Relationship Between Water Quality Indicators And Benthic Macroinvertebrate Assemblages In The Kuywa River, Kenya

Joash Nyakora Oruta
Hydrologist, Water Resources Management Authority, P.O Box 666 Kisumu

Boniface O. Oindo
Dean School of Environment and Earth Sciences, Maseno University, Private Bag Maseno,

Esnah K. Bosire
Senior Lecturer, Maseno University, Private bag maseno

ABSTRACT

The Kuywa River watershed has undergone riparian vegetation planting since 2006 in order to improve the river health. The planted riparian buffer zone vegetation was to improve channel stability, promote biodiversity, and improve water quality. Studies undertaken elsewhere have investigated how environmental factors affect ecosystem processes and functionalities but fail to show how water quality indicators influence the structure of the benthic macroinvertebrates. However, this study investigated the relationship between water quality indicators and benthic macroinvertebrate assemblages in the Kuywa River. Nine sites were assessed of different riparian vegetation zone conditions, consisting riparian assessment, benthic macroinvertebrate collection, and water quality sample collections for analysis for different parameters. PCA was performed on physico-chemical indicators and then BIO-ENV BEST to determine any significant relationship between physico-chemical and benthic macroinvertebrates. ANOSIM was employed to test the hypothesis for variations between the sampling sites in terms of physico-chemical parameters and riparian vegetation cover. Spearman rank correlation indicated that some water quality indicators area related to benthic macroinvertebrate assemblages while others did not respond. Elassoneuria($r=0.78$), Ephemerella ($r=0.75$), Synclita ($r=0.79$), Macrobdella ($r=0.90$) and Actnonaias ($r=0.91$) were highly correlated with dissolved oxygen ($\rho<0.05$). Hexatoma was highly correlated with percentage canopy cover ($r=0.83$, $\rho<0.01$) at the nine sampling sites. However, Actnonaias was also highly correlated with DO ($r=0.91$, $\rho<0.01$). Macrobdella was negatively correlated ($r=-0.85$, $\rho=0.002$) to Temperature(T) and Total Nitrogen (TN). In general, macroinvertebrate communities were dominated by two orders, Ephemeroptera (32.3%) and Diptera (53.2%). The study revealed that the diversity and evenness from the most disturbed habitat (A and K1) were much less than in the site from the less disturbed habitat (KS, E and T2). ANOSIM test indicated a significant difference in sites in terms of physico-chemical parameters ($R=392$, $p=0.1$) while macroinvertebrates were not significant. BIO-ENV BEST indicated discharge, sulphate and phosphate to have had greatest influence on benthic macroinvertebrate assemblages (BEST $p=0.515$). In this way, we rejected our null hypothesis that water quality indicators have no relationship with benthic macroinvertebrate assemblages in the Kuywa River. This knowledge is important to the community, government and water resource managers as they plan in investing resources to rehabilitate the catchments with a view of improving the health of equatorial rivers.

Keywords: Macroinvertebrates, Physico-chemical Indicators, River Health, Kuywa River

1. INTRODUCTION

Globally, studies have been carried out to establish the importance of riparian zone vegetation in determining geomorphology, structure of river valley floor landforms and riparian interactions (Gregory, Swanson, McKee, & Cummins, 1991). In USA, Naiman and Decamps (1997) established how riparian zone vegetation interacts with macroinvertebrates by observing the linkages between macroinvertebrates, vertebrates and microinvertebrates. However, much of the studies in USA are devoted to response of macroinvertebrates to urbanization (Chadwick, Thiele, Huryn, Benke, & Dobberfuhl, 2012; Paul & Meyer, 2001; Utz & Hilderbrand, 2011; Walsh et al., 2005). Other studies in Queensland Australia and New Zealand which deal with restored riparian vegetation observe the functions rendered by the macrophytes in processes such as denitrification and biomass changes (Fellows et al., 2006; Kennedy & Turner, 2011; Parkyn, Davies-Colley, Halliday, Costley, & Croker, 2003; Sheldon, Boulton, & Puckridge, 2002; Sheldon et al., 2012). In these studies, environmental factors are observed on how they affect the ecosystem processes and functionalities. However, the studies do not show how water quality indicators influence the structure of the benthic macroinvertebrates.

2. RESEARCH OBJECTIVE

To establish the relationship between water quality indicators (e.g. turbidity, pH, total nitrogen) and benthic macroinvertebrate assemblages in the Kuywa River

3. LITERATURE REVIEW

Aquatic macroinvertebrates have been identified as excellent indicators of river health due to their rapid response to environmental changes. They take part in important ecological processes, such as decomposition and nutrient cycling, and play a major role in food webs as both consumers and prey (Mendes, Calapez, Elias, Almeida, & Feio, 2014; Perera, Wattavidanage, & Nilakarawasam, 2012). However, physico-chemical factors have dominated studies which tend to assess the health of rivers, and more so in the equatorial countries. Moreover, physico-chemical factors are known also to affect the functionalities of benthic macroinvertebrates (M. Greenway, 2004; Sheldon, Boulton, & Puckridge, 2002). Physico-chemical factors known to affect benthic macroinvertebrates are those biophysical processes that may cause some organisms to increase in abundance, taxonomic richness and evenness while prohibiting others. Different benthic macroinvertebrate communities react differently to changes in biophysical environment and have a broad range of tolerance to stressors (Flinders, Horwitz, & Belton, 2008), while they are important in various food webs (USEPA, 2009). Thus, recent studies have shifted to combining physico-chemical and biological parameters in the studies for the river health.

In the last one decade, physical habitat structure and water quality have received much attention in studies as important elements of environmental quality and as agents structuring aquatic biotic assemblages (Ligeiro et al., 2013; Sály, Takács, I. Kiss, Bíró, & Erõs, 2011). Assessing and understanding the interactions among physical habitat features, water chemistry, and aquatic assemblages are essential to the conservation of headwater streams (Nerbonne & Vondracek, 2001; Pinto, Araujo, Rodrigues, & Hughes, 2009). At a stream point scale, physical complexity (e.g. structure cover, substrates, and water flow) influences assemblage composition, richness and temporal stability and ecological processes (Hughes, Herlihy, & Kaufmann, 2010; Kaufmann & M., 2012; Kovalenko, Thomaz, & Warfe, 2012). These processes are the core factors for structuring the aquatic ecosystems.

Many of the factors that contribute to diverse invertebrate communities may only be achieved over long time scales (Parkyn, Davies-Colley, Halliday, Costley, & Croker, 2003). Although some water quality variables (e.g., visual clarity) can recover quickly from fencing livestock out of streams, rehabilitation of shade and temperature may take decades, and the structure and habitat function of woody debris in streams may take centuries to develop (Reisinger, Blair, Rice, & Dodds, 2013). Inadequate decolonisation sources or pathways may limit invertebrate community rehabilitation, even when habitat is suitable (Newham, Fellows, & Sheldon, 2011; Parkyn et al., 2003; Sheldon et al., 2012). Rehabilitation of streams will be most successful when planting riparian zones begins from the headwaters down through the catchment and a continuous buffer length is achieved (Nyakora & Ngaira, 2014; Zhao, Mu, Tian, Jiao, & Wang, 2013). Rehabilitation process whether for the catchment or riparian zone require careful planning in order to realise the river health status anticipated.

The riparian conditions that lead to deterioration in stream water quality have been found to correlate negatively to macroinvertebrate assemblages. For instance, agricultural landscape with intensive grassland as the riparian vegetation has shown to correlate well with stream suspended solid concentrations (sediment supply), which in turn is negatively correlated to stream macroinvertebrate indices (Stutter, Langan, & Demars, 2007). This is attributed by the fact that when fine sediment supply from riparian and catchment exceeds in-stream transport and particle sorting capacity, the river-bed becomes clogged (Schalchli, 1992), thus affecting macroinvertebrate assemblages (Arthington, Naiman, Mcclain, & Nilsson, 2010; Larsen, Pace, & Ormerod, 2011). Riparian zone vegetation should be heterogeneous such that the sediments are trapped from reaching the stream at the same time there is enough leaf litter for the benthic macroinvertebrates to feed on.

Other physic-chemical parameters such as altitude and temperature are indirectly related to riparian vegetation and thus affect macroinvertebrate assemblages. Studies carried out at Moiben River in Kenya established a positive correlation between altitude and the abundance of macroinvertebrates (F. O. Masese, Muchiri, & Raburu, 2009). The same study indicated a negative correlation between taxon richness and water temperature, conductivity, depth, biological oxygen demand (BOD) and discharge, while taxon evenness was positively correlated with depth, width and discharge.

