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Full Length Research Paper

Passive Biomonitoring of Mazowe and Yellow Jacket Rivers, Zimbabwe: Use of the South African Scoring System and Habitat Assessment Index

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This study assessed the water quality of Mazowe and Yellow Jacket Rivers which passes through Mazowe citrus estate and Iron Duke Pyrite mine (IDPM) respectively using physicochemical analysis and passive biomonitoring. The physicochemical parameters analysed were temperature, dissolved oxygen, pH, conductivity, phosphates, nitrates and heavy metals. Passive biomonitoring involved assessing benthic macroinvertebrate communities using the South African System Version 5 and habitat quality was assessed using the Habitat Assessment Index (HAI). The physicochemical parameters analysed were lowest at sites 2 and 3, when compared to the other sites. The concentration of nitrates, and phosphates was significantly higher at site 3-6 ($p<0.05$), when compared to the other sites. The concentrations of heavy metals were significantly higher ($p<0.05$) at sites 2-4 when compare to the other sites. Site 1 and 5 had higher SASS score and number of families, while the score per taxon (ASPT) was highest at site 1, 5 and 6. The results suggest that habitat quality in the Yellow Jacket River was poor compared to Mazowe Rives as a result of acidity and heavy metal contamination from IDPM. In Mazowe River high nitrates and phosphate concentrations from the estate, may affect the water quality.

Keywords: biomonitoring, river water quality, habitat assessment index, pollution, macroinvertebrates.

INTRODUCTION

River pollution has increased substantially in the last decades due to a plethora of industrial, agricultural and domestic waste, effluents, as well as hazardous

List of Abbreviations

EMA- Environmental Management Agency ; ASPT- Average Score per Taxon ; HAI- Habitat Assessment Index ; IDPM- Iro Duke Pyrite Mine ; SASS- South African Scoring System.

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substances (Valavanadis and Vlachogianni, 2010). This wide range of effluents, including toxicants generated in catchment areas often finds their way into river systems and affecting their integrity. The ecological integrity of a river is its ability to support and maintain a balanced, integrated and adaptive composition of physicochemical characteristics with a biological diversity, composition, and functional organization on a temporal and spatial scale that are comparable to those of natural aquatic ecosystems in the region (Todd and Roux, 2000).

Effluents are the main source of direct and continuous input of pollutants into rivers and represent a major threat to communities in receiving aquatic ecosystems (Wepener et al. 2006). Efforts to monitor the presence of pollutants and assess water quality in aquatic ecosystems started in the 1960s, concerned with the chemical and physical parameters (Phiri, 2000; Rohr et al. 2006). While chemical monitoring provides sensitive indicators of substances that have been selected for study, this information alone lacks essential biological information. Physical and chemical parameters in many aquatic systems may vary considerably on a spatial and temporal scale. Chemical data are instantaneous and reveal only the conditions of the stream or river at the time of sampling. Large numbers of measurements would be needed to assess maximum and minimum variations in water quality.

The assessment of biota in rivers and streams is a widely recognized means for determining the condition or 'health' of rivers (Dickens and Graham, 2002).This is because the interactions among chemical, physical and biological stressors and their cumulative impacts emphasize the need to directly assess the biota as indicators of actual water course impairments (Dickens and Graham, 2002). Parallel chemical monitoring must also be done under all circumstances to reveal causative factors. Biological monitoring therefore complements chemical and physical monitoring. The biomonitoring indicates whether a problem is present, and the chemical monitors identify its nature (Hyne and Maher, 2003). This is a shift from isolated surrogate methods based on single species toxicity tests and chemical measurements to a more integrated and holistic exposure assessment that reflects the health of ecosystems (Wepener, 2008). The holistic approach incorporates the physicochemical, toxicological and ecological lines of evidence into an integrated assessment of aquatic systems integrity (Wepener, 2008).

