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Macroinvertebrate assemblages as biological indicators of water quality in the Moiben River, Kenya

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Benthic macroinvertebrate assemblages at eight stations in the Moiben River, corresponding to different catchment land uses, were assessed in 2006 as indicators of water quality. The relative abundance per taxon, diversity index, richness index, evenness, dominance, percentage of five dominant taxa and percentage Ephemeroptera + Plecoptera + Trichoptera (EPT) individuals were determined per sampling period per station. Significant spatio-temporal variation was observed in relative abundance, with Diptera dominating the study area. Ephemeroptera, Plecoptera and Trichoptera dominated the headwater stations, whereas Coleoptera, Oligochaeta and Chironomidae dominated further downstream. Significant relationships were recorded between physico-chemical parameters — conductivity, BOD, temperature, and discharge — and the occurrence of specific taxa, mainly *Heptagenia*, *Caenis*, *Baetis*, *Branchiobdella*, *Potamon*, *Ilyocoris*, *Elmis* and *Chironomus*. Significant changes in macroinvertebrate assemblages were primarily due to changes in water quality. As elsewhere, macroinvertebrate communities proved to be good indicators of water quality and should be used as bioindicators in long-term monitoring of this river.

Keywords: bioindicators, freshwater macroinvertebrate communities, Moiben River, water quality

Introduction

Rapid human population growth in developing countries, together with an enormous increase in the amount of waste produced, are placing demands on aquatic ecosystems. Streams and rivers continue to bear the consequences of pollution. This trend, coupled with freshwater scarcity in most countries, including Kenya (Postel 1992), continues to strain available aquatic resources. As a result these delicate ecosystems have increasingly lost their integrity, leading to sedimentation and eutrophication of the receiving water bodies such as lakes and reservoirs (Osano et al. 2003, Raburu 2003, GEF 2004).

As integrators of the effects of land-use practices within their catchments, streams and rivers can help in the diagnosis of the environmental health of the landscapes that they drain (Dallas and Day 1993). Changes anywhere on the landscape that influence rivers are reflected in the composition of resident biota (Rosenberg and Resh 1993, Harding et al. 1999). Aquatic communities are sensitive indicators of pollution as they integrate and reflect the effects of stress, both natural and human induced, over extended periods of time (Rosenberg and Resh 1993, Barbour et al. 1999). Therefore, by assessing the composition of these groups of organisms, the water quality status of streams and rivers can be determined. In so doing they act as early warning signals of pollutant loads that degrade water quality and overall ecological integrity.

Aquatic macroinvertebrates have gained prominence as bioindicators of environmental quality in lotic systems and information about their responses to changes in environmental quality continues to grow (Lenat and Crawford 1994, Baker and Sharp 1998). Their functional and structural composition varies, both spatially and temporally, in relation to environmental factors (Tate and Heiny 1995). These factors include discharge, substrate type, dissolved substances, turbidity, riparian vegetation, land use, temperature, altitude and latitude (Giller and Malmqvist 1998). However, human activities influence the effects of these factors, which in turn affects the composition and distribution of macroinvertebrates.

On the Kenyan side of the Lake Victoria basin, land-use practices are causing widespread degradation of aquatic ecosystems (Raburu 2003, GEF 2004, Okungu and Opango 2005). Increased agricultural activities in the Nzoia River Basin, (GEF 2004), the largest catchment in the Kenyan section of the Lake Victoria basin, has caused an increase in nutrient enrichment, pesticide contamination and sedimentation, not only of the streams and rivers (Osano et al. 2003) but also of Lake Victoria (Okungu and Opango 2005). In the face of changing and intensifying human activity in catchments draining into the lake, there is a need to assess the current status of water quality in the rivers and to test protocols for future monitoring. This study was therefore designed to generate information that could be used to strengthen management strategies for the Nzoia River.

Methods

Study area

The Moiben River Basin, 35°06′–35°34′ E and 00°37′–00°62′ N, with an estimated area of 1 050 km² (Gok 1973) (Figure 1), is part of the upper catchment of the 12 903 km² Nzoia River watershed. The Moiben River originates on the western side of the Kerio escarpment at 2 400 m asl. The river is approximately 81 km long from its source in the Kipkunnur forest to its confluence with the Kapolet River, where they join to form the larger Nzoia River (GoK 1973). The altitude of catchment boundaries in the highland areas varies between 2 400 and 1 500 m asl. The watershed has a highland equatorial climate with diverse relief features. The mean annual rainfall in the area is 1 124 mm, which occurs in one long season from March to September with two distinct peaks in May and August (Jaetzold and Schmidt 1983). The average air temperature in the region is 18 °C during the wet season, with a maximum of 28 °C during the dry season and a minimum of 7 °C in the coolest season. February is the hottest period, while June to July is the coldest (Jaetzold and Schmidt 1983).

The land-use systems and practices in the basin broadly range from forestry, small-scale farming to large-scale mechanised agriculture. The basin is an area of high agricultural potential and is densely populated, which influences land use. The river drains a forested area at its upper reaches before entering a valley where mixed farming is practiced. Station M1 (the most upstream sampling station) is located in a forested area (Figure 1) where human impacts are minimal. This site was therefore selected as a reference point. Other sampling stations (M2–M6) were impacted in various ways, as summarised in Table 1.

Sampling design

Triplicate samples of physico-chemical water parameters and macroinvertebrate assemblages were collected on a monthly basis at each station for six months from February to July 2006, covering part of the dry season (February to March) and part of the wet season (April to July).

