Macroinvertebrate population dynamics, community composition and diversity patterns of two coastal lakes in northern KwaZulu-Natal, South Africa

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Abstract

The 2018 South African National Biodiversity Assessment (NBA) identified eight freshwater lakes of national ecological importance and a lack of understanding of their biology. The assessment further called for baseline foundational data for their conservation. Aquatic invertebrates are considered to be reliable and sensitive biological indicators of environmental and water quality changes, and understanding aquatic invertebrate dynamics in these systems will provide a comprehensive understanding of how they can be better protected. The NBA also highlighted a gap in data associated with ecological response to landscape developments and climate change (mainly below average precipitation and increased temperatures) and how this contributes to aquatic resource conservation. This further complicates the modelling of important ecological thresholds and hampers the prediction of possible responses of these ecosystems to environmental changes. This gap informed the aims and rationale of this dissertation; to identify longer-term spatiotemporal trends in aquatic invertebrate communities in Lake Sibaya and Lake Mzingazi and to determine whether the surrounding land use changes could lead to long-term changes in aquatic invertebrate communities of both lakes by comparing recent survey data with historical datasets.

In Chapter 2, this dissertation investigated the population dynamics of freshwater shrimp, *Caridina africana* in Lake Sibaya and Mzingazi and compared the data to that of 1975 study published by Hart (1981). This was done to assess any changes in the *C. africana* populations due to the considerable changes in land use and weather patterns that have occurred in the last 48 years in and around the systems. Results from Lake Sibaya and Lake Mzingazi were also compared to determine any differences in urban and agricultural stressors presented to *C. africana* populations. This chapter hypothesised that increases in anthropogenic pollution, invasive species and other habitat modifications at Lake Sibaya and Lake Mzingazi would lead to (1) reductions in shrimp densities and changes in population dynamics when comparing with the 1975 data from Hart (1981). Additionally, it was predicted that (2) *Caridina africana* abundances found at Lake Mzingazi would be lower than those found at Lake Sibaya (3) due to different water quality variables associated with land use.

Results showed that *C. africana* population densities at Lake Sibaya and lake level recordings had experienced significant decreases since 1975 with densities being significantly lower in 2021. Additional differences seen in 2021 when compared to 1975 were that females were more abundant than males, individuals between the sizes of 3mm and 5mm were most abundant instead of those in the smallest size class (<0.83mm – 1.67mm) and females only dominated size classes above 4mm instead of all size classes above 2.5mm. Populations at Lake Sibaya were negatively correlated with nitrate concentrations in 2021 and populations at Lake Mzingazi were negatively correlated with temperature according to generalised linear models. These results emphasized the importance of pollution mitigation, sustainable water abstraction and the maintenance of natural water temperature ranges in the conservation of lentic *C. africana* populations. There was also no evidence that urbanisation and agriculture presented different threats to freshwater shrimp populations.

In Chapter 3, this dissertation aimed to quantify the littoral aquatic invertebrate diversity and assemblage patterns from Lake Sibaya and Lake Mzingazi to provide comprehensive baseline datasets for these coastal systems. This chapter also aimed to investigate the impacts of landscape developments and habitat change on aquatic invertebrate communities by understanding significant water quality parameters as drivers of community variation. Predictions for Chapter 3 were that increases in agricultural and anthropogenic disturbance and habitat modification will lead to (1) aquatic invertebrate community composition at lakes Sibaya and Mzingazi being structured according to water quality variables that stem from surrounding land-use activities, leading to (2) differing community structures at each lake. Lastly, it was hypothesised that (3) the presence of the invasive snail *Tarebia granifera* would likely be affecting the aquatic invertebrate diversity and composition of both lakes.

According to linear models, aquatic invertebrate abundance at Lake Sibaya was negatively affected by salinity, lake level and phosphate concentration, and positively associated with temperature. Taxa richness and Pielou's evenness at the lake were negatively associated with conductivity and nitrate concentrations respectively. The aquatic invertebrate community at Lake Sibaya also followed typical seasonal patterns. At Lake Mzingazi, Pielou's evenness was negatively associated with nitrate and ammonium concentrations and no typical seasonal patterns were evident in the community composition. Communities at Lake Mzingazi also exhibited resilience despite changes in physicochemical parameters, emphasising the difficulty in predicting aquatic community response to habitat modification due to lake-specific community resilience. *Tarebia granifera* populations at Lake Sibaya were found to negatively affect invertebrate diversity scores according to generalised linear models. Additionally, no individuals of *Melanoides tuberculata* were found in either system indicating the possibility that these native snails may have been outcompeted by their invasive counterpart. The prevalence of significant stressors associated with habitat disturbance and the unexpected results seen at Lake Mzingazi emphasized the importance of monitoring aquatic invertebrate communities in response to climate change and associated land use developments to adequately understand the long-term threats these changes pose to freshwater ecosystems and biodiversity conservation.

Keywords: Bioindicator, Landscape Development, Sibaya, Mzingazi, Freshwater Lake, Biological Monitoring, Biodiversity, Physicochemistry, Spatio-Temporal Assessment

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This thesis is the result of the author's own work, except where acknowledged or specifically stated in the text. It has not been submitted for any other degree or examination at any other university or academic institution.

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Chapter 1

Introduction to global change in South African freshwater systems with a focus on the Maputaland-Pondoland-Albany hotspot

1.1. Introduction

Freshwater ecosystems are important natural resources for both socio-economic and ecological well-being (Dudgeon et al., 2006). However, these ecosystems have been historically understudied, leaving them even more vulnerable to the ongoing ecosystem changes being noted due to global change. Furthermore, the Millennium Ecosystem Assessment (2005) found that 15 of 24 key ecosystem services were being degraded or overexploited. Therefore, this chapter seeks to highlight the ongoing consequences of global change in freshwater ecosystems to clarify how we may better conserve our aquatic biodiversity.

1.2. Global change in freshwater ecosystems

Freshwater systems are considered important natural resources with important ecosystem services, and they provide direct and indirect benefits to humans (and wildlife) around the world (Dudgeon et al., 2006; Vörösmarty et al., 2010). This includes the provision of major goods such as fishing, both for commercial and subsistence consumption, as well as drinking water and tourism attractions (South African Department of Science and Technology (SADST), 2017). The biodiversity of these systems also provides important scientific and educational tools for future generations (SADST, 2017). Key ecosystem services are categorized into four classes, (1) Support; through nutrient and water cycling, soil formation, photosynthesis and primary production (organism accumulation of energy or nutrients). (2) Provisioning of resources such as food and fresh water, (3) Regulation of climate, floods or disease and (4) through cultural importance such as aesthetics, spirituality and education (Millennium Ecosystem Assessment, 2005). Freshwater systems contribute substantially (and disproportionally) to all four classes of ecosystem services while also being exceptionally vulnerable (Jackson et al., 2016). According to the Millennium Ecosystem Assessment (2005), 15 of the 24 ecosystem services are considered degraded, which leaves people depending on these resources at risk.

Global change refers to a variety of phenomena, mainly driven by anthropogenic activities which are responsible for habitat modification, alien invasion, natural resource over-exploitation and pollution, all of which have changed most of the earth's ecosystems (Dudgeon et al., 2006; Mead, 2011; Reid et al., 2019). Global change affects freshwater ecosystems in a variety of ways, necessitating a comprehensive understanding of the drivers, causes and impacts of its associated stressors, which is one of the major concerns of the 21st century. This is important because freshwater systems occupy less than 1% of the Earth's surface but contain more than 10% of the earth's total biodiversity while providing important ecosystem services (Strayer and Dudgeon, 2010). Freshwater systems are vulnerable to environmental stressors acting on multiple levels (Jackson et al., 2016; Wangai et al., 2016; Fouchy et al., 2019; Orimoloye et al., 2020) and are also considered some of the most altered and threatened systems on earth (Carpenter et al., 2011; Tickner et al., 2020) further emphasising the importance of understanding climate and land-use alterations within freshwater systems.

1.3. Causes and impacts of global change

Dudgeon et al. (2006) reviewed the global change threats to freshwater ecosystems and identified five major causes of declines in organism populations and distribution changes. These were: overexploitation, water pollution, flow modification, destruction or degradation of habitat and invasion by exotic species. More than a decade later, Reid et al. (2019) produced a revised version identifying 12 emerging threats facing freshwater ecosystems globally. These were: changing climates, e-commerce, alien invasions, infectious diseases, harmful algal blooms, expanding hydropower, emerging contaminants, emerging nanomaterials, microplastic pollution, light and noise pollution, freshwater salinisation, declining calcium concentration and the accumulation of these stressors. Below, the major drivers are discussed and highlighted, with particular attention given to South Africa and related regions.

1.3.1. Overexploitation and flow modification

Freshwater ecosystems in Southern Africa are considered critical for important food and water provisioning services (Ashton, 2010). The increasing human population and demand for the exportation of raw materials, locally and abroad, are leading to the overexploitation of resources (Combrink et al., 2011; Jackson et al., 2016; Wangai et al., 2016; Buah-Kwofie and Humphries, 2017; Fouchy et al., 2019). Approximately 45% of freshwater fish species are considered to be over-harvested in Africa, posing a significant threat to food security in these vulnerable areas (Jackson et al., 2016; Fouchy et al., 2019). Overexploitation, specifically of long-lived species, coupled with other human-induced changes in freshwater systems may lead to local extirpation of species considered economically and ecologically important (Darwall et al*.,* 2011). The increasing settlement of humans in natural areas has resulted in competition for both resources and space. A common freshwater example of this is that of Nile crocodile (*Crocodylus niloticus*) populations within Lake Sibaya. Historically crocodiles have been killed for their skin but in recent times their decline, at Lake Sibaya specifically, seems to be associated with competition with humans. This includes excessive water abstraction and destroying nests (essentially destroying habitats) or large floods due to damming – which when released, floods nesting sites (Combrink et al., 2011). The over-abstraction of water is a growing threat to vulnerable South African freshwater systems (Kundzewics and Doll, 2009). Chronic overuse of water due to both household use and increasing agricultural demands combined with changes in average precipitation and river flow have created significant water stress on freshwater systems that ultimately may prove to be unsustainable (Kundzewics and Doll, 2009; Vörösmarty et al., 2010; Jackson et al., 2016).

Another consequence of the growing human demand for water resources has resulted in the construction of dams to create a more reliable supply of water in areas where this is not naturally possible (Mantel et al., 2010; Grill et al., 2015; Fouchy et al., 2019; Reid et al., 2019). These dams create river fragmentation, altering the natural flow regimes of rivers, which has led to the complete loss of certain regionally distinct flow regimes (Mantel et al., 2010; Grill et al., 2015; Fouchy et al., 2019). Flow modification in the form of dams has been well-researched with dams reducing the discharge of water, impairing fish migration and decreasing connectivity between rivers and floodplains (Mantel et al., 2010; Juracek, 2015; Fouchy et al., 2019). The biological consequences of this have shown to be impacting both water quality and quantity in South African rivers, with increases in total dissolved solids (TDS), expected to be a result of increased evaporation created by damming and a decrease or change in river flow attributed to high evaporation or flow diversions (Mantel et al., 2010). Grill et al. (2015) conducted a review of the effects of damming on freshwater systems and found that species migration, dispersal and their role in community structure tended to change as longitudinal connectivity was reduced in river habitats (Ziv et al., 2012; Altermatt, 2013).

Flow fragmentation/regulation affects just under 50% of all river volume worldwide (Grill et al., 2015). The recent expansion of hydropower as a sustainable energy source (3700 hydropower dams according to Reid et al. (2019)), may lead to over 90% of worldwide river volume being fragmented or regulated. This adds to the already established threat of damming worldwide, leading to alterations in water discharge, water temperature, total dissolved solids, sediment transport, invertebrate composition and fish migration (Bredenhand and Samways, 2009; Mantel et al., 2010; Grill et al., 2015; Juracek, 2015; Fouchy et al., 2019). Changes in thermal regimes (Poole and Berman, 2001) were also noted with the hypothesis that this may lead to losses of endemic species and an increase in the prevalence of invasive species which would ultimately affect biodiversity (Grill et al., 2015). Barbarossa et al. (2020) conducted a global species-level assessment concerning the degree of dam-induced river fragmentation with specific reference to the migration of lotic fish species. Their study found that South Africa already has one of the highest levels of fragmentation worldwide, along with the United States, Europe, India and China, with all four countries also hosting the lowest connection index values (Barbarossa et al., 2020). This makes river connectivity a key consideration in the commission of new hydropower dams within South Africa in order to conserve biodiversity.

Zilihona et al. (2004) conducted a study on Coleoptera within three habitat types of the Kihansi Gorge in Tanzania before and after the building of a hydropower plant and showed that the flow diversion created by the plant impacted the spray habitat (the mist zone created by a high falling waterfall) most significantly. The richness of coleopterans was found to decrease in all habitats after the building but Shannon-Wiener diversity indices showed the greatest change in diversity was within the spray zone, which was reduced by 95% after the introduction of the hydropower plant (Zilihona et al., 2004). The drying of this spray zone and loss of 90% of the water flow to the gorge has left the endemic Kihansi Spray Toad (*Nectophronoides asperginis*) highly threatened along with other resident biota (Zilihona et al., 2004). Bredenhand and Samways (2009) also conducted a study on aquatic invertebrate species diversity in the Kleinplaas dam on the Eerste River South Africa, and found significant differences above and below the dam wall, with the area below only hosting half the diversity as above the dam.

1.3.2. Habitat destruction/degradation

Habitat destruction is a well-known driver behind the loss of biodiversity and population declines within freshwater ecosystems (SADST, 2017), but its threat to these ecosystems is expected to be compounded by other emerging threats (Jackson et al., 2016; SADST, 2017; Reid et al., 2019). The nature of water pollution is changing swiftly as new water pollutants emerge and as the human population increases, increasing the use of these contaminants and the consequent spillage/leakage associated with them (Carter et al., 2007). Common pollutants found in freshwater systems as a result of increasing human settlement are nitrogen and phosphorous and their introduction into freshwater systems can result in high nutrient loading (Ndlela et al., 2016; Visser et al., 2016). Excessive increases in wastewater output can be harmful to many aquatic organisms (Ashton, 2010; Matthews et al., 2014; Ndlela et al., 2016; Visser et al., 2016). These harmful effects can span the entire trophic web such as the large crocodile die-offs that have been seen in the Olifants River and found to be related to pansteatitis. Pansteatitis is attributed to the ingestion of rancid fish after fish die-offs, which are typically a result of high levels of pollution in the river (Ashton, 2010).

South Africa's climate is considered semi-arid which makes it ideal for a variety of crop cultivation. Around 13% of South Africa's 122 million hectares of land is used for the cultivation of crops (Horak et al., 2021). This places large stressors on the freshwater ecosystems providing services required for cultivation – such as water use, land clearance and the use of pesticides (Kundzewics and Doll, 2009; Vörösmarty et al., 2010, Horak et al., 2021). South Africa is considered "the largest agrochemical user in sub-Saharan Africa with over 3000 registered pesticide products" (Horak et al., 2021, pg. 1). Although these pesticides do reduce the loss of crops, an important action for the food security of the country, they also affect non-target organisms via run-off into aquatic systems, spray drift or leaching into the soils (Horak et al., 2021). Low concentrations of pesticides are ever-present in the environment which can lead to toxic effects on the populations (both human and animal) experiencing this chronic or semi-chronic exposure (Horak et al., 2021).

Pesticides are biologically available for uptake by many aquatic species through their food and surface areas (such as their skin or gills) and can bioaccumulate within the trophic food web (Galindo-Reyes et al., 1999). Pesticides are known as endocrine disruptors which can alter normal hormone functioning in organisms exposed. This can change the physiology of these organisms while disturbing homeostasis which can lead to diabetes or respiratory problems along with changes in reproductive development in humans (Jacobson et al., 2012; Han et al., 2020; Teng et al., 2020) and certain neurodegenerative disorders in animals (Kass et al., 2020). Endocrine-disrupting chemicals have been detected in South Africa's largest floodplain (Pongola River Floodplain) as well as heritage sites containing biodiversity not found anywhere else in the world, such as the iSimangaliso wetland park (Horak et al., 2021). The Western Cape province in South Africa is home to large agricultural areas. These areas consist of orchards which are sprayed periodically with pesticides (mainly organophosphates) which are transported to non-target areas by rain as runoff (Schulz, 2001; Reinecke and Reinecke, 2007). A study found that total suspended solids in the Lourens River and its tributaries increased from 32 to 520 mg/l in the aftermath of a 28.8mm rainstorm in December of 1998 – exceeding national water quality standards as well as the standards set out by the US Environmental Protection Agency (Schulz, 2001). The contaminated runoff was found to enter the tributaries that bordered orchards with the suspended sediments persisting for around three and a half months after the rainfall without any additional input (Schulz, 2001) and up to six months within the soil following a spraying event (Reinecke and Reinecke, 2007). This further emphasizes the need to understand agricultural pollution and the effects it has on freshwater systems and the biodiversity found within these ecosystems.

1.3.3. Invasion by exotic species

Gallardo et al. (2016) conducted a global meta-analysis investigating separate cases of the introduction of invasive species into aquatic ecosystems between 1994 and 2014. Authors found that macrophytes, zooplankton and fish are significantly affected by the presence of invasive organisms such as the common carp (Ellender and Weyl, 2014), the Nile Perch (Goldshmidt et al., 1993) and *Tarebia granifera* (Appleton and Nadasan, 2002). Invasive species typically have rapid growth rates, broad environmental tolerances, high production (and success) of offspring with little parental input and early sexual maturity (Bownes and McQuaid, 2006; Zardi et al., 2018). With these traits, there are few environmental conditions considered unfavourable, which is why invasive species are typically able to outcompete native species with ease (Bownes and McQuaid, 2006; Mainka and Howard, 2010;

Zardi et al., 2018). Alien invasion can be further enhanced by increasing global temperatures and most stressors associated with anthropogenic pollution and climate change (Mainka and Howard, 2010).

Several invasive species have been introduced into Southern African freshwater systems, such as the common carp (*Cyprinus carpio*), which is considered a vector of pathogens and parasites (Ellender and Weyl, 2014; Jackson et al., 2016). Such co-introduced parasites have been found to affect the native sport fish, *Labeobarbus aeneus*, in South Africa (Ellender and Weyl, 2014; Jackson et al., 2016). Further, the common carp alters colonised habitats by resuspending sediments while using its bottom-grubbing feeding strategy, ultimately increasing turbidity and nutrient loading, thus completely changing the environment (Ellender and Weyl, 2014; Jackson et al., 2016). A worst-case scenario for introduced species is that of the Nile perch (*Lates niloticus*) which was introduced into Lake Victoria in 1954. This led to an explosive population increase around the 1980s which caused the predation and extirpation of around 200 native fish species in the Lake. This is considered to be the largest vertebrate extinction event to occur during the 20th century (Goldschmidt et al. 1993; Jackson et al., 2016).

South African freshwater systems in the eastern side of the country are being threatened by the invasive snail *Tarebia granifera*, which was first reported in 1999 in a concrete-lined Mandeni reservoir (Appleton and Nadasan, 2002). Since then it has become widespread throughout the North-eastern coastline of the country, mainly in KwaZulu-Natal and now spreading inland into Mpumalanga and Limpopo provinces of South Africa. *Tarebia granifera* invaded approximately five degrees of latitude (25°S-30°S) in 10 years (Appleton et al., 2009) and has been associated with the loss of two native gastropod species in Puerto Rico (mechanism unknown) (Appleton et al., 2009). This event could be repeated in the case of Southern African lakes where native species, such as *Melanoides tuberculata*, have been shown to occur in lower numbers when co-occurring alongside *T. granifera* (Appleton et al., 2009). It has been suggested that temperature may be a highly significant distribution predictor with *T. granifera* consistently being found in waters above 24°C (Appleton et al., 2009). *Tarebia granifera* has also been shown to colonize both brackish and moderately saline habitats and a wide variety of natural and man-made substrata (i.e. sand, mud, rock, concrete bridge formations, concrete walls and floors of dams, irrigation canals and ponds) (Appleton et al., 2009*). Tarebia granifera* are also able to disperse upstream due to their high tolerance to turbulent waters (Appleton et al., 2009) which only further emphasizes their resilience, dispersal ability and ultimately their threat to South African freshwater biodiversity.

1.3.4. *Changing climates*

Climate change has been the most prevalent in the form of increases in global average temperatures (Dallas and Rivers-Moore, 2014), increases in carbon dioxide (CO_2) (Carter et al., 2007) and changing rainfall patterns (Horak et al., 2021), which have altered flow regimes, created changes in river flow patterns (Whitfield et al., 2016) and caused reductions in lake levels (Hickley et al., 2004; Jackson et al., 2016; Carnie, 2020; Whitelaw and Van Rensburg, 2020). Africa is one of the continents considered most vulnerable to changing climates due to its low adaptive ability, particularly to changes in water availability, and the intensity of predicted climate change impacts in the region (Carter et al., 2007). These effects can cause organism distribution and phenological shifts, harmful algal blooms and changes in interspecific interactions (Mead, 2011; Dallas and Rivers-Moore, 2014; Ndlela et al., 2016; Visser et al., 2016; Reid et al., 2019; Tanentzap et al., 2020).

Climate change has both direct and indirect consequences on freshwater ecosystems. These effects can include; the favoured growth of cyanobacteria over other eukaryotic algae (Matthews, 2014; Ndlela et al., 2016; Visser et al., 2016), distribution changes in lotic environments due to thermal tolerance limits (Mead, 2011; Sydeman et al., 2015; Whitfield et al., 2016; Sintayehu, 2018), increased stratification (Kraemer et al., 2015; Reid et al., 2019; Woolway et al., 2021), phenology changes (such as spring phytoplankton blooms) (Winder and Schindler, 2004; Tanentzap, 2020), increases in the incidence of pathogens or disease transfer (Ziervogel et al., 2014; Ndlela et al., 2016; Visser et al., 2016) and an increase in invasion and extinction events (Ziervogel et al., 2014; Jackson et al., 2016). Lastly, other studies have also observed changes in interspecific interactions, in terms of trophic hierarchy, mismatches in phenology and possibly changes in ecosystem productivity (Winder and Schindler, 2004; Scheffers et al., 2016). Rising temperatures combined with flooding have been said to increase the risk of contracting infectious waterborne or vector-borne diseases for populations considered vulnerable (Dallas and Rivers-Moore, 2014). However, Ziervogel et al. (2014) found that modelling showed no overall increase in the incidence of malaria in South Africa preceding their study, which concurs with the findings of Reid et al. (2019) that the severity of the threat of pathogen and disease to freshwater biodiversity is still yet to be completely understood. The biological consequences of climate change are discussed in further detail in section 1.4 below.

Changes in precipitation include increased rainfall variation in a country where rainfall is already highly variable (Horak et al., 2021). Reduced rainfall can cause reduced connectivity when aquatic systems dry up. This can lead to isolation, possibly recruitment failures and ultimately local extinctions of aquatic organisms (Dallas and Rivers-Moore, 2014). Four perennial rivers in the South-Western Cape of South Africa that were studied showed consistent declines in the availability of habitat for aquatic invertebrates with an increase in habitat fragmentation, biotope isolation, low river velocity and shallow biotopes becoming more dominant (Dallas and Rivers-Moore, 2014). It has been predicted that large decreases in rainfall will cause many perennial rivers to transform into intermittent rivers leaving dependent aquatic organisms vulnerable (Reid et al., 2019).

1.3.5. Harmful algal blooms

Pollution in lentic freshwater systems has resulted in high nutrient loading, mainly nitrogen and phosphorous, from poor wastewater treatment (Ndlela et al., 2016; Visser et al., 2016). This has led to an increase in the occurrence of harmful algal blooms within freshwater ecosystems (Matthews, 2014). Increasing temperatures and CO_2 concentrations combined with increasing precipitation and the consequent runoff of increasing anthropogenic waste caused by global change favour the growth of cyanobacteria (Paerl and Huisman, 2009). Waste runoff and endorheic systems (with little to no outflow) provide ideal conditions for the growth of algae and cyanobacteria (Matthews, 2014; Kock et al., 2019). The increasing frequency of cyanobacterial blooms in freshwater lakes and reservoirs is of concern because certain species produce cyanotoxins, which can be harmful to people drinking the water, using the water for irrigation of crops, commercial agriculture or those subsistence fishing in the area (Paerl and Huisman, 2009; Visser et al., 2016; Reid et al., 2019). Twenty-three of the 50 largest standing water bodies in South Africa are said to have medium to severe cyanobacterial coverage (Matthews, 2014). Similarly, Ndlela et al. (2016) found that cyanobacteria blooms are recorded in almost all known water resources within the country, creating large concerns for the future of South African aquatic systems.

1.3.6. Microplastic pollution

Comprehensive assessments of microplastic pollution within freshwater systems is relatively limited, specifically at global, regional and basin scales (Zandaryaa, 2021). Most studies tend to focus on the direct effects on a single species, resulting in limited research concerning the effects of microplastics on different functional groups, species interactions and general ecosystem processes within freshwater systems (Ockenden et al., 2021). Studies have shown microplastic concentration increases in freshwater systems (Castañeda et al., 2014; Ockenden et al., 2021), but potential impacts are more than often adapted from marine studies or from lab studies that form results based on median concentrations up to six times higher than naturally occurring concentrations (Ockenden et al., 2021).

Various studies have shown that microplastics are ingested by freshwater biota (Eerkes-Medrano et al., 2015), such as invertebrates (Windsor et al., 2019), fish (Campbell et al., 2017) and birds (Holland et al., 2016) but research associated with the effects of this ingestion on organisms is limited (Eerkes-Medrano et al., 2015; Ockenden et al., 2021; Zandaryaa, 2021). Research on the effects of the chemical additives in plastics within freshwater systems only formed 6% (out of 146 papers) of studies reviewed by Ockenden et al. (2021). Previous reviewers of this topic have recommended more ecologically relevant studies that consider species interaction, functional feeding groups, ecotoxicological effects, methods of exposure and bioaccumulation to form a solid knowledge base for policy makers (Eerkes-Madrano et al., 2015; Ockenden et al., 2021; Zandaryaa, 2021). Some aquatic invertebrates have been studied for microplastic ingestion ability with reports showing inflammatory responses in the mussel *Mytilus edulis* and reduced membrane stability in digestive cells (von Moos et al., 2012) as well as translocation of particles from the digestive system to the circulatory system (Browne et al., 2008). Freshwater *Daphnia* spp. were found to translocate microplastics into cells, which then move into oil storage droplets (Rosenkranz et al., 2009) and the fish *Oryzias latipes* exhibited bioaccumulation, glycogen depletion, early tumour formation, single cell necrosis and fatty vacuolation after ingesting microplastics (Rochman et al., 2013). These case studies indicate that not only do freshwater biota ingest microplastics but there are also harmful consequences to this ingestion that threaten the residents of freshwater ecosystems.

1.3.7. Freshwater salinisation and declines in calcium availability

Freshwater salinisation is expected to increase in both rate and scale as a result of climate change with estimates showing that wetlands and forests have been affected by salinisation across the world (Wicke et al., 2011; Reid et al., 2019). Peatlands are also susceptible to salt-water intrusion via sea-level rise (Henman and Poulter, 2008). Areas that are already experiencing reduced rainfall due to climate change are at risk because of increased evaporation rates with little water recharge leading to an increase in solute concentration (Mills et al., 2013). Reductions in the growth, fecundity and diversity of freshwater invertebrates have also been reported with increases in salt concentration (Pinder et al., 2005). Increases in salt concentration lead to increased sodium and chloride ion concentrations within the cells of freshwater taxa, reducing their ability to ingest ions and water which are essential for their survival, thus causing loss of diversity and trophic cascades (Finlayson et al., 2013).

Salinity poses risks to the hatching success of crustacean eggs (specifically Anostraca, Notostraca, Spinicaudata, Copepoda, Cladocera and Ostracoda) (Mabidi et al., 2018). Taxa richness and abundance showed significant declines in salinities higher than 2.5 g/l according to Mabidi et al. (2018) with only Copepoda, Daphniidae and Ostracoda noted in the highest tested salinity concentration of 10 g/l. These densities were lower than those found at lower salinities, leading to the conclusion that salt-water intrusion risks the ecological balance of freshwater systems by the potential reduction in the abundance of large branchiopods (Mabidi et al., 2018).

Most studies focused on the effects of declining Calcium (Ca) concentrations within freshwater systems have focused on the Cladocera because of their commonality as keystone species within lake communities (Reid et al., 2019). Some *Daphnia* species have been found to have fairly high calcium requirements, with some failing to persist if calcium concentrations fall below optimal (e.g. 1.5 mg/l) (Jeziorski and Yan, 2006; Ashforth and Yan, 2008). Invertebrates with high Ca requirements (mainly Crustacea) may begin to struggle as Ca concentrations decline within freshwater systems due to their reliance on Ca for the structuring of their exoskeleton (Stevenson et al., 1985; Jeziorski and Smol, 2016). It is important to note that some organisms (such as the freshwater crayfish *Orconectes virilis*) may not show a strong relationship between growth or whole-body calcium content and calcium concentrations within studied systems (Edwards et al., 2016). This indicates that some taxa are more resilient to Ca declines in freshwater systems than others and highlights the importance of studying the effects of Ca decline on individual taxa as opposed to creating generalisations.

1.3.8. Cumulative stressors

The combination of global change stressors has led to the term 'polycrisis' (SADST, 2017). This term defines crises that are mutually reinforced by one another (SADST, 2017; Reid et al., 2019). An example of this is an increase in temperature, CO_2 and nutrient input facilitating the occurrence of harmful algal blooms (Ndlela et al., 2016; Visser et al., 2016). Almost all of the above-discussed threats to freshwater biodiversity can compound with one or many of the other threats, creating stress within freshwater systems that can be both direct such as temperature and CO_2 changes affecting organism physiology (Scheffers et al., 2016), and indirect including the changing of interspecific interactions due to temperature increases and mismatches in adaptation throughout the trophic food web (Tanentzap et al., 2020) which are discussed further in section 1.4 below. The world does not experience global change uniformly. Certain areas are more buffered due to environmental differences and others experience higher rates of change due to their environmental differences – making the predictability of the impacts of global change difficult. The combination of stressors in different ways across the world will only add to this prediction difficulty and once again informs the necessity of the holistic understanding of global change stressors in South African freshwater systems.

1.4. Measuring global change impacts

1.4.1. Physiological/morphological changes

Distributional changes occur when environmental conditions exceed the physiological tolerance of a species (Scheffers et al., 2016). Organisms with narrow environmental tolerances are therefore more affected by climate change and to survive, must migrate to an area within their physiological boundaries (Mead, 2011; Dallas and Rivers-Moore, 2014, Reid et al., 2019). This ultimately affects community composition, bringing about new communities (Laws, 2017; Sintayehu, 2018), novel species interactions (Sintayehu, 2018; Tanentzap et al*.,* 2020) and if keystone species are forced to migrate, trophic cascades are likely (Scheffer et al., 2005; Halpern et al., 2008). Interspecific interactions are affected under increased warming because unless all organisms within all trophic levels respond to the temperature change in the same way, the structure of that trophic web is altered (Tanentzap et al., 2020).

An example of this is how temperature affects metabolic processes. An increase in temperature can result in a higher metabolic rate and therefore a higher requirement for sustenance (Scheffers et al., 2016). This increased need for food can lead to an increase in the rate of prey consumption by higher trophic level organisms without a corresponding increase in prey species' metabolic rate. Studies have shown increases in the ingestion rates of consumers to be proportionally higher than any increases in their prey's ingestion rate (O'Connor et al., 2009; Kratina et al., 2012). This then results in a decrease in the abundance of those lower trophic-level species. This trend has been observed in Lake Maggiore, where temperatures increased by up to 2°C per year from 1981 to 1996. Populations of *Daphnia* species were recorded to experience a reduction in their abundance in correspondence with increased water flea (*Bythotrephes* sp.) abundance due to temperature increases between 1981 and 1996 (Tanentzap et al., 2020). Supporting South African literature for this trend seems to be limited.

Many studies have also shown a general trend where warm-water organisms are more positively influenced and cold-water organisms are more negatively influenced by increasing temperatures (Mead, 2011; Dallas and Rivers-Moore, 2014; Sydeman et al., 2015; Whitfield, et al., 2016; Sintayehu, 2018). One such example is that of the declining cool-water walleye (*Sander vitreus*) populations and increasing warm-water bass (*Micropterus salmoides*) populations seen in Wisconsin lakes (Hansen et al., 2017). Most of the research available is on migratory fish species and those that are economically important and there are also very few studies done in a South African context (Dallas and Rivers-Moore, 2014). Some studies have shown that increasing temperatures can cause shifts in organism sex ratios via temperature-dependent sex determination (Scheffers et al., 2016) as well as homogenising invertebrate communities (Baker et al., 2021; Sabater et al., 2022). There is also evidence showing that morphological changes occur due to increasing temperatures with species exhibiting smaller body sizes as the larger surface-volume ratio created by smaller bodies is typically favoured in warmer conditions (Scheffers et al., 2016).

1.4.2. Phenology and stratification changes

Increased stratification occurs due to climate change because the temperature of surface waters increases, which can cause a deeper thermocline (Kraemer et al., 2015; Reid et al., 2019). This affects resident biota and the quality of the water source (Kraemer et al., 2015). It has been found that lakes experiencing high temperatures and low wind exposure experience yearly stratification onset earlier than those with lower temperatures and higher wind exposure (Woolway et al., 2021). The earlier stratification was found to bring the onset of phytoplankton blooms earlier (Woolway et al., 2021) and would then require the adaptation of all trophic levels to sustain the current trophic structure (Poloczanska, et al., 2013; Laws, 2017; Tanentzap et al., 2020). Should all the trophic levels not respond in the same way, the entire trophic web may be restructured (Tanentzap et al., 2020). In a rare extended observational study aiming to detect the impacts of global warming, it was found that *Daphnia* grazers were unable to track the earlier peak in diatom abundance caused by an earlier spring bloom in Lake Washington, USA (Winder and Schindler, 2004). This resulted in a steady population decline over 40 years (Winder and Schindler, 2004; Tanentzap et al., 2020). However, studies regarding stratification and/or phenology changes are mainly concerned with the Northern Hemisphere and literature available on freshwater phenology changes in response to climate change in South Africa is scarce (Woods et al., 2021). Again, this shows the importance of visiting these topics in South African freshwater systems to better understand how these communities may be conserved.

1.4.3. Biodiversity loss

Extinction events have become increasingly more common over the last decades (Fouchy et al., 2019; Reid et al., 2019). Freshwater systems have been shown to experience the highest proportion of species threatened with extinction (Dallas and Rivers-Moore, 2014), while also experiencing more population decline (83%) than both the marine and terrestrial environments (Reid et al., 2019). Fang et al. (2006) found a variety of biodiversity changes associated with global change in central Yangtze lakes and the current status of African freshwater systems shows a general biodiversity decline and disturbed ecological processes, with possible effects on ecosystem productivity (Winder and Schindler, 2004; Fouchy et al., 2019, Reid et al., 2019). This can be attributed to habitat loss because of disruptions in sedimentation, changes in nutrient, sediment and lake levels and their exchange along the river system or blocking/ loss of migratory pathways (Fouchy et al., 2019). The worst vertebrate extinction event of the 20th century occurred in Lake Victoria in the 1980s due to the invasive Nile Perch, mentioned in section 1.3.3. above. The explosive population increase caused the predation and extirpation of around 200 native fish species in the Lake.

The IUCN red list has 31 inland freshwater species from sub-Saharan Africa labelled as extinct (IUCN, 2023). Notable extinctions from the 31 listed by the IUCN that are classified as resulting from global change are that of *Barbus* (now *Labeobarbus*) *microbarbis* (a large cyprinid species found in Lake Luhonda, Rwanda) which seems to have disappeared following the introduction of *Tilapia* in 1935 (De Vos et al., 1990; FishBase team RMCA and Geelhand, 2016). *Pantanodon madagascariensis* is thought to have been lost due to habitat loss caused by converting swamp habitat into rice fields as well as the introduction of *Gambusia* sp. (Sparks, 2016a). The cichlid *Ptychochromis onilahy* is thought to have been lost due to habitat loss caused by deforestation, increased sedimentation, the competition created by the introduction of tilapia and overfishing (Sparks, 2016b). Finally, the freshwater mussel *Unio malgachensis* is thought to have been lost due to mud and pesticides resulting from rice field cultivation along rivers (Van Damme, 2016). Although complete eradication of species as of now have been rather rare, the large biodiversity losses seen in various organisms due to climate change, anthropogenic pollution and invasive species (Barnosky et al., 2011; Weijters et al., 2009) indicates the need for monitoring and conservation of species before extinction events become more common.

1.5. Knowledge gaps in freshwater lake research

There are many gaps in knowledge surrounding global change within freshwater ecosystems in South Africa. Gaps in knowledge are mainly associated with minorly studied/emerging subjects such as declining calcium availability and freshwater salinisation or a lack of ecologically relevant studies as in the case of microplastic pollution within freshwater systems. Declining calcium availability is a slow process and requires either long-term monitoring or paleolimnological evidence to fully understand the driver behind the decline (Molot and Dillon, 2008). Cladocera is a heavily focused group with regard to calcium decline because of their importance as keystone species within lake communities (Reid et al., 2019). There has been no global synthesis of data on the increasing incidence of freshwater salinisation (Reid et al., 2019) and freshwater phenology changes in South Africa are also fairly understudied (Woods et al., 2020). Emerging nano-materials (i.e. Titanium Dioxide, Zinc Oxide and Silver) also have little long-term data to show the risks of their presence in freshwater ecosystems (Reid et al., 2019).

The 2018 South African National Biodiversity Assessment (Skowno et al., 2019) found a lack of aquatic invertebrate foundational data on some of the unique and ecologically important freshwater lakes in South Africa. To formally conserve these systems, the current ecological status and species checklists should be conducted and identified. Another knowledge gap comes from the lack of baseline/historical studies which makes it difficult to quantify present changes as there is no previous information to compare with (Jeziorski and Smol, 2016). There was also a gap in data associated with ecological response to climate change and the tendency of stressors to act in conjunction with one another which makes modelling important ecological thresholds and projected responses difficult (Skowno et al., 2019). Additionally, gaps exist in the knowledge of lake food webs and the effects of invasive species on these systems, the biogeochemical cycles (such as dissolved organic matter/carbon) of lake systems and long-term monitoring data for freshwater lake systems (Skowno et al., 2019). Reducing these gaps in knowledge and understanding the effects of global change on freshwater ecosystems is of utmost importance if freshwater ecosystems, the ecosystem services they offer and the biota within them are to be conserved.

1.6. Using historical data to address current research gaps in lake systems

Understanding the changes in ecosystem functionality in response to global change is an important step in the prediction of ecosystem response to global change. If we can understand how systems have changed from their historical state, we can use this information to predict how ecosystems may respond in the future and to inform future conservation decisions (Scheffers et al., 2016). An important tool for comparative conservation assessment, especially in light of ongoing habitat disturbance, is the use of long-term monitoring data or historical studies (Stuble et al., 2021). The presence of such studies enables the comparison of historical and current trends, which is imperative in understanding the effects of global change.

Stuble et al. (2021) explored the accuracy of snapshot resampling in assessing the effects of global change on biological systems. Using long-term datasets, such as long-term monitoring efforts, to track ecosystem changes under global change has been considered generally unfeasible due to the high associated costs (Stuble et al., 2021). Snapshot resampling is said to provide a more cost-effective way to track the changes within ecosystems and can create a clear picture of the ecological effects of global change. In worst cases, snapshot sampling can lead to "spurious conclusions" (Stuble et al., 2021, pg. 1) concerning this knowledge of changing ecosystems under global change conditions. However, it was ultimately concluded that this sampling can be an important tool in learning how ecosystems have changed and will continue to change in response to global change (Stuble et al., 2021).

Historical resampling has been used to assess the response of small mammals to global change (Moritz et al., 2008), shifts in the breeding range of birds due to temperature changes (Tingley et al., 2012) and community-level responses to climate change in restricted/endemic flora (Damschen et al., 2010). Plant-pollinator interaction shifts have also been observed after resampling in response to changes in the phenology of pollinators, species extinctions and landscape modification (Burkle et al., 2013). Distribution changes and abundance losses have been studied in response to temperature changes by resampling bumble bees (Pyke et al., 2016). Finally, losses of cool-weather species and gains in warm-weather species have also been observed by resampling bee communities in Munich Botanical Gardens (Hofmann et al., 2018).

The effect of global change on target organisms or ecosystems is key to understanding the impact of global change, thus efforts can be implemented towards mitigation with the ultimate goal of conservation. One such study is that of Hart (1981), which conducted a population assessment on the

Caridina shrimp in Lake Sibaya in 1975 to create a baseline of the productivity and life history strategies of the dominant tropical shrimp – *Caridina nilotica*. Comparing the present population dynamics of shrimp populations in Lake Sibaya to that of Hart (1981)'s results could show how the shrimp populations adapt to changing environmental conditions over time but additionally, in light of identifiable developments around the lake over the last forty years or so, a new baseline study is imperative for future comparison.

1.7. The Maputaland-Pondoland-Albany hotspot

The Maputaland-Pondoland-Albany (MPA) is a centre of global biodiversity found on the Eastern coast line of Southern Africa covering Mozambique, Swaziland and three provinces of South Africa (KwaZulu-Natal, Mpumalanga and the Eastern Cape) (Steenkamp et al., 2004). It is one of 34 global biodiversity hotspots, three of which are found in Southern Africa (The MPA in conjunction with the Cape Floristic Region and Succulent Karoo) (Myers et al., 2000; Steenkamp et al., 2004). Further, the MPA hotspot is the point at which six of South Africa's eight biomes meets including forest, thicket, grassland, nama- and succulent Karoo, fynbos and savanna (or bushveld) biomes (Steenkamp et al., 2004) and hosts three of the country's six marine bioregions (Delagoa, Natal and Agulhas/Indo-West Pacific ecoregions (Spalding et al., 2007; Porter et al., 2013)). The MPA is approximately 17 000 km^2 in size and hosts important conservation areas such as the iSimangaliso Wetland World Heritage Site, five RAMSAR sites including Kosi Bay, Lake Sibaya, St Lucia System, Turtle Beaches/ Coral Reefs of Tongaland and Ndumo Game Reserve. It is considered an important biodiversity region for plant (Steenkamp et al., 2004), vertebrate (Steenkamp et al., 2004; Perera et al., 2011) and invertebrate endemism (Perera et al., 2013) as well as one of the biological wonders of the world, hosting levels of diversity and endemism of global significance (SANBI, 2010) and "ecosystems that characterise the world's image of Africa" (SANBI, 2010, pg. vii).

The most diverse vertebrate group is birds, with over 500 bird species being commonly found within the hotspot (Steenkamp et al., 2004; Perera et al., 2011). Reptiles and amphibians are also highly diverse groups with 21.3% endemism in amphibians and 14.3% endemism in reptiles (Perera et al., 2011). Mammals are represented by a total of 193 species with at least three being endemic (Steenkamp et al., 2004; Perera et al., 2011). Initially, Steenkamp et al. (2004) found 20 endemic freshwater fish species out of 73 indigenous species but Perera et al. (2011) revisited this topic and found 10 endemics and 10 near-endemic freshwater fish species that are represented out of a total number of 97 indigenous species. Invertebrates account for a large portion of animal diversity on earth and within the MPA, velvet worms (Onychophora), butterflies (specifically Lycaenidae), dung beetles (Scarabaeinae) and earthworms (Microchaetidae) have been identified as groups with high diversity and endemism levels (Steenkamp et al., 2004). Perera et al. (2013) looked at invertebrate diversity within the MPA and found 108 endemic species out of 323 sampled invertebrate species. It is important to note that many centres of invertebrate endemism were found to be within forested areas, with only one centre of endemism within the savanna biome (Perera et al., 2013). These swamp forests are currently highly threatened (van Deventer et al., 2021), adding to the necessity of long-term biodiversity assessment of these forests.

Freshwater systems within the hotspot are considered some of the most diverse waters in Southern Africa, having been split into two broader ecoregions including the Zambezian Lowveld Freshwater ecoregion and the Amatola-Winterberg Freshwater ecoregion (Abell et al., 2008). These two ecoregions host species richness' that have been ranked near that of the Okavango Delta (SANBI, 2010). There are more than 12 freshwater fish species that are considered critically endangered within the MPA including species of *Barbus and Pseudobarbus* (SANBI, 2010) which can be attributed partially to changes in flow regimes of rivers and estuaries as well as the destruction of riverine habitat (both in-stream and riparian habitat) (Dallas and Rivers-Moore, 2014).

The area's groundwater system is highly dependent on rainfall and supports terrestrial and aquatic ecosystems (Carnie, 2020; Orimoloye et al., 2020; Whitelaw and Van Rensburg, 2020). There is a rough population of 18.4 million people living within the MPA hotspot, with 10 million mostly depending on natural resources for water, grazing for livestock and other resources such as wood, vegetation, and medicine (Smith et al., 2008). Rapid population increases have thus placed large pressure on the groundwater systems (Allanson, 1979; Grundling et al., 2013; Humphries and Benitez-Nelson, 2013; Miranda and Perissinotto, 2014; Buah-Kwofie and Humphries, 2017; Bate et al., 2018; Jansen Van Rensburg et al., 2019; Kock et al., 2019), thus understanding these pressures and their effects on resident biota cannot be overlooked.

1.8. Coastal lakes of Maputaland

Lake Sibaya is South Africa's largest natural freshwater lake found in the iSimangaliso Wetland Park on the eastern coast of KwaZulu-Natal, approximately 430 km North of Durban (Hart, 1981; Humphries, 2013; Kock et al., 2019). The lake is endorheic, oligo- to mesotrophic with a high chloride content, low plant nutrient concentrations and high transparency (Hart, 1981). The lake has an average depth of 13 m and is 41-43 m at its deepest point, covering a total surface area of around 60-70 km^2, sitting about 22 metres above sea level (Hart, 1981; Kock et al., 2019; Carnie, 2020; Whitelaw and Van Rensburg, 2020). Historically a lagoon/estuary, the lake has been separated from the ocean by large dunes covered in vegetation for about 5000 years (Whitelaw and Van Rensburg, 2020), meaning it was once an estuary and has a variety of unique estuarine relict fauna (Hart, 1981; Humphries and Benitez-Nelson, 2013; Kock et al., 2019). The lake used to host a large diversity of fauna and flora, such as the second largest Nile Crocodile and Hippopotamus populations in the KwaZulu-Natal province (Humphries, 2013; Kock et al., 2019). Unfortunately, these crocodile population numbers have declined rapidly over the years as human-wildlife conflict increased mainly due to grazing pressures and human population increases (Combrink et al., 2011).

Lake Sibaya consists of five parts – the main basin, the southern basin, the south-western basin, the western arm and the northern arm (Kock et al., 2019). Until recently the 5 parts were one entity, however, the Southern basin became separated from the main basin in around 2015 because of groundwater depletion due to increased commercial plantation, sustained below-average rainfall and warmer temperatures in the region (Carnie, 2020). Lake Sibaya is fed with an underground aquifer (at an estimated 21 106 m^3 per year) and precipitation at around 900mm per year during the summer months. Large amounts of water extraction required by Mbazwana and Sodwana Bay towns and their recent population increase threaten its water level (Humphries and Benitez-Nelson, 2013; Carnie, 2020). Along with human consumption, the recent and rapid water level drop of Lake Sibaya, resulting in the separation of the southern basin is primarily attributed to increases in commercial afforestation in the catchment (Carnie, 2020). Deep-rooted gum trees (*Eucalyptus* sp.) have been found to use twice as much groundwater compared to indigenous trees which could reduce the groundwater inflow to the lake by almost 30% in extended drought periods (SANBI, 2010; Carnie, 2020). If the lake level continues to drop, it has been projected that Lake Sibaya is at risk of salt-water intrusion from the Indian Ocean (Carnie, 2020), which would have negative social and ecological impacts on the system.

Lake Mzingazi is also a freshwater, estuarine-relict coastal lake and is found in the same region (i.e. MPA) as Lake Sibaya. It is a much smaller system with a surface area of 10.3 km^2 (Van Tonder et al., 1986). Unlike Lake Sibaya, Lake Mzingazi does not have a natural clear break with marine or estuarine connection - there is a saltwater barrier 1.4 km downstream of Lake Mzingazi, which prevents saltwater intrusion into the lake (Moloi, 2012). The lake was once connected to the saline Mzingazi River but became separated in 1955 by a weir built to sustain the need for potable drinking water (Moloi, 2012). This weir prevented the migration of euryhaline fish and crustaceans between the estuarine/marine environment and the freshwater of Lake Mzingazi. As a result of this, in 2005-2006, a new weir was built which had a pool and weir fishway to assist in the migration of organisms in and out of the lake (Moloi, 2012; Weerts et al., 2014). Lake Mzingazi has typically housed a unique combination of algae, plankton, benthic fish species and crustaceans originating from the Lake's estuarine connection history (Reavell and Cyrus, 1989). Very little research has been conducted on the coastal lakes of the Richard's Bay area but due to impending water pollution threats by residents, industry and mining in the area, the system is in need of baseline comparison data.

1.9. Aims and hypotheses

This dissertation aims to understand the temporal variability of *Caridina africana* populations and aquatic invertebrate communities over a 12-month period at Lake Sibaya and 10 months in the case of Lake Mzingazi. This is done with the intention of understanding how these communities respond to both natural variation within the systems and the ongoing consequences of the above-discussed landscape developments within and around freshwater coastal lakes to understand how we may better our aquatic biodiversity conservation efforts.

Chapter 2 aims to investigate the population dynamics of the freshwater shrimp *Caridina africana* found within Lake Sibaya and to compare the current population trends in Lake Sibaya over one year (March 2021 to February 2022) to those found in 1975 by Hart (1981). Secondly, this chapter aims to compare the population dynamics of *Caridina* shrimp at Lake Sibaya and Lake Mzingazi over a 10-month period from March 2021 to December 2021. This chapter hypothesises that increases in anthropogenic/agricultural pollution, invasive species and other habitat modification at Lake Sibaya and Lake Mzingazi will lead to (1) shrimp densities at Lake Sibaya being lower with differences in population dynamics when comparing the current study to that of Hart's (1981) results from 1975. (2) *Caridina africana* densities found at Lake Mzingazi will be lower than those found at Lake Sibaya and (3) that these differences will be due to different water quality variables associated with land use.

Chapter 3 aims to quantify the littoral aquatic invertebrate diversity and assemblage patterns from Lake Sibaya and Lake Mzingazi over a 10-month period (March-December 2021) to form a comprehensive baseline dataset for use in future comparison. Additionally, this chapter aims to investigate the impacts of landscape developments and habitat change on aquatic invertebrate communities by understanding significant water quality parameters as drivers of aquatic invertebrate community variation in Northern KwaZulu-Natal lakes. Predictions for Chapter 3 are that increases in agricultural and anthropogenic disturbance and habitat modification will lead to (1) invertebrate community composition at lakes Sibaya and Mzingazi being structured according to water quality variables that stem from surrounding land-use activities which will (2) lead to different community structures at each lake. Finally, it is hypothesised that (3) the invasive snail *Tarebia granifera* will likely be affecting the aquatic invertebrate diversity and composition of both lakes.

Chapter 2

Revisiting the freshwater shrimp of coastal Lakes Sibaya and Mzingazi, Northern KwaZulu-Natal, South Africa

2.1. Global change effects on coastal lakes, Sibaya and Mzingazi

In the last 40+ years since the 1975 study of Hart (1981), several changes have occurred around Lake Sibaya in particular and coastal lakes in general within the Maputaland Pondoland Albany (MPA) hotspot. These global change impacts are discussed and highlighted below, as they form the basis of our comparative assessment of the temporal changes in *Caridina* shrimp population dynamics.

2.1.1. Groundwater aquifer depletion

The southern basin of Lake Sibaya has become detached from the main basin of the lake due to increased water demand stemming from increasing commercial plantations and anthropogenic water demands (Carnie, 2020; Whitelaw and Van Rensburg, 2020). The drop in water level occurred drastically between 2015 and 2016. If the water level continues to drop there are fears of saltwater intrusion from the ocean, which would compromise domestic water use, and completely change the system back to its historic saline state. Rapidly declining water levels lead to losses in the total inundated littoral flora, leaving organisms that typically inhabit those areas stranded or forced to face higher inter- and intraspecific competition as well as predation in the areas that do remain inundated (Gaeta et al., 2014).

Mathenjwa (2006) observed that communities around Lake Mzingazi had introduced large plantations of gum trees and sugar cane on the banks of the lake, leaving the lake susceptible to high water loss through evapotranspiration. Lake Mzingazi has a water balance of -84.3 million m^3 per year (calculated using abstraction rates and historical yields for Lake Mzingazi from a WRA report), indicating a clear imbalance in the use and replenishment of natural water even before these plantations were introduced (Mallory, 2000). The addition of water removal for irrigation of the golf course on the eastern side of the lake also contributes to this imbalance.

2.1.2. Anthropogenic pollution (Nutrient enrichment)

Although historically oligotrophic, studies have found increased nutrient loads (i.e. nitrogen and phosphorous) in Lake Sibaya (Bate et al*.,* 2018; Kock et al., 2019) which can mainly be attributed to increased human settlement and subsequent poor waste management within the catchment (DWAS, 2004; Humphries and Benitez-Nelson, 2013; Kock et al., 2019). Kock et al. (2019) found Lake Sibaya to be in a mesotrophic state, according to guidelines set out by the Department of Water Affairs and Forestry (DWAF) (1996) with high inorganic nitrogen and phosphate concentrations. Algal blooms were reported within Lake Sibaya prior to the current study (Humphries and Benitez-Nelson, 2013), indicating periods of excess inorganic nitrogen and/or phosphorous input. This pollution allows the growth of large blooms of algae (harmful algal blooms - HABs) which block light and consume most of the oxygen available in the water column due to high demand (Paerl and Huisman, 2009; Matthews, 2014; Ndlela et al., 2016; Visser et al., 2016).

These blooms are of particular concern in lentic waterbodies and more specifically wetlands as these areas tend to act as a sink, accumulating any waste or nutrients that find their way into the water which then cannot be removed as there is no outflow associated with the system (Dallas and Day, 2004; Dalu and Froneman, 2016; Kock et al., 2019) (as in the case of Lake Sibaya). Cyanobacteria are also known to produce cyanotoxins which can enter the water system and subsequently be mixed with water used for drinking, irrigation or subsistence fishing (Paerl and Huisman, 2009; Visser et al*.,* 2016; Reid et al., 2019). Eleven African countries have reported toxic and physicochemical parameters that are considered to be a result of harmful algal blooms (Ndlela et al., 2016). South Africa has produced the majority of this information due to the outbreaks of illness and animal deaths associated with HABs (Ndlela et al*.,* 2016).

Increases in human settlement in the area surrounding Lake Mzingazi have resulted in communities being as close as three meters from the lakeshore in some areas, drastically reducing the buffer zone that had been set by the Mandlazini Trust in 1997 and increasing the potential for both agricultural and anthropogenic pollution (Mathenjwa, 2006). Some communities living around the area of both Lake Sibaya and Lake Mzingazi also make use of pit latrines which are dug to an average depth of 1.5 m (Mathenjwa, 2006; Bate et al., 2018). Untreated waste can seep into the groundwater which, in the case of Lake Sibaya, is the main supply of water to the lake (Mathenjwa, 2006; Humphries and Benitez-Nelson, 2013). This can lead to increased nutrient loading in the system (Humphries, 2013; Humphries and Benitez-Nelson, 2013; Bate et al., 2018; Kock et al., 2019). Lake Mzingazi water treatment works received multiple complaints in 2002 about an odour and bad taste in the water which was found to be from *Anabaena laxa* and *Actinomycetes* sp. which are surface-growing algae aided by increasing nutrient concentrations (Mathenjwa, 2006). This indicates previous potability issues within the lake and highlights the necessity for long-term monitoring in this regard.

2.1.3. Landscape developments: Agriculture and forestry plantations

Agricultural pollution in the form of pesticides affects non-target organisms via run-off into aquatic systems, spray drift or leaching into the soils (Bate et al., 2018; Horak et al., 2021). Agricultural areas can create runoff of up to 300 kg/km^2 of phosphorous per year (Horne and Goldman, 1994). Commercial timber production has also caused large habitat transformation, such as degradation and fragmentation, within the MPA hotspot and this destruction is usually irreversible. Timber plantations can use between 500 million and 1 500 million hectares of water every year and at present cover almost 31% of the hotspot (SANBI, 2010; Carnie, 2020). This is much more than the natural vegetation these plantations are replacing which has ultimately led to a reduction in stream flow between 50 and 150mm per year at Lake Sibaya (SANBI, 2010; Carnie, 2020). Areas with significant timber plantations are in the KwaZulu-Natal Midlands, the Highland Grasslands and Zululand and areas with significant sugarcane plantations in KwaZulu-Natal and Swaziland (now Eswatini) (SANBI, 2010). These plantations place large amounts of stress on the water systems supporting them (Kundzewics and Doll, 2009; Vörösmarty et al., 2010; Jackson et al., 2016; Carnie, 2020). Levels of water extraction and flow modification for agricultural uses have been considered unsustainable within South African freshwater systems (SANBI, 2010), once again highlighting the importance of long-term monitoring and bioassessment of these important ecosystem services.

2.1.4. Alien invasive species

The invasive freshwater snail, *Tarebia granifera*, has been reported at both Lakes Sibaya and Lake Mzingazi (Appleton and Nadasan, 2002; Appleton et al., 2009; Miranda et al., 2011; Miranda and Perissinotto, 2014). *Tarebia granifera* was first reported in 1999 in a concrete-lined reservoir in Mandeni, KwaZulu-Natal (Appleton and Nadasan, 2002). Since then it has become widespread throughout KwaZulu-Natal and has been reported in various inland water bodies of Mpumalanga (Appleton et al., 2009) and Limpopo (Makherana et al., 2022). *Tarebia granifera* can reach over 1000 individuals per m^2 (Miranda et al., 2011). This invasive snail is known to have high salinity and temperature tolerances (Miranda et al., 2010), with populations driven by high pH, total dissolved solids (TDS), conductivity, rainfall and temperatures (Appleton and Nadasan, 2002; Jones et al., 2017; Makherana et al., 2022). The association of the snails with high conductivity and TDS is said to be related to their reliance on detritus and algal biofilms (Jones et al., 2017; Majdi et al., 2022), which form a large portion of their diet. Studies have also found decreases in both the growth and survival of the gastropods in environments with low conductivity (Larson et al., 2020). Their previously mentioned association with high pH is said to be related to the reduction in calcification when faced with increasing acidity (Makherana et al., 2022).

Tarebia granifera has been known to displace native gastropod species such as *Melanoides tuberculata* with predictions also showing their likelihood to restructure benthic communities and ultimately affect aquatic biodiversity (Appleton et al., 2009). Miranda and Perissinotto (2014) reported that the dominating taxon within one uninvaded lake was freshwater shrimp (*Caridina* sp.) while *T. granifera* dominated the similarity measures (between 32% and 99%) at invaded lakes (Miranda and Perissinotto, 2014). The results of this study indicate changes within littoral aquatic communities when in the presence of *T. granifera* invasion. There have, however, been studies that found that these benthic community structures were driven more (statistically) by environmental factors than the presence of *T. granifera* (Miranda and Perissinotto, 2014), highlighting the importance of physicochemical considerations in the understanding of macroinvertebrate community composition.

Other studies conducted at Lake Sibaya were unable to find the native snail *Melanoides tuberculata*, which was previously known to occur in Lake Sibaya (Miranda et al., 2011; Miranda and Perissinotto, 2014). Both studies do note the potential for these snails to have moved into deeper waters (which were not sampled) due to extraordinary competition by the invasive *T. granifera*. Other invertebrates threatened by *T. granifera* are bivalves, dragonfly-nymphs, polychaetes, isopods, amphipods and decapods (both shrimp and crabs) (Appleton et al., 2009). Large numbers of *T. granifera* will result in competition with other littoral invertebrates that feed on detritus and periphyton/biofilms. High densities (±500 individuals/m^2) were related to strong decreases in the availability of Chlorophyll-a within the benthos which was hypothesised to be a result of grazing of the invasive gastropod (Makherana et al., 2022). This confirms the increased dietary competition created by large *T. granifera* populations to organisms which share the same dietary niche.

Tarebia granifera snails have also been observed in Lake Mzingazi with densities reaching over 1000 individuals per m^2 at peak densities and 800 individuals per m^2 on average (Jones, 2014). The increasing human settlement around Lake Mzingazi has included a golf estate which was found to be illegally extracting water from the lake and had likely resulted in the additional invasion of water lettuce (*Pistia stratiotes*) through backwash - the plant was also found in the estate's ponds (Mathenjwa, 2006). Water lettuce is a free-floating invasive plant and can cover entire areas of water, creating de-oxygenated zones as with harmful algal blooms (Mathenjwa, 2006).

2.2. An overview of South African freshwater shrimp

Caridina shrimp are widely distributed around the country occurring in almost all provinces in rivers, estuaries and in lakes/dams (Kensley, 1981; Richard and Clarke, 2009) but most commonly in KwaZulu-Natal. *Caridina* shrimp are considered a taxonomically challenging genus with over 300 described species around the world (Wood et al., 2018). The taxonomy of South African specimens is no different with many changes occurring in the last 50 years. Kensley (1981) described three freshwater Atyid species found in South Africa i.e. *Caridina africana and C. typus*, both noted to occur in Zululand (North-Eastern, KwaZulu-Natal) and *C. nilotica* noted to occur in Free State, KwaZulu-Natal (KZN), and what was previously known as the Transvaal (now Gauteng, North West and Limpopo), which were revised by Richard and Clarke (2009) to include four new species; *Caridina natalensis* from KZN, *C. belazoniensis* paratype from the Usustu River, KZN, *C. susuroflabra* noted in Zululand and *C. umtatensis* in Mthatha, Eastern Cape. *Caridina susuroflabra* was described on seven type specimens collected during Richard and Clarke's (2009) study and has since been determined morphologically and genetically to be a junior synonym of *C. africana* using over 100 collected specimens (Wood et al., 2018). A 2010 study then described *Caridina serratirostris* from the Umbilo River in KZN and *C. brachydactyla* replacing *C. nilotica var. natalsensis* and *C. nilotica var. brevidactyla* from the Umgeni and Mthatha rivers (Richard and Clarke, 2010).

In addition to the freshwater *Caridina* species found within South Africa, Kensley (1981) described five freshwater *Macrobrachium* species in South Africa and one solely freshwater species of *Palaemon* both in the Palaemonidae family. *Macrobrachium equidens, M. Idella, M. lepidactylus, M. petersii and M. rude* were all noted to occur on the eastern coastline of South Africa in KZN, bordering Mozambique while *Palaemon capensis* was noted between Hermanus in the Western Cape and Port Elizabeth in the Eastern Cape (Kensley, 1981). These species do require brackish water during a period of their lives and are also found in estuaries boasting a higher salinity content.

Shrimp and other invertebrates form part of the diet of many fish species such as *Glossogobius* sp., *Croilia* sp., *Clarius* sp. and *Oreochromis* sp. (Blaber and Whitfield, 1977; Bruton, 1978; Howard-Williams, 1979; Adite and Winemiller, 1997; Dalu et al., 2017). The production of these fish is thought to be closely correlated with the production of detritivores, which play an important role in linking primary benthic resources and secondary consumers (Odum, 1970; Bowen, 1979). A study of *Sarotherodon mossambicus* (now *Oreochromis mossambicus*) in Lake Sibaya showed accelerated growth rates for juveniles feeding on the shallow sand shelves of the lake while adults feeding in deeper waters, were found to be suffering malnutrition with much slower growth rates (Bowen, 1979). This was concluded to be a result of much lower protein levels of organic matter as the lake deepened, associated with a reduction in available food sources in this area (Bowen, 1979), highlighting the importance of invertebrates as the base of the trophic web.

