# DECLARATION

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**Declaration by the Supervisors**

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

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University of Eldoret

# DEDICATION

I dedicate this work to you my dear children Cynthia Wanza and Dion Mwema.

You are my inspiration.

God bless you.

## ABSTRACT

Sustainability of culture-based fisheries (CBF) development in small water bodies (SWBs) largely depends on their ecological conditions and productivity. Studies were conducted from November 2010 to July 2012 in Kesses and Kerita dams in Uasin Gishu and Mauna and Yenga dams in Siaya. Sampling for water quality, phytoplankton, macroinvertebrates and fish parasites was conducted once a month. Phytoplankton and macroinvertebrates were collected using plankton and scoop nets respectively. Water quality parameters were measured *in-situ* using electronic meters. Parasitological examination was done according to standard procedures. Significant differences in temperature ($F=17.38$; $p=0.000$), DO ($F=8.76$; $p= 0.000$) and TN ($F= 6.34$; $p=0.01$) were found between Uasin Gishu and Siaya dams. Water pH in Kesses was higher during the wet season ($F=14.44$; $p= 0.000$) while TN and TP were higher during the dry season ($F=9.38$; $p=0.02$) and $F=5.02$; $p=0.023$ respectively). Similarly in Kerita, pH was higher during the wet season ($F=15.98$; $p=0.01$) while TN and TP were high during the dry season ($F=43.21$; $p=0.00$ and $F=20.03$; $p=0.003$ respectively). In Siaya, DO was higher during the rainy season ($F=6.60$; $p= 0.01$). During the wet season, Bacillariophyceae was more abundant in Kerita dam ($H=5.45$; $p=0.02$) and Desmidiaceae ($H=4.07$; $p=0.044$) in Kesses dam. The relative abundance of Ephemeroptera ($H=16.07$; $p=0.003$) and Plecoptera ($H= 16.07$; $p=0.000$) was higher during the wet season in Kesses while Hemiptera ($H= 16.07$; $p=0.000$) and Pulmonata ($H= 9.04$; $p=0.002$) was high during the dry season. *Diplostomum* parasites were associated with high TP levels in Kesses ($rs= 0.621$; $p=0.0001$) and with temperature ($rs= 0.63$; $p= 0.002$) in Mauna. Parasites affected the condition factor depending on seasonality and sex of fish. In Kesses, the condition factor of fish infected with *Tylodelphys* during the dry season was found to be significantly higher than un-infected ones ($F= 18.86$; $p=0.000$). While it is evident that seasonality and geographical location based on altitude have an effect on some aspects of water quality, phytoplankton, macroinvertebrates and parasites of fish in these SWBs, they are considered suitable for fish production as they are rich in phytoplankton and macroinvertebrate communities which serve as food. Development of mitigation measures against habitat degradation and water quality deterioration combined with strong community support for sustainable culture-based fisheries in these SWBs are recommended.

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# LIST OF ABBREVIATIONS, ACRONYMS AND SYMBOLS

| | |
|---|---|
| **ANOVA** | Analysis of Variance |
| **APHA** | American Public Health Association |
| **BMP** | Best Management Practices |
| **CBF** | Culture-Based Fisheries |
| **CCA** | Canonical Correspondence Analysis |
| **Cm** | Centimeter |
| **DO** | Dissolved Oxygen |
| **EIA** | Environmental Impact Assessment |
| **EPT** | Ephemeroptera, Plecoptera, Trichoptera grouped together |
| **ERA** | Environmental Risk Assessment |
| **FAO** | Food and Agriculture Organization |
| **GPS** | Geographical Positioning System |
| **IEE** | Initial Environmental Examination |
| **LVB** | Lake Victoria Basin |
| **MA** | Mean Abundance |
| **MDG** | Millennium Development Goal |
| **MI** | Mean Intensity |
| **PCA** | Principal Component Analysis |
| **RA** | Relative Abundance |
| **SEM** | Standard error of mean |
| **SWB** | Small Water Body |

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# CHAPTER ONE

# INTRODUCTION

## 1.1 Background

The United Nations Millennium Declaration of September 2000 made a pledge which turned into the eight Millennium Development Goals (MDGs) for 2015. The first of these goals which states "to eradicate extreme poverty and hunger" (UNDP, 2014), is becoming more and more elusive as hunger and poverty continue to threaten the development agenda in the developing world. The impacts of climate change on agricultural production, increasing competition for land, water and energy and the need to maintain regulatory environmental services only serve to make the challenge for feeding current and future populations harder. The pressures on the global food system requires a focused, coherent response that links the food sector with major strands of public policy; leading to concerns for food security development agenda (Allison, 2011). As the world population continues to grow, capture fisheries will continue to stagnate as demand for fisheries products increases.

It is expected that fisheries and aquaculture will continue to provide food and income for many developing countries and close to two thirds of global food consumption by 2030. Changes in global fish production and consumption indicate that fish consumption in the developing world has increased over tenfold between 1981 and 1997 (De Silva *et al.*, 2006). Consequently, there has been an increasing emphasis on the development of inland fisheries as a significant contributor in narrowing the growing gap between supply and demand for food fish (De Silva, 2003). While projections indicate that fastest supply growth is likely to come from tilapia, carp and catfish; global tilapia production is expected to almost double between 2010 and 2030 (World Bank, 2013). The fisheries sector acts as an economic multiplier in marginal rural areas where increased fisheries development from small reservoirs has been identified as an important means to address many national policy objectives including food security, economic empowerment, optimal economic benefit from water and poverty. As a result, development of small reservoir fisheries as an important resource for small rural communities, particularly those living close to water bodies has recently been identified by the Organization of African Unity as a priority investment area for food security, poverty alleviation and regional economic development (Richardson *et al.*, 2009).

Culture-based fisheries (CBF) is a form of extensive aquaculture conducted in small water bodies (SWBs) generally less than 100ha and includes small water reservoirs, dams or lakes of not more than 10km$^2$, large ponds, canals, swamps which may be permanent or temporary, natural or man-made (De Silva *et al.*, 2006). This form of fisheries has received a lot of attention in recent times as an alternative for increased fish production. The contribution of SWBs to fish production has been on the rise as a result of cage fish culture in these water bodies and also because of the fact that water bodies built for other purposes other than fisheries or aquaculture can be used for culture-based fisheries. A major difference between culture-based fisheries and traditional stock enhancement practices is stock ownership in the former as opposed to open access in the latter.

Ownership of the culture-based fishery stock and the need for some degree of caring of the stock makes it a form of aquaculture in accordance with the Food and Agriculture Organization of the United Nations' (FAO) definition of aquaculture (De Silva *et al.*, 2006).

Although small water bodies (SWBs) have a high potential for aquaculture production especially from cage fish culture, they remain relatively under-utilized in this respect (FAO, 1994). Due to size, proximity to local communities and ease of improved management, SWBs have high potential for enhanced aquaculture production, with the success of fish production depending on a number of factors. Globally, 60 percent of the global land mass is considered too dry for perennial small water bodies and would negatively impact fish production (high risk), about 11 percent could support small water bodies with cost-adding modifications (moderately high risk) while some 23 percent would have ample water (very low risk) (FAO, 1994).

Since limnological characteristics affect the recruitment and survival of fish, physical and biological aspects of the water bodies as well as socio-economic conditions of communities in the vicinity, potential practitioners and primary stakeholders are key factors that influence suitability of water bodies for culture based fisheries (De Silva *et al.*, 2006). The role of environment on fishery recruitment of most water bodies must be understood for the potential of small water bodies to be exploited sustainably for aquaculture production. Ownership in terms of the authorities responsible for both environmental protection and fishery management and stakeholders should be clearly defined. The reservoir should be accessible to facilitate movement of personnel and materials to and from the reservoir. Reservoirs of 300 -700 ha are easier for fisheries development and their impact is more noticeable than larger ones. Reservoirs that are selected for fisheries enhancement should have macrophyte coverage in the range of 25-30% as excessive plant coverage hinders netting and impinges upon ecosystem productivity (Cowx, 1999).

The water temperature should be conducive to the feeding and growth of fish as at lower temperatures, fish stop feeding and growth is slowed down (Escalera-Vazquez and Zambrano, 2010). Extremely high water transparency is an indication of low primary production and should be avoided (Vass *et al*., 2009). Too high or too low little dissolved oxygen is unfavorable for fish. According to Mwaura (2006), the pH of the water should be mildly alkaline as acidic or highly alkaline pH affects fish growth. The concentrations of total alkalinity, nitrates and phosphates impinge upon primary production and the density of plankton, which form the base of the food chain. Nutrient levels should be reasonable as too little or too much is detrimental to fish. Natural food items that form the food chain should be established so as to identify vacant food niches that can be exploited with management interventions for fish production.

The type of fish in the ecosystem should be determined as it is essential for guiding the enhancement intervention. Fish should be stocked in environments suitable for their sustenance and growth by being efficient in utilizing natural food. Although fish species that feed low in the food chain are preferred, they should also offer good eating, economic value and potential for marketing. The sustainability and conservation of indigenous species, especially ecologically sensitive species should not be endangered by enhanced production in reservoirs (Vass *et al.*, 2009). According to Mustapha (2011), shallow tropical reservoirs are characteristically eutrophic, do not thermally stratify, are polymictic with short but varying water residence time especially in the dry season and possess high watershed /water body area, high shoreline development index with high level of water fluctuations due to seasonal influences. They are also high in phytoplankton and zooplankton which are food for fish. Although their assemblage is influenced by several abiotic and biotic factors, they have high fish biodiversity due to abundance of food, shallow depth and high productivity. Aquatic macrophytes are conspicuous features of such reservoirs; providing refuge for zooplankton against predation by fish, change the nutrient dynamics and prevent resuspension of sediments. As a result, they tend have high potential fish yield; with estimates of up to125.72kg/ha.

Some of the problems associated with African reservoirs include thermal stratification and turbidity (Mustapha 2011). In the dry season, thermal stratification may occur in the reservoirs and could result in deoxygenation of the hypolimnion; reducing water quality and consequently decreasing fish stocks. Turbidity can create severe problems of fertility, productivity and pollution for the fish and water body. In addition to these problems, invasion and explosiveness of aquatic macrophytes as a result of eutrophication also produces severe negative impacts on fish production. Furthermore, these reservoirs critically exhibit longitudinal (up lake and down lake) gradients in turbidity, nutrient concentration, mixing depth, euphotic depth, with sufficient light for photosynthesis, flushing rates, chlorophyll concentration, plankton productivity, fish standing stocks, macrophytes abundance, benthic community structure and other limnological and biological variables and are classified as river-lake hybrids due to their characteristic riverine-lacustrine environments (Mustapha 2011).

According to Mustapha (2011), African reservoirs are often made up of three zones namely the lotic (riverine zone), lentic (reservoir zone) and the transitional zone which is found between the riverine and reservoir zones. Each of the zones is characterized by different transparency, different causes of light attenuation, different nutrient regimes and different biota. Transparency is low in the riverine section as a result of washing of debris into it by flood, while light attenuation is high in the lacustrine and transition zones due to shallow nature and high productivity of the zones. The transition zone is also the area with high nutrient and diverse fish and plankton assemblages. Building and utilization of nutrients; known as bottom-up and top-down mechanisms as well as seasonality occur between phytoplankton, zooplankton and fish components; termed as trophic cascade of these reservoirs. Shallow reservoirs are sensitive to eutrophication due to intense exchange of nutrients between water column and sediments.

The use of existing, perennial and non-perennial water bodies for culture- based fisheries is mostly done communally. Some of the advantages of culture-based fisheries include that they are more often less capital intensive, require limited technical expertise and not labor intensive, often environmentally friendly and tend to involve women and youth (De Silva *et al*., 2006; Erina, 2010). Whilst stocking has the potential to yield substantial benefits, the actual outcomes in terms of production, distribution of benefits and sustainability can be influenced adversely. Limited prior knowledge of physical, biological, technical and institutional characteristics of individual sites; complexities involving dynamic interactions between the biological characteristics of the resource; technical interventions and the people involved are some underlying reasons for unexpected and sometimes undesirable outcomes of stocking in small water bodies (Lorenzen and Garaway, 1998; Garaway and Lorenzen, 2001).

Small water bodies cover 2.5% of Kenya's total area; with fish production from estimated at 2000kg/ha for natural systems and 9000kg/ha for intensive ones (Mwaura, 2006). The Lake Victoria Basin (LVB) is one of the country's regions that is endowed with numerous perennial rivers that drain into the lake. The Basin lies between altitudes of 900 and 1800 meters above sea level and is located in the upper reaches of the Nile basin. The lake is shared by Kenya, Uganda and Tanzania as well as Rwanda and Burundi which form part of its drainage basin. In Kenya, the catchment area of the lake covers the entire former Nyanza and Western provinces and drains extensive sections of the eastern slopes of the Rift Valley, an area that extends from Cherangani hills to Mau forest and includes Masai Mara game reserve in the Rift Valley province. It covers an area of approx. 251,000km$^2$; of which 69,000km$^2$ is the lake itself which is shared by Kenya, Uganda and Tanzania. Lake Victoria is the largest freshwater lake in Africa and the second largest in the world. Over 80% of Lake Victoria waters are from direct precipitation in the riparian areas. In Kenya, the main influent rivers are Sio, Nzoia, Yala, Nyando, Sondu Miriu, Kuja, Migori and Mara River (Raburu *et al.*, 2012). Combined, these rivers contribute much more to the lake waters than the largest single influent; river Kagera in Tanzania. The Lake outflows through White Nile and Katonga; both of which are part of the upper Nile (http://www.britannica.com/, 2013). Along the river drainage basins are important natural small lakes and swamps and man-made reservoirs and dams built during the pre-independence era and stocked with various fish species. High population in the basin has inevitably led to high eutrophication in Lake Victoria, land degradation, loss of biodiversity, disease and malnutrition. Fish catches from Lake Victoria have been on a sharp decline. In view of this, focus on small water bodies to boost fish production for food and generation of income is expected to rise.

## 1.2 Problem Statement

In the wake of this economic volatility, the question of how to produce and distribute enough food for a projected global population of 9 billion people in 2050 has become a central concern (Allison,2011). Small-scale aquaculture has great potential for increasing production of valuable protein and poverty alleviation for rural communities in Sub-Saharan Africa. In Kenya, fisheries development in small water bodies presents an opportunity to increase fish production for food and income to communities. For sustainable fisheries development, the ecological conditions and productivity of the water bodies should be understood as they impact on fish production. While limnological characteristics and water retention affect the recruitment and survival of fish in that ecosystem, the role of environment on fishery recruitment and production of most water bodies is largely unknown. For this reason, the physico-chemical parameters associated with these water bodies should be established to assess their overall suitability for fish production.

While other aspects of fish ecology have been fairly well studied, information on parasites remains scanty in Eastern Africa compared to Western, Central and Southern Africa (Aloo, 2002). It is a well-known fact that development of reservoir fisheries has been associated with transmission of parasites from wild to cage fish; with introduction of large numbers of caged fish to a system having dramatic effects on disease agents and often causes severe infestation and heavy mortalities (Knaap, 1994). Fish diseases and parasites have economic consequences associated with mortality, rejection of infected fish, retarded growth and weight losses of the infected fish. Furthermore, some fish parasites are potentially pathogenic to man when the parasitized fish are consumed not well cooked, cold smoked, marinated or just raw (Florio *et al.*, 2009).

## 1.3 Justification

Studies on tilapia farming systems in reservoirs and temporary water bodies in Eastern Africa indicate that both wild and farmed fish are vulnerable to parasitic infections (Florio *et al.*, 2009). However, the mechanisms of spread of these diseases remain poorly understood compared with other aspects of fish ecology in Kenya (Aloo, 2002). Although studies on small water bodies in Lake Victoria basin, Kenya have been conducted (Maithya, 1998; 2008; Maithya *et al.*, 2012; Ngodhe *et al.*, 2013) they have not addressed the aspect of fish parasites. Compared to other aspects of fish ecology, information on parasites remains scanty in eastern Africa (Aloo, 2002). It is a well-known fact that development of reservoir fisheries has been associated with transmission of parasites (Florio *et al.*, 2009). However, the mechanisms of spread of these diseases remain poorly understood compared with other aspects of fish ecology in Kenya (Aloo, 2002). Apart from studies on influence of water quality on parasite prevalence levels in small water bodies in Uasin Gishu (Ochieng *et al.*, 2012), there is little or no other work that has been done on small water bodies in Lake Victoria basin. There is need fro information on fish parasites of tilapia, which is the most commonly cultured fish in the region; considering that some pose a risk of being zoonotic to consumers.

Since physical factors are not easily controlled, selection of sites for aquaculture should be conducted in harmony with existing climatic and geologic regimes. Among the most important weather/climatic variables is air temperature; which is mostly moderated by elevation; with mountainous areas in the tropics having a relatively cool climate. In tropical climates, water temperatures are high all year, but difference in air temperature between wet and dry seasons may affect them.

There is need to fill information gaps on ecological aspects and parasites affecting tilapia (*Oreochromis niloticus*) production in small water bodies. This study therefore conducted to provide information on the role of seasonal variations on physical and chemical aspects of water quality, biological variables and consequently on aquaculture production in small water bodies in Uasin Gishu and Siaya

## 1.4 Objectives of the study

### 1.4.1 Overall Objective

The overall objective of this study was to assess the ecological suitability of small water bodies in Uasin Gishu and Siaya Counties for culture-based fisheries and the parasitological status of tilapia (Orechromis niloticus)found in the same systems.

### 1.4.2 Specific Objectives

1. To determine selected water quality physico-chemical parameters and nutrient levels in the SWBs in Uasin Gishu and Siaya counties.
2. To estimate composition, abundance and estimate phytoplankton primary productivity in the SWBs.
3. To determine the composition, abundance and biomass of macroinvertebrates in the SWBs
4. To identify fish parasites and establish their prevalence and mean intensity levels in *O. niloticus* in SWBs in Siaya and Uasin Gishu.
5. To determine the effect of seasonality on water quality, phytoplankton, macro-invertebrates and fish parasites in the SWBs in Uasin Gishu and Siaya counties.

## 1.5 Hypotheses

$H_o$: There is no difference in water quality physic-chemical parameters and nutrient levels between SWBs in Uasin Gishu and Siaya counties.

$H_o$: There is no difference in composition, abundance and primary productivity of phytoplankton in SWBs in Uasin Gishu and Siaya counties.

$H_o$: There is no difference in composition, abundance and biomass of macroinvertebrates in SWBs in Uasin Gishu and Siaya counties

$H_o$: There is no difference in prevalence and intensity of parasites of *Oreochromis niloticus* in SWBs in Siaya and Uasin Gishu.

$H_o$: Water quality has no effect on abundance of phytoplankton, macro-invertebrates and intensity of fish parasites in SWBs in in Uasin Gishu and Siaya counties.

$H_o$: Seasonality has no effect on water quality, phytoplankton, macro-invertebrates and fish parasites in the SWBs

# CHAPTER TWO

# LITERATURE REVIEW

## 2.1 Overview

Since the Roman times, reservoirs were popular for water supply in the Mediterranean region and were introduced in Western Europe in the Late Middle Ages (Mwaura, 2006). They have since served different purposes worldwide; with 48% used for irrigation, 20% for hydropower generation while others are used for flood control, recreation, fisheries, and receptacles for treated municipal water, domestic and industrial water supply

Culture- based fisheries (CBF) in reservoirs has received a lot of attention as an effective way of increasing fish supplies while providing additional income especially in rural areas, therefore contributing to poverty alleviation (Lorenzen and Garaway,1998). Culture based fisheries has been successful in Sri Lanka (Wijenayake *et al.* 2005); Vietnam (Nguyen *et al.* 2005) and Thailand (Garaway and Lorenzen 2001). Most culture-based fisheries that have been effective in stocking fish that were either absent before enhancement, or their abundance had been reduced by overfishing. Chinese reservoirs have been associated with remarkable success of culture-based fisheries with yields increasing from 150 to 750 kg/ha/yr; mainly attributed to stocking of species that are able to make good use of available food resources. The success of culture-based fisheries is has been linked to ecosystem productivity (De Silva *et al.*, 1991). Some of the studies that have been conducted have focused on the influence of trophic levels and stocking density on fish yield in small water bodies (Lorenzen and Garaway, 1998).

Appropriate selection of suitable sites is a necessity for success and sustainability of fisheries in water bodies and for reducing potential conflicts between aquaculture and land usage. To qualify for aquaculture, sites must be of a reasonable size, more or less permanent water sources of good quality water. Ownership is an important requirement in selection of small water bodies for fishery enhancement. Stakeholder definition for the purpose of management and environmental protection is important, with all concerned parties giving consent for any proposed development that entails intervening in the ecosystem (Vass *et al.*, 2009).

## 2.2 Water quality

### 2.2.1 Physico-chemical Parameters

The various chemicals dissolved in the water, as well as the temperature and other physical attributes of water, all combine to form what is referred to as water quality and can strongly influence fish production (Diana *et al.* 1997; Mwaura, 2006).

One of the most dynamic attributes of water bodies is dissolved oxygen (DO) which is influenced by different environmental factors such as temperature, elevation and chemical nature of bottom sediments. Dissolved oxygen is a very vital indicator of the ability of water to support aquatic life as it plays a big role in survival of aquatic life. Sources of DO in water are absorption from atmosphere or from photosynthetic aquatic plants and algae. Solubility of oxygen in water depends on a number of factors. The amount of oxygen dissolved in a water body is affected by salinity, altitude, groundwater inflow, and water temperature. Of these, temperature most directly affects DO in water bodies, with high temperature resulting in low dissolved oxygen levels (Addy and Green 1997).

Although not usually a concern in many freshwater bodies, salinity can affect oxygen solubility in estuaries, brackish waters, bogs and water bodies in agricultural areas. Groundwater, especially if it does not come into contact with the atmosphere, typically has lower levels of DO than surface waters (Mwaura, 2006). When groundwater enters a lake, DO concentrations are initially reduced near the spring. However, groundwater is generally colder than surface waters and can therefore hold more oxygen. Due to the temperature-dependent attribute of water density, warm water which is of lower density remains at the surface forming the epilimnion while cooler, denser water sinks to the bottom to form the hypolimnion. A layer of rapid temperature change known as thermocline separates the two. As organic matter load in the hypolimnion increases, the hypolimnion becomes depleted of oxygen due to decomposition. In nutrient rich water bodies, the process is enhanced by increased microbial activity and is often the cause of oxygen depletion in the hot season.

Dissolved oxygen plays an important role in the survival of aquatic life. Most aerobic life in water has DO requirements of $8mgl^{-1}$, which is hard to maintain due to increasing pollution and eutrophication (Templer *et al.*, 1998). Most aquatic organisms may not survive prolonged low dissolved oxygen levels of less than $2mgl^{-1}$. Dissolved oxygen levels of less than 5 to $6mgl^{-1}$ may cause stress and increase susceptibility of aquatic animals to infection by opportunistic pathogens. Total depletion of oxygen gives rise to anaerobic conditions; often characterized by the smell of rotten eggs as a result of anaerobic activity of microorganisms on nitrates and sulfates. Dissolved oxygen affects biogeochemical processes such as nitrogen cycle and ultimately the productivity of aquatic ecosystems. Studies by Mwaura (2006) indicate that dissolved oxygen levels in small water bodies in Kenya range between 2.7 to 7.2mgl$^{-1}$; with high levels attributed to elevation and high photosynthetic activity of algae and submerged macrophytes such as *Ceratophyllum demersum* while low dissolved oxygen levels are associated with limited turbulence during the dry season, low atmospheric temperature, temperature stratification and high organic matter load. The effect of altitude on dissolved oxygen levels is as a result of partial pressure and degree of saturation of oxygen; with dissolved oxygen levels decreasing with altitude due to decreased relative pressure (Magagula *et al*. 2010). Water is often associated with higher levels of dissolved oxygen. On the other hand, slow moving or stagnant water with high loads of suspended solids, nutrient levels and increased turbidity could result in decreased dissolved oxygen.

Temperature probably has the greatest influence on growth, development, health, distribution and survival of fish. Fish community structure depends on biotic interactions and abiotic variables; with the latter playing an important role in highly variable freshwater systems such as tropical wetlands that exhibit marked seasonal hydroperiod (dry and wet seasons) resulting in modification in water quality and quantity; often resulting in differential survival of fish and consequently affecting species abundance and richness. Specific conductance or electrical conductance; which is water’s capacity to conduct electric currents and is related to the concentration of free ions and water temperature, provides a quick, convenient estimate of the ionic content as well as ionic pollutants (Escalera-Vazquez and Zambrano 2010).

Water pH has been referred to as the master variable in the chemistry of aquatic systems (Vass *et al*. 2009).While most water bodies have pH range from 6 to 9, Kenyan reservoirs have pH ranges from 5-10; with variations attributed to wet and dry conditions; particularly in arid and semi-arid areas where high pH is attributed to elevated temperatures leading to high evaporative rates in the dry season (Mwaura, 2006). Closed drainage basins or endorheic systems are also associated with high pH while changes in photosynthetic intensity contribute to fluctuations in pH. Geology, soils as well as presence of lacustrine sediments in the vicinity of the water body were also found to contribute to high pH. Low pH was attributed to strong buffering effect by both catchment vegetation and riparian macrophytes against material import especially soil; further emphasizing the influence of human activity on water quality deterioration. Low pH of 4 to 5 influences biodiversity by favoring or discouraging the presence of certain forms of life. Below 5, pH can severely reduce aquatic species diversity. Water pH should be mildly alkaline as acidic or highly alkaline pH affects fish growth; with persistent pH between 4 and 6 inhibiting fish reproduction. Turbulent upwelling is an important feature for shallow water bodies as it enables water mixing and prevents chemical stratification.

Availability of water is the first most important requisite for aquaculture development and is largely depend on depth and surface area of the water body. Shallow, tropical water bodies are often characterized by homogeneity in physico-chemical parameters due to complete mixing within the water column (Mkare *et al*., 2010). A saucer-shaped bottom is most preferred and should have should have a depth of at least 4-5 meters to ensure good fish growth. The depth of a water body indicates environmental health and ecological productivity of a water body. Particulate matter is reflected by levels of water transparency and is often influenced by amount of algae, particulate and dissolved matter in water. Secchi disc depth (SDD) is, therefore, a good indicator of environmental health and ecological productivity, depths of less than 1m indicating heavy sediment and nutrient loads Very high transparency indicates low plankton populations, reflecting low primary production (Mwaura, 2006; Vass *et al.* 2009).

The surface area of the water body is an important determinant of fish yield (Wijenayake *et al*. 2005). A minimum surface area of 3 hectares is recommended and should smaller reservoirs be used, they would require aeration and waste removal options (Maleri 2009). Although it has a direct influence on amount of oxygen, the surface area alone does not determine levels of oxygen but rather a combination of factors such as nutrient load, stocking density of fish and ammonium nitrogen levels.

While as some water characteristics such as dissolved mineral content, pH, alkalinity, and hardness are strongly influenced by water source and soil type, geological and climatic conditions of the watershed region also play an important role (Addy and Green 1997). Human and natural activities shape the physico-chemical parameters of water such as pH and water temperature and determine the occurrence distribution of aquatic organisms; with biological components being confined to parts of the water where chemical and physical conditions are conducive for them. Phosphorus and nitrogen from agricultural farms may enter water through runoff and affect oxygen concentration, resulting in over-fertilization of the water and subsequently excess aquatic plant growth. When these plants die and decay they consume and reduce oxygen levels in the aquatic environment (Addy and Green 1997; Masese *et al*., 2009a). Agricultural practice has been demontrated to influence nitrogen concentration, alkalinity and total dissolved solids (TDS). Land use practice has also been associated with phosphorus and nitrogen levels in water. Human activities such as deforestation and cultivation may contribute to high water temperature, conductivity, total dissolved solids and turbidity (Johnson *et al*. 1997). Removal of trees and plants that grow along the edge of streams and rivers decreases shading and lowers dissolved oxygen levels due to warmer water temperature. Therefore, although access to water bodies is an important consideration for their selection for culture-based fisheries, locations close to human settlements could potentially influence water quality due to direct disposal of untreated domestic sewage (Mwaura, 2006).

### 2.2.2 Nutrients

Availability and amount of nutrients plays a major role in the chemistry of aquatic systems as well as on phytoplankton productivity and composition in water bodies (Kalff 2002). Nutrients such as phosphorus and nitrogen are often in short supply and limit primary productivity in aquatic systems. Supply of nutrients in small water bodies influences species composition and abundance of phytoplankton. Consequently, the dissolved oxygen levels and aquatic animals are also affected (Harper 1992). Total phosphorus in water consists of particulate and dissolved phases; both consisting of several components. A large proportion of phosphorus in freshwater systems occurs as organic phosphates and cellular constituents in the biota are adsorbed to inorganic and dead particulate organic materials. Particulate phosphorus consists of phosphorus in organisms, mineral phases of rocks and soil and phosphorus adsorbed into dead particulate organic matter. There are other processes and organisms that influence availability and resupply of dissolved nutrients (Trommer *et al*., 2013). As such, their ratios are not sufficient to establish actual nutrient limitations that affect natural phytoplankton populations. Therefore the influence of other factors such as food web composition and interactions and nutrient recycling in nutrient limitation patterns should be considered.

Nitrogen and phosphorus are the most important nutrient factors causing high productivity in water bodies. Limitation of any one of phosphorus, nitrogen or carbon; found in plant organic matter of aquatic algae and macrophytes results in excess amounts of the other two nutrients; with contribution of phosphorus, nitrogen and carbon being 500, 71 and 12 times its weight respectively in living algae. Phosphurus and nitrogen are usually the first to impose limitation on freshwater sytem; with phosphorus regulating productivity of most freshwater systems (Schindler *et al*., 1987). On the other hand, high levels of nitrates and phosphates in water bodies may contribute to the process of eutrophication (Swierk and Szpakowska 2011).

Dissolved phosphorus on the other hand is composed of orthophosphate, polyphosphates and low-molecular weight phosphate esters (Boyd, 1990). In most cases, total phosphates refers to inorganic soluble phosphorus or orthophosphate. Most uncontaminated surface waters have total phosphorus concentrations of between 10 – 50 µgp $l^{-1}$. A large proportion of phosphorus in freshwater systems occurs as organic phosphates and cellular constituents in the biota are adsorbed to inorganic and dead particulate organic materials. Most of the phosphorus delivered to lakes is potentially unable to stimulate the production of epilimnetic phytoplankton because much of it accumulates in the water column and not in the epilimnion or metalimnion. Furthermore, direct uptake particularly by cyprinids; which consume part of the aquaculture waste from the cages, further reduces the amount of phosphorus for phytoplankton (Bristow *et al.* 2008).

Variation in phosphorus concentrations in aquatic systems is attributed to several factors. The geographical characteristics of an area influence concentrations; being generally higher in mountainous regions of crystalline bedrock and increase in lowland areas derived from sedimentary rock deposits. Lakes rich in high organic matter tend to exhibit high total phosphorus concentrations. With a few exceptions, the amount of phosphorus tends to increase with productivity (Wetzel 1983). The phosphorus content of precipitation is ususally high in heavily fertilized agricultural regions. In temperate regions, variations in nutrient concentrations are attributed to among other factors, accumulation in snowpacks and ice during winter and their release in large amounts during spring.

Surface runnoff in lakes and streams serves as a major source of phosphorus; with quantities varying depending on amount of phosphorus in the soils, topography, vegetative cover, quantity and duration of runoff, land use and pollution systems (Goldman and Horne 1983).The type of parent rock material from which the soils are derived will vary in phosphorus content. Soil composition, based on quantities of organic and inorganic colloids influence the exchange capacity of soils for phosphorus. Phosphorus is most available from soils of pH of 6-7 and combines with aluminium, iron and manganese at lower pH and with calcium at higher pH. Topography of catchment area influences the extent of erosion and subsequent export of nutrients whereby flat lands with little runoff have lower nutrient load from runoff than steeper areas. Erosion is further influenced by type of vegetation and land use practices such as application of fertilizers and land management in agriculture and forestry.

Amounts of phosphorus discharged in surface waters may be influenced by population density of urban areas in surrounding areas. Other factors such as storm drainage, residential fertilization, domestic sewage and industrial inputs are known to contribute to large inputs of phosporus in surface waters. Levels of phosphorus in freshwater systems are also influenced by domestic cleaning detergents that are released to these systems (Goldman and Horne 1983).Hypertrophic reservoirs and shallow lakes serve as sinks for phosphates and nitrates and result in internal loading of these nutrients. High phosphate levels may also result from high temperatures especially during the dry season, contributing to decomposition of organic matter, thus releasing orthophosphate in water. On the other hand, low phosphate and nitrate levels are associated with stripping by floating and emergent macrophytes. Phosphate and nitrate levels in reservoirs have fluctuate seasonally; with high peaks in the rainy season and low levels in the dry season (Kaggwa *et al.* 2011).

Phosphorus, nitrogen and silica coupled with environmental factors such as temperature, light, grazing pressure, morphometric features and water mixing regimes of the water body basin influence phytoplankton species composition and productivity (Fujimoto *et al.*, 1997; Kalff, 2002). Changes in structure and function of phytoplankton communities have been observed as a result of uncontrolled nutrient inflows into small water bodies (Krienitz *et al.*, 2001). In eutrophic systems, phytoplankton growth is limited by nutrients. Phytoplankton assemblages in tropical freshwater systems are known to be shaped by many factors, among them is nutrient levels (Trebisan and Forsberg, 2007; Souza *et al*., 2008). The effect of nutrient dynamics on phytoplankton is more pronounced in the tropics than temperate regions, where nutrient enhancement results in replacement of original phytoplankton inhabitants with chlorophytes (Lueangthuwapranit *et al.*, 2011). Phytoplankton communities exhibit nutrient limitations that are primarily dependent on the temporal and spatial distribution of dissolved nutrients (Trommer *et al.* 2013); with a tendency for freshwater systems being phosphorus limited whereas estuarine and marine systems are more nitrogen limited (Hecky and Kilham 1988).

In Brazil, diatoms and dinoflagelletes have been associated with the presence of nutrient-rich waters (Goncalves-Araujo *et al.* 2012). In oligotrophic tropical waters, phytoplankton are mainly composed of small nanoflagelletes and cyanobacteria; with strong water column stability found to favor the presence of nanoflagelletes (Goncalves-

Araujo *et al*. 2012). Seasonal succession of phytoplankton communities is affected significantly by total phosphorus in lakes (Pereira *et al*., 2011). With global warming, greater impacts are expected on these communities (Cermeno, *et al*. 2013). In Lake Victoria, nutrient levels have been associated with changes in phytoplankton assemblages (Mugidde *et al*., 2003). Changes in nutrient amounts are attributed to increased nutrient loading as a result of high human population and subsequent land degradation in the lake's catchment (Hecky 1993). As a result of increased burial of diatoms in the sediments, silica concentration have reduced and has led to a shift from dominance by diatoms to blue green algae in the lake. It is anticipated that this shift will have a direct effect on the lake's water quality due to production of cyanobacterial toxins by some these algae (Krienitz *et al.* 2001; (Vershuren *et al*., 2002).

Among the nutrients, dissolved reactive phosphorus concentrations have significant implications for nutrient dynamics and phytoplankton composition and hence on fish production systems (Mkare *et al*.; 2010). Although addition of phosphorus to lakes or ponds results in rapid increase in algal productivity, this productivity is not sustained due to losses in colloidal fraction and sedimentation of particulate phosphorus(Kaggwa *et al.*, 2011) Therefore to sustain increased productivity, inputs of nutrients should be maintained (Goldman and Horne 1983). The effect of cage aquaculture on concentrations of phosphorus, and hypolimnetic dissolved oxygen on freshwater systems vary; with some indicating relatively low levels. Potential ecological impacts associated with phosphorus enrichment by cage aquaculture must therefore be established prior to expansion of the industry in freshwater systems (Bristow *et al*,. 2008). This is especially so in arid and semi-arid areas where the trophic status is related to seasonal fluctuations in the hydrology of the system controlled by rainfall (Chaves *et a*l., 2013).

## 2.3 Phytoplankton Composition, Abundance and Primary Productivity

### 2.3.1 Composition

Composition of phytoplankton communities is as critical in small waterbodies as it is in pond culture because fish that are largely dependent on natural foods are positively influenced by primary productivity (Boyd 1990). Ecosystems with low diversity of phytoplankton communities are usually unstable as a result of massive die-offs and may lead to oxygen depletion and death of fish. Phytoplankton communities exhibit nutrient limitations that are primarily dependent on the temporal and spatial distribution of dissolved nutrients, with a tendency for freshwater systems to be phosphorus limited whereas estuarine and marine systems are nitrogen limited (Hecky and Kilham 1988; Trommer *et al*., 2013). Dissolved reactive phosphorus concentrations have significant implications for nutrient dynamics and phytoplankton composition (Hecky and Kilham; 1988; Mkare *et al.*, 2010).

In the tropics, the effect of tidal variation and nutrient dynamics on phytoplankton is more pronounced than temperate regions. This results in a strong gradient in environmental variability and consequently replacement of original inhabitants with chlorophytes due to nutrient and turbidity enhancement (Lueangthuwapranit *et al.,* 2011). In Brazil nutrient-rich waters have been associated with the presence of diatoms and dinoflagelletes with low biomass; probably as a result of grazing pressure. In contrast, oligotrophic tropical waters are associated with small nanoflagelletes and cyanobacteria; with strong water column stability found to favor the presence of nanoflagelletes (Goncalves-Araujo *et al.*, 2012). In seasonal karstic Irish lakes, seasonal succession of phytoplankton communities has been attributed to seasonal flooding, with total phosphorus (TP) and mean depth/color of lakes being the key determinants of phytoplankton composition (Pereira *et al.*, 2011).

Variation in phytoplankton composition has been associated with seasons, geographical location, variations in salinity and turbidity. Turbid freshwater habitats tend to be rich in chlorophytes, cyanobacteria and euglenophytes, whereas diatoms and dinoflagellates dominate along salinity gradients of clear estuarine environment (Lueangthuwapranit *et al.*, 2011). Geographical location, seasonality and pollutant substances affecting water quality also influence density and richness of phytoplankton species whose assemblages are also known to be shaped by stratification, reduced light, increased nutrients and seasonal and daily temperature variation in tropical freshwater systems (Souza *et al.* 2008; Trebisan and Forsberg, 2007). In seasonal lakes, seasonality, total phosphorus and mean depth of water are the main variables affecting phytoplankton composition. Dynamic hydrological environments of seasonal lakes that are often associated with turbulent, nutrient rich waters that are not phosphorus-limited are often dominated by r-selected phytoplankton such as cryptophytes and diatoms (Pereira *et al*, 2011). Grazing pressure by zooplankton may also contribute to the composition of phytoplankton communities in such environments. Soft, acid waters tend to be loaded with humic substances and result in poor light penetration while hard water tends to be clearer due to coagulation and precipitation of colloids by calcium and magnesium, therefore influening phytoplankton communities(Neustupa *et al.*, 2011).

### 2.3.2 Abundance

Abundance of phytoplankton is an important aspect for fish production particularly in production systems based on natural foods (Boyd 1990).Changes in the abundance of individual species of phytoplankton are common and are attributed to numerous factors including pH, temperature, nutrient concentration, light, weather, diseases, grazing by fish and zooplankton, competition between species, natural algal toxins, climate change and chance(Boyd 1990; Pereira *et al.*, 2011; Cermeno *et al.*, 2013).Seasonality of water quality affects the relative proportions of phytoplankton in different taxa. Even in ponds which have been treated alike, the genera and species of phytoplankton may differ greatly due to reasons that are still unknown. Ponds with high total alkalinity of 68-148mgl$^{-1}$ have high proportions of blue-green algae than green algae. In brackish water, a shift from diatoms to green or blue-green algae has been attributed to the greater role assumed by nitrogen as a limiting factor (Boyd 1990).

Differences in nutrient levels between and even within water bodies exist and have an impact on phytoplankton communities (Maitland, 1990).Uncontrolled inflow of nutrients into the small water bodies could modify the community structure of phytoplankton (Krienitz *et al.,* 2001). Variation in nutrient levels may result in non-linear growth patterns that are distinguished among different genera of algal communities (Reynolds *et al.*, 2001).

In Lake Victoria, changes in phytoplankton abundance have been attributed to differences in nutrients and light availability (Mugidde *et al.*, 2003). Reduced silica levels has been a major factor in the shift from dominance by diatoms to that of blue green in the open waters of the lake due to burial of diatoms in the sediments (Verschuren *et al.*, 2002. It is anticipated that as the blue green algae and associated algal blooms in the lake increase, production of cyanobacterial toxins will continue to affect the lake water quality (Krienitz *et al.*, 2001).

### 2.3.3 Primary Productivity

Primary productivity in aquatic systems consists of the build-up of complex organic compounds by macrophytes, algae and certain bacteria from inorganic compounds such as carbon dioxide ($CO_2$) and water. Phytoplankton are primary producers and form the base of the food chain in most aquatic ecosystems (Marshal *et al*., 2005). The values of primary productivity and plankton density provide a rough estimate of anticipated production in the ecosystem. The concentrations of total alkalinity, nitrates and phosphates impinge upon primary production (Hecky and Kilham, 1988; Mkare *et al*., 2010). While extremely high or low primary productivity and plankton density do not favor fish growth (Vass *et al.*, 2009), high levels of Chlorophyll-α in small water bodies have been associated with significantly high fish yields (Wijenayake *et al.*, 2005).

While as the biomass of phytoplankton is more likely to be nutrient limited, transport time may be more critical to its growth (Townsend *et al.*, 2012).In Lake Victoria Increased nutrient loading into the lake, associated with increased human population in the lake catchment and the resultant land degradation has been blamed for increased eutrophication and changes in phytoplankton productivity and assemblage (Hecky, 1993).

Primary production at lower trophic levels by phytoplankton and zooplankton has a strong influence on fish production levels. The supply of nutrients to water mass determines which plant species and what quantity of plant material exist and thrive, which in turn control the oxygen concentrations and animal species (Harper 1992). Depending on nutrient levels, aquatic ecosystems may be low in nutrients (oligotrophic), contain moderate amount of nutrients (mesotrophic) or be fully overloaded with nutrients (eutrophic). Oligotrophic systems contain few phytoplankton and macroinvertebrates and the geomorphology of their beds is primary rocks. Eutrophic SWBs are rich in phytoplankton and macroinvertebrates and are found in naturally fertile lowland regions in which human activity causes increased supply of nutrients. Eutrophic systems are productive, support a great variety of aquatic life and are high in fish yield (FAO, 1994)

## 2.4 Macroinvertebrates Composition, Abundance and Biomass

### 2.4.1 Composition

Macroinvertebrates are an important energy link in the food web and form a significant component of the diet of fish; therefore play an important role in fisheries production. Herbivorous fish such as *Oreochromis niloticus* has a diet that includes mostly insects, algae, fish, molluscs and detritus (Njiru *et al.*, 2004). Macroinvertebrates are major consumers of primary production and form part of the secondary trophic level (Mushi *et* *al.*, 2005). They show high diversity, wide distribution and are able to exploit most of the habitats In SWBs (Merritt & Cummins, 1996).

The community structure of macroinvertebrates is influenced by spatial and temporal variation of environmental factors such as substrate type, dissolved substances, turbidity, riparian vegetation, land-use, temperature, altitude and latitude (Richards *et al.*, 1993). Among gastropods, the effects of human activities are diverse. Sites disturbed by degradation of forests result in lower abundance of prosobranch snails (Schilthuizen *et al.*, 2005; Kappes 2006). Habitat destruction, water pollution, pesticides and collection of larger snails the hotel industry has been shown to upset inter-tidal prosobranch community assemblages (Ludwig et al., 2007). Salinization of freshwater has also been identified as a serious issue affecting macroinvertebrate communities (Dunlop *et al.*, 2008; Hassel *et al.*, 2006).

High species diversity of macroinvertebrates is associated with complex interactions involving energy transfer, competition and niche apportionment (Brower *et al.*, 1990). Composition and richness of macroinvertebrates is used to by ecologists to detect changes in water quality. Due to sensitivity to different chemical and physical conditions, they provide an estimate of the health of aquatic systems. Macroinvertebrates live in water for all or part of their lives, so their survival is related to the water quality. Unlike fish which have the ability to move away from unfavorable conditions, macroinvertebrates are confined to limited areas and act as continuous monitors within freshwater systems (Magagula *et al* 2010; Ojunga *et al.*, 2010). The effects of water quality on the benthic macroinvertebrate communities has been widely studied (Wynes and Wissing 1981; Mathooko and Mavuti 1992; Baker and Sharp 1998; Raburu 2003; Okungu and Opango 2005; Mwaura, 2006; Kibichii *et al.*, 2007; Masese *et al.*, 2009a; Masese *et al.*, 2009b; Magagula *et al.*, 2010).