Physico-chemical parameters

On each sampling occasion, physical and chemical parameters were measured before macroinvertebrates were sampled. Conductivity was measured *in situ* using a conductivity meter (OAKTON®, Model WD-35607-10, Singapore) while temperature and pH were also measured *in situ* by a combined pH/temperature-meter (OAKTON®, Model pH/Mv/°C METER, Singapore). Water velocity was determined as an average of velocities taken at three points across the river by the floatation method (Herschy 1978). Water depth and width were also measured at three equally-spaced points along a 10 m section of river and averages were calculated, which were used in determining discharge. Discharge was then calculated as a function of cross-sectional area (width x depth) and water velocity. Dissolved oxygen (DO) and biological oxygen demand (BOD) were determined using the Winkler method (APHA 1992).

Macroinvertebrates samples

At each station a 100 m reach of stream was selected for sampling. Macroinvertebrates were collected from three microhabitats (pools, riffles and runs) that were identified according to Jeffries and Mills (1990). To avoid bias due to spatial variations or patchiness, three random samples were collected from each of the three microhabitats by establishing a transect at each sampling reach with five equally-spaced points from which a sampling point was selected using random numbers. This procedure was replicated three times for each microhabitat, making nine samples per reach. Sampling was done using a 0.09 m² surber sampler with a 250 μm mesh size and samples were pooled to make one composite sample per habitat per station. The samples were preserved in 10% formal-dehyde solution prior to transporation to the laboratory for sorting.

Laboratory sample processing

In the laboratory, samples were washed through a 250 μm mesh sieve, sorted, and counted using a stereo microscope. They were identified to the lowest taxomic level possible, mostly genus, according to Macan (1977), Scholtz and Holm (1985), Merritt and Cummins (1996), Nilson (1996, 1997) and Verschuren (1997); taxonomic lists of species known to be present in Kenya (e.g. Johanson 1992, Mathooko 1998) were also helpful. Voucher specimens were preserved in 75% ethanol and stored with the Moi University Department of Fisheries and Aquatic Sciences collections.

The macroinvertebrate assemblage composition was determined for each sampling station and sampling occasion using number of taxa (*S*), total number of individuals, and relative abundance of each taxon. The Shannon-Wiener diversity index (*H′*) as described by Magguran (1988) was used to assess diversity as follows:

$$H' = \sum ((n/N) \times \ln (n/N))$$

where n = number of individuals of a taxon, and N = total number of individuals in the sample. An associated evenness H'/H'_{max} (Pielou 1975) was also calculated. As an extra measure of evenness, the percentage of total numbers accounted for by the five most abundant taxa at each station was also used (Barbour et al. 1999). To assess compositional differences among sites, the percentage of intolerant taxa — Ephemeroptera + Plecoptera + Trichoptera (EPT) — which is widely used as an indicator of disturbance to stream communities (Lenat and Crawford 1994), was also calculated.

The Simpson Index (D_s) (Simpson 1949) was used as a measure of taxon richnesss. The index is given by:

$$D_S = \sum_{i=1}^{i=n} \frac{\{ n_1 (n_1 - 1)\}}{\{ N (N - 1)\}}$$

where n_1 is the number of species in the sample and N is the total number of individuals in the station.

