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

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Abstract

The study aimed to assess the impact of alluvial diamond mining on macroinvertebrate community composition at four selected sites in lower Vaal River in 2016 with the application of the South African Scoring System 5 (SASS5). Macroinvertebrates and physico-chemical water parameters such as pH, turbidity, electrical conductivity and dissolved oxygen were measured seasonally. Turbidity levels were high 5.98.0±0.63 NTU but never exceeded the water quality guidelines of aquaculture, whilst pH values did not vary much and were generally alkaline throughout the study. The macroinvertebrate community structure varied at all sites during the study period. Site 4 was the most impacted site dominated by the more tolerant macroinvertebrates such as Simuliidae, Baetidae and Gyrinidae. The study concludes that although alluvial diamond mining had a negative impact on macroinvertebrate community in the short-term period, they appeared to re-establish quickly once the mining operations stopped.

Keywords: Alluvial diamond mining, bioindicators, community structure, lower Vaal River macroinvertebrates, water quality

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Introduction

Mining is an unsustainable activity as it is extractive without replacing the resource, and minerals cannot 'regrow'. It is the largest gross domestic product (GDP) contributor to the economy of the Northern Cape Province in South Africa, followed by agriculture, the largest land surface user. The lower Vaal, Harts, Riet, and middle Orange rivers and their tributaries have been the focus of alluvial diamond mining operations for over 130 years (Norton *et al.*, 2007). The diamonds from these deposits are of exceptional size and quality, originating from the many diamond bearing kimberlites that lie within the current and ancient watersheds of these drainages (Norton *et al.*, 2007). Due to the meandering nature of the Vaal River, diamond deposits are spread in the riparian zones and river channel beds formed by erosion that occurred millions of years ago (Chutter, 1968). The alluvial diamond deposits are vigorously extracted by informal, small- and large-scale miners (Naidoo-Vermaak, 2006). 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 riverbanks, destroying vast areas of riparian vegetation (Heath *et al.*, 2004; Naidoo-Vermaak, 2006; Ramollo, 2011).

Sediments such as rock, gravel and sand are often stockpiled in the riparian zone and next to the river, where after it is left un-rehabilitated (Chutter, 1968). Heavy rainfall events often erode these dumps (Heath *et al.*, 2004; Ramollo, 2011) washing exposed soils into the waterways, increasing turbidity and Total Dissolved Solids (TDS) of rivers (Jones *et al.*, 2012). The level of erosion depends on the duration and intensity of the rain, and the slope of the dumps and surrounding area (Newcombe and MacDonald, 1991). The alluvial diamond mining does not use toxic chemicals in processing the diamonds as compared to many gold mines that discard their toxic waste in the natural water bodies. Previous studies at the mined sites in the lower Vaal River have showed water quality remained in a good to fair condition (Ferreira, 2008; Malherbe, 2013), yet little is known about how alluvial diamond mining affects river condition in general and aquatic fauna such as macroinvertebrate community structure in particular.

Macroinvertebrates live either permanently or during part of their life cycles in freshwater ecosystems, and are active at or near to the water surface, within fringing vegetation, and the benthos of aquatic ecosystems (Benetti and Garrido, 2010). They are considered good indicators of river integrity because they are localised and largely immobile compared to fish (Dickens and Graham, 2002). Furthermore, based on their tolerance or intolerance to water pollution, their abundance and diversity in streams can indicate the health of the aquatic ecosystems (Parmar *et al.*, 2016). Furthermore, they are easier to sample and can be identified to Family level. The main aim of this study was to determine the impact of alluvial diamond mining on the macroinvertebrate community composition at four selected sites in the lower Vaal River, Northern Cape Province. The objectives of the study were to determine the water quality and habitat suitability and relate these to instream macroinvertebrate community structure.

