Relationship Between Riparian Vegetation Cover And Macroinvertebrate Assemblages In Kuywa River, Kenya

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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 cover and benthic macroinvertebrates were collected. Spatial characteristics were obtained by averaging four rounds of field sampling. Descriptive statistics employed included Richness Index (S), Abundance Index (N), Margalef Richness (d), Shannon Index (H), Simpson diversity (λ) and Pielou. Bray-Curtis similarity measure and Multi-dimensional Scaling (MDS) were applied. To test the hypothesis, whether variations between the sites are significant, ANOSIM analysis was applied. Further, the effect of planted riparian zone vegetation was tested by the percentage of EPT. The study classified the study sites into 'Excellent', 'Good', and 'Poor'. Three genus were found to be positively correlated ($p<0.01$) with canopy cover and two negatively correlated. On the site richness, KG, K1 and T2 were found to have the highest family richness (15, 14, 14 respectfully) and A (8) the least. Our study found a significant difference between sites in terms of macroinvertebrate assemblages ($R=0.94$, $p<0.01$). For the sensitive species, K2, T2 and KM (14, 13, 11, respectfully) had the highest richness in terms of species, families and abundance. Site A had the lowest intolerant species (8). ANOSIM hypothesis testing indicated variations between sites were statistically different ($R=0.94$, $p<0.1$). SIMPROF test indicated that the MDS clusters generated for the nine sites were statistically significant ($Pi=3.215$, $p=0.001$). Our study concluded that the loss of large woody debris provided by riparian vegetation reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage; reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability and community function. The land use above the site has a considerable influence on the river health. For management and restoration actions to be effective, we must diagnose cause as well as assess harm, which requires an improved understanding of the mechanisms through which land use impacts stream ecosystems. This knowledge is important to the community and water resource managers as it will yield information on effect of planted riparian zone vegetation on protecting the river health which may lead to the replication of the same project in other watersheds.

Keyword: Benthic Macroinvertebrates, Riparian Vegetation Cover, River Health, Kuywa River

I. INTRODUCTION

The ecological health of rivers and streams is a fundamental and increasingly important water management issue in both developed and developing world (Bunn, Davies, & Mosisch, 1999; Masese et al., 2013; Wantzen, Ramirez, & Winemimller, 2006). The health of a river in ecological perspective is considered by looking at its sustainability, resilience to stress and ecological integrity (Naiman & Decamp, 1997). The ecological integrity in this case may refer to the capacity to support and maintain a balanced, integrated, adaptive biologic system having the full range of elements and processes expected in the natural habitat of a region (Naiman & Decamp, 1997). With this understanding, programs such as AusRivAS, which measure river health by making comparison with a nearby natural reference site (Smith et al., 1999) has been established. Moreso in South Africa, SAAS programme classify health of rivers using the composition of macroinvertebrates (Masese et al., 2014). The aim of these programes is to establish rivers which mimic the characteristics of a pristine conditions with all processes and benthic macroinvertebrates assemblages being at optimum.

Prior to the 1990s, river health assessment mainly relied on water quality measures; however, in more recent times, assessment programs have focused on the direct measurement of characteristics of the biota (mainly benthic macroinvertebrates, algae, vegetation, and fish) or ecosystem processes ,(e.g.,(Angradi et al., 2011; Bunn et al., 2010; Metcalfe-Smith, 1996). A holistic approach to assessing the health of a river system is to apply multimetric methods which combine indicators that represent the biological, chemical, and physical aspects of ecosystems (e.g., (Bunn et al., 2010; Davies, Harris, Hillman, & Walker, 2010; Ladson & White, 1999; Zhao, Yang, & Yao, 2005). Nevertheless, the riparian vegetation provides both physical and biological aspects to determine river health. The physical aspects may include regulation of temperature through its canopy, while biological aspects may include instream primary production.

More especially in small streams, the riparian provide instream primary production through light interception and provides allochthonous organic matter used as food and habitat by aquatic consumers (Monoury, Gilbert, & Lecerf, 2014). Vegetation cover also affects the exposure of the river to other disturbances such as cattle grazing. Sabo et al. (2005) established that changes in riparian forest age, canopy structure and plant communities may modify the composition of stream communities and functional roles played at community levels.

Riparian zone vegetation normally comprises diverse, dynamic and complex systems, such that the ecosystem functioning depends on the composition and characteristics of flora (Kim, Yeom, & An, 2014; Masese et al., 2013; Sheldon & Fellows, 2010). Among other functions of riparian zone vegetation are; provision of critical habitat and corridors for terrestrial wildlife (Naiman & Decamps, 1997), provision of habitat for aquatic organisms, ensures protection of surface water quality and quantity (Growns, Rourke, & Gilligan, 2013), and the control of the temperature of streams through canopy shades (Zhao, Mu, Tian, Jiao, & Wang, 2013). Furthermore, it prevents bank erosion by stabilizing bank soils through roots (Naiman & Decamps, 1997; Nyakora & Ngaira, 2014). This complexity sustain both simple and complex food-web in the rivers.

