Land Use and Water Resources Research

Land-use impacts on river water quality in lowveld sand river systems in south-east Zimbabwe

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Abstract

We examined spatial patterns of physico-chemico characteristics and nutrients of freshwater lotic systems in impaired waters in sand river systems in a semi-arid tropical lowveld region of south-east Zimbabwe, from August 2003 to April 2004. These changes were viewed in the context of effluent discharge and land use. The high concentrations of dissolved compounds in the Runde River shows that the chemical composition of this river water is determined largely by the inflowing waters from surrounding irrigated lands. The elevated levels in nutrients recorded along the Runde River suggest that agricultural activities are a key determinant of ecosystem structure and function, especially of algal and invertebrate biomass and productivity in the south-east river systems. This study also shows that nutrient eutrophication of lotic systems may have indirect beneficial impacts on the fish population through increased fish food supply. Lotic ecosystems should be viewed as being influenced profoundly by agricultural activities.

Introduction

The effects of land use in tropical semi-arid areas have many noticeable impacts on aquatic ecosystems that pose water conservation challenges. A growing body of literature has focused on riffled streams (Gratwick, 1998/1999; Benfield *et al*., 2000) predominantly in temperate climates where environmental conditions are somewhat different from semi-arid sand river systems in lowlands. Novotny *et al*. (2005) have illuminated the role that biological monitoring can have in water quality and conservation. The growing need to preserve and restore the physical, chemical and biological integrity of rivers and other waterbodies is implicit in several ongoing studies (Van der Merwe *et al.*, 1993; Buermann *et al.*, 1995; Marx and Avenant-Oldewage, 1998; Don-Pedro *et al.*, 2004; Novotny *et al*., 2005). This has become necessary to maintain the diversity of species and water quality. Stricter legislation has already been enforced worldwide on mining, industry and agriculture to protect natural waterbodies. Prior to the promulgation of *The Environmental Management Bill* on 03 March 2003 in Zimbabwe, the lack of harmonised legislation led to the current problem of polluted rivers, the sandy Runde River in south-east Zimbabwe being an example.

Rivers as sinks of pollutants are the effective transport medium from factory sites and sewage works (Jackson and Jackson, 1998). Within a flowing stream there is a constant attrition of organic and inorganic materials to downstream areas. Drift material may be exploited either as food or as elemental nutrients by various trophic levels in the stream, contributing to enrichment and eutrophication of the eventual receiving water (Grant, 1975; Schneider *et al.*, 2000, Wetzel, 2001; Whitton, 1975). In flood, the south-east lowveld rivers are brown, silty and fast-flowing, with a high content of suspended silt from upstream catchments. The rivers pick up soil and, by dissolving, eroding and leaching, acquire additional dissolved components, the concentrations of which vary due to changes in flow and other seasonal effects.

The south-easterly flowing Runde River has one of the largest catchments in Zimbabwe. Although plankton, fish and other fauna have been studied following pollution inputs in South African lowveld river systems (Buermann *et al.*, 1995; Heritage and Niekerk, 1995; Heritage and Large, 2001; Van der Merwe *et al*., 1993; Marx and Avenant-Oldewage, 1998), a census of the present status of the macrobenthic community in the Runde River and its low-order streams is necessary due to increased pollution, population explosion and anthropogenic activities in and around the catchment.

The composition, diversity and density of aquatic organisms maybe affected by various stressors. Aquatic organisms respond to their immediate stresses such as lack of food, exposure to toxic effects of pollutants, including pH or temperature, and absence of an adequate habitat; they do not respond directly to stresses in the catchment, such as diffuse pollution. These stresses are transmitted through various pathways, modified, attenuated and transformed into the stresses that affect the aquatic biota directly. The stresses may be long-term, transient, periodic and random (Novotny *et al.*, 2005). These parameters define the niche of the macroinvertebrates which may change throughout the year. In polluted water environments of south-east Zimbabwe, environmental factors may affect presence and absence of aquatic taxa. The human disturbance of the river waters through point source and diffuse pollution impacts on various aquatic organisms by altering the physical, chemical and biological integrity of rivers.

The sand river systems have been described as being considerably impoverished in organic matter. As Marshall (1972) puts it, the fauna of a polluted, sandy stream- bed is poor in species, numbers and biomass. This is exacerbated by the scouring effect brought on by floods in the rainy season and there is little opportunity for organic matter to accumulate. This in contrast to weirs, dams and other barriers that may allow organic matter to accumulate, such that these areas support a greater biomass than unobstructed streams.

The Runde River is enriched with nutrients while passing through an agricultural area and then receives sewage effluent that will support an algal bloom. Nitrates, phosphates and sulphates are applied in fertilisers to grow crops (Hippo Valley, 1995). Elevated nutrients in the watercourses present additional feeding resources and challenges to aquatic organisms. Foraging decisions, or opportunities, affect the fitness of individual nymphs and this may be reflected in the wide size range of contemporary nymphs found in field populations (Williams *et al*., 1993). Variation in foraging efficiency by predators and evasion success by prey, across substrate types, is thought to contribute to the well-known microdistribution patterns of species observed in lotic communities (Williams *et al*., 1993; Novotny *et al.*, 2005). These habitat shifts have been interpreted as the result of the optimal foraging and antipredatory behaviour, and they can be decisive for the outcome of species interactions (Haertel *et al*., 2002).

