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Historic water quality patterns in endemic schistosomiasis regions of southern Africa: Implications for disease vector distribution

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ABSTRACT

Alterations in atmospheric temperature and rainfall have a major influence on the distribution patterns of vector-borne diseases, including schistosomiasis, by changing the optimum habitat of the intermediate hosts (vectors) that transmit these diseases. Water quality degradation and the creation of artificial freshwater habitats may widen or restrict the distribution of disease vectors. Understanding the influence of water quality on ecosystems is essential in determining the environments that vectors thrive in and where potential outbreaks of the disease they carry will occur. This study aims to analyse the changes in various water quality variables between 1977 and 2009 in the Mbombela river systems and how this may have altered the presence of schistosomiasis snail vectors in these areas. Historical water quality data for the Sabie, Kaap, Komati and Crocodile rivers were obtained from the Department of Water and Sanitation monitoring stations and separated into three decades (1977–1987; 1988–1998 and 1999–2009). The Physicochemical Driver Assessment Index (PAI) was used to determine the ecological category of the rivers in each decade. A Principal Component Analysis was also created for the different river systems to identify possible changes in water quality over the 3 decades. According to the PAI, the Sabie River remained in a relatively good state while the Crocodile, Komati and Kaap rivers had a low water quality to begin with and became further degraded over the three decades. The PCAs indicated the Sabie and Crocodile rivers had increased levels of salts, nutrients and pH between 1988-1998 and 1999–2009 while the Kaap River had high salt content and pH between 1977 and 1998 and high nutrients during the decade of 1999–2009. These results indicate that there has been a change in the water quality of Mbombela river systems with notable increases in salt, pH and nutrient levels. As a result, the aquatic biota have likely been negatively affected and this poses severe impacts on the presence and abundance of schistosomiasis vectors as the snails are sensitive to changes in salinity, nutrients and pH.

1. Introduction

Variability of micro-climate parameters (rainfall and ambient temperatures) has drastically changed waterbodies' dynamics such as flow, pH, salinity, electrical conductivity, and the overall water quality (Merolla, 2011; Adekiya et al., 2020). Global changes, such as climate change, population growth, urbanization, industrial development and the expansion of agriculture put huge pressure on natural resources, particularly water (Okello et al., 2015). In Southern Africa where environmental sustainability issues are increasingly coming into conflict with human development objectives and where data are scarce, water is an important resource for social and economic well-being of the predominantly rural populations (Okello et al., 2015). Water quality is affected by both point and nonpoint sources of pollution in rural and urban areas (Glińska-Lewczuk et al., 2016). Some of these sources include sewage discharge, industrial discharge and agricultural run-off (Khatri and Tyagi, 2015). Extreme weather events may alter chemical parameters of water by increasing variables such as nitrates, phosphates, and sodium chloride (Nichols et al., 2018). Floods and droughts, as well as lack of awareness among end-users' aspects including hygiene, environment sanitation, storage and disposal need to be considered for the maintenance of water resources (Edokpayi et al., 2017; Manyangadze et al., 2021). These natural and human induced changes have led to decreased surface layer nutrient concentrations and prolonged

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oxygen depletion in river systems (Parry, 1998).

Natural processes such as soil weathering, hydrological factors and biotic factors, can cause fast-flowing water, leading to run-off that can degrade water quality by changing the physical and chemical composition of water (Barakat et al., 2016; Adekiya et al., 2020). Human contributions such as agriculture have both direct and indirect effects on both surface water quality and groundwater quality (Shabalala et al., 2013). The agri-business is one of the principal causes of water quality degradation, particularly in rural areas and this is as a result of the excessively used agrochemicals (Zia et al., 2013; Omer, 2019). Agro-pollutants associated with degrading water quality include nitrate- and phosphate-rich fertilizers, metals, pathogens, pesticides, irrigation, and the removal of natural habitats for farms and plantations (Shabalala et al., 2013). Modification of river systems has led to an increase in salts, organic and inorganic materials which degrade the water quality (El Deeb et al., 2017). Consequently, both acute and long-term degradation lead to changes in water quality parameters such as pH, salinity and nutrients (Khatri and Tyagi, 2015).

Water that comes from natural sources generally contains organisms that are a part of the biogeochemical cycles of aquatic systems (Walz et al., 2015; Koushali et al., 2021). Many ecosystems are sensitive to small changes in the physical and chemical composition of the water body, resulting in degradation of the ecosystem (Ghosh et al., 2020). Minerals and dissolved salts are necessary components of good quality water as they help maintain the health and vitality of organisms that rely on this ecosystem service (Khatri and Tyagi, 2015; Jones et al., 2021). Changes in water quality parameters, such as pH, salinity and nutrients, has resulted in changes in productivity rate, species composition, distribution and organism abundance (Waide et al., 1999; Oehri et al., 2017). These changes may also influence disease patterns including population dynamics of vector and water-borne diseases such as schistosomiasis (de Necker, 2020). Schistosomiasis is found within countries located in the tropical and subtropical regions, primarily sub-Saharan Africa including South Africa (Olkeba et al., 2020). Schistosomiasis continues to be neglected as a health threat, especially in rural populations located within provinces in the north-eastern parts of the country (Paull and Johnson, 2011; Adekiya et al., 2020). The Mbombela local municipality in the Mpumalanga Province falls under this endemic area. The municipality is located in the eastern parts of Mpumalanga, also known as the Lowveld, which are reported to have high prevalence of schistosomiasis (NICD, 2022). Mbombela's popularity is attributed to its favourable moderate climate, which makes it suitable for agricultural activities. However, this, combined with other natural factors, contributes to alterations in water quality. This places stress on freshwater ecosystems such as rivers and dams where various aquatic biota are found including molluscs that transmit schistosomiasis (Malan et al., 2009; Döll and Zhang, 2010). One of the major drivers of water quality deterioration within the Mbombela area (Nelspruit and Matsulu) is high pH and salinity levels which occurs as a result of a combination of wastewater effluent and runoff from fertilizers used for the intensively irrigated sugar cane and subtropical fruits (DWS, 2013; Khatri and Tyagi, 2015). Human influenced ecological changes due to irrigation schemes and dam creation for agricultural purposes, discharge of industrial, domestic wastewater and agricultural drainage have been reported to cause degradation of water quality (Stensgaard et al., 2013; Barakat et al., 2016).

Agricultural activities in the Mbombela area are predominantly in peri-urban areas and many of these occur near rivers and dams (Shabalala et al., 2013; Khatri and Tyagi, 2015). Small scale farming does not largely affect schistosomiasis distribution unlike large scale farming where agrochemicals are widely used, such as sugarcane plantation and the forestry industry that are found near river systems within Mbombela (Dallas and Day, 2004; DWS, 2013; Hoover et al., 2020). According to the National Freshwater Snail Collection (NFSC), 72% of *Biomphalaria pfeifferi* snails are found in perennial waterbodies within Mbombela, while *Bulinus globosus* snails were found to prefer permanently inundated aquatic habitats (Appleton et al., 1995; Malan et al., 2009). Any water quality changes due to environmental or human interference within this region can affect the snail species due to their prevalence which in turn influences the distribution of schistosomiasis (Merolla, 2011; McCreesh and Booth, 2014).

The variables mentioned above can vary from one waterbody to another and from one ecological zone to another. Changes in water quality variables will differ between ecosystems and ecological regions and may have either a positive or negative effect on the presence and abundance of disease vectors (Lefcort et al., 2015; Yu et al., 2022). Research on how water quality changes over time is key to understanding how this may alter the abundance and distribution of intermediate host snails of schistosomiasis. This will provide vital information for effective intervention strategies for such diseases in endemic areas. This study aims to analyse how the changes in various water quality variables between 1977 and 2009 in the Mbombela river systems between 1977 and 2009 in order to understand how this may have influenced the distribution of schistosomiasis vectors in the region.

2. Materials and methods

2.1. Study area

The Mbombela local municipality is located in the eastern part of South Africa in the Mpumalanga Province and forms part of the Ehlanzeni District. It is one of the four municipalities in the district, making up approximately a third of its geographical area with a surface area of 7 152 km^2. Mbombela is a regional centre catering for people from as far as Bushbuckridge in the north, Mozambique (Maputo) in the east, the Kingdom of Eswatini (Mbabane) in the southeast and Lydenburg in the west. The Municipality includes Mbombela City (Nelspruit), the capital town of Mpumalanga, the former Transitional Local Councils (TLCs) of Nsikazi, Hazyview and White River. Matsulu, Kabokweni, Daantjie and kaNyamazane make up the Municipality's urban and peri-urban nodes. The rural nodes include Hazyview, Mganduzweni, Ngodwana, Luphisi/Mpakeni, Nsikazi and Elandshoek in the planning areas (Mbombela IDP, 2012). The municipality is one of four in the district of Ehlanzeni and is endowed with several advantages, notably fertile land, a strategic location, great infrastructure and a beautiful environment (City of Mbombela, 2011). The Municipality also serves as a gateway to some of the best eco- and adventure activities in southern Africa. The moderate climate within the local Municipality makes it a preferred tourist destination all year round.