Aquatic biomonitoring is the science of inferring the ecological condition of rivers, lakes, streams, and wetlands by examining the effects of xenobiotics to organisms. The advantage is that living organisms identify that interactions have taken place between contaminants and the organisms, and they measure sub-lethal effects, whereas chemical analysis can measure only a fraction of the contaminants, but reveals nothing about their adverse effects on living organisms (Barbour et al. 1999). The assessment of biological organisms is also the most effective means of evaluating cumulative impacts from non-point sources of pollution which may involve habitat degradation, chemical contamination, or water withdrawal (Barbour et al. 1999). Biological communities integrate the effects of different stressors and thus provide a broad measure of their aggregate impact. Communities integrate the stresses over time and provide an ecological measure of fluctuating environmental conditions. Most importantly, the status of biological communities is of direct interest to the public as a measure of a pollution free environment. Living organisms also act as short-term predictors of long-term ecological effects through integration into a suite of measurements and understanding at different levels of contamination.

Biomonitoring typically takes two approaches: passive biomonitoring and active biomonitoring. Passive biomonitoring uses organisms in their natural environment to evaluate environmental health, while, active biomonitoring includes all methods which insert organisms under controlled conditions into the site to be monitored. It is important for the purposes of comparing individual populations in contaminated environments to those in non-contaminated environments (Muposhi, 2009). Indigenous organisms are considered continuous monitors of environmental quality and can help in the detection of short-term environmental variations (Schmitt and Dethloff, 2007). In aquatic ecosystems, these assessments often focus on invertebrates, algae, macrophytes (aquatic plants), fish, or amphibians (Wepener, 2008). Passive biomonitoring is usually done together with habitat assessment. Biological potential is limited by the quality of the physical habitat, forming the template within which biological communities develop (Dias- Silva et al. 2010). The basic structure of the surrounding physical habitat influences the quality of the water resource and the condition of the resident aquatic community (Dwarf, 2007; Kleyhans et al. 2008). Habitat integrity assessment is therefore an integral part of any determination of the ecological reserve of a river. Benthic macroinvertebrates were used as indicator organisms in passive biomonitoring since they are ubiquitous (Dallas, 2007) inexpensive to sample, and found in nearly all aquatic ecosystems. Important inferences about the health of the stream can be made by examining the relative abundance and diversity of macroinvertebrates, as well as community responses to specific types of disturbance, which are termed "biological response signatures" (Barbour et al., 1999). Their sedentary nature helps in the detection of point source pollution or localized disturbance effects (Dickens and Graham 2002).

In this study, a combination of physicochemical characteristics, habitat assessment index, and the South African Scoring System Version 5 (SASS5) as passive biomonitoring tools were used to monitor the integrity of

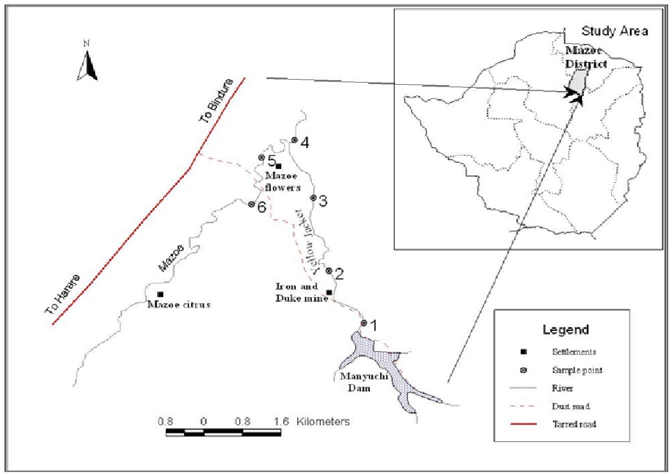

Figure 1. Map showing the study area in Mazowe and the six sampling sites along the two Rivers

the Yellow Jacket and Mazoe rivers. The SASS is a warning system for detecting pollution events that has advantages over traditional approaches because it can give some idea of the effect of stress on a community (Roux, 1997). It can also detect intermittent episodes of chemical pollution that might be missed using routine chemical analysis methods (Gratwicke, 1999). The objective of this stud was to assess the water quality of Mazowe and Yellow Jacket Rivers using physicochemical analysis and passive biomonitoring.

MATERIALS AND METHODS

Study Area

The study area is located 17° 28' S and 31° 15' E, about 50 km North of Harare. The area is part of the highveld of Zimbabwe, and receives an average rainfall of 800 to 1000 mm per annum, which normally falls between the months of November and April. The study area experiences dry cold winters from May to July during which average temperatures are 15°C and the summer season runs from November to April with average daily temperatures of 30°C.

