2	2246	0° 27’ 08’’N, 35° 24’ 17’’E	
	Kipsinende River	2	2232	0° 27’ 13’’N, 35° 23’ 31’’E	
	Kipsenganyi upstream	3	2433	0° 22’ 22’’N, 35° 28’ 56’’E	
Agriculture	Chepkoilel River at Turbo Bridge before confluence	4	1806	0° 37’ 48’’N, 35° 03’ 38’’E	-Agricultural catchment with > 60% under agriculture -Riparian zone less intact with agricultural activities along the banks -Mucky water with a lot of sedimentation -Some other human activities include water abstraction, bathing, washing clothes, animal grazing and small scale fishing and sand harvesting
	Sosiani River at Turbo before confluence	4	1805	0° 37’ 29’’N, 35° 03’ 25’’E	
	Kipkaren at Kipkaren-Sosiani confluence	5	1697	0° 35’ 10’’N, 35° 00' 21"E	
	Sosiani at Kipkaren-Sosiani confluence	5	1700	0° 35’ 20’’N, 35° 00’ 18’’E	
	Sosiani-Chep confluence at Turbo	5	1798	0° 37’ 46’’N, 35° 03’ 04’’E	
	Kipkaren River at Kipkaren Bridge	6	1651	0° 36’ 24’’N, 34° 57’ 53’’E	
Mixed	Cheboen upstream	1	2558	0° 17’ 05’’ N, 35° 32’ 28’’ E	-Located in agricultural catchment but agriculture and forest both are less than 60% -Agricultural practices at the
	Naiberi River at Sisibo Bridge	3	2227	0° 27’ 08’’N, 35° 24’ 17’’E	

	Kapsenegerute	3	2292	0° 25’ 37’’N, 35° 25’ 48’’E	catchment but some with intact riparian zones. -Some with banks with undercuts. -Used for water abstraction for domestic use and animal watering points -Sugar and paper industries along the river -Mixed canopy of both indigenous and exotic trees
	Mlango tributary	4	2065	0° 24’ 46’’N, 35° 13’ 38’E	
	Kipkaren River at Airport	4	2014	0° 23’ 34’N, 35°11’ 58’’E	
	Ngara falls river	3	2015	0° 23’ 36’’N, 35° 11’ 55’’E	
Urban	Sosiani at Nairobi Bridge	4	2105	0° 29’ 52’’N, 35° 18’ 10’’E	-Located in urban centres with human settlement -Eroded banks with undercuts with overhanging litter and litter deposits in the stream (plastic bags and bottles) and town sewage discharge -Greenish turbid smelly water -Some other human activities include; agriculture activities on the banks(maize and kales), burning and clearing of bushes, water abstraction and animal grazing grounds and watering points
	Sosiani at Elgon View	4	2080	0° 30’ 26’’N, 35° 17’ 03’’E	
	Sosiani River Sokoni	4	2064	0° 30’ 56’’N, 35° 16’ 17’’E	
	Sosiani River at Mille 4	4	1972	0° 32’ 38’’N, 35° 13’ 13’’E	

CHEPKOILEL (FOREST LANDUSE) SABOR (FOREST LANDUSE)

KIPKAREN (MIXED LANDUSE) KIPKAREN (MIXED LANDUSE)



KIPKAREN (AGRICULTURE LANDUSE) KIPKAREN (AGRICULTURE LANDUSE)

SOSIANI (URBAN LANDUSE) SOSIANI (URBAN LANDUSE)

Plate 2: Different stream sections in the four land-use categories (Forest, Mixed, Agriculture and Urban) along the Sosiani-Kipkaren River

3.3 Variation of Water and Habitat Quality

3.3.1 Water physico-chemistry

Sampling was done twice; during the wet season (July-August, 2018) and during the dry season (February-March, 2019) with the wet season being sampled during the short-light rains. At each sampling site, before sampling of macroinvertebrates, several variables were recorded *in situ* by directly inserting a YSI multi-probe water quality meter (556 MPS, Yellow Springs Instruments, Ohio, USA) into the stream. Some of the variables measured included; water temperature, dissolved oxygen concentration (DO), pH as well as electric conductivity (EC). For nutrient analyses, Duplicate filtered water samples were taken per site in acid-washed HDEP bottles, instantly fixed with sulphuric acid, and stored in a cooler before being transported to the laboratory, where they were stored at 4 ^{o}C before analysis. A total of 70 water samples were collected; 42 samples during the wet season and 28 samples during the dry season. For total suspended solids (TSS) and particulate organic matter (POM), at each sampling site, Known volumes of water samples were filtered through pre-combusted Whatman GF/F; Glass fiber filters of 0.42mm thickness, 0.7 µm pore size and 47mm diameter. The GF/F filters holding the suspended matter were wrapped in aluminium foil and stored in a cooler box before being transported to the University of Eldoret laboratory for processing.

3.3.2 Nutrients and cations laboratory analyses

Standard colorimetric procedures (APHA, 2005) were used in the lab to analyze the water column nutrients. The soluble nutrients; nitrites (NO_2), soluble reactive phosphorous (SRP), nitrates (NO_3), and ammonium (NH_4) were analyzed from filtered water samples. The ascorbic acid technique was used to analyze the SRP, with absorbance being read at a wavelength of 885 nm (APHA, 2005). The salicylate method was used to analyze NO_2 and NO_3, with the spectrophotometric absorbance being read at a wavelength of 543 nm (APHA, 2005). The reaction between sodium salicylate and hypochlorite solutions was used to analyze NH_4 with the spectrophotometric absorbance of the treated sample being read at a wavelength of 655 nm (APHA, 2005). Cations (Sodium (Na) and Potassium (K)) were determined using flame atomic absorption spectrometry (FAAS), detected at 589.0nm and 766.5 nm, respectively. The nutrients and cations concentrations were determined using equations generated from the standard calibration curves obtained from the absorbance values.

3.3.3 TSS and POM determination

GF/F filters with embedded sediments were dried at 60 ºC for 72 hours to attain constant weight. The filters were then re-weighed using an analytical balance and subtracting the filters weight for TSS determination.

TSS (mgL-1) = ((A – B)/V) * 106……………………………………Equation 1

Where: A = mass of filter + dried residue (g), B = dry mass of filter (g), and V = volume of sample filtered (L).

The filters were then ashed at 450 ºC for 4 hours in a muffle furnace and re-weighed for the determination of POM as the difference between TSS and ash-free-dry mass/weight.

Equation: POM (mgL-1) = ((C-B)/V) * 106………………………………Equation 2

Where: B = dry mass of filter (g), C = Weight of ashed filter (g) and V = volume of sample filtered (L).

3.3.4 Measuring of stream and habitat variables, and organic matter

At each sampling site, reach characterization was done by measuring the width of the stream, depth of the water, flow velocity and stream discharge along a reach. A measuring tape was used to determine stream width while a 1-m ruler was used to measure water depth at a number of points determined by channel shape and width along the river reach. Velocity was measured using a mechanical flow meter (General Oceanics; 2030 Flow meter, Miami, Florida) while discharge was calculated using velocity–area method (Wetzel & Likens, 2000).

Discharge (Q) = River cross-sectional area (A) × Velocity (v)……………Equation 3

Riparian and in-stream habitat assessment was also done at each site qualitatively to determine habitat quality and diversity using the Qualitative Habitat Evaluation Index (QHEI) adopted from Rankin (1995) and modified for the LVB (Masese *et al*., 2009b; Raburu & Masese, 2012) (Appendix 3). Habitat quality variables assessed included; substrate, instream cover, channel morphology, riparian zone and bank erosion and pool/glide and riffle/run quality, and at each point the percentage of substratum at each biotope was also estimated. The overall QHEI score is obtained as a summation of the aforementioned variables and it categorizes sites into three condition classes/levels; > 67.5 the habitat quality is excellent or very good, 52.5-67.5 it is good and <52.5 the habitat is described as marginal or poor.

For each site, the percentage of streambed covered by different substrate types was estimated for each biotope sampled for invertebrates. A biotope was divided into nine sub-sampling units of similar size, and counting the number of units occupied by various types of substrates. The dominant substratum was the particle size (boulders, cobbles, pebbles, gravel, sand and mud) that made up 50% or more of the streambed surface within the quadrat when classified according to a modified Wentworth scale into one of the size classes (Mykrä *et al*., 2007). Data on water depth and velocity were also collected for each sampled biotope.

In addition to substrate types, the biomass (standing stock) of course particulate organic matter (CPOM), was estimated by collecting CPOM samples in triplicates from each sampling site using a quadrat (0.5*0.5 m2) and placed in zip lock bags for transportation to the laboratory for processing. The CPOM collected was mainly composed of sticks, leaves, seeds, fruits and flowers. Percentage coarse particulate organic matter (% CPOM) was determined as the % coverage of the CPOM in the stream bed. In the laboratory, the CPOM was dried at 68°C for 48 h to constant mass and different fractions (leaves, sticks, seeds, fruits, and flowers) were weighed separately with a Sartorius balance to the nearest 0.1 mg (SECURA224-1ORU; Sartorius, Goettingen, Germany). The CPOM standing stock biomass was determined as the summation of the dry weight of the various fractions (Masese *et al*., 2014a).

3.4 Macroinvertebrates sampling and processing

3.4.1 Macroinvertebrates sampling

Macroinvertebrate sampling was conducted using a semi-quantitative kick-net sampling method (Dickens & Graham, 2002). Three major biotopes were delineated and sampled within each site: 1) GSM: gravel, sand and mud; (2) STONES: bedrock, boulders, cobbles and pebbles, either under flowing or non-flowing conditions; (3) VEG: submerged and marginal vegetation (Masese *et al*., 2021). The sampling process involved kicking/ disturbing the benthos an area of approximately 1 m^2 upstream of the kick net (500-µm mesh size), so that water current can wash the dislodged macroinvertebrates into the net. Three replicates per biotope were collected where kicking was carried out on for about a 1 minute per biotope. Larger substrate such as boulders and cobbles were disturbed by hand and washed into the net. Macroinvertebrate samples per biotope were preserved in 75% ethyl ethanol for further processing in the laboratory.

3.4.2 Macroinvertebrates sorting and identification

Macroinvertebrates samples were washed in running water in sieves (from 500µm) and transferred into sorting trays where they were counted and identified mainly to the genus level, with the aid of identification keys (Day *et al*., 2002 a, b; de Moor *et al*., 2003a, b; Stals & de Moor 2007; Merritt *et al*., 2008; Appendix I & II). Macroinvertebrates biomass was then determined by oven drying the macroinvertebrates at 103 ^{o}C for four hours and thereafter weighed using an analytical balance (Sartorius, Secura 124-1S, 0.0001g) (Mason *et al*., 1983).

Macroinvertebrate FFGs richness was obtained as the number of/count of species belonging to the different FFGs in the different land-use types while numerical abundance was determined as the number of individuals per species in the different FFGs.

3.4.3 Allocation of macroinvertebrate FFGs

Allocation of FFGs was done using the literature (Merritt & Cummins, 1996; Graca *et al*., 2001; Dobson *et al*., 2002; Polegatto & Froehlich, 2003; Molina, 2004; Masese *et al*., 2014a). Surrogates of ecosystem attributes were derived from ratios of the various FFGs (Vannote *et al.*, 1980; Merritt *et al.*, 2002; Cummins *et al.*, 2005) as:

i. Autotrophy vs. heterotrophy (production/respiration [P/R]) index which was determined as the scrapers to (shredders + total collectors) ratio;

ii. Linkage between stream food webs and allochthonus inputs; coarse particulate organic matter and fine particulate organic matter (CPOM/FPOM) index was calculated as the ratio of shredders to total collectors;

iii. The ratio of predators to prey (total of all other groups) was used to calculate the top-down control index;

iv. The filterers to gatherers ratio was used to determine the transport fine particle organic matter and benthic fine particulate organic matter (TFPOM/BFPOM) index;

v. Scrapers + Filterers: Shredders + Gatherers ratio was used to determine the stable channel index.

3.5 Statistical analyses

Statistical analyses were performed with R version 3.3.3. (R Development Core Team, 2017), using the packages vegan (Oksanen *et al.,* 2013), sem (Fox, 2006), and deSolve (Soetaert *et al*., 2010). Figures were created in SigmaPlot (Version 12), MS Office Excel (2016) and R version 3.3.3 (R-Development-Core-Team, 2017).

Descriptive statistics (means ± standard deviation) and plots were used to present spatial and temporal variation in water and habitat quality variables at different site categories.

Two-way analysis of variance (ANOVA) was used to test for differences in physico-chemical and habitat variables among land-uses (forested, mixed, agricultural, and urban) and seasons (dry and wet) with land-use and seasons as main factors and land-use × season interaction term. Where there were no significant seasonal differences, data were pooled and one-way ANOVA used to test for differences among land-uses followed by Tukey multiple post hoc comparisons of the means. Prior to analysis count data were log (x+1) transformed while the rest of the response variables were log-transformed to meet normality assumptions.

Principal Component Analysis (PCA) was used to reduce the dimensionality of the physico-chemistry and habitat variables data. Two PCs were included to describe water quality physico-chemical variables and habitat quality variables separately. PCAs were statistically assessed using PERMANOVA (permutational analysis of variance), based on Bray-Curtis similarity matrices (McArdle & Anderson, 2001).

Habitat quality and diversity was determined by getting the mean of the QHEI metrics/components at the different land-use site categories and assessing them against the three condition classes/levels; excellent or very good, good and marginal or poor.

Community structure was described in terms of taxon richness, abundance, biomass and community indices. Species occurrence (presence-absence) and distribution data were summarized for each site and means calculated for each land-use category using the number of taxa (S) and the total relative abundances. Several reach- scale diversity indices were calculated for each study site and means calculated for each site category. Shannon's diversity index (H′) was derived as a measure of diversity (Magurran, 2004), and an associated H′/H′max index (Pielou, 1975) was used as a measure of evenness. The reciprocal form of the Simpson index (1-Ds) (Simpson, 1949) was used as a measure of species richness. Hill's number (i.e., gamma diversity; Hill, 1973) and Fisher's alpha (Fisher *et al*., 1943) were used as extra measures of macroinvertebrates diversity. Hill's number was calculated as the ratio between H′ and 1/D (Hill, 1973). Margalef’s species richness index was also calculated as an extra measure of taxon richness.

Two-way ANOVA was then used to test for differences in total abundance, biomass, and taxon richness of all taxa between seasons (dry and wet) and the four land-use categories (forested, mixed, agricultural and urban) with the main factors as season and land-use with a season × land-use interaction.

The average rank similarities of macroinvertebrate FFGs were compared between the wet and dry seasons using two-way nested analysis of similarities (ANOSIM), with replicate land-uses nested within seasons. ANOSIM calculates the R-statistic, which is a test statistic that varies between 0 and 1; higher values indicate bigger differences between factors.

Non-metric multidimensional scaling (NMDS) was then used to visualize functional composition of macroinvertebrates in different land-uses and seasons (Clarke & Gorley, 2006). Using Bray–Curtis (Bray & Curtis, 1957) coefficients, dissimilarity matrices were derived for 2 sets of data: un-transformed abundances data and presence–absence data for the FFGs. The magnitude of the associated stress value (< 0.2 corresponding to a good ordination) was used to determine the ordination's fit (Kashian *et al.*, 2007).

Similarity percentages analysis (SIMPER) was performed to establish which key macroinvertebrates were accountable for the variations observed between land-uses (indicator macroinvertebrates for changes in land-use, habitat and water quality). The %contribution FFGs to the overall dissimilarity was quantified between land-uses per season. SIMPER is a restrictive pairwise analysis between two factor levels (Clarke & Warwick, 2001), and in this case, comparisons were done between forested and mixed, forested and agricultural, and finally forested and urban land-use sites.

Redundancy analysis (RDA) was used to elucidate relationships between macroinvertebrate assemblages and environmental variables. The output was displayed as triplots, in which the plotted points for taxa and FFGs and land-use categories could be related to physico-chemical and habitat variables that were represented as rays. Before RDA was performed the gradient length in standard deviation (SD) units was estimated using Detrended Correspondence Analysis to test the suitability of a Canonical Correspondence Analysis (Ter Braak & Smilauer, 1998). Because the gradient length was less than 3 SD, RDA was employed instead of CCA to determine which factors were responsible for the structure or groupings of FFGs among site categories (Ter Braak & Smilauer, 2004).

The ratios of the macroinvertebrate FFGs from the four land-use categories were used to calculate mean values of stream ecosystem attributes. The ratios were derived from macroinvertebrate FFGs numerical abundance and biomass during both the wet and dry seasons.

To assess longitudinal trends in metrics used as surrogates of ecosystem attributes with changes in stream size (river width), generalized additive models (GAMs) (Wood, 2017) was used. GAMs usually incorporates smooth functions that are more flexible in modelling nonlinear relationships (Hastie & Tibshirani, 1990). GAMs were selected over more commonly used linear regression techniques because typical patterns in compositions of FFGs along streams and rivers are hypothesized to be nonlinear (e.g., Vannote *et al.,* 1980). GAMs were built using penalized cubic regression splines with degrees of freedom automatically identified based on the generalized cross-validation score (GCV). GAMs were fitted using the R-package *mgcv* (Wood & Wood, 2015).

CHAPTER FOUR

RESULTS

4.1 Water physico-chemistry and nutrients

Both season and land-use change played significant roles in influencing water and habitat quality variables in the study. There was a significant decline in habitat quality, mainly riparian zone quality and instream cover from forested to urban land-uses (Table 3). Significant differences were recorded across the variables with higher values being recorded during the dry season than the wet season in all the physico-chemical and nutrient variables except for dissolved oxygen (DO), TSS, POM and depth. The Percentage of CPOM coverage and its standing stock biomass also declined among land-uses though did not differ significantly (Table 3).

Highest temperature levels were recorded in agricultural sites (19.5 ± 0.3 ◦C) and (22.6 ± 0.4 ◦C) while forested sites recorded the lowest (14.6 ± 0.4 ◦C) and (15.6 ± 1.0 ◦C) during the wet and dry seasons, respectively. Forested sites recorded significantly higher DO concentrations during both wet (7.1 ± 0.4) and dry (6.5 ± 0.9) seasons. Highest conductivity was recorded in the agricultural sites during the dry season (241 ± 25.9 µS/cm) while the lowest was recorded in the forested sites (32 ± 2.1 µS/cm) during the wet season (Table 3). The concentrations of nutrients were lowest in forested sites in both the dry and wet seasons while agricultural and urban sites recorded highest levels of nitrates and SRP during both seasons.

Qualitative habitat evaluation index (QHEI) mainly focuses on five major components and the less disturbed forest sites scored highly on the instream cover (13 ± 0.516), substrate (14 ± 0.792) and riparian zone quality (15 ± 1.183) metrics while the disturbed urban and agricultural sites scored poorly in those metrics (Table 3). Forest land-use sites varied significantly from the other land-use sites in instream cover and riparian zone and bank quality but substrate and channel morphology was not significantly different across the different site categories. Agriculture and urban sites had high scores in the pool/glide and riffle/run quality (Table 3). From the total scores, forest land-use sites had good habitat quality (QHEI Score of 56) while mixed (score = 46), agriculture (score = 45) and urban (score = 42) all had marginal or poor habitat quality (Table 3).

The PCA biplot combining water physico-chemistry and habitat quality data (Figure 2) collected during both the dry and wets seasons indicated clear seasonal gradients and hence further analysis was done for each season separately (Figure 3). The Principal component 1 (PC 1) of the PCAs explained 34.9-35.1% of the total variation, while PC 2 explained 18.5-20.5% of the total association (Figures 2).

Table 3: Means (± SE) variation of physico-chemical variables, habitat quality and stream size variables in the different land-use categories. EC= electrical conductivity, SRP= soluble reactive phosphorus, TSS= total suspended solids, POM= particulate organic matter and CPOM=course particulate organic matter

Variable	Season	Forest	Mixed	Agriculture	Urban	F-Value	*p-value*
Seasonal variation							
Temperature (°C)	Wet	14.6±0.4^{b}	18.3±0.6^{a}	19.5±0.3^{a}	18.7±0.4^{a}	19.3	*0.001**
	Dry	15.6±1^{c}	17.8±0.4^{b}	22.6±0.4^{a}	21.5±0.3^{a}	29.7	*0.001**
EC (µS/cm)	Wet	32±2.1^{b}	82±7.5^{a}	101±4.1^{a}	96±7.2^{a}	25.99	*0.001**
	Dry	58±7.7^{c}	102±12.9^{c}	241±25.9^{a}	165±12.3^{b}	23.61	*0.001**
Dissolved oxygen (mg/L)	Wet	7.1±0.4^{a}	6.3±0.1^{ab}	6.4±0.1^{ab}	5.9±0.2^{b}	4.25	*0.008**
	Dry	6.5±0.9^{a}	5.9±0.2^{a}	4.5±0.5^{b}	4.6±0.4^{b}	3.86	*0.042**
pH	Wet	6.7±0.1^{a}	6.4±0.13^{ab}	6.7±0.147^{a}	5.9±0.2^{b}	5.87	*0.001**
	Dry	7.1±0.04^{b}	7.1±0.05^{b}	7.3±0.03^{a}	7.2±0.02^{a}	8.33	*0.001**
Nitrites (mg/L)	Wet	0.1±0.01^{a}	0.1±0.01^{a}	0.1±0.01^{a}	0.1±0.02^{a}	4.49	*0.208*
	Dry	0.2±0.1^{a}	0.6±0.3^{b}	0.1±0.11^{a}	0.1± 0.03^{a}	1.63	*0.008**
Nitrates (mg/L)	Wet	0.1±0.02^{c}	0.2±0.02^{bc}	0.3±0.1^{a}	0.3±0.1^{ab}	6.88	*0.001**
	Dry	1.3±0.4^{a}	1.9±0.7^{a}	3.8±0.9^{b}	2.7±0.6^{c}	1.89	*0.001**
SRP (mg/L)	Wet	0.04±0.02^{b}	0.05±0.04^{b}	0.1±0.01^{a}	0.1±0.01^{ab}	9.44	*0.001**
	Dry	0.02±0.01^{b}	0.02±0.01^{b}	2.2±1.2^{a}	0.2±0.1^{b}	6.79	*0.002**
Ammonia (mg/L)	Wet	0.01±0.01^{a}	0.01±0.01^{a}	0.03±0.02^{a}	0.02±0.01^{a}	0.86	*0.47*
	Dry	0.2±0.1^{a}	0.3±0.1^{b}	0.4±0.1^{b}	0.4±0.1^{b}	6.68	*0.002**

TSS (mg/L)	Wet	20.4±7.5^{b}	33.9±19^{ab}	43.1±6.1^{a}	27.3±7.4^{ab}	3.63	_0.034*_
	Dry	7.0±0.3^{a}	12.2±11.7^{a}	26.3±0.9^{a}	17.0±7.1^{a}	1.79	_0.212_
POM (mg/L)	Wet	8.4±4.9^{a}	13.5±4.6^{a}	13.7±3.9^{a}	10.8±5.2^{a}	1.55	_0.238_
	Dry	3.1±0.1^{a}	3.6±3.2^{a}	6.6±1.4^{a}	4.6±2.2^{a}	0.86	_0.492_
Depth (m)	Wet	0.3±0.03^{b}	0.2±0.03^{b}	0.7±0.12^{a}	0.3±0.08^{b}	7.43	_0.02*_
	Dry	0.3±0.13^{a}	0.2±0.04^{a}	0.2±0.03^{a}	0.2±0.04^{a}	0.69	_0.58_
No seasonal variation							
Sodium (mg/L)		1.8±0.3^{b}	2.8±0.2^{ab}	2.1±0.3^{ab}	3.3±0.6^{a}	3.04	_0.038*_
Potassium (mg/L)		1.3±0.2^{a}	2.3±0.3^{a}	1.7±0.3^{a}	1.9±0.4^{a}	2.28	_0.092_
Width (m)		5.4±0.9^{b}	4.9±0.6^{b}	15.7±1.8^{a}	10.4±2.7^{ab}	10.41	_0.001*_
Discharge (m^{3}/s)		0.4±0.01^{a}	0.1±0.03^{a}	0.1±0.01^{a}	0.3±0.08^{a}	2.72	_0.061_
Substrate type		13.2±0.8^{a}	13.8±1.0^{a}	13.2±1.1^{a}	13.3±1.1^{a}	0.06	_0.979_
Instream cover		14.2±0.5^{a}	9.8±1.2^{b}	9.0±0.9^{b}	6.8±0.7^{b}	8.56	_0.001*_
Channel morphology		7.4±0.3^{a}	6.8±0.3^{a}	6.8±0.3^{a}	6.5±0.2^{a}	0.93	_0.449_
Riparian zone and bank erosion		15.4±1.2^{a}	13.8±1.1^{ab}	9.3±1.3^{b}	9.3±1.1^{ab}	4.69	_0.015*_
Pool/Glide Riffle/Run Quality		5.8±0.5^{a}	5.5±0.3^{a}	7.2±0.8^{a}	6.8±0.5^{a}	2.96	_0.061_
Total Habitat Quality score (max score)		56.0±2.8^{a}	49.8±1.0^{ab}	45.5±2.0^{b}	42.5±2.3^{b}	7.55	_0.002*_
% CPOM		52.5±6.4^{a}	48.6±6.3^{a}	41.3±5.7^{a}	37.7±5.5^{a}	0.55	_0.652_
CPOM standing stock (g/m^{2})		60.2±13.9^{a}	45.7±7.3^{a}	28.7±6.5^{a}	35.3±8.9^{a}	2.1	_0.108_

*Means that do not share a letter are significantly different, Tukey posthoc tests

*P-values marked with asterisks are significantly different among site categories at $p < 0.05$

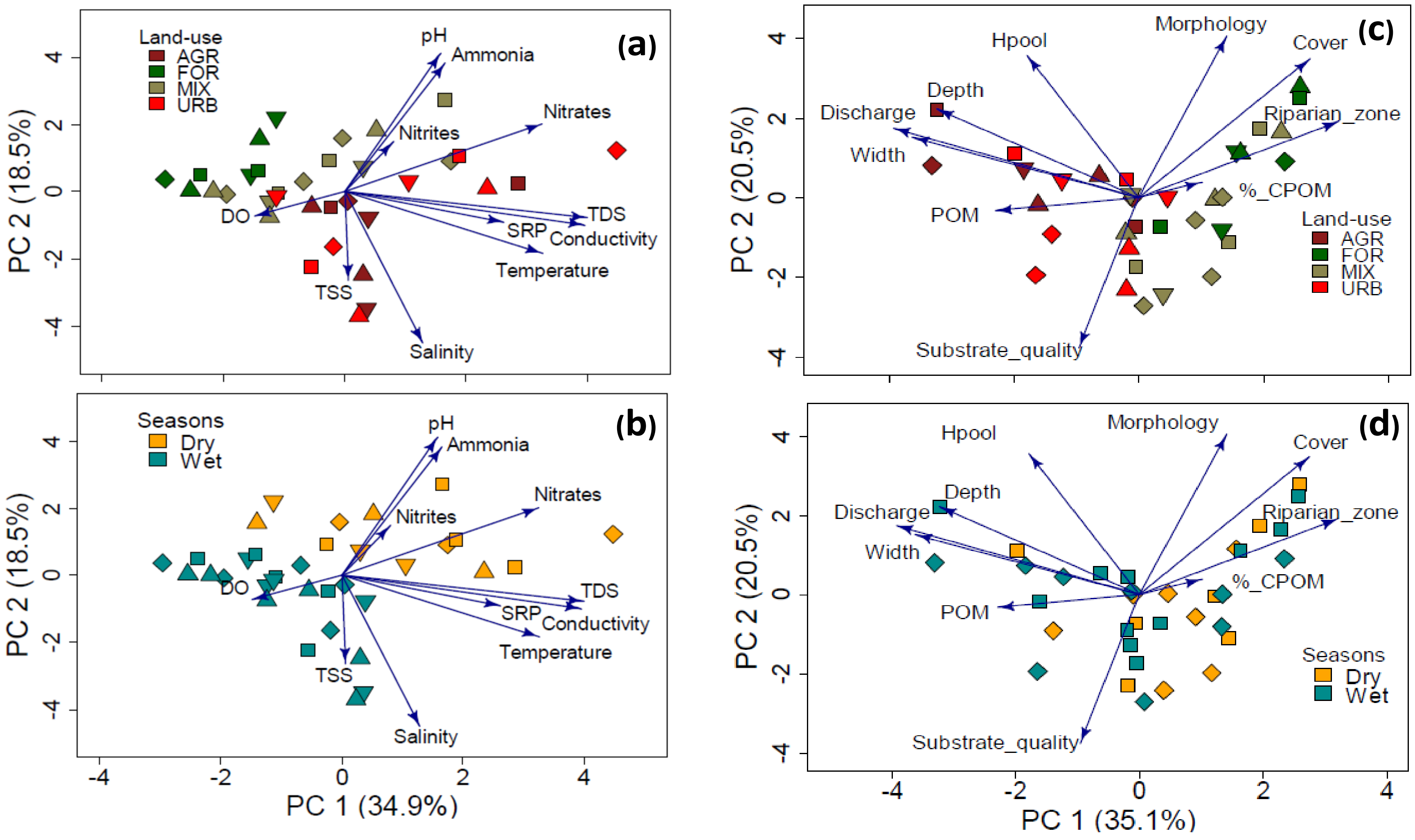


Figure 2: PCA biplot for all data for water quality variables (a, b) and habitat quality and stream size variable (c, d) for land-use (a, c) and season (b, d) in the Sosiani-Kipkaren River

Ordination of habitat quality and stream size variables indicated that forested and some mixed sites had protected riparian areas with stable instream substrate. Principal component 1 (PC 1) of the PCA explained 29.3-38.6% of the total variation in habitat and stream size data, while PC 2 explained 21.2-25.5% of the total association (Figure 3a, b). During both seasons % CPOM, substrate quality, instream cover, riparian zone and bank erosion and morphology were associated with the forested and mixed land-use types, while POM (read TSS), discharge and river width were associated with agriculture and urban sites.