Other functionalities of riparian vegetation that affect the health of a river include the provision of organic inputs, such as woody debris and leaf litter, to the stream ecosystem (Newham, Fellows, & Sheldon, 2011). The woody debris creates a habitat and provides nutrients to stream organisms, dissipates energy and traps moving materials (Naiman & Decamps, 1997; Sheldon, Boulton, & Puckridge, 2002; Xu & Liu, 2014). These functions lead to assessing the effect of riparian vegetation on river health by monitoring the characteristics of benthic macroinvertebrates.

Benthic macroinvertebrates are widely used for assessments of river health (Johnson, Widerholm, & Rosenberg, 1993). Benthic macroinvertebrates reside in the benthic habitat for at least part of their life, relatively immobile, and very sensitive, therefore any disturbances in the aquatic environment may cause them to disappear or reduce diversity (Hilsenhoff, 1988; Morse et al., 2007; Zamora-Muñoz, Sáinz-Cantero, Sánchez-Ortega, & Alba-Tercedor, 1995). The advantage of using benthic macroinvertebrates in river health monitoring have been discussed by Rosenberg and Resh (1993) and Ndebele-Murisa (2012) as they provide continuous monitors of the condition of the waters they inhibit. Generally, benthic macroinvertebrates are sound to be used for river health assessment due to their taxonomic soundness (easy to be recognized by non-specialists); wide or cosmopolitan distribution; low mobility (local indication); well-known ecological characteristics; Numerical abundance; exhibit diversity and are sensitive to pollution; high sensitivity to environmental stressor (s); and high ability for quantification and standardization (Füreder & Reynolds, 2003; Hilty & Merenlender, 2000; Rosenberg & Resh, 1993).

In Africa, the concept of stream restoration by planting riverine vegetation is new. The South African River Health Monitoring Program investigates the macroinvertebrate assemblages in different rivers under different disturbance gradients (Nojiyeza, 2013). Both in Uganda and Tanzania, the Lake Victoria Environmental Management Programgram investigated the characteristics of macroinvertebrate and microinvertebrates assemblages in Lake Victoria and its rivers in natural and disturbed gradients. Further, studies have been carried out in upland rivers of the Usambara Mountains of Tanzania to evaluate the impacts of tea plantations in the catchment of rivers on macroinvertebrates assemblage (Biervliet, 2009). These previous studies in Africa focused on disturbances on natural river vegetation and their impact on macroinvertebrates. However, no study has been carried out to investigate the effect of planted riparian vegetation cover on benthic macroinvertebrate assemblages attributes.

In Kenya, a number of studies have been undertaken by using macroinvertebrates as an indicator of river health. In the Mara River physico-chemical water quality parameters under different land have been investigated and how they affect spatial distribution of benthic macroinvertebrates (Kilonzo et al., 2014; Minaya, MClain, Moog, Omengo, & Singer, 2013). Further, in the Mara River classification of shredder using gut contents has been carried out at different pollution gradients (Masese et al., 2013). In the Njoro River, Makoba, Shivoga, Muchiri, and Miller (2008) investigated the influence of seasonality and point source effluent pollution on the water chemistry and the structure of benthic invertebrate. On the other hand, Raburu, Masese, and Mulanda (2009) developed a macroinvertebrate Index of Biotic Integrity for monitoring rivers in the upper catchments of Nyando and Nzoia Rivers. Furthermore, Ndaruga, Ndiritu, Gichuki, and Wamich (2004) established the relationship between water quality parameters and macroinvertebrate assemblages in Getharaini drainage in central Kenya.

Despite a wide range of studies in Kenya dealing with microinvertebrates and land use practices none of them has documented the influence of canopy cover on benthic macroinvertebrate composition. Our study aimed at determining the relationship between riparian vegetation cover and benthic macroinvertebrate assemblage attributes as an ecosystem measure of river health in the Kuywa River. This knowledge is important to the community and water resource managers as it will yield information on effect of planted riparian zone vegetation on protecting the river health which may lead to the replication of the same project in other watersheds.

2. 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	Acronym	LAT	LONG	ALT (M)	Land use	Local watershed erosion	Category of vegetation
Alumuli	A	0.58395	34.6908	1440	Sugar cane plantation	Moderate	Sugar cane plantation
Kibingei	KG	0.73628	34.68845	1534	Agricultural, eucalyptus bank vegetation	Heavy	Planted eucalyptus
Kibisi	KS	0.75534	34.65931	1533	Agricultural	Heavy	Natural conserved
Kuywa Market	K1	0.75068	34.6403	1548	Agricultural	Heavy	Planted mature

Kuywa Nakoyonjo	K2	0.78208	34.6121	1574	Agricultural, open grazing	Heavy	Planted mature
Teremi	T1	0.81716	34.58682	1956	Grazing	Heavy	Fenced and retired from grazing
Emia	E	0.82473	34.58234	1970	Agricultural	Moderate	Planted young
Teremi confluence	T2	0.8244	34.58379	1960	Agricultural	Moderate	Planted mature
Kimurio	KM	0.8873	34.58733	2304	Forest, open grazing	Moderate	Natural conserved

Nine sampling sites were objectively identified (Raburu et al., 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 (M. O. 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**

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.