Detritivores play an important role in freshwater systems by recycling nutrients for primary consumers via decomposition (Hart, 1981; Allanson, 1990). Correlation between habitat diversity and decomposition rates in various streams across the world showed a positive relationship with the conclusion that the extinction of detritivores would affect the decomposition rates of aquatic habitats (Boyero et al., 2021). The decomposition of dead organic matter is vital within lake systems because of the lack of outflow associated with the systems (Buah-Kwofie and Humphries, 2017; Kock et al., 2019). Increases in dead organic matter with no breakdown can lead to nutrient loading which can lead to eutrophication (Humphries, 2013; Humphries and Benitez-Nelson, 2013; Siméon et al., 2014; Bate et al., 2018; Kock et al., 2019). The loss of *Caridina africana* and detritivores from this decomposition role would likely lead to increased eutrophication events due to reduced nutrient recycling (Boyero et al., 2021) which is a large threat to the ecology of closed systems, such as Lake Sibaya.

A study was conducted on *Caridina nilotica* (now *Caridina africana* according to Richard and Clarke (2009)) population dynamics of Lake Sibaya in 1975 but little data on the *Caridina* shrimp in Lake Sibaya exists post-1980s, with the last in-depth work on their populations in the lake being Hart (1981). This 1975 study found, in summary, that the shrimp bred perennially, females grew larger than males on average and that population densities fluctuated between months (Hart, 1981). It was also noted that sampling attempts became increasingly challenging due to increasing lake levels and vegetation inundation creating difficulty in finding populations (Hart, 1981). The production estimates of the shrimp population in Lake Sibaya were higher than any other recorded shrimp population production estimates according to Hart (1981).

Mackay and Cyrus (2001) reported finding 16 individuals of *Caridina nilotica* (*Caridina africana*) as well as noting the presence of *Macrobrachium lepidactylus* in Lake Mzingazi. The presence of *Macrobrachium* in Lake Mzingazi but not in Lake Sibaya is understandable as *Macrobrachium* species require brackish water for a portion of their lives and so should only be found in an area where migration into brackish water is possible (Reavell and Cyrus, 1989). Weerts et al. (2014) recorded the presence of *M. equidens, M idella, M. lepidactylus, M. rude, M. scabriculum, C. africana, C. indistincta* and *C. nilotica* during their sampling in 2002 of Lake Mzingazi.

In the last 40+ years, drastic declines in the water level of Lake Sibaya, as discussed in Chapter 1, have occurred due to various influences (Carnie, 2020). The sampling sites of Hart (1981) all occurred within the southern basin which has since become detached from the main basin due to the lake's water level dropping (Carnie, 2020). *Caridina* shrimp are one of many invertebrate groups that inhabit the littoral zone (Allanson, 1979) and are thus threatened by lake level decline, which would result in the loss or reduction of littoral invertebrate biodiversity (Hunt and Jones, 1972). They would likely move to areas where there is still shallow water and submerged vegetation, resulting in large amounts of both inter- and intra-specific competition in these areas as densities of littoral aquatic invertebrates would be much higher. The lack of life found when attempting to sample the Southern Basin in 2021 reiterates the concern for littoral invertebrates reliant on inundated vegetation.

Caridina shrimp have been relatively well studied with regard to their use as bioindicators due to their wide distribution and range of habitats (lotic and lentic water bodies), range of life history strategies, comprehensive global taxonomic research, the fact that they are readily available, highly abundant, ecologically important (forming part of the diet of many fish species and aiding in the recycling of organic matter) and their sensitivity to environmental changes (Allanson, 1990; Ketse, 2006; Moulton et al., 2012; Siméon et al., 2014; De Grave et al., 2015; Boyero et al., 2021). *Caridina* shrimp have been concluded to be suitable bioassessment organisms for various salinity measures (Ketse, 2006), anthropogenic disturbance (Siméon et al., 2014; Jansen Van Rensburg et al., 2019), pesticide exposure (Jansen Van Rensburg et al., 2022) and wide temperature fluctuations (Hart, 1985). Further, *Caridina* populations have been said to reflect changes in anthropogenic and agricultural pollution making them good candidates to quantify landscape developments in freshwater systems (Siméon et al., 2014; De Grave et al., 2015; Jansen Van Rensburg et al., 2019). Finally, *Caridina africana* are native to South Africa and have been studied in depth at Lake Sibaya (Hart, 1981; Hart, 1985; Allanson, 1990).

The above evidence satisfies the guidelines set out by Rand (1995) for the use of bioindicator organisms. These being that the organism should be (1) sensitive, (2) widely available and abundant, (3) it should be native to the system in question, (4) it should be ecologically important and (5) background information should be adequately available.

2.3. Aims and hypotheses

This chapter aims to quantify the current temporal population dynamics of shrimp species at Lake Sibaya and Lake Mzingazi for one year (2021-2022) and further investigate the impacts of water level reduction and landscape developments/habitat change on *Caridina africana* populations by comparing the 1975 population dynamics found by Hart (1981) with the current 2021 study in Lake Sibaya.

This chapter predicts that increases in anthropogenic disturbance, invasive species and other habitat modification at Lake Sibaya and Lake Mzingazi will lead to the following: (1) *Caridina africana* densities at Lake Sibaya will be lower and population dynamics will differ in 2021 when compared to the 1975 historical dataset from Hart (1981). (2) *Caridina africana* densities will be lower at Lake Mzingazi than at Lake Sibaya and (3) that these differences will be due to differences in land use and the related differences in physicochemical parameters.

2.4. Materials and methods

2.4.1. Study sites

Lakes Sibaya and Mzingazi are coastal lakes found in Northern KwaZulu-Natal, South Africa (see Chapter 1, Section 1.7 for full study area description). Three sites were selected for data collection to replicate the study design of Hart (1981). At Lake Sibaya, sites 1-3 consisted of a sandy substrate. Site 1 (27,30222°S; 32,6765°E) consisted of dense *Imperatus* grass with little wave action, site 2 (27,33558°S; 32,70569°E) was dominated by fragmented *Phragmites* reeds with slightly more wave action than site 1 and site 3 (27,37319°S; 32,6736°E) also consisted of *Phragmites* reeds with more wave action than site 1 but less than site 2 (Figure 2.3). Comparatively, sites sampled in 1975 (Hart, 1981) were quantified in terms of the emergent hydrophytes and water-side plants. Site 1 consisted of *Typha latifolia capensis* and *Cyperus natalensis*. Site 2 consisted of *Phragmites mauritianus* and *Polygonum cylindrica*. Site 3 consisted of *Cyperus natalensis*. Further descriptions state that hydrophyte cover decreased from sites 1 to 3, contributing to significant differences in shrimp density seen between sites in this case, whereas wind and wave exposure increased from sites 1 to 3 (Hart, 1981).

At Lake Mzingazi, site 1 (28,760861°S; 32,079049°E) and site 2 (28,763113°S; 32,079933°E) consisted of a sandy substrate while site 3 (28,752812°S; 32,08249°E) had a rocky bottom. Site 1 was characterised by dense *Imperatus* grass and high wave action, site 2 was characterised by very fragmented waterlilies and little wave action, while site 3 consisted of dense *Imperatus* grass and high wave action.

Figure 2.1: Satellite map of Lake Sibaya in the KwaZulu-Natal province of South Africa. Showing three sampled sites from 2021 (solid outlines) (Site 1: 27.30222°S; 32.6765°E, site 2: 27.33558S; 32.70569°E, site 3: 27.37319°S; 32.6736°E) and three sampled sites from 1975 (broken outlines), commercial forestry areas (outlined in red) and human settlements (white text with black outlines)

Figure 2.2: Satellite map of Lake Mzingazi in the KwaZulu-Natal province of South Africa. Three sampled sites are indicated with solid outlines (Site 1: 28.760861°S; 3.079049°E, site 2: 28.763113°S; 32.079933°E, site 3: 28.752812°S; 32.08249°E). Notable human settlements are indicated in white text with black outlines

2.4.2. Physicochemical parameters

At each site, a handheld multimeter (Eutech PCS Tester 35 model) was used to record pH, salinity (Sal, ppm), conductivity (EC, µS/m) and water temperature (˚C) in the field. Additionally, a 500 ml water sample was collected from each site for ex-situ water nutrient analysis including nitrate (NO_3^-, mg/l), ammonium (NH_4^+, mg/l) and phosphate (PO_4^{3-}, mg/l). Phosphate concentrations were measured in the laboratory using the Hanna HI83399 Bench Photometer for Aquaculture. The phosphate high range (HR) measurement procedure uses the amino acid method adapted from Standard Methods for the Examination of Water and Wastewater, 18th edition and requires phosphate HR reagents A (Reagent code: HI93717A-0) and B (Reagent code: HI93717B-0). Nitrate and ammonium concentrations were measured by placing two YSI ProQuatro multiparameter meters fitted with an ISE sensor in the collected water samples. Calibration of the ISE sensor for nitrate and ammonium measurements follows a two-point calibration using hte standards that differ by 2 orders of magnitude (i.e. 1 mg/l and 100 mg/l).

Lake level records for Lake Sibaya were acquired from the South African Environmental Observation Network (SAEON) platform which has a long-term monitoring programme in place for the lake. Lake Mzingazi's water level data was obtained from the South African Department of Water and Sanitation (DWS) hydrological services (https://www.dws.gov.za/Hydrology/Default.aspx; station ID: W1R004). Dissolved oxygen (DO) would ideally form part of this analysis but equipment limitations meant we were unable to take DO in the field. Therefore, dissolved oxygen did not form a part of the environmental measurements.

2.4.3. Sampling of shrimp populations

In order to compare historical and current *Caridina africana* populations, this study replicated the methodology of Hart (1981) with minor modifications. Three littoral zone sites were selected at each lake to represent as much habitat variability as possible. The first modification from Hart (1981), who sampled the southern basin of Lake Sibaya in 1975, was that this basin during the present study was detached from the main basin with an exceptionally low water level. Sampling of the Southern basin was attempted to compare sites directly but no life was found and thus this study identifies three sites around the main lake. The second modification was Hart's (1981) study used a 0.1 m^2 metal cuboid with no top and bottom ends, however, this was challenging to procure so it was replaced by a $0.093m^2$ plastic bin with both ends open (Figure 2.4). Both sampling methods randomly pushed the sampling device into vegetated biotopes and used a hand net (≤500 µm) to catch *Caridina* shrimps within the sampling device area.

Collected *Caridina* shrimps were placed in a 10% ethanol solution for 30 minutes to render the shrimp insensible and induce respiratory depression (American Veterinary Medical Association, 2020). Thereafter, specimens were transferred into an 80% ethanol solution for later identification and long-term storage. In the laboratory, monthly male and female *Caridina africana* abundances and egg-bearing females were counted. Additionally, each specimen's rostral and carapace length was measured for quantification of population size structures and size-specific sex ratios as was done by Hart (1981).

Data from Lake Sibaya are based on 12 months of sampling between March 2021 and February 2022 whereas data for Lake Mzingazi were based on 10 months of sampling between March 2021 and December 2021.

b).

Figure 2. 3: Showing 0.093m diameter plastic bin with no top and bottom ends which was used for plunge sampling and *Caridina* shrimp collection during the 2021 study

2.4.4. Population parameter estimates

For comparability, population parameters were calculated as was done by Hart (1981) using the methodology of Cummins et al. (1969). Mean clutch size (Ê) (50 eggs per female) was determined using Hart (1981) instead of counting individual eggs. Embryos per unit volume (E) were calculated as the percentage of egg-bearing females found during quantitative sampling. Developmental time (D) was taken from Hart (1980) and adjusted according to recorded temperatures. Finite birth rate (B) was calculated using the following equation: $B = \frac{E}{(D)(N_0)}$ where N_0 is the population density at a given time. Instantaneous birth rate (b) was calculated as the natural log of (1+B). Population change (r) was calculated using the following equation: $\frac{(Natural \log N_t) - (Natural \log N_0)}{t}$, where N_0 is the initial population density and N_t is the final population density at a given time. Finally, instantaneous death rate was calculated as the difference between instantaneous birth rate and population change ($d = b - r$).

2.4.5. Statistical methods

All statistical analyses were conducted using R Studio (2009-2022 RStudio, PBC). Preceding all statistical methods was the checking of assumptions and the distributions of the data.

All data involved in the statistical analysis were checked for normality using Shapiro-Wilk normality tests (Function: “shapiro.test” from Package: “stats”) and were found to not be normal, necessitating the use of non-parametric tests, where necessary. Thus, Kruskal-Wallis tests were conducted on month, site and season variables using the function “Kruskal.test” (package: “stats”) to test the significance of categorical variables on variation in shrimp densities. Significant variables from the Kruskal-Wallis test results were then used in Dunn’s post hoc test using the Hochberg method (Function: “dunnTest” specifying method = “hochberg”, Package: “FSA”) for the adjusting of P-values to determine which category groups (month, site or season) were contributing to this significance. Wilcoxon rank sum tests were also conducted to compare the density of shrimp, lake level and water temperature at Lake Sibaya between years (1975 and 2021) as well as the density of shrimp and water temperature of Lake Sibaya and Lake Mzingazi using the “wilcox.test” function (package: “stats”).

A Principle Components Analysis was conducted using the “princomp” function (Package: “stats”) to assess the temporal variation in physicochemical parameters between Lake Sibaya and Lake Mzingazi. A biplot of the results was then created using the “autoplot” function (Package: “ggplot2”).

An important note is that the replication of Hart’s methods followed in 1975 (i.e. the use of monthly replicate sampling) incorporates the concern of pseudoreplication. Lack of independence in terms of both spatial and temporal realms will therefore play a significant role in the choice of validation procedures used for the models in this study. Davies and Gray (2015) state conclusions of pseudoreplication should come with appropriate effort to demonstrate the conclusion to be both statistically valid and ecologically meaningful. Therefore, it was decided that this lack of independence of samples would have to be proven statistically before attempting to replicate the study in a way that both accurately replicated the baseline study and avoided all possible pseudoreplication. The independence of samples was confirmed through the three steps. (1) Checking of residual heterogeneity, identifying no patterns within the residual plots of models. (2) The checking of autocorrelation, which is defined as the degree to which time-series data and lagged versions of that same time-series are dependent on one another. And, finally, (3) The addition of mixed effects models to be compared, via likelihood ratio tests, with initial nested models to assess whether significant improvements in the case of lack of independence were made by the addition of any mixed effects. No models with unacceptable levels of residual heterogeneity or autocorrelation are to be utilised to avoid the possibility of a type-I statistical error. The decision was also made to follow the idea that one should always try to explain the most variation in the data with the simplest statistical models to avoid models with singularity errors and the possibility of a type-II statistical error.

Generalised Linear Models (GLMs) were conducted using two families to compare AICc values and to choose the best-fitting model. All predictor variables were scaled using the “scale” function (Package: “Base”) to avoid any problems related to the different scales of the variables before the GLMs. The “scale” function (Package: “Base”) centres (χ-column mean) the data and then divides the centred value by the column’s standard deviation, producing a Z-Score ($Z = \frac{x - mean}{SD}$). The Negative Binomial (“glm.nb” using Package: “MASS”) and Poisson (“glm” using Package: “stats”, specifying “Family = poisson”) models were chosen due to the positively skewed count data for which both families are appropriate. The Poisson models operate under the assumption that the means and variances of the data are equal and violating this assumption can lead to over-dispersion, which is catered for in the Negative Binomial models. The model with the lowest AICc was chosen and this was the family used on the final model (Negative binomial) (Table S1). Variance inflation factors were checked to ensure no collinear variables using the “vif” function (Package: “car”) and all collinear variables (VIF>5) were removed.

Once these variables had been removed, two generalised linear mixed models, one using site and one using month as random effects, were conducted using the function “glmer.nb” (Package: “lme4”). Model performance was compared to the initial (nested) model’s performance using likelihood ratio tests through the “anova” function (Package: “stats”) and comparing the AICc values of all models using the “AICc” function (Package: “MuMIn”) (Table S2). This showed that both full models (mixed effects models) performed identically and that neither full model outperformed the nested model (P>0.05). To further confirm these results, residual plots using the “plot” function (Package: “Graphics”) were created to assess any improvements in residual patterns after the addition of either month or site as random effects and none were found. To assess the addition of these random effects on the temporal autocorrelation within the models, autocorrelation function (acf) plots (Function: “acf” from package: “stats”) were also created for residual values. In no case did the acf plot associated with the full (mixed) model outperform the nested (initial) model. The lack of significant likelihood ratio tests, improved AICc values, lack of residual pattern and lack of significant autocorrelation according to acf plots confirmed the use of the initial nested models going forward.

The final step was finding the most parsimonious model, which is defined as the model using the fewest variables to explain the most variation. This was determined using the “dredge” function (Package: “MuMIn”). Final models were then compared to the initial models to confirm the better-fitting model suggested by the dredge function. In an effort to include temperature in Lake Mzingazi’s model, the temperature variable was added to the most parsimonious model and VIF values were rechecked, showing no VIFs>5. The significance of variables was determined using Wald's test with the function "Anova" (Package: "car"). Temporal autocorrelation was once again assessed and found to be acceptable for all final models. The final models can be found in Table S3.

2.5. Results

2.5.1. Lake Sibaya: An overview of current and historic trends

Lake Sibaya experienced similar shrimp densities at all three sites throughout 2021 with site 2 experiencing slightly higher densities than the other sites (except for September where there is a substantial increase in shrimp density) (Figure 2.4A). There was no distinct seasonality seen throughout the year with populations fluctuating slightly from month to month. The patterns exhibited throughout 2021 follow similar patterns to the unpredictable fluctuations seen at the three sampled sites of 1975 (Figure 2.4B). The three sites sampled in both studies are different and no direct comparison of the densities at the three sites in both years is possible so these sites have been treated as replicates. Densities of *Caridina africana* found at Lake Sibaya in 2021 were generally much lower (ranging from 0 to around 1000 individuals) than those found in 1975 (ranging between 0 and over 3500 individuals). Shrimp densities found in 2021 in all three replicates were closest in value to Site/rep. 3 of 1975.

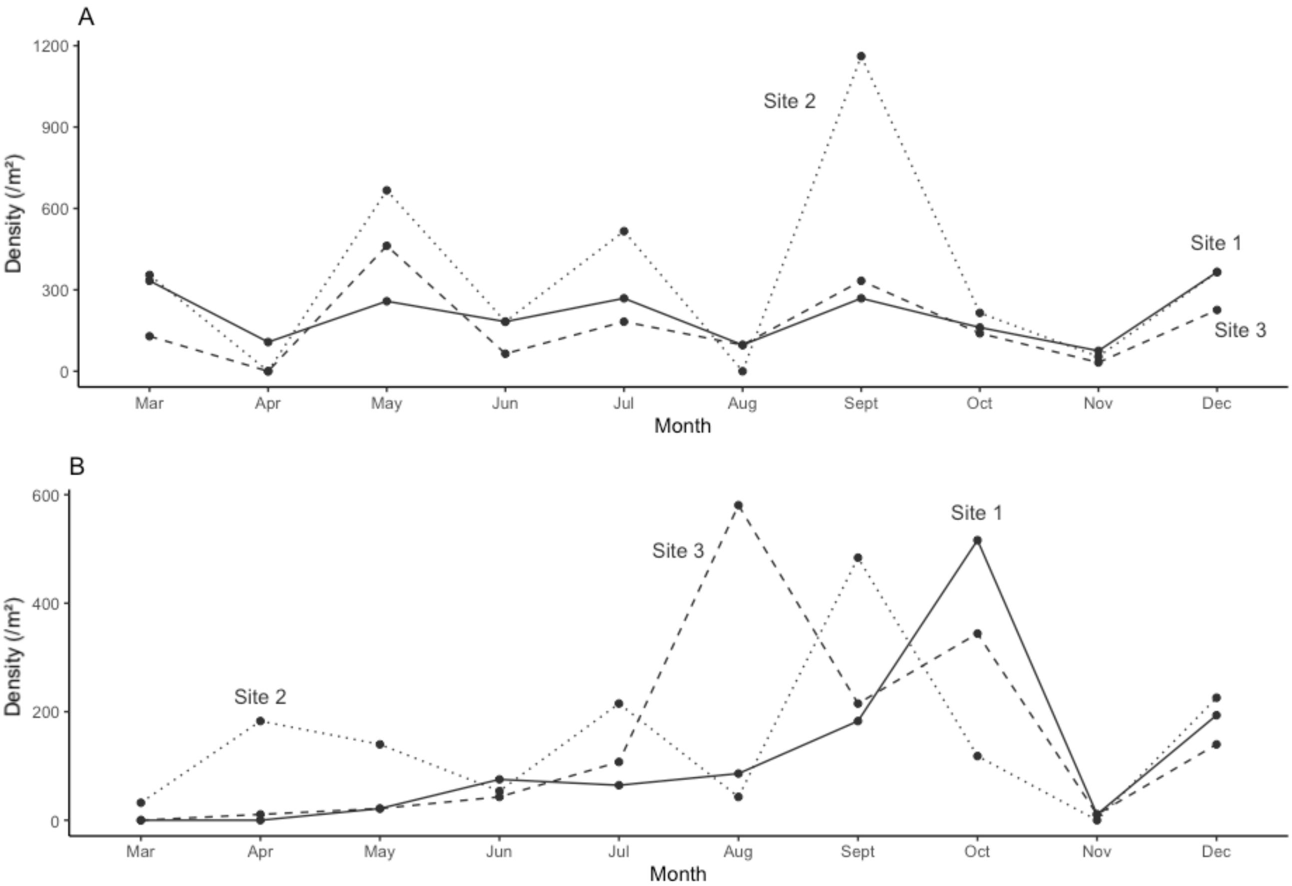

Figure 2. 4: Monthly densities (individuals/m^2) of *Caridina africana* at 3 sampled sites of Lake Sibaya: A – March 2021 to February 2022, and B- January 1975 to December 1975 (Hart, 1981)

Significant differences were seen in shrimp densities and lake level between 1975 and 2021 (Figure 2.5A/C and Table 2.1). Average shrimp densities declined by an average of 75% when comparing results from 1975 and 2021. Lake level has declined by an average of 9m (46%) when comparing 1975 and 2021. There was no significant difference seen in water temperature over time (Figure 2.5B). The 2021 temperature also maintained a similar seasonal pattern to that of 1975.

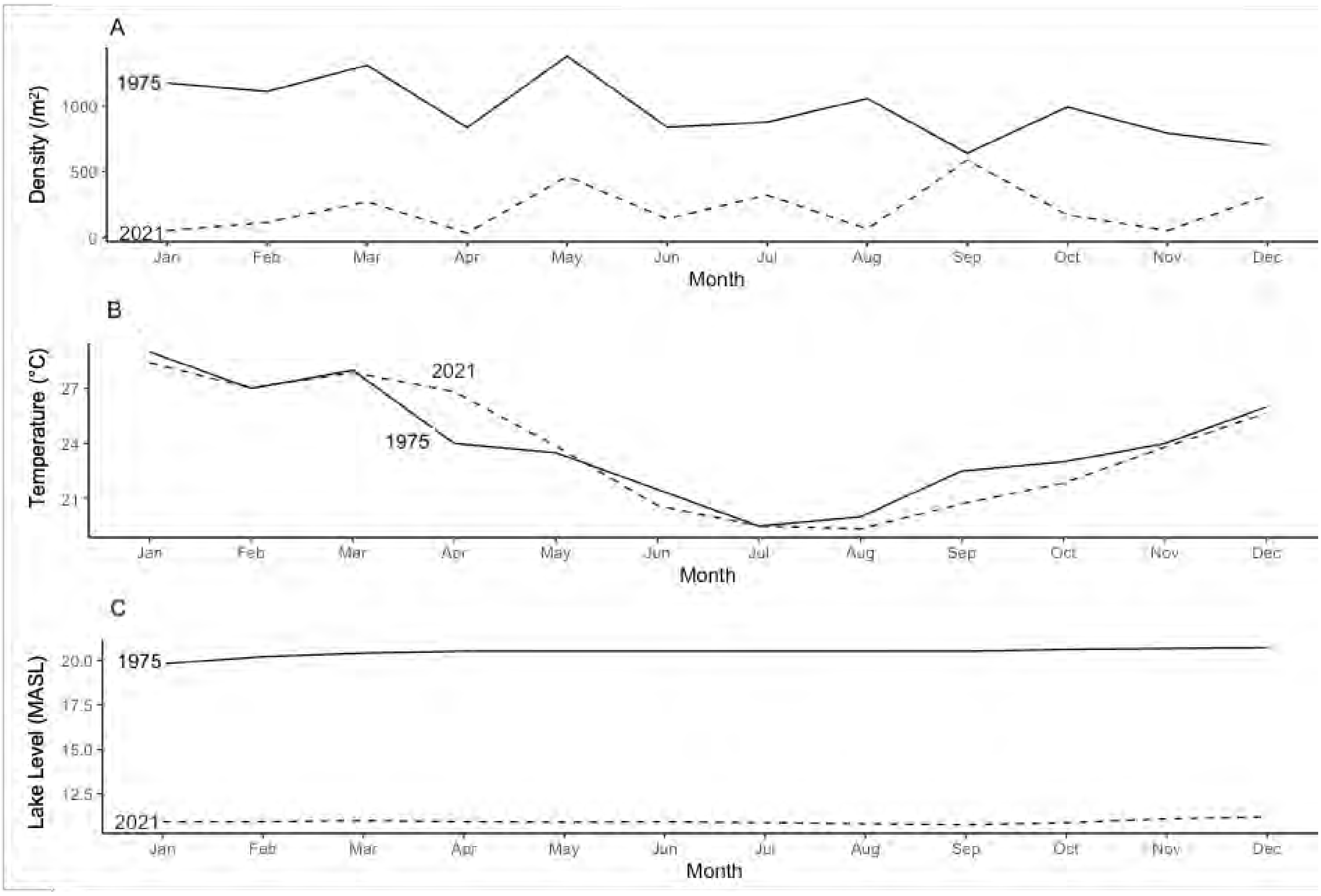

Figure 2. 5: Showing comparisons of A-Shrimp density (individuals/m^2), B- Water temperature (°C) and C - Average lake level (MASL) between 1975 (solid line) and 2021 (dotted line) at Lake Sibaya

Table 2. 1: Showing Wilcoxon Rank Sum test results conducted on density, temperature and lake level data collected in 2021 and historical data collected (Hart, 1981)

Factors (comparison)	W	P-Value
2021 Density vs 1975 Density	779	p<0.001
2021 Temperature vs 1975 Temperature	480	p>0.05
2021 Lake level vs 1975 Lake level	900	p<0.001

2.5.2. *Caridina shrimp population dynamics at Lake Sibaya: Current and historic trends*

Lake Sibaya's *Caridina africana* population showed no distinct seasonality at any site throughout the study period. Shrimp between the sizes of 3mm and 5mm CL (Carapace Length) were most abundant in all 12 months (Figure 2.6). Male shrimp dominated the smaller size classes (1.5 - 4mm CL while female shrimp dominated the larger size classes (4 – 7.5mm CL), and overall males were more abundant than females (Figure 2.6). No monthly or seasonal patterns are evident in the size-frequency distribution of *Caridina africana*, with winter months (May, June and July) producing similar values to the summer and spring months sampled.

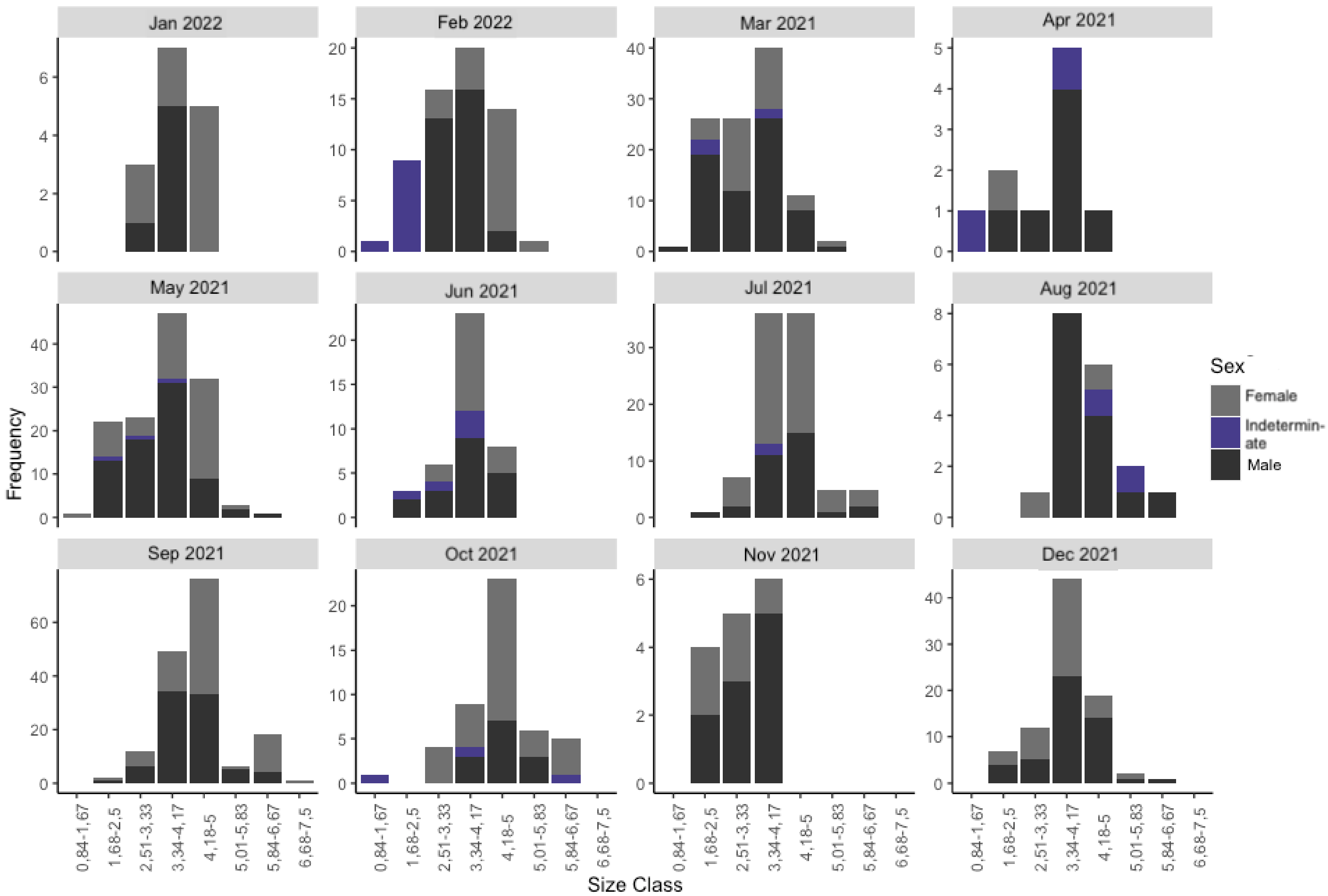

Figure 2.6: Showing monthly population abundances of different sexes and size classes of *Caridina africana* at Lake Sibaya from March 2021 to February 2022. Values are totals from all three sampled sites

Similarly, during the 1957 study, Lake Sibaya's *Caridina africana* population showed no monthly or seasonal patterns in terms of size-frequency distribution (Figure 2.6). Shrimp between the sizes of less than 0.83mm - 1.67mm CL (Carapace Length) were most abundant throughout the 12 months (Figure 2.7), unlike the 2021 period where the medium-size classes were most abundant (Figure 2.6). Male shrimp dominated the smaller size classes (1.5 – 4mm CL) whereas female shrimp dominated all size classes above 2.5mm CL (Figure 2.7). This is similar to 2021 where males dominated the smaller size classes and females dominated larger size classes but in 2021 females only became dominant within size classes larger than 4mm (Figure 2.6), whereas they often dominated size classes from 2.5mm in 1975 (Figure 2.7). Females generally grew larger than males as was evident in 2021 (Figure 2.6) and seemed to be more abundant, unlike 2021 where males were found to be most abundant (Figure 2.6).

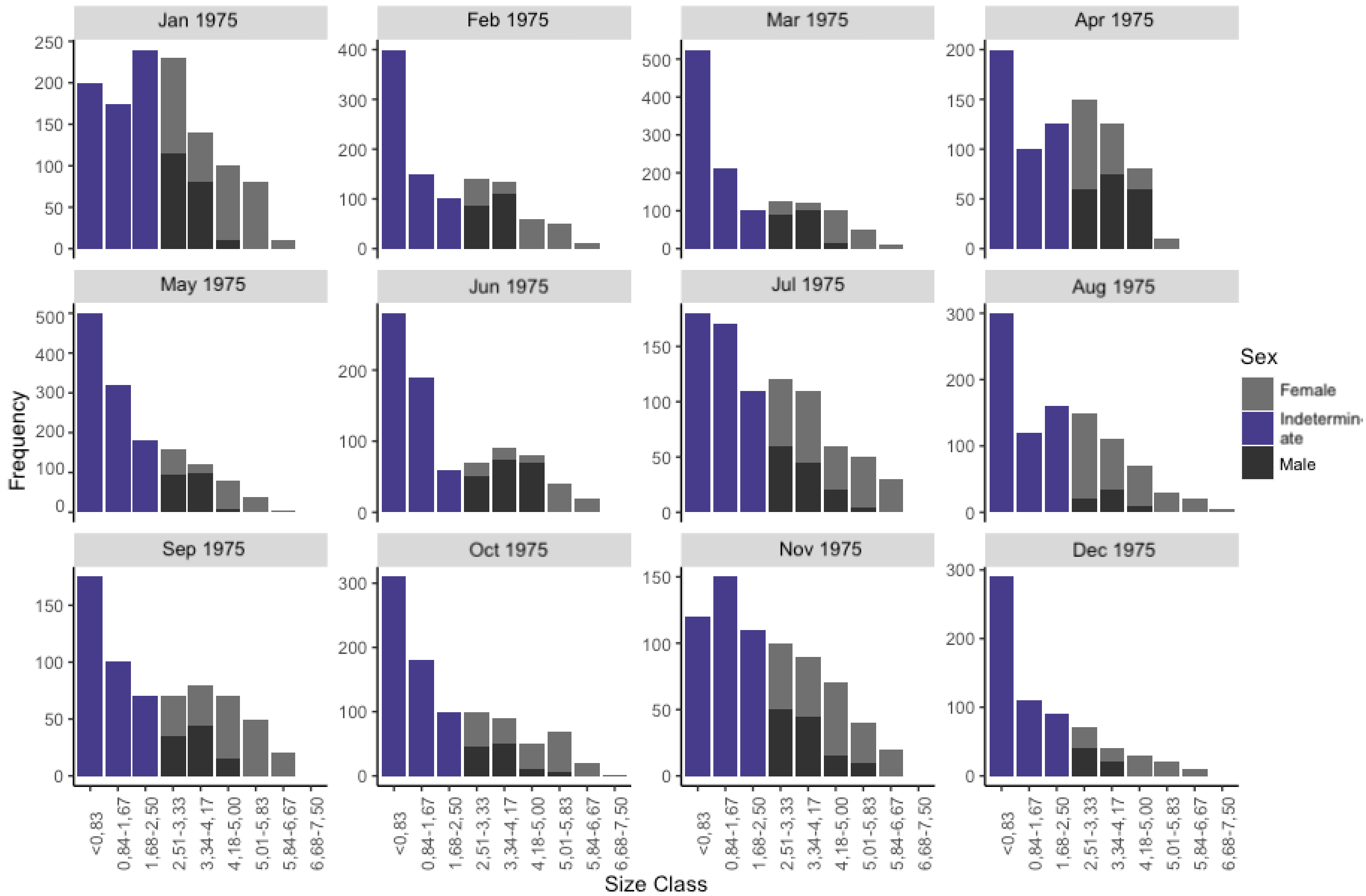

Figure 2. 7: Showing monthly population abundances of different sexes and size classes of *Caridina africana* at Lake Sibaya from January 1975 to December 1975 (Hart, 1981). Values are totals from all three sampled sites

The 3.33mm-4.17mm and 4.17mm-5mm size classes were the most abundant at Lake Sibaya in 2021 at all three sites (Figure 2.8A). Hart (1981) found the smaller size classes (<0.83mm CL) to dominate percentage contributions in the 1975 sampling period with similar patterns observed at all three sites sampled in 1975 (Figure 2.8B).

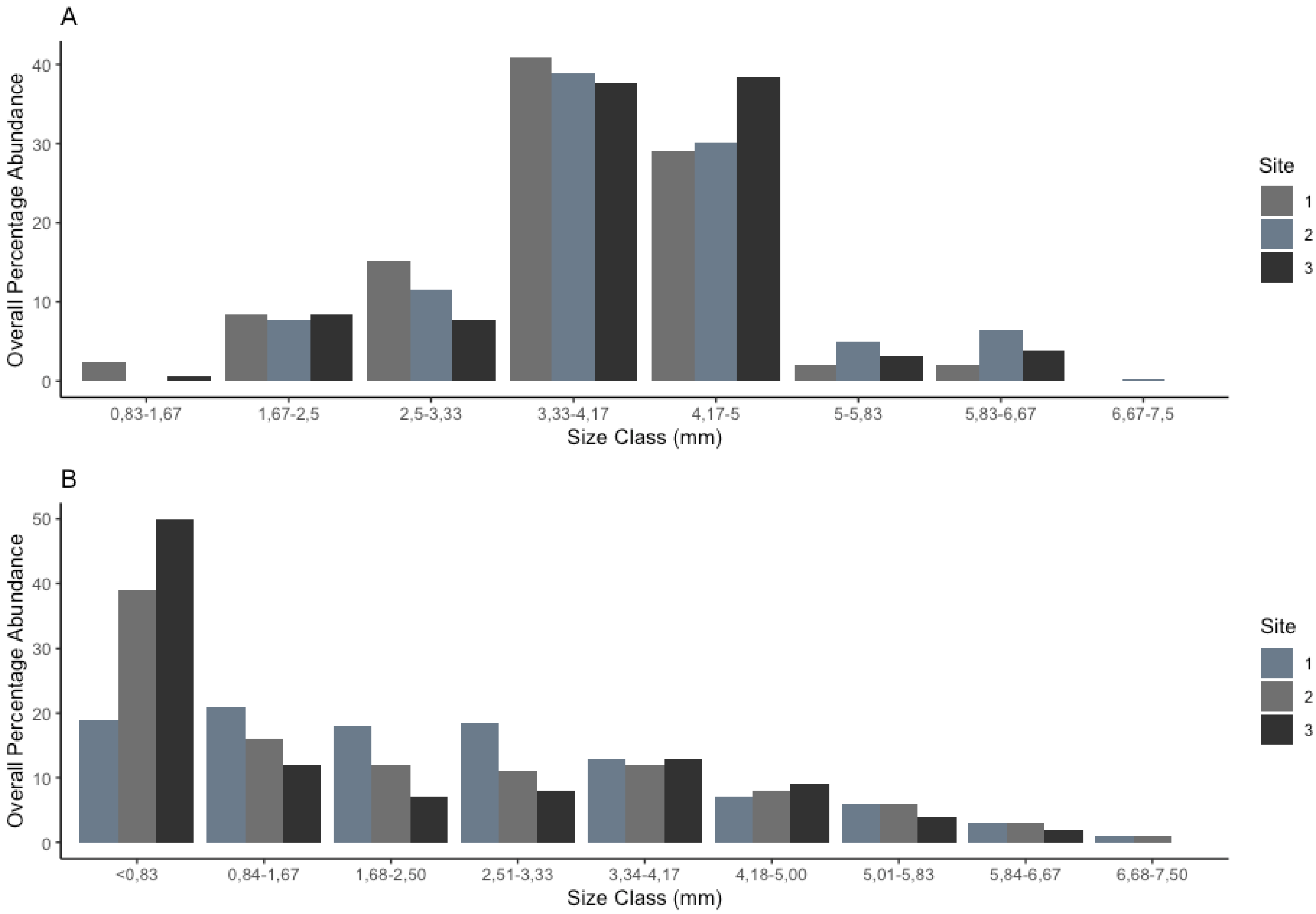

Figure 2. 8: Percentage abundances of *Caridina africana* size classes observed at 3 sampled sites in Lake Sibaya from (A) – March 2021 to February 2022 and (B) - January 1975 to December 1975 (Hart, 1981)

Indeterminate specimens form the bulk of the lowest size class in 2021 with males contributing largely to the smaller size classes and females contributing mostly to the larger size classes in 2021 (Figure 2.9A). Hart's (1981) results also found females to grow larger than males in the 1975 sampling period (Figure 2.9B) but found males to make up a smaller contribution to the smaller and medium-sized classes than was found in 2021. Females were also found to be more abundant overall during the 1975 sampling period whereas males were found to be more abundant overall in 2021.

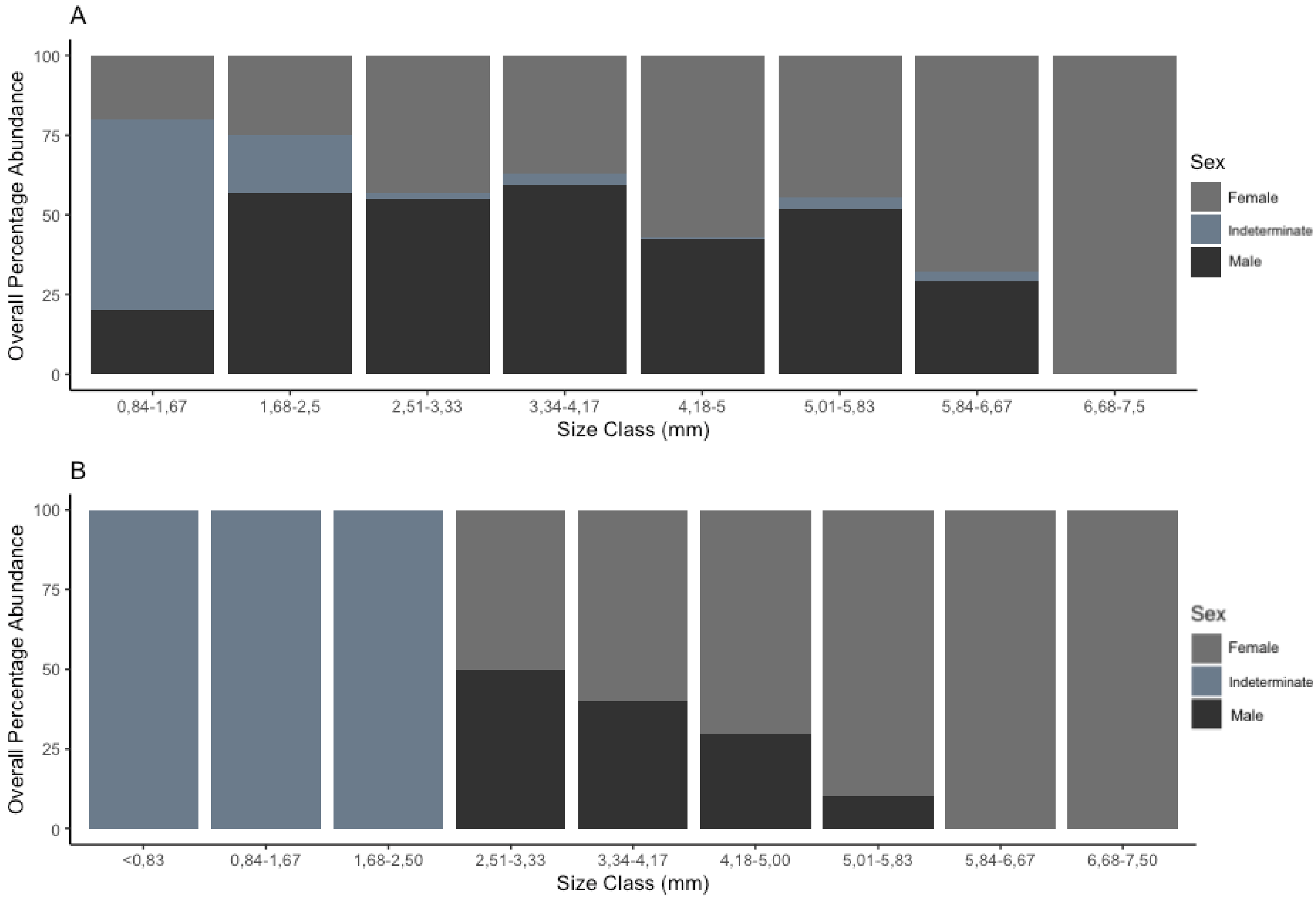

Figure 2. 9: Percentage abundances of female, indeterminate and male *Caridina africana* shrimp within the observed size classes at Lake Sibaya during the (A) – March 2021 to February 2022 and (B) - January 1975 to December 1975 (Hart, 1981)

Berried females (egg-bearing females) were found every month except in June 2021 with the highest density of berried females being in May and September 2021 (Figure 2.10A). The number of berried females found at Lake Sibaya in 2021 was slightly lower than those found in 1975, excluding May, July and September (Figure 2.10B). Berried females were also found every month in 1975, unlike in 2021 (Figure 2.10B). General peaks seem to occur in relatively similar periods with 2021 and 1975 both experiencing slight peaks in abundance in July, September and October (Figure 2.10 A & B).

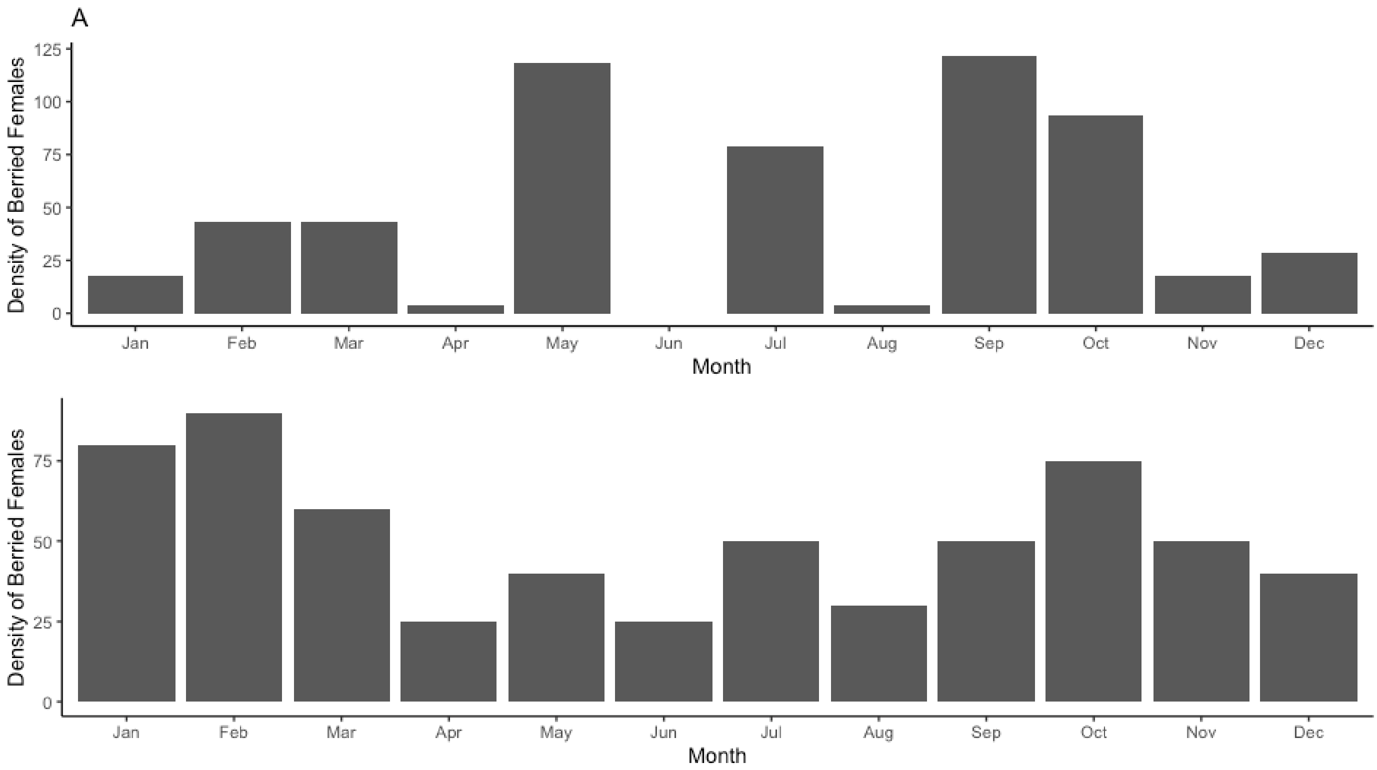

Figure 2. 10: Showing monthly densities (individuals /m^2) of berried females (*Caridina africana*) at Lake Sibaya during (A) – March 2021 to February 2022 sampling period and (B) January 1975 to December 1975 (Hart, 1981)

Instantaneous birth rates for *Caridina africana* populations observed in 2021 were relatively stable with small fluctuations from month to month. Large rates of population change were observed between months which were reflected by large fluctuations in the population's mortality rates (Figure 2.11A), indicating shrimp populations in 2021 were structured more according to mortality than birth rate. Whereas in 1975, instantaneous birth and death rates were similar for shrimp populations with a rate of population change fluctuating along a small gradient and instantaneous birth/death rates being very similar throughout the study period (Figure 2.11B).

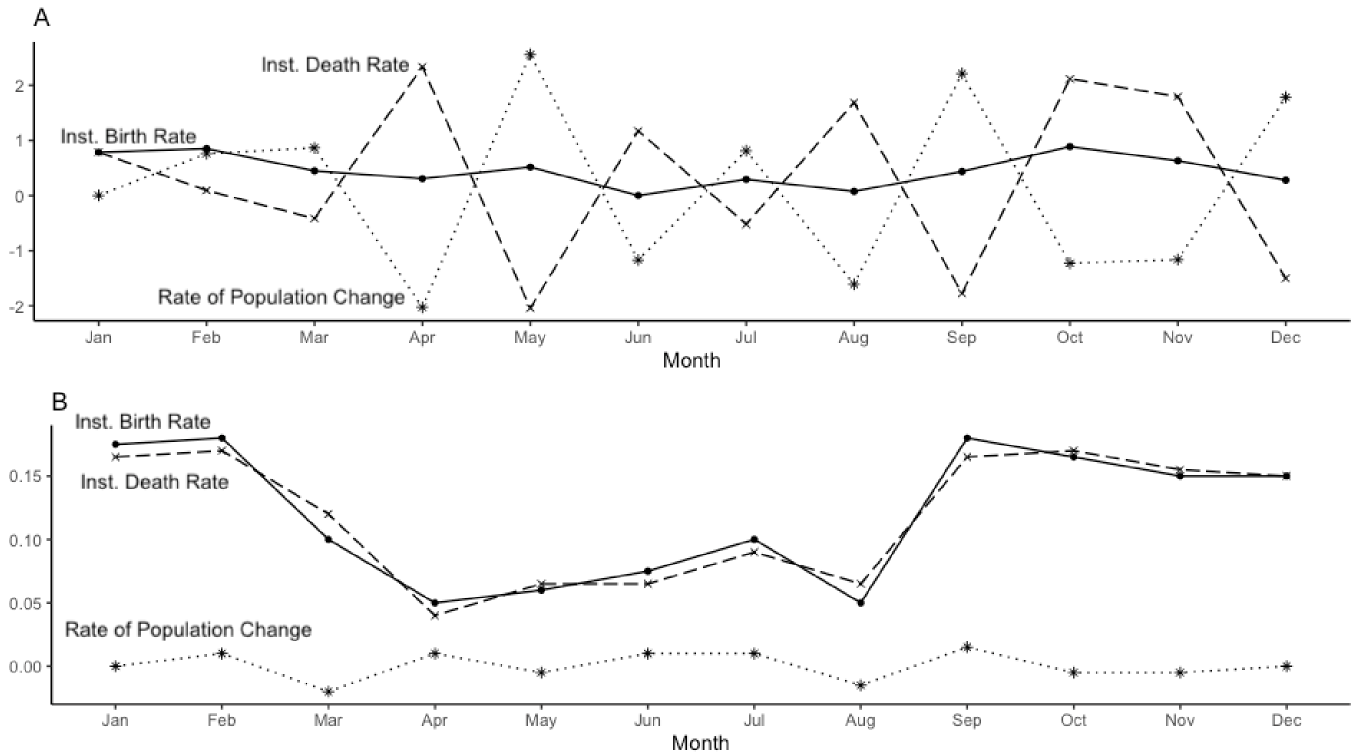

Figure 2. 11: Showing instantaneous birth and death rates as well as the rate of population change of *Caridina africana* populations observed in 2021 (A) and 1975 (B). Calculations done following the methods of Cummins et al. (1969) as done by Hart (1981)

2.5.3. Physicochemical variations at Lake Sibaya and Lake Mzingazi

pH at Lake Sibaya was high at the beginning of the year at 10.5 in January and dropped and became stable with a range of between 9 and 9.5 for the rest of the year (Figure 2.12A). Salinity experienced fluctuations throughout the year starting at around 350 ppm and fluctuating between 300 and 400 ppm until August when it dropped to 200 ppm and then fluctuated between 100 ppm and 300 ppm for the rest of the year (Figure 2.12B). Ammonium concentration fluctuated between 0.4 and 0.6 mg/l throughout the year with a large decrease in August to 0.2 mg/l, followed by an increase in September again (Figure 2.12C). Nitrate concentration generally remained stable between 0.8 and 1 mg/l throughout the year but there was an increase in April followed by a decrease in May again and a large decrease in August and an increase in September (Figure 2.12D). Phosphate concentration remained stable in the first few months of the year at around 0.1 mg/l until May, when it increased, followed by a sharp drop in June, a gradual increase in July returning to 0.1 mg/l in August, where it remained fairly stable until December (excluding another sharp decrease in October) (Figure 2.12E). Water temperature at Lake Sibaya experienced standard seasonal fluctuations with higher water temperatures in the spring/summer months (i.e. March, April, October and December) and lower water temperatures in the winter months (i.e. May, June, July and August) (Figure 2.12F). Conductivity generally remained between 800 and 850 μS/m with a drop to 700 and 750 μS/m in both March and September (Figure 2.12G). Lake level remained stable in the early summer months, experiencing a slow decrease from around 11 metres above sea level (MASL) to 10.8 MASL over the winter/non-rainfall months and a large increase during the later summer months again (Figure 2.12H).

Comparatively in Lake Mzingazi, pH was 8 in March and then declined in both April and May to 7 followed by a steady increase until August when it was almost 9, with a slight decrease in September but again returned to 9 in October until December (Figure 2.12A). Salinity was stable at 300 ppm from March until September. Then there was a sharp drop in October to close to 0 ppm but increased to 300 ppm over the last two sampling months (Figure 2.12B). Ammonium concentration was 0.4 mg/l in March and steadily increased to about 0.6 mg/l in July. There was a 0.2 mg/l decrease in August followed by an equivalent increase in September back to around 0.6 mg/l. Concentrations then decreased sharply again in October, rising slightly in both November and December to 0.4 mg/l (Figure 2.12C). Nitrate concentrations also remained relatively stable at the beginning of the year fluctuating between 0.8 mg/l and 0.9 mg/l from March until September. There was a decrease in October to 0.4 mg/l but this was back to 0.8 mg/l for the remainder of the sampling period (Figure 2.12D). Phosphate concentrations started at 0.2 mg/l in March and were the same until May. There was a steady decline from June until October where concentrations were around 0 mg/l, then a slight increase to 0.1 mg/l in November and December (Figure 2.12E). Water temperatures were around 30°C in March after which it slightly declined to 28°C in September and further decreased in October to 24°C and increased in November and December to 28°C (Figure 2.12F). Conductivity followed a similar pattern to water temperature, starting at its highest in March at around 600 µS/m and then declining over the course of the study period to around 500 µS/m in September. There was a drop in October to 400 µS/m, which remained the same for November and December (Figure 2.12G). Lake level started at 3 MASL in March and slightly increased to around 3.3 MASL which was observed in June, and a further 0.3 level decline until August where it was observed to be around 3 MASL. A further 0.2 increase was noted from September to December making a level of 3.2 MASL for the remainder of the study period (Figure 2.12H).

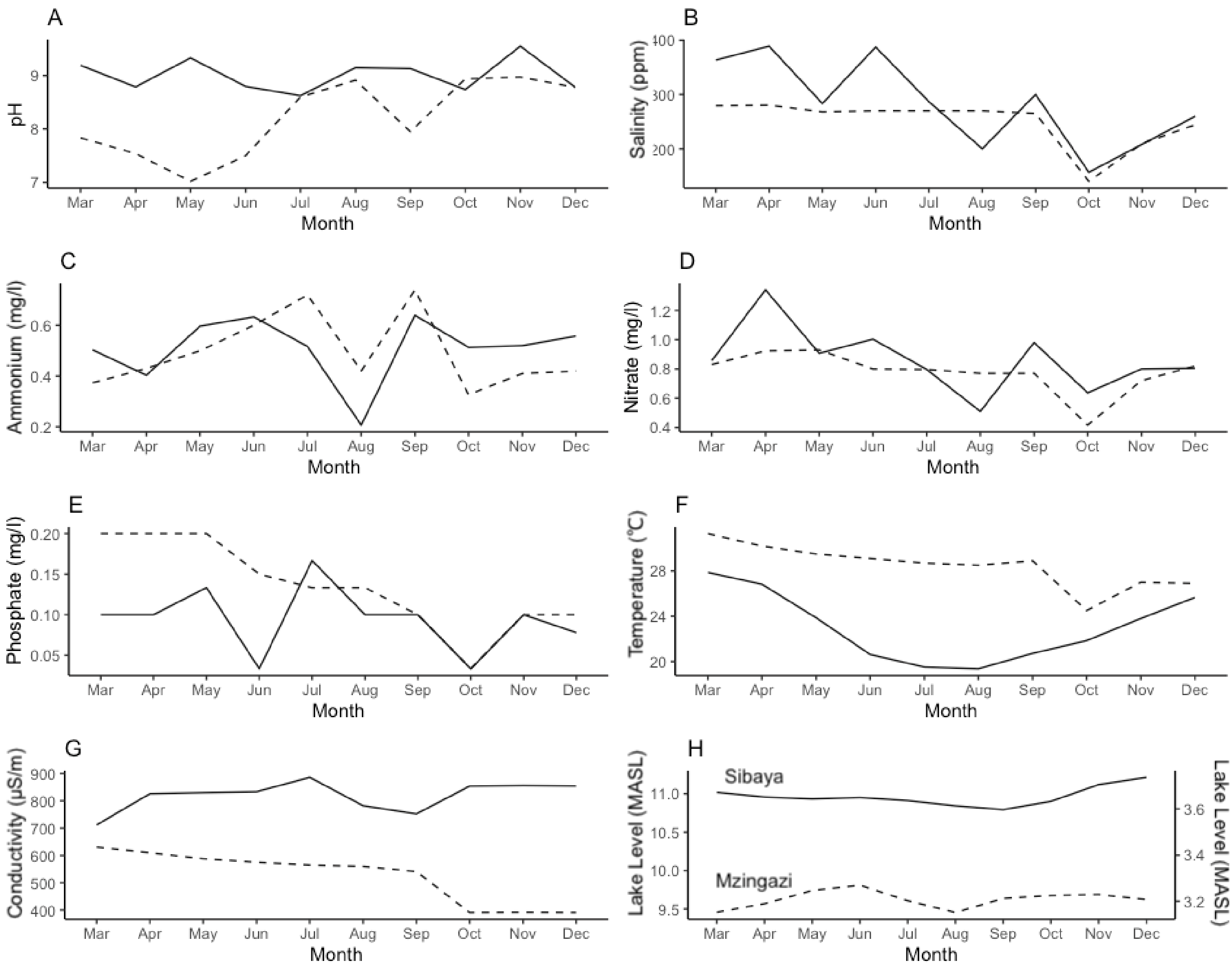

Figure 2. 12: Showing physicochemical variable measurements taken at Lake Sibaya (solid line) and Lake Mzingazi (dotted line) (A- pH, B- Salinity (ppm), C- Ammonium concentration (mg/l), D- Nitrate concentration (mg/l), E- Phosphate concentration (mg/l), F- Temperature (°C), G- Conductivity (µS/m), H-Lake Level (Metres Above Sea Level - MASL))

A principle components analysis (PCA) was conducted on the physicochemical variables from Lake Sibaya and Lake Mzingazi (Figure 2.13). The plot showed two distinct groups created by the two lakes but no clear grouping within the sampled sites was shown. The biplot shows most of the variation comes from the second component (y-axis) with both lakes being spread across component one (x-axis). Lake Sibaya sits in the positive portion of component two whereas Lake Mzingazi sits in the negative portion. Component loading scores of principle components and initial physicochemical variables showed pH (0.667), temperature (-0.788) and conductivity (0.794) to be the main contributors to component 2, therefore contributing to the difference between the two lake's physicochemical characteristics. Wilcoxon Rank Sum testing found significant differences in *Caridina africana* densities and temperatures at Lake Mzingazi and Lake Sibaya in 2021 (Table S4).

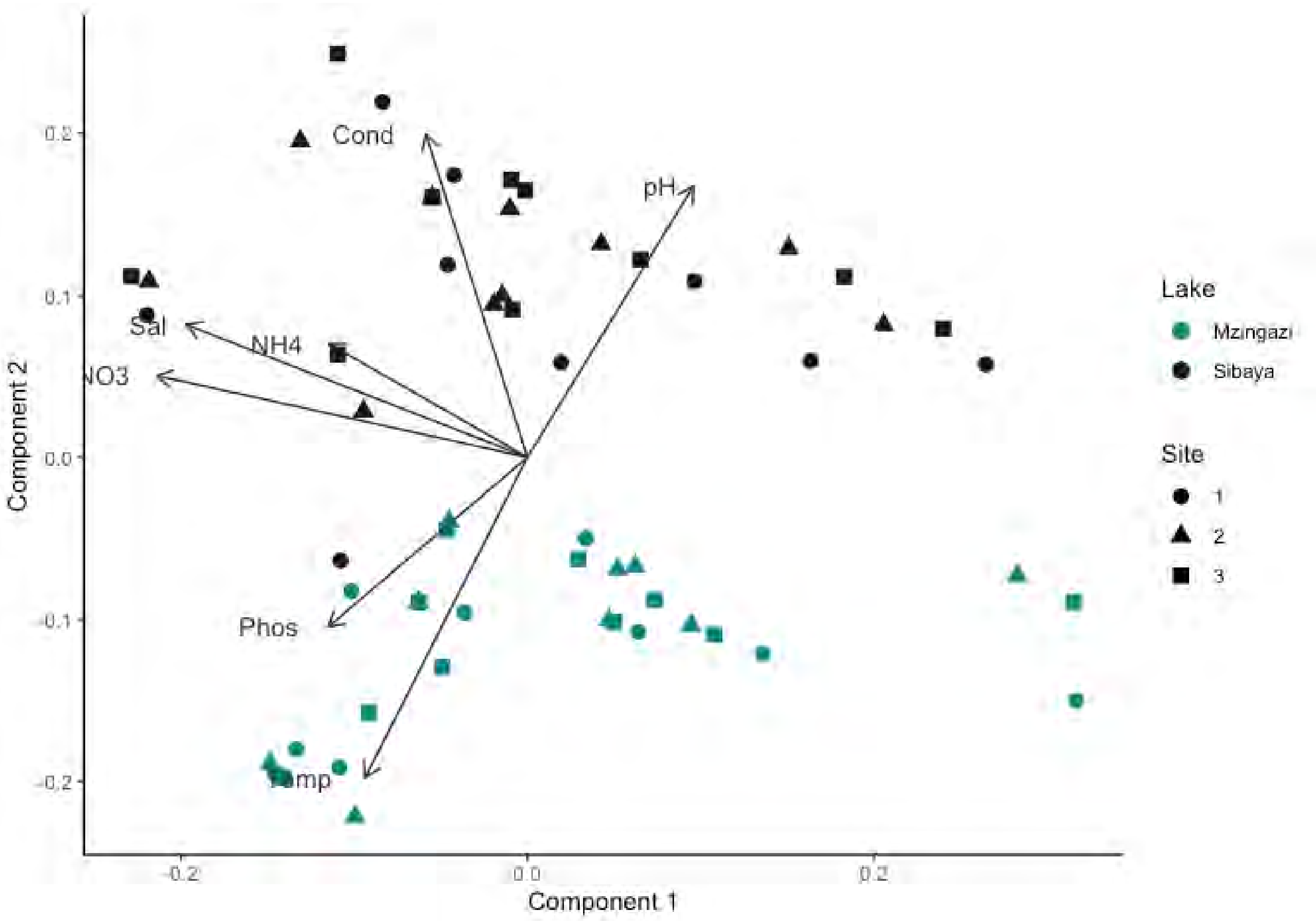

Figure 2.13: Biplot showing results from a Principle Components Analysis conducted on physicochemical variables from Lake Sibaya and Lake Mzingazi

2.5.4. Population dynamics at Lake Mzingazi vs Lake Sibaya

Caridina africana population in Lake Mzingazi showed no distinct seasonality, as was found at Lake Sibaya, but peaks in density were seen in late winter/early spring at Lake Mzingazi (Figure 2.14B) which is similar timing as the peak experienced by Lake Sibaya in September (Figure 2.14A). Both populations experienced fluctuations from month to month throughout the year but overall Lake Mzingazi's population experienced low densities at the beginning of the year, which slowly increased over time towards the end of the study period and Lake Sibaya experienced month to month fluctuation following a fairly uniform pattern.