The distribution of aquatic organisms may also be mediated by metals (Van der Merwe *et al.,* 1993; Marx and Avenant-Oldewage, 1998; Whitton, 1975; Wetzel, 2001). Because the aquatic ecosystems are often the sink for human-generated pollutants, discharges of metals should be monitored and captured, so as to maintain and conserve the biota dependent on it (Van der Merwe *et al.*, 1993; Don-Pedro *et al.*, 2004). Metals have an indirect effect on biological responses affecting density, diversity, community structure and species composition of populations (Van der Merwe *et al.*, 1993; Don-Pedro *et al.*, 2004; Wetzel, 2001). The type and extent of impact depends primarily on the heavy metal concentration in the water and sediment since above certain tolerable levels, it becomes toxic to aquatic organisms (Van der Merwe *et al.*, 1993; Don-Pedro *et al.*, 2004).

One of the potential problems is that the Runde River water is likely to have a very high rate nutrient loading in the form of artificial fertilisers from agricultural runoff. Should this happen, the river water will become hypereutrophic, leading to algal blooms. This loss of water quality will not affect irrigation until the aquatic weeds interfere with pumping operations. The biological productivity of aquatic ecosystems is usually determined by the concentration of phosphorous and nitrogen and unless well planned and monitored, domestic sewage overflow and discharge from the sugar estate sectional housing units could affect river water quality.

It has therefore become necessary to assess the effect of habitat destabilisation and modification caused by agricultural runoff and sewage discharge on water quality and aquatic organisms in the sandy river systems of south-east Zimbabwe lowveld, with a view to assessing the problem, proffer solutions and prevent defaunation. Measuring the stream biota provides a direct assessment of the resource condition because the characteristics of the biota reflect the influence of human activity (Novotny *et al.*, 2005). If biological assemblages are not present at the level expected then almost certainly human influences are degrading the streams. Water pollution monitoring in the river systems of south-east Zimbabwe can help provide a useful basis for early warning systems for pollution events propagating downstream.

Study area

The study area included the area drained by the Chiredzi, Mtirikwi, Tokwe and Runde Rivers, about 350 km south-east of Harare (Fig.1). The study area is underlain by granite and gneiss, but in the east volcanics and rocks of Karoo age form the geological template. This study area was selected because it covers a wide range of typical Zimbabwean land uses, including sugarcane production at Triangle Sugar Estates and Hippo Valley Estates, and sugarcane processing mills at Hippo Valley and Triangle.

The Runde River is typically an East Coast River with sandy bottoms and little aquatic plant coverage (Bell-Cross and Minshull, 1988). The Runde River and the other eastward flowing rivers such as the Save River probably derived their fauna from the large south-eastern Zaire tributary — the Lualaba — plus an element from the Nile system (Bell-Cross and Minshull, 1988). Seasonal trends are clear in both the precipitation and the flow regime (Magadza *et al.*, 1993). Drying up of the river waters is a frequent and important hazard . Surface water extent varies dramatically over time, with dry periods characterised by extreme habitat shrinkage and fragmentation. The flow regime is characteristic of semi-arid watercourses; extremes of discharge occur with low winter baseflows of 1 or 2 cumecs and occasional high summer flood flows exceed 100 cumecs. Climatic variability has been identified for the lowveld (Tyson, 1987). During winter base flow, the rivers carry clear water on alluvial braided channels. The distribution of sediment varies in the channels, giving rise to different geomorphic conditions along the rivers. The tributaries of the Runde River that include the Chiredzi, Mtirikwi and Tokwe (Fig. 1) pass through low input peasant agricultural areas and these streams are characterised as reference

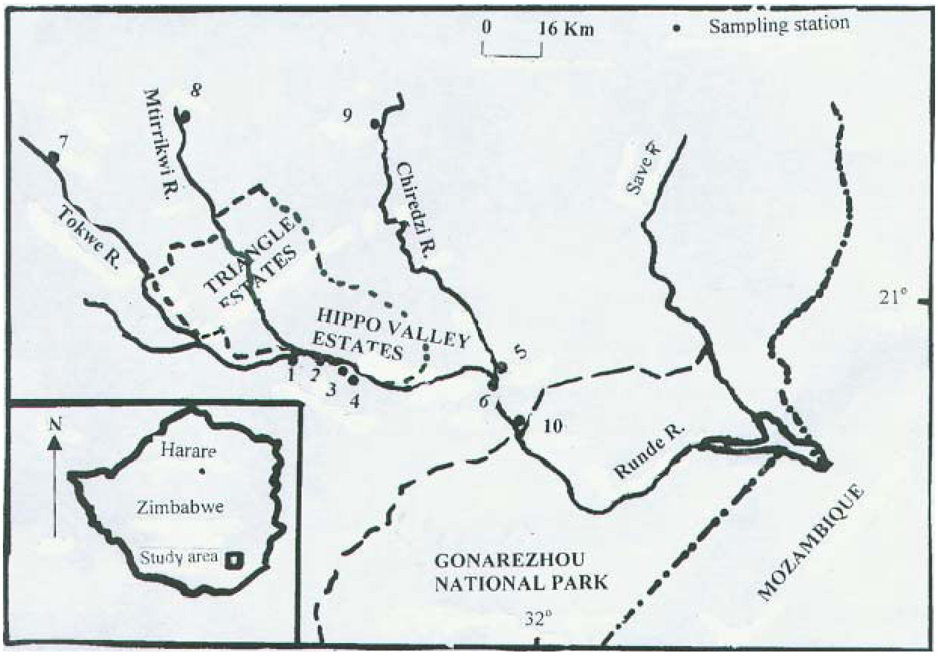

Figure 1. Map showing situation of the study area. The inset map shows the situation of the study area in southeast Zimbabwe lowveld

streams in this river study. The Runde River tributaries flow from their source to the Runde River, passing through the incised granite plain of the Lebombo zone and lowveld zone. Table 1 shows that the Runde River mean daily flows vary greatly between 0 $m^3\ s^{-1}$ in winter and under drought situations and 75 $m^3\ s^{-1}$ in summer.

