IMPACT OF ALLUVIAL DIAMOND MINING ON MACROINVERTEBRATE COMMUNITY STRUCTURE IN THE LOWER VAAL RIVER, NORTHERN CAPE PROVINCE.

By

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RESEARCH REPORT

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ABSTRACT

Macroinvertebrates and physico-chemical parameters were assessed at four sites in the lower Vaal River. The primary aim of the study was to determine the impact that alluvial diamond mining activities have on aquatic macroinvertebrate community structure using the South African Scoring System version five (SASS 5), as an index of the river's ecological status. The macroinvertebrates were sampled using the SASS5 method and the results were analysed together with selected physico-chemical water parameters and Integrated Habitat Assessment. The results indicated that habitat played a major role in the presence of macroinvertebrates. Macroinvertebrate diversity was calculated using the Shannon-Wiener Index. A total of 28,167 macroinvertebrates belonging to 36 families were recorded throughout the sampling seasons. The highest abundance was recorded at Site 4, the most impacted site, but in turn this site had the lowest diversity. Chironomidae was the most abundant family (2,588 individuals) and the least abundant were the Chlorocyphidae and Velidae, each having five individuals. High numbers of taxa were noted amongst the Simuliidae, Corbiculidae, Physidae, and Oligochaeta. Site 4 was the only site where the presence of livestock was seen, and construction trucks were observed driving over the biotopes crossing over banks of the river, further adding pressure on the existing alluvial diamond mining impacts that contributed to low presence of macroinvertebrates. Generally, the alluvial diamond mining activities had severe impacts on the riverbed and changed the river flow regime and water quality. The hypotheses that the water quality of the Vaal River is negatively impacted by the alluvial diamond mining practices, and that the mining activities negatively impacted on macroinvertebrate community structure were supported.

DECLARATION

"I declare that the research report hereby submitted to the University of the Witwatersrand, Johannesburg for the degree Master of Science in Environmental Sciences has not previously been submitted by me for a degree at this or any other university; that it is my work in design and in execution, and that all material contained herein has been duly acknowledged and referenced accordingly''

Signature 28/05/2018

Date

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CHAPTER 1

1. Introduction

Water quality assessments in South Africa and elsewhere have primarily been determined by conducting water chemical analyses and measurement of physical water parameters (Roux *et al*. 1993). Chemical analyses of water can provide a measure of the concentrations or loads of individual substances in the aquatic ecosystem. However, since chemical analyses only considers the water flowing at the time of sample collection (Davies and Day 1998), they can only be accurate for that particular sampling time (Bertasso 2004). Furthermore, chemical and physical water analyses are costly and need trained and expert personnel. Taking this into consideration, other techniques for determining the water quality of rivers, dams and wetlands have been developed globally. One of these alternative methodologies is biomonitoring, which is the use of biological organisms as time-integrated indicators of the health of the environment (Dickens and Graham 2002).

Macroinvertebrates live either permanently or during part of their life cycles in freshwater ecosystems (Dickens and Graham 2002; Benetti and Garrido 2010) and are active at or near to the water surface, within fringing vegetation, and the benthos of aquatic ecosystems (Hauer and Resh 1996; Gerber and Gabriel 2002). As macroinvertebrates are relatively easy to sample and identify to family level, they are considered good indicators of river biota integrity because they are localised and largely immobile compared to fish (Dickens and Graham 2002). Furthermore, based on their tolerance or intolerance to pollution, their diversity in streams can indicate the health of the systems Parmar *et al.* (2016). Indeed, macroinvertebrates are commonly used to assess the quality of aquatic ecosystem health due to their great diversity of form and habits (Rosenberg and Resh 1993).

The South African Scoring System (SASS) is a long-standing macroinvertebrate monitoring tool to assess river health. Dickens and Graham (2002) adapted the South African Scoring System version 4 (SASS4) developed by Chutter (1994) and improved it into the South African Scoring System version five (SASS5). The SASS5 is a rapid bioassessment tool used to determine the health of flowing rivers, also relying on organisms whose occurrence, absence, richness and behaviour show the effect of a pollutant on the plants and animals of a certain area or region (Bonada *et al.* 2006). This bioassessment tool is used by river health practitioners nationally to provide reliable information regarding the ecological status of rivers and assessing water quality. Indeed, SASS5 is considered the backbone of the River Health Programme (RHP) now called the River Ecostatus Monitoring Programme (REMP), which falls within the National Aquatic Ecosystem Health Monitoring Programme (NAEHMP).

When environmental conditions deteriorate, sensitive macroinvertebrates can disappear whilst the most tolerant organisms will remain in the system. Therefore, variations in the assemblage composition of aquatic organisms raises red flags, but without necessarily pinpointing the source of pollution (Benetti *et al.* 2012). Aquatic biota are naturally, heavily reliant on the water bodies they inhabit (Chutter 1998). However, the physical and chemical quality of water can be described by a large number of variables. It is often very difficult to determine which single parameters, or combination of parameters cause the observed biological responses. The geomorphological features, hydrological regime, physico-chemical water quality, nature of the riparian and in-stream habitats represent some key factors that determine the health of a river ecosystem (Balance *et al*. 2001). It is difficult to assess each one of aforementioned factors in detail so the REMP focuses on selected drivers and ecological indicators that give a full representation of the larger ecosystem health.

Since normal biota reflect the changes of chemical and physical impacts done over a specific period of time, they are viewed as good bioindicators of overall ecological integrity (Dallas 2007). The diversity and abundance of macroinvertebrates also varies based on seasonal changes (Metcalfe 1989; Freund and Petty 2007). The abundance of macroinvertebrates depends on the quality of organic matter, substrate and water quality (Hussain and Pandit 2012). A factor such as habitat availability is a determining factor because different species occupy different habitats; for example species composition can vary between different substrates (Dickens and Graham 2002). This is because species composition depends on environmental factors such as the availability of habitat integrity which have a strong influence on the final SASS score, water flow regime and water quality (Dallas and Day 2007).

The SASS5 index can therefore provide an overall overview of both the previous and current situations in a stream and the long-term changes in water quality. This is mainly because the macroinvertebrates that are existing in a watercourse must have been in a position to endure any conditions the stream has been exposed to in the past and be able to provide a direct and cumulative status of river health (Davies and Day 1998; Dallas and Day 2004).

The quality of water in the Vaal River and some of its streams feeding into it downstream of the Vaal Dam are impacted by urban and industrial discharges, and mining return flows (Braune and Rodgers 1987). The upper Vaal River is moderately modified by urban development whilst the middle reaches are impacted mostly by mining and agricultural activities. A study by DWAF (2009a) showed a significant decrease in the water quality of the upper, middle and lower reaches, which eventually affects the general ecosystem health status of the system.

The lower Vaal River Water Management area lies below the Bloemhof Dam to confluence with the Orange River in Douglas, drainage areas of Northern Cape Province, the North West Province, south-west of the Free State Province and northern border of Botswana (DWAF 2004). The lower Vaal Water Management Area is less developed and the predominant land uses are agriculture, urbanisation, industrial growth and alluvial diamond mining (Ochse 2007; Mboweni and De Crom 2016). However, in 2000, Du Preez *et al.* (2000) showed that the water quality of the lower Vaal River had declined over the past 20 years and predicted that it would continue to deteriorate further due to wastewater coming from Johannesburg the economic hub of the country. In contrast, Ramollo and Moalusi (2010), using macroinvertebrates and water quality to bioassess the status of selected sites in the lower Vaal River, showed that the ecological health of the lower Vaal River was in good to fair condition, sometimes good to pristine condition in different seasons.

Essentially, these on-going developments have collectively led to the deterioration in the water quality of the Vaal River system (DWAF 2004). Such river deterioration is unacceptable considering South Africa being a drought-prone country, and requiring involvement from management authorities to ensure good water quality is accessible to all users in the system, and especially since the development will continue to increase in the Vaal area (DWAF 2009a). For example, the irrigation in the lower Vaal River contributes for 80% of the water use (DWAF 2011). The water in the lower Vaal River is diverted to the largest irrigation scheme (Vaalharts) where it is mainly used for the cultivation of maize, lucerne, pecan nuts, olives, potatoes, wheat and livestock farming (DWAF 2011; Van Rensburg *et al.* 2011).

The Department of Water Affairs, now the Department of Water and Sanitation (DWS), conducted a detailed water quality assessment of the Vaal River to develop an integrated water quality management protocol

(DWAF 2006). The study pinpointed a variety of water quality issues across the entire Vaal River that were either general in nature while other environmental issues were more restricted. The study confirmed that high levels of Total Dissolved Solids (TDS) and a noticeable increased salinity had the huge impact on water usage in the Vaal River. It also identified the localisation of emerging microbiological pollutants and increased concentrations of certain metals (DWAF 2006).

Nutrient pollution was the other serious water quality challenge in the Vaal River system. The problem was exacerbated by partially treated sewage water, raw sewage water inflows, and agricultural returns flows which increase nutrient loads. These were associated with increased algal blooms and growth of the invasive water hyacinth (*Eichhornia crassipes*). The study concluded that the upper Vaal River had good water quality but found that the Vaal Barrage, middle Vaal River, and lower Vaal River just downstream the confluence of the Harts River had only fair water quality with extremely high levels of TDS and nutrients (DWAF 2006).

According to DWAF (2006) the lower Vaal River was also impacted by upstream activities and cumulative impacted water coming from the Harts River. The Harts River originates in the North West Province near Lichtenburg, passes Taung, Pampierstaad, and eventually flows into Spitskop Dam from where it forms a confluence with the Vaal River at Delpoortshoop in the Northern Cape Province (DWAF 2009a). The water quality of the Harts River is highly impacted by sewage inflows from Sannieshof and agricultural return flows from the Vaalharts Irrigation Scheme (DWAF 2006). These caused the water quality to be nutrient-rich and have high salinity. This was corroborated by the studies conducted by Ferreira (2008) and Malherbe (2013). The macroinvertebrates studies of 2003 indicated that the overall water quality status of the upper reaches of Harts River remained in a good to fair condition while the lower Harts River was in a fair to poor condition (DWAF 2009a).

Mining plays the second biggest part in the economy of the Northern Cape Province, after agriculture, but poor management practices make many operations unsustainable. Due to the meandering nature of the Vaal River, diamond deposits are spread in the riparian zones and river channels beds by erosion that occurred millions of years ago (Chutter 1968). The alluvial diamond deposits are vigorously extracted by informal, large- and small-scale miners (Naidoo-Vermaak 2006; Ramollo 2011). Large-scale alluvial diamond mining operations use heavy machinery such as bulldozers and trucks to isolate potential gravel deposits yielding diamonds. They dredge instream, braided islands and on the riverbanks, destroying a vast area of riparian vegetation (Naidoo-Vermaak 2006; Heath *et al.* 2004; Ramollo 2011).

Sediments such as rock, gravel and sand are often stockpiled in the riparian zone and next to the river, and the area left unrehabilitated (Chutter 1968). Heavy rainfall events often erode these dumps into the rivers (Ramollo 2011) but the level of erosion is dependent on the duration, intensity of the rain, and the slope of the area (Newcombe and MacDonald 1991). If the soil is exposed to direct rainfall it is washed into the waterway very quickly thereby increasing turbidity and TDS (Jones *et al.* 2012). Barring two studies by Ferreira (2008) and Malherbe (2013) at the two sites in the lower Vaal River that showed water quality to be in fair condition, little is known how this quality affects aquatic fauna, such as macroinvertebrate community structure, and river health in general.

1.1. Aim of the study

1.1.1. To determine the impact of alluvial diamond mining on river health, using the community composition of macroinvertebrates in the lower Vaal River, Northern Cape Province.

1.2. Research questions

The major research questions of the study include:

1.2.1 Is the water quality of the lower Vaal River negatively impacted by alluvial diamond mining?

1.2.2 Does alluvial diamond mining impact on macroinvertebrate community structure in the lower Vaal River?

1.3. Objectives

The specific objectives of the study are;

1.3.1. To determine the water quality at the four selected sites in the lower Vaal River.

1.3.2. To determine the extent to which alluvial diamond mining contributes to the deterioration of water quality of the Vaal River.

1.3.3. To determine the community composition of macroinvertebrate taxa at the four selected sites.

1.3.4. Identify the presence and absence of macroinvertebrates at the four selected sites.

1.3.5. To assess the ecological status of the river based on the use of aquatic invertebrate taxa as indicators of ecosystem health.