Mbombela local municipality is responsible for the huge former homeland trust area known as Nsikasi. The northern Nsikasi area receives chlorinated/disinfected (not purified) water from the Sabie, Crocodile and Komati River with a few water monitoring stations located along these river systems (Brown, 2005a; see Fig. 1). The main river systems within Mbombela are main tributaries to the Incomati basin (van der Zaag and Carmo Vaz, 2003). The basin falls within the summer (October–March) rainfall region with a mean annual precipitation of approximately 740 mm per annum, which generally increases from east to west (Mogebisa, 2021). The mean annual potential evaporation for the basin is about 1900 mm per annum, which generally decreases from east to west. Consequently, the deficit between rainfall and potential evaporation increases from west to east, irrigation becoming more important for crop production towards the east including exotic forest plantations (van der Zaag and Carmo Vaz, 2003). The most important anthropogenic changes in the river environment are caused by dams and reservoirs, water abstractions from these and interbasin transfers. The resulting modified river flow regime affects structural and functional attributes of the biotic communities (van der Zaag and Carmo Vaz 2003). In terms of land- and water use and economy, two crops dominate the basin, namely rain-fed commercial tree plantations (some 340,000 ha) and irrigated sugarcane cultivation (42, 800 ha). Excluded from this is 10,800 ha in the Umbeluzi basin which is

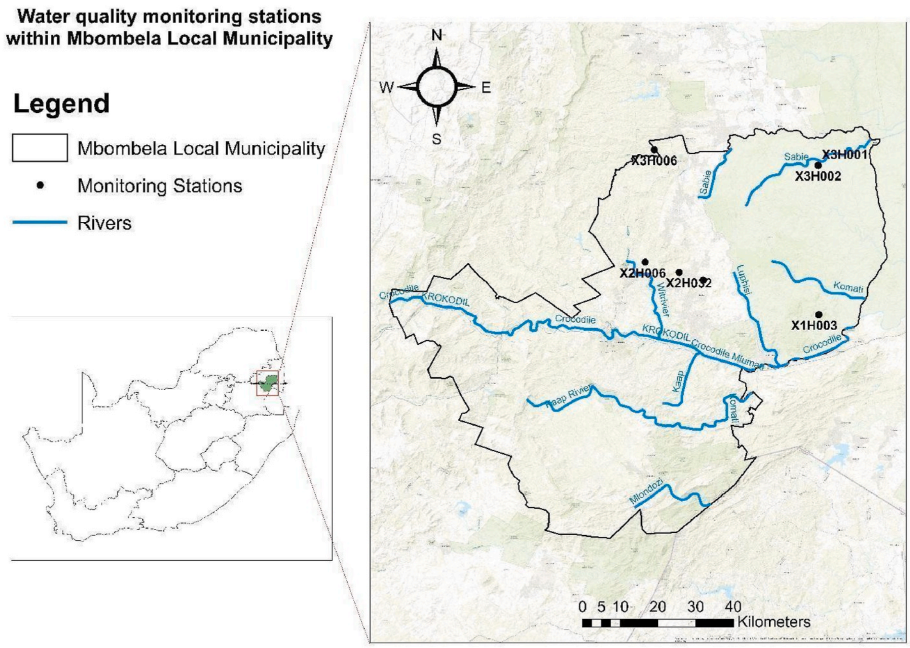

Fig. 1. A map indicating the largest rivers within Mbombela local municipality, South Africa, as well as the municipal gauging stations that monitor water quality in the region.

irrigated with Incomati water and the related sugar industry (Smithers et al., 2001; van der Zaag and Carmo Vaz, 2003). Other water uses include domestic, municipal, and industrial use as well as water for livestock and game.

According to the Community Survey 2016, the municipality is home to 729,464 people. This constitutes 35% of the Ehlanzeni District population, making it the biggest municipality in the district of Ehlanzeni (Statistics South Africa, 2018). In the last 15 years, the municipality has seen tremendous growth in its population, even when compared with the other municipalities in the Ehlanzeni District (Mbombela IDP, 2015–16). The municipality has also become a destination for approximately 40.67% of all immigrants coming into Ehlanzeni District Municipality (Statistics South Africa, 2011) meaning that the municipality is being greatly challenged to ensure that the communities are receiving appropriate water channels that offer quality water (Calfucoy et al., 2009). Small communities in Mbombela mainly use water for activities such as subsistence farming but due to the large population growth caused by immigrants, there is a great deal of pressure on the region and water availability (DWS, 2013). Due to limited water resources such as taps, people in the area make use of dams and rivers to access water and this has increased exposure to water-related diseases such as schistosomiasis.

3. Data collection

The historical water quality data for Sabie, Kaap, Komati and Crocodile rivers were obtained from the Department of Water and Sanitation (DWS) monitoring stations under the National Chemical Monitoring Programme for Surface Water for the years 1977–2009 (Table 1). Seven monitoring stations located under Mbombela local municipality were found to have sufficient data and included four river systems namely the Komati River, Kaap River, Crocodile River and Sabie River. The Komati, Crocodile and Sabie rivers are located on the Incomati River basin and are main sub-catchments contributing approximately 94% of natural discharge, with an area of 61% of the basin. These rivers, along with the Kaap River that is a tributary of the Crocodile River, are surrounded by small- and large-scale farmers with small communities (van der Zaag and Carmo Vaz, 2003; Okello et al., 2015) One site was located on the Komati River (X1H003), one site on the Kaap River (X2H022), two sites on the Crocodile River (X2H006 and X2H032) and three on the Sabie River (X3H001, X3H002 and X3H006; Table 1). The collection and analysis of the water quality data included sodium (Na; $mg.L^{-1}$), electrical conductivity (EC; $\mu S.cm^{-1}$), calcium (Ca; $mg.L^{-1}$), magnesium (Mg^{+}; $mg.L^{-1}$), chloride (Cl^{-}; $mg.L^{-1}$), sulfates (SO_4^{2-}; $mg.L^{-1}$), nitrate nitrogen/nitrite nitrogen (NO_3^{-}/NO_2^{-} $mg.L^{-1}$), orthophosphate (PO_4^{3-}; $mg.L^{-1}$), ammonium nitrogen (NH_4^{+}; $mg.L^{-1}$), total dissolved solids (TDS; $mg.L^{-1}$) and pH.

Table 1
Department of Water and Sanitation (DWS) monitoring stations used in this study including names, station numbers, years for which the water quality data were available and site names as appearing in the results.

River name	DWS Monitoring Station Number and Location	Years for which Historical Data is Available	Site names
Komati River	X1H003 (Komati River at Tonga)	1977–2009	Komati
Kaap River	X2H022 (Kaap River at Dolton)	1977–2009	Kaap
Crocodile River	X2H006 (Crocodile River at Karino)	1977–2009	Crocodile 1
Crocodile River	X2H032 (Crocodile River at Weltevrede)	1977–2009	Crocodile 2
Sabie River	X3H001 (Sabie River at Sabie)	1977–2009	Sabie 1
Sabie River	X3H002 (Klein Sabie River at Sabie)	1977–2009	Sabie 2
Sabie River	X3H006 (Sabie River at Perry's Farm)	1977–2009	Sabie 3

4. Data analysis

Data analysis was conducted using IBM SPSS Statistics 23 (IBM Corp. Released, 2015. IBM Statistics for Windows, Version 23.0). Data were separated into three decades namely 1977–1987, 1988–1998 and 1999–2009 and analysed for spatio-temporal variability in the water quality. For each site, the monthly records for each year were averaged to annual records and data are reported as an annual average per year. The Physicochemical Driver Assessment Index (PAI) was used to determine the water quality within the local municipality for each decade. The PAI is used in South Africa as part of the eco-classification determination process; for a specific site the tool can determine the state of chemical and physical water quality (Rossouw, 2011; de Necker et al., 2020). Microsoft Excel-based weighting is used in the PAI, and this allows for the assessment of individual variables through rating and ranking. This is based on a multicriteria decision analysis that considers several different water quality characteristics which include salts, pH and turbidity (Kleynhans et al., 2005; Divisi et al., 2017). The PAI was used to determine the water quality in the different time frames (1977–1987; 1988–1998 and 1999–2009) for each river system (Sabie, Crocodile, Komati and Kaap) and this is rated based on the degree to which water quality parameters have shifted from reference conditions, as well as the significance of each variable regarding the potential biotic responses it may cause. The program calculates a PAI score (Table 2) in % from 0 to 100 and an ecological category rating from A to F where A and 100% are the best or unmodified and F and 0% show the worst and most impacted ecosystem.

Descriptive statistics were done to test any significant differences in the water quality parameters over the three decades for all the variables at each site. All the river sites were compared between the three decades (1977–1987; 1988–1998 and 1999–2009). Data normalities were first tested using the D'Agostino Pearson normality test to determine if data were normally distributed (Ghasemi and Zahediasl, 2012). Following this, either one-way ANOVAs or Kruskal-Wallis H tests were conducted to assess whether any significant differences were present between the three decades (de Necker et al., 2020). Data that met the assumptions of normality (parametric distribution) were tested for significance using the one-way ANOVA with Tukey's post-hoc test while data that did not meet this assumption (non-parametric distribution) were tested for significance using the Kruskal-Wallis H test with Dunn's multiple comparisons test (Brown 2005b; de Necker et al., 2020). The significance value was set at $p < 0.05$.