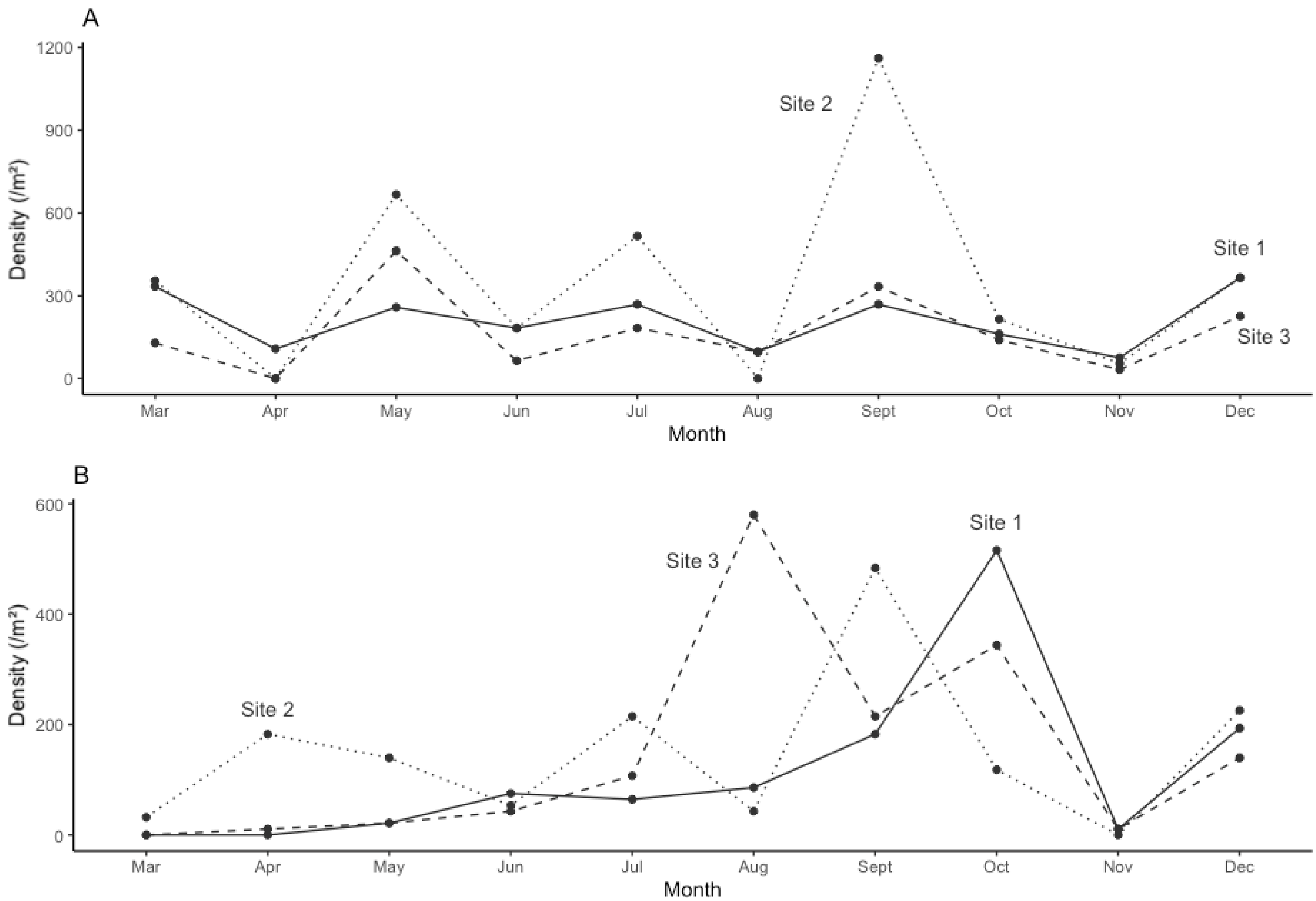

Figure 2. 14: Monthly densities (individuals/m^2) of *Caridina africana* at 3 sampled sites of Lake Sibaya (A) and Lake Mzingazi (B) from March 2021 to December 2021

The 3.33 – 4.17mm CL size class was the most abundant size class at Lake Mzingazi for most of the study period excluding August where the 1.67 – 2.5mm CL size class was most abundant and September where the 4.17 – 5mm CL size class was most abundant (Figures 2.15). Males dominated all size classes between 1.5mm CL and 5mm CL, whereas females dominated the 5 - 7mm CL size classes (Figure 2.15). The 3.33 – 4.17mm CL size class was the most abundant at all 3 sites in both lakes (Figure 2.16). Similar size-class frequency patterns are evident at Lake Sibaya and Lake Mzingazi with the medium-size classes being the most common, males being more abundant than females and no monthly or seasonal patterns evident in size-frequency distribution (Figures 2.15 and 2.16).

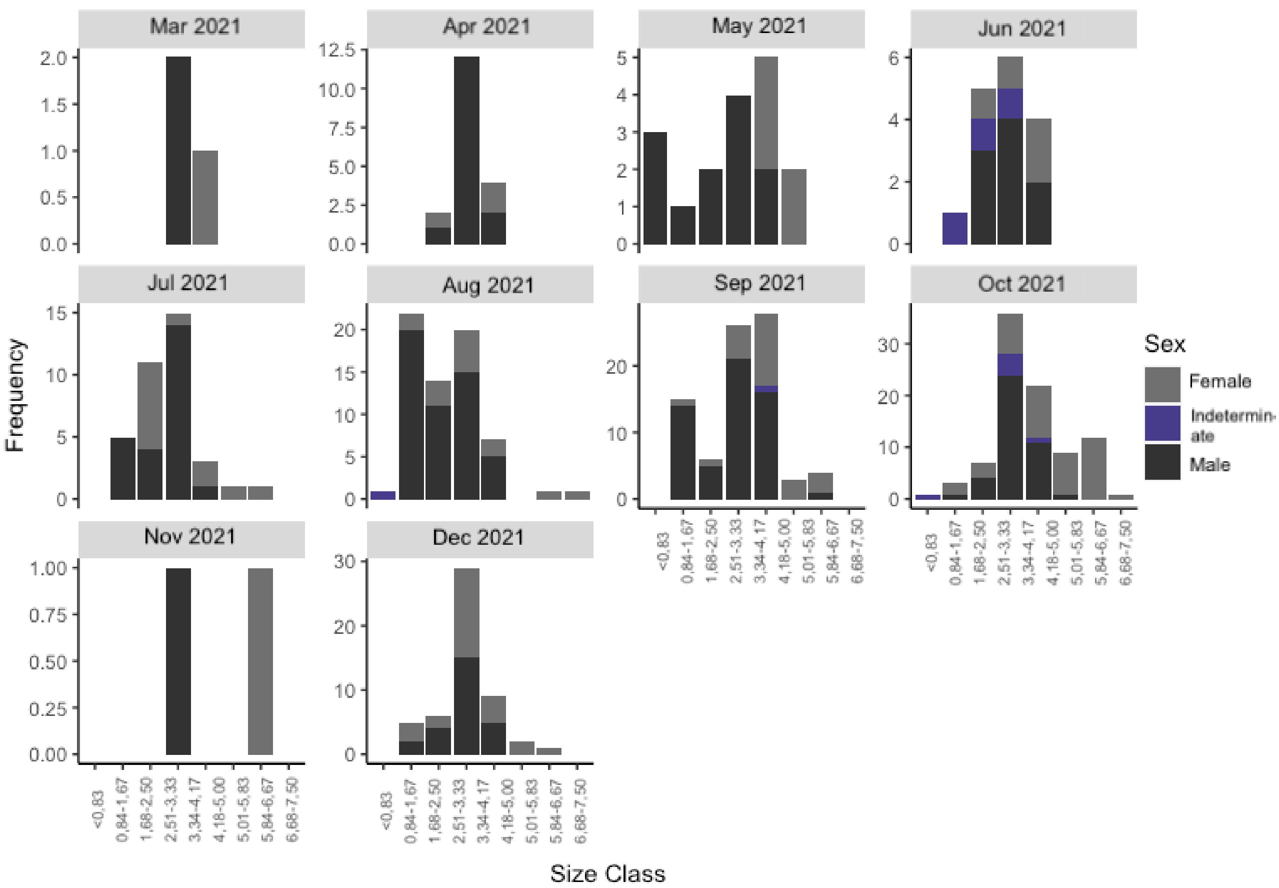

Figure 2. 15: Showing monthly population abundances of different sexes and size classes of *Caridina africana* at Lake Mzingazi from March 2021 to December 2021. Values are totals from all three sampled sites

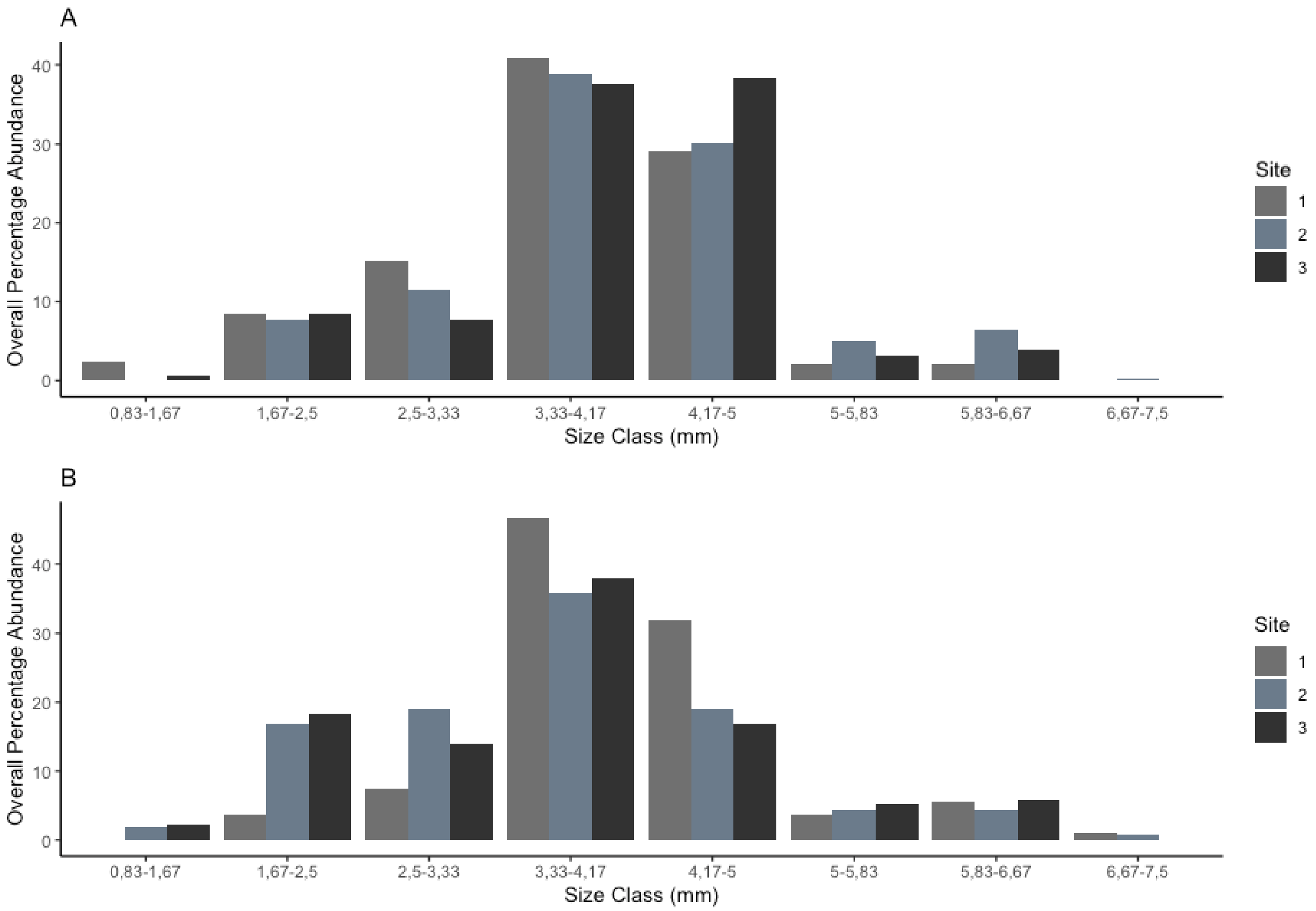

Figure 2. 16: Percentage abundances of *Caridina africana* size classes observed at 3 sampled sites of Lake Sibaya (A) and Lake Mzingazi (B) from March 2021 to December 2021

Males dominated all of the smaller size classes between 1.67mm CL and 4.17mm CL and females were larger, dominating the 4.17-7.5mm CL size classes at both lakes (Figure 2.17). Berried females (egg-bearing females) were found at Lake Mzingazi every month except March, June and November with the highest abundance found in October (Figure 2.18B). Comparatively, berried females were found every month except in June at Lake Sibaya with peak abundances in May and September (Figure 2.18A).

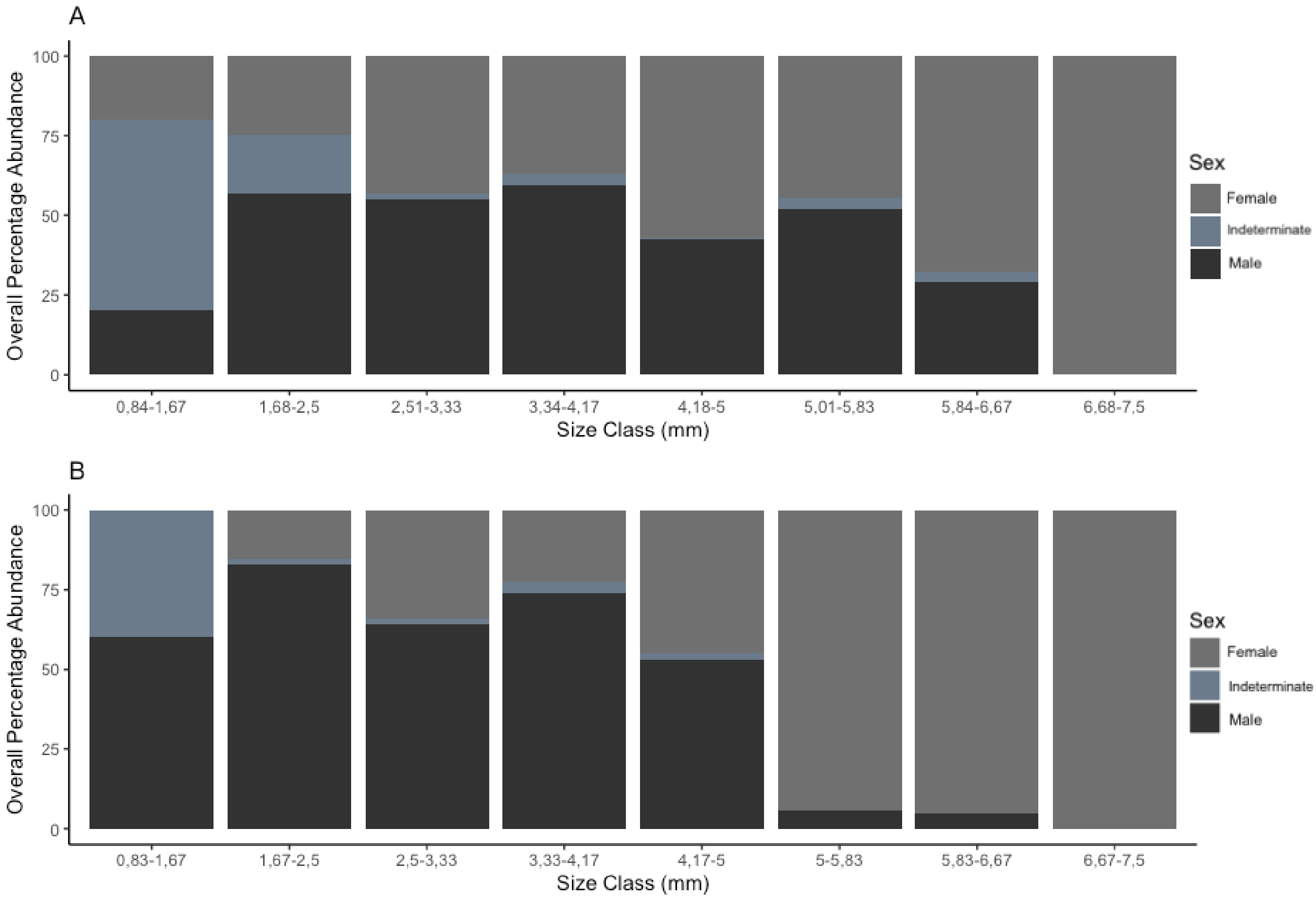

Figure 2. 17: Percentage abundances of female, indeterminate and male *Caridina africana* shrimp within the observed size classes at Lake Sibaya (A) and Lake Mzingazi (B) from March 2021 to December 2021

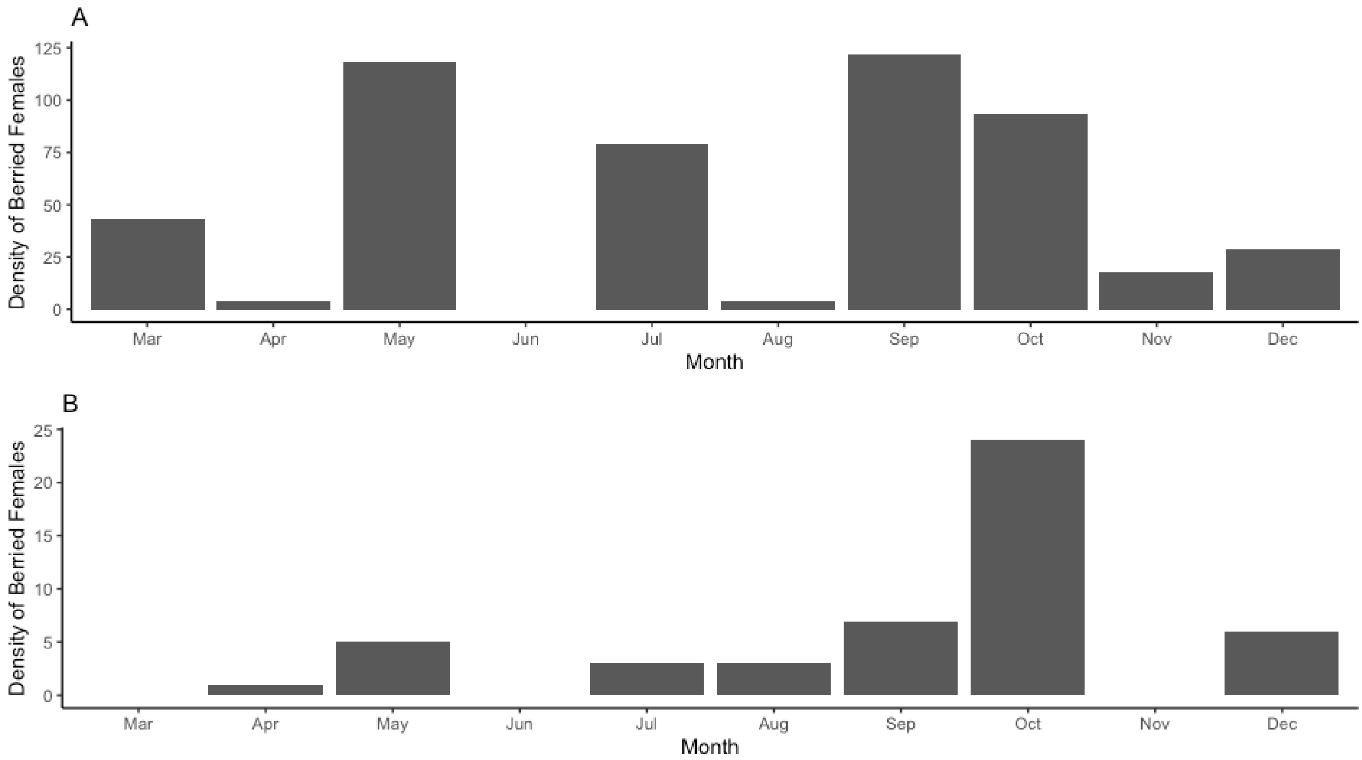

Figure 2. 18: Showing monthly densities (individuals/m^2) of berried females (*Caridina africana*) at Lake Sibaya (A) and Lake Mzingazi (B) from March 2021 to December 2021

2.5.5. Physicochemical parameters as drivers of Caridina africana population dynamics

Kruskal-Wallis tests conducted on month, site and season showed densities to differ significantly between months but not between sites at both lakes. Seasons were found to be significantly different at only Lake Mzingazi (Table 2.2) but post hoc tests showed no significance for any variables when using adjusted p-values (Table S5).

Table 2. 2: Showing results of Kruskal-Wallis tests conducted on *Caridina africana* density with Month, Site and Season as factors at Lake Sibaya and Lake Mzingazi respectively

Lake	Factors	Chisq	DF	P-Value
Sibaya	Month	22.874	9	p<0,01
	Site	-1.7977	2	p>0.05
	Season	2.9915	3	p>0.05
Mzingazi	Month	20.024	9	p<0.05
	Site	0.87531	2	p>0.05
	Season	7.9389	3	p<0.05

Generalised Linear Model (GLM) results showed a significant correlation between *Caridina africana* density and water nutrients including ammonium, nitrate and phosphate concentrations at Lake Sibaya (Table 2.3). Lake Mzingazi GLM results showed *Caridina africana* density to be significantly correlated with ammonium concentration and water temperature (Table 2.3).

Table 2. 3: Showing significant results from the determined most parsimonious generalised linear model conducted on data from Lakes Sibaya and Mzingazi

Lake	Parameters	Chisq	Df	P-Value
Sibaya	NH_4^+	21.0307	1	p<0.001
	NO_3^-	6.0097	1	p<0.05
	PO_4^{3-}	6.2744	1	p<0.05
Mzingazi	NH_4^+	7.2915	1	p<0.01
	Temperature	4.4593	1	P<0.05

Ammonium and phosphate concentrations showed a positive correlation with *Caridina africana* density at Lake Sibaya (Figure 2.19A & 2.19C). *Caridina africana* density and nitrate concentrations at Lake Sibaya showed a negative correlation, where a decrease in *C. africana* density was accompanied by an increase in nitrate concentrations (Figure 2.19B). It is important to note that the majority of measured density values occur where phosphate concentrations sit close to 0 mg/l. As with Lake Sibaya, Lake Mzingazi experienced an increase in *C. africana* density as increases in ammonium concentration occurred (Figure 2.19D) and also experienced low densities when temperatures were high (Figure 2.19E).

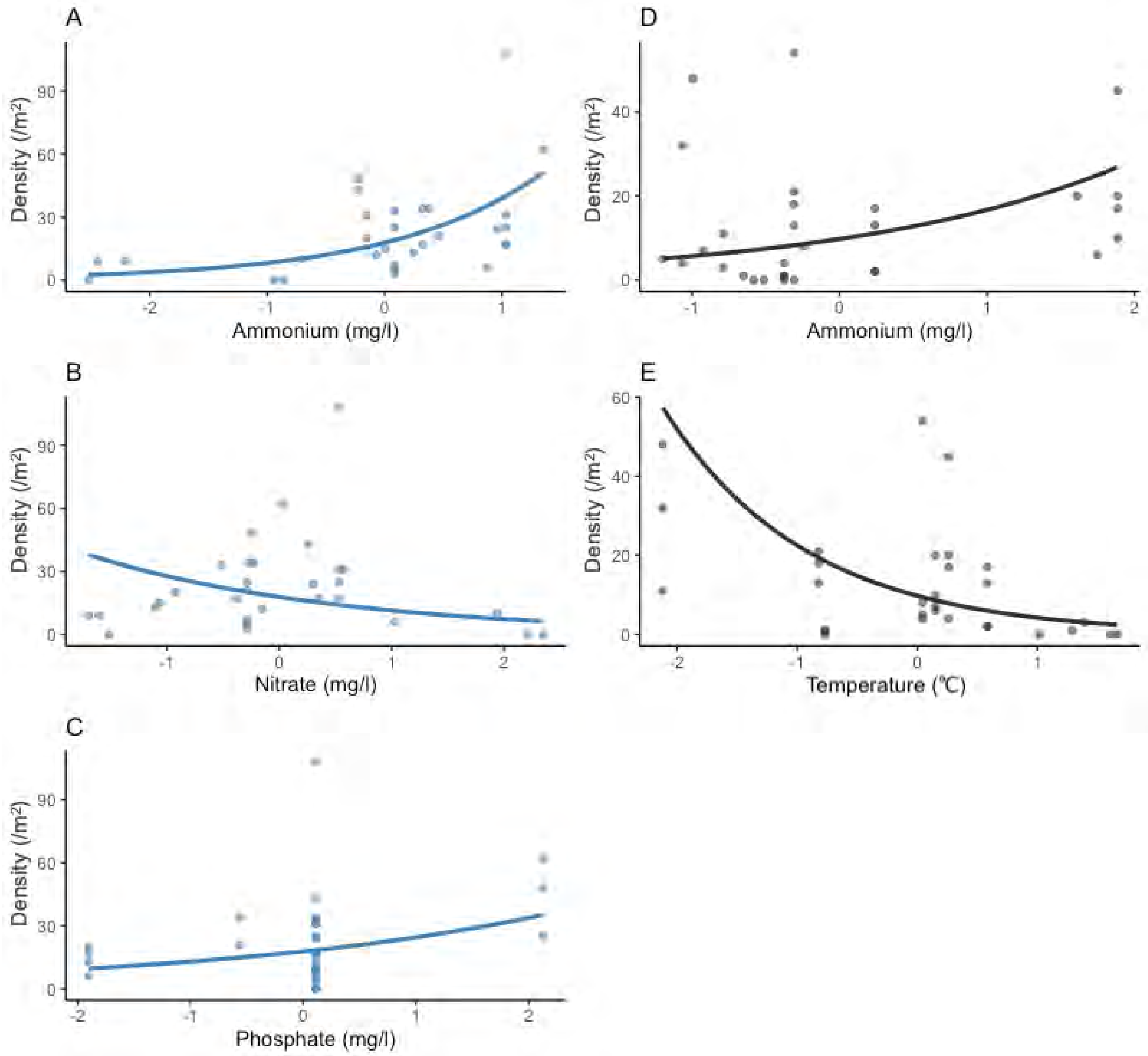

Figure 2. 19: Showing plotted effects of Z scores for environmental variables found to be contributing significantly to *Caridina africana* density at Lake Sibaya (Blue) (A- Ammonium concentration (mg/l), B- Nitrate concentration (mg/l), C- Phosphate concentration (mg/l)) and at Lake Mzingazi (Grey) (D- Ammonium concentration (mg/l), E- Temperature (°C)) according to Generalised Linear Models

2.6. Discussion

2.6.1. Lake Sibaya: 40 years on, physiochemistry

Historical water chemistry data (Allanson, 1979; Humphries and Benitez-Nelson, 2013) showed Lake Sibaya with slightly lower pH values (around 8,3) than what was found in the current study (between 8,5 and 9,5 on average). Small pH changes can cause variations in the concentration and toxicity of metal complex molecules (metal ions such as aluminium, copper, cadmium, lead, etc. surrounded by other ions) (DWAF, 1996). The effect of pH changes on aquatic communities is dependent on the degree, length and timing of the changes and should be interpreted accordingly (DWAF, 1996). In the case of the current study, pH values do not vary more than 0.5 units throughout the year, which is within South African water quality guidelines (DWAF, 1996) but the similarity of pH values reported by Allanson (1979) in 1967 and 1979, and again in 2015 (Bate et al., 2018), does point to a more recent increase in pH of Lake Sibaya, which should probably be monitored to ensure this increase does not continue.

Nitrate concentrations have increased from between 0.02 and 0.05 mg/l (Allanson, 1979) to between 0.5 and 1.3 mg/l in the current study. This concentration of inorganic nitrogen (between 0.5 and 2.5 mg/l) puts the lake in a mesotrophic state (DWAF, 1996). The approximate total inorganic nitrogen concentrations (excluding nitrite) found were similar to those found by Kock et al. (2019) indicating a move from the previously oligotrophic state of the lake to the current mesotrophic state. In comparison with the values found by Kock et al. (2019), the findings of this study show more recent stability in the inorganic nitrogen concentrations of the lake following the increase in nitrogen concentrations seen by Kock et al. (2019) in 2015/2016. According to the water quality guidelines set out by the Department of Water Affairs and Forestry (1996), mesotrophic conditions are normally associated with high biodiversity and production levels. These conditions do promote the growth of blue-green algae but these blooms are rarely toxic. The increasing concentrations do, however, point to the necessity of long-term monitoring in this regard to ensure the lake does not enter eutrophic conditions, which would promote harmful algal blooms (Paerl and Huisman, 2009; Visser et al., 2016; Reid et al., 2019).

The mesotrophic conditions may also explain the increase in the pH of the lake over time as these increases have been associated with increases in biological/algal activity (DWAF, 1996). Increases in agricultural runoff into Lake Sibaya would likely be shown in the phosphorous content of the lake (Reinecke and Reinecke, 2007; Horak et al., 2021). Allanson (1979) found less than 0.1 mg/l phosphate in 1969 and Bate et al. (2018) found orthophosphate as phosphorous concentrations to be 0.03 mg/l (convertible to 0.09 mg/l orthophosphate) when sampling in 2015. When this is compared with the 0.1-0.2 mg/l found in this study, there seems a fairly consistent concentration with slight increases and decreases from time to time. On the contrary, Kock et al. (2019) found much higher concentrations of orthophosphate (between 1 and 1.5 mg/l) at sites sampled in similar areas of the lake to this study. This result does seem to be an outlier when compared to Allanson (1979), Bate et al. (2018) and the current study. This is likely related to the rapid water level decrease experienced in 2015 and differences in sampled sites when compared to the sites sampled by Bate et al. (2018) in the same year.

Receding lake levels reduce the extent of littoral vegetation, ultimately reducing the habitat available to and likely increasing the competition between littoral invertebrates (Hunt and Jones, 1972; Gaeta et al., 2014). Increasing water abstraction by both anthropogenic and agricultural sources combined with drought, reducing aquifer recharge, has led to large reductions in the water level of Lake Sibaya (Carnie, 2020; Whitelaw and Van Rensburg, 2020). When comparing lake levels recorded in 1975 and 2021, a statistically significant difference in the level of Lake Sibaya can be seen. This indicates that monitoring lake levels and ideally reducing water abstraction rates should be made a priority to ensure adequate inundation of littoral vegetation and to reduce competition being experienced in these habitats. Compounded with the invasion of and high competition offered by *Tarebia granifera*, this notion becomes even more important in the conservation of Lake Sibaya and its biodiversity (further discussion in section 2.6.2 below).

2.6.2. Lake Sibaya: 40 years on; shrimp dynamics

It was hypothesised in Section 2.3 on Page 35 that *Caridina africana* densities would be lower and population dynamics would differ in 2021 when compared with 1975. In comparison with the current study, it can be said that general population dynamics are similar to those found by Hart (1981). Significant differences did exist when comparing current shrimp densities at Lake Sibaya when compared to the historical study. Generally, males formed a larger percentage of the smaller size classes and larger individuals were less abundant than smaller individuals as was found by Hart (1981). Contrary to Hart (1981)'s findings, males were more abundant than females but this could be related to the larger number of indeterminate specimens found in 1975 when compared to 2021. Hart (1981) found the most common size class in 1975 to be <0.83mm CL whereas in the current study, no individuals within this size class were found and the dominant size classes were 3.33 - 4.17mm CL and 4.18 – 5mm CL. There is a possibility that due to decreasing water levels and less habitat availability, fish stocks within the lake have decreased as seen by Gaeta et al. (2014). This would lead to lower levels of predation and could result in higher abundances of larger shrimp as the need to avoid predation decreases along with decreasing predation pressure. There is also a small possibility that the lake houses areas more suitable to juveniles and berried females but the fairly similar number of berried females found in the current study when compared to Hart (1981) does suggest this is not the case. It is more likely that individuals of this size were not seen and therefore not picked out of the sieve which highlights probably the largest limitation of this sampling method – hand sorting. This is further supported by the rate of population change which seemed to be driven more by mortality than lack of recruitment. A far more in-depth analysis of this high level of mortality and the potential drivers would be required for any further conclusions to be made on predation pressure and size-frequency distributions.

One of the larger threats to the ecosystem of Lake Sibaya is the invasive snail *Tarebia granifera*. This invasive snail species has been noted at Lake Sibaya by multiple previous studies (Appleton and Nadasan, 2002; Appleton et al, 2009; Miranda et al., 2011; Miranda and Perissinotto, 2014) as well as the current study (See also Chapter 3). Appleton et al. (2009) state that *T. granifera* is a threat to decapod shrimp. *Tarebia granifera* occupies the same dietary niche as *Caridina* shrimp mainly feeding on biofilm and detritus, and in high densities would create high competition in food-limited environments (Appleton et al., 2011; Moulton et al., 2012). Makherana et al. (2022) found strong reductions in the availability of benthic chlorophyll-a when snail populations were over 500 individuals/m^2 which was hypothesised to be due to grazing of *T. granifera*. Miranda and Perissinnotto (2014) found *T. granifera* to dominate community similarity indices at Lake Sibaya which makes it possible for the invasive snail to have created a food-limited environment. However, considering the mesotrophic state of the lake, ample nutrition to support both species should be available and it is therefore unlikely that *Tarebia granifera* alone is the reason for the reduction seen in shrimp densities since 1975.

Rather, it is more likely that the compounding of this higher dietary competition with the additional reductions in habitat availability due to receding lake levels as well as the long-term increase seen in nitrate concentration of the lake would contribute to lower shrimp densities. This can be concluded because the low phosphate concentrations indicate low agricultural runoff into the lake so it is unlikely this runoff is contributing to the reductions in shrimp density. This point is further confirmed by the positive association seen between phosphate concentrations and *Caridina africana* densities in the current study. The same is true for ammonium concentrations which were positively associated with *Caridina africana* densities and are unlikely to be contributing to these large reductions seen over time. Therefore, the maintenance of stable water levels within Lake Sibaya and the long-term monitoring of their response to increasing nitrate concentrations and the invasive population of *T. granifera* is crucial for the survival and productivity of the *Caridina africana* population living within the lake. This conclusion once again reiterates the importance of sustainable water abstraction from Lake Sibaya to minimise habitat loss, decrease competition and increase dilution capacity to mitigate nutrient enrichment (further discussed in section 2.6.4).

2.6.3. Lake Sibaya vs Lake Mzingazi

It was hypothesised in Section 2.3. on Page 35 that *Caridina africana* population densities would be lower at Lake Mzingazi than at Lake Sibaya and that these differences would be due to differences in land use and the related differences in physicochemical parameters. *Caridina africana* populations of Lake Sibaya and Lake Mzingazi experienced different annual density patterns over the study period and shrimp densities were significantly higher at Lake Sibaya than at Lake Mzingazi.

The annual patterns for the water chemistry parameters are generally similar but there is separated grouping of the lakes, which was related to differences in pH, conductivity and water temperature. The higher temperature at Lake Mzingazi is to be expected considering the current warming conditions being experienced globally (Dudgeon et al., 2006; Dallas and Rivers-Moore, 2014; Reid et al., 2019). Smaller, shallow water bodies have less of an ability to maintain constant temperature despite changes in air temperature. A general increase in the density of shrimp at Lake Mzingazi over the year is met with decreasing temperatures. At temperatures from 29.5°C to 31.5°C (March, April and May), there were very low densities of *Caridina africana* but when this temperature drops to 28.7°C in June, shrimp populations begin to increase and continue to do so under the cooler conditions experienced throughout the rest of the year. *Caridina nilotica* (now *C. africana;* See Richards & Clark 2009) has been known to experience drops in survivorship at temperatures over 30°C (Hart, 1985). In conjunction with the significant GLM result, this information leads to the assumption that temperature was the largest contributing factor in structuring *Caridina africana* populations at Lake Mzingazi. The area of Lake Mzingazi (10,3 km^2) would afford it much less adaptive capacity than Lake Sibaya with an area between 60 and 70 km^2.

The higher pH values of Lake Sibaya are also to be expected with the proven existence of algal blooms (Humphries and Benitez-Nelson, 2013) and the decomposing organic matter around the lake. Both biological activity and decomposition of organic matter are known to affect pH values (DWAF, 1996). The higher conductivity at Lake Mzingazi is likely related to the higher temperature of the lake (temperature and conductivity are highly collinear). The above information leads to the conclusion that the differences experienced in *Caridina africana* population densities are more related to temperature differences as a result of physical lake characteristics (mainly size and depth) than to physicochemical parameters related to land-use characteristics.

It seems agriculture and urbanisation exhibit similar effects on nitrogen concentration in freshwater lake systems, with similar ammonium and nitrate concentrations present at both lakes throughout the study period. There was decomposing plant matter at all sites as well as peatlands at Lake Sibaya which would increase nitrate concentrations in the lake and could be responsible for the similar values experienced when compared to the urban land-use and increased waste water found around and within Lake Mzingazi. For a more accurate conclusion to be made about the effects of agricultural and urban stressors on freshwater lake systems, it would be better to measure nutrient concentrations of lakes that would not experience nutrient enrichment from natural sources such as peatlands or to assess lakes that exhibit similar levels of natural nutrient enrichment, which is not the case in this study. Comparing lakes with similar natural nutrient enrichment would allow easier comparison because nutrient enrichment would be attributed solely to urbanisation or agriculture instead of trying to estimate the levels of natural enrichment vs unnatural enrichment. Phosphate concentrations were higher at Lake Mzingazi from March to July and this difference could be attributed to the increased phosphorous output related to anthropogenic pollution (Horne and Goldman, 1994; Horak et al., 2021). Both lakes do, however, have similar observed values from July onwards. As previously mentioned, it seems the main differences experienced between the lakes were a result of temperature and the smaller size of Lake Mzingazi when compared to Lake Sibaya instead of differences in agriculture/urban stressors.

The higher temperatures at Lake Mzingazi do not only pose a risk to *Caridina africana* populations because of their thermal tolerances (see: Hart, 1985; who found survivorship to dwindle dramatically above temperatures of 30°C) but also because the invasive snail *Tarebia granifera* is present within the lake (Jones, 2014). *Tarebia granifera* is known to exploit conditions of high temperature as well as high pH, conductivity and rainfall (Miranda et al., 2010), which may pose a threat to shrimp populations (Appleton et al., 2011; Moulton et al., 2012; Makherana et al., 2022). *Caridina africana* is known to struggle under increased temperatures and conductivity (Hart, 1985; Siméon et al., 2014) and would therefore, be unable to outcompete or even just compete with *Tarebia granifera* for habitat or food under these conditions. This highlights the importance of monitoring shrimp populations within Lake Sibaya as high pH and conductivity values were recorded during this study but also the importance of monitoring shrimp populations within smaller lake systems, such as Lake Mzingazi. These systems may be more impacted by increasing global temperatures in a shorter space of time than others and in the case of Lake Mzingazi, this has resulted in exceeding the thermal tolerance limit of *Caridina africana* in March and April, drastically reducing their ability to compete with invasive species. The shrimp seem to recover over the course of the year but the importance of revisiting this study to assess if populations are recovering to numbers found before extreme heat events cannot be overlooked.

Previously, both *Macrobrachium* sp. and *Palaemon* sp. have been found at Lake Mzingazi (Mackay and Cyrus, 2001; Moloi, 2012; Weerts et al., 2014) but none were found in this study. Both are estuarine species and were found at the entrance to the fishway which is the connection between the lake and the saline Mzingazi river. The fishway was not sampled in this study and all sites sampled at Lake Mzingazi would be considered typical freshwater habitats, making the presence of *Palaemon* sp. unlikely. *Macrobrachium* sp. is known to rely on estuarine conditions in their larval stage and for post-larval metamorphosis but this is followed by migration into freshwater (Read et al., 1985). A study conducted using multiple sampling techniques found only *Caridina* shrimp when using a small seine net, only noting the presence of both *Palaemon* sp. and *Macrobrachium* sp. when using an electro-shocker (Weerts et al., 2014), which is highly likely to be the reason no specimens of *Macrobrachium* sp. were found in this study.

2.6.4. Physicochemical parameters as drivers of shrimp density

Ammonium, nitrate and phosphate are three of the more common water quality parameters found within aquatic systems and can be a result of human and animal effluent, agricultural fertilisers and organic/industrial waste (Smith et al., 1987; DWAF, 1996; USGS, 1999; Revenga et al., 2000; Ashton, 2010; Matthews et al., 2014; Ndlela et al., 2016; Visser et al., 2016). Significant correlations existed between shrimp density and ammonium, nitrate and phosphate concentrations at Lake Sibaya and ammonium concentration and temperature at Lake Mzingazi. The correlation with ammonium is positive at both lakes with high shrimp densities occurring alongside high ammonium concentrations. This is to be expected as ammonium is not recorded as being harmful to aquatic organisms (DWAF, 1996) and it is known to be a result of biological activity. Inorganic nitrogen poses a risk to aquatic invertebrate communities mainly due to the harmful effects associated with nitrogen ions, specifically ammonia (not measured in this study) and nitrate. Nitrate was found to be negatively correlated with shrimp density at Lake Sibaya which agrees with a previous study that found shrimp communities to be sensitive to increases in nitrate concentration (Siméon et al., 2014). Increase in inorganic nitrogen and phosphorous concentrations have been found to be associated with an increase in agricultural input (Smith et al., 1987; USGS, 1999; Revenga et al., 2000) which could explain the significance of nitrate and phosphate concentrations on shrimp densities at Lake Sibaya and not at Lake Mzingazi but this is unlikely considering the potential nitrate input associated with anthropogenic waste (DWAF, 1996; DWAF, 2004) and the positive correlation seen between phosphorous and shrimp densities at Lake Mzingazi.

It is unlikely that human effluent is contributing to inorganic nitrogen and phosphorous concentrations seen at Lake Sibaya as urbanisation in the direct lake area is minimal. However, the livestock that utilise the lake's surrounding for grazing, as well as the agricultural plantations around the lake, could be contributing to these nutrient concentrations. Hippos are also abundant at Lake Sibaya (Kock et al., 2019) and their dung is a key component in the transfer of terrestrial trophic resources into aquatic ecosystems (Dawson et al., 2016). The effects of this organic waste input depend on the ratio of input to utilisation with dung being classed as a potential threat to the productivity of aquatic benthic systems (Dawson et al., 2016). The significance of *Caridina africana* densities and nitrate concentrations at Lake Sibaya is to be of concern because most of the sources responsible for this nutrient enrichment are likely natural (hippos and peatland leaching) which indicates not that nutrient enrichment should (or could) be minimised necessarily but that the role of nutrient recyclers in the removal of nitrates should be monitored more closely. The possibility exists that nitrate concentrations are not contributing to the reductions in *Caridina africana* densities but rather that the reduction in *Caridina africana* (and other nutrient recyclers) is leading to a lack of removal of naturally occurring nitrate within the system but far more research would have to be conducted to form a solid conclusion on this point.

The *Caridina africana* populations at Lake Sibaya and Lake Mzingazi do not seem to be threatened by current ammonium or phosphate concentrations with correlations at both lakes being positive or non-significant and it can be assumed that the ratios of input: utilisation of inorganic phosphorous and ammonium at the recorded concentrations are adequate for the systems to function successfully. As previously mentioned, the maintenance of stable water levels within Lake Sibaya and the long-term monitoring *of Caridina africana*'s response to increasing nitrate concentrations and the invasive population of *Tarebia granifera* seem to be the most important conservation strategies for enhancing the survival and productivity of the *C. africana* populations living within the lake. This conclusion emphasizes the importance of sustainable water abstraction from Lake Sibaya to minimise habitat loss, decrease competition and increase dilution capacity to mitigate natural nutrient enrichment.

2.6.5. Conclusions

Over the last 40 years, drastic reductions in the lake level of Lake Sibaya have likely forced many littoral invertebrates into slightly deeper (and inaccessible to sampling) vegetation while increasing the competition experienced in these habitats. This combined with the invasion of *Tarebia granifera* and the competition they offer to the dietary niche of *Caridina africana* has likely led to the 75% reduction in *Caridina africana* population density within the lake over time. This study found differences in size-frequency and sex-frequency distributions with medium-sized *Caridina africana* specimens being most abundant in 2021 in comparison to the smallest size class which was most abundant in 1975 but this is likely related to the sampling method.

Caridina africana population densities were driven mostly by nitrate concentrations at Lake Sibaya in the current study. This combined with the historical lake level drop, 75% decrease in *Caridina africana* densities observed during the study and the move from an oligotrophic to mesotrophic system over time emphasises the importance of global change mitigation, specifically water abstraction in times of drought and nitrogen pollution, in the survival of *Caridina africana* populations within the lake. Lake Mzingazi's *Caridina africana* populations were mainly driven by water temperature in 2021 indicating the importance of monitoring smaller lake systems during global warming. There was no clear indication that urbanisation and agriculture present different environmental threats/stress to freshwater lakes with the temperature being the main differentiating factor between the two systems in this study.

Finally, endorheic systems rely largely on nutrient recyclers to prevent eutrophic conditions and the importance of monitoring the organisms that perform this ecological role should not be underestimated. Fluctuating water chemistry conditions (inorganic nitrogen and phosphate concentrations specifically) when compared with previous studies show variation in waste or water input or possibly (considering the decreases in shrimp density seen between the current and 1975 studies) in the productivity of nutrient-recyclers within Lake Sibaya. The exact cause of these fluctuating parameters should be investigated in more detail to assess the risks being posed to the lake.

Chapter 3

Biodiversity Survey of Littoral Aquatic Invertebrates of Two Coastal Lakes of Northern KwaZulu-Natal, South Africa

3.1. Introduction

3.1.1. Littoral fauna of South African freshwater systems

Aquatic invertebrates found most commonly in Southern African lakes are a variety of crustacea (Tanaidacea, Decapoda, Amphipoda, Isopoda, Cladocera and Copepoda) as well as Gastropoda, Diptera, Ephemeroptera, Trichoptera, Odonata, Hemiptera and Coleoptera (Allanson, 1990; Hart and Appleton, 1997; Mackay et al., 2010; Ferreira et al., 2012, Coetzee et al., 2014; Farrell et al., 2015; Burger et al., 2018; de Necker et al., 2021; Dalu et al., 2022). Baetidae, Corixidae, Coenogrionidae, Dytiscidae and Physidae were found to dominate aquatic invertebrate communities of the Nlyslvey Wetland when sampled by Dalu et al. (2022), Corixidae and Notonectidae were found to dominate aquatic invertebrate communities in endorheic depression wetlands in Gauteng when sampled by Burger et al. (2018) and Naucoridae, Lestidae, Nepidae, Notonectidae and Dixidae were found in the slow-moving sections of the Wilge River by Farrell et al. (2015).

Aquatic invertebrates offer vital ecosystem services to humans (Cardinale et al., 2000; Loreau, 2000). These are mainly through support of aquatic systems including nutrient cycling and primary production as well as the provisions these systems offer (Millennium Ecosystem Assessment, 2005). Additionally, these organisms support aquatic resources by providing important ecological information on the state and health of these systems by acting as bioindicators (Lenat, 1988; Ketse et al., 2006; Bredenhand and Samways, 2009; Siméon et al., 2014; Dalu et al., 2017; Jansen Van Rensburg et al., 2019). The assessment of aquatic invertebrates to determine the ecological state of river systems has been well-recognised around the world (Carlisle and Clements, 1999; Hodkinson and Jackson, 2005; Leigh et al., 2013; Firmiano et al., 2021; Kumar et al., 2023) and in South Africa (Dickens and Graham, 2002; Ketse et al., 2006; Bredenhand and Samways, 2009; Dalu et al., 2013; Dalu et al., 2017; Jansen Van Rensburg et al., 2019). A variety of bioassessment methods exist for this purpose. Two of the most common methods are the South African Scoring System (SASS – Currently version 5) (Dickens and Graham, 2002) and the Dragonfly Biotic Index (DBI) (Samways and Simaika, 2015). The fifth version of the South African Scoring System is a method based on scoring the presence of either pollution-sensitive and/or pollution-tolerant taxa found at a site of interest. Pollution-sensitive taxa have a higher SASS score (or quality value) close to 15, whereas pollution-tolerant taxa have a SASS score close to zero (Dickens and Graham, 2002). The DBI is a similar system scoring adult Odonata species (Samways and Simaika, 2015). As with SASS5, each known adult (usually male) Odonatan has an assigned DBI score based on the species distribution (common/localised or endemic), threat status and sensitivity to habitat modification (Samways and Simaika, 2015). Based on the number of taxa collected (i.e. aquatic macroinvertebrates and adult dragonflies), the SASS and DBI scores are computed by dividing the total scores by the number of taxa to get the site-specific water quality and habitat health scores.

The application of aquatic invertebrates for monitoring natural standing waterbodies (lakes, dams, pans and wetlands) is less obvious and still widely debated (White and Irvine, 2003; Poikane et al., 2016). Studies have shown success in determining ecological changes resulting from global change impacts using aquatic invertebrate communities of temporary and perennial pans in South Africa (Ferreira et al., 2012; Foster et al., 2015; de Necker et al., 2016; Dalu et al., 2021), but less is known of the aquatic invertebrates in lake systems (Skowno et al., 2019). Additionally, aquatic invertebrates have further been considered to be sensitive to changes or disturbances such as acidification, eutrophication, hydro-morphological changes and the combination of all three stressors in European lakes (Poikane et al., 2016). However, Bird et al. (2013) showed that regional aquatic invertebrate diversity indices were not successful in indicating human disturbance in the South-Western Cape, Mediterranean region of South Africa. Authors further state this could be likely due to the inherent spatial and temporal variation present within aquatic invertebrate communities of temporary pans (Bird et al., 2013). The permanent nature of Lakes Sibaya and Mzingazi is more than likely to reduce the inherent temporal variation typically experienced by endorheic pan communities. In summary, the advantages and success of using aquatic invertebrate community responses to monitor human disturbance in freshwater ecosystems have been said to far outweigh the disadvantages found (Foster et al., 2015).

As discussed in Chapter 2, aquatic invertebrates form part of the diet of many fish species (Blaber and Whitfield, 1977; Bruton, 1978; Howard-Williams, 1979; Adite and Winemiller, 1997; Dalu et al., 2017). The production of fish is thought to be closely correlated with the production of detritivores, which play an important role in linking primary benthic resources and secondary consumers (Odum, 1970; Bowen, 1979; Boyero et al., 2021). A study of *Sarotherodon mossambicus* (now *Oreochromis mossambicus*) in Lake Sibaya showed accelerated growth rates for juveniles feeding on the shallow sand shelves of the lake while adults, that feed in deeper waters, were found to be suffering malnutrition with much slower growth rates (Bowen, 1979). This was concluded to be a result of much lower protein levels of organic matter as the lake deepened, associated with a reduction in available food sources in the pelagic area (Bowen, 1979). This study specifically highlights the importance of aquatic invertebrates at the lower levels of the trophic food web within freshwater ecosystems. Additionally, aquatic invertebrates form an important part of energy transfer between aquatic and terrestrial environments (Nakano and Murakami, 2001), being a significant portion of the diet of birds such as woodpeckers, flycatchers and warblers (Shelton et al., 2016). Therefore, the loss of aquatic invertebrate abundance would pose a threat to many terrestrial biota (mostly birds) that find a large portion of their diet in aquatic invertebrates.

While forming a crucial bottom-up role in aquatic food webs, aquatic invertebrates also play an important role in recycling nutrients, particularly nitrogen and carbon, through feeding and decomposition (Hart, 1981; Allanson, 1990; Covich et al., 1999; Moulton et al., 2012; Boyero et al., 2021; Makherana et al., 2022). The decomposition of dead organic matter is important within endorheic lakes because of the lack of outflow associated with these systems (Buah-Kwofie and Humphries, 2017; Kock et al., 2019). Increases in decaying organic matter with no breakdown leads to nutrient loading and can lead to eutrophication (Humphries, 2013; Humphries and Benitez-Nelson, 2013; Siméon et al., 2014; Bate et al., 2018; Kock et al., 2019). These findings reiterate the importance of nutrient recyclers and detritivores, most of which are aquatic invertebrates, within freshwater systems and more specifically within endorheic systems.

3.1.2. Littoral aquatic invertebrates of Lake Sibaya and Lake Mzingazi

Previous studies of littoral aquatic invertebrates in Lake Sibaya found *Tarebia granifera*, *Melanoides tuberculata, Corbicula* sp., *Apseudes digitalis*, *Cyathura* sp., *Caridina* sp., *Corophium* sp., *Grandidierella* sp., *Bulinus natalensis* as well finding Amphipods, Polychaetes, Oligochaetes and Chironomid larvae to be the dominant taxa (Allanson, 1990; Miranda and Perissinotto, 2014). Various taxa-specific studies have found iSimangaliso wetland park to be a hotspot of aquatic invertebrate endemism and biodiversity, specifically Odonata (Hart et al., 2014) and Coleoptera (Perissinotto et al., 2016; Bird et al., 2017). Hydrochidae, Spercheidae, Hydrophilidae, Hydraenidae, Scirtidae, Heteroceridae, Curculionidae, Gyrinidae, Haliplidae, Noteridae and Dytiscidae beetles have all been noted in the park (Perissinotto et al., 2016; Bird et al., 2017) along with Gomphidae, Aeshnidae, Corduliidae and Libellulidae (Hart et al., 2014). As can be seen in the above studies, all-encompassing studies of littoral (as opposed to benthic) freshwater invertebrates are rare and often out of date, which further informs the need for current and continuous monitoring studies on littoral aquatic invertebrates in South African lakes and specifically Lake Sibaya as highlighted by the 2018 National Biodiversity Assessment (Skowno, et al., 2019).

Lake Mzingazi shows a variety of decapod species including *Pennaeus* sp., *Palaemon* sp., *Macrobrachium* sp., *Cardina* sp., *Potomonautes* sp., *Gastrosaccus* sp., *Callichirus* sp., *Clibanarius* sp., *Hymenosoma* sp., *Scylla* sp., *Dorippe* sp., *Dotilla* sp., *Sesarmid* sp., *Uca* sp. and *Varuna litterata* as well as Isopods (*Dies monodi, Apseudes digitalis and Excorollana* sp.), Amphipods (*Grandidierella lignorum, Corophium triaenonyx, Orchestia ancheidos, Hyale grandicornis* and *Bolttsia minuta*), Nematodes, Annelids (*Limnatus fenestrate, Capitellidae* sp.), Gastropods (*Biomphalaria* sp. and *Bulinus* sp.), Ephemeropterans (mainly *Baetis* sp.), Dytiscidae, Chironimidae and Trichoptera (*Leptocerius* sp.) (Fowles and Archibald, 1987; Mackay and Cyrus, 2001; Moloi, 2012; Weerts et al., 2014). The majority of these studies are presence-absence studies and are a decade old or older which once again informs the need for a more recent baseline aquatic invertebrate inventory for the lake to make comparisons and for understanding the impact of global change stressors (defined in Chapter 1) on freshwater systems.

3.1.3. Environmental drivers of aquatic invertebrate community composition in lakes

Biotic and abiotic variables are known to structure aquatic invertebrate communities (Dalu et al., 2022). Various studies focused on population drivers of aquatic invertebrate or benthic communities and have found links between water nutrients, habitat and aquatic invertebrate community composition (Pinel-Alloul et al., 1996; Dalu et al., 2013; Dalu et al., 2017; Dalu and Chauke, 2019; Majdi et al., 2022). Conductivity levels, typically reflecting dissolved ion content, were found to contribute significantly to variation in the aquatic invertebrate community composition of the shallow and oligotrophic Lake Saint-François in Canada (Pinel-Alloul et al., 1996) and the Sambandou wetlands (Dalu and Chauke, 2019). Dissolved ion content increases when external nutrient loading increases within freshwater systems, typically as a result of surrounding land-use activities such as urbanization and agriculture (Dalu and Chauke, 2019).

pH has been found to affect community composition of lake communities at a regional level but not on a local scale (Dalu and Chauke, 2019). Berezina (2001) found that species diversity tended to drop when pH was less than 4 and greater than 9 with the greatest diversity found at pH values between 4.09 and 8.65. It was also found that aquatic invertebrates had greater tolerance to pH changes when total dissolved solids (TDS) increased (Berezina, 2001). Decreases in pH are known to affect sodium uptake in both freshwater fish and *Daphnia* sp. (Aladin and Potts, 1995), likely contributing to the structuring of invertebrate communities.

Additionally, harmful algal blooms (HABs) due to enriched nutrient conditions can produce cytotoxins and create deoxygenated zones (Paerl and Huisman, 2009; Ndlela et al., 2016) such as those created by the invasive free-floating plants; water lettuce (*Pistia stratiotes)* and water hyacinth (*Eichhornia crassipes*)(Chamier et al., 2012). These deoxygenated zones have been found to reduce aquatic invertebrate movement, respiration, feeding, growth, and reproduction rates (Lee et al., 2023) while toxins from HABs can affect aquatic invertebrates (specifically filter feeders) when digested (Turner et al., 2021). These consumed toxins then also move through the trophic web (Turner et al., 2021).

A study by Gallardo et al. (2016) found that macrophytes, zooplankton and fish are significantly affected by the presence of invasive organisms. Such examples are the common carp (Ellender and Weyl, 2014), the Nile Perch (Goldshmidt et al., 1993) and *Tarebia granifera* (Appleton and Nadasan, 2002). Invasive species typically have rapid growth rates, broad environmental tolerances, high production (and success) of offspring with little parental input and early sexual maturity (Bownes and McQuaid, 2006; Zardi et al., 2018). Alien invasion can be further enhanced by increasing global temperatures and most stressors associated with anthropogenic pollution and climate change (Mainka and Howard, 2010).

Tarebia granifera has invaded the majority of Northern KZN coastal lakes (Appleton and Nadasan, 2002; Appleton et al., 2009; Miranda et al., 2011; Jones et al., 2014; Miranda and Perissinotto, 2014). *Tarebia granifera* has been known to displace native gastropod species such as *Melanoides tuberculata* with predictions also showing their likelihood to restructure benthic communities and ultimately affect biodiversity (Appleton et al., 2009). Additionally, populations of the invasive snail have been found to reach over 1000 individuals per $m^{2.}$ (Miranda et al., 2011), with high densities (+-500 individuals/m^2) being found to be related to strong decreases in the availability of chlorophyll-a (Makherana et al., 2022). This further confirms the increased dietary competition created by large *T. granifera* populations to organisms which share their dietary niche.

Other drivers of aquatic invertebrate abundance are high temperatures, high fish abundance, large hippo populations and water level fluctuation. High temperatures can lead to exceedance of the thermal tolerance limits of organisms, ultimately reducing aquatic invertebrate abundances (de Necker et al., 2016). High fish abundance can lead to increased predation pressure on aquatic invertebrates (Mallory et al., 1994; Dalu et al., 2022). Areas with large hippo populations can experience nutrient loading because of the dung entering the water which affects more sensitive organisms (Dawson et al., 2016). Water level fluctuations influence the extent of littoral vegetation cover (Gaeta et al., 2014), which would increase competition if less habitat is available and would therefore lead to higher mortality. Additionally, hydrophyte cover, sedimentation rates, availability of benthic algae and wave action have also been shown to contribute to the structuring of littoral freshwater communities (McLachlan and McLachlan, 1971; Marshall, 1978; Allanson, 1990; Dalu et al., 2022; Makherana et al., 2022).

3.1.4. Study rationale

There is large variation in the response of aquatic invertebrate communities to global change-induced stress and as a result, there are many recommendations to further study community response to land-use changes in surrounding catchment areas (Allanson, 1990; Dalu et al., 2013; Dalu and

Chauke, 2019; Dalu et al., 2022; Makherana et al., 2022). This recommendation informs the use of Lake Sibaya and Lake Mzingazi as study sites because of the differences in land use in their surrounding catchment areas (Agriculture at Lake Sibaya and urbanization at Lake Mzingazi). There are also recommendations to test the generality of studies on larger spatiotemporal scales to assess the true nature of aquatic invertebrate community responses to global change (Dalu et al., 2021). This recommendation informs the length of the study period (10 months) which allows more understanding of the natural temporal changes that occur in both systems and the opportunity to compare these changes on a spatial scale with two coastal lakes found at a distance of approximately 160km apart. The 2018 South African National Biodiversity Assessment (Skowno et al., 2019) found a lack of invertebrate foundational data and specifically those that are considered sensitive to changes in environmental conditions (such as aquatic invertebrates). There was also a gap in data associated with ecological response to climate change and its cumulative effects on ecosystems (Skowno et al., 2019). This lack of information complicates the modelling of important ecological thresholds and hampers the prediction of possible responses of those ecosystems to environmental changes (Skowno et al., 2019). This lack of baseline information on environmentally sensitive organisms and their response to land-use changes in combination with the previously mentioned need for more recent studies over larger spatiotemporal scales informs the main rationale for this chapter – to create a 10-month baseline dataset of the aquatic invertebrate communities of Lake Sibaya and Lake Mzingazi to aid in the closing of these knowledge gaps, provide data usable in future comparison studies and possibly aid in the prediction of aquatic invertebrate community response to global change.

3.2. Aims and hypotheses

This chapter aims to quantify the littoral aquatic invertebrate diversity and assemblage patterns from Lake Sibaya and Lake Mzingazi over a 10-month period (March-December 2021) to form a comprehensive baseline dataset for use in future comparison. Additionally, this chapter aims to investigate the impacts of landscape developments and habitat change on aquatic invertebrate communities by understanding significant water quality parameters as drivers of aquatic invertebrate community variation in Northern KwaZulu-Natal lakes.