CHAPTER 2

2. Materials and methods

2.1. Study area

The Vaal River is a vital water resource with a number of important tributaries along its length. It originates at Sterkfontein near Breyten in the Drakensberg escarpment, Mpumalanga Province. The Vaal River is controlled through the Grootdraai Dam in Mpumalanga, Vaal Dam (Vaal-Barrage Dam) in the Gauteng Province and Bloemhof Dam in the North West Province. It flows 1,415 km in the south west to meet with the Orange River at Douglas in the Northern Cape Province. Most of the tributaries of the Vaal River downstream from the Vaal Dam are in a critical state of ecological decline (DWAF 2006). The river upstream and downstream of the Vaal Dam, in the upper Vaal Water Management Area (WMA) is modified by anthropogenic activities such as agricultural return flows, sewage waste inflows, industrial and mining return flows (DWAF 2004). The localities of the four sampling sites, that are the focus of this study, fell within this low gradient area (Figure 1). The sites were sampled each season between April 2016 and December 2016; autumn, winter, spring and summer.

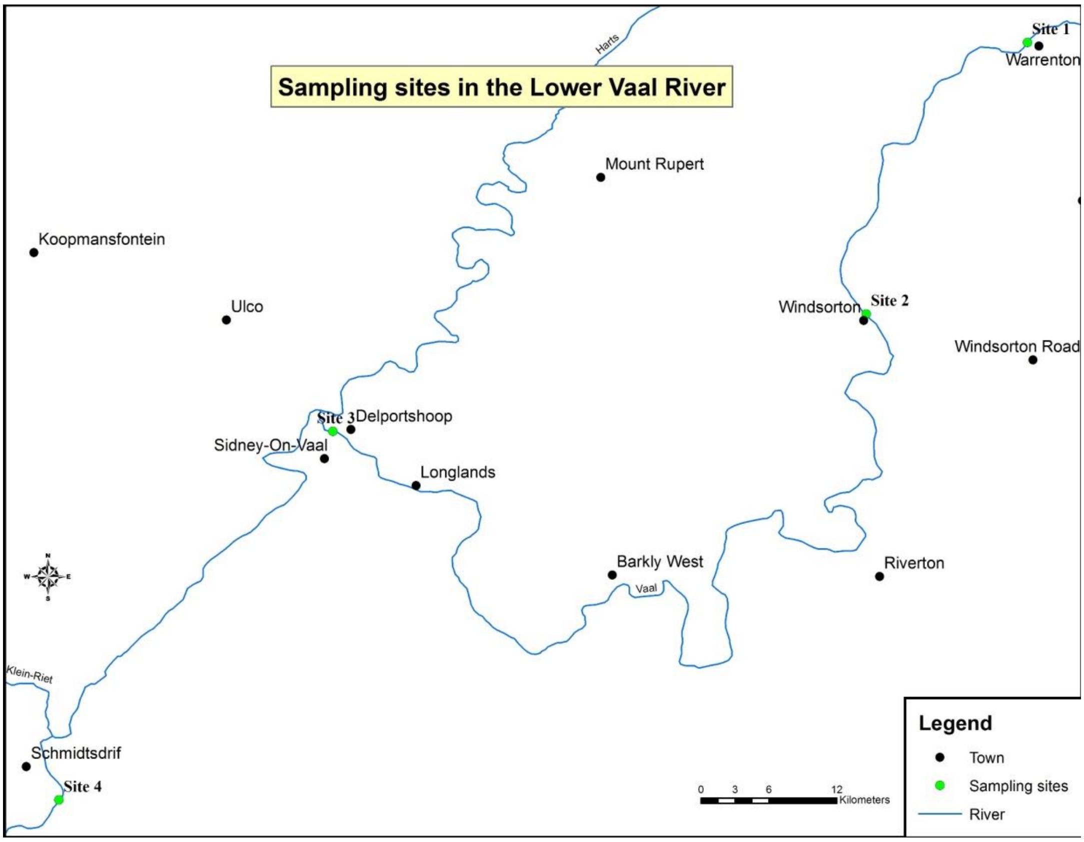

Figure 1: Showing the localities of the four sampling sites along the lower Vaal River.

2.2. Sampling sites

Site 1 (Warrenton) **Co-ordinates: 28°19'28.8S; 24°42'46.8E**

This site is located below the old single lane low-level Margaretha Prinsloo Bridge and the new bridge constructed by South African National Roads Agency Limited (SANRAL) on the N18 Road to Vryburg, in the year 2013. The river at this site was braided with boulders and bedrock. The vegetation type includes reeds (*Phragmites australis),* bulrush *(Typha capensis)* and river star (*Gomphistigma virgatum).* The riparian vegetation on both banks of the river is still intact and consists of mixture of vegetation e.g. cape willow (*Salix macronata*), buffalo thorn (*Ziziphus macronata*) and white karee *(Searsia pendulina)* but is infested by river red gum (*Eucalyptus camaldulensis)* and Spanish gold (*Sesbania punicea*). The instream of the river is colonised by invasive water hyacinth

(*Eichhornia crassipes*) and indigenous water grass *(Potamogeton pectinatus).*

Figure 2: Image of the uppermost site (Site 1) in Warrenton; reference site.

Site 2 in Windsorton Co-ordinates: 28°19'29.7S; 24°42'54.8E

The site is located about 500 m downstream of the bridge to Warrenton. The riparian vegetation was severely degraded by alluvial diamond mining. The river was diverted to prospect for diamonds and there is active mining activity that destroys and removes riparian vegetation, in-stream and out of stream. The disturbed riparian zone is colonized by invasive sponge-fruit salt bush *(Atriplex spp.)* and Spanish gold (*Sesbania punicea).*

Figure 3: Image of Site 2 located in Windsorton.

Site 3 in Delpoortshoop   Coordinates: 28°25'06.5S; 24°17'25.7E

The site is located below the culverts of an illegal road to alluvial diamond diggings that impede the river flow. The road is constructed with concrete culverts, steel pipes and scrap iron materials. Small stands of sweet thorn (*Vachellia karroo)* and sedges (*Cyperus papyrus)* occur at this site. The site is seriously modified by alluvial diamond extraction. The site is comprised of a length of cobbles created by alluvial diamond diggings.

Figure 4: Image of Site 3 located in Delpoortshoop

Site 4 in Schmidtsdrift Co-ordinates: 28°42’42.9S; 24°04ʾ20.4E

The site is located downstream of Schmidtsdrift Weir in the Vaal River. It is dominated by filamentous algae. The riparian vegetation is severely modified due to alluvial diamond mining activities. The remaining and recovering vegetation comprises of patches of reeds (*Phragmites australis*), bulrush (*Typha capensis*), sedges *(Cyperus papyrus)* and water grass (*Potamogeton pectinatus)* submerged in the waterway. The site is used informally for livestock drinking and fishing.

Figure 5: Image of Site 4 located at Schmidtsdrift

2.3. Data Collection

2.3.1. Physico-chemical parameters

The pH, water temperature, dissolved oxygen and electrical conductivity (EC) were determined *in situ* by means of a handheld multi-parameter instrument (YSI model 54 Combo meter) at all sampling sites. Total Suspended Solids (TSS) was measured using a Secchi disc and turbidity using a clarity tube.

2.3.2 Invertebrate Habitat Assessment System (IHAS)

The Invertebrate Habitat Assessment System (IHAS) was applied to determine the suitable major biotopes conditions for macroinvertebrates and to assist in interpreting the results of SASS5 (McMillan 1998). Scores for the IHAS index were interpreted using the guidelines prescribed by McMillan (1998) as follows:

- If the score is below 65% it is insufficient for supporting a diverse aquatic macro-invertebrate community.
- If the score is 65%-75% it is enough to support a relatively diverse macro-invertebrate community.
- If is more than 75% it is highly suitable for supporting a diverse macro-invertebrate community.

2.3.3. Macroinvertebrate collection

Macroinvertebrates were collected in different biotopes (stones in current and out of current, marginal and aquatic vegetation and gravel/sand/mud) using the South African Scoring System version 5 (SASS5) protocol as described by Dickens and Graham (2002). A 30 cm x 30 cm SASS net, a white flat-bottomed tray (approximately 30 cm x 45 cm size and 10 cm deep), waders, forceps, field identification book (Gerber and Gabriel 2002) and sample collection bottles were used for sampling. Once the collection was completed from the available habitats, the samples were washed down to the bottom of the net (repeatedly until the water passing through the net ran clear), then carefully tipped into the tray by inverting the net as per the standard protocol of Dickens and Graham (2002). The net was flushed out with clean water to make sure that biota did not remain in it. Some lingering macroinvertebrates on the net were put into the tray using forceps.

Sufficient clean water was then added into the white tray to immerse the sample as stated by Dickens and Graham (2002). Larger leaves, twigs, stones and other debris that hindered flow were removed and examined for macroinvertebrates and placed in another tray for counting and identification. They were shaken in the water and checked for clinging biota before being removed. Invertebrates were identified in the field with the help of a hand lens to family level and were released back into the river after counting and identification. Those macroinvertebrates that could not be identified on site were preserved in collection bottles with 70% ethanol for further identification in the laboratory.

2.4. Macroinvertebrate data analysis

The SASS 5 scores were calculated and the Ecological Categories (A-F) were determined as appears in Figure 6 and Table 1.

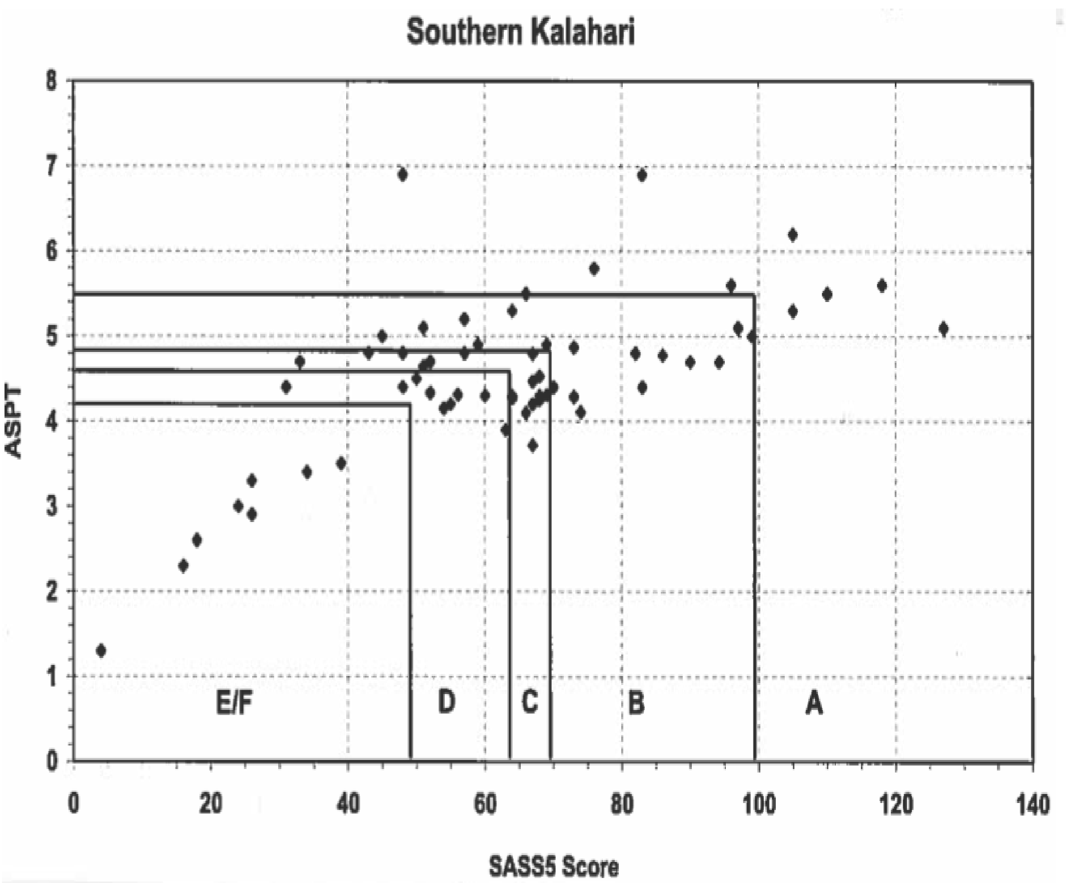

Figure 6: SASS 5 classification using biological bands calculated from percentiles for the Southern Kalahari Aquatic Ecoregion (Dallas 2007).