3. DATA ANALYSIS

To determine the relationship between percentage planted riparian zone vegetation cover and benthic macroinvertebrate assemblage attributes, sites with planted riparian vegetation were compared with the control sites. Pearson rank correlation was used to show which species had a significant relation with percentage riparian vegetation cover. Spatial characteristics were obtained by averaging for rounds of field sampling, representing both dry period and wet period. This was accomplished by employing descriptive statistics which involved computing the Richness Index (S), Abundance Index (N), Margalef Richness (d) (Margalef, 1956), Shannon Index (H) (Maguran, 1988), Simpson diversity (λ) and Pielou evenness for benthic macroinvertebrates at different sites. Inferential statistics was also performed to determine Bray-Curtis similarity measure. The similarity was visualised using Multi-dimensional Scaling (MDS) in Primer v6, (Clarke, Somerfield, & Chapman, 2006) on benthic macroinvertebrates when factored with percentage riparian vegetation cover. To test the hypothesis, whether variations between the sites are significant, ANOSIM analysis was applied. Further, the effect of planted riparian zone vegetation was tested by the percentage of EPT.

4. RESULTS

Using the four criteria developed for the quantification of percentage riparian zone vegetation cover (Table 2), along with photographs taken in the field (Figure 4), percentage riparian zone vegetation cover scores were obtained (Table 5). The scores ranged between 1 and 4, whereby 1 indicated a "poor" riparian condition, and 4 an "excellent" riparian zone condition.

Table 3: Summary of riparian zone vegetation classification scores and resulting site riparian classification for the nine sites sampled along the Kuywa River.

Sites	A	KG	KS	K1	K2	T1	T2	E	KM
% planted riparian vegetation cover score	3	1	3	3	4	1	3	3	4
Category of vegetation	Sugar cane plantation (PS)	Planted eucalyptus (PE)	Natural conserved (PY)	Planted mature (PM)	Planted mature (PM)	Fenced and retired from grazing (F)	Planted young (PY)	Planted mature (PM)	Natural conserved (NC)
Site classification	Good	Poor	Good	Good	Excellent	Poor	Good	Good	Excellent

Two sites, Site KM and Site K2 had "Excellent" site classification. These sites had more than 30m of riparian covered with vegetation. Site KM was naturally conserved while Site K2 was rehabilitated by the community and fenced off to avoid the interference by the animals. Sites with intermediate score of and thus classified as "Good" included A, KS, K1, T2 and E. These sites had patchy vegetation with some spots within 100m ridge not well covered with vegetation or not reaching 30m as given by the Water Act, 2002. The sites which scored lowest were Site A and Site T1. Site A was covered by exotic trees (eucalyptus trees) however, some sections at the upstream of the river stretch was covered by indigenous vegetation. Site T1 was recently been fenced off, but still not recovered from the effects of animal grazing.

The different site classes had distinct canopy cover especially according to the type of vegetaion and its structure. However, when the assessment was done on the relationship between the canopy cover and the benthic macroinvertebrates, only five species has a significant correlation as indicated in Table 4.

*Table 4: Correlation between percentage Canopy Cover and species abundance in Kuywa River. * means statistically significant, ** very statistically significant at 95% confidence. Other species were not significantly correlated*

	Canopy Cover r	ρ
Hexatoma	0.83**	0.003
Belostoria	0.73*	0.013
Ariacalis	-0.63*	0.034
Simulium	0.62*	0.037
Notonectidae	-0.69*	0.02

At 95% confidence, Hexatoma (Diptera) was very positively correlated (r=0.83, ρ=0.003) with canopy cover, while Belostoria (Hemiptera) and Simulium (Diptera) were just positively correlated (r=0.73, ρ=0.013 and r=0.62, ρ=0.02 respectfully). We also established that Ariacalis (Plecoptera) and Notonectidae (Hemiptera) were negatively correlated with canopy cover (r=-0.63,ρ=0.034 and r=-0.69,ρ=0.02) respectfully.

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.

The analysis of taxonomic composition revealed that, the two orders, Ephemeroptera (32.3%) and Diptera (53.2%) dominated the sites. 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, whereby Simulidae was the most abundant genus possessing 46.6% at site KM.

The highest taxonomic richness was recorded at KG (34) and T2 (35) while the lowest was recorded at A (Table 5). Low value Shannon diversity index was recorded at K1(0.89). In both Shannon and Simpson diversity indices, site E (2.55) recorded highest values followed by T2 (2.51) and KS(2.51). The lowest scores were awarded to A(1.52) and K1(0.89). It was surprising that the reference site KM was the third lowest in both diversity indices. As the trend in diversity also in evenness and richness, A and T1 scored the lowest followed by the reference site KM. However, KG (5.75) was found to have the highest richness score followed by T2 (5.40). On the side of evenness, KS (0.73) scored the highest value followed by E (0.72).

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'(E_H)	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; E_H=Shannon evenness; 1-λ=Simpson diversity.

To test the null hypothesis that there are no assemblage differences between the sites with excellent, Good and Poor riparian vegetation cover, ANOSIM (analysis of similarity) was used and gave the result, Global R=0.94 and p=0.037 ($p<0.1$). Thus our study rejected the null hypothesis for there was significant difference between the sites in terms of macroinvertebrate assemblages. The role of individual species in contributing to the dissimilarity of these nine sites with different riparian vegetation cover was implemented in the SIMPER (Similarity Percentages) procedure and scores separating sites classified as 'Poor' and 'Excellent' and those classified as 'Poor' and 'Good' are tabulated in Table 6 and Table 7.