This chapter predicts that increases in agricultural and anthropogenic disturbance and habitat modification will lead to (1) aquatic invertebrate community composition at lakes Sibaya and Mzingazi being structured according to water quality variables that stem from surrounding land-use activities which will (2) lead to different community structures at each lake. Finally, it is hypothesised that (3) the invasive snail *Tarebia granifera* will likely be affecting the aquatic invertebrate diversity and composition of both lakes.

3.3. Materials and Methods

3.3.1. Study sites

Aquatic invertebrates were collected at three sites per lake as described in Chapter 2 on pages 35-37 (Section 2.4.1).

3.3.2. Physicochemical parameters

All physicochemical variables collected during the study are described in Chapter 2 on page 37 (Section 2.4.2).

3.3.3. Sampling of aquatic invertebrate communities

Aquatic invertebrates were collected following the methods described in Mlambo et al. (2011) using a 1-metre SASS net (30 x 30 cm square frame, 1mm mesh size) (Dickens and Graham, 2002). Twenty-seven sweeps were done per site incorporating three sampling biotopes (emergent vegetation, submerged vegetation and open water) where each was swept nine times. Three biotopes x nine sweeps equated the 27 sweeps completed per site. Collected aquatic invertebrates were transferred into a 1L sample jar and immediately preserved with an 80% ethanol solution. In the laboratory, aquatic invertebrate samples were further sorted and identified to the lowest possible taxonomic level using various identification keys from chapters of "Guides to Freshwater Aquatic Invertebrates of South Africa" (Griffiths and Stewart, 2001; Hart et al., 2001; Kensley, 2001; Appleton, 2002; Dippenaar, 2002; Oosthuizen and Siddall, 2002; Barber-James and Lugo-Ortiz, 2003; De Meillon and Wirth, 2003; Reavell, 2003; Samways and Wilmot, 2003; Harrison, 2003; Biström, 2007; Endrödy-Younga and Stals, 2007; Stals, 2007). Additionally, specimens were sorted into Functional Feeding Groups (FFGs) according to Merritt et al. (1996), Cummins et al. (2005) and Fry (2021).

3.3.4. Statistical Methods

As with Chapter 2, the use of this monthly replicate sampling incorporates the concern of pseudoreplication and the methods as described on page 40 are used in this chapter to ensure no models with unacceptable levels of residual heterogeneity or autocorrelation are utilised to avoid the possibility of a type-I statistical error. Additionally, the simplest models were used to describe the most variation to avoid models with singularity errors and the possibility of a type-II statistical error.

All statistical analyses were conducted using R-Studio (2009-2022 RStudio, PBC) excluding calculations for taxa richness and Pielou's evenness which were calculated on Microsoft Excel. Taxa richness was calculated using Microsoft Excel's "countif" function specifying to count if the value present is greater than 0. Shannon's diversity index values were calculated using the "diversity" function (Package: "vegan") in R Studio. Thereafter, Pielou's evenness was calculated using the formula: “=Shannon’s diversity score/log (Richness)” in Microsoft Excel and finally, Bray-Curtis dissimilarity matrices were calculated using the “vegdist” function (Package: “vegan” specifying “method = “bray””) in R Studio.

Data concerning diversity indices (Taxa richness, Pielou’s evenness and Shannon’s diversity index) as well as aquatic invertebrate abundance were checked for normality using a Shapiro-Wilk normality test (Function: “shapiro.test”, “Package: “Stats”) and were found to not be normal ($P<0.05$). Thus, a non-parametric test, in this case a Kruskal-Wallis test, using the function “Kruskal.test” (package: “stats”) was conducted to assess whether diversity indices were different between months, sites and seasons. Dunn’s post hoc test using the Hochberg method for adjusting of p-values (Function: “dunnTest”, specifying “method = “Hochberg”” from package: “FSA”) was then done to assess which factor levels were contributing significantly to variation in aquatic invertebrate diversity indices.

Principle Co-ordinates Analyses were conducted to assess temporal variation in aquatic invertebrate community composition at both lakes using the “cmdscale” function (Package: “stats”) and results were plotted using the functions “ggplot” and “geom_path” (Package: “ggplot2”).

To determine the contribution of spatial (site and lake) and temporal (month) variation as drivers of variation within aquatic invertebrate communities, Permutational Multivariate Analyses of Variance (PERMANOVA) analyses were conducted using the “adonis2” function (Package: “vegan”). A PERMANOVA is similar to a standard MANOVA but is based on permutations of supplied distance matrices. The PERMANOVA run by the “adonis2” function fits linear models to distance matrices and partitions distance matrices among sources of variation, which can be either categorical or continuous (Oksanen et al., 2022). PERMANOVAs have been found to be more powerful than ANOSIM and Mantel tests at highlighting changes in community structure (Anderson and Walsh, 2013) and are also highly flexible, catering to both complex designs and small sample sizes, with very few assumptions (Walters and Coen, 2006). The strength of PERMANOVA analyses are drastically reduced in the case of unbalanced study designs, which is the case with the season variable (i.e. one summer month vs three months for all other seasons) and season was therefore not included in PERMANOVA analyses. Furthermore, a SIMPER analysis (Function: “simper”, Package: “vegan”) was used to determine the taxa that significantly contributed to community dissimilarity. All genera contributing up to 80% of community dissimilarity measures were considered significant.

To test the significance of numeric physicochemical variables contributing to aquatic invertebrate abundance, richness, evenness and diversity, Linear Mixed Models (LMMs) were conducted using the “lmer” function (Package: “lme4”). All predictor variables (i.e. pH, salinity, conductivity, temperature and phosphate, nitrate and ammonium concentrations) were scaled preceding all LMMs using the function “scale” (Package: Base) which centres (χ-column mean) the data and then divides the centred value by the column’s standard deviation, producing a Z-Score ($Z = \frac{x - mean}{SD}$). Three models for every response variable (i.e. abundance, richness, evenness and diversity) were created. The first was a standard linear model with no random effects (the nested model), the second was a mixed model with site as a random effect (first full model) and the third was also a mixed model with month as a random effect (second full model). These three models were compared with one another to find the best-performing model for each response variable. Variance inflation factors were checked to ensure there were no collinear variables using the “vif” function (Package: “car”) and all collinear variables (VIF>5) were removed before deciding on the best-fitting model. Comparisons of the nested and full models were done using the “anova” function (Package: “Stats”) to perform a likelihood ratio test (Table S6).

If the LMs and LMMs did not produce acceptable acf or residual plots (Table S7), then generalised linear (mixed) models (GL(M)Ms) were performed. The predictor variables were scaled before any models were run as with the above linear models. In deciding on the exponential family to be used in the GLMs, various models using different families were run and their AICc values were assessed to choose the best-performing model (Table S8). The Negative Binomial (“glm.nb” using Package: “MASS”) and Poisson (“glm” using package: “stats”) models were chosen for Lake Sibaya’s aquatic invertebrate abundance due to the positively skewed count data for which both families are appropriate. The Poisson models operate under the assumption that the means and variances of the data are equal and violating this assumption can lead to over-dispersion, which is catered for in the Negative Binomial models. The model with the lowest AICc was chosen and this was the family used on the final model (Negative Binomial). Models for Pielou’s evenness at Lake Sibaya and Shannon’s diversity index at both lakes were all tested using the Gamma and Inverse Gaussian families because these datasets all consisted of positive continuous values. The models run using the Gamma family all outperformed the models run using the Inverse Gaussian family. Therefore, GLMs for Lake Sibaya’s evenness and diversity values for both lakes were run using Gamma as the selected family (Table S8).

As with the L(M)Ms, three models for each response variable were created (no random effect, random effect of month and random effect of site). Initial models (no random effect) were compared with both mixed models in separate likelihood ratio tests to determine if either mixed model outperformed the initial model (Table S9). Acf and residual plots were also compared for the three models to confirm the results of the likelihood ratio tests (Table S7). Following acceptable AICc, VIF and acf values for the above models, the “dredge” function from Package: “MuMIn” was then used to find the most parsimonious model (defined as the model using the fewest variables to explain the most variation) and this model was used as the final model.

Generalised Linear Mixed Models (GLMMs) were conducted using the same steps as above with the function "glm" (package: "lme4") when assessing the effects of *T. granifera* on community diversity. Various exponential families were tried and the family with the lowest AICc value was chosen for the final model family (Table S10). The models used for both Lake Sibaya and Lake Mzingazi were conducted using the Gamma and Inverse Gaussian families because this data also consisted of positive continuous values. The addition of the random effects for month or site did not improve AICc, residual plots or autocorrelation factor plots (Table S11) so these models were run as standard generalised linear models. The final models and their parameters can be found in Table S12. All models contain at least the 3 highest contributing variables for the variation seen in the model according to the output from the "dredge" function.

3.4. Results

3.4.1. Temporal variation in aquatic invertebrate diversity and community composition of lakes Sibaya and Mzingazi

Lake Sibaya's aquatic invertebrate diversity ranged between 0.25 and 1.4, aquatic invertebrate abundance between 50 and 500, taxa richness between 3 and 8 and evenness between 1.0 and 2.0 during the study (Figure 3.1). The highest aquatic invertebrate abundance was recorded in August, September and October (Figure 3.1 A), whereas taxa richness and diversity were highest in March with the highest evenness observed in May (Figure 3.1 B & D). Aquatic invertebrate abundance, taxa richness and diversity were found to be the lowest in July, and in November for evenness (Figure 3.1). Lake Mzingazi on the other hand experienced ranges of between 0.5 and 1.75 for diversity, between 4 and 350 for aquatic invertebrate abundance, between 3 and 7 for taxa richness and between 0.8 and 3.0 for evenness (Figure 3.1). The highest aquatic invertebrate abundance and diversity values for Lake Mzingazi were found in April (Figure 3.1 A & C), the highest taxa richness in March and July (Figure 3.1 B) and the highest evenness in November (Figure 3.1 D). Aquatic invertebrate abundance, taxa richness and diversity were lowest in August and the lowest evenness was found in October (Figure 3.1).

When comparing the lakes, Lake Sibaya generally had higher aquatic invertebrate abundances (excluding July) and taxa richness (excluding April, July, October and December) throughout the study period (Figure 3.1 A & B). Whereas, Lake Mzingazi had higher diversity in April, June, July, November and December, with Lake Sibaya having higher diversity in March, May, August, September and October (Figure 3.1 C). Evenness scores were generally higher at Lake Mzingazi than at Lake Sibaya (excluding July) (Figure 3.1 D).

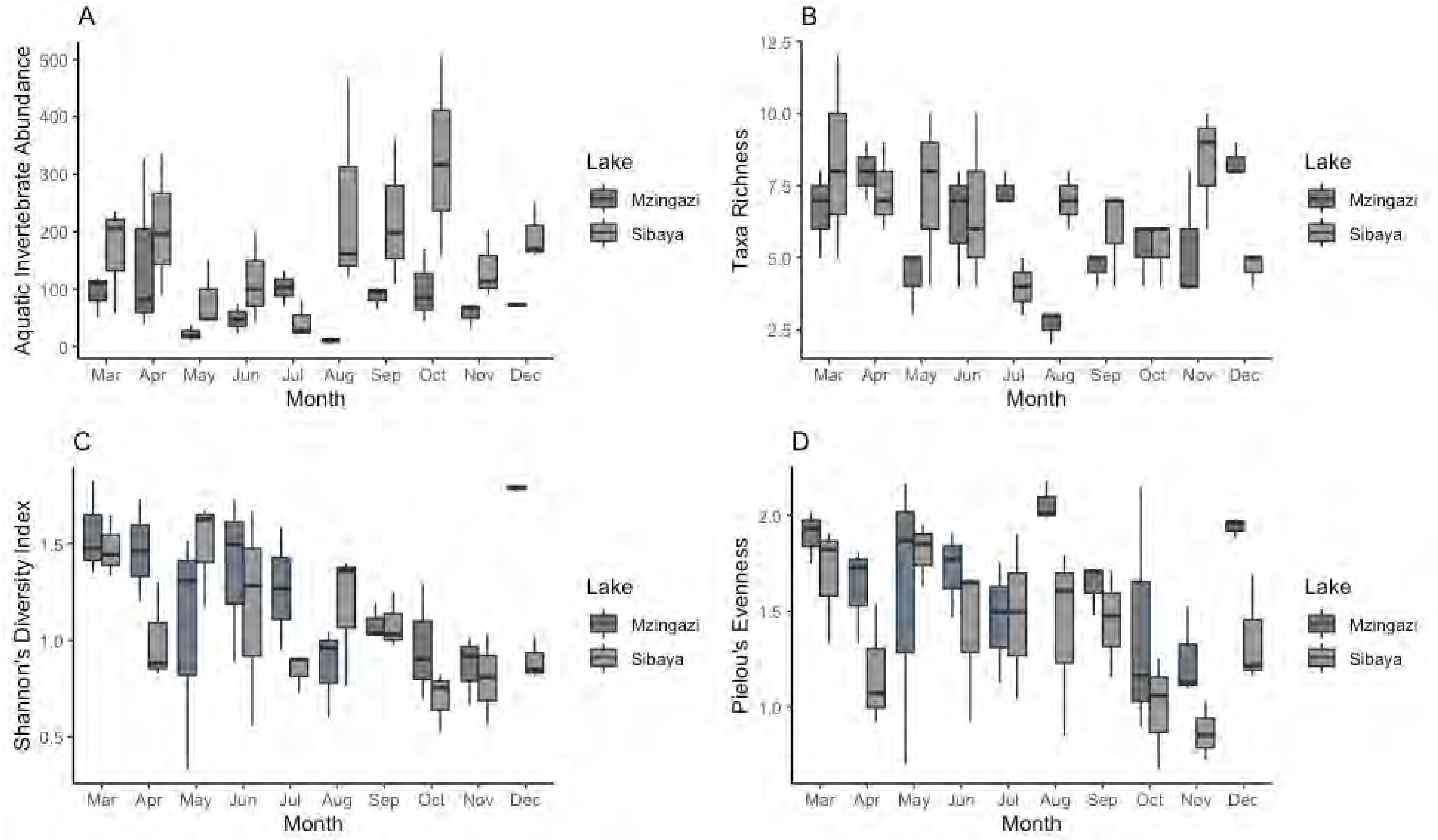

Figure 3. 1: Showing monthly biodiversity indices collected from lakes Sibaya and Mzingazi over the 10-month (March – December 2021) study period. Aquatic invertebrate abundance (A), taxa richness (B), Shannon's diversity index (C) and Pielou's evenness (D)

Kruskal-Wallis tests showed season to be contributing significantly to variation in aquatic invertebrate diversity at Lake Sibaya and Lake Mzingazi (Table 3.1). While month was found to be contributing significantly to aquatic invertebrate abundance and taxa richness values only at Lake Mzingazi (Table 3.1). Site was not found to be significantly contributing to variation in any diversity measures. When assessing the significance of categorical variables using Dunn's posthoc test, significant differences in aquatic invertebrate richness were only found between August and December at Lake Mzingazi (Table S13). Significant differences in aquatic invertebrate diversity were found between autumn and spring for Lake Sibaya and between spring and summer for Lake Mzingazi (Table S13). No significant differences were found using adjusted p-values for aquatic invertebrate abundance (Table S13).

Table 3. 1: Showing significant results found when assessing measured biodiversity indices (i.e. Aquatic invertebrate abundance, Taxa richness, Pielou's evenness and Shannon's diversity index) using Kruskal-Wallis tests from Lake Sibaya and Lake Mzingazi with site, month and season as factors

Lake	Response variable	Predictor variable	DF	Chisq	P-Value
Sibaya	Shannon's diversity index	Season	3	8.151	P<0.05
Mzingazi	Aquatic invertebrate abundance	Month	9	17.353	P<0.05
	Taxa richness	Month	9	19.939	P<0.05
	Shannon's diversity index	Season	3	12.418	P<0.01

Principal coordinate analysis (PCoA) results showed slight spatial and temporal differences in aquatic invertebrate community composition in Lake Sibaya (Figure 3.2). Site 1 had minor changes in composition from March to June, and a large shift in composition from June to July and July to August (Figure 3.2). From August through October more minor shifts in community composition were shown followed by larger shifts from October to November and November to December (Figure 3.2). The only clear seasonal grouping seen was between the final months moving from winter into spring and summer (August, September, October and December), with early autumn (March and April) into winter months (May and June) experiencing similarities in aquatic invertebrate composition (Figure 3.2). Site 2 experienced changes every month with only May, June and July showing one group and August and September showing another, indicating no large aquatic invertebrate variation that could be attributed to seasonal differences except for the cool months (May and June) (Figure 3.2). Site 3 experienced grouping in March, April, September and December and variation between the other months (Figure 3.3), showing the only seasonal grouping within the warmer months mentioned above and more community variation in the cooler months (Figure 3.2). When checking the correlation between genera and the results from PCoA, it was observed that *T. granifera* and *Bulinus* sp. were the most species correlated with Axis 1 (Correlation = -0.59 and -0.87 respectively), while *Caridina africana* had the highest correlation (0.76) with Axis 2.

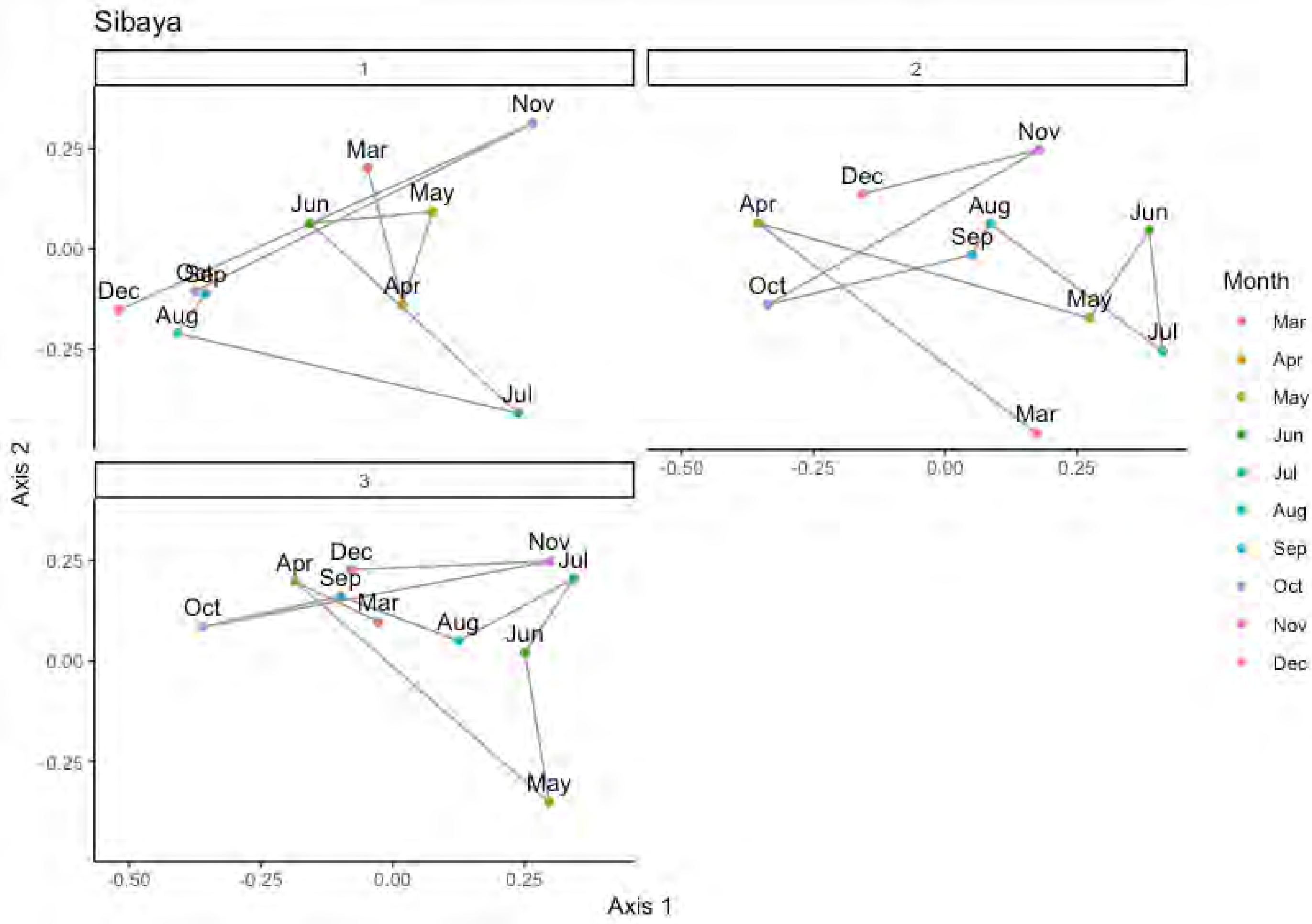

Figure 3. 2: Principle Co-ordinates Analyses performed on Bray-Curtis dissimilarity matrices showing monthly community variation of the aquatic invertebrates found at sites 1-3 at Lake Sibaya

Principal coordinates analysis (PCoA) results from Lake Mzingazi showed monthly changes in aquatic invertebrate community composition at sites 1 to 3 (Figure 3.3). Site 1 experienced grouping and similarity in aquatic invertebrate composition in March, September, October, November and December as well as in April, June and July and large variation in community composition between other months showing little seasonal grouping except for warm months and in the coolest months (June and July) (Figure 3.3). Site 2 did not experience any clear seasonal grouping, with similar variation between months. There was a larger variation seen between July and August, and August and September, which is unlikely to be related to seasonal differences (Figure 3.3). Site 3 showed three main groups with March, September and October forming the first group, July, November and December forming the second group, and April, June, May and August forming the third (figure 3.3). This indicates similarity in aquatic invertebrate community structure during warmer (group 1) and cooler (group 2) months but also indicates overlap within warm and cool months shown by the third group (Figure 3.3). Checking the correlation between genera and the PCoA, *Cloeon* sp. was highly correlated (0.77) with Axis 2 while *T. granifera* was mildly correlated with Axis 1 (-0.48) and Axis 2 (-0.50).

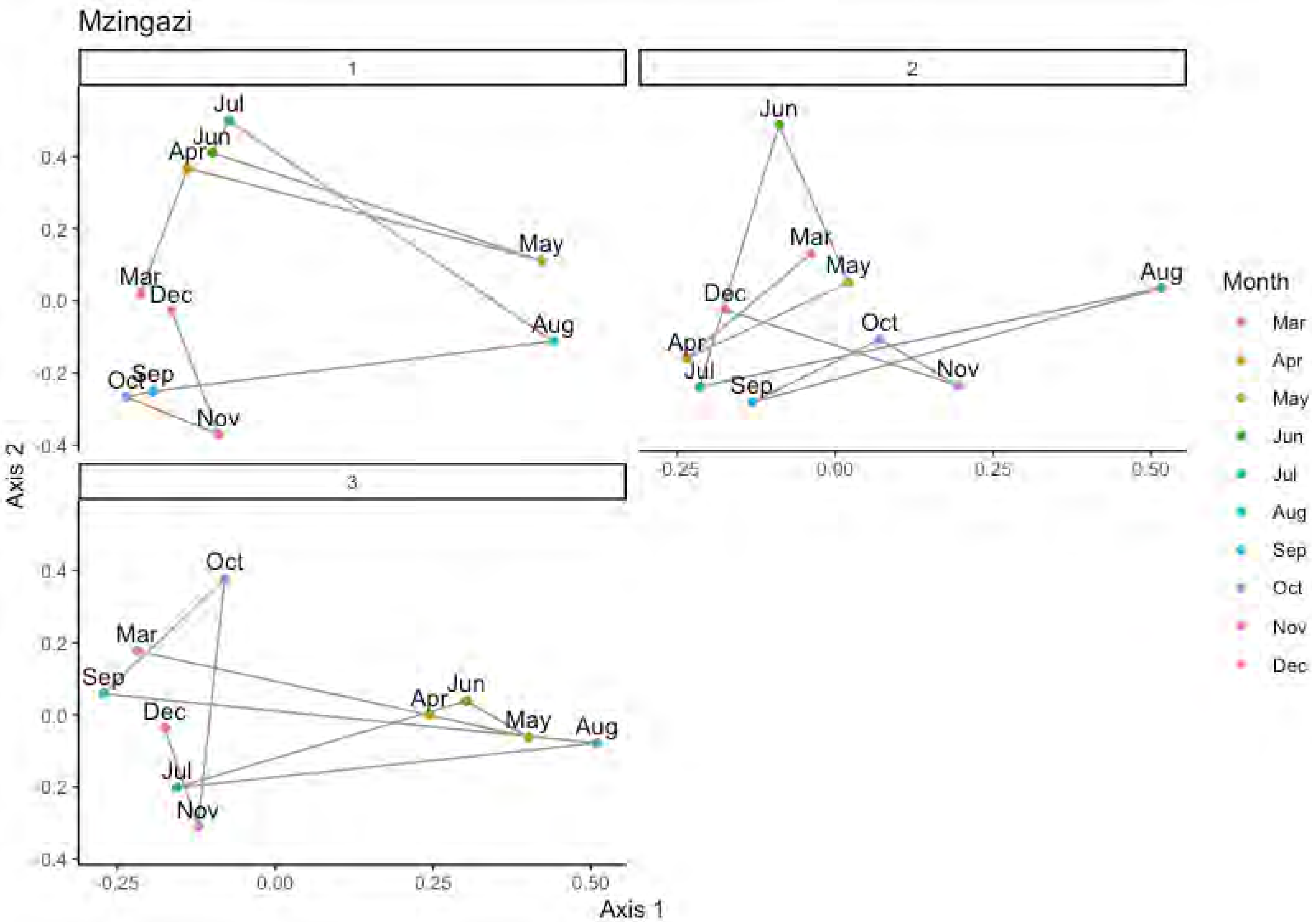

Figure 3. 3: Principle Co-ordinates Analyses performed on Bray-Curtis dissimilarity matrices showing monthly community variation of the aquatic invertebrates found at sites 1-3 at Lake Mzingazi

PERMANOVA analyses conducted on aquatic invertebrate community composition found at both lakes revealed that month was contributing significantly ($P<0.05$) to the variation seen in aquatic invertebrate community composition at Lake Sibaya and Lake Mzingazi (Table 3.2). Season could not be included due to the unbalanced nature of the variable (see page 76 for further explanation) and site was found to be insignificant.

Table 3. 2: Showing significant results of PERMANOVA analyses testing aquatic invertebrate community variation at three sampled sites over 10 months at Lake Sibaya and Lake Mzingazi

Lake	**Significant Variable**	**DF**	**F**	**P-Value**
Sibaya	Month	9	2.6395	$P \leq 0.001$
	Site	2	1.7538	$P>0.05$
Mzingazi	Month	9	2.1664	$P \leq 0.001$
	Site	2	0.9548	$P>0.05$

Taxa that were found to be contributing 80% of community composition variation within each lake according to SIMPER analysis were *T. granifera*, which was found to account for most of the variation within lakes (around 60%), and *C. africana*, *Cloeon* sp. and *Bulinus* sp. which contributed the rest of the 80%. Lake Sibaya had higher abundances of *Bulinus* sp., *C. africana* and *T. granifera* (Figure 3.4 B-D) throughout most of the study excluding April for *Bulinus* sp. (Figure 3.4A) and July and November for *T. granifera* (Figure 3.4D). *Cloeon* sp. was found to be more abundant at Lake Mzingazi than Lake Sibaya in March, April, July, October and December and more abundant at Lake Sibaya in May, June, August, September and November (Figure 3.4C). Both lakes show a general decrease in the abundance of all major contributing taxa during the cooler months of the study (Figure 3.4).

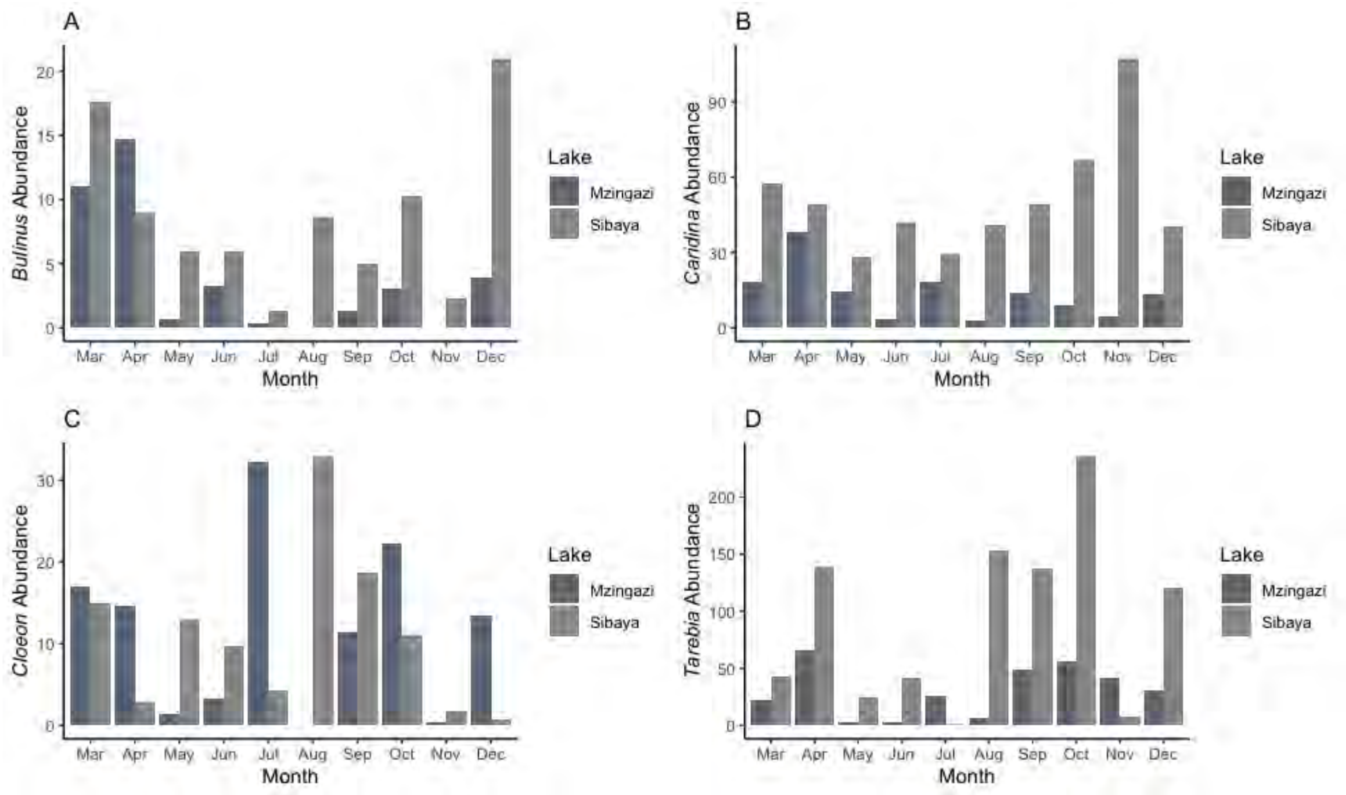

Figure 3. 4: Showing abundances of genera found to be significantly contributing to variation in aquatic invertebrate community composition within lakes Sibaya and Mzingazi over the study period (A – Bulinus sp., B – Caridina sp., C- Cloeon sp., D- Tarebia sp.)

3.4.2. Relationship between physicochemical drivers and aquatic Invertebrate diversity in lakes Sibaya and Mzingazi

Generalised Linear (Mixed) Models (GL(M)Ms) showed salinity, lake level, water temperature, conductivity and phosphate and nitrate concentrations to be contributing significantly to aquatic invertebrate abundances and diversity indices at Lake Sibaya (Table 3.3, Figure 3.5). Salinity (Figure 3.5A), lake level (Figure 3.5B) and phosphate concentration (Figure 3.5D) showed significant negative effects on aquatic invertebrate abundance and temperature (Figure 3.5C) showed a significant positive effect on aquatic invertebrate abundance. Conductivity had a significant negative effect on taxa richness (Figure 3.5E). Nitrate concentration exhibited a significant negative relationship with Pielou's evenness (Figure 3.5G), while salinity (Figure 3.5F) and phosphate concentrations (Figure 3.5H) showed a positive significant relationship with Pielou's evenness. Salinity also had a positive relationship with Shannon's diversity index (Figure 3.5I). At Lake Mzingazi GLM(M)s showed salinity, nitrate and ammonium to be contributing significantly to aquatic invertebrate abundances and diversity indices (Table 3.3, Figure 3.6). Salinity had a significant positive effect on Pielou's evenness (Figure 3.6A) and Shannon's diversity index (Figure 3.6D) while nitrate and ammonium concentrations both showed significant negative relationships with Pielou's evenness (Figure 3.6B and C).

Table 3. 3: Showing significant results found when assessing measured biodiversity indices (Aquatic invertebrate abundance, taxa richness, Pielou's evenness and Shannon's diversity index) using (generalised) linear (mixed) models from Lake Sibaya and Lake Mzingazi in response to all measured numeric variables. F-Values (and their associated P-Values) are in italics and Chisq values are in standard text

Lake	**Response variable**	**Predictor variable**	**DF**	**Chisq/** ***F- Value***	**P-Value**
Sibaya	Aquatic invertebrate abundance	Salinity	1	12.562	P<0.001
		Lake Level	1	9.186	P<0.01
		Temperature	1	12.482	P<0.001
		Phosphate	1	17.015	P<0.001
	Taxa richness	Conductivity	1	*5.537*	*P<0.05*
	Pielou's evenness	Salinity	1	12.387	P<0.001
		Nitrate	1	5.910	P<0.05
		Phosphate	1	4.273	P<0.05
	Shannon's diversity index	Salinity	1	5.623	P<0.05
Mzingazi	Pielou's evenness	Salinity	1	*18.297*	*P<0.001*
		Nitrate	1	*5.506*	*P<0.05*
		Ammonium	1	*5.046*	*P<0.05*
	Shannon's diversity index	Salinity	1	6.511	P<0.01

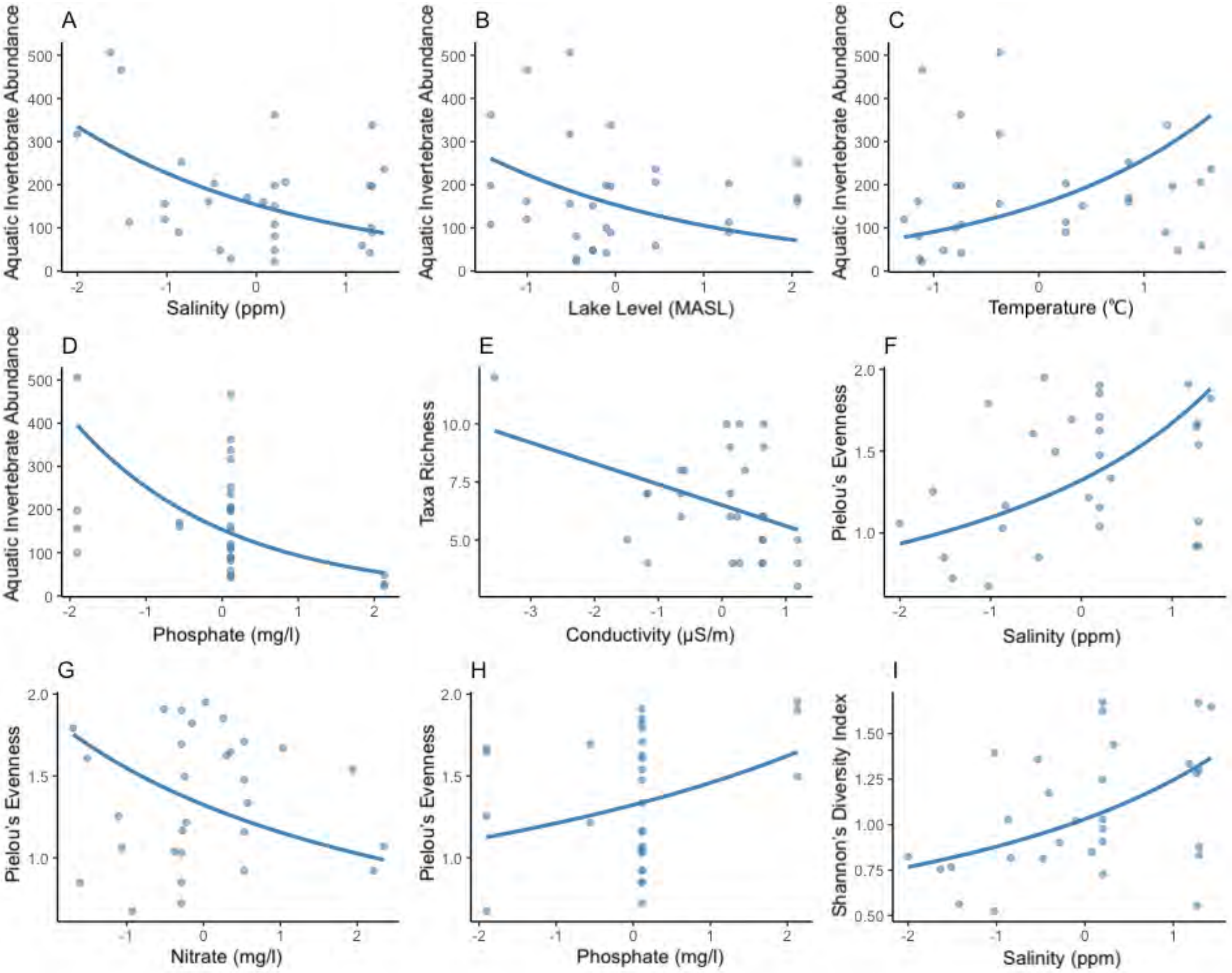

Figure 3. 5: Showing effect plots of Z scores for significant predictor variables found in (generalised) linear (mixed) models conducted on diversity indices for Lake Sibaya (A- Aquatic invertebrate abundance vs salinity (ppm), B- Aquatic invertebrate abundance vs lake level (Metres Above Sea Level - MASL), C – Aquatic invertebrate abundance vs temperature (°C), D- Aquatic invertebrate abundance vs phosphate concentration (mg/l), E- Taxa richness vs conductivity (µS/m), F- Pielou's evenness vs salinity (ppm), G- Pielou's evenness vs nitrate concentration (mg/l), H- Pielou's evenness vs phosphate concentration (mg/l), I- Shannon's diversity index vs salinity (ppm))

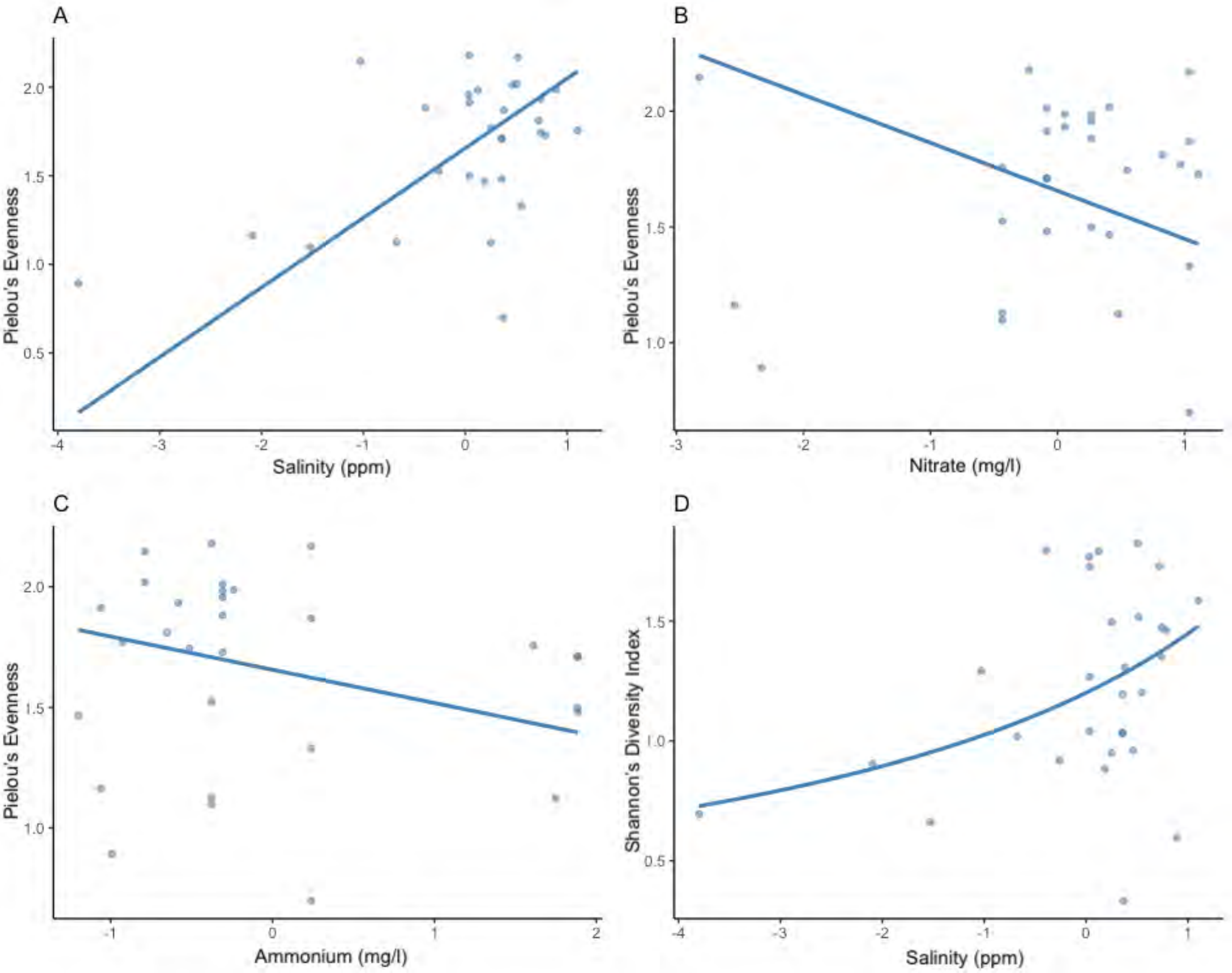

Figure 3. 6: Showing effect plots of Z scores for significant predictor variables found in (generalised) linear (mixed) models conducted on diversity indices for Lake Mzingazi (A- Pielou's evenness vs salinity (ppm), B- Pielou's evenness vs nitrate concentration (mg/l), C- Pielou's evenness vs ammonium concentration (mg/l), D- Shannon's diversity index vs salinity (ppm)

3.4.3. *Functional Feeding Group (FFG) composition from lakes Sibaya and Mzingazi*

Grazer/scrapers and collector-gatherers were the most abundant functional feeding groups at both lakes during the study with grazer/scrapers dominating in April, September, October and December (Making up over 50% of community composition (CC) in all months) at Lake Sibaya. Grazer/scrapers dominated at Lake Mzingazi in August, September and November (making up 50%+ CC during all months). Collector-gatherers dominated in March, May, June, July and November (over 50% CC) and both were similarly dominant in August at Lake Sibaya (48% and 50% CC) (Figure 3.7A). At Lake Mzingazi, collector-gatherers dominated from March till July (over 50% CC in all months) and grazer/scrapers dominated all other months (50%+ CC) except December where both collector-gatherers and grazer/scrapers were equally dominant (50% and 48% CC respectively) (Figure 3.7B). Both FFGs were found in similar percentage compositions at Lake Sibaya and Lake Mzingazi (Averages of 45% and 43% respectively for collector-gatherers and averages of 44% and 41% respectively for grazer/scrapers) (Figure 3.7). Predators were found in relatively low percentage compositions throughout the year at both lakes Sibaya and Mzingazi (yearly average contributions of 3% and 4% respectively) (Figure 3.7). Collector filterers were found in a very small percentage at Lake Sibaya in April and July (averages of less than 1% CC) (Figure 3.7A) and larger percentages in August (13% CC) and November (15% CC) at Lake Mzingazi. Collector-filterers made up a larger percentage of the yearly FFG composition at Lake Mzingazi when compared with Lake Sibaya (averages of 3% and less than 1% respectively) (Figure 3.7). No shredders were found at either lake during the study. There also does not seem to be a seasonal component to FFG structure during the year at either lake (Figure 3.7).

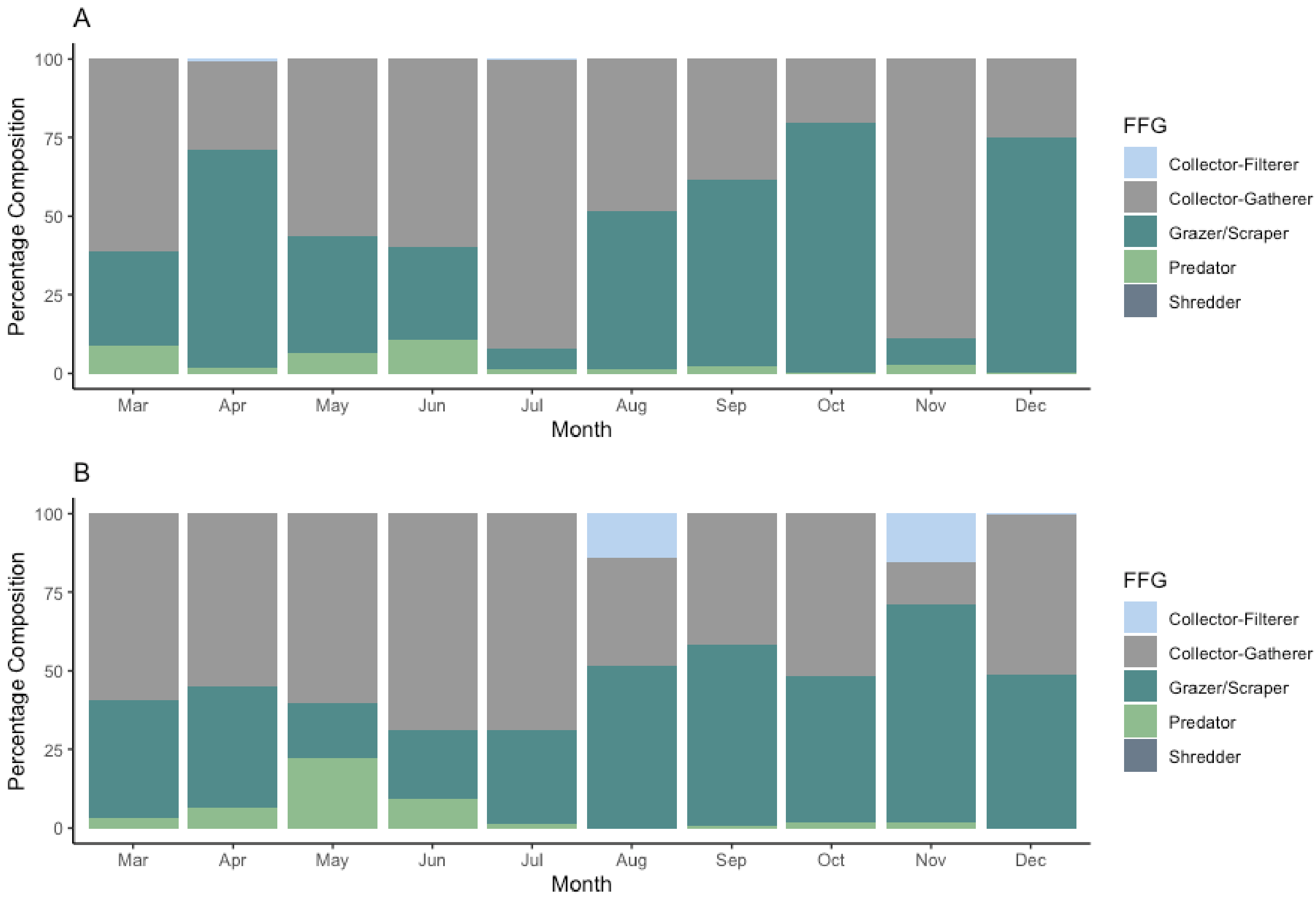

Figure 3. 7: Bar plots showing the Percentage Composition of Functional Feeding Group (FFG) at lakes Sibaya (A) and Mzingazi (B) from March to December 2021

3.4.4. The effect of Tarebia granifera on aquatic invertebrate community composition of Lake Sibaya and Lake Mzingazi

Both Lake Sibaya and Lake Mzingazi experienced a decline in diversity and evenness in the presence of increasing *T. granifera* abundances (Figure 3.8). The dominance of *T. granifera* at both lakes surpasses that of native snails throughout the study, occurring in much higher abundances than both *Bellamya* sp. and *Bulinus* sp. (excluding July at Lake Sibaya and June at Lake Mzingazi where *Bulinus* sp. was most abundant) (Figures 3.9 and S1). Lake Sibaya had higher abundances of *T. granifera* than Lake Mzingazi (excluding July) but Lake Sibaya hosted higher abundances of native gastropods when compared with Lake Mzingazi (Figure 3.9). In general *T. granifera* percentage composition was over 75% of snail composition in all months, excluding March and July at Lake Sibaya and March and June at Lake Mzingazi (Figure S1). Generalised Linear Models conducted on Shannon diversity scores versus *T. granifera* abundances showed a significant negative relationship between Shannon's diversity index and *T. granifera* at Lake Sibaya but not at Lake Mzingazi (Table 3.4).

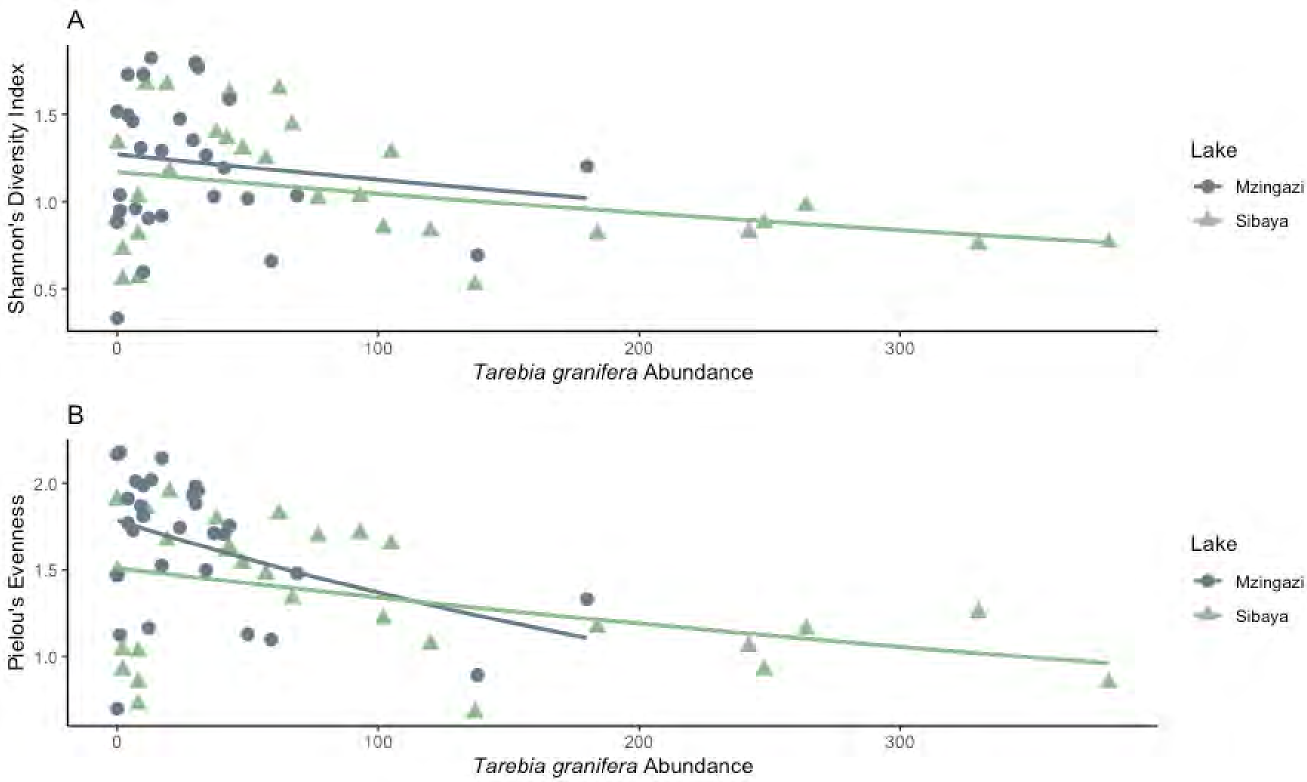

Figure 3. 8: Scatterplots showing (A) Shannon's diversity index and (B) Pielou's evenness in response to *Tarebia granifera* abundance at lakes Sibaya and Mzingazi

Table 3. 4: Showing results from generalised linear models conducted on native aquatic invertebrate community diversity in response to Tarebia granifera abundance at Lake Sibaya and Lake Mzingazi

Lake	**DF**	**Chisq**	**Pr(>Chisq)**
Sibaya	1	5.616	P<0.05
Mzingazi	1	0.042	P>0.05

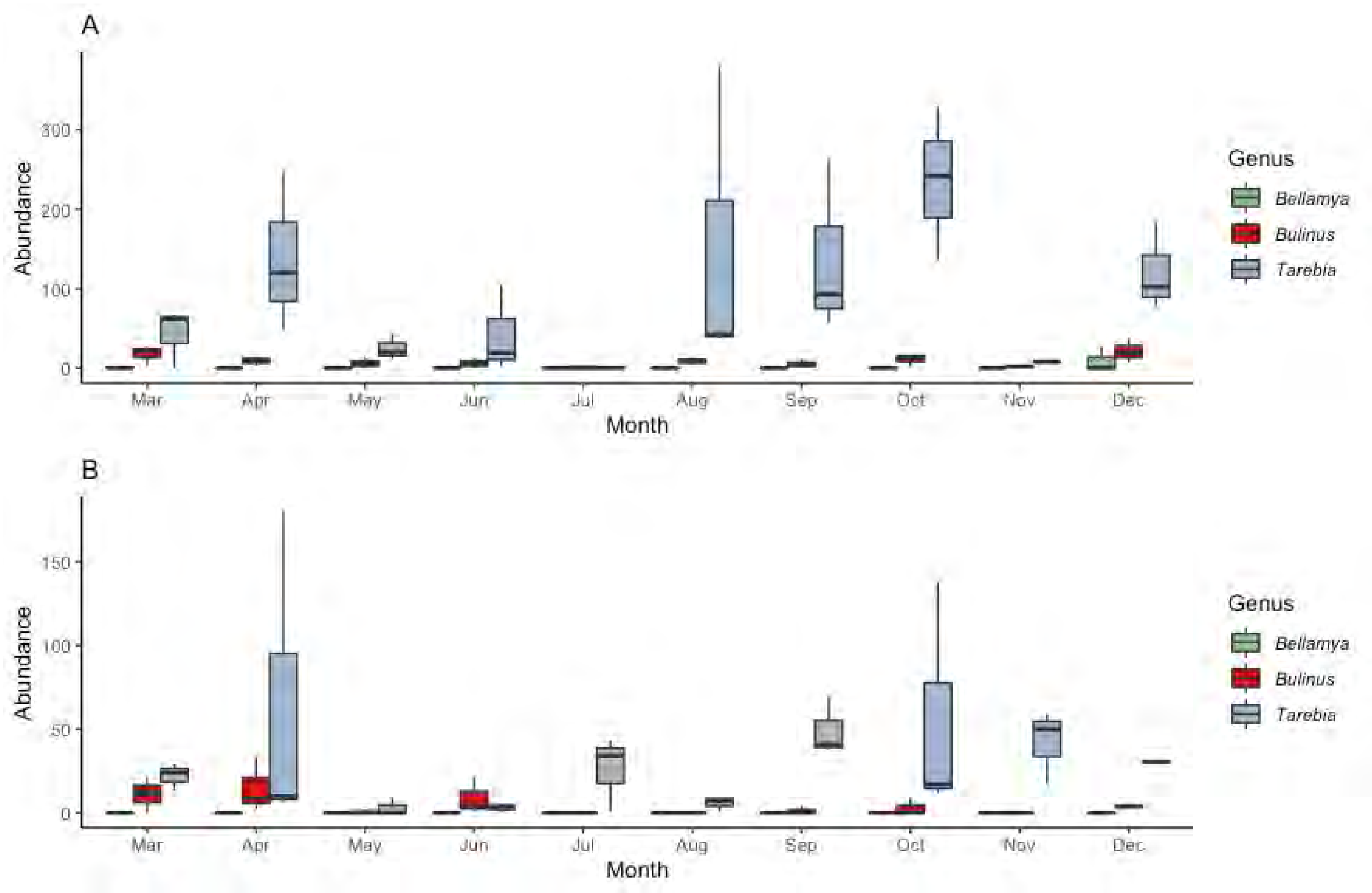

Figure 3. 9: Showing monthly snail abundances at lakes Sibaya (A) and Mzingazi (B) throughout the study period

3.5. Discussion

3.5.1. Drivers of aquatic invertebrate diversity patterns of Lakes Sibaya and Mzingazi

It was hypothesized in Section 3.2 that Lake Sibaya and Lake Mzingazi would have different physicochemical variables driving community structures due to differing land-use in their respective catchments. This was partially true with differences in significantly contributing physicochemical variables to aquatic invertebrate abundance and taxa richness at both lakes. In contrast to the above hypothesis, Pielou's evenness and Shannon's diversity index were significantly correlated with similar variables at both lakes. Lake Sibaya is situated in the iSimangaliso wetland park, a protected area which hosts high levels of diversity and endemism (Steenkamp et al., 2004; Perera et al., 2011; Perera et al., 2013), making the generally higher community abundance and taxa richness seen at the lake when compared with Lake Mzingazi expected and also in line with the hypotheses for this chapter. The higher evenness and sometimes diversity seen at Lake Mzingazi can be explained by the fact that *Tarebia granifera* does not dominate over the other present taxa as it does at Lake Sibaya, which would increase evenness and the associated diversity equation. In addition to these results, significant differences were found in the community composition of the two lakes, further emphasising the importance of designated national heritage sites in the conservation of resident biota and diversity while also showing the threat invasive species pose to biodiversity and evenness of lake systems.

Aquatic invertebrate community abundance tends to decrease during winter months as found by previous studies (Davies, 1982; Bredenhand and Samways, 2009; Makherana et al., 2022) putting the positive association between aquatic invertebrate community abundance and temperature at Lake Sibaya in line with previous conclusions. Lake Sibaya also showed a general decrease in all major contributing taxa (*Bulinus* sp., *Caridina africana*, *Cloeon* sp. and *T. granifera*) during the cooler months which further agrees with the findings of previous studies but this trend was not seen at Lake Mzingazi. The decrease in community abundance with an increase in lake level seen at Lake Sibaya could be explained by the fact that rising lake levels increase habitat availability (Hunt and Jones, 1972; Gaeta et al., 2014) and would therefore increase the spread of aquatic invertebrate communities around available habitat, leading to lower abundances.

Conductivity is said to affect aquatic organisms because it is indicative of the nutrient loading occurring in a system due to both anthropogenic and agricultural stressors (Johnson, 2007; Nhiwatiwa et al., 2017; Majdi et al., 2022). Similar results were found for conductivity levels of Lake Saint-François, which were found to contribute significantly to variation in aquatic invertebrate community composition (Pinel-Alloul et al., 1996). Similarly, in this study, conductivity was found to be negatively correlated with taxa richness values at Lake Sibaya. High salinities have also previously been linked to high anthropogenic effluent discharge (Majdi et al., 2022) as well as agricultural runoff causing excess input of dissolved ions into the system (Nhiwatiwa et al., 2017). Salinity and its corresponding ion input have been found to disrupt previously studied aquatic invertebrate communities (Nhiwatiwa et al., 2017; Riddel et al., 2019; Majdi et al., 2022). This corresponds with results found for aquatic invertebrate abundance at Lake Sibaya which decreased as salinity values increased but also contrasts the observed increases in diversity and evenness when salinity values increased at both lakes. These results could be attributed to the estuarine history of both lakes and the estuarine taxa that consequently inhabit these systems but the negative correlation seen between aquatic invertebrate abundance at Lake Sibaya and lack of significant correlation with abundance at Lake Mzingazi and richness at both lakes does seem to contrast this idea. The months with higher salinity values at Lake Sibaya did have lower abundances of *Tarebia granifera* and a more even community which would lead to higher Shannon's diversity index values (as evenness forms a part of the calculation of these values) so this trend is likely responsible for the pattern seen at Lake Sibaya. The salinity values recorded for Lake Mzingazi showed a generally consistent pattern through the year except for October when there was a 50% decrease seen in the salinity recorded and November when the salinity recording was still lower than the rest of the year. During these months, the community at Lake Mzingazi was less even (likely due to the dominance of *Cloeon* sp. and *T. granifera* in October and the dominance of *T. granifera* in November) which would again result in lower diversity values and could be the reason for the positive correlation seen between salinity and these values at Lake Mzingazi.

Common ions found within freshwater systems due to nutrient loading are inorganic phosphorous and nitrogen ions (DWAF, 1996). Increasing phosphate input is a well-known driver of aquatic invertebrate communities (Pinel-Alloul et al., 1996; Dalu et al., 2017; Majdi et al., 2022), often causing biomass loss (Dalu et al., 2017; Majdi et al., 2022), which seems to be shown in Lake Sibaya's invertebrate abundance decreases when phosphate levels are higher but is in contrast to results found for Pielou's evenness at the lake. This indicates that monitoring these concentrations would be important to gain a deeper understanding of the effects that varying phosphate concentrations exhibit on the aquatic invertebrate communities of Lake Sibaya.

Nitrogen input has also been linked with increasing agricultural runoff (Smith et al., 1987, USGS, 1999; Revenga et al., 2000) which is a present threat at Lake Sibaya and could explain the increase in inorganic nitrogen concentrations found at the lake when compared to historical data (discussed in Chapter 2). The association seen between aquatic invertebrate abundance and nitrogen is normally due to organism sensitivity to these compounds (Pinel-Alloul et al., 1996; Dalu et al., 2017). Nitrate and ammonium concentrations were found to be negatively correlated with evenness at Lake Mzingazi and evenness was found to be negatively correlated with nitrate at Lake Sibaya. These results agree with previous studies that found ammonium and nitrate concentrations to structure aquatic invertebrate communities (Pinel-Alloul et al., 1996; Dalu et al., 2017) and highlights the importance of monitoring and reducing inorganic nitrogen input within freshwater aquatic systems. The above-presented results inform the necessity of mitigating nutrient input within freshwater aquatic systems and also reiterates the importance of maintaining stable water levels and annual temperatures of lake ecosystems.

Diversity was significantly different in autumn and spring at Lake Sibaya and spring and summer at Lake Mzingazi, which is likely to be a result of the increase in diversity associated with the spring blooms most taxa experience (Davies, 1982) and the increase in diversity as temperatures increase into the summer period (Davies, 1982; Bredenhand and Samways, 2009; Makherana et al., 2022). Month was found to be contributing significantly to variation in invertebrate community abundance and taxa richness at Lake Mzingazi but not at Lake Sibaya. Although there are monthly differences, these do not necessarily follow any seasonal pattern and were only found to be significant when comparing taxa richness in August and December (the lowest and highest richness values found) and diversity in spring and summer according to the adjusted p-values of post hoc testing. Expected spring biomass increases and winter decreases were not observed in Lake Mzingazi's aquatic invertebrate richness and diversity patterns as would be expected, with autumn and winter sharing similar diversity and richness values. Additionally, spring diversity and richness were lower than in autumn and most of winter. This is likely a result of minimal temperature reductions from autumn to the winter period and unexpected temperature reductions following winter moving into the summer period. This indicates that temperature may have played a significant role in aquatic invertebrate diversity as found by previous studies (Davies, 1982; Bredenhand and Samways, 2009) but that these changes did not occur seasonally as would be expected. It is, therefore, more likely that these differences are more related to the changes in significant physicochemical variables discussed above as opposed to season. This finding again emphasizes the importance of pollution mitigation within freshwater ecosystems due to the effects on littoral invertebrate communities.