Depending on their tolerance, macroinvertebrates respond differently to water quality. Key among the water quality parameters that shape the assemblages of benthic and macroinvertebrate organisms is dissolved oxygen (Heatherly *et al.*, 2007; Magagula *et al.* 2010). The Ephemeroptera- Plecoptera-Tricoptera (EPT) groups are considered "clean" water groups; declining in abundance and richness in sites associated with water quality deterioration and habitat degradation (Wynes and Wissing, 1981;Klemm *et al.*, 2003). However, tolerance to environmental conditions varies among these aquatic insects; with emeregence of Ephemeroptera being limited by enviromental variables such as pH while Trichoptera is able to tolerate wider ranges of environmanetal conditions (Freitag 2004). Among dragonflies (Insecta: Odonata), while as diversity is strongly correlated with good water quality containing low turbidity, moderate conductivity and high dissolved oxygen (Corbet 1999), correlations with temperature are more or less coincidental (D'Amico *et al.* 2004).

In Lake Victoria, poor water conditions are associated with chironomids, hirudnea and soft bodied, non-insect individuals such as oligochates and planarians which are more tolerant (Mwambungu *et al.*, 2005). The requirement for shelter is an important factor affecting macroinvertebrate communities. Seasonal El-Nino events have been associated with changes in availability of cover and food and are linked to variation in abundance of the gastropod *Turbo torquatus* in New South Wales (Ettinger-Epstein and Kingsford, 2008).Abiotic factors also play a role in microinvertebrate species composition. Due to their sensitivity, dragonflies (Insecta: Odonata) have been used to monitor changes in abiotic environmental conditions (Simaika and Samways 2011). High similarity of the environment represents little variation in abiotic conditions (Juen and De Marco, 2011).

Anthropogenic activities at the catchment and local scales have serious implications for water quality and can restructure benthic communities mostly by reducing the total number of taxa, as well as shifting to more unevenly distributed communities (Jones *et al.*, 2002; Masese *et al.*, 2009b). Industrial waste discharge from pulp, paper, textile, cotton and sugar processing plants plays a role in deterioration of water quality and affects macroinvertebrates and fish adversely (Ojunga *et al.*, 2010). Although spatial patterns, local habitat characteristicts, geomorphology and water chemistry are associated with macroinvertebrate assemblages, temporal variability has been found to have a greater impact (Marshall *et al*., 2005). Rainfall, salinity, landuse and instream habitat are the most significant factors influencing distribution of aquatic invertebrates (Kay *et al.*, 2001). Rainfall patterns, especially heavy rainfall influence macrinvertebrate populations, probably as a result of sweeping away of aquatic larval stages of these organisms during heavy rains.

### 2.4.2 Abundance

Abundance and richness of macroinvertebrates is attributed to riparian vegetation cover and habitat quality (Raburu 2003). In water having low dissolved oxygen as a result of high organic loads, organisms that are less tolerant are replaced by more resistant ones (Ojunga *et al.,* 2010). Although abundance and richness of 'EPT' groups declines in sites associated with water quality deterioration and habitat degradation, the response of each group to degradation is different; with sensitivity decreasing from Plecoptera to Trichoptera. Low abundance of Ephemeroptera is attributed to water quality deterioration and habitat degradation in these water bodies (Wynes and Wissing; 1981).

The type of food ingested has been linked to abundance levels in gastropods. Reduced growth and fecundity in the Prosobranch *Potamopyrgus antipodarum*have been recorded following ingestion of *Planktothrix agardii*; indicating negative effects of toxic cyanobacteria on natural communities of freshwater gastropds (Lance *et al.,* 2008).The presence of predatory organisms such as fish, competitive interactions and availability of food has also been seen to influence abundance of macroinvertebrates (Fincke 1992; Braccia *et al.*, 1992).

Landuse activities such as animal watering, unpaved roads, and human settlements are factors in increased wastes and sediments; causing smothering and disturbance of substrates that are grazed by certain macroinvertebrates. Owing to increased sedimentation during rainy seasons in combination with landuse activities, spatial and temporal variation in abundance and richness of macroinvertebrate communities is manifested. Shifts in environmental conditions can be detected by the proportions of macroinvertebrates such as scrapers to filterers which represent the balance between food sources. An increase in relative abundance of filtering collectors indicates existence of organic wastes from animals deposited directly in the water; thus increasing availability of fine particulate matter. On the other hand, the relative abundance of gatherers is a useful measure of general degradation since these are genaralists that thrive in areas abundant with fine particulate organic matter (Masese *et al.*, 2009b).

Although tolerant to a broad range of environmental conditions such as salinity (Kay *et al.,* 2001; Dunlop *et al.*, 2008) and dissolved oxygen (Ojunga *et al.*, 2010; Wynes and Wissing; 1981), variation in these conditions affects a large number of macroinvertebrates. As such, assessment of the effects of environmental conditions on macroinvertebrate communities is necessary for environmental management procedures.

### 2.4.3 Biomass

Macroinvertebrates contribute significantly to biomass of freshwater communities. There exist a strong correlation between fish yield and both primary and secondary production in lakes and dams (Oglesby, 1977). Numerous methods exist for measuring secondary production and the relationship between the biomass-specific growth rate and population biomass is implicit to all. Secondary production of aquatic invertebrates is normally calculated by following a cohort through time. The production rate of each of the different orders is however inferred from measures of the number of organisms and the rate of growth in each of these orders, thus giving the formation of animal biomass over time (Benke & Huryn, 2006). The most important variables that influence benthic biomass in a given water body are the mean depth and water column productivity, with biomass inversely related to the former and directly to the latter (Rasmussen and Kalff, 1987).

Studies in Quebec, Canada on the effect of macrophytes on macroinvertebrates indicate that the type of habitat also affects macroinvertebrate biomass, as well abundance and richness, which have been found to be significantly higher in submerged than in emergent and floating-leaved habitats. This has been attributed to substrate preferences of herbivores, mostly Gastropoda (Cremona *et al.*, 2008). In the same study, high water levels were associated with more macroinvertebrate densities; which may have negative implications with decreasing water levels influenced with global climatic changes. Another factor in variation of macroinvertebrate biomass densities is seasonal changes of their habitats. In the Amazon region, density of intertidal benthic macrofauna of rare rocky increases during the rainy season and is attributed to differences in larval supply, recruitment and settlement processes (Morais and Lee, 2014).

Macroinvertebrates are known to influence fish populations. High values of macroinvertebrate biomass indicate high productivity, good freshwater habitat and adequate amounts of food for fish and have been correlated to high density and biomass of brown trout (Salmo trutta) (Morante *et al.*, 2012). An increase in macroinvertebrate biomass may be explained by an increase in food availability, allowing consumers to invest less energy in searching for food, thus increasing their feeding efficiency (Wallace and Webster, 1996).

## 2.5 Abundance, Prevalence and Mean Intensity of Fish Parasites

### 2.5. 1. Abundance of fish parasites

Parasites are important components of host biology, survival, population structure and ecosystem functioning and serve as important indicators of animal diversity, abundance and trophic interactions (Schludermann *et al.*, 2003). Parasitism can influence wild fish population structures either directly through mortality or indirectly through, reduced fecundity, alteration in host behavior, reduced swimming speed, or increased risk of predation (Longshaw *et al.*, 2010). In fish farming, especially in conditions of intensive monoculture, mortalities of young fish have been reported as a result of mass invasion of *Diplostomum spathaceum* cercariae (Field & Irwin, 1994). Eye flukes of the genera *Diplostomum* and *Tylodelphys* are common parasites of wild freshwater fishes. In commercial farms operating intensive monoculture systems; these parasites can be significant pathogens causing a range of disease symptoms, including growth reduction and high fish mortality levels.

Parasites and diseases pose major threats to populations of endangered species (McCallum & Dobson, 1995). Fish parasites; particularly fish-borne zoonotic trematodes, not only pose a risk to human food safety and health, but also cause substantial losses in aquaculture as a result of restrictions on export and reduced consumer demand due to food safety concerns (Nguyen *et al.*, 2009). Several parasites found in fish are potentially transmissible to humans through consumption of raw fish (Hong *et al.*, 1996; Tiewchaloern *et al.*, 1999; Aloo, 2000; Wiwanitkit e*t al.*, 2001; Dzikowski *et al.*, 2003; Thu *et al.*, 2007; Nguyen *et al.*, 2009; Park *et al.*, 2009; Sohn, 2009).

Parasite communities may vary among localities or among host species. The ecological factors which cause the parasite communities to differ include host diet, range, abundance, size and longevity. Among helminth species, diversity is strongly correlated with host body size and to a lesser extent with longevity and host diet; suggesting that the diversity of internal fauna supported by a host is determined primarily by its ability to maintain a parasite population, and secondarily by its liability to acquire it. Freshwater parasites tend to have an aggregated distribution which is attributed to increased reproductive efficiency in some species of adult parasites and occupation of a different niche; enhancing the chances of infecting the host. This pattern of distribution can influence the evolutionary history of parasites due to completion for food, space and reproductive success (Campos *et al.*, 2009).

To describe the structure of parasite infra-communities, data related to parasite abundance and prevalence is used as well as the species richness. The diversity of parasite communities can be defined by a diversity index chosen to describe it. Such an index weighs the relative evenness of distribution of each species and is dependent on species richness and evenness. The contribution of each species may determine by equal weighting without regard to the number of individuals of each species (Bush *et al.*, 1997).

The dynamics of fish parasitic fauna depends on several factors which include habitat geography, seasonality, water characteristics, fauna present, feeding behavior of fish, direct activity area and life cycle of parasite and availability of intermediate and definitive hosts (Ibiwoye *et al.*, 2004; Elsheikha & Elshazly, 2008; Campos *et al.*, 2009; Singhal and Gupta 2012). Seasonal cycles in water quality characteristics such as temperature, salinity and flow rate lead to changes in parasitic indices such as intensity, abundance and prevalence (Gonzalez, at al., 2009). In heliminths, host feeding ecology, habitat use, host sex, evolutionary and historical factors and distance between populations are key factors affecting their population structures (Bell and Burt, 1991; Mwita and Nkwengulila, 2008; Campos *et al.*, 2009). In addition, the role of host-parasite coevolution and phylogenetic relatedness of host species within an area, coupled with host specificity helminth populations has been emphasized (Overstreet *et al.*, 1998).

Diversity and abundance of the internal fauna supported by a host is determined primarily by its ability to maintain a parasitic population and secondarily by its liability to acquire it. Interesting, the size of the host has been associated with both positive and negative correlations with level of parasitism. Large hosts sustain a more diverse parasite community as they tend to be long- lived and piscivorous (Bell and Burt, 1991). The larger surface area and hence more niches and an increasingly animal diet in larger fish allows more parasites. However, large fish sometimes have lower parasite levels as a result of elimination of parasites acquired in early stages of fish at adult stage as well and changes in ingestion of forage species by different age brackets and dynamics of intermediate hosts (Campos *et al.*, 2009). In Lake Victoria, the structure of parasite communities has been linked to ecological factors, host size and diet, with a richer and diverse parasite community found in high population densities of clariid fishes (Mwita and Nkwengulila, 2008).

Parasite infection levels are influenced by water quality parameters such as temperature, pH, dissolved oxygen and conductivity (Puinyabati *et al.*, 2013). Water flow rates and turbulence also contribute to the success of parasite transmission. In *Diplostomum* for instance, outbreaks of diplostomiasis are related to reduced water flow rates and temperature (Chappell *et al.*, 1994). In freshwater systems, temperature is a key factor influencing emergence, survival and infectivity of cercarial transmission stages of trematodes (Fried *et al.*, 2002; Fried & Ponder, 2003; Thieltges & Rick, 2006). Water salinity has also been found to influence parasite intensities in fish (Rogowski & Stockwell, 2006).

### 2.5.2. Fish Parasite Prevalence

Parasitic infection levels are dependent on many factors. Key among them are distribution of intermediate hosts, age of host and life cycle of parasite species (Puinyabati *et al.*, 2013). The lifecycle pattern, availability and infectivity of intermediate and definitive hosts are important factors in variation of prevalence rates of parasites (Khurshid & Ahmad, 2012). In small water bodies in South Africa, the presence of three species of snails *Bulinus africanus*, *Bulinus stropicus* and *Lymnaea natalensis* was associated with high levels of strigeid metacercariae in muscles of the southern mouth brooder (*Pseudocrenilabrus philander*), eye diplostomatids in the branded tilapia (*Tilapia sparrmanii*) and the southern mouth brooder (*Pseudocrenilabrus philander*), cranial cavity diplostomatids of catfish (C*larias gariepinus*), *Clinostumum tilapia* in the gills of *T. sparrmanii*, and metacercariae of *Euclinostomum heterostomum* in muscle of *P. philander* (King, 2007).

Differences in prevalence levels for *Contracaecum* sp. in *Oreochromis leucosticta* in lake Naivasha's Oloidien Bay have attributed to feeding habits of fish; with benthic feeders yielding higher prevelances due to high presence of larval stages of parasites in the diet (Aloo, 2002). Susceptible to parasitic infection is higher in stressed fish, contributing to higher prevalences (Muchiri, 1990; Mwita and Nkwengulila, 2008). Size, age and sex of fish also influences prevalence levels with large, older males having higher rates due to consumption of larger food quantities, longer exposure and physiological attributes of male and female fish (Aloo, 2002). Parasite-mediated sexual selection influence gender variation in parasitic infection levels in fish (Batra, 1984; Takemoto & Paravelli, 2000; Reimchen & Nosil, 2001; Ibiwoye *et al.*, 2004; Maan *et al.*,2006; Singhal & Gupta, 2009; Gupta *et al.*, 2012). In freshwater murrel, higher infectivity by *Genarchopsis*is linked to positive stimulus which preferentially attracts cercariae and other parasites to females (Singhal and Gupta 2009). On the other hand, a stronger in-built resistance to the infection contributes to establishment of fewer parasites in males than in females (Gupta *et al.*, 2012). In Lake Victoria, higher and more viable parasite loads in males are attributed to a male specific trade-off between immune defence and reproductive investment mediated through carotenoid-based male breeding coloration (Gupta *et al.*, 2012). Over and above water quality physical chemical conditions, diet and availability of definitive and intermediate host, the role of salinity has been emphasized in marine and brackish water fish (Ogbeibu *et al.*, 2014)

### 2.5.3 Fish Parasite Mean Intensity

Dynamics of life cycle patterns of parasites influence degree of intensity in fish hosts. In trematodes for instance, the intensity of parasitism is influenced largely by age of the host as a result of repeated seasonal exposure to cercariae and metacercariae. In cases where older fish harbor fewer parasites, it is attributed to recent colonization of species that preferentially invade young fry (Chappell *et al.*, 1994). The diversity in feeding habits of fish provides intermediate or definitive hosts for many helminth parasites (Bell & Burt, 1991).

Gender variation in parasite intensity has also been linked to parasite-mediated sexual selection (Batra, 1984; Takemoto & Paravelli, 2000; Reimchen & Nosil, 2001; Ibiwoye *et al.* 2004; Maan *et al.*, 2006; Singhal & Gupta, 2009; Gupta *et al.*, 2012). Males tend to have a stronger in-built resistance to the infection than females, leading to establishment of fewer parasites in males (Gupta *et al.*, 2012). In Lake Victoria, changes in male coloration during breeding is linked to higher susceptibility to parasitic infection and in return contributes to higher and more viable parasite loads.

Ecological differences between genders also contribute to differences in patterns of parasitic infection in male and female fish. Spatial and dietary niche differences between male and female fish may be driven by intraspecific competition for resources and facilitated by the opportunity for females to exploit pelargic niches with little or no competitors (Maan *et al.*, 2006). In cichlids, higher helminth infections in males have been associated with consumption of larger food quantities; leading to higher probabilities of consuming infected food items than females (Batra 1984). Ecological factors structure the parasite communities of parasites in Lake Victoria, In lake Naivasha, differences in mean intensity levels of *Contracaecum* parasites in the main lake and Oloidien Bay have been recorded at 2 and 15 parasites per fish respectively and are attributed to feeding habits of fish and stress from salinity at the bay (Aloo, 2002). Disparities in relative parasitism in threespine stickleback *Gasterosteus aculeatus* have been attributed to differences in feeding habits whereby females had higher frequencies of pelagic food items such as *Schistocephalus solidus* and nematodes while as males had benthic items such as *Cyathocephalus truncates* dominate the diet (Maan *et al.*, 2006).

## 2.6 Effect of Water Quality and Seasonality on Biota

Water quality in reservoirs is an important aspect of water resource management as it is a determinant of spatio-temporal dynamics of aquatic organisms and drives various uses of aquatic ecosystems. Water quality characteristics influence phytoplankton abundance, diversity, species richness, distribution and density, as well as species composition of aquatic communities and sensitive life stages of macroinvertebrates (Welch, 1980).

### 2.6.1 Effect Water Quality and Seasonality on Phytoplankton

The relationship between water quality and phytoplankton differs from place to place depending on a number of factors. In certain cases, strong correlations exist between physic-chemical water quality aspects such as temperature. Seasonal variation in phytoplankton species composition is strongly linked to temperature (Harris, 1986). Assimilation rates of nutrients in certain species of phytoplankton such as the diatom, *Skeletonema costatum* increase at lower temperatures and subsequently increase biomass (Goldmman, 1977). In the same way, seasonal variations in dissolved inorganic nutrients have a bearing on composition, abundance and biomass of phytoplankton in Tanzania (Hamisi *et al.*, 2014). Similarly, higher phytoplankton diversity during the rainy season is attributed to additional recruitment resulting from scouring of by floodwaters in a Nigerian creek (Onyema, 2007). Interestingly, while dry and wet seasons affect phytoplankton abundance and diversity in the coastal region of Malaysia, water quality conditions had no influence (Al-Gahwari, 2003).

Productivity of phytoplankton is greatly influenced by amount of light and nutrients and is highest when the two are at optimal. Because photosynthesis takes place in the photic zone, nutrients are often depleted in this zone as a result of utilization by phytoplankton and other organisms. The depth at which maximum production for phytoplankton occurs is therefore a function of availablity of light amount of nutrients especially in the face of vertical mixing (Yentsch, 1981).

The implication of the relationship between water quality and phytoplankton, particularly temperature is grim in the face of global warming. This is because primary production and zooplankton biomass are projected to decrease due to increased stratification in response to a warming climate, particularly in the tropics (Chust *et al.*, 2014).

### 2.6.2 Effect Water Quality and Seasonality on Macroinvertebrates

Macroinvertebrate communities consist of groups tolerant to a broad range of environmental conditions such as salinity (Kay *et al.* 2001; Dunlop *et al.* 2008) pH, nutrients, the mean depth (Ortiz and Puig, 2007) and dissolved oxygen (Ojunga *et al.* 2010; Wynes and Wissing; 1981). Several macroinvertebrate groups especially Pelecypoda, Hirudnea and Gastropoda do not occur in low pH (5.0) while on the other hand, the bivalve *Pisidium* sp. is found in acidic water (pH 5.2) (Schell and Kerekes, 1989). In addition, high water levels influence macroinvertebrates; resulting in high biomass densities (Morais and Lee, 2014).

Aquatic biota have developed flexible physiological and lifecycle characteristics that enable them to survive fluctuations in hydrological, habitat and ecological conditions. The mechanisms they have adapted include using environmental cues for critical life history activities, drought refugia and life history strategies that resist dessication.

Implementaion of environmental management procedures therefore requires assessment of the effects of environmental conditions on species physiological responses.

Sensitivity to environmental parameters varies among different groups of macroinvertebrates. Dragonflies (Insecta: Odonata) are known for their sensitivity to changes in environmental conditions and have been used for assessment of environmental change (Simaika and Samways, 2011). The degree of dissimilarity, also referred to as Beta-diversity patterns in Odonata has been attributed to environmental variables; with low index values attributed to high similarity of the environment which represents little variation in abiotic conditions (Juen and De Marco, 2011). High diversity of Odonata species is strongly correlated with good water quality containing low turbidity, moderate conductivity and high dissolved oxygen (Corbet 1999). Correlations for Odonate species with temperature have however been found to be more or less coincidental. (D'Amico *et al.*, 2004). The presence of predatory organisms such as fish, competitive interactions and availability of food also influences abundance of Odonata (Braccia *et al.*, 1992; Fincke 1992). Tolerance to environmental conditions varies among aquatic insects; with emeregence of Ephemeroptera being limited by enviromental variables such as pH while Trichoptera is able to tolerate wider ranges of environmanetal conditions (Freitag 2004).

Although spatial patterns, local habitat characteristicts, geomorphology and water chemistry are associated with macroinvertebrate assemblages, temporal variability has been found to have a greater impact (Marshall *et al.*, 2005). In their studies Kay *et al.*,

(2001) demonstrated that rainfall, salinity, landuse and instream habitat are the most significant factors influencing distribution of aquatic invertebrates. Seasonal habitat changes contributes to changes in macroinvertebrate biomass densities. In the Amazon region, densities of intertidal benthic macrofauna of rare rocky increase during the rainy season, a factor attributed to differences in larval supply, recruitment and settlement processes. Macroinvertebrate biomass levels in Nyando wetlands in Lake Victoria basin (Orwa *et al.*, 2012) and in small water bodies in Siaya and Uasin Gishu (Ngodhe *et al.*, 2013) have been attributed to high nutriet levels. On the other hand, the effects of nutrients on macroinvertebrate densities have been found to be inconsistent, withhigh nutrients levels associated with both increase and decrease in macroinvertebrate densities (Ortiz and Puig, 2007; Morais and Lee, 2014). Though not well understood, the reasons for such variations are linked to high food availability and effects of toxic compounds (Wallace *et al.*, 1996). The role of food availability in biomass density of macroinvertebrates has been emphasized by Morais and Lee, (2014), indicating that differences in larval supply, recruitment and settlement processes in wet and dry seasons is responsible for changes in macroinvertebrate biomass. Nutrient enrichment decreases macroinvertebrate richness by elimination of sensitive taxa mostly represented by the orders Ephemeroptera, Plecoptera and Trichoptera (EPT) (Paul & Meyer, 2001). The amount of food available has been an important factor associated with density and biomass of macroinvertebrates (Morante *et al.*, 2012).

### 2.6.3 Effect Water Quality and Seasonality on Parasites

Pathological conditions of fish have been on the increase as a result of overcrowding, importation of seed for re-stocking, sources of food and water quality deterioration in intensive fish culture facilities (Florio *et al.*, 2009). The role of temperature and seasonality has been underscored by several authors (Batra, 1984; Paperna, 1996; Aloo, 2002; Dzikowski, Diamanti, *et. al.*, 2003). Seasonality of parasitic infection is considered to be mechanism for high transmission success through synchronization with chronobiological behavior of the fish host (Faltynkova *et al*., 2009).

Modulation of extent and intensity of parasitism by climatic conditions can be explained by the emergence of some parasitic diseases. In digenes for instance, the emergence of cercariae is associated with temporal distribution of hosts which is controlled by seasonality (Faltynkova *et al.*, 2009).

Seasonal cycles in water quality characteristics such as temperature influence parasitic indices such as prevalence rates in fish (Khurshid and Ahmad, 2012; (Puinyabati *et al.*, 2013). Studies on the parasite *Eustrongylides africanus* in *Clarias gariepinus* indicate that incidences of infection increased during the dry season and were higher in females than in males in both dry and wet seasons (Ibiwoye *et al.*, 2004). Densities of *Contracaecum* parasites were found to be higher during the dry seasons, a factor attributed to high availability of food and salinity (Ogbeibu *et al.*, 2014). Vincent and Font (2003) attribute seasonal fluctuations in helminth parasites of fish is to heavy rains associated with the winter season which decreases parasite transmission by flushing infected hosts, intermediate copepod hosts and possibly free-living infective worm stages downstream.

High prevalences of helminths in the dry season is attributed to behaviour of fish in relation to habitat types and disruption of parasite transmission by water currents and depth respectively (Akoll *et al.* 2012). The authors further suggest that cages are safer for rearing fish in deep reservoirs as parasites are more in the bottom of reservoirs. Decrease in water volume during the dry season is associated with nutrient imbalances in the water; leading to less production of fish food organisms (Gupta *et al.*, 2012) and higher consumption of parasitic organisms. Parasite prevalences are also influenced by changes in water quality parameters as a result of human activities (Ochieng *et al.*, 2012).

Distinct seasonal patterns in prevalence and abundance of the helminth *Camallanus oxycephalus* during summer have been demonstrated by Steinauer and Font (2003) and are attributed to release of juveniles in spring, the increase of copepod intermediate hosts into the environment and increased fish foraging in the summer. In the same study, the decline of prevalence and abundance of the parasite in the fall season could be due to death of infected hosts or competitive interarctions among individual helminths. Interstingly, temperature did not seem to play a significant role in this study.

On the other hand, lack of seasonality in parasitic infection has been observed and been attributed to undefined seasons in the tropics, leading to continuous parasite life cycles (Aloo, 2002). Non-seasonal fluctuations in parasites could be due to highly variable nature of the water body; with frequent, unpredictable changes in the physical conditions causing unstable populations that fluctuate unpredictably regardless of seasonal effects (Aloo, 2002). Water shrinkage associated with rapid water quality deterioration and low water turn-over during the dry season may result in environment-related host stress; increasing susceptibility to parasites (Akoll *et al.*, 2012).

## 2.8 Information Gaps

For sustainable fisheries development, the ecological conditions and productivity of the water bodies should be well understood. While limnological characteristics and water retention affect the recruitment and survival of fish in that ecosystem (Mustapha, 2011), the role of environment on ecosystem and fishery recruitment of most water bodies is largely unknown (Cowx, 1999). Information on relationships between seasonal variation in weather and climatic conditions, water quality, phytoplankton, macroinvertebrate as well as fish parasites is lacking. As an example, studies as recent as 2014 indicate that there are uncertainties on the role of nutrient levels on macroinvertebrate communities (Morais and Lee, 2014). Conflicting results on the effect of high nutrients levels have been associated with both increase and decrease in macroinvertebrate biomass density. Although the reasons for such inconsistency are not well understood, they are associated with variations in food availability in wet and dry seasons (Wallace *et al.*, 1996).

Although studies on small water bodies in Lake Victoria basin, Kenya have been conducted (Maithya, 1998; 2008; Maithya *et al.*, 2012; Ngodhe *et al.*, 2013) they have not addressed the aspect of fish parasites. Compared to other aspects of fish ecology, information on parasites remains scanty in eastern Africa (Aloo, 2002). It is a well-known fact that development of reservoir fisheries has been associated with transmission of parasites from wild to cage fish (Florio *et al.*, 2009), with introduction of large numbers of caged fish to a system having dramatic effects on disease agents and often causes severe infestation and heavy mortalities (Knaap, 1994). However, the mechanisms of spread of these diseases remain poorly understood compared with other aspects of fish ecology in Kenya (Aloo, 2002). Apart from studies on influence of water quality on parasite prevalence levels in small water bodies in Uasin Gishu (Ochieng *et al.*, 2012), there is little or no other work that has been done on small water bodies in Lake Victoria basin. Further to this, interactions between water quality, phytoplankton, macroinvertebrates and fish parasites have not been conducted in the region.

# CHAPTER THREE

# MATERIALS AND METHODS

## 3.1 Study Area

The Small Water Bodies in Uasin Gishu and Siaya counties are located in the upper catchment of the Lake Victoria Basin (LVB), which is (Figure 3.1) is located in the upper reaches of the Nile basin and lies between altitudes of 900 and 1800 meters above sea level. It is well endowed with numerous perennial rivers that drain into the lake. Lake Victoria lies across the equator between latitudes $0^{o}$ 31'N and $3^{o}$ 54'S, longitudes $31^{o}$ 18'E and $34^{o}$ 54'E with an average depth of 80m. The lake is shared by Kenya, Uganda and Tanzania as well as Rwanda and Burundi which from part of its drainage basin.


**Figure 3.1 A map of the Lake Victoria basin and its catchment area (Source: Koyombo and Jorgensen, 2007)**

In Kenya, the catchment area of the lake covers the entire Nyanza and Western provinces and drains extensive sections of the eastern slopes of the Rift Valley, an area that extends from Cherangani hills to Mau forest and includes Masai Mara game reserve in the former Rift Valley province. It covers an area of approx. 251,000km$^2$; of which 69,000km$^2$ is the lake itself. In Kenya, the main influent rivers are Sio, Nzoia, Yala, Nyando, Sondu Miriu, Kuja, Migori and Mara River (Raburu *et al.*, 2012). The Lake outflows through White Nile and Katonga; both of which are part of the upper Nile (http://www. britannica). The basin is one of the most densely populated places in Eastern Africa, with over 30 million people living and deriving their livelihoods directly or indirectly from its resources (Kizza *et al.,* 2009).

There are considerable variations in the average yearly rainfall over the Lake and the surrounding area. Rainfall in the eastern side of the Lake is between 500 and 700mm per year. Westward from this area, rainfall increases to an annual average of over 2000mm in areas around Bukoba and the Ssese islands. The Southern part of the lake in Mwanza region rainfall is 750 to 1100 mm and in the eastern part of the Mara region rainfall in between 750-1000mm increasing to 1600mm on the highland areas of Tarime (Ministry of Water, 2008). The basin in composed of many sub-basins. In Kenya, the main river basins are the Sio (1450km$^2$), Nzoia (12676km$^2$), Yala (2388km$^2$), Nyando, Sondu Miriu (3450km$^2$) and the Gucha (6600km$^2$) while the Mara is shared between Kenya and Tanzania (Mutua, 1980). The Nzoia River basin in which this study was conducted receives rainfall in three main rainy seasons: March to May, July to September and October to December, with the July to September season being mild in the lower parts of the basin.

Due to good soils and rainfall on the middle and upper parts of this basin, diverse and intense farming activities have adversely affected the flow characteristics on the lower parts of the basin. As a consequence there has been an increased frequency of flooding in the basin. However, the seasons are not exactly concurrent in all parts of the basin. The most intense season is March to May and often causes flooding in the lower parts of the basin. Evaporation over the basin is almost constant throughout the year except for the period of April to August when the rates are slightly lower due to lower temperatures at this time of the year. The basin has soil-water surpluses and deficits during the rainy seasons and other times of the year respectively (Mutua, 1980).

### 3.1.1 Preliminary Survey for Selection of Small Water Bodies

A preliminary survey was conducted between January and March 2010 in Uasin Gishu and Siaya counties. The counties fall within two Landsat TM images, namely, Path 169 Row 060 and Path 170 Row 060. A mosaic of two available images of 1988 was created and images representing the two districts extracted. These images were processed to identify all sites including dams and large ponds, as well as wetlands with potential for aquaculture. The coordinates for the centers of all identified sites were extracted for purposes of guiding their location during fieldwork.

A total of 17 dams were surveyed; 7 in Uasin Gishu and 10 in Siaya districts. In Uasin Gishu the following small water bodies were included in the survey: Kesses, Kerita, Airport, Sugutek/Chepkanga, Kiara/ Chepkanga, Koitoror/ Chepkanga and Ziwa. In Siaya it involved: Kalanjuok, Kamasawa, Ochilo, Uranga, Abura, Apidi, Futro, Kapodo, Yenga and Ukwala. A survey on the sites was conducted to assess water quality, biodiversity and socio-economic characteristics (Table 3.1).

**Table 3.1**. **Targeted parameters for assessment in small water bodies in Siaya and Uasin Gishu counties in during a survey conducted from January to March 2010**

| **Parameter** | **Measurable attributes** |
|---|---|
| Site characteristics | Land use, water body size, depth, water body uses, etc |
| Water quality | pH, conductivity, dissolved oxygen, nutrients, |
| Biodiversity | Aquatic macrophytes and microphytes, fish and bird species, etc |
| Socioeconomic characteristics | Population, uses for water bodies |

To select sites for the study, criteria for site suitability was developed based on optimal conditions for aquaculture development. Based on available literature and experience, sites that were considered suitable had pH levels above 7 and minimum of 5mgl$^{-1}$ DO, 1m turbidity and 4 ha in surface area. The water bodies were also required to have an average depth of 2-3m.

Dissolved oxygen, pH and turbidity were measured following the procedure outlined in section 3.3. The depth of the water body was determined using a measuring tape at three locations while the area of the water bodies was established from secondary and primary information from the ministries of agriculture, water, and fisheries as well as from Kenya marine and fisheries research institute. A structured questionnaire was used to obtain information from communities living near the water bodies. The questionnaire was structured to gain information on the following aspects of the community: land use, water sources, sources of income, education background, knowledge of aquaculture (Appendix 1).

### 3.1.2. Small Water Bodies Selected in Uasin Gishu

Uasin Gishu County is a high altitude area at a height of between 1500m and 2200m above sea level at Kipkaren and Timboroa respectively. The county has two rainfall seasons with average annual rainfall of 900mm to 1200 mm. Temperatures range between 8.4$^{o}$ C to 27$^{o}$ C. The county is located in the upper catchment of the Lake Victoria Basin and covers an area of 2,955 km$^{2,}$ It is located on latitude 00$^{o}$ 30' 00'' N and longitudes 35$^{o}$ 20' 0'' E. Trans-Nzoia and Keiyo Counties neighbor Uasin Gishu to the north and north east respectively while Kakamega and Nandi Counties are found bordering it to the west, Kericho and Baringo Counties to the south and east respectively (www.investkenya.com/uasingishu-county, 2014). (Figure 3.2)

Kesses dam is located in Uasin-Gishu county; at an altitude of about 7234 feet above sea level; 00$^{0}$ 17. 263'N, and 035$^{0}$ 19.852'E, (GPS readings, Etrex Garmin model). The littoral zone is dominated by *Typha latifolia* and *Cyperus papyrus* species. In the shallow littoral areas, water lilies, *Nyphae lotus*, water fern, *Azolla* spp., water cabbage, *Pistia stratoites*, *Salvinia*, *Lemna*, *Ceratophylum dermersum*, *Potamogeton*, *Ultricularia* and *Najas* species are found. The dam is drained by two rivers from the east; Endaragwa and Endaragweta. It has a surface area of approximately 500 acres and a maximum depth of 4.48 m.

Kerita dam is also a high altitude water body in Uasin Gishu, is located at $00^0$ 19. 263'N, and $035^0$ 24.329'E, at an altitude of about 7392 feet (GPS readings, etrex Garmin model). The most noticeable emergent macrophytes in the littoral zone are dominated by *Typhae latifolia* and *Cyperus papyrus*. Others macrophytes found in the dam are *Ceratophylum dermersum*, water lilies *Nyphae lotus*, sedges and reeds. It is fed by two rivers i.e. River Chebolol entering in the south-east direction and Kabiyemit which enters the dam in the south-west direction. It has a surface area of approximately 25 acres and a maximum depth of 3 m.

### 3.1.3. Small Water Bodies Selected in Siaya

Siaya County is a low altitude area between 1140m to 1400m above sea level and covers an area of 2,946.1km$^2$. Average annual rainfall ranges between 1170mm and 1450mm while temperatures are between 15$^o$C and 30$^o$C. It lies in the lower catchment of the basin between latitudes $0^o26^{'}$N to $0^o$ 90' S and longitudes $33^o$ 58' E and $34^o$ 33' W. Siaya borders Kakamega and Vihiga counties to the north-east, Kisumu to the south-east and Busia to the north (www.investkenya.com/siaya-county, 2014). (Figure 3.2).

Mauna represents a low altitude water body located in Siaya county at $00^0$ $12^{'}$ $358^{''}$N; $034^0$ $09^{'}$ $433^{''}$E at an altitude of about 3900 feet. It has steep sided edges. The littoral zones are composed of sandy and rocky bottoms with emergent macrophytes such as *Typha*, papyrus reeds, and sedges dominated the zone. It has a surface area of approximately 35 acres and a maximum depth of 4m.

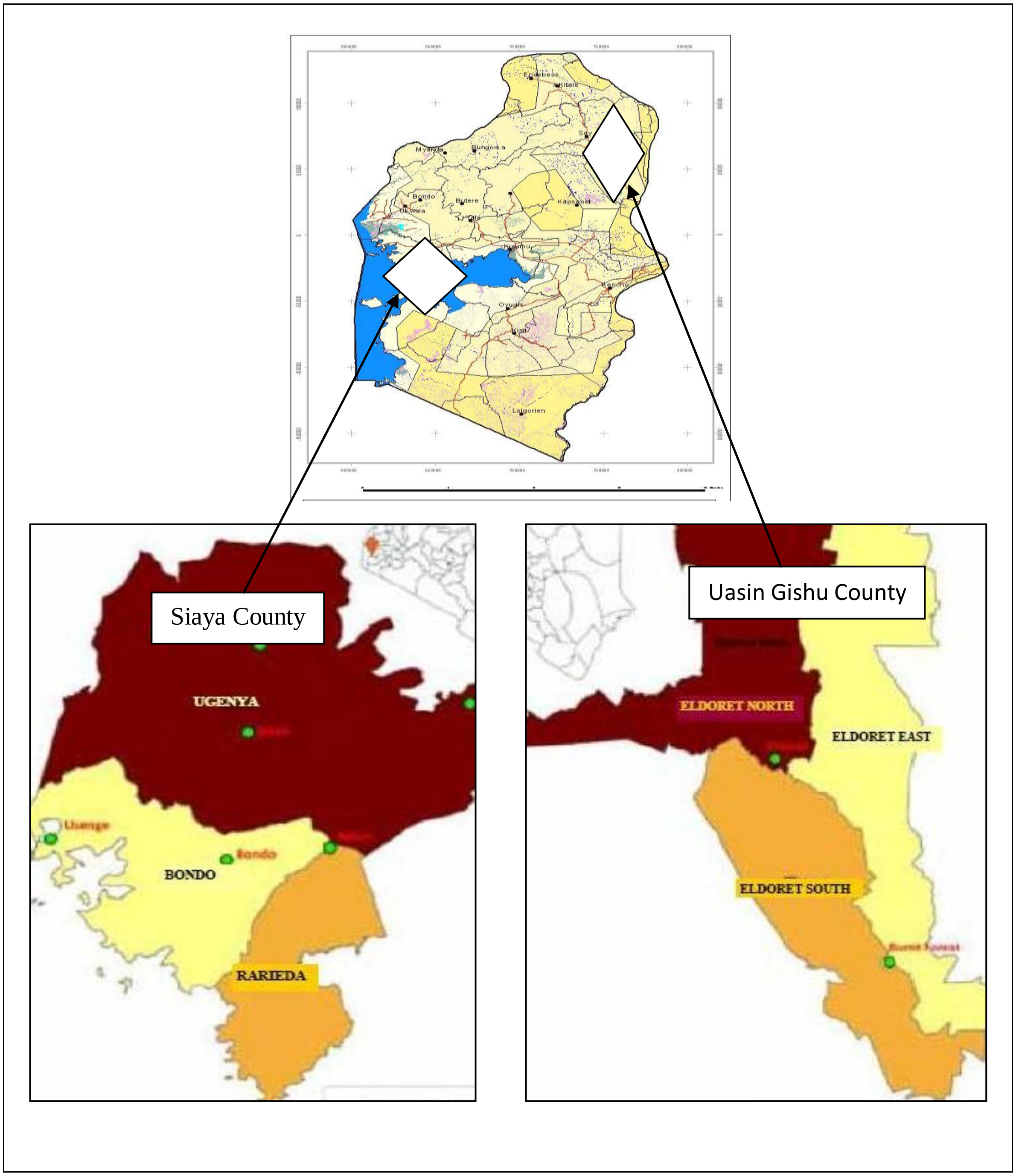



**Figure 3.1 The Lake Victoria Basin on the Kenyan side showing Uasin Gishu and Siaya Counties (Source: Ministry of Environment and Natural Resources, Lake Victoria Environment Management Project , 2006)**

Located in Siaya county at $00^0$ $13^'$ $03^{''}$N; $034^0$ $12^'$ $44^{''}$E, Yenga also represents a low altitude water body at an altitude of about 4022 feet. It has steep sided edges. The littoral zones are composed of sandy and rocky bottoms with loose macrophytic detrital and animal manure deposits brought in by surface runoff from the catchment characterize the littoral zones. The dam is dendritic in shape, has a seasonal feeder stream called Ugege and a permanent spillway. It has a surface area of approximately 15 acres and an average depth of 4.5 m.

## 3.2 Study Design

The relatively cooler, high altitude part of the upper catchment area was represented by Uasin Gishu. Siaya represented the warmer, low altitude part of the lower catchment of the basin. Kesses and Kerita dams in Uasin Gishu and Mauna and Yenga in Siaya served as plots for each stratum. Each of the two plots in each region are within short distances of less than 20km apart. Data was collected on a monthly basis from November 2010 to December 2011 from each of the four dams for assessment of water quality aspects and parasitological status of fish.

## 3.3 Sampling Design

A stratified sampling design was employed to collect data from two areas of the Lake Victoria basin that are geographically and climatically distinct with respect to altitude and temperature. Each of the four water bodies was divided into different stations representing the inlet, outlet, open waters and littoral zone. Kerita (Plate 3.2), Mauna (Plate 3.3) and Yenga (Plate 3.4) had three stations each while Kesses (Plate 3.1), due to its size and diverse activities had six stations. Samples for water quality, nutrients, phytoplankton and macroinvertebrates were collected in triplicate at each station. On each sampling date, fish for isolation of parasites were sampled randomly using a two 100m long seine in each water body. Data collection of physico-chemical parameters, nutrients, macro invertebrates and phytoplankton were conducted once in a month, for eight months from November 2010 to June 2011. Fish parasite data was collected from November 2010 to March 2012.

Rainfall data for Uasin Gishu and Siaya was obtained from Climate Kenya historical weather records website at www.tutiempo.net. The data for Siaya was based on rainfall records from Kisumu weather station 637080 (HKKI) at latitude 0.1’ N, longitude 34’74$^{o}$ E and altitude 1157 m above sea level. Uasin Gishu records were based on Eldoret airport weather station 636880 at latitude 0.4 N’, longitude 35’23$^{o}$ E and altitude of 2104 m above sea level. Sampling months were categorized into two based on rainfall levels: those with rainfall between 0 to 49 mm were considered dry season and those that had rainfall of 50 mm and above were considered wet. In Uasin Gishu, the dry season was from November 2010 to February 2011 while the wet season was from March to December 2011. In Siaya, the dry season was in November 2010, January, February and July 2011 and the wet season was from August to December 2011.