The study area lies below the 600 m contour and is hot and semi-arid. The climate of the lowveld is hot and wet from mid-November to April, cool and dry from May to August, and hot and dry from September to mid November. The temperatures range from 8.1°C in July to 50°C in January, with a mean of 24°C. A significant feature of the rainfall is its unreliability, both in terms of quantity and duration. The variation from year to year is so great that the annual rainfall can range from 20% to 200% of normal. The rainfall varies considerably from a low of 92.7 mm in 1991/92 to a high of 834.0 mm in 1977/78 (Magadza *et al.*, 1993). Tropical depression and cyclone activity can produce unprecedented rainfall and floods, as recorded in March 2000 across the eastern African subcontinent (Heritage *et al*., 2001).

Table 1 Runde River mean daily discharge rate, $m^3\ s^{-1}$

Month	*92/93*	*91/92*	*90/91*
October	0	0	11.4
November	0.471	0	0.038
December	179	2.39	7.23
February	75.5	3.49	7.32
March	352	0.226	30.1
April	24.3	0	28.5
May	1.93	0	0.748
June	0.007	0	0
July	1.18	0	0
August	4.42	0	0
September	0.001	0	0

The Runde watershed is situated within an area of intensive commercial farming area characterised by the Hippo Valley and Triangle Sugar Estates. Farmers in the sugar estates use appreciable quantities of lime to neutralise the soil pH which is normally around 5. The application of lime on agricultural soils is expected to raise the Runde River waters well above the current levels in due course, hence the need to monitor the physico-chemico aspects of the river waters.

Methods and materials

Stratified random sampling in which the rivers were subdivided according to land-use zones was undertaken. Sampling positions were randomly selected within stratified zones thus ensuring that chances of missing any general biotic associations are extremely small. Sample stations were to be readily accessible. The sampling stations represent a wide range of water quality conditions in the study area. Six sample sites — 1,2,3,4,6 and 10 — were selected on the impacted zone of the Runde River. Site 2 receives domestic and agricultural effluent discharged into the lower section of the Mtirkwi river. Site 4 is a discharge point of pre-treated sewage. Site 10 is situated at the edge of the Gonarezhou National Park. Sample sites 7, 8 and 9 were selected on the reference streams and are situated in peasant agricultural areas. Site 5 was selected at the confluence of Chiredzi River and Runde River. Site 5 receives agricultural runoff from the small scale intensive farms in the proximity of lower Chiredzi river. Other sampling sites along the Runde River receive inflow from surrounding areas. Due to difficulties in finding alternative sites because of terrain conditions, site 8 is situated on obstructed Mtirikwi River. Samples taken from upstreams of pollution of Chiredzi, Mtirikwi and Tokwe establish expected biological conditions in the absence of the pollution source. Benthic samples were taken in the dry season August 2003 and wet season of April 2004 at Hippo Valley and Triangle sugar estates, using a $0.1m^2$ Van-Veen grab. All accumulated material from the samplers were collected and preserved immediately upon retrieval. The collected organisms were identified at the family level and genus. List of families collected from the rivers was compiled.

The temperature of the water and the concentration of dissolved oxygen was measured with YSI dissolved oxygen meter. Water was collected in thoroughly rinsed bottle containers and immediately stored in a freezer for later analysis. Measurements of water temperature, pH, dissolved oxygen, light penetration, and conductivity were undertaken in August during the dry season and in April during the wet season. Conductivity, pH, oxygen and temperature were measured with portable measuring devices (WTW). Light penetration was measured using a Secchi disc. Other data collected include substrate composition such as sand, gravel, stones, etc) and description of surrounding area (amount of tree cover and land use, etc). In the laboratory, the samples were filtered through Whatman GF/C fibre glass filters. The concentrations of total phosphorous were then determined by the reactive molybdate method, ammonia by the indophenol method, and nitrate and total nitrogen by the sulphanilamide method, using a cadmium reduction column and a Hitachi 100-40 spectrophotometer (Golterman *et al.*, 1978). The concentrations of inorganic cations (lead, magnesium, cadmium, copper, zinc and calcium) were determined by atomic absorption spectrophotometry. Potassium was determined by flame photometry. Physical analyses of water were carried out in the field. Water and sediment were analyzed for metals and physical and chemical water parameters.

Discriminant analyses (DA) were applied to the nutrient data collected in the study plots, nutrient data classified as 'polluted' or 'unpolluted'. Linear Discriminant Analysis (LDA) determines the linear combination of predictor variables that best classifies cases into one of several known groups and cases whose groups are known. The variables in the 'polluted' and 'unpolluted groups were tested for significant differences using the t-test. The assumption in the t-test is that any difference in response is due to the treatment or lack of treatment and not to other factors. SPSS for Windows (SPSS,1996) was used to process raw data in the stepwise DA.

The following references were used for the taxonomical determination of species: Parish (1975), Pennak (1978), Thirion *et al.* (1995), Picker *et al.* (2002) and Appleton (1996).