The Yellow Jacket and Mazowe Rivers run through a range of mountain called the Iron Mask Mountain Range, which stretches from the Iron Cap Mine area in the South to the Shamva area in the North-East, forming an actuate structure (Ravengai et al. 2005). The Yellow Jacket River cuts through the Iron mask range at Iron Duke Mine and it flows north into the larger Mazoe River, which flows along the Western edge of the range (Ravengai et al. 2005).

The study area is covered Miombo woodland and the dominant tree species include *Brachystegia spiciformis, Brachystegia bohemii and Julbernadia globiflora,* with some *Acacia* species scattered within the woodland. Patches of dense, impenetrable *Lantana camara* are found along the Yellow Jacket and Mazoe riverbanks. Tall thick grasses characterize the area during the wet season. Fersiallite deep red clay (*Inceptisols*) derived from ipidiorite is the dominant soil type (Mapanda et al. 2007). Commercial crops grown include citrus, tobacco, cotton, vegetables, wheat, maize and floriculture farming.

Study Design

The study was carried out between January and March 2011 which coincided with the high flow period. A total of 6 sites were sampled; three along the Yellow Jacket River, two along the Mazowe River and the other site after the confluence of the two rivers (Figure 1). The sites were approximately 2 km from each other. Site one saved as the control while sites 2 to 6 were the experimental sites.

Water Sampling

Physicochemical parameters were measured fortnightly during the three month sampling period. Water samples were collected in 500 ml plastic bottles which had been washed with a phosphorous- free detergent, rinsed with distilled water and left to stand overnight in hydrochloric acid before being rinsed again in distilled water. Samples for metal analysis were filtered onsite through a 0.45 µm membrane filter and fixed for preservation with 1% (HNO_3) to pH just below 2. Filtering was done to remove suspended organic and inorganic solids that could react with dissolved ions and alter the chemistry of the water. The water bottles were then put on ice blocks and taken to the Environmental Management Agency (EMA) Laboratory in Harare for chemical analyses. The concentrations of metals iron (Fe), manganese

Table. 1. Habitat Integrity Classes (Dallas, 2005).

Class	Condition	Description	% Score
A	**Excellent**	Unmodified, natural.	**90-100**
B	**Very Good**	Largely natural with few modifications. A small change in natural habitats and biota may have taken place, but the assumption is that ecosystem functioning is essentially unchanged.	**80-89**
C	**Good**	Moderately modified. A loss or change in natural habitat and biota has occurred, but basic ecosystem functioning appears predominantly unchanged.	**60-79**
D	**Fair**	Largely modified. A loss of natural habitat and biota and a reduction in basic ecosystem functioning functioning is assumed to have occurred.	**40-59**
E	**Poor**	Seriously modified. A loss of natural habitat and ecosystem functioning is extensive.	**20-39**
F	**Very Poor**	Modifications have reached a critical level, and there has been an almost complete loss of natural habitat and biota.	0-19

Table 2. The scores used to determine the condition of a site using the three indices of SASS 5 adapted from Thirion et al. (1995) and Phiri (2000).

Condition	Habitat Score	SASS Score	Average Score per Taxon
Excellent	>100	>140	7
Good	80-100	100-140	5-7
Fair	60-80	60-100	3-5
Poor	40-60	30-40	2-3
Very poor	< 40	< 30	<2

(Mn), lead (Pb), copper (Cu), cadmium (Cd), cobalt (Co), nickel (Ni) and zinc (Zn) were determined using flame atomic absorption spectrophotometry (ASS). Nitrate and phosphate concentrations were determined using the spectrophotometric method (Greenberg et al. 1992).

At each site temperature (℃) and conductivity (µS/cm) were measured *in situ,* using a YSI-30 Salinity-Conductivity-Temperature meter. Dissolved oxygen (DO) (mg/L) was measured using a YSI-55 DO meter, and pH was measured with a Fisher Scientific Acumen AP60 portable pH meter. These measurements were taken three times on each sampling occasion and average readings recorded.