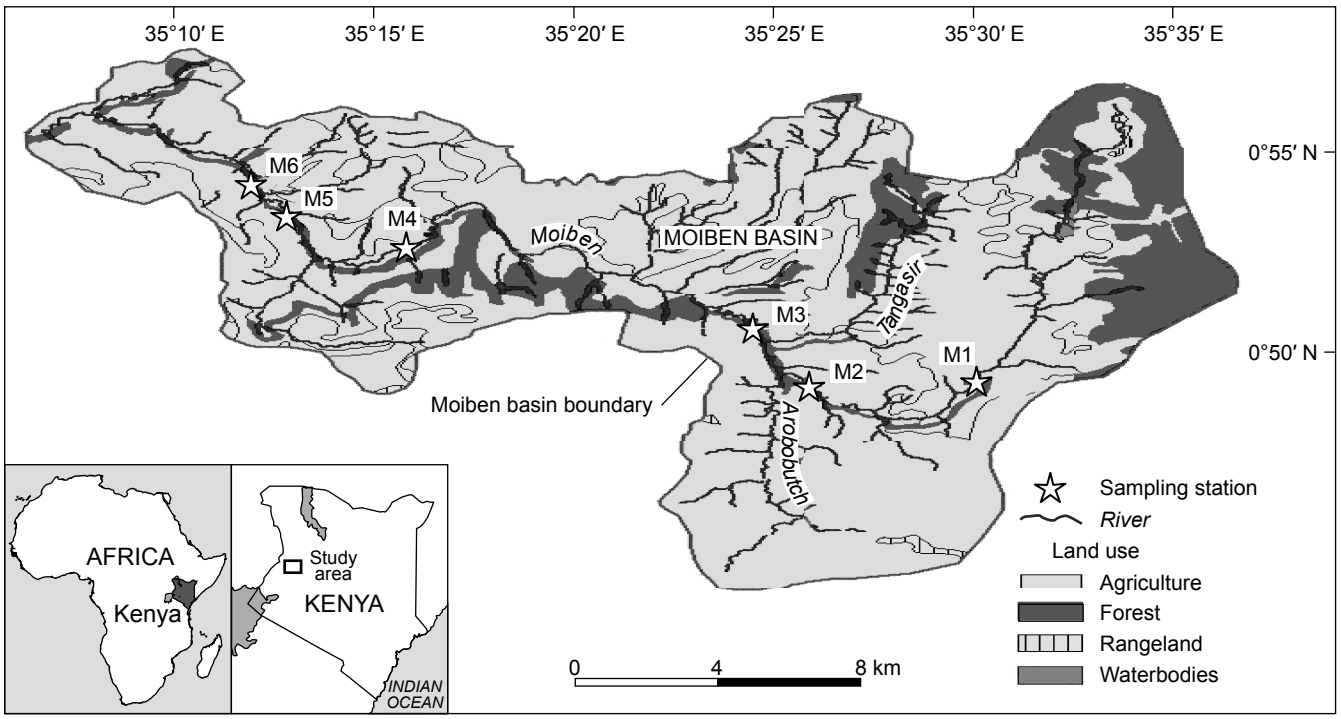

Figure 1: Land-use map of the Moiben watershed and positions of sampling stations (M1–M6) in the Moiben River

Table 1: Sampling stations along the Moiben River, with details of site characteristics and human impacts (see Figure 1 for map of localities)

Sampling station no.	Site description and land use in immediate catchment area
M1	The most upstream site, located in a forested area; minimal human impacts (reference site)
M2	In a region with semi-intensive, small-scale mixed farming; intensive use of animals on the banks, which were eroded and devoid of marginal vegetation
M3 and M4	Areas of intensive maize farming; pockets of forestry also common; Station M4 had stable banks with marginal vegetation and the riparian zone was >15 m wide.
M5 and M6	Intensive maize farming and animal production in the riparian zone; at Station M6 additional human impacts included sand-winning and waste-dumping from a nearby shopping centre

Statistical analysis

Physico-chemical parameters were expressed as means ± SE for each sampling station. All macroinvertebrate count data were $\log_{10}(x+1)$ transformed to meet the statistical criteria for normality. One-way analysis of variance was used to test for differences between stations for each parameter. Multiple comparisons of means were done *post hoc* using Duncan's Multiple Range Test (DMRT) (Zar 2001) to distinguish the stations and sampling occasions that differed significantly from one another. Community indices were used to compare diversity, richness, evenness and dominance of macroinvertebrates between the different stations and sampling occasions. Pairwise Spearman's rank correlation analysis (Zar 2001) was performed to investigate the relationship between community attributes and physico-chemical parameters. Data analysis was done using SPSS for Windows (Version 13.0, SPSS Inc. Chicagio, Illinois) and all significant differences for all inference tests were accepted at 95% confidence level.

Results

Physico-chemical parameters

Mean values obtained per station for physico-chemical parameters are presented in Table 2, and temporal variability is presented in Figure 2. Conductivity, velocity, temperature, discharge and river width showed significant variation in both space and time ($p < 0.05$) while depth, DO, and BOD showed significant variation only with respect to time ($p < 0.05$). The pH values varied significantly ($p < 0.05$) temporally, but not spatially.

Conductivity increased downstream, with Stations M5 and M6 registering the highest values. Highly significant differences ($p < 0.001$) were observed between sampling occasions, with sampling occasion 1 (February) differing from the others. There was a general decline in conductivity over time. The lowest temperature was recorded at Station M1 and this differed from the rest of the stations.

Table 2: Summary of the mean (± SE) physico-chemical properties of the study stations on the Moiben River, February–July 2006

	Station					
	M1	M2	M3	M4	M5	M6
Physical characteristics						
Air temperature (°C)	23.3 ± 2.8	25 ± 2.2	23.6 ± 3.3	23.7 ± 0.8	25.2 ± 2.2	25.0 ± 2.7
Water temperature (°C)	17.2 ± 0.4	17.9 ± 2.6	18 ± 0.1	18.4 ± 0.8	20 ± 0.9	20.1 ± 0.5
Water depth (m)	0.3 ± 0.08	0.4 ± 0.13	0.3 ± 0.1	0.6 ± 0.21	0.5 ± 0.15	0.6 ± 0.24
Width (m)	2.6 ± 0.2	3.3 ± 0.4	7.5 ± 0.5	4.5 ± 0.3	5.8 ± 0.8	4.3 ± 0.2
Current velocity (m s^{-1})	0.5 ± 0.07	0.6 ± 0.05	0.5 ± 0.07	0.6 ± 0.07	0.7 ± 0.09	0.7 ± 0.09
Discharge (m^3 s^{-1})	0.6 ± 0.3	1.0 ± 0.4	1.3 ± 0.7	1.9 ± 1.0	2.2 ± 1.2	2.2 ± 1.2
Chemical characteristics						
pH	9.4 ± 0.4	9.2 ± 0.7	8.8 ± 0.3	9.9 ± 0.7	9.7 ± 0.5	10.6 ± 0.4
Dissolved oxygen (mg l^{-1})	7.6 ± 0.5	7.8 ± 0.1	7.1 ± 0.8	6.8 ± 0.9	7.3 ± 0.5	7.32 ± 0.5
DO (% saturation)	78.6 ± 5.4	82.2 ± 1.6	75.6 ± 8.0	71.4 ± 8.8	81.1 ± 6.5	79.95 ± 3.8
BOD (mg l^{-1})	1.4 ± 0.3	2.1 ± 0.5	1.6 ± 0.6	1.8 ± 0.5	2.4 ± 0.4	2.2 ± 0.4
Conductivity (μS cm^{-1})	70.3 ± 16.6	83.6 ± 15.1	110.5 ± 20.3	113.2 ± 12.7	150.7 ± 18.3	178.7 ± 27.7