Ordination of water quality physico-chemistry data indicated that principal component 1 (PC 1) of the PCA explained 38.7-44.2% with PC 2 explaining 19.4-24.9% of the total variation. During the dry season, high levels of DO were associated with forested and mixed sites, while higher levels of nutrients (SRP and nitrate), conductivity, TSS, temperature were associated agricultural and urban sites. During the wet season, similar trends were noted but with increased levels of electrical TSS, conductivity and nutrients in agricultural and urban sites (Figure 3c, d).

Habitat quality and organic matter characteristics had significant differences between seasons (PERMANOVA, F = 2.1, df = 1, p = 0. 05), among land-uses (PERMANOVA, F = 4.3, df = 3, p = 0. 001), but without a significant season*land-use interaction (PERMANOVA, F =1.3, df = 3, p = 0. 21). For water quality physico-chemical variables, stronger seasonal (PERMANOVA, F =18.1, df = 3, p = 0. 001) and among land-uses (PERMANOVA, F = 7.9, df = 3, p = 0.001) differences were obtained, with a significant season*land-use interaction (PERMANOVA, F = 3.4, df = 3, p = 0.001).

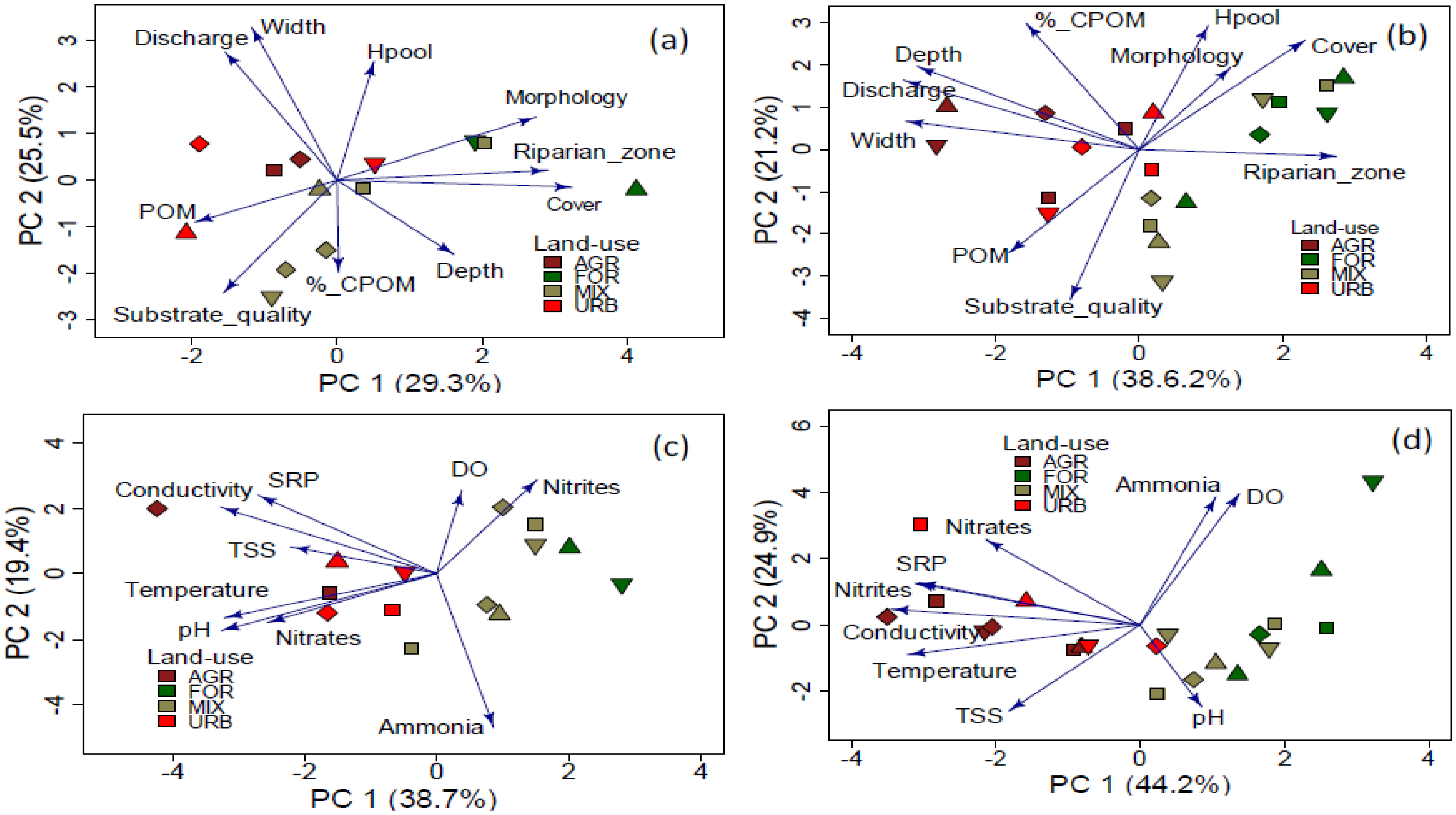


Figure 3: PCA biplot for habitat quality (a, b) and water quality physico-chemical variables (c, d) during the dry season (a, c) and wet season (b, d) in the Sosiani-Kipkaren River. EC = electrical conductivity, DO = dissolved oxygen, SRP = soluble reactive phosphorus, TSS-total suspended solids

4.1.1 Nutrient concentrations

During both the dry and wet season, nitrates and SRP were highest in the agricultural and urban sites (Figure 4). Nitrites and nitrates did not significantly vary during the dry season (*p* > 0.05) (Table 3). Nitrates recorded highest values than nitrites in both seasons with much variation during the dry season. The highest nitrates level were recorded in the dry season 3.8 ± 0.864 mg/L (Table 3). During the rainy season all nutrients recorded low concentration (0.01 ± 0.005 - 0.3 ± 0.051 mg/L) as compared to the dry season (0.1 ± 0.026 - 3.8 ± 0.864mg/L) (Figure 4).

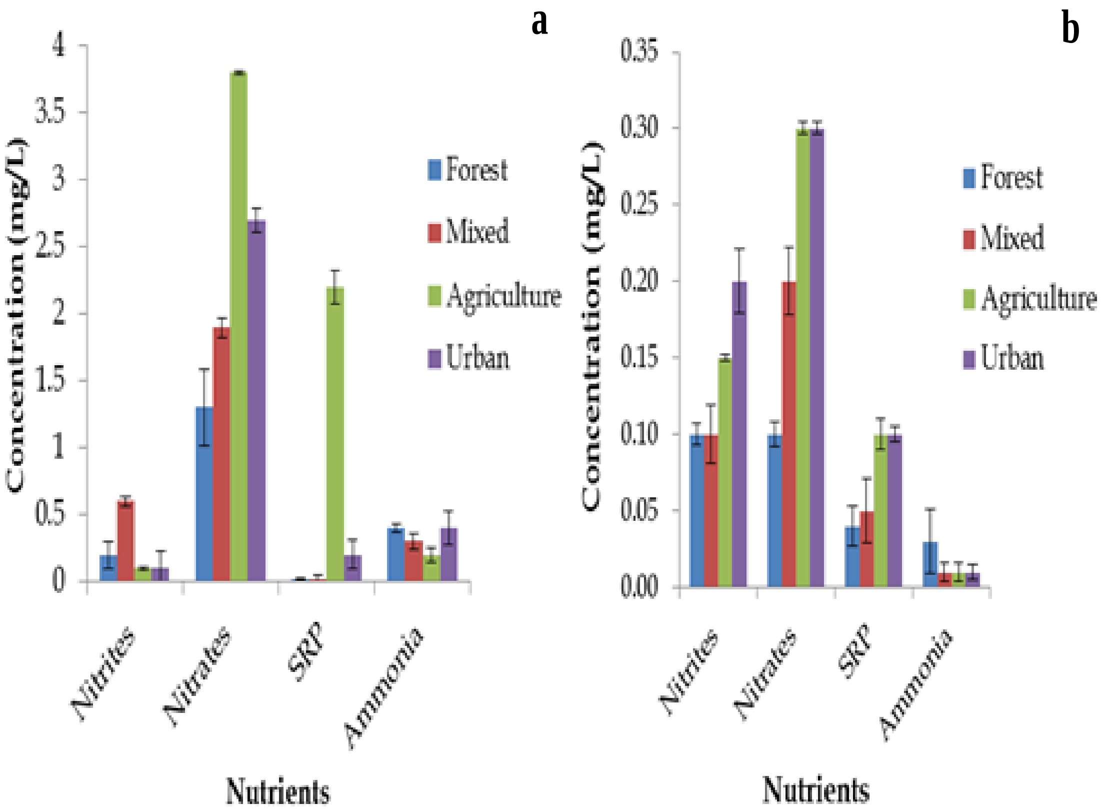


Figure 4: Nutrient concentration across the different land-use sites; dry season (a) (February-March 2019) and wet (b) (July-August 2018) season along the Sosiani-Kipkaren River. Error bars represent standard Error

4.1.2 Cations

Cation concentrations varied across the different land-use sites (Table 3). Mixed and urban land-use sites had high levels of both sodium (2.8 ±0.24 and 3.3 ±0.645 mg/L, respectiely) and potassium (2.3 ±0.267 and 1.9 ±0.353 mg/L, respectively) while the forest land-use sites had low levels of both ions; sodium (1.8 ±0.291 mg/L) and potassium (1.3 ±0.239 mg/L) (Figure 5). Sodium recoreded higher levels than potassium across the different land-use sites (Figure 5).

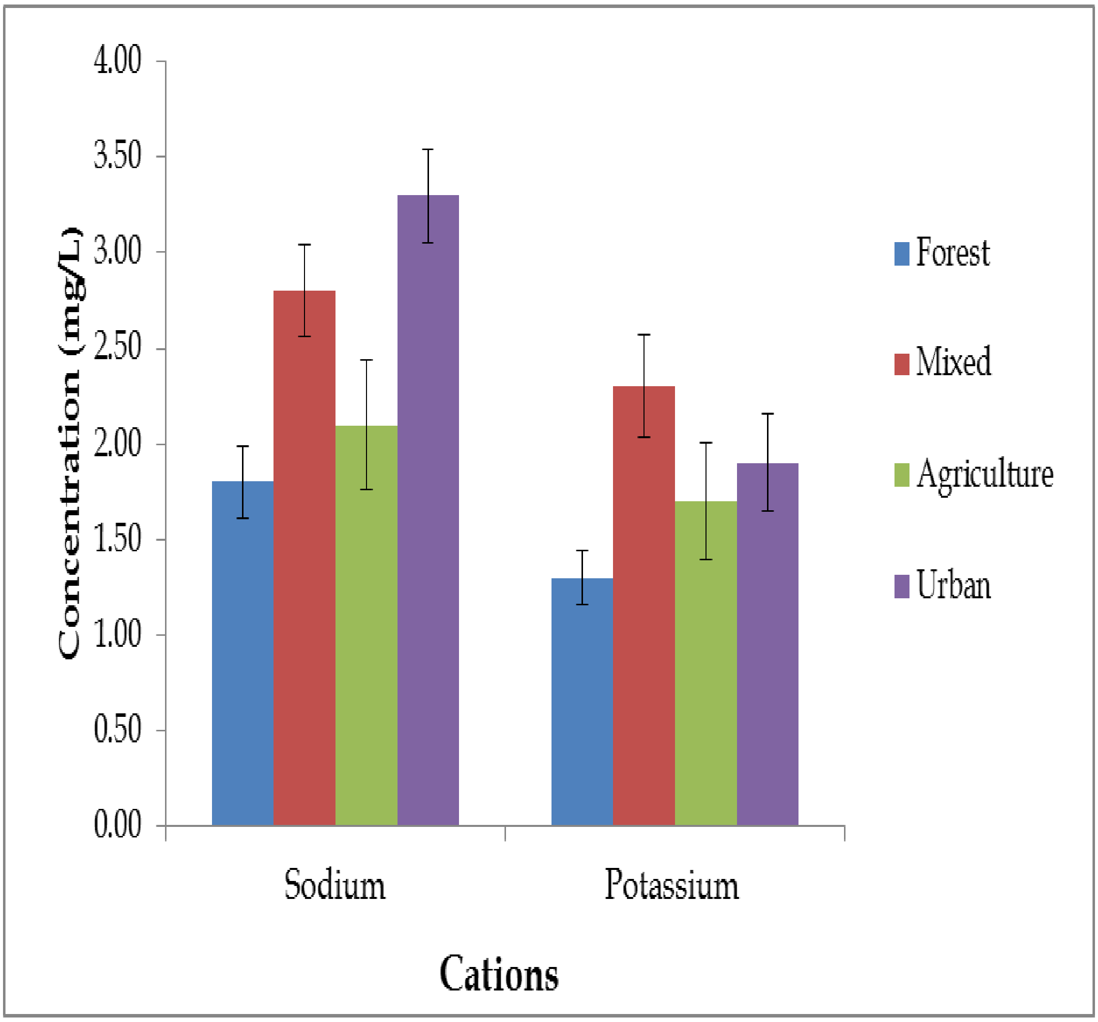


Figure 5: Cations concentration across the different land-use sites along the Sosiani-Kipkaren River. Error bars represent standard Error

4.2 Macroinvertebrates structural and functional composition, diversity and biomass

A total of 43,479 macroinvertebrate individuals were collected in the study area. Total abundance was higher during the wet season (35,827) compared with the dry season (7,652). A total of 15 orders, 68 families and 98 genera were collected during the wet season while 13 orders, 53 families and 67 genera were collected during the dry season. During the wet season, Ephemeroptera were the most abundant with 13,692 individuals, followed by Diptera (5586) then Tricladida (4950). The least abundant orders were Arachnida, Lepidoptera and Collembola with 33, 31 and 1 individuals respectively (Figure 6).

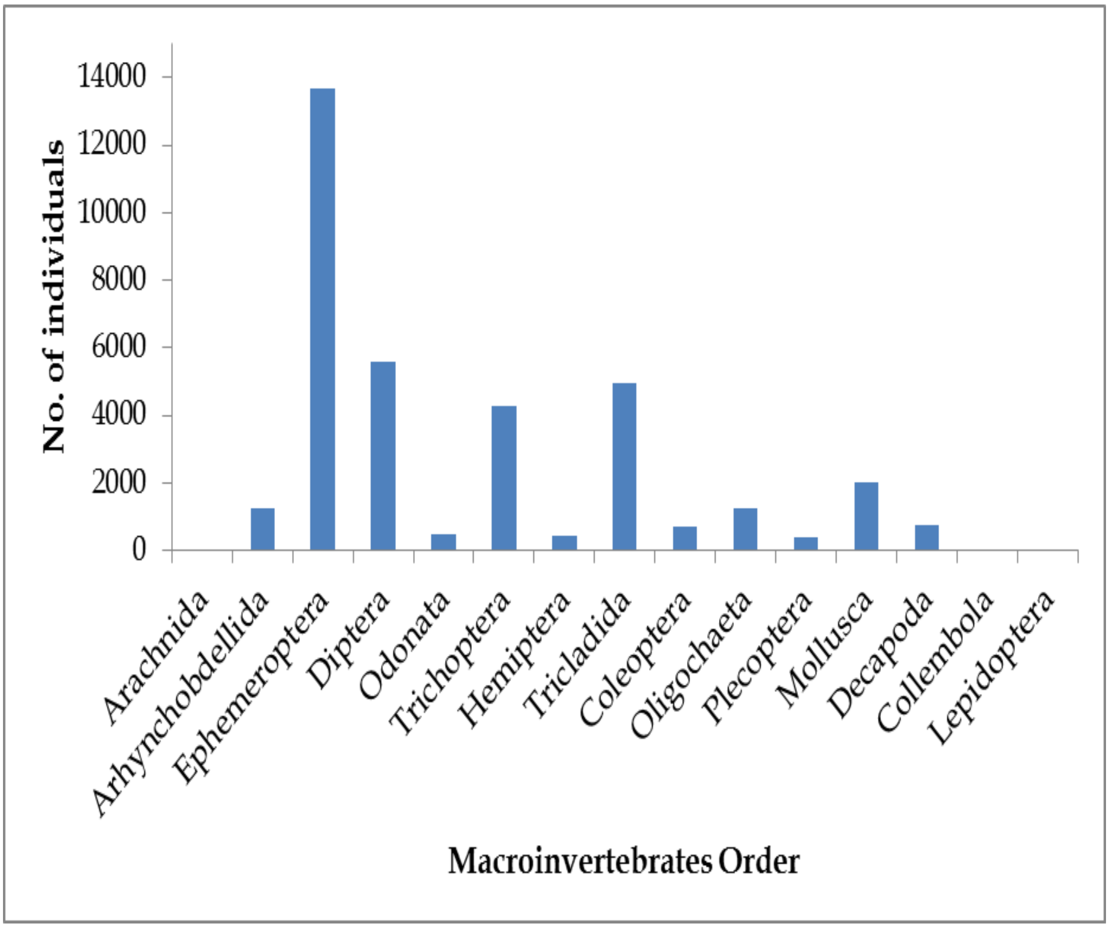


Figure 6: Abundance of macroinvertebrate orders during the wet season in July-August 2018 in the Sosiani-Kipkaren River

During the dry season, Diptera were the most abundant with 3611 individuals followed by Ephemeroptera with 1363 individuals, then Mollusca with 926 individuals. The least abundant orders were Tricladida, Lepidoptera, and Arhynchobdellida with 12, 10 and 7 individuals, respectively (Figure 7).

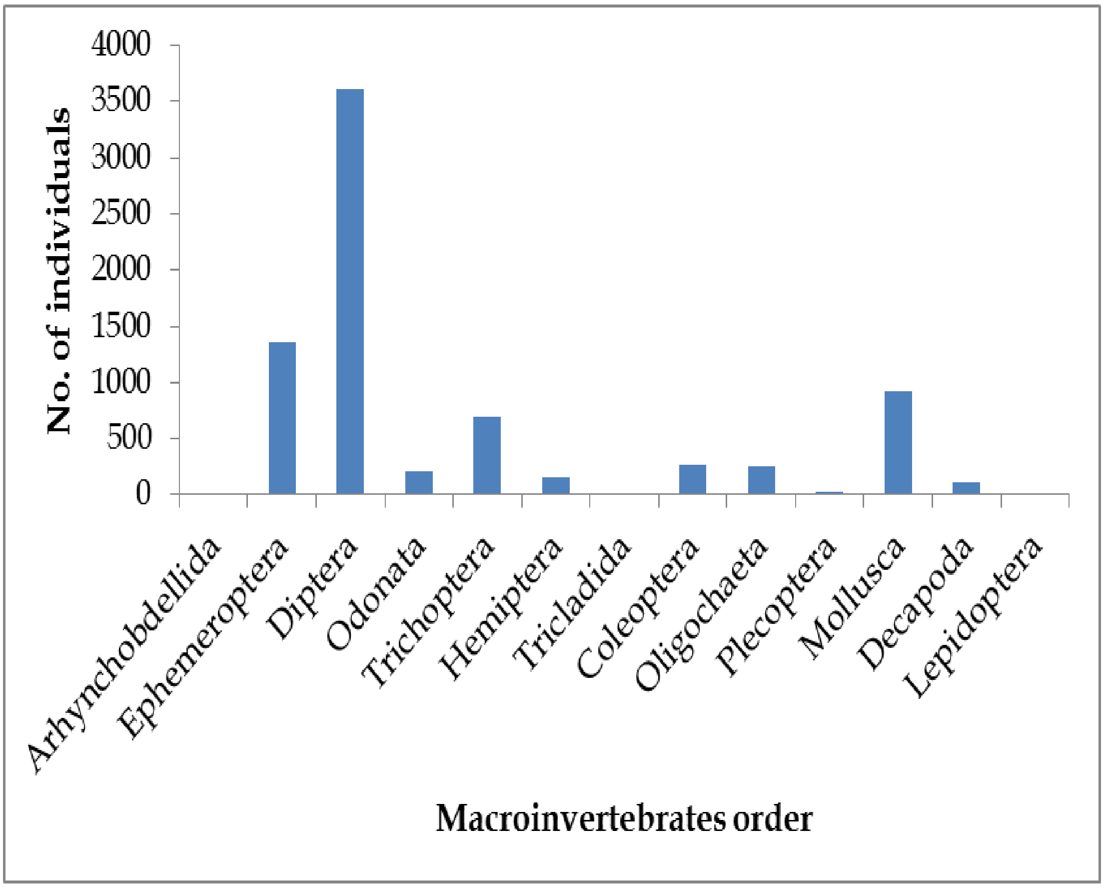


Figure 7: Abundance of macroinvertebrate orders during the dry season in February-March 2018 in the Sosiani-Kipkaren River

4.2.1 Spatio-temporal patterns of macroinvertebrates community composition

During the dry season, Ephemeroptera, Plecoptera and Trichoptera (EPT taxa) dominated the forest sites and decreased in the urban and agricultural sites (Figure 8), a pattern that was exhibited by the Coleopterans too (Figure 8). Dipterans were the dominant taxa in the agriculture and urban sites (Figure 8).

During the wet season, forest sites were still dominated by the EPT taxa that reduced through to the urban sites (Figure 8). The agriculture and urban sites were dominated by other macroinvertebrates taxa including the Odonata, Mollusca and Tricladida (Figure 8). Coleoptera taxon was highly abundant in the forest land-use sites (Figure 8).

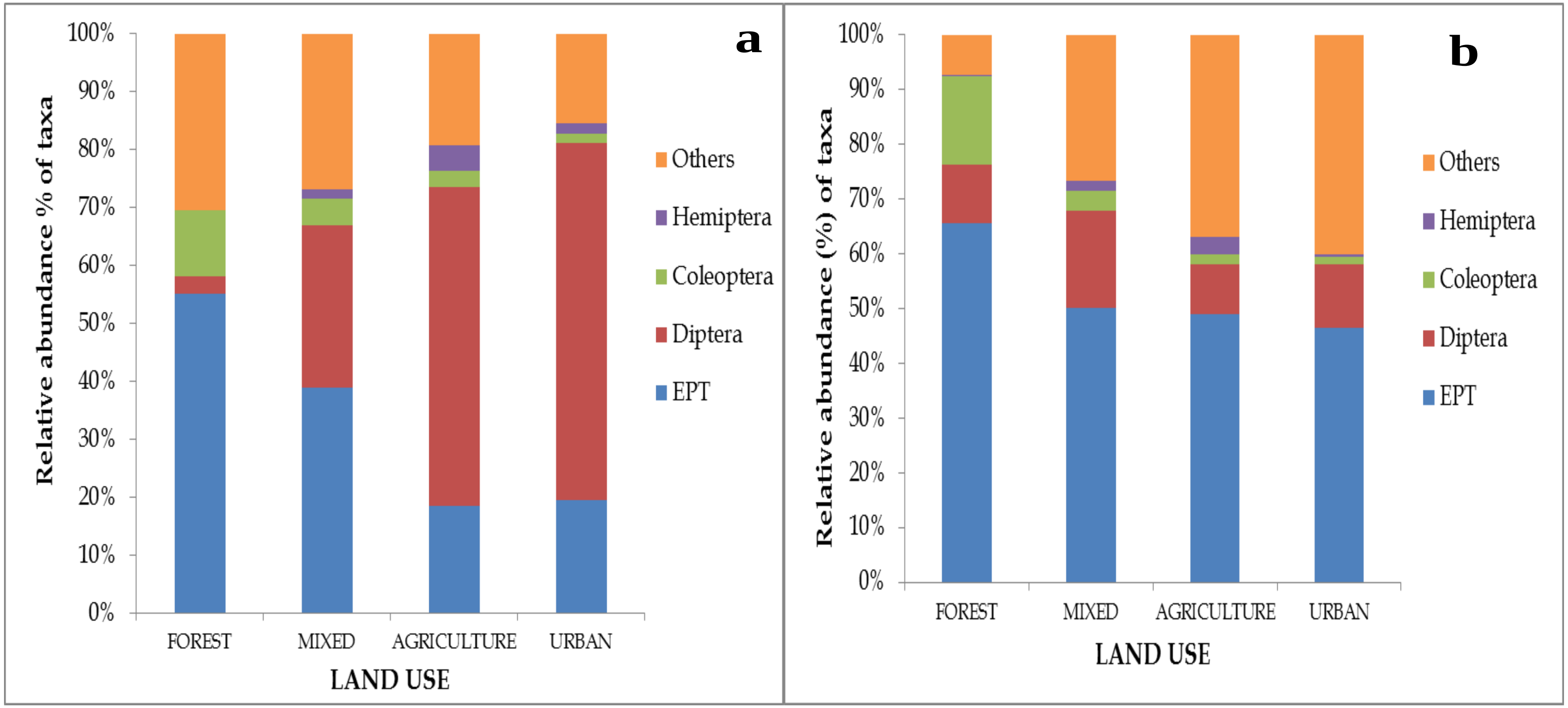


Figure 8: Relative abundance of macroinvertebrates taxa across the different land-use site categories during the dry season (a) in February-March 2019 and the wet season (b) in July-August 2018. EPT= Ephemeroptera, Plecoptera and Trichoptera

There were mixed results in diversity indices used to measure community structure of macroinvertebrates across the four land-use categories during the wet and dry seasons, with some showing wide ranges, such as taxon richness, dominance, Margalef's species richness index and Fisher's alpha diversity, while the rest showed narrow ranges (Table 4). During both seasons, Shannon diversity index was higher (3.15 and 3.07) in forested sites while urban had the least diversity (2.53 and 2.09) (Table 4). Similarly, Simpson index (1/*Ds*) had the same trends with higher values in forested sites and least in urban sites during both seasons (Table 4).