**Plate 3.1 Kesses dam showing sampling stations KES 1, KES 2, KES 3, KES 4, KES 5, KES 6 and KES OW (Open waters). (Source: Author, 2015)**

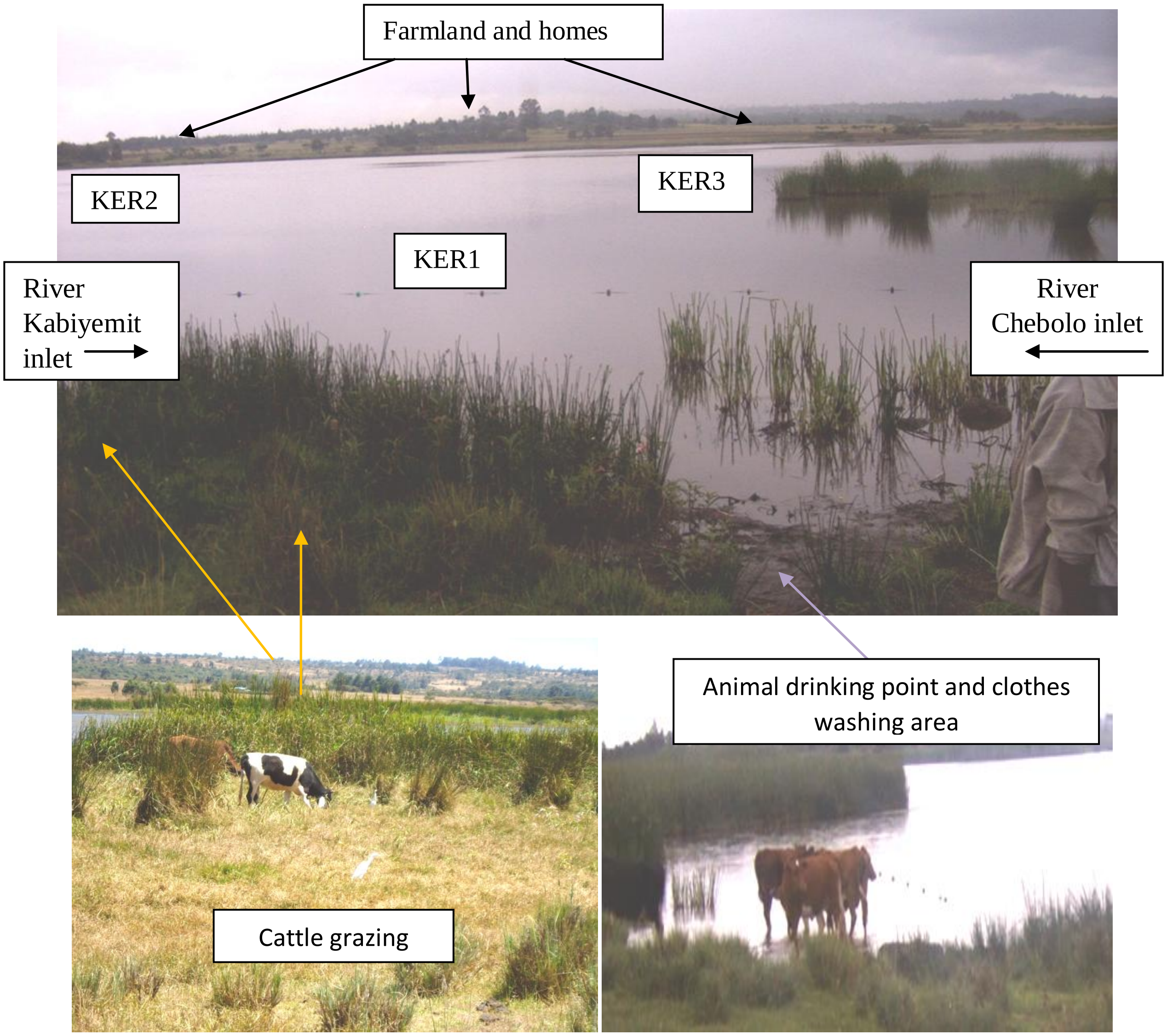



**Plate 3.2 Kerita dam showing sampling stations KER 1, KER 2 and KER 3 (Source: Author, 2015)**

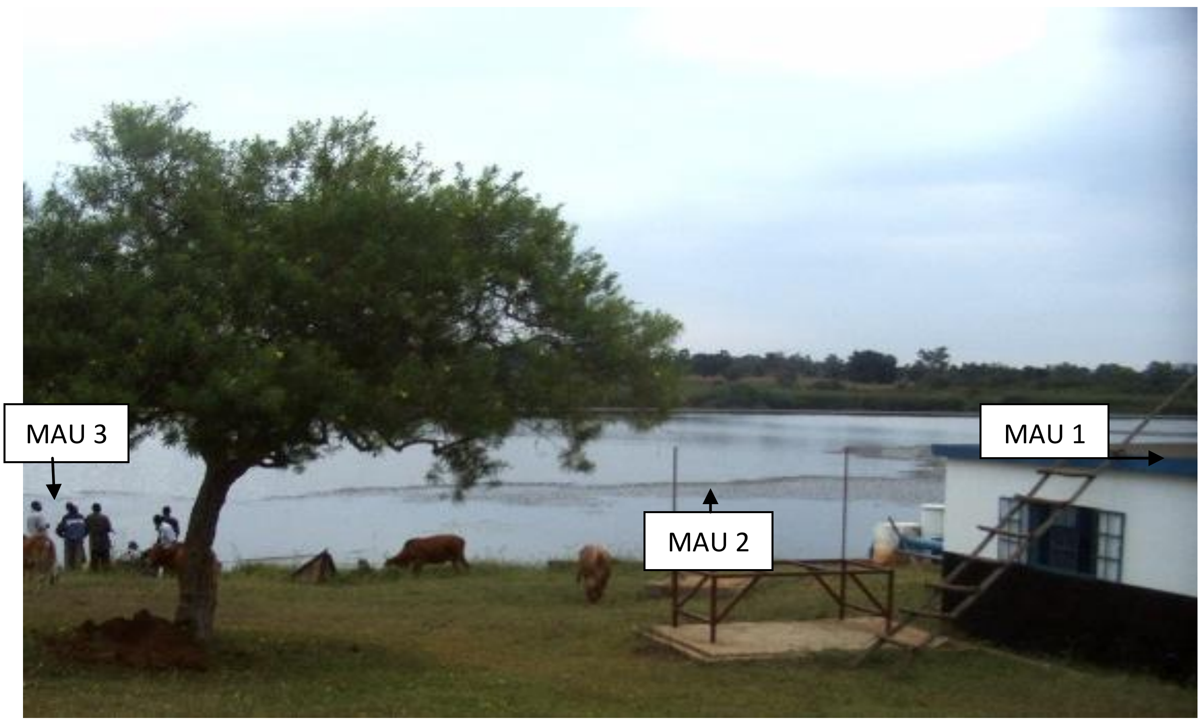



**Plate 3.3 Mauna dam showing sampling stations MAU, MAU2 and MAU3. (Source: Author, 2015)**

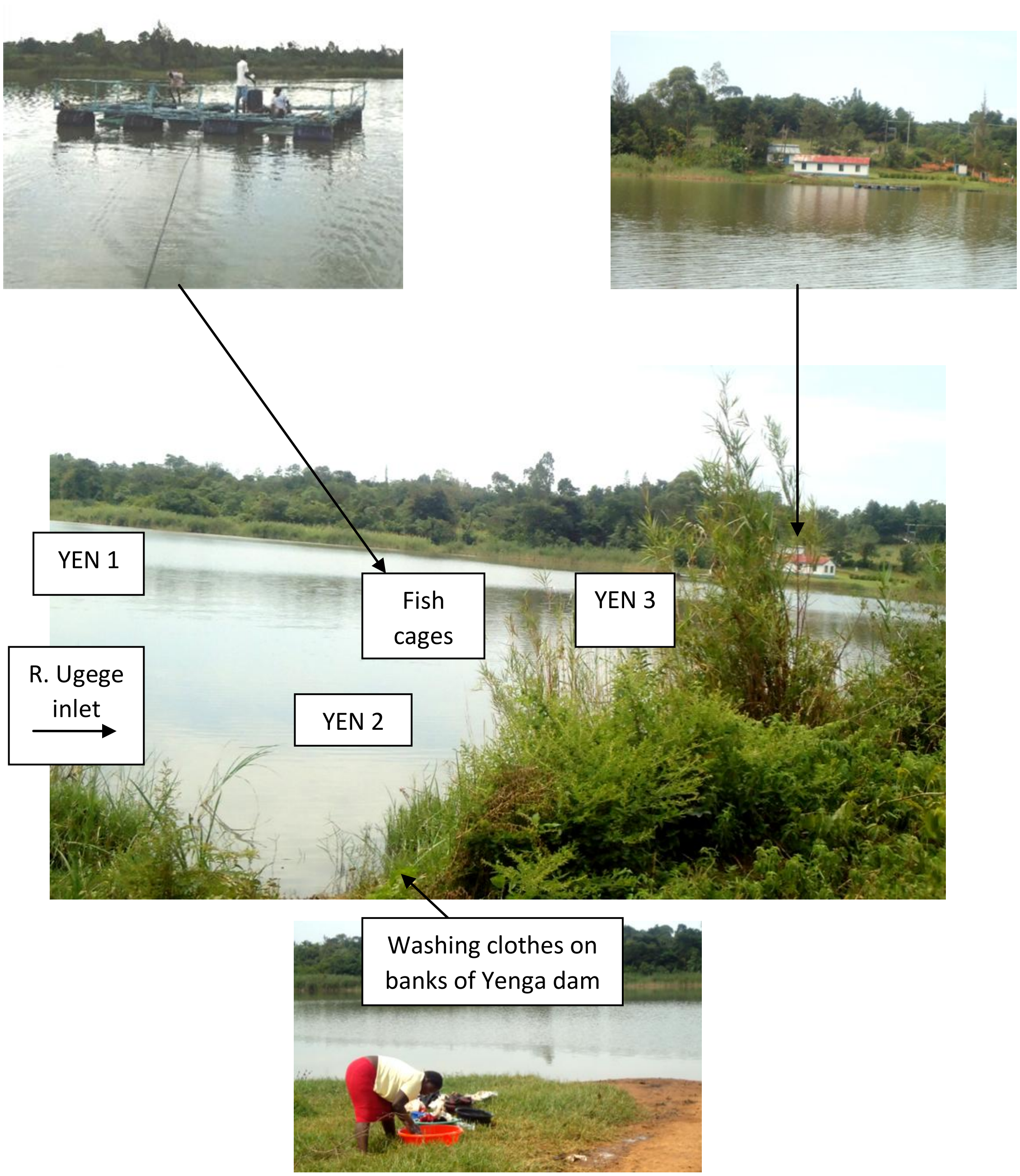



**Plate 3.4 Yenga dam showing sampling stations YEN 1, YEN 2 and YEN 3. (Source: Author, 2015)**

## 3.4 Determination of Water Quality

### 3.4.1 Physico-chemical Water Quality Parameters

Triplicate temperature and pH readings were taken *in situ* by a combined pH-and-temperature-meter, (OAKTON$^R$, Model pH/Mv/ºC METER, Singapore) at a depth of 10 cm below the water surface for each of the macro-habitats and recorded.

The Winkler method (APHA, 1998) was used to determine dissolved oxygen (DO) and biological oxygen demand (BOD). Two sets of triplicate samples were collected in glass stoppered bottles at each sampling station. The first set used to determine DO was fixed using 2ml manganous sulphate followed by 2ml of Winkler's reagent. Dissolved oxygen was then determined by titrating 150ml of sample with a standardized 0.025M sodium thiosulphate solution. The amount of dissolved oxygen (DO) in mgl$^{-1}$ was calculated as:

DO = ________

where

$C$ = the volume of thiosulphate used

$C_b$ = concentration of thiosulphate

$S_v$ = volume of sample used

$S_b$ = volume of manganese+ Winkler's reagent added

Samples for BOD were wrapped using aluminium foil immediately after collection, stored in a dark box and transferred to a dark cabinet in the laboratory. On the fifth day the amount of DO was calculated as explained above and the difference between the initial and final concentration of oxygen in sample was used to derive BOD.

### 3.4.2 Nutrients

Sampling for nutrients (phosphorus, nitrogen) was done by collecting triplicate samples from each sampling station using plastic sampling bottles before sampling for macro invertebrates to prevent contamination. The samples were fixed with 3 drops of concentrated sulphuric acid in the field and transported in a cool-box to maintain the nitrogen balance to the laboratory for further analyses.

Total Nitrogen was determined using the Kjeldahl method (APHA, 2000). A 50 ml water sample was taken in a conical flask to which 2 ml of ammonium chloride solution was added. The solution was then mixed well. The first 10 ml of the sample was run through the cadmium column and discarded, and the following 25 ml of solution was collected in a conical flask to which 0.5 ml of sulphanilamide solution was added. After about 5 minutes, 0.5 ml of n-1 naphthyl- ethylene diaminedihydrochloride was added and the solution was mixed well. After 1.5 hours, the absorbance of the solution was measured at a wavelength of 543 nm in a Spectrophotometer (Pharmacia Biotech model, 65455). A 50 ml of distilled water and different standard solution were also treated as above. Total nitrogen (TN) in $mgl^{-1}$ was calculated as follows:

where

$E_0$ = absorbance of sample without reductant

$E_{B1}$ = absorbance of distilled water + reagent

$E_1$ = absorbance of sample with reagent

Total phosphorus was measured using the Persulfate digestion method (APHA, 2000). A 100 ml of mixed reagent was added to a flask with 100 ml of sample. Simultaneously, 10 ml of mixed reagent without reductant was added to another flask with 100 ml of sample and the sample thoroughly mixed. After 1.5 hours, extinction coefficient of the solution was measured at a wavelength of 885 nm in a spectrophotometer (Pharmacia Biotech model, 65455). The absorbance of the reagent and distilled water blank was also measured. The total phosphorus content of the sample (TP) in µg/l was calculated as follows:

where

F = Sample concentration of in µg/L

absorbance of sample without reductant

absorbance of sample with reductant

absorbance of distilled water + reagent

## 3.5 Phytoplankton Community Structure and Primary Productivity

### 3.5.1 Community Structure of Phytoplankton

Phytoplankton were collected using plankton net of mesh size 20µm from a depth of 10 cm below the surface and immediately preserved using 1ml Lugol's solution. Phytoplankton was concentrated in the laboratory by sedimentation by leaving the samples in the bottles un-disturbed and the supernatant decanted off leaving a final volume of about 20 ml. The known volume of the concentrated sample was used to identify and count the phytoplankton, utilizing an inverted microscope (IMT-2, Model) at X100 magnification. The phytoplankton species were identified using methods described by (APHA, 2003).

### 3.5.2 Primary Productivity of Phytoplankton

Primary productivity was determined monthly using the modified Winkler's dark and light bottle method as described by Wetzel and Likens (1991). Net primary productivity (NPP), gross primary productivity (GPP) and respiration rates were calculated from differences in oxygen concentrations. Two ground stoppered and leak-proof BOD bottles and one dark painted, wax coated BOD bottles were used. Of the first two bottles one served as control bottle while the other as light bottle. Water samples were taken and used to fill 250 ml BOD bottles. Before the start of the experiment the control bottle (100 ml) was filled with water samples (using a plastic tube) and fixed immediately with $MnSO_4$ and alkaline KI to determine $O_2$ concentration at time zero ($t_o$) using Winkler method as described in section 3.2.1.

The experimental bottles were filled and incubated for a period of 4-6 hours in the light incubator (light bottles, LB) and in the dark (DB), at constant water temperature. The $O_2$ concentrations were determined in all bottles (Winkler method). The changes in $O_2$ concentrations in the light bottles were due to photosynthesis and respiration. The changes in $O_2$ concentrations in the dark bottles were due to respiration only.

When oxygen concentration ($mgl^{-1}$) was calculated using the formula in section 3.2.1, primary production (NP and GP) and respiration (R) were calculated as follows:

$$R\ [mgO_2 l^{-1} hr^{-1}] = C_{T0} - C_{DB} \quad \ldots\ldots(i)$$

Where

$C_{T0}$ = oxygen concentration at time zero

$C_{DB}$ = oxygen concentration in the dark bottle

T = incubation time (hr)

Net production (NP) is calculated from the equation:

$$NP\ [mgl^{-1} hr^{-1}] = C_{LB} - C_{T0} \quad \ldots\ldots(ii)$$

where

$C_{LB}$ = oxygen concentration in the light bottles at the end of the incubation

$C_{T0}$ = oxygen concentration at time zero

Gross production (GP) is calculated as follows:

$$GP\,[mgO_2 l^{-1} hr^{-1}] = NP + R = C_{LB} - \text{..............................................................................(iii)}$$

Where

$C_{LB}$ = oxygen concentration in the light bottles at the end of the incubation

## 3.6 Macroinvertebrates Community Structure and Biomass

### 3.6.1 Macroinvertebrate Community Structure

Macroinvertebrates were collected using a scoop-net of 0.5mm mesh size with 0.4 m diameter on a monthly basis for a period of seven months from November 2010 to May 2011. Triplicate random samples of macro-invertebrates in each of the water bodies were taken from each station. During sampling, three standard sweeps from around the sediments along the plant stems to the water surface in a one- square metre were made for two minutes at each locality. The net was moved back and forth and to and fro during these sweeps. Macroinvertebrate samples were sorted alive in a white plastic tray and placed into vials and preserved with 70% ethanol. Macroinvertebrate samples were then transported to the laboratory for further sorting, counting and identification which was done using identification keys (Merritt & Cummins, 1996). All the specimens were identified at genera levels.

Relative abundance (R.A.) was calculated as the proportionate percentage (by numbers) of each taxon in a sample. The relative abundance was calculated according to Roy, *et. al*., (2001) as:

RA =

Where

n = Number of individuals of one taxon

N = Total number of individuals in a station

$p_i$ = Proportion of the *i*th species

100 = Percentage conversion

Taxon diversity was done in all stations using the Shannon-Weiner and Simpsons diversity indices which were calculated as follows:

Shannon-Weiner Index of diversity as described by Ludwig and Reynolds (1988):

where

$H'$ = Shannon diversity,

ln = natural logarithm

$p_i$ = Proportion of the *i*th species

Simpson's index of diversity was calculated as follows (Ludwig and Reynolds, 1988):

where

n = total number of individuals in the sample

ni= number of individual in the ith species in the sample

### 3.6.2 Biomass of Macroinvertebrates

Samples of macroinvertebrates were washed through a screen with mesh size of 6 mm and then hand sorted. They were then separated into taxonomic groups, counted and weighed. They were dried with blotting paper then wet weight of each group was recorded per station. They were then dried at 105ºC for 24 hours. The live weight of the most abundant genera, which include inorganic shells and the water trapped within the body cavities, was multiplied by the conversion factor to convert them to dry weights. The factor was obtained by drying a total number of all the representative species at 105ºC for 24 hours then a regression analysis conducted to determine the dry-wet relationship. Biomass for *Limnaea sp*. and *Gammarus sp.* in Uasin-Gishu and Siaya counties respectively were obtained directly by multiplying their wet weights by the conversion factor since they were the most abundant taxa. Apart from Limnaeaidae which do not lose weight when preserved, all the other other taxa do (Johnson and Brinkhurst, 1971); indicating that their weights were likely to be underestimated. However, since they constituted a small fraction, it was assumed that this did not affect the total biomass significantly.

## 3.7 Fish Parasites Prevalence and Mean Intensity

### 3.7.1 Isolation of Fish Parasites

Fish for parasite examination were collected from November 2010 to July 2012. Monthly samples of fish were obtained using seine nets of 2, 3 and 3.5 inch mesh-size. Fish were killed by cerebral commotion and sex determination by observation of genitals. They were measured to obtain total weight (g) and standard length (cm), which were used to determine the Fulton's condition factor as follows:

where

weight

length

Pasitological examination was conducted using standard necropsy procedures (Thoesen, 1994). Internal examinations were done through stretching out the intestines and stomach in a Petri dish and cut open longitudinally. The contents were examined under a compound microscope at a magnification of X100. Internal organs were examined by cutting a small piece from each, put on glass slide, squashed by use of a cover slip and observed under a compound microscope. Metacercariae of trematodes were released from their cysts for better examination by teasing them out of the membranous coat. All parasites observed were counted, identified and recorded. Isolated parasites were fixed and stored in 70% alcohol and preserved in 10% formalin (Paperna, 1996) for identification. Staining and mounting of parasite specimens were done by procedures described by Pritchard and Kruse (1982). Specimens were passed through a graded series of concentrations of alcohol from 70, 50, 35% and finally in distilled water to bring them to the level of the stain. They were stained using Mayer's hematoxylin stain then dehydrated through a series of graded alcohol concentrations of 35, 50, 70, 85, 95 and absolute alcohol. Clearing of specimens was done with xylene after which they were mounted on glycerin. Further identification of parasites was by use of morphological features and identification keys described by Paperna (1996), Yamaguti (1971) and Khalil *et al*., (1994).

### 3.7.2 Parasite Prevalence Rates

Prevalence levels of tilapia parasites in Kesses, Kerita, Mauna and Yenga were quantified according to Bush *et al.*, (1997) as follows:

### 3.7.3 Mean Intensity Levels of Parasites

Mean intensity (MI) of parasites in each of the water bodies was determined according to Bush *et al.*, (1997) as follows:

### 3.8 Data Analysis

Water quality physico-chemical parameters were summarized for the study period as mean ± SE values for each sampling station. Spatial variation in macroinvertebrate and phytoplankton community and physico-chemical parameters was tested using a two-way ANOVA at 95% confidence limits. Physico-chemical parameters and nutrient levels were correlated with macroinvertebrate and phytoplankton composition, abundance and diversity using Spearman's correlation analysis.

Each parasite's importance value was calculated based on prevalence and used to verify the importance of each species of parasites. Species with a prevalence higher than 66.66% were considered as central, between 33.33 and 66.66% secondary, and less than 33.33% satellite (Flores & Baccala, 1998); (Campos *et al.*, 2009). The relationship between host total length, parasite abundance and condition factor was determined by Spearman rank correlation.

Rainfall levels were classified into two seasons. Low rainfall season corresponded to rainfall between 0 and to 99mm while high was 100mm and above. The relationship between seasonality and parasite abundance was evaluated by the Kruskal Wallis Test. Analysis of variance (ANOVA) was used to determine the effect of parasites during low and high rainfall in male and female fish. Mann Whitney (U) test was used to determine the effect of host sex on abundance of each species. Principal component analyses (PCA) and Canonical Correspondence Analyses (CCA) (Shostak *et al.*, 1987; Flores & Baccala,

1998) of water quality, phytoplankton, macroinvertebrates and parasites were used to determine water quality components affecting each water body. Only parasites with prevalences higher than 10% were included in analyses (Bush *et al.* 1997). Data was log transformed before analysis was done. Analysis was done using MINITAB™ version 14.0 and Statgraphics 2.1 statistical programs. All statistical significances were tested at 95% levels.

# CHAPTER FOUR

# RESULTS

## 4.1 Water Quality

### 4.1.1 Physico-chemical Parameters

Mean physico-chemical water quality parameters at Kesses, Kerita, Yenga and Mauna dams from November 2010 to December 2011 are as presented in Table 3.1. At Kesses, water quality parameters between different stations did not have any significant differences. The highest pH recorded was at station KES 2 which had the highest concentration of emergent and submerged aquatic plants with a mean of 7.623 ± 0.34. Station KES 5 recorded the lowest pH of 7.08±0.23. The highest mean temperature was recorded at station KES 6 (22.68±0.55 ºC). The open waters (station KES OW) recorded the lowest mean temperature of 21.55±0.40$^{o}$C. KES 5 station near the inlet of River Endaragweta had the highest mean dissolved oxygen (9.79±2.82 mgl$^{-1}$) and the lowest was recorded at station KES 1 near the dam's outlet (6.90±0.69 mgl$^{-1}$). Similar to pH, the highest mean BOD was recorded at station KES 2 (4.09±0.82 mgl$^{-1}$) and lowest at station KES 3 near the landing site at 2.82±0.443 mgl$^{-1}$ (Table 4.1).

In Kerita, there were no significant differences between the stations for pH, DO and BOD. Station KER 2 which was found close to farmland and homesteads and with a lot of aquatic vegetation had the highest pH (7.58±0.15) and temperature (24.10±0.76). The lowest pH station (7.29±.11) was at KER 3 close to River Cheboloi inlet. The mean dissolved oxygen was highest in station KER 2 (7.08±1.24) while station KER 1, which was close to the washing and animal drinking point recorded the lowest (6.62±1.29).

Likewise, BOD was highest in station KER 3 (3.80±0.40) and lowest in station KER 2 (2.21±0.80).There was a significant difference between the sampling stations (F=10.49, p=0.002) in DO in Kerita (Table 4.1). In Siaya, the highest mean pH was 7.72 ± 0.14 in Mauna at station MAU 2, which was found close to the water treatment works and its staff quarters. The lowest pH was 7.29 ±0.14 in Yenga at station YEN 2 near the animal drinking and washing point (Table 4.1). Temperature increased gradually from February and peaked in June in all the water bodies (Figure 4.1).

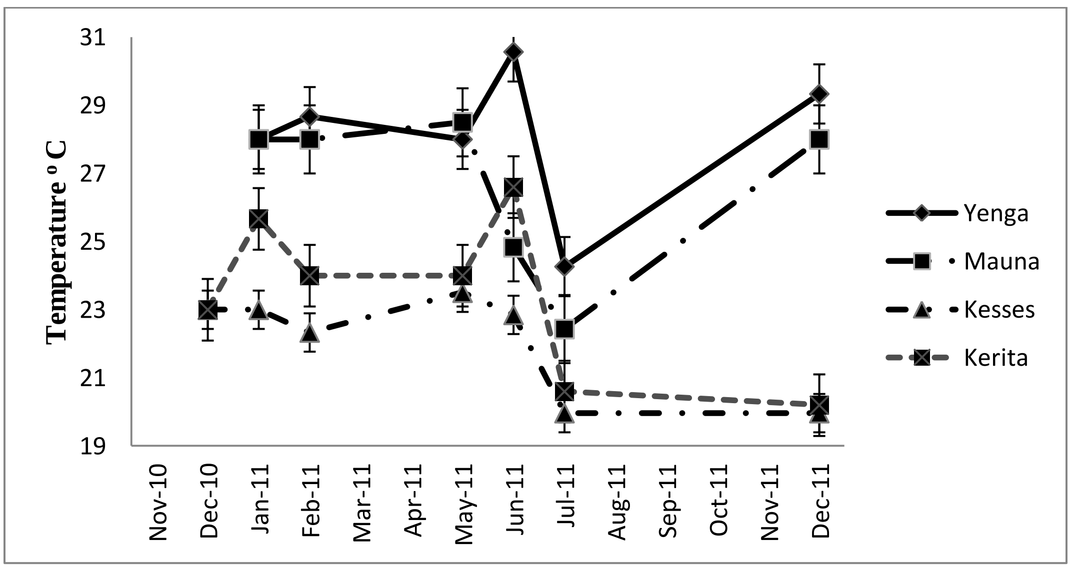



**Figure 4.1 Monthly variation in mean temperature (in $^{o}$C ) (± SEM) in Kesses and Kerita dams in Uasin Gishu and Mauna and Yenga dams in Siaya during the period between November 2010 and December 2011.**

**Table 4.1 Mean (± SEM) of physico-chemical water quality parameters measured in of Kesses, Kerita, Yenga and Mauna Dams at different sampling stations during the period between November 2010 and June 2011.**

| County | Station | Physico-chemical parameters | | | |
|---|---|---|---|---|---|
| | | **pH** | **Temp.** | **BOD** | **DO** |
| **Uasin-Gishu** | KES 1 | 7.585 ±0.41 | 21.95 ±0.37 | 3.56 ±0.634 | 6.90 ±0.69 |
| | KES 2 | 7.623 ±0.34 | 22.6 ±0.92 | 4.09 ±0.82 | 8.17 ±0.96 |
| | KES 3 | 7.575 ±0.33 | 22.07 ±0.63 | 2.82 ±0.443 | 7.09 ±0.44 |
| | KES 4 | 7.448 ±0.25 | 21.93 ±0.71 | 3.87 ±0.92 | 7.63 ±1.16 |
| | KES 5 | 7.085 ±0.23 | 21.75 ±0.57 | 3.28 ±1.14 | 9.79 ±2.82 |
| | KES 6 | 7.33 ±0.24 | 22.68 ±0.55 | 4.08 ±0.69 | 8.33 ±0.71 |
| | KES OW | 7.58 ±0.34 | 21.55 ±0.4 | 4.04 ±0.43 | 8.04 ±0.31 |
| | KER 1 | 7.38 ±0.07 | 23.50 ±0.78 | 3.16 ±0.80 | 6.62 ±1.29 |
| | KER 2 | 7.58 ±0.15 | 24.10 ±0.76 | 2.21 ±0.80 | 7.08 ±1.24 |
| | KER 3 | 7.29 ±0.11 | 23.40 ±0.97 | 3.80 ±0.40 | 6.79 ±1.07 |
| **Siaya** | YEN 1 | 7.29±0.21 | 25.44±1.49 | 3.67±0.65 | 7.40±0.27 |
| | YEN 2 | 7.29±0.14 | 25.84±1.43 | 3.61±0.65 | 7.24±0.38 |
| | YEN 3 | 7.45±0.15 | 26.10±1.39 | 3.45±0.80 | 6.75±0.48 |
| | MAU 1 | 7.72±0.14 | 24.18±0.64 | 2.91±0.69 | 5.63±0.51 |
| | MAU 2 | 7.58±0.22 | 23.40±0.49 | 2.94±0.53 | 5.24±0.84 |
| | MAU 3 | 7.32±0.18 | 24.48±0.59 | 2.61±0.54 | 4.98±0.88 |

The highest mean temperature was 26.10±1.39$^{o}$C recorded at station YEN 3 next to the pump house while the lowest was 23.40±0.49 $^{o}$C in MAU 2. The highest mean BOD of 3.67±0.65 mgl$^{-1}$ was at station YEN 1 where macrophyte cover was high compared to other parts of the dam and also had low human and animal activity dam. The lowest BOD was at station MAU 3 (2.61±0.54 mgl$^{-1}$) in Mauna. Mean dissolved oxygen was highest in station YEN 1 (7.40±0.27 mgl$^{-1}$) close to the inlet in Yenga. Station MAU 3 which had a dense cover of submerged plants, had dark water color and foul small had the DO (4.98±0.88 mgl$^{-1}$) (Table 4.1). There were no significant differences between the stations within Siaya dams for all water quality parameters.

There was variation in temperature at the four dams; the lowest mean temperature measurement was observed in Kesses dam, it ranged between 21$^{o}$C in December and 23$^{o}$C in the months of February, March and July (Table 4.2a). Kerita dam also had the lowest temperature of 21$^{o}$C during the month of April but had a high mean measurement of 24$^{o}$C in November and May. Yenga dam had the highest mean temperature that ranged between 25$^{o}$C in November and February and 28$^{o}$C in April and May. Significant differences in temperature were found between Siaya and Uasin Gishu ($F = 17.38$; $p = 0.000$). The range of pH measurement at Kesses dam was 7.1 to 7.8 with high mean measurement in January (7.8 ±0.2) and April (7.7 ±0.40) (Table 4.2a) and these too were the highest mean measurements observed in all of the dams. The highest BOD measurements were in Kesses at 4.1±0.10 mgl$^{-1}$ in November 2010 and 4.5±0.12 mgl$^{-1}$ in May 2011 (Table 4.2a).

**Table 4.2a Mean monthly Physico-chemical water quality parameters in Kesses and Kerita dams in Uasin Gishu from November 2010 to June 2011.**

| Month/Year | Kesses Dam | | | | Kerita Dam | | | |
|---|---|---|---|---|---|---|---|---|
| | **pH** | **Temp ($^oC$)** | **DO ($mgl^{-1}$)** | **BOD ($mgl^{-1}$)** | **pH** | **Temp ($^oC$)** | **DO ($mgl^{-1}$)** | **BOD ($mgl^{-1}$)** |
| **Nov-10** | 7.4±0.22 | 22±0.44 | 7.5±0.45 | 4.1±0.10 | 7.1±0.10 | 24±0.90 | 6.5±0.55 | 3.1±0.10 |
| **Dec-10** | 7.3±0.31 | 21±0.25 | 7.1±0.48 | 3.5±0.12 | 7.4±0.15 | 23±0.15 | 7.2±0.40 | 3.2±0.15 |
| **Jan-11** | 7.8±0.20 | 22±0.20 | 6.9±0.52 | 3.6±0.23 | 7.2±0.12 | 22±0.58 | 6.7±0.50 | 3.5±0.18 |
| **Feb-11** | 7.5±0.23 | 23±0.33 | 7.0±0.50 | 3.4±0.15 | 7.3±0.09 | 22±0.18 | 7.0±0.45 | 3.0±0.12 |
| **Mar-11** | 7.6±0.12 | 23±0.42 | 7.4±0.46 | 3.8±0.25 | 7.5±0.18 | 23±0.80 | 7.2±0.80 | 2.8±0.20 |
| **Apr-11** | 7.7±0.40 | 21±0.36 | 6.5±0.42 | 3.9±0.18 | 7.5±0.15 | 21±0.90 | 6.8±0.25 | 3.0±0.14 |
| **May-11** | 7.4±0.15 | 22±0.45 | 7.5±0.41 | 4.5±0.12 | 7.4±0.11 | 24±0.85 | 6.7±0.10 | 3.4±0.16 |
| **Jun-11** | 7.2±0.11 | 23±0.38 | 7.1±0.38 | 3.7±0.13 | 7.3±0.16 | 23±0.55 | 6.5±0.30 | 2.9±0.16 |

Dissolved oxygen also varied in the different water bodies. Mauna dam had the lowest DO at 5.6±0.45 mg/ in November and5.1±0.40 $mgl^{-1}$ in December 2010 (Table 4.2b). All other dams had DO levels between 6.5 – 7.5 $mgl^{-1}$. Mauna also had the lowest BOD measurements at 2.6±0.45 $mgl^{-1}$ in May 2011 and 2.7±0.20 $mgl^{-1}$ in November 2010. In Kesses, DO were highest in January then dropped to the lowest in February and peaking again in June followed by a large drop in July. In Kerita, the trend was similar to Kesses only that drop in DO in July was greater.

**Table 4.2b Mean monthly water quality physico-chemical parameters in Mauna and Yenga dams in Siaya during the period between November 2010 to June 2011.**

| Month/Year | Yenga Dam | | | | Mauna Dam | | | |
|---|---|---|---|---|---|---|---|---|
| | **pH** | **Temp (°C)** | **DO (mgl$^{-1}$)** | **BOD (mgl$^{-1}$)** | **pH** | **Temp (°C)** | **DO (mgl$^{-1}$)** | **BOD (mgl$^{-1}$)** |
| **Nov-10** | 7.4±0.15 | 25±0.55 | 7.0±0.20 | 3.5±0.20 | 7.6±0.30 | 24±0.55 | 5.6±0.45 | 2.7±0.20 |
| **Dec-10** | 7.3±0.20 | 26±0.75 | 6.9±0.35 | 3.2±0.10 | 7.5±0.25 | 23±0.60 | 5.1±0.40 | 2.9±0.40 |
| **Jan-11** | 7.5±0.18 | 27±0.70 | 7.1±0.30 | 3.6±0.15 | 7.3±0.20 | 24±0.77 | 5.2±0.85 | 3.0±0.65 |
| **Feb-11** | 7.4±0.25 | 25±0.65 | 7.2±0.15 | 3.5±0.11 | 7.4±0.20 | 26±0.73 | 5.4±0.56 | 3.2±0.30 |
| **Mar-11** | 7.2±0.10 | 26±0.40 | 7.2±0.10 | 3.4±0.25 | 7.4±0.10 | 25±0.75 | 5.4±0.20 | 2.8±0.35 |
| **Apr-11** | 7.3±0.15 | 28±0.45 | 6.7±0.25 | 3.6±0.27 | 7.2±0.15 | 24±0.45 | 5.2±0.10 | 3.2±0.45 |
| **May-11** | 7.3±0.12 | 28±0.60 | 7.1±0.20 | 3.7±0.22 | 7.2±0.12 | 24±0.35 | 5.4±0.15 | 2.6±0.45 |
| **Jun-11** | 7.2±0.16 | 27±0.50 | 7.1±0.30 | 3.5±0.12 | 7.4±0.16 | 25±0.54 | 5.5±0.35 | 2.8±0.56 |

The lowest mean pH measurement was at Kerita dam during the month of November and measured 7.1 ± 0.1.Levels of pH increased from the month of February and reached their peak in June and then dropped to their lowest in July in all the water bodies (Figure 4.2).

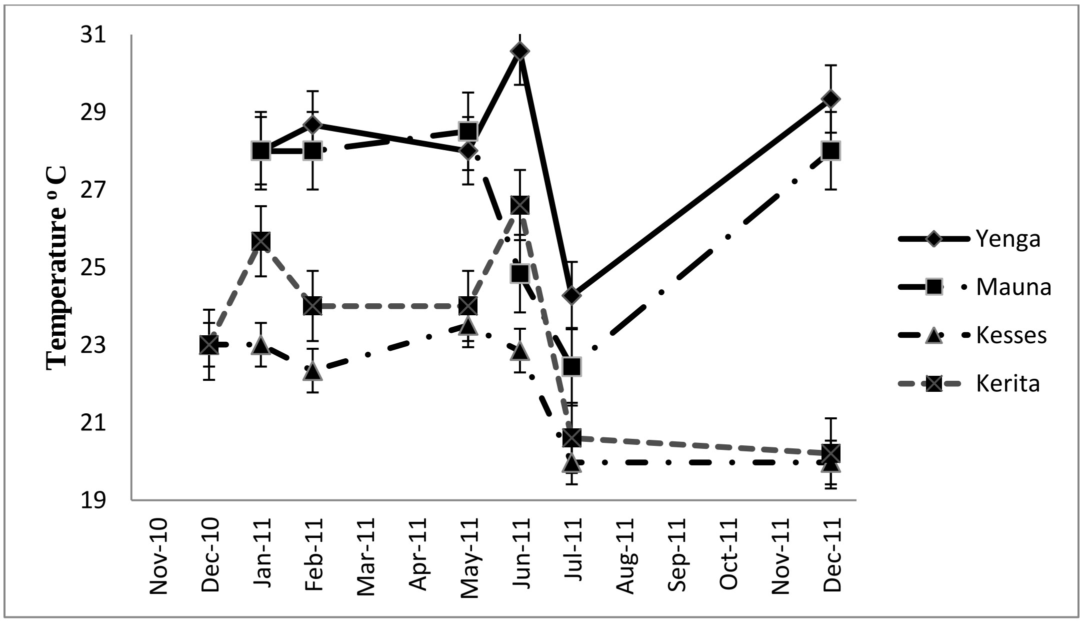



**Figure 4.2 Monthly variation of mean pH (± SEM) in Kesses and Kerita dams in Uasin Gishu County and Mauna and Yenga dams in Siaya County during the period between November 2010 and December 2011.**

Yenga had the least fluctuations in DO. Mauna had high peaks in DO in May and low levels in July (Figure 4.3). Variation in pH and BOD was observed although not statistically different between dams (Figure 4.4).

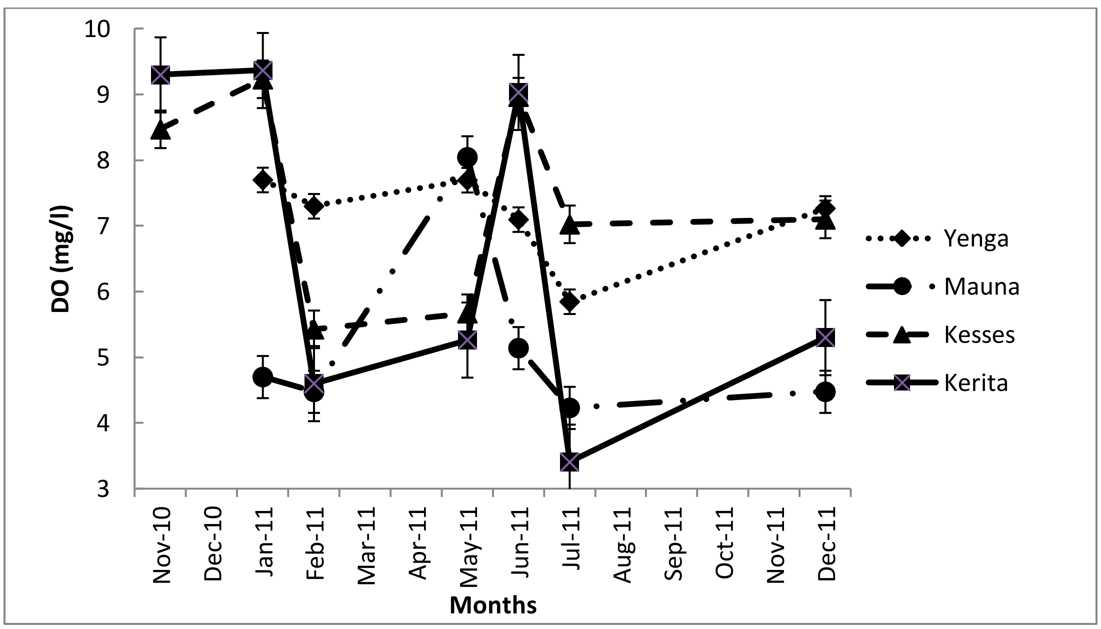



**Figure 4.3 Monthly variation of mean DO (in $mgl^{-1}$) (± SEM) in Kesses and Kerita dams in Uasin Gishu County and Mauna and Yenga dams in Siaya County during the period between November 2010 and December 2011.**

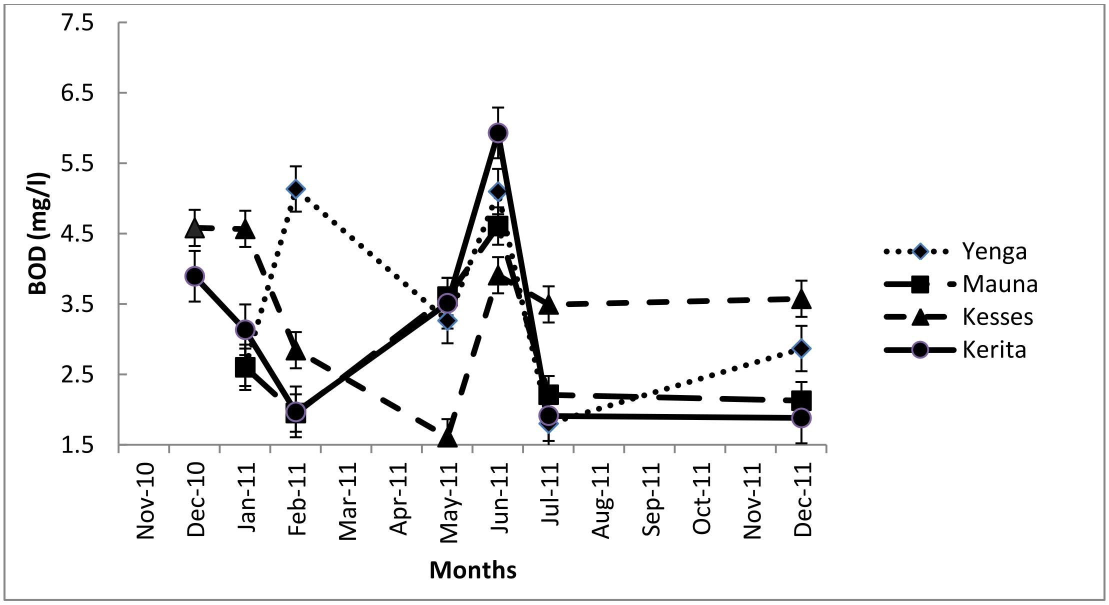



**Figure 4.4 Monthly variation of mean BOD (in mgl$^{-1}$) (± SEM) in Kesses and Kerita dams in Uasin Gishu County and Mauna and Yenga dams in Siaya County during the period between November 2010 and December 2011.**

### 4.1.2. Nutrients

Total nitrates in Uasin Gishu were relatively higher compared to Siaya. Total nitrates levels were generally higher than phosphates in all stations of the four dams in this study. Ranges for total nitrates were between 1.3±0.001 to 1.81±0.05 mgl$^{-1}$ in Uasin Gishu while in Siaya the range was 0.6± 0.05 to 1.2±0.10 (Table 4.3). Total phosphorus levels in the two counties ranged from 0.06±0.008 mgl$^{-1}$ to 0.18±0.009 mgl$^{-1}$ in Uasin Gishu and 0.06± 0.002 mgl$^{-1}$ and 0.18± 0.010 mgl$^{-1}$ in Siaya.

**Table 4.3 Mean monthly Total Nitrates (TN) and Total phosphate (TP) levels (in $mgl^{-1}$) in Kesses and Kerita dams in Uasin Gishu and Yenga and Mauna in Siaya from November 2010 to June 2011.**

| Month/Year | Kesses | | Kerita | | Yenga | | Mauna | |
|---|---|---|---|---|---|---|---|---|
| | TN | TP | TN | TP | TN | TP | TN | TP |
| Nov-10 | 1.6±0.05 | 0.18±0.005 | 0.9±0.01 | 0.10±0.001 | 0.9±0.01 | 0.10±0.001 | 1.1±0.05 | 0.15±0.004 |
| Dec-10 | 1.5±0.08 | 0.09±0.002 | 0.8±0.02 | 0.09±0.001 | 0.8±0.02 | 0.09±0.001 | 0.9±0.07 | 0.15±0.005 |
| Jan-11 | 1.6±0.10 | 0.12±0.009 | 1.1±0.02 | 0.06±0.002 | 1.1±0.02 | 0.06±0.002 | 1.1±0.04 | 0.18±0.003 |
| Feb-11 | 1.3±0.06 | 0.20±0.012 | 0.6±0.03 | 0.09±0.003 | 0.6±0.03 | 0.09±0.003 | 0.7±0.05 | 0.16±0.001 |
| Mar-11 | 1.4±0.12 | 0.18±0.009 | 1.1±0.05 | 0.08±0.002 | 1.1±0.05 | 0.08±0.002 | 0.8±0.09 | 0.11±0.005 |
| Apr-11 | 1.5±0.05 | 0.15±0.007 | 0.9±0.02 | 0.10±0.001 | 0.9±0.02 | 0.10±0.001 | 1.2±0.10 | 0.18±0.010 |
| May-11 | 1.6±0.15 | 0.14±0.006 | 0.6±0.01 | 0.06±0.004 | 0.6±0.01 | 0.06±0.004 | 1.1±0.08 | 0.14±0.002 |
| Jun-11 | 1.4±0.09 | 0.16±0.008 | 0.8±0.05 | 0.08±0.002 | 0.8±0.05 | 0.08±0.002 | 0.6±0.03 | 0.16±0.009 |

Fluctuations in total nitrates were more pronounced in Uasin Gishu than in Siaya (Figure 4.5). In both Uasin Gishu and Siaya, total nitrates were high between November and May while low levels were observed between June and July. (Figure 4.5). While Kesses did not have any significant difference with either of the Siaya dams, Kerita had higher total nitrate levels than both Mauna ($F = 6.34$; $p = 0.01$) and Yenga ($F= 9.22$; $P=0.000$;).