To further compare the water quality within the four river systems, a Principal Component Analysis (PCA) was created for each of the different river systems (Sabie, Kaap, Komati and Crocodile) to determine possible changes in the water quality of each sampling site over the 3 decades. A PCA is based on a linear response model which relates a single variable, such as water quality data, to a particular site (Jolliffe and Cadima, 2016). A PCA is a method for lowering the dimensionality of such datasets, improving interpretability while reducing information loss (Solidoro et al., 2004). A diagram of the data for each site is visualized on a two-dimensional basis (de Necker et al., 2020). The placement of the sites on the diagram signifies either similarities or dissimilarities between the measured variable and sites based on the angle of the variables to one another. Variables at $\leq 90^\circ$ are positively correlated while variables $>90^\circ$ are negatively correlated (Šmilauer and Lepš, 2014).

Table 2
Ecological categories used for rating Physicochemical Driver Assessment Index (PAI) scores (Adapted from Kleynhans and Louw, 2008; de Necker et al., 2020).

Ecological Category	Description	Score
A	**Unmodified**, natural.	90–100
B	**Largely natural with few modifications**. A small change in natural habitats and biota may have taken place, but the ecosystem functions are essentially unchanged.	80–89
C	**Moderately modified.** Loss and change of natural habitat and biota have occurred, but the basic ecosystem functions are still predominantly unchanged.	60–79
D	**Largely modified.** A large loss of natural habitat, biota and basic ecosystem functions has occurred.	40–59
E	**Seriously modified.** Extensive loss of natural habitat, biota and basic ecosystem functions.	20–39
F	**Critically/Extremely modified.** Modifications have reached a critical level and the system has been modified completely, with an almost complete loss of natural habitat and biota. In the worst instances the basic ecosystem functions have been destroyed and the changes are irreversible.	0–19

5. Results

The historic water quality of the Sabie River was in an unmodified state (A-rating) between 1977 and 1987 at two of the sites (Sabie 1 and 2) with one of the sites being in a largely natural state (B-rating; Fig. 2a). For Sabie 1, the ecological category remained largely the same over all three decades (from 1977 to 2009) with the rating remaining between A and B, while the category for Sabie 2 and 3 declined to moderately modified state (C-rating) by 2009 with Sabie 2 having a lower overall score (75.92%) than Sabie 3 (79.13%). The decline in ecological category for all sites on the Sabie River, particularly Sabie 2 and Sabie 3 in 1988–1998 and 1999–2009 was driven by an increase in salinity (measured by EC, TDS, Mg^{+}; Ca and Cl^{-}) as well as pH and organic nutrients (measured by NO_3^-/NO_2^-; Appendix B and C). The PAI% for Crocodile 1 (84.86%) and Crocodile 2 (81.68%) indicated that the ecological category of the river was largely natural from 1977 to 1987 (Fig. 2b). The ecological category declined to a moderately modified state between 1999 and 2009 for Crocodile 1 (72.06%). The PAI% for Crocodile 2 (66.54%) indicated a decline in ecological category from 1999 to 2009 which was driven by an increase in salts and pH. The ecological category for Kaap River and Komati River remained moderately modified (rating C) over the three decades (from 1977 to 2009; Fig. 2c) with the PAI% of the Kaap River always lower; 1977–1987 (74.42%), 1988–1998 (67.12%), 1999–2009 (63.85%) compared to Komati River; 1977–1987 (78.5%), 1988–1998 (77.94%), 1999–2009 (71.59%).

Significance tests were conducted to assess changes in water quality variables for the three decades within Mbombela local municipality. Fluctuations were evident in the water quality variables for all the sampling sites with a general increase occurring over all sites across all three decades (Appendix A-G). The significance tests determined several significant changes in these water quality variables at all the monitoring stations.

In the Sabie River there was a significant increase in pH and the measured salts, including EC, TDS, and Mg^{+} from 1977 to 2009 at all three sampling locations ($p < 0.05$; see Appendix A-C for specific p-values). In addition, Ca significantly increased at two of the sites along with NO_3^-/NO_2^- and NH_4^+ (Sabie 1 and Sabie 2) while SO_4^{2-} significantly decreased from 1977 to 2009 ($p < 0.05$; Appendix A and B). There was also a significant increase in Cl^{-} and PO_4^{3-} at two of the Sabie River sites (Sabie 1 and Sabie 3; $p < 0.05$; Appendix A and C). Salinity, including EC, Na, TDS, Ca, and Cl^{-}, significantly increased for the Crocodile River system (1 & 2) between 1999 and 2009 along with pH ($p < 0.05$). Other water variables such as NO_3^-/NO_2^- SO_4^{2-}, PO_4^{3-} and NH_4^+ showed significant increases for the monitoring station within Crocodile River 1 (Appendix D & E). The Kaap and Komati rivers both showed a similar trend regarding pH which increased significantly from 1977 to 2009 ($p < 0.0001$; Appendix F and G). The results further indicated a significant increase in PO_4^{3-} in the Kaap River from 1977 to 2009 and significant increases in EC, TDS, Na, Mg^{+}, Ca, Cl^{-}, NO_3^-/NO_2^-, and SO_4^{2-} in the Komati River (Appendix G).

The PCA indicated a similar trend across all monitoring sites where there was a distinct separation between each of the three decades. The PCA of the first Sabie River site (Sabie 1) compared water quality variables over the three decades and explains 69.04% of data variation (Fig. 3a). Separation on the first axis, which explains 52.53% of data variation, was a result of differences in water quality between the 1977–1987 decade and the other two decades (1988–1998 and 1999–2009). This was driven by salts (TDS, EC, Ca, Na and Mg^{+}) and pH that were generally much higher in the decades of 1988–1998 and 1999

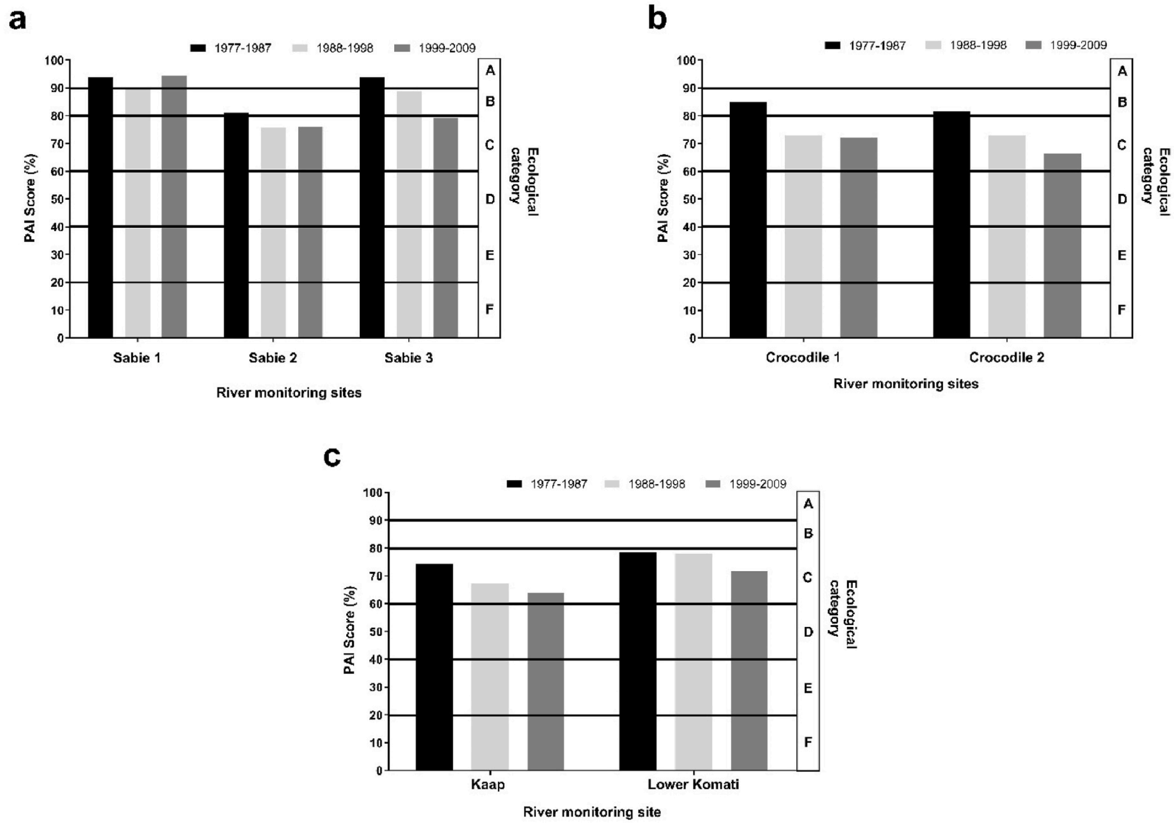

Fig. 2. Physicochemical Driver Assessment Index (PAI) scores (%) and ecological categories calculated from monitoring stations located on the a) Sabie River, b) Crocodile River and c) Komati River catchment in Mbombela local municipality.

to 2009 than in 1977–1987. Separation on the second axis explained 16.51% of data variation and this was driven largely by much higher values of nutrients (NO_3^-/NO_2^-; PO_4^{3-} and NH_4^+) in the 2000s compared to the other years.