The importance of macroinvertebrate as an indicator of river health has been underscored (Growns, Rourke, & Gilligan, 2013; Mendes et al., 2014; Perera et al., 2012; Sheldon et al., 2012; Xu & Liu, 2014). Previous studies have indicated that factors that affect macroinvertebrate assemblage such as physical habitat structure, chemical properties and biological relationships (Flinders, Horwitz, & Belton, 2008; Ligeiro et al., 2013; F. O. Masese et al., 2009; Nerbonne & Vondracek, 2001; Nyakora & Ngaira, 2014; Parkyn et al., 2003; Pinto et al., 2009; Sály et al., 2011; USEPA, 2009; Zhao et al., 2013). In these studies, environmental factors are observed on how they affect the ecosystem processes and functionalities. However, the studies do not show how water quality indicators influence the structure of the macroinvertebrates.

This study aims to outline the relationship between water quality indicators, such as pH, Dissolved Oxygen (DO), Nitrites (NO2) and Nitrates (NO3) and aquatic benthic macroinvertebrate assemblages in the Kuywa River, Kenya. The objective is to increase knowledge of these equatorial rivers, as well as to provide evidence regarding the effect of physico-chemical variables on river health. The present study attempts to provide baseline data that will aid conservation and management efforts into these unique systems.

4. MATERIALS AND METHODS

The Kuywa catchment is bounded by latitude 034º 32` 53" E and 34º 45' 32" E and 0º 25' 24" N and 1º 50' 40" N. The entire river system is approximately 110km long. It originates from Mt. Elgon forest and discharges its waters into Nzoia River, a major river draining into Lake Victoria (Government-of-Kenya, 1984). The Kuywa River drains an estimated area of 580 square km2. The Kuywa River receives much of its runoff from the springs, which are perennial, and a stable ground water recharge as evidenced by the 13 boreholes and 150 hand-dug wells in the catchment (Water Resources Management Authority, 2011).

This study adopted longitudinal and cross-sectional descriptive design. Data was collected at the same sites quarterly over a period of one year beginning January, 2016 and ending October, 2016. This enabled the searchers to detect changes in the characteristics of the target population at both group and the individual levels, which led to establishment of sequence of events thus suggesting cause-and-effect relationship.

Table 1: The spatial distribution of the study sites (GPS, land use and physical characteristics) in Kuywa River and its major tributaries during the study period.

Station	LAT	LONG	ALT (M)	Landuse	Local watershed erosion	Category of vegetation
A	0.58395	34.6908	1440	Sugar cane plantation	Moderate	Sugar cane plantation
KG	0.73628	34.68845	1534	Agricultural, eucalyptus bank vegetation	Heavy	Planted eucalyptus
KS	0.75534	34.65931	1533	Agricultural	Heavy	Natural conserved
K1	0.75068	34.6403	1548	Agricultural	Heavy	Planted mature
K2	0.78208	34.6121	1574	Agricultural, open grazing	Heavy	Planted mature
T1	0.81716	34.58682	1956	Grazing	Heavy	Fenced and retired from grazing
E	0.82473	34.58234	1970	Agricultural	Moderate	Planted young
T2	0.8244	34.58379	1960	Agricultural	Moderate	Planted mature
KM	0.8873	34.58733	2304	Forest, open grazing	Moderate	Natural conserved

Nine sampling sites were objectively identified (Raburu, Masese, & Mulanda, 2009) in the Kuywa River basin, which represented a range of planted riparian vegetation buffer and are spatially separated to cover as much of the catchment as possible. Sites chosen were those rehabilitated or have been retired from grazing for at least two years. Each site that has been rehabilitated by re-vegetating or fencing off (to eliminate grazing) the buffer zone was compared with an un-vegetated or actively grazed riparian zone. Since the extent of the buffer zone may influence the stream properties, a 100m distance from the sampling site upstream were surveyed at each paired site and physical, biological and water quality parameters measured. Of these nine sampling sites, one site (KM) with near-natural riparian condition (Barbour & Stribling, 1993; Raven, Fox, Everard, Holmes, & Dawson, 1997) was chosen to be control or reference site, while another site (A) which was within a sugar cane plantation and riparian not conserved served as the second control site. The remaining seven sampling sites with a re-vegetated riparian buffer zone were test or study sites. Control sites provided reference standard to which the study sites were compared (Jungwirth, Muhar, & Schmutz, 2002). The sampling of these nine sites was carried out between August 2015 and July 2016. This ensured that both the rainy and dry periods were captured so as to investigate both the spatial and temporal effectiveness of the planted riparian buffer vegetation.

Riparian vegetation cover condition characterization

At each of the nine sites (planted, open and reference sites), the condition of the riparian zone vegetation was observed as per the critaria in Table 2. The percentage canopy cover was visually estimated and determined over 100m upstream (Masese et al., 2014; Raburu et al., 2009). According to SEPA (2003), Törnblom et al. (2011) and Lazdinis and Angelstam (2005) the width of 30m riparian vegetation condition has an effect on biology of the stream. Therefore, the riparian zone considered included a two-30m wide zones on either side of the stream over a 100 range along the stream.

Using riparian vegetation conditions, the indicators for stream health were assessed by adapting (Ladson & White, 1999) methods. The characteristics included were as follows: capacity to filter input, such as light, sediment, and nutrients, to streams; capacity to act as a source of input, such as woody debris and leaves, to streams; and capacity to provide a habitat for terrestrial animals. These characteristics, when broken down into detailed assessment, were found to be numerous, and thus necessitated detailed criteria, which reduced the characterization to four characteristics as shown in Table 2.

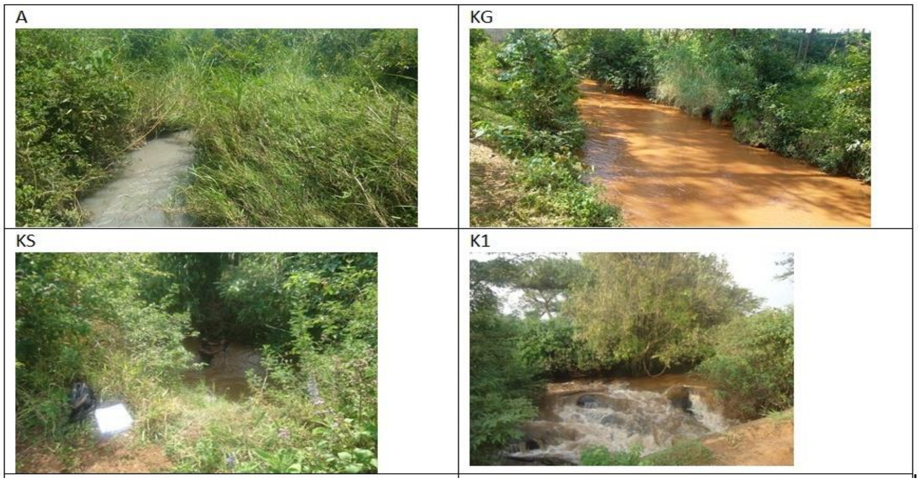
To develop a metric for the "riparian condition", a dimensionless rating was given for each indicator based on the proximity each indicator had to the reference condition; a value of 1 was given when the indicator was completely different to what would be expected under the reference condition, and a value of 4 when the indicator was the same (Table 2). The differences in riparian zone conditions for each site can be seen.

Table 2: Criteria for percentage riparian zone vegetation cover characterization:

Riparian classification and description of conditions	Rating
Excellent No exotic vegetation within 100 m of the riparian zone; natural vegetation intactness > 80%; width of the stream with vegetation > 40%; has more than 90% vegetated bank length within 100 m upstream on both sides	4
Good Exotic vegetation cover within 100m of riparian zone <30%; width of the streamside vegetation 25–40%; longitudinal continuity of indigenous vegetation within 100m upstream 65–80%; structural intactness of the riparian vegetation 60–80% at least on one bank	3
Fair Within 100m exotic vegetation cover 30–60%; width of streamside zone with vegetation 5–25%; longitudinal continuity of indigenous vegetation within 100m upstream 40–65%; and structural intactness of the riparian vegetation 40–60%	2
Poor Within 100m exotic vegetation cover >60%; width of streamside with vegetation <5% (may be characterized by collapsed river banks without vegetation); longitudinal continuity of indigenous vegetation <40%; structural intactness < 40%	1

Physico-chemical parameters

Stream water physico-chemical variables which include: temperature (T), conductivity (Cond), dissolved oxygen (DO), pH, turbidity, and total dissolved solids were measured in situ using a multiprobe water quality meter manufactured by Hydrolab company. Total suspended solids (TSS) was measured in the lab using filtration method (APHA, 1998). Sulphates were measuring in using a turbidity meter through turbidimetric method (applying balium chloride) (APHA, 1998). Phosphates were measured using spectrophometric method (Ascopic acid) while nitrates (NO3 -) and nitrites (NO2 -) were measured using cadium column reduction method then spectrophotometric technology (APHA, 1998). Total phosphates (TP) and total nitrates (TN) were measured in the laboratory using spectrophotometric method (persulphate) (APHA, 1998). Stream discharge (Q) was measured by wadding method using a FlowTracker Handheld Acoustic Doppler Velocimeter (ADV) manufactured by a YSI Environmental Company. The same ADV was used to measure average stream depth, width and average water velocity.