Table 6: SIMPER scores for group Poor and Excellent. Average dissimilarity was 34.92

	Group Poor	Group Excellent				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Simulium	2.9	6.51	4.49	4.29	12.85	12.85
Gompus	2.54	1.35	1.5	1.43	4.29	17.14
Baetis	6.3	5.14	1.45	2.06	4.15	21.29
Macrobdella	0.41	1.08	1.35	1.21	3.88	25.16
Afronurus	1.77	1.05	1.25	1.55	3.58	28.74
Leptophlebiidae	0.9	0.98	1.12	1.73	3.21	31.95
Lepidostoma	1.52	0.8	1.11	1.14	3.19	35.14
Elmnae	0	0.76	0.94	10.32	2.71	37.84
Haplogenis	1.84	1.13	0.88	4.36	2.51	40.36
Caenis	1.27	0.6	0.84	3.4	2.41	42.77
Tricorythus	2.18	1.54	0.8	1.34	2.3	45.06
Oligoneuridae	0.61	0	0.78	0.87	2.22	47.29
Megalagrion	0.54	1.16	0.76	1.01	2.19	49.48

Table 7: SIMPER scores for group Good and Poor. Average dissimilarity was 40.61

	Group Good	Group Poor				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Simulium	4.95	2.9	3.66	1.05	9.01	9.01
Baetis	4.19	6.3	2.76	1.68	6.79	15.8
Elassoneuria	1.45	0.74	1.7	1.15	4.18	19.98
Afronurus	1.25	1.77	1.42	1.4	3.49	23.47
Gomphus	1.92	2.54	1.39	1.16	3.41	26.88
Chironomous	2.9	2.36	1.32	1.16	3.24	30.13
Tricorythus	1.21	2.18	1.31	1.19	3.22	33.34
Meso	0	0.96	1.21	2.16	2.98	36.32
Leptophlebiidae	0.54	0.9	1.11	1.17	2.73	39.05
Lestes	1.76	2.01	1.07	1.6	2.64	41.7

Ephemerella	0.92	0	1.07	0.53	2.63	44.32
Lepidostoma	0.94	1.52	1.01	1.2	2.48	46.81
Haplogenis	1.27	1.84	0.99	1.41	2.45	49.25

In Table 6, the average of the Bray-Curtis dissimilarity between the pairs of the nine sites was 34.92. This was found to be made up of 4.49 from Simulium (Diptera), 1.5 from Gomphus (Odonata), 2.06 from Baetis (Ephemeroptera) and the rest had insignificant contributions. The Simulium contributed 12.85% of the total of 34.92, Gompus gave 4.29% of this total and Baetis gave 4.15% of the total. Simulium declines strongly in abundance in poor vegetation cover (6.51 to 2.9), whereas, Gompus increases in poor vegetation cover (1.35 to 2.54). Further, in Table 7, the average Bray-Curtis dissimilarity between all nine pairs of sites in the group was 40.61, made up of from Simulium (3.66 i.e. 9%), Baetis (2.76 i.e. 6.79%), Elassoneuria (Ephemeroptera) (0.74 i.e. 4.18%) and the rest being less than 3.4% contribution. However, the bubble plot for the MDS for specific genus indicated patterns in Figure 1.