Table 1: Eco-classification model for determining the Present Ecological State (PES) for the Southern Kalahari aquatic ecoregion based on SASS score and Average Score Per Taxon (Dallas 2007).

SASS5 Score	ASPT	Condition	Class
>100	>5.5	Natural/unmodified	A
70-100	4.8-5-5	Minimally modified	B
65-69	4.6-4.7	Moderately modified	C
50-64	4.2-4.5	Largely Modified	D
<49	0-4.1	Seriously Modified	E/F

Macroinvertebrates collected were analysed using the Shannon-Wiener Index (H) to evaluate species diversity. The index was determined by both the number of species and the evenness (E = H' / log S) of distribution of individuals among those species (relative dominance). It was calculated using the following equation: $H = -\sum P_i * \ln P_i$, where Pi (relative abundance) = ni/N, ni = the number of individuals within a species (Magurran 1988), N = total number of individuals in all species, ln = natural log, = sum of the values for each species. Therefore percentage (%Pi) = the number of individuals within a species (ni) divided by the total number of individuals (N) present in the entire sample x 100. The relative abundance allowed comparisons to be made of diversity at all the sampled sites.

CHAPTER 3

3. Results

3.1. Physico-chemical parameters

3.1.1 pH

There are currently no targeted pH values for the aquatic ecosystem water quality guidelines (DWAF 1996c), instead the ranges for domestic use were used to interpret the results. The pH during this study ranged between 8.3 and 8.9. During April 2016, autumn, the pH values at Site 1 and 2 exceeded the Targeted Water Quality Range (TWQR) of 8.5 as described by DWAF (1996a) water quality guidelines for domestic use. In turn, at Sites 3 and 4 pH values were within range (Appendix A; Figure 7). During winter, the pH values at all sites were within the acceptable TWQR for domestic use. In spring, the pH values at Sites 1 and 4 were above the TWQR, whilst at Sites 2 and 3 values were within the TWQR as described by DWAF water quality guidelines for domestic use (1996a). During summer, the pH at all sites exceeded TWQR for domestic use. The water quality of the Vaal River at the four selected sites was generally alkaline and did not vary much with changes in seasonality.

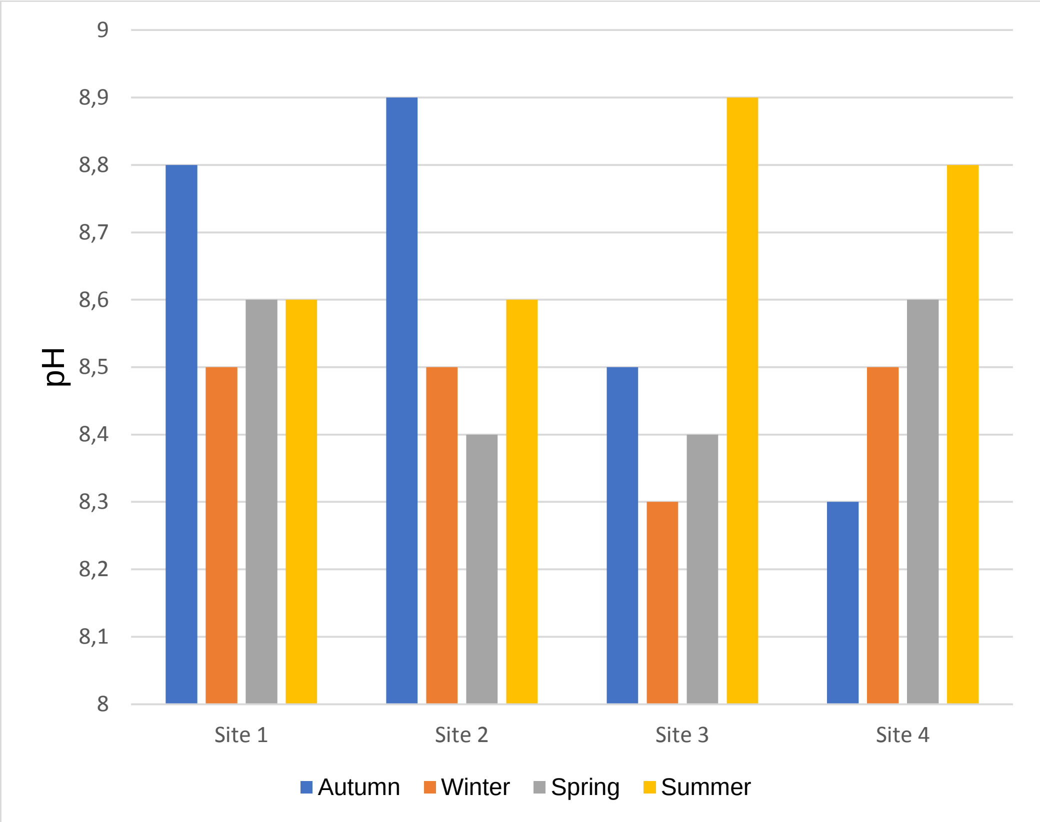

Figure 7: pH values recorded at the four sites

3.1.2 Dissolved oxygen (DO)

Generally, the dissolved oxygen (DO) mean values ranged between 4.6 - 8.3 mg l^{-1}. DO levels recorded at all sites throughout the sampling period were not considerably different except at Site 4 which ranged between 4.2 to 4.9 mg l^{-1} (Appendix A; Figure 8). The DO concentrations at Site 4 were relatively low throughout the study period but not considered to be hypoxic at the time of study; this was a possibility during night time when there is no instream photosynthesis.

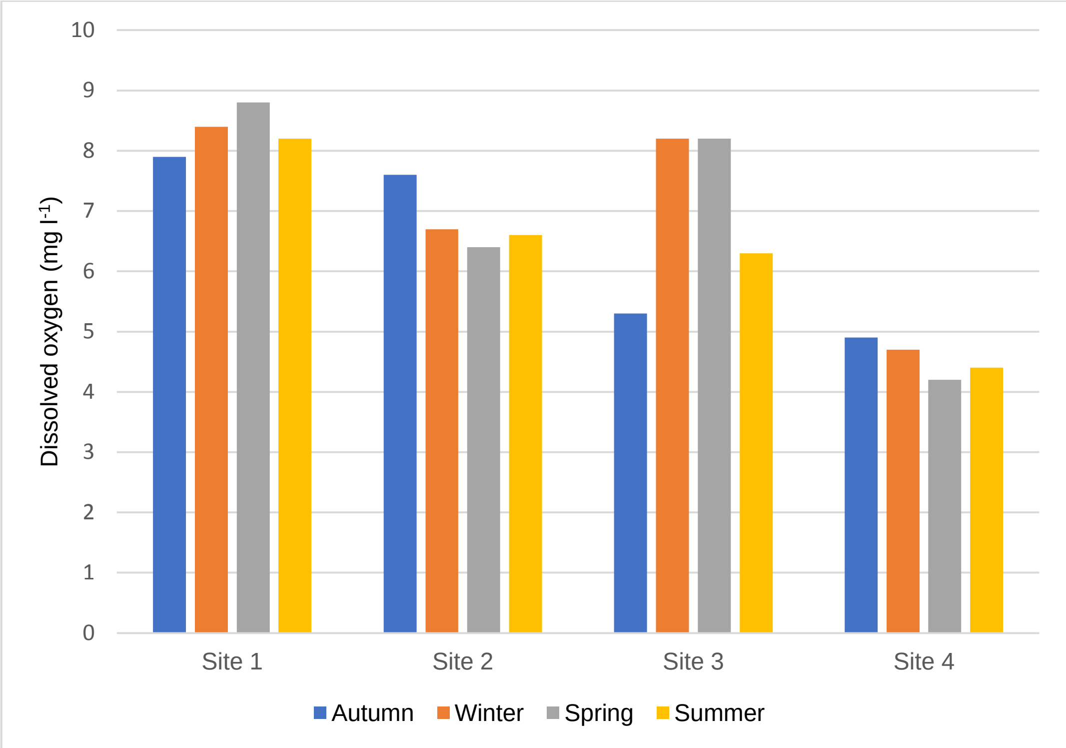

Figure 8: Seasonal Dissolved oxygen concentrations recorded at the four sites

3.1.3. Electrical conductivity

The Electrical Conductivity (EC) levels during the study period ranged between 76 mS m^{-1} to 120 mS m^{-1} (Figure 9), which would taste brackish. However, there are no electrical conductivity values listed in the aquatic ecosystem water quality guideline (DWAF 1996c), instead the values of domestic guidelines were once again used to interpret the results. Overall the highest levels were recorded at Site 4 throughout the study period (Appendix A: Figure 9).

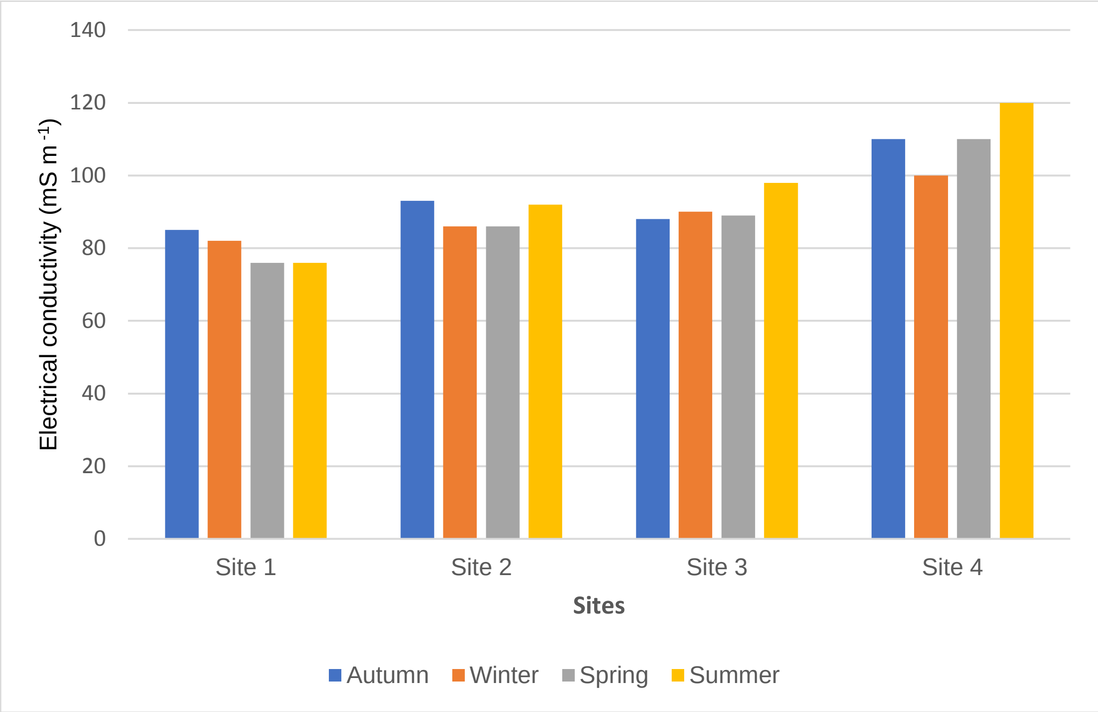

Figure 9: Seasonal Electrical Conductivity (EC) values at the four sampling sites

3.1.4 Water temperature

The water temperature ranged from 14°C to 27.2°C (Appendix A: Figure 10). As expected, lower temperatures were recorded in winter and the highest temperature in summer (Figure 10). Winter water temperatures did not exceed 18°C across sites and had an average temperature of 16.5°C. Summer temperatures were all above 25°C across sites with an average temperature of 26.4°C (Appendix A; Figure 10).