Pielou's evenness index displayed the lowest response across the land-use and seasonal gradient with values of (0.29 and 0.38) at forested sites and (0.20 and 0.16) at urban (Table 4). In contrast, Fisher's alpha diversity showed the widest range with the highest value (12.43 and 11.19) at forested sites during the wet and dry seasons respectively with again urban sites recording the least values (Table 4).

Dominance followed the opposite trend as Fisher's alpha diversity index with the highest values (0.14 and 0.32) at urban sites and the lowest values (0.06 and 0.07) at forested sites for the wet and dry seasons respectively. Forested sites had the highest number of taxa (79 and 56) with mixed sites having the least taxa (62 and 27) during both seasons (Table 4).

Table 4: The diversity indices of macroinvertebrate communities in the Sosiani-Kipkaren River during the dry season in February-March 2019 and wet season in July-August 2018

	Wet season				Dry season			
	Forest	**Mixed**	**Agriculture**	**Urban**	**Forest**	**Mixed**	**Agriculture**	**Urban**
Taxa_S	79	62	70	63	56	27	46	51
Individuals	9,124	7,421	3,461	14,148	1,657	810	1,150	4,035
Dominance_D	0.06	0.08	0.11	0.14	0.07	0.10	0.23	0.32
Simpson_1-D	0.94	0.92	0.89	0.86	0.93	0.90	0.77	0.68
Shannon_H	3.15	2.88	2.89	2.53	3.07	2.64	2.39	2.09
Pielou's evenness (*J’*)	0.29	0.28	0.26	0.20	0.38	0.32	0.24	0.16
Margalef’s species richness index	8.55	6.96	8.47	6.49	7.42	3.88	6.39	6.02
Equitability_J	0.72	0.69	0.68	0.61	0.76	0.80	0.62	0.53
Fisher_alpha	12.43	9.45	11.89	8.49	11.19	5.38	9.59	8.23
Hill's number (gamma diversity)	4.97	4.32	3.28	2.87	4.56	3.70	1.83	1.51

4.2.2 Spatio-temporal patterns of macroinvertebrates functional composition

Five functional feeding groups (FFGs) collected in the study sites along the Sosiani-Kipkaren River were scrapers, collector-gatherers (gatherers), collector-filterers (filterers), predators and shredders.

During the dry season, predators (3,641 individuals) were the most abundant, followed by filterers (1427) and scrapers (1314) (Figure 9a). Shredders were the least abundant (442 individuals). During the wet season, scrapers (11700) were the most abundant followed by predators (9144) and filterers (8005) (Figure 9b), again, shredders being the least abundant with 1546 individuals (Figure 9b).

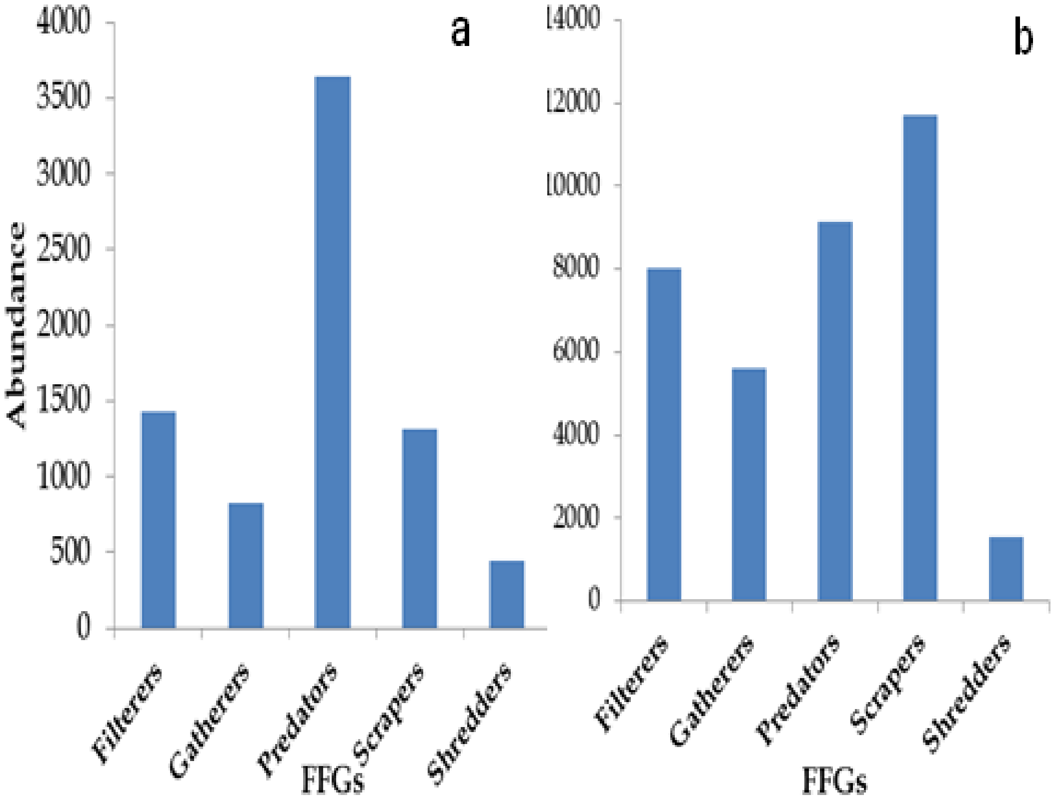


Figure 9: Macroinvertebrate functional feeding groups abundance along the Sosiani-Kipkaren River during the dry season (a) (February-March, 2019) and wet season (b) (July-August, 2018)

Seasonality strongly influenced the abundance of FFGs with higher numerical abundance during the wet season (2-way ANOVA, $F_{1,3}$= 9.08, $p < 0.05$). Shredder abundance (6.1% and 8.9%) was highest in forested sites while predators numerical abundance increased with increase in disturbance from the forested sites (9.7% and 8.9%) to urban sites (40.1% and 62.3%) during the wet and dry seasons respectively (Figure 10A). During the wet season, scrapers were more abundant in mixed (29.5%) and agricultural (30.5%) sites while during the dry season, they were more abundant in forested (26.8%) and mixed (22.9%) sites. During the wet season, collector-filterers were more abundant in forested (30.2%) and mixed (27.3%) sites, while during the dry season, they displayed a decreasing trend from forested (26.8%) to urban (11%) sites (Figure 10A).

Total macroinvertebrate biomass differed among the land-uses (One-way ANOVA, F_3= 1.74, $p < 0.05$), but did not significantly differ between seasons (One-way ANOVA, F_1= 2.69, $p = 0.11$). Shredder biomass was highest in forested sites (90.5% and 80.7%) and decreased gradually along the degradation gradient with the lowest biomass being recorded in the urban sites (23% and 28.8%) during wet and dry seasons respectively (Figure 10B). During the wet season, predators (32%) had the highest biomass in the agricultural sites while filterers (33.7%) were dominant at the urban sites. There was a gradual increase in the biomass of scrapers along the degradation gradient with the highest biomass recorded in urban sites (13.7% and 38.6%) and the least biomass recorded in forested sites (1.3% and 5.6%) during both wet and dry seasons respectively (Figure 10B).

Taxon richness was dominated by predators in all land-uses during both the wet and dry seasons (Figure 10C). During the wet season, filterers had the lowest taxon richness across all land-uses (Figure 10C).

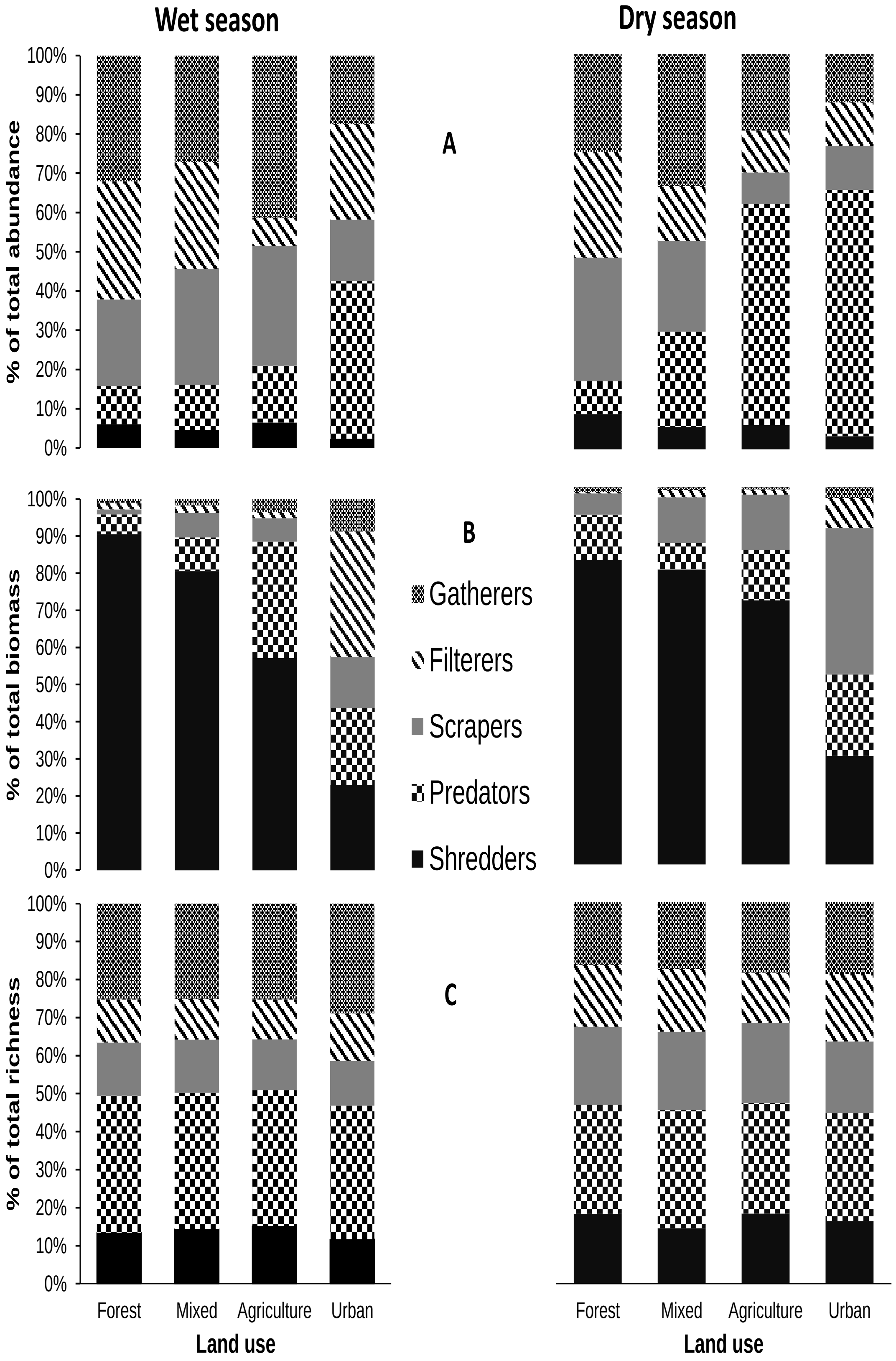


Figure 10: Percentage composition of macroinvertebrates FFGs in terms of total abundance (A), biomass (B), and taxon richness (C) in the different land-use sites during the wet and dry seasons

4.3 Relationships between water and habitat quality and macroinvertebrates assemblages

ANOSIM indicated significant differences in macroinvertebrate assemblages for un-transformed abundance data among land-uses (R-statistic = 0.24, $p < 0.004$), and between seasons (R-statistic = 0.30, $p < 0.007$). These findings suggest a stronger effect of “seasons” across land-uses as compared to “land-use” effect across seasons. Both the abundance and presence-absence data-based NMDS had good ordination with stress values < 0.2 (Figure 11a-d). Both abundance and presence-absence data of FFGs grouped land-uses similarly, although there were some overlaps. Sensitive taxa mainly belonging to EPT (Baetidae, Afronurus, Adenophlebia, Hydropsyche, Calamoceratidae and Oligoneuridae), Potamonautidae, Perlidae and Coleoptera clustered in the forested sites while the tolerant taxa of mainly Oligochaeta, Diptera, Odonata, Planariidae and Hirudinae clustered among the disturbed Urban and Agricultural sites (Figure 11).

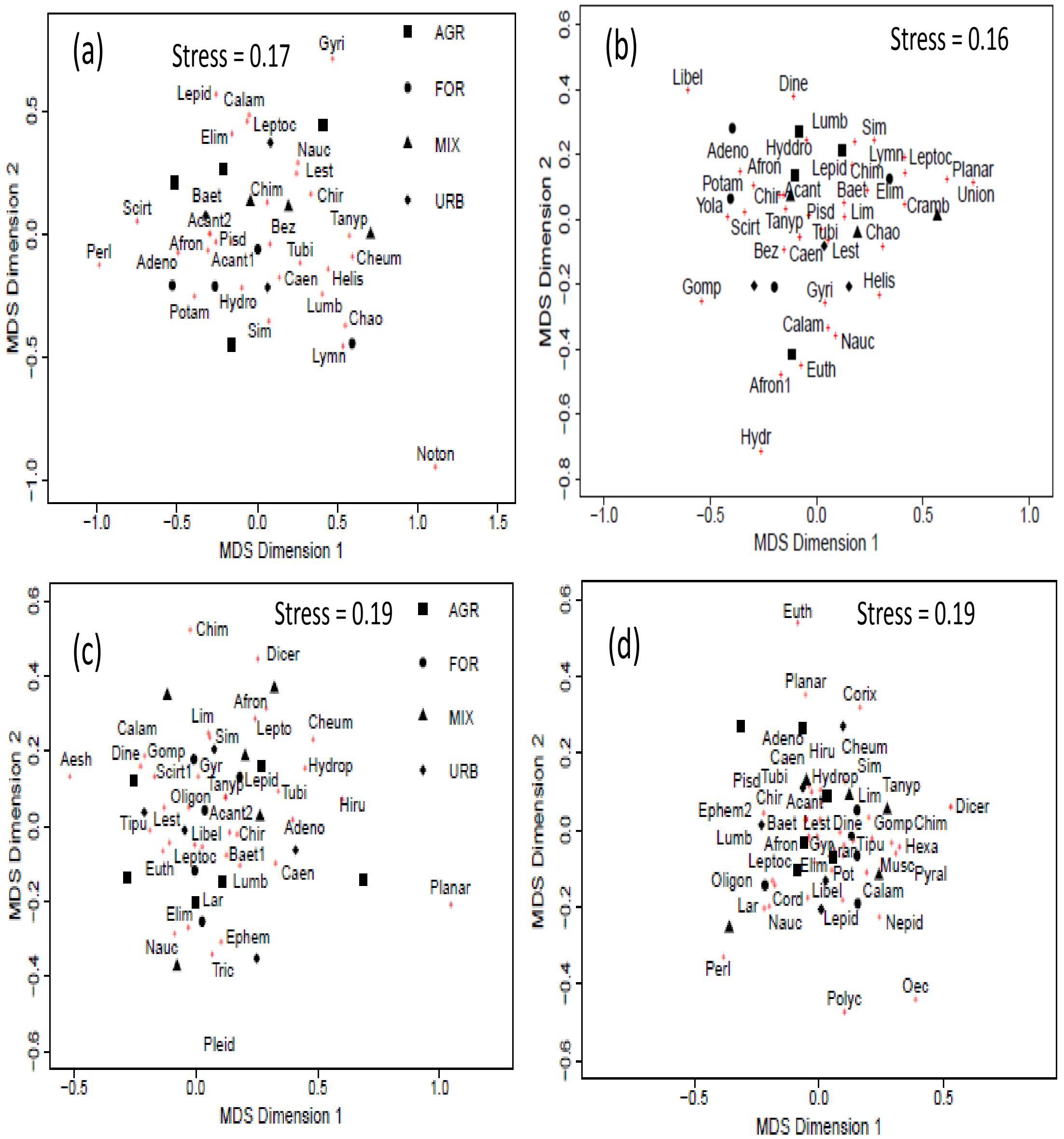


Figure 11: NMDS based on abundance (a, c) and presence-absence data (b, d) of the macroinvertebrates taxa during the wet season (a, b) and dry season (c, d). Lepid = Lepidostomatidae, Calam = Calamoceratidae, Elim = Elmidae, Leptoc = Leptoceridae, Nauc = Naucoridae, Lest = Lestidae, Baet = Baetidae, Scirt = Scirtidae, Acanth = Acanthiops, Perl = Perlidae, Adeno = Adenophlebia, Afron = Afronurus, Bez = Bezzia, Tany = Tanypodinae, Potam = Potamonautidae, Hydro = Hydropsychidae, Sim = Simuliidae, Lumb = Lumbriculidae, Chao = Chaoboridae, Lymn = Lymnaedae, Caen = Caenidae, Cheum = Cheumatopsyche, Musc = Muscidae, Tipu = Tipulidae, Aesh = Aeshnidae, Gomp = Gomphidae, Oligon = Oligoneuridae, Adeno = Adenophlebiidae, Dicer = Dicercomyzon, Nepid = Nepidae, Hiru = Hirudinae, Chim = Chimarra, Tubi = Tubifex, Gyr = Gyrinidae, Cord = Corduliidae, Pisd = Pisdium, Planar = Planariidae, Cramb = Crambidae

Abundance-based SIMPER’s pair-wise comparison of forested sites with agricultural sites during the wet season identified Simuliidae (16.3%) and Baetidae (16.2%) to contribute the greatest dissimilarity between forested and agricultural sites, with higher abundance in forested sites (Table 5). *Afronurus sp.* (11.9%) and Simuliidae (10.2%) contributed greatest dissimilarity between forested and mixed sites with Afronurus having higher abundance in the mixed sites while Simuliidae had higher abundance in the forested sites. *Baetis sp.* (14.7%), *Planaria sp.* (21.1%) and again Simuliidae (12.4%) accounted for greater dissimilarity between urban and forested sites (Table 5).

During the dry season Tanypodinae (30.0%) and Baetidae (12.0%) accounted for the dissimilarity between forested and agricultural sites with Tanypodinae having higher abundance in Agricultural sites while Baetidae being more abundant in Forested sites (Table 5). Pisidium (16.5%) and Baetidae (16.3%) contributed greatest dismiliarity between forest and mixed sites with both having higher abundance in the forested sites. Tanypodinae (25.7%) (Higher abundance in urban sites) and Baetidae (9.4%) (Higher abundance in forested sites) accounted for the dissimilarity between forested and urban sites (Table 5).

Unlike the abundance based-SIMPER, biomass-based SIMPER’s pair-wise comparison indicated that *Potamonautes sp.* (fresh water crabs) were responsible for the observed dissimilarity between forested and all the other three site categories with higher biomass in the forested sites (Table 6).

Table 5: Macroinvertebrates taxa-ranked abundance-based results of SIMPER analysis for mean abundance of macroinvertebrates

	Wet season				**Dry season**		
Taxon	**Mean Forest**	**Mean Agriculture**	**Contrib. %**	**Taxon**	**Mean Forest**	**Mean Agriculture**	**Contrib. %**
Simuliidae	198	4	16.3	Tanypodinae	4	262	30.0
Baetis sp.	400	242	16.2	*Baetis sp.*	99	30	12.0
Pisidium sp.	108	13	7.8	*Pisidium sp.*	90	65	8.6
Hydropsyche sp.	112	19	7.5	*Caenis sp.*	46	13	6.3
Chironomidae	92	44	6.7	Scirtidae	37	1	5.5
Tubificidae	88	11	6.4	Naucoridae	0	21	2.9
Afronurus sp.	91	17	6.0	*Neoperla sp.*	18	0	2.8
Planaria sp.	55	1	4.2	*Chironomus sp.*	4	22	2.3
Caenis sp.	54	32	2.7	*Afronurus sp.*	12	0	2.1
Potamonautes sp.	36	3	2.7	*Adenophlebia sp.*	16	1	2.1
Neoperla sp.	16	21	2.1	*Potamonautes sp.*	10	12	2.1
Leptophlebia sp.	28	0	2.0	*Hydropsyche sp.*	6	14	1.8
	Mean Forest	**Mean Mixed**	**Contrib. %**		**Mean Forest**	**Mean Mixed**	**Contrib. %**
Afronurus sp.	91	195	11.9	*Pisidium sp.*	90	36	16.5
Simuliidae	198	111	10.2	*Baetis sp.*	99	27	16.3
Chironomidae	92	110	7.7	*Caenis sp.*	46	4	7.9
Pisidium sp.	108	151	6.7	Scirtidae	37	1	7.2
Hydropsyche sp.	112	100	6.5	*Afronurus sp.*	12	37	6.0
Baetis sp.	400	274	12.6	Simuliidae	3	29	4.4
Tubificidae	88	16	5.0	Tanypodinae	4	27	4.0
Planaria sp.	55	17	3.8	*Chimarra sp.*	23	2	4.0
Potamonautes sp.	36	30	2.8	*Neoperla sp.*	18	0	3.9
Hirudinidae	23	40	2.7	Tubificidae	17	1	3.4
Tanypodinae	20	50	2.6	*Adenophlebia sp.*	16	4	2.7
Atyidae	0	51	2.1	*Hydropsyche sp.*	6	17	2.6

	Mean Forest	**Mean Urban**	**Contrib. %**		**Mean Forest**	**Mean Urban**	**Contrib. %**
Planaria sp.	55	1120	21.1	Tanypodinae	4	557	25.7
Baetis sp.	400	524	14.7	*Baetis sp.*	90	9	9.4
Simuliidae	198	219	12.4	*Caenis sp.*	46	50	8.0
Hydropsyche sp.	112	355	7.8	*Pisidium sp.*	90	47	6.9
Caenis sp.	54	284	6.1	*Chimarra sp.*	2	50	6.9
Chimarra sp.	2	126	5.1	*Afronurus sp.*	12	41	5.5
Chironomidae	92	132	4.4	Scirtidae	37	0	4.1
Hirudinidae	23	203	4.2	Chaoboridae	0	24	2.6
Pisidium sp.	108	37	3.2	*Helisoma sp.*	0	19	2.4
Tubificidae	88	26	3.0	Calamoceratidae	0	14	2.1
Afronurus sp.	91	131	3.0	*Neoperla sp.*	18	0	2.1
Cheumatopsyche sp.	13	108	2.4	Lestidae	1	19	2.0

Table 6: Macroinvertebrates taxa-ranked abundance-based results of SIMPER analysis for mean biomass of macroinvertebrates

Wet season				Dry season			
Taxon	**Mean Forest**	**Mean Agriculture**	**Contrib. %**	**Taxon**	**Mean Forest**	**Mean Agriculture**	**Contrib. %**
Potamonautes sp.	33.40	2.82	76.4	*Potamonautes sp.*	11.30	9.39	66.1
Limonia sp.	0.50	0.09	3.0	Libellullidae	0.96	0.00	6.6
Libellullidae	0.77	0.15	2.5	Naucoridae	0.00	0.44	3.3
Tipula sp.	0.17	0.32	2.0	Scirtidae	0.49	0.01	2.9
Belostomitadae	0.00	0.45	2.0	*Helisoma sp.*	0.00	0.42	2.3
Hexatoma sp.	0.19	0.02	1.7	*Neoperla sp.*	0.35	0.00	2.2
Neoperla sp.	0.31	0.42	1.7	Belostomitadae	0.00	0.48	2.1
Gomphidae	0.41	0.21	1.4	Tanypodinae	0.00	0.21	1.8
Hydropsyche sp.	0.34	0.06	1.2	Gomphidae	0.11	0.34	1.6
Naucoridae	0.00	0.25	1.0	Lymnaeidae	0.00	0.17	1.6
				Calamoceratidae	0.00	0.15	1.3
				Leptoceridae	0.20	0.11	1.3
				Hexatoma sp.	0.00	0.27	1.0
				Lestidae	0.01	0.15	1.0
	Mean Forest	**Mean Mixed**	**Contrib. %**		**Mean Forest**	**Mean Mixed**	**Contrib. %**
Potamonautes sp.	33.40	28.20	73.3	*Potamonautes sp.*	11.30	6.42	67.2
Gomphidae	0.41	2.04	6.2	Libellullidae	0.96	0.12	8.1
Planorbis sp.	0.00	1.62	4.2	Lymnaeidae	0.00	0.66	4.9
Libellullidae	0.77	0.20	2.0	Scirtidae	0.49	0.01	3.8
Limonia sp.	0.50	0.36	1.8	*Neoperla sp.*	0.35	0.00	3.0
Tipula sp.	0.17	0.38	1.2	Gomphidae	0.11	0.13	1.6
Hexatoma sp.	0.19	0.25	1.2	Leptoceridae	0.20	0.00	1.5
Hirudinidae	0.12	0.21	0.9	Lestidae	0.01	0.14	1.4
Hydropsyche sp.	0.34	0.30	0.9	*Caenis sp.*	0.18	0.01	1.3

	Mean Forest	**Mean Urban**	**Contrib. %**		**Mean Forest**	**Mean Urban**	**Contrib. %**
Potamonautes sp.	33.40	1.88	69.8	*Potamonautes sp.*	11.30	1.41	51.2
Chimarra sp.	0.02	1.30	3.7	Lymnaeidae	0.00	1.67	9.7
Caenis sp.	0.22	1.14	3.6	Libellullidae	0.96	0.00	6.8
Hydropsyche sp.	0.34	1.07	2.9	*Helisoma sp.*	0.00	0.82	4.9
Limonia sp.	0.50	0.08	2.3	Tanypodinae	0.00	0.45	3.4
Libellullidae	0.77	0.00	2.1	*Chimarra sp.*	0.02	0.51	3.3
Hexatoma sp.	0.19	0.03	1.4	Scirtidae	0.49	0.00	3.0
Gomphidae	0.41	0.20	1.2	Notonectidae	0.00	0.53	2.8
Tipula sp.	0.17	0.00	1.0	Calamoceratidae	0.00	0.37	2.5
				Neoperla sp.	0.35	0.00	2.3
				Caenis sp.	0.18	0.20	1.6
				Lestidae	0.01	0.24	1.5
				Leptoceridae	0.20	0.11	1.2
				Gomphidae	0.11	0.08	1.0

The wet season RDA indicated distinct spatial patterns in macroinvertebrate community composition associated with water quality variables (Figure 12a, b) and stream size, organic matter and habitat variables (Figure 12c, d). For the water quality, RDA axis 1 accounted for 21.4 % of the total variation while the second axis accounted for 17.7% (Figure 12a, b), while for the habitat and stream size RDA, Axis 1 accounted for 23.3-24.2% of the total variation while Axis 2 accounted for 16.2-16.7%. Both abundance and presence-absence data indicated that, increased DO levels, %CPOM, substrate quality, riparian and cover were associated with Potamonautidae, *Acanthiops sp.*, Calamoceratidae, Crambidae, Elmidae, *Euthraulus sp*. and Lepidostomatidae in the forested sites while higher temperature levels, conductivity, TSS, POM, discharge, sodium, potassium and nutrient levels was associated with *Cheumatopsyche sp.*, Libellulidae, Simuliidae, *Hydropsyche sp.*, Lumbriculidae and Planariidae in the urban and agricultural sites (Figure 12a-d).