Habitat Assessment

This was done using The United Nations Environmental Protection Agency (USEPA)'s Habitat Assessment Index (HAI) using the Habitat Assessment Field Data sheets designed by Barbour et al. (1999). The assessment was done monthly from January to March.

The assessment was done concurrently with sampling for macroinvertebrates. The assessment involved rating 10 habitat parameters (epifaunal substrate/ available cover, embeddedness, velocity/depth regime, sediment deposition, channel flow status, channel alteration, frequency of riffles, bank stability, vegetative protection and riparian vegetative zone width) as optimal, suboptimal, marginal and poor. All these parameters were evaluated and rated on a numerical scale from 0-20 for each sampling reach (20 m along the river on either side) at each site. The ratings were then totaled and expressed as a percentage to provide the final habitat quality ranking using Kleynhans's, (1999).

Macroinvertebrate Sampling

Each site was sampled three times for benthic macro invertebrates. A month was allowed between sampling occasions so that the sites would recover from the previous sampling occasion. A KC Denmark square hand frame of dimensions, 0.3 m length, 0 .3 m height x 0.3 bag depth, and 250 µm mesh size with a 2 m wooden handle was used for collecting the macroinvertebrates. The net was positioned facing upstream and an area of 0.5 m x 0.5 m, (0.25 m^2) was then vigorously kicked for three minutes. In areas where the flow was insufficient to carry materials into the net, the net was swept through the disturbed area to collect dislodged materials. Live sorting and preliminary counting of readily identifiable taxa was carried out in trays within the field before preserving the sorted macroinvertabrates in 70% isopropyl alcohol for laboratory identification using

Table 3. Water physicochemical parameters measured in situ at the six sites during the six week sampling period.

Site	Temperature (℃)	pH	Conductivity (µS/cm)	Dissolved oxygen (mg/L)
1	22.9a	7.7a	310.5a	5.0a
2	26.1b	5.0b	696.6b	4.5bc
3	25.3bc	5.5c	677.1c	3.9b
4	24.9c	6.9cd	346.3a	4.8ac
5	25.1bc	7.6a	366.9a	4.7c
6	24.9c	7.6a	406.8a	4.9c

Columns with different letters are significantly different (P < 0.05).

Table 4. Table 4. Metal and nutrient concentration (mg/L) along the sampling sites.

Site	**1**	**2**	**3**	**4**	**5**	**6**	**Lsd (P<0.05)**
Cd	0.01	0.45	0.51	0.20	0.02	0.02	0.08
Co	0.02	0.32	0.33	0.36	0.30	0.29	0.04
Cu	0.02	1.54	1.53	1.16	0.04	0.04	0.79
Fe	0.77	5.26	5.40	1.92	0.91	0.90	0.39
Mn	0.08	0.35	0.34	0.11	0.41	0.41	0.09
Ni	0.18	0.61	0.54	0.33	0.31	0.30	0.12
Pb	0.13	0.20	0.19	0.16	0.12	0.15	0.05
Nitrates	0.5	2.5	2.7	11.0	12.0	14	3.1
Phosphates	0.1	0.25	0.2	2.1	3.9	4.0	0.5

Lsd = the least significant difference (P<0.05).

identification keys.

All benthic macro invertebrates samples were examined with an Olympus binocular microscope, using an objective lens with a zoom magnification of 0.7 to 0.4 and eye pieces of 15x.Where necessary, an adapter lens was fitted to the objective lens to double the magnification. The SASS protocol envisages recognition of benthic macroinvertebrates at the family level with the aid of identification keys and manuals. The SASS index was calculated by assigning a score to each of the families present based on their sensitivity to pollution (Day and de Moore, 2002; Tafangenyasha and Dube 2008). The SASS score was the sum of the scores of each invertebrate taxon present while the Average Score Per Taxon (ASPT) is the SASS score divided by the number of families. The condition of each site was the determined according to the criteria in Table 2.