Results

Water temperature measured in August in the lowveld rivers varied between 17.1°C and 25°C (Table 1). Water temperature measured in April in the lowveld rivers varied between 18.4°C and 34°C (Table 2). High but wide variations in dissolved oxygen concentration were measured in August (Table 1, Fig. 2). Dissolved oxygen concentration varied between 8.9 and 16.7 mg l^{-1} (Table 1). Wide variations in dissolved oxygen concentration were recorded at all sample sites (2.4 mg l^{-1} to 6.3 mg l^{-1}) (Table 2). Dissolved oxygen is markedly higher in August of the dry season than in April of the wet season (Fig.2). Figure 2 shows very low levels of dissolved oxygen recorded at site 1, suggesting extreme conditions of oxygen depletion in the wet season. However,

Table 2 Water quality parameters recorded at nine sample stations in the month of August. N/A=not available

Parameter		*Sites* 1	2	3	4	5	6	7	8	9
Temp (Celcius)	max	24.5	23	24.2	26.5	26.1	23.4	25	25	21.1
	mean	21.1±2.16	21.3±0.95	22.3±1.16	25±0.84	23.8±1.39	21.6±1.32	21.4±1.97	20.7±2.15	19.4±0.91
	min	17.1	19.7	20.2	23.6	21.3	19	18.2	18.2	18
	n	3	3	3	3	3	3	3	3	3
pH	max	7.9	8	8.1	7.4	7	6.9	7	8.2	8
	mean	7.8±0.09	7.8±0.17	7.8±0.21	7.2±0.19	6.5±0.27	6.4±0.26	6.8±0.18	7.7±0.32	7.6±0.23
	min	7.6	7.5	7.4	6.8	6.1	6	6.4	7.1	7.2
	n	3	3	3	3	3	3	3	3	3
Dissolved Oxygen (mg l^{-1})	max	9.8	11.8	13	15.3	14.8	14.5	14.3	16.7	15.3
	mean	9.4±0.26	10.9±0.43	12.2±0.50	15.4±0.38	14.5±0.20	13.6±0.44	13.7±0.35	12.4±2.58	12.3±1.77
	min	8.9	10.1	11.3	14.6	14.1	13	13.1	7.8	9.2
	n	3	3	3	3	3	3	3	3	3
Conductivity (µS cm^{-1})	max	107	107	205	182	250	208	123	183	298
	mean	106.7±0.58	103.7±4.04	204.2 ±2.0	180±3.21	249.6±2.12	207.1±2.65	121.7 ±3.51	182.4±2.0	296.3±5.86
	min	106	100	201	179	248	206	120	181	295
	n	3	3	3	3	3	3	3	3	3
Secchi disc (cm)	max	22	94	110	90.2	23.1	14	33.7	110.2	98.2
	mean	21±0.58	90.7±1.71	100.5±5.35	86±2.09	19.7±1.75	10.7±1.74	32.0±0.95	102±4.73	94.3±2.38
	min	20	88.2	91.5	83	17.2	8.1	30.4	93.6	90
	n	3	3	3	3	3	3	3	3	3

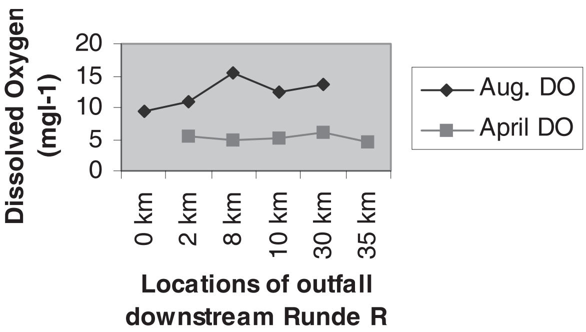

Figure 2. Dissolved oxygen concentration curve along the Runde River

the typical septic zone was not present along the Runde River. Dissolved oxygen concentration did not exceed 20% saturation in a river that was 0–2m deep.

The pH measured in the river waters in August varied between 6.0 and 8.2 (Table 1). The pH measured in April deviated substantially from the 6.0 to 8.2 (Table 2). Conductivity in the river waters varied from 100 to 298 µS cm^{-1} in August (Table 1). Conductivity values tend to fluctuate among the sampling sites. Variation in conductivity tend to follow a predictable pattern of substantial decreases during a period of high inflows. The level of conductivity in water gives a good indication of the amount of ionisable substances dissolved in it, such as phosphates, nitrates and nitrites which are washed into streams and ponds after fertiliser is applied on surrounding fields or are present in effluent from sewage-treatment facilities.

Mean Secchi depths measured in August varied between 10.7 cm and 102 cm (Table 1). Light penetration often reached the substrate in the majority of pools in the dry season. Mean Secchi depth recordings in April varied between 14.1 cm and 63.6 cm (Table 2), indicating less frequent occurrences of clear water and more turbid conditions since the periodic inputs of suspended solids decrease water clarity and light penetration.

The variations in the physico-chemico characteristics were monitored on the Runde River at one sampling station over 24 hrs to include day and night changes (Figs 3–7). Dissolved oxygen concentration was lowest at 1600 hrs, measuring 1 mg l^{-1}. Dissolved oxygen concentration increased to a peak of 2.5 mg l^{-1} at 2400 hrs and thereafter decreased to 0700 hrs before reaching a peak at 2.5 mg l^{-1} at 1000 hrs (Fig. 3). Temperature was lowest at 0600 hrs but increased to peak of 1500 hrs at 1500 hrs (Fig.4).