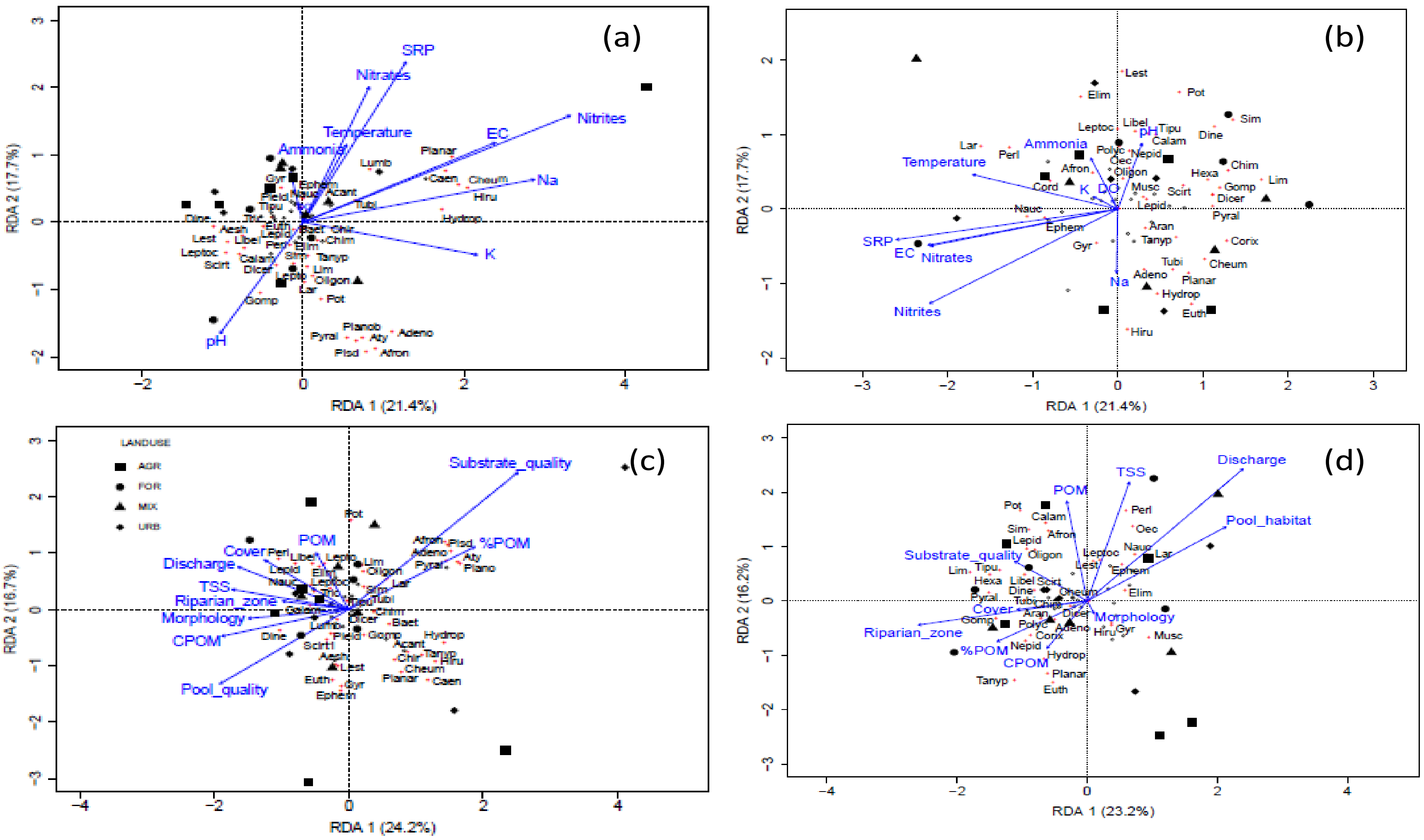


Figure 12: Redundancy analysis (RDA) triplot of macroinvertebrates structural composition based on abundance (a, c) and presence-absence (b, d) data in relation to water quality (a, b) and stream size, habitat quality variables (c, d) during the wet season in the Sosiani-

Kipkaren River. Lepid = Lepidostomatidae, Calam = Calamoceratidae, Elim = Elmidae, Leptoc = Leptoceridae, Nauc = Naucoridae, Lest = Lestidae, Baet = Baetidae, Scirt=Scirtidae, Acanth = Acanthiops, Perl = Perlidae, Adeno = Adenophlebia, Afron = Afronurus, Bez = Bezzia, Tany = Tanypodinae, Potam = Potamonautidae, Hydro = Hydropsychidae, Sim = Simuliidae, Lumb = Lumbriculidae, Chao = Chaoboridae, Lymn = Lymnaedae, Caen = Caenidae, Cheum = Cheumatopsyche, Musc = Muscidae, Tipu = Tipulidae, Aesh = Aeshnidae, Gomp = Gomphidae, Oligon = Oligoneuridae, Adeno = Adenophlebiidae, Dicer = Dicercomyzon, Nepid = Nepidae, Hiru = Hirudinae, Chim = Chimarra, Tubi = Tubifex, Gyr = Gyrinidae, Cord = Corduliidae, Pisd = Pisdium, Planar = Planariidae, Cramb = Crambidae, EC = electrical conductivity, DO = dissolved oxygen, SRP = soluble reactive phosphorus, K = potassium, Na = sodium, POM = particulate organic matter, CPOM = coarse particulate organic matter, TSS = Total suspended solids

RDA still indicated distinct spatial patterns in macroinvertebrate community composition associated with water quality variables (Figure 13a, b) and stream size, organic matter and habitat variables (Figure 13c, d) during the dry season. The RDA Axis 1 explained the most variance (% explained variance, range 21.5-32.9 %) while the second RDA Axis was responsible for 17.4-24.4% of the variation (Figure 13a-d). Just like the wet season, dry season data indicated that for both abundance and presence-absence data, increased DO levels, %CPOM, substrate quality, riparian and cover were associated with Potamonautidae, *Acanthiops sp.*, Calamoceratidae, Crambidae, Elmidae, *Euthraulus sp*. and Lepidostomatidae in the forested sites while higher temperature, conductivity, TSS, POM, discharge and nutrient levels were associated with *Cheumatopsyche sp.*, Libellulidae, Simuliidae, *Hydropsyche sp.*, Lumbriculidae and Planariidae in the urban and agricultural sites (Figure 13a-d).

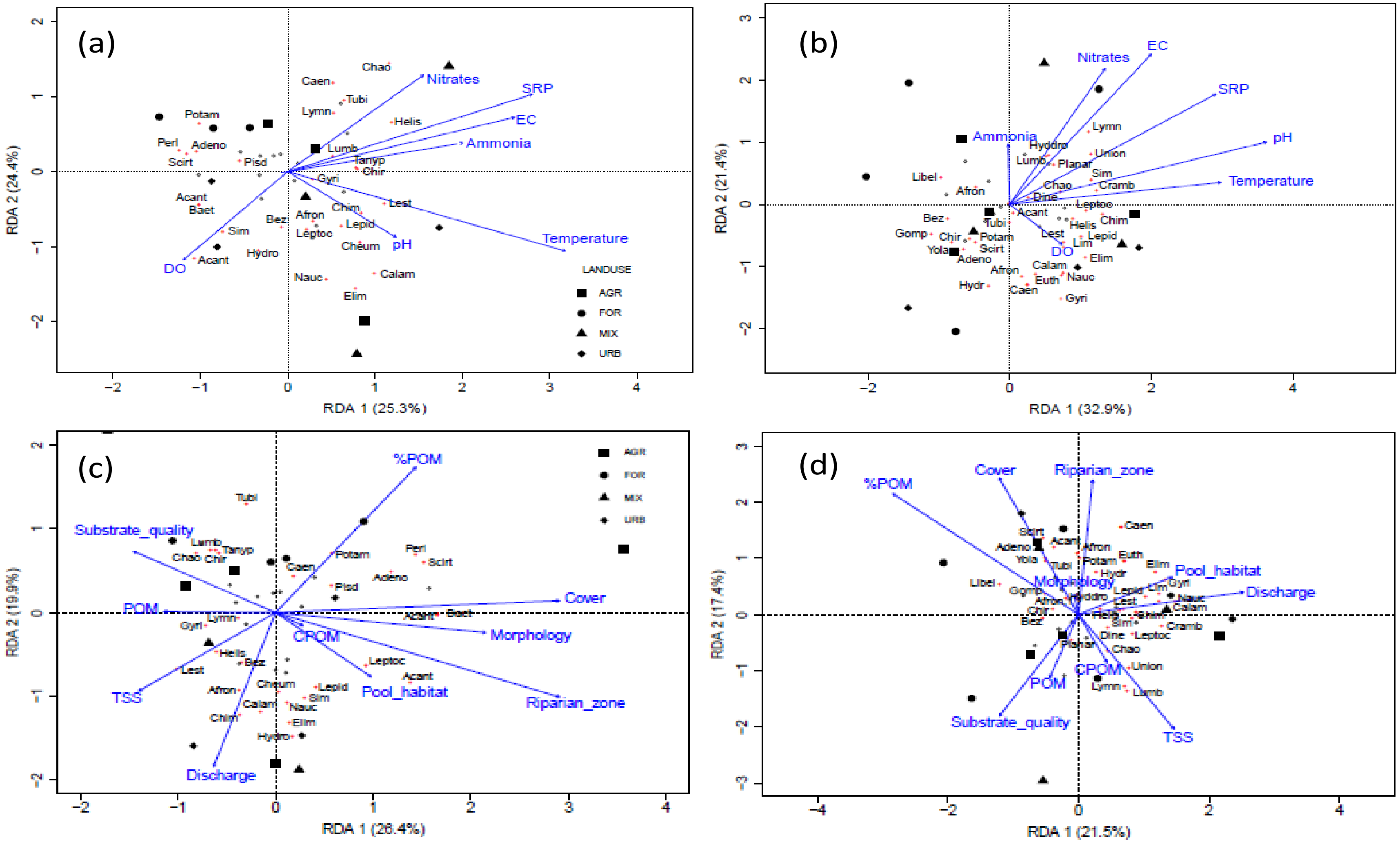


Figure 13: Redundancy analysis (RDA) triplot of macroinvertebrates structural composition based on abundance (a, c) and presence-absence (b, d) data in relation to water quality (a, b) and stream size, habitat quality variables (c, d) during the dry season in the Sosiani-

Kipkaren River. Lepid = Lepidostomatidae, Calam = Calamoceratidae, Elim = Elmidae, Leptoc = Leptoceridae, Nauc = Naucoridae, Lest = Lestidae, Baet = Baetidae, Scirt = Scirtidae, Acanth = Acanthiops, Perl = Perlidae, Adeno = Adenophlebia, Afron = Afronurus, Bez = Bezzia, Tany = Tanypodinae, Potam = Potamonautidae, Hydro = Hydropsychidae, Sim = Simuliidae, Lumb = Lumbriculidae, Chao = Chaoboridae, Lymn = Lymnaedae, Caen = Caenidae, Cheum = Cheumatopsyche, Musc = Muscidae, Tipu = Tipulidae, Aesh = Aeshnidae, Gomp = Gomphidae, Oligon = Oligoneuridae, Adeno = Adenophlebiidae, Dicer = Dicercomyzon, Nepid = Nepidae, Hiru = Hirudinae, Chim = Chimarra, Tubi = Tubifex, Gyr = Gyrinidae, Cord = Corduliidae, Pisd = Pisdium, Planar = Planariidae, Cramb = Crambidae, EC = electrical conductivity, DO = dissolved oxygen, SRP = soluble reactive phosphorus, POM = particulate organic matter, CPOM = coarse particulate organic matter, TSS = Total suspended solids

The macroinvertebrate FFGs ANOSIM indicated significant differences for un-transformed abundance data among land-uses (R-statistic = 0.14, $p < 0.028$), and between seasons (R-statistic = 0.66, $p < 0.001$). These findings suggest a stronger effect of “seasons” across land-uses as compared to “land-use” effect across seasons. Both the abundance and presence-absence data-based NMDS had good ordination with stress values < 0.2 (Figure 14). Both abundance and presence-absence data of FFGs grouped land-uses similarly, although there were some overlaps. Predators (Pred) and scrapers (Scr) clustered among agricultural sites (AGR) and Urban (URB) sites, collector-gatherers (Colg) clustered among mixed (MIX) sites, while shredders (Shr) clustered among forested (FOR) sites. Collector-filterers (Colf) were more associated with agricultural sites but overlapped across the different land-uses (Figure 14).

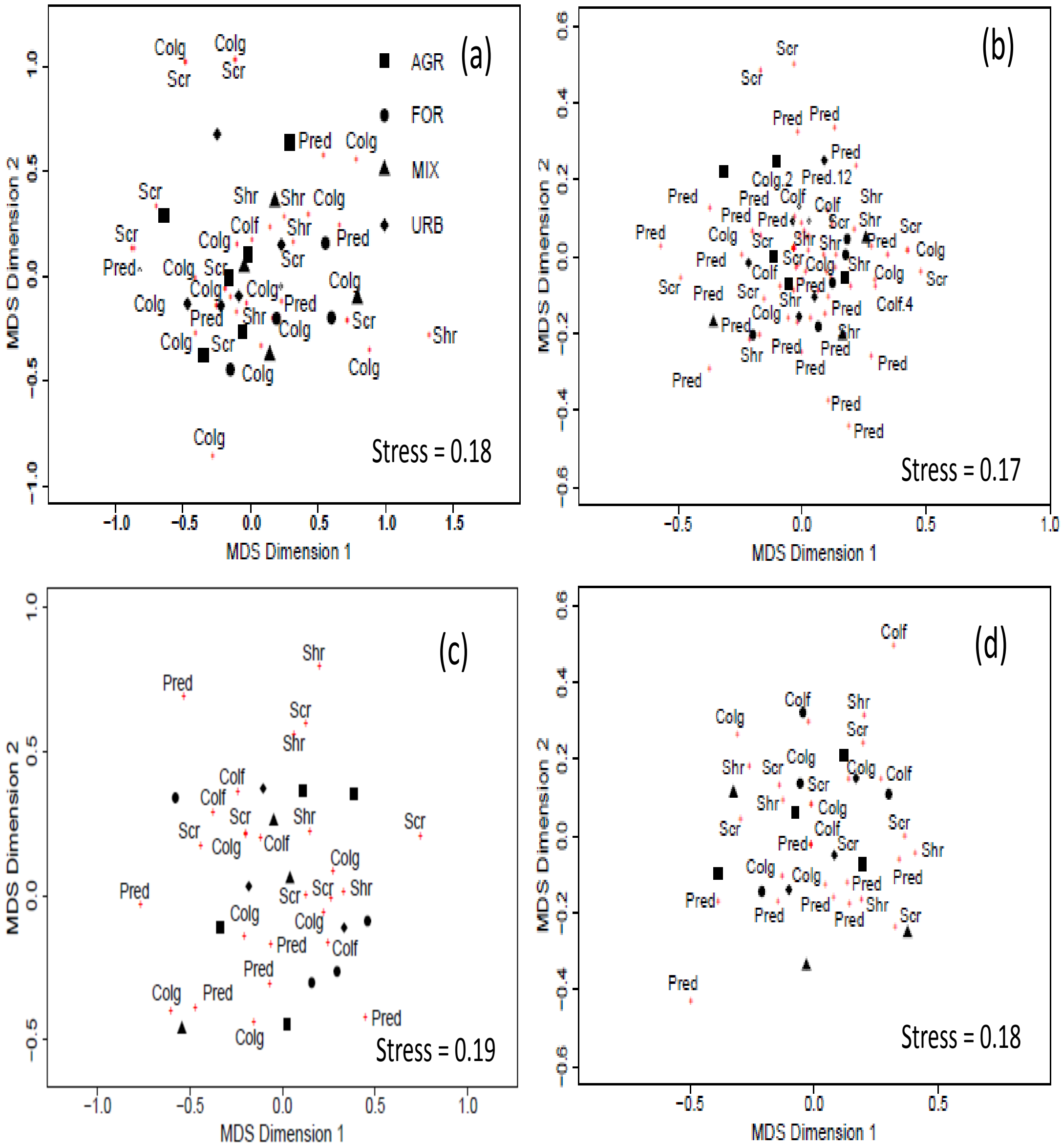


Figure 14: NMDS based on abundance (a, c) and presence-absence (b, d) data of the macroinvertebrate FFGs during the wet (a, b) and dry (c, d) seasons. Pred = predators, Shr = shredders, Scr = scrapers, Colf = collector-filterers, Colg = collector-gatherers

Abundance-based SIMPER's pair-wise comparison of forested sites with agricultural sites during the wet season identified collector-filterers (20.6 %) and collector-gatherers (17.2%) to contribute the greatest dissimilarity between forested and agricultural sites, with higher abundance in forest sites (Table 7). Collector-filterers (13.8 %) and collector-gatherers (14.7 %) still contributed the greatest dissimilarity between forested and mixed streams, with higher abundance in forest sites, while collector-filterers (20.7 %) and predators (19.1 %) contributing the greatest dissimilarity between forested and urban sites, with higher abundance in urban sites.

During the dry season, predators (25.0 %) and scrapers (16.6 %) accounted for the greatest dissimilarity between forested and agricultural sites, with higher abundance in the agricultural sites. Collector-filterers (19.7 %) and scrapers (18.7 %) were identified to contribute the dissimilarity between forested and mixed streams, with higher abundance in forested sites, while predators (29.8 %) and scrapers (17.0 %) contributed to the greatest dissimilarity between forested and urban sites, with higher abundance in urban sites (Table 7).

Unlike the abundance data, biomass-based SIMPER’s pair-wise comparison indicated that during the wet season, shredders and predators contributed greatest dissimilarity among all the land-uses. Shredder biomass differentiated among the land-uses with more than 70 % total contribution to dissimilarity with higher biomass in the forested land-use while predators accounted for 11-14 % with higher biomass in the agricultural, mixed and urban sites (Table 7).

For the dry season, differentiation of the forested sites from agricultural sites was by shredders (69.1 %) and predators (20.0 %), with higher biomass in the agricultural sites. Shredders (55.8 %) with higher biomass in the forested sites and scrapers (19.6 %) with higher biomass in the urban sites contributed greatest dissimilarity between forested and urban sites, while shredders (70.5 %) and predators (15.7 %) differentiated forested from mixed sites, with higher biomass in the forest sites (Table 7).

Table 7: FFGs-ranked abundance- and biomass-based SIMPER analysis for mean abundance and mean biomass of macroinvertebrate FFGs for all sites per land-use

	Wet Season				Dry Season		
Mean abundance							
FFGs	**Forest**	**Agriculture**	**% contribution**	**FFGs**	**Forest**	**Agriculture**	**% contribution**
Filterers	448	41	20.6	Predators	56	225	25
Gatherers	476	240	17.2	Scrapers	120	55	16.6
Scrapers	328	176	10.6	Gatherers	83	58	11.1
Predators	145	83	8.7	Filterers	82	92	10.4
Shredders	90	38	3.8	Shredders	25	30	4.7
	Forest	**Mixed**			**Forest**	**Mixed**	
Gatherers	476	412	14.7	Filterers	82	113	19.7
Filterers	448	416	13.8	Scrapers	120	50	18.7
Scrapers	328	449	12.7	Predators	56	67	15.5
Predators	145	175	7.5	Gatherers	83	37	13.7
Shredders	90	71	3	Shredders	25	18	4.5
	Forest	**Urban**			**Forest**	**Urban**	
Filterers	448	865	20.7	Predators	56	629	29.8
Predators	145	1419	19.1	Scrapers	120	112	17
Gatherers	476	615	13.4	Filterers	82	123	12.4
Scrapers	328	553	9.2	Gatherers	83	111	11.2
Shredders	90	84	2.7	Shredders	25	34	4.8
				Mean biomass			
	Forest	**Agriculture**			**Forest**	**Agriculture**	
Shredders	34.2	3.3	81.9	Shredders	9.7	11.6	69.1
Predators	2	1.8	12.9	Predators	1.5	1.9	20

Filterers	0.7	0.1	2.2	Scrapers	0.7	0.7	7.9
Gatherers	0.4	0.2	1.5	Filterers	0.04	0.2	1.7
Scrapers	0.5	0.4	1.4	Gatherers	0.2	0.1	1.3
	Forest	**Mixed**			**Forest**	**Mixed**	
Shredders	34.2	17.8	77.9	Shredders	9.7	6.6	70.5
Predators	2	2.8	13.8	Predators	1.5	0.6	15.7
Scrapers	0.5	1.5	4.1	Scrapers	0.7	0.8	11
Filterers	0.7	0.5	2.2	Gatherers	0.2	0.1	1.5
Gatherers	0.4	0.4	2	Filterers	0.04	0.2	1.3
	Forest	**Urban**			**Forest**	**Urban**	
Shredders	34.2	2.1	73.7	Shredders	9.7	2	55.8
Predators	1.9	2	10.8	Scrapers	0.7	2.7	19.6
Filterers	0.7	3.1	8.8	Predators	1.5	1.5	19.3
Scrapers	0.5	1.2	3.8	Filterers	0.04	0.6	3.6
Gatherers	0.4	0.8	2.8	Gatherers	0.2	0.2	1.7

Redundancy analysis indicated distinct spatial patterns in macroinvertebrate community composition associated with water quality variables (Figure 15) and stream size, organic matter and habitat variables (Figure 16) for both dry and wet seasons. During the wet season, RDA Axis 1 accounted for the greatest variance (% explained variance, range 20.5% - 28.8 %) in the data. Both presence-absence and abundance data showed similar trends to water quality and nutrients during both wet and dry seasons.

The first RDA axis (RDA 1) explained between 23.8% - 29.8% of the association for both abundance and presence/absence data. The second RDA axis (RDA 2) explained between 16.3% and 23.2% of the association. The RDA ordination showed similar associations of FFGs with specific land-uses as observed with the NMDS ordination for both wet and dry seasons. Nutrients (SRP and nitrates), higher temperature levels, electrical conductivity and TSS were associated with predators and scrapers in urban and agricultural sites while higher DO levels occurred in forested sites and were associated with shredders and collector-gatherers (Figure 15).

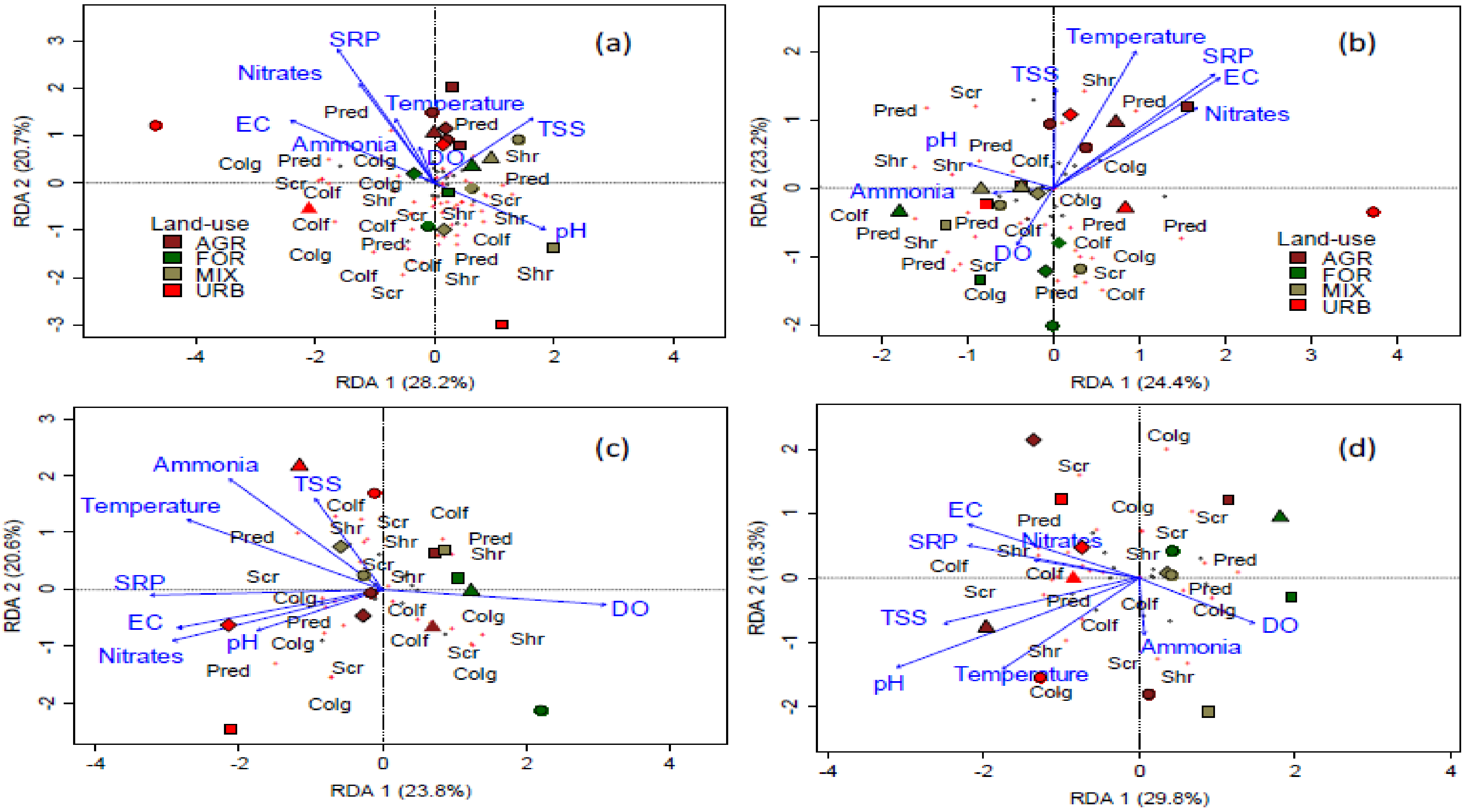


Figure 15: Redundancy analysis (RDA) triplot of macroinvertebrate FFGs based on abundance (a, c) and presence-absence (b, d) data in relation to water quality variables for the wet (a, b) and dry (c, d) seasons in Sosiani-Kipkaren River. Pred = predators, Shr = shredders, Scr

= scrapers, Colf = collector-filterers, Colg = collector-gatherers, EC = electrical conductivity, DO = dissolved oxygen, SRP = soluble reactive phosphorus, K = potassium, Na = sodium

For both the abundance and presence-absence data, there were clear associations of specific FFG taxa with hydraulic parameters and other physical variables (Figure 16). The association were explained by (22.6% - 26.1%) in axis 1 while axis 2 explained between (18.4% - 21.5%) of the association (Figure 16). Abundance and presence-absence data indicated that variables; instream cover, riparian zone and substrate quality were associated with RDA 1 (Figure 16). Substrate quality, cover, discharge, TSS, and riparian zone quality were the most important variables describing the variation of the macroinvertebrates FFGs in the various land-uses in both seasons. Increased amounts of CPOM and instream cover were positively associated with shredders at the forested and mixed land-use sites. Scrapers and collector-gatherers were mainly associated with agricultural sites and were negatively influenced by increased TSS and POM levels, but positively associated with increased conductivity and nutrient levels. Pool quality, stream depth and stream width were associated with predators at urban sites (Figure 16).