Data Analysis

Statistical analysis was performed using Statistical Package for Social Scientists (SPSS) version 16.0. Data were first tested for normality and homogeneity of variance using the Kolmogorov- Smirnoff and Levene's tests respectively. A one way analysis of variance was computed for water physicochemical parameters, habitat assessment and SASS data to ascertain the differences over time and site. The least significant differences (lsd) was used to separate means at P < 0.05. All values are expressed as mean ± standard error.

RESULTS

Physicochemical Parameters

The water in the Yellow Jacket River at sites 2 and 3 appeared yellowish/ brownish due to a mixture of yellow-boy/ochre and sediment, while the water at site 1 was clear. The water in Mazoe River at sites 5 and 6 appeared brownish-greenish due to sediment and algae. Physicochemical parameters that were measured *in situ* at all the sites and in the control are given in Table 3. Sites 2 and 3 generally showed high water temperature and electrical conductivity but low pH and dissolved oxygen. The mean temperature of the water at site 1 was significantly lower (P < 0.05) than the mean water temperature at sites 2, 3, 4, 5 and 6.

The mean concentrations of selected metals at the sampling sites are shown in Table 4. The heavy metal concentrations were lowest at control sites, which was comparable to site 5 and 6 except for Mn. The heavy metal concentrations were significantly higher at site 2 and 3 when compared to the other sites, Table 4. The mining activity at IDPM was associated with an increase

Table 5. Average SASS score and number of macroinvertebrates families and average score per taxon (ASPT) recorded at the six sampling sites.

Site	1	2	3	4	5	6	Lsd
SASS score	74.7a	18.3d	25.7d	46.7c	70a	67.3b	7.4
No of families	21.0a	7.0d	8.7d	14.7c	19.7a	19.0b	1.8
ASPT score	3.6a	2.6d	2.9c	3.2b	3.6a	3.5a	0.2

Lsd = Least significant differences (P<0.05)

in heavy metal concentrations along the river. Although Mn increased at site 2 and 3, the highest concentrations were not at the IDPM, but at site 5 and six.

The levels of nitrates and phosphates measured along the sampling sites are shown in Table 4. The concentration of all the nutrients was significantly lower at Site 1 ($P < 0.05$) than the rest of the sites. The concentration along the river from site 1 and the highest concentration of nutrients were recorded at sites 4, 5 and 6.

Habitat Assessment

The SASS score at site 1 and site 6 was significantly higher ($P < 0.05$) than the rest of the other sites. The habitat score was lowest at site 2 and 3. The number of macroinvertabrates families showed the same trend as the SASS score. Site 1 and 6 had the highest number of families, while site 2 and 3 had the lowest number of families (Table 5). The Average Score Per Taxon (ASPT) was highest at site 1, 5 and 6 and was least at site 2. The habitat assessment shows lower score at the disturbed site 2 and 3 and was highest at the control site and the less disturbed site 5 and 6.

Benthic Macroinvertebrate using SASS 5 classification

The benthic macroinvertebrates classes, orders and families that were identified at all the sampling sites and their scores based on their sensitivity to pollution are shown in Table 6. Site 1 recorded the highest values in all the three SASS 5 parameters (SASS score, number of families and average score per taxon) while site 2 had the lowest values.

The *Dipteran chironomidae* was the most abundant family at all the sites while the relatively pollution sensitive *Caenidae*, *Cordulidae*, *Gomphidae*, *Dytiscidae* and *Gyrinidae* were absent at Sites 2 and 3.

DISCUSSION

The water in the Yellow Jacket River was clear at Site 1, but turned yellowish/ brownish downstream of the IDPM. This was most likely due to a mixture of ochre produced by acid mine drainage and sediment from stream bank cultivation as well as gold panning in the river banks. The water in Mazowe River at Site 5 and 6 appeared brown-greenish, suggesting pollution by sediment, slight eutrophication and algal growth as a result of high nitrates and phosphorous concentrations. The mean pH, of the water at site 1 was slightly alkaline (7.67), though previous studies (Magadza and Masendu, 1986; Gratwicke 1999; Ravengai et al., 2005) reported neutral pH at this site. Site 2 and 3 recorded the lowest mean pH values of 5.0 and 5.5, respectively. Water acidity is known to increase the solubility, availability and toxicity of metals in aquatic ecosystems (Akpor and Muchi, 2011). These pH values were outside the WHO guidelines for drinking water and EMA acceptable range of 6-9 and also outside the range associated with most natural waters (6.5 – 8.5) as well as the pH range for aquatic life (6.5 to 9.0) (Chidya et al. 2010).