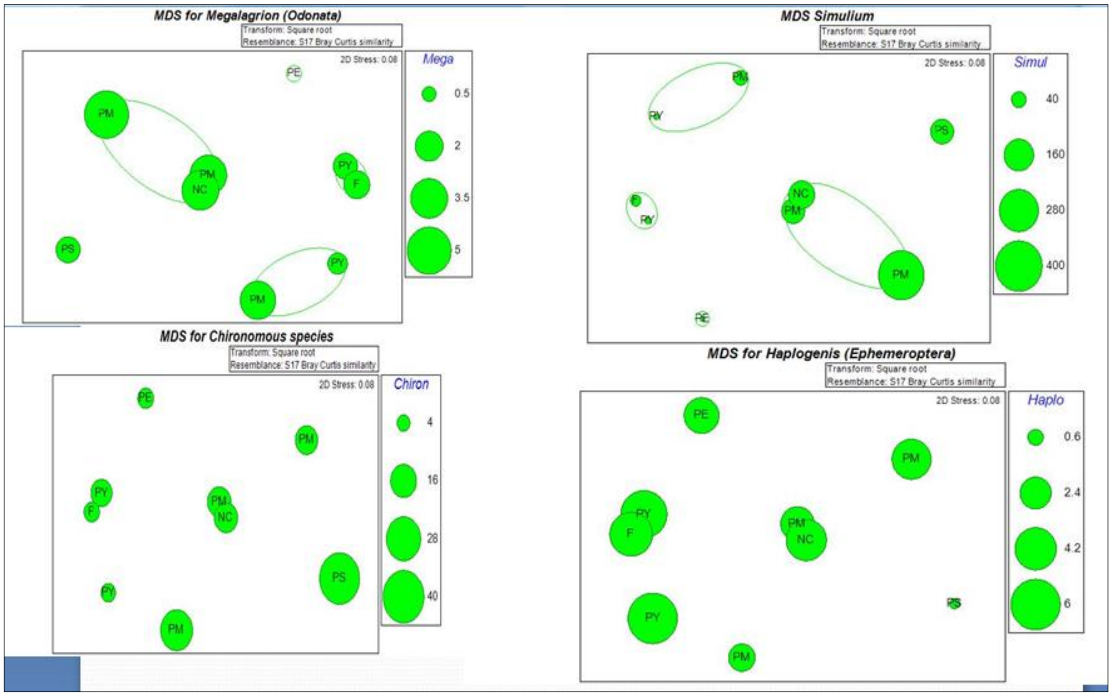

Figure 1: Bubble plot for Megalogrion, Simulium, Haplogenis, and Chironomous. PM=planted mature vegetation, PY=planted young vegetation, PE=planted eucalyptus, NC=Natural conserved vegetation, PS=planted sugarcane, F=fenced off from animals

MDS for Megalagrion (Odonata) species indicated that sites with mature planted vegetation cover and those with natural conserved clustered together had a greater abundance of individuals than the sites which had young planted, planted sugarcane, planted eucalyptus trees and that which was fenced off. The same abundance trend was found in Simulium genus (Diptera). On the other hand MDS for Chironomous (Diptera) species included the riparian with planted sugarcane in the same cluster with mature and natural conserved riparian vegetation cover. MDS for Haplogenis (Ephemeroptera) was opposite of Chironomous in that planted sugarcane riparian vegetation cover had the least macroinvertebrates compared to other sites. Most of the genus were found not to be very sensitive to this metric. Nevertheless, the clusters generated above were tested whether they were statististically significant clusters from each of a number of sites. SIMPROF (similarity profile) performed gave Pi=3.215 and p=0.001, therefore the null hypothesis rejected (Figure 2).

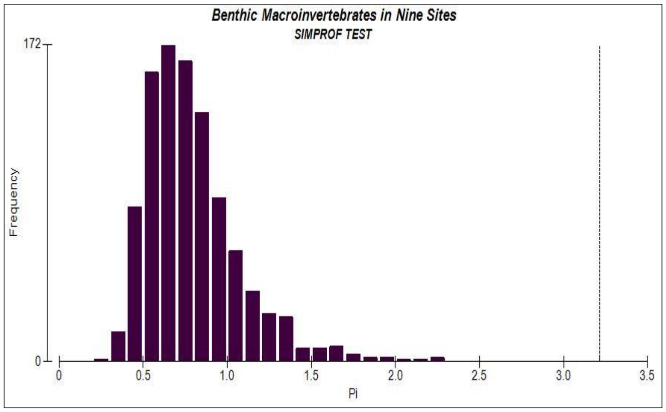

Figure 2: Macroinvertebrates Kuywa River. Simulated distribution of the test statistic Pi under the hypothesis Ho of no site differences within each riparian vegetation cover: the observed Pi is 3.215 at 0.001 confidence.

To assess the effect of riparian vegetation cover on sensitive taxa, EPT indices were applied on the nine sampling sites and the results are presented in Figure 3, 4 and 5.

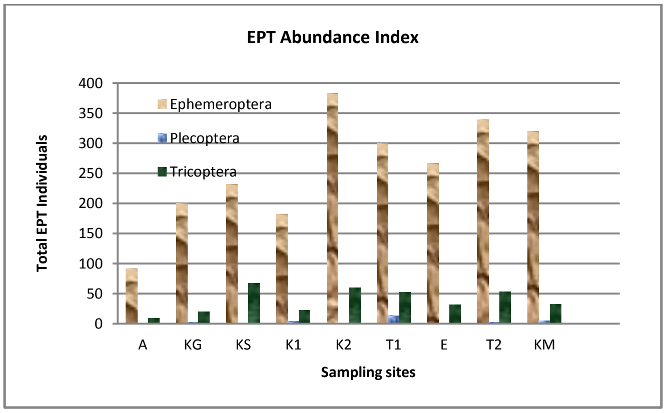

Figure 3: Ephemeroptera, Plecoptera and Tricoptera abundance for the nine Kuywa River sites during the study period January-October, 2016

In the nine sites of Kuywa River, we collected 2,687 (36% of individuals collected) EPT individuals distributed in 22 genus (Table 8). K2 was found to have the highest number of individuals in EPT, 444 (16.5% of total EPT) which comprised of 383 Ephemeroptera, 60 Tricoptera and only one Plecoptera (Figure 3). The reference (control) site KM, had 357 (13.2% of EPT) individuals made of 320 Ephemeroptera, 32 Tricoptera and 5 Plecoptera. It was unexpected that site A which had good riparian vegetation cover sustained the least number of EPT compared to other eight sites. The total individual in site A was 100 (0.04%) comprising of 91 Ephemeroptera, 9 Tricoptera and zero Plecoptera. Also K1 had second least number of EPT individuals, 208 (0.08%) comprising of 182 Ephemeroptera, 22 Tricoptera and 4 Plecoptera. K1 had mature planted riparian vegetation which appeared to be excellent in terms of vegetation cover. T1 which was fenced off from animal interference had a fair representation of Ephemeroptera (300), Plecoptera (13) and Tricoptera (52)(Figure 3).