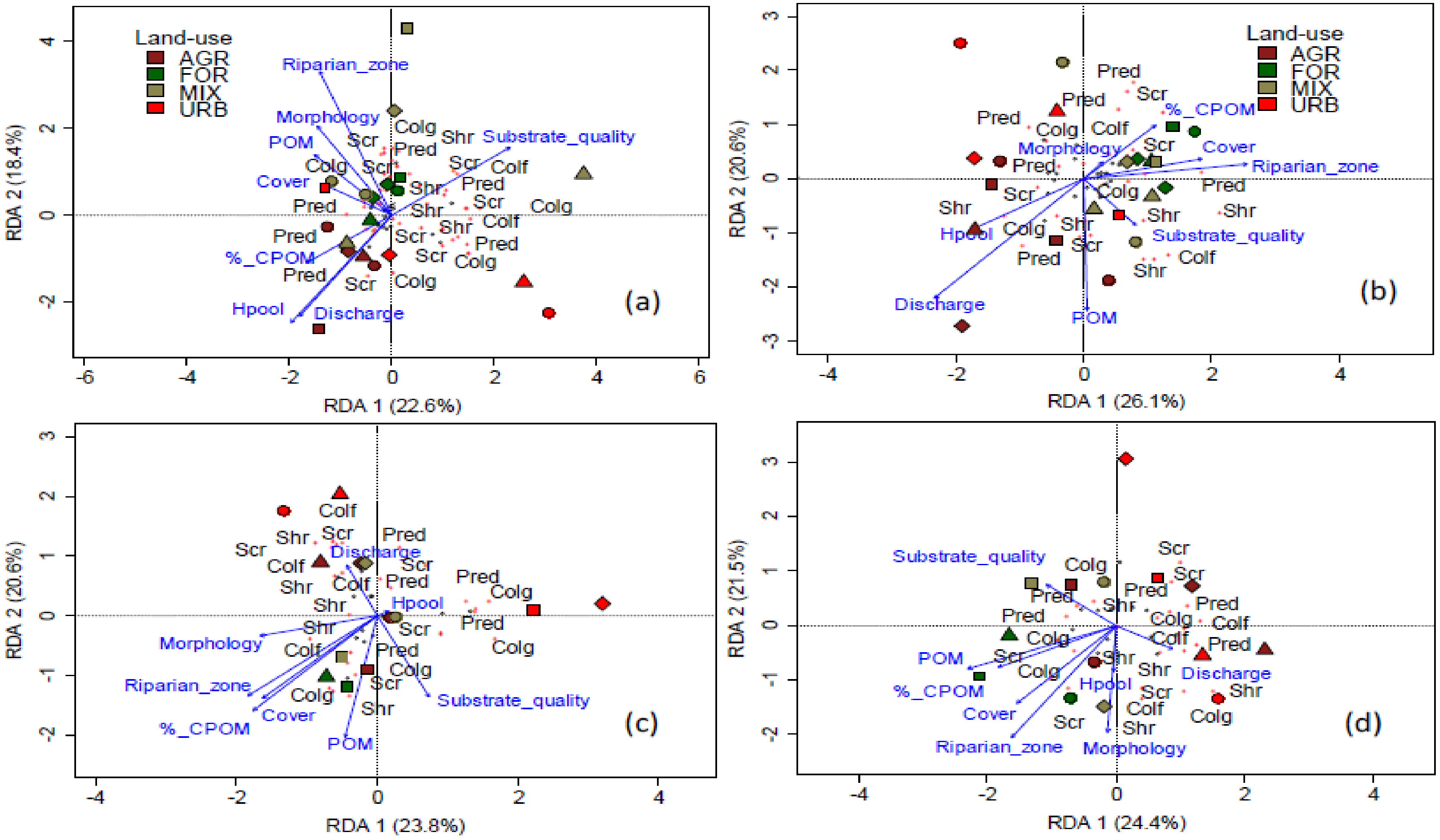


Figure 16: Redundancy analysis (RDA) triplot of macroinvertebrate FFGs based on abundance (a, c) and presence-absence (b, d) data in relation to stream size, habitat quality variables in wet (a, b) and dry (c, d) seasons in the Sosiani-Kipkaren River. Pred = predators, Shr =

shredders, Scr = scrapers, Colf = collector-filterers, Colg = collector-gatherer, POM = particulate organic matter, CPOM = coarse particulate organic matter, TSS = Total suspended solids

4.4 Abundance- vs. biomass-based ratios of macroinvertebrates FFGs as surrogates of ecosystem attributes

4.4.1 Spatial-temporal patterns

Land-use and stream size had strong influences on the ecosystem attributes (Table 8, Figure 17). Abundance- and biomass-based metrics displayed mixed outcomes for many of the ecosystem attributes assessed, with opposite results for the CPOM/FPOM index and agreement for channel stability and balance between FPOM in transport and FPOM deposited in the benthos. With the CPOM/FPOM index, biomass data showed a strong linkage between food webs in the river and the riparian zone (CPOM > FPOM) at all site categories and during both the dry and wet seasons. On the contrary, abundance-based metrics did show no such linkage (CPOM < FPOM, Table 8).

Seasonality played a major role in a lack of congruence between abundance- and biomass-based metrics of ecosystem functioning, with more disagreements during the wet than dry season. For instance, abundance and biomass data agreed that forested sites were heterotrophic (P < R), but differed on the rest of the sites with abundance data showing that they were autotrophic (P > R) while biomass data showing that they are heterotrophic (P < R).

During both seasons top-down control index derived from both abundance and biomass data showed that other than the forest sites, the rest of the sites had an overabundance of predators. Irrespective of site category, all sites had a stable channel and more fine particulate matter being transported from upstream than that being deposited in the benthos during both seasons as indicated by the TFPOM/BFPOM index (Table 8).

Table 8: Numerical abundance- and biomass-based derived mean values of stream ecosystem attributes in the different land-use categories along the Sosiani-Kipkaren River during the two seasons. *and bold face identify those values above the threshold values for that metric

	Abundance based attributes					**Biomass based attributes**				
Land-use	**P/R ratio**	**CPOM/ FPOM**	**Top-down control**	**Channel stability**	**TFPOM/ BFPOM**	**P/R ratio**	**CPOM/ FPOM**	**Top-down control**	**Channel stability**	**TFPOM/ BFPOM**
Wet season										
Forest	0.67	0.14	0.12	**3.00***	**2.12***	0.02	**38.40***	0.12	0.06	**2.34***
Mixed	**0.79***	0.18	**0.28***	**2.36***	**1.20***	0.46	**42.39***	**0.66***	**0.62***	**1.26***
Agriculture	**1.88***	0.22	0.17	**2.93***	0.43	0.26	**10.13***	**1.19***	0.35	**0.69***
Urban	**0.95***	0.2	**0.82***	**5.30***	**5.15***	0.33	**0.81***	**0.28***	**1.76***	**15.73***
Dry season										
Forest	0.41	0.23	0.10	**4.63***	**4.12***	0.06	**70.53***	0.16	0.07	**0.66***
Mixed	0.74	0.17	**0.59***	**4.14***	**4.57***	0.29	**99.20***	**1.12***	0.43	**1.73***
Agriculture	0.30	0.24	**1.22***	**1.85***	**2.26***	0.37	**45.32***	**0.44***	**0.85***	**1.83***
Urban	**0.86***	0.20	**1.93***	**2.26***	**4.29***	**4.67***	**3.62***	**0.45***	**5.94***	**14.30***

[#] Threshold values for the attributes are: P/R: >0.75, CPOM/FPOM: >0.25, Top/Down Control: >0.20, TFPOM/BFPOM: >0.50 and stable channel: >0.50 (Cummins *et al.*, 2005).

There were strong seasonal differences in performance of abundance- vs. biomass-based metrics of ecosystem functioning (Figure 17). Stream size influence on the metrics varied between the abundance- and biomass-based metrics. The P/R and CPOM/FPOM ratios were better explained with the biomass-based metrics while top-down control, channel stability and TFPOM/BFPOM ratios were better explained by the abundance-based metrics (Figure 17).

Unexpectedly, abundance-based P/R metric indicated that during the dry season in the mid reaches, the system was heterotrophic. Similarly, with the CPOM/FPOM metric, the abundance-based data indicated poor linkage between food webs in the river and the riparian zone in forested sites during the wet season. Biomass-based channel stability metric showed that forest sites were not stable during both the wet and dry seasons (Figure 17).

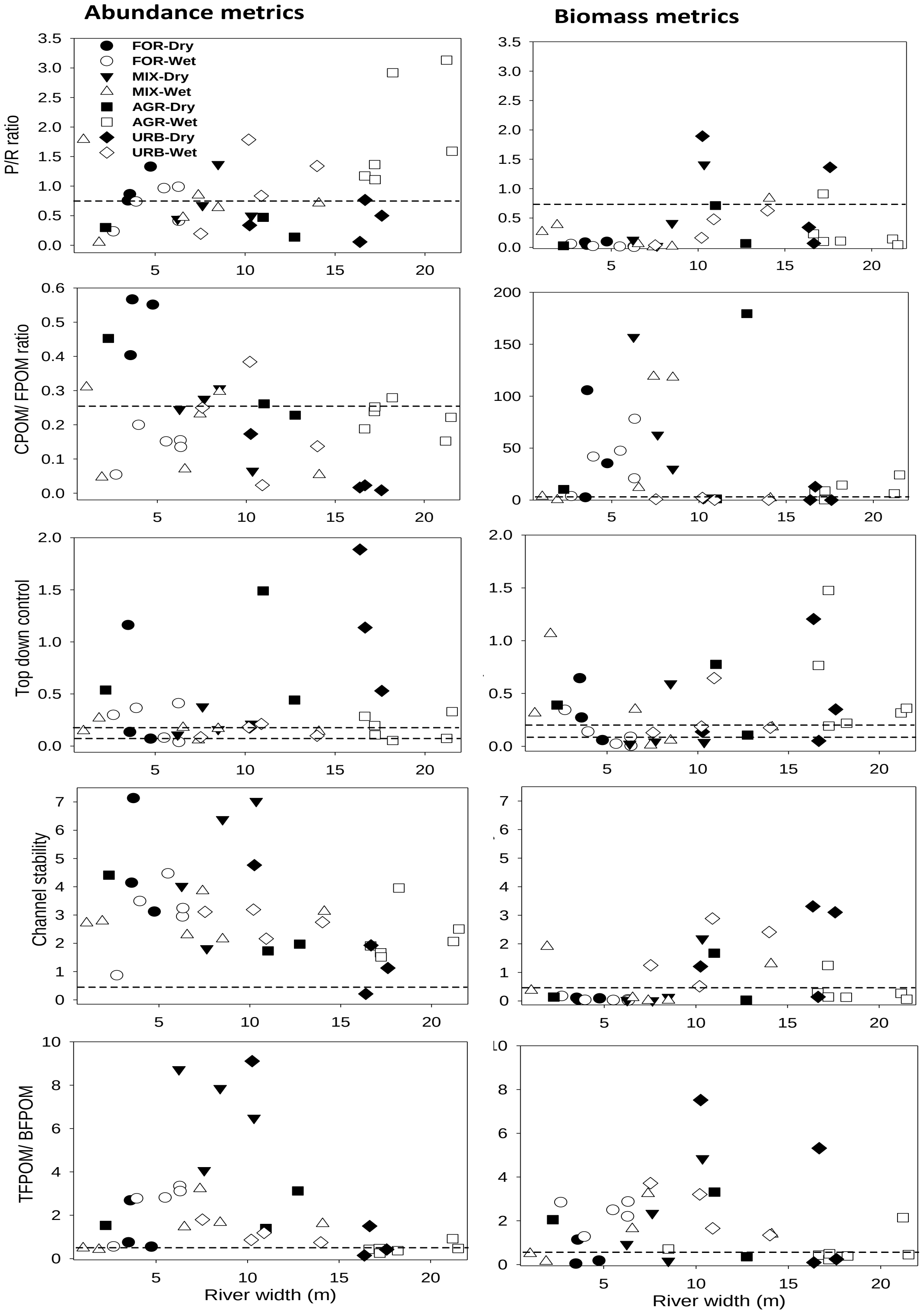


Figure 17: Performance of abundance- vs. biomass-based metrics of ecosystem function in the Sosiani-Kipkaren River as determined by different ratios of macroinvertebrate functional feeding groups. The dashed line show thresholds indicating a change in ecosystem attributes for each metric as shown in Table 8. Note the difference in the y-axis scale for CPOM/ FPOM metric between the abundance and biomass metrics because of the high biomass of freshwater crabs

4.4.2 Longitudinal patterns

To investigate longitudinal changes in ecosystem functioning with increasing stream size, relationships were explored using GAMS between FFG metrics of ecosystem attributes and stream/ river width with a smoother for land-use interacting with stream width (Figure 18). Abundance and biomass data were plotted separately during the dry and wet seasons. There were both seasonal differences in the responses of abundance- and biomass-based attributes of ecosystem functioning with relationships clearer during the dry season than during the wet season (Figure 18).

However, there were no significant interactions between the land-use smoother and stream width (data not shown), indicating that land-use did not influence longitudinal patterns in the FFGs metrics of ecosystem functioning. Abundance-based attributes performed better than biomass-based attributes with CPOM/FPOM and channel stability showing significant decrease with increasing stream size during the low flows (Figure 18b, c).

During the rainy season abundance-based P/R ratio significantly increased with streams size, while TFPOM/BFPOM decreased with as increase in stream size (Figure 18e, h). In comparison, biomass-based attributes showed significant longitudinal relationships with stream size only during the dry season, with P/R ratio and channel stability increasing with an increase in stream size (Figure 18i, k) while CPOM/FPOM ratio showed an opposite relationship with increase in stream size.

There was incongruence in abundance- and biomass-based attributes of ecosystem functioning for most of the attributes that showed significant or marginally significant ($p < 0.1$) relationships with stream size/ width. There were disagreements in the P/R ratio and channel stability with abundance-based showing a decrease with stream size, while biomass-based attributes showing an increase with stream size. The ratio of FPOM in transport to FPOM deposited in the benthos (TFPOM/BFPOM) showed a consistent non-linear relationship with higher amount of FPOM in the benthos in both low and high order streams and lower amount in mid-sized streams.

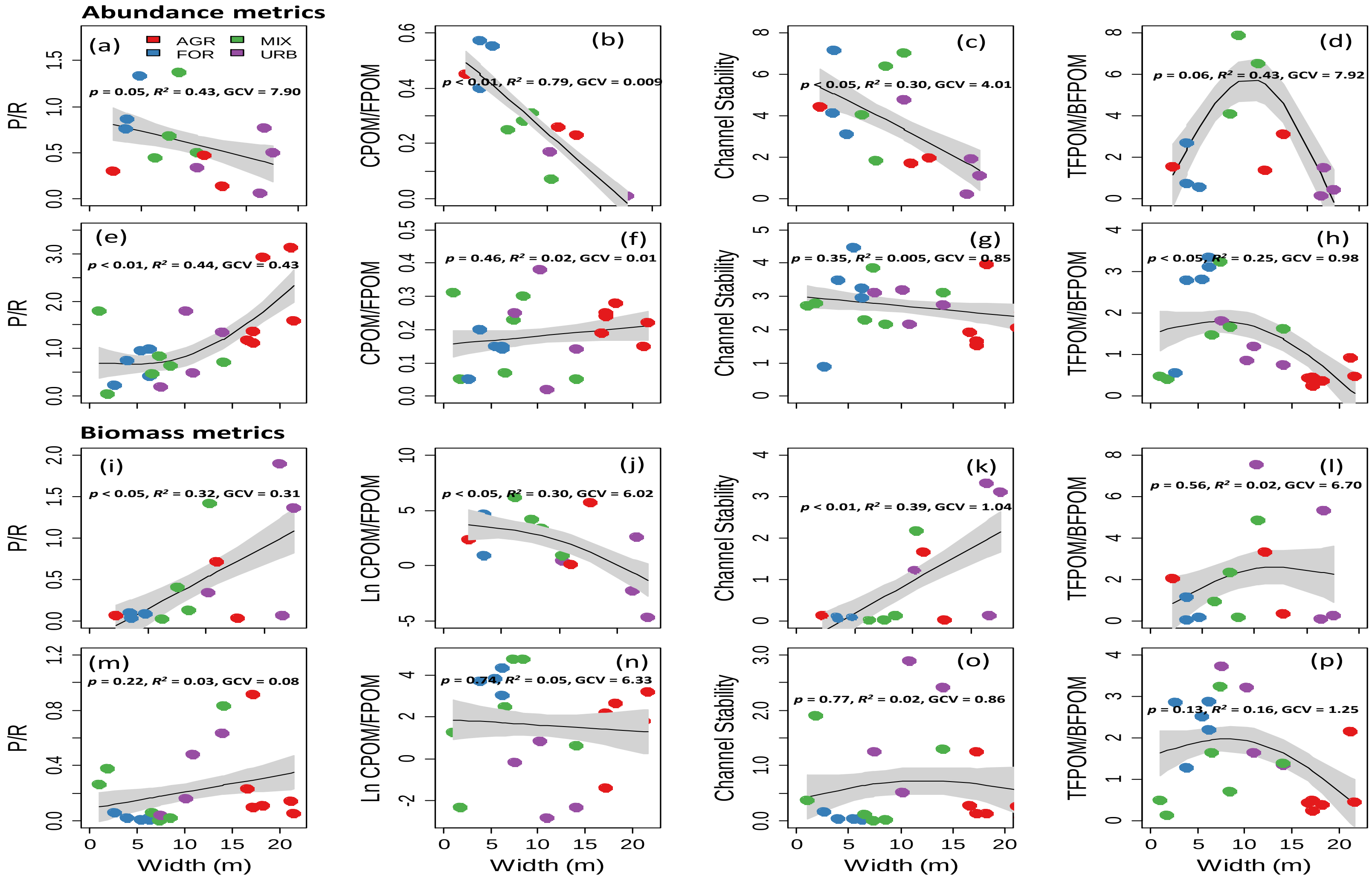


Figure 18: Longitudinal variability in ecosystem attributes derived from the composition of FFG along the Sosiani-Kipkaren River during the dry (a-d and i-l; n =14) and wet (e-h and m-p; n =21) seasons. The black line with grey shaded area represents smoother mean and s.e.; smoother significance, R^2 and GCV are supplied in the figures. Note the changes in y-axis values across graphs. Ln CPOM/FPOM for biomass data are as a result of the high biomass values of freshwater crabs among shredders

CHAPTER FIVE

DISCUSSION

The Sosiani-Kipkaren River and its tributaries serve as critical water sources for various uses and users. However, a number of anthropogenic activities along the river, such as bathing, washing clothes, water collection for domestic purposes (using motorbikes, hand carts and women and children carrying on their backs and heads), swimming, irrigation, sand harvesting, washing motorbikes and vehicles, and discharge of waste water from urban centres (Eldoret, Turbo and Kipkaren) pose threats to water quality. This study shows that the five major macroinvertebrate FFGs identified (scrapers, shredders, predators, filterers and gatherers) displayed both spatial and temporal variability in abundance, biomass and taxon richness in response to land-use driven changes. Seasonality amplified the land-use effects on water quality, with low dissolved oxygen and increased concentrations of nutrients and higher electrical conductivity associated with agricultural and urban sites.

Macroinvertebrates responded similarly with higher shredder abundance, biomass and taxon richness being recorded in forested sites, irrespective of the season, while predators and scrapers increased in agricultural and urban sites where water temperature and nutrient concentrations were higher. Biomass of FFGs responded more strongly to changes in land-use type than abundance. For shredders, decrease in biomass along the land-use gradient was attributed to the presence of macroconsumers (freshwater crabs) which are very sensitive to deforestation and deterioration of habitat and water quality (Lancaster *et al*., 2008).

Both abundance- and biomass-based metrics displayed responses to land-use and stream size, as hypothesized, although there was incongruence in how some metrics evaluated ecological condition and ecosystem functioning. Most notably, biomass-based metrics were poor predictors of ecological condition while abundance-based metrics were a better approximation of assessments based on physical measures of water and habitat quality. Nevertheless, the functional metrics used offered insights into the ecological condition and functioning of the river.

5.1 Variation of Water and Habitat Quality

Both season and land-use played significant roles in influencing water quality, organic matter and habitat variables in the study. Changes in water quality across the land-use sites were indicated by decreasing DO levels, increasing temperature, pH, conductivity, nutrients, cations and TSS levels (Table 3). The higher mean temperature in the agricultural and urban streams can be attributed to open canopy cover along the riparian zones, while the lower mean temperature at forested sites were due to dense vegetation cover. Vegetation cover on river margins limit solar radiation reaching the water thus reducing fluctuations in water temperature in forested streams (Mathooko & Kariuki, 2000; Aura *et al*., 2011; López-Carr & Burgdorfer, 2013; Masese *et al*., 2017).

The lower conductivity recorded in forested sites than other site categories during both seasons can be attributed to the fact that undisturbed catchments are characterized by very low in-stream ionic concentrations. The higher conductivity recorded in agricultural and urban streams during the wet season can be attributed to runoff from farmlands and urban areas (Minaya *et al*., 2013; Mwaijengo *et al*., 2020). The study contrasted with similar works around the tropics (Rodrigues *et al*., 2018;

Mwaijengo *et al*., 2020) with lower conductivity being recorded during the wet than dry season which can be attributed to increased livestock using these streams as watering points during the dry season. But these watering do not comprise to a greater extend these streams to be termed as impaired.

The higher levels of nitrates in agricultural sites could be attributed to nitrogenous fertilizers used in farmlands used for maize and wheat production. Increase in dissolved fractions of nitrogen, sodium, potassium and conductivity are indicators of disturbance that have been attributed to change in land-use type from forestry to agriculture in the region (Minaya *et al*., 2013; Jacobs *et al*., 2017). In urban sites the high levels of nitrates and ammonia are attributed to runoff and leakages from sewerage facilities and waste disposal from agro-industrial activities (Osano *et al*., 2003; Aura *et al*., 2011).

Habitat quality and organic matter characteristics had significant differences among land-uses, without a significant interaction with seasonality. Forested sites had good riparian zone quality, instream cover and a higher percentage coverage and biomass of CPOM whose quality declined among land-uses. This can be ascribed to the fact that forest streams are less interfered with, therefore still containing in-stream vegetation with intact riparian zones. It was evident from the study that water and habitat quality worsened with change in land-use type from forestry to urbanization and agricultural land-use.

5.2 Patterns in structural composition of macroinvertebrates

The study identified Ephemeroptera, Diptera and Trichoptera (EPT) as the dominant taxa along the river. Observations that were similar to earlier studies of Masese *et al.*

(2009) and Aura *et al*. (2011) while on working in the same catchment. However, there were differences in macroinvertebrate composition and distribution among land-use site categories, which can be explained by different macroinvertebrate taxa's tolerance levels to adverse environmental conditions.

The total abundance of macroinvertebrates was much larger during the wet than the dry season. These findings corroborate those of Harrison & Hynes (1988) and Masese *et al*. (2009) who reported more taxa in the wet season than in the dry one. However, the results contradict other similar studies within the region (Tumwesigye *et al*., 2000; Arimoro *et al*., 2012) that have found an increase in taxa during the dry than the wet season. The high abundance during the wet season can be attributed to two main reasons; first, more sites were sampled during the wet season than the dry there by bringing a disparity in effort and secondly flow differences caused by seasonality (Table 3).

During both the wet and dry seasons EPT taxa dominated (percent abundance and richness) the forest sites and decreased in the urban and agricultural sites, a pattern that was exhibited by the Coleopterans too, while Dipterans were the dominant taxa in the agriculture and urban sites (Figure 8). Similar results have been reported by earlier studies; Raburu *et al*. (2009), Masese *et al*. (2009) and M'erimba *et al*. (2014). The agricultural and urban sites also recorded a higher abundance of other macroinvertebrates taxa including the Odonata, Mollusca and Tricladida (Figure 8). In the Lake Victoria Basin, these taxa are thought to be among the most tolerant to organic pollution (Kobingi *et al*., 2009; Masese *et al*., 2009).

The lower numbers of EPT taxa in the urban and agricultural land-use sites coincided with water quality degradation, EPT taxa can therefore be referred to as good indicators of water quality change (Figure 8). Several studies have recognized the significant correlation between land-use and macroinvertebrate communities, indicating that the total number of taxa and the percentage of groups like Ephemeroptera, Plecoptera, Trichoptera and Coleoptera decreasing while those Oligochaeta and Diptera increasing as the pollution and alterations in the river quality increases (Masese *et al*., 2009; Hussain & Pandit, 2012; Masese *et al*., 2014). The urban and agricultural land-use had high nutrient levels, TSS, POM and temperature (Table 3). Several studies (such as; Hawkins & Vinson, 2000; Hyslop & Brown, 2012) have indicated that high nutrient enrichment and sedimentation are known to favor Dipterans, primarily Chironomids and Oligochaetes, at the expense of snails, algae piercing Trichoptera, Ephemeroptera, and Plecoptera.

Forested land-use sites had the highest richness and diversity (Table 4) during both seasons. The higher diversity of macroinvertebrates in these areas compared to agricultural and urban sites can attributed to habitat diversity and complexity in these sites given that these sites are areas with little or no human disturbance with some having intact riparian zones. Streams with minimally disturbed riparian vegetation have been reported to shed leaves and large wood to the streams that increase habitat complexity and produce habitats that favour increased abundance and diversity of macroinvertebrates (Mathooko, 2001; Kaufmann & Faustini, 2012; Anyona *et al*., 2014).