Mauna had sharp fluctuations in total phosphates, increasing between January and February then dropping drastically in May (Figure 4.6). The mean total nitrates was highest in KES3 station (1.71±0.08 $mgl^{-1}$) and lowest was at KES OW station (1.22±0.161 $mgl^{-1}$) (Figure 4.7a). The highest mean total phosphorus was 0.25 ± 0.02 $mgl^{-1}$ at KES2 while the lowest TP was 0.03±0.007 $mgl^{-1}$ was recorded at station KES1.

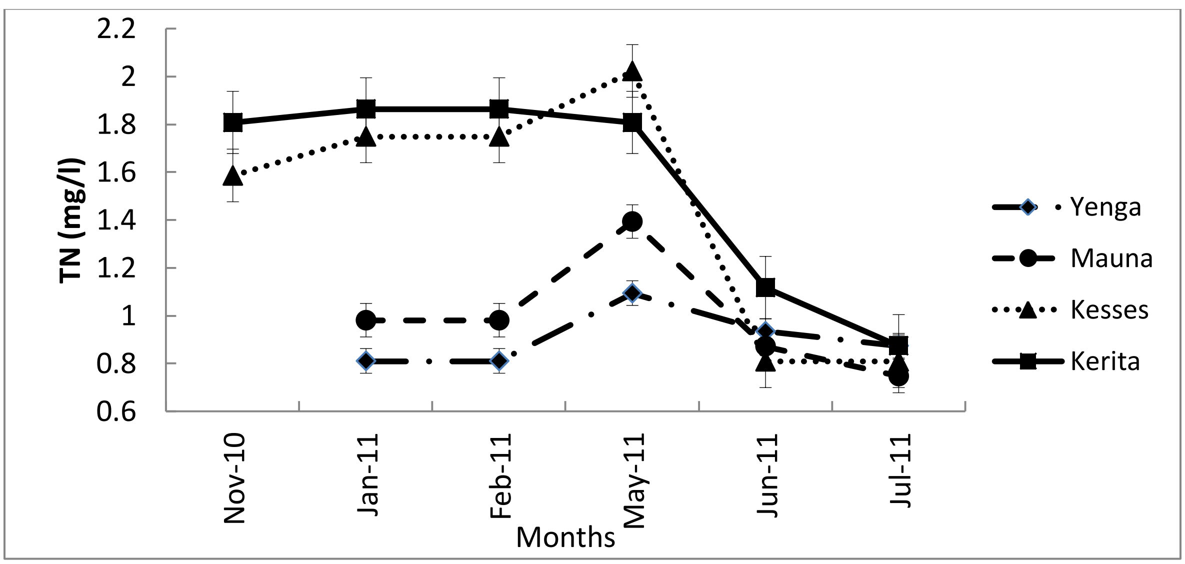



**Figure 4.5 Monthly variation of mean total nitrates (in $mgl^{-1}$) (± SEM) in Kesses and Kerita dams in Uasin Gishu County and Mauna and Yenga dams in Siaya County during the period between November 2010 and December 2011.**

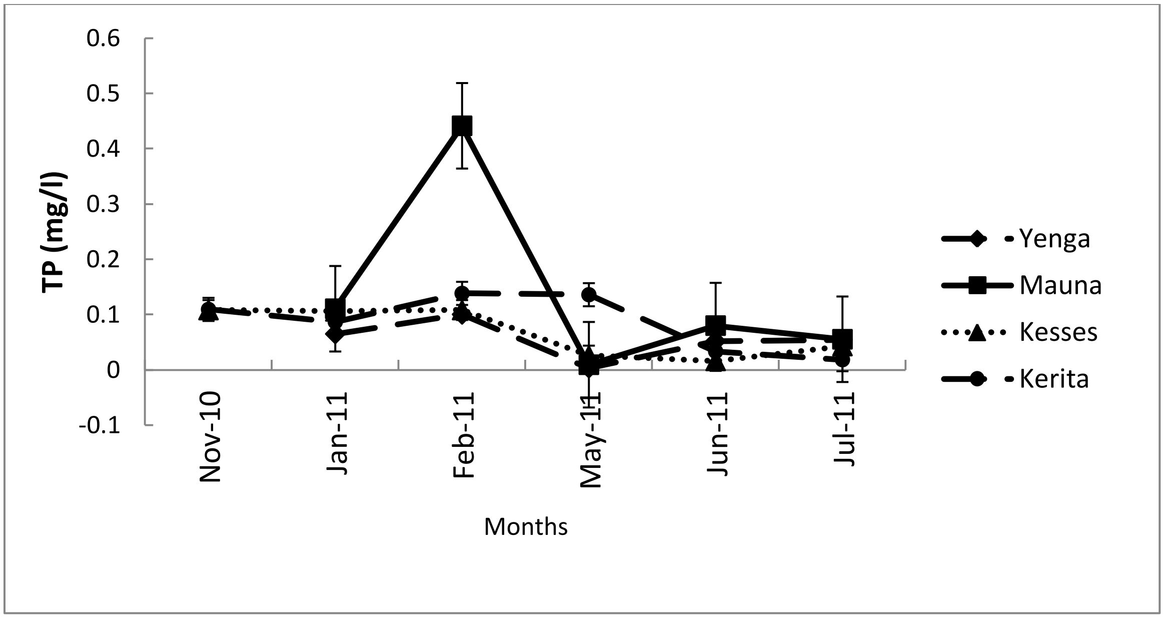



**Figure 4.6 Monthly variation of total phosphates (in mgl$^{-1}$) (± SEM) in Kesses and Kerita dams in Uasin Gishu County and Mauna and Yenga dams in Siaya County during the period between November 2010 and December 2011.**

In Kerita, the stations with the highest and lowest total nitrate and phosphate levels. Were station KER (1.68±0.07 mgl$^{-1}$) and KER 1 (1.553±0.100 mgl$^{-1}$) respectively while the highest total phosphorus was KER 3 (0.0937±0.0319 mgl$^{-1}$) and the lowest KER1 (0.0778±0.0195 mgl$^{-1}$) (Figure 9a). In Siaya, the station with the highest total nitrates and phosphates was MAU 2 (1.05±0.065 mgl$^{-1}$) and MAU 3 (0.17±0.035 mgl$^{-1}$) respectively (Figure 4.7b). The lowest total nitrate level was in YEN 3 (0.78±0.060 mgl$^{-1}$) (Figure 4.7b). There were no significant differences in TN and TP within stations in water bodies in both Mauna and Yenga dams.

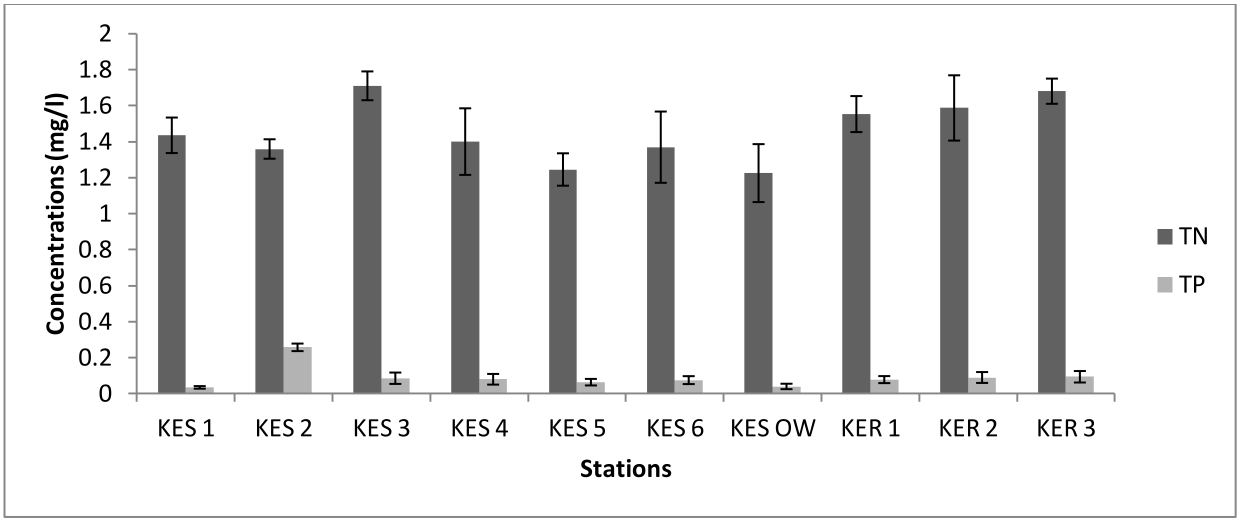



**Figure 4.7a Mean (± SEM) of TN and TP for the Sampling Stations in Kesses and Kerita in Uasin Gishu during this study.**

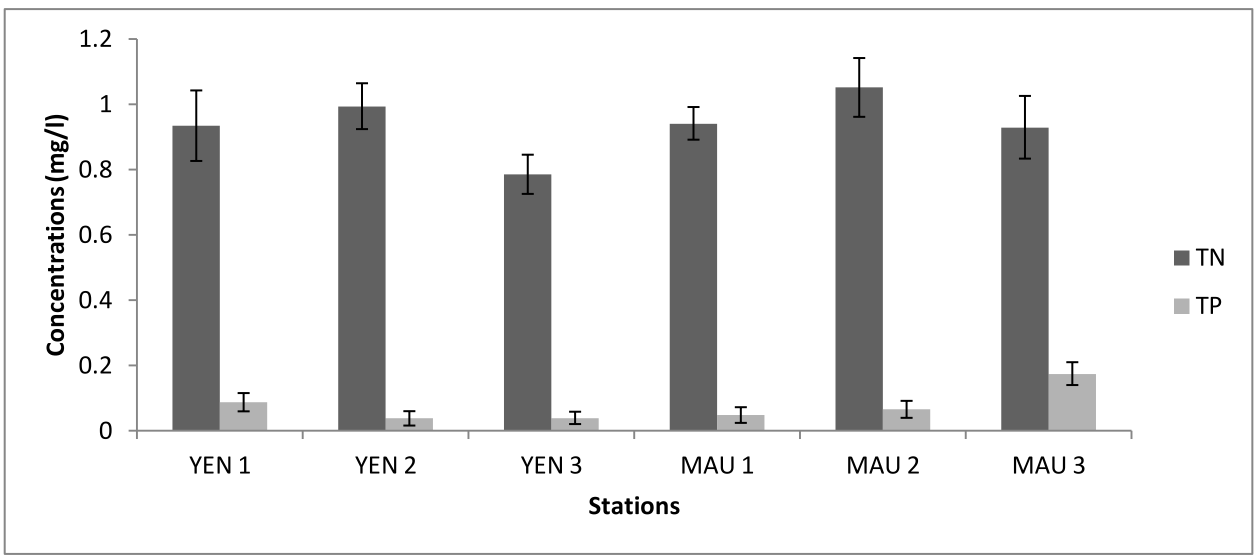



**Figure 4.7b Mean (± SEM) of TN and TP for the Sampling Stations in Yenga and Mauna dams in Siaya during this study.**

Principal component analysis (PCA) showed that both the first and second components described 65.22% of the variation in water quality parameters at the four dams during the sampling period. Component 1 and 2 explained 36.86% and 28.36% variation in water quality (Table 4.4), with eigen values of 2.21 and 1.70 for the two components respectively (Figure 4.8).

**Table 4.4 Principle components contributing to variation in water quality physico-chemical parameters in SWBs in Uasin Gishu and Siaya during this study.**

| PC | Eigenvalue | % variance |
|---|---|---|
| 1 | 2.21173 | 36.862 |
| 2 | 1.70156 | 28.359 |
| 3 | 0.973235 | 16.221 |
| 4 | 0.753502 | 12.558 |
| 5 | 0.204682 | 3.4114 |
| 6 | 0.155288 | 2.5881 |

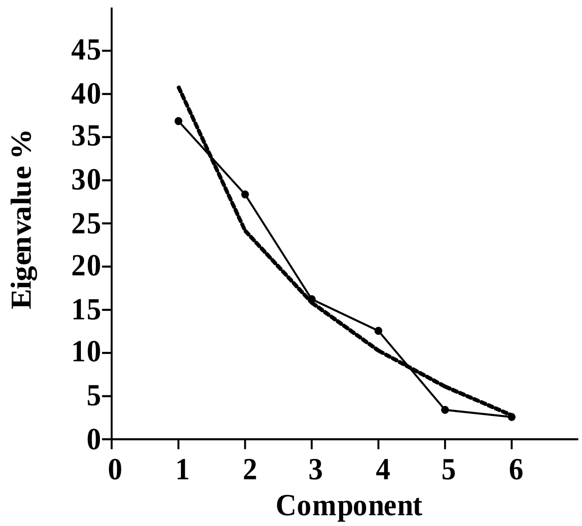



**Figure 4.8 Scree plot analysis for significant principle components contributing to variation in water quality physico-chemical parameters in small SWBs in Uasin Gishu and Siaya during this study.**

Correspondence analysis of water quality parameters indicated that temperature had strong correlations with Yenga and Mauna with the first component explaining 36.86% of the correlation in water quality parameters (Figure 4.9). These correlations in Yenga were associated with the months of April, May and June 2011, which correspond to the wet season in Siaya. In Mauna, the months of February, March and June of the same year and also in the wet season were associated with strong correlations with temperature. In Kesses dam, TN, DO, BOD and pH had strong correlations with component 1 in the months of November 2010, January, April and May 2011, which was in the dry season. Similar observations were made in Kerita in the month of December 2010, February and April 2011 also in the dry season. Component 2 which explained 28.36% of variation in water quality had a strong correlation with TP at Mauna dam in November and December

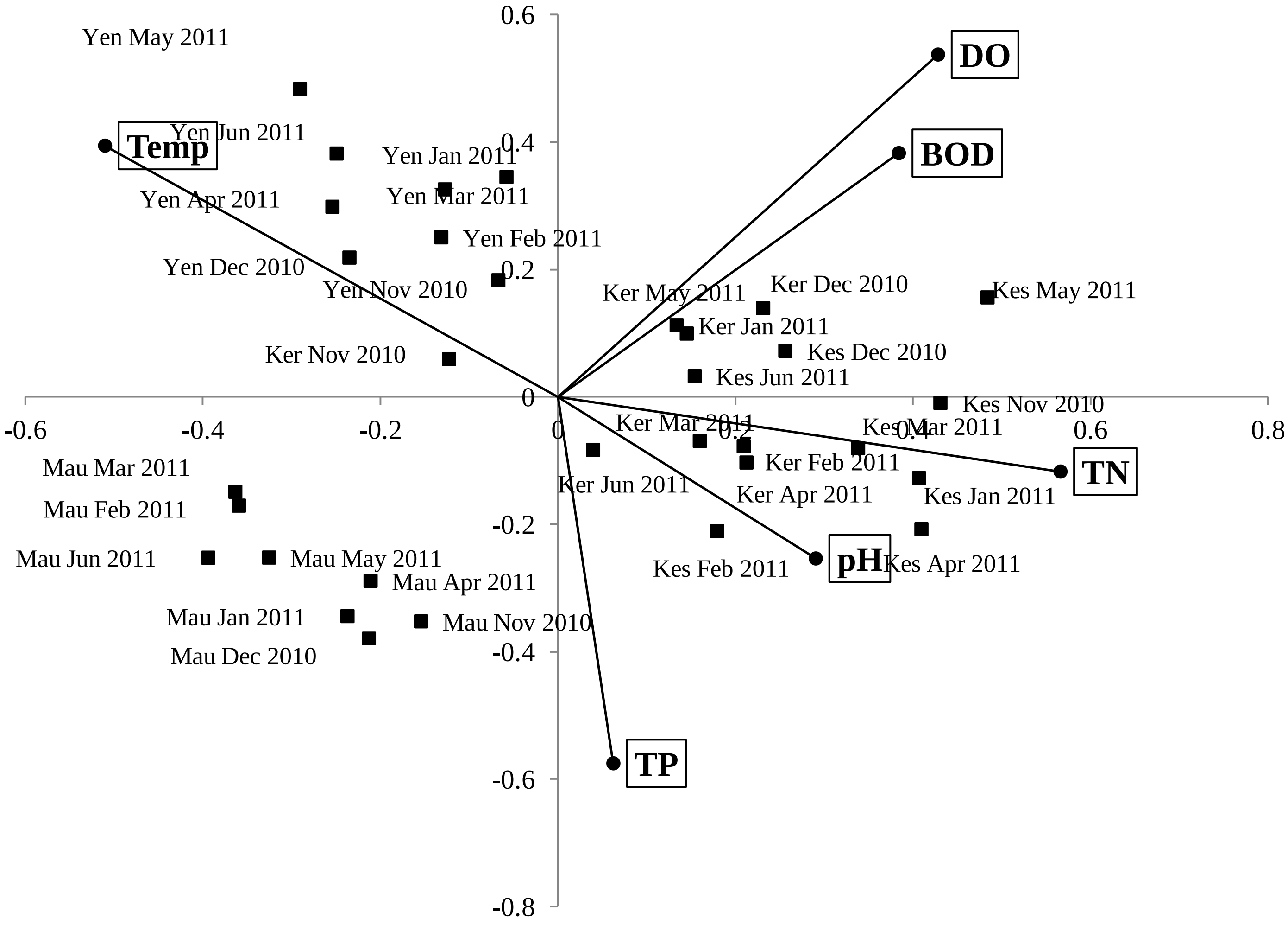



**Figure 4.9. Cannonical correspondence analysis (CCA) ordination plot of temperature (Temp), pH, dissolved oxygen (DO), biological oxygen demand (BOD), total nitrates (TN) and total phosphates (TP) in Kesses, Kerita, Mauna and Yenga dams during the period between November 2010 and December 2011.**

2010, January and April 2011 and also with Kesses dam in the months of February and April 2011.

Comparing the dams at 95% ellipse with the measured water quality parameters, Mauna dam was significantly uncorrelated with the other dams, the furthest being with Yenga dam. Kesses and Kerita dam had a stronger correlation with the first and second component vector matrix. Kerita and Yenga dam were also correlated much with the second component along the vertical axis. This indicated that the dams at Uasin-Gishu county had closer relationship in their water quality parameters measured than the dams at Siaya county (Figure 4.10).

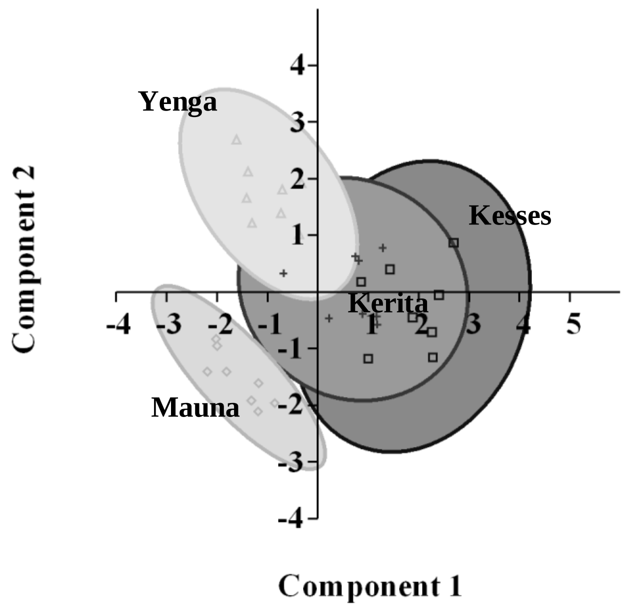



**Figure 4.10 Principle component analysis of ellipse at 95% confidence limits for water quality in Kesses and Kerita in Uasin Gishu and, Mauna and Yenga in Siaya during this study.**

## 4.2 Phytoplankton

### 4.2.1 Community Structure of Phytoplankton

A total of 21 taxa represented seven families were observed (Table 4.5). Genera representing Bacillariophyceae were *Synedra sp., Navicula sp., Melosira sp., Frustilia sp., Tabellaria sp., Cyclotella sp. and Diatoma sp, Taxa* representing Chlorophyceae were *Botryoccoccus sp., Crucigenia sp., Cladophora sp., Coelastrum sp., Tetraspora sp., Spirogyra sp., Pediastrum sp., Scenedra sp.* and *Scenedesmus sp.* Cyanophyceae was represented by *Phormidium, sp.* and *Coelosphaerium sp.* while Desmidiaceae was by *Omarion sp., Gonatozygon sp.* and *Closterium sp.*

**Table 4.5 Genera of phytoplankton found in Kesses, Kerita, Mauna and Yenga during this study. P = Present A = Absent in study site**

| Family | Genera | Kesses | Kerita | Mauna | Yenga |
|---|---|---|---|---|---|
| **Bacillariophyceae** | *Synedra sp.* | P | P | A | A |
| | *Navicula sp.* | P | P | P | A |
| | *Melosira sp.* | P | P | P | A |
| | *Frustilia sp.* | P | P | P | A |
| | *Diatoma sp.* | P | P | P | A |
| | *Cyclotella sp.* | P | P | A | A |
| | *Tabellaria sp.* | P | A | A | A |
| **Chlorophyceae** | *Botryoccoccus sp.* | P | P | P | P |
| | *Crucigenia sp.* | P | A | A | A |
| | *Cladophora sp.* | P | A | A | A |
| | *Coelastrum sp.* | P | A | A | P |
| | *Tetraspora sp.* | P | A | A | P |
| | *Spirogyra sp.* | P | A | P | A |
| | *Pediastrum sp.* | P | A | A | A |
| | *Scenedra sp.* | P | A | A | A |
| | *Scenedesmus sp.* | P | P | P | A |
| **Cyanophyceae** | *Phormidium sp.* | P | P | P | P |
| | *Coelosphaerium sp.* | A | A | P | A |
| **Desmidiaceae** | *Cosmarion sp.* | P | A | P | A |
| | *Gonatozygon sp* | P | A | A | A |
| | *Closterium sp.* | P | A | A | A |
| **Euglenophyceae** | *Phecus sp.* | A | P | A | P |
| | *Euglena sp.* | A | A | A | P |
| | *Closterium sp.* | P | A | A | A |
| **Euglenophyceae** | *Phecus sp.* | A | P | A | P |
| | *Euglena sp.* | A | A | A | P |

With a total of 18 taxa, Kesses had the highest number of phytoplankton with 20 taxa followed by Kerita and Mauna with 10 taxa each and lastly was Yenga with 6 taxa. Chlorophyceae and Cyanophyceae were found in all the dams while Bacillariophyceae was found in all dams except Yenga. The highest number of phytoplankton genera and individuals was 20 (Figure 4.11) and 1392 (Figure 4.12) respectively both in Kesses.

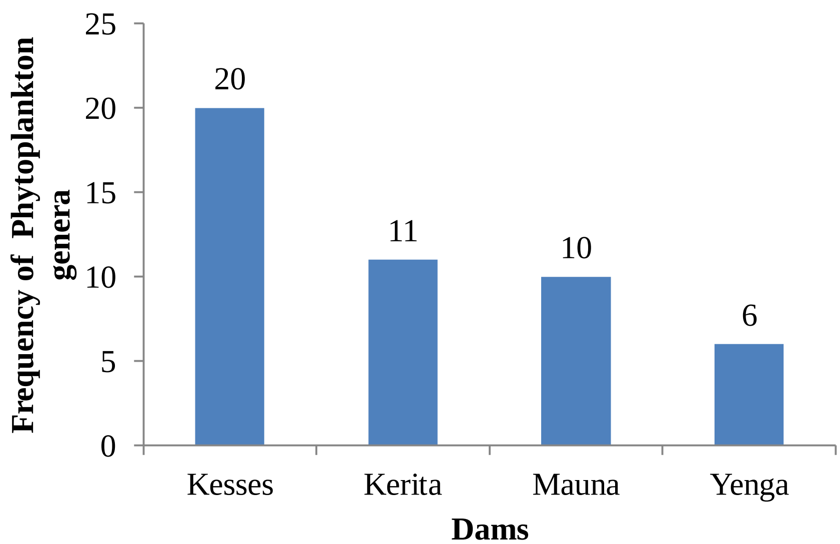



**Figure 4.11 Frequency of phytoplankton genera in Kesses, Kerita, Mauna and Yenga during this study.**

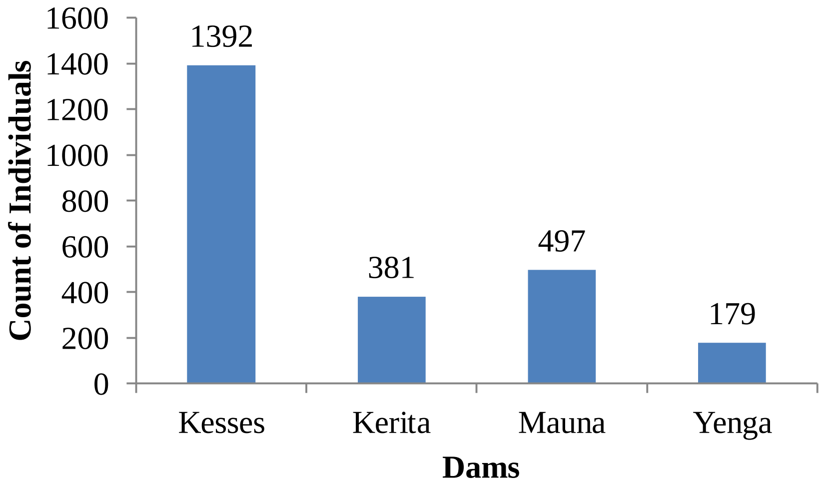



**Figure 4.12 Number of individual phytoplankton in Kesses, Kerita, Mauna and Yenga during this study.**

Abundance levels of phytoplankton in Kesses (Figure 4.13a), Kerita (Figure 4.13b), Mauna (Figure 4.13c) and Yenga (Figure 4.13d) indicated that Euglenophyceae and Desmidiaceae were in high abundance and dominated in Mauna and Yenga respectively. Euglenophyceae and Chlorophyceae had high relative abundance levels in Kesses and Kerita and were the dominant phytoplankton in these dams.

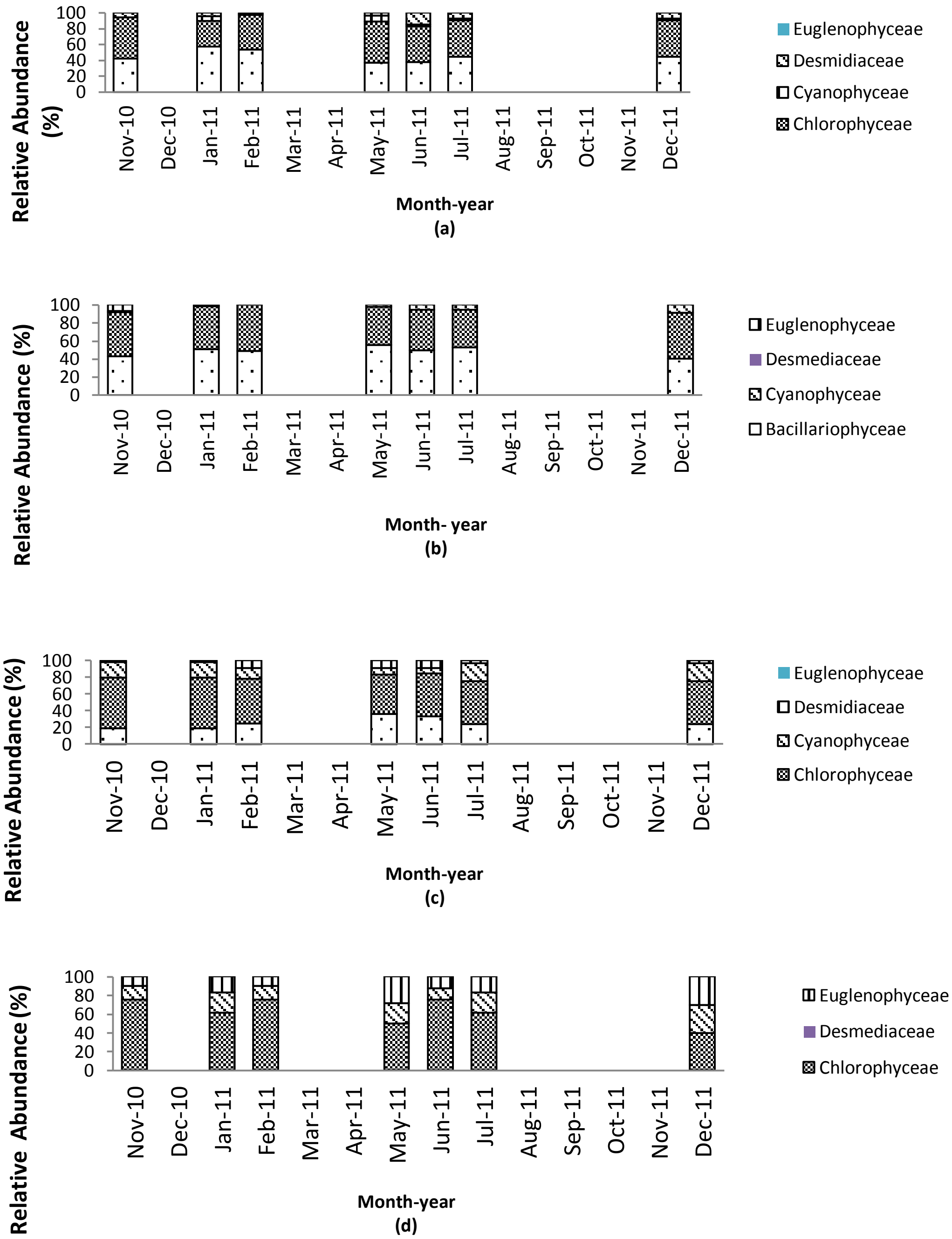



**Figure 4.13 Monthly relative Abundance (%) of phytoplankton in (a) Kesses (b) Kerita (c) Mauna and (d) Yenga dams during the period of November 2010 to December 2011**

Yenga had the highest dominance and lowest species diversity of phytoplankton, having 0.73 and 1.53 values of Simpson and Shannon diversity index (Table 4.6). There were significant differences between dams in abundance of phytoplankton. Chlorophyceae was higher in Kesses than Kerita (W= 1506; p= 0.018), Mauna (W= 1665; p= 0.0006) and Yenga (W = 351; p = 0.012). Cyanophyceae was found in higher abundance in Siaya than Uasin Gishu; with Mauna having higher abundance than Kesses (W= 1114; p= 0.0008 and Kerita (W= 340; p= 0.0054) and Yenga also higher than both Kesses (W= 1186; p= 0.0034) and Kerita (W= 384; p= 0.025) (Table 4.7).

**Table 4.6 Dominance (D), Evenness (e^H/S), Shannon (H) and Simpson (1-D) diversity indices for phytoplankton taxa in Kesses, Kerita, Mauna and Yenga dams during this study.**

| DAMS | INDICES | | | |
|---|---|---|---|---|
| | Dominance (D) | Evenness (e^H/S) | Shannon (H) | Simpson (1-D) |
| Kesses | 0.07 | 0.25 | 0.92 | 2.72 |
| Kerita | 0.19 | 0.11 | 0.80 | 1.97 |
| Mauna | 0.17 | 0.13 | 0.81 | 1.88 |
| Yenga | 0.27 | 0.26 | 0.73 | 1.53 |

**Table 4.8 Mann Whitney tests to compare relative abundance of phytoplankton between Kesses, Kerita, Mauna and Yenga dams during this study. ([1] and [2] indicate significantly higher abundance of phytoplankton in first and second dam respectively (p=0.05). Absence of phytoplankton in dam is indicated by - .**

| | Kesses and Kerita | | Kesses and Mauna | | Kesses and Yenga | | Kerita and Mauna | | Kerita and Yenga | | Mauna and Yenga | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | W | P | W | p | W | P | W | P | W | P | W | p |
| Bacillalliophyceae | 1452 | 0.115 | 1501 | 0.023[1] | - | - | 473 | 0.5973 | - | - | 2856 | 0.066 |
| Chlorophyceae | 1506 | 0.018[1] | 1665 | 0.0006[1] | 351 | 0.012[1] | 437 | 0.623 | 627 | 0.001[1] | 325 | 0.752 |
| Cyanophyceae | 154 | 0.217 | 1114 | 0.0008[2] | 1186 | 0.003[2] | 340 | 0.005[2] | 384 | 0.025[2] | 114 | 0.256 |
| Desmidiacea | - | - | 254 | 0.456 | - | - | - | - | - | - | - | - |
| Euglenophyceae | - | - | - | - | - | - | - | - | 355 | 0.0005[2] | - | - |

Cluster analysis on phytoplankton grouped both Kerita and Yenga together; indicating that they are similar in terms of composition and abundance of phytoplankton. Mauna was intermediate while Kesses was most distant from the other three dams; showing that it was most different (Figure 4.14).

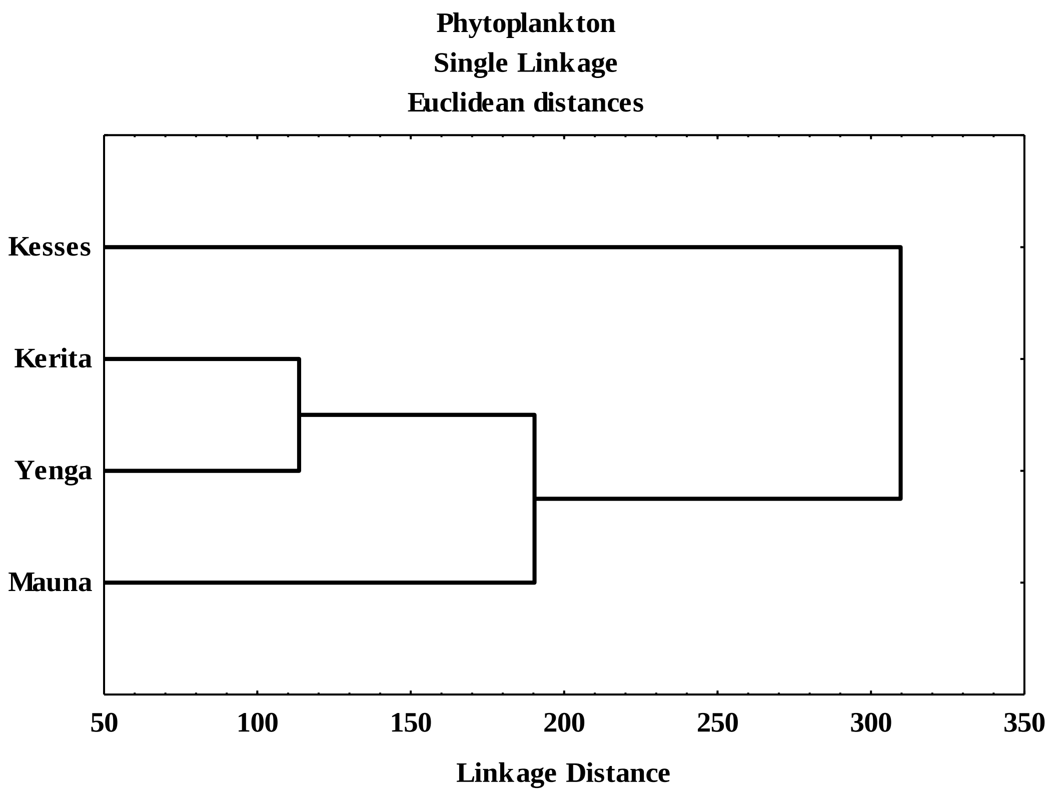



**Figure 4.14 Cluster analysis of phytoplankton abundance at Kesses, Kerita, Mauna and Yenga dams during the period between November 2010 and June 2011.**

Correspondence analysis gave strong loadings of Diatoms, *Phacus, Euglena* and *Cyclotella* in both Kerita and Yenga. Mauna had a strong loading of *Coelosphaerium, Cosmarium* and *Spirogyra.* Kesses had strong loadings of *Closterium, Tetraspora, Coalastrum* and *Synedra* (Figure 4.15).

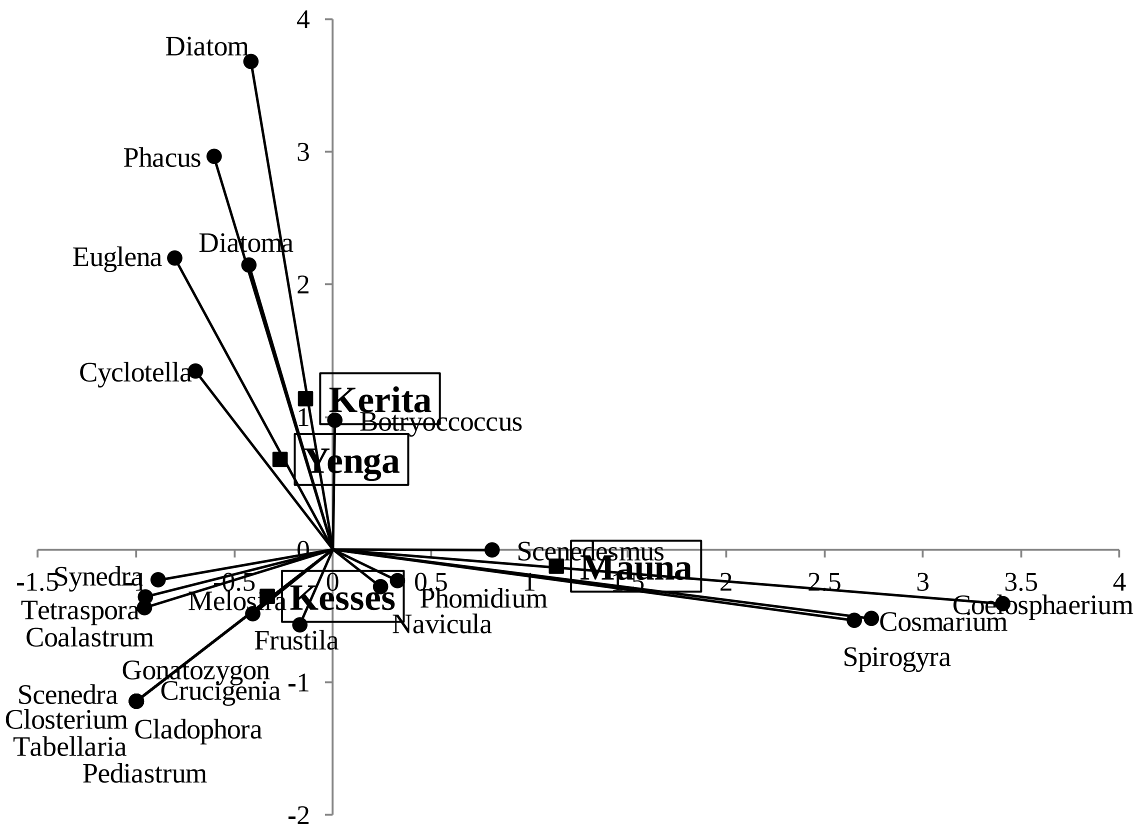



**Figure 4.15 Cannonical correspondence analysis (CCA) ordination plot for distribution of phytoplankton in Kesses, Kerita, Mauna and Yenga dams during this study.**

### 4.2.2 Primary Productivity of Phytoplankton

Kesses Dam recorded an estimated mean productivity of 0.33±0.049 $mgO_2$/l/hr (4,897gC/$m^2$/hr), Kerita Dam a mean of 0.26±0.025 $mgO_2$/l/hr (6,944gC/$m^2$/hr), Yenga 0.28±0.019 $mgO_2$/l/hr (6,365gC/$m^2$/hr) while Mauna had a mean value of 0.31±0.005 $mgO_2$/l/hr (5,787gC/$m^2$/hr) (Table 4.8). While there was no significant difference between the two regions (Uasin Gishu and Siaya) in primary productivity, there were significant differences between Kesses and Kerita Dams ($F=34.09$; $p=0.002$) in Uasin Gishu and between Yenga and Mauna Dams ($F=47.87$; $p=0.0031$) in Siaya.

**Table 4.8 Net primary productivity (NPP) for Kesses, Kerita, Mauna and Yenga during this study and other selected water bodies.**

| | NPP | Source |
|---|---|---|
| Kesses | 7.92 mg $O_2$ $m^{-2}$ $day^{-1}$ | This study |
| Kerita | 6.72 mg $O_2$ $m^{-2}$ $day^{-1}$ | This study |
| Mauna | 7.44 mg $O_2$ $m^{-2}$ $day^{-1}$ | This study |
| Yenga | 6.22 mg $O_2$ $m^{-2}$ $day^{-1}$ | This study |
| Lake Bogoria | 31.9±4.2 $O_2$ g $m^{-2}$ $day^{-1}$ | (Oduor and Schageri, 2007) |
| Lake Nakuru | 60.4± 2.6 g $O_2$ $m^{-2}$ $day^{-1}$ | (Oduor and Schageri, 2007) |
| Lake Elementaita | 28.6 ±4.3 g $O_2$ $m^{-2}$ $day^{-1}$ | (Oduor and Schageri, 2007) |
| Sagana Fish ponds | 0.1-11.9 g C $m^{-2}$ $day^{-1}$ | (Veverica *et al.*, 2001) |
| Wakuluba | 7 mg $O_2$ $m^{-2}$ $day^{-1}$ | (Kaggwa *et al.*, 2009) |
| Gaba | 4.2 mg $O_2$ $m^{-2}$ $day^{-1}$ | (Kaggwa *et al.*, 2009) |

## 4.3 Macroinvertebrate Composition, Abundance and Biomass

### 4.3.1 Composition and Abundance

A total of 12 macroinvertebrate orders were found in the four dams: Odonata, Diptera, Coleoptera, Ephemeroptera, Plecoptera, Trichoptera, Amphipoda, Hirudnea, Hemiptera, Pulmonata, Prosobranchiata and Lamelliobranchiata. Some macroinvertebrates are were found in low abundance such as *Aeshna, Gramarus, Haliplus, Hydrometra, Hyponeura, Physa, Promoresia* and *Tipula* in Kesses dam, *Chironomus, Pantala, Pisisdium, Psephenus, Simulium* and *Sphaerium* in Kerita dam, *Anodonta, Caenis, Coenagrion, Dytiscus, Ephemera* in Mauna and *Nobus, Notonecta* and *Platambus* in Yenga dam.

There were species that were relatively more abundant at some sites such as *Valvata* in Kesses dam with percentage relative abundance of 26.6%, Baetis (17.2%) and *Corixa* (16.5%) in Kerita, while other species were abundant in two dams like *Limnaea* in Kesses and Mauna with 42.2 % and 38.9% relative abundance, *Agrion* in Kerita and Mauna (23.6% and 14.3% respectively) and *Gammarus* in Mauna and Yenga ((11.3% and 53.4% respectively) (Figure 4.16). Kesses dam had 29 genera with 6187 individuals, the highest count in all the four dams. Kerita dam had 27 genera and 2674 individual counts sampled in it while Yenga dam also had 27 genera but with fewer individuals (2272) than Kerita. Mauna dam had the least number of genera (21) but with more individuals (2860) than Kerita and Yenga.

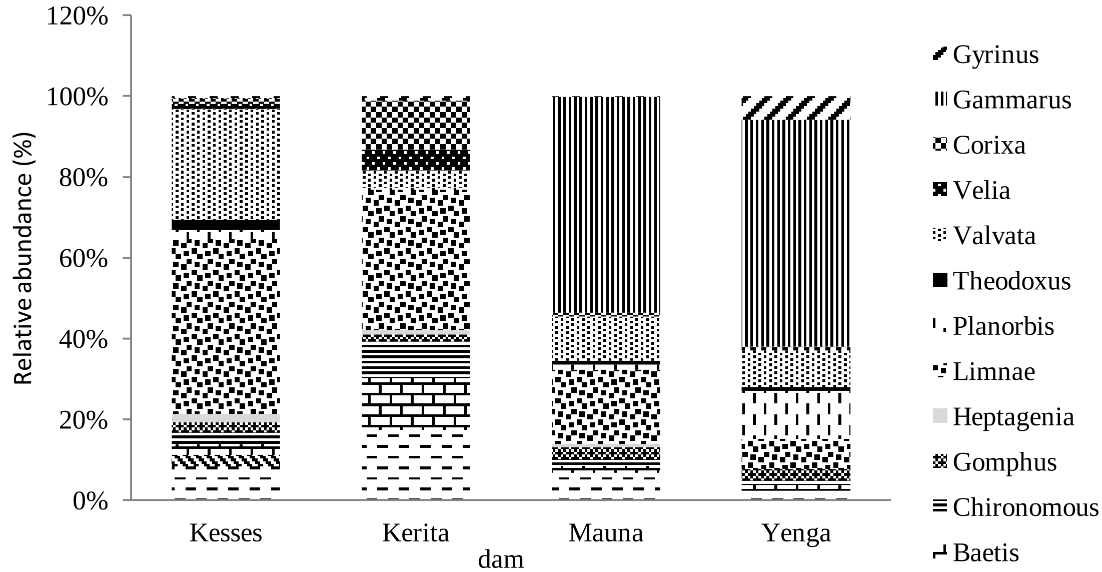



**Figure 4.16 Relative abundance of macroinvertebrates in Kesses and Kerita (Uasin Gishu) and Mauna and Yenga (Siaya) during this study.**

The macroinvertebrate genera *Agrion, Baetis, Chironomus* and *Corixa* had a higher median value and interquartile range at Kesses and Kerita dam than at Mauna and Yenga dam (Figure 4.17). *Gammarus* had the highest abundance range in Mauna while other macroinvertebrates were low in abundance (Figure 4.18).