The EPT richness index in the nine sites ranged between 0 and 8 (Figure 4). T1 had a fair balance of the EPT richness (7,2,5 respectfully) compared to the other sites. The reference site KM had the richness of Ephemeroptera (6), Plecoptera (2) and Tricoptera (3) (Figure 4). Site A and E had no Plecoptera. In general there were more taxa of Ephemeroptera followed by Tricoptera and Plecoptera was rare. The same trend as that of EPT richness appeared in the family richness whereby Ephemeroptera was the most rich in families followed by Tricoptera and then Plecoptera (Figure 5). Sites KG and E had the highest Ephemeroptera family richness (9) followed by K1, K2 and T2. The reference site had Ephemeroptera family richness of 6. Tricoptera families were more in KS (6) followed by KG, K1 and T2 each having richness of 5. There were no Plecoptera families in site A and E.

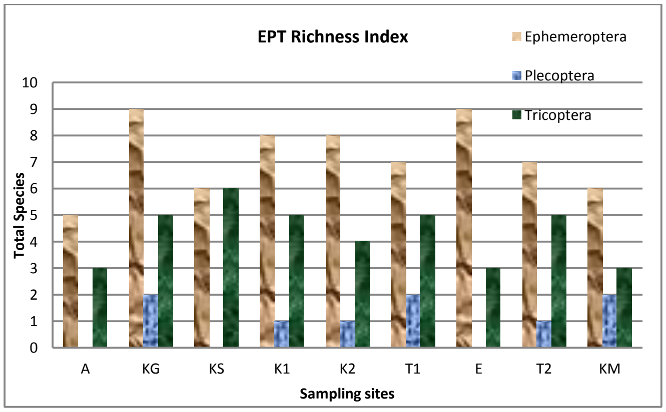

Figure 4: Ephemeroptera, Plecoptera and Tricoptera richness index for the nine Kuywa River sites during the study period January-October, 2016

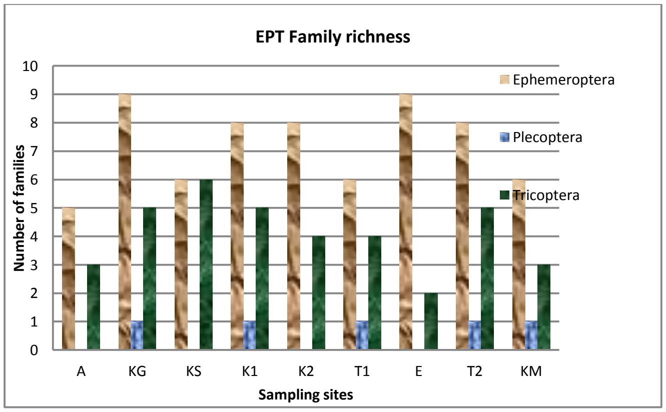

Figure 5: EPT genus richness comparisons among 27 subsamples of the 9 sample sites

5. DISCUSSION

Our results suggested an influence of riparian vegetation cover on benthic macroinvertebrate assemblages in the Kuywa River. Further the results suggested a differential health status of the Kuywa River as per the riparian vegetation cover. Headwater near pristine, site KM, Showed 'Excellent' classification at the site scale. This was as expected because this site was just outside the conserved national park. Site K2 which was also 'Excellent' had planted riparian vegetation which was mature and fenced well avoiding the grazing of animals. Site KS,T2, K1,and E had vegetation but not continuous either due to human interference of still too young to exert the influence. This may be as a result of trees and shrubs taking too long to cover the ground (Sheldon et al., 2002; Sponseller, Benfield, & Valett, 2001) and mimic the natural habitat. Even the rehabilitated riparian normally have the foot-prints of human interference indicated by habitat modification (Sanchez-Arguello, Cornejo, Pearson, & Boyero, 2010). The 'Poor' sites were site T1and A due to having the exotic species of vegetation not preferred by the benthic macroinvertebrates. Further, exotic species of vegetation have been found not allowing indigenous undergrowth whose allochthonous organic matter afford habitat and food for aquatic macroinvertebrates(Monoury et al., 2014).

Not all species of aquatic benthic macroinvertebrate respond the same with the type of riparian vegetation cover. Hexatoma (Diptera) which was highly positive correlated (R=0.83), and Simulium (Diptera) and Belostoria (Hemiptera) positive correlated (r=0.62 and r=0.73 respectfully) are very sensitive to oxygen concentration and like clear water. Further, Belostoria and Hexatoma are very intolerant to any form of pollution. The sites with higher density of riparian vegetation cover, had higher canopy cover which lowered the water temperature and thus increasing oxygen concentration. Lower temperature and high oxygen are important conditions that support diverse aquatic organisms (Narangarvuu, Hsu, Shieh, Wuc, & Yang, 2014). Similarly, sites which had low riparian vegetation cover and thus low canopy cover might have had microbial growths which deprived water of oxygen making these macroinvertebrates impossible to survive. This finding is similar to Bourque and Pomeroy (2001), Findlay, Quinn, Hickey, Burrell, and Downes (2001), and Stauffer, Goldstein, and Newman (2000) who established that riparian clearing/canopy opening reduces shading, causing increases in stream temperatures, light penetration, and plant growth. Conversely, Ariacalis (Plecoptera) and Notonectidae (Hemiptera) were negative significantly correlated as they have a wide ecological tolerant to stressors. Most of the species were not significantly correlated which might be attributed by the frequent habitat disturbance which eliminate intolerant species from the river (Dudgeon et al., 2006).