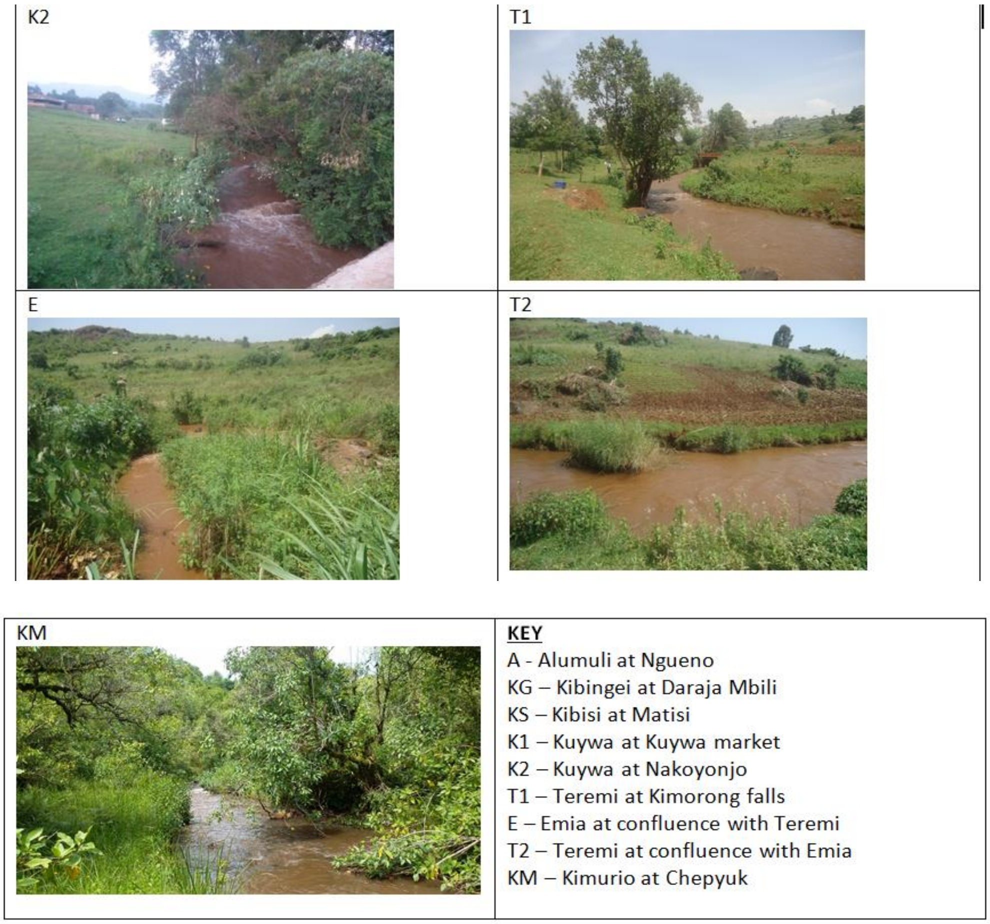


Figure 1: Riparian vegetation condition for the nine sampling sites in the Kuywa catchment. Site T1 and KM are the control sites

Benthic macroinvertebrate sampling

From each site, triplicate samples representing microhabitats (such as riffle, pool and run) were taken making a total of 27 samples at one sampling phase. Before sampling at the riffle and run, the debris was disturbed using kicks. The benthic macroinvertebrates were sampled using a standardized 250µm mesh dip net. The sampling distance was about 10m. Time taken for each sampling was 60 seconds to produce a representative sample. In the field, collected benthic macroinvertebrate samples were preserved in well labelled polythen bags with a 10% formalin solution and transported to the laboratory. The benthic macroinvertebrates were identified in the laboratory according to specific procedures (Acũna, Díez, Flores, Meleason, & Elosegi, 2013; AustralianGovernment, 2001; Mathooko, 1998).

In the laboratory, samples were washed using a 250µm sieve and sorted into well labelled plastic bottled containing 10% formalin solution. During identification, samples were displayed on a sorting tray and sorted under a stereo dissecting microscope and further preserved in the bottles containing 70% methylated spirit.

Samples were identified according to orders, families and genus using standard published and in-house taxonomic identification keys and guides for South Africa, and the abundance of each taxon recorded.

5. DATA ANALYSIS

To establish the relationship between water quality indicators and benthic macroinvertebrate assemblages in the Kuywa River, inferential statistics were employed. In inferential statistics Principal Component Analysis (PCA) was performed on physico-chemical parameters averaged for the four seasons and then tested with riparian vegetation condition as a factor. On macroinvertebrate data, transformation used was square root and log v. BIO-ENV BEST was performed using spearman rank correlations to determine any significant relationship between water quality and benthic macroinvertebrates. Further, one-way Analysis of Similarity (ANOSIM) was employed to test the hypothesis for variations between the sampling sites in terms of water quality parameters and measured riparian vegetation cover. On the other hand, Bubble plots of key water quality parameters were overlaid on top of the Multi-dimensional Scaling (MDS) plot to give an indication of which parameters were important in determining differences between sites.

6. FINDINGS

Physicochemical variables

Physico-chemical parameters varied considerably among the nine sampling sites. The altitude of the sites ranged between 1440m at site A and 2304m at site KM above sea level. The average stream depth at the sampling sites ranged between 0.15m (A) to 0.61m (KM), while the stream width ranged between 1.8m (A) and 7.02m (K2). Site K1 was found to have the highest discharge (3.09m3/s) followed by K2 (2.69m3/s) and the lowest being A (0.11m3/s). The mean values and SE of physicochemical variables are summarised in Table 3.

All nine sites were well oxygenated (>6.97mg/l)(Table 3) and within the neutral pH (7.3-8.4) for all of the sampling sites. In comparison with the reference site (KM), mean values which were lower included: at site A turbidity (57.4NTU), TSS (23.6), Sulphate (4.7), phosphate (0.01) and TP (0.078); at site T1 sulphate (5.1), phosphate (0.04), nitrate (0.04) and TP (0.08); at site E sulphate (3.1), phosphate (0.03); at site T2 sulphate (2.8), nitrate (0.43); at site K1 nitrate (0.07); and at site K2 nitrite (0.32). All the study sites had higher values of TN compared to the reference site (KM). Furthermore, percentage canopy cover at site A, KG and K1 (50,60, 80 respectfully) were equal or greater than the reference site (KM)(Table 3).

Table 4 indicate the values of t-test for the eight sites. As compared to the reference site (KM), mean temperature and DO of all sites except site T1, E and T2 were statistically significant ($p<0.05$). Ec and TDS were the difference was significant ($p<0.05$) for sites KG, KS, K1 and K2. pH was only significant ($p<0.05$) at site T2. However, TSS was significant ($p<0.05$) at site K1 only. Moreover, turbidity, nitrites, nitrates, TP and TN had no significant difference with the reference site (KM).

Table 3: Mean (± SE) values for physicochemical characteristics of the nine study sites.