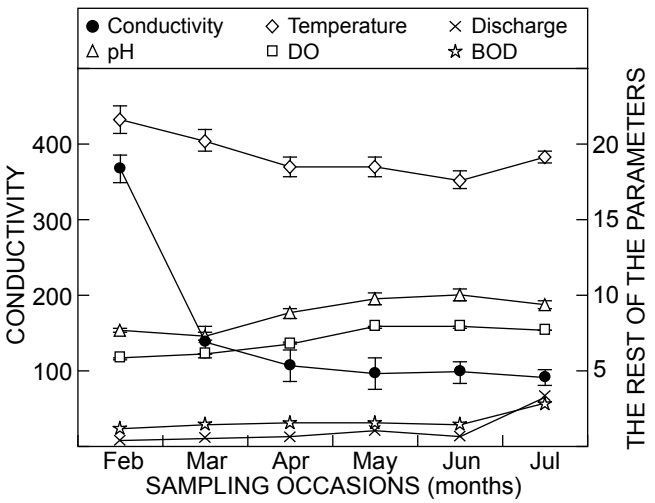

Figure 2: Monthly variation (mean ± SE) in physico-chemical parameters in the Moiben River during the study period, February–July 2006. February and March = dry season, April to July = rainy season (units of measurement for 'Rest of parameters': temperature (°C); pH (no units); DO (mg 1^{-1}); discharge (m^3 s^{-1}); BOD (mg l^{-1})

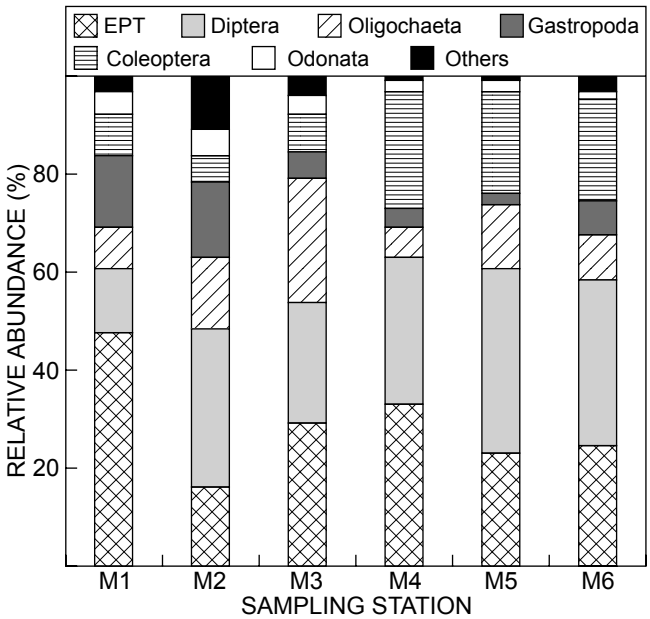

Figure 3: Relative abundance of the main taxonomic groups of macroinvertebrates at each sampling station from February to July 2006

The highest temperature was recorded during February and the lowest in July. The highest pH value was recorded in June 2006 and the lowest in February 2006. The lowest DO was also recorded during February and this differed from the rest of the sampling occasions ($p < 0.05$). In terms of BOD, Station M1 recorded the lowest value and this differed only from Station M5, which recorded the highest value ($p < 0.05$). With respect to time, the highest BOD was recorded on the last sampling occasion (July); this value was significantly different from those measured during all other sampling occasions ($p < 0.05$).

Macroinvertebrate assemblages

A total of 7 333 macroinvertebrate specimens were collected, comprising 70 taxa belonging to 13 orders and 50 families (see Appendix). The relative abundance of taxonomic groups encountered is shown in Figure 3. Diptera was the commonest order in the study area, occurring at all stations. Other orders that were sampled at all stations included Ephemeroptera, Coleoptera, Oligochaeta, Trichoptera, Gastropoda, Odonata and Hemiptera. There were significant differences in abundance between the stations and the sampling periods ($p < 0.001$).

Station M1 had the highest mean abundance per sample (291.7 ± 63.5) while Station M6 had the lowest (128.5 ± 34.1) (Figure 4). A general downstream decline in mean abundance per sample was observed, except at Station M4. There was an increase in the number of genera in the middle stations with Station M4 recording the highest (41) and Station M6 (31) recording the lowest (Figure 4). Plecoptera were not represented at Stations M2 or M3 (Table 2). Hirudinea occurred at Stations M1, M2 and M5, while Crustacea did not occur at Station M6. Arachnida

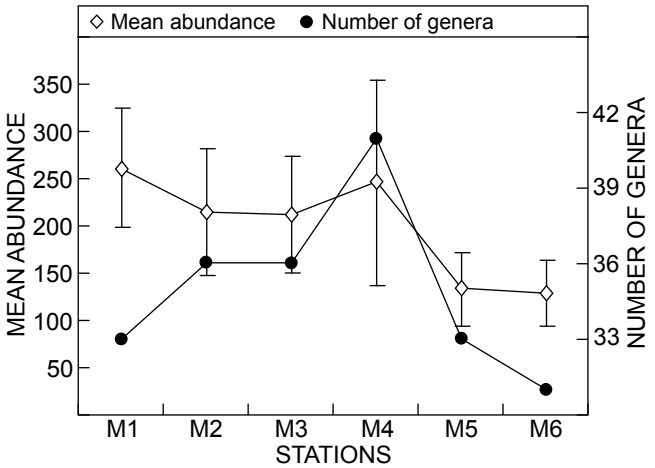

Figure 4: Spatial variation (±SE) in mean abundance and number of genera (per 0.09 m^2) among the sampling stations during February to July 2006