The PCA for Sabie 2 (Fig. 3b) explained 60.79% of data variation, with 35.94% explained on the first axis and 24.85% explained on the second axis. All three decades separated from one another as a result of significant changes in water quality (see Appendix A-C). The second decade, 1988–1998, was associated with EC, TDS, Mg^+ and Ca that were all significantly higher in this decade compared to the previous one. The third decade, 1999–2009, showed a positive association with most nutrients including NO_3^-/NO_2^-, PO_4^{3-} and NH_4^+ as well as pH while SO_4^{2-} was positively associated with 1977–1987 since it was also highest during this decade. The PCA for Sabie 3 (Fig. 3c) explained a total of 66.40% of data variation, with 51.84% explained on the first axis and 14.56% explained on the second axis. The third decade, 1999–2009, was associated with EC, TDS, pH and Na that was significantly higher in this decade compared to the other two. There were high levels of SO_4^{2-} over the decade of 1977–1987, and an association between pH and 1988–1998 due to the significant differences in water quality between this decade and 1977–1987 (see Appendix C).

The PCA of the first Crocodile River site (Crocodile 1) explained 78.95% of total data variation with 67.49% explained on the first axis and 11.46% on the second axis, (Fig. 4a). Separation on this graph was a result of most water quality variables that were significantly different between the three decades, particularly all salts, pH and nutrients (excluding NO_3^-/NO_2^- and NH_4^+), that were significantly higher in the 2000s compared to the other two decades (1977–1987 and 1988–1998). The PCA of the second Crocodile River site (Crocodile 2) accounted for 76.96% of data variation, with 64.56% explained on the first axis and 12.40% explained on the second axis (Fig. 4b). Separation on this graph was similar to that of Crocodile 1, with separation largely as a result of significantly higher water quality variables, particularly salts and pH in the later decade (1999–2009) compared to earlier decades (1977–1987).

The PCA for the Kaap River explained 70.98% of data variation, with 59.02% explained on the first axis and 11.96% on the second axis. Separation on this graph was as a result of most water quality variables, excluding PO_4^{3-}, NO_3^-/NO_2^-, SO_4^{2-} and EC, that were significantly higher in the later decades, particularly the 2000s, compared to the 1970s and 1980s (Appendix F). The PCA results for the Komati River were similar to that of the Kaap River, with a large difference in water quality in the 1970s and 1980s (1977–1987) compared to the later decades (Fig. 5). The PCA for the Komati River explained 77.91% of data variation, with 67.93% explained on the first axis and 9.97% explained on the second axis. Several water quality variables including pH, Ca, TDS, Cl^- and EC, decreased in the Komati River during 1977–1987 and then increased over the other two decades. The water quality between 1988 and 1998 had moderately higher nutrients (PO_4^{3-}, NH_4^+), salt (Ca, TDS, Cl^-) and pH compared to previous years (Fig. 6).

6. Discussion

This study assessed how the changes in chemical water composition, including salts and nutrients, between 1977 and 2009 had an impact on water quality in the Mbombela river systems using a set of ecological factors and water parameters. The main findings of the study showed that the water quality in the Sabie, Crocodile, Kaap and Komati River systems changed significantly from 1977 to 2009. There was a noticeable change in nutrients and salts over the three decades with 1999–2009 showing the highest concentrations in most cases. There was a decrease in river ecological category for most of the river systems with Sabie and Crocodile River having a high PAI% at monitoring points from 1988 to 2009 along with a steady increase in pH, EC and salinity, and

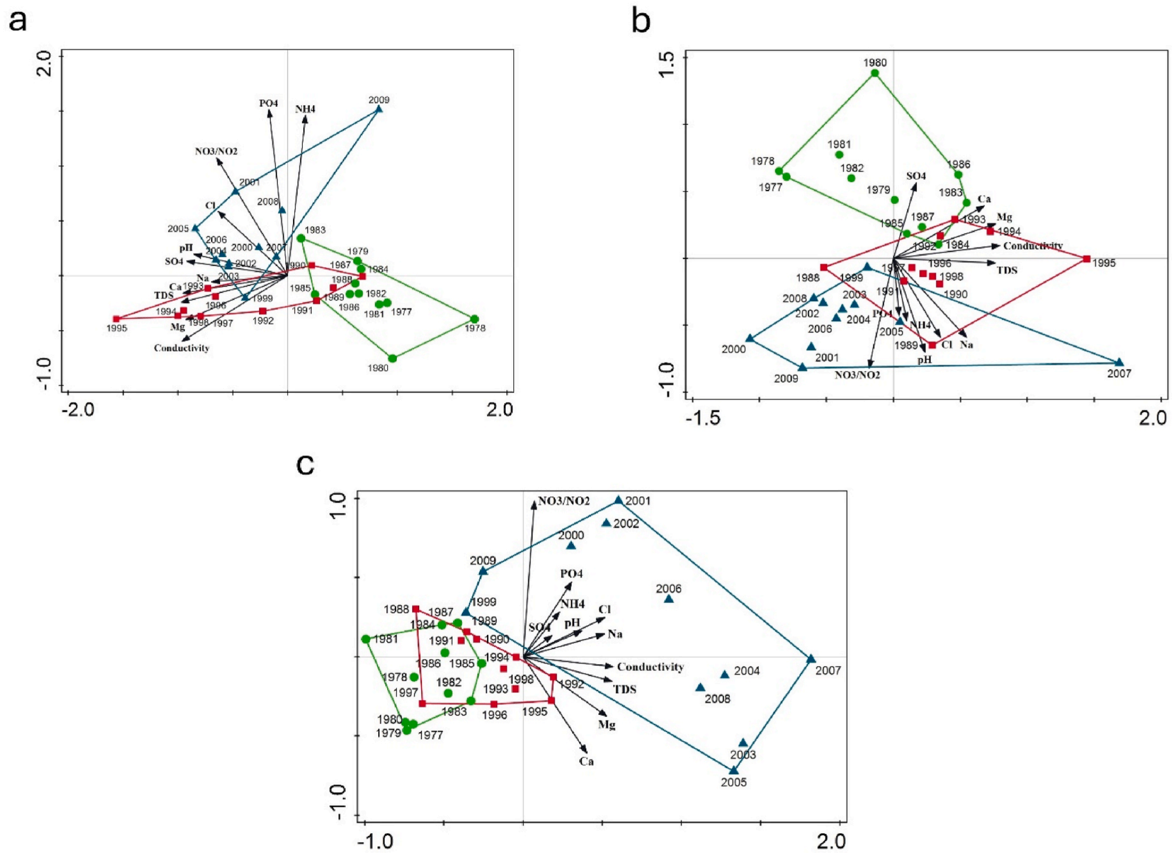

Fig. 3. Principal Component Analysis (PCA) plot of chemical water variables measured from 1977 to 1987 (green circles and lines); 1988–1998 (red squares and lines) and 1999–2009 (blue triangles and lines) at the a) Sabie 1 (X3H001); b) Sabie 2 (X3H002) and c) Sabie 3 (X3H006) Department of Water and Sanitation (DWS) monitoring stations located on the Sabie River system. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

changes in organic and inorganic nutrients, with salinity levels significantly increasing for the four river systems.

6.1. Historic water quality patterns of Mbombela local municipality

Water quality in South Africa is considered a difficult issue, and the country has struggled to supply many rural and local municipalities (LeMarie et al., 2006). Rapid urbanization and growing levels of pollution in our river systems are among the key reasons why South Africa is facing major challenges with its water quality (Nkomo and van der Zaag, 2004). However, some municipalities and districts have little funding for basic chemicals to clean wastewater, while others lack sound management and expertise which has led to people using water from rivers and streams for basic needs. Water is critically important to the economies and social well-being of the predominantly rural populations of southern Africa, where environmental sustainability issues are increasingly coming into conflict with human development objectives, and where data are also scarce (Pollard et al., 2011; Riddell et al., 2013). The local economies and livelihoods of many southern African communities are strongly dependent on agriculture and fisheries, and water availability remains one of the main constraints to development in Africa (Pollard and du Toit, 2009).