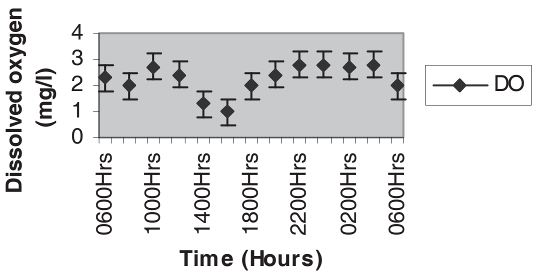

Figure 3. Day and night monitored changes in dissolved oxygen concentration

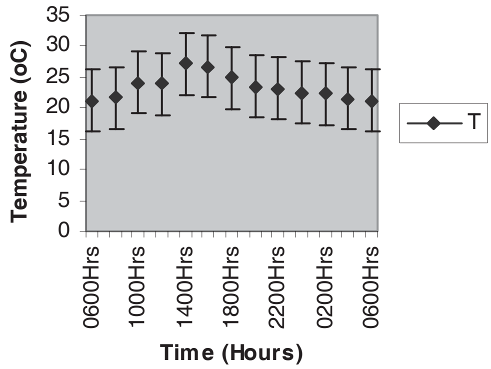

Figure 4. Day and night monitored changes in temperature

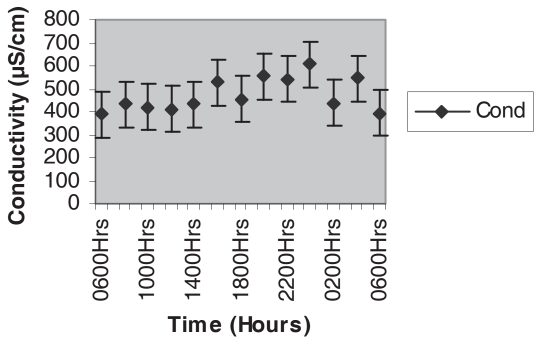

Figure 5. Day and night monitored changes in conductivity

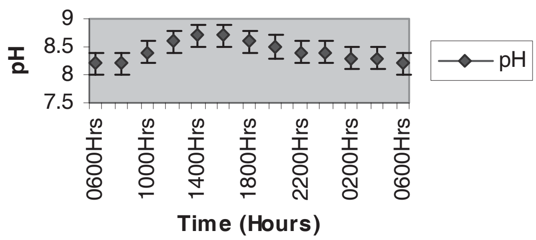

Figure 6. Day and night monitored changes in pH

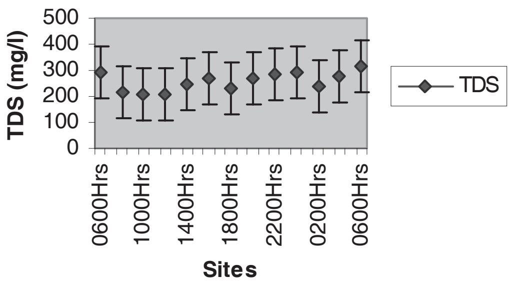

Figure 7. Day and night monitored changes in Total dissolved solids (TDS)

Conductivity was lowest at 0600 hrs, measuring 400 µS cm^{-1} but gradually increased to a peak of 600 µS cm^{-1} between 2000 hrs and 2400 hrs (Fig. 5). pH was lowest at 0600 hrs, measuring 8.2, but this increased to a peak of 8.7 at 1500 hrs (Fig 6). TDS was lowest at 1200 hrs at 200 mg l^{-1} but this increased to a peak of 300 mg l^{-1} at 0600 hrs (Fig.7).

Table 3 Summary of families of benthic macro-invertebrates identified in sample stations

Order	*Family*	*Genera*
Acari	Hydrachnidae	Hydrachna
Coleoptera	Hydrophilidae	Hydrophillus
Crustacea	Arrenuridae	Arrenura
Diptera	Chironomidae	Xenospace
Ephemeroptera	Heptageniidae	Afronurus
	Baetidae	Baetis
Hemiptera	Belostamitadae	Belostoma
	Gerridae	Gerris
	Veliidae	Velia
Mollusca	Corbicula	Corbiculidae
	Limnaediae	Limnaea
	Planorbidae	Planorbis
		Helisoma
		Gyraulus
		Promenetus
	Physidae	Physa
	Sphaeriidae	Sphaerium
		Pisidium
		Eupera
		Quadrula
	Unionidae	Coelatura
Odonata	Aeshnidae	Aeshna
		Anax

A total of 10 orders, 20 families and 27 genera were identified in all sample stations (Table 3). The mollusca were the most diversified group with four families and five genera and this was followed by the Odonata with four families and four genera. The least diversified group include Acari, Coleoptera, Crustacea and Diptera.

Figure 8 shows abundance of benthic macroinvertebrates collected at nine sampling stations. Stations 4, 5, 6 and 9 have very high numbers of individuals and few families in summer and at these stations elevated nitrates and total nitrogen were recorded in sediments (Fig.9), while elevated total phosphates, total nitrogen and nitrates were recorded

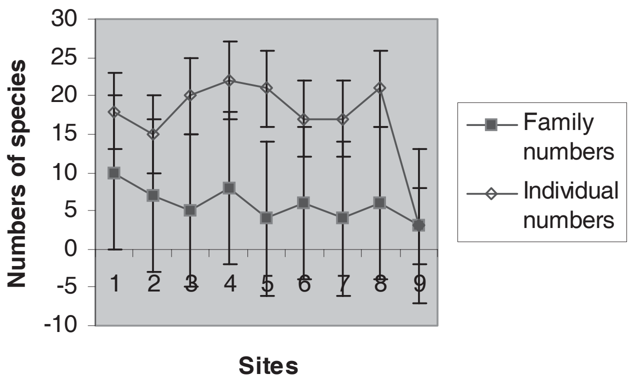

Figure 8. Distribution of benthic macroinvertebrates at nine sampling sites in August