Sites	A	KG	KS	K1	K2	T1	E	T2	KM
Altitude (m)	1440	1534	1533	1548	1574	1956	1970	1960	2304
Depth (m)	0.1523±0.032	0.6113±0.125	0.4823±0.012	0.5647±0.0418	0.394±0.024	0.3306±0.031	0.5457±0.074	0.4267±0.008	0.2957±0.026
Width (m)	1.85±0.253	5.175±0.259	4.86±0.826	6.45±0.318	7.2±0.473	7.025±0.464	3.75±0.328	4.925±0.502	4.6±0.376
Discharge(M3/sec	0.1078±0.029	0.7129±0.1084	1.6585±0.6061	3.0971±0.9638	2.6922±0.9762	2.9753±1.2642	0.9842±0.4018	1.4767±0.3862	0.6881±0.2882
% canopy cover	50	20	60	80	40	5	40	25	50
pH	7.30±0.20	7.40±0.20	7.39±0.09	7.59±0.16	7.58±0.07	7.65±0.18	7.98±0.20	8.20±0.17	7.79±0.24
DO (Mg/L)	7.325±0.264	6.970±0.293	7.877±0.268	7.365±0.085	7.457±0.066	7.925±0.298	8.137±0.279	7.900±0.262	8.437±0.186
TDS	30.25±2.95	79.75±8.08	68.25±6.34	46.75±3.90	48.00±3.24	37.25±1.80	40.75±4.21	36.00±3.11	35.50±1.85
Turbidity (NTU)	57.40±18.12	347.75±178.91	156.95±72.88	148.25±53.00	126.25±36.18	125.50±34.58	125.35±47.95	72.47±21.60	64.47±14.89
Temperature (oC)	21.73±0.90	20.52±0.41	19.72±0.48	17.51±0.36	18.42±0.30	16.51±1.12	17.55±0.81	15.19±1.00	13.72±0.70
TSS	23.67±6.20	135.50±51.58	111.16±25.46	120.91±5.39	108.54±7.48	86.58±15.53	95.25±18.53	65.50±19.52	37.12±10.70
Sulphate	4.74±2.32	8.31±2.74	9.85±1.03	5.13±1.27	5.83±0.60	11.90±6.92	3.17±1.58	2.80±0.98	5.53±0.78
Phosphate	0.015±0.005	0.086±0.021	0.064±0.019	0.056±0.027	0.056±0.021	0.040±0.006	0.036±0.011	0.067±0.022	0.042±0.009
Nitrite	1.399±0.796	0.908±0.566	0.810±0.459	0.424±0.240	0.328±0.179	0.399±0.226	0.464±0.258	0.458±0.285	0.348±0.219
Nitrate	0.319±0.254	0.113±0.077	0.143±0.082	0.071±0.052	0.150±0.067	0.044±0.021	0.096±0.058	0.043±0.016	0.077±0.041
TP	0.078±0.030	0.196±0.036	0.206±0.053	0.211±0.053	0.173±0.028	0.087±0.023	0.165±0.016	0.186±0.043	0.152±0.039
TN	1.675±0.878	1.172±0.722	1.375±0.732	0.967±0.473	1.008±0.600	0.928±0.529	0.938±0.520	0.933±0.535	0.905±0.525

*Table 4: Paired t-test compared with the reference site (KM). Values in table are two-tailed p-values. * means statistically significant, ** very statistically significant at 95% confidence. DO=Dissolved Oxygen, TSS=Total Suspended Solids, TP=Total Phosphates, TN=Total Nitrogen*

Parameter	A	KG	KS	K1	K2	T1	E	T2
Temp	0.00036**	0.0027**	0.0104*	0.0201*	0.0177*	0.0721	0.0641	0.4272
pH	0.1686	0.2443	0.2555	0.4533	0.412	0.4637	0.1464	0.0323*
Ec	0.2815	0.0031**	0.0176*	0.0393*	0.0144*	0.474	0.1784	0.9124
TDS	0.3154	0.0196*	0.0143*	0.043*	0.0126*	0.4222	0.1965	0.8788
DO	0.002**	0.0167*	0.029*	0.0181*	0.0152*	0.1632	0.2421	0.2904
Turb	0.7758	0.1996	0.259	0.153	0.13	0.0558	0.3163	0.7645
TSS	0.4513	0.1896	0.1143	0.0046**	0.0237*	0.0023	0.0828	0.3699
Phos	0.0111*	0.152	0.4282	0.7004	0.6532	0.8861	0.7675	0.3681
NO_2	0.1763	0.2053	0.1711	0.2535	0.8429	0.5509	0.3205	0.1966
NO_3	0.3432	0.4061	0.2071	0.813	0.2748	0.2458	0.4634	0.3364
TP	0.2084	0.4591	0.4367	0.4257	0.5148	0.1637	0.7855	0.359
TN	0.1662	0.2809	0.158	0.4015	0.2888	0.7872	0.4278	0.1995
Q	0.1642	0.9513	0.0575	0.1168	0.062	0.1012	0.0916	0.0053**

DO=Dissolved Oxygen, TSS=Total Suspended Solids, TP=Total Phosphates, TN=Total Nitrogen, Temp=Temperature, Ec=Electrical conductivity, TDS=Total Dissolved Solids, Turb=Turbidity, Phos=Phosphorous, NO_2=Nitrites, NO_3=Nitrates, Q=Discharge

Macroinvertebrate communities

A total of 7,444 macroinvertebrate individuals belonging to 73 taxa of 41 families in the 9 insect orders Odonata, Ephemeroptera, Plecoptera, Tricoptera, Coleoptera, Hemiptera, Diptera, Lepidoptera, three orders from class annelids (Hirudinea, Herodinea, and Oligochaeta) and Decapoda were collected from the nine sites during the study period.

In general, macroinvertebrate communities were dominated by two orders, Ephemeroptera (32.3%) and Diptera (53.2%). The rest of the 10 orders represented 14.5%. Ephemeroptera was the most diverse and abundant order which possessed 14 taxa and comprised about half percentage of total abundance in the Kuywa watershed. Tricoptera showed a lower diversity and abundance than Ephemeroptera. Diptera possessed 11 taxa. Among them, Simulidae was the most abundant genus possessing 46.6% at site KM.

Table 5: Macroinvertebrate metrics calculated from data collected between January to October, 2016 to discriminate the 9 Kuywa River sites in terms of their absolute numbers, abundance, richness, diversity and evenness.

Site	S	N	d	J'(EH)	H' (loge)	1-λ'
A	26	707	3.81	0.47	1.52	0.64
KG	34	312	5.75	0.68	2.41	0.80
KS	32	495	5.00	0.73	2.51	0.88
K1	31	1858	3.99	0.26	0.89	0.32
K2	35	1026	4.90	0.55	1.96	0.75
T1	29	607	4.37	0.69	2.34	0.82
E	34	843	4.90	0.72	2.55	0.88
T2	35	543	5.40	0.71	2.51	0.86
KM	28	1052	3.88	0.58	1.93	0.73

S=Richness index; N=Abundance index; d=Margalef richness; J'=Plelou's evenness; H'=Shannon index; EH=Shannon evenness; 1-λ=Simpson diversity.

The study revealed that the diversity and evenness from the most disturbed habitat (A and K1) were much less than in the site from the less disturbed habitat (KS, E and T2). It was also noticeable that the reference site (KM) had less index value of diversity and evenness. The less disturbed sites not only has a greater number of species present, but the individuals in the community are distributed more equitably among these species, except for site KS which had no Plecoptera. Moreover, it was observed that site KS which had natural conserved riparian vegetation had the highest species evenness index that all the other eight sites.

Relationships between macroinvertebrate assemblages and physichochemical variables

Macrobdella (Gynathobdellida), Ephemerella(Diptera) and Ellassoneuria(Ephemeroptera) were highly correlated with altitude, DO and pH ($\rho<0.01$) (Table 5). Hexatoma (Diptera) was highly correlated with percentage canopy cover ($\rho<0.01$) at the nine sampling sites. However, Actnonaias(Unionoida) was also highly correlated with DO (r=0.91, ρ<0.01). Macrobdella was negatively correlated (r=-0.85, ρ=0.002) to Temperature(T) and Total Nitrogen (TN).

Table 6: Correlation between macroinvertebrate genus and physicochemical variables

	Alt		Wid		Q		CC		pH		DO		TDS	
	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ
Megal														
Dipla														
Baet	0.77*	0.008							0.58*	0.05	0.68*	0.021		
Euth											-0.63*	0.034		
Hydrov	-0.73*	0.014							-0.73*	0.014				
Gerr	-0.59*	0.046			-0.59*	0.046			-0.59*	0.046	-0.73*	0.013		
Polypo	-0.63*	0.035									-0.68*	0.023		
Goer														
Lepido			0.65*	0.030	0.59*	0.048								
Ecno														
Asta			-0.70*	0.018										
Elm														
Hexa							0.83**	0.003						
Belos							0.73*	0.013						
Haplo														
Leptop	0.68*	0.022												
Oligo					0.71*	0.017								
Elass	0.83**	0.003							0.85**	0.002	0.78**	0.007		
Hydrop											0.64*	0.031		
Leptoc													0.59*	0.047
Ariac							-0.63*	0.034						
Mesop														
Epheme	0.82**	0.003							0.78**	0.006	.75**	0.01		
Simuli							.062*	0.037						
Limon														
Leac									0.73*	0.013				
Nepus									0.71*	0.017				
Nauc					-0.59*	0.048								
Notone			0.73*	0.013			-0.69*	0.02						
Pleid													0.78**	0.006
Tubif														
Lumb														
Sync	0.71*	0.016									0.79**	0.006		
Macrob	0.87**	0.001							0.85**	0.002	0.90**	0.000		
Mesov														
Actnon	0.68*	0.022							0.60*	0.046	0.91**	0.000		