The water quality variables alternated in concentration within the three studied decades, with overall water quality deteriorating from the 1970s to the 2000s. In general, even though there were significant increases in salinity and pH, sites on the Sabie River were the least impacted of all other river sites, with the PAI of this river decreasing the least of all the studied sites. In particular, the first site on the Sabie River (Sabie 1) remained between an Unmodified (Ecological Category A) and Largely Natural category (Ecological Category B). A significant proportion of the Sabie River catchment lies within Kruger National Park (KNP), and hence the river is considered to be less modified compared to the other river systems in the region (Pollard et al., 2011; Gerber et al., 2015; Majdi et al., 2022). Furthermore, KNP has had significant influence on water management outside of the park (Shikwambana et al., 2021) which has likely allowed for reduced negative impacts on the Sabie River. The increase in nutrients and salinity at the monitoring sites on this river was likely caused by the impoundments and increased human activities located on the Sabie catchment outside the KNP. It is well-known that dams have the potential to cause negative impacts on downstream water quality, including nutrient loading into downstream ecosystems (Marcinkowski and Grygoruk, 2017; de Necker et al., 2020). Other activities that contribute to increases in salinity and nutrients in riverine ecosystems include natural surface runoff, and human activities, such as farming and livestock (Carpenter et al., 1998; Dabrowski et al., 2014). Subsistence and commercial farming areas have increased in the past few decades around KNP along with increased population growth and urbanization (Rivers-Moore et al., 2005) and runoff from these cultivated and urbanised areas, along with water abstraction, siltation and sediment erosion have likely reduced water flow and increased nutrient and salinity loading into this ecosystem. Similar findings were reported by Shikwambana et al. (2021) who made use of diatom communities to monitor water quality changes of KNP and determined that water quality of the Sabie River has seen significant changes, with a definitive shift in the diatom communities as a result of much increased EC and pH, particularly from 1983 to 2015.

There was a significant increase in nutrients and salinity in the water at both sites of the Crocodile River from 1977 to 2009 which resulted in

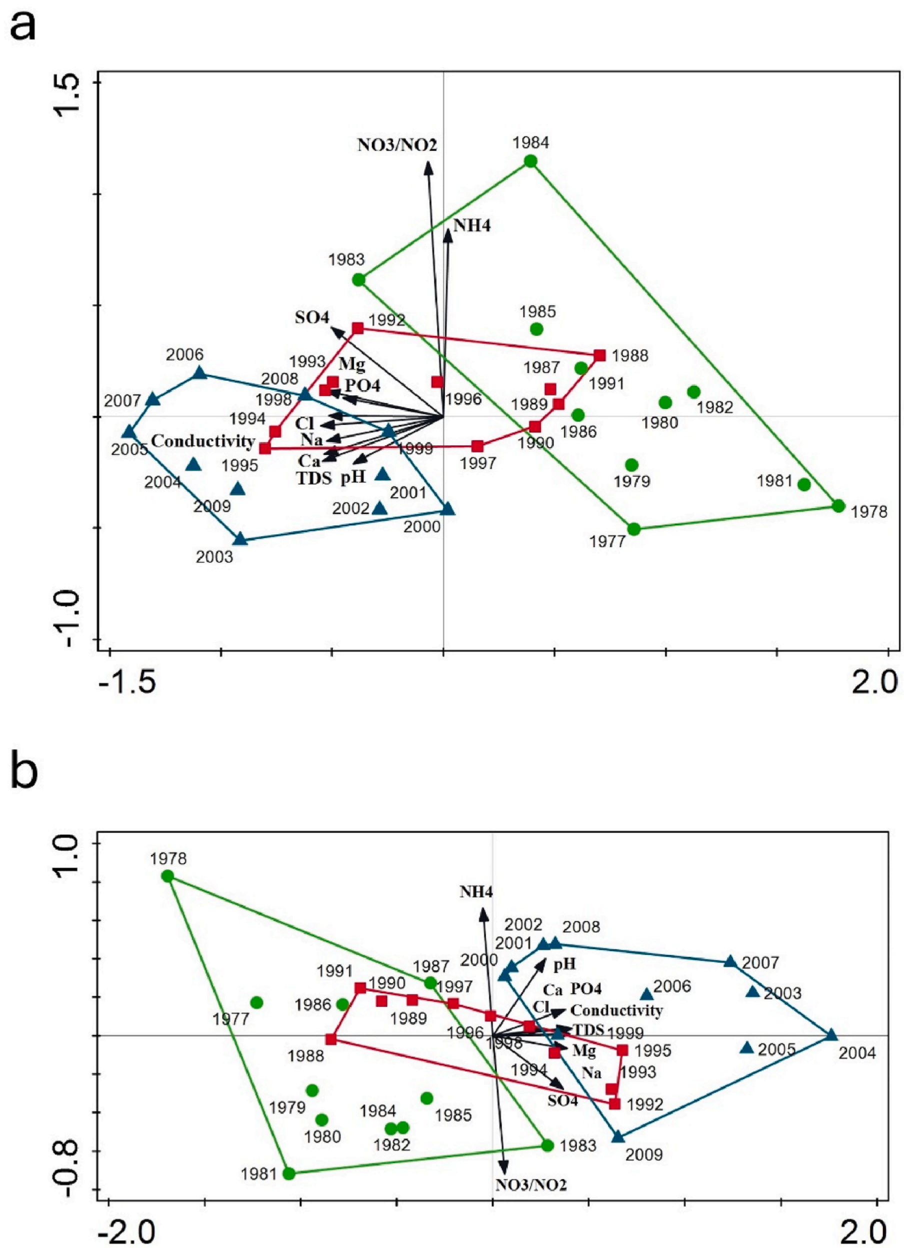

Fig. 4. Principal Component Analysis (PCA) plot of chemical water variables measured from 1977 to 1987 (green circles and lines); 1988–1998 (red squares and lines) and 1999–2009 (blue triangles and lines) at the a) Crocodile 1 (X2H006) and b) Crocodile 2 (X2H032) Department of Water and Sanitation (DWS) monitoring stations located on the Crocodile River system. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

a decline in the river's PAI scores from Largely natural (Ecological Category B) to Moderately modified (Ecological Category C). This was likely as a result of increased human development and activities surrounding the Crocodile River catchment that has occurred since the 1970s (Bunn and Arthington, 2002; Sauka, 2016). Indeed, the Crocodile River system is considered as one of the most impacted and water-stressed ecosystems in South Africa as it is surrounded by intense agricultural activities, including 12,500 ha of sugarcane and several commercial crop farms for mangoes, citrus, avocadoes, and bananas (Bate et al., 1999; DWAF 2004; Van der Laan et al., 2012). These are all known to use a great deal of nitrogen and phosphate rich fertilizers (Sahula, 2015). During periods of high rainfall these fertilizers leach from the soils into nearby waterbodies, such as rivers, thus increasing the nutrient levels in these ecosystems (Shabalala et al., 2013). Large exotic pine and eucalyptus forests (approximately 1775 km^2) and numerous rural and urban communities also surround the catchment of this river, further contributing toward greater water demand and increased salinity and eutrophication in this ecosystem (Bate et al., 1999; DWAF 2004; Van der Laan et al., 2012). Similarly, a study by Majdi et al. (2022) determined that water quality and aquatic invertebrate community structures were much lower in the Crocodile River compared to other river sites flowing through the KNP (Sabie and Olifants rivers), with this system containing much higher levels of various nutrients (phosphates, nitrates and ammonium). These findings were also attributed to the numerous anthropogenic impacts on this river system occurring outside of KNP.

The water quality of the Kaap River appears to have been negatively affected for several decades as the PAI scores for this river was already in a Largely modified state (Ecological Category C) in the 1970's with a further decline observed towards 2009. This resulted from significant

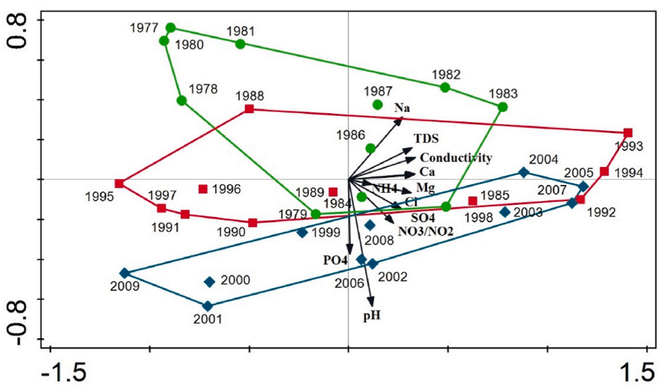

Fig. 5. Principal Component Analysis (PCA) plot of chemical water variables measured from 1977 to 1987 (green circles and lines); 1988–1998 (red squares and lines) and 1999–2009 (blue triangles and lines) at the Kaap River, Dolton (X2H022) Department of Water and Sanitation (DWS) monitoring stations located on the Kaap River system. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