5.3 Patterns in functional composition of macroinvertebrates

Shredder biomass and abundance dominated in forested sites compared to sites in the other land-uses. This observation is in agreement with earlier studies in the tropical region, including Dobson *et al*. (2002), Cheshire *et al*. (2005), Uwadiae, (2009) and

Masese *et al.* (2014a). Shredders (most of them being Trichoptera) and scrapers (mostly Ephemeroptera) are more sensitive to environmental changes, while collectors and predators are more tolerant to disturbance and organic pollution (Boyero *et al.*, 2009; Masese *et al.*, 2014a; Masese & Raburu, 2017). Abundance based SIMPER identified collectors (filterers and gathers) during the wet season and predators during the dry season in terms of numerical abundance as the major FFGs contributing to the dissimilarity among the land-uses. Shredders and predators co-dominated the biomass and contributed to differences among the land-uses. There was dominance in terms of richness and abundance of predators in urban streams, which can be attributed to tolerant taxa, among Odonata and Hemiptera (Masese *et al.*, 2021). The availability of other tolerant prey taxa such as Oligochaeta allowed these taxa to dominate in urban sites (Barbee, 2005).

Seasonality was a major driver of functional organization of macroinvertebrates in the study area. There was high abundance, biomass and richness of FFGs during the wet season than the dry season, which can be attributed to the increase in habitats as marginal vegetation are flooded and a broad diversity of flow velocities are available for the flow velocities are maintained for both rheophilic taxa and pool taxa (Dallas, 2007; Munoz-Mas, 2019; Masese *et al.*, 2021). Food resources are also abundant and diverse during the wet season from run-off from terrestrial sources (Masese *et al.*, 2009a). For example, the study indicated the lowest DO concentration but with highest nutrient levels and electrical conductivity during the dry season, indicative of point sources of pollution, which in urban areas emanates from wastewater treatment facilities and outfalls from agro-industrial facilities (Walsh *et al.*, 2005).

There were distinct spatial patterns in macroinvertebrate community composition associated with water quality variables and stream size characteristics, organic matter and habitat variables for both dry and wet seasons (Figures 10). For instance, abundance and richness of predators dominated the dry season whose proportion increased with increase in disturbance from the forested sites to the degraded urban sites and were positively correlated with increased levels of cations, nutrients and electrical conductivity. Anthropogenic activities influence on the riparian land of urban sites created conditions that could only accommodate a narrow range of macroinvertebrates (mainly tolerant FFG taxa) that can withstand and are opportunistic in the degraded water and habitat conditions.

Shredders were the least abundant and diverse but had the highest biomass that was >80 % of all taxa in forested and mixed sites. Earlier studies by Pearson *et al*. (1989), Cheshire *et al*. (2005) and Camacho *et al*. (2009) indicated that Shredder composition in the tropical streams and rivers fluctuate over time and space. The low numbers of shredders in agricultural and urban sites can be attributed to the deforestation and clearance of indigenous riparian vegetation, water pollution and habitat disturbance caused by farming activities and urbanization in the region (Masese *et al*., 2009; Raburu *et al*., 2009; Aura *et al*., 2011). Furthermore, exotic tree species dominates the remaining riparian vegetation (mainly Eucalyptus, Cypress and Pine cover). The inadequate quality and quantity of leaf litter input may jointly cause the scarcity of shredders (Cummins *et al*., 1989; Tiegs & Peter, 2008; Jiang *et al*., 2011; Masese *et al*., 2014a).

The longitudinal distribution of FFGs at Sosiani-Kipkaren River varied widely among abundance, biomass and taxon richness of the various FFGs, and did not meet the expectations of the River Continuum Concept (RCC) (Vannote *et al.*, 1980). While the abundance data showed a mixed trend in the distribution of shredders from upstream sites (forested) to downstream sites (agricultural and urban), biomass data had a clear distribution that noticeably matched the RCC predictions. The richness-based metric did not show any systematic longitudinal patterns and was less responsive to change in land-use type. This can be attributed to the replacement of intolerant by tolerant taxa across FFGs (Masese *et al.*, 2021). Abundance data indicated that collector-gatherers and collector-filterers co-dominated the upper reaches while predators dominated the mid and lower reaches. The biomass-based metric showed a clear trend in the distribution of shredders conforming to the RCC prediction.

Although in low numbers, shredders numerical abundance was highest at the upper forested reaches and decreased from upstream to downstream sites. The biomass of total collectors (filtering and gathering collectors) and scrapers increased from upstream to downstream, results that conform to similar studies within the region by Dobson *et al*. (2002) and Masese *et al*. (2014a). The predominance of scrapers and collectors throughout the river has been reported in other studies in tropical streams (Tomanova *et al.*, 2006; Jiang *et al.*, 2011). The shredder biomass which differed significantly ($p < 0.05$) from the other functional feeding groups was contributed significantly by crabs of the genus *Potamonautes*, Tipulids, and Trichopterans (*Pisulia* sp., *Triaenodes* sp., *Adicella* sp., and *Lepidostoma* sp.) which are large-bodied and have been reported to be highly abundant in East African streams (Dobson *et al*., 2002; Masese *et al*., 2014a).

5.4 Abundance- vs. biomass-based ratios of macroinvertebrate FFGs as surrogates of ecosystem attributes

The spatio-temporal dynamics of macroinvertebrate FFGs in the river in response to disturbance rendered them quite amenable as surrogates of ecosystem functioning. The various ratios of FFGs (metrics) used were able to track shifts in ecosystem integrity and functioning as a result of changes in seasonality and land-use/ disturbance. However, abundance- and biomass-based indices did not quite agree in all attributes and periods (wet and dry seasons), but there were interesting agreements in others, with land-use change and stream size having strong influences on the ecosystem attributes (Table 8).

Abundance- and biomass-based metrics displayed opposite results for the P/R index, CPOM/FPOM index and channel stability, but near perfect agreement for top-down control and the balance between FPOM in transport and FPOM deposited in the benthos (Table 8). Abundance data indicated that while the forested sites were heterotrophic (P/R < 0.75), sites in the all other land-uses were autotrophic; an observation that conforms to earlier studies in Kenyan streams (Masese *et al*., 2014, 2017). On the contrary, biomass data indicated that all sites were heterotrophic, which is an indication of poor sensitivity of this metric to changes in stream size or land-use. Surprisingly, the river was more heterotrophic during the dry than the wet season. This is contrary to expectations because the river was more turbid (higher TSS concentrations) during the wet season, which, in addition to cloud cover and scouring, would reduce primary production and turn the river heterotrophic (Griffiths *et al*., 2013; Masese *et al*., 2017). Thus, autotrophy at most of the sites during the wet season, and only at urban sites during the dry season, is a significant departure from expectations based on other measures of ecosystem functioning.

Another instance of poor performance of the biomass-based metric was on the CPOM/ FPOM index, whereby the abundance-based results captured the removal of riparian vegetation along urban and agricultural streams, but the biomass-based index showed a strong linkage between food webs and the riparian zone (CPOM > FPOM) at all site categories and seasons. This poor performance of biomass-based metrics can be attributed to the presence of large-bodied shredders, especially *Potamonautes* sp. and Tipulids, whose presence, even in small numbers, can disproportionately shift the P/R ratio toward greater heterotrophy and the CPOM/FPOM ratio metric to identify sites as having a well-protected and functioning riparian zone, when in essence they are not. These findings highlight concerns regarding potential bias in biomass-based ecosystem functioning metrics when large-bodied macroconsumers like freshwater crabs are present. Although macroconsumers, such as crabs, crayfish and shrimps are often classified as shredders and play major roles in organic matter processing in tropical streams (Crowl *et al*., 2001; Schofield *et al*., 2001; Masese *et al*., 2014), they are also omnivores with a diverse diet, implying that their classification as shredders when calculating metrics of ecosystem functioning may be misleading. Moreover, macroconsumers can exert strong top-down controls on other invertebrates (Lancaster *et al*., 2008), which would disadvantage their relative roles as indicators of ecosystem condition and functioning.

There was an increase in predator driven top-down control with land-use change as indicated by both abundance- and biomass-based metrics. Urban and agricultural streams had an overabundance of predators, such as Hemiptera, Coleoptera and

Odonata. Increased abundance of predators, especially large bodied Odonates, beetles and bugs, which are fast colonizers and tolerant to poor water quality (Boulton & Lake, 2008), have been reported in disturbed streams, especially during droughts. Some Coleopterans (beetles) and Hemipterans (bugs) are also known to persist in drying pools and their high mobility enables them to escape and seek refuge in larger and permanent ones (Velasco & Millan, 1998). Some Odonate species are also tolerant to flow variation and temperature (Stewart & Samways, 1998; Hardersen, 2008), and this can partly explain their high abundance and diversity at the urban and agricultural sites.

There was close correspondence in the performance of channel stability and TFPOM vs. BFPOM metrics as measures of ecological condition. Across the four land-uses, both abundance- and biomass-based metrics showed that there was more FPOM in transport as compared to that deposited on the streambed, while the abundance-based metric showed that all sites had stale instream substrate. These two metrics relied mainly on Hydropsychidae and Simuliidae, which have been found to thrive in moderately disturbed sites with organic material utilized by Simuliidae that are the main prey for predatory Hydropsychidae (Rivers-Moore *et al.*, 2007; Masese & Raburu, 2017). This tolerance to organic pollution and cosmopolitan distribution of these two families biased assessment of the geomorphological condition of the river against visual evidence of erosion and sedimentation, especially at agricultural sites.

The better performance of abundance- vs. biomass-based metrics of ecological condition and function in this river was supported by the visual assessment of sites based on water quality variables and the qualitative habitat quality index. The poor performance of biomass-based metrics in this study raises interesting questions on the importance of macroconsumers and other large bodied individuals, such as some Odonates, bugs and beetles as bioindicators of ecological condition in streams and rivers. In the context of biodiversity and ecosystem functioning, evidence shows that species richness and diversity can enhance ecosystem functioning (Hooper *et al*., 2005; Cardinale *et al.,* 2012). This can be interpreted to imply that a diverse community of taxa belonging to various FFGs would be a better predictor of ecological condition and ecosystem function compared to a less diverse community of dominated by a few taxa. In a system with macroconsumers as ours, abundance better approximates species richness, than biomass. For instance, freshwater crabs can constitute more than 80% of the biomass of invertebrates (Masese *et al*., 2014), which significantly diminishes the presence and role of other invertebrates. Moreover, poor performance of biomass-based metrics and some abundance-based metrics could be as a result of using threshold values that were developed for temperate streams (Merritt *et al*., 2017), and may not be appropriate for tropical streams where the composition of FFGs and tolerance of taxa to different forms of disturbance are different (Boyero *et al*., 2009; Masese & Raburu, 2017).

The study recorded notable longitudinal shifts in metrics of ecosystem attributes in response to increasing stream size (Figure 18). From the RCC predictions it's expected that change in stream size would affect the composition of FFGs, and hence the metrics used here as measures of ecosystem functioning. In some instances, the ecosystem attributes agreed with the predictions of the RCC (Vannote *et al*., 1980), and in some instances there were total disagreement, but these were dependent on seasons and on whether the metrics were abundance- or biomass-based. For instance, headwaters streams were heterotrophic and mid-reaches were autotrophic, which is in agreement with RCC concept. Moreover, for most biomass-based metrics relationships were weak or non-existent, further highlighting the weaknesses of using biomass as a measure of ecosystem integrity. The weak longitudinal relationship imply that changes in stream size did not play a significant role in influencing the functional composition of macroinvertebrates as surrogates of ecosystem functioning, but rather the changes were as a result of confounding factors of seasonality and reach scale influences caused by agricultural activities and urbanization in the vicinity of the sampling sites.

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 CONCLUSIONS

This study shows that land-use change and seasonality affected water and habitat quality variables with higher values being recorded in the disturbed urban and agricultural sites. The results indicate that the functional organization of macroinvertebrates in headwater streams is subject to deterministic processes through the occurrence of gradients caused by changes in environmental conditions, such as organic matter, water and habitat quality. These differences are then amplified or ameliorated by flow variations caused by seasonality in these streams. Changes in land-use type from forestry to agricultural and urbanization resulted in deterioration of water and habitat quality, which subsequently affected the different macroinvertebrate FFGs differently.

The sensitive EPT taxa and shredders were found in the less polluted forested streams while the tolerant taxa were in the polluted streams. Shredder biomass was most negatively responsive to change in land-use type, while the rest of the functional feeding groups seemed to thrive in modified stream conditions. Higher shredder abundance, biomass and taxon richness was recorded in forested sites, irrespective of the season, while predators and scrapers increased in agricultural and urban sites where water temperature and nutrient concentrations were higher.

This study also adds to the increasing body of knowledge about macroinvertebrates' functional organization in tropics and their potential use as bioindicators of ecosystem health both at spatial and temporal scales. Although there were disagreements between abundance- vs. biomass-based metrics used as surrogates of ecosystem functioning, some metrics were in agreement, also with some of the predictions of the RCC, which confirms the potency of this approach as a means of assessing both the ecological integrity and functioning of tropical streams. The abundance-based metrics were more reliable as they agreed with visual assessments of the sites based on water and habitat quality, and cases of disagreements from expected patterns would be explained by the confoundment caused by land-use change and point sources of pollution (urbanizations and industries).

Despite the challenges of deposition and sedimentation along the river gradient during the wet season, and unbalanced sampling during the dry and wet seasons, the results from this study are reliable.

6.2 RECOMMENDATIONS

1. The study showed that catchment land-use can modify riparian quality and in-stream habitat conditions in turn affecting stream macroinvertebrate communities, therefore, riparian corridors along streams should be protected and this knowledge used in management actions of the Sosiani-Kipkaren River and similar other streams and rivers in the region.

2. Results from the study indicated clearer patterns during the dry than the wet season. Therefore, sampling for biomonitoring should be done during the base flow where the confounding effects of flow variation are minimal.

3. Overall, the use of FFG-based metrics as surrogates of ecological condition and ecosystem functioning in tropical streams holds promise, but there is a need to evaluate and update threshold values for the various metrics originally developed for temperate streams, as these may not be applicable.

REFERENCES

Abdul, N. H., & Rawi, C. S. M. (2019). Functional Feeding Group (FFG) of Aquatic Macroinvertebrate in Middle Reach of Kerian River Basin of North Malaysia Peninsula. *Tropical Life Sciences Research*, *30*(2). https://doi.org/10.21315/tlsr2019.30.2.1

Alberts, J. M., Fritz, K. M., & Buffam, I. (2018). Response to basal resources by stream macroinvertebrates is shaped by watershed urbanization, riparian canopy cover, and season. *Freshwater Science, 37(3),* 640-652. https://doi.org/10.1086/699385

Allan J. D. (1995) *Stream Ecology: Structure and Function of Running Waters*. Chapman & Hall, London.

Allan, J. D. (2004). Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics, 35*, 257-284. https://doi.org/10.1146/annurev.ecolsys.35.120202.110122

Amezaga, J. M., Santamaría, L., & Green, A. J. (2002). Biotic wetland connectivity—supporting a new approach for wetland policy. *Acta oecologica*, *23*(3), 213-222. https://doi.org/10.1016/S1146-609X(02)01152-9

Anyona, D. N., Abuom, P. O., Dida, G. O., Gelder, F. B., Onyuka, J. O., Matano, A. S., & Ofulla, A. V. (2014). Effect of anthropogenic activities on physico-chemical parameters and benthic macroinvertebrates of Mara river tributaries, Kenya. *Merit research journal of Environmental science and Toxicology*, *2*(5), 98-109.

APHA-American Public Health Association. WPCF-Water Pollution Control Federation (2005) Standard methods for the examination of water and wastewater 21st edn. *Washington, DC*.

Arimoro, F. O., Ikomi, R. B., & Iwegbue, C. M. (2007). Water quality changes in relation to Diptera community patterns and diversity measured at an organic effluent impacted stream in the Niger Delta, Nigeria. *Ecological indicators*, *7*(3), 541-552. . https://doi.org/10.1016/j.ecolind.2006.06.002

Arimoro, F. O. (2009). Impact of rubber effluent discharges on the water quality and macroinvertebrate community assemblages in a forest stream in the Niger Delta. *Chemosphere*, *77*(3), 440- 449. https://doi.org/10.1016/j.chemosphere.2009.06.031

Arimoro, F. O., Ikomi, R. B., Ajuzieogu, I. O., & Nwadukwe, F. O. (2011). Temporal and spatial variability in macroinvertebrate community structure in relation to environmental variables in Ajijiguan Creek, Niger Delta, Nigeria. *African journal of aquatic science*, *36*(1), 57-66. https://doi.org/10.2989/16085914.2011.559687

Arimoro, F. O., Obi-Iyeke, G. E., & Obukeni, P. J. (2012). Spatiotemporal variation of macroinvertebrates in relation to canopy cover and other environmental factors in Eriora River, Niger Delta, Nigeria. *Environmental monitoring and assessment*, *184*(10), 6449-6461. https://doi.org/10.1007/s10661-011-2432-9

Aschalew, L., & Moog, O. (2015). Benthic macroinvertebrates based new biotic score “ETHbios” for assessing ecological conditions of highland streams and rivers in Ethiopia. *Limnologica, 52*, 11-19. https://doi.org/10.1016/j.limno.2015.02.002

Aura, C. M., Raburu, P. O., & Herrmann, J. (2011). Macroinvertebrates’ community structure in rivers Kipkaren and Sosiani, river Nzoia basin, Kenya. *Journal of Ecology and the Natural Environment*, *3*(2), 39-46.

Baptista, D. F., Buss, D. F., Egler, M., Giovanelli, A., Silveira, M. P., & Nessimian, J. L. (2007). A multimetric index based on benthic macroinvertebrates for evaluation of Atlantic Forest streams at Rio de Janeiro State, Brazil. *Hydrobiologia*, *575*(1), 83. https://doi.org/10.1007/s10750-006-0286-x

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

Barbour, M. T., Gerritsen, J., Snyder, B. D., & Stribling, J. B. (1999). Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. *US Environmental Protection Agency, Office of Water*. Washington, DC, 339(1), 1-7.

Beasley, G., & Kneale, P. (2002). Reviewing the impact of metals and PAHs on macroinvertebrates in urban watercourses. *Progress in Physical Geography*, *26*(2), 236-270. https://doi.org/10.1191/0309133302pp334ra

Beche, L. A., Mcelravy, E. P., & Resh, V. H. (2006). Long- term seasonal variation in the biological traits of benthic- macroinvertebrates in two Mediterranean- climate streams in California, USA. *Freshwater Biology*, *51*(1), 56-75. https://doi.org/10.1111/j.1365-2427.2005.01473.x

Benstead, J. P., & Pringle, C. M. (2004). Deforestation alters the resource base and biomass of endemic stream insects in eastern Madagascar. *Freshwater Biology*, *49*(4), 490-501. https://doi.org/10.1111/j.1365-2427.2004.01203.x

Bogan, M. T., & Lytle, D. A. (2007). Seasonal flow variation allows ‘time- sharing’by disparate aquatic insect communities in montane desert streams. *Freshwater Biology*, 52(2), 290-304. https://doi.org/10.1111/j.1365-2427.2006.01691.x

Bogan, M. T., Boersma, K. S., & Lytle, D. A. (2013). Flow intermittency alters longitudinal patterns of invertebrate diversity and assemblage composition in an arid- land stream network. *Freshwater Biology*, 58(5), 1016-1028. https://doi.org/10.1111/fwb.12105

Boulton, A. J., & Lake, P. S. (2008). Effects of drought on stream insects and its ecological consequences. *Aquatic insects: challenges to populations. CABI Publishing, Wallingford, UK*, 81-102. https://doi.org/10.1079/9781845933968.0081

Bouwer, H. (2000). Integrated water management: emerging issues and challenges. *Agricultural water management*, *45*(3), 217-228. https://doi.org/10.1016/S0378-3774(00)00092-5

Boyero, L., Ramírez, A., Dudgeon, D., & Pearson, R. G. (2009). Are tropical streams really different? *Journal of the North American Benthological Society*, *28*(2), 397-403. https://doi.org/10.1899/08-146.1

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

Boyero, L., Pearson, R.G., Albariño, R.J., Callisto, M., Correa-Araneda, F., Encalada, A.C., Masese, F., Moretti, M.S., Ramírez, A., Sparkman, A.E. & Swan, C.M., (2020). Identifying Stream Invertebrates as Plant Litter Consumers. In *Methods to Study Litter Decomposition,* (455-464). Springer, Cham. https://doi.org/10.1007/978-3-030-30515-4_50

Bray, J. R., & Curtis, J. T. (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs, 27*(4), 326-349. https://doi.org/10.2307/1942268

Bunn, S. E., & Arthington, A. H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management, 30*(4), 492-507. https://doi.org/10.1007/s00267-002-2737-0

Burdon, F. J., McIntosh, A. R., & Harding, J. S. (2013). Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. *Ecological Applications, 23*(5), 1036-1047. https://doi.org/10.1890/12-1190.1

Buss, D. F., Carlisle, D. M., Chon, T. S., Culp, J., Harding, J. S., Keizer-Vlek, H. E., ... & Hughes, R. M. (2015). Stream biomonitoring using macroinvertebrates around the globe: a comparison of large-scale programs. *Environmental Monitoring and Assessment 187*(1), 4132. https://doi.org/10.1007/s10661-014-4132-8

Camacho, R., L. Boyero, Cornejo, A., Ibáñez, A. & Pearson, R. G. (2009). Local variation in shredder numbers can explain their oversight in tropical streams. *Biotropica, 4,* 625–632. https://doi.org/10.1111/j.1744-7429.2009.00519.x

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., ... & Kinzig, A. P. (2012). Biodiversity loss and its impact on humanity. *Nature, 486*(7401), 59-67. https://doi.org/10.1038/nature11148

Carpenter, S. R., Stanley, E. H., & Vander Zanden M. J (2011) State of the world’s freshwater ecosystems: physical, chemical, and biological changes. *Annual Review of Environmental Resources, 36,* 75-99. https://doi.org/10.1146/annurev-environ-021810-094524

Chará-Serna, A. M., Chará, J. D., Zúñiga, M. d. C., Pedraza, G. X., & Giraldo, L. P. (2010). Clasificación trófica de insectos acuáticos en ocho quebradas protegidas de la ecorregión cafetera Colombiana. *Universitas Scientiarum, 15*, 27-36. https://doi.org/10.11144/javeriana.SC15-1.tcoa

Cheshire K., Boyero L. and Pearson R.G. (2005) Foodwebs in tropical Australian streams: shredders are not scarce, *Freshwater Biology, 50*(5), 748-769. https://doi.org/10.1111/j.1365-2427.2005.01355.x

Clarke, K. R., & Gorley, R. N. (2006). Primer. *PRIMER-e,* Plymouth.

Clarke, K. R., Warwick R. M. (2001). Change in marine communities: an approach to statistical analysis and interpretation. *PRIMER-e*, Plymouth, UK.

Collier, K. J., & Quinn, J. M. (2003). Land- use influences macroinvertebrate community response following a pulse disturbance. *Freshwater Biology, 48*(8), 1462-1481. https://doi.org/10.1046/j.1365-2427.2003.01091.x

Cooper, S. D., Lake, P. S., Sabater, S., Melack, J. M., & Sabo, J. L. (2013). The effects of land use changes on streams and rivers in mediterranean climates. *Hydrobiologia*, *719*(1), 383-425. https://doi.org/10.1007/s10750-012-1333-4

Crowl, T. A., McDowell, W. H., Covich, A. P., & Johnson, S. L. (2001). Freshwater shrimp effects on detrital processing and nutrients in a tropical headwater stream. *Ecology, 82*, 775-783. https://doi.org/10.1890/0012-658(2001)082[0775:FSEODP]2.0.CO;2

Cummins, K. W., & Klug, M. J. (1979). Feeding ecology of stream invertebrates. *Annual review of ecology and systematics*, *10*(1), 147-172. https://doi.org/10.1146/annurev.es.10.110179.001051

Cummins, K. W., Wilzbach, M.A., Gates, D. M., Perry, J. B., Taliaferro, W. B., (1989). Shredders and riparian vegetation. *BioScience, 39,* 24-30. https://doi.org/10.2307/1310804

Cummins, K. W., Merritt, R. W., & Andrade, P. C. (2005). The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in south Brazil. *Studies on Neotropical Fauna and Environment, 40*(1), 69-89. https://doi.org/10.1080/01650520400025720

Dallas, H. F. (2007). The influence of biotope availability on macroinvertebrate assemblages in South African rivers: implications for aquatic bioassessment. *Freshwater biology*, *52*(2), 370-380. https://doi.org/10.1111/j.1365-2427.2006.01684.x

Day, J. A., and I. J. de Moor. (2002a). Guides to the freshwater invertebrates of southern Africa. Volume 5: Non-arthropods (the protozoans, Porifera, Cnidaria, Platyhelminthes, Nemertea, Rotifera, Nematoda, Nematomorpha, Gastrotrichia, Bryozoa, Tardigrada, Polychaeta, Oligochaeta and Hirudinea). WRC Report No.TT 167/02. *Water Research Commission, Pretoria, South Africa.*

Day, J. A., and I. J. de Moor. (2002b). Guides to the freshwater invertebrates of southern Africa. Volume 6: Arachnida and Mollusca (Araneae, Water Mites and Mollusca). WRC Report No.TT 182/02. *Water Research Commission, Pretoria, South Africa.*

de Moor, I. J., Day, J. A., & De Moor, F. C. (2003a). Guides to the freshwater invertebrates of southern Africa. Volume 7: Insecta I: Ephemeroptera, Odonata and Plecoptera. WRC Report No. TT 207/03. *Water Research Commission, Pretoria, South Africa*.

de Moor, I.J., Day, J.A., de Moor, F.C., (2003b). Guides to the freshwater invertebrates of southern Africa. Volume 8: Insecta II. Report No. TT 214/03. *Water Research Commission, Pretoria, South Africa.*

Dickens, C. W., & Graham, P. M. (2002). The South African Scoring System (SASS) version 5 rapid bioassessment method for rivers. *African Journal of Aquatic Sciences, 27*(1), 1-10. https://doi.org/10.2989/16085914.2002.9626569

Dobson, M., Magana, A., Mathooko, J. M., & Ndegwa, F. K. (2002). Detritivores in Kenyan highland streams: more evidence for the paucity of shredders in the tropics?. *Freshwater biology*, *47*(5), 909-919. https://doi.org/10.1046/j.1365-2427.2002.00818.x

Dolédec, S., Phillips, N., & Townsend, C., (2011). Invertebrate community responses to land use at a broad spatial scale: trait and taxonomic measures compared in New Zealand rivers. *Freshwater Biology, 56*, 1670-1688. https://doi.org/10.1111/j.1365-2427.2011.02597.x

Dudgeon, D. (1984). Longitudinal and temporal changes in functional organization of macroinvertebrate communities in the Lam Tsuen River, Hong Kong. *Hydrobiologia*, *111*(3), 207-217. https://doi.org/10.1007/BF00007201

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Lévêque, C., & Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological reviews*, *81*(2), 163-182. https://doi.org/10.1017/S1464793105006950

Dudgeon, D. (2010). Prospects for sustaining freshwater biodiversity in the 21st century: linking ecosystem structure and function. *Current Opinion in Environmental Sustainability*, *2*(5-6), 422-430. https://doi.org/10.1016/j.cosust.2010.09.001

Eady, B. R., Hill, T. R., & Rivers-Moore, N. A. (2014). Shifts in aquatic macroinvertebrate community structure in response to perenniality, southern Cape, South Africa. *Journal of Freshwater Ecology*, *29*(4), 475-490. https://doi.org/10.1080/02705060.2014.910146

Engineers, C. C. (2002). *Integrated water quality/limnological study for Lake Victoria. Lake Victoria Environmental project*. Part II Technical Report. Kisumu.