	Tb		T		TSS		SO3		PO4		NO2		NO3		TP		TN	
	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ	r	ρ
Megal											-0.70*	0.018	0.61*	0.039				
Dipla							0.73*	0.013							.655*	0.039		
Baet			-0.67*	0.025							-0.93**	0.000					-0.72*	0.015
Euth			0.65*	0.029							0.72*	0.015						
Hydrov			0.62*	0.037							0.62*	0.037						
Gerr			0.71*	0.017							0.71*	0.017					0.59*	0.046
Polypo																		
Goer							0.71*	0.017										
Lepido																		
Ecno							0.71*	0.017										
Asta	-0.64*	0.032					-0.78**	0.007										
Elm											-0.60*	0.043						
Hexa																		
Belos																		
Haplo			-0.58*	0.050														
Leptop															-0.766*	0.013		
Oligo															0.764*	0.014		
Elass			-0.66*	0.028													-0.72*	0.015
Hydrop																		
Leptoc									.700*	0.018			-0.65*	0.029				
Ariac																		
Mesop							0.633*	0.034										
Epheme							-0.60*	0.042					0.70*	0.017	-0.76*	0.014	-0.60*	0.042
Simuli																		
Limon															-0.63*	0.047		
Leac							-0.73*	0.013										
Nepus							-0.71*	0.017										
Nauc											0.67*	0.0248						
Notone							0.64*	0.032										
Pleid	0.78**	0.006			0.78**	0.006												
Tubif							-0.84**	0.002										
Lumb							-0.67*	0.024										
Sync																	-0.71*	0.016
Macrob			-0.85**	0.002									0.83**	0.003			-0.85**	0.002
Mesov																		
Actnon													0.62*	0.037			-0.63*	0.035

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

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

Figure 2: Acronym used for species

Species	Acronym	Species	Acronym	Species	Acronym	Species	Acronym
Gomphus	Gom	Hexatoma	Hexa	Polypotomus	Polypo	Ephemerella	Epheme
Zygopteran	Zyg	Belostoria	Belos	Goerodes	Goer	Simulium	Simuli

Megalagrion	Megal	glossiphoria	glossi	Lepidostoma	Lepidos	Limnophora	Limnop
Diplacodes	Dipla	Haplogenis	Haplo	Chimarra	Chima	Limonia	Limon
Aeshnidae	Aesh	Tricorythus	Trico	Ecnomus	Ecno	Athericidae	Ather
Crocothemis	Croco	Leptophlebiidae	Leptop	Leptocerus	Leptoc	leach	leac
Lestes	Lest	Teleganodidae	Telega	Astacus	Asta	Nepus	Nepus
Baetis	Baet	Oligoneuridae	Oligo	Potamonaute	Potamon	Naucoris	Nauc
Caenis	Caen	Elassoneuria	Elass	Cyrinus	Cyri	Micronecta	Micro
Leptohyphidae	Lepto	Hydropsyche	Hydrop	Potamodyte	Potamod	Notonectidae	Notone
Afronurus	Afro	Leptoceridae	Leptoc	Elmnae	Elm	Pleidae	Pleid
Euthraulus	Euth	Amphinenva	Amphin	Proteneura	Prote	Tubifex	Tubif
Hydrovatus	Hydrov	Ariacalis	Ariac	Chrysomelidae	Chrys	Lumbricus	Lumb
Gerries	Gerr	Perlidae	Perli	Chironomous	Chiron	Synclita	Sync
Belostoma	Belos	Mesoperla	Mesop	Tanypodinae	Tany	Macrobdella	Macrob
Actnonaias	Actnon	Mesovelia	Mesov				

Dendrogram was constructed to establish the groupings of the nine sites of Kuywa River in terms of physical variables and macroinvertebrate assemblages and Figure 3 and Figure 4 emerged. Group II (Figure 4) which comprised sites E and T2 as per physicochemical variables appeared to be grouped together in macroinvertebrate assemblages as group III (Figure 3). Furthermore, K2 and KG which formed group IV (Figure 4) in macroinvertebrate assemblages were found to be in the same group (II) (Figure 3) in physicochemical variables.

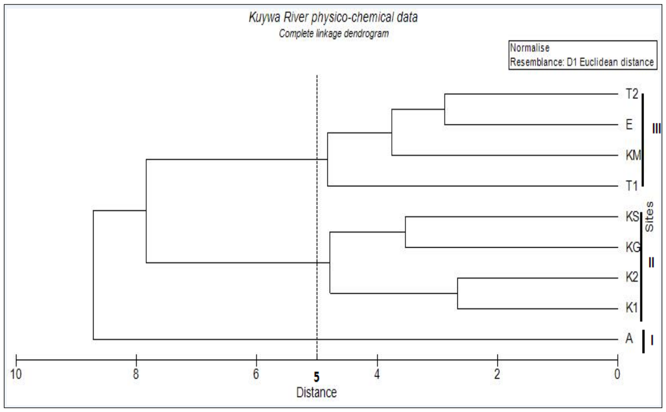


Figure 3: Dendrogram of the 9 Kuywa River sites based on complete linkage hierarchical cluster analysis from Bray-Curtis similarities on log x transformed. The three clusters of sites are separated at 50% similarity threshold.

Emerging general pattern in macroinvertebrate indicators

In Group I cluster of Figure 3, the sites had excellent riparian but heavily utilised for agricultural practices. Group II sites comprised the site with planted mature and young riparian vegetation. Group III sites had young planted and open grazed sites. Group IV where the "reference" site was included, comprised the site with mature planted vegetation mixed with eucalyptus trees and K2 with had mature planted riparian vegetation.

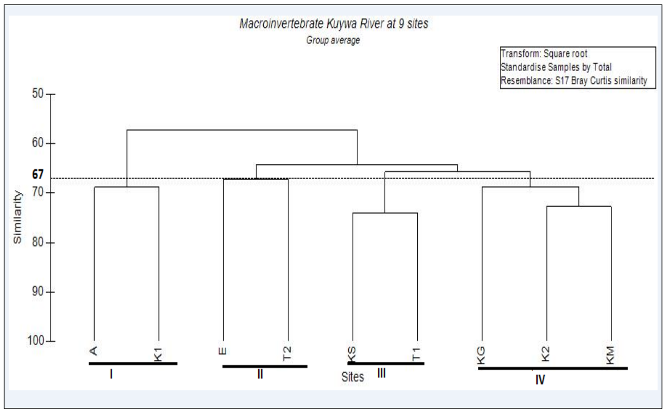


Figure 4: Figure 4: Dendrogram of the 9Kuywa River sites based on group average hierarchical cluster analysis from Bray-Curtis similarities on square-root-transformed abundances. The four clusters of sites are separated at 67% similarity threshold.

To visualise the data in 2 dimensions so as to investigate whether the data had any meaningful structure at this 67% similarity, the cluster members in the dendrogram were found to be the same as those in the MDS at 65% similarity (Figure 4). The stress of 0.06 indicated a good structure in the configuration. ANOSIM analysis on physico-chemical variables indicated that there was a minimal significant differences between sites ($R = 0.391$, $p = 0.1$). However, ANOSIM analysis on benthic macroinvertebrate assemblages showed no significant difference ($R=0.094$, $\rho=0.611$). PCA followed by BIO-ENV procedure indicated that discharge (Q), Sulphate (SO_3) and Phosphate (PO_4) have a greater influence on benthic macroinvertebrate assemblages than other water quality variables(BEST P=0.515)(Figure 5).

ANOSIM analysis showed no statistical difference between the mid-pH cluster sites and the low-pH cluster sites (R statistic¼0.365, P¼0.1)

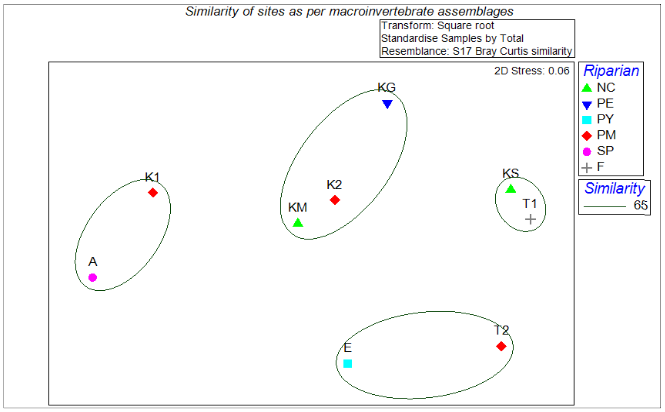


Figure 5: The 9 Kuywa River sites macroinvertebrates' MDs with the averaged abundances, the sites group into four major clusters. KM was the "reference" site. NC=Natural Conserved riparian, PE=Planted vegetation with Eucalyptus, PY=Planted Young vegetation, PM=Planted Mature riparian vegetation, SP=Sugar cane Plantation and F=Fenced riparian.