Material and Methods

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. Vaal River flow is controlled through the Grootdraai Dam in Mpumalanga, the Vaal Dam in Gauteng, and the Bloemhof Dam in the North West Province. It flows 1,415 km in a south westerly direction 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 localities of the four sampling sites (Warrenton, Windsorton, Delpoortshoop and Schmidtsdrift Weir) that are the focus of this study, fell within the lower reaches of the Vaal River (Figure 1) and where chosen based on their accessibility. The sites were sampled once off each season in April, July, October and December 2016; autumn, winter, spring and summer respectively for macroinvertebrates and physico-chemical water parameters. The study area receives a greater amount of rainfall in summer. The geology of the area is covered with alluvial deposits and clay soils covering residual soils and bedrock belonging to the Karroo, Transvaal and Ventersdorp supergroups with the post-karroo dolerite dykes and silts intruded into these geological formations. The western part of the system is underlained by Ghaap group, Campbell rand formation, Transvaal supergroup, sedimentary rocks with dispersed patches of shale, dolerite and andesite.

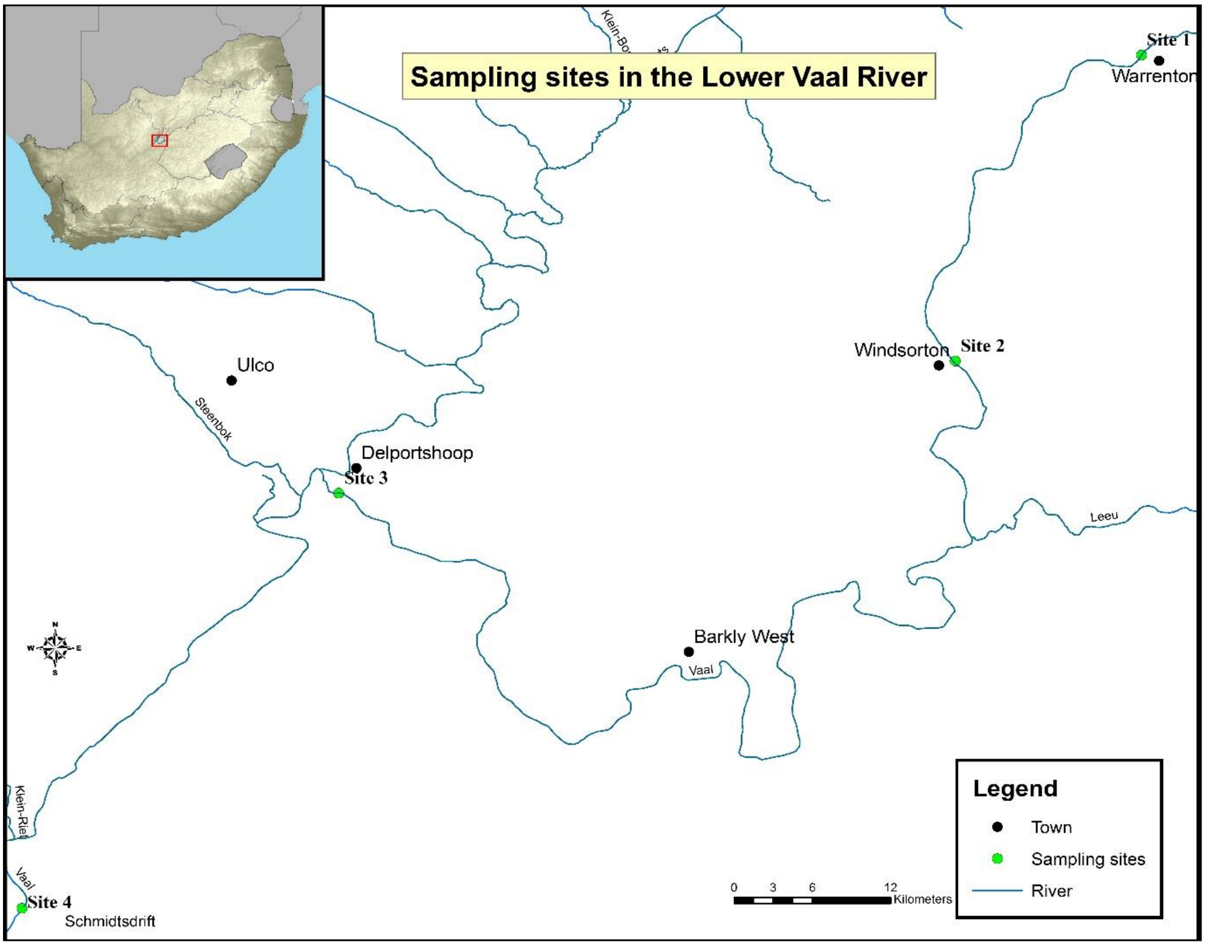

Figure 1: Locations of four sampling sites along the lower Vaal River.

Sampling sites

Site 1 (Warrenton): 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 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 a 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 part of the river is colonised by invasive water hyacinth (*Eichhornia crassipes*) and indigenous water grass *(Potamogeton pectinatus).*

Site 2 (Windsorton): 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 is diverted to prospect for diamonds and there is active mining that destroys and removes riparian vegetation, in- and out-of-stream. The disturbed riparian zone is colonized by invasive sponge-fruit salt bush *(Atriplex* sp.*)* and Spanish gold (*Sesbania punicea).*

Site 3 (Delpoortshoop): 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 material. Small stands of sweet thorn (*Vachellia karroo)* and sedges (*Cyperus papyrus)* occur at this site. The site is heavily modified by alluvial diamond extraction, causing a length of cobbles created by alluvial diamond diggings.

Site 4 (Schmidtsdrift): 28°42'42.9S; 24°04'20.4E