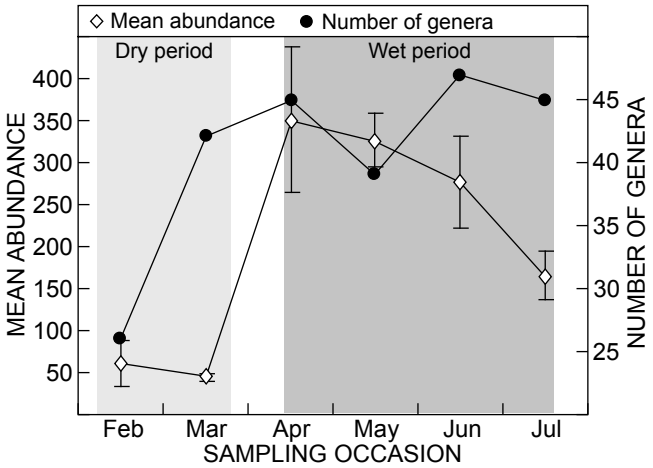

Figure 5: Temporal variation (±SE) in mean abundance and number of genera (per 0.09 m^2) among the sampling occasions during February to July 2006

Table 3: The diversity measures (±SE) of macroinvertebrate communities at the study stations on the Moiben River, February–July 2006

	Station					
Community attributes	1	2	3	4	5	6
Number of genera	33	36	36	41	33	31
Dominance	0.13 ± 0.01	0.15 ± 0.01	0.19 ± 0.03	0.15 ± 0.2	0.32 ± 0.11	0.18 ± 0.03
Richness index $1/D_S$	9.67 ± 1.6	8.23 ± 1.3	6.86 ± 1.3	8.4 ± 1.6	6.16 ± 1.1	6.57 ± 0.5
Diversity index (H')	2.54 ± 0.1	2.15 ± 0.1	2.16 ± 0.1	2.24 ± 0.1	1.9 ± 0.2	2.1 ± 0.1
Evenness index	0.46 ± 0.05	0.46 ± 0.05	0.78 ± 0.4	0.48 ± 0.06	0.41 ± 0.03	0.46 ± 0.03
% 5 most dominant taxa	95.45	87.98	91.07	92.75	94.0	89.46
% EPT	53.09	17.52	30.43	33.22	23.21	24.52

were sampled only at Stations M2 and M3 while Lepidoptera occurred only at Station M1 (see Table 1 for a description of these sampling stations).

With respect to temporal variation in abundance, results from February and March differed from those of April and May, when the highest mean abundances were obtained (Figure 5), but results from June and July did not differ from either of these groups. The number of genera encountered was lowest on the first sampling occasion (February), then rose during March and April, declined in May, rose to its highest levels in June, and then declined slightly in July (Figure 5). During the dry period (February and March) low numbers of pollution-sensitive EPT were collected, while the numbers of pollution-tolerant taxa such as Oligochaeta, Gastropoda and Diptera were very high. As the rains started and increased a succession was evident: EPT, Coleoptera, Hemiptera and Odonata taxa and abundances increased while the abundances of Oligochaeta, Gastropoda and Diptera declined. However, there was an overall decrease in abundance for all taxa during the heavy rains (June and July).

Community attribute results at each sampling site (described in Table 1) are presented in Table 3. The Shannon-Wiener diversity index was highest at Station M1 followed by Station M4 while Station M5 had the lowest value. The evenness index was highest at Station M3 and lowest at Station M5. Mean taxon richness was highest at Station M1 ith 9.67 ± 1.6 followed by Station M4 with 8.4 ± 1.6 while the lowest was recorded at Station M5 with 6.16 ± 1.1. The percentage of the five most dominant taxa was highest at Station M1 with 95.5% while it was lowest at Station M2 with 88%. There were significant differences in relative abundance of EPT spatially ($p < 0.05$) but not temporally. Of the total abundance of macroinvertebrate individuals collected at Station M1, 52.2% belonged to the EPT while a general decline was observed downstream (Table 3).

Relationship between physico-chemical parameters, macroinvertebrate abundance and community attributes

Table 4 shows pairwise Spearman's rank correlations between physico-chemical parameters and macroinvertebrate community attributes. Altitude was negatively correlated with temperature, conductivity, depth, width and discharge, but was positively correlated with abundance ($p < 0.05$). Abundance was negatively correlated with most of the physico-chemical parameters considered, except altitude and pH. Macroinvertebrate abundance was lowest during the dry period in February. It increased progressively during the onset of the rainy season but started to decline during the peak and spates in May. During this period the river was characterised by large quantities of suspended matter and high sediment loads. Discharge had also increased more than ten times. Taxon richness was negatively correlated with

Table 4: Spearman's rank correlation coefficients observed among physico-chemical parameters, and between physico-chemical parameters and selected taxa and community structure attributes. Units are identical to those in Table 2 (* = significant correlation at $p = 0.05$)