The analysis of the vector values for the water quality variables showed clearly that total dissolved solids (TDS) and turbidity (Tb) appears to be the key variable determining differences in macroinvertebrate composition between sites (Fig. 4 and 5). However, the discharge of a stream (Q) was also found to influence aquatic benthic macroinvertebrate assemblages. Cluster overlays (resemblance level 65%) clearly separated the sites with vegetated riparian zone from that which was bare and sugar cane plantation (Figure 4). Fig. 5 also shows discharge (Q) as one of three variables accounting for 50% of the variation in community structure.

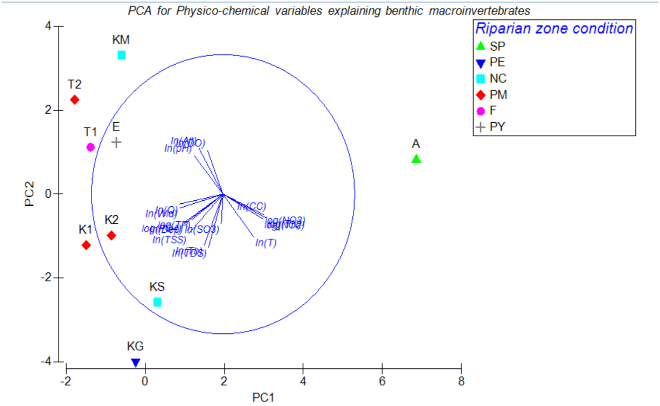


Figure 6: PCA showing differences between sites in regards to physico-chemical variables, 86% captured in the PC1-PC3; with the PC1 dominating 41.3%, PC2 32.8% and PC3 11.9%. PC1: TP(-0.232), PC2: TDS(-0.381), Tb(-0.364) and PC3: CC(-0.589), SO_3(0.525), TP(-0.44).

7. DISCUSSION

Our results suggested on-going effects of pollution on macroinvertebrate communities in Kuywa River despite introduction of riparian vegetation cover at most parts of the river stretch. It was expected that headwater communities at the pristine Site KM would show lowest densities, and the highest taxonomic richness at the site scale. However, site KM had a comparable richness to other sites but being higher than site A and highest abundance together with site K1. Site K1 had mature planted riparian vegetation which mimicked the pristine site KM. These two sites were not completely free from human interference. The result is comparable to other studies (Belsel, Usseglio-Polatera, & Moreteau, 2000; Sponseller, Benfield, & Valett, 2001) which have established the importance of riparian vegetation in aquatic colonization of macroinvertebrates.

The Order Diptera was the most abundant during the study period, being predominant in the process of decomposition of detritus. Another role of these organisms has been reported in other studies, which represent them as essential in the detritus recycling. Thus, our results corroborate the findings of Moretti, Goncalves, Ligeiro, and Callisto (2007), Ligeiro, Moretti, Goncalves, and Callisto (2010), Goncalves, Rezende, Franca, and Callisto (2012) and Biasi, Tonin, Restello, and Hepp (2013) which also pointed out the Chironomidae (Diptera) dominance associated with plant substrate in the decomposition process. Because this group is important for nutrient recycling, it might be responsible for structuring entire community of benthic macroinvertebrate, as they are considered generalists, allowing them to colonize different types of detritus, regardless of its quality (Goncalves et al., 2012).

Physicochemical variables were significantly related to a number of macroinvertebrate genus. Macrobdella (Gnathobdellida) which is highly intolerant to pollution (Sawyer, 1986) was found to be negatively correlated to NO3, NO2 TN and TP. This is due to the fact that these nutrients lead to oxygen deficiency in pool water, which is mostly the habitat for the Macrobdella. The main effect of excess nitrogen and phosphorous in the water body is that they stimulate the growth of aquatic plants and algae blooms making the water deficient in oxygen (Kadlec & Wallace, 2009). However, other genus such as Gompus and Lestes (Odonata) were abundant since they are more tolerant to pollution and other anthropogenic disturbances. However, we must be cautious when interpreting the deference between site KM and downstream sites, as these could also be at least partly due to a natural altitudinal gradient (Site KM was located at more than 2300 m asl while Site A was 1440 m asl). Altitude was one of the abiotic variables most related to variation in macroinvertebrate densities and richness between Site KM and the other sites. Stream width, which generally co-varies with altitude, was also related to this variation, as well as canopy cover, alkalinity, and dissolved solids. These variables may show some natural variation along an altitudinal gradient (particularly canopy cover) but are also frequently related to human activities (Sheldon et al., 2012). Moreover, some other variables commonly associated to organic pollution (pH, nitrates and phosphates) were strongly related to deference in taxonomic richness between Sites T1 and E and the other sites. Our results are similar to other studies that have found significant increase in the specific conductance with reduced riparian vegetation (Greenway, 2006). In stream ions occur naturally in water as a result of the local geology and the subsequent geomorphic process. However, the higher levels detected in streams surrounded by increased land use suggest an anthropogenic effects.

Water quality and the relative occurrence of habitat types also affect macroinvertebrate assemblage composition. For example, Hydropsyche (Trichoptera: Hydropsychidae) are found more frequently in slowly flowing waters than in pools and faster moving water where they build catchnets at downstream of woods and rocks to trap the prey. This might have been the reason for site A to be very different by having less of the Tricoptera compared to the other sites. However, as argued by Baptista et al. (2007), and Ferreira, Paiva, and Callisto (2011), the quality of the habitat for the benthic macroinvertebrates may be affected by dissolved oxygen (DO) and level and conductivity. Moreover, these intolerant macroinvertebrates act as indicator organisms and their absence represent poor water quality.

The presence of a high number of EPT taxa at some sites within Kuywa River, compared with other waterways in Lake Victoria Basin (generally contain 1–4 EPT: usually Baetidae, Caenidae, Leptoceridae and Ecnomidae) indicate that the Kuywa River in the present study is of good ecological health (Duivenvoorden et al., 2008;

Duivenvoorden, Price, Noble, & Carroll, 2003). The presence of Plecoptera especially provides support for their good ecological condition as Plecoptera are generally absent from polluted streams.

TSS of water at K1and K2 the sites which had mature planted riparian vegetation had a significant value (ρ=0.0046 and ρ=0.023 respectfully) compared to the test site as the degradation of water quality may not only be related to the riparian zone vegetation condition at a given stream stretch. The type of land use within the sub-catchment has a great influence on the river physicochemical variables. Not only physicochemical variables are affected by land use but in general the river health including factors such as water quality, habitat quality and biotic communities (Lichtenberg & Shapiro, 1997; Smart, Jones, & Sebaugh, 1985). Ecological prosperity which ensures biological biodiversity require natural fluctuations in hydrological parameters. However, in the Kuywa River, hydrological regimes have been subjected to anthropogenic changes as a result of land use changes (Nyakora & Ngaira, 2014). As a direct result, many aquatic ecosystems are presently subject to either insufficient or extreme discharge that are frequently destructive to aquatic life. During the study period, Site A and KG were highly eroded after the heavy downpours of May, 2016. This finding is similar to the one undertaken by Adámek, Konečná, Podhrázská, Všetičková, and Jurajdová (2016), which established that the extreme rainfall episodes and subsequent sudden flow pulses had an impact on biota communities.

SO_4, PO_4 and Q were linked to differences in benthic aquatic macroinvertebrate composition in Kuywa River between sites in the our study. PO4 was determined to be the Key variable with significant correlation recorded in a single BIO-ENV correlation. Conversely, pH, altitude and dissolved oxygen were negatively correlated with temperature, therefore the difference in benthic aquatic macroinvertebrate assemblages linked altitude, pH and oxygen are likely linked to decrease in temperature rather than direct altitude, pH and oxygen. Connolly, Crossland, and Pearson (2004) and Kaller and Kelso (2007) observe that there is no effect of dissolved oxygen on benthic macroinvertebrate assemblages since most taxa are tolerant of oxygen levels <10% saturation except for Ephemeroptera, which are generally sensitive to low oxygen. Our study adds support the idea of having a combination of factors to determine the aquatic benthic macroinvertebrate assemblages.

Further, distinct type of land use in a sub catchment of a river changes the community structure and composition of the colonizer' aquatic macroinvertebrates. Site KG and K2 were in the same cluster in terms of benthic macroinvertebrates since their riparian vegetation conditions were near to that of site KM (reference site). This was attributed by the fact that streams with more preserved characteristics usually have a higher densities of riparian vegetation on the banks, leading to increase in allochthonous leaves' input, which provide food and shelter for many insect larvae (Ayres-Peres, Sockolowicz, & Santos, 2006). This study was consistent with Ramirez and Gutlerrez-Fonseca (2014) and Oliveira and Nessimian (2010) who established that high densities of riparian zone vegetation is normally associated with physical and chemical characteristics of aquatic environment thus providing conditions for establishment of fragmenting macroinvertebrates.