3.5.2. Functional Feeding Group (FFG) composition

Farrell et al. (2015) found collector-gatherers to be the dominant feeding group in the Wilge River, Mpumalanga and Burger et al., (2018) found collector-gatherers to be the second most abundant FFG in depression wetlands of Gauteng. Additionally, Dalu et al. (2022) found scrapers to be the most abundant FFG in the Nylsvely wetland, Limpopo. This corresponds with results found in this study which found collector-gatherers and grazer/scrapers to be the most abundant functional feeding groups (FFGs) present at both lakes as well as Miranda and Perissinotto (2014) who found the most abundant taxa at various lakes to be *T. granifera*, *Caridina* sp., isopods and amphipods which all fall into the aforementioned FFGs. Collector-gatherers and grazer/scrapers would all benefit from the increasing nutrient availability created by mesotrophic conditions found at both lakes (DWAF, 1996; Kock et al., 2019) which is likely the reason for their dominance of community FFG percentage composition.

Previous studies have found shredders to be one of the least abundant FFGs in aquatic systems (Farrell et al., 2015; Burger et al., 2018, Dalu et al., 2022), which was the case in this study and is further understandable considering the minimal overhanging vegetation at both lakes which would reduce leaf litter. Predators were found to be the second most dominant taxa in the Nylsvely wetland (Dalu et al., 2022), the dominant FFG in depression wetlands in Gauteng (Burger et al., 2018) and one of the most abundant FFGs in the Wilge River (Farrell et al., 2015) which contrasts the results of this study finding predators to be in higher percentage contributions than only shredders and collector-filterers. Miranda and Perissinotto (2014), however, did not find any taxa considered a part of the predator FFG to dominate Bray-Curtis similarity measures when sampling Lake Sibaya which does agree with the findings of this study. There is enough prey available to predators considering the high number of collector gatherers found at both lakes so the lack of predators in comparison to previous studies is unexpected. Studying this trend in more detail would be recommended to ascertain the reasons behind these composition differences. The low percentage contribution and rare occurrence of collector-filterers (*Corbicula* sp.) indicates they have likely inhabited areas that were not sampled in this study. This may be to avoid competition or because other areas host better habitat conditions but more research would have to be conducted to formulate a conclusion on this point.

3.5.3. Aquatic invertebrate community composition of Lake Sibaya

Minor spatial differences in aquatic invertebrate community variation were seen between the three sites sampled at Lake Sibaya but these differences were not found to be statistically significant. Site 1 was shown to have two groups of similar monthly compositions with warmer months forming a tight group, indicating minor changes in community composition during early spring and summer months when compared with autumn and winter months. This trend was also seen at site 3 but was not seen at site 2. Both sites 1 and 2 experienced loose grouping of the cooler months but this trend was not seen at site 3. The spatial differences exhibited by the aquatic invertebrate communities do not seem to be a result of differing physicochemical parameters at the different sites with all three sites showing similar measurements throughout the year. These differences also do not seem to be a result of habitat differences with sites 2 and 3 not showing similar community composition grouping despite sharing similar wave action, vegetation and distance to the main basin. Site 1 shared grouping of one season with both sites 2 and 3 despite having the lowest wave action and different vegetation present when compared with both other sites.

Additionally, *Apseudes digitalis*, *Corophium* sp., *Grandidierella* sp., Polychaetes, Oligochaetes and *Corduliidae* sp. were previously found at Lake Sibaya and were not found in this study. All taxa except *Corduliidae* sp. are typically found in benthic habitats making their presence in this littoral study unlikely. These results show the spatial differences possible within one system and emphasise the importance of using both benthic and littoral sampling methods when attempting to fully quantify the aquatic invertebrate communities of freshwater systems.

Temporal differences in community composition were seen at all three sites with sites 1 and 3 showing differences in community composition between warm and cool months. No grouping of cool months occurred at site 3 as was found at sites 1 and 2 and no grouping of warm months occurred at site 2 as was found at sites 1 and 3. Previous studies have shown lower temperatures to result in reductions in aquatic invertebrate abundances (Davies, 1982; Bredenhand and Samways, 2009). This could explain the grouping of these months and the significant differences seen in composition between spring and autumn and between July and August but similar temperatures were recorded during these two months making these significant differences unexpected. There were, however, reductions in the abundances of the major contributors to community dissimilarity during the cool months with reductions in the abundances of *Bulinus* sp., *Caridina africana* and *Tarebia granifera* following warm months at the beginning of the year and an increase in these taxa during the warmer months at the end of the year (with only August being an outlier). This corresponds with the previously mentioned studies that found reductions in aquatic invertebrate communities during winter months and biomass increases in the spring (Davies, 1982; Bredenhand and Samways, 2009; Makherana et al., 2022).

As previously discussed, ammonium and nitrate concentrations have been found to structure aquatic invertebrate communities (Pinel-Alloul et al., 1996; Dalu et al., 2017). This could explain the difference in composition between July and August despite similar seasonal considerations as there is a large decrease in both ammonium and nitrate concentrations between the two months. The reductions in salinity and conductivity at the same time further emphasize that these differences are likely a result of a decrease in nutrient loads as opposed to seasonal differences (Johnson, 2007; Nhiwatiwa et al., 2017; Riddel et al., 2019; Majdi et al., 2022). Increases in the abundance of *Bulinus sp*., *Caridina africana* and *Cloeon sp*. are seen from July to August. These results further confirm the composition changes as a result of nutrient loading as both *Caridina africana* and *Cloeon* sp. are known to be sensitive to increases in inorganic nitrogen concentration and would therefore be positively affected by reductions in nitrogen concentration (Bredenhand and Samways, 2009; Siméon et al., 2014). The above results show that the aquatic invertebrate communities of Lake Sibaya seem to follow typical seasonal patterns experienced by aquatic invertebrate communities, which confirms the results from Chapter 2 finding the effects of global warming to not be a significant concern within the lake at present when comparing current and historical water temperatures (see Hart, 1981; Allanson, 1990). There are, however, indications that the aquatic invertebrate communities at the Lake Sibaya are affected by the increases in inorganic nitrogen concentration seen within the lake when compared to historical nitrogen concentrations (see Chapter 2 for discussion of results found by Allanson, 1990 and Kock et al., 2019).

3.5.4. Aquatic invertebrate community composition of Lake Mzingazi

Minor spatial differences are seen between the three sites sampled at Lake Mzingazi but this variation was not found to be statistically significant. Sites 1 and 3 exhibited a grouping of some warm and cool months and site 2 only showed larger variation in July and August and then August and September. As at Lake Sibaya, these differences could be a result of slightly different habitat types with sites 1 and 3 sharing dense *Imperatus* grass and high wave action while site 2 experienced less wave action but with little protection in the form of fragmented water lilies. The difference in substrate (site 1 with sand and site 3 with rocks) makes these results slightly unexpected but these sites were along the same edge of the lake which could explain the shared communities between them.

Spatial differences were also exhibited when comparing the findings of this study to previous studies with a large number of aquatic invertebrates not found in this study when compared to the studies of Fowles and Archibald (1987), Mackay and Cyrus (2001), Moloi (2012) and Weerts et al. (2014). Various brackish Decapod species, Isopods, Amphipods, Nematodes, Annelids, one Gastropod species (*Biomphalaria* sp.) and one Trichopteran (*Leptocerius* sp.) were missing from the results of this study. There were differences in sampled habitat (benthic vs the littoral habitat sampled in this study) when comparing with Fowles and Archibald (1987) and Mackay and Cyrus (2001), once again showing the importance of sampling both benthic and littoral habitats when consolidating aquatic invertebrate communities. Differences in sampling methods were also seen with Weertz et al. (2014) utilising a variety of techniques (multiple seine nets, cast nets, gill nets and electro-shockers) and Moloi (2012) who used funnel traps, dip nets and seine nets. Moloi (2012) also sampled primarily within and in the immediate surroundings of the fishway (the connection to the saline environment of the Mzingazi River), which would host a larger number of brackish species than would be found in the standard freshwater environments sampled in this study. Once again, these findings show the spatial differences possible within a single lake system due to differences in habitat conditions.

Assessing temporal differences in aquatic invertebrate community structure at Lake Mzingazi indicated a common grouping of warmer months and grouping of cooler months which corresponds with findings from previous studies that found temperature to structure aquatic invertebrate communities (Davies, 1982, Bredenhand and Samways, 2009). However, there was also overlap in grouping with some cool and warm months grouping which does not fit the findings of those studies. It is possible that physicochemical variables are driving the communities more than seasonal considerations but most of the unexpected grouping could not be explained by similarities in physicochemical parameters. Temperatures at Lake Mzingazi did not seem to follow typical seasonal patterns with higher temperatures in autumn and winter than in summer, adding the possibility that temperature was driving these unexpected changes but grouping also did not correspond with similarities in temperature. Additionally, there was variation in almost all of the physiochemistry variables found for the grouped months which indicates some resilience to variation in physicochemical parameters exhibited by the aquatic invertebrate community of Lake Mzingazi. This trend should be studied more in-depth to understand what may be promoting this resilience of the aquatic invertebrate community.

On the contrary, some differences in temperature and physiochemistry were found for months that showed significantly different taxa richness (August and December). The three variables found to be different in those two months were temperature with a 2°C difference, conductivity with a 150μS/m difference and phosphate concentration with a 1mg/l decrease. The 2°C difference is likely not enough of a temperature difference to justify significant differences in community composition being a result of seasonality so the decreases in phosphate concentration and conductivity seen between these two months are more likely the cause of these differences in community composition. Similar results were found at Lake Saint-François (Pinel-Alloul et al., 1996), where aquatic invertebrate community composition was found to be significantly affected by variation in phosphate concentrations. Conductivity structuring aquatic invertebrate communities has been discussed in section 3.5.1 above and is typically a result of increased ion content within the water (Johnson, 2007; Nhiwatiwa et al., 2017; Majdi et al., 2022) so the similarities seen in pH, salinity and ammonium and nitrate concentrations in those two months does make this result slightly more unexpected as salinity has also been found to reflect ion content and the increasing ion content should be partially a result of increasing ammonium and nitrate concentrations (Dalu et al., 2013; Nhiwatiwa et al., 2017; Majdi et al., 2022).

The significance of differences found in aquatic invertebrate diversity between spring and summer does make slightly more sense when compared with the above results as there is a larger temperature difference (4°C) combined with differences in pH, phosphate concentration and conductivity in the two seasons which have all been discussed as drivers of aquatic invertebrate community composition (Davies, 1982; Pinel-Alloul et al. (1996); Blinn et al., 2004; Bredenhand and Samways, 2009; Dalu et al., 2013; Nhiwatiwa et al., 2017; Majdi et al., 2022; Makherana et al., 2022). The lack of difference in salinity, ammonium and nitrate concentrations does once again make this result unexpected as with the above results. Looking at the major contributing taxa to community dissimilarity does not aid in making sense of this temporal variation as there does not seem to be any seasonal component to the abundances of *Bulinus* sp., *Caridina africana*, *Cloeon* sp. *and Tarebia granifera* over the year despite the findings of previous studies showing biomass decreases in winter and increases in spring (Davies, 1982, Bredenhand and Samways, 2009). Overall the aquatic invertebrate communities of Lake Mzingazi showed unexpected variation in response to various physicochemical and seasonal changes which would require further study to fully understand what may be driving the differences seen in temporal composition.

3.5.5. Effects of Tarebia granifera invasion on aquatic invertebrate community composition

It was hypothesized in Section 3.2. that *Tarebia granifera* would likely be affecting the aquatic invertebrate diversity and composition of Lake Sibaya and Lake Mzingazi. *Tarebia granifera* was found in higher abundances than all native snails throughout the year at both lakes, excluding June at Lake Mzingazi, where *Bulinus* sp. were the most abundant snail (although in very low numbers) and July at Lake Sibaya where all three genera were not detected. These results are similar to Miranda et al. (2011), Miranda and Perissinotto (2014) and Makherana et al. (2022) who found that *T. granifera* tends to dominate the littoral community of freshwater aquatic systems, particularly the coastal systems. Another important note is that the native snail *Melanoides tuberculata* although previously known to occur in KZN coastal lakes, was not found in either lake throughout the study period. This matches the findings from Miranda et al. (2011) and Miranda and Perrissinotto (2014) who found no individuals of *M. tuberculata* at Lake Sibaya. This is likely due to the high competition created by high densities of *T. granifera* once populations have been established (Appleton et al., 2009; Miranda and Perrissinotto, 2012; Majdi et al., 2022; Makherana et al., 2022). *Melanoides tuberculata* may have been out-competed by its invasive counterpart but there is also the possibility that the snails have simply moved into deeper waters (Miranda et al., 2011) to avoid the competition created in the littoral zone by these exploding populations of *T. granifera* since their arrival to the South African coast in 1999. Either way, this finding is rather concerning for the future of *M. tuberculata* and similar native snails within South African freshwater systems.

Further adding to this concern, Shannon's diversity index was significantly correlated with *Tarebia granifera* abundance at Lake Sibaya with aquatic invertebrate diversity decreasing as *T. granifera* abundances increased as was predicted. Simper analyses conducted on the native community of Lake Sibaya showed *Cloeon* sp., *Caridina* sp., *Bulinus* sp. and Chironomidae to be contributing to 80% of the variation experienced within the native community of the lake, indicating that these species are most impacted by large established populations of *T. granifera*. These results match the predictions of Appleton et al. (2009) who stated that native gastropods and shrimp would be most affected by the invasion of *T. granifera* in aquatic systems and indicates the importance of invasive species mitigation in freshwater coastal lakes before these threatened populations are out-competed as it seems *M. tuberculata* has been. At Lake Mzingazi, however, although no individuals of *M. tuberculata* were found, Shannon's diversity index was not significantly associated with *Tarebia granifera* abundance. This indicates that not all aquatic invertebrate communities within freshwater systems respond to *T. granifera* invasion in the same way.

3.5.6. Conclusions

This chapter predicted that, firstly, aquatic invertebrate community composition at both lakes Sibaya and Mzingazi would be structured according to the water quality variables attributed to land-use activities in the surrounding catchments. This leading to differences in community composition between the two lakes. Results were partially in agreement with our hypothesis with physicochemical parameters typically a result of land-use structuring aquatic invertebrate abundance and taxa richness at Lake Sibaya, Pielou's evenness and Shannon's diversity index at both lakes and significant differences in the community composition between lakes being found. Aquatic invertebrate abundance at Lake Sibaya seemed to follow more typical seasonal patterns with temperature being a significant contributor unlike at Lake Mzingazi. Phosphate played a more significant role at Lake Sibaya with significant contributions to declines in aquatic invertebrate abundance, increases in Pielou's evenness and no significance at Lake Mzingazi. Finally, ammonium played a larger role at Lake Mzingazi with a significant contribution to declines in Pielou's evenness and no significance at Lake Sibaya. Contrary to the hypothesis of differing physicochemical parameters as a result of differing land-use was salinity contributing significantly to evenness and diversity at both lakes and nitrate concentration being significantly associated with a decline in evenness at both lakes.

These conclusions suggest that agricultural activities and urbanisation exhibit similar effects on aquatic invertebrate communities with the exception of the effect of phosphate concentrations on communities at Lake Sibaya and the effect of ammonium concentrations on communities at Lake Mzingazi. Results did, however, show that higher aquatic invertebrate abundance and taxa richness occurred in the national heritage site of Lake Sibaya when compared with Lake Mzingazi, indicating the importance of these demarcations in conserving aquatic invertebrate communities of South African freshwater coastal lakes. The negative effects of physicochemical parameters associated with land use on aquatic invertebrate abundance, taxa richness and Pielou's evenness seen in this study and in previous studies (specifically phosphate, nitrate and ammonium concentrations and the corresponding effects of salinity and conductivity), shows that these variables should be monitored more closely on a long-term basis to understand how we can mitigate their negative effects on freshwater invertebrate communities and ensure the survival of aquatic invertebrate communities under threat from land-use changes. Additionally, the lack of any clear pattern when looking at the aquatic invertebrate community composition of Lake Mzingazi in terms of season and physiochemistry variables indicates the need for more long-term studies to investigate drivers of aquatic invertebrate community composition within the lake and what may drive the apparent resilience to changes in environmental conditions.

It was also hypothesised that communities would also be affected by competition offered by *Tarebia granifera*. This hypothesis was found to be correct at Lake Sibaya with *T. granifera* abundances being significantly and negatively correlated with aquatic invertebrate diversity measures but was not found to be the case at Lake Mzingazi where *T. granifera* did not dominate over the native aquatic invertebrate community as it did at Lake Sibaya. The absence of *M. tuberculata* which has previously been found at Lake Sibaya and the dominance of the invasive *T. granifera* should act as a warning of the risks for South African freshwater systems in the presence of invasive species.

The above-presented information leads to the conclusion that the combination of anthropogenic/agricultural pollution as well as the invasion of exotic species does pose a risk to the aquatic invertebrate communities of South African freshwater coastal lakes. However, it is clear that these aquatic invertebrate communities do exhibit some resilience despite changes in physicochemical variables from month to month and the mechanisms of this resilience should be studied more closely to understand how freshwater littoral communities may resist the impacts of habitat disturbance going forward. Comparing the national heritage site of iSimangaliso Wetland Park in which Lake Sibaya is found with Lake Mzingazi also shows the efficacy of protected areas in preserving the abundance and richness of littoral aquatic communities. Unfortunately, this demarcation has not seemed to prove effective in combatting the invasion of species such as *Tarebia granifera*, which is now a well-established species within Lake Sibaya, dominating over all native taxa found.

Chapter 4

General Discussion and Conclusions

4.1. General discussion

4.1.1. Introduction and study rationale/predictions

Freshwater systems are considered important natural resources with important ecosystem services, providing direct and indirect benefits to humans and wildlife around the world (Dudgeon et al., 2006; Dallas and Rivers-Moore, 2014; Orimoloye, et al., 2020). The Maputaland-Pondoland-Albany (MPA) centre is a global hotspot found on the Eastern Coast of Southern Africa covering Mozambique, three provinces of South Africa (Mpumalanga, KwaZulu-Natal and the Eastern Cape) as well as Swaziland (now Eswatini) (Steenkamp et al., 2004). It is one of 34 global biodiversity hotspots, three of which are found in Southern Africa. The MPA hosts important conservation areas such as the iSimangaliso Wetland World Heritage Site and five RAMSAR sites and is considered an important biodiversity region for plant (Steenkamp et al., 2004), vertebrate (Steenkamp et al., 2004; Perera et al., 2011) and invertebrate endemism (Perera et al., 2013).

Lake Sibaya is South Africa's largest natural freshwater lake and is one of the lakes identified as nationally important by the National Biodiversity Assessment (Skowno et al., 2019). Lake Sibaya is fed largely by an underground aquifer and precipitation and, consequently, large amounts of water extraction required by the nearby towns of Mbazwana and Sodwana Bay, commercial plantations and the recent human population increase in its catchment combined with an ongoing drought is threatening its sustainability. Lake Mzingazi is a smaller system, situated in the middle of a sprawling urban environment, unlike Lake Sibaya which is a much larger system in a more rural setting.

Aquatic invertebrates offer vital ecosystem services to humans (Cardinale et al., 2000; Loreau, 2000). These are mainly through support (nutrient cycling) for aquatic systems as well as provisioning services (Millenium Ecosystem Assessment, 2005). These organisms support aquatic resources and other organisms such as fish but also offer important information on the ecological state of these systems by acting as bioindicators (Lenat, 1988; Ketse et al., 2006; Bredenhand and Samways, 2009; Siméon et al., 2014; Dalu et al., 2017; Jansen Van Rensburg et al., 2019).

The decomposition of dead organic matter is important within lake systems because of the low outflow associated with the systems (Buah-Kwofie and Humphries, 2017; Kock et al., 2019). Increases in dead organic matter with no breakdown lead to nutrient loading and can lead to eutrophication (Humphries, 2013; Humphries and Benitez-Nelson, 2013; Siméon et al., 2014; Bate et al., 2018; Kock et al., 2019). The South African National Biodiversity Assessment (Skowno et al., 2019) found a lack of invertebrate foundational data for endemic species as well as a gap in data associated with ecological response to climate change and the tendency of stressors to act in conjunction with one another which makes modelling important ecological thresholds and projected responses difficult (Skowno et al., 2019).

The application of aquatic invertebrates in the monitoring of human disturbance in lake systems is less understood and more widely debated than that of river systems (White and Irvine, 2003; Poikane et al., 2016). Studies have shown success in determining ecological changes resulting from global change impacts using the aquatic invertebrate communities of perennial pans in northern provinces of South Africa (Ferreira et al., 2012; Foster et al., 2015; de Necker et al., 2016; Dalu et al., 2021). On the other hand, using regional aquatic invertebrate indices as indicators for human disturbance has been shown to be unsuccessful in the South-Western Cape, Mediterranean region (Bird et al., 2013). This finding was said to be a result of the inherent spatial and temporal variation present within aquatic invertebrate communities of temporary pans leading to the assumption that patterns in aquatic invertebrate community structure associated with human disturbance would likely be easier to detect on a local scale (Bird et al., 2013). Further, the advantages of studying aquatic invertebrate community responses to human disturbance have been said to far outweigh the disadvantages (Foster et al., 2015). The permanent nature of Lake Sibaya and Lake Mzingazi will more than likely reduce the inherent temporal variation typically experienced by endorheic pan communities.

The above-presented information informed the aims and rationale of this dissertation; to identify longer-term spatiotemporal trends evident in aquatic invertebrate communities of Lake Sibaya and Lake Mzingazi. Secondly to determine how agriculture and urbanisation may affect them as well as to identify any long-term changes in the aquatic invertebrate communities of both lakes using historical baseline datasets for comparison. Additionally, this dissertation aimed to understand if urbanisation and agriculture present different environmental stressors to these freshwater systems and the communities inhabiting them.

4.1.2. Has habitat change occurring at Lake Sibaya led to long-term impacts on *Caridina africana*?

Chapter 2 aimed to quantify the current temporal population dynamics of shrimp species at Lake Sibaya and Lake Mzingazi for one year (March 2021 - February 2022) and further investigate the impacts of water level reduction and landscape developments/habitat change on *Caridina africana* populations by comparing the 1975 population dynamics found by Hart (1981) with the current 2021 study in Lake Sibaya.

It was hypothesised that increases in anthropogenic disturbances, invasive species and other habitat modification at Lake Sibaya and Lake Mzingazi would lead to (1) *Caridina africana* densities at

Lake Sibaya being lower with differing population dynamics in 2021 when compared to the historical dataset from Hart (1981) due to drastic landscape changes and habitat modification and (2) *Caridina africana* densities would be lower at Lake Mzingazi than at Lake Sibaya because of (3) differences in land use and the related differences in physicochemical parameters.

Caridina africana population densities at Lake Sibaya have decreased significantly since 1975 along with habitat modification and associated lake level drop and possibly the invasion of the invasive snail *Tarebia granifera*. Significant declines in *C. africana* with increases in nitrate concentration at Lake Sibaya could indicate the necessity of pollution mitigation within the system but considering the peatlands that surround the lake, it is more likely that the increasing nitrate concentration is a result of leaching from the peatlands combined with decreasing lake level and therefore, decreasing dilution capacity. This leads to the conclusion that water abstraction specifically in times of drought by commercial plantations and other climate-related stresses is the larger threat to the *C. africana* population of Lake Sibaya. The positive correlation of ammonium and phosphate concentration with *C. africana* abundance further supports this idea. Furthermore, the invasive *Tarebia granifera* population cannot be ignored as the competition created by the dietary overlap between it and *C. africana* is more than likely affecting the productivity of the shrimp (see section 4.1.4 below). The significant declines in *C. africana* abundance seen since 1975 at Lake Sibaya emphasize the importance of finding a solution to the increasing water demand from the lake as well as the explosive invasive *T. granifera* population to ensure the survival of lentic *Caridina africana* populations.

The medium-size classes of *C. africana* (3.33mm-5mm) were most abundant in 2021 at Lake Sibaya whereas the lowest-size class (0.83mm-2.5mm) was dominant in 1975. This difference could be related to predation pressure with declining fish stocks over time, leading to an increase in the size of prey or this could be a result of the sampling method and the limitations that arise through hand sorting of specimens when removed from the sampling bin. Males made up a larger contribution to the smaller-size classes than females did in 2021 unlike in 1975 which could be attributed to a higher number of indeterminate specimens found in 1975 when compared to the current study. This could also explain why males were overall more abundant than females in this study which contrasted the findings of Hart (1981). Females did, however, grow larger than males at Lake Sibaya in both 2021 and 1975. Abundances of berried females were lower in 2021 than in 1975, which is to be expected as population densities have decreased significantly. However, peaks in the frequency of berried females were experienced in July, September and October in both study years which shows that reproductive timing has not been affected by the surrounding land-use changes. Overall, populations in 2021 seemed to be driven more by mortality instead of lack of reproduction whereas in 1975 population mortality and instantaneous death rates were similar throughout the year. The high level of mortality seen may discount the possibility that declining predation pressure has resulted in larger-sized shrimp but the drivers of this high mortality would have to be studied more closely to make any conclusions regarding predation pressure and size-frequency distributions.

4.1.3. Did urbanisation at Lake Mzingazi compared with the agricultural stress at Lake Sibaya lead to different effects on *Caridina africana* populations?

Although *Caridina africana* abundances at Lake Mzingazi were significantly lower than the abundances found at Lake Sibaya in 2021, there was no clear indication that urbanisation and rural or agricultural land use presented different physicochemical stressors to freshwater *Caridina africana* populations. Temperature was the main differentiating factor between the two systems with the exceedance of the thermal tolerance limit of *C. africana* (see Hart, 1980) at Lake Mzingazi leading to significantly lower densities at temperatures found over 29°C. It is interesting to note the speed at which populations seem to begin recovering following decreases in temperature and further investigation into population recovery following thermal zone exceedance would clarify the long-term severity of this threat. The lower lake level of Lake Mzingazi when compared to Lake Sibaya would also likely result in less habitat available to *C. africana* populations, increasing competition and ultimately reducing densities in those areas.

Similar size-class and sex ratio frequency patterns were evident at Lake Sibaya and Lake Mzingazi with the medium-size classes being the most common and both the lowest and highest-size classes being the least common. Males dominated smaller size classes while females dominated the larger size classes at Lake Mzingazi as was found at Lake Sibaya. Lake Mzingazi's *C. africana* population also showed no distinct seasonality, as was found at Lake Sibaya, but peaks in abundance were seen in September at both lakes, likely to be a result of spring population increases typically experienced by aquatic invertebrate communities (Davies, 1982; Makherana et al., 2022). Populations at both lakes also experienced fluctuations from month to month throughout the year with Lake Sibaya fluctuating up and down on a monthly basis and Lake Mzingazi's populations experiencing a slow increase over time correlated with the temperature changes of the lake. These results further support the conclusion that the differences seen in *Caridina africana* densities between the lakes were more a result of water temperature as opposed to differing effects of urbanisation and agriculture.

In conclusion, endorheic systems rely largely on nutrient recyclers (such as *C. africana*) to prevent eutrophic conditions and the importance of monitoring these organisms specifically those with long-term historical studies and the traits to act as successful bioassessment tools, that perform important ecological roles should not be underestimated. These organisms are particularly important in systems known to experience fluctuations between trophic states (such as the move from oligo- to mesotrophic) and generally large variations in nutrient loads such as those shown in the study when compared to recent publications.

4.1.4. Did land-use activities structure the aquatic invertebrate communities of Lake Sibaya and Lake Mzingazi?

The aim of Chapter 3 was to quantify the littoral aquatic invertebrate diversity and assemblage patterns from Lake Sibaya and Lake Mzingazi over a 10-month period (March-December 2021) to form a comprehensive baseline dataset for use in future comparison. This chapter also aimed to investigate the impacts of landscape developments and habitat change on aquatic invertebrate communities by understanding significant water quality parameters as drivers of aquatic invertebrate community variation. It was hypothesised that increases in agricultural and anthropogenic disturbance and habitat modification will lead to (1) aquatic invertebrate community composition at lakes Sibaya and Mzingazi being structured according to water quality variables associated with surrounding land use which (2) would result in different communities at the two lakes. Additionally, it was hypothesised that (3) the invasive snail Tarebia granifera would likely be affecting the native aquatic invertebrate diversity of both lakes.

The hypothesis that aquatic invertebrate communities would be structured by physicochemical parameters related to land-use changes in surrounding catchments proved partially true. Significantly contributing variables to variation in aquatic invertebrate community compositions of Lake Sibaya and Lake Mzingazi were all variables known to be derived from activities related to both urbanisation and agriculture. There was, however, some overlap in variables despite the differing land-use changes occurring around Lake Sibaya and Lake Mzingazi. The main differences exhibited in the aquatic invertebrate communities of the lakes were the significance of phosphate vs ammonium concentrations at Lake Sibaya and Lake Mzingazi respectively and the lack of significant physicochemical parameters in driving community abundance and richness at Lake Mzingazi. This led to the conclusion that urbanisation and agricultural stress present similar threats to freshwater invertebrate communities and that inorganic phosphate and nitrogen concentrations and corresponding nutrient-ion variables such as conductivity and salinity should all be monitored closely within these systems to understand how we may mitigate the effects of pollution on freshwater systems in future and ensure the survival of aquatic invertebrate communities currently under threat. The finding of pollution-sensitive aquatic invertebrates (specifically *Caridina africana* and *Cloeon* sp.) as highly abundant taxa found at both lakes and being in higher abundance than the pollution-tolerant taxa (specifically Chironomidae) is, however, a positive sign for the health of both systems.

Additionally, it was suggested that these differences would lead to differing communities between the two lakes which was proven true in the finding of significant differences in the aquatic invertebrate community compositions of the two lakes. This finding provided evidence emphasising the success of national heritage sites in promoting and conserving aquatic invertebrate communities with Lake Sibaya situated in the protected iSimangaliso wetland park boasting higher aquatic invertebrate abundance and taxa richness than the unprotected Lake Mzingazi. These differences could also be attributed to the lack of clear seasonal patterns at Lake Mzingazi unlike the patterns seen at Lake Sibaya but the resilience of the aquatic communities of Lake Mzingazi to differing physicochemical conditions throughout the year does prove an interesting topic for further investigation.

Functional feeding group composition was similar to those of previous studies (see Miranda and Perissinotto, 2014; Farrell et al., 2015; Burger et al., 2018; Dalu et al., 2022) but also differed slightly as predators were previously found as dominant taxa (Farrell et al., 2015; Burger et al., 2018; Dalu et al., 2022) which was not the case in this study. The results of Miranda and Perissinotto (2014) from Lake Sibaya did, however, agree with these results where the dominant taxa (according to Bray-Curtis similarity measures) found in their study did not form a part of the predator FFG. There also seemed to be no seasonal component structuring functional feeding group patterns at either lake.

There was also a significant negative correlation between *T. granifera* abundance and the diversity of the native aquatic invertebrates found at Lake Sibaya and this was not the case at Lake Mzingazi where *T. granifera* did not dominate over the native community as it did at Lake Sibaya. The native snail *Bulinus* sp. being found in relatively high abundance is positive considering the abundance of the invasive *T. granifera* but the lack of any *Melanoides tuberculata* individuals in either system is worrying as it shows *T. granifera*'s ability to outcompete native Thiaridae snails in South African freshwater systems.

4.2. General conclusions

This dissertation provides 10-month baseline datasets covering all seasons for *Caridina africana* populations (12 months in the case of Lake Sibaya) as well as aquatic invertebrate communities of Lake Sibaya and Lake Mzingazi to aid in closing the gaps identified by the 2018 South African National Biodiversity Assessment (Skowno et al., 2019). Additionally, the results from this dissertation could aid in future projections and in-situ biological responses to global change in freshwater lake systems. This study showed the value of using historical studies to assess biological changes associated with global change but also showed the complexity of identifying specific drivers accounting for these changes. This dissertation provides an evidence-based case study on how lake level variation, increasing temperatures, invasive species and anthropogenic and agricultural pollution (specifically inorganic phosphorous and nitrogen) can be drivers of littoral freshwater aquatic invertebrate communities and how these variables should be monitored closely, specifically in vulnerable ecosystems. Smaller lake systems also seem more vulnerable to the physical effects of global warming with less adaptive capacity and further research on this topic is highly recommended as there does seem to be some form of resilience exhibited in terms of biological response to these stressors. Finally, the importance of mitigation efforts and further research into aquatic invertebrate responses to these global change impacts to preserve South African freshwater lake systems and the aquatic invertebrate biodiversity associated with them cannot be emphasized enough.

4.3. References

Abell, R., Thieme, M. L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S. C., Bussing, W., Stiassny, M. L. J., Skelton, P., Allen, G. R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J. V, Heibel, T. J., Wikramanayake, E., Olson, D., López, H. L., Reis, R. E., Lundberg, J. G., Sabaj Pérez, M. H. & Petry, P., 2008. Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. BioScience 58:403–414.

Adite, A. & Winemiller, K. O., 1997. Trophic ecology and ecomorphology of fish assemblages in coastal lakes of Benin, West Africa. Écoscience 4:6–23.

Aladin, N. V & Potts, W. T. W., 1995. Osmoregulatory capacity of the Cladocera. Journal of Comparative Physiology B 164:671–683.

Allanson, B. R., 1979. Lake Sibaya. Monographiae Biologicae, Vol 36. Dr W Junk, London, UK. 386 pp.

Allanson, B. R., Hart, R. C., O'Keeffe, J. H. & Robarts, R. D., 1990. Planktonic and benthic invertebrates - Inland Waters of Southern Africa: An Ecological Perspective. Monographiae Biologicae, Vol 64. Kluwer Academic Publisher Group, Dordrecht, Netherlands. 458 pp.

Altermatt, F., 2013. Diversity in riverine metacommunities: A network perspective. Aquatic Ecology 47:365–377.

American Veterinary Medical Association., 2020. AVMA Guidelines for the Euthanasia of Animals. Schaumburg, Illinois. 121 pp.

Anderson, M. J. & Walsh, D. C. I., 2013. PERMANOVA, ANOSIM, and the Mantel test in the face of heterogeneous dispersions: What null hypothesis are you testing? Ecological Monographs 83:557–574.

Appleton, C. C. & Nadasan, D. S., 2002. First report of *Tarebia granifera* (Lamarck, 1816) (Gastropoda: Thiaridae) from Africa. Journal of Molluscan Studies 68:399–402.

Appleton, C. C., Forbes, A. T. & Demetriades, N. T., 2009. The occurrence, bionomics and potential impacts of the invasive freshwater snail *Tarebia granifera* (Lamarck, 1822) (Gastropoda: Thiaridae) in South Africa. Zoologische Mededelingen, 83(4):525–536.

Appleton, C.C., 2002. Mollusca. In: De Moor, I.J. & Day, J.A. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 6: Arachnida and Mollusca. WRC Report No. TT 182/02. Water Research Commission, Pretoria, 141 pp.

Ashforth, D. & Yan, N. D., 2008. The interactive effects of calcium concentration and temperature on the survival and reproduction of *Daphnia pulex* at high and low food concentrations. Limnology and Oceanography 53:420–432.

Ashton, P. J., 2010. The demise of the Nile crocodile (*Crocodylus niloticus*) as a keystone species for aquatic ecosystem conservation in South Africa: The case of the Olifants River. Aquatic Conservation Marine and Freshwater Ecosystems 20:489–493.

Baker, N. J., Pilotto, F., Haubrock, P. J., Beudert, B. & Haase, P., 2021. Multidecadal changes in functional diversity lag behind the recovery of taxonomic diversity. Ecology and Evolution 11:17471–17484.

Barbarossa, V., Schmitt, R. J. P., Huijbregts, M. A. J., Zarfl, C., King, H. & Schipper, A. M., 2020. Impacts of current and future large dams on the geographic range connectivity of freshwater fish worldwide. Proceedings of the National Academy of Sciences 117(7): 3648-3655

Barber-James, H.M. & Lugo-Ortiz, C.R., 2003. Ephemeroptera. In: De Moor, I.J., Day, J.A. & De Moor, F.C., (eds), Guides to the Freshwater Invertebrates of Southern Africa: Volume 7 Insecta I—Ephemeroptera, Odonata and Plecoptera. WRC Report No. TT 207/03. Water Research Commission, Pretoria, 288 pp.

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O. U., Swartz, B., Quental, T. B., Marshall, C., McGuire, J. L., Lindsey, E. L., Maguire, K. C., Mersey, B. & Ferrera, E. A., 2011. Has the Earth's sixth mass extinction event already arrived? Nature 471:51–57.

Barton, K., 2022. MuMIn: Multi-Model Inference (1.47.1).

Bate, G., Mkhwanazi, M. & Simonis, J., 2018. The effects of population growth on the groundwater quality of a sandy rural catchment. Transactions of the Royal Society of South Africa 73:243–253.

Bates, D., Machler, M., Bolker, B. & Walker, S., 2015. Fitting Linear Mixed Models using {lme4}. Journal of Statistical Software 67:1–48.

Berezina, N. A., 2001. Influence of ambient pH on freshwater invertebrates under experimental conditions. Russian Journal of Ecology 32:343–351.

Bird, M. S., Bilton, D. T. & Perissinotto, R., 2017. Diversity and distribution of polyphagan water beetles (Coleoptera) in the Lake St Lucia system, South Africa. ZooKeys 656:51-84.

Bird, M.S., Mlambo, M.C. & Day, J.A., 2013. Macroinvertebrates as unreliable indicators of human disturbance in temporary depression wetlands of the south-western Cape, South Africa. Hydrobiologia 720, 19–37.

Biström, O., 2007. Dytiscidae. In: Stals, R. & de Moor, I. J. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 10: Coleoptera. WRC Report No. TT 320/07. Water Research Commission, Pretoria. 262 pp.

Blaber, S. J. M. & Whitfield, A. K., 1977. The biology of the burrowing goby *Croilia mossambica* Smith (Teleostei, Gobiidae). Environmental Biology of Fishes 1:197–204.

Blinn, D., Halse, S., Pinder, A. & Shiel, R., 2004. Diatom and micro-invertebrate communities and environmental determinants in the western Australian wheatbelt: a response to salinization. Hydrobiologia 528:229–248.

Bowen, S. H., 1979. A Nutritional Constraint in Detritivory by Fishes: The Stunted Population of *Sarotherodon mossambicus* in Lake Sibaya, South Africa. Ecological Monographs 49:17–31.

Bownes, S. J. & McQuaid, C. D., 2006. Will the invasive mussel *Mytilus galloprovincialis* Lamarck replace the indigenous *Perna perna* L. on the south coast of South Africa? Journal of Experimental Marine Biology and Ecology 338:140–151.

Boyero, L., López-Rojo, N., Tonin, A. M., Pérez, J., Correa-Araneda, F., Pearson, R. G., Bosch, J., Albariño, R. J., Anbalagan, S., Barmuta, L. A., Basaguren, A., Burdon, F. J., Caliman, A., Callisto, M., Calor, A. R., Campbell, I. C., Cardinale, B. J., Jesús Casas, J., Chará-Serna, A. M., Chauvet, E., Ciapała, S., Colón-Gaud, C., Cornejo, A., Davis, A. M., Degebrodt, M., Dias, E. S., Díaz, M. E., Douglas, M. M., Encalada, A. C., Figueroa, R., Flecker, A. S., Fleituch, T., García, E. A., García, G., García, P. E., Gessner, M. O., Gómez, J. E., Gómez, S., Gonçalves, J. F., Graça, M. A. S., Gwinn, D. C., Hall, R. O., Hamada, N., Hui, C., Imazawa, D., Iwata, T., Kariuki, S. K., Landeira-Dabarca, A., Laymon, K., Leal, M., Marchant, R., Martins, R. T., Masese, F. O., Maul, M., McKie, B. G., Medeiros, A. O., Erimba, C. M. M., Middleton, J. A., Monroy, S., Muotka, T., Negishi, J. N., Ramírez, A., Richardson, J. S., Rincón, J., Rubio-Ríos, J., Dos Santos, G. M., Sarremejane, R., Sheldon, F., Sitati, A., Tenkiano, N. S. D., Tiegs, S. D., Tolod, J. R., Venarsky, M., Watson, A. & Yule, C. M., 2021. Impacts of detritivore diversity loss on instream decomposition are greatest in the tropics. Nature Communications 12 (1): 3700.

Bredenhand, E. & Samways, M. J., 2009. Impact of a dam on benthic macroinvertebrates in a small river in a biodiversity hotspot: Cape Floristic Region, South Africa. Journal of Insect Conservation 13: 297–307.

Browne, M. A., Dissanayake, A., Galloway, T. S., Lowe, D. M. & Thompson, R. C., 2008. Ingested Microscopic Plastic Translocates to the Circulatory System of the Mussel, *Mytilus edulis* (L.). Environmental Science & Technology 42: 5026–5031.

Bruton, M. N., 1978. The habitats and habitat preferences of *Clarias Gariepinus* (Pisces: Clariidae) in a clear coastal lake (Lake Sibaya, South Africa). Journal of the Limnological Society of Southern Africa 4: 81–88.

Buah-Kwofie, A. & Humphries, M. S., 2017. The distribution of organochlorine pesticides in sediments from iSimangaliso Wetland Park: Ecological risks and implications for conservation in a biodiversity hotspot. Environmental Pollution 229: 715–723.

Burger, M., Van Vuren, J. H. J., De Wet, L. & Nel, A., 2018. A comparison of water quality and macroinvertebrate community structure in endorheic depression wetlands and a salt pan in the Gauteng province, South Africa. Environmental Monitoring and Assessment 191: 14.

Burkle, L.A., Marlin, J.C. & Knight, T.M., 2013. Plant-pollinator interactions over 120 years: Loss of apecies, co-occurrence, and function. Science 339:1611-1615.

Campbell, S. H., Williamson, P. R. & Hall, B. D., 2017. Microplastics in the gastrointestinal tracts of fish and the water from an urban prairie creek. FACETS 2:395–409.

Cardinale, B. J., Nelson, K. & Palmer, M. A., 2000. Linking species diversity to the functioning of ecosystems: on the importance of environmental context. Oikos 91:175–183.

Carlisle, D.M. & Clements, W.H., 1999. Sensitivity and variability of metrics used in biological assessments of running waters. Environmental Toxicology and Chemistry, 18: 285-291.

Carnie, T., 2020. The Slow Death of Lake Sibaya. The Water Wheel 19(2): 22-27.Carpenter, S. R., Stanley, E. H. & Vander Zanden, M. J., 2011. State of the World's Freshwater Ecosystems: Physical, Chemical, and Biological Changes. Annual Review of Environment and Resources 36:75–99.

Carter, T.R., Jones, R.N., Lu, X., Bhadwal, S., Conde, C., Mearns, L.O., O' Neil, B.C., Rounsevell, M.D.A & Zurek, M.B., 2007. New methods and the characterisation of future conditions. In: Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. M. L. Parry, O.

F. Canziani, P. Van der Linden, & C. E. Hanson (eds). Cambridge University Press, Cambridge, UK: 133-171.

Castañeda, R. A., Avlijas, S., Simard, M. A. & Ricciardi, A., 2014. Microplastic pollution in St. Lawrence River sediments. Canadian Journal of Fisheries and Aquatic Sciences 71:1767–1771.

Chamier, J., Schachtschneider, K., Le Maitre, D. C., Ashton, P. J. & Van Wilgen, B. W., 2012. Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. Water SA 38:345–356.

Coetzee, J. A., Jones, R. W., & Hill, M. P., 2014. Water hyacinth*, Eichhornia crassipes* (Pontederiaceae), reduces benthic macroinvertebrate diversity in a protected subtropical lake in South Africa. Biodiversity and Conservation 23(5): 1319–1330.

Combrink, X., Korrûbel, J. L., Kyle, R., Taylor, R. & Ross, P., 2011. Evidence of a declining Nile crocodile *(Crocodylus niloticus*) population at Lake Sibaya, South Africa. South African Journal of Wildlife Research 41:145–157.

Covich, A. P., Palmer, M. A. & Crowl, T. A., 1999. The Role of benthic invertebrate species in freshwater ecosystems: Zoobenthic species influence energy flows and nutrient cycling. BioScience 49:119–127.

Cummins, K. W., Costa, R. R., Rowe, R. E., Moshiri, G. A., Scanlon, R. M. & Zajdel, R. K., 1969. Ecological energetics of a natural population of the predaceous zooplankter *Leptodora kindtii Focke* (Cladocera). Oikos 20(2): 189–223.

Cummins, K. W., Merritt, R. W., & Andrade, P. C., 2005. The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in south Brazil. Studies on Neotropical Fauna and Environment 40(1): 69–89.

Dallas, H. F. & Day, J. A., 2004. The effect of water quality variables on aquatic ecosystems : A review. WRC Report No. TT 224/04. Water Research Commission, Pretoria. 222 pp.

Dallas, H. F. & Rivers-Moore, N., 2014. Ecological consequences of global climate change for freshwater ecosystems in South Africa. South African Journal of Science 110 (5-6): 01-11.

Dalu, T. & Chauke, R., 2019. Assessing macroinvertebrate communities in relation to environmental variables: the case of Sambandou wetlands, Vhembe Biosphere Reserve. Applied Water Science 10(1): 1-11.

Dalu, T. & Froneman, P. W., 2016. Diatom-based water quality monitoring in southern Africa: Challenges and future prospects. Water SA 42:551–559.

Dalu, T., Clegg, B. W. & Nhiwatiwa, T., 2013. A study of the ichthyofauna of a small tropical reservoir, south-eastern lowveld, Zimbabwe. African Journal of Aquatic Science 38:105–113.

Dalu, T., Cuthbert, R. N., Methi, M. J., Dondofema, F., Chari, L. D. & Wasserman, R. J., 2022. Drivers of aquatic macroinvertebrate communities in a RAMSAR declared wetland system. Science of The Total Environment 818:151683.

Dalu, T., Dlamini, P., Wasserman, R. J., Mokgoebo, M. J., Mutshekwa, T., Dondofema, F. & Cuthbert, R. N., 2021. Effects of environmental variables on littoral macroinvertebrate community assemblages in subtropical reservoirs. Chemistry and Ecology 37:419–436.

Dalu, T., Wasserman, R. J., Tonkin, J. D., Alexander, M. E., Dalu, M. T. B., Motitsoe, S. N., Manungo, K. I., Bepe, O. & Dube, T., 2017. Assessing drivers of benthic macroinvertebrate community structure in African highland streams: An exploration using multivariate analysis. Science of The Total Environment 601–602:1340–1348.

Damschen, E.I., Harrison, S. & Grace, J.B., 2010. Climate change effects on an endemic-rich edaphic flora: resurveying Robert H. Whittaker's Siskiyou sites (Oregon, USA). Ecology, 91: 3609-3619.

Darwall, W., Smith, K., Allen, D., Holland, R., Harrison, I., & Brooks, E., 2011. The diversity of life in African freshwaters: underwater, under threat: an analysis of the status and distribution of freshwater species throughout mainland Africa. The World Conservation Union, Gland, Switzerland. xii + 348 pp.

Davies, B. R., 1982. Studies on the zoobenthos of some southern Cape coastal lakes. Spatial and temporal changes in the benthos of Swartvlei, South Africa, in relation to changes in the submerged littoral macrophyte community. Journal of the Limnological Society of Southern Africa 8:33–45.

Davies, G. M. & Gray, A., 2015. Don’t let spurious accusations of pseudoreplication limit our ability to learn from natural experiments (and other messy kinds of ecological monitoring). Ecology and Evolution 5:5295–5304.

Dawson, J., Pillay, D., Roberts, P. J. & Perissinotto, R., 2016. Declines in benthic macroinvertebrate community metrics and microphytobenthic biomass in an estuarine lake following enrichment by hippo dung. Scientific reports 6:1–8.

De Grave, S., Smith, K. G., Adeler, N. A., Allen, D. J., Alvarez, F., Anker, A., Cai, Y., Carrizo, S. F., Klotz, W., Mantelatto, F. L., Page, T. J., Shy, J.-Y., Villalobos, J. L. & Wowor, D., 2015. Dead shrimp blues: A global assessment of extinction risk in freshwater shrimps (Crustacea: Decapoda: Caridea). PLOS ONE 10:e0120198.

De Meillon, B. & Wirth, W.W., 2003. Ceratopognidae. In: Day, J.A., Harrison, A.D. & de Moor, I.J. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 9: Diptera. WRC Report No. TT 201/02. Water Research Commission, Pretoria. 199 pp.

de Necker, L., Brendonck, L., van Vuren, J. Wepener, V. & Smit, N. J., 2021. Aquatic invertebrate community resilience and recovery in response to a supra-seasonal drought in an ecologically important naturally saline lake. Water 13(7): 948.

de Necker, L., Ferreira, M., Van Vuren, J. H. J. & Malherbe, W., 2016. Aquatic invertebrate community structure of selected endorheic wetlands (pans) in South Africa. Inland Waters 6:303–313.

De Vos, L., Snoeks, J. & van den Audenaerde, D.T., 1990. The effects of *Tilapia* introductions in Lake Luhondo, Rwanda. Environmental Biology of Fishes 27: 303–308.

Department of Water Affairs and Forestry (DWAF)., 1996. South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems. Department of Water Affairs and Forestry, Pretoria.

Dickens, C. W. S. & Graham, P. M., 2002. The South African Scoring System (SASS) Version 5 rapid bioassessment method for rivers. African Journal of Aquatic Science 27:1–10.

Dippenaar, S., 2002. Araneae. In: De Moor, I.J. & Day, J.A. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 6: Arachnida and Mollusca. WRC Report No. TT 182/02. Water Research Commission, Pretoria, 141 pp.

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z.-I., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A.-H., Soto, D., Stiassny, M. L. J. & Sullivan, C. A., 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81:163–182.

Edwards, B. A., Jackson, D. A. & Somers, K. M., 2016. Effects of declining calcium availability on the survival, growth and calcium content of a freshwater crayfish, *Orconectes virilis*. Freshwater Biology 61:914–922.

Eerkes-Medrano, D., Thompson, R. C. & Aldridge, D. C., 2015. Microplastics in freshwater systems: A review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. Water Research 75:63–82.

Ellender, B. R. & Weyl, O. L. F., 2014. A review of current knowledge, risk and ecological impacts associated with non-native freshwater fish introductions in South Africa. Aquatic Invasions 9(2): 117-132.

Endrödy-Younga, S. & Stals, R., 2007. Haliplidae. In: Stals, R. & de Moor, I. J. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 10: Coleoptera. WRC Report No. TT 320/07. Water Research Commission, Pretoria. 262 pp.

Fang, J., Wang, Z., Zhao, S., Li, Y., Tang, Z., Yu, D., Ni, L., Liu, H., Xie, P., Da, L., Li, Z. & Zheng, C., 2006. Biodiversity changes in the lakes of the Central Yangtze. Frontiers in Ecology and the Environment 4:369–377.

Farrell, K., Van Vuren, J. H. J. & Ferreira, M., 2015. Do aquatic macroinvertebrate communities respond to land-use effects in the Wilge River, Mpumalanga, South Africa? African Journal of Aquatic Science 40:165–173.

Ferreira, M., Wepener, V. & Van Vuren, J. H. J., 2012. Aquatic invertebrate communities of perennial pans in Mpumalanga, South Africa: A diversity and functional approach. African Invertebrates 53:751–768.

Finlayson, C. M., Davis, J. A., Gell, P. A., Kingsford, R. T. & Parton, K. A., 2013. The status of wetlands and the predicted effects of global climate change: The situation in Australia. Aquatic Sciences 75:73–93.

Firmiano, K. R., Castro, D. M. P., Linares, M. S. & Callisto, M., 2021. Functional responses of aquatic invertebrates to anthropogenic stressors in riparian zones of neotropical savanna streams. Science of The Total Environment 753:141865.

FishBase team RMCA & Geelhand, D., 2016. *Barbus microbarbis*. The IUCN Red List of Threatened Species 2016:e.T61247A47242030. https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T61247A47242030.en. Accessed on 10 October 2023.

Foster, L., Malherbe, W., Ferreira, M. & Van Vuren, J. H. J., 2015. Macroinvertebrate variation in endorheic depression wetlands in North West and Mpumalanga provinces, South Africa. African Journal of Aquatic Science 40:287–297.

Fouchy, K., McClain, M. E., Conallin, J. & O'Brien, G., 2019. Multiple stressors in African freshwater systems. In: Sergi Sabater, Arturo Elosegi, Ralf Ludwig (eds), Multiple Stressors in River Ecosystems, Elsevier: 179-191.

Fowles, B. K. & Archibald, C. G. M., 1987. Zooplankton and estuarine relict (benthic) fauna in Lake Mzingazi, a freshwater coastal lake of Natal. Journal of the Limnological Society of Southern Africa 13:66–74.

Fox, J. & Weisberg, S., 2019. An {R} Companion to Applied Regression. 3rd Ed. Sage.

Fry, C. 2021. Field Guide to the Freshwater Macroinvertebrates of Southern Africa. Jacana Media (Pty) Limited, Sunnyside, Johannesburg. 212 pp.

Gaeta, J. W., Sass, G. G. & Carpenter, S. R., 2014. Drought-driven lake level decline: effects on coarse woody habitat and fishes. Canadian Journal of Fish and Aquatic Sciences 71:315–325.

Galindo-Reyes, J. G., Fossato, V. U., Villagrana-Lizarraga, C. & Dolci, F., 1999. Pesticides in water, sediments, and shrimp from a coastal lagoon on the gulf of California. Marine Pollution Bulletin 38:837–841.

Gallardo, B., Clavero, M., Sánchez, M.I. & Vilà, M., 2016. Global ecological impacts of invasive species in aquatic ecosystems. Global Change Biology 22: 151-163.

Goldschmidt, T., Witte, F. & Wanink, J., 1993. Cascading effects of the introduced Nile Perch on the detritivorous/ phytoplanktivorous species in the sublittoral areas of Lake Victoria. Conservation Biology 7:686–700.

Griffiths, C. & Stewart, B.A., 2001. Amphipoda. In: Day, J.A., Stewart, B.A., de Moor, I.J. & Louw, A.E. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 4: Crustacea III. Bathynellacea, Amphipoda, Isopoda, Spelaeogriphacea, Tanaidacea and Decapoda. WRC Report No. TT 141/01. Water Research Commission, Pretoria. 141 pp.

Grill, G., Lehner, B., Lumsdon, A. E., Zarfl, C., MacDonald, G. K. & Reidy Liermann, C., 2015. An index-based framework for assessing patterns and trends in river fragmentation and flow regulation by global dams at multiple scales. Environmental Research Letters 10(1): 015001.

Grundling, A. T., Van den Berg, E. C. & Price, J. S., 2013. Assessing the distribution of wetlands over wet and dry periods and land-use change on the Maputaland coastal plain, north-eastern KwaZulu-Natal, South Africa. South African Journal of Geomatics 2:120–138.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R. & Watson, R., 2008. A global map of human impact on marine ecosystems. Science 319:948–952.

Han, X., Zhang, F., Meng, L., Xu, Y., Li, Y., Li, A., Turyk, M. E., Yang, R., Wang, P., Zhang, J., Zhang, Q. & Jiang, G., 2020. Exposure to organochlorine pesticides and the risk of type 2 diabetes in the population of East China. Ecotoxicology and Environmental Safety 190:110e125.

Hansen, G. J. A., Read, J. S., Hansen, J. F. & Winslow, L. A., 2017. Projected shifts in fish species dominance in Wisconsin lakes under climate change. Global Change Biology 23:1463–1476.

Harrison, A.D., 2003. Chironomidae. In: Day, J.A., Harrison, A.D. & de Moor, I.J. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 9: Diptera. WRC Report No. TT 201/02. Water Research Commission, Pretoria. 199 pp.

Hart, L. A., Bowker, M. B., Tarboton, W. & Downs, C. T., 2014. Species composition, distribution and habitat types of Odonata in the iSimangaliso Wetland Park, KwaZulu-Natal, South Africa and the associated conservation implications. PLoS One 9:e92588.

Hart, R. C. & Appleton, C. C., 1997. A limnological synopsis of Bhangazi South, a dystrophic coastal lake in the greater St Lucia Wetland Park (Kwazulu-Natal), with comments on its conservation value. Southern African Journal of Aquatic Science 23(2): 34-54.

Hart, R. C., 1980. Embryonic duration and post-embryonic growth rates of the tropical freshwater shrimp *Caridina nilotica* (Decapoda:Atyidae) under laboratory and experimental field conditions. Freshwater Biology 10:297–315.

Hart, R. C., 1981. Population dynamics and production of the tropical freshwater shrimp *Caridina nilotica* (Decapoda: Atyidae) in the littoral of Lake Sibaya. Freshwater Biology 11:531–547.

Hart, R. C., 1985. Temperature tolerances and southern African distribution of tropical freshwater shrimp *Caridina nilotica* (Decapoda: Atyidae). South African Journal of Zoology 18(2): 67–70.

Hart, R., Stewart, B.A. & Bickerton, I., 2001. Decapoda. In: Day, J.A., Stewart, B.A., de Moor, I.J. & Louw, A.E. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 4: Crustacea III. Bathynellacea, Amphipoda, Isopoda, Spelaeogriphacea, Tanaidacea and Decapoda. WRC Report No. TT 141/01. Water Research Commission, Pretoria. 141 pp.

Henman, J. & Poulter, B., 2008. Inundation of freshwater peatlands by sea level rise: Uncertainty and potential carbon cycle feedbacks. Journal of Geophysical Research: Biogeosciences 113: G01011.

Hickley, P., Muchiri, M., Boar, R., Britton, R., Adams, C., Gichuru, N. & Harper, D., 2004. Habitat degradation and subsequent fishery collapse in Lakes Naivasha and Baringo, Kenya. Ecohydrology and Hydrobiology 4(4):503-517.

Hodkinson, I.D., Jackson, J.K., 2005. Terrestrial and aquatic invertebrates as bioindicators for environmental monitoring, with particular reference to mountain ecosystems. Environmental Management 35: 649–666.

Hofmann, M.M., Fleischmann, A. & Renner, S.S., 2018. Changes in the bee fauna of a German botanical garden between 1997 and 2017, attributable to climate warming, not other parameters. Oecologia 187: 701–706.

Holland, E. R., Mallory, M. L. & Shutler, D., 2016. Plastics and other anthropogenic debris in freshwater birds from Canada. Science of The Total Environment 571:251–258.

Horak, I., Horn, S. & Pieters, R., 2021. Agrochemicals in freshwater systems and their potential as endocrine disrupting chemicals: A South African context. Environmental Pollution 268:115718.

Horne, A. J. & Goldman, C. R., 1994. Lake Ecology Overview. Limnology (2nd edition). McGraw-Hill Science/Engineering/Math, New York, USA. 480 pp.

Howard-Williams, Davies, C. & Davies, B. R., 1979. The rates of dry matter and nutrient loss from decomposing *Potamogeton pectinatus* in a brackish south-temperate coastal lake. Freshwater Biology 9:13–21.

Humphries, M. S. & Benitez-Nelson, C. R., 2013. Recent trends in sediment and nutrient accumulation rates in coastal, freshwater Lake Sibaya, South Africa. Marine and Freshwater Research 64:1087–1099.

Humphries, M. S., 2013. DDT residue contamination in sediments from Lake Sibaya in northern KwaZulu-Natal, South Africa: Implications for conservation in a World Heritage Site. Chemosphere 93(8): 1494-1499.

Hunt, P. C. & Jones, J. W., 1972. The effect of water level fluctuations on a littoral fauna. Journal of Fish Biology 4:385–394.

IUCN., 2023. The IUCN Red List of Threatened Species. Version 2022-2. https://www.iucnredlist.org. Accessed 10 October 2023.

Jackson, M. C., Woodford, D. J. & Weyl, O. L. F., 2016. Linking key environmental stressors with the delivery of provisioning ecosystem services in the freshwaters of southern Africa. Geography and Environment 3:e00026.

Jacobsen, P. R., Axelstad, M., Boberg, J., Isling, L. K., Christiansen, S., Mandrup, K. R., Berthelsen, L. O., Vinggaard, A. M. & Hass, U., 2012. Persistent developmental toxicity in rat offspring after low dose exposure to a mixture of endocrine disrupting pesticides. Reproductive Toxicology 34:237–250.

Jansen Van Rensburg, G., Bervoets, L., Smit, N. J., Wepener, V. & Van Vuren, J., 2019. Biomarker responses in the freshwater shrimp *Caridina nilotica* as indicators of persistent pollutant exposure. Bulletin of Environmental Contamination and Toxicology 104: 193-199.

Jansen Van Rensburg, G., Wepener, V., Horn, S. & Greenfield, R., 2022. Oxidative stress in the freshwater shrimp *Caridina africana* following exposure to atrazine. Bulletin of Environmental Contamination and Toxicology 109(3): 443-449.

Jeziorski, A. & Smol, J. P., 2016. The ecological impacts of lakewater calcium decline on softwater boreal ecosystems. Environmental Reviews 25:245–253.

Jeziorski, A. & Yan, N. D., 2006. Species identity and aqueous calcium concentrations as determinants of calcium concentrations of freshwater crustacean zooplankton. Canadian Journal of Fisheries and Aquatic Sciences 63:1007–1013.

Johnson, R. K., Furse, M. T., Hering, D. & Sandin, L., 2007. Ecological relationships between stream communities and spatial scale: Implications for designing catchment-level monitoring programmes. Freshwater Biology 52:939–958.

Jones, R. W., 2014. Aquatic invasions of the Nseleni River system: Causes, consequences and control. Rhodes University. PhD Thesis. 179 pp.

Jones, R. W., Hill, J. M., Coetzee, J. A., Avery, T. S., Weyl, O. L. F. & Hill, M. P., 2017. The abundance of an invasive freshwater snail *Tarebia granifera* (Lamarck, 1822) in the Nseleni River, South Africa. African Journal of Aquatic Science 42:75–81.

Juracek, K. E., 2015. The aging of America's reservoirs: In-reservoir and downstream physical changes and habitat implications. Journal of the American Water Resources Association 51:168–184.

Kass, L., Gomez, A. L. & Altamirano, G. A., 2020. Relationship between agrochemical compounds and mammary gland development and breast cancer. Molecular Cell Endocrinology 508:110e789.

Kensley, B., 1981., On the Zoogeography of southern African Decapod Crustacea, with a distributional checklist of the species (338th edition). Smithsonian Institution Press, City of Washington, USA. 64 pp.

Kensley, B., 2001. Isopoda. In: Day, J.A., Stewart, B.A., de Moor, I.J. & Louw, A.E. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 4: Crustacea III. Bathynellacea, Amphipoda, Isopoda, Spelaeogriphacea, Tanaidacea and Decapoda. WRC Report No. TT 141/01. Water Research Commission, Pretoria. 141 pp.

Ketse, N., 2006. The effects of selected reference toxicants on embryonic development of the freshwater shrimp *Caridina Nilotica* (Decapoda: Atyidae). Rhodes University. Master's Thesis. 135 pp.

Kock, A., Taylor, J. C. & Malherbe, W., 2019. Diatom community structure and relationship with water quality in Lake Sibaya, KwaZulu-Natal, South Africa. South African Journal of Botany 123:161–169.