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.

Data Collection

Physico-chemical parameters

The pH, water temperature, dissolved oxygen (DO) and electrical conductivity (EC) were determined *in situ* by means of a handheld multi-parameter instrument (YSI model 54 Combo meter) and turbidity using a clarity tube at all sampling sites. There are no clear water quality guidelines for aquatic ecosystems (DWAF, 1996a) with regards to the physico-chemical parameters measured but thresholds of potential concern will include dissolved oxygen (<5 mg m^{-1}; likely to affect fish community composition and death if exposure is sustained for a prolonged period) and electrical conductivity (>70 mS m^{-1} = water has a distinctly salty taste) based on Davis (1975) and DWAF (1996b) respectively.

Macroinvertebrates sampling

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). The macroinvertebrate taxa are allocated scores of between 1 for tolerant taxa and 15 for intolerant taxa. The data interpretation is based on SASS Score, which is the sum of the sensitivity/tolerance scores for taxa present at a site, and average score per taxon (ASPT), which is the SASS Score, divided by the number of taxa.

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, a 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. 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 (Dickens and Graham, 2002).

Results

Physico-chemical parameters

The pH values measured during the study was alkaline, ranged from 8.3 to 8.9, and there was very little change during the year. The highest pH values were recorded in summer (8.7 ± 0.07) and lowest in winter (8.57 ± 0.05) (Table 1). Water temperature ranged from 14ºC to 27.2ºC, the lowest was recorded in winter (16.5 ± 0.96ºC) and highest recorded in summer (26.4 ± 0.48ºC). There were no discernible spatial trends in pH or temperature.

Dissolved oxygen concentrations ranged from 4.2 mg l^{-1} to 8.8 mg l^{-1}, with the lowest concentrations being measured at site 4; these were low and can be regarded as being hypoxic (<5 mg l^{-1}).

Electrical conductivity ranged from 76 mS m^{-1} to 120 mS m^{-1}, which exceeds the threshold of 70 mS m^{-1} throughout the study, and there was an increase in mean EC from 79.8 ± 2.3 mS m^{-1} at site 1 to 110.0 ± 4.1 mS m^{-1} at site 4. This increase indicates an increase in the amount of ions in the water.