The low pH suggests pollution by acid mine drainage since these sites are close to Iron Duke Mine evaporation ponds. Lower pH values of between 3 and 4 were recorded in previous studies (Gratwicke, 1999; Ravengai, et al. 2005). The slightly higher pH values recorded during the study could due be to the dilution effects of the increased water volumes since this study was carried out during the rainy season when river water flow was high. Site 4 which was after the confluence of the Yellow Jacket and Mazoe Rivers recorded a relatively neutral mean pH of 6.9. This could be attributed to the dilution as this site was after the confluence of the two rivers.

Sites 2 and 3 recorded the highest conductivity due to pollution by metals that are solubilised by increased acidity due to acid mine drainage from IDM. Intensive gold panning going on around these sites could also have contributed to the elevated conductivity by increasing AMD and sediments in the water. Melembe, (2010) showed increased conductivity of water in Nhamucuarara River due to gold panning. Elevated conductivity values at Sites 4, 5 and 6 could be due dissolved salts, especially fertilizers from the surrounding farms and estate. Electrical conductivity at all sites was, however, found to be within the WHO and EMA acceptable limit of <1000 µS/cm.

Dissolved oxygen indicates the ability of a water body

Table 6. Macroinvertebrates classes, orders and families identified at the sampling sites

Class	Order	Family	SASS Score
INSECTA	Ephemeroptera	Caenidae	6
	Trichoptera	Hydropsychidae	4
	Odonata	Coenagrridae	4
		Cordulidae	8
		Libelludidae	4
		Gomphidae	6
	Hemiptera	Belostomatidae	3
		Gerridae	5
	Coleoptera	Dytiscidae	5
		Gyrinidae	5
	Diptera	Chironomidae	2
		Culicidae	1
		Heleidae	3
OLIGOCHAETA	Lumbriculida	Not identified	1
GASTROPODA	Mollusca	Hydrobiidae	3
		Physidae	3
		Plarmobidae	3
CRUSTACEA	Decapoda	Not identified	3
PLECYCOPODA	Heterodonta	Sphaeridae	3
HIRUDINEA	Rhynchobdellida	Glossiphoniidae	3

to support aquatic life (Amutenya et al. 2010) since it is required for respiration by aerobic life forms (Akpor and Muchie 2011). The mean DO at Sites 2 and 3 was significantly lower ($p < 0.05$) than at the rest of the sites. This is perhaps a result of the high temperature of the water at these sites since high temperature reduces the solubility of oxygen in water. The actual quantity of oxygen that can be present in solution is also governed by partial pressure of the atmosphere and the concentration of impurities such as metals and suspended solids in water (Amutenya et al. 2010). The low DO at Sites 5 and 6 could be a result of eutrophication which is cause by high nitrogen and phosphorus from surrounding farms. The DO at all the sites was however within the EMA minimum acceptable limits.

Heavy metals are naturally present in small quantities in all aquatic environments; it is almost exclusively through human activities that their concentrations are increased to toxic levels. The mean concentration of most of the metals was significantly high at Sites 2, 3 and 4. Elevated metal levels at these sites were likely due to the high acidity which might be a result of the effect of acid mine drainage (AMD) from IPM. Jennings et al. (2008) observed that the concentration of metals such as copper, cadmium, zinc, iron and a range of trace metals such as lead, arsenic, aluminium and manganese become elevated in waters with low pH. Previous studies (Gratwicke, 1999; Williams and Smith, 2000, Mapanda et al. 2007) also reported high concentrations of Al, Zn, Cu, Co, Ni, V, Cr and Cd at Sites 2 and 3 in the Yellow Jacket River.