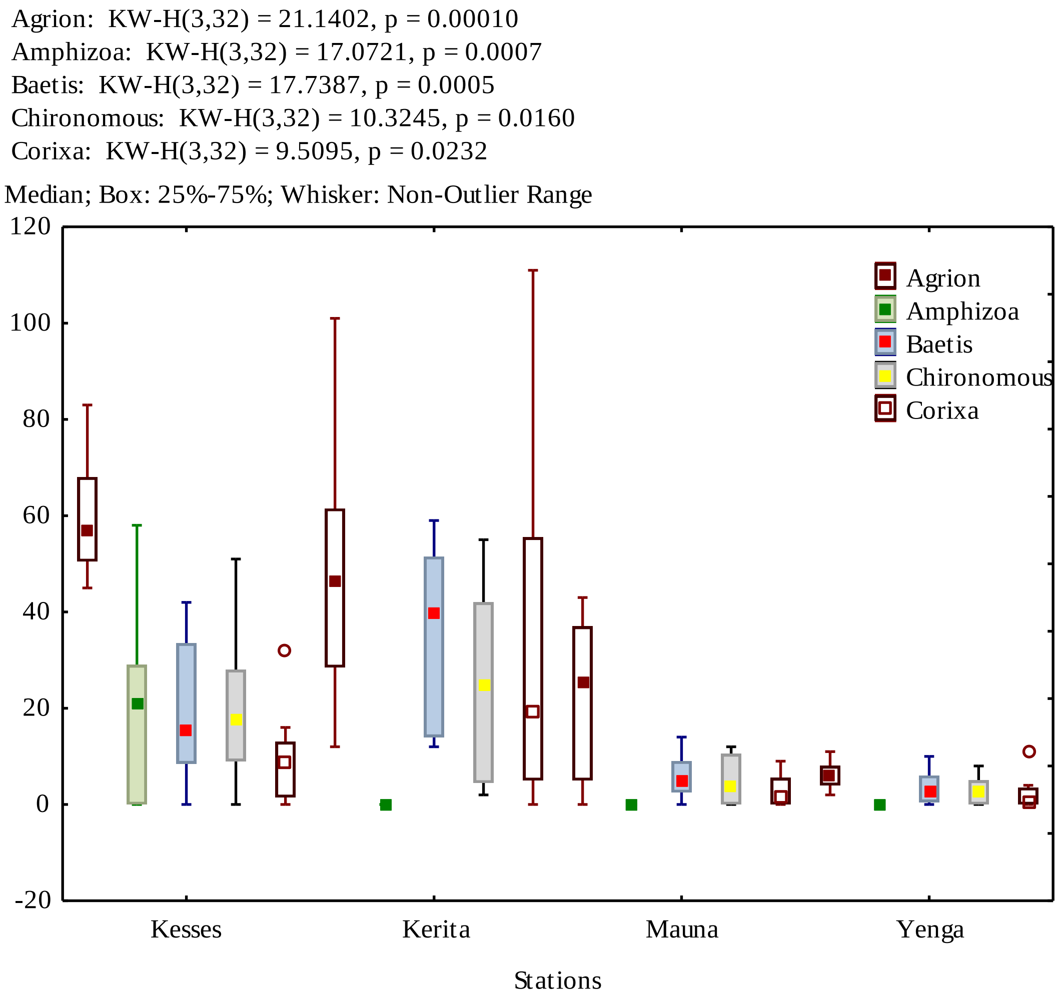



**Figure 4.17 Mean measurements and interquartile range for the macroinvertebrates Agrion, Amphizoa, Baetis, Chironomus, and Coriza in Kesses, Kerita, Mauna and Yenga during this study.**

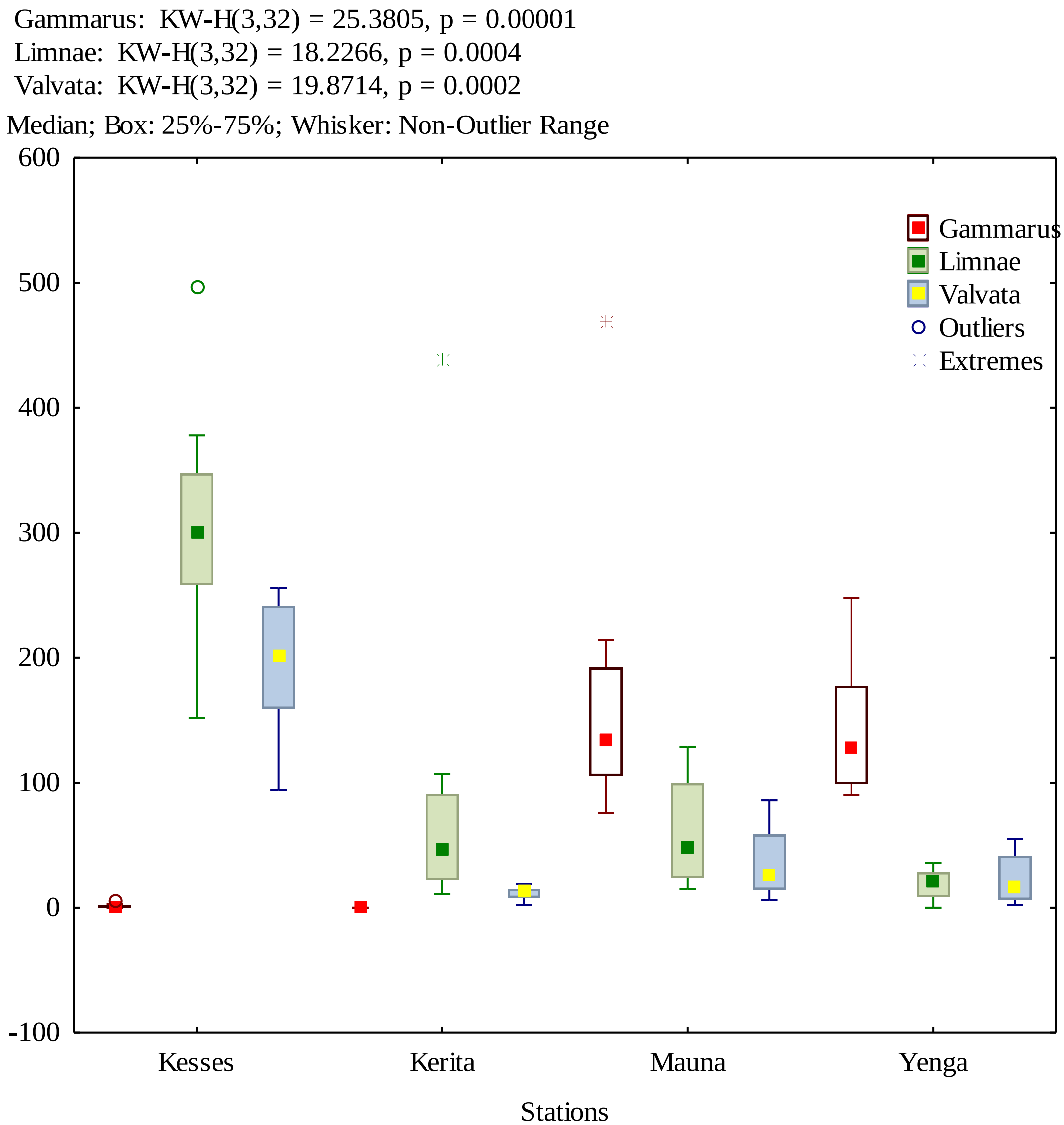



**Figure 4.18 Mean measurements and interquartile range for the macroinvertebrates *Gammarus, Limnae* and *Valvata i*n Kesses, Kerita, Mauna and Yenga during this study.**

Mauna and Yenga dams in Siaya had significant differences in abundance of Odonata; Mauna having higher values than Yenga (W=622; p=0.0016). Kerita dam had the highest abundance of Ephemeroptera; being significantly higher than Kesses (W= 993; p= 0.000), Mauna (W= 36; p= 0.000) and Yenga (W= 679; p= 0.000).Dams in Uasin Gishu were more abundant in Hemiptera; with Mauna having lower abundance than both Kesses (W= 1582; p= 0.000) and Kerita (W= 586; p= 0.0007). Pulmonata was more abundant in Kesses than Mauna (W= 1659; p=0.000) and Yenga (W= 1659; p= 0.000). Abundance of Pulmonata was higher in Mauna than Yenga (W= 591; p= 0.0145). Kesses dam had higher abundance of Prosobranchiata than Kerita (W= 1659; p=0.000), Mauna (W= 1587; p= 0.0004) and Yenga (W= 1713; p= 0.000). There were no differences in abundance of Coleoptera between all dams (Table 4.9).

Kruskal Wallis test for differences in abundance of macroinvertebrates showed that the following were all significantly different at the four dams as follows: *Gammarus* (H = 25.38; p = 0.000), *Limnaea* (H= 18.23; p = 0.0004), *Valvata* (H = 19.83; p = 0.0002), *Agrion* (H =21.14; p= 0.00010, *Amphizoa* (H = 17.07; p = 0.0007), *Baetis* (H = 17.73; p = 0.0005), *Chironomus* (H = 10.32; p = 0.016) and *Corixa*, (H = 9.509; p = 0.023).Mann Whitney tests indicated that abundance of Odonata was higher in Kerita than Kesses (W=1173; p=0.0129) and Yenga (W= 699; p=0.000). Kesses had a higher abundance of Odonata compared to Yenga (W= 17731; p= 0.000) (Table 4.9).

**Table 4.9 Mann Whitney tests to compare relative abundance of macroinvertebrate abundance between Kesses, Kerita, Mauna and Yenga dams. [1] indicates macroinvertebrate abundance higher in first dam and [2] indicates phytoplankton abundance higher in second dam. – indicates absence of phytoplankton in one of the dams**

| | Kesses and Kerita | | Kesses and Mauna | | Kesses and Yenga | | Kerita and Mauna | | Kerita and Yenga | | Mauna and Yenga | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | W | P | W | p | W | P | W | P | W | P | W | p |
| Odonata | 1173 | 0.013 | 296 | 0.201 | 1731 | 0.000[1] | 194 | 0.301 | 699 | 0.000[1] | 622 | 0.001[1] |
| Diptera | 311 | 0.263 | 174 | 0.161 | 1596 | 0.012 | 532 | 0.043[1] | 667 | 0.000 | 577 | 0.032 |
| Coleoptera | 532 | 0.625 | 781 | 0.521 | 489 | 0.691 | 511 | 0.366 | 410 | 0.602 | 533 | 0.614 |
| Ephemeroptera | 990 | 0.000[2] | 356 | 0.123 | 1557 | 0.0068[1] | 636 | 0.000[1] | 679 | 0.000[1] | 567 | 0.011[1] |
| Plecoptera | - | - | - | - | - | - | - | - | - | - | - | - |
| Trichoptera | - | - | - | - | - | - | - | - | - | - | - | - |
| Amphipoda | - | - | 903 | 0.000[2] | 903 | 0.000[2] | - | - | - | - | 627 | 0.001[2] |
| Hirudnea | 1011 | 0.000[2] | 381 | 0.476 | 1101 | 0.000[2] | 527 | 0.425 | 463 | 0.448 | 662 | 0.401 |
| Hemiptera | 231 | 0.211 | 1582 | 0.0004[1] | 314 | 0.189 | 586 | 0.0007[1] | 555 | 0.025[1] | 289 | 0.190 |
| Pulmonata | 179 | 0.246 | 1659 | 0.000[1] | 1658 | 0.000[1] | 360 | 0.071 | 298 | 0.081 | 591 | 0.014[1] |
| Prosobranchiata | 1659 | 0.000[1] | 1587 | 0.0004[1] | 1713 | 0.000[1] | 370 | 0.043[2] | 357 | 0.011[2] | 302 | 0.057 |
| Lamelliobranchiata | - | - | - | - | - | - | 367 | 0.201 | 248 | 0.193 | 580 | 0.020[2] |

Cluster analysis of the abundance of macroinvertebrates at the four sites with linked Euclidean distance show Yenga and Mauna dam to be closely similar in the species composition and abundance while Kerita dam is intermediate and Kesses being most different from the rest (Figure 4.19). Correspondence analysis of macroinvertebrates also showed a close relationship between Mauna and Yenga. Kerita and Kesses differed with respect to abundance and composition of macroinvertebrates. *Corixa*, *Velia*, *Chironomus*, *Baetis* and *Agrion* were most abundant in Kerita dam while *Amphizoa*, *Heptagenia*, *Limnae*, *Theodoxus* and *Valvata* were more abundant in Kesses dam (Figure 4.20).

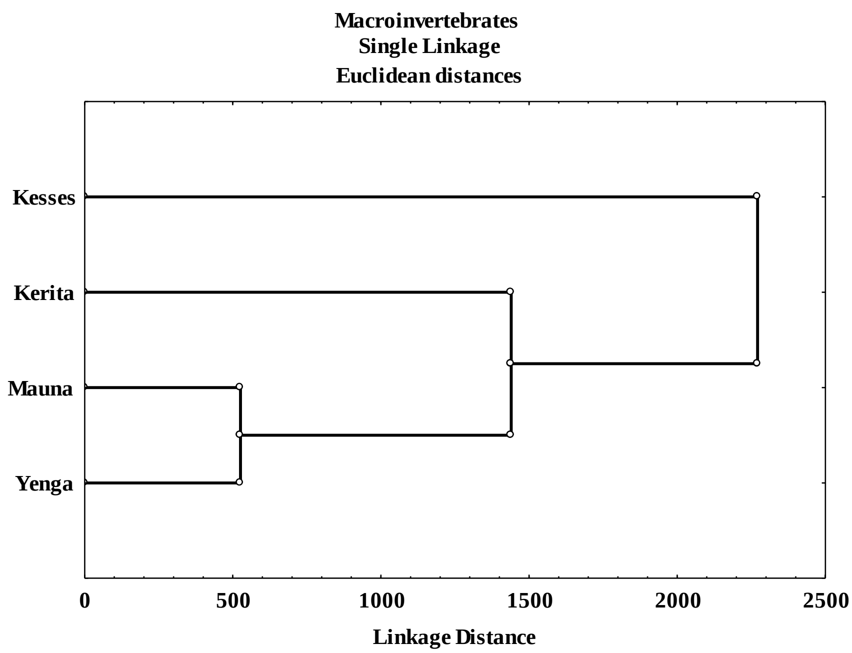



**Figure 4.19 Cluster analysis for macroinvertebrates in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and June 2011.**

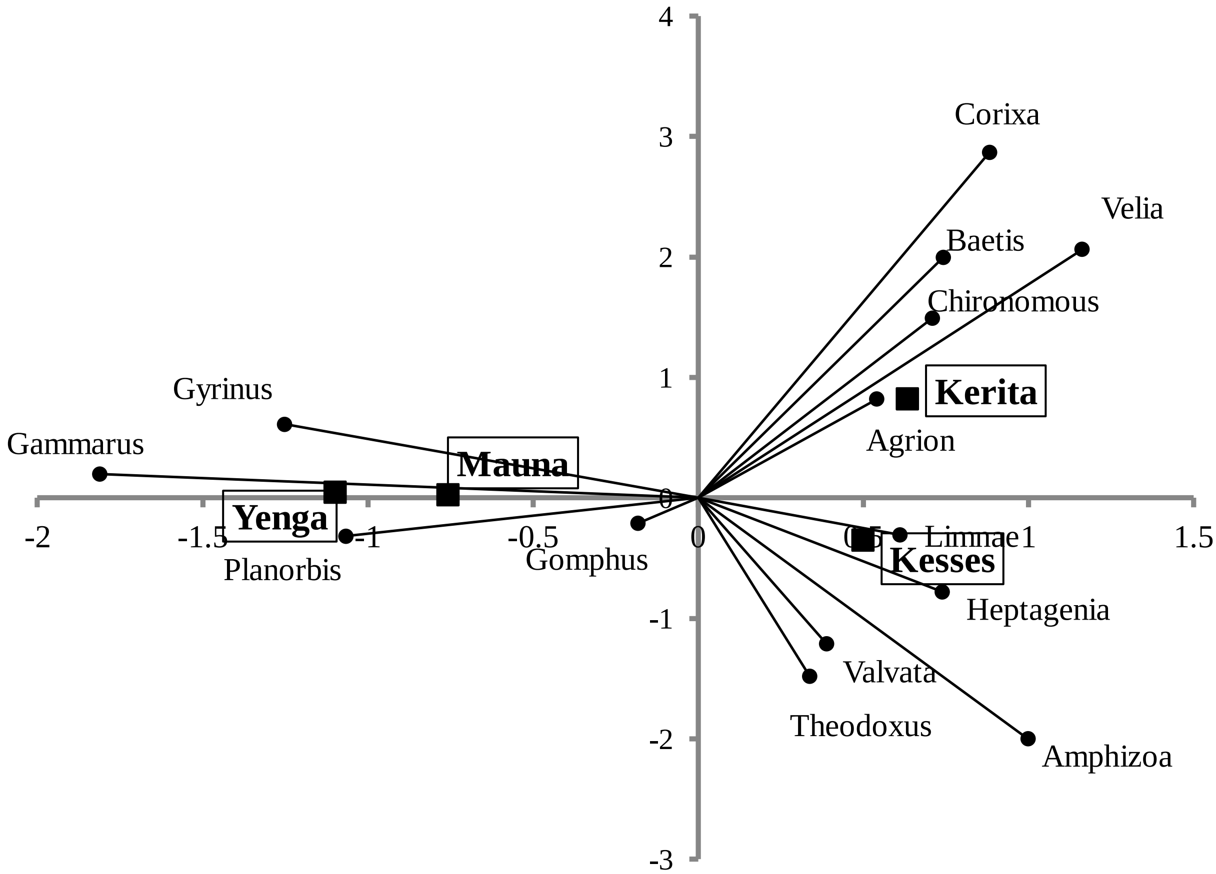



**Figure 4.20 Correspondence analysis for the most abundant macroinvertebrates in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and July 2011.**

Mauna dam had high dominance (D=0.29) than Yenga dam (D=0.28) and Kesses (D=0.27) (Table 4.10). Yenga had the lowest dominance of 0.14 while it had the highest evenness (0.43) indicating that, the macroinvetebrates were relatively even in distribution during the sampling period and were not dominated by some genera. Despite the high number of individual counts at Kesses dam, evenness and dominance index indicated that the dam had some species more abundant than others during the sampling period. Diversity of macroinvertebrates was higher in Uasin Gishu than Siaya. Shannon (H) and Simpson (1-D) indices of diversity were highest in Kesses (2.05 and 0.77 respectively) and lowest in Mauna (1.79 and 0.71 respectively) (Table 4.10).

**Table 4.10 Dominance (D), Evenness (e^H/S), Shannon (H) and Simpson (1-D) diversity indices for macroinvertebrate taxa in Kesses, Kerita, Mauna and Yenga dams during this study.**

| DAMS | INDICES | | | |
|---|---|---|---|---|
| | Dominance (D) | Evenness (e^H/S) | Shannon (H) | Simpson (1-D) |
| Kesses | 0.23 | 0.27 | 2.05 | 0.77 |
| Kerita | 0.14 | 0.43 | 2.46 | 0.86 |
| Mauna | 0.29 | 0.29 | 1.79 | 0.71 |
| Yenga | 0.28 | 0.27 | 1.98 | 0.72 |

### 4.3.2. Macroinvertebrate Biomass

Regression analysis of wet weight and dry weights of macroinvertebrates were subjected to a regression model. This resulted in a very strong relationship between the wet weight and the dry weight ($R^2$=0.937). The conversion factor (0.237) was used to convert wet weight into dry weight (biomass) which was used to estimate the secondary productivity of macroinvertebrates in SWBs. Mean macroinvertebrates biomass was highest in Mauna (0.47±0.178 g/$m^2$), followed by Kesses (0.33±0.195 g/$m^2$). Kerita registered a mean of 0.26±0.225 g/$m^2$ while Yenga had the lowest biomass mean 0.24±0.212 g/$m^2$ (Table 4.11). There was no significant difference in macroinvertebrates biomass between all the dams. The primary productivity and biomass estimation in this study were compared to that of other studies (Table 4.11).

**Table 4.11 Mean macroinvertebrate biomass (MMB) for Kesses, Kerita, Mauna and Yenga during this study and other water bodies.**

| | MMB | Source |
|---|---|---|
| Kesses | 0.26 ±0.225 g $m^{-2}$ | This study |
| Kerita | 0.33±0.195 g $m^{-2}$ | This study |
| Mauna | 0.47±0.178 g $m^{-2}$ | This study |
| Yenga | 0.24±0.212 g $m^{-2}$ | This study |
| Lake Michigan | 4.4 g/$m^{-2}$ | (Nalepa, 1989) |
| Lake Nakuru | 145.8 kg. $ha^{-1}yr^{-1}$ | (Oduor and Schageri, 2007) |

## 4.4 Fish Parasites Prevalence and Mean Intensity

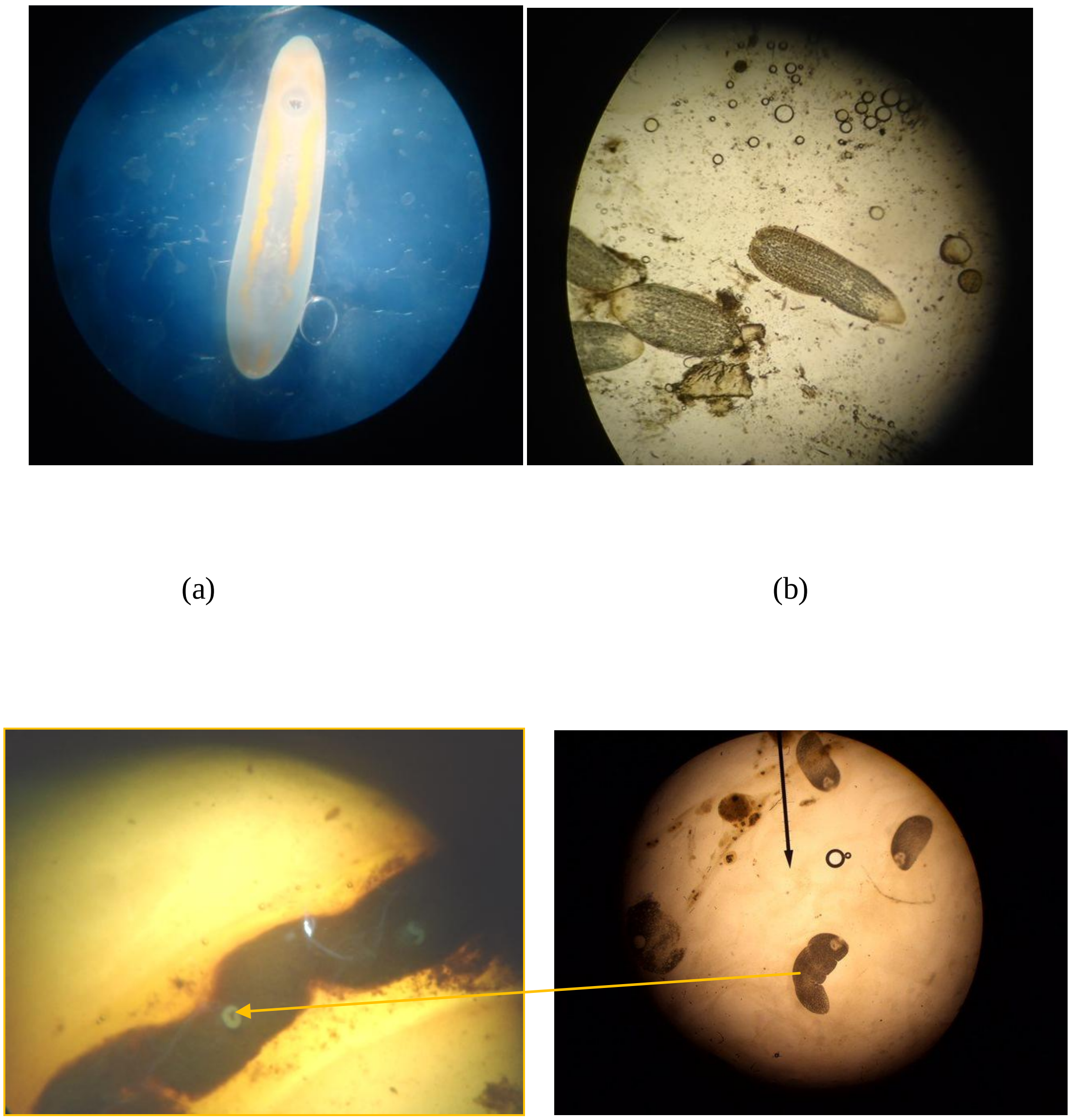
### 4.4.1 Abundance

A total of 1611 tilapia (*Oreochromis niloticus)* fish were sampled for fish parasites in the period between November 2010 and March 2012. The total number of fish infected was 1422; 906 males and 7004 females. There were no external parasites found on fish. A total of 29,574 internal helminth (endohelminth) parasites were found. The parasitic groups represented were: cestodes, digenea, nematodes, acanthocephalans and monogenea; contributing to 61.5%, 30%, 8.2%, 0.2% and 0.006 % of the total number of parasites found respectively. While there were no central species, the importance value based on prevalence indicated that 50% of the parasites (*Sanguinicolla*, *Acanthocestis, Cichlidogyrus* and *Bolbophorus)* were satellite and the rest (*Clinostomum*, *Contracaecum*, *Tylodelphys and Amirthalingamia)* were secondary parasite species (Table 13). Black spots associated with metacercariae of *Bolbophorus* sp. were found encysted on the skin and occasionally in the muscle. *Bolbophorus sp.*, S*anguinicolla sp.*, *Cichlidogyrus sp.* and *Acanthocestis* sp. had prevalence levels below 10% (Table 4.12) and were therefore excluded from further statistical analysis (Bush *et al*., 1990).

**Table 4.12 Summary of parasite data in 1611 *Oreochromis niloticus* from Kesses and Kerita dams in Uasin Gishu and Mauna and Yenga dams in Siaya counties during the period between November 2010 and March 2012. DS Developmental stage (M-Metacercaria; L- larvae; A– adult); IF - Number of infected fish; NP -number of parasites; MI - mean intensity; MA - mean abundance of parasites; AA - amplitude of abundance variation; P (%) – prevalence; I – species importance based on prevalence (P > 60% = central (Ce); P < 33% = satellite (Sa); P between 33% and 66% = secondary (Se).**

| Parasite | DS | IF | NP | MI | MA | AA | P(%) | I |
|---|---|---|---|---|---|---|---|---|
| *Clinostomum* | M | 755 | 2088 | 2.7 | 1.26 | 0-17 | 46.87 | Se |
| *Contracaecum* | L | 579 | 2400 | 4 | 1.4 | 0-22 | 35.94 | Se |
| *Tylodelphyes* | M | 1134 | 6684 | 9.5 | 3.25 | 0-58 | 34.45 | Se |
| *Amirthalingamia* | L | 802 | 18192 | 19.1 | 9.51 | 0-744 | 49.78 | Se |
| *Sanguinicolla* | L | 8 | 19 | 2.1 | 0.01 | 0-7 | 0.49 | Sa |
| *Acanthocestis* | A | 42 | 84 | 2 | 0.05 | 0-6 | 2.61 | Sa |
| *Cichlidogyrus* | A | 1 | 2 | 1 | 0.001 | 0-2 | 0.06 | Sa |
| *Bolbophorus* | M | 13 | 102 | 7.3 | 0.06 | 0-24 | 0.81 | Sa |

Un-encysted metacercariae of *Diplostomum* and *Tylodelphys* spp. were found in the lens and vitreous humor of the eye respectively. Larval cestodes of genus *Amirthalingamia* were occasionally found in association with Acanthocephala of the type *Acanthocestis* spp. in the gut (Plate 4.1).

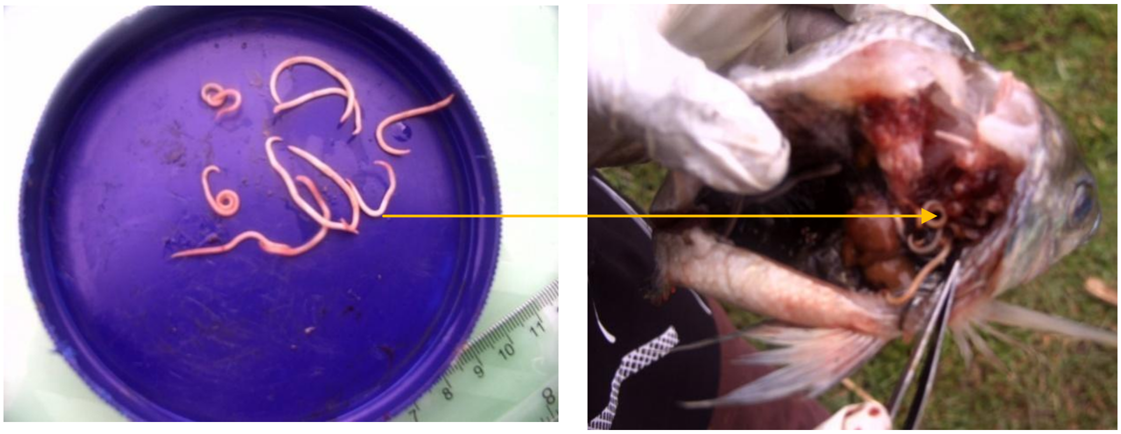


(a) (b)

(c) (d)

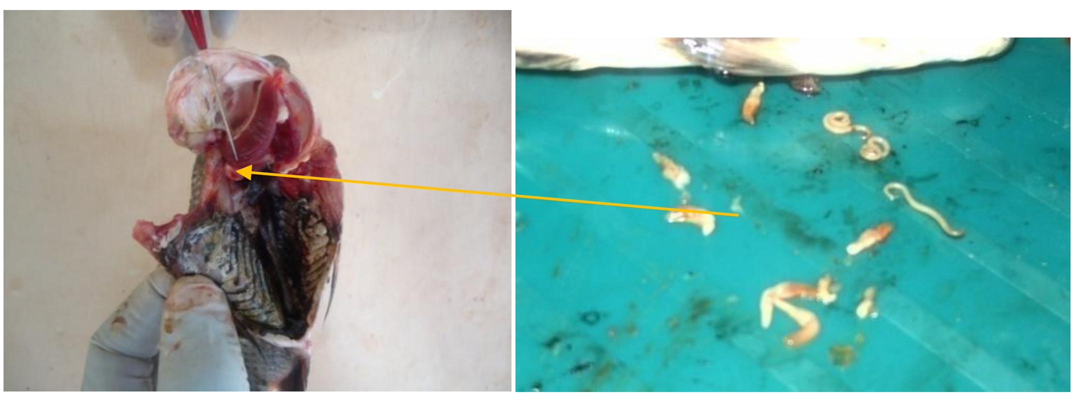
**Plate 4.1 Parasites of tilapia (*Oreochromis niloticus*) during the study period between November 2010 and March 2012. (a) *Diplostomum* sp. from eye lens (b) *Tylodelphys* sp. from eye vitreous from Kesses dam. (c) and (d) *Amirthalingamia* from the small intestine of fish from Yenga dam. (Source: Author, 2015)**

Strigeoid metacercariae stage digeneans were found in various organs of fish. *Sanguinicolla* were usually found together with *Contracaecum* sp. larvae in the pericardial cavity (Plate 4.2a and 4.2b). *Clinostomum phalacrocorasis* metacercaria were found encysted behind the gill arch while (Plate 4.3a and 4.3b). Heavy infection with *Amirthalingamia* caused distension of the abdominal cavity while *Diplostomum* and *Tylodelphys* parasites in tilapia were associated with cloudy eyes leading to blindness (Plate 4.4).



(a) **(b)**

**Plate 4.2 (a) and (b) *Contracaecum* parasites from buccal cavity of tilapia (*Oreochromis niloticus*) from Kesses dam during this study. (Source: Author, 2015)**



(a) (b)

**Plate 4.3 (a) and (b) Metacercarial stage *Clinostomum* found behind the gills of tilapia (*Oreochromis niloticus*) from Kesses dam during this study. (Source: Author, 2015).**

(a) (b)

**Plate 4.4 Some effects of parasitic infection on tilapia (*Oreochromis niloticus*) observed during this study. (a) Distension of abdominal cavity due to heavy *Amirthalingamia* infection (b) Cloudy eyes leading to blindness due to infection by *Diplostomum sp*. and *Tylodelphys sp.* and fin erosion due to fungal infection in Yenga dam, Siaya county. (Source: Author, 2015).**

There were notable differences in abundance of parasites in the dams. Abundance of *Clinostomum* and *Contracaecum* were highest in Kesses (Figure 4.21a and 4.21b). *Amirthalingamia* had its highest abundance in Yenga (Figure 4.21c). *Tylodelphys* and *Diplostomum* were highest in Kerita (Figure 4.21d and 4.21e).

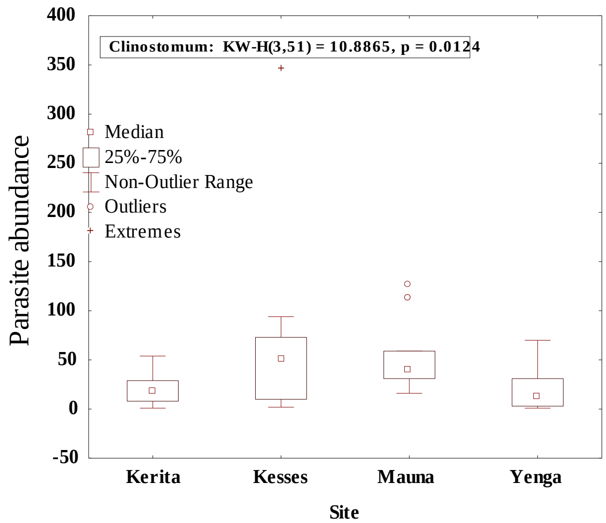



**Figure 4.21a. Median measurement and interquartile range for *Clinostomum* parasites of tilapia (*Oreochromis niloticus*) in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and March 2012.**

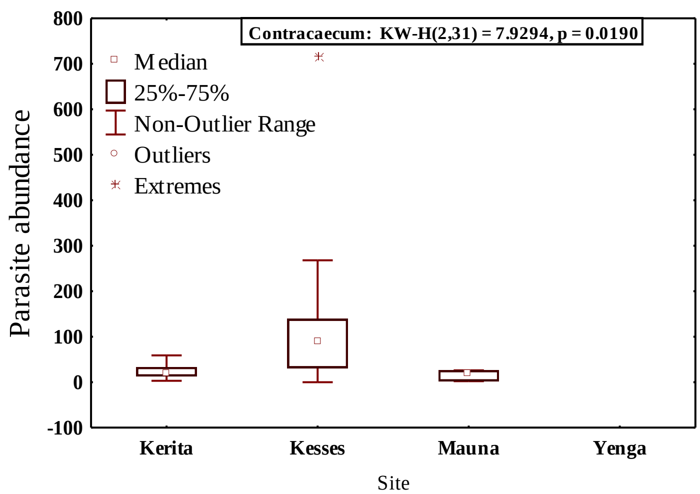



**Figure 4.21b Median measurement and interquartile range for *Contracaecum* parasites of tilapia (*Oreochromis niloticus*) in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and March 2012.**

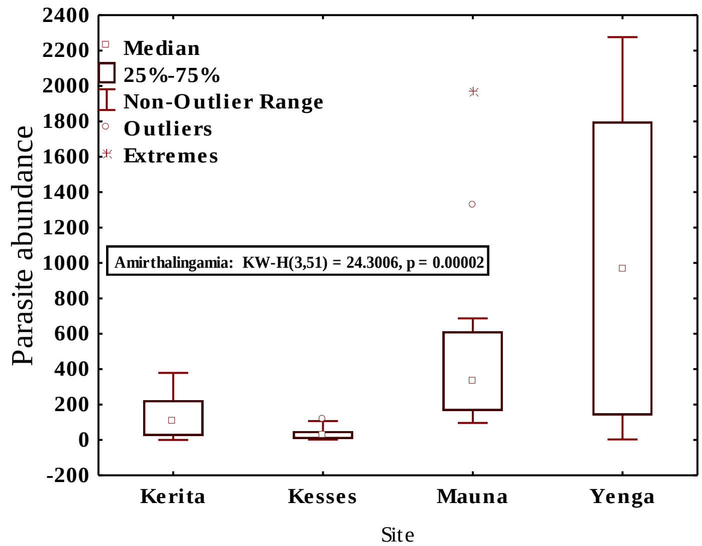



**Figure 4.21c. Median measurement and interquartile range for *Amirthalingamia* parasites of tilapia (*Oreochromis niloticus*) in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and March 2012.**

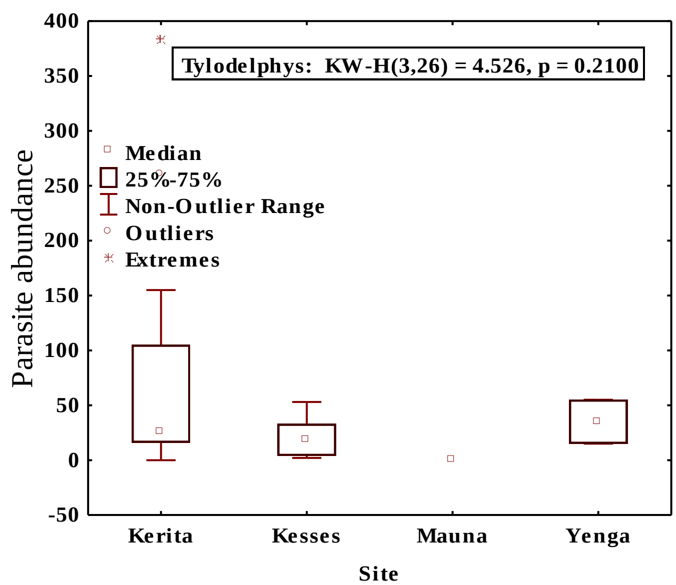



**Figure 4.21d. Median measurement and interquartile range for *Tylodelphys* parasites of tilapia (*Oreochromis niloticus*) in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and March 2012**

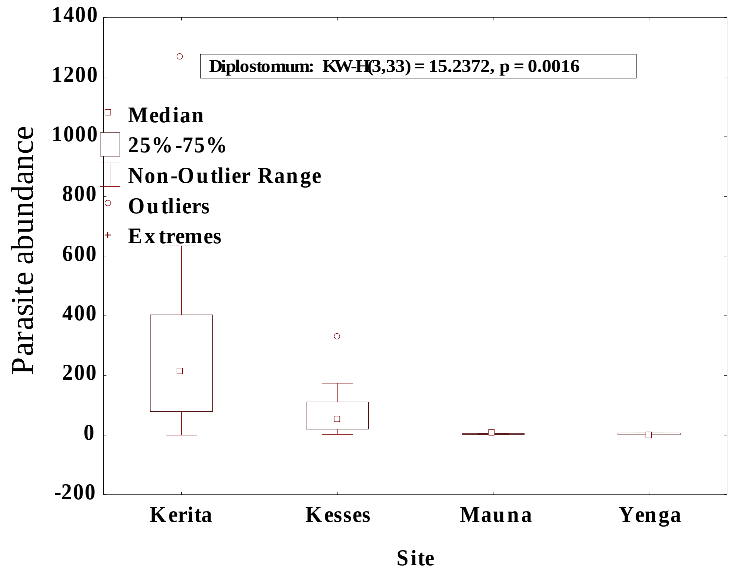



**Figure 4.21e. Median measurement and interquartile range for *Diplostomum* parasites of tilapia (*Oreochromis niloticus*) in Kesses, Kerita, Mauna and Yenga during the period between November 2010 and March 2012.**

### 4.4.2 Mean Intensity and Abundance

The parasite with the highest mean intensity in Kesses and Kerita was *Tylodelphys* at 5.4 and 13.0 parasites per fish respectively (Table 4.13a). *Amirthalingamia* had the highest mean intensity in Mauna and Yenga at 12.3 and 74.3 parasites per fish respectively (Table 4.13b). Although there was not much difference in prevalence, mean intensity for *Amirthalingamia* was higher in Mauna (12.3 parasites per fish) and Yenga (74.3 parasites per fish than in Kesses (4.4 parasites per fish) and Kerita 37.4 parasites per fish). There was no C*ontracaecum* in Yenga (Table 4.13b). In Uasin Gishu, *Tylodelphys* had the highest mean abundance at 13.0 parasites per fish in Kerita (Table 4.13a) while in Siaya, *Amirthalingamia* had the highest mean abundance at 50.4 parasites per fish in Yenga (Table 4.13b). The highest mean parasite abundance in both Kesses and Kerita was in

*Tylodelphys* while in Siaya it was *Amirthalingamia*. One fish was found with 744 larval stage *Amirthalingamia* cestodes in Yenga dam in Siaya County, which was the highest number of parasites in a single fish recorded during this study.

Parasitic intensity was found to differ significantly between dams for all parasites except *Clinostomum.* Differences in mean intensity of parasites were found within and between regions. In Uasin Gishu, Kesses had a higher mean intensity for *Contracaecum* than Kerita (U= 1722; p=0.000). *Tylodelphys* was higher in Kerita than Kesses (U = 1101; p = 0.0004). Similarly in Siaya, Yenga had a higher intensity of *Amirthalingamia* than Mauna (U = 384; p =0.025). Siaya dams were higher in *Amirthalingamia* than Uasin

Gishu with both Mauna and Yenga having significantly higher mean intensities for *Amirthalingamia* than Kesses (U=903; p=0.000 and U = 303; p = 0.0002 respectively) and Kerita (U = 1137; p =0.0003 and U = 339; p= 0.001 respectively). Yenga had higher mean intensity of *Amirthalingamia* than Kesses and Kerita. There was no difference in mean intensity of *Diplostomum* within the two regions

**Table 4.13a Prevalence (%), Mean Intensity (MI) and Mean Abundance (MA) of parasites in Kesses and Kerita in Uasin Gishu.**

| *Dam* | *Total number of fish* | *Parasite* | *Prevalence (%)* | *Mean Intensity* | *Mean Abundance* |
|---|---|---|---|---|---|
| *Kesses* | *485* | *Clinostomum* | *56.9* | *3.1* | *2.4* |
| | | *Contracaecum* | *84.3* | *5.2* | *4.1* |
| | | *Tylodelphys* | 43.3 | 5.4 | 2.2 |
| | | *Diplostomum* | 31.0 | 3.2 | 2.4 |
| | | *Amirthalingamia* | 52.8 | 4.4 | 1.3 |
| Kerita | 355 | *Clinostomum* | 38.9 | 2.0 | 1.1 |
| | | *Contracaecum* | 45.4 | 1.7 | 1.4 |
| | | *Tylodelphys* | 66.3 | 13.0 | 12.3 |
| | | *Diplostomum* | 34.3 | 10.4 | 9.0 |
| | | *Amirthalingamia* | 58.0 | 7.4 | 5.4 |

**Table 4.13b Prevalence (%), Mean Intensity (MI) and Mean Abundance (MA) of parasites Mauna and Yenga in Siaya.**

| *Dam* | *Total number of fish* | *Parasite* | *Prevalence (%)* | *Mean Intensity* | *Mean Abundance* |
|---|---|---|---|---|---|
| Mauna | 585 | *Clinostomum* | 47.5 | 2.3 | 1.1 |
| | | *Contracaecum* | 1.5 | 5.1 | 0.1 |
| | | *Tylodelphys* | 1.0 | 1.7 | 0.0 |
| | | *Diplostomum* | 5.0 | 1.6 | 1.7 |
| | | *Amirthalingamia* | 53.0 | 12.3 | 6.5 |
| Yenga | 186 | *Clinostomum* | 33.3 | 2.5 | 0.8 |
| | | *Contracaecum* | 0.0 | 0.0 | 0.0 |
| | | *Tylodelphys* | 16.7 | 3.1 | 0.5 |
| | | *Diplostomum* | 7.1 | 1.3 | 0.5 |
| | | *Amirthalingamia* | 67.7 | 74.3 | 50.4 |

### 4.4.3 Parasite Prevalence

The highest parasitic prevalence in Kesses, Kerita, Mauna and Yenga was *Contracaecum* (84.3%), *Tylodelphys* (66.3%), *Amirthalingamia* (53.0%) and 67.7% respectively. (Table 4.13a and 4.13b). Prevalence for *Diplostomum* and *Tylodelphys* in Mauna was 5.0 % and 1.0% respectively and in Yenga 7.1% and 16.7% respectively. In Kesses prevalence for *Diplostomum* and *Tylodelphys* was 31 % and 43.3 % respectively and in Kerita 34.3 % and 66.3 % respectively. Similarly, prevalence levels for *Amirthalingamia* were relatively in Yenga (67.7%) compared to Kesses (52%) and Kerita (58%).