The turbidity levels were very low (2.6 NTU - 7.8 NTU) during autumn at reference site 1, and only increased slightly at the other three more downstream sites. During winter, site 4 recorded a reasonably high level of turbidity (9.6 NTU) (Addendum A Table 1) as compared to other three sites. During spring turbidity level ranged between 5.4 NTU - 9.9 NTU (Addendum A Table 1). During summer, low levels were recorded at site 1 (2.4 NTU) as compared to other sites where there was slight increase. There are no turbidity target 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.

Macroinvertebrates

A total of 29167 individual macroinvertebrates, comprising of 36 taxa were recorded at all sites during the study period. During autumn, site 4 had the highest number of macroinvertebrates, followed by sites 1, 3 and 2 respectively (Figure 2). The taxa at site 4 showed a low diversity and were dominated by Simuliidae contributing 95.8% abundance, Corbiculidae (2.5%), Hirudinea (0.4%), Hydropsychidae (0.3%) and Oligochaeta (0.3%) respectively (Addendum B Table 2a-2d). Overall, Simuliidae showed a high relative abundance at all sites except at site 2 during the autumn. Planorbinae was only recorded at site 3 during autumn and spring and contributed only 0.5% of the site abundance (Addendum B Table 2a-2d). Turbellaria was only recorded at site 4 in very low abundance. Baetidae was the most common taxon recorded at all the sites during autumn (Addendum B Table 2a-2d).

During winter, site 4 recorded the highest number of macroinvertebrates followed by sites 3, 1 and 2 (Addendum B Table 2a-2d). Chironomidae and Elmidae were also recorded in high numbers at site 1 amongst other sites. Ecnomidae occurred only at sites 2 and 3 during autumn and winter. Tricorythidae, Chlorocyphidae, Nepidae, Notonectidae, Velidae, Lymnaeidae and Planorbinae were not recorded at any site during the winter, whilst Potamonautidae, Culicidae, Muscidae, Gomphidae and Tabanidae were recorded in low numbers (Addendum B Table 2a-2d). During this season, one of the 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 (Addendum B Table 2a-2d). 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 (1.1%), Hydropsychidae sp. (0.6%) and Oligochaeta (0.4%) (Addendum B Table 2a-2d). 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.

During summer, site 4 again had the highest number of macroinvertebrates, followed by sites 1, 2 and 3 respectively (Addendum B Table 2a-2d). The macroinvertebrates were dominated by more tolerant taxa such as Gyrinidae, Physidae, Corbiculidae, Simuliidae and Baetidae. respectively because of high diversity recorded at site 1 (Addendum B Table 2a-2d). 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 Heptageniidae, Leptophlebiidae, Chlorocyphidae, Baetidae, Hydropsychidae. Overall site 1 had a high mean value of 2993, followed by sites 2 (1621), 3 (1343) and 4 (1085) (Figure 2). 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.