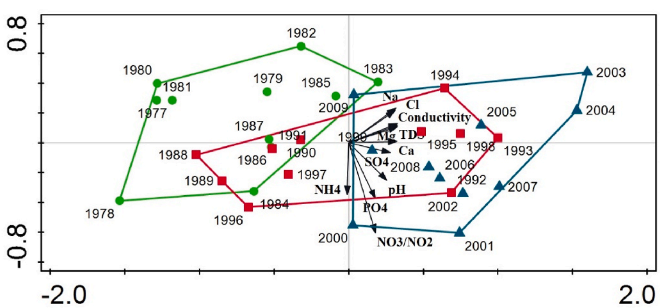

Fig. 6. Principal Component Analysis (PCA) plot of chemical water variables measured from 1977 to 1987 (green circles and lines); 1988–1998 (red squares and lines) and 1999–2009 (blue triangles and lines) in Tonga (X2H022) Department of Water and Sanitation (DWS) monitoring stations located on the Komati River system. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

increases in salinity, pH and nutrients over the three decades and was likely due to changes in land use patterns and continued misuse of this riverine ecosystem. Increases in EC as a result of high anthropogenic inputs, such as agriculture and acid mine drainage (AMD) from old gold mines along the river course and river water evaporation, are the most likely causes for the observed high concentration of salts and nutrients in this river (Van der Laan et al., 2012; Dlamini et al., 2021). The anthropogenic salinisation of inland waters is manifested by an increase in the TDS which, among others, is caused by mining activity (Pond et al., 2008; Sowa et al., 2019). The Kaap River also flows past several pine tree forests which likely further contributed to increased levels of salts, nutrients, and pH from 1999 to 2009, as forestry reduces the dilution capacity of effluents and contributes significantly to quantities of salt in river systems (Van der Laan et al., 2012). Similarly, DWAF (2004) reported that, based on several ecological indicators including fish and aquatic invertebrate presence, water quality, riparian vegetation and instream integrity, the Present Ecological State (PES) of the Kaap River was a C/D, making it Moderate to Largely modified river ecosystem with several changes in not only ecosystem processes but a substantial loss of natural habitat and fauna.

There was a significant increase in nutrients and salts at the Komati River monitoring site from 1988 to 2009 and this was likely due to new developments and irrigation projects that started in the early 1990s including cattle and game farming near the river (Rahman et al., 2013). These, and irrigation activities from sugar mills, contribute significantly to nutrient inputs in this river, while the mines, such as sand mining, also found in the river's catchment likely contributed to the increased salinity, pH, and other organic nutrients (Van der Laan et al., 2012). The water quality of this river is also severely impacted by water diversification at the Tonga Weir, located upstream of the monitoring point, which has led to substantial increases in salinity during low-flow periods as well as complete cessation of water flow on several occasions (AfriDev, 2005). These impacts resulted in the river PES being classified as E in 2005, which is a Seriously modified ecosystem with crucial changes in ecosystem function and loss of natural habitat and fauna (AfriDev, 2005).

6.2. *Implications of water quality alterations on disease vectors in the Mbombela local municipality*

Changes in water quality parameters of freshwater ecosystems have been known to have several negative impacts on the aquatic biota associated with these ecosystems (Dallas and Rivers-Moore, 2014). Anthropogenic and natural factors cause disturbances to the aquatic environment that contribute to the introduction of foreign objects that alter water quality which can cause freshwater species, including disease vectors, to experience stress when water quality parameters are altered (Sowa et al., 2019). These factors include, but are not limited to, wastewater pollution, changes in temperature, pH, and salinity, increases in sedimentation and alterations to microhabitat structure (Edokpayi et al., 2017). Findings of the current study indicated that water quality of the river systems within Mbombela local municipality has degraded since the 1970s. This has likely impacted the distribution and abundance of aquatic biota of these ecosystems, including the freshwater snails (vectors) that transmit schistosomiasis, which is prevalent in the Mbombela region (Malan et al., 2009; Riddell et al., 2019). Two snail species are known to be vectors for schistosomiasis in the Mbombela region namely *Bulinus globosus* and *Biomphalaria pfeifferi* (De Kock et al., 2004; De Kock and Wolmarans 2005).

It has been reported that these snail vectors are sensitive to alterations in water pH and salinity, with research indicating a negative correlation between snail presence and pH levels <6 and >8.5 (Kazibwe et al., 2006). Further, the vector *Bul. globosus* reportedly prefers much lower salinity (100–300 ppm) than *Biom. pfeifferi* (601–1000 ppm) meaning that *Biom. pfeifferi*, would likely be able to survive higher levels of salinity reported from the studied river sites, while *Bul. globosus*, would not (Adekiya et al., 2020; Manyangadze et al., 2021). Increased nutrients observed within the four river systems most likely would not have any direct effects on the freshwater snails, although there would be indirect effects through alterations of the aquatic plants they feed - and live on. Aquatic plant density and presence, food availability, and competition can affect freshwater snail distribution (McCreesh et al., 2015) and there exists a mutual relationship between submerged plant presence and snail presence (Tan et al., 2015). Increases in water nutrients stimulate plant growth, particularly submerged plants, and these plants provide a good habitat for snail that feed on periphyton, with both the density and biomass of snails being greater where plants are abundant (Wang et al., 2006; Mo et al., 2017). However, increased nutrient loading in an ecosystem, as was observed in the studied river systems over the three decades, may also result in toxic algal blooms (van Ginkel, 2011), which will cause reduced oxygen in the water, leading to severely negative effects on not only aquatic vegetation, but the aquatic biota that rely on them (NRC, 2000; Frost & Sullivan, 2010), including freshwater snails.

The much-increased nutrients observed over the three decades in the present study could have severe consequences for the distribution and abundance of schistosomiasis vectors in the region. Limited information is available regarding the current status of *Bul. globosus* and *Biom. pfeifferi* in the three studied rivers, as few comprehensive distribution studies have taken place since the 1980s and comprehensive assessments of this statement are therefore not possible. However, a recent investigation by Majdi et al. (2022) into the aquatic invertebrates of the Sabie, Crocodile, and Olifants rivers in the Kruger National Park did not detect the presence of these snails in any of the studied rivers. Their study attributed the decline in biofilm-dwelling organisms, including these snail species, to increased nutrient levels and changes in other water quality parameters and the presence of the invasive snail species *Tarebia granifera*, found in exceedingly high abundances in these rivers. *Tarebia granifera* is more tolerant to increased salinity, nutrient levels, and temperatures compared to other snail species, including those that transmit schistosomiasis (Pearson et al., 2024). This tolerance allows *T. granifera* to outcompete other invertebrate and snail species, potentially altering their distribution patterns (Pearson et al., 2024). This finding suggest that changes in abiotic environmental factors observed in our study are indirectly influencing the distribution and abundance of schistosomiasis vectors in the study region, potentially affecting the prevalence of the disease as well.

7. Conclusion

This study employed multivariate statistical analyses and a water quality index to investigate historical water quality trends of four ecologically and economically important river systems and their impact on schistosomiasis vector distribution. The results revealed a decline in water quality since the 1970s, with increased pH, salinity, and nutrient levels in the four river systems. The results from this study further provide valuable insight into the long-term water quality trends and effect of degrading water quality within the main river systems in the Mbombela local municipality. To manage schistosomiasis vectors and aquatic ecosystems, it is essential to update distribution records of freshwater snails, particularly those of economic importance and improve the water quality of aquatic ecosystems where these species occur. It is also necessary to improve climate surveillance systems, raise public awareness of schistosomiasis and its health impacts, and implement appropriate control measures at local scales. These strategies are vital for effective management and mitigation of schistosomiasis and other disease vectors.

CRediT authorship contribution statement

F. Letlaila: Conceptualization, Data curation, Methodology, Writing – original draft, Writing – review & editing. **N. Ayob:** Writing – original draft. **N. Nkosi:** Writing – original draft. **L. de Necker:** Conceptualization, Formal analysis, Methodology, Writing – original draft.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Lizaan de Necker reports financial support was provided by Water Research Commission.

Data availability

Data will be made available on request.