Entrekin, S. A., Rosi, E. J., Tank, J. L., Hoellein, T. J., & Lamberti, G. A. (2020). Quantitative Food Webs Indicate Modest Increases in the Transfer of Allochthonous and Autochthonous C to Macroinvertebrates Following a Large Wood Addition to a Temperate Headwater Stream. *Frontiers in Ecological Evolution*, *8*, 114. https://doi.org/10.3389/fevo.2020.00114

FAO, (2013). The State of Food Insecurity in the World: The multiple dimensions of food security.

Ferreira, V., Encalada, A. C., & Graça, M. A. (2012). Effects of litter diversity on decomposition and biological colonization of submerged litter in temperate and tropical streams. *Freshwater Science*, *31*(3), 945-962. https://doi.org/10.1899/11-062.1

Fierro, P., Bertrán, C., Tapia, J., Hauenstein, E., Peña-Cortés, F., Vergara, 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. https://doi.org/10.1016/j.scitotenv.2017.07.197

Fisher, R. A., Corbet, A. S. & Williams, C. B. (1943). The relation between the number of species and the number of individuals in a random sample of an animal population. *Journal of Animal Ecology, 12,* 42–58.

Fox, J. (2006). Teacher's corner: structural equation modeling with the sem package in R. *Structural equation modelling, 13*(3), 465-486. https://doi.org/10.1207/s15328007sem1303_7

Freitas, S., Hatosy, S., Fuhrman, J. A., Huse, S. M., Welch, D. B. M., Sogin, M. L., & Martiny, A. C. (2012). Global distribution and diversity of marine Verrucomicrobia. *The ISME journal*, *6*(8), 1499-1505. https://doi.org/10.1038/ismej.2012.3

Fugère, V., Jacobsen, D., Finestone, E. H., & Chapman, L. J. (2018). Ecosystem structure and function of afrotropical streams with contrasting land use. *Freshwater Biology, 63*(12), 1498-1513. https://doi.org/10.1111/fwb.13178

García, L., Cross, W. F., Pardo, I., & Richardson, J. S. (2017). Effects of landuse intensification on stream basal resources and invertebrate communities. *Freshwater Biology, 36*(3), 609-625. https://doi.org/10.1086/693457

GEF (Global Environmental Facility). 2004. Western Kenya integrated ecosystem project; Executive Summary.

Graça, M. S., Cressa, C. M. O. G., Gessner, T. O., Feio, M. J., & Callies, K. A. (2001). Food quality, feeding preferences, survival and growth of shredders from temperate and tropical streams. *Freshwater biology*, *46*(7), 947-957. https://doi.org/10.1046/j.1365-2427.2001.00729.x

Greathouse, E. A., Pringle, C. M., McDowell, W. H., & Holmquist, J. G. (2006). Indirect upstream effects of dams: consequences of migratory consumer extirpation in Puerto Rico. *Ecological Applications*, *16*(1), 339-352. https://doi.org/10.1890/05-0243

Griffiths, N. A., Tank, J. L., Royer, T. V., Roley, S. S., Rosi-Marshall, E. J., Whiles, M. R., ... & Johnson, L. T. (2013). Agricultural land use alters the seasonality and magnitude of stream metabolism. *Limnology & Oceanography*, *58*(4), 1513-1529. https://doi.org/10.4319/lo.2013.58.4.1513

Hardersen, S. (2008). Dragonfly (Odonata) communities at three lotic sites with different hydrological characteristics. *Italian Journal of Zoology*, *75*(3), 271-283. https://doi.org/10.1080/11250000801925227

Harrison, A.D., & Hynes, H.B.N., (1988). Benthic fauna of Ethiopian streams and rivers. *Archive fur Hydrobiology, 81*, 1–36.

Hastie, T. J., & Tibshirani, R. J. (1990). Generalized additive models (Vol. 43). *CRC press*.

Hawkins, C. P., & Vinson, M. R. (2000). Weak correspondence between landscape classifications and stream invertebrate assemblages: implications for bioassessment. *Journal of the North American Benthological Society*, *19*(3), 501-517. https://doi.org/10.2307/1468111

Hawkins, C. P., Mykrä, H., Oksanen, J., & Vander Laan, J. J. (2015). Environmental disturbance can increase beta diversity of stream macroinvertebrate assemblages. *Global Ecology and Biogeography*, *24*(4), 483-494. . https://doi.org/10.1111/geb.12254

Hepp, L. U., Urbim, F. M., Tonello, G., Loureiro, R. C., Sausen, T. L., Fornel, R., & Restello, R. M. (2016). Influence of land-use on structural and functional macroinvertebrate composition communities associated on detritus in Subtropical Atlantic Forest streams. *Acta Limnologica Brasiliensia*, *28*. https://doi.org/10.1590/S2179-975X0616

Hill, M. O. (1973). Diversity and evenness: a unifying notation and its consequences. *Ecology, 54*, 427-432. https://doi.org/10.2307/1934352

Hooper, D. U., Chapin Iii, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., ... & Schmid, B. (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecological Monographs, 75*(1), 3-35. https://doi.org/10.1890/04-0922

Hussain, Q. A., & Pandit, A. K. (2012). Macroinvertebrates in streams: A review of some ecological factors. *International Journal of Fisheries and Aquaculture*, *4*(7), 114-123. https://doi.org/10.5897/IJFA11.045

Hyslop, E., & Hunte-Brown, M. (2012). Longitudinal variation in the composition of the benthic macroinvertebrate fauna of a typical North coast Jamaican river. *Revista de Biología Tropical*, *60*(1), 291-303. https://doi.org/10.15517/rbt.v60i1.2762

Imberger, S. J., Thompson, R. M., & Grace, M. R. (2011). Urban catchment hydrology overwhelms reach scale effects of riparian vegetation on organic matter dynamics. *Freshwater Biology*, *56*(7), 1370-1389. https://doi.org/10.1111/j.1365-2427.2011.02575.x

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. https://doi.org/10.1016/j.scitotenv.2017.06.100

Jiang, X. M., Xiong, J., Qiu, J. W., Wu, J. M., Wang, J. W., & Xie, Z. C. (2010). Structure of macroinvertebrate communities in relation to environmental variables in a subtropical Asian river system. *International Review of Hydrobiology*, *95*(1), 42-57. https://doi.org/10.1002/iroh.200811131

Jiang, X., Xiong, J., Xie, Z., & Chen, Y. (2011). Longitudinal patterns of macroinvertebrate functional feeding groups in a Chinese river system: A test for river continuum concept (RCC). *Quaternary International, 244*(2), 289-295. https://doi.org/10.1016/j.quaint.2010.08.015

Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river floodplain systems. In D.P. Dodge (Ed.), *Proceedings of the International Large River Symposium*. *Canadian Journal of Fisheries &. Aquatic Science, 106,* 110–127.

Junker, J. R., & Cross, W. F. (2014). Seasonality in the trophic basis of a temperate stream invertebrate assemblage: Importance of temperature and food quality. *Limnology & Oceanography, 59*(2), 507-518. https://doi.org/10.4319/lo.2014.59.2.0507

Kaaya, L. T., Day, J. A., & Dallas, H. F. (2015). Tanzania River Scoring System (TARISS): a macroinvertebrate-based biotic index for rapid bioassessment of rivers. *African Journal of Aquatic Sciences*, *40*(2), 109-117. https://doi.org/10.2989/16085914.2015.1051941

Kashian, D. R., Zuellig, R. E., Mitchell, K. A., & Clements, W. H. (2007). The cost of tolerance: Sensitivity of stream benthic communities to UV- B and metals. *Ecological Applications*, *17*(2), 365-375. https://doi.org/10.1890/06-0396

Kaufmann P.R. and Faustini J.M., (2012). Simple measures of channel habitat complexity predict transient hydraulic storage in streams. *Hydrobiologia*, *685*, 69–95. https://doi.org/10.1007/s10750-011-0841-y

Kazanci, N & Dügel, M. (2010). Determination of influence of heavy metals on structure of benthic macroinvertebrate assemblages in low order Mediterranean streams by using canonical correspondence analysis. *Review of Hydrobiology*, *3*(1), 13-26.

KEBS (Kenya Bureau of Standards). 2015. Portable water specifications-KS EAS 12:2014, Kenya, Kenya Bureau of Standards.

Kenya Gazette Supplement No. 68. 2006. Environmental Management and Co-ordination (Water Quality) Regulations, 2006 Act No. 8 of 1999.

Ko, N. T., Suter, P., Conallin, J., Rutten, M., & Bogaard, T. (2020). The Urgent Need for River Health Biomonitoring Tools for Large Tropical Rivers in Developing Countries: Preliminary Development of a River Health Monitoring Tool for Myanmar Rivers. *Water*, *12*(5), 1408. https://doi.org/10.3390/w12051408

Kobingi, N., Raburu, P.O., Masese, F.O and Gichuki, J. (2009). Assessment of pollution impacts on the ecological integrity of the Kisian and Kisat rivers in Lake Victoria drainage basin, Kenya. *African Journal of Environmental Science and Technology*, *3*, 097-107.

Lancaster, J., Dobson, M., Magana, A. M., Arnold, A., & Mathooko, J. M. (2008). An unusual trophic subsidy and species dominance in a tropical stream. *Ecology*, *89*(8), 2325-2334. https://doi.org/10.1890/07-0553.1

Lawrence, J. M., & Gresens, S. E. (2004). Foodweb response to nutrient enrichment in rural and urban streams. *Journal of Freshwater Ecology, 19*(3), 375-385. https://doi.org/10.1080/02705060.2004.9664910

Leigh, C. (2013). Dry-season changes in macroinvertebrate assemblages of highly seasonal rivers: responses to low flow, no flow and antecedent hydrology. *Hydrobiologia,703*(1), 95-112. https://doi.org/10.1007/s10750-012-1347-y

López-Carr, D., & Burgdorfer, J. (2013). Deforestation drivers: population, migration, and tropical land use. *Environment, 55*(1), 3-11. https://doi.org/10.1080/00139157.2013.748385

Lorion, C. M., & Kennedy, B. P. (2009). Relationships between deforestation, riparian forest buffers and benthic macroinvertebrates in neotropical headwater streams. *Freshwater Biology, 54*(1), 165-180. https://doi.org/10.1111/j.1365-2427.2008.02092.x

M’merimba, 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*.

Magurran, A. E. (2004). Measuring biological diversity. *Blackwell Publishing*, Oxford, UK.

Makaka, C., Muteveri, T., Makoni, P., Phiri, C., & Dube, T (2018). Longitudinal distribution of the functional feeding groups (FFGs) of aquatic macroinvertebrates and ecosystem integrity of Tokwe River, Zimbabwe. *Journal of Biodiversity and Environmental Sciences*, *13*(1), 16-33.

Marchant, R., Metzeling, L., Graesser, A., & Suter, P. (1985). The organization of macroinvertebrate communities in the major tributaries of the LaTrobe River, Victoria, Australia. *Freshwater Biology, 15*(3), 315-331. https://doi.org/10.1111/j.1365-2427.1985.tb00204.x

Masese, F. O., Kitaka, N., Kipkemboi, J., Gettel, G. M., Irvine, K., & McClain, M. E. (2014a). Macroinvertebrate functional feeding groups in Kenyan highland streams: evidence for a diverse shredder guild. *Freshwater Science, 33*(2), 435-450. https://doi.org/10.1086/675681

Masese, F. O., Kitaka, N., Kipkemboi, J., Gettel, G. M., Irvine, K., & McClain, M. E. (2014b). Litter processing and shredder distribution as indicators of riparian and catchment influences on ecological health of tropical streams. *Ecological Indicators, 46*, 23-37. https://doi.org/10.1016/j.ecolind.2014.05.032

Masese, F. O., Muchiri, M., & Raburu, P. O. (2009a). Macroinvertebrate assemblages as biological indicators of water quality in the Moiben River, Kenya. *African Journal of Aquatic Sciences, 34*(1), 15-26. https://doi.org/10.2989/AJAS.2009.34.1.2.727

Masese, F. O., P. O. Raburu, and M. Muchiri. (2009b). A preliminary benthic macroinvertebrate index of biotic integrity (B-IBI) for monitoring the Moiben River, Lake Victoria Basin, Kenya. *African Journal of Aquatic Sciences, 34*, 1–14. https://doi.org/10.2989/AJAS.2009.34.1.1.726

Masese, F. O., Salcedo-Borda, J. S., Gettel, G. M., Irvine, K., & McClain, M. E. (2017). Influence of catchment land use and seasonality on dissolved organic matter composition and ecosystem metabolism in headwater streams of a Kenyan river. *Biogeochemistry, 132*(1-2), 1-22. https://doi.org/10.1007/s10533-016-0269-6

Masese, F. O., & Raburu, P. O. (2017). Improving the performance of the EPT Index toaccommodate multiple stressors in Afrotropical streams. *African Journal of Aquatic Sciences*, *42*(3), 219-233. https://doi.org/10.2989/16085914.2017.1392282

Masese, F. O., Abrantes, K. G., Gettel, G. M., Irvine, K., Bouillon, S., & McClain, M. E. (2018). Trophic structure of an African savanna river and organic matter inputs by large terrestrial herbivores: a stable isotope approach. *Freshwater Biology, 63*(11), 1365-1380. https://doi.org/10.1111/fwb.13163

Masese, F. O., Achieng, A. O., O’Brien, G. C., & McClain, M. E. (2021). Macroinvertebrate taxa display increased fidelity to preferred biotopes among disturbed sites in a hydrologically variable tropical river. *Hydrobiologia, 848*(2), 321-343. https://doi.org/10.1007/s10750-020-04437-1

Mason, W. T., Lewis, P. A., & Weber, C. I. (1983). An evaluation of benthic macroinvertebrate biomass methodology. *Environmental Monitoring and Assessment*, *3*(1), 29-44. https://doi.org/10.1007/BF00394030

Mathers, K. L., Rice, S. P., & Wood, P. J. (2017). Temporal effects of enhanced fine sediment loading on macroinvertebrate community structure and functional traits. *Science of the Total Environment, 599*, 513-522. https://doi.org/10.1016/j.scitotenv.2017.04.096

Mathooko, J. M., & Kariuki, S. T. (2000). Disturbances and species distribution of the riparian vegetation of a Rift Valley stream. *African Journal of Ecology*, *38*(2), 123-129. https://doi.org/10.1046/j.1365-2028.2000.00225.x

Mathooko, J. M. (2001). Temporal and spatial distribution of the baetid Afroptilum sudafricanum in the sediment surface of a tropical stream. *Hydrobiologia*, *443*(1-3), 1-8. https://doi.org/10.1023/A:1017502421985

McArdle, B. H., & Anderson, M. J. (2001). Fitting multivariate models to community data: a comment on distance based redundancy analysis. *Ecology, 82*(1), 290-297. https://doi.org/10.1890/0012-9658(2001)082[0290:FMMTCD]2.0.CO;2

McDowell, R. W., & Sharpley, A. N. (2001). Approximating phosphorus release from soils to surface runoff and subsurface drainage. *Journal of Environmental Quality, 30*(2), 508-520. https://doi.org/10.2134/jeq2001.302508x

Merritt, R. W., Cummins, K. W., Berg, M. B., Novak, J. A., Higgins, M. J., Wessell, K. J., & Lessard, J. L. (2002). Development and application of a macroinvertebrate functional-group approach in the bioassessment of remnant river oxbows in southwest Florida. *Journal of the North American Benthological Society*, *21*(2), 290-310. https://doi.org/10.2307/1468416

Merritt, R. W., & Cummins, K. W. (2006). Trophic relationships of macroinvertebrates. *Methods in stream ecology*, *2*, 585-601. https://doi.org/10.1016/B978-012332908-0.50036-X

Merritt, R. W., & Cummins, K. W. (Eds.). (1996). *An introduction to the aquatic insects of North America*. Kendall Hunt.

Merritt, R. W., Cummins, K. W., & Berg, M. B. (2008). An Introduction to the Aquatic Insects of North America. 4th (Edition) Kendall Hunt Publishing. *Dubuque, Iowa, U. S. A.*

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. https://doi.org/10.1016/B978-0-12-416558-8.00020-2

Minaya, V., McClain, M. E., Moog, O., Omengo, F., & Singer, G. A. (2013). Scale-dependent effects of rural activities on benthic macroinvertebrates and physico-chemical characteristics in headwater streams of the Mara River, Kenya. *Ecological Indicators*, *32*, 116-122. https://doi.org/10.1016/j.ecolind.2013.03.011

Minshall, G. W. (1978). Autotrophy in stream ecosystems. *BioScience*, *28*(12), 767-771. https://doi.org/10.2307/1307250

Molina Arzabe, C. I. (2004). Estudio de los rasgos biologicos y ecologicos en poblaciones de los ordenes: Ephemeroptera, Plecoptera y Trichoptera (clase insecta), en un rio al pie del glaciar Mururata.

Muñoz-Mas, R., Sánchez-Hernández, J., McClain, M. E., Tamatamah, R., Mukama, S. C., & Martínez-Capel, F. (2019). Investigating the influence of habitat structure and hydraulics on tropical macroinvertebrate communities. *Ecohydrology & Hydrobiology*, *19*(3), 339-350. https://doi.org/10.1016/j.ecohyd.2018.07.005

Mwaijengo, G. N., Vanschoenwinkel, B., Dube, T., Njau, K. N., & Brendonck, L. (2020). Seasonal variation in benthic macroinvertebrate assemblages and water quality in an Afrotropical river catchment, northeastern Tanzania. *Limnologica*, *12*,57-80. https://doi.org/10.1016/j.limno.2020.125780

Mykrä, H., Heino, J., & Muotka, T. (2007). Scale- related patterns in the spatial and environmental components of stream macroinvertebrate assemblage variation. *Global Ecology and Biogeography*, *16*(2), 149-159. https://doi.org/10.1111/j.1466-8238.2006.00272.x

Naiman, R. J., Bechtold, J. S., Drake, D. C., Latterell, J. J., O'keefe, T. C., & Balian, E. V. (2005). Origins, patterns, and importance of heterogeneity in riparian systems. In *Ecosystem function in heterogeneous landscapes* (pp. 279-309). Springer, New York, NY. https://doi.org/10.1007/0-387-24091-8_14

NEMA (National Environmental Management Authority). (2006). *Environmental Management and Coordination,* (Water Quality) Regulations 2006 in Kenya.

Njiru M, Kazungu J, Ngugi CC, Gichuki J, Muhoozi L (2008). An overview of the current status of Lake Victoria fshery: Opportunities, challenges and management strategies. *Lake and reservoir management, 13,* 1-12.

Nyadawa, M. O., & Mwangi, J. K. (2011). Geomorphologic characteritics of Nzoia river basin. *JOURNAL OF AGRICULTURE, SCIENCE AND TECHNOLOGY*, *12*(2).

O’Brien, A., K. Townsend, R. Hale, D. Sharley, & V. Pettigrove, (2016). How is ecosystem health defined and measured? A critical review of freshwater and estuarine studies. *Ecological Indicators, 69,* 722-729. https://doi.org/10.1016/j.ecolind.2016.05.004

O'Brien, G. C., Dickens, C., Hines, E., Wepener, V., Stassen, R., Quayle, L.… & Landis, W. G. (2018). A regional-scale ecological risk framework for environmental flow evaluations. *Hydrology and Earth System Sciences, 22*(2), 957-975. https://doi.org/10.5194/hess-22-957-2018

Oksanen, J., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, P. R., O’hara, R. B., & Oksanen, M. J. (2013). Package ‘vegan’. Community ecology Package, version, 2(9), 1-295.

Okungu J, Opango P. (2005). Pollution loads into Lake Victoria from the Kenyan catchment. In: *Mallya CA (ed.),* Knowledge and experiences gained from managing the Lake Victoria ecosystem. Dar es Salaam: Regional Secretariat, Lake Victoria Environmental Management Project (LVEMP). pp 90–108.

Oruta, J. N., Oindo, B. O., & Bosire, E. K. (2017). Relationship between riparian vegetation cover and macroinvertebrate assemblages in Kuywa River, Kenya.

Osano, O., Nzyuko, D., Admiraal, W. (2003). The fate of chloroacetalinide herbicides and their degradation products in the Nzoia Basin, Kenya. *AMBIO: A Journal of the Human Environment, 32,* 424-427. https://doi.org/10.1579/0044-7447-32.6.424

Palmer, M. A., E. S. Bernhardt, J. D. Allan, P. S.Lake, G. Alexander, S. Brooks, ... & D. L. Galat, (2005). Standards for ecologically successful river restoration. *Journal of Applied Ecology, 42,* 208-217. https://doi.org/10.1111/j.1365-2664.2005.01004.x

Palmer, R. W., & Taylor, E. D. (2004). The Namibian Scoring System (NASS) version 2 rapid bio-assessment method for rivers. *African Journal of Aquatic Science*, *29*(2), 229-234. https://doi.org/10.2989/16085910409503814

Pavluk, T. I., bij de Vaate, A., & Leslie, H. A. (2000). Development of an index of trophic completeness for benthic macroinvertebrate communities in flowing waters. *Hydrobiologia*, *427*(1), 135-141. https://doi.org/10.1023/A:1003911109416

Pearson, R. G., R. K. Tobin, L. J. Benson, and R. E. W. Smith. (1989). Standing crop and processing of rainforest litter in a tropical Australian stream. *Archiv für Hydrobiologie*, *115,* 481–498.

Petchey, O.L., Hector, A., Gaston, K.J., (2004). How do different measures of functional diversity perform? *Ecology, 85*, 847-857. https://doi.org/10.1890/03-0226

Pielou, E. C. 1975: Ecological Diversity. Wiley InterScience, New York.

Polegatto, C. M., & Froehlich, C. G. (2003). Feeding strategies in Atalophlebiinae (Ephemeroptera: Leptophlebiidae), with considerations on scraping and filtering. *Research Update on Ephemeroptera and Plecoptera'.(Ed. E Gaino.) pp*, 55-61.

R Development Core Team (2017). R: *A language and environment for statistical computing*; R Foundation for Statistical Computing, Vienna, Austria. URL http://www. R-project.org/.

Raburu P.O. (2003). *Water quality and the status of aquatic macroinvertebrates and ichthyofauna in River Nyando, Kenya*. PhD thesis, Moi University, Kenya.

Raburu, P. O., & Masese, F. O. (2012). Development of a fish based index of biotic integrity (FIBI) for monitoring riverine ecosystems in the Lake Victoria drainage Basin, Kenya. *River Research and Applications, 28*(1), 23-38. https://doi.org/10.1002/rra.1428

Raburu, P. O., Okeyo-Owuor, J. B., & Masese, F. O. (2009). Macroinvertebrate-based index of biotic integrity (M-IBI) for monitoring the Nyando River, Lake Victoria Basin, Kenya. *Scientific Research and Essays, 4*(12), 1468-1477.

Rajele, M. J. (2004). *A Comparison of SAAS and chemical monitoring of the rivers of the Lesotho highlands water project* (Doctoral dissertation, University of the Western Cape).

Ramirez, A., Pringle, C. M., & Wantzen, K. M (2008). Tropical stream conservation. In: Dudgeon D (ed). In *Tropical Stream Ecology*. Elsevier, London, 284–304. https://doi.org/10.1016/B978-012088449-0.50012-1

Rankin, E. T. (1995). Habitat indices in water resource quality assessments. Biological assessment and criteria: *Tools for water resource planning and decision making, 1995,* 181-208.

Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T., ... & Smol, J. P. (2019). Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biological Reviews, 94*(3), 849-73. https://doi.org/10.1111/brv.12480

Resh, V. H. (1995). Freshwater benthic macroinvertebrates and rapid assessment procedures for water quality monitoring in developing and newly industrialized countries. In W. S. Davis & T. P. Simon (Eds.), Biological assessment and criteria: Tools for water resources planning and decision making (pp 167–177).

Resh, V. H. (2008). Which group is best? Attributes of different biological assemblages used in freshwater biomonitoring programs. *Environmental Monitoring & Assessment, 138*(1-3), 131-138. https://doi.org/10.1007/s10661-007-9749-4

Resh, V. H., & Rosenberg, D. M. (Eds.). (1993). *Freshwater biomonitoring and benthic macroinvertebrates* (No. 504.4 FRE). New York, NY, USA:: Chapman & Hall.

Rivers Moore, N. A., De Moor, F. C., Morris, C., & O'keeffe, J. (2007). Effect of flow variability modification and hydraulics on invertebrate communities in the Great Fish River (Eastern Cape province, South Africa), with particular reference to critical hydraulic thresholds limiting larval densities of Simulium chutteri Lewis (Diptera, Simuliidae). *River research and applications*, *23*(2), 201-222. https://doi.org/10.1002/rra.976

Roach, K. A., & Winemiller, K. O. (2015). Hydrologic regime and turbidity influence entrance of terrestrial material into river food webs. *Canadian Journal of Fisheries & Aquatic Science, 72*(7), 1099 -1112. https://doi.org/10.1139/cjfas-2014-0459

Rodrigues, V., Estrany, J., Ranzini, M., de Cicco, V., Martín-Benito, J. M. T., Hedo, J., & Lucas-Borja, M. E. (2018). Effects of land use and seasonality on stream water quality in a small tropical catchment: The headwater of Córrego Água Limpa, São Paulo (Brazil). *Science of the total environment*, *622*, 1553-1561. https://doi.org/10.1016/j.scitotenv.2017.10.028

Rosenberg, D. M., Resh, V. H., & King, R. S. (2008). Use of aquatic insects in biomonitoring. pp. 123–138. *Merritt, RW, KW Cummins y MB Berg. An introduction to the aquatic insects of North America. Fourth edition. Kendall/Hunt Publications. Dubuque*.

Sangale F, Okungu J, Opango P. (2005). Variation of flow of water from Rivers Nzoia, Yala and Sio into Lake Victoria. In Knowledge and Experiences gained from Managing the Lake Victoria Ecosystem, Mallya GA, Katagira FF, Kang'oha G, Mbwana SB, Katunzi EF, Wambede JT, Azza N,Wakwabi E, Njoka SW, Kusewa M, Busulwa H (eds). Regional Secretariat, *Lake Victoria Environmental Management Project (LVEMP)*: Dar es Salaam; 691–702.

Savić, A., Đorđević, M., Jušković, M., & Pešić, V. (2018). Ecological analysis of macroinvertebrate communities based on functional feeding groups: a case study in southeastern Serbia. *Biologica Nyssana, 8*(2).

Schlünz, B., & Schneider, R. R. (2000). Transport of terrestrial organic carbon to the oceans by rivers: re-estimating flux-and burial rates. *International Journal of Earth Sciences, 88*(4), 599-606. https://doi.org/10.1007/s005310050290

Schofield, K. A., Pringle, C. M., Meyer, J. L., & Sutherland, A. B. (2001). The importance of crayfish in the breakdown of rhododendron leaf litter. *Freshwater Biology,* 46, 1191–1204. https://doi.org/10.1046/j.1365-2427.2001.00739.x

Simpson, E. H. 1949: Measurement of diversity. *Nature, 163,* 688.