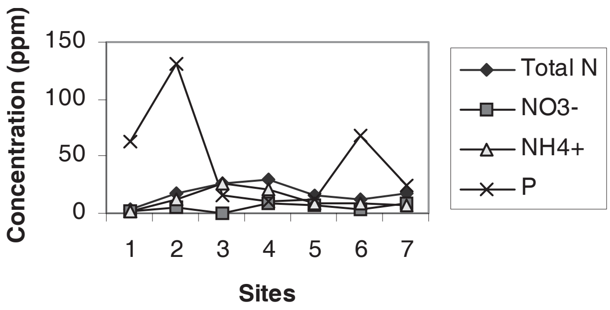

Figure 9. The change of the concentration of total nitrogen, nitrates, ammonia and total phosphates in river water sediments at sampling sites

in the water column (Fig. 10). Stations 4 and 5 have very high numbers of individuals and few families in winter at a time dissolved oxygen concentration is expected to be at its lowest (Fig. 11). The numbers of individuals and families of organisms tend to fluctuate among the sampling sites (Figs. 8 and 12). Benthic macroinvertebrates were not found in the streams in the wet month of April.

The results show high spatial and temporary variability in nutrients in the dry season and wet season (Figs 10-13). A t-test of nutrient concentrations showed significant differences between 'unpolluted' sampling sites and 'polluted' Runde sampling sites in total nitrogen, ammonia and total phosphates (Table 4). Using LDA, only total nitrogen and ammonia showed significant difference among

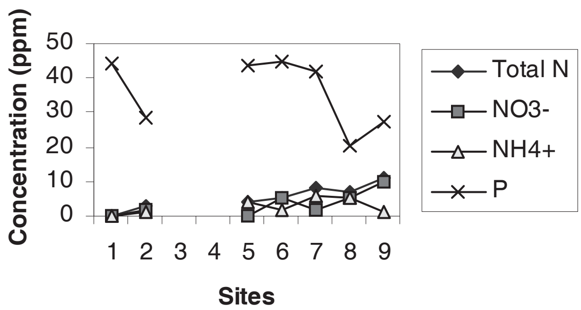

Figure 10. The change of the concentration of total nitrogen, nitrates, ammonia and total phosphates in the water column of the sampling sites

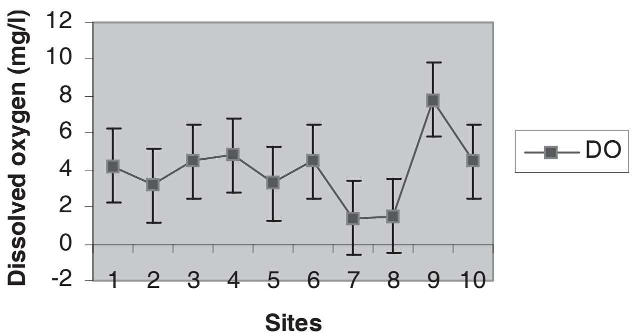

Figure 11. [caption missing - author to provide, please]

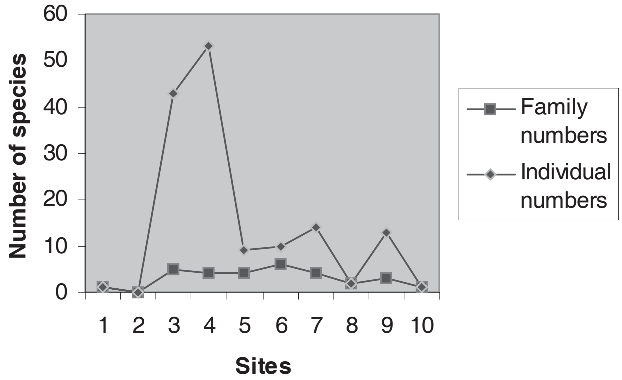

Figure 12. Distribution of benthic macroinvertebrates at nine sampling sites in May

Table 4 Summary of t-test for variables in the nutrient data collected for analysis. The significance of differences between reference sampling sites and polluted Runde sampling sites by t-test is shown as: NS, not significant; **P<0.05; **P<0.005*

Variable	*Significance*
Total nitrogen	*
Nitrates	NS
Ammonia	* *
Total phosphates	*

dependent variable means for each independent variable. The percentage of the variance accounted for by each discriminant function shows that only one function, total nitrogen, is highly significant ($P<0.000846$, df=14). DA correctly classifies 87.5% of the displacement outcomes. The model is definitely adequate and the hypothesis of the equality of the group is rejected.

Discussion

The physico-chemico characteristics in the sand river systems of the lowveld are shown to vary from locality to locality, depending on season and external factors, with the benthic macroinvertebrates distribution and abundance seeming to respond to these changes. The variability in the physico-chemico characteristics and spatial occurrence of benthic macroinvertebrates may be a feature of the lowveld river systems. Variability and unpredictability are therefore characteristics of the aquatic ecosystems, hydrologic patterns and climate of the largely dry lowland region that encompasses the lowveld ecosystems. Table 1 shows that lowveld rivers in south-east Zimbabwe can undergo long drying spells in some years and this affects stream discharge and habitat quality.

Severe droughts appear to be the result of a prolonged El Nino/Southern Oscillation event that results in a 38% reduction for expected rainfall for the lowveld in eastern southern Africa (Heritage and Niekerk, 1995). In some years the lowveld region can be subjected to floods by cyclonic rainfall. The physico-chemico characteristics tend to follow the same pattern of great variability among sampled sites across seasons. Concentrations of dissolved oxygen in the river waters vary between 2.4 mg l^{-1} and 16.7 mg l^{-1}.