Parameter	Altitude	Temp.	Cond.	pH	Velocity	Width	Depth	BOD	Discharge
Altitude		–0.54*	–0.77*	–0.35	–0.23	–0.45*	–0.47*	–0.21	–0.60*
DO	0.24	–0.25	–0.52*	0.27	0.13	–0.08	0.06	0.26	0.02
Dominance	0.04	0.15	0.06	–0.22	–0.03	–0.16	–0.24	0.02	–0.27
Diversity index	–0.01	–0.13	–0.19	0.22	0.13	0.15	0.37	0.09	0.32
Evenness	–0.23	0.07	0.03	0.14	0.20	0.41*	0.55*	0.23	0.60*
Taxon richness	0.35*	–0.35*	–0.53*	–0.10	–0.24	–0.23	–0.42*	–0.28	–0.43*
Abundance	0.61*	–0.47*	–0.38*	–0.26	–0.43*	–0.44*	–0.78*	–0.57*	–0.81*
Caenis sp.	0.31	–0.06	–0.42*	–0.06	–0.20	0.22	0.03	–0.39*	0.02
Baetis sp.	0.09	–0.21	–0.05	–0.03	–0.13	–0.09	–0.39*	0.44*	–0.37*
Heptagenia sp.	0.31	–0.38*	–0.25	–0.29	–0.48*	0.18	–0.22	–0.53*	–0.16
Branchiobdella sp.	0.35	–0.44*	–0.28	0.16	–0.25	–0.55*	–0.66*	–0.24	–0.69*
Glossiphonia sp.	0.34	–0.19	–0.39*	0.02	0.21	–0.53*	–0.32	–0.12	–0.42*
Potamon sp.	0.55*	–0.48*	–0.50*	–0.09	–0.17	–0.16	–0.48*	–0.30	–0.51*
Gomphus sp.	0.58*	–0.47*	–0.54*	–0.14	–0.33	–0.15	–0.32	–0.26	–0.38*
Ilyocoris sp.	–0.45*	0.22	0.48*	0.17	0.12	0.13	0.36*	0.17	0.27
Elmis sp.	0.30	–0.17	–0.13	–0.14	–0.24	–0.44*	–0.55*	–0.33	–0.63*
Chironomus sp.	0.16	0.07	–0.11	–0.12	–0.05	–0.45*	–0.34	–0.01	–0.45*

water temperature, conductivity, depth, BOD and discharge, while it was positively correlated with altitude ($p < 0.05$). Evenness showed a positive relationship with width, depth and discharge.

Among the macroinvertebrate taxa, *Caenis* sp. exhibited a negative relationship with conductivity and BOD. *Baetis* sp. displayed a negative relationship with depth and discharge but a positive relationship with BOD. *Heptagenia* sp. showed a negative relationship with temperature, velocity and BOD. *Branchiobdella* sp. exhibited negative relationships with temperature, width, depth and discharge. *Eriocheir* sp. and *Gomphus* sp. both showed a negative relationship with temperature, conductivity and discharge, but were positively correlated with altitude. *Ilyocoris* sp. was positively correlated with conductivity and depth, but negatively with altitude. *Elmis* sp. showed a negative relationship with width, depth and discharge. *Chironomus* sp. was negatively correlated with width and discharge. Intolerant taxa were negatively correlated with BOD. Discharge influenced most taxa negatively, by reducing their abundance.

Discussion

Physico-chemical variables

The spatial differences in mean BOD along the river could be explained by changes in human activity. The water clarity was very high upstream and it was possible to see the stream bottom, but further downstream there was an accumulation of sediments and other wastes that resulted in higher BOD values, which reached its peak at Station M6. The lowest BOD values, obtained at Station M1, could be attributed to low sedimentation as a result of minimal human impacts. An increase in BOD that registered a peak in July during the rainy season, might have been caused by increased runoff, which transports organic matter and sediments from the catchment into the river (Morris et al. 2003).

The uniformity of DO along the river must have been the result of the mixing of water caused by a significant drop in altitude between the stations (Busulwa and Bailey 2004). Significant differences recorded between the sampling periods could have resulted from changes in water velocity resulting from varying discharge rates.

Water temperature showed both spatial and temporal variation. The significantly lower temperature recorded at Station M1 was due to good riparian vegetation cover at that station. Vegetation cover limits solar radiation reaching the water thus contributing to minimal fluctuations of temperature (Giller and Malmqvist 1998). High temperature during the first sampling period was due to high solar radiation caused by lack of cloud cover and low water volume during the dry period.

The water chemistry differed temporally with respect to most parameters. Conductivity, DO and BOD, which were positively correlated with river width, depth, water velocity and discharge, showed significant variation between the dry and rainy seasons. These findings can be attributed to seasonal effects of non-point sources of pollution, which are mobilised during the rainy season through runoff and leaching, especially from agricultural areas (Sundblad et al. 1994, Moreau et al. 1998, Huber et al. 2000).

Spatio-temporal variations in composition, distribution and community structure

Diptera and Ephemeroptera dominated the study area, accounting for more than 59 per cent of all taxa by numbers. Similar findings have been obtained in highland tropical streams. For instance, Mathooko and Mavuti (1992), while investigating Mount Kenya streams, found the benthic communities to be dominated by *Baetis* sp. (Ephemeroptera: Baetidae) and *Simulium* sp. (Diptera: Simuliidae). Similar results were obtained in studies carried out in the Njoro River, Kenya where Kibichii et al. (2007) found that *Baetis* sp. and Simuliidae comprised 69 per cent, by number, of all benthic taxa identified.