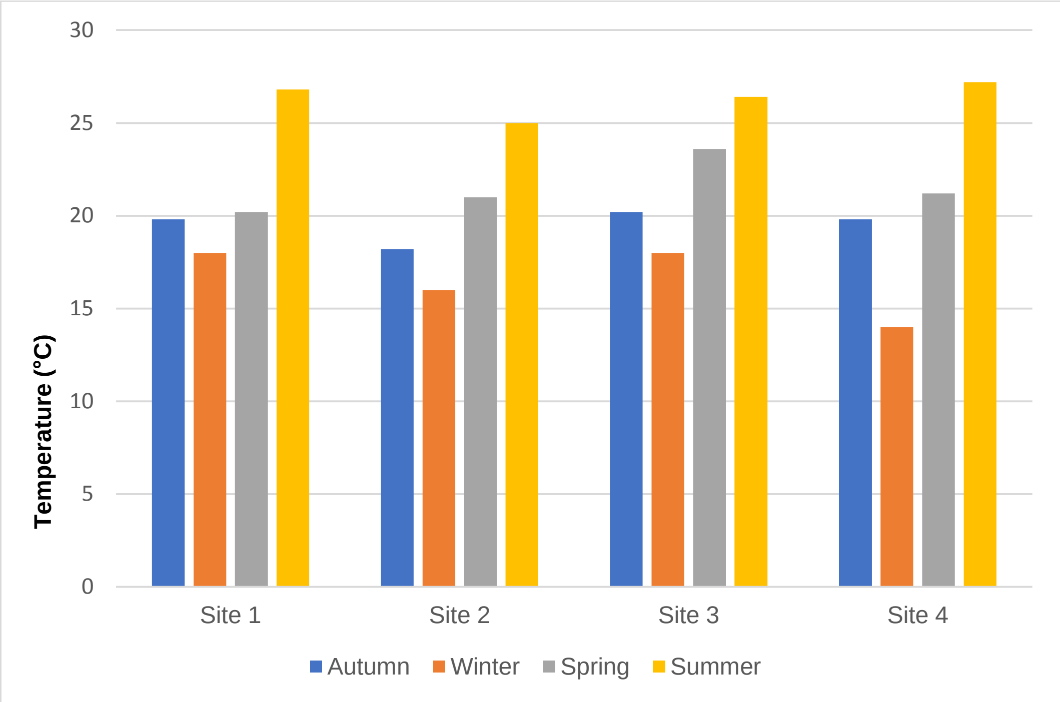

Figure 10: Seasonal water temperature at the four sampling sites

3.1.5. Total suspended solids

Mean Total Suspended Solid (TSS) concentrations differed from 18 mg l^{-1} to 21 mg l^{-1} between the four sites (Table 6; Figure 11). In autumn and winter, the TSS concentrations at all sites were constant (18 mg l^{-1}). The TSS concentrations at the three sites were constant in spring except at Site 4 which increased slightly to 20 mg l^{-1}. During summer, Sites 1 and 3 recorded a constant concentration of 20 mg l^{-1} whilst Site 2 and 4 recorded 18 mg l^{-1} and 28 mg l^{-1} (Appendix A; Figure 11), respectively. Overall, Site 4 recorded the highest concentration of 28 mg l^{-1} during summer. According to DWAF (1996c), the guidelines for aquatic ecosystem is <100 mg l^{-1} and must not exceed 10% of the background TSS concentrations at the specific locality and time. Generally, the TSS mean values showed that there were no considerable seasonal variations at all four sites during the study period (Figure 11).

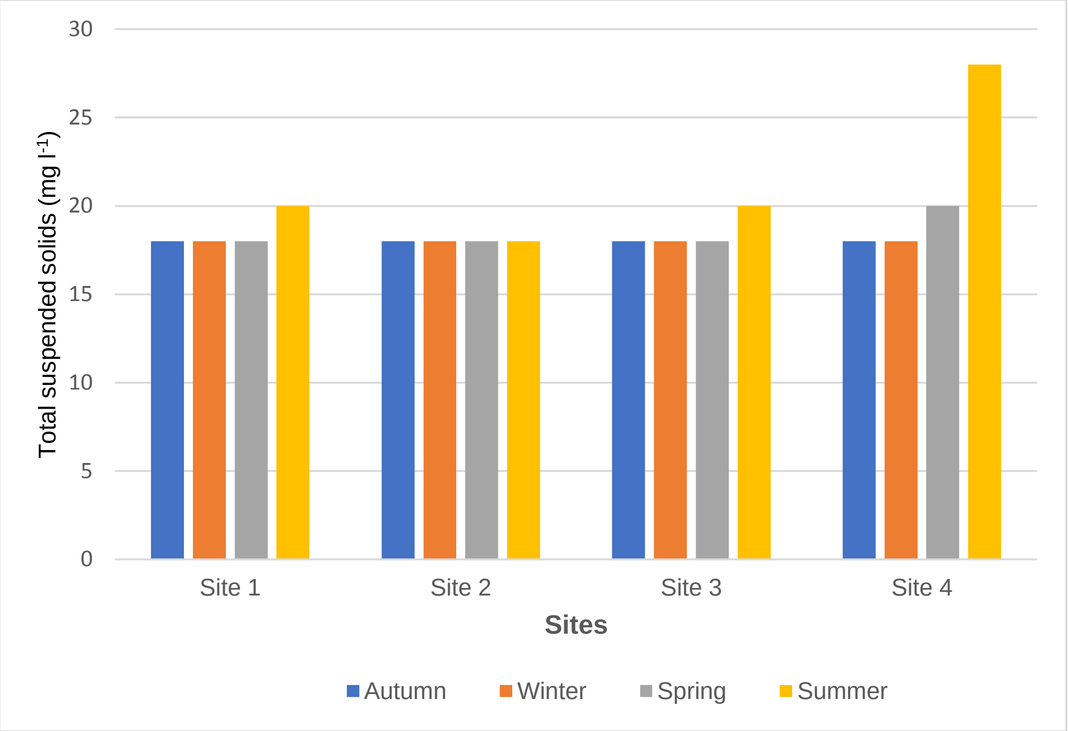

Figure 11: Seasonal Total Suspended Solids (TSS) concentrations at the four sites

3.1.6. Turbidity

The turbidity levels were very low (2.6 NTU - 7.8 NTU) in autumn at reference Site 1, and only increased slightly at the other three more downstream sites. In winter, Site 4 recorded a reasonably high level of turbidity (9.6 NTU) (Table 7) as compared to other three sites. And in spring the levels ranged between 5.4 NTU - 9.9 NTU (Figure 12). In summer, Site 1 recorded low levels of 2.4 NTU as compared to other sites which increased slightly. There are no turbidity targeted range for aquatic ecosystems (DWAF 1996c), however the aquaculture guidelines indicate that less than 25 NTU are tolerable turbidity values for clear water fish species (DWAF 1996b). Overall the turbidity levels never exceeded 25 NTU during the study period.

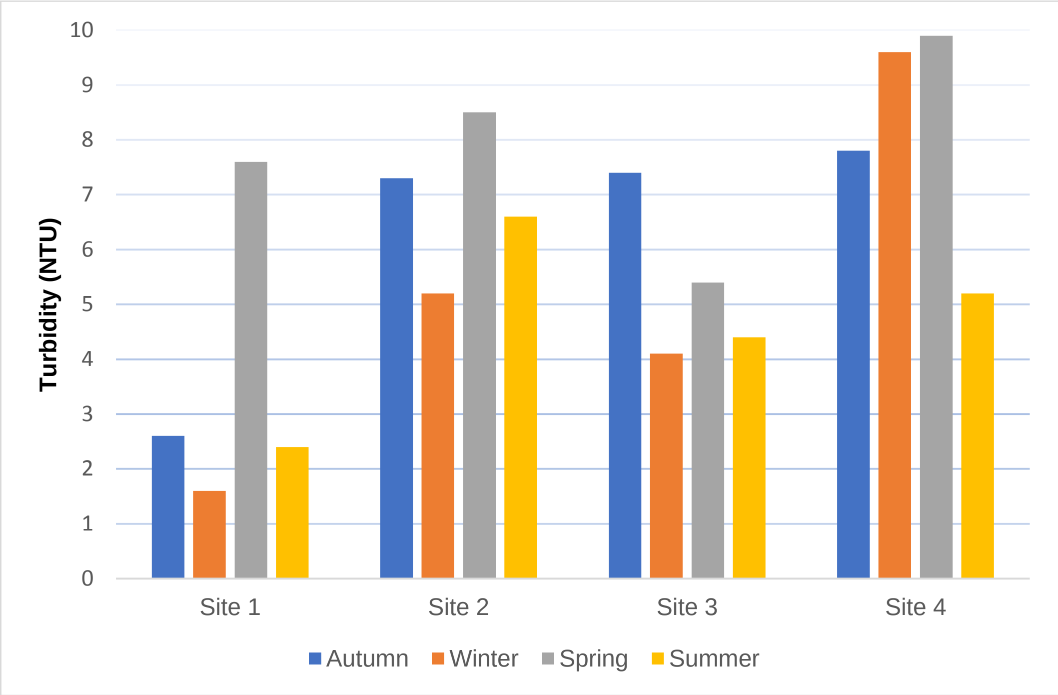

Figure 12: Seasonal Turbidity levels at the four sites

3.2. Integrated Habitat Assessment System (IHAS) at the four sampling sites.

The IHAS scores at Site 1 (81) was category B (good) for all four seasons. The scores of IHAS at Site 2 ranged from 58 - 64, and it showed that the habitat availability at Site 2 was category C (fair) in all seasons except in autumn which recorded a lowest score of 58 indicating a poor status (D) (Table 2). The IHAS scores at Site 3 ranged from 68 - 74, and it was category C in all four seasons. The IHAS scores at Site 4 ranged from 62 - 64 and indicated that the category was C (fair status) during the four seasons (Table 2).

Table 2: The seasonal IHAS scores recorded at the four sampling sites

		Autumn		
	Site 1	**Site 2**	**Site 3**	**Site 4**
Stones in Current	22	20	24	18
Vegetation	11	5	4	7
Other habitats	14	10	12	12
Stream condition	34	23	32	28
IHAS score	81	58	72	65
Status	B	D	C	C
		Winter		
Stones in Current	22	20	24	18
Vegetation	11	5	4	6
Other habitats	14	12	10	13
Stream condition	34	**23**	30	28
IHAS score	81	60	68	65
Status	B	C	C	C
		Spring		
Stones in Current	22	20	24	18
Vegetation	11	8	4	8
Other habitats	14	13	12	10
Stream condition	34	23	32	28
IHAS score	81	64	72	64
Status	B	C	C	C
		Summer		
Stones in Current	22	22	25	18
Vegetation	11	6	5	4
Other habitats	14	13	12	12
Stream condition	34	23	32	28
IHAS score	81	64	74	62
Status	B	C	C	C

3.3. River Health Status

The present ecological status or present ecological health of Site 1 (Warrenton) fell within category B during autumn, winter and spring, but improved to category A in summer. The SASS score at Site 2

(Windsorton) was lowest in autumn, only increasing slightly in winter, spring and summer. The ecological status at Site 2 was characterised within category A in autumn, category B in winter, and category A during both spring and summer.

The ecological status at Site 3 (Delpoortshoop) fell within category A during autumn and winter, respectively, and declined slightly in spring and summer to category B as per the ecoregion classification adapted from Dallas (2007) (Figures 1 and 2). The ecological category at Site 4 (Schmidtsdrift) was very poor (category E/F) in autumn, increased/improved slightly to category D in winter, declining to category E/F in spring, and finally showing a slight improvement to category D in summer where 15 taxa were recorded. The lowest number of taxa recorded at Site 4 was nine during spring. Overall the ASPT scores at this site were low (ranging between 3.8 and 4.8), and did not differ considerably throughout the study.

Table 3: Variation of SASS Scores at all sampled sites during autumn (28 April 2016), winter (28 July 2016), spring (14 October 2016) and summer (5 December 2016). ASPT = Average Score Per Taxon

Site	SASS 5 score	Autumn	Winter	Spring	Summer
Warrenton	**SASS**	96	91	121	109
	No. of taxa	19	18	21	19
	ASPT	5.1	5.1	5.0	5.7
	Category	B	B	B	A
Windsorton	**SASS**	95	112	131	141
	No. of Taxa	16	21	22	22
	ASPT	5.9	5.3	6.0	6.4
	Category	A	B	A	A
Delpoortshoop	**SASS**	102	89	81	97
	No. of Taxa	18	14	16	15
	ASPT	5.7	6.4	5.1	6.5
	Category	A	A	B	B
Schmidtsdrift	**SASS**	49	79	35	64
	No. of Taxa	13	18	9	15
	ASPT	3.8	4.4	3.9	4.3
	Category	E/F	D	E/F	D

3.4. Macroinvertebrate diversity and community structure

According to Ghosh and Biswas (2015) the Shannon-Wiener index (H') ranges between 0–5, where higher values indicate the higher diversity and the lower scores indicate pollution.