Soetaert, K. E., Petzoldt, T., & Setzer, R. W. (2010). Solving differential equations in R: package deSolve. *Journal of statistical software,* 33. https://doi.org/10.18637/jss.v033.i09

Stals, R., de Moor, I.J., 2007. Guides to the Freshwater Invertebrates of Southern Africa, Volume 10: Coleoptera. *WRC report No. TT 320/07,* South Africa.

Sterling, J. L., Rosemond, A. D., & Wenger, S. J. (2016). Watershed urbanization affects macroinvertebrate community structure and reduces biomass through similar pathways in Piedmont streams, Georgia, USA. *Freshwater. Science, 35*(2), 676-688. https://doi.org/10.1086/686614

Stewart, D. A., & Samways, M. J. (1998). Conserving dragonfly (Odonata) assemblages relative to river dynamics in an African savanna game reserve. *Conservation Biology, 12*(3), 683-692. https://doi.org/10.1046/j.1523-1739.1998.96465.x

Stone, M. L., Whiles, M. R., Webber, J. A., Williard, K. W., & Reeve, J. D. (2005). Macroinvertebrate communities in agriculturally impacted southern Illinois streams. *Journal of Environmental Qualit, 34*(3), 907-917. https://doi.org/10.2134/jeq2004.0305

Taylor, J. C., de la Rey, P. A., & van Rensburg, L. (2005). Recommendations for the collection, preparation and enumeration of diatoms from riverine habitats for water quality monitoring in South Africa. *African Journal of Aquatic Science, 30*(1), 65-75. https://doi.org/10.2989/16085910509503836

Ter Braak, C. J. F., & Smilauer, P. (2004). Canoco for Windows 4.53 Biometris-Plant Research International. *Wageningen University and Research Centre, the Netherlands*.

Ter Braak, C. T., & Smilauer, P. (1998). CANOCO reference manual and user's guide to Canoco for Windows: software for canonical community ordination (*version 4*).

Tiegs, S.D., Peter, F.D., (2008). Leaf decomposition and invertebrate colonization responses to manipulated litter quantity in streams. *Journal of North American Benthological Society, 27*(3), 21-331. https://doi.org/10.1899/07-054.1

Tomanova, S., Goitia, E., & Helešic, J. (2006). Trophic levels and functional feeding groups of macroinvertebrates in neotropical streams. *Hydrobiologia*, *556*(1), 251-264. https://doi.org/10.1007/s10750-005-1255-5

Townsend, C. R., Doledec, S., & SCARSBROOK, M. (1997). Species traits in relation to temporal and spatial heterogeneity in streams: a test of habitat templet theory. *Freshwater Biology*, *37*(2), 367-387. https://doi.org/10.1046/j.1365-2427.1997.00166.x

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. https://doi.org/10.1046/j.1365-2028.2000.00212.x

Twesigye, C. K., Onywere, S. M., Getenga, Z. M., Mwakalila, S. S., & Nakiranda, J. K. (2011). The impact of land use activities on vegetation cover and water quality in the Lake Victoria watershed. *The Open Environmental Engineering Journal*, *4*(1).

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

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The river continuum concept. *Canadian journal of fisheries and aquatic sciences*, *37*(1), 130-137. https://doi.org/10.1139/f80-017

Velasco, J., & Millan, A. (1998). Insect dispersal in a drying desert stream: effects of temperature and water loss. Southwest. *Nature,* 80-87.

Verberk, W.C.E.P., van Noordwijk, C.G.E.,& Hildrew, A.G., (2013). Delivering on a promise: integrating species traits to transform descriptive community ecology into a predictive science. *Freshwater Science, 32,* 531- 547. https://doi.org/10.1899/12-092.1

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., ... & Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, *467*(7315), 555-561. https://doi.org/10.1038/nature09440

Wagenhoff, A., Townsend, C. R., & Matthaei, C. D. (2012). Macroinvertebrate responses along broad stressor gradients of deposited fine sediment and dissolved nutrients: a stream mesocosm experiment. *Journal of Applied Ecology, 49*(4), 892-902. https://doi.org/10.1111/j.1365-2664.2012.02162.x

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: current knowledge and the search for a cure. *Journal of North American Benthological Society, 24*(3), 706-723. https://doi.org/10.1899/04-028.1

Wantzen, K. M., Yule, C. M., Tockner, K. & Junk,W. J. (2008) Riparian wetlands of tropical streams. In: Dudgeon, D. (Ed.). *Tropical stream ecology*. USA: Elsevier Inc., 199–217. https://doi.org/10.1016/B978-012088449-0.50005-4

Wenger, S. J., Roy, A. H., Jackson, C. R., Bernhardt, E. S., Carter, T. L., Filoso, S., ... & Meyer, J. L. (2009). Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. *Journal of the North American Benthological Society, 28*(4), 1080-1098. https://doi.org/10.1899/08-186.1

Wetzel RG, & Likens GE. (2000). Limnological analysis. 3rd edition. Springer-Verlag, New York. https://doi.org/10.1007/978-1-4757-3250-4

Wilhm, J. L., & Dorris, T. C. (1968). Biological parameters for water quality criteria. *Bioscience*, 477-481. https://doi.org/10.2307/1294272

Wood, S. N. (2017). Generalized additive models*: An introduction with R*. CRC press. 850. https://doi.org/10.1201/9781315370279

Wood, S., & Wood, M. S. (2015). Package ‘mgcv’. *R package version, 1,* 29.

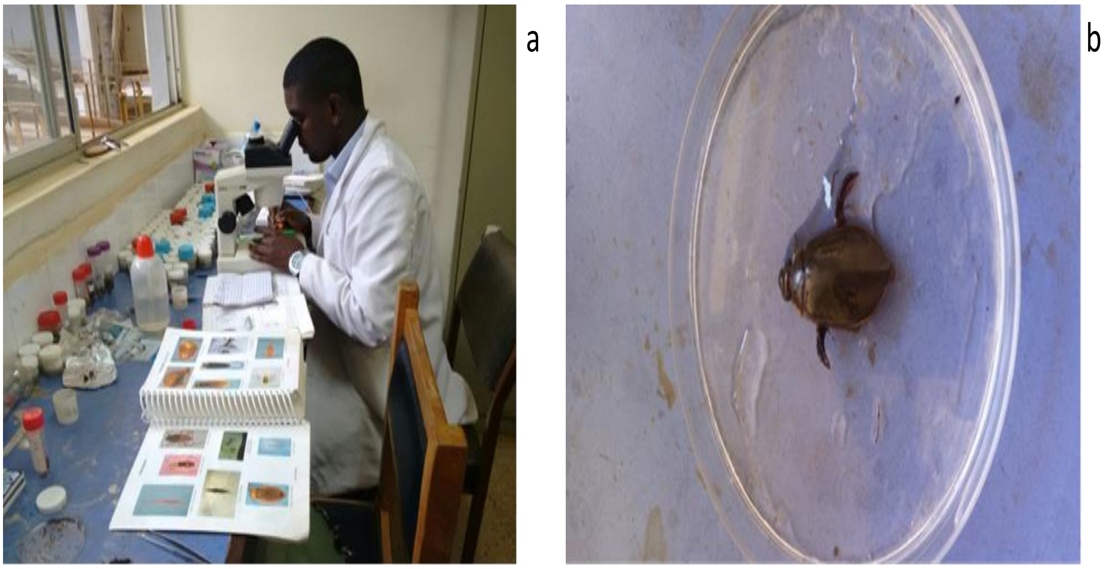
Yule, C.M. (1996). Trophic relationships and food webs of the benthic invertebrate fauna of two seasonal tropical streams on Bougainville Island, Papua New Guinea. *Journal of Tropical Ecology, 12,* 517-534. https://doi.org/10.1017/S0266467400009755

Yule, C.M., Leong, M.Y., Ratnarajah, L., Schmidt, K., Wong, H.M., Pearson, R.G., Boyero, L., (2009). Shredders in Malaysia: abundance and richness are greater in cool highland tropical streams. *Journal of North American Benthological Society, 28*, 404–415. https://doi.org/10.1899/07-161.1

Zeug, S. C., & Winemiller, K. O. (2008). Evidence supporting the importance of terrestrial carbon in a large river food web. *Ecology, 89,* 1733–1743. https://doi.org/10.1890/07-1064.1

APPENDICES

Appendix I: a; Identification of macroinvertebrates in the laboratory under a stereomicroscope with the aid of several keys, b; an identified Dytiscidae



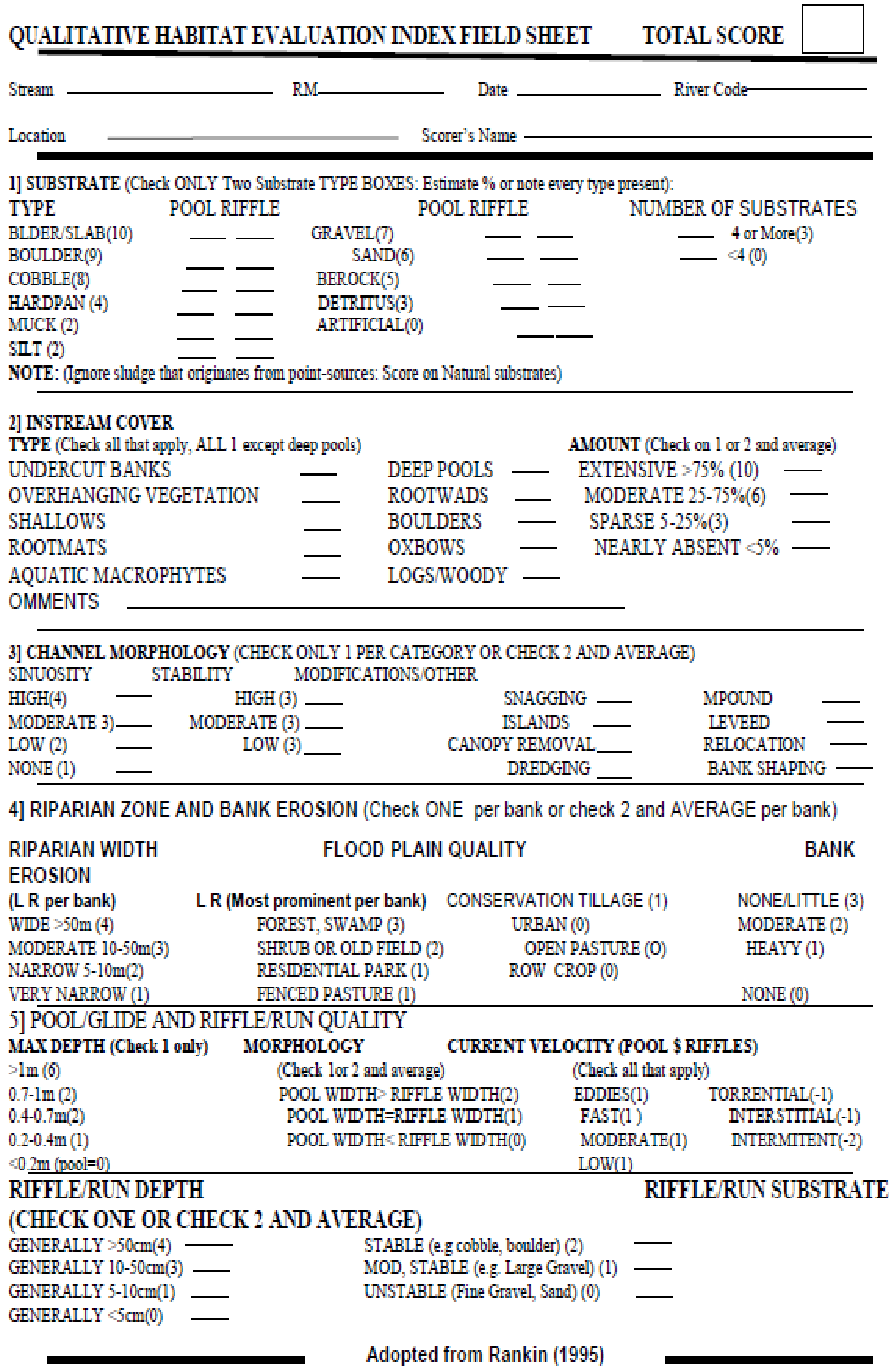
Appendix II: Macroinvertebrates FFG assignment (Merritt & Cummins, 1996; Dobson *et al.*, 2002, Molina, 2004; Masese *et al.*, 2014a) collected in the Sosiani-Kipkaren River during the wet and dry seasons. PRD = Predators, SCR = Scrapers, CG = Collector-gatherers, CF = Collector-filterers, SHR = Shredders

ORDER	FAMILY	GENUS	FFG
Arachnida	Araneae	Araneae	**PRD**
Arhynchobdellida	Hirudinae	*Hirudo sp.*	**PRD**
Coleoptera	Amphizoidae	Amphizoidae	**PRD**
	Dytiscidae	*Yola sp.*	**PRD**
	Elmidae	Elminae	**SCR**
		Larainae	**SHR**
	Gyrinidae	*Dineutus sp.*	**PRD**
		Gyrinus sp.	**PRD**
	Hydraenidae	*Hydraena sp.*	**PRD**
	Hydrophilidae	Hydrohilinae	**PRD**
		Hydrophilus sp.	**PRD**
	Noteridae	Noteridae	**PRD**
	Scirtidae	*Elodes sp.*	**SCR**
Collembola	Collembola	Collembola	**CG**
Decapoda	Atyidae	Atyidae	**CG**
	Cambaridae	*Cambarus sp.*	**CG**
	Potamonautidae	*Potamonaute sp.*	**SHR**
Diptera	Athericidae	Athericidae	**PRD**
	Ceratopogonidae	*Bezzia sp.*	**PRD**
		Culicoides	**PRD**
	Chironomidae	*Chironomus sp.*	**CG**
		Orthocladiinae	**CG**
		Tanypodinae	**PRD**
	Muscidae	*Musca sp.*	**CG**
	Nepidae	Nepidae	**PRD**
	Psychodidae	Psychodidae	**CG**
	Simuliidae	*Simulium sp.*	**CF**
	Tabanidae	*Tabanus sp.*	**PRD**
	Tanyderidae	Tanyderidae	**CG**
	Tipulidae	*Antocha sp.*	**SHR**
		Hexatoma sp.	**PRD**
		Limonia sp.	**SHR**
		Tipula sp.	**SHR**
Ephemeroptera	Baetidae	*Acanthiops sp.*	**SHR**
		Baetis sp	**SCR**
		Centroptiloides	**CG**
		Tsitsa sp.	**CG**

	Caenidae	*Afrocaenis sp*	**CG**
		Caenis sp	**CG**
	Ephemerythidae	*Ephemerythus sp.*	**CG**
	Heptageniidae	*Afronurus sp.*	**SCR**
	Leptophlebiidae	*Adenophlebia sp*	**CG**
		Euthraulus sp.	**SCR**
		Leptophlebia sp.	**CG**
	Oligoneuriidae	*Oligoneuriopsis sp.*	**CF**
	Polymitarcyidae	*Povila sp.*	**CG**
	Tricorythidae	*Dicercomyzon sp.*	**CG**
		Tricorythus sp.	**CG**
Hemiptera	Belostomatidae	*Belostoma sp.*	**PRD**
	Corixidae	*Corixa sp.*	**PRD**
		Micronecta sp.	**PRD**
	Gerridae	*Eurymetra sp.*	**PRD**
		Gerris sp.	**PRD**
		Metrobates sp.	**PRD**
	Hydrometridae	*Hydrometra sp.*	**PRD**
	Mesoveliidae	*Mesovelia sp.*	**PRD**
	Naucoridae	Naucoridae	**PRD**
	Nepidae	Nepidae	**PRD**
	Notonectidae	*Notonecta sp.*	**PRD**
		Plea sp.	**PRD**
Lepidoptera	Crambidae	Crambidae	**SHR**
		Parapoynx sp.	**SHR**
		Synclita sp.	**SHR**
Mollusca	Lymnaeidae	Lymnaeidae	**SCR**
	Planorbidae	*Planorbis sp.*	**SCR**
	Sphaeriidae	*Pisidium sp.*	**CF**
	Thiaridae	Thiaridae	**SCR**
Odonata	Aeshinidae	*Aeshna sp.*	**PRD**
	Chlorolestidae	*Chlorolestidae*	**PRD**
	Corduliidae	*Cordulia sp.*	**PRD**
		Corduliidae	**PRD**
		Phyllomacromia sp.	**PRD**
	Gomphidae	*Gomphus sp.*	**PRD**
	Lestidae	*Lestes sp.*	**PRD**
	Libelluliidae	Libelluliidae	**PRD**
	Protoneuridae	Protoneuridae	**PRD**
Oligochaeta	Lumbriculidae	*Lumbricus sp.*	**CG**
	Tubificidae	*Tubifex sp.*	**CG**
Plecoptera	Perlidae	*Neoperla sp.*	**PRD**
Trichoptera	Calamoceratidae	*Anisocentropus sp.*	**SHR**
	Ecnomidae	*Ecnomus sp.*	**CF**
	Hydropsychidae	*Cheumatopsyche sp.*	**PRD**
		Hydropsyche sp.	**CF**
	Hydroptilidae	Hydroptilidae	**SCR**

	Lepidostomatidae	*Lepidostoma sp.*	**SHR**
	Leptoceridae	Atheripceides	**SHR**
		Leptocerus sp	**SHR**
		Oecetis sp	**PRD**
	Philopotamidae	*Chimarra sp.*	**CF**
	Pisuliidae	*Pisulia sp.*	**SHR**
	Polycentropodidae	*Polycentropus sp.*	**PRD**
Tricladida	Planariidae	*Planaria sp.*	**PRD**

Appendix III: The QHEI form used for qualitative evaluation of sites along the Sosiani-Kipkaren River (Adopted from Rankin, 1995)



QUALITATIVE HABITAT EVALUATION INDEX FIELD SHEET TOTAL SCORE ☐

Stream ____________ RM__________ Date __________ River Code__________

Location ______________ Scorer's Name ______________

1] **SUBSTRATE** (Check ONLY Two Substrate TYPE BOXES: Estimate % or note every type present):

TYPE	POOL	RIFFLE		POOL	RIFFLE	NUMBER OF SUBSTRATES	
BLDER/SLAB(10)	____	____	GRAVEL(7)	____	____	____	4 or More(3)
BOULDER(9)	____	____	SAND(6)	____	____	____	<4 (0)
COBBLE(8)	____	____	BEROCK(5)	____	____		
HARDPAN (4)	____	____	DETRITUS(3)	____	____		
MUCK (2)	____	____	ARTIFICIAL(0)	____	____		
SILT (2)	____	____					

NOTE: (Ignore sludge that originates from point-sources: Score on Natural substrates)

2] **INSTREAM COVER**

TYPE (Check all that apply, ALL 1 except deep pools)				**AMOUNT** (Check on 1 or 2 and average)	
UNDERCUT BANKS	____	DEEP POOLS	____	EXTENSIVE >75% (10)	____
OVERHANGING VEGETATION	____	ROOTWADS	____	MODERATE 25-75%(6)	____
SHALLOWS	____	BOULDERS	____	SPARSE 5-25%(3)	____
ROOTMATS	____	OXBOWS	____	NEARLY ABSENT <5%	____
AQUATIC MACROPHYTES	____	LOGS/WOODY	____		

OMMENTS ______________________

3] **CHANNEL MORPHOLOGY** (CHECK ONLY 1 PER CATEGORY OR CHECK 2 AND AVERAGE)

SINUOSITY		STABILITY		MODIFICATIONS/OTHER			
HIGH(4)	____	HIGH (3)	____	SNAGGING	____	MPOUND	____
MODERATE 3)	____	MODERATE (3)	____	ISLANDS	____	LEVEED	____
LOW (2)	____	LOW (3)	____	CANOPY REMOVAL	____	RELOCATION	____
NONE (1)	____			DREDGING	____	BANK SHAPING	____

4] RIPARIAN ZONE AND BANK EROSION (Check ONE per bank or check 2 and AVERAGE per bank)

RIPARIAN WIDTH	FLOOD PLAIN QUALITY		BANK EROSION
(L R per bank)	**L R (Most prominent per bank)**	CONSERVATION TILLAGE (1)	NONE/LITTLE (3)
WIDE >50m (4)	FOREST, SWAMP (3)	URBAN (0)	MODERATE (2)
MODERATE 10-50m(3)	SHRUB OR OLD FIELD (2)	OPEN PASTURE (O)	HEAVY (1)
NARROW 5-10m(2)	RESIDENTIAL PARK (1)	ROW CROP (0)	
VERY NARROW (1)	FENCED PASTURE (1)		NONE (0)

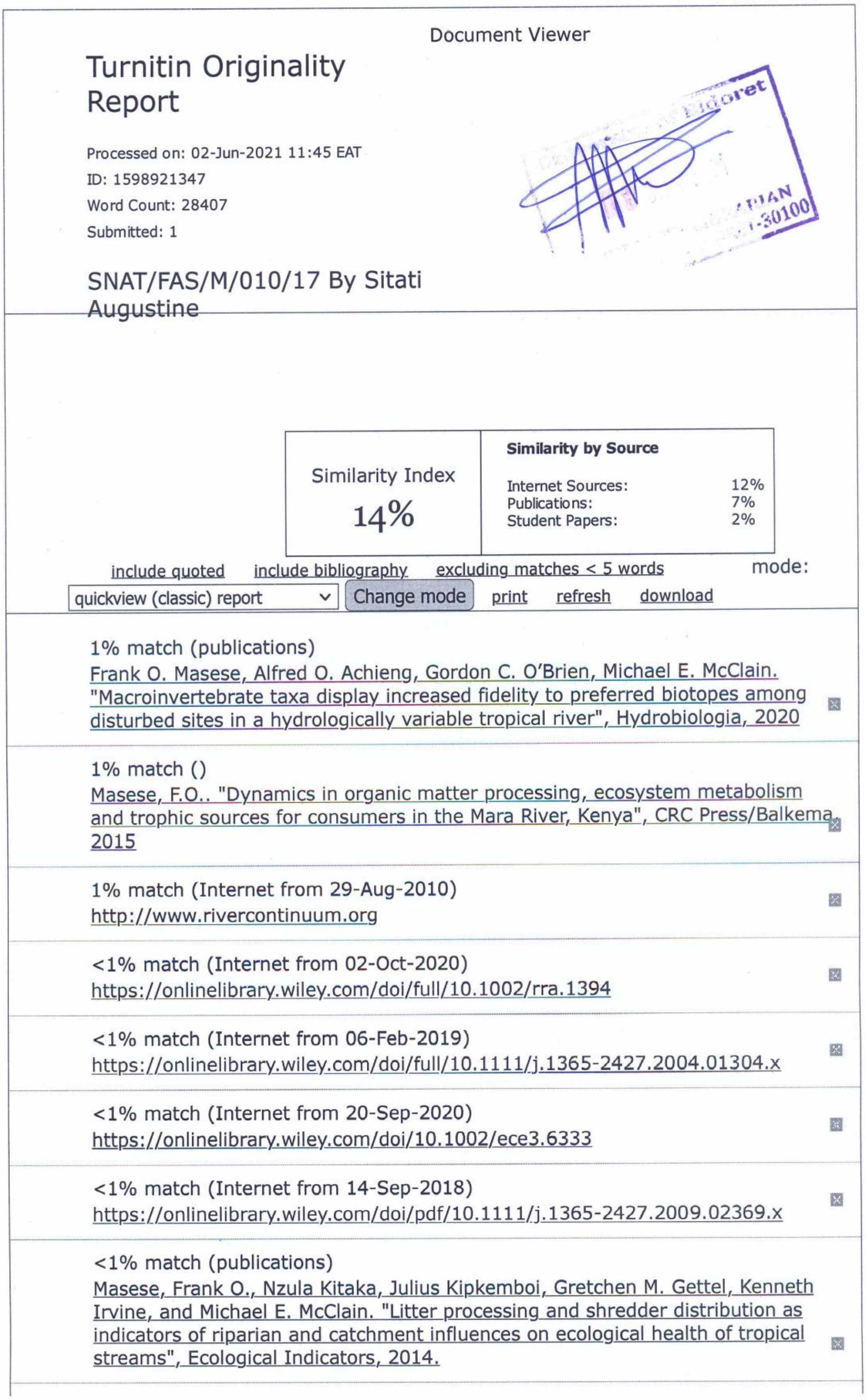
5] POOL/GLIDE AND RIFFLE/RUN QUALITY

MAX DEPTH (Check 1 only)	**MORPHOLOGY**	**CURRENT VELOCITY (POOL $ RIFFLES)**	
	(Check 1or 2 and average)	(Check all that apply)	
>1m (6)	POOL WIDTH> RIFFLE WIDTH(2)	EDDIES(1)	TORRENTIAL(-1)
0.7-1m (2)	POOL WIDTH=RIFFLE WIDTH(1)	FAST(1)	INTERSTITIAL(-1)
0.4-0.7m(2)	POOL WIDTH< RIFFLE WIDTH(0)	MODERATE(1)	INTERMITENT(-2)
0.2-0.4m (1)		LOW(1)	
<0.2m (pool=0)			

RIFFLE/RUN DEPTH (CHECK ONE OR CHECK 2 AND AVERAGE)		RIFFLE/RUN SUBSTRATE	
GENERALLY >50cm(4)	____	STABLE (e.g cobble, boulder) (2)	____
GENERALLY 10-50cm(3)	____	MOD, STABLE (e.g. Large Gravel) (1)	____
GENERALLY 5-10cm(1)	____	UNSTABLE (Fine Gravel, Sand) (0)	____
GENERALLY <5cm(0)	____		

Adopted from Rankin (1995)

Appendix IV: Similarity Report



Turnitin Originality Report

Processed on: 02-Jun-2021 11:45 EAT
ID: 1598921347
Word Count: 28407
Submitted: 1

SNAT/FAS/M/010/17 By Sitati Augustine

Similarity Index	Similarity by Source	
14%	Internet Sources:	12%
	Publications:	7%
	Student Papers:	2%

1% match (publications)
Frank O. Masese, Alfred O. Achieng, Gordon C. O'Brien, Michael E. McClain. "Macroinvertebrate taxa display increased fidelity to preferred biotopes among disturbed sites in a hydrologically variable tropical river", Hydrobiologia, 2020

1% match ()
Masese, F.O.. "Dynamics in organic matter processing, ecosystem metabolism and trophic sources for consumers in the Mara River, Kenya", CRC Press/Balkema, 2015

1% match (Internet from 29-Aug-2010)
http://www.rivercontinuum.org

<1% match (Internet from 02-Oct-2020)
https://onlinelibrary.wiley.com/doi/full/10.1002/rra.1394

<1% match (Internet from 06-Feb-2019)
https://onlinelibrary.wiley.com/doi/full/10.1111/j.1365-2427.2004.01304.x

<1% match (Internet from 20-Sep-2020)
https://onlinelibrary.wiley.com/doi/10.1002/ece3.6333

<1% match (Internet from 14-Sep-2018)
https://onlinelibrary.wiley.com/doi/pdf/10.1111/j.1365-2427.2009.02369.x

<1% match (publications)
Masese, Frank O., Nzula Kitaka, Julius Kipkemboi, Gretchen M. Gettel, Kenneth Irvine, and Michael E. McClain. "Litter processing and shredder distribution as indicators of riparian and catchment influences on ecological health of tropical streams", Ecological Indicators, 2014.