Male fish appeared to have relatively higher prevalence levels for *Clinostomum* and *Contracaecum* while females had high prevalence levels for *Diplostomum* and *Amirthalingamia* in Kesses (Figure 4.22a). In Kerita, prevalence was high for *Diplostomum* and *Amirthalingamia* in females (Figure 4.22b). Prevalence was high in males for *Clinostomum* and in females for *Amirthalingamia* in Mauna (Figure 4.22c) with a similar trend for Yenga (Figure 4.22d).

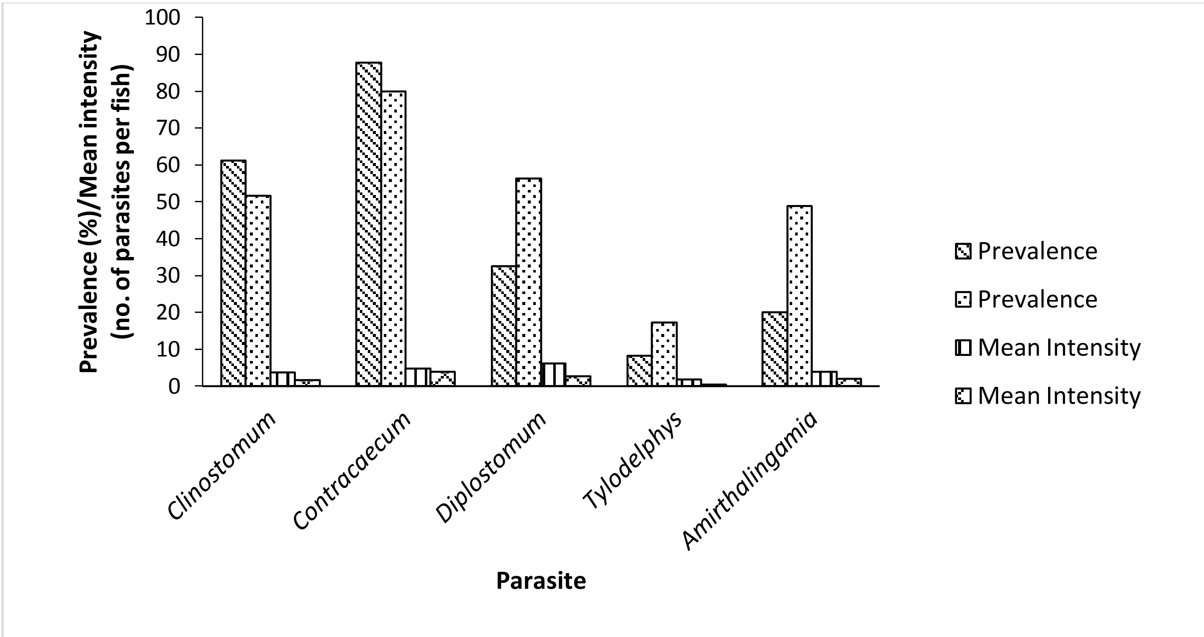



**Figure 4.22a. Prevalence (%) and Mean Intensity of *Clinostomum, Contracaecum, Diplostomum* and *Amirthalingamia* parasites in Kesses dam, Uasin Gishu County**

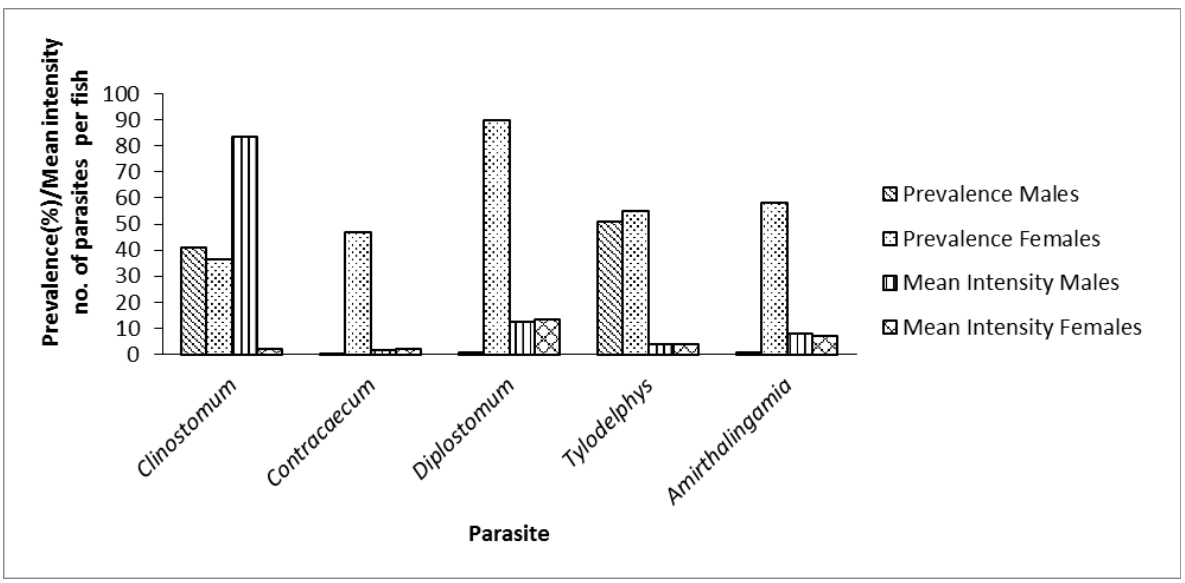



**Figure 4.22b. Prevalence (%) and Mean Intensity of *Clinostomum, Contracaecum, Diplostomum* and *Amirthalingamia* parasites in Kerita dam, Uasin Gishu County**

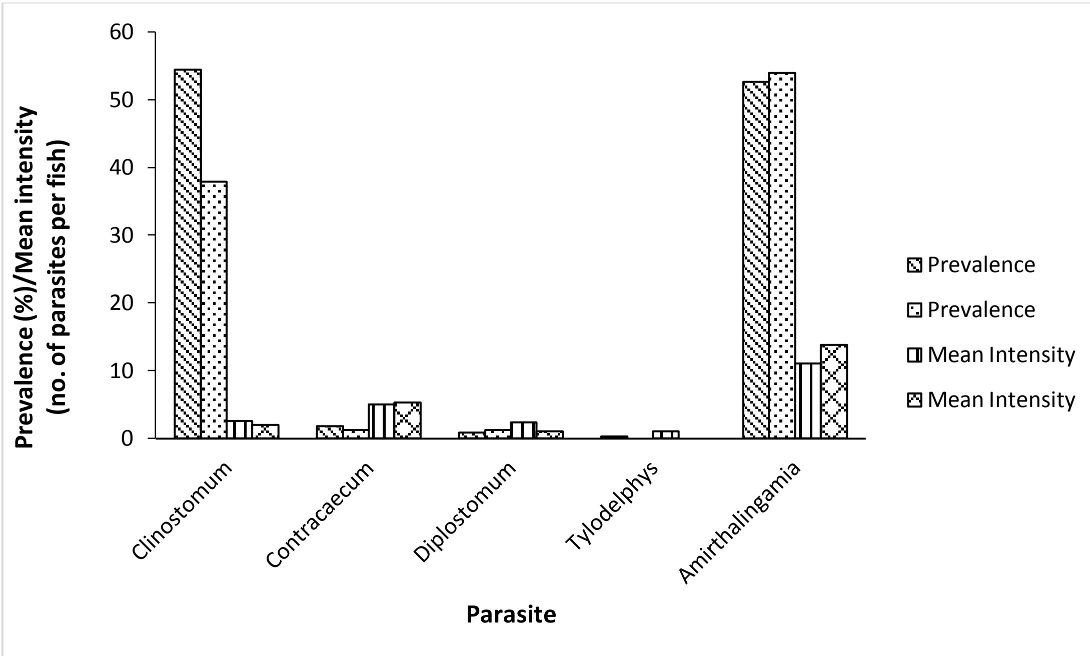



**Figure 4.22c**. Prevalence **(%) and Mean Intensity of *Clinostomum, Contracaecum, Diplostomum* and *Amirthalingamia* parasites in Mauna dam, Siaya County**

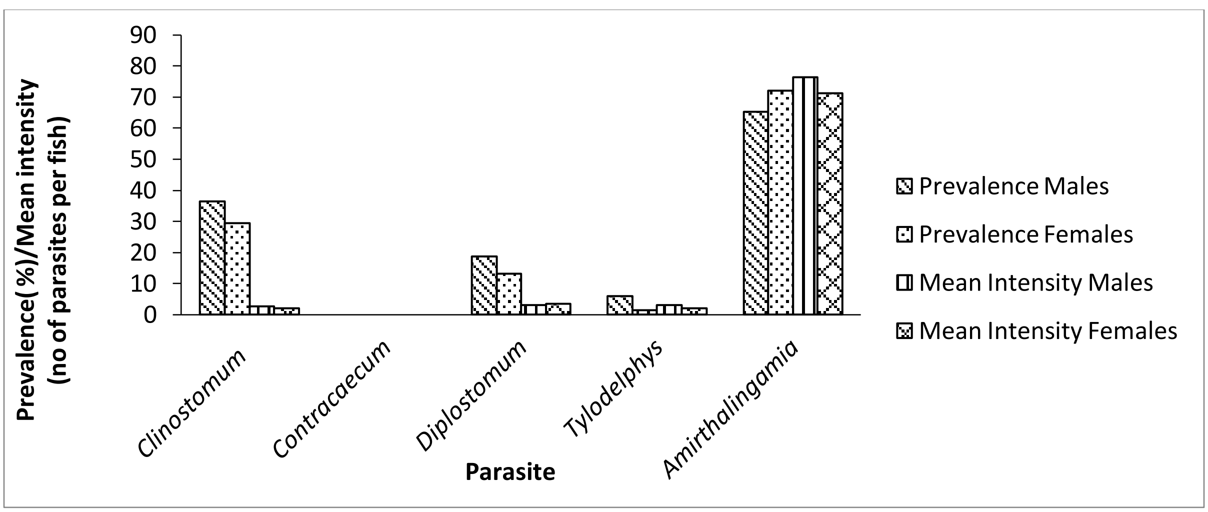



**Figure 4.22d**. **Prevalence (%) and Mean Intensity of *Clinostomum, Contracaecum, Diplostomum* and *Amirthalingamia* parasites in Yenga dam, Siaya County**

Correspondence analysis revealed that there were strong loadings of *Tylodelphys* and *Diplostomum* at Kerita, *Contracaecum* and *Clinostomum* at Kesses dam and *Amirthalingamia* at both Yenga and Mauna in Siaya (Figure 4.23). Kerita and Kesses dam were grouped closely in terms of parasites composition and abundance by cluster analysis, with Yenga dam being the most distant from the other dams (Figure 4.24). Yenga and Mauna dam also must have differed in the abundance of *Amirthalingamia* parasite since cluster analysis grouped them differently. Mauna dam was intermediate between Yenga and both Kesses and Kerita, which were grouped together.

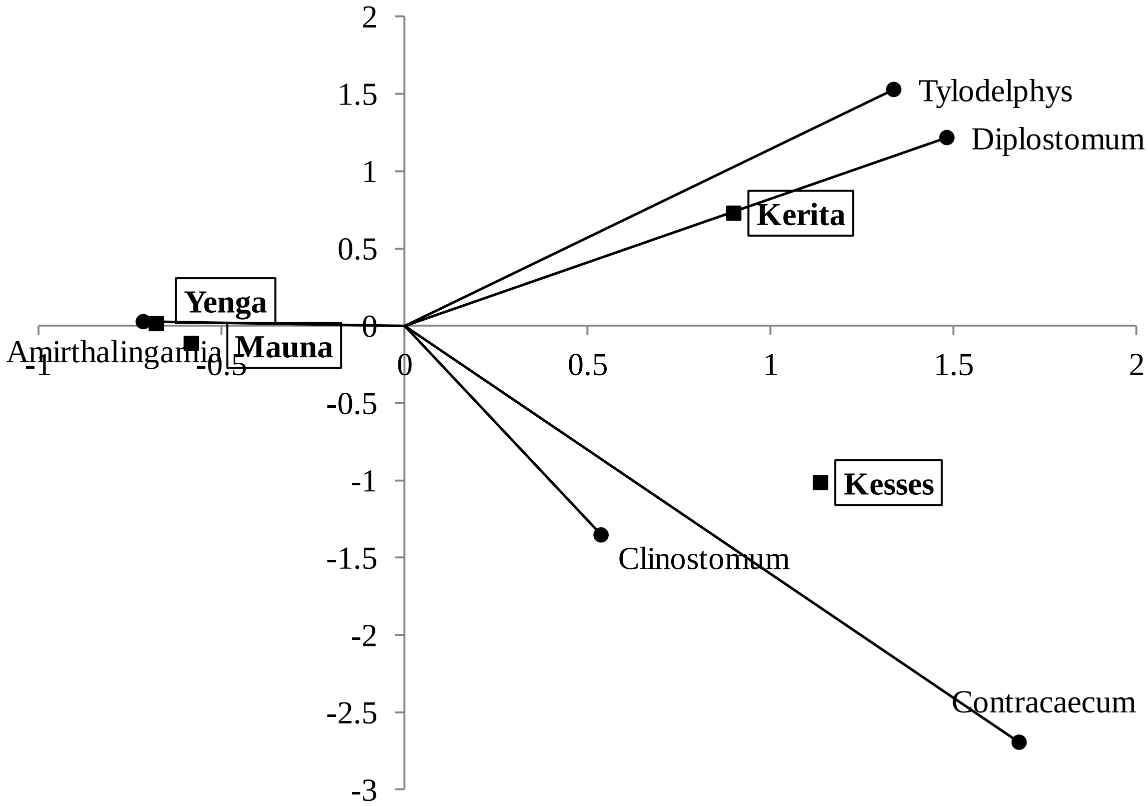



**Figure 4.23 Correspondence analysis ordination plot of *Tylodelphys, Diplostomum, Contracaecum, Clinostomum* and *Amirthalingamia* parasites in Kesses, Kerita, Mauna and Yenga dams during this study.**

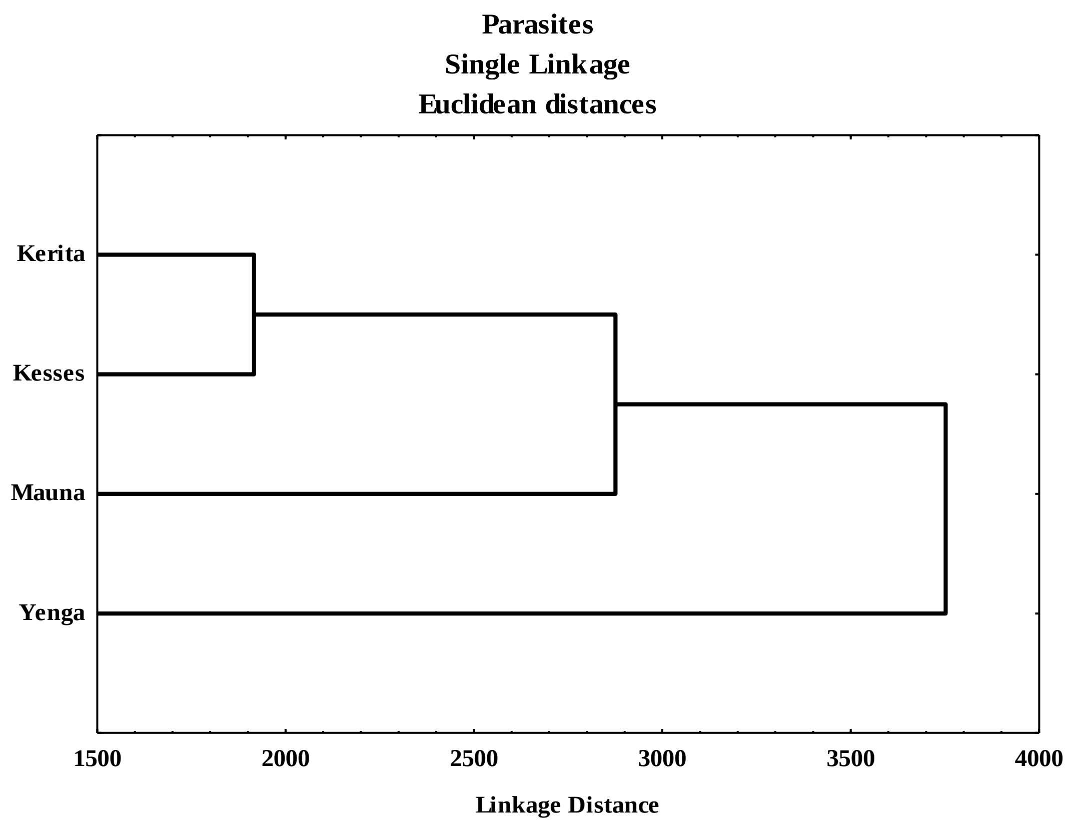



**Figure 4.24 Cluster analysis of *Tylodelphys, Diplostomum, Contracaecum, Clinostomum* and *Amirthalingamia* parasites in Kesses, Kerita, Mauna and Yenga dams during this study.**

Kesses dam had the lowest dominance and highest diversity (Shannon's) of parasites at 0.22 and 1.41 respectively. The highest dominance was found in Mauna (0.25) while the lowest diversity was at Yenga (0.08) respectively (Table 4.14).

**Table 4.14 Dominance (D), Evenness (e^H/S), Shannon (H) and Simpson (1-D) diversity indices for parasites in Kesses, Kerita, Mauna and Yenga dams during this study.**

| DAMS | INDICES | | | |
|---|---|---|---|---|
| | Dominance (D) | Evenness (e^H/S) | Shannon (H) | Simpson (1-D) |
| Kesses | 0.22 | 1.13 | 1.41 | 0.75 |
| Kerita | 0.24 | 0.86 | 0.59 | 1.18 |
| Mauna | 0.25 | 0.84 | 0.24 | 0.32 |
| Yenga | 0.23 | 0.89 | 0.08 | 0.09 |

## 4.5 Effect of Water Quality on Biota

### 4.5.1 Effect of Water Quality on Phytoplankton

Spearman rank correlation indicated that there were significant relationships between water quality components and phytoplankton. Phytoplankton abundance in all dams was influenced by physico-chemical parameters and nutrients. In Kesses, Bacillariophyceae, Cyanophyceae and Desmidiaceae were affected while Chlorophyceae was not significantly affected by water quality. Bacillariophyceae was affected significantly by TP (r= -0.29; p=0.000) and pH (r=0.372; p=0.013). DO had a significant negative effect on Desmidiaceae (r= -0.26; p=0.025) and Cyanophyceae (r= -0.18; p= 0.027) (Table 16). In Kerita dam the abundance of Bacillariophyceae had a significant relationship with TN (r=0.533, p=0.0042) and BOD (r=-0.11, p=0.000). Among the parameters that had significant influence on phytoplankton were Temperature and TN; both having positive relationships with Euglenophyceae (r=0.11, p=0.029 and r=0.28, p=0.001 respectively). Chlorophyceae was significantly influenced by pH (r=0.20, p=0.035) (Table 4.15).

In Yenga, Chlorophyceae was significantly affected by BOD (r=0.2, p=0.002) while Euglenophyceae was influenced by TP (r=0.32, p=0.000). (Table 12). In Mauna, pH, BOD, TP and TN were the most influential parameters of phytoplankton. BOD was found to be significantly related to Bacillariophyceae (r=0.13, p=0.045), pH and TP to Chlorophyceae (r=0.1; p=0.045; r=0.24; p=0.014 respectively).

**Table 4.15 Spearman rank (rs) correlation between abundance of phytoplankton and pH, temperature, Biological oxygen demand (BOD), dissolved oxygen (DO) total nitrates (TN) and total phosphates (TP) in Kesses, Kerita, Mauna and Yenga dams during the period between November 2010 and June 2011 (* = r is significant at p=0.05)**

| | | Taxa | pH | Temp | BOD | DO | TN | TP |
|---|---|---|---|---|---|---|---|---|
| **Uasin-Gishu** | Kesses | Bacillariophyceae | 0.372* | -0.114 | -0.052 | -0.074 | 0.117 | -0.29* |
| | | Chlorophyceae | 0.095 | -0.013 | 0.051 | 0.024 | -0.013 | -0.289 |
| | | Cyanophyceae | 0.188 | -0.15* | 0.03* | -0.18* | -0.07* | -0.131 |
| | | Desmidiaceae | -0.108 | 0.053 | -0.235 | -0.26* | 0.265 | -0.227 |
| | | Euglenophyceae | _ | _ | _ | _ | _ | _ |
| | Kerita | Bacillariophycaea | 0.079 | -0.284 | -0.11* | 0.075 | 0.533* | -0.309 |
| | | Chlorophyceae | 0.20* | -0.006 | 0.389 | 0.17 | -0.502 | -0.364 |
| | | Cyanophyceae | 0.609 | -0.026 | 0.135 | 0.024 | -0.336 | -0.342 |
| | | Desmidiaceae | _ | _ | _ | _ | _ | _ |
| | | Euglenophyceae | 0.125 | 0.11* | 0.064 | -0.271 | 0.28* | -0.284 |
| **Siaya** | Yenga | Bacillariophyceae | - | - | - | - | - | - |
| | | Cyanophyceae | -0.135 | -0.209 | 0.03 | -0.199 | -0.168 | 0.256 |
| | | Chlorophyceae | 0.374 | -0.358 | 0.2* | 0.323 | -0.065 | -0.472 |
| | | Euglenophyceae | -0.089 | 0.155 | -0.09 | 0.016 | -0.102 | 0.32* |
| | Mauna | Bacillariophyceae | 0.34 | 0.163 | 0.024* | 0.115 | -0.188 | -0.223 |
| | | Chlorophyceae | 0.1* | 0.316 | -0.205 | -0.179 | -0.198 | 0.258 |
| | | Cyanophyceae | 0.17* | -0.282 | -0.337 | -0.497 | 0.005* | -0.02* |
| | | Desmidiaphyceae | -0.462 | 0.198 | -0.13* | -0.225 | 0.241* | 0.619* |

### 4.5.2 Effect of Water Quality on Macroinvertebrates

Following Spearman's rank correlations, several significant relationships were observed between water quality and macroinvertebrate. In Kesses, the order pulmonata was positively related with pH, temperature, BOD, DO and Total nitrogen (r=0.017, 0.240, 0.178, 0.305, 0.380 and 0.138) respectively while it was negatively related with total phosphorus (r=-0.044). C Abundance of prosobranchiata was positively related with pH, BOD and TN (r=0.245, 0.154 and 0.05) respectively with a negative correlation with the rest of the parameters, the order had a significant negative correlation with DO (r=0.06, p=0.000) and a significant positive correlation with TN (r=0.05, p=0.0167). All orders had both negative and positive relationship with different physico-chemical parameters and nutrients except order Hirudinea which registered all negative relationship with physico-chemical parameters and nutrients (Table 4.16a).

In Kerita Dam the order pulmonata was negatively related to all parameters and nutrients except for pH which recorded a weak positive relationship of 0.046, there was a significant positive correlation with temperature and DO (r=0.590, p=0.043, and 0.473, p=0.007) respectively. Some of the orders were negatively related with physico-chemical parameters and nutrients while others had a negative relationship. There were some significant differences in the correlation between some macroinvertebrates orders and some physico-chemical parameters (p<0.05) (Table 4.16a).

**Table 4.16a**. **Spearman rank (rs) correlations between macroinvertebrate abundance and water quality parameters in Kesses and Kerita Dams in Uasin Gishu during the study period (* indicates that r is significant at p<0.05).**

| Taxa | pH | Temp | BOD | DO | TN | TP |
|---|---|---|---|---|---|---|
| **KESSES** | | | | | | |
| Amphipoda | 0.320 | 0.027 | -0.066 | 0.021 | -0.091 | -0.167 |
| Coleoptera | -0.216 | 0.079 | 0.379 | 0.356 | 0.079 | 0.213 |
| Diptera | -0.299 | 0.183 | -0.005 | -0.024 | 0.054 | 0.041 |
| Ephemeroptera | -0.018 | -0.365 | 0.146 | 0.090* | -0.040* | 0.020 |
| Hemiptera | -0.032 | -0.160* | 0.006 | 0.018 | -0.175 | -0.057 |
| Hirudinea | -0.262 | -0.359 | -0.254 | -0.314 | -0.245 | -0.118 |
| Lamellibranchiata | 0.092 | -0.138 | -0.049 | -0.033 | 0.189 | -0.234 |
| Odonata | -0.227 | 0.155* | -0.158 | -0.424 | -0.063 | 0.312* |
| Plecoptera | -0.038 | 0.208 | 0.143 | 0.102 | 0.022 | 0.204 |
| Prosobranchiata | 0.245 | -0.214 | 0.154 | -0.060* | 0.050* | -0.015 |
| Pulmonata | 0.017 | 0.240* | 0.178 | 0.305 | 0.138 | -0.044 |
| Trichoptera | 0.442 | -0.180 | 0.064 | 0.137 | -0.326 | -0.376 |
| **KERITA** | | | | | | |
| Amphipoda | _ | _ | _ | _ | _ | _ |
| Coleoptera | -0.387 | -0.035 | -0.169 | -0.180 | 0.109 | 0.268 |
| Diptera | -0.237 | 0.243 | 0.217 | 0.529* | 0.169 | 0.419 |
| Ephemeroptera | -0.33 | -0.309 | -0.360 | 0.125 | -0.187 | 0.058 |
| Hemiptera | 0.164 | -0.390 | -0.115 | 0.560* | -0.172 | -0.190 |
| Hirudinea | 0.206 | -0.057 | 0.165 | 0.073 | -0.145 | 0.030 |
| Lamellibranchiata | 0.248 | 0.019 | -0.003 | -0.327 | -0.132 | -0.357 |
| Odonata | -0.136 | 0.496* | 0.478* | -0.101 | -0.355 | -0.142 |
| Plecoptera | _ | _ | _ | _ | _ | _ |
| Prosobranchiata | -0.375 | -0.340 | -0.293 | -0.109 | -0.066 | 0.241 |
| Pulmonata | 0.046 | 0.590* | -0.439 | 0.473* | -0.374 | -0.326 |
| Trichoptera | 0.130 | 0.563* | -0.364 | -0.391 | -0.457 | -0.363 |

In Siaya, Spearman's, relationships between macroinvertebrates and physico-chemical parameters were also evident. In Yenga Dam the order amphipoda was negatively related to pH and positively related to nutrients and other parameters while the order pulmonata was negatively related with BOD, TN. Pulmonat was significantly related to DO (r=-0.64, p=0.000) while it was positively related with pH, Temperature and TP (p>0.05).

Abundance of prosobranchiata was positively correlated with all water quality parameters while all other had both negative and positive relationship with different physico-chemical parameters and nutrients (Table 4.16b).

Both positive and negative correlations between macroinvertebrates and water quality physico-chemical parameters and nutrients were observed in Mauna (Table 4.16b). The order amphipoda had negative relationships with DO and TP and positive relationship with all other physico-chemical parameters. Pulmonata had a significant positive relationship with pH ($r=0.791$, $p=0.024$) while it was negatively correlated with temperature, BOD, DO and nutrients. Prosobranchiata had a significant positive correlation with pH and TP ($r=0.712$, $p=0.021$ and $r=0.18$, $p=0.015$ respectively) and negative correlations with temperature, BOD, DO and TN with no significant relationship.

**Table 4.16b Spearman rank (rs) correlations between macro-invertebrate abundance and water quality parameters in Yenga and Mauna Dams in Siaya during the study period (* = r is significant at p<0.05)**

| Taxa | pH | Temp | BOD | DO | TN | TP |
|---|---|---|---|---|---|---|
| **YENGA** | | | | | | |
| Amphipoda | -0.338 | 0.055 | -0.400* | 0.447 | 0.531 | 0.247 |
| Coleoptera | -0.253 | 0.089 | -0.069 | -0.235 | -0.265 | -0.181 |
| Diptera | 0.075 | 0.500* | 0.125 | 0.137 | 0.079 | -0.411 |
| Ephemeroptera | 0.227 | 0.099 | -0.153 | 0.001 | -0.253 | 0.069 |
| Hemiptera | -0.561 | 0.227 | -0.364 | 0.247 | -0.052 | -0.149 |
| Hirudinea | -0.188 | -0.203 | -0.164 | -0.140 | -0.217 | -0.273 |
| Lamellibranchiata | 0.205 | -0.148 | 0.083 | -0.342 | -0.489 | -0.308 |
| Odonata | 0.280* | 0.287 | -0.006 | -0.134 | 0.102 | 0.200* |
| Plecoptera | 0.041 | 0.038 | -0.203 | -0.072 | 0.299 | 0.335 |
| Prosobranchiata | 0.181 | 0.148 | -0.050* | -0.211 | 0.182 | 0.276 |
| Pulmonata | 0.463 | 0.096 | 0.438 | -0.330 | -0.140 | 0.119 |
| **MAUNA** | | | | | | |
| Amphipoda | 0.146 | 0.229 | -0.089 | 0.473 | 0.118 | -0.149 |
| Coleoptera | -0.367 | 0.090* | -0.065 | 0.251 | -0.127 | 0.181 |
| Diptera | 0.407 | -0.020 | 0.292 | 0.360 | -0.366 | -0.090* |
| Ephemeroptera | 0.134 | -0.141 | 0.017 | 0.425 | -0.265 | -0.203 |
| Hemiptera | 0.561* | -0.253 | 0.587* | -0.208 | -0.499 | -0.125 |
| Lamellibranchiata | -0.360* | 0.403 | -0.094 | -0.070* | 0.166 | -0.110* |
| Odonata | 0.771* | -0.365* | 0.908* | 0.056 | -0.530* | -0.303 |
| Prosobranchiata | 0.712* | -0.472 | 0.593* | -0.275 | -0.165 | 0.180* |
| Pulmonata | 0.791* | -0.497 | 0.737* | -0.032 | -0.489 | -0.296 |

### 4.5.3 Effect of Water Quality on Abundance, Prevalence and Mean Intensity of Fish Parasites

Strong correlations between parasitic abundance levels and some water quality parameters were observed. In Kesses, significant strong positive correlations were found between *Tylodelphys* and pH (rs = 0.677; p= 0.0001) and *Diplostomum* and TP (rs= 0.619; p=0.0001). Negative correlations were found between TN and *Tylodelphys* (rs=-0.621; p=0.0001) and TP and both *Tylodelphys* and *Diplostomum* (rs= -0.737; p= 0.0001). (Table 4.17a). In Kerita, *Clinostomum* had a strong positive correlation with temperature (rs= 0.634; p= 0.002) and strong negative correlations between *Contracaecum* and temperature (rs= -0.589; p= 0.0048), BOD (rs= -0.701; p=0.0003) and DO (rs=-0.509; p=0.018). In Mauna, strong positive correlations were found between temperature and *Diplostomum* (rs = 0.63; p = 0.002). A strong negative correlation was found between *Tylodelphy*s and TP (rs = -0.53; p = 0.005) (Table 4.17b). In Yenga, positive correlations were found between *Clinostomum* and temperature (rs = 0.56; p = 0.004) and *Amirthalingamia* and TP (rs = 0.51; p = 0.01). *Amirthalingamia* and pH had a negative correlation (rs = -0.47; p = 0.033).

**Table 4.17a**. **Spearman rank (rs) correlation between parasite abundance and water quality parameters in small water bodies in Kesses and Kerita in Uasin Gishu. (* indicates significant correlations; p =0.05).**

| | pH | | Temp | | BOD | | DO | | TN | | TP | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | rs | p | rs | p | S | p | rs | p | Rs | p | rs | p |
| **Kesses** | | | | | | | | | | | | |
| *Clinostomum* | -0.36 | 0.019* | 0.10 | 0.521 | -0.46 | 0.002* | -0.70 | 0.009 | 0.24 | 0.129 | 0.12 | 0.412 |
| *Contracaecum* | 0.00 | 1.00 | -0.24 | 0.115 | 20.04 | 0.763 | -0.283 | 0.069 | -0.14 | 0.370 | 0.12 | 0.439 |
| *Amirthalingamia* | 0.12 | 0.444 | 0.023 | 0.882 | 0.175 | 0.268 | 0.318 | 0.04* | -0.02 | 0.922 | -0.43 | 0.785 |
| *Tylodelphys* | 0.68 | 0.0001* | -0.28 | 0.068 | -0.11 | 0.504 | 0.011 | 0.946 | -0.62 | 0.0001* | -0.75 | 0.0001* |
| *Diplostomum* | -0.37 | 0.0174 | -0.05 | 0.760 | 0.28 | 0.068 | 0.27 | 0.076 | 0.30 | 0.053 | 0.62 | 0.0001* |
| **Kerita** | | | | | | | | | | | | |
| *Clinostomum* | -0.01 | 0.957 | 0.63 | 0.002* | 0.141 | 0.543 | -0.07 | 0.743 | 0.34 | 0.123 | 0.29 | 0.197 |
| *Contracaecum* | -0.42 | 0.056 | -0.58 | 0.004* | -0.70 | 0.0003* | -0.51 | 0.018* | -0.27 | 0.221 | -0.26 | 0.256 |
| *Amirthalingamia* | 0.049 | 0.834 | 0.19 | 0.441 | -0.26 | 0.261 | -0.41 | 0.065 | -0.32 | 0.125 | -0.29 | 0.191 |
| *Tylodelphys* | 0.152 | 0.509 | -0.04 | 0.853 | 0.14 | 0.545 | 0.047 | 0.841 | -0.44 | 0.045* | -0.45 | 0.042* |
| *Diplostomum* | -0.04 | 0.867 | 0.12 | 0.60 | -0.33 | 0.145 | -0.69 | 0.004* | -0.19 | 0.399 | -0.07 | 0.748 |

**Table 4.17b. Spearman rank (rs) correlation between parasite abundance and water quality parameters in small water bodies Mauna and Yenga. (* indicates significant correlations; p= 0.05).**

| | pH | | Temp | | BOD | | DO | | TN | | TP | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Rs | p | Rs | P | Rs | p | Rs | P | Rs | P | rs | p |
| **Mauna** | | | | | | | | | | | | |
| *Clinostomum* | -0.02 | 0.911 | 0.36 | 0.114 | 0.31 | 0.171 | 0.35 | 0.115 | 0.15 | 0.499 | -0.38 | 0.087 |
| *Contracaecum* | -0.32 | 0.144 | 0.18 | 0.417 | -0.10 | 0.649 | -0.105 | 0.649 | -0.07 | 0.758 | -0.19 | 0.398 |
| *Amirthalingamia* | 0.10 | 0.643 | -0.27 | 0.231 | -0.13 | 0.576 | -0.13 | 0.576 | 0.14 | 0.545 | 0.29 | 0.197 |
| *Tylodelphys* | -0.15 | 0.523 | 0.47 | 0.029* | 0.11 | 0.624 | 0.11 | 0.624 | 0.07 | 0.73 | -0.58 | 0.005* |
| *Diplostomum* | -0.29 | 0.194 | 0.63 | 0.002* | 0.045 | 0.848 | 0.04 | 0.847 | 0.18 | 0.428 | -0.18 | 0.415 |
| **Yenga** | | | | | | | | | | | | |
| *Clinostomum* | -0.13 | 0.533 | 0.56 | 0.004* | 0.35 | 0.088 | 0.24 | 0.242 | 0.08 | 0.699 | 0.26 | 0.209 |
| *Contracaecum* | - | - | - | - | - | - | - | - | - | - | - | - |
| *Amirthalingamia* | -0.47 | 0.033* | 0.11 | 0.591 | 0.17 | 0.421 | 0.38 | 0.061 | 0.17 | 0.403 | 0.51 | 0.01* |
| *Tylodelphys* | - | - | - | - | - | - | - | - | - | - | - | - |
| *Diplostomum* | -0.29 | 0.163 | -0.06 | 0,771 | 0.20 | 0.336 | 0.41 | 0.046* | 0.28 | 0.180 | 0.60 | 0.002* |

## 4.6 Effect of Seasonality on Water Quality and Biota

### 4.6.1 Effect of Seasonality on Water Quality

During this study, rainfall data records indicated that the dry season for Uasin Gishu November 2010 to February 2011 and in Siaya was November 2010, January, February and July 2011. The wet season for Uasin Gishu was November to December 2010 and March to June 2011 and for Siaya was August to December 2011.

Differences in water quality were observed between dry and wet seasons (Table 4.18). In the wet season, pH was higher than in the dry season in Kesses ($t=14.44$; $p=0.000$) and Kerita ($t=47.5$; $p=0.010$). Dissolved oxygen was higher during the wet season in Mauna ($t= 6.6$; $p= 0.000$) and Yenga ($t= 0.51$; $p= 0.035$). Similarly, BOD was higher during the wet season in Mauna ($t= 5.82$; $p= 0.001$) and Yenga ($t= 6.88$; $p= 0.011$). Total nitrates were higher during the dry season in Kesses ($t=9.38$; $p=0.002$) and in Kerita ($t = 9.46$; $p = 0.000$). Total phosphates were also high during the dry season in Kesses ($t= 5.02$; $p= 0.023$) and Kerita ($t= 7.21$; $p= 0.003$).

**Table 4.18 T-tests for water quality parameters between wet and dry seasons in Kerita, Kesses, Mauna and Yenga dams. ( [1] and [2] indicates parameters higher in dry season and wet season respectively; p= 0.05).**

| | Kesses | | Kerita | | Mauna | | Yenga | |
|---|---|---|---|---|---|---|---|---|
| | T | P | t | P | t | P | T | p |
| pH | 14.44 | 0.000[2] | 47.5 | 0.010[2] | 0.18 | 0.833 | 0.56 | 0.967 |
| Temperature | 2.84 | 0.096 | 0.03 | 0.570 | 2.67 | 0.419 | 0.99 | 0.163 |
| DO | 1.5 | 0.219 | 0.55 | 0.676 | 6.60 | 0.000[2] | 0.51 | 0.035[2] |
| BOD | 0.48 | 0.681 | 0.55 | 0.192 | 5.82 | 0.001[2] | 6.88 | 0.011[2] |
| TN | 9.38 | 0.002[1] | 9.46 | 0.000[1] | 0.05 | 0.900 | 0.15 | 0.768 |
| TP | 5.02 | 0.023[1] | 7.21 | 0.003[1] | 1.75 | 0.293 | 0.02 | 0.130 |

### 4.6.2 Effect of Seasonality on Phytoplankton

Dry and wet seasons affected the relative abundance of phytoplankton; with Kruskal Wallis tests indicating that in Kesses, Desmidiaceae was more abundant during the wet season than the dry season (H = 4.07; p = 0.044 respectively) while in Kerita, Bacillariophyceae was also more abundant during the wet season (H = 5.45; p = 0.738) (Table 4.19).

**Table 4.19 Kruskal Wallis (H) test on the effect of dry and wet seasons on relative abundance of phytoplankton in Kesses, Kerita, Mauna and Yenga dams. ([1] and [2] indicate significantly higher abundance in the dry season wet season respectively; p= 0.05). – indicates absence of phytoplankton in respective dam during this study.**

| | Kesses | | Kerita | | Mauna | | Yenga | |
|---|---|---|---|---|---|---|---|---|
| | H | P | H | P | H | P | H | p |
| Bacillariophyceae | 0.11 | 0.738 | 5.45 | 0.02[2] | 0.01 | 0.907 | - | - |
| Chlorophyceae | 2.14 | 0.144 | 2.42 | 0.119 | 2.93 | 0.087 | 0.04 | 0.835 |
| Cyanophyceae | 0.03 | 0.856 | 0.22 | 0.640 | 2.07 | 0.150 | 0.07 | 0.788 |
| Desmidiaceae | 4.07 | 0.044[2] | - | - | 0.61 | 0.436 | - | - |
| Euglenophyceae | - | - | 0.05 | 0.815 | - | - | 0.001 | 0.975 |

### 4.6.3. Effect of seasonality on macroinvertebrates

Kruskal-Wallis test on differences in seasonal abundance of macroinvertebrates indicated seasonal differences in abundance of macroinvertebrates (Table 4.20). In Kesses, Hemiptera ($H= 16.07$; $p= 0.000$) and Pulmonata ($H= 9.04$; p 0.003) were higher during the dry than wet season while the opposite was true for Odonata ($H= 25.1$; $p=0.00$), Coleoptera ($H= 9.04$; $p=0.0003$) and Ephemeroptera ($H= 9.04$; $p=0.003$). In Yenga, Plecoptera ($H= 16.07$; $p= 0.000$), Amphipoda ($H= 9.04$; $p= 0.000$) and Prosobranchiata ($H= 25.12$; $p=0.000$) were in higher during the wet than dry season. In Yenga, Plecoptera ($H= 4.61$; $p=0.032$), Hirudnea ($H= 4.61$; $p= 0.032$) and Lamelliobranchiata ($H= 5.83$; $p= 0.016$) were higher during the dry season while Ephemeroptera ($H= 10.37$; $p= 0.006$) and Pulmonata ($H= 5.83$; $p= 0.016$) in the rainy season. Pulmonata was found in higher abundance in the wet than dry season in Mauna ($H= 4.42$; $p= 0.035$). There were no seasonal variations in abundance of macroinvertebrates between seasons in Kerita dam.

**Table 4.20. Kruskal Wallis (H) test on the effect of dry and wet seasons on macroinvertebrate abundance in Kesses, Kerita, Mauna and Yenga. ([1] and [2] indicate higher abundance in dry season and wet seasons respectively; p= 0.05). – indicates absence of macroinvertebrates in respective dam during this study.**

| | Kesses | | Kerita | | Mauna | | Yenga | |
|---|---|---|---|---|---|---|---|---|
| | H | p | H | p | H | p | H | P |
| Odonata | 25.12 | 0.000[1] | 0.12 | 0.726 | 1.96 | 0.161 | 1.80 | 0.18 |
| Diptera | 1.00 | 0.316 | 1.10 | 0.293 | 3.07 | 0.080 | 0.00 | 1.00 |
| Coleoptera | 9.04 | 0.003[2] | 0.12 | 0.726 | 4.42 | 0.036[2] | 1.15 | 0.283 |
| Ephemeroptera | 9.04 | 0.003[2] | 0.12 | 0.726 | 1.96 | 0.161 | 10.37 | 0.001[2] |
| Plecoptera | 16.07 | 0.000[2] | - | - | - | - | 4.61 | 0.032[1] |
| Trichoptera | 1.00 | 0.316 | - | - | - | - | - | - |
| Amphipoda | 9.04 | 0.003[2] | - | - | 0.000 | 1.000 | 0.07 | 0.788 |
| Hirudnea | 1.00 | 0.316 | 0.00 | 1.000 | 1.10 | 0.293 | 4.61 | 0.032[1] |
| Hemiptera | 16.07 | 0.000[1] | 0.12 | 0.726 | 4.42 | 0.036[1] | 0.07 | 0.788 |
| Pulmonata | 9.04 | 0.003 | 1.10 | 0.293 | 4.42 | 0.036[1] | 5.83 | 0.016[2] |
| Prosobranchiata | 25.12 | 0.000[2] | 0.49 | 0.484 | 0.49 | 0.484 | 0.07 | 0.788 |
| Lamelliobranchiata | - | - | 0.00 | 1.000 | 1.96 | 0.161 | 5.83 | 0.016 |

### 4.6.4 Effect of Seasonality on Fish Parasites

Rainfall seasonality was found to affect parasite abundance during this study. Abundance of *Clinostomum* parasites was found to be higher during the dry than the wet season in Kesses (H= 16.07; p= 0.000) while in Kerita and Mauna it was higher during the wet season (H= 4.42; p= 0.036; H= 4.42; p= 0.036 respectively). Abundance of *Amirthalingamia* was higher during the wet season in Kesses (H= 4.02; p= 0.045) while *Tylodelphys* was higher in the rainy season in Kesses (H= 25.12; p= 0.000). (Table 4.21).