Table 4: Diversity (Shannon-Wiener index H') and evenness index (E) of macroinvertebrates at all sites during the four seasons

		Autumn		
	Site 1	**Site 2**	**Site 3**	**Site 4**
Taxa richness (d)	18	16	18	13
Shannon-Wiener Index (H ')	0.58	2.07	1.39	0.24
Evenness (E)	0.20	0.75	0.48	0.09
		Winter		
Taxa richness (d)	18	21	15	18
Shannon-Wiener Index (H ')	2.03	2.74	0.48	0.47
Evenness (E)	0.70	0.90	0.18	0.16
		Spring		
Taxa richness (d)	21	21	16	11
Shannon-Wiener Index (H')	2.32	2.15	2.0	0.94
Evenness (E)	0.76	0.71	0.72	0.39
		Summer		
Taxa richness (d)	20	24	15	15
Shannon-Wiener Index (H')	2.34	2.34	1.88	1.45
Evenness (E)	0.78	0.74	0.69	0.53

During this study in autumn, the H' values were slightly elevated at Site 2 (H' = 2.04), 3 (H' = 1.39), whilst at Site 1 (H' = 0.58) and 4 (H' = 0.24) were relatively low (Table 4). The taxa richness amongst the four sites ranged between 13 -18, while the evenness was low at Site 4 that showed a high dominance of more tolerate taxa. In winter, high H' values were recorded at Sites 1 (H' = 2.03) and Site 2 (H' = 2.74) as compared to Site 3 (H' = 0.48) and Site 4 (H' = 0.47) (Table 4). Again, the taxon evenness was evenly distributed in winter at Sites 1 and 2; and low at Sites 3 and 4 as reflected in evenness index. In spring, Sites 1, 2, and 3 had the highest H' values (2.32, 2.15 and 2.0) respectively; reflecting the highest macroinvertebrates diversity. Site 4 had the lowest H' value of 0.94 (Table 4). The three sites except Site 4 showed an even distribution of taxon between the sites and high taxa richness. In summer, all the sites recorded the highest H' and E values, and high taxa richness were recorded at Sites 1 (20) and 2 (24), whilst Sites 3 and 4 recorded a uniform value of 15 (Table 4).

There was seasonal variation in macroinvertebrate community structure at the four sampling sites along lower Vaal River. A total of 29 167 individual macroinvertebrates, comprising 36 taxa, were recorded at all sites during the study period (refer to Appendix B).

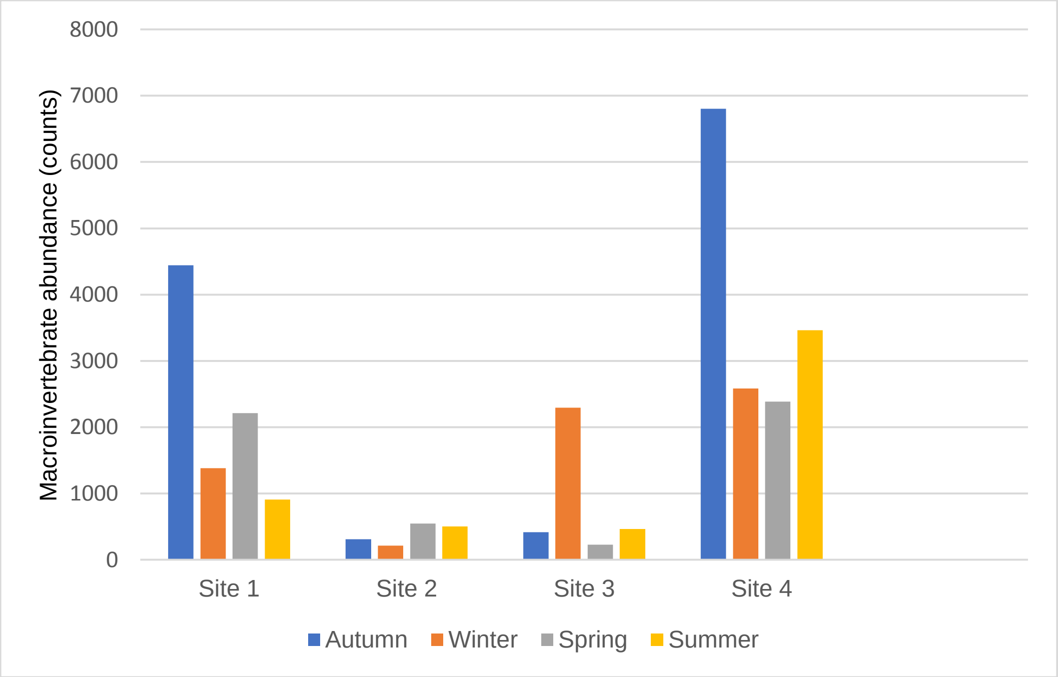

Fig 13: Seasonal abundance of macroinvertebrates across all four sites

During autumn, Site 4 had the highest number of macroinvertebrates, followed by Sites 1, 3 and 2 respectively (Appendix B). The taxa at Site 4 showed a low diversity and therefore were dominated by Simuliidae contributing 95.8% abundance, Corbiculidae (2.5%), Hirudinea (0.4%), Hydropsychidae >2 species (spp.) (0.3%) and Oligochaeta (0.3%) respectively (Appendix C). Overall, Simuliidae showed a high relative abundance at all sites except at Site 2 throughout the autumn season. Planorbinae was only recorded at Site 3 during autumn and spring and contributed 0.5% of the site abundance (Appendices C and E). During this study Turbellaria was only recorded at Site 4 in very low abundance. Baetidae >2 spp. was the most common taxa recorded at all the sites during the autumn (Appendices C and E).

During winter, Site 4 recorded the highest number of macroinvertebrates followed by Sites 3, 1 and 2 respectively (Appendix B). Chironomidae and Elmidae were also recorded in high numbers at Site 1 amongst other sites. Ecnomidae occurred only at Sites 2 and 3 in autumn and winter. Tricorythidae, Chlorocyphidae, Nepidae, Notonectidae, Velidae, Lymnaeidae and Planorbinae were not recorded at any site during the winter season, whilst Potamonautidae, Culicidae, Muscidae, Gomphidae and Tabanidae were recorded in low numbers (Appendices C and E). During this study, Leptophlebiidae, the most sensitive taxon, was recorded at all sites except at Site 4.

During spring, Site 4 recorded the highest number of macroinvertebrates followed by Sites 1, 2 and 3 respectively (Appendix B). The reason for high variance at Site 4 is because of high numbers of more tolerant taxa to water pollution e.g. Corbiculidae (67.5%), Physidae (23.6%), Simuliidae (5.7%), Baetidae 1 spp. (1.1%), Hydropsychidae 1 spp. (0.6%) and Oligochaeta (0.4%) respectively (Appendix C). Overall the Corbiculidae (67.5%) was the most dominant taxon recorded at Site 4. Turbellaria, Belostomatidae, Nepidae, Culicidae and Tabanidae were absent at all sites during spring. Planorbinae was only restricted to Site 3 during this study period and contributed 4.8% abundance (Appendices C).

During summer, Site 4 again had the highest number of macroinvertebrates, then followed by Sites 1, 2 and 3 respectively (Appendix B). The macroinvertebrates structure was dominated by more tolerant taxa such as Gyrinidae, Physidae, Corbiculidae, Simuliidae and Baetidae single (1) species (spp.) respectively because of high diversity recorded at Site 1 (Appendices F). The diversity at Site 4 was low as it was dominated by more common, tolerant taxa, while at Sites 2 and 3 there was high diversity, and the presence of more sensitive taxa like Heptagenidae, Leptophlebiidae, Chlorocyphidae, Baetidae >2 spp, Hydropsychidae >2 spp (Appendices F). Overall Site 1 had a high mean value of 2993, followed by Sites 2 (1621), 3 (1343) and 4 (1085) (Appendix B). Turbellaria, Gomphidae, Belostomatidae, Notonectidae, Ecnomidae, Culicidae, Muscidae, Tabanidae and Planorbinae were not recorded at all sites during summer season perhaps this can be attributed to their life cycles/seasonality on the impacted water quality.

CHAPTER 4

4. Discussion

4.1. Physico-chemical parameters

4.1.1. pH

Water pH is an indicator of the relative concentration of hydrogen ions (H^+) and hydroxide ions (OH^-) which reflects the acidity / alkalinity of water. The pH of an aquatic ecosystem is important because it influences biological productivity (Dallas and Day 2004). According to DWAF (1996a) the pH range of aquatic ecosystem water quality guidelines range between 6.0 and 8.5, but large variations may occur because of catchment geology. When the pH of the water is below the targeted water quality range of a pH <6, it renders the metals present in water toxic and stressful to fish and macroinvertebrates (Dallas and Day 2004). As the pH increases the proportion of toxic ammonia to harmless ammonium increases. The pH during this study was alkaline (8.3 - 8.9) and unlikely to pose any threat to macroinvertebrates. The high pH can be attributed to elevated levels of major cations and anions, e.g. magnesium, sodium, calcium and sulphate, which can result in alkaline systems (DWAF 1996a). Again, it can be ascribed to the biogeochemistry of the Vaal River coupled with the high photosynthetic activity of the algae. If these conditions are persistent they may have a limiting effect on aquatic biota at this site.

4.1.2. Dissolved oxygen

Dissolved Oxygen (DO) is important for all forms of life (Terry *et al.* 2017). The DO in the river depends on the balance between the flux of organic carbon and the rate at which the heterotrophic bacteria use up oxygen in the decomposition of organic material, and the daily inputs of oxygen by diffusion from the atmosphere as well as via photosynthesis by macrophytes and phytoplankton (Dallas and Day 2004). Low DO in the water can cause physiological stress for fish and macroinvertebrates. Concentrations below 5 mg l^{-1} may have adverse disturbances on the functioning and survival of aquatic communities while the concentrations below 2 mg l^{-1} may lead to the death of most fish (DWAF 1996c). DO concentrations in unpolluted water normally range between 8 mg l^{-1} and 10 mg l^{-1} and the saturation is dependent on temperature (Watson *et al.* 1985).

The low DO levels at Site 4 can be attributed to the organic material from the livestock (pigs and horses) from upstream water. As 2016 was a drought year, and the water level was very shallow and flowed slowly at this site as compared to other sites, the low DO levels could perhaps also be due to seasonal temperature differences. High temperatures tend to heat and deplete the oxygen in the water; indeed, seasonal variations in DO in water arise from changes in temperature and biological activity (Dallas and Day 2004). The observed limited flow due to drought at site 4 might have stimulated high algal growth that prevented the habitat of macroinvertebrates. In general, the low DO concentrations at Site 4 throughout the study was in part due to the shallow water heating and the organic decomposition in the water, livestock and vehicle disturbances all of which appear to decrease macroinvertebrate diversity, with the sensitive ones being first to disappear.

4.1.3. Electrical conductivity

Electrical Conductivity (EC**)** is a numerical expression of the ability of water to conduct an electrical current, resulting from the presence of charged species in solution (DWAF 1996a). The EC of water is directly proportional to the total dissolved solids. The major contributing ions are carbonate, bicarbonate, chloride, sulfate, nitrate, sodium, potassium, calcium and magnesium (DWAF 1996a; Dallas and Day 2004). There are no EC guidelines set for aquatic ecosystems in South Africa (DWAF 1996c) and the effects of EC on aquatic biota are not well known (Dallas and Day 2004). The EC guidelines for domestic use ranges between 0 mS m^{-1} **-** 70 mS m^{-1} (DWAF 1996a) and during this study the values exceeded the TWQR set for domestic use at all sites (76 mS m^{-1} to 120 mS m^{-1}). According to DWAF (1996a) the high EC in water is due to elevated levels of major cations and anions whilst low EC or reduced level is due to the dissolved ions. The elevated EC levels recorded at Site 4 throughout the study period might be due to the accumulation of dissolved ions emanating from a tributary of the Vaal River just below Site 3, and also increased by upstream land use activities in the upper parts of the Vaal River.

4.1.4. Water temperature

Temperature plays a crucial part in chemical reactions of water by affecting the metabolism of organisms and eventually their distribution. The general range of water temperature for inland waters in South Africa is around 5 - 30°C (DWAF 1996c). Thermal characteristics of running waters are dependent on various hydrological, climatic, and structural features within regions, catchments, and rivers (Dallas 2008). TWQR for temperature should not vary by > 2°C or by > 10% for the specific site and time. The changes in water temperature may lead to variations in the abundance, diversity and composition of aquatic communities. In this study the water temperatures appeared to be normal, and did not exceed the TWQR throughout the study.