Despite the overall dominance of Baetidae and Simuliidae, marked changes in relative abundance of various taxa were observed downstream. Headwater stations were dominated

by taxa associated with pristine waters and by pollution-sensitive macroinvertebrates such as Ephemeroptera, Plecoptera and Trichoptera, which declined downstream. In Stations M2 and M3 pollution-sensitive taxa were replaced by *Chironomus* sp., *Hydropsyche* sp., *Simulium* sp., *Baetis* sp., Elmidae and Oligochaeta. The high numbers of these taxa can be attributed to organic pollution as a result of enrichment and sedimentation caused by agricultural activities and excretion by livestock in the riparian areas (Buss et al. 2002). High nutrient enrichment and sedimentation have been shown to favour some Chironomidae, net-spinning Trichoptera, Mollusca and Oligochaeta at the expense of Ephemeroptera and Plecoptera (Quinn et al. 1997). Stations M1 and M2 are devoid of riparian vegetation cover and are used as watering points for animals, thus receiving higher amounts of animal and agricultural wastes, compared to Station M1 located in an area with a well-protected riparian zone with limited animal access and other human activities. *Chironomus* sp. dominated Stations M5 and M6 that were characterised by increased animal use in the riparian zones and other human activities like washing, bathing and sand mining. *Chironomus* sp. are known to occur in greater abundance in areas with environmental stress as the genus is able to colonise water with low oxygen concentration because they are able to use haemoglobin as a means of respiring more efficiently during these low-oxygen conditions (Johnson et al. 1993).

High diversity of macroinvertebrates was obtained at Station M4 despite there being point and non-point sources of pollution upstream. This station had well-protected banks with vegetation cover that offered wider habitat diversity to aquatic biota. This explains the high abundance and taxon richness at this station as compared to other stations downstream. This observation concurs with Ruburu's findings during his studies carried out at the Nyando River, where high macroinvertebrate diversity recorded below point sources of pollution was attributed to riparian vegetation cover and instream habitat quality (Raburu 2003).

Apart from anthropogenic impacts, natural stress such as spates and floods may also influence macroinvertebrate assemblage distribution in the tropics. In this study temporal differences in taxon richness and abundance were recorded between the dry and rainy seasons. Low taxon richness was recorded during the dry period and at the onset of the rains and a significant reduction in abundance was recorded during the long rainy season and during spates in July. This was consistent with the findings of Shivoga (200l) in two tropical streams in Kenya where macroinvertebrate abundance was highest at the onset of the rains and declined progressively as the rainfall increased. Although natural conditions influence taxon richness and abundance, in the current study it seems likely that human activities exacerbated the effects of these conditions, leading to the observed distribution and abundance of the various taxa. For instance, land-use activities — relating to road building, agriculture and settlements — influence the quantity of runoff and sediments that enter streams and rivers during the rains (Wang and Lyons 2003); lower runoff levels mean that lower volumes of sediments and nutrients from disturbed land surfaces enter streams. However, during the peak of the dry season, conditions can worsen in streams and rivers because, as discharge declines, pH also declines (as recorded in North American streams: Bowman et al. 2006), temperatures rise and dissolved oxygen becomes limiting. This scenario can explain the temporal and spatial variation in taxon richness, composition and abundance of macroinvertebrates in the study area.

There was no clear trend in taxon richness with distance downstream. Stations at the upper reaches of the river had lower numbers of genera, which increased at Station M4 and then decreased at the lower stations. This is the trend expected in most riverine systems (Vannote et al. 1980) as they reflect changes in stream order and other factors that influence community composition and structure. On the other hand, high diversity at Station M1 can be attributed to good habitat quality and high water quality in the upper reaches of the Moiben River. The forests surrounding Station M1 were a good source of allochthonous organic matter for stream biota. Canopy cover at this station helped maintain low water temperatures and provided diverse habitats for a variety of macroinvertebrates, leading to increased diversity. However, the downstream decrease in diversity can be linked to agricultural land-use intensification in downstream catchment areas (Wang and Lyons 2003).

Ecologically-unimpaired stations could be separated from impaired ones by assessing the composition of their EPT. For instance, the relative abundance of EPT at pristine Station M1 was 53.1% as compared to Stations M2 and M5 with 17.5% and 23.2% respectively, both with degraded riparian zones and instream habitats. This concurs with other studies (Lemly 1982, Baker and Sharp 1998, Raburu 2003) in which low relative abundances of EPT were observed in degraded areas. Despite the sensitivity of Plecoptera to pollution, their representation in the current study area was low, compared to that of Ephemeroptera and Trichoptera. With only two taxa and a total of 20 individuals sampled in the whole study area, plus their absence at Stations M2 and M3, their use as indicators of water quality in the current study area is lmited. Similar low numbers in tropical streams have also been reported (Thorne and Williams 1997). Durand and Leveque (1981), cited in Thorne and Williams (1997), reported only one plecopteran species in the whole of West Africa. However, this order was well-represented in the Nyando River, Lake Victoria basin in Kenya (Raburu 2003) where it formed a larger fraction of the percentage EPT than in the current study. Their low number of genera and abundance in the Moiben River can, therefore, be attributed to degradation. The instream habitats in the Moiben River comprise mostly sandy and muddy bottoms known to affect plecopteran distribution, diversity and abundance (Lemly 1982).

Overall, this study revealed that macroinvertebrate communities responded to changes in water quality along the river. Despite little apparent change in taxon richness observed at the different stations, there were marked shifts in dominance and composition. Headwater stations were dominated by taxa associated with pristine or unimpacted waters with a decrease in the pollution-intolerant genera, like EPT, downstream. An increase in the fraction of Diptera and Oligochaeta downstream also indicated a decline in water quality. This response is a clear indication that macroinvertebrate communities in the

river are good candidates for assessing water quality and general ecosystem integrity. With intensification of agricultural activities in the watershed there is a need to consider macroinvertebrate assemblages in future water quality monitoring programs in the river. To date, biomonitoring protocols using macroinvertebrates have not been developed in Kenya, and since this approach is cost effective, we strongly recommend that this be incorporated in the continuous monitoring of surface water quality.