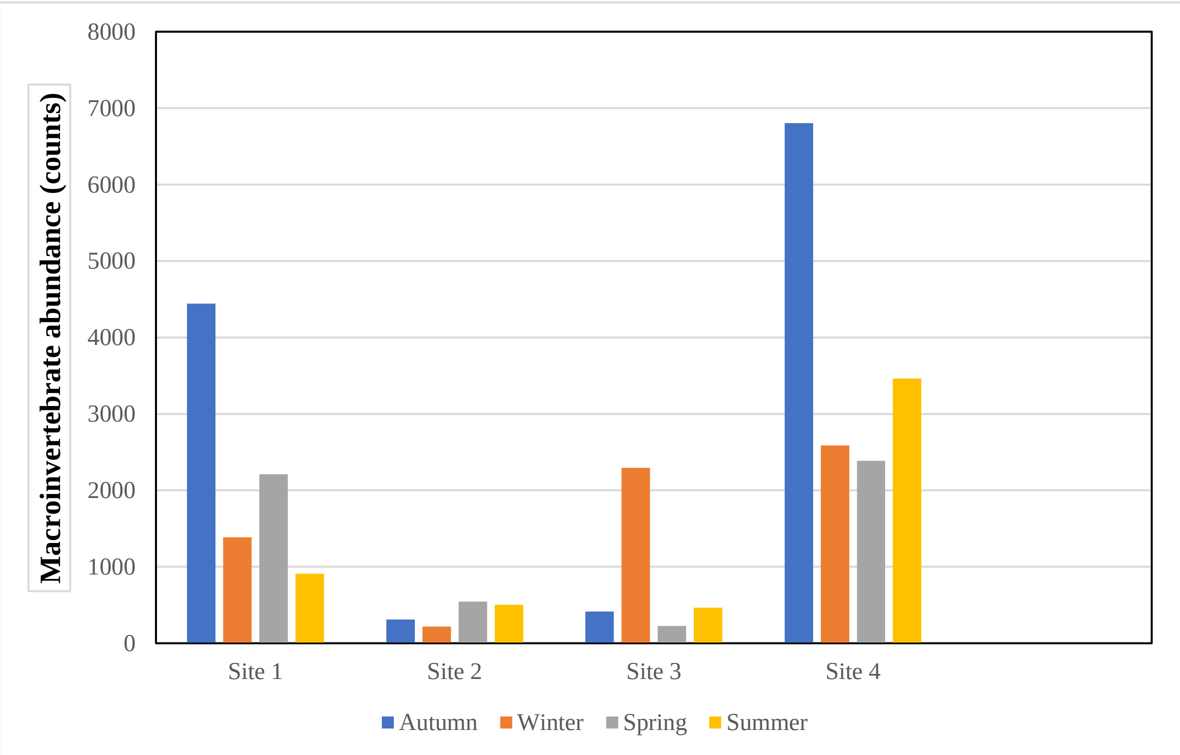

Fig 2: Seasonal abundance of macroinvertebrates across all four sites.

Discussion

Physico-chemical parameters

pH

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). 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, which can result in alkaline systems (DWAF, 1996a). Again, this can be attributed to the fact that the pH value above 8 in aquatic body is produced by photosynthetic rate that demands more carbon dioxide than quantities provided by respiration and decomposition (Wani and Subla, 1990; Soni and Thomas, 2013).

Dissolved oxygen

Dissolved oxygen (DO) is important for all forms of life (Terry *et al.,* 2017). 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, 1996b). 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). 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 low DO levels at site 4 can be attributed to the organic material from the livestock (pigs and horses) and from organic matter coming in from upstream. 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. 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.

Electrical conductivity

The EC of water is directly proportional to the total dissolved solids. There are no EC guidelines set for aquatic ecosystems in South Africa (DWAF, 1996b) 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 Targeted Water Quality Range (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 levels 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.

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, 1996b). The changes in seasonality and water temperature may lead to variations in the abundance, diversity and composition of aquatic communities. In this study the water temperatures appeared to be within a normal range, and did not exceed the TWQR throughout the study, therefore it can be assumed that the temperature and seasonality did not affect macroinvertebrates diversity.

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 and other microorganisms (DWAF, 1996b; Dallas and Day, 2004). There are no water quality guidelines for turbidity in aquatic ecosystems so aquaculture guidelines were used as an alternative, which indicated that 0 NTU - 25 NTU is a tolerable turbidity range for fish species (DWAF, 1996b). The turbidity range recorded during the study was 1.6 NTU - 9.9 NTU and never exceeded 25 NTU (Addendum A Table 1). The turbidity levels recorded by Malherbe (2013) at site 3 were very high compared to this study: 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 as vegetation stripping. The study concluded that alluvial diamond mining activities contributes to an increase in turbidity and subsequent loss in aquatic biodiversity in the river. This study corroborates the finding of Naidoo-Vermaak (2006) where the increase in turbidity was most probably the result of elevated erosion and associated total suspended solids coming from discharge of water used to wash diamonds coupled with upstream activities.