In this way, we rejected our null hypothesis that water quality indicators have no relationship with benthic macroinvertebrate assemblages in the Kuywa River. The effects herein detected and discussed above could be influenced by lowering input of allochthonous materials and also due to increased input of fine sediments and other chemicals in streams draining adjacent area.

8. CONCLUSION

This study established that physico-chemical parameters influences the structure of benthic macroinvertebrate assemblages, leading to changes and reductions (1) in taxonomic richness, (2) abundance of organisms, and (3) the proportion of sensitive taxa in the community. Diptera in general dominated the study sites which was consistence with other studies done within the Nzoia basin. Intolerant species such as Macrobdella were found to be negatively correlated with with nutrients such as TN, TP and NO3 as they favour the growth of aquatic algae thus depriving water of oxygen. Our study found relatively higher number of EPT at some sites compared to other waterways of Lake Victoria Basin, suggesting a fair river health than others. In this way, the present study confirms that benthic macroinvertebrate assemblages as an important tool to assess the health of streams. Despite the existing and planted riparian vegetation at some of the streams, sensitive taxa communities reflected the damage caused due to conversion of the native forest to agricultural land use which finally negatively impact on the river physico-chemical variables. Further, we observed that streams with more preserved characteristics (e.g. KG) have a higher densities of macroinvertebrate communities as a result of increased allochthonous leaves' input which provide food and shelter for insect larvae. Thus, we recommend improved ecological assessment programs, watershed management practices, aquatic ecosystem rehabilitation measures, and protection strategies for aquatic biota of the head waters of equatorial streams. Additionally, best land use practices should be employed at watershed level to minimize the sediment carrying, pesticides, and fertilizers in the runoff into the stream, and so effectively conserving the aquatic fauna. This knowledge is important to the community, government and water resource managers as they plan in investing resources to rehabilitate the catchments with a view of improving the health of equatorial rivers.

9. REFERENCES

Acũna, V., Díez, J. R., Flores, L., Meleason, M., & Elosegi, A. (2013). Does it make eco-nomic sense to restore rivers for their ecosystem services? . *Journal Appl. Ecol.*, 1365-2664.

Adámek, Z., Konečná, J., Podhrázská, J., Všetičková, L., & Jurajdová, Z. (2016). Response of Small-Stream Biota to Sudden Flow Pulses Following Extreme Precipitation Events. *Pol. J. Environ. Stud., 25*(2 (2016)), 495-501.

APHA. (1998). *Standard method for the Examination of Water and Wastewater, 2nd Edition APHA,* . Washington D.C.

Arthington, A., Naiman, J. R., Mcclain, M., & Nilsson, C. (2010). Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biology, 55*, 1-16.

AustralianGovernment. (2001). *Queensland Australian River Assessment System (AusRivAS) Sampling and Processing Manual, August 2001*.

Ayres-Peres, L., Sockolowicz, C. C., & Santos, S. (2006). Diversity and abundance of the benthic macrofauna in lotic environments from the central region of the Rio Grande do Sul State, Brazil. *Biota Neotropica, 6*.

Baptista, D. F., Buss, D. F., Egler, M., Giovanelli, A., Silveira, M. P., & Nessimian, J. L. (2007). A multimetric index based on benthic macroinvertebrates for evaluation of Atlantic Forest streams at Rio de Janeiro State, Brazil. *Hydrobiologia, 575*, 83-94.

Barbour, M. T., & Stribling, J. B. (1993). A technique for assessing stream habitat structure. In: Jungwirth, M., Muhar, S., & Schmutz, S. (2002). Re-establishing and assessing ecological integrity in riverine landscapes. *National Association of Conservation Districts, Washington, DC.*, 156-178.

Belsel, J. N., Usseglio-Polatera, P., & Moreteau, J. C. (2000). The spatial heterogeneity of river bottom: a key factor determining macroinvertebrate communities. *Hydrobiologia, 442*(423), 163-171.

Biasi, C., Tonin, A. M., Restello, R. M., & Hepp, L. U. (2013). The colonisation of leaf litter by Chironomidae (Diptera): The influence of chemical quality and exposure duration in sub tropical stream. *Limnological-Ecology and Management of Inland Waters, 43*, 427-433.

Chadwick, M. A., Thiele, J. E., Huryn, A. D., Benke, A. C., & Dobberfuhl, D. R. (2012). Effects of urbanization on macroinvertebrates in tributaries of the St. Johns River, Florida, USA. Urban Ecosyst, 15, 347-365.

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

Duivenvoorden, L. J., Houston, W., Kinnear, S., Black, R., Goedee, A., Stitz, L., . . . Lowry, R. (2008). Ecological considerations of spray control for Hymenachne. In 'Ecological, economic and social considerations of spray control for Hymenachne'. *Full technical report prepared for Land and Water Australia, Australia*.

Duivenvoorden, L. J., Price, M. D., Noble, R. N., & Carroll, C. (2003). Assessment of ecological risk associated with irrigation in the Fitzroy Basin: Phase 2 - Analysis and characterisation of risk with emphasis on effects on macroinvertebrates *Project No UCQ3: Final Report to Land and Water Australia, Australia*.

Ferreira, W. R., Paiva, L. T., & Callisto, M. (2011). Development of a benthic multimetric index for biomonitoring of a Neotropical watershed. *Brazilian Journal of Biology, 71*, 15-25.

Fellows, C. S., Clapcott, J. E., Udy, J. W., Bunn, S. E., Harch, B. D., Smith, M. J., & Davies3, P. M. (2006). Benthic metabolism as an indicator of stream ecosystem health. Hydrobiologia, 572, 71-87. doi: 10.1007/s10750-005-9001-6

Flinders, C., Horwitz, R., & Belton, T. (2008). Relationship of fish and macroinvertebrate communities in the mid-Atlantic uplands: implications for integrated assessments. *Ecol Indic, 8*, 588-598.

Goncalves, J. J. F., Rezende, R. S., Franca, J., & Callisto, M. (2012). Invertebrate colonisation during leaf processing of native, exotic, and artificial detritus in a tropical stream. *Marine Freshwater Research, 63*, 428-439.

Government-of-Kenya. (1984). *Kimilili 88/1, Bungoma 88-3, Elgon 74-3. East Africa 1:50,000 (Kenya)*. Nairobi: Survey of Kenya, Nairobi.

Greenway, M. (2004). Constructed Wetlands for Water Pollution Control - Processes, Parameters and Performance. *Dev. Chem. Eng. Mineral process, 12*(5/6), 1-14.

Greenway, M. (2006). The role of macrophytes in nutrient removal using constructed wetlands. In S. N. Singh & R. D. Tripathi (Eds.), *Bioremediation-A novel Technology*. Lucknow, India: National Botanic Research Institute.

Gregory, S. V., Swanson, F. J., McKee, W. A., & Cummins, K. W. (1991). An Ecosystem Perspective of Riparian Zones: Fucus on links between land and water. BioScience, 41(8).

Growns, I., Rourke, M., & Gilligan, D. (2013). Toward river health assessment using species distributional modeling. *Ecological Indicators, 29*, 138-144. doi: 10.1016/j.ecolind.2012.12.024

Hughes, R. M., Herlihy, A. T., & Kaufmann, P. R. (2010). An evaluation of qualitative indexes of physical habitat applied to agricultural streams in ten U.S. States. *Journal of the American Water Resources Association, 46*, 792–806.

Jungwirth, M., Muhar, S., & Schmutz, S. (2002). Re-establishing and assessing ecological integrity in riverine landscapes. *Freshwater Biology, 47*, 867-887.

Kadlec, R. H., & Wallace, S. (2009). Treatment wetlands. *CRC Press Boca Raton, FL.*

Kaller, M. D., & Kelso, W. E. (2007). Association of macroinvertebrate assemblageswithdissolvedoxygenconcentrationandwoodsurfacearea in selected subtropical streams of the southeastern USA. *Aquatic Ecology, 41*, 95-110.

Kaufmann, P. R., & M., F. J. (2012). Simple measures of channel habitat complexity predict transient hydraulic storage in streams. *Hydrobiologia, 685*, 69-95.

Kennedy, T. L., & Turner, T. F. (2011). River channelization reduces nutrient flow and macroinvertebrate diversity at the aquatic terrestrial transition zone. Ecosphere, 2(3).

Kovalenko, K. E., Thomaz, S. M., & Warfe, D. M. (2012). Habitat complexity: approaches and future directions. *Hydrobiologia, 685*, 1–17.