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Appendix: Occurrence of macroinvertebrate taxa at six stations on the Moiben River, February to July 2006 (∗ = occurrence)

Taxon	Station					
	M1	M2	M3	M4	M5	M6
Ephemeroptera						
Baetidae						
Baetis sp.	∗	∗	∗	∗	∗	∗
Acentrella sp.	–	–	–	∗	–	–
Caenidae						
Caenis sp.	∗	∗	∗	∗	∗	∗
Ephemeridae						
Ephemera sp.	–	–	–	∗	∗	∗
Heptageniidae						
Heptagenia sp.	∗	∗	∗	∗	∗	∗
Rhithrogena sp.	–	∗	∗	–	–	∗
Oligoneuridae						
Lachlania sp.	∗	–	–	–	–	–
Plecoptera						
Nemouridae						
Nemoura sp.	∗	–	–	∗	–	∗
Perlodidae						
Neoperla sp.	∗	–	–	–	∗	∗
Trichoptera						
Hydroptilidae	∗	–	–	∗	∗	–
Leptoceridae	∗	∗	–	–	–	–
Polycentropodidae	∗	∗	–	∗	–	–
Hydropsychidae						
Hydropsyche sp.	∗	∗	∗	∗	∗	∗
Psychomyiidae						
Lype sp.	–	∗	–	–	–	–
Phryganeidae						
Lepidostoma sp.	–	–	–	∗	–	–
Philopotamidae						
Philopotamus sp.	–	–	∗	∗	–	–
Hirudinea						
Erpobdellidae						
Erpobdella sp.	–	–	∗	–	–	
Glossiphonidae						
Branchiobdella sp.	∗	∗	∗	–	∗	∗
Glossiphonia sp.	∗	∗	–	∗	∗	–
Hellobdella sp.	–	∗	–	–	–	–
Oligochaeta						
Lumbricidae						
Lumbricus sp.	∗	∗	∗	∗	∗	∗
Gastropoda						
Limnaeidae						
Limnaea sp.	∗	–	–	∗	–	–
Planorbiidae						
Planorbis sp.	∗	∗	∗	∗	∗	∗
Sphaeriidae						
Pisidium sp.	∗	∗	∗	∗	∗	∗
Sphaerium sp.	∗	∗	–	–	∗	–
Unionidae						
Anodonta sp.	–	–	–	–	–	∗
Crustacea						
Decapoda						
Eriocheir sp.	∗	∗	∗	∗	∗	–
Arachnida	–	∗	∗	–	–	–
Odonata						
Coenagrionidae						
Enallagma sp.		∗	∗	∗	∗	∗
Gomphidae						
Gomphus sp.	∗	∗	∗	∗	∗	–
Epertogomphus sp.	∗	∗	∗	–	–	–
Aeshna sp.	–	∗	∗	∗	∗	∗

Appendix: (cont.)

Taxon	Station					
	M1	M2	M3	M4	M5	M6
Lepidoptera						
Pyralidae						
Elophila sp.	*	–	–	–	–	–
Hemiptera						
Belostomatidae						
Belostoma sp.	–	*	*	*	–	*
Corixidae						
Corisella sp.	*	*	*	–	–	*
Gerridae						
Metrobates sp.	–	*	*	–	*	*
Naucoridae						
Ilyocoris sp.	–	–	–	*	*	*
Corixidae						
Notonecta sp.	–	–	*	–	–	–
Corixa sp.	–	–	–	*	–	–
Veliidae	–	–	–	–	–	*
Microvelia sp.	–	–	–	–	–	*
Trochopus sp.	–	–	*	–	–	–
Nepidae						
Nepa sp.	–	–	*	–	–	–
Coleoptera						
Elmidae						
Ancyronyx sp.	–	*	–	–	–	–
Elmis sp.	*	*	*	*	*	*
Lara sp.	*	–	–	–	–	–
Eterlimnius sp.				*	*	*
Oulimnius sp.	*	*	*	*	*	*
Microcylloepus sp.	–	–	–	–	–	*
Dytiscidae						
Eretes sp.	–	–	–	*	*	*
Gyrinidae						
Gyrinus sp.	–	*	–	*	*	*
Dineutus sp.	*	–	*	*	*	–
Hydrophilidae						
Enochrus	–	–	–	*	–	–
Dibolocelus sp.	–	–	–	*	*	*
Helodidae						
Helodid sp.	–	–	*	–	–	–
Noteridae						
Hydrocanthus sp.	–	–	*	*	–	–
Noterus sp.	–	–	–	*	–	–
Diptera						
Athericidae						
Atrichops sp.	–	–	–	*	–	–
Chaoboridae						
Chaoborus sp.	–	*	*	–	–	–
Chironomidae						
Ablabesmyia sp.	–	*	*	–	*	*
Chironomus sp.	*	*	*	*	*	*
Ceratopogonidae						
Bezzia sp.	–	*	*	*	*	–
Curicoides sp.	*	–	–	–	–	–
Emphididae						
Hemerodromia sp.	–	–	–	–	*	–
Psychodidae						
Ulomya sp.	*	–	–	–	–	–
Rhagionidae						
Chrysophilus sp.	–	–	–	*	–	–
Simulidae						
Simulium sp.	*	*	*	*	*	*

Appendix: (cont.)

Taxon	Station					
	M1	M2	M3	M4	M5	M6
Tipulidae						
Antocha sp.	*	*	–	–	–	–
Pedicia sp.	*	*	–	*	–	–
Tabanidae						
Chrysops sp.	–	–	*	–	–	–