High but wide variations in dissolved oxygen concentration were measured at the sample stations in August of the hot dry season. Concentrations of dissolved oxygen in the river waters vary between 2.4 mg l^{-1} and 16.7 mg l^{-1}. Low but wide variations in dissolved oxygen concentration were measured in the wet season. Standards in South Africa require that dissolved oxygen concentrations should be >4.0 mg l^{-1} and the target guideline warm-water species is 5.0 mg l^{-1} (Buermann *et al*., 1995).

Dissolved oxygen in waterbodies varies in response to biological activity, with higher levels being associated with presence of aquatic plants (Moyo, 1997; Schneider *et al.*, 2000). Dissolved oxygen concentration levels measured in the wet season of April are lower than concentration levels measured in the dry season of August. This may be due to the loss of photosynthetic aquatic plants by faster current. Oxygen depletion results from the eutrophication of natural water that receives excessive amounts of nutrients normally limiting to plant growth.

The pH in all sampling stations (Table 2) did not deviate substantially from the 6.1 to 8.2 range expected for most surface freshwater systems. Fawzi *et al*. (2002) suggests that geological, atmospheric, biological and anthropogenic influences may all affect the proportion of major ions in waterbodies, and hence pH. Biological activity, in particular consumption of carbon dioxide, results in an increase in pH during photosynthesis by abundant aquatic plants, and released carbon dioxide resulting in decreased pH during respiration and decomposition (Schneider *et al.*, 2000). Biological activity and human activity would be the primary determinant of pH with human activities triggering nutrient loading while biological activity plays a secondary role. The Runde River is enriched with nitrate while passing through an agricultural area and then receives sewage effluent rich in phosphate (Figs 9 and 10) that will support an algal bloom.

After catchment water releases by sudden heavy rains, the conductivity of the Runde River decreased and Secchi readings decreased greatly. This indicates that under erosive conditions, rains and flood waters add proportionally more insoluble material than soluble electrolytes. The decrease in conductivity and TDS indicates a dilution effect of the poor quality Runde River water that result in lower water quality. Russell (1999) showed substantial short-term declines in salinity during high runoff periods, suggesting that dry season pollution events of waterbodies should be of concern.

Mean Secchi depths measured in August varied between 10.7 cm and 100.5 cm with a range of 8.1 to 110.2 cm (Table 2). Light penetration often reached the substrate in the majority of pools in the dry season as these were generally less than 2 m in depth. Mean Secchi depth recordings varied between 14.1 to 81.3 cm (Table 5), indicating less frequent occurrences of clear water conditions and turbid conditions in the wet season. The periodic inputs of sediment decrease water clarity and light penetration leading to reduced biological diversity. Suspended solids in flooded river waters contribute to oxygen depletion (Vesilind *et al.*, 1994).

Table 5 Water quality parameters recorded at nine sample stations in the month of April

Parameter		*Sites* 1	2	3	4	5	6	7	8	9	10
Temp	max		29.6	29	31.3	30.2	30.7	24.3	31.2	30.5	34
(Celcius)	mean		26.2±2.20	26.4±1.37	28.1±1.82	28.3±1.07	28.7±1.25	22.5±1.24	25.3±3.41	24.5±3.49	28.4±2.91
	min		22.1	24.3	25.1	26.5	26.4	20.1	19.4	18.4	24.2
	n		3	3	3	3	3	3	3	3	3
pH	max		7.9	8.1	7.4	7.4	7.3	7	7	8.2	8
	mean		7.8±0.09	7.8±0.21	7.2±0.19	7.2±0.19	6.8±0.36	6.4±0.26	6.8±0.19	7.7±0.32	7.6±0.23
	min		7.8	7.4	6.8	6.8	6.1	6	6.4	7.1	7.2
	n		3	3	3	3	3	3	3	3	3
Dissolved	max		6.3	6	5.4	2.9	6.5	4.8	5.9	6.1	5.6
Oxygen	mean		5.9±0.48	5.3±0.38	4.9±0.29	2.6±0.15	6.3±0.10	4.6±0.27	5.5±0.23	5.7±0.22	4.7±0.47
(mg/l)	min		4.9	4.7	4.4	2.4	6.2	4.1	5.1	5.4	4
	n		3	3	3	3	3	3	3	3	3
Conductivity	max		106.3	209	179	298	213	111	128	190	308
(µScm^{-1})	mean		98.6±4.36	196±6.56	172.7±3.48	241±28.39	199.7±6.89	106±4.04	118.3±5.24	175.7±8.69	293.3±7.69
	min		90.9	188.1	166.5	210	190	98	110	160	282
	n		3	3	3	3	3	3	3	3	3
Secchi	max		66	80	33.7	15.9	86.3	30.1	42	32.4	35.4
Disc	mean		63.6±1.27	71.0±4.9	29.5±2.45	14.1±0.93	81.3±2.60	27.7±1.48	35.1±5.73	27.5±2.80	32.3±2.43
(cm)	min		61.7	63.1	25.2	12.8	77.6	25	23.7	22.7	27.5
	n		3	3	3	3	3	3	3	3	3

TDS varied between 126 and 841mg l^{-1} with a mean of 375mg l^{-1} in winter months at the sampling stations. The limited tolerance range of organisms in freshwater should be between 350–550 mg l^{-1}, with a threshold of 800 mg l^{-1} cited for some lowveld rivers such as the Olifants River (Buermann *et al*., 1995). TDS concentrations recorded in the winter season during this study did not exceed the recommended threshold for survival. Under erosive conditions, rains and flood waters, TDS may drastically overshoot the recommended threshold.