4.1.5. Total suspended solids

Total Suspended Solids (TSS) determine the solid content dissolved or suspended in water. The presence of suspended silt causes a high-water turbidity. The nature and levels of suspended matter influences the turbidity and clearness of the water (Dallas and Day 2004). The sources of suspended matter are plankton, clay, silt, fine particles of organic and inorganic matter (Fouche *et al*. 2013). Normal functioning of aquatic ecosystems is affected when concentrations of TSS in water are relatively high (Dallas and Day 2004). The mining in the lower Vaal River occurs 10 meters from the water course. The dumps are stockpiled next to the river potentially impacting on water quality thereby increasing the siltation of the water or suspended solids which could affect the spawning sites and habitat for fish and macroinvertebrates (Naidoo-Vermaak 2006; Ramollo 2011).

According to Heath *et al.* (2004) the alluvial diamond mining does not cause a major impact in water quality, but in some areas along the river where the river course is severely impacted it does affect macroinvertebrate diversity. Elevated sedimentation in the river reduces light penetration in the water and can result in decreased photosynthesis, ultimately causing a decline in oxygen in the water column that negatively affects macroinvertebrates (Dallas and Day 2004; Naidoo-Vermaak 2006). According to DWAF (1996a) the concentration of TSS in rivers should be<100 mg l^{-1} and must not exceed a background of 10% of TSS concentrations at a given time and site. The increased TSS at Site 4 could be ascribed to the disturbance of the biotopes by pigs and horses that constantly make use of the river.

According to the study by Naidoo-Vermaak (2006) the TSS concentrations measured at Site 1 over a period of 10 months was <21 mg l^{-1}, then decreased slightly to <18 mg l^{-1} throughout that study period. The TSS concentrations at Barkly West (not far from Sites 2 and 4) indicated the concentrations of <18 mg l^{-1} measured over a period of 10 months (July 2004 - May 2005) and only increased in November 2004 to <21 mg l^{-1} and in May 2005 to <44 TSS mg l^{-1}, respectively (Naidoo-Vermaak 2006). The study concluded that alluvial diamond mining activities contributes significantly to an increased TSS. The results of this study conducted in 2016 do not show a considerable variation in TSS concentrations when compared to the study conducted by Naidoo-Vermaak (2006). The concentrations of TSS during this study period appeared to be constant, and can be ascribed to the fact that 2016 was a drought year, characterized by shallow water with reduced flows, and with considerably less runoff of sediments such as silts and clay into the river by rainfall. This indicated that the contribution of TSS concentrations by alluvial diamond mining was low in these conditions.

Normally when the river discharge is higher it washes a lot of suspended solids, and as the flow of the stream decreases the TSS settles out. However, settling out is also dependent on the size of particles and of the river velocity. In addition, it appeared that the discharged waste water (with suspended materials) used to wash diamonds settled rapidly in the river. When these sediments settle they cover the area and change the habitat for macroinvertebrates (Dallas and Day 2004).

4.1.6. Turbidity

Turbidity is an expression of certain light scattering and light absorbing properties of the water sample caused by the presence of clay, silt, suspended matter, colloidal particles, plankton and other microorganisms (Chapman and Kimstach 1996; DWAF 1996b; Dallas and Day 2004). Turbidity can be measured by turbidity and nephelometry. Turbidity of water affects other water quality parameters such as colour when it is imparted by colloidal particles and influences both the quantity and quality of light penetration into the water (DWAF 1996b; Dallas and Day 2004). The concentration of suspended solids increases with the discharge of sediment washed into rivers or dams, due to rainfall and re-suspension of deposited sediment (Chapman and Kimstach 1996; Dallas and Day 2004).

There are no water quality guidelines of turbidity for aquatic ecosystems, however aquaculture guidelines were used and thus indicate that 0 NTU - 25 NTU is a tolerable turbidity for fish species (DWAF 1996b). The turbidity recorded during the study ranged from 1.6 NTU - 9.9 NTU and never exceeded 25 NTU. The turbidity levels recorded by Malherbe (2013) at Site 3 were very high as compared to this study. The levels were as follows: in 2007 (38 NTU), 2008 (9 NTU) and 2009 (28 NTU). The study conducted by Naidoo-Vermaak (2006) in the Lower Vaal River in Vaalbos not far from Site 3 showed an increase in turbidity and depletion of oxygen levels caused by alluvial diamond mining activities such vegetation stripping. The study concluded that alluvial diamond mining activities contributes to an increase turbidity and subsequent loss in aquatic biodiversity in the river. This study corroborates the finding of Naidoo-Vermaak (2006) by alluding that the increase in turbidity was most probably as a result of elevated erosion and associated high TSS coming from discharge of water used to wash diamonds coupled with upstream activities, which ranged between mean 18 mg l^{-1} - 21 mg l^{-1} (Appendix A; Figure 11).

4.2. Integrated Habitat Assessment System (IHAS) at the four sampling sites

The alluvial diamond mining impacts (riparian destruction, siltation and river diversions) were clearly visible during this study. Site 1 was in a good state in all seasons with moderate river flows. The condition of the stream at this site was natural with minimal impacts as compared to the other three sites.

The habitat and stream condition at Site 2 was highly modified by active instream alluvial diamond mining activities, with the tailings water pumped in the river downstream of the sampling point. The active mining activity created a non-natural length of cobbles that served as a habitat for macroinvertebrates. The instream mining changed the natural river flows, creating rapids, runs and sometimes a mixture of runs/rapids. Overall the IHAS category at this site was category C for all the sampling seasons, and the low scores can be attributed to the alluvial diamond mining.

Site 3 consisted of an artificial length of cobbles located in rapids, riffles and pools which was recolonised by sensitive macroinvertebrates such as Leptophlebiidae, Tricorythidae and Heptagenidae. The length of stone in current biotope was increased by the alluvial diamond mining which has happened in the river bed. The vegetation at this site was severely destroyed. The destruction of the habitat was seen as a limiting factor for the diversity of macroinvertebrates. Overall the IHAS scores at this site was category C during the four seasons.

Site 4 had limited stones in current as compared to the other three sites. The IHAS scores at this site ranged from 62 - 65 during the four seasons. These low scores can be ascribed to mining activities, presence of livestock and construction vehicles which added pressures on the diversity of macroinvertebrates. The synergistic impacts of livestock, construction trucks and alluvial diamond mining were the limiting factor on the presence of sensitive and moderate macroinvertebrates, and favoured the more tolerant taxa. Overall the status was category C throughout the study.

4.3. River health status

The present ecological status at the reference site (Site 1) was rated as minimally modified (B) during the three seasons (autumn, winter and spring) and improved to unmodified/natural (A) in summer according to Southern Kalahari ecoregion interpretations (Dallas 2007). The riparian vegetation at the reference was dominated by reeds while the in-stream vegetation was dominated by water hyacinth. The Working for Water Programme clears the water hyacinth with chemical application whilst the water reeds are controlled through burning. The clearance of the water reeds and water hyacinth might have resulted in reduced diversity and absence of more sensitive taxa. However, the results were consistent with the DWAF study (2011) that indicated below the Bloemhof Dam to the Douglas confluence the ecological state of the river falls within category A/B, and thus minimally impacted.

Site 2 recorded high SASS scores as reflected on the high Average Score Per Taxon (5.3 - 6.4) throughout the study. The highest SASS scores were recorded during spring (131) and summer (141) respectively. The present ecological status was A during autumn, decreased slightly to category B and improved to A category in spring and summer according to Southern Kalahari ecological bands (Dallas 2007). The IHAS score was largely modified to moderately modified. The results of this study are consistent with the study conducted by Ramollo and Moalusi (2010) during winter and spring which showed the ecological category as being B. This site had no marginal vegetation in current and out of current during autumn and the site was mostly dominated by the cobbles biotope, which indicated to have influenced the SASS score. The cobble biotope at this site was not natural but as the result of alluvial diamond mining diggings which appears to have created suitable habitat for sensitive mayflies.

In other seasons the vegetation recovered and marginal vegetation was dominated by patches of sedges and in-stream water grass, with a few macroinvertebrates recorded as associated with this vegetation. The presence of sensitive macroinvertebrates at this site ultimately increased the overall SASS scores (Ramollo and Moalusi 2010). The present ecological status of Site 3 was A during autumn and winter but did decline slightly to indicate some biotope modification (B) for spring and summer conditions. The results were in agreement with the studies conducted by Ferreira (2008), Ramollo and Moalusi (2010); DWAF (2011) and Malherbe (2013), which also recorded category B. The most suitable habitat for macroinvertebrates at this site was stones-in-current and stones-out-of current and these were generally covered with filamentous algae, and had few patches of sedges and in-stream pondweed (Ramollo and Moalusi 2010).

Overall the site had little marginal or in-stream vegetation due to in-stream alluvial diamond diggings and river obstruction by culverts (Ferreira 2008). Site 3 had no gravel/sand/mud (GSM) biotopes so no sampling of GSM was done throughout the study period. This low/reduced habitat diversity (moderately modified) appeared to have influenced the SASS score and as a result the diversity was low but dominated by more sensitive taxa recorded in cobbles (Malherbe 2013). The cobble biotope at this site is not natural but due to in-stream alluvial diamond diggings and the construction of an illegal crossing road bridge built using scrap materials, concrete culverts, and steel culverts (Ferreira 2008; Malherbe 2013).

The present ecological status at Site 4 was rated as E/F during autumn and spring, yet it improved to category D under the more extreme seasons of winter and summer. Results from Site 4 varied with the study conducted by Ramollo and Moalusi (2010) who recorded a B category during low flows and high flows. This study also differed with a study conducted at the lower reaches of the Vaal River near Douglas Weir, downstream of Site 4, indicating that the site was in category C (Kimberg and Rall 2005). Moreover, the ecological assessments conducted by DWAF (2011) in the whole of Vaal River, showed that the PES of the Vaal River below the Bloemhof Dam to the Douglas Weir falls within category A/B; this is not in agreement with the current study. Site 4 appeared to be highly impacted, had low SASS scores, and had lost much of its capacity to support diversity of taxa sensitive to water pollution. This might indeed have led to the adverse changes in faunal structure observed at this site and it is suggested that stricter rules be implemented to prevent pigs and other livestock or animals in the waterway.

Overall all sites had a good representation of families of macroinvertebrates with a clear exception of Site 4. The poor ecological status at Site 4 was due to a combination of poor habitat, impacted water quality coming from the Vaal-Harts rivers, and also the presence of horses and pigs which defecate in the water and forage on the freshwater mussels and pondweed disturbing the establishment of macroinvertebrates. The little vegetation present varied from other reaches. The in-stream vegetation at this site was dominated mostly by pondweed, whilst marginal vegetation was dominated by sedges and reeds. It appeared the observed slow to moderate flows at this site created a suitable habitat for an in-stream mat of pondweed.

4.4. Macroinvertebrate diversity and community structure

Overall site 4 indicated the lowest Shannon-Wiener Index values (H’) and evenness index (E) values during this study as compared to the other three sites. The high H' values at the aforementioned sites is an indication of a more diverse macroinvertebrates communities (Shimba *et al*. 2018). The high abundance of Simuliidae at Sites 1 and 4 can be attributed to the availability of microplankton food and habitat (riffles). According to Schmitt *et al.* (2016) the food availability is the most obvious factor that controls the occurrence and abundance of species. The high abundance can also be ascribed to low levels of predation by carnivorous/ insectivorous taxa like Hydropsychidae, Hirudinea and others (Chutter 1968). The reason for variances and low abundance of Simuliidae at Site 2 can be attributed to in-stream alluvial diamond mining that destroyed a length of biotope cobbles and vegetation which functioned as a breeding and recruitment area for Simuliidae and other taxa (Chutter 1968; De Moor 1982).

The in-stream alluvial diamond mining activities might have limited refugial space causing few taxa to drift, again in-stream alluvial diamond prospecting might have disrupted the macroinvertebrates life cycle, impacting on the food chain and imposing physiological stresses on more tolerant taxa (Adakole and Annue 2003). According to Biol *et al.* (2011), some species of Baetidae are tolerant towards nutrient rich waters, sedimentation, and wide ranges of change in river flows. Baetidae are common in low lying rivers and tend to prefer gently flowing water and are found between cobbles and in vegetation (Slabbert 2007); this observation was supported during this study.