Macroinvertebrate community

Overall site 4 had the lowest diversity of families compared to the other three sites. The high abundance of Simuliidae at sites 1 and 4 can be attributed to the availability of microplankton and habitat (riffles). According to Schmitt *et al.* (2016), 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 that functioned as a breeding and recruitment area for Simuliidae and other taxa (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 macroinvertebrate's 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 large variation in river flows. Baetidae are common in low-lying rivers so tend to prefer gently flowing water and some in still waters, and during this study they were found in cobbles and vegetation.

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, stable flows and low abundances of their natural enemies/ predators are likely reasons for the high abundance of Simuliidae (De Moor, 1982). According to Kiel *et al.* (1998) other species of Simuliidae are able to colonise substrates within hours and can reach densities of thousands of individuals in just a few days.

During winter most of Simuliidae population concentrate in the aquatic environment and during summer will be in adult stage (De Moor, 1982). All the sites during summer recorded low abundances of Simuliidae as compared to other seasons which can be ascribed to the fact that a large population of the Simuliidae during 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 most likely due to high abundances of tolerant taxa e.g. Simuliidae (91.5%) and Corbiculidae (3.4%), 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 different species of Leptophlebiidae can have different habitat requirements. Gyrinidae, Atyidae and Coenagrionidae were dominant throughout with a preference for marginal vegetation.

Heptageniidae was recorded only at site 2 and 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 impacted (Dallas, 2007). The aforementioned taxa are very sensitive to changes in water quality, variations in flows, turbidity and substrate conditions (Golder, 2019), so once the water was impacted their abundance declined dramatically**.** The presence of Leptophlebiidae, Chlorocyphidae and Baetidae. at site 2 during this study showed that the site was recovering from mining impacts that lasted for eight months.

The water level at site 4 was extremely low during the study period. As a result, trucks of local mining companies were seen crossing the river, using it as a short cut to access their mining operation. This is likely to have impact on macroinvertebrate diversity by killing in-stream macroinvertebrates and stirring up fine sediments. At this site, more than fifteen pigs were seen feeding on the freshwater clams (Corbiculidae) and water grass (*Potamogeton pectinatus*). The pigs' presence could have caused a decline in dissolved oxygen levels that could ultimately have resulted in low macroinvertebrates diversity at the site. The pigs graze all day defecating in the river channel. Their faeces may have contributed to nutrient loading, which is likely to have placed a higher demand on dissolved oxygen and affecting macroinvertebrate diversity.

Horses were also observed disturbing the gravel, sand and mud with their hooves, which might also have resulted in sediment mobilisation and deoxygenation of the river water. The resuspension of sediments can also increase erosion and release of nutrients and heavy metals. 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 allowing them to survive in hypoxic conditions (Tyokumbur *et al.*, 2002). The complete absence of Chironomidae at sites 3 and 4 during summer suggests the heavy smothering of sediments on their biotopes emanating from upstream activities might have prevented them colonising these sites. The gravel, sand and mud (GSM) biotope was always low in macroinvertebrate diversity at all sites, and only a few individuals of the Gomphidae, Oligochaeta and Ceratopogonidae taxa were recorded. Site 4 (Schmidtsdrift) was the only site where Turbellaria were recorded in low numbers.

The GSM biotope was 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 Coenagrionidae and Gyrinidae were 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 macroinvertebrates community structures, as a paucity of habitat resulted in fewer macroinvertebrates being recorded at almost all sites. Overall, all sites except site 4 recorded a reasonable high family diversity throughout the study.

Limitations of the study

It was also observed that the macroinvertebrate community at all sites differed seasonally. These can be attributed to the fact that macroinvertebrate assemblages are best considered by combining the data collected from different periods of the year/seasons. Because of the variability in life cycles and changes in different macroinvertebrate groups, the taxa with seasonal distribution were sampled in one season but not in other seasons.