Kraemer, B.M., Anneville, O., Chandra, S., Dix, M., Kuusisto, E., Livingstone, D.M., Rimmer, A., Schladow, S.G., Silow, E., Sitoki, L.M. & Tamatamah, R., 2015. Morphometry and average temperature affect lake stratification responses to climate change. Geophysical Research Letters 42:4981–4988.

Kratina, P., Greig, H. S., Thompson, P. L., Carvalho-Pereira, T. S. A. & Shurin, J. B., 2012. Warming modifies trophic cascades and eutrophication in experimental freshwater communities. Ecology 93:1421–1430.

Kumar, D. K., Dobriyal, A. & Rana, A., 2023. Aquatic Invertebrates Serve As An Effective Tool For Monitoring and Assessing Water Quality. In: Rawat, A. (ed), Water Science & Technology. ABS Books, Nudh Vihar Phase-2, Delhi: 11-20.

Kundzewicz, Z. W. & Döll, P., 2009. Will groundwater ease freshwater stress under climate change? Hydrological Sciences Journal 54:665–675.

Larson, M. D., Dewey, J. C. & Krist, A. C., 2020. Invasive *Potamopyrgus antipodarum* (New Zealand mud snails) and native snails differ in sensitivity to specific electrical conductivity and cations. Aquatic Ecology 54:103–117.

Laws, A. N. 2017., Climate change effects on predator–prey interactions. Current Opinion in Insect Science 23:28–34.

Lee, Y., Byeon, E., Kim, D. H., Maszczyk, P., Wang, M., Wu, R. S. S., Jeung, H. D., Hwang, U. K. & Lee, J. S., 2023. Hypoxia in aquatic invertebrates: Occurrence and phenotypic and molecular responses. Aquatic Toxicology 263:106685.

Leigh, C., Stubbington, R., Sheldon, F. & Boulton, A. J., 2013. Hyporheic invertebrates as bioindicators of ecological health in temporary rivers: A meta-analysis. Ecological Indicators 32:62–73.

Lenat, D. R. 1988., Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7:222–233.

Loreau, M. 2000., Biodiversity and ecosystem functioning: recent theoretical advances. Oikos 91:3–17.

Mabidi, A., Bird, M. S. & Perissinotto, R., 2018. Increasing salinity drastically reduces hatching success of crustaceans from depression wetlands of the semi-arid Eastern Cape Karoo region, South Africa. Scientific Reports 8:5983.

Mackay, C. F. & Cyrus, D. P., 2001. Available information on macroinvertebrates of the Mhlathuze coastal lakes: Setting the ecological reserve (lake water requirements). African Journal of Aquatic Science 26:85–98.

MacKay, F., Cyrus, D., & Russell, K. L., 2010. Macrobenthic invertebrate responses to prolonged drought in South Africa's largest estuarine lake complex. Estuarine, Coastal and Shelf Science 86(4): 553–567.

Mainka, S. A. & Howard, G. W., 2010. Climate change and invasive species: Double jeopardy. Integrative Zoology 5:102–111.

Majdi, N., De Necker, L., Fourie, H., Loggenberg, A., Netherlands, E. C., Bunte-Tschikin, J., Traunspurger, W. & Du Preez, G. C., 2022. Diversity and distribution of benthic invertebrates dwelling rivers of the Kruger National Park, South Africa. Koedoe 64(1): 1-18.

Makherana, F., Cuthbert, R. N., Dondofema, F., Wasserman, R. J., Chauke, G. M., Munyai, L. F. & Dalu, T. 2022., Distribution, drivers and population structure of the invasive alien snail *Tarebia granifera* in the Luvuvhu system, South Africa. River Research and Applications 38(8): 1362-1373.

Mallory, M. L., Blancher, P. J., Weatherhead, P. J. & McNicol, D. K., 1994. Presence or absence of fish as a cue to macroinvertebrate abundance in boreal wetlands. Hydrobiologia 279:345–351.

Mallory, S. J. L., 2000. Appendix B. In: Steyl, I., Versfeld, D.B. and Nelson, P.J (eds), Strategic environmental assessment for water use. Mhlathuze Catchment – KZN. Water Resource Assessment Report No. SEA 01/2000. Department of Water Affairs and Forestry, Pretoria. 16 pp.

Mantel, S. K., Hughes, D. A. & Muller, N. W., 2010. Ecological impacts of small dams on South African rivers Part 1: Drivers of change – Water quantity and quality. Water SA 36:351–360.

Marshall, B. E., 1978. Aspects of the ecology of benthic fauna in Lake McIlwaine, Rhodesia. Freshwater Biology 8:241–249.

Mathenjwa, C. M., 2006. Evaluation of drinking water quality in Lake Mzingazi in Richards Bay. University of KwaZulu-Natal. Master's Thesis. 138 pp.

Matthews, M. W. 2014., Eutrophication and cyanobacterial blooms in South African inland waters: 10 years of MERIS observations. Remote Sensing of Environment 155:161–177.

McLachlan, A. J. & McLachlan, S. M., 1971. Benthic fauna and sediments in the newly created Lake Kariba (Central Africa). Ecology 52:800–809.

Mead, A., 2011. Climate and bioinvasives: Drivers of change on South African rocky shores? University of Cape Town. PhD Thesis. 304 pp.

Merritt, R. W., Wallace, J. R., Higgins, M. J., Alexander, M. K., Berg, M. B., Morgan, W. T., Cummins, K. W. & Vandeneeden, B., 1996. Procedures for the functional analysis of invertebrate communities of the Kissimmee River-floodplain ecosystem. Florida Scientist 59:216–274.

Millennium Ecosystem Assessment., 2005. Ecosystems and human well-being: Wetlands and water synthesis. World Resources Institute, Washington, DC. 68 pp.

Mills, K., Gell, P., Gergis, J., Baker, P. J., Finlayson, C. M., Hesse, P. P., Jones, R., Kershaw, P., Pearson, S., Treble, P. C., Barr, C., Brookhouse, M., Drysdale, R., McDonald, J., Haberle, S., Reid, M., Thoms, M. & Tibby, J., 2013. Paleoclimate studies and natural-resource management in the Murray-Darling Basin II: Unravelling human impacts and climate variability. Australian Journal of Earth Sciences 60:561–571.

Miranda, N. A. F. & Perissinotto, R., 2014. Effects of an alien invasive gastropod on native benthic assemblages in coastal lakes of the iSimangaliso Wetland Park, South Africa: Ecology. African Invertebrates 55:209–228.

Miranda, N. A. F., Perissinotto, R. & Appleton, C. C., 2011. Population structure of an invasive parthenogenetic gastropod in coastal lakes and estuaries of Northern KwaZulu-Natal, South Africa. PLOS ONE 6:e24337.

Miranda, N., Renzo, P. & Appleton, C., 2010. Salinity and temperature tolerance of the invasive freshwater gastropod *Tarebia granifera* : Research article. South African Journal of Science 106:1–7.

Mlambo, M. C., Bird, M. S., Reed, C. C. & Day, J. A., 2011. Diversity patterns of temporary wetland macroinvertebrate assemblages in the south-western Cape, South Africa. African Journal of Aquatic Science 36:299–308.

Moloi, P. C. 2012., The Mzingazi gauging weir and its effects on the fish and macrocrustacean communities of Lake Mzingazi. University of Zululand. Master’s Thesis. 172 pp.

Molot, L. A. & Dillon, P. J., 2008. Long-term trends in catchment export and lake concentrations of base cations in the Dorset study area, central Ontario. Canadian Journal of Fisheries and Aquatic Sciences 65: 809–820.

Moritz, C., Patton, J.L., Conroy, C.J., Parra, J.L., White, G.C. & Beissinger, S.R., 2008. Impact of a century of climate change on small-mammal communities in Yosemite National Park, USA. Science 322: 261-264.

Moulton, T. P., Souza, M. L., Brito, E. F., Rosário Almeida Braga, M. & Bunn, S. E., 2012. Strong interactions of *Paratya australiensis* (Decapoda : Atyidae) on periphyton in an Australian subtropical stream. Marine and Freshwater Research 63:834–844.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A. B. & Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403:853–858.

Nakano, S. & Murakami, M., 2001. Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs. Proceedings of the National Academy of Sciences 98:166–170.

Ndlela, L. L., Oberholster, P. J., Wyk, J. H. Van & Cheng, P. H., 2016. An overview of cyanobacterial bloom occurrences and research in Africa over the last decade. Harmful Algae 60:11–26.

Nhiwatiwa, T., Dalu, T. & Brendonck, L., 2017. Impact of irrigation based sugarcane cultivation on the Chiredzi and Runde Rivers quality, Zimbabwe. Science of The Total Environment 587–588:316–325.

O'Connor, M. I., Piehler, M. F., Leech, D. M., Anton, A. & Bruno, J. F., 2009. Warming and resource availability shift food web structure and metabolism. PLoS Biology 7:e1000178.

Ockenden, A., Tremblay, L. A., Dikareva, N. & Simon, K. S., 2021. Towards more ecologically relevant investigations of the impacts of microplastic pollution in freshwater ecosystems. Science of The Total Environment 792:148507.

Odum, W. E., 1970. Pathways Of Energy Flow In A South Florida Estuary. University of Miami. PhD Dissertation. 205 pp.

Oksanen, J., Simpson, G., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., O'Hara, R., Solymos, P., Stevens, M., Szoecs, E., Wagner, H., Barbour, M., Bedward, M., Bolker, B., Borcard, D., Carvalho, G., Chirico, M., De Caceres, M., Durand, S., Evangelista, H., FitzJohn, R., Friendly, M., Furneaux, B., Hannigan, G., Hill, M., Lahti, L., McGlinn, D., Ouellette, M., Cunha, R., Smith, T., Stier, A., Ter Braak, C. & Weedon, J., 2022. Vegan: Community Ecology Package.

Oosthuizen, J. H. & Siddall, M. E., 2002. Hirudinea. In: De Moor, I.J. & Day, J.A. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 5: Non-Arthropods. The Protozoans, Porifera, Cnidaria, Platyhelminthes, Nemertea, Rotifera, Nematoda, Nematomorpha, Gastrotrichia, Bryozoa, Tardigrada, Polychaeta, Oligochaeta and Hirudinea. WRC Report No. TT 167/02. Water Research Commission, Pretoria, 293 pp.

Orimoloye, I. R., Kalumba, A. M., Mazinyo, S. P. & Nel, W., 2020. Geospatial analysis of wetland dynamics: Wetland depletion and biodiversity conservation of iSimangaliso Wetland, South Africa. Journal of King Saud University 32:90–96.

Paerl, H. W. & Huisman, J., 2009. Climate change: A catalyst for global expansion of harmful cyanobacterial blooms. Environmental Microbiology Reports 1:27–37.

Perera, S. J., 2013. Patterns of animal endemism in the Maputaland-Pondoland-Albany biodiversity hotspot. University of KwaZulu-Natal. PhD Thesis. 194 pp.

Perera, S., Dayani, R. P. & Serban, P., 2011. Vertebrate distributions indicate a greater Maputaland-Pondoland-Albany region of endemism : Research article. South African Journal of Science 107:1–15.

Perissinotto, R., Bird, M. S. & Bilton, D. T., 2016. Predaceous water beetles (Coleoptera, Hydradephaga) of the Lake St Lucia system, South Africa: Biodiversity, community ecology and conservation implications. ZooKeys 595: 85.

Pinder, A. M., Halse, S. A., McRae, J. M. & Shiel, R. J., 2005. Occurrence of aquatic invertebrates of the wheatbelt region of Western Australia in relation to salinity. Hydrobiologia 543:1–24.

Pinel-Alloul, B., Méthot, G., Lapierre, L. & Willsie, A., 1996. Macroinvertebrate community as a biological indicator of ecological and toxicological factors in Lake Saint-François (Québec). Environmental Pollution 91:65–87.

Poikane, S., Johnson, R. K., Sandin, L., Schartau, A. K., Solimini, A. G., Urbanič, G., Arbačiauskas, K., Aroviita, J., Gabriels, W. & Miler, O., 2016. Benthic macroinvertebrates in lake ecological assessment: A review of methods, intercalibration and practical recommendations. Science of the total environment 543:123–134.

Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. V, O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F., Thompson, S. A. & Richardson, A. J., 2013. Global imprint of climate change on marine life. Nature Climate Change 3:919–925.

Poole, G. C. & Berman, C. H., 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental management 27:787–802.

Porter, S. N., Branch, G. M. & Sink, K. J., 2013. Biogeographic patterns on shallow subtidal reefs in the western Indian Ocean. Marine Biology 160:1271–1283.

Pyke, G. H., Thomson, J. D., Inouye, D. W., & Miller, T. J,. 2016. Effects of climate change on phenologies and distributions of bumble bees and the plants they visit. Ecosphere 7(3):e01267.

Rand, G. M., 1995. Fundamentals of aquatic toxicology: Effects, environmental fate and risk assessment. Taylor & Francis Inc., Washington, USA. 1148 pp.

Read, G. H. L., 1985. Aspects of larval, post-larval and juvenile ecology of *Macrobrachium petersi* (Hilgendorf) in the Keiskamma estuary, South Africa. Estuarine, Coastal and Shelf Science, 21(4): 501-510.

Reavell, P. E. & Cyrus, D. P., 1989. Preliminary observations on the macrocrustacea of coastal lakes in the vicinity of Richards Bay, Zululand, South Africa. Southern African Journal of Aquatic Sciences 15:103–128.

Reavell, P.E., 2003. Hemiptera. In: In: De Moor, I.J., Day, J.A. & De Moor, F.C., (eds), Guides to the freshwater invertebrates of southern Africa. Volume 8: Insecta II. Hemiptera, Megaloptera, Neuroptera, Trichoptera and Lepidoptera. WRC Report No. TT 214/03. Water Research Commission, Pretoria. 220 pp.

Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T. J., Kidd, K. A., MacCormack, T. J., Olden, J. D., Ormerod, S. J., Smol, J. P., Taylor, W. W., Tockner, K., Vermaire, J. C., Dudgeon, D. & Cooke, S. J., 2019. Emerging threats and persistent conservation challenges for freshwater biodiversity. Biological Reviews 94:849–873.

Reinecke, S. A. & Reinecke, A. J., 2007. The impact of organophosphate pesticides in orchards on earthworms in the Western Cape, South Africa. Ecotoxicology and Environmental Safety 66:244–251.

Revenga, C., Brunner, J., Henninger, N., Kassem, K. & Payne, R., 2000. Freshwater systems. World Resources Institute Washington, DC. 65 pp.

Richard, J. & Clark, P. F., 2009. African *Caridina* (Crustacea: Decapoda: Caridea: Atyidae): Redescriptions of *C. africana* Kingsley, 1882, *C. togoensis* Hilgendorf, 1893, *C. natalensis*

Bouvier, 1925 and *C. roubaudi* Bouvier, 1925 with descriptions of 14 new species. Zootaxa 1:3–75.

Richard, J. & Clark, P. F., 2010. *Caridina* H. Milne Edwards, 1837 (Crustacea: Decapoda: Caridea: Atyoidea: Atyidae)–fFreshwater shrimps from eastern and southern Africa. Zootaxa 2372:305–337.

Riddell, E. S., Govender, D., Botha, J., Sithole, H., Petersen, R. M. & Shikwambana, P., 2019. Pollution impacts on the aquatic ecosystems of the Kruger National Park, South Africa. Scientific African 6:e00195.

Rochman, C. M., Hoh, E., Kurobe, T. & Teh, S. J., 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Scientific Reports 3:3263.Rosenkranz, P., Chaudhry, Q., Stone, V. & Fernandes, T. F., 2009. A comparison of nanoparticle and fine particle uptake by *Daphnia magna*. Environmental toxicology and chemistry 28:2142–2149.

RStudio Team, 2022. RStudio: Integrated Development Environment for R. RStudio, PBC, Boston, Massachusetts, USA.

Sabater, S., Freixa, A., Jiménez, L., López-Doval, J., Pace, G., Pascoal, C., Perujo, N., Craven, D. & González-Trujillo, J. D., 2022. Extreme weather events threaten biodiversity and functions of river ecosystems: Evidence from a meta-analysis. Biological Reviews 98(2): 450-461.

Samways, M. J. & Simaika, J. P., 2015. Manual of freshwater assessment for South Africa: Dragonfly biotic index. *Suricata* 2. South African National Biodiversity Institute, Pretoria. 224 pp.

Samways, M.J. & Wilmot, B.C., 2003. Odonata. In: De Moor, I.J., Day, J.A. & De Moor, F.C., (eds), Guides to the Freshwater Invertebrates of Southern Africa: Volume 7 Insecta I—Ephemeroptera, Odonata and Plecoptera. WRC Report No. TT 207/03. Water Research Commission, Pretoria, 288 pp.

Scheffer, M., Carpenter, S., de Young, B., 2005. Cascading effects of overfishing marine systems. Trends in Ecology and Evolution 20: 579-581.

Scheffers, B. R., De Meester, L., Bridge, T. C. L., Hoffmann, A. A., Pandolfi, J. M., Corlett, R. T., Butchart, S. H. M., Pearce-Kelly, P., Kovacs, K. M., Dudgeon, D., Pacifici, M., Rondinini, C., Foden, W. B., Martin, T. G., Mora, C., Bickford, D. & Watson, J. E. M., 2016. The broad footprint of climate change from genes to biomes to people. Science 354: aaf7671.

Schulz, R., 2001. Rainfall-induced sediment and pesticide input from orchards into the Lourens river, Western Cape, South Africa: importance of a single event. Water Research 35:1869–1876.

Shelton, J. M., Bird, M. S., Samways, M. J. & Day, J. A., 2017. Non-native rainbow trout (*Oncorhynchus mykiss*) occupy a different trophic niche to native Breede River redfin (*Pseudobarbus burchelli*) which they replace in South African headwater streams. Ecology of Freshwater Fish 26:484–496.

Siméon, T., Gideon, A., Dramane, D., Idrissa, C. A., Mexmin, K. K. & Pierre, N., 2014. Impact of anthropogenic activities on water quality and freshwater shrimps diversity and distribution in five rivers in Douala, Cameroon. Journal of Biodiversity and Environmental Sciences 4:183–194.

Simpson, S. D., Purser, J. & Radford, A. N., 2015. Anthropogenic noise compromises antipredator behaviour in European eels. Global Change Biology 21:586–593.

Sintayehu, D. W., 2018. Impact of climate change on biodiversity and associated key ecosystem services in Africa: a systematic review. Ecosystem Health and Sustainability 4:225–239.

Skowno, A., C.J. Poole, Raimondo, D., K.J. Sink, Van Deventer, H., Van Niekerk, L., Harris, L., Smith-Adao, L., Tolley, K., Zengeya, T., Foden, W., Midgley, G. & Driver, A., 2019. National Biodiversity Assessment 2018: The status of South Africa's ecosystems and biodiversity. Synthesis report. South African National Biodiversity Institute, Pretoria. 214 pp.

Smith, R. A., Alexander, R. B. & Wolman, M. G., 1987. Water-quality trends in the nation's rivers. Science 235:1607–1615.

Smith, R. J., Easton, J., Nhancale, B. A., Armstrong, A. J., Culverwell, J., Dlamini, S. D., Goodman, P. S., Loffler, L., Matthews, W. S., Monadjem, A., Mulqueeny, C. M., Ngwenya, P., Ntumi, C. P., Soto, B. & Leader-Williams, N., 2008. Designing a transfrontier conservation landscape for the Maputaland centre of endemism using biodiversity, economic and threat data. Biological Conservation 141:2127–2138.

South African Department of Science and Technology (SADST), 2017. 10-Year Global Change Research Plan for South Africa. Department of Science and Technology, Pretoria. 43 pp.

South African National Biodiversity Institute (SANBI), 2010. Ecosystem Profile: Maputaland-Pondoland-Albany Biodiversity Hotspot. Department of Science and Technology, Pretoria. 128pp.

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdaña, Z. A., Finlayson, M., Halpern, B. S., Jorge, M. A., Lombana, A., Lourie, S. A., Martin, K. D., McManus, E., Molnar, J., Recchia, C. A. & Robertson, J., 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Sparks, J.S., 2016A. *Pantanodon madagascariensis*. The IUCN Red List of Threatened Species 2016: e.T15949A58297104. https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T15949A58297104.en. Accessed on 10 October 2023.

Sparks, J.S., 2016B. *Ptychochromis onilahy*. The IUCN Red List of Threatened Species 2016: e.T44504A58308545. https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T44504A58308545.en. Accessed on 10 October 2023.

Stals, R., 2007. Gyrinidae. In: Stals, R. & de Moor, I. J. (eds), Guides to the freshwater invertebrates of southern Africa. Volume 10: Coleoptera. WRC Report No. TT 320/07. Water Research Commission, Pretoria. 262 pp.

Steenkamp, Y., Van Wyk, B., Victor, J., Hoare, D., Smith, G., Dold, T. & Cowling, R., 2004. Maputaland-Pondoland-Albany. In: Mittermeier, R. A., Gil, P. R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J. & Da Fonseca, G. A. B (eds), Hotspots revisited: Earth's biologically richest and most endangered ecoregions. CEMEX, Mexico DF: 219–228.

Stevenson, J. R., Bliss, D. E. & Mantel, L. H., 1985. Dynamics of the integument. The Biology of Crustacea 9:1–42.

Strayer, D. L. & Dudgeon, D., 2010. Freshwater biodiversity conservation: recent progress and future challenges. Journal of the North American Benthological Society 29:344–358.

Stuble, K. L., Bewick, S., Fisher, M., Forister, M. L., Harrison, S. P., Shapiro, A. M., Latimer, A. M. & Fox, L. R., 2021. The promise and the perils of resurveying to understand global change impacts. Ecological Monographs 91:e01435.

Sydeman, W. J., Poloczanska, E., Reed, T. E. & Thompson, S. A., 2015. Climate change and marine vertebrates. Science 350:772–777.

Tanentzap, A. J., Morabito, G., Volta, P., Rogora, M., Yan, N. D. & Manca, M., 2020. Climate warming restructures an aquatic food web over 28 years. Global Change Biology 26:6852–6866.

Teng, M., Wang, C., Song, M., Cheng, X., Zhang, J. & Wang, C., 2020. Chronic exposure of zebrafish (*Danio rerio*) to flutolanil leads to endocrine disruption and reproductive disorders. Environmental Resources 184:109e310.

Tickner, D., Opperman, J. J., Abell, R., Acreman, M., Arthington, A. H., Bunn, S. E., Cooke, S. J., Dalton, J., Darwall, W., Edwards, G., Harrison, I., Hughes, K., Jones, T., Leclère, D., Lynch, A. J., Leonard, P., McClain, M. E., Muruven, D., Olden, J. D., Ormerod, S. J., Robinson, J., Tharme, R. E., Thieme, M., Tockner, K., Wright, M. & Young, L., 2020. Bending the curve of global freshwater biodiversity loss: An emergency recovery plan. BioScience 70:330–342.

Tingley, M.W., Koo, M.S., Moritz, C., Rush, A.C. & Beissinger, S.R., 2012. The push and pull of climate change causes heterogeneous shifts in avian elevational ranges. Global Change Biology, 18: 3279-3290.

Turner, A. D., Lewis, A. M., Bradley, K. & Maskrey, B. H., 2021. Marine invertebrate interactions with harmful algal blooms - Implications for one health. Journal of invertebrate pathology 186:107555.

United States Geological Survey, 1999. The quality of our nation's waters: Nutrients and pesticides. US Geological Survey Circular 1225. United States Geological Survey, Virginia, USA. 82 pp.

Van Damme, D., 2016. *Unio malgachensis*. The IUCN Red List of Threatened Species 2016: e.T91292882A91292891. https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T91292882A91292891.en. Accessed on 10 October 2023.

Van Deventer, H., Adams, J. B., Durand, J. F., Grobler, R., Grundling, P. L., Janse van Rensburg, S., Jewitt, D., Kelbe, B., MacKay, C. F., Naidoo, L., Nel, J. L., Pretorius, L., Riddin, T. & Van Niekerk, L., 2021. Conservation conundrum – Red listing of subtropical-temperate coastal forested wetlands of South Africa. Ecological Indicators 130:108077.

Van Tonder, G., Botha, J. & Müller, J., 1986. The problem of sea-water intrusion near Lake Mzingazi at Richards Bay. Water SA 12:83–88.

Venables, W. N. & Ripley, B. D., 2002. Modern Applied Statistics with S. (4th edition). Springer.

Visser, P. M., Verspagen, J. M. H., Sandrini, G., Stal, L. J., Matthijs, H. C. P., Davis, T. W., Paerl, H. W. & Huisman, J., 2016. How rising CO2 and global warming may stimulate harmful cyanobacterial blooms. Harmful Algae 54:145–159.

Von Moos, N., Burkhardt-Holm, P. & Köhler, A., 2012. Uptake and effects of microplastics on cells and tissue of the blue mussel *Mytilus edulis* L. after an experimental exposure. Environmental science & technology 46:11327–11335.

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A. & Liermann, C. R., 2010. Global threats to human water security and river biodiversity. Nature 467:555–561.

Walters, K. & Coen, L. D., 2006. A comparison of statistical approaches to analyzing community convergence between natural and constructed oyster reefs. Journal of Experimental Marine Biology and Ecology 330:81–95.

Wangai, P. W., Burkhard, B. & Müller, F., 2016. A review of studies on ecosystem services in Africa. International Journal of Sustainable Built Environment 5:225–245.

Weerts, S., Mackay, C. & Cyrus, D., 2014. The potential for a fish ladder to mitigate against the loss of marine–estuarine–freshwater connectivity in a subtropical coastal lake. Water SA 40:27–38.

Weijters, M.J., Janse, J.H., Alkemade, R. & Verhoeven, J.T.A., 2009. Quantifying the effect of catchment land use and water nutrient concentrations on freshwater river and stream biodiversity. Aquatic Conservation: Marine and Freshwater Ecosystems 19: 104-112.

White, J. & Irvine, K., 2003. The use of littoral mesohabitats and their macroinvertebrate assemblages in the ecological assessment of lakes. Aquatic Conservation: Marine and Freshwater Ecosystems 13:331–351.

Whitelaw, G. & Van Rensburg, S. J., 2020. Lake Sibaya and the beginning of the iron age in Kwazulu-Natal. The Digging Stick 37.

Whitfield, A. K., James, N. C., Lamberth, S. J., Adams, J. B., Perissinotto, R., Rajkaran, A. & Bornman, T. G., 2016. The role of pioneers as indicators of biogeographic range expansion caused by global change in southern African coastal waters. Estuarine, Coastal and Shelf Science 172:138–153.

Wicke, B., Smeets, E., Dornburg, V., Vashev, B., Gaiser, T., Turkenburg, W. & Faaij, A., 2011. The global technical and economic potential of bioenergy from salt-affected soils. Energy & Environmental Science 4:2669–2681.

Wickham, H., 2016. Package 'ggplot2'. Create elegant data visualisations using the grammar of graphics. Version 2(1). 189 pp.

Winder, M. & Schindler, D. E., 2004. Climate change uncouples trophic interactions in an aquatic ecosystem. Ecology 85:2100–2106.

Windsor, F. M., Tilley, R. M., Tyler, C. R., Ormerod, S. J., 2019. Microplastic ingestion by riverine macroinvertebrates. Science of the Total Environment 646: 68-74.

Wood, L. E., Daniels, S. R. & De Grave, S., 2018. *Caridina susuroflabra* Richard & Clark, 2009 is a junior synonym of the widespread *Caridina africana* Kingsley, 1882 (Decapoda, Atyidae). Crustaceana 91:243–249.

Woods, T., Kaz, A. & Giam, X., 2021. Phenology in freshwaters: A review and recommendations for future research. Ecography 44:1–14.

Woolway, R. I., Sharma, S., Weyhenmeyer, G. A., Debolskiy, A., Golub, M., Mercado-Bettín, D., Perroud, M., Stepanenko, V., Grant, Z. T. L., Ladwig, R., Mesman, J., Moore, T. N., Shatwell, T., Vanderkelen, I., Austin, J. A., DeGasperi, C. L., Dokulil, M., Fuente, S. La, Mackay, E. B., Schladow, S. G., Watanabe, S., Thiery, R. M. W. & Jennings, E., 2021. Phenological shifts in lake stratification under climate change. Nature Communications 12:1–11.

Zandaryaa, S., 2021. Freshwater microplastic pollution: the state of knowledge and research. In: Stock, F., Reifferscheid, G., Brennholt, N., Kostianaia, E. (eds), Plastics in the Aquatic Environment- Part I:. The Handbook of Environmental Chemistry, vol 111. Springer, Cham: 255–272.

Zardi, G. I., McQuaid, C. D., Jacinto, R., Lourenço, C.R., Serrão, E. A.. & Nicastro, K. R., 2018. Re-assessing the origins of the invasive mussel *Mytilus galloprovincialis* in southern Africa. Marine and Freshwater Research 69:607–613.

Ziervogel, G., New, M., Archer van Garderen, E., Midgley, G., Taylor, A., Hamann, R., Stuart-Hill, S., Myers, J. & Warburton, M., 2014. Climate change impacts and adaptation in South Africa. WIREs Climate Change 5:605–620.

Zilihona, I. J. E., Niemelä, J. & Nummelin, M., 2004. Effects of a hydropower plant on Coleopteran diversity and abundance in the Udzungwa Mountains, Tanzania. Biodiversity & Conservation 13:1453–1464.

Ziv, G., Baran, E., Nam, S., Rodríguez-Iturbe, I. & Levin, S. A., 2012. Trading-off fish biodiversity, food security, and hydropower in the Mekong River Basin. Proceedings of the National Academy of Sciences 109:5609.

Supplementary Material

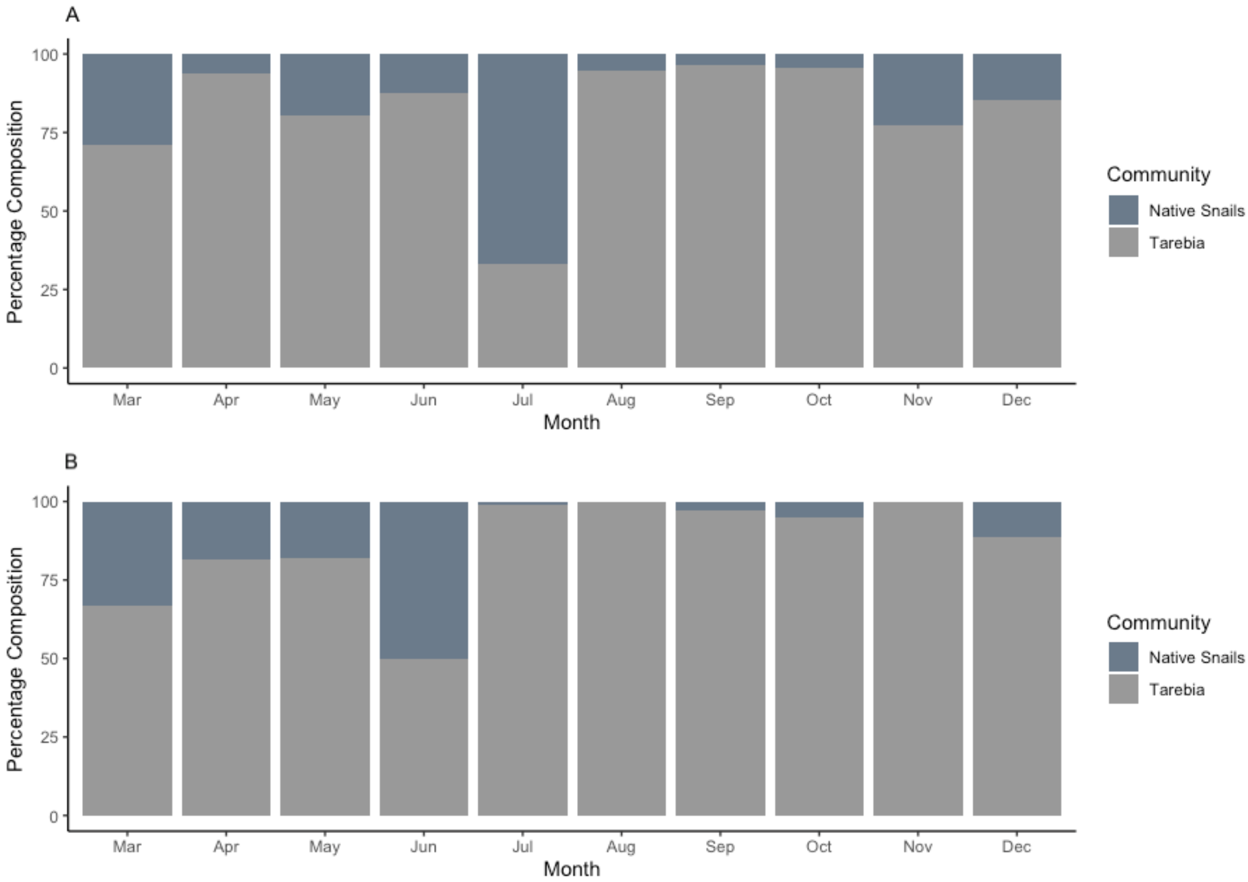

Figure S 1: Showing the percentage composition of the native snail communities and Tarebia granifera over the study period at lakes Sibaya (A) and Mzingazi (B)

Table S 1: Showing model comparison tables used to determine the family to be used for generalised linear models testing the correlation between *Caridina africana* abundance and physiochemistry parameters

Lake	**Model Family**	**LogLik**	**AICc**
Sibaya	Poisson	-190.497	398.99
	Negative Binomial	-113.305	246.61
Mzingazi	Poisson	-141.260	309.52
	Negative Binomial	-89.716	211.01

Table S 2: Showing likelihood ratio test results comparing full (mixed) and nested (initial) generalised linear models testing the correlation between *Caridina africana* abundance and physiochemistry parameters

Lake	Model	Df	P-Value
Sibaya	**Abundance~Predictors**	10	1
	vs		
	Abundance~Predictors + (1|Site)	11	
	Abundance~Predictors	10	1
	vs		
	Abundance~Predictors + (1|Month)	11	
Mzingazi	**Abundance~Predictors**	10	1
	vs		
	Abundance~Predictors + (1|Site)	11	
	Abundance~Predictors	10	1
	vs		
	Abundance~Predictors + (1|Month)	11	

Table S 3: Showing final generalised linear models used when assessing the effect of physicochemical variables on *Caridina africana* abundances

Lake	Response Variable	Predictor Variables	Mixed Effect	Type of Model
Sibaya	*Caridina africana* abundances	Ammonium + Nitrate + Phosphate + Conductivity	-	GLM (Family = Negative Binomial)
Mzingazi	*Caridina africana* abundances	Temperature + Ammonium + Nitrate + Phosphate	-	GLM (Family = Negative Binomial)

Table S 4: Showing results of Wilcoxon Rank Sum tests conducted on *Caridina africana* abundances (n =30) and temperatures (n =30) *recorded* at Lake Sibaya and Lake Mzingazi in 2021

Factors (comparison)	**W**	**P-Value**
Lake Mzingazi abundance vs Lake Sibaya abundance	233	p<0.01
Lake Mzingazi temperature vs Lake Sibaya temperature	843	p<0.001

Table S 5: Showing unadjusted significant results of Dunn's post hoc tests conducted on significant results from the Kruskal-Wallis tests results found in Table 2.2. P-values were adjusted using the Hochberg method

Lake	**Comparison**	**Z**	**P (Unadjusted)**	**P (Adjusted)**
Sibaya	Apr-Dec	-2.55397529	P<0.05	P>0.05
	Aug-Dec	-2.32179572	P<0.05	P>0.05
	Apr-Jul	-2.29857776	P<0.05	P>0.05
	Aug-Jul	-2.06639819	P<0.05	P>0.05
	Apr-Mar	-2.01996228	P<0.05	P>0.05
	Apr-May	-2.83259078	P<0.01	P>0.05
	Aug-May	-2.60041121	P<0.01	P>0.05
	Dec-Nov	2.46110346	P<0.05	P>0.05
	Jul-Nov	2.20570594	P<0.05	P>0.05
	May-Nov	2.73971895	P<0.01	P>0.05
	Apr-Sep	-2.73971895	P<0.01	P>0.05
	Aug-Sep	-2.50753938	P<0.05	P>0.05
	Nov-Sep	-2.64684712	P<0.01	P>0.05
Mzingazi	Autumn-Spring	-2.1732902	P<0.05	P>0.05

	Autumn-Summer	-2.3525529	P<0.05	P>0.05
	Aug-Mar	1.97506896	P<0.05	P>0.05
	Dec-Mar	2.48626327	P<0.05	P>0.05
	Aug-Nov	1.99830506	P<0.05	P>0.05
	Dec-Nov	2.50949938	P<0.05	P>0.05
	Apr-Oct	-2.04477727	P<0.05	P>0.05
	Mar-Oct	-2.74186043	P<0.01	P>0.05
	Nov-Oct	-2.76509654	P<0.01	P>0.05
	Apr-Sep	-2.04477727	P<0.05	P>0.05
	Mar-Sep	-2.74186043	P<0.01	P>0.05
	Nov-Sep	-2.76509654	P<0.01	P>0.05

Taxa richness at Lake Sibaya and aquatic invertebrate abundance and Pielou's evenness at Lake Mzingazi were run as standard linear models as neither model using a random effect outperformed the initial model (Table S6). Taxa Richness at Lake Mzingazi was run as a linear mixed model using month as a random effect because the model using site as a random effect did not outperform the model using month (Table S6).

Table S 6: Showing likelihood ratio test results comparing full (mixed) and nested (initial) linear (mixed) models conducted on taxa richness at lake Sibaya and aquatic invertebrate abundance, taxa richness and Pielou's evenness at lake Mzingazi in response to all measured numeric variables

Lake	Model	Df	P-Value
Sibaya	**Richness~Predictors** **vs** **Richness~Predictors + (1|Site)**	10 11	P>0.05
	Richness~Predictors **vs** **Richness~Predictors + (1|Month)**	10 11	P>0.05

Mzingazi	**Abundance~Predictors** **vs** **Abundance~Predictors + (1|Site)**	10 11	P>0.05
	Abundance~Predictors **vs** **Abundance~Predictors + (1|Month)**	10 11	P>0.05
	Richness~Predictors **vs** **Richness~Predictors + (1|Site)**	10 11	P>0.05
	Richness~Predictors **vs** **Richness~Predictors + (1|Month)**	10 11	P<0.05
	Evenness~Predictors **vs** **Evenness ~Predictors + (1|Site)**	10 11	P>0.05
	Evenness ~Predictors **vs** **Evenness ~Predictors + (1|Month)**	10 11	P>0.05

Table S 7: Showing model comparison tables used for (Generalised) Linear (Mixed) Models conducted on diversity indices (Aquatic invertebrate abundance, taxa richness, Pielou's evenness and Shannon's diversity index) in response to all measured numeric variables. Generalised linear models are in italics and linear models in standard text

Lake	**Model**	**AICc**	**Maximum Acceptable Autocorrelation**	**Highest Model Autocorrelation**
Sibaya	**Invertebrate Abundance ~Predictors**	385.478	0.4	-0.5
	Invertebrate Abundance ~Predictors + (1|Site)	319.961	0.4	-0.5
	Invertebrate Abundance ~Predictors + (1|Month)	319.961	0.4	-0.5
	Invertebrate Abundance ~Predictors	368.833	0.39	-0.35

	Invertebrate Abundance ~Predictors + (1|Site)	373.921	0.39	-0.35
	Invertebrate Abundance ~Predictors + (1|Month)	373.921	0.39	-0.35
	Richness~Predictors	149.553	0.39	-0.3
	Richness~Predictors+ (1|Site)	154.814	0.39	-0.3
	Richness~Predictors+ (1|Month)	155.028	0.39	-0.37
	Evenness~Predictors	43.723	0.39	-0.4
	Evenness~Predictors+ (1|Site)	79.987	0.39	-0.39
	Evenness~Predictors+ (1|Month)	80.362	0.39	-0.4
	Evenness~Predictors	42.599	0.39	-0.4
	Evenness~Predictors+ (1|Site)	48.030	0.39	-0.37
	Evenness~Predictors+ (1|Month)	48.291	0.39	-0.39
	Diversity~Predictors	38.577	0.39	-0.4
	Diversity ~Predictors+ (1|Site)	76.756	0.39	-0.39
	Diversity ~Predictors+ (1|Month)	74.912	0.39	-0.39
	Diversity~Predictors	37.867	0.39	-0.38
	Diversity ~Predictors+ (1|Site)	43.682	0.39	-0.39
	Diversity ~Predictors+ (1|Month)	40.828	0.39	-0.39
Mzingazi	**Invertebrate Abundance ~Predictors**	351.208	0.39	-0.3
	Invertebrate Abundance ~Predictors + (1|Site)	299.703	0.39	-0.3
	Invertebrate Abundance ~Predictors + (1|Month)	296.097	0.39	-0.5
	Richness~Predictors	140.917	0.39	0.4
	Richness~Predictors+ (1|Site)	139.911	0.39	0.4
	Richness~Predictors+ (1|Month)	128.294	0.39	-0.3

	Evenness~Predictors	33.637	0.39	0.39
	Evenness~Predictors+ (1|Site)	66.218	0.39	-0.39
	Evenness~Predictors+ (1|Month)	66.218	0.39	-0.39
	Diversity~Predictors	45.714	0.39	-0.39
	Diversity ~Predictors+ (1|Site)	75.074	0.39	-0.39
	Diversity ~Predictors+ (1|Month)	70.693	0.39	-0.4
	Diversity~Predictors	48.731	0.39	0.37
	Diversity ~Predictors+ (1|Site)	55.099	0.39	0.39
	Diversity ~Predictors+ (1|Month)	53.310	0.39	0.35

The very high AICc value found for the Poisson regression used to assess aquatic invertebrate abundance at Lake Sibaya prompted the checking of the Gamma family to be sure of the data distribution (Table S8). The negative binomial model outperformed both Poission and Gamma families and was, therefore, the selected model for this analysis.

Table S 8: Showing model comparison tables used to determine the family to be used for generalised linear models conducted on invertebrate abundance, Pielou's evenness and Shannon's diversity index at lake Sibaya and Shannon's diversity index at lake Mzingazi in response to all measured numeric variables

Lake	Diversity Index	Model Family	AICc	LogLik
Sibaya	**Invertebrate Abundance**	Negative Binomial	368.8	-168.627
		Gamma	377.2	-172.810
		Poisson	1442.4	-707.716
	Pielou's Evenness	Gamma	42.6	-7.799
		Inverse Gaussian	45.5	-9.251
	Shannon's Diversity	Gamma	37.9	-3.144
		Inverse Gaussian	45.5	-4.333
Mzingazi	**Shannon's Diversity**	Gamma	48.731	-10.865
		Inverse Gaussian	54.963	-13.982

Aquatic invertebrate abundance at Lake Sibaya and Shannon's Diversity Index at both lakes were run as standard generalized linear models as neither model using a random effect outperformed the initial model (Table S9). Pielou's evenness at Lake Sibaya was run as a generalised linear mixed model using site as a random effect despite the results of the likelihood ratio test because the model run using site as a random effect had the lowest ACF value (Table S7). The model using month did not outperform the model using site (Table S7).

Table S 9: Showing likelihood ratio test results comparing full (mixed) and nested (initial) generalised linear (mixed) models conducted on invertebrate abundance, Pielou's evenness and Shannon's diversity index at lake Sibaya and Shannon's diversity index at lake Mzingazi in response to all measured numeric variables

Lake	Model	Df	P-Value
Sibaya	**Abundance~Predictors**	10	P>0.05
	vs		
	Abundance~Predictors + (1|Site)	11	
	Abundance~Predictors	10	P>0.05
	vs		
	Abundance~Predictors + (1|Month)	11	
	Evenness~Predictors	10	P>0.05
	vs		
	Evenness~Predictors + (1|Site)	11	
	Evenness~Predictors	10	P>0.05
	vs		
	Evenness ~Predictors + (1|Month)	11	
	Diversity~Predictors	10	P>0.05
	vs		
	Diversity ~Predictors + (1|Site)	11	
	Diversity ~Predictors	10	P>0.05
	vs		
	Diversity ~Predictors + (1|Month)	11	
Mzingazi	**Diversity~Predictors**	10	P>0.05
	vs		

	Diversity ~Predictors + (1|Site)	11	
	Diversity ~Predictors	10	P>0.05
	vs		
	Diversity ~Predictors + (1|Month)	11	

Table S 10: Showing model comparison tables used to determine the family to be used for generalised linear models conducted on native invertebrate diversity of lakes Sibaya and Mzingazi in response to the invasive snail *Tarebia granifera*

Lake	**Model Family**	**LogLik**	**AICc**
Sibaya	Gamma	-1.989	40.6
	Inverse Gaussian	-1.989	40.6
Mzingazi	Gamma	-12.727	48.3
	Inverse Gaussian	-7.858	113.9

Table S 11: Showing likelihood ratio test results comparing full (mixed) and nested (initial) generalised linear (mixed) models conducted on native invertebrate diversity of lakes Sibaya and Mzingazi in response to the invasive snail *Tarebia granifera*

Lake	**Model**	**Df**	**P-Value**	**Maximum Acceptable Autocorrelation**	**Model Autocorrelation**
Sibaya	**Diversity~*Tarebia granifera***	8	1	0.39	0.3
	vs				
	Diversity~*T. granifera*+ (1|Site)	9		0.39	0.3
	Diversity~*T. granifera*	8	1	0.39	0.3
	vs				
	Diversity~*T. granifera* + (1|Month)	9		0.39	0.3
Mzingazi	**Diversity~*T. granifera* ~**	8	1	0.39	0.35
	vs				
	Diversity~*T. granifera* + (1|Site)	9		0.39	0.35
	Diversity~*T. granifera*	8	1	0.39	0.35
	vs				
	Diversity~*T. granifera* + (1|Month)	9		0.39	0.35

Table S 12: Showing model parameters used for final (generalised) linear (mixed) models conducted on aquatic invertebrate abundance, taxa richness, Pielou's evenness and Shannon's diversity index in response to all measured numeric variables and Shannon's diversity index of the native invertebrate community in response to Tarebia granifera abundances and all measured numeric variables at lakes Sibaya and Mzingazi

Lake	Response Variable	Predictor Variables	Mixed Effect	Type of Model
Sibaya	**Aquatic invertebrate Abundance**	Salinity + Lake Level + Temperature + Phosphate Concentration	-	GLM (Family = Negative Binomial)
	Taxa richness	Conductivity + Phosphate Concentration + Nitrate Concentrate	-	LM
	Pielou's evenness	Salinity + Nitrate Concentration + Phosphate Concentration	Site	GLMM (Family = Gamma)
	Shannon's diversity index	Salinity + Nitrate Concentration + Conductivity	-	GLM (Family = Gamma)
	Native aquatic invertebrate diversity	Conductivity + *Tarebia granifera* abundance	-	GLM (Family = Gamma)
Mzingazi	**Aquatic invertebrate Abundance**	Salinity + Nitrate Concentration + Phosphate Concentration	-	LM
	Taxa richness	pH + Nitrate Concentration + Lake Level	Month	LMM
	Pielou's evenness	Salinity + Ammonium Concentration + Nitrate Concentration	-	LM
	Shannon's diversity index	Salinity + Ammonium Concentration	-	GLM (Family = Gamma)
	Native aquatic invertebrate diversity	Salinity + Ammonium Concentration + *Tarebia granifera* abundance	-	GLM (Family = Gamma)

Table S 13: Showing all significant (Unadjusted P -Values) results of Dunn's post hoc tests conducted on significant results from Kruskal-Wallis tests found in Table 3.1. P-values were adjusted using the Hochberg method and significant adjusted P-values are in bold

Lake	**Comparison**	**Z**	**P (Unadjusted)**	**P (Adjusted)**
Sibaya	Autumn-Spring (Shannon's Diversity Index)	2.731	P<0.01	**P<0.05**
Mzingazi	Autumn-Spring (Shannon's Diversity Index)	2.382	P<0.05	P>0.05
	Spring-Summer (Shannon's Diversity Index)	-3.181	P<0.01	**P<0.01**
	Summer-Winter (Shannon's Diversity Index)	2.423	P<0.05	P>0.05
	Apr-Aug (Invertebrate Abundance)	2.389	P<0.05	P>0.05
	Aug-Dec (Invertebrate Abundance)	-2.065	P<0.05	P>0.05
	Aug-Jul (Invertebrate Abundance)	-2.807	P<0.01	P>0.05
	Aug-Mar (Invertebrate Abundance)	-2.644	P<0.01	P>0.05
	Apr-May (Invertebrate Abundance)	2.041	P<0.05	P>0.05
	Jul-May (Invertebrate Abundance)	2.459	P<0.05	P>0.05
	Mar-May (Invertebrate Abundance)	2.296	P<0.05	P>0.05
	Aug-Oct (Invertebrate Abundance)	-2.459	P<0.05	P>0.05
	May-Oct (Invertebrate Abundance)	-2.111	P<0.05	P>0.05
	Aug-Sep (Invertebrate Abundance)	-2.413	P<0.05	P>0.05
	May-Sep (Invertebrate Abundance)	-2.065	P<0.05	P>0.05

	Apr-Aug (Richness)	3.127	P<0.01	P>0.05
	Aug-Dec (Richness)	-3.409	P<0.001	**P<0.05**
	Aug-Jul (Richness)	-2.639	P<0.01	P>0.05
	Aug-Jun (Richness)	-2.069	P<0.05	P>0.05
	Aug-Mar (Richness)	-2.304	P<0.05	P>0.05
	Apr-May (Richness)	2.186	P<0.05	P>0.05
	Dec-May (Richness)	2.468	P<0.05	P>0.05
	Apr-Sep (Richness)	1.998	P<0.05	P>0.05
	Dec-Sep (Richness)	2.281	P<0.05	P>0.05

Table S 14: Showing raw physicochemistry data and *Caridina africana* abundance and density over 12 sampled months from Lake Sibaya

Month	Site	Lake	pH	Salinity (ppm)	NH_4^+ (mg/l)	NO_3^- (mg/l)	PO_4^{3-} (mg/l)	Temperature (°C)	Conductivity(μS/m)	Lake Level (MASL)	*C. africana* Density (/m²)	*C. africana* Abundance	Season
Jan	1	Sibaya	10,8	231	0,5	0,83	0,1	28,4	785	10,949	140	13	Summer
Jan	2	Sibaya	10,8	305	0,5	0,83	0,1	28,4	785	10,949	0	0	Summer
Jan	3	Sibaya	10,8	296	0,5	0,83	0,1	28,4	785	10,949	22	2	Summer
Feb	1	Sibaya	9,33	223	0,50	0,83	0,10	28,40	785,00	10,956	140	13	Summer
Feb	2	Sibaya	10,8	298	0,50	0,83	0,10	28,40	785,00	10,956	183	17	Summer
Feb	3	Sibaya	9,88	285	0,50	0,83	0,10	28,40	785,00	10,956	22	2	Summer
Mar	1	Sibaya	7,81	310	0,49	0,99	0,1	27,751	616	11,018	333	31	Autumn
Mar	2	Sibaya	10,8	380	0,52	0,75	0,1	27,801	734	11,018	355	33	Autumn
Mar	3	Sibaya	8,92	400	0,5	0,83	0,1	28,072	785	11,018	129	12	Autumn
Apr	1	Sibaya	8,53	389	0,42	1,29	0,1	26,744	826	10,956	108	10	Autumn
Apr	2	Sibaya	8,9	389	0,39	1,35	0,1	26,793	826	10,956	0	0	Autumn
Apr	3	Sibaya	8,93	389	0,4	1,38	0,1	26,94	826	10,956	0	0	Autumn
May	1	Sibaya	9,9	300	0,63	0,93	0,1	24,3	823	10,932	258	24	Autumn
May	2	Sibaya	8,92	250	0,68	0,87	0,2	27,1	828	10,932	667	62	Autumn
May	3	Sibaya	9,17	300	0,48	0,92	0,1	20,2	839	10,932	462	43	Autumn
Jun	1	Sibaya	8,58	387	0,64	0,94	0	20,579	832	10,950	183	17	Winter
Jun	2	Sibaya	8,86	387	0,64	0,98	0,1	20,722	834	10,950	183	17	Winter
Jun	3	Sibaya	8,95	388	0,62	1,09	0	20,555	834	10,950	65	6	Winter
Jul	1	Sibaya	8,64	300	0,52	0,8	0,2	19,555	886	10,910	269	25	Winter
Jul	2	Sibaya	8,58	260	0,48	0,81	0,2	19,508	886	10,910	516	48	Winter
Jul	3	Sibaya	8,65	300	0,55	0,78	0,1	19,46	886	10,910	183	17	Winter
Aug	1	Sibaya	9,15	160	0,2	0,51	0,1	19,579	782	10,842	97	9	Winter
Aug	2	Sibaya	9,13	240	0,19	0,53	0,1	19,45	782	10,842	0	0	Winter
Aug	3	Sibaya	9,16	200	0,23	0,49	0,1	19,032	782	10,842	97	9	Winter
Sep	1	Sibaya	9,13	300	0,64	0,98	0,1	20,698	752	10,792	269	25	Spring
Sep	2	Sibaya	9,13	300	0,64	0,98	0,1	20,722	752	10,792	1161	108	Spring
Sep	3	Sibaya	9,13	300	0,64	0,98	0,1	20,722	752	10,792	333	31	Spring
Oct	1	Sibaya	8,72	120	0,51	0,63	0,1	21,843	854	10,900	161	15	Spring
Oct	2	Sibaya	8,8	200	0,49	0,66	0	21,843	854	10,900	215	20	Spring

Oct	3	Sibaya	8,69	150	0,54	0,62	0	21,843	854	10,900	140	13	Spring
Nov	1	Sibaya	9,55	168	0,52	0,8	0,1	23,8	856	11,1183	75	7	Spring
Nov	2	Sibaya	9,55	245	0,52	0,8	0,1	23,8	856	11,1183	54	5	Spring
Nov	3	Sibaya	9,55	213	0,52	0,8	0,1	23,8	856	11,1183	32	3	Spring
Dec	1	Sibaya	8,78	215	0,56	0,80	0,10	25,65794556	855	11,2125	366	34	Summer
Dec	2	Sibaya	8,78	290	0,55	0,81	0,07	25,65794556	855	11,2125	366	34	Summer
Dec	3	Sibaya	8,78	275	0,57	0,80	0,07	25,65794556	855	11,2125	226	21	Summer

Table S 15: Showing raw physicochemistry data and *Caridina africana* abundance and density over 10 sampled months from Lake Mzingazi

Month	Site	Lake	pH	Salinity (ppm)	NH_4^+ (mg/l)	NO_3^- (mg/l)	PO_4^{3-} (mg/l)	Temperature (°C)	Conductivity(μS/m)	Lake Level (MASL)	*C. africana* Density (/m²)	*C. africana* Abundance	Season
Mar	2	Mzingazi	7,13	272	0,35	0,84	0,2	31	630	3,152	32	3	Autumn
Mar	3	Mzingazi	8,79	283	0,38	0,79	0,2	31,5	630	3,152	0	0	Autumn
Apr	1	Mzingazi	7,46	285	0,42	0,94	0,2	30,3	610	3,190	0	0	Autumn
Apr	2	Mzingazi	7,02	274	0,5	0,93	0,2	29,5	610	3,190	183	17	Autumn
Apr	3	Mzingazi	8,14	282	0,37	0,9	0,2	30,8	610	3,190	11	1	Autumn
May	1	Mzingazi	7,02	272,5	0,5	0,93	0,2	29,5	588	3,247	22	2	Autumn
May	2	Mzingazi	7,02	265,5	0,5	0,93	0,2	29,5	588	3,247	140	13	Autumn
May	3	Mzingazi	7,02	266	0,5	0,93	0,2	29,5	588	3,247	22	2	Autumn
Jun	1	Mzingazi	8,3	260	0,33	0,92	0	28,7	603	3,270	75	7	Winter
Jun	2	Mzingazi	8,15	257	0,29	0,84	0	28,5	603	3,270	54	5	Winter
Jun	3	Mzingazi	7,95	250	0,31	0,77	0	28,9	603	3,270	43	4	Winter
Jul	1	Mzingazi	8,6	260	0,72	0,85	0,2	28,7	565	3,203	65	6	Winter
Jul	2	Mzingazi	8,61	300	0,7	0,72	0,1	28,7	565	3,203	215	20	Winter
Jul	3	Mzingazi	8,58	250	0,74	0,82	0,1	28,7	565	3,203	108	10	Winter
Aug	1	Mzingazi	8,9	290	0,43	0,79	0,20	28,50	560	3,151	86	8	Winter
Aug	2	Mzingazi	8,94	250	0,41	0,75	0,10	28,50	560	3,151	43	4	Winter
Aug	3	Mzingazi	8,9	270	0,42	0,77	0,10	28,50	560	3,151	581	54	Winter
Sep	1	Mzingazi	7,95	265	0,74	0,77	0,1	28,9	541	3,213	183	17	Spring
Sep	2	Mzingazi	7,95	265	0,74	0,77	0,1	28,9	541	3,213	484	45	Spring
Sep	3	Mzingazi	7,95	265	0,74	0,77	0,1	28,9	541	3,213	215	20	Spring
Oct	1	Mzingazi	8,96	70	0,32	0,45	0,1	24,5	391	3,226	516	48	Spring
Oct	2	Mzingazi	8,92	200	0,35	0,38	0	24,5	391	3,226	118	11	Spring
Oct	3	Mzingazi	8,94	150	0,31	0,42	0	24,5	391	3,226	344	32	Spring
Nov	1	Mzingazi	8,97	176,5	0,41	0,72	0,1	27	392	3,230	11	1	Spring
Nov	2	Mzingazi	8,97	236	0,41	0,72	0,1	27	392	3,230	0	0	Spring
Nov	3	Mzingazi	8,97	216,5	0,41	0,72	0,1	27	392	3,230	11	1	Spring
Dec	1	Mzingazi	8,78	229,75	0,42	0,82	0,1	26,9	391	3,208	194	18	Summer
Dec	2	Mzingazi	8,78	254	0,42	0,82	0,1	26,9	391	3,208	226	21	Summer
Dec	3	Mzingazi	8,78	249,75	0,42	0,82	0,1	26,9	391	3,208	140	13	Summer

Table S 16: Showing raw *Caridina africana* population data collected from Lake Sibaya in 2021

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
2,51-3,33	1	Jan	12	4	3	f	no
2,51-3,33	1	Jan	13	3	3	f	no
3,34-4,17	1	Jan	6	3,5	3,5	m	no
3,34-4,17	1	Jan	7	4,5	3,5	m	no
3,34-4,17	1	Jan	4	4	4	f	yes
3,34-4,17	1	Jan	9	4	4	f	no
3,34-4,17	1	Jan	5	5	4	m	no
3,34-4,17	1	Jan	8	4	4	m	no
3,34-4,17	1	Jan	11	4,5	4	m	no
4,18-5	1	Jan	10	5	4,5	f	no
4,18-5	1	Jan	1	5	5	f	yes
4,18-5	1	Jan	2	5	5	f	yes
4,18-5	1	Jan	3	5	5	f	yes
2,51-3,33	3	Jan	2	3,5	3	m	no
4,18-5	3	Jan	1	5,5	4,5	f	yes
0,84-1,67	1	Feb	1	2	1,5	ind	no
1,68-2,5	1	Feb	2	2,5	2	ind	no
1,68-2,5	1	Feb	3	3	2,5	ind	no
2,51-3,33	1	Feb	4	3,5	3	m	no
2,51-3,33	1	Feb	5	3,5	3	m	no
2,51-3,33	1	Feb	6	3,5	3	m	no
3,34-4,17	1	Feb	7	4	3,5	m	no
3,34-4,17	1	Feb	8	4	3,5	m	no
3,34-4,17	1	Feb	9	4	3,5	m	no
3,34-4,17	1	Feb	10	4	3,5	m	no
3,34-4,17	1	Feb	11	4	3,5	m	no
3,34-4,17	1	Feb	12	4	3,5	m	no
3,34-4,17	1	Feb	13	4,5	4	f	no
3,34-4,17	1	Feb	14	4,5	4	f	no
3,34-4,17	1	Feb	15	4,5	4	f	no