There is a pattern of physico-chemico trends over 24 hours (Figs 3–7) in the monitored river waters in this study that have been reported for a variety of freshwater rivers (Moyo, 1997). The trends suggests that the physico-chemico characteristics of lotic freshwater systems reflect the composition of and solubilities of materials in the rock, soil, primary production and inflows that the water flows through. The physico-chemico patterns may be influenced more by influents from land drainage than natural river processes as this study tends to show.

The fauna of polluted sites on the sandy stream bed of the Runde River is poor in species, high in numbers and biomass which has not been expected for sites with in-channel bar growth and naturally low productivity. This observation suggests that inflows along the length of the Runde allow organic matter to accumulate to support a greater biomass of benthic macroinvertebrates. The mollusca were the most diversified group, with four families and five genera, and this was followed by the Odonata with four families and four genera. Mollusca abundance is more pronounced on the Runde River and sections of reference streams passing through the intensive agricultural zone. Dominating bivalves were Sphaeriidae, filter feeders. Such suspension feeding bivalves can be very efficient consumers of primary production (algae) (Josefson and Rasmussen, 2000). Sites 1 and 4 recorded very high levels of benthic macroinvertebrates (Fig. 8). Nutrient loading increases bivalve productivity (Josefson and Rasmussen, 2000). This abundance of bivalves may be considered to be a good indicator of secondary production.

In the reference stations 7, 8 and 9, benthic macroinvertebrate population (Fig. 8) may have been kept low due to substrate instability and variable discharge. Following the wet season floods, no macroinvertebrates were found at sample stations. The river waters probably dislodged the animals into drift. An increased silt load and an increased stream velocity can result in the disappearance of stream invertebrates due to increased drift (Magadza and Masendu, 1986).

Changes in physico-chemico characteristics were viewed in the context of effluent discharge and landuse. Variation in nitrate-nitrogen content is probably due to rainfall and leaching from agricultural land and this tends to result in concentration increases downstream.The variation in nutrients can influence niche change by macroinvertebrates through fluctuations in food supply. Nutrients in the inflow may trigger a positive feedback mechanism in benthic macroinvertebrates by increasing diversity and abundance as the food supply increases. The Runde River may be described by its sensitivity to potential changes to land use, especially those including a change in application of fertilisers, and hydrologic patterns such as amount and timing of streamflow. The high concentration of dissolved compounds in the Runde River shows that the chemical composition of this river water is chiefly determined by the inflowing waters from surrounding irrigated lands. The high concentrations of total salts (conductivity), alkalinity, pH and phosphate lead to strong increases in these quantities downstream.

LDA and t-test results revealed that appreciable amounts of sedimenting organic matter are a major source of nitrogen from influents, probably from both surface land and from groundwater sources. Pollution of the Runde River may be attributed to agricultural runoff, mobilisation of river sediments and domestic and industrial effluents. Large quantities of sewage are produced by villages on the estates and result in increased loading on the effluent ponds (Hippo Valley Estates Environmental Report, 1995). Typically, organic pollution leads to a decrease in species but an increase in individuals, unless conditions deteriorate to the extent that no animals can survive (Jackson and Jackson, 1998; Marshall, 1972). Septic conditions were not recorded along the Runde River, presumably because of high rates of denitrification and volatilisation of ammonia due to the high temperatures that prevail. The data show that the most productive test sites experience agricultural runoff and both domestic and sewage discharges and that these sites are characterised by an abundance of families and individuals of organisms.

Increased primary production is not bad in and of itself; indeed, many lakes and ponds are artificially fertilised to increase commercial fish production (Ricklefs, 2001). Macroinvertebrates are important fish food. This study shows that nutrient eutrophication of the Runde River has indirect beneficial impacts on the fish population through increased fish food supply. But excessive eutrophication can lead to imbalance when natural regeneration processes cannot handle the increased demands for cycling of organic matter. The problem maybe heightened in winter, when photosynthesis rates are low and little oxygen is generated within the water column. This type of pollution can deplete oxygen all the way to the surface, causing fish and other obligately aerobic organisms to suffocate.

The chemical measurements support the classification of the Runde River as a hypertrophic stream. The parameters measured are supported by inflow in a definite way (Figs 9 and 10). As Josefson and Rasmussen (2000) put it, hypertrophic waterbodies may be highly productive to benthic macroinvertebrates up to a point, depending on the physico chemico conditions of the river water quality. The high productivity of eutrophic waterbodies increases abundance fish species that attract a thriving fishing community. During our sampling of the river waters, thriving fishing communities were noted on the Runde River.The loss of water quality on the Runde River and lower sections of the reference rivers may not affect irrigation and not until the aquatic weeds interfere with pumping operations, will the implications in the loss of water quality be felt.

Noticeable presence of spreading water hyacinth *Eichornia crissipes* can be found on the obstructed lower reaches of Mtirikwi River. Although downstream dispersal, by seeds or by vegetatively reproducing propagules or fragments, is easy in a river, it is difficult in unobstructed waterbodies with a scouring effect brought about by floods in the rainy season.

Within the south-east lowveld, the major threat to the quality of water in the Runde River is as a result of point and nonpoint sources of agricultural activity and sewage discharge, which may increase nutrient levels in the water, such as nitrogen, nitrates, phosphorous, seriously affecting water quality and distribution of benthic macroinvertebrates. The findings in the study suggest that agricultural runoff and sewage is a key attribute affecting water quality and distribution of benthic macroinvertebrates

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