Ladson, A. R., & White, L. J. (1999). *An Index of Stream Condition: Reference Manual*. (9780731143184). Victoria. Dept. of Natural Resources Environment. Waterways Unit: Waterways Unit, Department of Natural Resources and Environment Retrieved from http://books.google.com.au/books?id=OaV_AAAACAAJ.

Larsen, S., Pace, G., & Ormerod, S. J. (2011). Experimental eff ects of sediment deposition on the structure and function of macroinvertebrate assemblages in temperate streams. *River Res. Appl., 27*, 257-267.

Lazdinis, M., & Angelstam, P. (2005). Functionality of riparian forest ecotones in the context of former Soviet Union and Swedish forest management histories. *Forest Policy and Economics, 7*(3), 321-332.

Lichtenberg, E., & Shapiro, L. K. (1997). Agriculture and nitrate concentrations in Maryland Community Water System Wells. *Journal of Environmental Quality, 26*, 145-153.

Ligeiro, R., Hughes, R. M., Kaufmann, P. R., Macedo, D. R., Firmiano, K. R., Ferreira, W., . . . Callisto, M. (2013). Defining quantitative stream disturbance gradients and the additive role of habitat variation to explain macroinvertebrate taxa richness. *Ecological Indicators, 25*.

Ligeiro, R., Moretti, M. S., Goncalves, J. J. F., & Callisto, M. (2010). What is more important for invertebrate colonization in streams with low-quality litter inputs: exposure time or leaf species? *Hydrobiologia, 654*, 125-136.

Masese, F. O., Muchiri, M., & Raburu, P. O. (2009). Macroinvertebrate assemblages as biological indicators of water quality inthe Moiben River, Kenya. *African Journal of Aquatic Science, 34*(1), 15-26.

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

Mathooko, J. M. (1998). Mayfly diversity in East Africa. *Africa Journal of Ecology, 36*, 368-370.

Mendes, T., Calapez, A. R., Elias, C. L., Almeida, S. F. P., & Feio, M. J. (2014). Comparing alternatives for combining invertebrate and diatom assessment in stream quality classification. *Marine and Freshwater Research, 65*, 612-623.

Moretti, M. S., Goncalves, J. J. F., Ligeiro, R., & Callisto, M. (2007). Invertebrates colonization on native tree leaves in a neo-tropical stream (Brazil). *International review of Hydrobiology, 92*, 199-210.

Naiman, R. J., & Decamps, H. (1997). The ecology of interfaces: Riparian Zones Annual Review of Ecology and Systematics, 28, 621-658.

Nerbonne, B. A., & Vondracek, B. (2001). Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. *Environmental Management, 28*, 87–99.

Newham, M. J., Fellows, C. S., & Sheldon, F. (2011). Functions of riparian forest in urban catchments: a case study from sub-tropical Brisbane, Australia. *Urban Ecosyst, 14*, 165-180. doi: 10.1007/s11252-010-0151-6

Nyakora, J. O., & Ngaira, J. (2014). Assessing the achievement of integrated watershed management tool for sustainable management of water resources in Kuywa River, Western Kenya. *Herald J. Geogr. Rgnl. Plann., 3*(4), 140 - 157

Oliveira, A. L. H., & Nessimian, J. L. (2010). Spatial distribution and functional feeding groups of aquatic insect communities in Serra da Bocaina streams southern Brazil. *Acta Limnologica Brasiliensia, 4*, 424-441.

Parkyn, S. M., Davies-Colley, R. J., Halliday, N. J., Costley, K. J., & Croker, G. F. (2003). Planted Riparian Buffer Zones in New Zealand: Do They Live Up to Expectations? . *Restoration Ecology, 11*, 436-447. Paul, M. J., & Meyer, J. L. (2001). Streams in the urban landscape. Annu Rev Ecol Syst, 32, 333-365.

Perera, L. G. R. Y., Wattavidanage, J., & Nilakarawasam, N. (2012). Assessing the health of an urban stream: a case study of Suzhou Creek in Shanghai, China. *Environ Monit Assess, 184*, 7425-7438.

Pinto, B. C. T., Araujo, F. G., Rodrigues, V. D., & Hughes, R. M. (2009). Local and ecoregion effects on fish assemblage structure in tributaries of the Rio Paraíba do Sul, Brazil. *Freshwater Biology, 54*, 2600–2615.

Raburu, P. O., Masese, F. O., & Mulanda, C. A. (2009). Macroinvertebrate Index of Biotic Integrity (M-IBI) for monitoring rivers in the upper catchment of Lake Victoria Basin, Kenya. *Aquatic Ecosystem Health & Management, 12*(2), 197-205.

Ramirez, A., & Gutlerrez-Fonseca, P. E. (2014). Functional feeding groups of aquatic insect families in Latin America: a critical analysis and review of existing literature. *International Journal of Tropical Biology and Conservation, 62*, 155-167.

Raven, P. J., Fox, O. P., Everard, M., Holmes, N. T. H., & Dawson, F. H. (1997). River habitat survey: a new system for classifying rivers according to their habitat quality. In:Jungwirth, M., Muhar, S., & Schmutz, S. (2002). Re-establishing and assessing ecological integrity in riverine landscapes. . *The Stationary Office, Edinburgh.*, 215-234.

Reisinger, A. J., Blair, J. M., Rice, C. W., & Dodds, W. K. (2013). Woody Vegetation Removal Stimulates Riparian and Benthic Denitrification in Tallgrass Prairie. *Ecosystems, 16*, 547-560.

Sály, P., Takács, P., I. Kiss, Bíró, P., & Erõs, T. (2011). The relative influence of spatial context and catchment- and site-scale environmental factors on stream fish assemblage in a human-modified landscape. *Ecology of Freshwater Fish, 20,* 251-262.

Sawyer, R. T. (1986). *Leech Biology and behaviour* (Vol. 2). Oxford England: Clarendon Press.

Schalchli, U. (1992). The clogging of coarse gravel beds by fi ne sediment. *Hydrobiologia, 235-236*, 189-197.

SEPA. (2003). Environmental Handbook for Monitoring: Site Descriptions (Version 1:5 2003-09-25), Program,Swedish. *Freshwater:*, 1-17

Sheldon, F., Boulton, A. J., & Puckridge, J. T. (2002). Conservation value of variable connectivity: aquatic invertebrate assemblages of channel and floodplain habitats of a central Australian arid-zone river, Cooper Creek. *Biological Conservation, 103*, 13-31.

Sheldon, F., Peterson, E. E., Boone, E. L., Sippel, S., Bunn, S. E., & Harch, B. D. (2012). Identifying the spatial scale of land use that most strongly influences overall river ecosystem health score. doi: 10.1890/11-1792.1

Smart, M. M., Jones, J. R., & Sebaugh, J. L. (1985). Streamwatershed relations in the Missouri Ozark Plateau Province. *Journal of Environmental Quality, 14*, 77 -82.

Sponseller, R. A., Benfield, E. F., & Valett, H. M. (2001). Relationship between land use, spatial scale and stream macroinvertebrate communities. *Freshwater Biology, 2001*(46), 1409-1424.

Stutter, M. I., Langan, S. J., & Demars, B. O. L. (2007). River sediments provide a link between catchment pressures and ecological status in a mixed land use Scottish River system. *Water Res., 41*, 2803-2815.

Törnblom, J., Angelstam, P., Degerman, E., Henrikson, L., Edman, T., & Temnerud, T. (2011). Catchment land cover as a proxy for macroinvertebrate assemblage structure in Carpathian Mountain streams. *Hydrobiologia, 2011*(673), 153-168.

USEPA (Producer). (2009, 7th November, 2014). Invertebrates as indicators. Biological indicators of watershed health. Retrieved from http://www.epa.gov/bioindicators/html/invertebrate.html

Utz, R. M., & Hilderbrand, R. H. (2011). Interregional variation in urbanization-induced geomorphic change and macroinvertebrate habitat colonization in headwater streams. J N Am Bentholl Soc, 30, 25-37. Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: current knowledge and the search for a cure J North Am Benthol Soc, 24, 706-723.

Water Resources Management Authority, W. (2011). *Water Abstraction Survey for Kuywa Watershed*. Kakamega: WRMA.

Xu, S., & Liu, Y. (2014). Assessment for river health based on variable fuzzy set theory. *Water Resources, 41*(2), 218-224. doi: 10.1134/S0097807814020134

Zhao, G. J., Mu, X. M., Tian, P., Jiao, J. Y., & Wang, F. (2013). Have conservation measures improved Yellow River health? *Journal of Soil and Water Conservation, 68*(6), 159A-161A. doi: 10.2489/jswc.68.6.159A