Conclusion

Macroinvertebrate community structures varied at all sites during the study period. The macroinvertebrate community in the Vaal River re-established quickly after three weeks due 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, the macroinvertebrate diversity were impacted, some of the macroinvertebrates moved with the river flow, perhaps that's why the macroinvertebrates were able to recover and establish quickly. The adult stages of some of the macroinvertebrates are terrestrial and always flew along the river banks which made it quick for the river to recover and start supporting sensitive macroinvertebrates. The alluvial diamond mining also created a length of cobbles that served as habitat of the macroinvertebrates. Therefore, it can be concluded that the alluvial diamond mining impact on macroinvertebrates was medium to low, whilst on riparian vegetation was highly severe.

Conflict of interest

The author declares that there are no conflicts of interest regarding the publication of this paper.

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Addendum A

Table 1. Physico-chemical parameters measured at the four sites across all seasons.

Date	Site	pH	DO (mg l^{-1})	EC (mS m^{-1})	Temperature (ºC)	Turbidity (NTU)
Apr. '16 (Autumn)	**1**	8.8	7.9	85	19.2	2.6
	2	8.8	7.6	93	18.2	7.3
	3	8.5	5.3	88	20.2	7.4
	4	8.3	4.9	110	19.8	7.8
Jul. '16 (Winter)	**1**	8.5	8.4	82	18	1.6
	2	8.5	6.7	86	16	5.2
	3	8.3	8.2	90	18	4.1
	4	8.5	4.7	100	14	9.6
Oct. '16 (Spring)	**1**	8.6	8.8	76	20.2	7.6
	2	8.4	6.4	86	21	8.5
	3	8.4	8.2	89	23.6	5.4
	4	8.6	4.2	110	21.2	9.9
Dec. '16 (Summer)	**1**	8.6	8.2	76	26.8	2.4
	2	8.6	6.6	92	25	6.4
	3	8.9	6.3	98	26.4	4.4
	4	8.8	4.4	120	27.2	5.5
	Avg ± SE	8.57±0.05	6.68±0.39	92.6±3.09	20.9±0.98	5.98.0±0.63

Addendum B

Table 2a: Total number and relative abundance (RA) values of macroinvertebrate taxa sampled at the three biotopes in autumn at the four selected sites.

Taxon	Site 1	RA%	Site 2	RA%	Site 3	RA%	Site 4	RA%	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**
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**
Libellulidae	1	0.0	2	0.6	0	0.0	0	0.0	**3**
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**
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**
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**
Ancylidae	18	0.4	3	1.0	0	0.0	0	0.0	**21**
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**

Table 2b: Total number and relative abundance (RA) (%) of macroinvertebrates sampled at the three biotopes in winter.

Taxon	Site 1	RA%	Site 2	RA%	Site 3	RA%	Site 4	RA%	Total
Turbellaria	0	0.0	0	0.0	0	0.0	16	0.6	**16**
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**
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**
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**
Physidae	12	0.9	2	0.9	0	0.0	19	0.7	**33**
Corbiculidae	236	17.0	16	7.4	5	0.2	88	3.4	**345**
Total	**1385**	**100**	**216**	**100**	**2296**	**100**	**2587**	**100**	**6484**

Table 2c: Total number and relative abundance (RA) (%) of macroinvertebrates sampled at the three biotopes in spring.

Taxon	Site 1	RA%	Site 2	RA%	Site 3	RA%	Site 4	RA%	Total
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
Naucoridae	2	0.1	3	0.5	2	0.9	4	0.2	11
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
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
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**

Table 2d: Total number and relative abundance (RA) (%) of macroinvertebrates sampled at the three biotopes in summer.

Taxon	Site 1	RA %	Site 2	RA%	Site 3	RA%	Site 4	RA%	Total
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**
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**
Veliidae	0	0.0	0	0.0	12	2.6	3	0.1	**15**
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**
Simuliidae	70	7.7	5	1.0	120	25.8	257	10.4	**452**
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**
Corbiculidae	17	1.9	107	21.3	0	0.0	593	24.1	**717**
Total	**908**	**100**	**503**	**100**	**466**	**100**	**2463**	**100**	**4340**