The concentration of metals decreased downstream from IDPM. This may be due to dissolved metals being removed from solution as they sorbed onto suspended particles to form colloids or precipitate on the stream bottom. In a similar study, Williams & Smith, (2000) observed that downstream from IDPM hydrous oxide precipitate formed the dominant mechanism of metal attenuation in water. The concentrations of iron, manganese and lead were observed to be in the extreme hazard zone of the EMA regulations, raising serious environmental and human health concerns. Because manganese is not as abundant naturally, an increase in manganese is always related to AMD, and is a good indicator of AMD impacted streams (Cherry et al. 2001).The danger of heavy and trace metal pollutants in water lies in that they persist in natural ecosystems for extended periods and they have the ability to accumulate in successive levels of biological food chains, producing potential human health risks and ecological disturbances (Akpor and Muchie, 2011). Of serious concern is the concentration of cadmium, which at most sites was in the extreme hazard zone and, therefore beyond the EMA maximum permissible concentration of 0.3 mg/L as well as the WHO permissible limits for drinking water.

Cadmium is of environmental and human health concern due to its carcinogenic and endocrine disrupting effects in humans (Rafau et al. 2007). It accumulates mainly in the kidney and liver and high concentrations have been shown to lead to chronic kidneys dysfunction as well as induction of cell injury and death by interfering with calcium regulation (Rafau et al. 2007). Apart from the health implications of lead and cadmium, these metals and other elements such as zinc form a toxic ‘soup’ that often acts synergistically in their harm to organisms (Rafau et al. 2007).

The concentrations of nitrates and phosphates were found to be significantly lower ($p < 0.05$) at site 1 compared to all the other sites. The concentrations of nitrates and phosphorous was highest at sites 5 and 6 which were along the estate, and this may be attributed to the use of nitrogenous and phosphorous based fertiliser at the estates.

Habitat Assessment

None of the sites had a habitat quality stable enough to be classified as good using the classification system shown in Table 1. Site 1 had the highest habitat score of 60.56 which is classified as fair. Site 1 was comparable to Site 1 was comparable to Site 5 which also had a fair habitat quality. Canopy cover shaded the water surface only partly. Riparian vegetation on one bank consisted of dense impenetrable banks, while the other consisted mainly of grasses. The low habitat scores at sites 2 and 3 was due to high levels of sediment due to ochre with smoothers the river bed with very fine silt. This produces an unstable and continuously changing environment that becomes unsuitable for many aquatic organisms (Dias-Silva et al. 2010). The sites had low pool variability, largely modified riparian vegetation, very low pool frequencies, few bends and poor bank stability. River banks around most sites were also severely degraded due to rampant gold panning as well as stream-bank cultivation.

Benthic Macroinvertebrates (SASS 5)

A total of six benthic macroinvertebrate classes, 11 orders and 21 families were identified at the sampling sites (Table 6). The highest number of benthic macroinvertebrate families recorded at site 1 was 21, and significantly higher ($p < 0.05$) than the number recorded at all the other sites. A fairly high mean SASS score of 74.67 was also recorded at site 1 and site 5. However, the ASPT was 3.56, and the water quality at the site was classified as being fair (Table 5). This was confirmed by the presence of a low number of benthic macroivertebrates orders associated with clean water such as *Ephemeropterans, Plecopterans and Coleopterans*. Only 2 *Ephemeroptra* families (*Baetadae* and *Caenidae*), 1 *Trichoptera* family *(Hydropsychidae)* and 2 *Coleoptera* families (*Dytiscidae* and *Gyrinidae*) identified at the site. The site scored an ASPT score of 5.2 ten years back (Gratwicke, 1999) and this deterioration in the ASPT over the years may be due to deteriorating in the overall water quality or failure of macroinvertebrate colonization. The site was however, fairly undisturbed when compared to the other sites since it is just below the dam and is in a gorge with little space for gold panning and human cultivation on the banks of the Yellow Jacket River.