**Table 4.21**. **Kruskal Wallis (H) on effect of dry and wet seasons on parasite abundance in Kesses and Kerita, Mauna and Yenga. ( $^{1}$ and $^{2}$ indicate abundance of parasite significantly higher in dry and wet season respectively; p= 0.05). – indicates absence of parasites in respective dam during this study.**

| | Kesses | | Kerita | | Mauna | | Yenga | |
|---|---|---|---|---|---|---|---|---|
| | H | p | H | p | H | p | H | p |
| *Clinostomum* | 16.07 | 0.000 $^{1}$ | 4.42 | 0.036 $^{2}$ | 4.42 | 0.036 $^{2}$ | 1.15 | 0.283 |
| *Contracaecum* | 2.26 | 0.133 | 7.855 | 0.005 $^{1}$ | 0.12 | 0.726 | - | - |
| *Amirthalingamia* | 4.02 | 0.045 $^{2}$ | 0.12 | 0,726 | 0.196 | 0.161 | 1.15 | 0.283 |
| *Tylodelphys* | 25.12 | 0.000 $^{2}$ | 1.96 | 0.161 | 3.07 | 0.080 | - | - |
| *Diplostomum* | 2.26 | 0.133 | 0.49 | 0.484 | 0.49 | 0.484 | - | - |

The effect of parasitic infection on the condition factor of fish was influenced by rainfall seasonality and sex of fish (Table 4.22). In Uasin Gishu, parasitic infection was mostly associated with higher condition factor of fish during the dry season and rainy season in Siaya. In Kesses, the condition factor of infected male fish was significantly higher than un-infected ones with *Tylodelphys* (F=18.86; p=0.000), *Diplostomum* (p=0.000; F= 46.10) and *Amirthalingamia* (F= 21.89; p=0.000) during the dry season. During the rainy season, males infected with *Contracaecum* had higher condition factor that un-infected ones (p= 0.002; F= 10.40).In the dry season, un-infected female fish had higher condition factor than infected with *Contracaecum* (F=5.58; p=0.02) while infected fish had higher condition factors with *Tylodelphys* (F=13.39; p=0.000), *Diplostomum* (F=23.64; p=0.000;) and *Amirthalingamia* (F= 13.97; p=0.000). Parasitic infection had no effect on condition of female fish during rainy season in Kesses.

In Kerita, condition of male fish was not affected by parasitic infection in both dry and rainy seasons. However, in females, it was significantly higher in infected than un-infected fish with *Contracaecum* (F=12.24; p= 0.001) and Tylodelphys (F= 6.38; p= 0.013) during the dry season. Infection with *Diplostomum* resulted in higher condition factor in infected females compared to un-infected ones in the rainy season (F=6.13; p=0.017) and lower during the dry season (F= 10.30; p=0.002) (Table 4.22).

**Table 4.22 ANOVA for effect of dry (D) and wet (W) seasons on condition factor of male (♂) and female (♀) *O. niloticus* in Kesses, Kerita, Mauna and Yenga dams. ( [1] and [2] indicate higher condition factor in infected and un-infected fish respectively; p= 0.05). - indicates absence of parasites during this study.**

| | | *Clinostomum* | | *Contracaecum* | | *Tylodelphys* | | *Diplostomum* | | *Amirthalingamia* | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | *F* | *p* | *F* | *p* | *F* | *p* | *F* | *p* | *F* | *p* |
| **Kesses** | | | | | | | | | | | |
| ♂ | D | 1.89 | 0.155 | 2.58 | 0.110 | 18.86 | 0.000[1] | 46.10 | 0.000[1] | 21.89 | 0.000[1] |
| | W | 0.83 | 0.368 | 10.40 | 0.002[1] | 2.78 | 0.102 | 0.00 | 0.982 | 0.17 | 0.680 |
| ♀ | D | 1.24 | 0.268 | 5.58 | 0.02 | 13.39 | 0.000[1] | 23.64 | 0.000[1] | 13.97 | 0.000[1] |
| | W | 0.00 | 0.958 | 3.66 | 0.062 | 0.77 | 0.385 | 1.48 | 0.230 | 0.46 | 0.502 |
| **Kerita** | | | | | | | | | | | |
| ♂ | D | 9.82 | 0.002[2] | 1.67 | 0.200 | 1.46 | 0.231 | 0.10 | 0.752 | 14.44 | 0.000[2] |
| | W | 0.18 | 0.672 | 0.89 | 0.350 | 0.11 | 0.739 | 0.85 | 0.361 | 0.87 | 0.354 |
| ♀ | D | 1.07 | 0.302 | 12.24 | 0.001[1] | 6.38 | 0.013[1] | 10.30 | 0.002[2] | 0.03 | 0.872 |
| | W | 1.82 | 0.184 | 0.93 | 0.341 | 0.04 | 0.852 | 6.13 | 0.017[1] | 0.01 | 0.933 |
| **Mauna** | | | | | | | | | | | |
| ♂ | D | 0.45 | 0.503 | - | - | - | - | 0.03 | 0.871 | 0.43 | 0.513 |
| | W | 0.02 | 0.880 | - | - | 0.08 | 0.783 | - | - | - | - |
| ♀ | D | 1.62 | 0.206 | - | - | - | - | - | - | - | - |
| | W | 0.11 | 0.736 | 1.21 | 0.274 | 0.44 | 0.844 | - | - | 4.80 | 0.032[2] |
| **Yenga** | | | | | | | | | | | |
| ♂ | D | 1.35 | 0.249 | - | - | 0.65 | 0.425 | 2.08 | 0.154 | 0.58 | 0.449 |
| | W | 12.59 | 0.001[1] | - | - | 0.30 | 0.584 | - | - | 40.96 | 0.000[1] |
| ♀ | D | 2.62 | 0.115 | - | - | 0.09 | 0.766 | 0.73 | 0.400 | 0.55 | 0.463 |
| | W | 3.79 | 0.061 | - | - | 0.00 | 0.995 | - | - | 320.17 | 0.000[1] |

While parasitic infection had no effect on condition factor of fish during the dry season in Yenga, there were some significant effects during the rainy season. Males infected with *Clinostomum* and *Amirthalingamia* had a higher condition factor than un-infected ones ($F= 12.50$; $p= 0.001$ and $F= 40.96$; $p=0.000$ respectively) during the rainy season while females infected with *Amirthalingamia* had a higher condition factor than un-infected ones ($F=320.17$; $p=0.00$). In Mauna, *Contracaecum* was only found in females during the rainy season. *Tylodelphys* was found in both males and females only during the rainy season while *Diplostomum* was found in males during the dry season only and absent in females in both dry and rainy seasons. The only parasite found to affect condition of fish in Mauna was *Amirthalingamia;* with significantly lower condition factor in infected female fish compared to un-infected ones ($F=4.80$; $p=0.032$).

# CHAPTER FIVE

# DISCUSSION

## 5.1 Water Quality

Differences in Siaya and Uasin Gishu with respect to temperature are attributed to climatic differences based on altitude in the two regions, with temperatures in Uasin Gishu being lower than Siaya. Water temperatures were higher in Siaya than Uasin Gishu. Siaya water bodies were influenced more by temperature during the months of March to June 2011, which corresponds to the wet season. Strong temperature fluctuations in this study correspond to dry and wet seasons as well as availability of vegetation cover to shade the water. Variation in temperature is a common feature especially in tropical freshwater systems that are characterized by marked dry and wet hydroperiods (Escalera-Vazquez & Zambrano, 2010). Higher temperatures in the rainy season have been associated with deep reservoirs that have a long residence time; favoring thermal and chemical stratification in the rainy season (Ayroza *et al*. 2013). Temperature was found to influence dissolved oxygen in this study. This is in agreement with Mwaura (2006) who noted that atmospheric temperature is one of the factors that influence dissolved oxygen. The effect of altitude on temperature and subsequently dissolved oxygen was demonstrated (Masese *et al*., (2009a).

Fluctuations in dissolved oxygen were evident in this study. In both Uasin Gishu and Siaya, these are caused by dry and wet seasons leading to higher DO in the rainy season. Higher water turbulence in Uasin Gishu and high water temperatures in Siaya contributed to higher DO levels in Uasin Gishu. Wynes and Wissing (1981) found that low turbulence especially in the dry season results in low disssolved oxygen. This is supported by findings in this study where Uasin Gishu which is found in higher elevation is a more windy place than Siaya especially during the rainy season. Although large water bodies tend to have higher dissolved oxygen levels due to higher turbulence, Kerita dam which was smaller than Yenga and Mauna had higher dissolved oxygen. This was attributed to higher elevation in Kerita. A similar observation was made by Pacini, 1994) who found DO levels at Masinga, a relatively large dam to be much lower at 0.2 -10.3 $mgl^{-1}$. This further explains the low DO levels in Siaya during the dry season. Lower dissolved oxygen levels during the dry season in Siaya is also attributed to higher water temperatures which lead to lower dissolved oxygen levels in water (Addy and Green, 1997). Furthermore, decreased shading and subsequent warmer water temperature may also contribute to lower dissolved oxygen concentrations and was more evident in Siaya where the water bodies had little or no trees around them compared to Uasin Gishu.

Contrary to findings in this study, Mwaura (2006) recorded higher dissolved oxygen levels during the dry season, a factor he attributed to high photosynthetic rate of algae and submerged macrophytes at high temperatures. Kaggwa *et al*. (2011) also found that high temperatures in small water bodies in Machakos were responsible for supersaturation of oxygen as a result of high net productivity; a feature common to tropical water bodies. Though temperatures in Siaya were high, DO levels were low. In Yenga, this could not have been the case as the water was generally much clearer than all other water bodies. Furthermore, submerged macrophytes and other aquatic plants were limited in this deep, steep walled dam. Mauna had a high floating and submerged plants cover which, combined with the grayish coloration of the water, observations that are indicative of anaerobic decomposition.

Fluctuations in pH were evident in this study and are a common feature to other aquatic ecosystems (Taub, 1996; Mwaura, 2006; Masese *et al.*, 2009a). High pH in Kesses and Kerita during the wet season is due to non-point sources of pollution that are mobilized during the rainy season through runoff and leaching from agricultural areas (Masese *et al.*, 2009a). According to Mwaura (2006), pH is usually low due to high evaporative concentration of ions in combination with changes in photosynthetic intensity during the dry season. Intense decomposition and respiration especially in low DO conditions also contributes to low pH (Ayroza *et al.* 2013). Other than seasonality, other factors such as topographic characteristics and soil type account for fluctuations in pH.

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The fluctuations in pH in this study can be as a result of several factors as explained by Boyd (1990). First is carbon dioxide solubility. Carbon dioxide content in water is a function of biological activity of respiration and photosynthesis. Although pH is lower at higher carbon dioxide concentrations, it is assumed that carbon dioxide cannot make water more acid than pH 4.5 as only a small proportion forms carbonic acid ($H_2CO_3$). Secondly, pH is influenced by formation of bicarbonate as a result of carbon dioxide in natural water reacting with bases in rocks and soils, exchange reactions between water and soil and ammonia release from fish and other aquatic organisms. Lastly, diel fluctuations of pH result from removal of carbon dioxide for photosynthesis by aquatic plants. During daylight, aquatic plants remove carbon dioxide from water for photosynthesis and in the night, both plants and animals release carbon dioxide into water by respiration. However, the rate of removal of carbon dioxide from water by aquatic plants is usually faster than can be replaced by respiration.

As carbon dioxide is removed, carbonate accumulates and hydrolyses (gives away hydroxyl ions); and pH increases. Plants can continue to use the small amounts of carbon dioxide available at pH values above 8.3, and bicarbonate may be absorbed by plants and some of the carbon from bicarbonate used in photosynthesis. When bicarbonate concentrations are low, waters are poorly buffered while as pH values of 9 to 10 are common during periods of intense photosynthesis. During the night, carbon dioxide accumulates and pH falls. Removal of carbon dioxide also causes slight shift in concentration of bicarbonate and carbonate during diel cycle. Conversion of carbon dioxide to organic carbon by photosynthesis may exceed its release by respiration especially during warm seasons; resulting in higher pH values.

Mwaura (2006) also points out that the type of soil and lacustrine sediments can be associated with bicarbonate and carbonate ions; resulting in high pH. Loss of macrophyte cover and other aquatic vegetation can lead to low buffering capacity especially during the dry when they are harvested or burnt down. Furthermore, during the dry season, water levels recede drastically from the macrophyte covered further lowering buffering capacity. Increase in pollutants due to evapo-concentration is yet another factor in high pH during the dry season (Marshall *et al*., 2005; Mwaura, 2006)

The influence of total nitrates in Uasin Gishu dams was evident in this study. It appears that Kesses and Kerita in Uasin Gishu have high rates of decomposition of organic matter since they have high levels of both TN and DO especially during the dry season. Siaya on the other hand, particularly Mauna had high TP in the dry season. This can be explained by low redox potential resulting from anoxia in the hypolimnion and bottom sediments (Boyd 1990). It does seem that that during the dry season, the nitrate- high Uasin Gishu water bodies are characterized by aerobic respiration. On the other hand, the Siaya water bodies which are high in phosphorus during the dry season are indicative of anaerobic respiration.

High amounts of phosphorus which arise from low dissolved oxygen levels in anoxic environments are associated with low decomposition rates in the water column, leading to more organic production and greater respiration rates, limited light penetration and eventually restriction of photosynthetic oxygen production to shallower depths; reducing the epilimnion and eventually restricting photosynthetic oxygen production to shallower depths. This explains why Mauna was strongly uncorrelated to all the other water bodies. Furthermore, unlike the other three water bodies in this study, Mauna had more catfish (*Clarias gariepinus*) caught than tilapia during this study, indicating low dissolved oxygen levels that favor this type of fish. Monitoring of such water bodies is critical as oxygen depletion may be inevitable especially in shallow waters where the sediment area to water volume ration is high (Kaggwa *et al.*, 2011).

## 5.2 Phytoplankton and Effects of Water Quality and Seasonality

The influence of water quality on phytoplankton composition and abundance was evident. In Uasin Gishu, phytoplankton abundance was influenced most by TN, temperature and pH, affecting abundance of Chlorophyceae, Bacillariophyceae and Euglenophyceae. Even within this region, there appeared to be differences in the effect of water quality on phytoplankton. For instance, In Kesses, Chlorophyceae is influenced most by temperature. The influence of pH on Chlorophyceae in Kerita and Mauna indicates that although they are in different geographic locations, they have some similar such light availability and nutrient dynamics (Huszar and Marino, 2000). That nitrogen-limited systems are not suitable for chlorophyceae (Reynolds, 1993) was seen in Mauna which had low notrogen and high cyanophyte abundance.

Seasonality of Phytoplankton was most evident in Uasin Gishu and is linked marked variations in temperature, light intensity and nutrient levels between dry and wet seasons. Water level changes also influence seasonality of phytoplankton especially during the dry season when access to nutrients from the sediments is limited. During the rainy season, Desmydiaceae was more abundant in Kesses while Bacillariophyceae was more abundant in Kerita. Although this study did not indicate lower temperatures during the rainy season, temperatures in Uasin Gishu are often higher than during the dry season; favoring the occurrence of Desmydiacea in this water body. This was enhanced further by total nitrate concentrations which were also found to be lower during the rainy season in Kesses. Although having the same geographical and seasonal conditions; higher abundance of Bacillariophyceae in Kerita may be attributed to the fact that it is much smaller and therefore more vulnerable to increased nutrient levels during the rainy season than the larger Kesses. In Siaya, there was no temporal variation in phytoplankton abundance.

Seasonal variation in temperature in Uasin Gishu and Siaya appears to be key determining factor in water quality and subsequently influences DO levels. In combination with total nitrates and phosphates, DO levels are most critical in determining the type of decomposition that takes place in the water and ultimately influences the type of biota. Anthropological activities such as farming, washing clothes, cutting of vegetation along the banks of the water body are linked to non-point pollution and further contribute to seasonal variation of water quality, nutrients and phytoplankton communities a view in agreement with that of Masese *et al*., (2009a).

Changes in nutrient levels were seen in Uasin Gishu and Siaya. In Uasin Gishu, total nitrate levels were higher during the dry season. This was attributed to inputs from neighbouring agricultural land followed by evapo-concentration of nutrients during the dry season. High temperatures during the dry season in Siaya are associated with high rates of decomposition of organic matter, relaeasing orthophosphates in the water. Nitrogen and phosphorus are the most important nutrient factors causing high productivity in water bodies. Limitation of any one of phosphorus, nitrogen or carbon which are found in plant organic matter of aquatic algae and macrophytes results in excess amounts of the other two nutrients; with phosphorus contributing as high as 500 times its weight in living algae while nitrogen and carbon contribute 71 12 times each respectively. Phosphorus and nitrogen are usually the first to impose limitation on freshwater sytem, with phosphorus regulating productivity of most freshwater systems (Schindler *et al.*, 1987). On the other hand, increased levels of nitrates and phosphates in water bodies may contribute to the process of eutrophication (Swierk and Szpakowska 2011).

Primary productivity was higher in Uasin Gishu than Siaya dams and this was attributed to several factors. First, the geographical location of Uasin Gishu in a high altitude area favors higher dissolved oxygen levels than Siaya. Furthermore, higher temperatures especially during the dry season in Siaya result in lower dissolved oxygen levels, leading to lower primary productivity of phytoplankton. Secondly, intensive agricultural activities involving application of chemical fertilizers contributes to high nutrient levels in water bodies, favoring high primary production in Uasin Gishu. Thirdly, dams in Uasin Gishu are shallower than Siaya, therefore allowing organic matter to be re-circulated in the system. Mixing in shallow lakes and reservoirs allows nutrients to be deposited to the photic zone, making them available for phytoplankton. This is supported by Thomas *et al*. (2000) who found shallow reservoirs to be more productive.

Deep lakes are on the other hand generally oligotrophic due to large volumes of water, limited mixing and thermal stratification. Algal growth in this case is limited to the epilimnion, rendering the nutrient rich hypolimnion out of reach for phytoplankton (Kotut, 1998). This explains the low productivity in Yenga dam which was the deepest of the four dams in this study. Phytoplankton biomass productivity in this study was higher than that in other small water bodies in Uganda (Kaggwa *et al*., 2009). Productivity in fertilized ponds is expected to be much higher than in this study as has been shown by Veverica *et al*., (2001). Spatial and temporal variation in phytoplankton biomass in this study can be explained by changes in temperature, light intensity, availability of organic carbon and availability of TN and TP.

The effect of geographical location, seasonality and nutrient levels on phytoplankton assemblages in freshwater systems have been demonstrated by Souza *et al*., (2008) and Trebisan and Forsberg (2007). Nutrient and turbidity enhancement has been found to result in replacement of original inhabitants with chlorophytes, therefore influence primary productivity. The effect of seasonal variation and nutrient dynamics on phytoplankton is more pronounced in the tropics than temperate regions; resulting in a strong gradient in environmental variability (Lueangthuwapranit *et al*. 2011). Phytoplankton communities exhibit nutrient limitations that are primarily dependent on the temporal and spatial distribution of dissolved nutrients (Trommer *et al*. 2013); with a tendency for freshwater systems to be phosphorus limited whereas estuarine and marine systems are more Nitrogen limited (Hecky and Kilham 1988). Nutrient-rich waters in Brazil have been associated with diatoms and dinoflagelletes. Phytoplankton in oligotrophic tropical waters is mainly composed of small nanoflagelletes and cyanobacteria(Goncalves-Araujo *et al*. 2012). Seasonal succession of phytoplankton communities has been shown to be affected significantly by total phosphorus in lakes (Pereira *et al*., 2011). The impacts of nutrient changes on phytoplankton communities are expected to be higher with global warming (Cermeno*etal*., 2013).

## 5.3 Macroinvertebrates and Effects of Water Quality and Seasonality

Some macroinvertebrates were found in low abundance like *Aeshna, Gramarus, Haliplus, Hydrometra, Hyponeura, Physa, Promoresia* and *Tipula* in Kesses dam, *Chironomus, Pantala, Pisisdium, Psephenus, Simulium* and *Sphaerium* in Kerita dam, *Anodonta, Caenis, Coenagrion, Dytiscus, Ephemera* in Mauna and *Nobus, Notonectus* and *Platambus* in Yenga dam. These species might have been affected by the difference in water quality in these dams and their low abundance could be an indicator of their sensitivity to changes in water quality. Occurrence of Prosobranchiata, Diptera, Hirudnea, Plecoptera, Coleoptera and Lamelliobranchiata in Kesses, Kerita and Yenga is an indication that dissolved oxygen levels are adequate. Their absence in Mauna, however indicates a different scenario whereby low dissolved oxygen levels as resulting from high organic loads cause a shift with less tolerant organisms being replaced by more tolerant ones as earlier observed by Ojunga *et al*. (2010) and Wynes and Wissing (1981).

Abundance of Amphipoda and Lamelliobranchiata in Mauna was associated with positive loadings of total phosphates and temperature. Low abundance of EPT taxa in Mauna is attributed to habitat and water quality deteriotation due to sedimentation, and cultivation on banks which are evident on site. Human activities including washing, bathing, animal watering and collection of domestic water are common at the site. Yenga dam did not appear to be affected by total phosphates and temperature. However, high phosphate levels in Mauna is attributed to animal manure deposits, chemical fertilizers and sediments washed down into the water by surface runoff; which lies in a low lying basin and is surrounded by farmland. On the other hand, Yenga is on a fairly flat terrain; with hardly any farm activities around it.

In Lake Victoria, the presence of molluscs is attributed to poor environmental conditions leading to presence of chironomids, hirudnea and soft bodied, non-insect individuals such as oligochates and planarians which are more tolerant (Mwambungu *et al.,* 2005). Studies by Lance *et al*. (2008) indicating that the type of food they ingested affects abundance of macroinvertebrates. Macroinvertebrates assemblages have been shown to be sensitive to human activities such as forest degradation (Schilthuizen *et al.,* 2005; Kappes 2006) habitat destruction, water pollution, use of pesticides and collection of organisms for food (Ludwig *et al.*,2007).

Raburu (2003) attributes abundance and richness of macro invertebrates to riparian vegetation cover and habitat quality. The fact that Plecoptera was in high abundance in Kesses, Kerita and Yenga is an indication of high dissolved oxygen levels. According to Klemm *et al*. (2003); Ephemeroptera- Plecoptera-Tricoptera (EPT) group are considered "clean" water forms of macroinvertebrates; with their abundance and richness declining in sites associated with water quality deterioration and habitat degradation. However; they respond differently to degradation; with sensitivity decreasing from Plecoptera to Trichoptera. Low abundance of Ephemeroptera is attributed to water quality deterioration and habitat degradation in these water bodies. Macro-invertebrates respond to water quality in different ways depending on their tolerance levels. A shift in environmental conditions can be detected by proportion of scrapers to filterers; which ideally represents the balance between food sources. An increase in relative abundance of filtering collectors indicates existence of organic wastes from animals deposited directly in the water; thus increasing availability of fine particulate matter. On the other hand, the relative abundance of gatherers is a useful measure of general degradation since these are genaralists that thrive in areas abundant with fine particulate organic matter (Masese *et al.,* 2009b).

The high abundance of Odonata and Ephemeroptera in Kerita dam in Uasin Gishu is linked to abiotic factors. Dragonflies (Insecta: Odonata) are sensitive to changes in environmental conditions and have been used for assessment of environmental change (Simaika and Samways 2011). High diversity of Odonata species is strongly correlated with good water quality containing low turbidity and high dissolved oxygen (Corbet 1999). The presence of predatory organisms such as fish, competitive interactions and availability of food could also have influenced abundance of Odonata (Fincke 1992; Braccia *et al.,* 1992). Emeregence of Ephemeroptera is usually limited by enviromental variables such as pH while Trichoptera is able to tolerate wider ranges of environmanetal conditions (Freitag 2004).

The effect of rainfall on macroinvertebrates was evident in all the water bodies in this study; resulting in higher abundance of Ephemeroptera, Pulmonata and Plecoptera during the rainy season in Uasin Gishu and of Pulmonata and Plecoptera during the dry season in Siaya. Although spatial patterns, local habitat characteristicts, geomorphology and water chemisrty are associated with macroinvertebrate assemblages, temporal variability has been found to have a greater impact (Marshall *et al*. 2005; Ortiz and Puig, 2007; Corbi and Strixino 2008 and Masese *et al.,* 2009b).

Rainfall is associated with seasonal changes in availability of cover and food and thus influences variation in abundance of macroinvertebrates (Ettinger-Epstein and Kingsford 2008). Some macroinvertebrates are able to survive changes in rainfall patterns as they have devoloped flexible physiological and lifecycle characteristics that enable them to survive fluctuations in hydrological, habitat and ecological conditions. The mechanisms they have adapted include using environmental cues for critical life history activities, drought refugia and life history strategies that resist dessication. Therefore, it is important that the effects of environmental conditions on species physiological responses be assessed prior to implementaion of environmental management procedures. The role of rainfall in influencing distribution of aquatic invertebrates has been shown by Kay *et al.* (2001) and Dunlop *et al*. (2008).

The fact that Uasin Gishu had a higher diversity of macroinvertebrates is further evidence that water quality had an effect on these organisms during this study. Environmental variables control beta-diversity patterns; or degree of dissimilarity of macroinvertebrates such as dragonflies (Odonata). High diversity of Odonata species is strongly correlated with good water quality containing low turbidity, moderate conductivity and high dissolved oxygen (Corbet 1999).

The biomass levels in this study were suitable for fish production and were within those found in studies by Kaggwa *et. al.*, (2011). Contrary to studies by Uwadiae and Ajose (2014) which indicate that primary productivity affects macroinvertebrate abundance, this was not reflected in this study. While macroinvertebrate biomass levels have been attributed to high nutriet levels (Orwa *et al.*, 2012; Ngodhe *et al.*, 2013), Kesses dam which recorded the highest net primary productivity had a much lower macroinvertebrate biomass than other dams. This could be as a result of agricultural activities (Ortiz and Puig, 2007; Corbi and Strixino 2008; Masese *et al*. 2009b). The effects of nutrients on macroinvertebrate densities have been found to be inconsistent, with high nutrients levels associated with both high increased and decreased macroinvertebrate densities (Ortiz and Puig, 2007; Morais and Lee, 2014).

Wallace *et al.* (1996) suggest that high food availability associated with the effects of toxic compounds may be the reason why high nutrient levels result in low macroinvertebrate biomass densities. The amount of food available has been an important factor associated with density and biomass of macroinvertebrates (Morante *et* *al.*, 2012). Food availability is associated with differences in larval supply, recruitment and settlement processes in wet and dry seasons and is also linked to changes in macroinvertebrate biomass density (Morais and Lee., 2014). In this case, low macroinvertebrate biomass density in Kesses can be linked to nutrient enrichment, causing decreased macroinvertebrate richness. According to Paul & Meyer (2001), this is attributed to elimination of sensitive taxa mostly represented by the orders Ephemeroptera, Plecoptera and Trichoptera (EPT). Furthermore, Kesses dam has also been a beneficiary of restocking exercise under the ministry of fisheries aquaculture stumulus programme during which it has had over 100,000 fish added to the already existing ones. This may have increased the predator pressure on macroinvertebrates in the dam, a premise that is supported by the findings of Braccia *et al*. (1992) and Fincke (1992).

## 5.4 Fish Parasites and Effects of Water Quality and Seasonality

High prevalence levels for *Tylodelphys* and *Clinostomum* in Uasin Gishu is associated with presence of birds that are linked to the life cycle of some of these parasites. Some of the common birds included the cattle egret, grey heron, pelicans and cormorants. These birds play an important role in transmission and life cycle of these parasites, hence the high prevalence levels. Piscivorous birds are definitive hosts for many metacercarial stage digenean parasites found in fish. Aquatic birds also contribute to the dispersal of aquatic snails which serve as intermediate hosts for many digenean parasites. Records from Paperna (1996) suggest that water bodies from the Jordan system throughout the

Nile to the Rift Valley lakes share common snails such *Lymnaea* and *Melanoides* which were evident in the study sites.

The water quality parameters influencing abundance of fish parasites in Uasin Gishu were pH, total nitrates and phosphorus in Uasin Gishu. Abundance was correlated positively with pH, negatively with TN and both positively and negatively depending on the type of parasite with TP in Uasin Gishu. In Siaya, temperature and total phosphates had the greatest influence where temperature; with parasite abundance correlated positively with temperature and negatively with TP. According to Khidr *et al.,* (2012), high temperature favours short induction periods, rapid growth and high egg production while high tolerance levels to pH are developed through acclimatization. Similarly El-Naggar *et al.* (1986) found temperature was the main fator affecting seasonal variation in prevalence and mean intensity of cichlidogyrids in tilapia. The role of temperature in the dynamics of parasite communities has been emphasized (Ernst *et al.,* 2005) and has been cited as one of the most important abiotic factors determining the seasonal dynamics of parasites of fish (Khidr *et al*.,, 2012 ).

Fish that were infected with *Tylodelphys, Diplostomum* and *Amirthalingamia* during the dry season and with *Contracaecum* during the wet season had a higher condition factor than un-infected ones in Uasin Gishu. In Siaya, fish infected with *Amirthalingamia* during the dry season were found to have higher condition factor than those without. Contrary to this, studies on farmed tilapia in Kisumu, Kenya have shown that the presence of *Diplostomum* parasites had no effect on condition factor of fish (Ndeda 2013). This difference is linked to the fact that since fish in culture environments are held under controlled conditions, they are not affected as much as those in the wild by factors such as seasonality of water quality parameters, food availability and presence of intermediate and definitive hosts. The importance of life cycle patterns including availability and infectivity of intermediate and definitive hosts in variation in prevalence rates of parasites has been emphasized by Khurshid and Ahmad (2012) who argue that transmission success is a key determinant of parasite fitness in any host-parasite system, especially in complex life cycles.

High infection rates during the rainy season are associated with large inputs of domestic sewage and pollutants in flood water (Khanum *et al.,* 2011). In this study, parasitic infection was associated with condition factor and season but not to sex of fish. In Uasin Gishu, both infected male and female fish had higher condition factors during the dry season than their un-infected counterparts while there was no difference during the rainy season. However, it was during the rainy season that infected males and females had higher condition factors than un-infected ones in Siaya. The reason has to do with size; whereby large fish have been found to have more helminth parasites than small ones. This is supported by Steinauer and Font (2003) who found larger fish to harbour more helminth species and more individuals of each species than smaller fish and suggests that larger and presumably older fish have a longer period of exposure to trematodes. They therefore ingest more prey items than smaller fish thus increasing their probabability of exposure to intermediate hosts of these parasites. Similar findings by Khidr *et al.*, (2012) and Khanum *et al.* (2011) show that fish with higher body weight tend to have more parasites.

Similarly, Akoll *et al*. (2012) have shown that size and not sex of fish affects helminth infection rates, an observation explained by accumulation and prolonged exposure in larger and older fish. In helminth parasites, age of fish hosts influences the intensity of parasitism as result of repeated seasonal exposure to cercariae and to the lifespan of metacercariae; with cases where older fish harboring fewer parasites being attributed to recent colonization of species that preferentially invade young fry (Chappell *et al*., 1994).

High prevalence of helminths in the dry season is linked to behaviour of fish in relation to habitat types and disruption of parasites transmission by water currents and depth respectively. For this reason, cages are considered to be safer for rearing fish in deep reservoirs as parasites are in higher numbers in the bottom. In the helminth parasite *Camallanus oxycephalus*, distinct seasonal patterns in prevalence and abundance during summer is attributed to release of juveniles in spring, the increase of copepod intermediate hosts and increased fish foraging in the summer (Steinauer and Font, 2003). On the other hand, decline of prevalence and abundance in fall has been asociated with death of infected hosts or competitive interarctions among individual helminths.

According to Akoll et. al. (2012), non-seasonal fluctuations in parasites could be due to highly variable nature of the water body; with frequent, unpredictable changes in the physical conditions causing unstable populations that fluctuate unpredictably regardless of seasonal effects. They further explanied that in cases where differences in infection rates occur between male and female tilapia (*Oreochromis niloticus*), they are attributed to differential exposure to parasites whereby females are confined to at least 1-2 weeks while incubating eggs and protecting fry; exposing them to infections.

Similar findings where body size but not sex of fish has an effect on prevalence and mean abundance of parasites were established by Vincent and Font(2003). They attribute this to differences in habitat prevalences of copepod intermediate hosts; with larger, adult fish generally shifting away from the shallow to deeper water that has higher populations of copepods that serve as intermediate hosts for these parasites

A different observation has been made by Ibiwoye *et al*. (2004) whereby parasitic incidences differ between both seasons and sex of fish. Parasite- mediated sexual selection has been identified as a factor in gender variation in parasitic infections of fish (Batra 1984; Takemoto and Paravelli, 2000; Reichem and Nosil, 2001; Ibiwoye *et al.*. 2004; Maan *et al.*, 2006; Reimchen and Singhal and Gupta 2009 and Takemoto and Gupta *et al.*, 2012). High infectivity in females has been linked to attraction of cercariae and other parasites by positive stimuli in freshwater murrel (Singhal and Gupta 2009). The influence of the sex of fish on parasite dynamics is also seen in some cestode parasites whereby fish spawning and female host hormone levels provide the stimulus for parasite egg production (Kennedy and Hine 1969).

In Lake Victoria, females have been found to have tendencies to select against heavily parasitized males; leading to higher and more viable parasite loads in males. This is linked to ecological differences between males and females and is associated with a male specific trade-off between immune defence and reproductive investment (Maan *et al.* 2006). Ecological differences between genders and not reproductive cost, are more important in patterns of parasitic infection (Reimchen and Nosil, 2001). Their study on threespine stickleback *Gasterosteus aculeatus* indicate that differences in dietary niches for male and females influenced relative parasitism by cestodes. Excess consumption of cladocerans by females increased their relative exposure to pelagic copepods infected by *S. solidus* while in males consumption of chironomids increased their exposure to benthic oligochaetes infected by *C. trancatus.*

The amount of food consumed has been linked to higher helminth infections in male cichlids as a result of higher chances of consuming infected food items (Batra 1984). However, this was not the case in this study; an indication of lack of intraspecific competition for resources and also shows that there were no micro-spatial niche-variations. This is supported by the fact that there were no significant differences between stations at each site with respect to macroinvertebrate organisms which serve as major food items for fish.

Variation in helminth parasites has been found to be positively correlated with water temperature (Khurshid and Ahmad 2012). Due to their sensitivity, helminths can serve as stress indicators in SWBs utilized for culture based fisheries. Seasonal cycles in water quality characteristics such as temperature influence parasitic indices such as prevalence rates in fish (Khurshid and Ahmad 2012; Puinyabati *et al.,* 2013. During the dry season, water shrinkage associated with rapid water quality deterioration and low water turn-over may result in environment-related host stress; increasing susceptibility to parasites (Akoll *et al*. 2012) and in nutrient imbalances that lead to less production of fish food organisms and higher consumption of parasitic organisms (Gupta *et al.,* 2012). Seasonal variation and not sex or size of fish host is the main reason for variation in dynamics of *Proteocephalus sagittus* in the stone loach *Barbatula barbatula* in the Czech Republic (Jarkovsky *et al*., 2004). Vincent and Font (2003) attribute seasonal fluctuations in helminth parasites of fish to flushing of infected hosts, intermediate hosts and free-living infective worm stages downstream by heavy rains.

# CHAPTER SIX

# CONCLUSION AND RECOMMENDATIONS

## 6.1 Conclusions

1. Dams in Uasin Gishu and Siaya Counties are different with respect to temperature which is attributed to geographical differences based on altitude. The two counties also differ with respect to TP, DO and TN. TP is higher in Siaya while DO and TN are higher in Uasin Gishu.

2. Seasonality of rainfall causes temporal variation in pH, DO, BOD, TP and TN between dry and wet seasons. In Uasin Gishu County, pH is higher during the wet season while TN and TP are higher in the dry season. In Siaya, DO and BOD are higher during the wet season.

3. Spatial-temporal variation in water quality is responsible for differences in composition and abundance of phytoplankton between Uasin Gishu and Siaya. In Uasin Gishu phytoplankton are influenced by temperature, pH and TN and by pH in county. Primary productivity was higher in Uasin Gishu than Siaya County.

4. Water quality influences macroinvertebrates communities. Low abundance of EPT in Mauna is an indication of water quality deterioration. Low abundance of EPT taxa in Mauna is attributed to habitat and water quality deteriotation due to

agricultural activities and high organic load. High abundance of Amphipoda and Lamelliobranchiata in Mauna was associated with high total phosphate levels and temperature.

5. Rainfall seasonality affects macroinvertebrate assemblages in Siaya and Uasin Gishu counties.

6. Primary and secondary production of SWBs in this study was between 4.2 - 7.44mg $O_2$ $day^{-1}$ and 0.24±0.121 – 0.47±0.178 g $m^{-2}$ and within suitable levels for fish production.

7. Variation in Temperature, pH, DO, BOD, TN, TP influences composition, abundance and biomass of macroinvertebrates in SWBs in Uasin Gishu and Siaya counties.

8. Variation in Temperature, pH, DO, BOD, TN, TP influences parasite abundance, prevalence and mean intensity in *O niloticus* in SWBs in Siaya and Uasin Gishu.

## 6.2 Recommendations

For enhancement of fisheries production in SWBs, the following are recommended:

1. Regular water quality monitoring should be conducted. Where water levels are affected significantly during the dry season, rapid water quality determination should be conducted to reduce environment-related host stress so as to minimize susceptibility to parasites
2. To minimize adverse conditions associated with seasonal fluctuations in water levels, withdrawal of water from the water bodies should be guided and regulated to maintain certain established minimum levels.
3. Enhancement of fish production in SWBs should be integrated with water quality, land use activities, topography, geographical location and other reservoir operations.
4. Enhancement of fisheries production in SWBs should take into consideration the role of vectors such snails found in the aquatic environment in transmission of both human and fish diseases.
5. Creation of awareness on fish health to farmers, researchers, service providers and fish consumers for disease management in commercial aquaculture production.
6. To prevent the risk of transmission of fish parasites to consumers, there is need to create awareness on public health concerns associated with fish parasites such as *Contracaecum* and *Clinostomum.*

## REFERENCES


Addy K., &. Green L. (1997). *Dissolved oxygen and temperature.* University of Rhode Islands. College of Resource Development. Cooperative extension. Natural Resources facts. Fact sheets No. 96-3.4pp.

Akoll, P., Fioravanti, M., Konecny, R., & Schiemer, F. (2012). Infection dynamics of *Cichlidogyrus tilapiae* and *C. sclerosus* (Monogenea Ancyrocephalinae) in Nile tilapia (*Oreochromis niloticus L.*) from Uganda. *Journal of Helminthology, 86*(3), 302-310.

Allison, H. E. (2011). *Aquaculture, fisheries, poverty and food security.* The WorldFish centre. 62pp.

Al-Gahwari, Y. A. (2003). *Use of phytoplankton and species diversity for monitoring coastal water quality* . M.Sc. Thesis, Universiti Sains Malaysia.

Aloo, P. A. (2000). Health problems associated with consumption of fish and the role of aquatic environments in the transmission of human diseases. *African Journal of health Sciences, 7*(3-4), 107-113.

Aloo, P. A. (2002). A comparative study of helminth parasites from the fish Tilapia zilli and *Oreochromis leucostictus* in lake Naivasha and Oloiden Bay, Kenya. *Journal of Helminthology, 76*, 95-102.

APHA. (1998). *Standard methods for the examination of water and wastewater* (20th ed.). American Public Health Association, Washington D.C.

APHA. (2000). *Standard methods for the analysis of water and waste water* (15th ed.). American Public Health Association, Washington D.C.

APHA. (2003). *Standard methods for the examination of water and waste water for the examination of water and waste water "Part 5210B: %-day Biochemical Oxygen Demand.* (L. S. Clesceri, & A. Rice, Eds.) American Public Health Association, Washington D.C.

Ayroza, D. M., G, N. M., Ayroza L M D S, Carvalho E D, Ferraudo A S, & Camargo A F M. (2013). Temporal and Spatial Variability of limnological characteristics in areas under the influence of Tilapia cages in the Chavantes reservoir, Paranepanema River, Brazil. *Journal of the World Aquaculture Society, 44*(6), 814-825.

Batra, V. (1984). Prevalence of helminth parasites in three species of cichlids from a man-made lake in Zambia. *Zoological Journal of the Linnean Society, 82*(3), 319-333.

Baker, S. C. & Sharp, H.E. (1998). Evaluation of the recovery of a polluted urban stream using Ephemeroptera-Plecoptera-Tricoptera Index. *Journal of Freshwater ecology*, 13, 229-234.

Bell, G., & Burt, A. (1991). The comparative biology of parasite species diversity: internal helminths of freshwater fish. *Journal of animal ecology, 60*(3), 1047-1063.

Benke, A. C., & Huryn, A. D. (2006). Secondary production of macroinvertebrates. In F. R. Hauer, & G. A. Lamberti, *Methods in stream ecology* (2 ed., pp. 691-710). San Diego: Academic Press.

Boyd, C. E. (1990). *Water quality in ponds for aquaculture*. Birmingham, Alabama: Birmingham Publishing Co.

Braccia, A. V., Reese , J., & Christman, V. D. (1992). The Odonata of newly constructed ponds with life history and production of dominant species. *Aquatic Insects*, 29 (2), 115-140.

Bristow, C. E., Morin , A., Hesslein, R., & Podemski, C. (2008). Phosphorus budget and productivity in an experimental lake during the initial three years of cage aquaculture. *Canadian Journal of Fisheries and Aquatic Science., 65*, 2485- 2495.

Brower, J. E., Zar, J. H., & Von, E. (1990). *Field and laboratory methods for general ecology*. Newy York, N. Y.: William C. Brown Publishers.

Bush, A. O., Lafferty, K., Lotz, J., & Shostak, A. (1997). Parasitology meets ecology on its own terms: Margolis *et al*. revisited. *Journal of Parasitology, 83*(4), 575-583.

Campos, C. M., Fonseca, V. E., Takemoto, R. M., & Moraes, F. R. (2009). Ecology of the parasitic endohelminth community of *Pseudoplatystoma fasciatum* (Linnaeus, 1776) (Siluriformes Pimelodidae) from the Aquidauana River, Pantanal, State of Mato Grosso du Sul, Brazil. *Braz. J. Biol, 69*(1), 93-99.

Cermeno, P., Moranon, E., & Romero, O. E. (2013). Response of marine diatom communities to late quartenary abrupt climate changes. Journal of Plankton Research, 35 (1), 12-21.

Chappell, L. H., Hardie , L., & Secombes , C. (1994). Diplostomiasis: the disease and host-parasite interactions. In A. W. Pike, & J. Lewis, *Parasitic diseases of fish* (pp. 59-86). Tresaith, Dyfed, Great Britain: Samara Publishing Ltd.

Chaves, F. I., Lima, P. F., Leitao, R. C., Paulino, W. C., & Santaella, S. T. (2013). Influence of rainfall on the trpohic status of a Brazilian semiarid reservoir. *Acta Scientiarum, 35*(4), 505-511.

Chust, G., Allen, J. I., Bopp, L., Schrum, C., Holt, J., Tsiaras, K., Zavatarelli, M., Chifflet, M., Cannaby, H., Dadou, I. O., Daewal, U., Wakelin, S. I., Machui, E., Pushpadaas, D., Butenschon, M., Artioli, Y., Petihakis, G., Smith, C., Garconi, C., Goubanova, K., Vui, B. L., Fach, B. A., Salihoglu, B., Clementis, E., Irigoieni, E, & Le Vu, B. (2014). Biomass changes and trophic amplification of plankton in a warmer ocean. *Global Change Biology, 20*, 2124-2139.

Corbet, P. S. (1999). *Dragonflies: Behaviour and ecology of Odonata.* Ithaca, New York: Coomstock Publishing Associates.

Corbi, J. J., & Strixino, S.T. (2008). Relationship between sugarcane cultivation and stream macroinvertebrate communities. *Braz. arch. biol. technol.*, 51 (4), 769-779.

Cowx, I. G. (1999). An appraisal of stocking strategies in the light of developing country constraints. *Fisheries management and ecology, 6*, 21 -34.

Cremona, F., Planas, D., & Lucotte, M. (2008). Biomass and composition of macroinvertebrate communities associated with different types of macrophyte architectures and habitats in a large fluvial lake. *Fundamental and Applied Limnology, 171*(2), 119-130.