3,34-4,17	1	Feb	16	4,5	4	m	no
Size Class	**Site**	**Month**	**Specimen**	**Rostral Length (mm)**	**Carapace Length (mm)**	**Sex**	**Eggs Present**
4,18-5	1	Feb	17	5	4,5	f	yes
4,18-5	1	Feb	18	5	4,5	f	yes
4,18-5	1	Feb	19	5	4,5	f	yes
4,18-5	1	Feb	20	5,5	5	f	yes
4,18-5	1	Feb	21	5,5	5	m	no
5,01-5,83	1	Feb	22	6	5,5	f	no
1,68-2,5	2	Feb	1	1,5	1,5	ind	no
1,68-2,5	2	Feb	2	1	1	ind	no
1,68-2,5	2	Feb	3	1,5	1,5	ind	no
1,68-2,5	2	Feb	4	2	2	ind	no
2,51-3,33	2	Feb	5	2,5	2,5	m	no
2,51-3,33	2	Feb	6	2,5	2,5	m	no
2,51-3,33	2	Feb	7	3	3	m	no
2,51-3,33	2	Feb	8	3	3	f	no
3,34-4,17	2	Feb	9	3,5	3,5	m	no
3,34-4,17	2	Feb	10	3,5	3,5	m	no
3,34-4,17	2	Feb	11	3,5	3,5	m	no
3,34-4,17	2	Feb	12	3,5	3,5	m	no
3,34-4,17	2	Feb	13	4	3,5	m	no
3,34-4,17	2	Feb	14	4,5	4	m	no
3,34-4,17	2	Feb	15	4,5	4	f	yes
4,18-5	2	Feb	16	5	4,5	f	no
4,18-5	2	Feb	17	5	4,5	f	no
1,68-2,5	3	Feb	1	2	1,5	ind	no
1,68-2,5	3	Feb	2	2	1,5	ind	no
1,68-2,5	3	Feb	3	2,5	2	ind	no
2,51-3,33	3	Feb	4	3	2,5	m	no
2,51-3,33	3	Feb	5	3	2,5	m	no
2,51-3,33	3	Feb	6	3	2,5	m	no
2,51-3,33	3	Feb	7	3,5	3	f	no
2,51-3,33	3	Feb	8	3,5	3	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
2,51-3,33	3	Feb	9	3,5	3	m	no
2,51-3,33	3	Feb	10	3,5	3	m	no
2,51-3,33	3	Feb	11	3,5	3	f	yes
2,51-3,33	3	Feb	12	3,5	3	m	no
3,34-4,17	3	Feb	13	4	3,5	m	no
3,34-4,17	3	Feb	14	4	3,5	m	no
3,34-4,17	3	Feb	15	4,5	4	m	no
4,18-5	3	Feb	16	4,5	4,5	f	yes
4,18-5	3	Feb	17	4,5	4,5	f	yes
4,18-5	3	Feb	18	4,5	4,5	f	yes
4,18-5	3	Feb	19	4,5	4,5	m	no
4,18-5	3	Feb	20	4,5	4,5	f	yes
4,18-5	3	Feb	21	5	5	f	yes
4,18-5	3	Feb	22	5	5	f	yes
0,83-1,67	1	Mar	27	1,5	1,5	ind	no
0,83-1,67	1	Mar	28	1,5	1,5	ind	no
0,83-1,67	1	Mar	26	2	1,5	m	no
0,83-1,67	1	Mar	29	1,5	1,5	m	no
1,67-2,5	1	Mar	31	1,5	2	f	no
1,67-2,5	1	Mar	23	2	2	m	no
1,67-2,5	1	Mar	24	2	2	m	no
1,67-2,5	1	Mar	25	2	2	m	no
1,67-2,5	1	Mar	30	2	2	m	no
2,5-3,33	1	Mar	19	2,5	3	m	no
2,5-3,33	1	Mar	20	3	3	m	no
2,5-3,33	1	Mar	21	3	3	m	no
2,5-3,33	1	Mar	22	2,5	3	m	no
3,33-4,17	1	Mar	16	3	3,5	f	no
3,33-4,17	1	Mar	15	3,5	3,5	m	no
3,33-4,17	1	Mar	3	6	4	f	yes
3,33-4,17	1	Mar	8	4,5	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	1	Mar	11	5	4	f	no
3,33-4,17	1	Mar	5	5	4	m	no
3,33-4,17	1	Mar	10	4	4	m	no
3,33-4,17	1	Mar	12	4	4	m	no
3,33-4,17	1	Mar	13	4	4	m	no
3,33-4,17	1	Mar	14	3,5	4	m	no
3,33-4,17	1	Mar	17	4	4	m	no
3,33-4,17	1	Mar	18	4,5	4	m	no
4,17-5	1	Mar	4	6,5	4,5	m	no
4,17-5	1	Mar	6	4,5	4,5	m	no
4,17-5	1	Mar	7	5,5	4,5	m	no
4,17-5	1	Mar	9	3,5	4,5	m	no
4,17-5	1	Mar	1	5	5	f	yes
5-5,83	1	Mar	2	5	5,5	f	yes
1,67-2,5	2	Mar	11	1	2	m	no
1,67-2,5	2	Mar	30	1	2	m	no
1,67-2,5	2	Mar	32		2	m	no
1,67-2,5	2	Mar	33	1	2	m	no
1,67-2,5	2	Mar	29	2	2	m	no
1,67-2,5	2	Mar	28		2,5	f	no
1,67-2,5	2	Mar	22	3,5	2,5	m	no
1,67-2,5	2	Mar	31	2	2,5	m	no
2,5-3,33	2	Mar	6	5	3	f	no
2,5-3,33	2	Mar	25	2	3	f	no
2,5-3,33	2	Mar	26	1,5	3	f	no
2,5-3,33	2	Mar	5	5	3	m	no
2,5-3,33	2	Mar	18	3	3	m	no
2,5-3,33	2	Mar	21		3	m	no
2,5-3,33	2	Mar	27	1,5	3	m	no
3,33-4,17	2	Mar	7	5,5	3,5	f	no
3,33-4,17	2	Mar	15	2,5	3,5	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	Mar	4	4	3,5	m	no
3,33-4,17	2	Mar	10	4,5	3,5	m	no
3,33-4,17	2	Mar	14	4	3,5	m	no
3,33-4,17	2	Mar	16	4	3,5	m	no
3,33-4,17	2	Mar	19	6	3,5	m	no
3,33-4,17	2	Mar	24	3,5	3,5	m	no
3,33-4,17	2	Mar	8	5	3,5	ind	no
3,33-4,17	2	Mar	23	3,5	3,5	ind	no
3,33-4,17	2	Mar	1	5	4	f	yes
3,33-4,17	2	Mar	2	5	4	f	yes
3,33-4,17	2	Mar	3	5	4	f	yes
3,33-4,17	2	Mar	17	4	4	f	no
3,33-4,17	2	Mar	9	5	4	m	no
3,33-4,17	2	Mar	12	4	4	m	no
3,33-4,17	2	Mar	13	4	4	m	no
3,33-4,17	2	Mar	20	4	4	m	no
3,33-4,17	3	Mar	8	5	3,5	f	no
3,33-4,17	3	Mar	9	4	3,5	m	no
3,33-4,17	3	Mar	11	3	3,5	m	no
3,33-4,17	3	Mar	4	5	4	f	yes
3,33-4,17	3	Mar	5	4	4	f	yes
3,33-4,17	3	Mar	10	3	4	f	no
3,33-4,17	3	Mar	6	5	4	m	no
3,33-4,17	3	Mar	12	5	4	m	no
4,17-5	3	Mar	1	4,5	4,5	f	yes
4,17-5	3	Mar	7	4	4,5	f	no
4,17-5	3	Mar	2	4,5	5	f	yes
4,17-5	3	Mar	3	4	5	f	yes
1,67-2,5	1	Apr	3	3	2,5	ind	no
2,5-3,33	1	Apr	4	3	3	m	no
2,5-3,33	1	Apr	5	2,5	3	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
2,5-3,33	1	Apr	4	3,5	3	m	no
3,33-4,17	1	Apr	2	4,5	3,5	m	no
3,33-4,17	1	Apr	3	3,5	3,5	m	no
3,33-4,17	1	Apr	2	3	3,5	ind	no
3,33-4,17	1	Apr	1	5	4	f	yes
3,33-4,17	1	Apr	1	4	4	m	no
4,17-5	1	Apr	1	6	5	m	no
0,83-1,67	1	May	24	1,5	1,5	ind	no
1,67-2,5	1	May	23	2	2	m	no
1,67-2,5	1	May	22	2,5	2,5	m	no
2,5-3,33	1	May	20	2,5	3	f	no
2,5-3,33	1	May	7	5	3	m	no
2,5-3,33	1	May	9		3	m	no
2,5-3,33	1	May	12	3,5	3	m	no
2,5-3,33	1	May	13	3,5	3	m	no
2,5-3,33	1	May	14	3,5	3	m	no
2,5-3,33	1	May	18	3	3	m	no
2,5-3,33	1	May	19	3	3	m	no
2,5-3,33	1	May	21	3	3	m	no
3,33-4,17	1	May	3		3,5	m	no
3,33-4,17	1	May	5	4	3,5	m	no
3,33-4,17	1	May	11	4,5	3,5	m	no
3,33-4,17	1	May	15	3,5	3,5	m	no
3,33-4,17	1	May	16	3,5	3,5	m	no
3,33-4,17	1	May	17	3	3,5	m	no
3,33-4,17	1	May	4	4	4	f	no
3,33-4,17	1	May	2	5	4	m	no
3,33-4,17	1	May	6	4	4	m	no
3,33-4,17	1	May	10	5	4	m	no
4,17-5	1	May	1	5	4,5	f	yes
4,17-5	1	May	8	4	5	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
1,67-2,5	2	May	59	2,5	2	m	no
1,67-2,5	2	May	60	2	2	m	no
1,67-2,5	2	May	62	2	2	m	no
1,67-2,5	2	May	50	2	2,5	f	no
1,67-2,5	2	May	56	2,5	2,5	m	no
1,67-2,5	2	May	57	3	2,5	m	no
1,67-2,5	2	May	58	2,5	2,5	m	no
1,67-2,5	2	May	61	2,5	2,5	m	no
2,5-3,33	2	May	27		3	m	no
2,5-3,33	2	May	37	3	3	m	no
2,5-3,33	2	May	48	2,5	3	m	no
2,5-3,33	2	May	51		3	m	no
2,5-3,33	2	May	52	3	3	m	no
2,5-3,33	2	May	53		3	m	no
2,5-3,33	2	May	54	3	3	m	no
2,5-3,33	2	May	55	3	3	m	no
3,33-4,17	2	May	35	4,5	3,5	f	no
3,33-4,17	2	May	42	3,5	3,5	f	no
3,33-4,17	2	May	44	3,5	3,5	f	no
3,33-4,17	2	May	49	2,5	3,5	f	no
3,33-4,17	2	May	31	4,5	3,5	m	no
3,33-4,17	2	May	34	5,5	3,5	m	no
3,33-4,17	2	May	38	3,5	3,5	m	no
3,33-4,17	2	May	40	4	3,5	m	no
3,33-4,17	2	May	43	3,5	3,5	m	no
3,33-4,17	2	May	45	4	3,5	m	no
3,33-4,17	2	May	46	3	3,5	m	no
3,33-4,17	2	May	47	3	3,5	m	no
3,33-4,17	2	May	41	4	3,5	ind	no
3,33-4,17	2	May	4	5	4	f	yes
3,33-4,17	2	May	28	4	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	May	29	4	4	f	no
3,33-4,17	2	May	23		4	m	no
3,33-4,17	2	May	24		4	m	no
3,33-4,17	2	May	25	4	4	m	no
3,33-4,17	2	May	32	5,5	4	m	no
3,33-4,17	2	May	33	5	4	m	no
3,33-4,17	2	May	36	4	4	m	no
3,33-4,17	2	May	39	4	4	m	no
4,17-5	2	May	3	5	4,5	f	yes
4,17-5	2	May	15	4,5	4,5	f	no
4,17-5	2	May	22	4,5	4,5	f	no
4,17-5	2	May	30	4,5	4,5	f	no
4,17-5	2	May	17	4,5	4,5	m	no
4,17-5	2	May	20	5	4,5	m	no
4,17-5	2	May	1	5	5	f	yes
4,17-5	2	May	2	5	5	f	yes
4,17-5	2	May	5	5	5	f	yes
4,17-5	2	May	7	4,5	5	f	yes
4,17-5	2	May	8	5	5	f	yes
4,17-5	2	May	10	5,5	5	f	yes
4,17-5	2	May	12	5	5	f	yes
4,17-5	2	May	13	4,5	5	f	yes
4,17-5	2	May	18	5	5	f	no
4,17-5	2	May	21	5	5	f	no
4,17-5	2	May	14	5	5	m	no
4,17-5	2	May	16	4,5	5	m	no
4,17-5	2	May	26	3,5	5	m	no
5-5,83	2	May	9	4,5	5,5	f	yes
5-5,83	2	May	11	4,5	5,5	f	yes
5-5,83	2	May	19	4,5	5,5	m	no
5,83-6,67	2	May	6	5	6	f	yes

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
0,83-1,67	3	May	43	1	1,5	ind	no
1,67-2,5	3	May	37	2	2	f	no
1,67-2,5	3	May	30	2	2	m	no
1,67-2,5	3	May	32	2	2	m	no
1,67-2,5	3	May	33	2,5	2	m	no
1,67-2,5	3	May	36	2	2	m	no
1,67-2,5	3	May	38	2	2	m	no
1,67-2,5	3	May	39	2	2	m	no
1,67-2,5	3	May	40	2,5	2	m	no
1,67-2,5	3	May	41	1,5	2	m	no
1,67-2,5	3	May	42	1,5	2	m	no
1,67-2,5	3	May	29	2,5	2,5	m	no
1,67-2,5	3	May	34	2,5	2,5	m	no
2,5-3,33	3	May	19	4	3	f	yes
2,5-3,33	3	May	31	3	3	f	no
2,5-3,33	3	May	26	3,5	3	m	no
2,5-3,33	3	May	27	3	3	m	no
2,5-3,33	3	May	35	3	3	m	no
3,33-4,17	3	May	10	4,5	3,5	f	yes
3,33-4,17	3	May	20	3,5	3,5	f	no
3,33-4,17	3	May	24	3,5	3,5	m	no
3,33-4,17	3	May	25	3,5	3,5	m	no
3,33-4,17	3	May	28	3,5	3,5	m	no
3,33-4,17	3	May	7	5	4	f	yes
3,33-4,17	3	May	8	4	4	f	yes
3,33-4,17	3	May	9	4	4	f	yes
3,33-4,17	3	May	12	3,5	4	f	yes
3,33-4,17	3	May	17	3,5	4	f	yes
3,33-4,17	3	May	18	5	4	f	yes
3,33-4,17	3	May	21	4,5	4	f	no
3,33-4,17	3	May	22	4	4	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	3	May	23	4	4	m	no
4,17-5	3	May	1	4	4,5	f	yes
4,17-5	3	May	2	5	4,5	f	yes
4,17-5	3	May	6	5	4,5	f	yes
4,17-5	3	May	11	4,5	4,5	f	yes
4,17-5	3	May	13	6	4,5	f	yes
4,17-5	3	May	14	3,5	4,5	f	yes
4,17-5	3	May	15	3,5	4,5	f	yes
4,17-5	3	May	3	4	5	f	yes
4,17-5	3	May	4	5	5	f	yes
4,17-5	3	May	5	3	5	f	yes
4,17-5	3	May	16	4	5	f	yes
1,67-2,5	1	Jun	14	3	2,5	ind	no
1,67-2,5	1	Jun	16	2,5	2,5	ind	no
2,5-3,33	1	Jun	11	3	3	f	no
2,5-3,33	1	Jun	15	3	3	m	no
2,5-3,33	1	Jun	17	2,5	3	m	no
3,33-4,17	1	Jun	13	4	3,5	f	no
3,33-4,17	1	Jun	4	4,5	3,5	m	no
3,33-4,17	1	Jun	7	3,5	3,5	m	no
3,33-4,17	1	Jun	10	3,5	3,5	m	no
3,33-4,17	1	Jun	9		4	f	no
3,33-4,17	1	Jun	3	4	4	m	no
3,33-4,17	1	Jun	6	5,5	4	m	no
3,33-4,17	1	Jun	12	4	4	m	no
3,33-4,17	1	Jun	8	4	4	ind	no
4,17-5	1	Jun	1	5,5	4,5	m	no
4,17-5	1	Jun	2	5,5	4,5	m	no
4,17-5	1	Jun	5	5,5	5	f	no
1,67-2,5	2	Jun	16		2	m	no
1,67-2,5	2	Jun	17	3	2	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
1,67-2,5	2	Jun	14	2	2,5	m	no
2,5-3,33	2	Jun	5	6	3	f	no
2,5-3,33	2	Jun	8	4	3	f	no
2,5-3,33	2	Jun	13	3	3	f	no
2,5-3,33	2	Jun	15	2	3	ind	no
3,33-4,17	2	Jun	1	4	4	f	no
3,33-4,17	2	Jun	2		4	f	no
3,33-4,17	2	Jun	9	4	4	f	no
3,33-4,17	2	Jun	11	4	4	f	no
3,33-4,17	2	Jun	10	3	4	m	no
3,33-4,17	2	Jun	12	4	4	m	no
4,17-5	2	Jun	4	4,5	4,5	f	no
4,17-5	2	Jun	6		4,5	f	no
4,17-5	2	Jun	3	5	5	f	no
4,17-5	2	Jun	7	4	5	m	no
2,5-3,33	3	Jun	6	4	3	ind	no
3,33-4,17	3	Jun	5	2,5	3,5	f	no
3,33-4,17	3	Jun	3		4	f	no
3,33-4,17	3	Jun	1	4	4	m	no
3,33-4,17	3	Jun	4	4	4	m	no
4,17-5	3	Jun	2	5,5	4,5	m	no
3,33-4,17	1	Jul	17	3,5	3,5	f	no
3,33-4,17	1	Jul	12		4	f	no
3,33-4,17	1	Jul	19	4	4	f	no
3,33-4,17	1	Jul	15	3	4	m	no
3,33-4,17	1	Jul	22	4	4	m	no
3,33-4,17	1	Jul	23	4	4	m	no
4,17-5	1	Jul	5		4,5	f	yes
4,17-5	1	Jul	7	6	4,5	f	yes
4,17-5	1	Jul	14	4,5	4,5	f	no
4,17-5	1	Jul	9	5,5	4,5	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
4,17-5	1	Jul	16	3,5	4,5	m	no
4,17-5	1	Jul	20	4	4,5	m	no
4,17-5	1	Jul	25	4	4,5	m	no
4,17-5	1	Jul	1	5	5	f	yes
4,17-5	1	Jul	2	4,5	5	f	yes
4,17-5	1	Jul	3	5	5	f	yes
4,17-5	1	Jul	10	4	5	f	no
4,17-5	1	Jul	11	6	5	f	no
4,17-5	1	Jul	18	4	5	f	no
4,17-5	1	Jul	24	4	5	f	no
4,17-5	1	Jul	8	5	5	m	no
4,17-5	1	Jul	13	4	5	m	no
5-5,83	1	Jul	4	5,5	5,5	f	yes
5,83-6,67	1	Jul	6	5	6	f	yes
5,83-6,67	1	Jul	21	4	6	f	no
1,67-2,5	2	Jul	48	2,5	2,5	m	no
2,5-3,33	2	Jul	31		3	m	no
2,5-3,33	2	Jul	42	2	3	m	no
2,5-3,33	2	Jul	45	2	3	m	no
2,5-3,33	2	Jul	46	3	3	m	no
2,5-3,33	2	Jul	47	2	3	ind	no
3,33-4,17	2	Jul	30	3,5	3,5	f	no
3,33-4,17	2	Jul	34	3,5	3,5	f	no
3,33-4,17	2	Jul	27	2,5	3,5	m	no
3,33-4,17	2	Jul	29	3,5	3,5	m	no
3,33-4,17	2	Jul	35	4,5	3,5	m	no
3,33-4,17	2	Jul	43	3	3,5	m	no
3,33-4,17	2	Jul	38	2	3,5	ind	no
3,33-4,17	2	Jul	11		4	f	yes
3,33-4,17	2	Jul	16	4	4	f	no
3,33-4,17	2	Jul	19	5	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	Jul	20	5	4	f	no
3,33-4,17	2	Jul	21	6	4	f	no
3,33-4,17	2	Jul	24	4	4	f	no
3,33-4,17	2	Jul	25	3,5	4	f	no
3,33-4,17	2	Jul	36		4	f	no
3,33-4,17	2	Jul	37	4,5	4	f	no
3,33-4,17	2	Jul	40		4	f	no
3,33-4,17	2	Jul	44	4	4	f	no
3,33-4,17	2	Jul	15	5	4	m	no
3,33-4,17	2	Jul	22	5	4	m	no
3,33-4,17	2	Jul	26	3	4	m	no
3,33-4,17	2	Jul	32	5	4	m	no
3,33-4,17	2	Jul	33	3	4	m	no
3,33-4,17	2	Jul	41	5	4	m	no
4,17-5	2	Jul	3		4,5	f	yes
4,17-5	2	Jul	12	5,5	4,5	f	yes
4,17-5	2	Jul	14	4,5	4,5	f	yes
4,17-5	2	Jul	28	4	4,5	f	no
4,17-5	2	Jul	39	4,5	4,5	f	no
4,17-5	2	Jul	17	5	4,5	m	no
4,17-5	2	Jul	2	5	5	f	yes
4,17-5	2	Jul	6	6	5	f	yes
4,17-5	2	Jul	9	5	5	f	yes
4,17-5	2	Jul	13	5,5	5	f	yes
4,17-5	2	Jul	4	6	5	m	no
4,17-5	2	Jul	18	4	5	m	no
5-5,83	2	Jul	5	7	5,5	f	yes
5-5,83	2	Jul	8	5,5	5,5	f	yes
5-5,83	2	Jul	10	6	5,5	f	yes
5,83-6,67	2	Jul	1		6	f	yes
5,83-6,67	2	Jul	7	7	6	f	yes

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
5,83-6,67	2	Jul	23	5	6	f	no
2,5-3,33	3	Jul	11	5,5	3	m	no
2,5-3,33	3	Jul	17	2,5	3	m	no
3,33-4,17	3	Jul	13	4,5	4	f	no
3,33-4,17	3	Jul	14	4	4	f	no
3,33-4,17	3	Jul	15	4	4	f	no
3,33-4,17	3	Jul	16	4	4	f	no
3,33-4,17	3	Jul	6	6	4	m	no
3,33-4,17	3	Jul	10	6,5	4	m	no
4,17-5	3	Jul	2	6	4,5	f	no
4,17-5	3	Jul	8	5,5	4,5	f	no
4,17-5	3	Jul	12	4,5	4,5	f	no
4,17-5	3	Jul	9	7	4,5	m	no
4,17-5	3	Jul	1	6	5	f	no
4,17-5	3	Jul	4	5	5	f	no
4,17-5	3	Jul	5	5	5	f	no
4,17-5	3	Jul	7	5	5	f	yes
5-5,83	3	Jul	3	6	5,5	f	yes
3,33-4,17	1	Aug	8	4	4	f	no
3,33-4,17	1	Aug	6	5	4	m	no
3,33-4,17	1	Aug	9	5	4	m	no
3,33-4,17	1	Aug	7	4	4	ind	no
4,17-5	1	Aug	2	5	5	m	no
4,17-5	1	Aug	5		5	m	no
5-5,83	1	Aug	1	5,5	5,5	m	no
5-5,83	1	Aug	3		5,5	m	no
5,83-6,67	1	Aug	4	5	6	m	no
3,33-4,17	3	Aug	4	5,5	3,5	m	no
3,33-4,17	3	Aug	5	6,5	4	m	no
3,33-4,17	3	Aug	7	5	4	m	no
3,33-4,17	3	Aug	8	4	4	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	3	Aug	2	4	4	ind	no
4,17-5	3	Aug	1	4	4,5	f	yes
4,17-5	3	Aug	3	5	4,5	m	no
4,17-5	3	Aug	6	5	4,5	m	no
4,17-5	3	Aug	9	6	5	m	no
2,5-3,33	1	Sep	24	2	3	m	no
3,33-4,17	1	Sep	7	4,5	3,5	f	no
3,33-4,17	1	Sep	5	3,5	3,5	m	no
3,33-4,17	1	Sep	22	5	4	f	no
3,33-4,17	1	Sep	4	4	4	m	no
3,33-4,17	1	Sep	6	5	4	m	no
3,33-4,17	1	Sep	11	4	4	m	no
3,33-4,17	1	Sep	18	4	4	m	no
3,33-4,17	1	Sep	19	4,5	4	m	no
3,33-4,17	1	Sep	20	3,5	4	m	no
3,33-4,17	1	Sep	23	3	4	m	no
3,33-4,17	1	Sep	25		4	m	no
4,17-5	1	Sep	8	5	4,5	m	no
4,17-5	1	Sep	15	4	4,5	m	no
4,17-5	1	Sep	16	4,5	4,5	m	no
4,17-5	1	Sep	17	4,5	4,5	m	no
4,17-5	1	Sep	21	3	4,5	m	no
4,17-5	1	Sep	1		5	f	yes
4,17-5	1	Sep	2	5	5	f	yes
4,17-5	1	Sep	3	6	5	f	no
4,17-5	1	Sep	9	4	5	m	no
4,17-5	1	Sep	10	4	5	m	no
4,17-5	1	Sep	12	5	5	m	no
4,17-5	1	Sep	13	5	5	m	no
4,17-5	1	Sep	14	4,5	5	m	no
1,67-2,5	2	Sep	35	2	2,5	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
2,5-3,33	2	Sep	22	4,5	3	f	no
2,5-3,33	2	Sep	29	2,5	3	f	no
2,5-3,33	2	Sep	74	3	3	f	no
2,5-3,33	2	Sep	101	1,5	3	m	no
2,5-3,33	2	Sep	34	3	3	m	no
2,5-3,33	2	Sep	65		3	m	no
2,5-3,33	2	Sep	73	2,5	3	m	no
2,5-3,33	2	Sep	100	3	3	m	no
2,5-3,33	2	Sep	102	2	3	m	no
3,33-4,17	2	Sep	59		3,5	f	no
3,33-4,17	2	Sep	97		3,5	f	no
3,33-4,17	2	Sep	64	3,5	3,5	m	no
3,33-4,17	2	Sep	86	5	3,5	m	no
3,33-4,17	2	Sep	94	4	3,5	m	no
3,33-4,17	2	Sep	98	3	3,5	m	no
3,33-4,17	2	Sep	99	3	3,5	m	no
3,33-4,17	2	Sep	11	5	4	f	yes
3,33-4,17	2	Sep	16	5	4	f	no
3,33-4,17	2	Sep	30	4	4	f	no
3,33-4,17	2	Sep	71	4	4	f	no
3,33-4,17	2	Sep	72	3,5	4	f	no
3,33-4,17	2	Sep	88	4	4	f	no
3,33-4,17	2	Sep	92	4	4	f	no
3,33-4,17	2	Sep	96	2	4	f	no
3,33-4,17	2	Sep	108	4	4	f	no
3,33-4,17	2	Sep	24	4	4	m	no
3,33-4,17	2	Sep	26		4	m	no
3,33-4,17	2	Sep	28	3	4	m	no
3,33-4,17	2	Sep	33	4	4	m	no
3,33-4,17	2	Sep	48	3	4	m	no
3,33-4,17	2	Sep	49	4	4	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	Sep	58	5	4	m	no
3,33-4,17	2	Sep	61	3	4	m	no
3,33-4,17	2	Sep	62	4,5	4	m	no
3,33-4,17	2	Sep	68		4	m	no
3,33-4,17	2	Sep	69	3	4	m	no
3,33-4,17	2	Sep	70	4	4	m	no
3,33-4,17	2	Sep	89	4	4	m	no
3,33-4,17	2	Sep	93	4	4	m	no
3,33-4,17	2	Sep	95	3	4	m	no
3,33-4,17	2	Sep	103	2	4	m	no
3,33-4,17	2	Sep	105	4	4	m	no
3,33-4,17	2	Sep	106		4	m	no
3,33-4,17	2	Sep	107	3	4	m	no
4,17-5	2	Sep	1	4,5	4,5	f	yes
4,17-5	2	Sep	14	4,5	4,5	f	yes
4,17-5	2	Sep	17	5,5	4,5	f	no
4,17-5	2	Sep	18	6	4,5	f	no
4,17-5	2	Sep	27	4,5	4,5	f	no
4,17-5	2	Sep	32	3	4,5	f	no
4,17-5	2	Sep	37	4	4,5	f	no
4,17-5	2	Sep	44	4,5	4,5	f	no
4,17-5	2	Sep	47	4,5	4,5	f	no
4,17-5	2	Sep	60	4	4,5	f	no
4,17-5	2	Sep	76	4	4,5	f	no
4,17-5	2	Sep	83	4,5	4,5	f	no
4,17-5	2	Sep	84	4,5	4,5	f	no
4,17-5	2	Sep	15	5,5	4,5	m	no
4,17-5	2	Sep	21	4	4,5	m	no
4,17-5	2	Sep	31	4,5	4,5	m	no
4,17-5	2	Sep	50	5,5	4,5	m	no
4,17-5	2	Sep	51	5	4,5	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
4,17-5	2	Sep	54	3,5	4,5	m	no
4,17-5	2	Sep	66	4,5	4,5	m	no
4,17-5	2	Sep	2	5,5	5	f	yes
4,17-5	2	Sep	3	5	5	f	yes
4,17-5	2	Sep	6		5	f	yes
4,17-5	2	Sep	7	5,5	5	f	yes
4,17-5	2	Sep	13	6	5	f	yes
4,17-5	2	Sep	19	4,5	5	f	no
4,17-5	2	Sep	38	3,5	5	f	no
4,17-5	2	Sep	39	4	5	f	no
4,17-5	2	Sep	40	5	5	f	no
4,17-5	2	Sep	46		5	f	no
4,17-5	2	Sep	52	5	5	f	no
4,17-5	2	Sep	53	5	5	f	no
4,17-5	2	Sep	67	4	5	f	no
4,17-5	2	Sep	75		5	f	no
4,17-5	2	Sep	81	5	5	f	no
4,17-5	2	Sep	82	4	5	f	no
4,17-5	2	Sep	25	4,5	5	m	no
4,17-5	2	Sep	55	5	5	m	no
4,17-5	2	Sep	57	4	5	m	no
4,17-5	2	Sep	63	4,5	5	m	no
4,17-5	2	Sep	85	4	5	m	no
4,17-5	2	Sep	90	4	5	m	no
4,17-5	2	Sep	91	5	5	m	no
4,17-5	2	Sep	104	2	5	m	no
5-5,83	2	Sep	10	5,5	5,5	f	yes
5-5,83	2	Sep	42	5	5,5	f	no
5-5,83	2	Sep	45	5	5,5	f	no
5-5,83	2	Sep	78	5,5	5,5	f	no
5-5,83	2	Sep	20	5	5,5	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
5,83-6,67	2	Sep	4	6,5	6	f	yes
5,83-6,67	2	Sep	5	5,5	6	f	yes
5,83-6,67	2	Sep	8	6	6	f	yes
5,83-6,67	2	Sep	9	5	6	f	yes
5,83-6,67	2	Sep	12	5,5	6	f	yes
5,83-6,67	2	Sep	41	5	6	f	no
5,83-6,67	2	Sep	43	5	6	f	no
5,83-6,67	2	Sep	56	5	6	f	no
5,83-6,67	2	Sep	77	4	6	f	no
5,83-6,67	2	Sep	87	6	6	f	no
5,83-6,67	2	Sep	23	4	6	m	no
5,83-6,67	2	Sep	79	4,5	6,5	f	no
5,83-6,67	2	Sep	80	4	6,5	m	no
6,67-7,5	2	Sep	36	5	7	f	no
3,33-4,17	3	Sep	29		3,5	m	no
3,33-4,17	3	Sep	31		3,5	m	no
3,33-4,17	3	Sep	14	6	4	f	yes
3,33-4,17	3	Sep	27	4	4	f	yes
3,33-4,17	3	Sep	24	5	4	m	no
3,33-4,17	3	Sep	26	4	4	m	no
4,17-5	3	Sep	13	4,5	4,5	f	yes
4,17-5	3	Sep	22	4	4,5	f	no
4,17-5	3	Sep	16	4	4,5	m	yes
4,17-5	3	Sep	28	4	4,5	m	no
4,17-5	3	Sep	30		4,5	m	no
4,17-5	3	Sep	2	5	5	f	yes
4,17-5	3	Sep	4	5	5	f	yes
4,17-5	3	Sep	5	5	5	f	yes
4,17-5	3	Sep	7	5	5	f	yes
4,17-5	3	Sep	9	6	5	f	yes
4,17-5	3	Sep	12	5	5	f	yes

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
4,17-5	3	Sep	18	4	5	f	no
4,17-5	3	Sep	15	4	5	m	yes
4,17-5	3	Sep	17	5	5	m	yes
4,17-5	3	Sep	19		5	m	no
4,17-5	3	Sep	20	4	5	m	no
4,17-5	3	Sep	21	5	5	m	no
4,17-5	3	Sep	23	5	5	m	no
4,17-5	3	Sep	25		5	m	no
5-5,83	3	Sep	6	5	5,5	f	yes
5,83-6,67	3	Sep	1	7	6	f	yes
5,83-6,67	3	Sep	3	5	6	f	yes
5,83-6,67	3	Sep	8	5	6	f	yes
5,83-6,67	3	Sep	10		6	f	yes
5,83-6,67	3	Sep	11	4	6	f	yes
1,67-2,5	1	Oct	14	2	2,5	f	no
2,5-3,33	1	Oct	9	4,5	3	m	no
2,5-3,33	1	Oct	13		3	m	no
2,5-3,33	1	Oct	15	2,5	3	ind	no
3,33-4,17	1	Oct	10	3,5	3,5	f	no
3,33-4,17	1	Oct	12	4	3,5	m	no
3,33-4,17	1	Oct	6	5	4	f	yes
3,33-4,17	1	Oct	8	6,5	4	f	yes
4,17-5	1	Oct	2		5	f	yes
4,17-5	1	Oct	3	6	5	f	yes
4,17-5	1	Oct	4	4	5	f	yes
4,17-5	1	Oct	5	6	5	f	yes
4,17-5	1	Oct	7	4	5	f	yes
4,17-5	1	Oct	11	5,5	5	m	no
5,83-6,67	1	Oct	1	5	6	f	yes
2,5-3,33	2	Oct	15	3	3	m	no
3,33-4,17	2	Oct	11	4	4	f	yes

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	Oct	17	5	4	m	no
3,33-4,17	2	Oct	20	6	4	m	no
4,17-5	2	Oct	1	4	4,5	f	yes
4,17-5	2	Oct	2	5	5	f	yes
4,17-5	2	Oct	7	4	5	f	yes
4,17-5	2	Oct	8	5	5	f	yes
4,17-5	2	Oct	9	6	5	f	yes
4,17-5	2	Oct	10	4	5	f	yes
4,17-5	2	Oct	12	4	5	f	yes
4,17-5	2	Oct	18	6	5	f	no
4,17-5	2	Oct	19	4	5	ind	no
5-5,83	2	Oct	5	5,5	5,5	f	yes
5-5,83	2	Oct	6	6	5,5	f	yes
5-5,83	2	Oct	13	5	5,5	f	yes
5,83-6,67	2	Oct	3	6	6	f	yes
5,83-6,67	2	Oct	4	5	6	f	yes
5,83-6,67	2	Oct	14	4	6	f	yes
5,83-6,67	2	Oct	16	5	6	m	no
3,33-4,17	3	Oct	13	3	4	f	no
3,33-4,17	3	Oct	11	5	4	m	no
4,17-5	3	Oct	9	5	4,5	f	no
4,17-5	3	Oct	12	7	4,5	m	no
4,17-5	3	Oct	1	5	5	f	yes
4,17-5	3	Oct	3	5	5	f	yes
4,17-5	3	Oct	6	6	5	m	no
4,17-5	3	Oct	8		5	m	no
4,17-5	3	Oct	10	6	5	m	no
4,17-5	3	Oct	7	4	5	ind	no
5-5,83	3	Oct	2	6	5,5	f	yes
5-5,83	3	Oct	4	6,5	5,5	f	yes
5-5,83	3	Oct	5	7	5,5	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
0,83-1,67	1	Nov	7	2	1,5	m	no
2,5-3,33	1	Nov	5	4	2,5	m	no
2,5-3,33	1	Nov	4	4,5	3	m	no
2,5-3,33	1	Nov	6	3	3	m	no
3,33-4,17	1	Nov	1	5	3,5	f	yes
3,33-4,17	1	Nov	2	5	3,5	f	yes
3,33-4,17	1	Nov	3	4,5	3,5	f	yes
2,5-3,33	2	Nov	2	5	2,5	m	no
2,5-3,33	2	Nov	3	3,5	2,5	m	no
2,5-3,33	2	Nov	5	4	3	m	no
3,33-4,17	2	Nov	4	4,5	3,5	m	no
3,33-4,17	2	Nov	1	5	4	f	yes
2,5-3,33	3	Nov	2	5	2,5	m	no
2,5-3,33	3	Nov	3	4,5	3	m	no
3,33-4,17	3	Nov	1	5	4	f	yes
1,67-2,5	1	Dec	33	2,5	2	m	no
1,67-2,5	1	Dec	29	3	2,5	f	no
1,67-2,5	1	Dec	27	3,5	2,5	m	no
1,67-2,5	1	Dec	28	2	2,5	m	no
1,67-2,5	1	Dec	30	3	2,5	m	no
1,67-2,5	1	Dec	31	2,5	2,5	m	no
2,5-3,33	1	Dec	20	4	3	f	no
2,5-3,33	1	Dec	24	2,5	3	f	no
2,5-3,33	1	Dec	26	3	3	f	no
2,5-3,33	1	Dec	34	3	3	f	no
2,5-3,33	1	Dec	23	3,5	3	m	no
2,5-3,33	1	Dec	32	2,5	3	m	no
3,33-4,17	1	Dec	18	3,5	3,5	f	no
3,33-4,17	1	Dec	25	3,5	3,5	f	no
3,33-4,17	1	Dec	11	7	3,5	m	no
3,33-4,17	1	Dec	16	3,5	3,5	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	1	Dec	17	4,5	3,5	m	no
3,33-4,17	1	Dec	21	3,5	3,5	m	no
3,33-4,17	1	Dec	22	3	3,5	m	no
3,33-4,17	1	Dec	7	4	4	f	no
3,33-4,17	1	Dec	10	4,5	4	f	no
3,33-4,17	1	Dec	13	5	4	f	no
3,33-4,17	1	Dec	14	3	4	f	no
3,33-4,17	1	Dec	15	3,5	4	f	no
3,33-4,17	1	Dec	19	3,5	4	f	no
3,33-4,17	1	Dec	9	4	4	m	no
3,33-4,17	1	Dec	12	5	4	m	no
4,17-5	1	Dec	8	5	4,5	f	no
4,17-5	1	Dec	3	5	4,5	m	no
4,17-5	1	Dec	4	4,5	4,5	m	no
4,17-5	1	Dec	5	5,5	4,5	m	no
4,17-5	1	Dec	6	4,5	4,5	m	no
4,17-5	1	Dec	1	7	5	f	yes
4,17-5	1	Dec	2	5	5	m	no
1,67-2,5	2	Dec	28	3,5	2	f	no
1,67-2,5	2	Dec	32	2,5	2,5	f	no
1,67-2,5	2	Dec	33	2,5	2,5	f	no
1,67-2,5	2	Dec	34	2,5	2,5	m	no
2,5-3,33	2	Dec	24	3,5	3	f	no
2,5-3,33	2	Dec	29	3	3	f	no
2,5-3,33	2	Dec	31	3	3	m	no
3,33-4,17	2	Dec	14		3,5	m	no
3,33-4,17	2	Dec	16	4,5	3,5	m	no
3,33-4,17	2	Dec	25	3	3,5	m	no
3,33-4,17	2	Dec	27	3,5	3,5	m	no
3,33-4,17	2	Dec	30	2,5	3,5	m	no
3,33-4,17	2	Dec	21	4	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	Dec	22	4,5	4	f	no
3,33-4,17	2	Dec	7	4	4	m	no
3,33-4,17	2	Dec	8	4	4	m	no
3,33-4,17	2	Dec	10	4	4	m	no
3,33-4,17	2	Dec	11	4,5	4	m	no
3,33-4,17	2	Dec	15	4	4	m	no
3,33-4,17	2	Dec	17	5	4	m	no
3,33-4,17	2	Dec	20	3,5	4	m	no
3,33-4,17	2	Dec	23	4	4	m	no
3,33-4,17	2	Dec	26	4,5	4	m	no
4,17-5	2	Dec	12	4	4,5	f	no
4,17-5	2	Dec	19	4	4,5	f	no
4,17-5	2	Dec	5	5	4,5	m	no
4,17-5	2	Dec	9	4	4,5	m	no
4,17-5	2	Dec	13	4,5	4,5	m	no
4,17-5	2	Dec	18	4,5	4,5	m	no
4,17-5	2	Dec	3	5	5	f	yes
4,17-5	2	Dec	4	5	5	f	yes
4,17-5	2	Dec	6	5	5	m	no
5-5,83	2	Dec	1	5	5,5	f	yes
5-5,83	2	Dec	2	5	5,5	f	yes
1,67-2,5	3	Dec	19	2	2,5	m	no
2,5-3,33	3	Dec	5	4	3	f	no
2,5-3,33	3	Dec	13	3,5	3	f	no
2,5-3,33	3	Dec	20		3	m	no
3,33-4,17	3	Dec	12	4	3,5	f	no
3,33-4,17	3	Dec	14	3	3,5	f	no
3,33-4,17	3	Dec	15	4	3,5	m	no
3,33-4,17	3	Dec	21	2,5	3,5	m	no
3,33-4,17	3	Dec	2	6	4	f	yes
3,33-4,17	3	Dec	7	4	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	3	Dec	8	4	4	f	no
3,33-4,17	3	Dec	10	4	4	f	no
3,33-4,17	3	Dec	16	4	4	f	no
3,33-4,17	3	Dec	6	4	4	m	no
3,33-4,17	3	Dec	9	4,5	4	m	no
3,33-4,17	3	Dec	17	5	4	m	no
3,33-4,17	3	Dec	18	4	4	m	no
4,17-5	3	Dec	3	3,5	4,5	f	yes
4,17-5	3	Dec	4	4,5	4,5	m	no
4,17-5	3	Dec	11	5,5	4,5	m	no
5,83-6,67	3	Dec	1	7	6	f	yes

Table S 17: Showing raw *Caridina africana* population data collected from Lake Mzingazi in 2021

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	2	Mar	2	3	4	m	no
3,33-4,17	2	Mar	1	4	4	m	no
4,17-5	2	Mar	1	5	5	f	no
2,5-3,33	2	Apr	16	NA	3	f	no
2,5-3,33	2	Apr	17	3	3	m	no
3,33-4,17	2	Apr	5	4,5	3,5	m	no
3,33-4,17	2	Apr	8	6	3,5	m	no
3,33-4,17	2	Apr	10	4,5	3,5	m	no
3,33-4,17	2	Apr	11	4,5	3,5	m	no
3,33-4,17	2	Apr	14	3,5	3,5	m	no
3,33-4,17	2	Apr	15	4	3,5	m	no
3,33-4,17	2	Apr	3	5	4	m	no
3,33-4,17	2	Apr	6	4	4	m	no
3,33-4,17	2	Apr	7	4	4	m	no
3,33-4,17	2	Apr	9	5,5	4	m	no
3,33-4,17	2	Apr	12	4	4	m	no
4,17-5	2	Apr	4	6	4,5	m	no
4,17-5	2	Apr	13	3,5	4,5	m	no
4,17-5	2	Apr	2	5	5	f	no
4,17-5	2	Apr	1	6	5	f	yes
3,33-4,17	3	Apr	1	5	4	m	no
3,33-4,17	1	May	1	3	4	m	no
4,17-5	1	May	2	3,5	4,5	f	yes
0,83-1,67	2	May	1	1	1,5	m	no
0,83-1,67	2	May	3	1	1,5	m	no
1,67-2,5	2	May	2	1,5	2	m	no
2,5-3,33	2	May	4	2,5	3	m	no
2,5-3,33	2	May	5	2,5	3	m	no
3,33-4,17	2	May	6	3	3,5	m	no
3,33-4,17	2	May	7	3,5	4	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
4,17-5	2	May	8	4	4,5	m	no
4,17-5	2	May	9	4	4,5	m	no
4,17-5	2	May	10	4	4,5	f	yes
4,17-5	2	May	11	4,5	5	f	yes
5-5,83	2	May	12	5	5,5	f	yes
5-5,83	2	May	13	5	5,5	f	yes
0,83-1,67	3	May	1	1	1,5	m	no
3,33-4,17	3	May	2	3,5	4	m	no
2,5-3,33	1	Jun	6	3,5	3	m	no
2,5-3,33	1	Jun	7	NA	3	m	no
2,5-3,33	1	Jun	4	4	3	ind	no
3,33-4,17	1	Jun	3	3,5	3,5	m	no
3,33-4,17	1	Jun	5	NA	3,5	ind	no
3,33-4,17	1	Jun	2	5	4	m	no
4,17-5	1	Jun	1	5	5	m	no
2,5-3,33	2	Jun	5	3	3	f	no
3,33-4,17	2	Jun	4	3	3,5	m	no
3,33-4,17	2	Jun	3	5	4	m	no
4,17-5	2	Jun	1	5	5	f	no
4,17-5	2	Jun	2	7	5	f	no
1,67-2,5	3	Jun	3	2	2,5	ind	no
2,5-3,33	3	Jun	4	3	3	m	no
3,33-4,17	3	Jun	2	4,5	3,5	f	no
4,17-5	3	Jun	1	5,5	4,5	m	no
2,5-3,33	1	Jul	6	6	3	m	no
3,33-4,17	1	Jul	5	5	4	f	no
3,33-4,17	1	Jul	2	3,5	4	m	no
3,33-4,17	1	Jul	3	5	4	m	no
3,33-4,17	1	Jul	4	5	4	m	no
4,17-5	1	Jul	1	6	5	m	no
1,67-2,5	2	Jul	18	2	2	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
1,67-2,5	2	Jul	19	1,5	2,5	m	no
1,67-2,5	2	Jul	20	1,5	2,5	m	no
1,67-2,5	2	Jul	11	1,5	2,5	m	no
1,67-2,5	2	Jul	16	1,5	2,5	m	no
2,5-3,33	2	Jul	3	3	3	f	no
2,5-3,33	2	Jul	6	3	3	f	no
2,5-3,33	2	Jul	7	3	3	f	no
2,5-3,33	2	Jul	12	3	3	f	no
2,5-3,33	2	Jul	13	2	3	f	no
2,5-3,33	2	Jul	17	2	3	f	no
2,5-3,33	2	Jul	8	2,5	3	m	no
2,5-3,33	2	Jul	10	2,5	3	m	no
2,5-3,33	2	Jul	15	2	3	m	no
3,33-4,17	2	Jul	2	4	3,5	m	no
3,33-4,17	2	Jul	4	3,5	3,5	m	no
3,33-4,17	2	Jul	5	3	3,5	m	no
3,33-4,17	2	Jul	9	2,5	3,5	m	no
3,33-4,17	2	Jul	14	2,5	3,5	m	no
3,33-4,17	2	Jul	1	6	4	m	no
2,5-3,33	3	Jul	9	3	3	f	no
3,33-4,17	3	Jul	7	5	3,5	m	no
3,33-4,17	3	Jul	8	3,5	3,5	m	no
3,33-4,17	3	Jul	10	4	3,5	m	no
3,33-4,17	3	Jul	5	4	4	m	no
3,33-4,17	3	Jul	6	NA	4	m	no
4,17-5	3	Jul	4	5	4,5	f	no
4,17-5	3	Jul	2	NA	5	f	yes
5-5,83	3	Jul	1	5,5	5,5	f	yes
5,83-6,67	3	Jul	3	6	6	f	yes
2,5-3,33	1	Aug	8	3,5	3	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	1	Aug	2	5,5	3,5	m	no
3,33-4,17	1	Aug	6	5	4	f	no
3,33-4,17	1	Aug	3	4,5	4	f	yes
3,33-4,17	1	Aug	4	5,5	4	m	no
4,17-5	1	Aug	5	4,5	4,5	m	no
4,17-5	1	Aug	1	6	5	m	no
4,17-5	1	Aug	7	5,5	5	m	no
2,5-3,33	2	Aug	4	3	3	f	no
2,5-3,33	2	Aug	3	3	3	m	no
4,17-5	2	Aug	2	6	5	m	no
6,67-7,5	2	Aug	1	NA	7	f	yes
0,83-1,67	3	Aug	54	1,5	1	ind	no
1,67-2,5	3	Aug	39	3	2	f	no
1,67-2,5	3	Aug	47	2,5	2	f	no
1,67-2,5	3	Aug	52	2	2	m	no
1,67-2,5	3	Aug	53	2	2	m	no
1,67-2,5	3	Aug	26	2,5	2	m	no
1,67-2,5	3	Aug	29	3	2	m	no
1,67-2,5	3	Aug	30	3,5	2	m	no
1,67-2,5	3	Aug	38	2,5	2	m	no
1,67-2,5	3	Aug	42	3	2	m	no
1,67-2,5	3	Aug	45	2	2	m	no
1,67-2,5	3	Aug	46	2	2	m	no
1,67-2,5	3	Aug	48	2	2	m	no
1,67-2,5	3	Aug	22	3	2,5	m	no
1,67-2,5	3	Aug	27	2,5	2,5	m	no
1,67-2,5	3	Aug	28	3	2,5	m	no
1,67-2,5	3	Aug	36	2,5	2,5	m	no
1,67-2,5	3	Aug	37	2,5	2,5	m	no
1,67-2,5	3	Aug	40	3	2,5	m	no

1,67-2,5	3	Aug	41	3	2,5	m	no
Size Class	**Site**	**Month**	**Specimen**	**Rostral Length (mm)**	**Carapace Length (mm)**	**Sex**	**Eggs Present**
1,67-2,5	3	Aug	49	2,5	2,5	m	no
1,67-2,5	3	Aug	50	2	2,5	m	no
1,67-2,5	3	Aug	51	2,5	2,5	m	no
2,5-3,33	3	Aug	13	2,5	3	f	no
2,5-3,33	3	Aug	31	3,5	3	f	no
2,5-3,33	3	Aug	7	3	3	m	no
2,5-3,33	3	Aug	20	3,5	3	m	no
2,5-3,33	3	Aug	23	3	3	m	no
2,5-3,33	3	Aug	24	2,5	3	m	no
2,5-3,33	3	Aug	25	3	3	m	no
2,5-3,33	3	Aug	32	3,5	3	m	no
2,5-3,33	3	Aug	35	4	3	m	no
2,5-3,33	3	Aug	43	3	3	m	no
2,5-3,33	3	Aug	44	3	3	m	no
3,33-4,17	3	Aug	12	3	3,5	f	no
3,33-4,17	3	Aug	19	3,5	3,5	m	no
3,33-4,17	3	Aug	33	3,5	3,5	m	no
3,33-4,17	3	Aug	34	3,5	3,5	m	no
3,33-4,17	3	Aug	5	2,5	4	f	no
3,33-4,17	3	Aug	11	3	4	f	no
3,33-4,17	3	Aug	2	6	4	m	no
3,33-4,17	3	Aug	3	4	4	m	no
3,33-4,17	3	Aug	4	3	4	m	no
3,33-4,17	3	Aug	8	4	4	m	no
3,33-4,17	3	Aug	10	3	4	m	no
3,33-4,17	3	Aug	15	4	4	m	no
3,33-4,17	3	Aug	16	4	4	m	no
3,33-4,17	3	Aug	17	4	4	m	no
3,33-4,17	3	Aug	18	3,5	4	m	no
3,33-4,17	3	Aug	21	4	4	m	no
4,17-5	3	Aug	9	3,5	4,5	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
4,17-5	3	Aug	14	2,5	4,5	m	no
4,17-5	3	Aug	6	3	5	f	no
5,83-6,67	3	Aug	1	6	6	f	yes
3,33-4,17	1	Sep	13	3,5	3,5	f	no
3,33-4,17	1	Sep	15	5,5	3,5	m	no
3,33-4,17	1	Sep	14	4	4	f	no
3,33-4,17	1	Sep	16	6	4	f	no
3,33-4,17	1	Sep	10	5	4	m	no
3,33-4,17	1	Sep	17	5	4	m	no
4,17-5	1	Sep	7	5,5	4,5	m	no
4,17-5	1	Sep	1	6	5	f	no
4,17-5	1	Sep	2	6	5	f	no
4,17-5	1	Sep	8	5	5	f	no
4,17-5	1	Sep	9	5	5	f	no
4,17-5	1	Sep	12	6	5	f	no
4,17-5	1	Sep	5	7	5	f	yes
4,17-5	1	Sep	6	5	5	f	yes
4,17-5	1	Sep	3	9	5	m	no
4,17-5	1	Sep	11	5	5	m	no
5-5,83	1	Sep	4	5	5,5	f	no
1,67-2,5	2	Sep	31	2	2	m	no
1,67-2,5	2	Sep	34	2,5	2	m	no
1,67-2,5	2	Sep	36	2	2	m	no
1,67-2,5	2	Sep	37	3	2	m	no
1,67-2,5	2	Sep	38	2,5	2	m	no
1,67-2,5	2	Sep	39	2	2	m	no
1,67-2,5	2	Sep	40	2	2	m	no
1,67-2,5	2	Sep	41	2	2	m	no
1,67-2,5	2	Sep	45	2	2,5	f	no
1,67-2,5	2	Sep	32	2,5	2,5	m	no
1,67-2,5	2	Sep	33	2,5	2,5	m	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
1,67-2,5	2	Sep	35	2,5	2,5	m	no
1,67-2,5	2	Sep	42	2,5	2,5	m	no
1,67-2,5	2	Sep	43	2,5	2,5	m	no
1,67-2,5	2	Sep	44	2	2,5	m	no
2,5-3,33	2	Sep	28	3	3	f	no
2,5-3,33	2	Sep	23	3,5	3	m	no
2,5-3,33	2	Sep	24	4	3	m	no
2,5-3,33	2	Sep	27	3,5	3	m	no
2,5-3,33	2	Sep	29	3	3	m	no
2,5-3,33	2	Sep	30	2,5	3	m	no
3,33-4,17	2	Sep	16	4,5	3,5	m	no
3,33-4,17	2	Sep	21	4	3,5	m	no
3,33-4,17	2	Sep	22	3,5	3,5	m	no
3,33-4,17	2	Sep	26	3,5	3,5	m	no
3,33-4,17	2	Sep	13	4	4	m	no
3,33-4,17	2	Sep	14	4	4	m	no
3,33-4,17	2	Sep	15	4	4	m	no
3,33-4,17	2	Sep	17	4,5	4	m	no
3,33-4,17	2	Sep	19	4,5	4	m	no
3,33-4,17	2	Sep	20	4	4	m	no
3,33-4,17	2	Sep	25	3,5	4	m	no
4,17-5	2	Sep	7	5	4,5	m	no
4,17-5	2	Sep	18	5	5	f	no
4,17-5	2	Sep	4	5,5	5	m	no
4,17-5	2	Sep	6	5	5	m	no
4,17-5	2	Sep	8	5	5	m	no
4,17-5	2	Sep	9	5,5	5	m	no
4,17-5	2	Sep	10	4,5	5	m	no
4,17-5	2	Sep	11	5	5	m	no
4,17-5	2	Sep	12	4,5	5	m	no
5-5,83	2	Sep	1	6	5,5	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
5,83-6,67	2	Sep	2	6,5	6	f	no
5,83-6,67	2	Sep	3	7	6	f	no
5,83-6,67	2	Sep	5	6	6	m	no
3,33-4,17	3	Sep	15	3,5	3,5	f	no
3,33-4,17	3	Sep	9	6	4	f	no
3,33-4,17	3	Sep	8	5	4	m	no
3,33-4,17	3	Sep	10	4	4	m	no
3,33-4,17	3	Sep	13	4	4	m	no
3,33-4,17	3	Sep	14	4	4	m	no
3,33-4,17	3	Sep	16	NA	4	m	no
3,33-4,17	3	Sep	17	4	4	m	no
3,33-4,17	3	Sep	19	NA	4	m	no
4,17-5	3	Sep	5	4	4,5	f	yes
4,17-5	3	Sep	11	4,5	4,5	m	no
4,17-5	3	Sep	20	NA	4,5	m	no
4,17-5	3	Sep	3	5	5	f	yes
4,17-5	3	Sep	4	5	5	f	yes
4,17-5	3	Sep	7	5	5	m	no
4,17-5	3	Sep	12	4	5	m	no
4,17-5	3	Sep	18	5	5	m	no
4,17-5	3	Sep	6	6	5	ind	no
5-5,83	3	Sep	2	5	5,5	f	yes
5,83-6,67	3	Sep	1	5	6	f	yes
1,67-2,5	1	Oct	44	NA	2,5	f	no
1,67-2,5	1	Oct	48	1,5	2,5	m	no
2,5-3,33	1	Oct	37	3,5	3	m	no
3,33-4,17	1	Oct	39	2,5	3,5	m	no
3,33-4,17	1	Oct	29	NA	4	f	no
3,33-4,17	1	Oct	34	4	4	f	no
3,33-4,17	1	Oct	35	3,5	4	f	no
3,33-4,17	1	Oct	36	4	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	1	Oct	38	3,5	4	f	no
3,33-4,17	1	Oct	16	NA	4	m	no
3,33-4,17	1	Oct	17	4	4	m	no
3,33-4,17	1	Oct	19	6	4	m	no
3,33-4,17	1	Oct	20	5	4	m	no
3,33-4,17	1	Oct	21	4	4	m	no
3,33-4,17	1	Oct	22	4	4	m	no
3,33-4,17	1	Oct	28	5	4	m	no
3,33-4,17	1	Oct	30	4	4	m	no
3,33-4,17	1	Oct	32	5	4	m	no
3,33-4,17	1	Oct	33	3,5	4	m	no
3,33-4,17	1	Oct	41	NA	4	m	no
3,33-4,17	1	Oct	43	3,5	4	m	no
3,33-4,17	1	Oct	47	4	4	m	no
3,33-4,17	1	Oct	40	3	4	ind	no
3,33-4,17	1	Oct	42	5	4	ind	no
4,17-5	1	Oct	25	5	4,5	f	no
4,17-5	1	Oct	23	6	4,5	m	no
4,17-5	1	Oct	26	4	4,5	m	no
4,17-5	1	Oct	27	NA	4,5	m	no
4,17-5	1	Oct	14	6	5	f	no
4,17-5	1	Oct	31	6,5	5	f	no
4,17-5	1	Oct	10	6,5	5	f	yes
4,17-5	1	Oct	11	5	5	f	yes
4,17-5	1	Oct	13	6	5	f	yes
4,17-5	1	Oct	15	7	5	m	no
4,17-5	1	Oct	18	6	5	m	no
4,17-5	1	Oct	45	4	5	m	no
4,17-5	1	Oct	46	4	5	m	no
4,17-5	1	Oct	24	6,5	5	ind	no
5-5,83	1	Oct	4	6	5,5	f	yes

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
5-5,83	1	Oct	5	6,5	5,5	f	yes
5-5,83	1	Oct	9	5	5,5	f	yes
5,83-6,67	1	Oct	1	6	6	f	yes
5,83-6,67	1	Oct	2	5	6	f	yes
5,83-6,67	1	Oct	3	6	6	f	yes
5,83-6,67	1	Oct	6	6	6	f	yes
5,83-6,67	1	Oct	7	6	6	f	yes
5,83-6,67	1	Oct	12	5	6	f	yes
6,67-7,5	1	Oct	8	8	8	f	yes
2,5-3,33	2	Oct	10	3	3	f	no
2,5-3,33	2	Oct	9	NA	3	m	no
2,5-3,33	2	Oct	11	3,5	3	m	no
3,33-4,17	2	Oct	5	4,5	4	f	no
3,33-4,17	2	Oct	7	3	4	f	no
3,33-4,17	2	Oct	8	NA	4	m	no
4,17-5	2	Oct	6	4	5	f	no
5-5,83	2	Oct	4	NA	5,5	f	yes
5,83-6,67	2	Oct	1	10	6	f	yes
5,83-6,67	2	Oct	2	8	6	f	yes
5,83-6,67	2	Oct	3	6	6	f	yes
0,83-1,67	3	Oct	33	NA	1,5	ind	no
1,67-2,5	3	Oct	31	2,5	2,5	f	no
2,5-3,33	3	Oct	27	3	3	f	no
2,5-3,33	3	Oct	30	3	3	f	no
2,5-3,33	3	Oct	29	3	3	m	no
3,33-4,17	3	Oct	16	5	3,5	m	no
3,33-4,17	3	Oct	21	4	3,5	m	no
3,33-4,17	3	Oct	22	3,5	3,5	m	no
3,33-4,17	3	Oct	18	3,5	3,5	ind	no
3,33-4,17	3	Oct	32	3,5	3,5	ind	no
3,33-4,17	3	Oct	17	3,5	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	3	Oct	19	4,5	4	m	no
3,33-4,17	3	Oct	23	4	4	m	no
3,33-4,17	3	Oct	24	3	4	m	no
3,33-4,17	3	Oct	25	4	4	m	no
3,33-4,17	3	Oct	26	5	4	m	no
3,33-4,17	3	Oct	28	4	4	m	no
4,17-5	3	Oct	15	NA	4,5	m	no
4,17-5	3	Oct	14	4	5	f	no
4,17-5	3	Oct	2	NA	5	f	yes
4,17-5	3	Oct	5	4,5	5	f	yes
4,17-5	3	Oct	12	7	5	m	no
4,17-5	3	Oct	13	6	5	m	no
4,17-5	3	Oct	20	NA	5	m	no
5-5,83	3	Oct	1	6	5,5	f	yes
5-5,83	3	Oct	3	7	5,5	f	yes
5-5,83	3	Oct	4	6	5,5	f	yes
5-5,83	3	Oct	6	5	5,5	f	yes
5-5,83	3	Oct	11	5	5,5	m	no
5,83-6,67	3	Oct	7	5	6	f	no
5,83-6,67	3	Oct	8	5	6	f	no
5,83-6,67	3	Oct	9	6	6	f	yes
3,33-4,17	1	Nov	1	4,5	4	m	no
5,83-6,67	3	Nov	1	6,5	4	f	no
1,67-2,5	1	Dec	33	2,5	2	m	no
1,67-2,5	1	Dec	29	3	2,5	f	no
2,5-3,33	1	Dec	23	3,5	3	m	no
2,5-3,33	1	Dec	32	2,5	3	m	no
3,33-4,17	1	Dec	18	3,5	3,5	f	no
3,33-4,17	1	Dec	25	3,5	3,5	f	no
3,33-4,17	1	Dec	11	7	3,5	m	no
3,33-4,17	1	Dec	7	4	4	f	no

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
3,33-4,17	1	Dec	10	4,5	4	f	no
3,33-4,17	1	Dec	13	5	4	f	no
3,33-4,17	1	Dec	14	3	4	f	no
3,33-4,17	1	Dec	15	3,5	4	f	no
3,33-4,17	1	Dec	19	3,5	4	f	no
3,33-4,17	1	Dec	9	4	4	m	no
4,17-5	1	Dec	5	5,5	4,5	m	no
4,17-5	1	Dec	6	4,5	4,5	m	no
4,17-5	1	Dec	1	7	5	f	yes
4,17-5	1	Dec	2	5	5	m	no
1,67-2,5	2	Dec	28	3,5	2	f	no
1,67-2,5	2	Dec	32	2,5	2,5	f	no
2,5-3,33	2	Dec	31	3	3	m	no
3,33-4,17	2	Dec	14		3,5	m	no
3,33-4,17	2	Dec	16	4,5	3,5	m	no
3,33-4,17	2	Dec	25	3	3,5	m	no
3,33-4,17	2	Dec	27	3,5	3,5	m	no
3,33-4,17	2	Dec	30	2,5	3,5	m	no
3,33-4,17	2	Dec	21	4	4	f	no
3,33-4,17	2	Dec	11	4,5	4	m	no
3,33-4,17	2	Dec	15	4	4	m	no
3,33-4,17	2	Dec	17	5	4	m	no
3,33-4,17	2	Dec	20	3,5	4	m	no
3,33-4,17	2	Dec	23	4	4	m	no
3,33-4,17	2	Dec	26	4,5	4	m	no
4,17-5	2	Dec	12	4	4,5	f	no
4,17-5	2	Dec	19	4	4,5	f	no
4,17-5	2	Dec	4	5	5	f	yes
4,17-5	2	Dec	6	5	5	m	no
5-5,83	2	Dec	1	5	5,5	f	yes
5-5,83	2	Dec	2	5	5,5	f	yes

Size Class	Site	Month	Specimen	Rostral Length (mm)	Carapace Length (mm)	Sex	Eggs Present
1,67-2,5	3	Dec	19	2	2,5	m	no
2,5-3,33	3	Dec	5	4	3	f	no
2,5-3,33	3	Dec	13	3,5	3	f	no
2,5-3,33	3	Dec	20		3	m	no
3,33-4,17	3	Dec	2	6	4	f	yes
3,33-4,17	3	Dec	7	4	4	f	no
3,33-4,17	3	Dec	8	4	4	f	no
3,33-4,17	3	Dec	10	4	4	f	no
3,33-4,17	3	Dec	16	4	4	f	no
3,33-4,17	3	Dec	6	4	4	m	no
3,33-4,17	3	Dec	9	4,5	4	m	no
4,17-5	3	Dec	11	5,5	4,5	m	no
5,83-6,67	3	Dec	1	7	6	f	yes

Table S 18: Showing raw data collected for aquatic invertebrate abundance and diversity indices at Lake Sibaya in 2021