The number of benthic macroinvertebrates families diminished by 300 % from Site 1 to 2 (21 to 7), while the ASPT dropped by 74 % from 3.56 at Site 1 to 2.62 at site 2. Basing on Table 2, site 2 had very poor quality. Gratwicke (1999) identified only five families at this site showing a gradual decrease in the benthic macroinvertebrate structure over time. The site is on the drainage side of IDPM, and could be heavily impacted by AMD. This study has also showed that the site has low pH and heavy metals. The highly acidic sulphate rich drainage also dissolves additional minerals and metals from within from within the surrounding rock strata or the stream bed (Last, 2001). Acidic pH and heavy metals individually are known to adversely affect benthic macroinvertebrate communities. No *Ephemeroptera, Plecoptera* and *Coleopteran* families were identified at the site since these are highly sensitive to AMD, and are almost eliminated from impacted streams (Last, 2001). Gratwicke (1999) also found a lower SASS score, fewer families and low ASPT at the same site. Elimination of *Ephemeroptera* and *Plecoptera* and an increase in tolerant organisms such as *Diptera*, (*Chironomidae*), *Megaloptera* (*Solidae*), *Hemiptera* and *Coleoptera* (*Dytiscidae*) also pararrels changes in community structure and composition seen in other studies of AMD (Soucek et al. 2000; Last, 2001) or metal impacted streams (Clements et al. 2000).

There was no significant difference in the SASS score and ASPT between site 2 and 3 as Site 3 was also presumed to be impacted by heavy metals and AMD. The site was also dominated by the acid tolerant families from the orders *Diptera* and *Coleoptera (*mainly the family *Chironomidae).* In a similar study, Soucek et al. (2000) found that the benthic macroinvertebrate community in an ADM impacted tributary of Monday Creek (Southeastern Ohio) consisted of species from only three orders: *Megaloptera, Diptera* and *Coleoptera*.

Site 5 and 6 had mean SASS scores of 70.00 and 67.33 as well as ASPT values of 3.56 and 3.54 respectively. Basing on Table 2, the condition and water quality at these sites can therefore be classified as fair. Gratwicke (1999) also found almost the same but slightly higher ASPT values (3.79 and 4.00) at the two sites, respectively. These low values in mean SASS score and ASPT could be a result of pollution of the Mazoe River by agrochemicals from the commercial farms in its catchments. Tafangenyasha and Dube (2007) also found low ASPT values in Runde River in a study to that examined the impacts of agricultural pollution on nutrient concentrations and benthic fauna in a semi-arid tropical lowveld region of Southeast Zimbabwe. Similarly, Chakona et al. (2008a) and Chakona et al. (2008b) found that intensive agricultural activities within the mid-reaches of the Nyaodza River in northern western Zimbabwe caused severe land degradation of the stream’s physical habitat which coincided with a significant decline in macroinvertebrate richness, diversity and abundance.

The ASPT did not show any significant variation among the sample sites. ASPT is a measure of the average sensitivity of all the macroinvertebrates families present to pollution. It ranged from 2.62 on the heavy impacted sites to 3.56 on the less impacted sites. When SASS scores are low (< 50), ASPT becomes variable and a single moderate scoring invertebrate family will have considerable influence on ASPT (Don et al. 2007). At many sites sampled, the SASS score was lower than 50, and ASPT was highly variable. This intrinsic variation could have had a confounding influence on statistical tests and there were no statistically significant differences in ASPT although the mean SASS score and mean number of families differed among sites.

All the sites were dominated by pollution tolerant macroinvertebrate families such as *Chironomidae*. The family *Chironomidae* includes species that are tolerant to various kinds of pollution (Muisa et al. 2010). Although *Chironomidae* were not identified below family level during sorting, differences in morphology revealed that the AMD impacted sites were often dominated by a few species, whereas the agricultural run-off impacted sites had more diverse assemblages. Site 4, which was after the confluence, had intermediate values in all the SASS score and ASPT as well as macroinvertebrate families, suggesting fair water quality after neutralisation of acidic water from the Yellow Jacket River by alkaline water from the Mazowe River.

CONCLUSIONS

The findings of this study suggest that Mazowe and Yellow Jacket River are negatively affected by mining and farming activities. The habitat scores at all the sites were low since the banks and catchment of the rivers are subject to various natural and anthropogenic disturbances that affect water quality. The SASS score and ASPT values were low except for site 1 and 5. The results strongly suggest that mining and agricultural communities are affecting the integrity of the two river systems. The South African Scoring System Version 5 (SASS 5) and the United States Environmental Protection Agency (UNEPA) Habitat Assessment Index (HAI) were effective and sensitive tools to assess water quality in aquatic systems.

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