During autumn, Sites 3 and 4 had the highest abundances of Simuliidae which can be attributed to the wide availability of stones in current and out of current which appeared to be the substrate most suitable for the colonization by Simuliidae larvae and pupae (De Moor 1982). The availability of food and low abundances of their natural enemies/ predators are likely reasons for the high abundance of Simuliidae (Chutter 1968). According to Kiel *et al.* (1998) other species of Simuliidae are able to colonise the substrates within hours and can reach a density of thousands of individuals in few days.

During winter most of Simuliidae population are concentrated in the aquatic environment and in summer will be in adult stage (De Moor 1982). All the sites in summer recorded low abundance of Simuliidae as compared to other seasons and this can be ascribed to the fact that a large population of the Simuliidae in summer would be in their adult stage (Chutter 1998; De Moor 1982).

The reason for Site 4 (highly impacted site) having a high macroinvertebrate abundance compared to Sites 1 and 2, is likely due to high abundances of the most tolerant taxa e.g. Simuliidae (91.5%) and Corbiculidae (3.4%), that increased the SASS score considerably, perhaps expected at such disturbed sites. Leptophlebiidae were recorded at all sites except at Site 4, and had a preference for cobble substrates while few individuals were recorded in woody snags. This supported the analysis of Thirion (2016) who stated that local and international literature indicated that different species of Leptophlebiidae can have different habitat requirements. Gyrinidae, Atyidae and Coenogrionidae were dominant throughout with a preference for marginal vegetation.

Heptageniidae was recorded only at Site 2 in very low numbers (with abundance of 0.5%). The complete absence of Heptageniidae at other sites can be related to the differences of in-stream environmental degradation along the river as a result of human activities, disturbance by alluvial diamond mining activities and impacted water quality. The presence of sensitive taxa such as Heptageniidae, Tricorythidae, Chlorocyphidae, Atyidae and Leptophlebiidae at Sites 1, 2 and 3 indicated that the water quality was minimally modified as shown in high Average Score Per Taxon scores of 6.5 (Dallas 2007). The aforementioned taxa are very sensitive to changes in water quality, variety of flow ranges/ modification, turbidity and substrate conditions, so once the water was impacted their abundance declined dramatically**.** The presence of Leptophlebiidae, Chlorocyphidae and Baetidae >2 spp. at Site 2 during this study showed that the site was recovering from mining impacts.

The water level at Site 4 was extremely low during the study period. As a result, the trucks of mining companies instead of taking a long-tarred and gravel road were seen crossing the river, using it as a short cut to access their mining operation. Driving over the biotopes is likely to have disturbed the macroinvertebrate diversity by killing in-stream macroinvertebrates and stirring up of mud which might have resulted in a decline in dissolved oxygen levels which can result in low diversity. At this site, more than fifteen pigs were seen feeding on the freshwater clams (Corbiculidae) and water grass (*Potamogeton pectinatus*). The presence of the pigs in the waterway is also likely to have caused a decline in dissolved oxygen levels that could ultimately have resulted in low macroinvertebrates diversity at the site. The pigs graze from the morning till sunset, when they defecate in the river/waterway their faeces also contributed to nutrient loading and depletion of oxygen.

Two horses were also observed disturbing the gravel, sand and mud with their hooves, which might also have resulted in sediments mobilisation and deoxygenation of the river water. The resuspension of sediments can also increase erosion and release of nutrients (phosphate and ammonium) and heavy metals such as copper, zinc, lead etc. This study showed that an increase in total abundance of macroinvertebrates does not necessarily indicate good ecological status/environment but rather indication of degradation that favours some tolerant taxa with subsequent decrease of sensitive taxa (Dallas and Day 2004).

The presence of least sensitive taxa to water pollution at all sites emphasized how good these taxa are at colonizing areas under a broad range of conditions. The Chironomidae are known to be able to tolerate a high level of organic pollution because they have a high haemoglobin content which makes them survive in hypoxic conditions (Tyokumbur *et al.* 2002). Their complete absence at Sites 3 and 4 in summer suggest the heavy smothering of sediments on their biotopes emanating from upstream activities that might have prevented their establishment colonising these sites. The gravel, sand and mud (GSM) biotope was always low in macroinvertebrate diversity at all sites, and only recorded a few individuals of the Gomphidae, Oligochaeta and Ceratopogonidae taxa. Site 4 (Schmidtsdrift) was the only site where Turbellaria were recorded in low numbers.

The GSM were dominated by Corbiculidae and Gomphidae. The least sensitive macroinvertebrates, e.g. Simuliidae, Hirudinea and Ancylidae were attached to the surface of stones biotopes. Taxa such as Atyidae, Coenogrionidae, Naucoridae, Belostomatidae and Gyrinidae were often recorded in the vegetation biotope while in some instances the Coenogrionidae and Gyrinidae where observed in the water column. A considerable dominance of Velidae and Gyrinidae were recorded at the surface of the river along the marginal and aquatic vegetation. Biotope availability also affected the final SASS scores, as a paucity of habitat resulted in less macroinvertebrates being recorded at most of the sites. Overall, all sites except Site 4 were in good condition throughout the sampling period as reflected by the high SASS scores.

5. GENERAL CONCLUSION

Physico-chemical parameters, river health status, macroinvertebrates diversity and community structure.

The macroinvertebrates in all but one sampling site, Site 4, indicated that the water quality of the lower Vaal River was in good to pristine condition. However, the integrity of habitats and biotopes was often poor and the physico-chemical water quality parameters were sometimes high (pH and TSS). The high abundance of macroinvertebrates in the gravel sand and mud biotope were dominated by the more tolerable Chironomidae, Corbiculidae, Hirudinea and Oligochaeta taxa, and were reflection of nutrient enrichment in the lower Vaal River. The alluvial diamond mining severely modified the riparian vegetation and riverbed that possibly affected the breeding and recruitment of macroinvertebrates resulting in relatively low SASS scores.

It was also observed that the macroinvertebrate community structures at all sites were affected by seasonal variation. It is normally believed that macroinvertebrate assemblages are best considered by combining the data collected from different times of the year/seasons. The increased in sampling efforts and a combination of seasonal data collection increases the number of families collected at a specific site and ensure that more habitats are sampled. Because of the variability in life cycles and changes in different macroinvertebrate groups, some families were sampled in one season but not in another seasons. This information could be crucial for management purposes to prevent further degradation of this important hard-working water resource.

Generally, this study has provided a detailed set of ecological data describing macroinvertebrate community structure along some reaches of the lower Vaal River. Macroinvertebrate organisms were shown to be potentially good water quality indicators at the four selected sites along the Vaal River, and the number of diverse taxa collected provided a valuable source of information. Out of the recorded taxa, Simuliidae were dominant throughout the study period probably because of their ability to tolerate organic pollution. The decline in dissolved oxygen levels, poor availability of vegetation, and elevated sediments loads eliminated sensitive macroinvertebrates from Site 4.

This study further highlighted the importance of understanding river health and provided knowledge of the abundance and distribution of macroinvertebrates. Observations made in this study further corroborate those by Ramollo and Moalusi (2010) in that the distribution and composition of aquatic macroinvertebrate communities in the system are significantly influenced by a variety of environmental factors such as poor habitat characteristics and impacted water quality. Poor water and habitat quality can further impact biological factors such as lowering macroinvertebrate competition and predation by indigenous fish, mainly mudfish (*Labeo capensis); sharptooth catfish (Clarias gariepinus*) and alien fish species like mosquito fish (*Gambusia affinis*) and common carp (*Cyprinus carpio*) (Ramollo and Moalusi 2010), further indicating the sensitive ecology of river ecosystems. Thus, as these environmental and biological factors were disturbed and changed over time, the macroinvertebrate community structure also changed.

Physico-chemical parameters such as temperature, pH, DO and TSS directly influenced the composition and abundance of macroinvertebrates. The variation in SASS5 scores observed may be attributed to changes in the habitat availability and suitability combined with the water quality. It is likely that impacts from farming activities, sewage and erosion were likely to have affected the aquatic ecological integrity at all sites. The low scores of invertebrate habitat assessment system showed that the habitat integrity at Sites 2, 3 and 4 was lacking and was therefore regarded as a limiting factor to macroinvertebrate existence. At sites where the vegetation biotope was highly modified or very poor, even the macroinvertebrates that are typically related with it were absent and that influenced the SASS scores. Overall the SASS5 scores showed a good to pristine ecological integrity (B/A) at all sites except at Site 4 as per ecoregions described by (Dallas 2007).

The macroinvertebrate community in the Vaal River re-established quickly. These can be attributed to the fact that the river is a big system with a width of roughly 100 - 300 metres. When the alluvial diamond miners diverted the river, some of the macroinvertebrates moved with the river flow, perhaps that's why the macroinvertebrates were able to recover and establish quickly. The miners diverted a certain portion of the river and when done with an exploration, channelled the river to another side and mined for diamonds, therefore it appears that the macroinvertebrate diversity was affected during the mining operation, but with the aforementioned reasons, coupled with the adult stages of the macroinvertebrates that are terrestrial, allowed the river to recover and start supporting sensitive macroinvertebrates. Therefore, it can be concluded that the alluvial diamond mining impact on macroinvertebrates was medium to low, whilst on riparian vegetation was high to severe.

The water quality of the lower Vaal River is still in a good state as reflected by the SASS and Average Score Per Taxon scores. Alluvial diamond prospecting does not add any chemical compounds in the processing of diamonds, therefore the tailings water discharged back into the river at Site 4 do not contain chemical pollutants, but rather consist mostly of silts that smothers the habitat of macroinvertebrates and fish. The research question that the water quality of the Vaal River is impacted by the alluvial diamond mining was supported with regards to the macroinvertebrate community. The research question related to the alluvial diamond mining impacted on macroinvertebrate community structure in the lower Vaal River was also supported, showing clear evidence of effects and therefore more studies are needed to separate the natural causes from water quality and habitat integrity. The human population continues to grow in the Vaal River catchment and will continue to impact the ecological status of the river.

In conclusion, this study has shown that the main factors that determined the macroinvertebrate fauna were habitat integrity and water pollution, which directly affected water quality. It is recommended that a further detailed survey be done in the lower Vaal River to achieve a more required baseline data of the desired conditions, revise and improve the ecological bands of the Southern Kalahari ecoregion of the lower Vaal River. There is an urgent need for a more thorough study on the entire length of the Vaal River to fully understand the effects of these impacts on the ecology of the river, a vital source of ecological services to support human wellbeing now, and in the future.

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Appendix A. Physico-Chemical variables at the four sites across all seasons

Autumn				
Variables	**Site 1**	**Site 2**	**Site 3**	**Site 4**
pH	8.8	8.9	8.5	8.3
Dissolved Oxygen (DO) mg l^{-1}	7.9	7.6	5.3	4.9
Electrical Conductivity (EC) mS m^{-1}	85	93	88	110
Water temperature (Temp)	19.8	18.2	20.2	19.8
Total Suspended Solids (TSS) mg l^{-1}	18	18	18	18
Turbidity (NTU)	2.6	7.3	7.4	7.8
Winter				
pH	8.5	8.5	8.3	8.5
Dissolved Oxygen mg l^{-1}	8.4	6.7	8.2	4.7
Electrical Conductivity (EC) mS m^{-1}	82	86	90	100
Water temperature (Temp)	18	16	18	14
Total Suspended Solids (TSS) mg l^{-1}	18	18	18	18
Turbidity (NTU)	1.6	5.2	4.1	9.6
Spring				
pH	8.6	8.4	8.4	8.6
Dissolved Oxygen (DO) mg l^{-1}	8.8	6.4	8.2	4.2
Electrical Conductivity (EC) mS m^{-1}	76	86	89	110
Water temperature (Temp)	20.2	21	23.6	21.2
Total Suspended Solids (TSS) mg l^{-1}	18	18	18	20
Turbidity (NTU)	7.6	8.5	5.4	9.9
Summer				
pH	8.6	8.6	8.9	8.8
Dissolved Oxygen (DO) mg l^{-1}	8.2	6.6	6.3	4.4
Electrical Conductivity (EC) mS m^{-1}	76	92	98	120
Water temperature (Temp)	26.8	25	26.4	27.2
Total Suspended Solids (TSS) mg l^{-1}	20	18	20	28
Turbidity (NTU)	2.4	6.4	4.4	5.5

Appendix B. Total number of macroinvertebrates recorded at all sampled sites

Surveys	Site 1	Site 2	Site 3	Site 4	Total
Autumn	4444	309	416	6803	11972
Winter	1385	216	2296	2587	6484
Spring	2211	546	227	2387	5371
Summer	908	503	466	3463	5340
Total	8948	1574	3405	15240	29167
Mean	2237	394	851	3810	7292

Appendix C: Total number and relative abundance (%) of macroinvertebrate taxa sampled at the three biotopes in autumn at the four selected sites.