Although the species different at various study sites, our study established the total benthic macroinvertebrates captured to be 7,444 (with 108 samples) belonging to 73 taxa. The abundance was comparable with other Kenyan Rivers. The study done by F. O. Masese, Muchiri, and Raburu (2009) in Moiben River which is also a tributary of Nzoia like Kuywa established a total of 7,333 individuals belonging to 70 taxa with 108 samples and Oruta (2016) established a total of 2,970 individuals from 57 samples on Sosiani River which is also a tributary of Nzoia River. Further, the dominance of order Diptera in Kuywa River, could probably be attributed to the presence of leaf litter and other course particulate organic matter (CPOM) in some sites such as KM and K1 and thus favoured the flourishing of Chironomidae family. Moreover, Ephemeroptera majority were from

Baetis genus who are scrappers, and the presence of algae especially at the sites with less canopy cover, favoured the flourishing of algae and thus supporting Baetis species of benthic macroinvertebrates. However, the less number of families such as Tipulidae, Potamonautidae and Lepidostomatidae may have been due to reduced riparian vegetation cover especially at sites T1, KS and KG, since they are shredders. These families had favourable temperatures as they are adapted to cold water and the tropical highlands are close to their thermal maxima (Baxter, Fausch, & Saunders, 2005).

The high values of taxonomic richness in sites T2 and KG might be due to the activities being undertaken on the upstream catchment and the characteristics of riparian vegetation cover. These two sites has less human captivities on their catchments. Furthermore, these two sites had continuous riparian vegetation for many kilometres and thus good connectivity for the organisms even when there is a disturbance. Studies carried out on the upper catchment of Nyando River (Orwa et al., 2013) established that, riparian vegetation cover has great impact on temperature and nutrient levels in a stream which consequently determines the integrity of the river system. In contrary, site A whose catchment was the sugar cane plantation had minimum taxonomic richness due to the nutrients emanating from the use of fertilizers on farms, which lead to competition of in-stream oxygen (Oruta, 2016) and thus reducing the sensitive species from the site. The low taxonomic value for the reference site KM may have come from the disturbance of wild animals about 150meters upstream of the sampling site.

The test of hypothesis indicated that there was significant benthic macroinvertebrate assemblage differences between the sites which had 'Excellent', 'Good' and 'poor' riparian vegetation cover ($p<0.1$). Simulium which feed on fine particulate organic matter (FPOM) from the water column using variety of filters, Gompus which are predate on other consumers and Baetis which are scrappers and consume algae and associated materials differentiated the sites between the 'poor' and 'Excellent'. This differentiation of sites as per macroinvertebrate assemblages may have been associated with the linkage that exist in riparian dominated headwater streams between course particulate organic matter (CPOM) and shredders and FPOM and collectors, and primary production and scrappers. This was evidenced by Leptophlebiidae and Lepidostoma species which are shredders being among the highest contributors (3.21% and 3.19% respectfully) to the dissimilarity of sites. These two species groups are very sensitive to water quality and can survive in good water quality only (Bunn, 1999). On the other hand the groups 'poor' and 'Good' were dissimilar (40.61) being differentiated by scrapper feeders (Baetis, Ellassoneuria, and Afronurus), Filter (Simulium) and gatherer feeders (Chironomous and Tricorythus). These species may have differentiated the sites as those sites with 'Good' riparian vegetation cover prevented sediments and other catchment materials from reaching the river, while those receiving sediments had plenty of gatherers. Other studies have shown that even modest riparian deforestation in highly forested catchments can result in degradation of stream habitat owing to sediment inputs (Sutherland, Meyer, & Gardiner, 2002). A comparison of two small catchments that were less than 3% non-forested with two that were 13% and 22% non-forested found the latter to have higher concentrations of suspended sediments, higher turbidity at baseflow, five to nine times greater bedload transport, and greater embeddedness (Sutherland et al. 2002). However, when this interpretation has been done, care should be taken as materials in the river may be transported from a distance and get deposited in a given site especially when encountered with the obstructions (Findlay et al., 2001). Furthermoer, as discussed above, the sites with closed canopy which were in sites KM and K2 prevented sun light penetration and might have limited the growth of algae (Greenway, 2004; Sheldon et al., 2012) necessary for the survival of scrappers. Nevertheless, the feeding of shredders on riparian litter affects dentrital processing in aquatic systems.