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Appendix A. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Sabie, Sabie 1 (X3H001) Department of Water and Sanitation (DWS) monitoring station located on the Sabie River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 12)	1988–1998 (n = 10)	1999–2009 (n = 11)	Significance
EC ($\mu S.cm^{-1}$)	95.46 ± 7.57^{ab} (80.50–110.3)	128.1 ± 22.41^{a} (96.09–169.9)	118 ± 21.71^{b} (60.54–138.8)	$p = 0.006$; $F_{(2,30)} = 9.481$
pH	6.92 ± 0.26^{ab} (6.44–7.37)	7.91 ± 0.13^{ac} (7.74–8.05)	7.66 ± 0.10^{bc} (7.5–7.85)	$p < 0.0001$; $F_{(2,30)} = 92.21$
TDS ($mg.L^{-1}$)	67.60 ± 6.10^{ab} (54.54–76.71)	92.97 ± 17.76^{a} (69.09–121.8)	82.98 ± 10.92^{b} (60.67–96.42)	$p = 0.0001$; $F_{(2,30)} = 12.23$
Na ($mg.L^{-1}$)	2.07 ± 0.64^{a} (1.48–3.92)	2.54 ± 1.01 (1.14–4.80)	2.53 ± 0.60^{a} (0.94–3.17)	$p = 0.0312$; $\chi^2 = 6.933$
Mg^{+} ($mg.L^{-1}$)	4.90 ± 0.51^{ab} (3.75–5.69)	6.48 ± 0.98^{a} (5.04–7.9)	5.65 ± 0.66^{b} (4.65–6.67)	$p < 0.0001$; $F_{(2,30)} = 12.90$
Ca ($mg.L^{-1}$)	7.98 ± 0.71^{ab} (6.63–8.91)	10.72 ± 2.67^{a} (7.04–14.10)	10.23 ± 1.15^{b} (8.5–12.36)	$p = 0.0002$; $F_{(2,30)} = 11.16$
Cl^{-} ($mg.L^{-1}$)	3.28 ± 1.41^{a} (1.29–4.98)	4.38 ± 1.00 (3.25–6.46)	4.57 ± 0.63^{a} (3.64–5.97)	$p = 0.0149$; $F_{(2,30)} = 4.852$
NO_3^-/NO_2^- ($mg.L^{-1}$)	0.19 ± 0.09^{a} (0.04–0.31)	0.25 ± 0.03 (0.18–0.30)	0.37 ± 0.08^{a} (0.25–0.44)	$p < 0.0001$; $F_{(2,30)} = 16.95$
SO_4^{2-} ($mg.L^{-1}$)	3.67 ± 1.92 (1.4–8.53)	7.81 ± 3.83^{a} (2.26–14.31)	7.69 ± 2.29^{b} (3.7–11.29)	$p = 0.0012$; $F_{(2,30)} = 8.445$
PO_4^{3-} ($mg.L^{-1}$)	0.01 ± 0.01 (0.004–0.03)	0.01 ± 0.004 (0.009–0.02)	0.02 ± 0.01 (0.01–0.05)	$p = 0.0387$; $F_{(2,30)} = 3.632$
NH_4^{+} ($mg.L^{-1}$)	0.06 ± 0.02 (0.02–0.22)	0.04 ± 0.02^{a} (0.02–0.08)	0.12 ± 0.09^{a} (0.03–0.34)	$p = 0.0108$; $\chi^2 = 9.061$

Appendix B. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Klein Sabie River, Sabie 2 (X3H002) Department of Water and Sanitation (DWS) monitoring station located on the Sabie River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 11)	1988–1998 (n = 11)	1999–2009 (n = 11)	Significance
EC ($\mu S.cm^{-1}$)	137.6 ± 14.39 (115.9–162.5)	146.6 ± 11.06^{a} (129.7–174.6)	130.4 ± 18.97^{a} (110.5–180.9)	$p = 0.0112$; $\chi^2 = 8.992$
pH	7.03 ± 0.30^{ab} (6.47–7.43)	7.79 ± 0.36^{a} (6.78–8.05)	7.70 ± 0.12^{b} (7.53–7.88)	$p < 0.0001$; $F_{(2,30)} = 25.06$
TDS ($mg.L^{-1}$)	90.47 ± 6.46^{a} (79.63–103.3)	101.8 ± 10.67^{ab} (82.73–125)	89.08 ± 14.33^{ab} (67.67–126.1)	$p = 0.0208$; $F_{(2,30)} = 4.419$

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Variable	1977–1987 (n = 11)	1988–1998 (n = 11)	1999–2009 (n = 11)	Significance
Na ($mg.L^{-1}$)	2.33 ± 0.72 (1.54–3.63)	2.92 ± 0.73 (2.09–4.74)	3.21 ± 1.28 (2.36–6.94)	$p = 0.0691$; $\chi^2 = 5.346$
Mg^+ ($mg.L^{-1}$)	6.9 ± 0.78^a (5.31–7.71)	7.23 ± 0.51^b (6.19–8.10)	6.03 ± 0.97^{ab} (4.87–8.62)	$p = 0.003$; $F_{(2,30)} = 7.076$
Ca ($mg.L^{-1}$)	11.72 ± 0.45 (11.10–12.36)	11.91 ± 1.14^a (10.17–14.52)	10.82 ± 1.0^a (9.54–13.06)	$p = 0.0195$; $F_{(2,30)} = 4.501$
Cl^- ($mg.L^{-1}$)	4.03 ± 1.94 (1.94–7.25)	5.18 ± 1.20 (3.28–7.72)	5.02 ± 1.04 (3.68–7.59)	$p = 0.1509$; $F_{(2,30)} = 2.015$
NO_3^-/NO_2^-($mg.L^{-1}$)	0.24 ± 0.08^{ab} (0.06–0.37)	0.30 ± 0.05^{ac} (0.21–0.39)	0.38 ± 0.07^{bc} (0.27–0.48)	$p = 0.0001$; $F_{(2,30)} = 12.22$
SO_4^{2-} ($mg.L^{-1}$)	13.98 ± 4.22 (6.40–21.85)	8.94 ± 2.86^a (6.2–16.66)	9.47 ± 2.12^b (6.6–12.21)	$p = 0.0013$; $F_{(2,30)} = 8.310$
PO_4^{3-} ($mg.L^{-1}$)	0.007 ± 0.004 (0.003–0.015)	0.01 ± 0.005 (0.008–0.022)	0.03 ± 0.04 (0.009–0.15)	$p = 0.0495$; $F_{(2,30)} = 3.327$
NH_4^+ ($mg.L^{-1}$)	0.06 ± 0.05 (0.02–0.20)	0.05 ± 0.04^b (0.02–0.16)	0.12 ± 0.08^{ab} (0.04–0.29)	$p = 0.0132$; $\chi^2 = 8.653$

Appendix C. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Perry's Farm, Sabie 3 (X3H006) Department of Waer and Sanitation (DWS) monitoring station located on the Sabie River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 11)	1988–1998 (n = 11)	1999–2009 (n = 11)	Significance
EC ($\mu S.cm^{-1}$)	102.4 ± 7.99^a (85.18–115)	112.2 ± 11.62^b (92.4–133.6)	180 ± 50.05^{ab} (100.6–241.1)	$p < 0.0001$; $F_{(2,30)} = 21.83$
pH	6.97 ± 0.31^{ab} (6.37–7.35)	7.67 ± 0.37^a (6.72–8.03)	7.79 ± 0.07^b (7.70–7.91)	$p < 0.0001$; $F_{(2,30)} = 27.92$
TDS ($mg.L^{-1}$)	70.21 ± 4.92^a (62.53–79.67)	80.81 ± 10.20^b (64.92–96.70)	113.4 ± 27.52^{ab} (72.57–151.2)	$p < 0.0001$; $F_{(2,30)} = 18.88$
Na ($mg.L^{-1}$)	3.59 ± 0.43^a (3.09–4.51)	3.63 ± 0.7^b (2.13–4.71)	11.22 ± 3.58^{ab} (4.21–16.32)	$p < 0.0001$; $F_{(2,30)} = 47.24$
Mg^+ ($mg.L^{-1}$)	4.84 ± 0.50^a (3.87–5.41)	5.27 ± 0.65^b (4.28–6.36)	7.02 ± 2.31^{ab} (3.77–9.81)	$p = 0.0026$; $F_{(2,30)} = 7.326$
Ca ($mg.L^{-1}$)	7.92 ± 0.61 (6.38–8.69)	8.33 ± 0.9 (7.03–9.72)	8.97 ± 2.21 (4.77–11.67)	$p = 0.268$; $F_{(2,30)} = 1.512$
Cl^- ($mg.L^{-1}$)	4.41 ± 1.74^a (2.38–6.95)	5.29 ± 0.86^b (4.08–6.43)	18.03 ± 6.79^{ab} (5.55–26.32)	$p < 0.0001$; $F_{(2,30)} = 38.41$
NO_3^-/NO_2^- ($mg.L^{-1}$)	0.18 ± 0.08 (0.09–0.32)	0.20 ± 0.06 (0.09–0.32)	0.28 ± 0.15 (0.07–0.53)	$p = 0.0824$; $F_{(2,30)} = 2.716$
SO_4^{2-} ($mg.L^{-1}$)	$^{5.11 \pm 2.08}$ (2.75–9.44)	5.52 ± 1.84 (2.18–8.29)	$^{5.62 \pm 1.31}$ (3.17–7.5)	$^{p} = 0.7725$; $F_{(2,30)} = 0.2603$
PO_4^{3-} ($mg.L^{-1}$)	0.007 ± 0.003^a (0.003–0.01)	0.02 ± 0.01 (0.007–0.04)	0.03 ± 0.03^a (0.008–0.12)	$p = 0.0002$; $\chi^{2=17.48}$
NH_4^+ ($mg.L^{-1}$)	0.04 ± 0.02 (0.02–0.09)	0.04 ± 0.02 (0.01–0.07)	0.05 ± 0.02 (0.03–0.09)	$p = 0.3043$; $F_{(2,30)} = 1.238$