Month	Site	Lake	Relative Community Abundance	Genera Richness	Shannon's Diversity Index	Pielou's Evenness
Mar	**1**	Sibaya	206	12	1,44	1,33
Mar	**2**	Sibaya	59	5	1,33	1,91
Mar	**3**	Sibaya	236	8	1,65	1,82
Apr	**1**	Sibaya	89	7	1,30	1,54
Apr	**2**	Sibaya	338	9	0,88	0,92
Apr	**3**	Sibaya	196	6	0,83	1,07
May	**1**	Sibaya	151	10	1,62	1,62
May	**2**	Sibaya	48	4	1,17	1,95
May	**3**	Sibaya	48	8	1,67	1,85
Jun	**1**	Sibaya	198	6	1,28	1,65
Jun	**2**	Sibaya	42	4	0,55	0,92
Jun	**3**	Sibaya	100	10	1,67	1,67
Jul	**1**	Sibaya	22	3	0,91	1,90
Jul	**2**	Sibaya	28	4	0,90	1,50
Jul	**3**	Sibaya	81	5	0,73	1,04
Aug	**1**	Sibaya	466	8	0,77	0,85
Aug	**2**	Sibaya	161	7	1,36	1,61
Aug	**3**	Sibaya	120	6	1,39	1,79
Sep	**1**	Sibaya	362	7	0,98	1,16
Sep	**2**	Sibaya	108	7	1,25	1,47
Sep	**3**	Sibaya	198	4	1,03	1,71
Oct	**1**	Sibaya	317	6	0,82	1,06
Oct	**2**	Sibaya	156	6	0,52	0,67
Oct	**3**	Sibaya	506	4	0,75	1,25
Nov	**1**	Sibaya	113	6	0,56	0,72
Nov	**2**	Sibaya	203	9	0,81	0,85
Nov	**3**	Sibaya	90	10	1,03	1,03
Dec	**1**	Sibaya	252	5	0,82	1,17
Dec	**2**	Sibaya	160	5	0,85	1,22
Dec	**3**	Sibaya	170	4	1,02	1,69

Table S 19: Showing raw data collected for aquatic invertebrate abundanceand diversity indices at Lake Mzingazi in 2021

Month	Site	Lake	Relative Community Abundance	Genera Richness	Shannon's Diversity Index	Pielou's Evenness
Mar	**1**	Mzingazi	119	7	1,47	1,74
Mar	**2**	Mzingazi	51	8	1,82	2,02
Mar	**3**	Mzingazi	109	5	1,35	1,93
Apr	**1**	Mzingazi	81	7	1,46	1,73
Apr	**2**	Mzingazi	329	8	1,20	1,33
Apr	**3**	Mzingazi	37	9	1,73	1,81
May	**1**	Mzingazi	11	5	1,52	2,17
May	**2**	Mzingazi	37	3	0,33	0,70
May	**3**	Mzingazi	19	5	1,31	1,87
Jun	**1**	Mzingazi	74	7	1,50	1,77
Jun	**2**	Mzingazi	47	4	0,88	1,47
Jun	**3**	Mzingazi	22	8	1,73	1,91
Jul	**1**	Mzingazi	132	7	0,95	1,12
Jul	**2**	Mzingazi	103	8	1,59	1,76
Jul	**3**	Mzingazi	72	7	1,27	1,50
Aug	**1**	Mzingazi	14	2	0,60	1,99
Aug	**2**	Mzingazi	4	3	1,04	2,18
Aug	**3**	Mzingazi	12	3	0,96	2,01
Sep	**1**	Mzingazi	101	5	1,04	1,48
Sep	**2**	Mzingazi	65	4	1,03	1,71
Sep	**3**	Mzingazi	96	5	1,19	1,71
Oct	**1**	Mzingazi	170	6	0,69	0,89
Oct	**2**	Mzingazi	43	4	1,29	2,15
Oct	**3**	Mzingazi	85	6	0,90	1,16
Nov	**1**	Mzingazi	73	4	0,66	1,10
Nov	**2**	Mzingazi	31	4	0,92	1,53
Nov	**3**	Mzingazi	68	8	1,02	1,13
Dec	**1**	Mzingazi	74	9	1,80	1,88
Dec	**2**	Mzingazi	72	8	1,79	1,98
Dec	**3**	Mzingazi	73	8	1,77	1,96

Table S 20: Showing raw abundance values of all collected aquatic invertebrates from Lake Sibaya and Lake Mzingazi in 2021

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
Acidocerinae Gen.	Mar	1	0	0
***Aliolimnatus* sp.**	Mar	1	0	0
***Appasus* sp.**	Mar	1	1	0
***Bellamya* sp.**	Mar	1	2	0
***Bulinus* sp.**	Mar	1	27	21
Caridina africana	Mar	1	89	54
Ceraptopogonidae Gen.	Mar	1	2	0
Chironimidae Gen.	Mar	1	9	5
***Cloeon* sp.**	Mar	1	5	11
***Corbicula* sp.**	Mar	1	0	0
***Crenigomphus* sp.**	Mar	1	0	0
***Cyathura* sp.**	Mar	1	0	0
***Haliplus* sp.**	Mar	1	0	0
***Laccocoris* sp.**	Mar	1	1	0
***Lestes* sp.**	Mar	1	0	0
***Mesovelia* sp.**	Mar	1	1	0
***Microvelia* sp.**	Mar	1	0	0
***Nepogomphoides* sp.**	Mar	1	0	0
Notonectidae Gen.	Mar	1	0	0
***Oncyogomphus* sp.**	Mar	1	0	0
***Paragomphus* sp.**	Mar	1	0	0
***Paramelita* sp.**	Mar	1	0	0
***Potomonautes* sp.**	Mar	1	0	0
***Povilla* sp.**	Mar	1	0	0
***Pseudagrion* sp.**	Mar	1	0	1
***Pseudospaerama* sp.**	Mar	1	0	3
***Ranatra* sp.**	Mar	1	1	0
***Sigara* sp.**	Mar	1	0	0
***Stenus* sp.**	Mar	1	0	0
Tarebia granifera	Mar	1	67	24
***Tetragnatha* sp.**	Mar	1	0	0
***Tetrathemis* sp.**	Mar	1	0	0
***Xiphoveloidea* sp.**	Mar	1	1	0
Acidocerinae Gen.	Mar	2	0	0
***Aliolimnatus* sp.**	Mar	2	0	0
***Appasus* sp.**	Mar	2	3	0
***Bellamya* sp.**	Mar	2	0	0
***Bulinus* sp.**	Mar	2	4	12
Caridina africana	Mar	2	0	0
Ceraptopogonidae Gen.	Mar	2	0	0
Chironimidae Gen.	Mar	2	25	8
***Cloeon* sp.**	Mar	2	19	9
***Corbicula* sp.**	Mar	2	0	0
***Crenigomphus* sp.**	Mar	2	0	0
***Cyathura* sp.**	Mar	2	0	2
***Haliplus* sp.**	Mar	2	0	0
***Laccocoris* sp.**	Mar	2	8	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Lestes* sp.**	Mar	2	0	0
***Mesovelia* sp.**	Mar	2	0	0
***Microvelia* sp.**	Mar	2	0	0
***Nepogomphoides* sp.**	Mar	2	0	1
Notonectidae Gen.	Mar	2	0	0
***Oncyogomphus* sp.**	Mar	2	0	3
***Paragomphus* sp.**	Mar	2	0	0
***Paramelita* sp.**	Mar	2	0	0
***Potomonautes* sp.**	Mar	2	0	0
***Povilla* sp.**	Mar	2	0	0
***Pseudagrion* sp.**	Mar	2	0	0
***Pseudospaerama* sp.**	Mar	2	0	3
***Ranatra* sp.**	Mar	2	0	0
***Sigara* sp.**	Mar	2	0	0
***Stenus* sp.**	Mar	2	0	0
Tarebia granifera	Mar	2	0	13
***Tetragnatha* sp.**	Mar	2	0	0
***Tetrathemis* sp.**	Mar	2	0	0
***Xiphoveloidea* sp.**	Mar	2	0	0
Acidocerinae Gen.	Mar	3	0	0
***Aliolimnatus* sp.**	Mar	3	0	0
***Appasus* sp.**	Mar	3	2	0
***Bellamya* sp.**	Mar	3	0	0
***Bulinus* sp.**	Mar	3	22	0
Caridina africana	Mar	3	84	1
Ceraptopogonidae Gen.	Mar	3	0	0
Chironimidae Gen.	Mar	3	33	37
***Cloeon* sp.**	Mar	3	21	31
***Corbicula* sp.**	Mar	3	0	0
***Crenigomphus* sp.**	Mar	3	0	0
***Cyathura* sp.**	Mar	3	0	0
***Haliplus* sp.**	Mar	3	0	0
***Laccocoris* sp.**	Mar	3	10	0
***Lestes* sp.**	Mar	3	0	0
***Mesovelia* sp.**	Mar	3	0	0
***Microvelia* sp.**	Mar	3	0	0
***Nepogomphoides* sp.**	Mar	3	0	0
Notonectidae Gen.	Mar	3	0	0
***Oncyogomphus* sp.**	Mar	3	0	0
***Paragomphus* sp.**	Mar	3	0	0
***Paramelita* sp.**	Mar	3	2	0
***Potomonautes* sp.**	Mar	3	0	0
***Povilla* sp.**	Mar	3	0	0
***Pseudagrion* sp.**	Mar	3	0	0
***Pseudospaerama* sp.**	Mar	3	0	11
***Ranatra* sp.**	Mar	3	0	0
***Sigara* sp.**	Mar	3	0	0
***Stenus* sp.**	Mar	3	0	0
Tarebia granifera	Mar	3	62	29

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Tetragnatha* sp.**	Mar	3	0	0
***Tetrathemis* sp.**	Mar	3	0	0
***Xiphoveloidea* sp.**	Mar	3	0	0
Acidocerinae Gen.	Apr	1	0	0
***Aliolimnatus* sp.**	Apr	1	0	0
***Appasus* sp.**	Apr	1	0	1
***Bellamya* sp.**	Apr	1	0	0
***Bulinus* sp.**	Apr	1	14	9
Caridina africana	Apr	1	16	20
Ceraptopogonidae Gen.	Apr	1	0	0
Chironimidae Gen.	Apr	1	1	0
***Cloeon* sp.**	Apr	1	8	37
***Corbicula* sp.**	Apr	1	0	0
***Crenigomphus* sp.**	Apr	1	0	0
***Cyathura* sp.**	Apr	1	1	0
***Haliplus* sp.**	Apr	1	0	0
***Laccocoris* sp.**	Apr	1	0	0
***Lestes* sp.**	Apr	1	0	0
***Mesovelia* sp.**	Apr	1	0	0
***Microvelia* sp.**	Apr	1	0	0
***Nepogomphoides* sp.**	Apr	1	0	0
Notonectidae Gen.	Apr	1	0	0
***Oncyogomphus* sp.**	Apr	1	0	0
***Paragomphus* sp.**	Apr	1	0	0
***Paramelita* sp.**	Apr	1	0	0
***Potomonautes* sp.**	Apr	1	0	0
***Povilla* sp.**	Apr	1	0	0
***Pseudagrion* sp.**	Apr	1	0	1
***Pseudospaerama* sp.**	Apr	1	0	7
***Ranatra* sp.**	Apr	1	0	0
***Sigara* sp.**	Apr	1	0	0
***Stenus* sp.**	Apr	1	0	0
Tarebia granifera	Apr	1	48	6
***Tetragnatha* sp.**	Apr	1	0	0
***Tetrathemis* sp.**	Apr	1	1	0
***Xiphoveloidea* sp.**	Apr	1	0	0
Acidocerinae Gen.	Apr	2	0	0
***Aliolimnatus* sp.**	Apr	2	0	0
***Appasus* sp.**	Apr	2	3	1
***Bellamya* sp.**	Apr	2	0	0
***Bulinus* sp.**	Apr	2	10	33
Caridina africana	Apr	2	62	90
Ceraptopogonidae Gen.	Apr	2	0	0
Chironimidae Gen.	Apr	2	0	0
***Cloeon* sp.**	Apr	2	1	6
***Corbicula* sp.**	Apr	2	8	0
***Crenigomphus* sp.**	Apr	2	0	0
***Cyathura* sp.**	Apr	2	0	0
***Haliplus* sp.**	Apr	2	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Laccocoris* sp.**	Apr	2	2	2
***Lestes* sp.**	Apr	2	1	0
***Mesovelia* sp.**	Apr	2	0	0
***Microvelia* sp.**	Apr	2	0	0
***Nepogomphoides* sp.**	Apr	2	3	0
Notonectidae Gen.	Apr	2	0	0
***Oncyogomphus* sp.**	Apr	2	0	0
***Paragomphus* sp.**	Apr	2	0	0
***Paramelita* sp.**	Apr	2	0	16
***Potomonautes* sp.**	Apr	2	0	0
***Povilla* sp.**	Apr	2	0	0
***Pseudagrion* sp.**	Apr	2	0	0
***Pseudospaerama* sp.**	Apr	2	0	1
***Ranatra* sp.**	Apr	2	0	0
***Sigara* sp.**	Apr	2	0	0
***Stenus* sp.**	Apr	2	0	0
Tarebia granifera	Apr	2	248	180
***Tetragnatha* sp.**	Apr	2	0	0
***Tetrathemis* sp.**	Apr	2	0	0
***Xiphoveloidea* sp.**	Apr	2	0	0
Acidocerinae Gen.	Apr	3	0	0
***Aliolimnatus* sp.**	Apr	3	0	0
***Appasus* sp.**	Apr	3	0	0
***Bellamya* sp.**	Apr	3	0	0
***Bulinus* sp.**	Apr	3	3	2
Caridina africana	Apr	3	69	4
Ceraptopogonidae Gen.	Apr	3	0	2
Chironimidae Gen.	Apr	3	1	0
***Cloeon* sp.**	Apr	3	0	1
***Corbicula* sp.**	Apr	3	0	0
***Crenigomphus* sp.**	Apr	3	0	0
***Cyathura* sp.**	Apr	3	0	0
***Haliplus* sp.**	Apr	3	0	0
***Laccocoris* sp.**	Apr	3	0	2
***Lestes* sp.**	Apr	3	0	0
***Mesovelia* sp.**	Apr	3	0	1
***Microvelia* sp.**	Apr	3	0	0
***Nepogomphoides* sp.**	Apr	3	0	0
Notonectidae Gen.	Apr	3	0	0
***Oncyogomphus* sp.**	Apr	3	2	0
***Paragomphus* sp.**	Apr	3	0	0
***Paramelita* sp.**	Apr	3	1	0
***Potomonautes* sp.**	Apr	3	0	0
***Povilla* sp.**	Apr	3	0	0
***Pseudagrion* sp.**	Apr	3	0	1
***Pseudospaerama* sp.**	Apr	3	0	14
***Ranatra* sp.**	Apr	3	0	0
***Sigara* sp.**	Apr	3	0	0
***Stenus* sp.**	Apr	3	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
Tarebia granifera	Apr	3	120	10
***Tetragnatha* sp.**	Apr	3	0	0
***Tetrathemis* sp.**	Apr	3	0	0
***Xiphoveloidea* sp.**	Apr	3	0	0
Acidocerinae Gen.	May	1	0	0
***Aliolimnatus* sp.**	May	1	0	0
***Appasus* sp.**	May	1	0	0
***Bellamya* sp.**	May	1	0	0
***Bulinus* sp.**	May	1	13	0
Caridina africana	May	1	56	4
Ceraptopogonidae Gen.	May	1	2	0
Chironimidae Gen.	May	1	0	0
***Cloeon* sp.**	May	1	18	2
***Corbicula* sp.**	May	1	0	0
***Crenigomphus* sp.**	May	1	0	0
***Cyathura* sp.**	May	1	0	0
***Haliplus* sp.**	May	1	0	2
***Laccocoris* sp.**	May	1	2	0
***Lestes* sp.**	May	1	0	0
***Mesovelia* sp.**	May	1	1	1
***Microvelia* sp.**	May	1	0	0
***Nepogomphoides* sp.**	May	1	0	0
Notonectidae Gen.	May	1	0	0
***Oncyogomphus* sp.**	May	1	0	0
***Paragomphus* sp.**	May	1	0	0
***Paramelita* sp.**	May	1	0	0
***Potomonautes* sp.**	May	1	0	0
***Povilla* sp.**	May	1	0	0
***Pseudagrion* sp.**	May	1	0	0
***Pseudospaerama* sp.**	May	1	14	0
***Ranatra* sp.**	May	1	0	0
***Sigara* sp.**	May	1	0	0
***Stenus* sp.**	May	1	1	0
Tarebia granifera	May	1	43	0
***Tetragnatha* sp.**	May	1	0	2
***Tetrathemis* sp.**	May	1	0	0
***Xiphoveloidea* sp.**	May	1	1	0
Acidocerinae Gen.	May	2	0	0
***Aliolimnatus* sp.**	May	2	0	0
***Appasus* sp.**	May	2	0	0
***Bellamya* sp.**	May	2	0	0
***Bulinus* sp.**	May	2	5	2
Caridina africana	May	2	19	34
Ceraptopogonidae Gen.	May	2	0	0
Chironimidae Gen.	May	2	0	0
***Cloeon* sp.**	May	2	4	1
***Corbicula* sp.**	May	2	0	0
***Crenigomphus* sp.**	May	2	0	0
***Cyathura* sp.**	May	2	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Haliplus* sp.**	May	2	0	0
***Laccocoris* sp.**	May	2	0	0
***Lestes* sp.**	May	2	0	0
***Mesovelia* sp.**	May	2	0	0
***Microvelia* sp.**	May	2	0	0
***Nepogomphoides* sp.**	May	2	0	0
Notonectidae Gen.	May	2	0	0
***Oncyogomphus* sp.**	May	2	0	0
***Paragomphus* sp.**	May	2	0	0
***Paramelita* sp.**	May	2	0	0
***Potomonautes* sp.**	May	2	0	0
***Povilla* sp.**	May	2	0	0
***Pseudagrion* sp.**	May	2	0	0
***Pseudospaerama* sp.**	May	2	0	0
***Ranatra* sp.**	May	2	0	0
***Sigara* sp.**	May	2	0	0
***Stenus* sp.**	May	2	0	0
Tarebia granifera	May	2	20	0
***Tetragnatha* sp.**	May	2	0	0
***Tetrathemis* sp.**	May	2	0	0
***Xiphoveloidea* sp.**	May	2	0	0
Acidocerinae Gen.	May	3	0	0
***Aliolimnatus* sp.**	May	3	0	0
***Appasus* sp.**	May	3	0	1
***Bellamya* sp.**	May	3	0	0
***Bulinus* sp.**	May	3	0	0
Caridina africana	May	3	10	5
Ceraptopogonidae Gen.	May	3	0	0
Chironimidae Gen.	May	3	3	0
***Cloeon* sp.**	May	3	17	1
***Corbicula* sp.**	May	3	0	0
***Crenigomphus* sp.**	May	3	1	0
***Cyathura* sp.**	May	3	0	0
***Haliplus* sp.**	May	3	0	0
***Laccocoris* sp.**	May	3	3	0
***Lestes* sp.**	May	3	0	0
***Mesovelia* sp.**	May	3	0	0
***Microvelia* sp.**	May	3	0	0
***Nepogomphoides* sp.**	May	3	0	0
Notonectidae Gen.	May	3	0	0
***Oncyogomphus* sp.**	May	3	0	0
***Paragomphus* sp.**	May	3	0	0
***Paramelita* sp.**	May	3	0	0
***Potomonautes* sp.**	May	3	0	0
***Povilla* sp.**	May	3	0	0
***Pseudagrion* sp.**	May	3	1	3
***Pseudospaerama* sp.**	May	3	0	0
***Ranatra* sp.**	May	3	0	0
***Sigara* sp.**	May	3	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
Stenus sp.	May	3	0	0
Tarebia granifera	May	3	11	9
Tetragnatha sp.	May	3	0	0
Tetrathemis sp.	May	3	0	0
Xiphoveloidea sp.	May	3	2	0
Acidocerinae Gen.	Jun	1	0	0
Aliolimnatus sp.	Jun	1	0	0
Appasus sp.	Jun	1	0	0
Bellamya sp.	Jun	1	0	0
Bulinus sp.	Jun	1	13	22
Caridina africana	Jun	1	48	10
Ceraptopogonidae Gen.	Jun	1	0	0
Chironimidae Gen.	Jun	1	0	7
Cloeon sp.	Jun	1	13	29
Corbicula sp.	Jun	1	0	0
Crenigomphus sp.	Jun	1	0	1
Cyathura sp.	Jun	1	0	0
Haliplus sp.	Jun	1	0	0
Laccocoris sp.	Jun	1	0	0
Lestes sp.	Jun	1	0	0
Mesovelia sp.	Jun	1	1	0
Microvelia sp.	Jun	1	0	0
Nepogomphoides sp.	Jun	1	0	0
Notonectidae Gen.	Jun	1	0	0
Oncyogomphus sp.	Jun	1	0	0
Paragomphus sp.	Jun	1	0	1
Paramelita sp.	Jun	1	0	0
Potomonautes sp.	Jun	1	0	0
Povilla sp.	Jun	1	0	0
Pseudagrion sp.	Jun	1	0	0
Pseudospaerama sp.	Jun	1	0	0
Ranatra sp.	Jun	1	0	0
Sigara sp.	Jun	1	0	0
Stenus sp.	Jun	1	0	0
Tarebia granifera	Jun	1	105	4
Tetragnatha sp.	Jun	1	0	0
Tetrathemis sp.	Jun	1	0	0
Xiphoveloidea sp.	Jun	1	18	0
Acidocerinae Gen.	Jun	2	0	0
Aliolimnatus sp.	Jun	2	0	0
Appasus sp.	Jun	2	0	1
Bellamya sp.	Jun	2	0	0
Bulinus sp.	Jun	2	0	4
Caridina africana	Jun	2	36	10
Ceraptopogonidae Gen.	Jun	2	0	0
Chironimidae Gen.	Jun	2	0	0
Cloeon sp.	Jun	2	3	32
Corbicula sp.	Jun	2	0	0
Crenigomphus sp.	Jun	2	1	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Cyathura* sp.**	Jun	2	0	0
***Haliplus* sp.**	Jun	2	0	0
***Laccocoris* sp.**	Jun	2	0	0
***Lestes* sp.**	Jun	2	0	0
***Mesovelia* sp.**	Jun	2	0	0
***Microvelia* sp.**	Jun	2	0	0
***Nepogomphoides* sp.**	Jun	2	0	0
Notonectidae Gen.	Jun	2	0	0
***Oncyogomphus* sp.**	Jun	2	0	0
***Paragomphus* sp.**	Jun	2	0	0
***Paramelita* sp.**	Jun	2	0	0
***Potomonautes* sp.**	Jun	2	0	0
***Povilla* sp.**	Jun	2	0	0
***Pseudagrion* sp.**	Jun	2	0	0
***Pseudospaerama* sp.**	Jun	2	0	0
***Ranatra* sp.**	Jun	2	0	0
***Sigara* sp.**	Jun	2	0	0
***Stenus* sp.**	Jun	2	0	0
Tarebia granifera	Jun	2	2	0
***Tetragnatha* sp.**	Jun	2	0	0
***Tetrathemis* sp.**	Jun	2	0	0
***Xiphoveloidea* sp.**	Jun	2	0	0
Acidocerinae Gen.	Jun	3	0	0
***Aliolimnatus* sp.**	Jun	3	0	0
***Appasus* sp.**	Jun	3	0	2
***Bellamya* sp.**	Jun	3	0	0
***Bulinus* sp.**	Jun	3	5	1
Caridina africana	Jun	3	43	9
Ceraptopogonidae Gen.	Jun	3	3	0
Chironimidae Gen.	Jun	3	0	3
***Cloeon* sp.**	Jun	3	13	0
***Corbicula* sp.**	Jun	3	0	0
***Crenigomphus* sp.**	Jun	3	0	0
***Cyathura* sp.**	Jun	3	0	0
***Haliplus* sp.**	Jun	3	0	0
***Laccocoris* sp.**	Jun	3	0	0
***Lestes* sp.**	Jun	3	0	0
***Mesovelia* sp.**	Jun	3	1	1
***Microvelia* sp.**	Jun	3	0	0
***Nepogomphoides* sp.**	Jun	3	0	0
Notonectidae Gen.	Jun	3	0	1
***Oncyogomphus* sp.**	Jun	3	0	0
***Paragomphus* sp.**	Jun	3	0	0
***Paramelita* sp.**	Jun	3	0	0
***Potomonautes* sp.**	Jun	3	0	0
***Povilla* sp.**	Jun	3	0	0
***Pseudagrion* sp.**	Jun	3	1	1
***Pseudospaerama* sp.**	Jun	3	0	0
***Ranatra* sp.**	Jun	3	1	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Sigara* sp.**	Jun	3	0	0
***Stenus* sp.**	Jun	3	2	0
Tarebia granifera	Jun	3	19	4
***Tetragnatha* sp.**	Jun	3	0	0
***Tetrathemis* sp.**	Jun	3	0	0
***Xiphoveloidea* sp.**	Jun	3	12	0
Acidocerinae Gen.	Jul	1	0	0
***Aliolimnatus* sp.**	Jul	1	0	0
***Appasus* sp.**	Jul	1	0	0
***Bellamya* sp.**	Jul	1	0	0
***Bulinus* sp.**	Jul	1	4	1
Caridina africana	Jul	1	4	3
Ceraptopogonidae Gen.	Jul	1	0	0
Chironimidae Gen.	Jul	1	0	1
***Cloeon* sp.**	Jul	1	0	93
***Corbicula* sp.**	Jul	1	0	0
***Crenigomphus* sp.**	Jul	1	0	0
***Cyathura* sp.**	Jul	1	0	11
***Haliplus* sp.**	Jul	1	0	0
***Laccocoris* sp.**	Jul	1	0	0
***Lestes* sp.**	Jul	1	0	0
***Mesovelia* sp.**	Jul	1	0	0
***Microvelia* sp.**	Jul	1	0	0
***Nepogomphoides* sp.**	Jul	1	0	0
Notonectidae Gen.	Jul	1	0	0
***Oncyogomphus* sp.**	Jul	1	0	0
***Paragomphus* sp.**	Jul	1	0	0
***Paramelita* sp.**	Jul	1	14	0
***Potomonautes* sp.**	Jul	1	0	0
***Povilla* sp.**	Jul	1	0	0
***Pseudagrion* sp.**	Jul	1	0	0
***Pseudospaerama* sp.**	Jul	1	0	22
***Ranatra* sp.**	Jul	1	0	0
***Sigara* sp.**	Jul	1	0	0
***Stenus* sp.**	Jul	1	0	0
Tarebia granifera	Jul	1	0	1
***Tetragnatha* sp.**	Jul	1	0	0
***Tetrathemis* sp.**	Jul	1	0	0
***Xiphoveloidea* sp.**	Jul	1	0	0
Acidocerinae Gen.	Jul	2	0	0
***Aliolimnatus* sp.**	Jul	2	0	0
***Appasus* sp.**	Jul	2	0	2
***Bellamya* sp.**	Jul	2	0	0
***Bulinus* sp.**	Jul	2	0	0
Caridina africana	Jul	2	19	26
Ceraptopogonidae Gen.	Jul	2	0	0
Chironimidae Gen.	Jul	2	0	0
***Cloeon* sp.**	Jul	2	6	1
***Corbicula* sp.**	Jul	2	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Crenigomphus* sp.**	Jul	2	0	0
***Cyathura* sp.**	Jul	2	0	8
***Haliplus* sp.**	Jul	2	0	0
***Laccocoris* sp.**	Jul	2	0	0
***Lestes* sp.**	Jul	2	0	0
***Mesovelia* sp.**	Jul	2	0	0
***Microvelia* sp.**	Jul	2	0	0
***Nepogomphoides* sp.**	Jul	2	0	0
Notonectidae Gen.	Jul	2	0	0
***Oncyogomphus* sp.**	Jul	2	0	0
***Paragomphus* sp.**	Jul	2	0	0
***Paramelita* sp.**	Jul	2	2	13
***Potomonautes* sp.**	Jul	2	0	4
***Povilla* sp.**	Jul	2	0	0
***Pseudagrion* sp.**	Jul	2	1	0
***Pseudospaerama* sp.**	Jul	2	0	6
***Ranatra* sp.**	Jul	2	0	0
***Sigara* sp.**	Jul	2	0	0
***Stenus* sp.**	Jul	2	0	0
Tarebia granifera	Jul	2	0	43
***Tetragnatha* sp.**	Jul	2	0	0
***Tetrathemis* sp.**	Jul	2	0	0
***Xiphoveloidea* sp.**	Jul	2	0	0
Acidocerinae Gen.	Jul	3	0	0
***Aliolimnatus* sp.**	Jul	3	0	0
***Appasus* sp.**	Jul	3	0	1
***Bellamya* sp.**	Jul	3	0	0
***Bulinus* sp.**	Jul	3	0	0
Caridina africana	Jul	3	65	26
Ceraptopogonidae Gen.	Jul	3	0	0
Chironimidae Gen.	Jul	3	0	1
***Cloeon* sp.**	Jul	3	7	3
***Corbicula* sp.**	Jul	3	1	0
***Crenigomphus* sp.**	Jul	3	0	0
***Cyathura* sp.**	Jul	3	0	0
***Haliplus* sp.**	Jul	3	0	0
***Laccocoris* sp.**	Jul	3	0	0
***Lestes* sp.**	Jul	3	0	0
***Mesovelia* sp.**	Jul	3	0	0
***Microvelia* sp.**	Jul	3	0	0
***Nepogomphoides* sp.**	Jul	3	0	0
Notonectidae Gen.	Jul	3	0	0
***Oncyogomphus* sp.**	Jul	3	0	0
***Paragomphus* sp.**	Jul	3	0	0
***Paramelita* sp.**	Jul	3	6	0
***Potomonautes* sp.**	Jul	3	0	4
***Povilla* sp.**	Jul	3	0	0
***Pseudagrion* sp.**	Jul	3	0	0
***Pseudospaerama* sp.**	Jul	3	0	3

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Ranatra* sp.**	Jul	3	0	0
***Sigara* sp.**	Jul	3	0	0
***Stenus* sp.**	Jul	3	0	0
Tarebia granifera	Jul	3	2	34
***Tetragnatha* sp.**	Jul	3	0	0
***Tetrathemis* sp.**	Jul	3	0	0
***Xiphoveloidea* sp.**	Jul	3	0	0
Acidocerinae Gen.	Aug	1	0	0
***Aliolimnatus* sp.**	Aug	1	0	0
***Appasus* sp.**	Aug	1	0	0
***Bellamya* sp.**	Aug	1	1	0
***Bulinus* sp.**	Aug	1	13	0
Caridina africana	Aug	1	19	4
Ceraptopogonidae Gen.	Aug	1	0	0
Chironimidae Gen.	Aug	1	17	0
***Cloeon* sp.**	Aug	1	31	0
***Corbicula* sp.**	Aug	1	0	0
***Crenigomphus* sp.**	Aug	1	0	0
***Cyathura* sp.**	Aug	1	0	0
***Haliplus* sp.**	Aug	1	0	0
***Laccocoris* sp.**	Aug	1	0	0
***Lestes* sp.**	Aug	1	0	0
***Mesovelia* sp.**	Aug	1	0	0
***Microvelia* sp.**	Aug	1	0	0
***Nepogomphoides* sp.**	Aug	1	0	0
Notonectidae Gen.	Aug	1	0	0
***Oncyogomphus* sp.**	Aug	1	0	0
***Paragomphus* sp.**	Aug	1	0	0
***Paramelita* sp.**	Aug	1	3	0
***Potomonautes* sp.**	Aug	1	0	0
***Povilla* sp.**	Aug	1	0	0
***Pseudagrion* sp.**	Aug	1	2	0
***Pseudospaerama* sp.**	Aug	1	0	0
***Ranatra* sp.**	Aug	1	0	0
***Sigara* sp.**	Aug	1	0	0
***Stenus* sp.**	Aug	1	0	0
Tarebia granifera	Aug	1	380	10
***Tetragnatha* sp.**	Aug	1	0	0
***Tetrathemis* sp.**	Aug	1	0	0
***Xiphoveloidea* sp.**	Aug	1	0	0
Acidocerinae Gen.	Aug	2	0	0
***Aliolimnatus* sp.**	Aug	2	0	0
***Appasus* sp.**	Aug	2	0	0
***Bellamya* sp.**	Aug	2	0	0
***Bulinus* sp.**	Aug	2	4	0
Caridina africana	Aug	2	54	2
Ceraptopogonidae Gen.	Aug	2	0	0
Chironimidae Gen.	Aug	2	2	0
***Cloeon* sp.**	Aug	2	54	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Corbicula* sp.**	Aug	2	0	1
***Crenigomphus* sp.**	Aug	2	0	0
***Cyathura* sp.**	Aug	2	0	0
***Haliplus* sp.**	Aug	2	0	0
***Laccocoris* sp.**	Aug	2	3	0
***Lestes* sp.**	Aug	2	0	0
***Mesovelia* sp.**	Aug	2	0	0
***Microvelia* sp.**	Aug	2	0	0
***Nepogomphoides* sp.**	Aug	2	0	0
Notonectidae Gen.	Aug	2	0	0
***Oncyogomphus* sp.**	Aug	2	0	0
***Paragomphus* sp.**	Aug	2	0	0
***Paramelita* sp.**	Aug	2	0	0
***Potomonautes* sp.**	Aug	2	0	0
***Povilla* sp.**	Aug	2	0	0
***Pseudagrion* sp.**	Aug	2	0	0
***Pseudospaerama* sp.**	Aug	2	2	0
***Ranatra* sp.**	Aug	2	0	0
***Sigara* sp.**	Aug	2	0	0
***Stenus* sp.**	Aug	2	0	0
Tarebia granifera	Aug	2	42	1
***Tetragnatha* sp.**	Aug	2	0	0
***Tetrathemis* sp.**	Aug	2	0	0
***Xiphoveloidea* sp.**	Aug	2	0	0
Acidocerinae Gen.	Aug	3	0	0
***Aliolimnatus* sp.**	Aug	3	0	0
***Appasus* sp.**	Aug	3	0	0
***Bellamya* sp.**	Aug	3	0	0
***Bulinus* sp.**	Aug	3	9	0
Caridina africana	Aug	3	50	3
Ceraptopogonidae Gen.	Aug	3	0	0
Chironimidae Gen.	Aug	3	8	0
***Cloeon* sp.**	Aug	3	14	0
***Corbicula* sp.**	Aug	3	0	2
***Crenigomphus* sp.**	Aug	3	0	0
***Cyathura* sp.**	Aug	3	0	0
***Haliplus* sp.**	Aug	3	0	0
***Laccocoris* sp.**	Aug	3	0	0
***Lestes* sp.**	Aug	3	0	0
***Mesovelia* sp.**	Aug	3	0	0
***Microvelia* sp.**	Aug	3	0	0
***Nepogomphoides* sp.**	Aug	3	0	0
Notonectidae Gen.	Aug	3	0	0
***Oncyogomphus* sp.**	Aug	3	0	0
***Paragomphus* sp.**	Aug	3	0	0
***Paramelita* sp.**	Aug	3	0	0
***Potomonautes* sp.**	Aug	3	0	0
***Povilla* sp.**	Aug	3	0	0
***Pseudagrion* sp.**	Aug	3	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Pseudospaerama* sp.**	Aug	3	0	0
***Ranatra* sp.**	Aug	3	0	0
***Sigara* sp.**	Aug	3	0	0
***Stenus* sp.**	Aug	3	0	0
Tarebia granifera	Aug	3	38	7
***Tetragnatha* sp.**	Aug	3	0	0
***Tetrathemis* sp.**	Aug	3	1	0
***Xiphoveloidea* sp.**	Aug	3	0	0
Acidocerinae Gen.	Sep	1	0	0
***Aliolimnatus* sp.**	Sep	1	0	0
***Appasus* sp.**	Sep	1	0	0
***Bellamya* sp.**	Sep	1	0	0
***Bulinus* sp.**	Sep	1	11	4
Caridina africana	Sep	1	37	9
Ceraptopogonidae Gen.	Sep	1	0	0
Chironimidae Gen.	Sep	1	22	0
***Cloeon* sp.**	Sep	1	24	0
***Corbicula* sp.**	Sep	1	0	0
***Crenigomphus* sp.**	Sep	1	0	0
***Cyathura* sp.**	Sep	1	0	6
***Haliplus* sp.**	Sep	1	0	0
***Laccocoris* sp.**	Sep	1	2	0
***Lestes* sp.**	Sep	1	0	0
***Mesovelia* sp.**	Sep	1	0	0
***Microvelia* sp.**	Sep	1	0	0
***Nepogomphoides* sp.**	Sep	1	0	0
Notonectidae Gen.	Sep	1	0	0
***Oncyogomphus* sp.**	Sep	1	0	0
***Paragomphus* sp.**	Sep	1	0	0
***Paramelita* sp.**	Sep	1	2	0
***Potomonautes* sp.**	Sep	1	0	0
***Povilla* sp.**	Sep	1	0	0
***Pseudagrion* sp.**	Sep	1	0	0
***Pseudospaerama* sp.**	Sep	1	0	13
***Ranatra* sp.**	Sep	1	0	0
***Sigara* sp.**	Sep	1	0	0
***Stenus* sp.**	Sep	1	0	0
Tarebia granifera	Sep	1	264	69
***Tetragnatha* sp.**	Sep	1	0	0
***Tetrathemis* sp.**	Sep	1	0	0
***Xiphoveloidea* sp.**	Sep	1	0	0
Acidocerinae Gen.	Sep	2	0	0
***Aliolimnatus* sp.**	Sep	2	0	0
***Appasus* sp.**	Sep	2	0	1
***Bellamya* sp.**	Sep	2	0	0
***Bulinus* sp.**	Sep	2	4	0
Caridina africana	Sep	2	33	16
Ceraptopogonidae Gen.	Sep	2	0	0
Chironimidae Gen.	Sep	2	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Cloeon* sp.**	Sep	2	7	0
***Corbicula* sp.**	Sep	2	0	0
***Crenigomphus* sp.**	Sep	2	0	0
***Cyathura* sp.**	Sep	2	0	0
***Haliplus* sp.**	Sep	2	0	0
***Laccocoris* sp.**	Sep	2	3	0
***Lestes* sp.**	Sep	2	0	0
***Mesovelia* sp.**	Sep	2	0	0
***Microvelia* sp.**	Sep	2	0	0
***Nepogomphoides* sp.**	Sep	2	0	0
Notonectidae Gen.	Sep	2	0	0
***Oncyogomphus* sp.**	Sep	2	0	0
***Paragomphus* sp.**	Sep	2	0	0
***Paramelita* sp.**	Sep	2	2	11
***Potomonautes* sp.**	Sep	2	0	0
***Povilla* sp.**	Sep	2	0	0
***Pseudagrion* sp.**	Sep	2	0	0
***Pseudospaerama* sp.**	Sep	2	0	0
***Ranatra* sp.**	Sep	2	0	0
***Sigara* sp.**	Sep	2	0	0
***Stenus* sp.**	Sep	2	0	0
Tarebia granifera	Sep	2	57	37
***Tetragnatha* sp.**	Sep	2	0	0
***Tetrathemis* sp.**	Sep	2	0	0
***Xiphoveloidea* sp.**	Sep	2	2	0
Acidocerinae Gen.	Sep	3	0	0
***Aliolimnatus* sp.**	Sep	3	0	0
***Appasus* sp.**	Sep	3	0	1
***Bellamya* sp.**	Sep	3	0	0
***Bulinus* sp.**	Sep	3	0	0
Caridina africana	Sep	3	78	17
Ceraptopogonidae Gen.	Sep	3	0	0
Chironimidae Gen.	Sep	3	0	0
***Cloeon* sp.**	Sep	3	25	34
***Corbicula* sp.**	Sep	3	0	0
***Crenigomphus* sp.**	Sep	3	0	0
***Cyathura* sp.**	Sep	3	0	0
***Haliplus* sp.**	Sep	3	0	0
***Laccocoris* sp.**	Sep	3	2	0
***Lestes* sp.**	Sep	3	0	0
***Mesovelia* sp.**	Sep	3	0	0
***Microvelia* sp.**	Sep	3	0	0
***Nepogomphoides* sp.**	Sep	3	0	0
Notonectidae Gen.	Sep	3	0	0
***Oncyogomphus* sp.**	Sep	3	0	0
***Paragomphus* sp.**	Sep	3	0	0
***Paramelita* sp.**	Sep	3	0	3
***Potomonautes* sp.**	Sep	3	0	0
***Povilla* sp.**	Sep	3	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Pseudagrion* sp.**	Sep	3	0	0
***Pseudospaerama* sp.**	Sep	3	0	0
***Ranatra* sp.**	Sep	3	0	0
***Sigara* sp.**	Sep	3	0	0
***Stenus* sp.**	Sep	3	0	0
Tarebia granifera	Sep	3	93	41
***Tetragnatha* sp.**	Sep	3	0	0
***Tetrathemis* sp.**	Sep	3	0	0
***Xiphoveloidea* sp.**	Sep	3	0	0
Acidocerinae Gen.	Oct	1	0	0
***Aliolimnatus* sp.**	Oct	1	0	1
***Appasus* sp.**	Oct	1	0	0
***Bellamya* sp.**	Oct	1	0	0
***Bulinus* sp.**	Oct	1	15	9
Caridina africana	Oct	1	31	18
Ceraptopogonidae Gen.	Oct	1	0	0
Chironimidae Gen.	Oct	1	0	0
***Cloeon* sp.**	Oct	1	27	0
***Corbicula* sp.**	Oct	1	0	0
***Crenigomphus* sp.**	Oct	1	0	0
***Cyathura* sp.**	Oct	1	0	0
***Haliplus* sp.**	Oct	1	0	0
***Laccocoris* sp.**	Oct	1	1	0
***Lestes* sp.**	Oct	1	0	0
***Mesovelia* sp.**	Oct	1	0	0
***Microvelia* sp.**	Oct	1	0	0
***Nepogomphoides* sp.**	Oct	1	0	0
Notonectidae Gen.	Oct	1	0	0
***Oncyogomphus* sp.**	Oct	1	0	0
***Paragomphus* sp.**	Oct	1	0	0
***Paramelita* sp.**	Oct	1	0	0
***Potomonautes* sp.**	Oct	1	1	3
***Povilla* sp.**	Oct	1	0	0
***Pseudagrion* sp.**	Oct	1	0	1
***Pseudospaerama* sp.**	Oct	1	0	0
***Ranatra* sp.**	Oct	1	0	0
***Sigara* sp.**	Oct	1	0	0
***Stenus* sp.**	Oct	1	0	0
Tarebia granifera	Oct	1	242	138
***Tetragnatha* sp.**	Oct	1	0	0
***Tetrathemis* sp.**	Oct	1	0	0
***Xiphoveloidea* sp.**	Oct	1	0	0
Acidocerinae Gen.	Oct	2	0	0
***Aliolimnatus* sp.**	Oct	2	0	0
***Appasus* sp.**	Oct	2	0	0
***Bellamya* sp.**	Oct	2	0	0
***Bulinus* sp.**	Oct	2	2	0
Caridina africana	Oct	2	9	8
Ceraptopogonidae Gen.	Oct	2	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
Chironimidae Gen.	Oct	2	0	0
***Cloeon* sp.**	Oct	2	6	5
***Corbicula* sp.**	Oct	2	0	0
***Crenigomphus* sp.**	Oct	2	0	0
***Cyathura* sp.**	Oct	2	0	0
***Haliplus* sp.**	Oct	2	0	0
***Laccocoris* sp.**	Oct	2	1	0
***Lestes* sp.**	Oct	2	0	0
***Mesovelia* sp.**	Oct	2	0	0
***Microvelia* sp.**	Oct	2	0	0
***Nepogomphoides* sp.**	Oct	2	0	0
Notonectidae Gen.	Oct	2	0	0
***Oncyogomphus* sp.**	Oct	2	0	0
***Paragomphus* sp.**	Oct	2	0	0
***Paramelita* sp.**	Oct	2	1	13
***Potomonautes* sp.**	Oct	2	0	0
***Povilla* sp.**	Oct	2	0	0
***Pseudagrion* sp.**	Oct	2	0	0
***Pseudospaerama* sp.**	Oct	2	0	0
***Ranatra* sp.**	Oct	2	0	0
***Sigara* sp.**	Oct	2	0	0
***Stenus* sp.**	Oct	2	0	0
Tarebia granifera	Oct	2	137	17
***Tetragnatha* sp.**	Oct	2	0	0
***Tetrathemis* sp.**	Oct	2	0	0
***Xiphoveloidea* sp.**	Oct	2	0	0
Acidocerinae Gen.	Oct	3	0	0
***Aliolimnatus* sp.**	Oct	3	0	0
***Appasus* sp.**	Oct	3	0	2
***Bellamya* sp.**	Oct	3	0	0
***Bulinus* sp.**	Oct	3	14	0
Caridina africana	Oct	3	161	1
Ceraptopogonidae Gen.	Oct	3	0	0
Chironimidae Gen.	Oct	3	0	0
***Cloeon* sp.**	Oct	3	0	62
***Corbicula* sp.**	Oct	3	0	0
***Crenigomphus* sp.**	Oct	3	0	0
***Cyathura* sp.**	Oct	3	0	0
***Haliplus* sp.**	Oct	3	0	0
***Laccocoris* sp.**	Oct	3	1	0
***Lestes* sp.**	Oct	3	0	0
***Mesovelia* sp.**	Oct	3	0	0
***Microvelia* sp.**	Oct	3	0	0
***Nepogomphoides* sp.**	Oct	3	0	0
Notonectidae Gen.	Oct	3	0	0
***Oncyogomphus* sp.**	Oct	3	0	0
***Paragomphus* sp.**	Oct	3	0	0
***Paramelita* sp.**	Oct	3	0	7
***Potomonautes* sp.**	Oct	3	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Povilla* sp.**	Oct	3	0	0
***Pseudagrion* sp.**	Oct	3	0	0
***Pseudospaerama* sp.**	Oct	3	0	0
***Ranatra* sp.**	Oct	3	0	0
***Sigara* sp.**	Oct	3	0	1
***Stenus* sp.**	Oct	3	0	0
Tarebia granifera	Oct	3	330	12
***Tetragnatha* sp.**	Oct	3	0	0
***Tetrathemis* sp.**	Oct	3	0	0
***Xiphoveloidea* sp.**	Oct	3	0	0
Acidocerinae Gen.	Nov	1	0	0
***Aliolimnatus* sp.**	Nov	1	0	0
***Appasus* sp.**	Nov	1	1	0
***Bellamya* sp.**	Nov	1	0	0
***Bulinus* sp.**	Nov	1	2	0
Caridina africana	Nov	1	98	7
Ceraptopogonidae Gen.	Nov	1	0	0
Chironimidae Gen.	Nov	1	0	0
***Cloeon* sp.**	Nov	1	3	0
***Corbicula* sp.**	Nov	1	0	6
***Crenigomphus* sp.**	Nov	1	0	0
***Cyathura* sp.**	Nov	1	0	0
***Haliplus* sp.**	Nov	1	0	0
***Laccocoris* sp.**	Nov	1	0	0
***Lestes* sp.**	Nov	1	0	0
***Mesovelia* sp.**	Nov	1	0	1
***Microvelia* sp.**	Nov	1	0	0
***Nepogomphoides* sp.**	Nov	1	0	0
Notonectidae Gen.	Nov	1	0	0
***Oncyogomphus* sp.**	Nov	1	0	0
***Paragomphus* sp.**	Nov	1	0	0
***Paramelita* sp.**	Nov	1	0	0
***Potomonautes* sp.**	Nov	1	0	0
***Povilla* sp.**	Nov	1	0	0
***Pseudagrion* sp.**	Nov	1	0	0
***Pseudospaerama* sp.**	Nov	1	1	0
***Ranatra* sp.**	Nov	1	0	0
***Sigara* sp.**	Nov	1	0	0
***Stenus* sp.**	Nov	1	0	0
Tarebia granifera	Nov	1	8	59
***Tetragnatha* sp.**	Nov	1	0	0
***Tetrathemis* sp.**	Nov	1	0	0
***Xiphoveloidea* sp.**	Nov	1	0	0
Acidocerinae Gen.	Nov	2	0	0
***Aliolimnatus* sp.**	Nov	2	0	0
***Appasus* sp.**	Nov	2	1	0
***Bellamya* sp.**	Nov	2	0	0
***Bulinus* sp.**	Nov	2	2	0
Caridina africana	Nov	2	156	1

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
Ceraptopogonidae Gen.	Nov	2	2	0
Chironimidae Gen.	Nov	2	31	0
***Cloeon* sp.**	Nov	2	0	0
***Corbicula* sp.**	Nov	2	0	12
***Crenigomphus* sp.**	Nov	2	0	0
***Cyathura* sp.**	Nov	2	0	1
***Haliplus* sp.**	Nov	2	0	0
***Laccocoris* sp.**	Nov	2	0	0
***Lestes* sp.**	Nov	2	0	0
***Mesovelia* sp.**	Nov	2	1	0
***Microvelia* sp.**	Nov	2	0	0
***Nepogomphoides* sp.**	Nov	2	0	0
Notonectidae Gen.	Nov	2	0	0
***Oncyogomphus* sp.**	Nov	2	0	0
***Paragomphus* sp.**	Nov	2	0	0
***Paramelita* sp.**	Nov	2	1	0
***Potomonautes* sp.**	Nov	2	0	0
***Povilla* sp.**	Nov	2	0	0
***Pseudagrion* sp.**	Nov	2	0	0
***Pseudospaerama* sp.**	Nov	2	1	0
***Ranatra* sp.**	Nov	2	0	0
***Sigara* sp.**	Nov	2	0	0
***Stenus* sp.**	Nov	2	0	0
Tarebia granifera	Nov	2	8	17
***Tetragnatha* sp.**	Nov	2	0	0
***Tetrathemis* sp.**	Nov	2	0	0
***Xiphoveloidea* sp.**	Nov	2	0	0
Acidocerinae Gen.	Nov	3	1	0
***Aliolimnatus* sp.**	Nov	3	0	0
***Appasus* sp.**	Nov	3	0	0
***Bellamya* sp.**	Nov	3	0	0
***Bulinus* sp.**	Nov	3	3	0
Caridina africana	Nov	3	68	6
Ceraptopogonidae Gen.	Nov	3	0	0
Chironimidae Gen.	Nov	3	0	5
***Cloeon* sp.**	Nov	3	2	1
***Corbicula* sp.**	Nov	3	0	0
***Crenigomphus* sp.**	Nov	3	0	0
***Cyathura* sp.**	Nov	3	1	0
***Haliplus* sp.**	Nov	3	0	0
***Laccocoris* sp.**	Nov	3	1	0
***Lestes* sp.**	Nov	3	0	0
***Mesovelia* sp.**	Nov	3	2	0
***Microvelia* sp.**	Nov	3	0	1
***Nepogomphoides* sp.**	Nov	3	0	0
Notonectidae Gen.	Nov	3	0	0
***Oncyogomphus* sp.**	Nov	3	0	0
***Paragomphus* sp.**	Nov	3	0	0
***Paramelita* sp.**	Nov	3	2	3

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Potomonautes* sp.**	Nov	3	0	0
***Povilla* sp.**	Nov	3	0	0
***Pseudagrion* sp.**	Nov	3	0	0
***Pseudospaerama* sp.**	Nov	3	2	1
***Ranatra* sp.**	Nov	3	0	1
***Sigara* sp.**	Nov	3	0	0
***Stenus* sp.**	Nov	3	0	0
Tarebia granifera	Nov	3	8	50
***Tetragnatha* sp.**	Nov	3	0	0
***Tetrathemis* sp.**	Nov	3	0	0
***Xiphoveloidea* sp.**	Nov	3	0	0
Acidocerinae Gen.	Dec	1	0	0
***Aliolimnatus* sp.**	Dec	1	0	0
***Appasus* sp.**	Dec	1	0	0
***Bellamya* sp.**	Dec	1	28	0
***Bulinus* sp.**	Dec	1	37	4
Caridina africana	Dec	1	0	14
Ceraptopogonidae Gen.	Dec	1	0	0
Chironimidae Gen.	Dec	1	0	2
***Cloeon* sp.**	Dec	1	0	13
***Corbicula* sp.**	Dec	1	0	1
***Crenigomphus* sp.**	Dec	1	0	0
***Cyathura* sp.**	Dec	1	0	1
***Haliplus* sp.**	Dec	1	0	0
***Laccocoris* sp.**	Dec	1	0	0
***Lestes* sp.**	Dec	1	0	0
***Mesovelia* sp.**	Dec	1	0	0
***Microvelia* sp.**	Dec	1	0	0
***Nepogomphoides* sp.**	Dec	1	0	0
Notonectidae Gen.	Dec	1	0	0
***Oncyogomphus* sp.**	Dec	1	0	0
***Paragomphus* sp.**	Dec	1	0	0
***Paramelita* sp.**	Dec	1	0	2
***Potomonautes* sp.**	Dec	1	2	0
***Povilla* sp.**	Dec	1	1	0
***Pseudagrion* sp.**	Dec	1	0	0
***Pseudospaerama* sp.**	Dec	1	0	3
***Ranatra* sp.**	Dec	1	0	0
***Sigara* sp.**	Dec	1	0	0
***Stenus* sp.**	Dec	1	0	0
Tarebia granifera	Dec	1	184	30
***Tetragnatha* sp.**	Dec	1	0	0
***Tetrathemis* sp.**	Dec	1	0	0
***Xiphoveloidea* sp.**	Dec	1	0	0
Acidocerinae Gen.	Dec	2	0	0
***Aliolimnatus* sp.**	Dec	2	0	0
***Appasus* sp.**	Dec	2	0	0
***Bellamya* sp.**	Dec	2	0	0
***Bulinus* sp.**	Dec	2	7	4

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
Caridina africana	Dec	2	49	13
Ceraptopogonidae Gen.	Dec	2	0	0
Chironimidae Gen.	Dec	2	0	2
***Cloeon* sp.**	Dec	2	0	13
***Corbicula* sp.**	Dec	2	0	0
***Crenigomphus* sp.**	Dec	2	0	0
***Cyathura* sp.**	Dec	2	0	1
***Haliplus* sp.**	Dec	2	0	0
***Laccocoris* sp.**	Dec	2	1	0
***Lestes* sp.**	Dec	2	0	0
***Mesovelia* sp.**	Dec	2	0	0
***Microvelia* sp.**	Dec	2	0	0
***Nepogomphoides* sp.**	Dec	2	0	0
Notonectidae Gen.	Dec	2	0	0
***Oncyogomphus* sp.**	Dec	2	0	0
***Paragomphus* sp.**	Dec	2	0	0
***Paramelita* sp.**	Dec	2	0	3
***Potomonautes* sp.**	Dec	2	0	0
***Povilla* sp.**	Dec	2	0	0
***Pseudagrion* sp.**	Dec	2	0	0
***Pseudospaerama* sp.**	Dec	2	0	3
***Ranatra* sp.**	Dec	2	0	0
***Sigara* sp.**	Dec	2	0	0
***Stenus* sp.**	Dec	2	1	0
Tarebia granifera	Dec	2	102	30
***Tetragnatha* sp.**	Dec	2	0	0
***Tetrathemis* sp.**	Dec	2	0	0
***Xiphoveloidea* sp.**	Dec	2	0	0
Acidocerinae Gen.	Dec	3	0	0
***Aliolimnatus* sp.**	Dec	3	0	0
***Appasus* sp.**	Dec	3	0	0
***Bellamya* sp.**	Dec	3	0	0
***Bulinus* sp.**	Dec	3	19	3
Caridina africana	Dec	3	72	13
Ceraptopogonidae Gen.	Dec	3	0	0
Chironimidae Gen.	Dec	3	0	2
***Cloeon* sp.**	Dec	3	2	13
***Corbicula* sp.**	Dec	3	0	0
***Crenigomphus* sp.**	Dec	3	0	0
***Cyathura* sp.**	Dec	3	0	1
***Haliplus* sp.**	Dec	3	0	0
***Laccocoris* sp.**	Dec	3	0	0
***Lestes* sp.**	Dec	3	0	0
***Mesovelia* sp.**	Dec	3	0	0
***Microvelia* sp.**	Dec	3	0	0
***Nepogomphoides* sp.**	Dec	3	0	0
Notonectidae Gen.	Dec	3	0	0
***Oncyogomphus* sp.**	Dec	3	0	0
***Paragomphus* sp.**	Dec	3	0	0

Genus	Month	Site	Lake Sibaya Abundance	Lake Mzingazi Abundance
***Paramelita* sp.**	Dec	3	0	3
***Potomonautes* sp.**	Dec	3	0	0
***Povilla* sp.**	Dec	3	0	0
***Pseudagrion* sp.**	Dec	3	0	0
***Pseudospaerama* sp.**	Dec	3	0	3
***Ranatra* sp.**	Dec	3	0	0
***Sigara* sp.**	Dec	3	0	0
***Stenus* sp.**	Dec	3	0	0
Tarebia granifera	Dec	3	77	31
***Tetragnatha* sp.**	Dec	3	0	0
***Tetrathemis* sp.**	Dec	3	0	0
***Xiphoveloidea* sp.**	Dec	3	0	0

Table S 21: Showing raw functional feeding group composition data collected at Lake Sibaya in 2021

Month	Site	Collector-Gatherer	Collector-Filterer	Predators	Shredders	Grazers/scrapers
Mar	1	103	0	7	0	96
Mar	2	44	0	11	0	4
Mar	3	140	0	12	0	84
Apr	1	26	0	1	0	62
Apr	2	63	8	9	0	258
Apr	3	71	0	2	0	123
May	1	88	0	7	0	56
May	2	23	0	0	0	25
May	3	30	0	7	0	11
Jun	1	61	0	19	0	118
Jun	2	39	0	1	0	2
Jun	3	56	0	20	0	24
Jul	1	18	0	0	0	4
Jul	2	27	0	1	0	0
Jul	3	78	1	0	0	2
Aug	1	70	0	2	0	394
Aug	2	112	0	3	0	46
Aug	3	72	0	1	0	47
Sep	1	85	0	2	0	275
Sep	2	42	0	5	0	61
Sep	3	103	0	2	0	93
Oct	1	59	0	1	0	257
Oct	2	16	0	1	0	139
Oct	3	161	0	1	0	344
Nov	1	102	0	1	0	10
Nov	2	189	0	4	0	10
Nov	3	75	0	4	0	11
Dec	1	3	0	0	0	249
Dec	2	49	0	2	0	109
Dec	3	74	0	0	0	96
Annual Average		69,300	0,300	4,200	0,000	100,333

Table S 22: Showing raw functional feeding group composition data collected at Lake Mzingazi in 2021

Month	Site	Collector-Gatherer	Collector-Filterer	Predators	Shredders	Grazers/scrapers
Mar	1	73	0	1	0	45
Mar	2	22	0	4	0	25
Mar	3	80	0	0	0	29
Apr	1	64	0	2	0	15
Apr	2	113	0	3	0	213
Apr	3	19	0	6	0	12
May	1	6	0	5	0	0
May	2	35	0	0	0	2
May	3	6	0	4	0	9
Jun	1	46	0	2	0	26
Jun	2	42	0	1	0	4
Jun	3	12	0	5	0	5
Jul	1	130	0	0	0	2
Jul	2	58	0	2	0	43
Jul	3	37	0	1	0	34
Aug	1	4	0	0	0	10
Aug	2	2	1	0	0	1
Aug	3	3	2	0	0	7
Sep	1	28	0	0	0	73
Sep	2	27	0	1	0	37
Sep	3	54	0	1	0	41
Oct	1	21	0	2	0	147
Oct	2	26	0	0	0	17
Oct	3	70	0	3	0	12
Nov	1	7	6	1	0	59
Nov	2	2	12	0	0	17
Nov	3	16	0	2	0	50
Dec	1	35	1	0	0	34
Dec	2	35	0	0	0	34
Dec	3	35	0	0	0	34
Annual Average		36,933	0,733	1,533	0,000	34,567

Table S 23: Showing raw functional feeding group percentage composition data collected at Lake Sibaya in 2021

Month	Site	Percentage Collector-Gatherer	Percentage Collector-Filterer	Percentage Predators	Percentage Shredders	Percentage Grazers/Scrapers
Mar	1	50,000	0,000	3,398	0,000	46,602
Mar	2	74,576	0,000	18,644	0,000	6,780
Mar	3	59,322	0,000	5,085	0,000	35,593
Apr	1	29,213	0,000	1,124	0,000	69,663
Apr	2	18,639	2,367	2,663	0,000	76,331
Apr	3	36,224	0,000	1,020	0,000	62,755
May	1	58,278	0,000	4,636	0,000	37,086
May	2	47,917	0,000	0,000	0,000	52,083
May	3	62,500	0,000	14,583	0,000	22,917
Jun	1	30,808	0,000	9,596	0,000	59,596
Jun	2	92,857	0,000	2,381	0,000	4,762
Jun	3	56,000	0,000	20,000	0,000	24,000
Jul	1	81,818	0,000	0,000	0,000	18,182
Jul	2	96,429	0,000	3,571	0,000	0,000
Jul	3	96,296	1,235	0,000	0,000	2,469
Aug	1	15,021	0,000	0,429	0,000	84,549
Aug	2	69,565	0,000	1,863	0,000	28,571
Aug	3	60,000	0,000	0,833	0,000	39,167
Sep	1	23,481	0,000	0,552	0,000	75,967
Sep	2	38,889	0,000	4,630	0,000	56,481
Sep	3	52,020	0,000	1,010	0,000	46,970
Oct	1	18,612	0,000	0,315	0,000	81,073
Oct	2	10,256	0,000	0,641	0,000	89,103
Oct	3	31,818	0,000	0,198	0,000	67,984
Nov	1	90,265	0,000	0,885	0,000	8,850
Nov	2	93,103	0,000	1,970	0,000	4,926
Nov	3	83,333	0,000	4,444	0,000	12,222
Dec	1	1,190	0,000	0,000	0,000	98,810
Dec	2	30,625	0,000	1,250	0,000	68,125
Dec	3	43,529	0,000	0,000	0,000	56,471
Annual Average		52,062	0,116	3,438	0,000	44,384

Table S 24: Showing raw functional feeding group percentage composition data collected at Lake Mzingazi in 2021

Month	Site	Percentage Collector-Gatherer	Percentage Collector-Filterer	Percentage Predators	Percentage Shredders	Percentage Grazers/Scrapers
Mar	1	61,345	0,000	0,840	0,000	37,815
Mar	2	43,137	0,000	7,843	0,000	49,020
Mar	3	73,394	0,000	0,000	0,000	26,606
Apr	1	79,012	0,000	2,469	0,000	18,519
Apr	2	34,347	0,000	0,912	0,000	64,742
Apr	3	51,351	0,000	16,216	0,000	32,432
May	1	54,545	0,000	45,455	0,000	0,000
May	2	94,595	0,000	0,000	0,000	5,405
May	3	31,579	0,000	21,053	0,000	47,368
Jun	1	62,162	0,000	2,703	0,000	35,135
Jun	2	89,362	0,000	2,128	0,000	8,511
Jun	3	54,545	0,000	22,727	0,000	22,727
Jul	1	98,485	0,000	0,000	0,000	1,515
Jul	2	56,311	0,000	1,942	0,000	41,748
Jul	3	51,389	0,000	1,389	0,000	47,222
Aug	1	28,571	0,000	0,000	0,000	71,429
Aug	2	50,000	25,000	0,000	0,000	25,000
Aug	3	25,000	16,667	0,000	0,000	58,333
Sep	1	27,723	0,000	0,000	0,000	72,277
Sep	2	41,538	0,000	1,538	0,000	56,923
Sep	3	56,250	0,000	1,042	0,000	42,708
Oct	1	12,353	0,000	1,176	0,000	86,471
Oct	2	60,465	0,000	0,000	0,000	39,535
Oct	3	82,353	0,000	3,529	0,000	14,118
Nov	1	9,589	8,219	1,370	0,000	80,822
Nov	2	6,452	38,710	0,000	0,000	54,839
Nov	3	23,529	0,000	2,941	0,000	73,529
Dec	1	47,297	1,351	0,000	0,000	45,946
Dec	2	48,611	0,000	0,000	0,000	47,222
Dec	3	47,945	0,000	0,000	0,000	46,575
Annual Average		50,10788542	2,998229117	4,5757726	0	41,81639539