Loss of large woody debris reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage; reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability and community function as evidences by the MDS plot of Megalagrion, Simulium, Chironomous and Haplogenis species. Chironomous and Simulium were more abundant at sites with either mature planted riparian vegetion or naturally conserved riparian vegetation. This may have been caused by Simulium getting enough food from the water column at these sites there is plenty of debris falling from the riparian vegetation into the water column. On the other hand the abundance of Chironomous may have been attributed by the presence of wood debris in the river at natural conserved and mature planted vegetation sites to trap food particles (L. B. Johnson, Breneman, & Richards, 2003; Stauffer et al., 2000). Although some species were sensitive riparian vegetation, Megalagrion and Haplogenis species were found to be tolerant to riparian vegetation absence. The sites with minimal riparian vegetation cover had more of these species which may be as a result of these species having a wider ecological tolerance. The clusters for the tolerant and intolerant species appeared to be statistically significant (Pi=3.215, $p<0.001$) indicating that they did not form by chance. Similar studies have shown that land use that deprives a stream of riparian vegetation affects the attributes of aquatic macroinvertebrates. A study carried out at Upper Wabash River in Indiana (Hrodey, Sutton, Frimpong, & Simon, 2009) revealed that intensive agricultural land use where riparian vegetation had been cleared led to remarkable changes in aquatic macroinvertebrate communities as well as to degraded water quality. Further, this study established that stream macroinvertebrates responded negatively to increased sedimentation and habitat loss.

The sensitive EPT group of benthic macroinvertebrates have been used to indicate the health of rivers/streams. Our study K2 and T2 to having the highest percentage of EPT (16.5% and 13.0%). This might have been due to the less disturbance at these two sites which had good riparian zone vegetation and less intensive agriculture on their catchment. The control site KM had also a comparable number of EPT to that of K2 and T2. Conversely, site A which had also good riparian zone vegetation had the lowest percentage EPT (0.04%), attributed by the effect of agricultural chemicals from the sugarcane plantation. The EPT are sensitive to disturbance and decrease with increase in nutrients levels (Mason, 2002). This was made more evident by the fact that sites A and E had no Plecoptera at all. Site E might have also missed Plecoptera due to high usage of fertilizers and pesticides on horticultural farms which were evident along the river. Site T1 had no grown vegetation but had a fair representation of EPT due to the as a result of being fences off to eliminate the stress exerted by animals. Generally, the relative abundance of the intolerant group in all sites was believed to be influenced by organic matter and the availability of food for consumption. Our study agrees with Aura, Raburu, and Herrmann (2010), Mason (2002) and Orwa et al. (2013) who found similar results and attributed it to the influence of organic matter and food distribution of aquatic macroinvertebrates.

The distribution of EPT species to different sites varied with Ephemeroptera being the richest, followed by Tricoptera and Plecoptera least. This was comparable with similar studies in Kenya (example; Orwa, Raburu, Ngodhe, and Kipkorir (2014) and Oruta (2016)). Furthermore, the EPT species richness was found to balance more in those sites with long coverage of riparian vegetation cover. The adjacent land use together with the riparian vegetation cover had great influence which led to site A, E and KS to lack Plecoptera species. Our result concurs with the one of Genito, Gburek, and Sharpley (2002) who established the decline of aquatic macroinvertebrates in intolerant taxa (e.g. EPT) with increased agricultural land cover. However, due to the high complexity of riparian vegetation and its services to ecosystem, there is always high colonization rates, which in turn support abundant and diverse aquatic macroinvertebrates (Belsel, Usseglio-Polatera, & Moreteau, 2000). In contrast, Sponseller et al. (2001) observe that the increased filamentous green algae production and the subsequent additional habitat it creates lead to a greater diversity of taxa in agricultural land use streams.

The family richness followed the same trend as the species richness, with KG and E having the highest scores of Ephemeroptera family richness. Some of the Ephemeroptera family members a wide ecological tolerance making them survive in stress prone areas. But still site A had lowest family richness due to the nutrients from sugar cane plantation.

6. CONCLUSION

This study established that, riparian vegetation cover had an influence on benthic macroinvertebrate assemblages. The sensitive species were found to be significantly correlated with the riparian vegetation cover type. The sites good vegetation cover and well maintained riparian vegetation supported more species and taxa compared to those with poor vegetation. However, the adjacent land use also affected the assemblages with those sites with more agricultural inputs having poor representation of species. Furthermore, our study established that there was a significant difference in macroinvertebrate assemblages between the sites which had 'Excellent', 'Good' and 'Poor' riparian vegetation cover. Sites with 'Poor' riparian vegetation cover suffered from lack of large woody debris which reduced substrate for feeding, attachment, and cover; caused loss of sediment and organic material storage; reduced energy dissipation and therefore impacted on the distribution of habitats. Our study concluded that, for management and restoration actions to be effective, we must diagnose cause as well as assess harm, which requires an improved understanding of the mechanisms through which land use impacts stream ecosystems. This may necessitate studies to examine the response of individual species, family and order to better connect the chain of influence from land use to stream response. Such innervations are of particular importance since riparian management has a direct influence on stream condition via well-documented path- ways and because it promises benefits that are highly disproportionate to the land area required. Such management should ensure continuity of the riparian vegetation cover to provide migratory routes for the colonizers and limit gaps where storm water may drain into streams bypassing the riparian zone and diminishing its effectiveness. This knowledge is important to the community and water resource managers as it will yield information on effect of planted riparian zone vegetation on protecting the river health which may lead to the replication of the same project in other watersheds.

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