D'Amico, F., Darblade, S., Avignon, S., Blanc-Manal, S., & Ormerod, S. J. (2004). Odonates as indicators of shallow lake respiration by liming: comparing adult and larval responses. *Restoration Ecology*, 12 (3), 439-446.

De Silva, S. S. (2003). Culture-based fisheries: an underutilized opportunity in aquaculture development. *Aquaculture , 221*, 221-243.

De Silva, S. S., Zhitang, Y., & Lin-Hu, X. (1991). A brief review of the status and practices of the reservoir fishery in mainland China. *Aquaculture and Fisheries Management, 7*, 513-521.

De Silva, S. S., Amarasinghe, U. S., & Nguyen, T. T. (2006). Better-practice approaches for culture-based fisheries development. *Australian Centre for International Agricultural Research, No. 120,* p. 96p.

Diana, J, S., Szyper, P., Batterson, T. R., Boyd, C. E., & Piedrahita, R. H. (1997). Water quality in ponds. In H. E. Egna, & C. E. Boyd. *Dynamics of pond aquaculture.* Boca Raton. new York: CRC press. 472pp.

Dunlop, J. E., Horrigan , N., McGregor , G., Kefford, B., Choy, S., & Prasad, R. (2008). Effect of spatial variation on salinity tolerance of macroinvertebrates in Eastern Australia and implications for ecosystem protection trigger values. *Environmental pollution, 151*, 621-630.

Dzikowski, R., Diamanti, A., & Paperna, I. (2003). Trematode metacercariae of fishes as sentinels of changing limnological environment. *Diseases of aquatic organisms, 55*, 145-150.

El-Naggar, M. M., & Khidr, A. (1986). Population dynamics of some monogeneans from the gills of three Tilapia spp. from Demieta Branch of the River Nile in Egypt. *Proc. Zool. Soc. A. R. Egypt, 12*, 275–286.

Elsheikha, H. M., & Elshazly , A. (2008). Host dependent variations in the seasonal prevalence and intensity of heterophyid encysted metacercariae (Digenea: Heterophyidea) in brackish water fish in Egypt. *Veterinary parasitology, 153*, 65-72.

Erina, N. (2010). How to fulll engage youth in enhancing fisheries and aquaculture in Sb-Saharan Africa. *the International Conference on Fisheries and aquaculture Development for Socio-economic growth in Malawi.*,). Jinja, Uganda. 7pp.

Erina, N. (2010). How to fully engage youth in enhancing fisheries and aquaculture in Sub-Saharan Africa. National Fisheries Resources Research Institute (NAFIRI), Jinja, Uganda.

Ernst, I. I., Whittington, D., Corneilite, S., & Talbot, C. (2005). Effects of temperature, salinity, desication and chemical treatments on egg embryonation and hatching success of *Benedenia seriolae* (Monogenea: Capsalidae), a parasite of farmed Seriola spp. *J. Fish Dis., 28*, 157–164.

Elsheikha, H. M., & Elshazly , A. (2008). Host dependent variations in the seasonal prevalence and intensity of heterophyid encysted metacercariae (Digenea: Heterophyidea) in brackish water fish in Egypt. *Veterinary parasitology, 153*, 65-72.

Escalera-Vazquez, L., & Zambrano, L. H. (2010). The effect of seasonal variation in abiotic factors on fish community structure in temporary and permanent pools in a tropical wetland. *Freshwater Biology , 55*(12), 2557-2569.

Ettinger-Epstein, P., & Kingsford, M. J. (2008). Effects of the El-Nino Southern oscillation on *Turbo Torquatus* (Gastropoda) and their kelp habitat. *Austral Ecology, 33*, 594-606.

Faltynkova, A., Karvonen, A., Jyrkka, M., & Vantonen, T. (2009). Being succesful in a world of narrow opportunities: Transmission patterns of the trematode *Ichthyocotylurus pileatus*. *Parasitology, 136*, 1375-1382.

FAO. (1994). *Small water bodies and their fisheries in South Africa.* Rome: FAO.

Field, J. S., & Irwin, S. (1994). The epidermiology, treatment and control of diplostomiasis on a fish farm in Northern Ireland. In A. W. Pike, & J. Lewis, *Parasitic diseases of fish* (pp. 87-100). Tresaith, Dyfed, Great Britain.

Fincke, O. M. (1992). Interspecific competition for trematodes: Consequences for mating systems and coexistence in Neotropical damselflies. *American Naturalist, 139* (1), 80-101.

Flores, V., & Baccala , N. (1998). Multivariate analyses in the taxonomy of two species of *Tylodelphys diesing*, 1850 (Trematoda: Diplostomidae) from the *Galaxias maculates* (Teleosteei; Galaxiidae). *Systematic Parasitology, 40*, 221-227.

Florio, D., Gustinelli , A., Caffara, M., Turci, F., Quaglio, F., Konecny, R., Nikowitz, T., Wathuta, E., Magana, A., Otachi, E.O., Matolla, G.K., Warugu, H. W., Liti, D., Mbaluka, R., Thiga, B., Munguti, J., Akoll, P., Mwanja, W., Asaminew, K., Tadeasse, Z., Fioravanti, M. (2009). Veterinary and public health aspects in Tilapia (*Oreoochromis niloticus*) aquaculture in Kenya, Uganda and Ethipia. *Ittiopatologia, 6*(1), 51-93.

Freitag, H. (2004). Composition and Longitudinal patterns of Aquatic insect emergence in small rivers of Palawan Island; the Philippines. *Internat. Rev. Hydrobiol., 4*, 375-391.

Fried, B., & Ponder , E. (2003). Effects of temperature on survival, infectivity and in vitro encystment of the cercariae of *Echinostoma caproni*. J*ournal of helminthology, 77*, 235-238.

Fried, B., LaTerra, R., & Kim, Y. (2002). Emergence of cercariae of *Echinostoma caproni* and *Schistosoma mansoni* from *Biomphalaria glabrata* under different laboratory conditions. *J. Helminthology 76*(4) 369-371.

Fujimoto, N., Ryuich, S., Suguira, N., & Inamori, Y. (1997). Nutient-limited growth of *Microcystis aerogunosa* and *pharmidium tenue* competion under various N:P supply ratios and temperatures. *Limnology and Oceanography, 2*, 250-256.

Garaway, C., & Lorenzen, K. (2001). Developing fisheries enhancement in small waterbodies: lessons from Lao PDR and Northeast Thailand. In S. S. De Silva (Ed.), *ACIAR Proceedings 98*, (pp. 227-234).

Goldman, C. R., & Horne, A. J. (1983). *Limnology* (international Student Edition ed.). Tokyo, Japan: McGraw-hill International Book Company.

Goldmman, J. C. (1977). Temperature effects on phytoplankton growth in continuous culture . *Limnology and Oceanography, 22*, 032-035.

Goncalves-Araujo R., De Souza M.S., Mendes C. R. B., Tavano V. M., Pollery R.C. and Garcia C.A. (2012). Brazil-Malvinas confluence: Effect of environmental variability on phytolankton community structure. *Journal of phytoplankton* research, 34 (5) 399-415.

Gonzalez, J. V., Macedo, M., Herrera, A., & Guerrero, S. (2009). Metazoan parasite community of blue sea catfish, *Sciades guatemalensis* (Ariidae) from Tres Palos Lagoon, Guerrero, Mexico. *Parasitol Res, 105*, 997-1005.

Gupta, N., Singhal , P., & Gupta, D. (2012). Population dynamics of a parasite *Pallisentis* in two species of fish *Channa punctatus* and *Channa striatus* . *Journal of environmental Biology, 33*, 195-199.

Hamisi, M. I., & Mamboya, F. A. (2014). Nutrient and Phytoplankton dynamics along the ocean road sewage discharge channel, Dar es Sallam, Tanzania. *Journal of Ecosystems*. Volume 2014: 1-8 Retrieved 20th, February 2014 from www.aginternetwork.org/

Harper, D. M. (1992). *Eutrophicatiion of freshwaters. Principles, problems and restoration.* Prentice Hall.

Harris, G. P. (1986). *Phytoplankton ecology: Structure, function and fluctuation.* London, UK.: Chapman and Hall.

Hassel, K. L., Kefford , B., & Nugegoda. (2006). Sub-lethal and chronic salinity tolerances of three freshwater insects: *Cloen sp,* and *Centroptilum sp*. (Ephemeroptera: Baetidae) ad *Chiromonus sp*. (Diptera: Chironomidae). *The Journal of experimental biology, 209*, 4024-4032.

Heatherly, T., Whiles, M. R., Royer, T. V., & David, M. B. (2007). Relationships between water quality, habitat quality, and macroinvertebrate assemblages in Illinois streams. *Journal of Environmental Quality, 36*, 1653-1660.

Hecky, R. E. (1993). The eutrophication of lake Victoria. *Verhandlungen der Interantionale Vereinigung fur Theoretische und Angewandte Limnologie , 25*, 39-48.

Hecky, R., & Kilham, P. (1988). Nutrient limitation of phytoplankton. *Limnol. Oceanogr, 33*, 796-822.

Hong, S. J., Chung, C., Lee, D., & Woo, H. (1996). One human case of natural infection by *Heterophyopsis continua* and three other species of intestinal trematodes . *Korean Journal of Parasitology , 34*(1), 87-89.

http://www.britannica.com/. (2013). *Encyclopedia Britannica*., from http://www.britannia.com/ EB checked /topic 1627661/Lake-Victoria.

Huszar, J., & Marino, R. (2000). Cyanoprokaryote assemblages in eight productive tropical Brazilian waters. *Hydrobiol, 424*, 67-77.

Ibiwoye, T. I., Balogun, A., Ogunsusi, R., & Agbontale, J. (2004). Determination of infection densities of mudfish Eustrongylides in *Clarias gariepinus* and *C. anguillaris* from Bida floodplains of Nigeria. *Journal of Applied Sciences and Environmental Management , 8*(2), 39-44.

Jarkovsky, J., Koubkova, B., Scholz, T., Prokes, M., & Barus, V. (2004). Seasonal dynamics of *Proteocephalus sagittus* in the stone loach *Barbatula barbatula* from the Hana River, Czech Republic. *Journal of Helminthology, 78*, 225-229.

Johnson, M. G., & Brinkhurst, R. O. (1971). Production of benthic invertebrates of Bay of Quinte and Lake Ontario. *J. Fish. Res. Board of Can. , 28*, 1699-1714.

Jones, C., Palmer, R. M., Matkaluk, S., & Walters, M. (2002). Watershed health monitoring: Emerging technologies for the paucity of shredders in the tropics. *Freshwater Biology, 47*, 909-919.

Juen, L., & De Marco, P. J. (2011). Odonate Biodiversity in Terra-firme forests streamlets in Central Amazonia: on the relative effects of neutral and niche drivers at small geographical extents. *Insect Conservation and Diversity, 4*, 265-274.

Kaggwa, M. N., Liti, D. M., & Schagerl, M. (2011). Small tropical reservoirs and fish caage culture: a pilot study conducted in Machakos district, Kenya. *Aquaculture International , 19*, 839-853.

Kaggwa, R. C., van Dam, A. A., Kansiime, F., Balirwa, J. S., & Denn, P. (2009). Increasing fish production from wetlands at Lake Victoria, Uganda using organically manured seasonal wetland fish ponds. *Wetlands Ecol Manage, 17*, 257-277.

Kalff, J. (2002). *Limnology: inland water ecosystems.* New Jersey, Upper saddle River: Prentice Hall.

Kappes, H. (2006). Relations between forest management and slug assemblages (Gastropoda) of deciduos regrowth forests. *Forest Ecology and Management, 237*, 450-457.

Kay, W. R., Halse, S., Scanlon , M., & Smith, M. J. (2001). Distribution and environmental tolerances of aquatic macroinvertebrate families in the agricultural zone of Southwestern Australia. *Journal of the North American Benthological Society, 20* (2), 182-199.

Kennedy, M., & Hine, P. (1969). Population biology of the cestode *Proteocephalus torulosus* (Batsch) in dace *Leuciscus leuciscus* (L.) of the River Avon. *Journal of Fish Biology, 1*, 209-219.

Khalil, L., Jones, A., & Bray, R. (. (1994). *Keys to the cestode parasites of vertebrates.* Wallingford: CAB International.

Khanum, H., Begum, S., & Begum, A. (2011). Seasonal prevalence, intensity and organal distribution of helminth parasites in *Macrognathus aculeatus*. *Dhaka Univ. J. Biol. Sci., 20*(2), 117-122.

Khidr, A. A., Said, A. E., Samak, O. A., & Sheref, S. E. (2012 ). The impacts of ecological factors on prevalence, mean intensity and seasonal changes of the monogenean gill parasite, *Microcotyloides sp.*, infesting the Teraponputa fish inhabiting coastal region of Mediterranean Sea at Damietta region. *The Journal of Basic & Applied Zoology, 65*, 109–115.

Khurshid, I., & Ahmad, F. (2012). Gasro-intestinal helminth infection in fishes relative to season from shallabugh wetland . *International Journal of Recent Scientific Research, 3*(4), 270-272.

Kibichii, S., Shivoga, W. A., Muchiri, M., & Miller, S. N. (2007). Microinvertebrate assemblages along a land-use gradient in the upper River Njoro watershed of Lake Nakuru drainage basin, Kenya. *Lakes & Rservoirs: Research Management , 12*, 107-117.

King, P. H. (2007). Freshwater snails as intermediate hosts for trematode parasites in South Africa. *Parassitologia. ISFP VII. 7th International Symposium on fish parasites. 24th -28th September 2007.* Viterbo, Italy: University of Rome .

Kizza, M., Rodhe, A., Xu, C. Y., Ntale, H. K., & Halldin, S. (2009). Temporal rainfal variability in the Lake Victoria Basin in East Africa during the twentieth century. *Theoretical and Applied Climatology, 98*, 119-135.

Klemm, D. L., Blocksom , K. A., Fulk, K. A., Herlihy, A. T., Hughes, R. M., Kaufmann, P. R., *et al*. (2003). Development and evaluation of a macroinvertrbrate biotic integrity index (MBII) for regionally assessing mid-Atlantic highland streams. . *Environmental Management, 31*, 656-669.

Knaap, M. V. (1994). *Status of fish stocks and fisheries of thirteen medium-sized African reservoirs.* Rome : FAO.

Kotut, K. (1998). *Phytoplankton and nutrient dynamics at Turkwel Gorge Reservoir, a new man-made lake in northern Kenya.* Kenyatta Univerity, Kenya.

Koyombo, S., & Jorgensen, S. E. (2007). *Lake Victoria: Experience and lessons learned brief. International Lake Environment Committee, Lake Basin Management Initiative* . Retrieved July 25, 2014, from http://www.iwlearn.net.publications/ll/lakevictoris_2005.pdf.

Krienitz, I., Ballot, A., Wiegand, C., Kotut, K., Codd, C., & Phlugmacher, S. (2001). Cyanotoxin-producing bloom of *Anabaena* flos-aquae, *Anabeana discoidea* and *Microcystis aeruginosa* (Cyanobacteria) in Nyanza Gulf Lake Victoria, Kenya. *Journal of Applied Botany, 76*, 179-193.

Lance, E., Bugajny, E., Bormans , M., & Gerard, C. (2008). Consumption of toxic cyanobacteria by *Potamopyrgus antipodarum* (Gastropoda; Prosobranchia) and consequences on life traits and microcystin accumulation. *Harmful Algae 7*, 464-472.

Longshaw, M., Frear, P., Nunn, A., Cowx, I., & Feist, S. (2010). The influence of parasitism on fish population success. *Fisheries management and ecology, 17*, 426-434.

Lorenzen, K., & Garaway, C. J. (1998). How predictable is the outcome of stocking? *Inland Fisheries Enhancements. FAO Fisheries Technical Paper, No. 374*, pp. pp. 133-152.

Lueangthuwapranit, C., Sampantarak, U., & Wongsai, S. (2011). Distribution and abundance of phytoplankton: Influence of salinity and turbidity gradients in the Na Thap River, Songkhla Province, Thailand. *Journal of Coastal Research, 27* (3), 585-594.

Ludwig, C., Naegel, A., & Lopez-Roche, J. A. (2007). The effect of temperature on growth of the inter-tidal purple snail *Plicopurpura pansa* (Gould 1853) under laboratory conditions. *Aquaculture Research, 38*, 493-497.

Ludwig, J. A., & Reynolds, J. F. (1988). *Statistsical ecology: A primer in methods and computing.* New York: J. Wiley & sons .

M.Muchiri, S. (1990). *The feeding ecology of tilapia and the fishery of Lake Naivasha* . Leicester University, U.K.

Maan, M. E., Spoel, M., Jimenez, P., Alphen, J., & Seehausen, O. (2006). Fitness correlates of male correlation in a Lake Victoria cichlid fish. *Behavioural ecology, Advance access publication* (2), 692-699.

Magagula, C. N., Mansuetus, A. B., & Tetteh, J. O. (2010). Ecological health of the Usuthu and Mbuluzi rivers in Swaziland based on selected biological indicators. *African Journal of Aquatic Science, 35*(3), 283-289.

Maithya, J. (1998). A survey of the ichthyofauna of Lake Kanyaboli and other small water bodies in Kenya: alternative refugia for endangered fish species . *NAGA ICLARM Quarterly*, *3*, 54-56.

Maithya, J., Njiru, M., Owuor, J. B., & Gichuki, J. (2012). Some aspects of the biology and life-history strategies of *Oreochromis variabilis* (Boulenger 1906)in the Lake Victoria basin. *Lakes and Reservoirs: Research and Management, 17*, 65-72.

Maitland, P. S. (1990). *Biology of fresh waters* (Second Edition. ed.). USA: Blackie and sons Limited .

Maleri, M. (2009). Site selection and production performance of rainbow. *Aquaculture Research, 40: 18-25.*

Marshall, B., & Maes, M. (1995). *Small water bodies and their fisheries in southern Africa.* Rome, FAO: CIFA Technical Paper No. 29. 68p.

Marshall, J. C., Sheldon , F., Thoms, M., & Choy, S. (2005). The macroinvertebrate fauna of an Australian dryland river: spatial and temporal patterns and environmental relationships. *Marine and Freshwater Research, 57*(1), 61-74.

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 Science, 34*(1), 15-26.

Masese, F. O., Raburu, P. O., & Muchiri, M. (2009b). A preliminary benthic macroinvertebrate index of biotic integrity (B-IBI) for monitoring the Moiben River, Lake Victoria Basin, Kenya. *African Journal of Aquatic Science, 34*(1), 1-14.

Mathooko, J. M., & Mavuti, K. M. (1992). Composition and seasonality of benthic invertebrates and drift in the Naro Moru River, Kenya. *Hydrobiologia, 232*, 47-56.

McCallum, H., & Dobson , A. (1995). Detecting disease aaand parasite threats to endangered species and ecosystems. *TREE, 10*(5), 190-193.

Merritt, R. W., & Cummins, K. (1996). *An introduction to aquatic insects of North America* (3rd ed.). Dubuque, Iowa, USA: Kendall/Hunt Publishing Company. 862pp.

Ministry of Environment and Natural Resources . (2006). *Assessment of the potential of land suitability mapping with environmental overlays and potential usefulness of spatial planning for managing the Lake Victoria Basin.* Lake Victoria Environmental Management Project Phase II.

Ministry of Water. (2008). *Nine water basins.* United Republic of Tanzania.

Mkare, T. K., Manyala, J., Mulanda , C., & Mbaru, E. (2010). Relation between phytoplankton composition and abundance aand physico-chemical characteristics of Chepkanga dam, Eldoret, Kenya. *Lakes and Reservoirs: Research aand management, 15*, 111-118.

Morais, G. C., & Lee, J. T. (2014). Intertidal benthic macrofauna in the Amazon region of rare rocky fragments in the Amazon region. *Rev. Biol. Trop. (Int. J. Trop. Biol), 62*(1), 69-86.

Morante, T., Garcia-Arberas, L., Ant'on, A., & Rallo, A. (2012). Macroinvertebrate biomass estimates in Cantabrian streams and relationship with brown troit *(Salmo trutta*) populations. *Limnetica, 31*(1), 85-94.

Muchiri, S. M. (1990). *The feeding ecologgy of tilapia and the fisherey of Lake Naivasha.* PhD Thesis, University of Leicester, U.K.

Mugidde, R., Hecky, R. E., Hendzel, L. L., & Taylor, W. D. (2003). Pelargic nitrogen fixation in Lake Victoria (East Africa). *Journal of Great Lakes Research. 29* (Supplement 2), 76-83.

Mushi, V. E., Oenga, D. E., & Mwanja, W. W. (2005). *The increasing demand for fish through development of aquaculture in Lake Victoria and their management* . (pp. 150-158). Jinja: Lake Victoria Fisheries Organization Secretariat.

Mustapha, M. (2011). Perspectives in the limonology of shallow tropical African reservoirs in relation to their fish and fisheries. *The Journal of Transdisciplinary Environmental Studies, 10*(1), 16-23.

Mutua, F. M. (1980). *A rainfall runoff model for the River Nzoia* . M. Sc Thesis, University of Nairobi.

Mwambungu, J. A., Mhitu , H., & Chande , A. (2005). Benthic Macroinvertebrates of the Tanzanian side of Lake Victoria and their role in fish production. In N. C. M, F. Orach-Meza, & J. Wamuongo, *Knowledge and experiences gained from managing Lake Victoria Ecoosystem* (pp. 369-378). Lake Victoria Environment Management Project.

Mwaura, F. (2006). Some aspects of water quality characteristics in small tropical man-made reservoirs in Kenya. *African Journal of Scienec and Technology* (No. 1), 82-96.

Mwita, C., & Nkwengulila, G. (2008). Determinants of the parasite community of clariid fished from Lake Victoria, Tanzania. *Journal of Helminthology, 83*, 7-16.

Nalepa, F. T. (1989). *Estimates of macroinvertebrate biomass in Lake Michigan.* from Agencies amd staff of the U.S. Department of commerce Paper 388: http://digitalcommons.unl.edu/usdeptcommerce/388 .

Ndeda, V. M. (2013). Effect of *Diplostomum* species on length-weight relationship of farmed Nile tilapia in Kibos are, Kisumu city, Kenya. *Fisheries and Aquaculture Journal, 60*, 1-7.

Neustupa, J., Cerna, K., & St'astny, J. (2011). The effects of a periodic dessication on the diversity of benthic desmid assemblages in a lowland peat bog. *Biodivers Conserv, 20*, 1695-1711.

Ngodhe, O. (2012). *Assessment of ecological suitability of small water bodies for increased fish production through stocing and cage culture in Lake Victoria basin, Kenya.* M.Sc Thesis. Moi University, Fisheries and Aquatic Sciences. Eldoret.

Ngodhe, S. O., Raburu, P. O., & Achieng, A. (2014). The impact of water quality on species diversity and richness of macroinvertebrates in small water bodies in Lake

Victoria Basin, Kenya. *Journal of Ecology and the Natural Environment, 6*(1), 32-41.

Ngodhe, S. O., Raburu, P., Kasisi, G. M., & Orwa, P. O. (2013). Assessment of water quality, macroinvertebrate biomass and primary productivity of small water bodies for increased fish production in the lake Victoria basin, Kenya. *Lakes and Reservoirs: Research and Management, 18*, 89-97.

Nguyen, H. S., Bui, A. T., Nguyen, D. Q., Truong, D. Q., Le, L. T., Abery, N. W., & de Silva, S. S. (2005). Culture-based fisheries in small reservoirs in northern Vietnam: effect of stocking density and species combinations. *Aquaculture Research, 36*, 1037-1048.

Nguyen, T. L., Nguyen, T., Murrel, K. D., Johansen, M., Dalsgaard, A., Thu, L. T., Chi, T. T., Thansborg, S. M. (2009). Animal resrvoir hosts and fish-borne zoonotic trematode infections on fish farms, Vietnam. *Emerging infectious diseases, 15*(4), 540-546.

Njiru, M., Okeyo-Owuor, J. B., Muchiri, M., & Cowx, I. G. (2004). Shift in feeding ecology of Nile Tilapia in Lake Victoria, Kenya. *African Journal of Ecology, 42*, 163-170.

Ochieng, V. O., Matolla, G. K., & Khyria, S. K. (2012). A study of *Clinostomum* affecting *Oreochromis niloticus* in small water bodies in Eldoret- Kenya. *International Journal of Scientific and Engineering Research, 3*(4), 1-6.

Oduor, S. O., & Schageri, M. (2007). Phytoplankton primary producitvity characteristics in response to photosynthetically active radiation in theree Kenya Rift Valley saline-alkaline lakes. *Journal of plankton reserach, 29*(12), 1041-1050.

Ogbeibu, A. E., Okaka, C. E., & Orihabor, B. J. (2014). Gastrointestinal helminth parasites community of fish species in a Niger delta tidal creek, Nigeria. *Journal of Ecosystems, Volume 2014, Article ID 246283*, 10.

Oglesby, P. T. (1977). Relationships of fish yield to take phytoplankton standing crop production and morphoedaphic factors. *Journal of Fisheries Research Board of Canada, 34*, 2271-2279.

Ojunga, S., Masese , F. O., Manyalla, J. O., Etiegni, L., Onkware , A. O., Senelwa, K., , Raburu, P. O., Balozi, B. K., Omutange , E. S. (2010). Impact of a Kraft pulp and paper mill effluent on phytoplankton and macroinvertebrates in River Nzoia, Kenya. *Water Qual. Res. J. Can.45* (2), 235-250.

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

Onyema, I. (2007). The phytoplankton composition, abundance and temporal variation of a polluted estuarine creek in Lagos, Nigeria. *Turkish Journal of Fisheries and Aquatic Sciences , 7*, 89-96.

Ortiz, J. D., & Puig, A. (2007). Point source effects on density, biomass and diversity of benthic macroinvertebrates in a mediterranean stream. *River Res. Applic., 23*, 155-170.

Orwa, P. O., Raburu, P., Njiru, J., & Okeyo-Owuor, J. B. (2012). Human infkluence on macroinvertebrate community structure within Nyando wetlands, Kenya. *Int. J. Aquat. Sci., 3*(2), 33-37.

Overstreet, B., Cribb , T., Kritsky, D., & Caira, J. (1998). Co-evolution collaboration for parasitic diseases. *Parasitology International, 47*(1), 97-100.

Pacini, N. (1994). *Coupling of land and water: phosphorus fluxes in the Upper Tana River Catchment, Kenya. Unpublished Doctor of Philosophy thesis, Leicester University.*

Paperna, I. (1996). *Parasites, infections and diseases of fish in East Africa. An update.* FAO, Rome.

Paperna, I., & Dzikowski, R. (2006). Digenea (Phylum Platyhelminthes). In P. T. Woo, *Fish diseases and disorders volume 1: Protozoan and metazoan infections. 2nd Edition* (p. 790 pp). Wallingford , Oxfordshire, UK.

Park, C. W., Kim, J., Joo, H., & Kim, J. (2009). A human case of *Clinostomum complanatum* infection in Korea. *Korean Journal of Parasitology*. *47*(4), 401-404.

Paul, M. J., & Meyer, J. L. (2001). Streams in the urban landscape. *Annual reviews of Ecology and Systemmatics, 92*, 333-365.

Pereira, H. C., Allott, N., Coxon, C., Naughton, O., Johnston, E., & Gill, L. (2011). Phytoplankton of turloughs (seasonal lakes). *Journal of Plankton Research, 33* (3), 385-403.

Pritchard, M., & Kruse, G. (1982). *The collection and preservation of animal parasites. 6th edition.* . Lincoln: University of Nebraska Press.

Puinyabati, H., Shomorendra, M., & Kar, D. (2013). Correlation of water's physico-chemical characteristics and trematode parasites. *Journal of Applied and Natural Science* (1), 190-193.

Raburu, P. O. (2003). Water quality and the status of aquatic macroinvertebrates and ichthyofauna in River Nyando, Kenya. Ph.D Thesis, Moi University, Kenya.

Raburu, P. O., Okeyo-Owuor, J. B., & Kwena, F. (2012). *Community Based Approach to the Management of Nyando Wetland, Lake Victoria Basin, Kenya.* Nairobi: KDC-VIRED-UNDP.

Rasmussen, J. B., & Kalff, J. (1987). Empirical models for zoobenthic biomass in lakes. *Can. J. Fish. Aquatic Sci.44*, 990-1001.

Reimchen, T. E., & Nosil, P. (2001). Ecological causes of sex-based parasitism in threespine stickleback. *Biological journal of the Linnean Society, 73*, 51-63.

Reynolds, C. S. (1993). *The Ecology of Freshwater Phytoplankton.* New York: Cambridge University Press.

Reynolds, C. S., Irish, A. E., & Elliot, J. A. (2001). The ecological basis for stimulating phytoplankton response to environmental change. *Ecolo. Model. 140*, 271-291.

Richards, C., Host, G. E., & Arthur, J. W. (1993). Identification of predominant environmental factoors structuring stream macroinvertebrate communities within a large agricultural catchment. *Freshwater ecology, 29*, 285-294.

Richardson, T. J., Booth, A. J., & Weyl, O. L. (2009). Rapid biological assesement of th fishery potential of Xonxa dam, near Queenstown, South Africa. *African Journal of Aquatic Science, 34*(1), 87-96.

Rogowski, D., & Stockwell, C. (2006). Parasites and salinity: costly tradeoffs in a threatened species. *Community Ecology 146(4)*, 615-622.

Roy, A., Leigh, D., Paul, M., & Wallace, J. (2001). Effects of changing land use on macroinvertebrate integrity. *Georgia water Resource Conference.* Athens: University of Georgia.

Schell, V. A., & Kerekes, J. J. (1989). Distribution, abundance and biomass of benthic macroinvertebrates relative to pH and nutrients in eight lakes of Nova Scotia, Canada. *Water, Air and Soil Pollution, 46*(1-4), 359-374.

Schilthuizen, M., Liew, T. S., & Elahan, B. B. (2005). Effects of Karst forest degradation on Pulmonata and Prosobranch land snail communities in Sabah, Malaysian Borneo. *Conservation Biology. 19* (3), 949-954.

Schindler, D. W., Fee E J, & Ruszczynski , T. (1987). Phosphorus input and its implications for phytoplankton standing crop and production in the Experimantal Lakes area and in similar lakes. *Journal of the Fisheries Resources Board of Canada, 35*, 190-196.

Schludermann, C., Konecny, R., Laimgruber, S., Lewis, J., Schiemer, F., Chovanec, A., & Sures , B. (2003). Fish macroparasites as indicators of heavy metal pollition in river sites in Austria. *Parasitology, 126*, S61-S69.

Shostak, A. W., Tompkins, J., & Dick , T. (1987). The identification and morphological variation of Diplostomum baeri bucculentum from two gull species, using metacercarial infections from least cisco from the Northwest Territories. *Canadian Journal of Zoology, 65*, 2287-2291.

Simaika, J. P., & Samways, M. J. (2011). Comparative assessment of indices of freshwater habitat conditions using different invertebrate taxon sets. *Ecological Indicators, 11*, 370-378.

Singhal, P., & Gupta, N. (2009). Genarchopsis infestation in relation to host length and sex in freshwater murrel, Channa. *Biospectra, 4*, 257-260.

Sohn, W. M. (2009). Fish-borne zoonotic trematode metacercariae in the Republic of Korea. *Korean Jornal of Parasitology, 47*, 103-113.

Souza, J., Barros, M. B., Barbosa, C. F., Hajnal, F., & Padisak, E. (2008). Role of atelomixis in replacement of phytoplankton assemblages in Dom Helvecio lake, South-East Brazil. *Hydrogiologia, 607*, 211-224.

Steinauer, M. L., & Font, W. (2003). Seasonal dynamics of the helminths of bluegill (Lepomis macrochirus) in a subtropical region. *Journal of Parasitology, 89*(2), 3324-328.

Swierk, D., & Szpakowska, B. (2011). The effect of environmental factors on micropollutants in small water bodies . *Ecological Chemistry Engineering , 18* (4), 545 - 569.

Takemoto, R. M., & Paravelli, G. (2000). Aspects of the ecology of proteocephalid cestodes parasites of Sorubim lima (Pimelodidae) of the upper Parana river, Brazil: I. structure and influence of host's size and sex. *Brazilian Journal of Biiology, 60*, 577-584.

Taub, F. B. (1996). *Lakes and reservoirs: Ecosystems of the world.* ( Series number 23). Amsterdam: Elsevier.

Templer, P., Findlay, S., & Wigand, C. (1998). Sediment Chemistry associated with native and non-native emergent macrophytes of a Hudson River marsh ecosystem. *Wetlands, 18*(1), 70-78.

Thieltges, D. W., & Rick, J. (2006). Effect of temperature on emergence, survival and infectivity of cercariae of the marine trematode *Renicola roscovitab* (Digenea: Renicolidae). *Diseases of aquatic organisms, 73*, 63-68.

Thoesen, J. (1994). *Sggested procedures for the detection and identification of certain finfish and shellfish pathogens. 4th edition Version 1.* (J. Thoesen, Ed.) Bethesda, Maryland:

Taub, F. B. (1996). *Lakes and reservoirs: Ecosystems of the world.* (Vol. Series number 23). Amsterdam: Elsevier. Fish health section. American Fisheries Society.

Thomas, S., Cecchi, P., Corbin, D., & Lemoalle, J. (2000). The different primary producers in a small African tropical reservoir during a drought: temporal changes and interactions. *Freshwat. Biol., 45*, 43-56.

Thu, N. D., Dalsgaard, A., Loan, L. T., & Murrell, K. D. (2007). Survey for zoonotic liver and intestinal trematode metacercariae in cultured and wild fish in An Giang province, Vietnam. *Korean Journal of parasitology, 45*(1), 45-54.

Tiewchaloern, S., Udomkijdecha, S., Suvouttho, S., Chunchamsri, K., & Waikagul, J. (1999). *Clinostomum* trematode from human eye. *Southeast Asian J. Trop. Med. Public Health 30*(2), 382-384.

Townsend, S. A., Przybylska, M., & Miloshisa, M. (2012). Phytoplankton composition and constraints to biomass in the middle reaches of an Autralian tropical river during base flow. *Marine and Freshwater Research, 63*, 48-59.

Trebisan, B. R., & Forsberg, G. V. (2007). Relationships among Nitrogen, Total Phosphorus, algal biomass and zooplankton density in the centrak Amazonian lakes. *Hydrobiologia, 586*, 356-365.

Trommer, G., Leynaert, A., Kleine, C., Naegelen, A., & Beker, B. (2013). Phytoplankton phosphorus limitation in a North Atlantic coastal ecosystem not predicted by nutrient load. *Journallof Plankton Research, 0* (0), 1-13.

www.tutiempo. *Climate -Kenya: historical weather records of world climate*. Retrieved August 2012, from http://tutiempo.net/en/climate/kenya/KE.html.

UNDP. (2014). www.undp.org/contenet/undp/en/home/mdgoverview.html#. Retrieved Thursday June 19th 2014,

Uwadiae, R. E., & Ajose, A. A. (2014). Does primary productivity affect benthic macroinvertebrate abundance and diversity in estuarine ecosystem? A case study in Lagos Lagoon, Nigeria. *International Journal of Marine Science, 4* (42), 1-11.

Vass, K. K., Shrivastava, N. P., Katiha, P. K., & Das, A. K. (2009). *Enhancing fishery production in small reservoir in India.* Technical Manual No. 1949., WorldFish Center, The WorldFish Center, Penang, Malaysia.

Vershuren, D., Johnson, T. C., Kling, H. J., Edgington, D. N., Leavitt, P. R., Brown, E. T., Talbot M. R. , & Hecky, R. E. (2002). *History and timing of human impact on Lake Victoria, East Africa.* London: The Royal Society.

Veverica, K., Liti, D., Were, E., & Bownam, J. (2001). Fish yields and economic benefits of tilapia/catfish polyculture in fertilized ponds receiving commericial feeds or pelleted agricultural by-products. In A. Gupta, K. McElwee, D. Burke, J. Burright, X. Cummings, & H. Egna, *Eighteenth Annual Technical Report. Pond Dynamics/Aquaculture CRSP.* Oregon State University. Corvallis, Oregon, Oregon.

Vincent, A. G., & Font , W. (2003). Host specificity and population structure of two exotic helminths, *Camallanus cotti (*nematoda) and *Bothriocephalus acheilognathi* (cestoda), parasitizing exotic fishes in Waianu stream, O'ahu , Hawai'i. *Journal of Parasitology, 89*(3), 540-544.

Wallace, J. B., & Webster, J. R. (1996). The role of macroinvertebrates in stream ecosystem function. *Annual Review of Entomology, 41*, 115-139.

Welch, E. B. (1980). *Ecological effects of wastewater.* Cambridge: Cambridge University Press.

Wetzel, G. W. (1983). *Limnology.* Winston, USA: CBS College Publishing.

Wetzel, R., & Likens, G. E. *Limnological analyses. Second edition* . New York: Springer- Verlag.

Wijenayake, W. M., Jayasinghe, U. A., Amarasinghe, U. S., Athula, J. A., Pushpalatha, K. B., & De Silva, S. S. (2005). Culture-based fisheries in non-perennial reservoirs. *Fisheries management and ecology, 12*, 249–258.

Wiwanitkit, V., Nithiuthai, V., Suwansaksri, J., Chongboonprasert, C., & Tangwattakanont, K. (2001). Survival of heterophyid metacercariae in uncooked Thai fish dishes. *Annals of Tropical Medicine and Parasitology 95* (7), 725-727.

World Bank. (2013). *Fish to 2030. Prospects for fisheries and aquaculture.* Washington DC: The World Bank.

*www.investkenya.com/uasingishu-county.* (2014). Retrieved July 27th 2014

Wynes, D. L., & Wissing, T. E. (1981). Effects of water quality on macroinvertebrate communities. *The Ohio Journal of Science, 81*(5-6), 259-267.

Yamaguti, S. (1971). *Synopsis of digenetic trematodes of vertebrates.* (Vol 1 ed.). Tokyo: Keigaku Publishing Co.

Yentsch, C. S. (1981). Vertical mixing, a constraint to primary production: An extension of the concept of an optimal mixing zone. In J. C. Nihoul, *Ecohydrodynamics* (pp. 67-78). Amsterdam, Netherlands: Elsevier Scientific Publishing.

# APPENDIX I.

Questionnaire for community data collection during preliminary survey of small water bodies in Uasin Gishu and Siaya

Date_______________________________

**NAME OF WATER BODY**

___________________________________________________

Division_________________________Location________________________

Latitude___________________ Longitude__________________________________

Approx. size of water body (acres)___________________

Permanent / seasonal______________________

Private / public____________________________

Current uses (make observations to identify any uses)

___________________________________________________________________________

___________________________________________________________________________

______________________________________________________________________

Land use activities_________________________________________________________________

___________________________________________________________________________

___________________________________________________________________________

___________________________________________________________________________

___________________________________________________________________________

________________________________________________________

1. **Respondents profile:**
    a. Male or female__________________
    b. Age:

       below 20____20 to 30yrs______31 to 40 yrs_____41 to 50 years_____

       above 50____
    c. Household size_______________________

d. Marital status: single _______married___________divorced_______separated_________

e. Education background

Primary_____ secondary _____ high school_____college_____ university_____

2. Main occupation of respondent________________________________

3. Water utilization activities by respondent

   a. ___________________________________

   b. ____________________________________

   c. _____________________________________

   d. ______________________________________

4. Water utilization activities by the others in the community?___________________________

   _______________________________________________________________

   _______________________________________________________________

5. Have you ever been heard about aquaculture? ______________________________________

6. If so, from where/ who?_________________________________________________________

7. Have you had any experience with fish farming:

   None at all_____________Some little experience____________________

   Moderate experience_________________A lot of experience___________________

8. If no experience or just a little, would you like to learn about fish farming?

   a. Yes

   b. No

   c. Not sure

9. If some or a lot of experience, what are the challenges you have experienced?

______________________________________________________________________

______________________________________________________________________

______________________________________________________________________

______________________________________________________________________

10. What do you think about introducing community aquaculture in the public water bodies in this region?______________________________________________________________

______________________________________________________________________

______________________________________________________________________

11. What issues do you consider to be hindrances to aquaculture in this region?

______________________________________________________________________

______________________________________________________________________

______________________________________________________________________

______________________________________________________________________

12. Do you think aquaculture would be of any benefit to this community?

______________________________________________________________________

If so, how?

13. If given an opportunity, would you consider fish farming as a source of food and income

______________________________________________________________________

Why?______________________

# APPENDIX II.

Abundance (A) and Relative abundance (RA) (%) of macroinvertebrates sampled in Kesses and Kerita (in Uasin Gishu) and Mauna and Yenga (in Siaya) during the period of November 2010 and July 2011.

| | Uasin-Gishu | | | | Siaya | | | |
|---|---|---|---|---|---|---|---|---|
| | Kesses | | Kerita | | Mauna | | Yenga | |
| | A | RA | A | RA | A | RA | A | RA |
| Aeshna | 2 | 0.0% | | | | | 7 | 0.3% |
| Agrion | 479 | 8.2% | 386 | 23.6% | 178 | 14.3% | 49 | 2.3% |
| Amphizoa | 158 | 2.7% | | | | | | |
| Anodonta | | | 19 | 1.2% | 1 | 0.1% | | |
| Baetis | 157 | 2.7% | 282 | 17.2% | 47 | 3.8% | 28 | 1.3% |
| Belostoma | 91 | 1.6% | 37 | 2.3% | | | | |
| Berosus | 31 | 0.5% | | | | | | |
| Bythinia | | | | | 68 | 5.4% | 26 | 1.2% |
| Caenis | | | | | 9 | 0.7% | | |
| Chironomous | 16 | 0.3% | 2 | 0.1% | 41 | 3.3% | 23 | 1.1% |
| Coenagrion | | | | 0.0% | 7 | 0.6% | | |
| Cordulegaster | 26 | 0.4% | 34 | 2.1% | 23 | 1.8% | | |
| Corixa | 79 | 1.4% | 271 | 16.5% | 23 | 1.8% | 19 | 0.9% |
| Dytiscus | | | | | 4 | 0.3% | | |
| Ephemera | | | | | 9 | 0.7% | 7 | 0.3% |
| Erpobdella | 2 | 0.0% | 2 | 0.1% | | | 8 | 0.4% |
| Gammarus | 1 | 0.0% | | | 141 | 11.3% | 1147 | 53.4% |
| Gomphus | 142 | 2.4% | 4 | 0.2% | 8 | 0.6% | 59 | 2.7% |
| Goniobasis | | | | | | | 31 | 1.4% |
| Gyrinus | 23 | 0.4% | 25 | 1.5% | 4 | 0.3% | 12 | 0.6% |
| Hagenius | | | | | | | 9 | 0.4% |
| Haliplus | 9 | 0.2% | 39 | 2.4% | 85 | 6.8% | 21 | 1.0% |
| Helmis | | | 6 | 0.4% | | 0.0% | | |
| Heptagenia | 118 | 2.0% | 23 | 1.4% | 19 | 1.5% | | |
| Hydrometra | 4 | 0.1% | | | 9 | 0.7% | 17 | 0.8% |
| Hydroporus | 11 | 0.2% | | | | | | |
| Hydropsyche | 79 | 1.4% | 19 | 1.2% | | | 15 | 0.7% |
| Hyponeura | 8 | 0.1% | 95 | 5.8% | | | 15 | 0.7% |
| Isogenus | 19 | 0.3% | | | | | | |
| Limnae | 2462 | 42.2% | 77 | 4.7% | 486 | 38.9% | 152 | 7.1% |
| Limnophora | | | 48 | 2.9% | | | | 0.0% |
| Nebus | | | | | | | 1 | 0.0% |
| Notonecta | | | | | | | 1 | 0.0% |
| Pantala | | | 1 | 0.1% | | | | |
| Perla | | | | | | | 9 | 0.4% |
| Physa | 7 | 0.1% | | | | | 7 | 0.3% |
| Pisidium | 11 | 0.2% | 1 | 0.1% | | | | |
| Planorbis | 127 | 2.2% | 11 | 0.7% | 33 | 2.6% | 241 | 11.2% |
| Platambus | | | | | | | 6 | 0.3% |
| Promoresia | 3 | 0.1% | 25 | 1.5% | | | | |
| Psephenus | | | 8 | 0.5% | | | | |
| Simulium | | | 8 | 0.5% | | | | |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Sphaerium | | | 2 | 0.1% | | | 32 | 1.5% |
| Theodoxus | 143 | 2.5% | | | 25 | 2.0% | 19 | 0.9% |
| Tipula | 1 | | 9 | 0.5% | | | | |
| Valvata | 1555 | 26.6% | 94 | 5.7% | 29 | 2.3% | 185 | 8.6% |
| Velia | 72 | 1.2% | 111 | 6.8% | | | | |
| | 5836 | | 1639 | | 1249 | | 2146 | |