Taxon	Site 1	%	Site 2	%	Site 3	%	Site 4	%	Total
Turbellaria	0	0,0	0	0,0	0	0,0	6	0,1	**6**
Oligochaeta	6	0,1	8	2,6	0	0,0	18	0,3	**32**
Hirudinea	60	1,4	0	0,0	5	1,2	26	0,4	**91**
Potamonautidae	2	0,0	0	0,0	5	1,2	0	0,0	**7**
Atyidae	60	1,4	0	0,0	0	0,0	0	0,0	**60**
Baetidae	198	4,5	88	28,5	14	3,4	14	0,2	**314**
Caenidae	2	0,0	0	0,0	1	0,2	0	0,0	**3**
Heptageniidae	0	0,0	2	0,6	2	0,5	0	0,0	**4**
Leptophlebiidae	8	0,2	44	14,2	44	10,6	0	0,0	**96**
Tricorythidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Chlorocyphidae	0	0,0	1	0,3	1	0,2	0	0,0	**2**
Coenagrionidae	0	0,0	1	0,3	9	2,2	6	0,1	**16**
Gomphidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Libellulidae	1	0,0	2	0,6	0	0,0	0	0,0	**3**
Belostomatidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Naucoridae	0	0,0	0	0,0	0	0,0	3	0,0	**3**
Nepidae	3	0,1	0	0,0	2	0,5	0	0,0	**5**
Notonectidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Veliidae	0	0,0	0	0,0	1	0,2	2	0,0	**3**
Ecnomidae	0	0,0	0	0,0	1	0,2	0	0,0	**1**
Hydropsychidae	20	0,5	32	10,4	8	1,9	22	0,3	**82**
Hydroptilidae	0	0,0	2	0,6	0	0,0	0	0,0	**2**
Dytiscidae	6	0,1	0	0,0	0	0,0	0	0,0	**6**
Elmidae	21	0,5	16	5,2	28	6,7	0	0,0	**65**
Gyrinidae	10	0,2	42	13,6	6	1,4	0	0,0	**58**
Ceratopogonidae	0	0,0	52	16,8	0	0,0	0	0,0	**52**
Chironomidae	8	0,2	3	1,0	12	2,9	5	0,1	**28**
Culicidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Muscidae	2	0,0	0	0,0	0	0,0	1	0,0	**3**
Simuliidae	3940	88,7	2	0,6	274	65,9	6514	95,8	**10730**
Tabanidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Ancylidae	18	0,4	3	1,0	0	0,0	0	0,0	**21**
Lymnaeidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Physidae	0	0,0	0	0,0	0	0,0	13	0,2	**13**
Planorbinae	0	0,0	0	0,0	2	0,5	0	0,0	**2**

Corbiculidae	79	1,8	11	3,6	1	0,2	172	2,5	**263**
Total	**4444**	**100**	**309**	**100**	**416**	**100**	**6802**	**100**	**11971**

Appendix D: Total number and relative abundance (%) of macroinvertebrates sampled at the three biotopes in winter.

Taxon	Site 1	%	Site 2	%	Site 3	%	Site 4	%	Total
Turbellaria	0	0,0	0	0,0	0	0,0	16	0,6	**16**
Oligochaeta	36	2,6	12	5,6	0	0,0	19	0,7	**67**
Hirudinea	62	4,5	15	6,9	0	0,0	13	0,5	**90**
Potamonautidae	1	0,1	0	0,0	0	0,0	0	0,0	**1**
Atyidae	3	0,2	6	2,8	2	0,1	4	0,2	**15**
Baetidae	55	4,0	13	6,0	24	1,0	12	0,5	**104**
Caenidae	0	0,0	2	0,9	1	0,0	0	0,0	**3**
Heptageniidae	0	0,0	12	5,6	0	0,0	0	0,0	**12**
Leptophlebiidae	84	6,1	27	12,5	130	5,7	0	0,0	**241**
Tricorythidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Chlorocyphidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Coenagrionidae	11	0,8	11	5,1	6	0,3	6	0,2	**34**
Gomphidae	0	0,0	0	0,0	0	0,0	2	0,1	**2**
Libellulidae	0	0,0	2	0,9	0	0,0	3	0,1	**5**
Belostomatidae	0	0,0	0	0,0	0	0,0	1	0,0	**1**
Naucoridae	0	0,0	0	0,0	0	0,0	2	0,1	**2**
Nepidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Notonectidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Veliidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Ecnomidae	0	0,0	7	3,2	1	0,0	0	0,0	**8**
Hydropsychidae	16	1,2	11	5,1	21	0,9	3	0,1	**51**
Hydroptilidae	0	0,0	0	0,0	3	0,1	0	0,0	**3**
Dytiscidae	4	0,3	2	0,9	1	0,0	8	0,3	**15**
Elmidae	173	12,5	14	6,5	12	0,5	0	0,0	**199**
Gyrinidae	3	0,2	13	6,0	5	0,2	1	0,0	**22**
Ceratopogonidae	4	0,3	9	4,2	0	0,0	1	0,0	**14**
Chironomidae	319	23,0	33	15,3	11	0,5	21	0,8	**384**
Culicidae	0	0,0	1	0,5	0	0,0	0	0,0	**1**
Muscidae	3	0,2	0	0,0	0	0,0	0	0,0	**3**
Simuliidae	351	25,3	0	0,0	2070	90,2	2368	91,5	**4789**
Tabanidae	0	0,0	1	0,5	0	0,0	0	0,0	**1**
Ancylidae	12	0,9	7	3,2	4	0,2	0	0,0	**23**
Lymnaeidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Physidae	12	0,9	2	0,9	0	0,0	19	0,7	**33**
Planorbinae	0	0,0	0	0,0	0	0,0	0	0,0	**0**

Corbiculidae	236	17,0	16	7,4	5	0,2	88	3,4	**345**
Total	**1385**	**100**	**216**	**100**	**2296**	**100**	**2587**	**100**	**6484**

Appendix E: Total number and relative abundance (%) of macroinvertebrates sampled at the three biotopes in spring.

Taxon	Site 1	%	Site 2	%	Site 3	%	Site 4	%	Total
Turbellaria	0	0,0	0	0,0	0	0,0	0	0,0	0
Oligochaeta	39	1,8	16	2,9	0	0,0	9	0,4	64
Hirudinea	160	7,2	12	2,2	2	0,9	16	0,7	190
Potamonautidae	0	0,0	0	0,0	1	0,4	0	0,0	1
Atyidae	7	0,3	6	1,1	0	0,0	4	0,2	17
Baetidae	205	9,3	84	15,4	44	19,4	25	1,1	358
Caenidae	4	0,2	108	19,8	0	0,0	0	0,0	112
Heptageniidae	0	0,0	3	0,5	0	0,0	0	0,0	3
Leptophlebiidae	6	0,3	105	19,2	6	2,6	0	0,0	117
Tricorythidae	0	0,0	6	1,1	0	0,0	0	0,0	6
Chlorocyphidae	0	0,0	1	0,2	0	0,0	0	0,0	1
Coenagrionidae	300	13,6	5	0,9	12	5,3	0	0,0	317
Gomphidae	0	0,0	0	0,0	0	0,0	2	0,1	2
Libellulidae	0	0,0	1	0,2	1	0,4	0	0,0	2
Belostomatidae	0	0,0	0	0,0	0	0,0	0	0,0	0
Naucoridae	2	0,1	3	0,5	2	0,9	4	0,2	11
Nepidae	0	0,0	0	0,0	0	0,0	0	0,0	0
Notonectidae	0	0,0	1	0,2	0	0,0	0	0,0	1
Veliidae	0	0,0	2	0,4	1	0,4	0	0,0	3
Ecnomidae	0	0,0	1	0,2	0	0,0	0	0,0	1
Hydropsychidae	5	0,2	8	1,5	0	0,0	14	0,6	27
Hydroptilidae	4	0,2	2	0,4	5	2,2	0	0,0	11
Dytiscidae	3	0,1	0	0,0	0	0,0	0	0,0	3
Elmidae	5	0,2	0	0,0	2	0,9	0	0,0	7
Gyrinidae	215	9,7	0	0,0	18	7,9	0	0,0	233
Ceratopogonidae	25	1,1	0	0,0	0	0,0	0	0,0	25
Chironomidae	511	23,1	96	17,6	3	1,3	0	0,0	610
Culicidae	0	0,0	0	0,0	0	0,0	0	0,0	0
Muscidae	8	0,4	0	0,0	0	0,0	2	0,1	10
Simuliidae	150	6,8	7	1,3	59	26,0	136	5,7	352
Tabanidae	0	0,0	0	0,0	0	0,0	0	0,0	0
Ancylidae	99	4,5	7	1,3	58	25,6	0	0,0	164
Lymnaeidae	31	1,4	0	0,0	0	0,0	0	0,0	31
Physidae	255	11,5	0	0,0	0	0,0	561	23,6	816
Planorbinae	0	0,0	0	0,0	11	4,8	0	0,0	11

Corbiculidae	177	8,0	72	13,2	2	0,9	1606	67,5	1857
Total	**2211**	**100**	**546**	**100**	**227**	**100**	**2379**	**100**	**5363**

Appendix F: Total number and relative abundance (%) of macroinvertebrates sampled at the three biotopes in summer.

Taxon	Site 1	%	Site 2	%	Site 3	%	Site 4	%	Total
Turbellaria	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Oligochaeta	5	0,6	8	1,6	0	0,0	1	0,0	**14**
Hirudinea	131	14,4	1	0,2	0	0,0	12	0,5	**144**
Potamonautidae	5	0,6	9	1,8	1	0,2	1	0,0	**16**
Atyidae	4	0,4	2	0,4	7	1,5	0	0,0	**13**
Baetidae	30	3,3	92	18,3	8	1,7	25	1,0	**155**
Caenidae	5	0,6	12	2,4	1	0,2	2	0,1	**20**
Heptageniidae	0	0,0	4	0,8	0	0,0	0	0,0	**4**
Leptophlebiidae	0	0,0	8	1,6	3	0,6	0	0,0	**11**
Tricorythidae	0	0,0	3	0,6	0	0,0	0	0,0	**3**
Chlorocyphidae	0	0,0	1	0,2	0	0,0	0	0,0	**1**
Coenagrionidae	28	3,1	23	4,6	14	3,0	1	0,0	**66**
Gomphidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Libellulidae	0	0,0	4	0,8	1	0,2	3	0,1	**8**
Belostomatidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Naucoridae	0	0,0	7	1,4	11	2,4	3	0,1	**21**
Nepidae	3	0,3	1	0,2	0	0,0	0	0,0	**4**
Notonectidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Veliidae	0	0,0	0	0,0	12	2,6	3	0,1	**15**
Ecnomidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Hydropsychidae	99	10,9	38	7,6	113	24,2	4	0,2	**254**
Hydroptilidae	12	1,3	0	0,0	12	2,6	0	0,0	**24**
Dytiscidae	1	0,1	2	0,4	0	0,0	3	0,1	**6**
Elmidae	10	1,1	6	1,2	1	0,2	0	0,0	**17**
Gyrinidae	116	12,8	111	22,1	50	10,7	900	36,5	**1177**
Ceratopogonidae	32	3,5	0	0,0	0	0,0	9	0,4	**41**
Chironomidae	118	13,0	29	5,8	0	0,0	0	0,0	**147**
Culicidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Muscidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Simuliidae	70	7,7	5	1,0	120	25,8	257	10,4	**452**
Tabanidae	0	0,0	0	0,0	0	0,0	0	0,0	**0**
Ancylidae	12	1,3	7	1,4	112	24,0	0	0,0	**131**
Lymnaeidae	10	1,1	5	1,0	0	0,0	0	0,0	**15**
Physidae	200	22,0	18	3,6	0	0,0	646	26,2	**864**
Planorbinae	0	0,0	0	0,0	0	0,0	0	0,0	**0**

Corbiculidae	17	1,9	107	21,3	0	0,0	593	24,1	**717**
Total	**908**	100	**503**	**100**	**466**	**100**	**2463**	**100**	**4340**

APPENDICES