Appendix D. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Karino, Crocodile (X2H006) Department of Water and Sanitation (DWS) monitoring station located on the Crocodile River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 11)	1988–1998 (n = 10)	1999–2009 (n = 12)	Significance
EC ($\mu S.cm^{-1}$)	142.4 ± 23.78^{ab} (111.2–188.3)	181.1 ± 31.25^{ac} (147.3–225)	227 ± 29.83BCE (185.1–275.6)	$p < 0.0001$; $F_{(2,30)} = 25.54$
pH	6.91 ± 0.33^{ab} (6.15–7.3)	7.76 ± 0.33^a (6.86–8)	7.77 ± 0.12b (7.59–7.93)	$p < 0.0001$; $F_{(2,30)} = 35.85$
TDS ($mg.L^{-1}$)	93.71 ± 13.20^{ab} (72.92–121.7)	119.2 ± 20.18^{ac} (95.92–148.6)	43.4 ± 21.10^{bc} (114–173.1)	$p < 0.0001$; $F_{(2,30)} = 20.60$
Na ($mg.L^{-1}$)	5.63 ± 1.20^a (4.23–8.6)	6.88 ± 1.42^b (4.64–9.45)	10.34 ± 2.22^{ab} (7.33–13.21)	$p < 0.0001$; $\chi^2 = 20.27$
Mg^+ ($mg.L^{-1}$)	7.04 ± 1.02^{ab} (5.39–9.27)	8.57 ± 1.4^a (7.13–10.75)	9.30 ± 1.31^b (6.83–11.16)	$p = 0.0006$; $F_{(2,30)} = 9.697$
Ca ($mg.L^{-1}$)	9.76 ± 1.63^{ab} (7.19–12.80)	12.03 ± 2.53^{ac} (9.27–15.40)	$^{14.44 \pm 1.97bc}$ (11.61–17.83)	$p < 0.0001$; $F_{(2,30)} = 14.92$
Cl^- ($mg.L^{-1}$)	5.84 ± 2.35^{ab} (2.95–9.72)	9.61 ± 1.38^{ac} (7.69–11.90)	15.83 ± 3.81^{bc} (10.25–20.62)	$p < 0.0001$; $F_{(2,30)} = 37.96$
NO_3^-/NO_2^-($mg.L^{-1}$)	0.87 ± 0.51 (0.20–1.95)	0.87 ± 0.28 (0.57–1.50)	$^{0.57 \pm 0.18}$ (0.31–0.82)	$p = 0.0816$; $F_{(2,30)} = 2.727$
SO_4^{2-} ($mg.L^{-1}$)	10.59 ± 5.70^a (3.20–22.30)	12.96 ± 4.46^b (7.63–19.33)	19.25 ± 5.00^{ab} (11.71–26.25)	$p = 0.0009$; $F_{(2,30)} = 8.92$
PO_4^{3-} ($mg.L^{-1}$)	0.02 ± 0.02^a (0.005–0.08)	0.03 ± 0.02 (0.02–0.08)	0.05 ± 0.02^a (0.03–0.10)	$p = 0.0009$; $\chi^2 = 13.95$
NH_4^+ ($mg.L^{-1}$)	0.09 ± 0.09 (0.01–0.33)	0.10 ± 0.05 (0.03–0.16)	0.09 ± 0.05 (0.04–0.19)	$p = 0.8967$; $F_{(2,30)} = 0.1094$

Appendix E. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Weltevrede, Crocodile 2 (X2H032) Department of Water and Sanitation (DWS) monitoring station located on the Crocodile River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 11)	1988–1998 (n = 10)	1999–2009 (n = 12)	Significance
EC ($\mu S.cm^{-1}$)	154.8 ± 22.83^{ab} (120.4–203.1)	188.3 ± 30.64^{ac} (155.2–230.8)	238.3 ± 45.68^{bc} (189.5–331.2)	$p < 0.0001$; $F_{(2,30)} = 16.66$
pH	7.02 ± 0.08^{ab} (6.40–7.40)	7.62 ± 0.29^a (6.93–7.97)	7.81 ± 0.09^b (7.67–7.96)	$p < 0.0001$; $F_{(2,30)} = 34.98$
TDS ($mg.L^{-1}$)	102 ± 12.55^{ab} (83.95–134)	97.92 ± 156.6^{ac} (126.6–22.65)	155.7 ± 30.03^{bc} (126.5–220)	$p < 0.0001$; $F_{(2,30)} = 15.48$
Na ($mg.L^{-1}$)	6.84 ± 1.37^a (5–10.27)	8.02 ± 1.70^b (5.09–10.98)	12.21 ± 3.76^{ab} (8.69–21.45)	$p < 0.0001$; $\chi^2 = 18.71$
Mg^+ ($mg.L^{-1}$)	7.43 ± 0.95^a (5.71–9.49)	8.63 ± 1.31 (7.13–10.94)	9.79 ± 2.08^a (7.54–14.01)	$p = 0.004$; $F_{(2,30)} = 6.691$
Ca ($mg.L^{-1}$)	10.47 ± 1.16^{ab} (8.85–13.47)	12.68 ± 2.45^{ac} (9.74–15.79)	15.06 ± 2.27^{bc} (12.73–19.09)	$p < 0.0001$; $F_{(2,30)} = 14.65$
Cl^- ($mg.L^{-1}$)	6.65 ± 2.17^{ab} (3.70–10.45)	9.71 ± 1.45^{ac} (7.83–13.02)	16.35 ± 4.16^{bc} (10.51–24.87)	$p < 0.0001$; F(2,30) = 33.27
NO_3^-/NO_2^-($mg.L^{-1}$)	0.88 ± 0.39 (0.22–1.35)	0.78 ± 0.25 (0.40–1.24)	0.74 ± 0.35 (0.42–1.76)	$p = 0.6144$; $F_{(2,30)} = 0.4948$
SO_4^{2-} ($mg.L^{-1}$)	11.89 ± 7.21^a (3.78–29.91)	13.75 ± 5.72 (7.01–20.93)	19.32 ± 4.61^a (12.69–26.30)	$p = 0.014$; $F_{(2,30)} = 4.937$
PO_4^{3-} ($mg.L^{-1}$)	0.05 ± 0.04 (0.01–0.17)	0.06 ± 0.04 (0.02–0.11)	0.07 ± 0.03 (0.04–0.13)	$p = 0.3855$; $F_{(2,30)} = 0.9842$
NH_4^+ ($mg.L^{-1}$)	0.05 ± 0.03 (0.02–0.10)	0.04 ± 0.01 (0.02–0.07)	0.05 ± 0.02 (0.03–0.09)	$p = 0.6742$; $F_{(2,30)} = 0.3995$

Appendix F. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Dolton, Kaap (X2H022) Department of Water and Sanitation (DWS) monitoring station located on the Kaap River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 11)	1988–1998 (n = 10)	1999–2009 (n = 12)	Significance
EC ($\mu S.cm^{-1}$)	540.7 ± 88.08 (439.8–701.8)	528.8 ± 154.7 (376–771.4)	559.8 ± 122.8 (375.4–743.3)	$p = 0.8386$; $F_{(2,30)} = 0.1771$
pH	7.85 ± 0.31^{ab} (7.45–8.53)	8.10 ± 0.17^{a} (7.68–8.29)	8.30 ± 0.06^{b} (8.20–8.39)	$p < 0.0001$; $F_{(2,30)} = 14.12$
TDS ($mg.L^{-1}$)	421.2 ± 76.92 (329.3–547.4)	417.5 ± 135.7 (273–636.8)	419.4 ± 119.2 (224.7–590.5)	$p = 0.9972$; $F_{(2,30)} = 0.0028$
Na ($mg.L^{-1}$)	47.51 ± 10.14 (35.90–68.91)	38.08 ± 13.81 (21.50–69.33)	41 ± 14.55 (22.98–62.74)	$p = 0.2459$; $F_{(2,30)} = 1.47$
Mg^{+} ($mg.L^{-1}$)	29.59 ± 7.02 (22.62–46.74)	31.05 ± 12.08 (18.20–51.70)	32.68 ± 7.69 (20.89–44.20)	$p = 0.7184$; $F_{(2,30)} = 0.3344$
Ca ($mg.L^{-1}$)	27.47 ± 3.82 (22.75–36.23)	29.81 ± 8.90 (22.60–43.79)	29.41 ± 4.81 (21.35–36.46)	$p = 0.6365$; $F_{(2,30)} = 0.4586$
Cl^{-} ($mg.L^{-1}$)	20.43 ± 4.62 (13.97–27.68)	21.49 ± 6.25 (12.81–31.90)	24.66 ± 6.55 (16.83–34.07)	$p = 0.2147$; $F_{(2,30)} = 1.620$
NO_3^-/NO_2^- ($mg.L^{-1}$)	0.53 ± 0.30 (0.18–1.29)	0.51 ± 0.31 (0.002–1.16)	0.63 ± 0.13 (0.35–0.82)	$p = 0.5065$; $F_{(2,30)} = 0.6945$
SO_4^{2-} ($mg.L^{-1}$)	39.15 ± 12.89 (24.06–71.80)	56.09 ± 26.33 (32.06–109.4)	55.53 ± 14.87 (35.84–76.62)	$p = 0.0678$; $F_{(2,30)} = 2.948$
PO_4^{3-} ($mg.L^{-1}$)	0.02 ± 0.02^{a} (0.004–0.08)	0.02 ± 0.007 (0.01–0.03)	0.03 ± 0.007^{a} (0.02–0.05)	$p = 0.0104$; $\chi^2 = 9.125$
NH_4^+ ($mg.L^{-1}$)	0.05 ± 0.03 (0.01–0.12)	0.05 ± 0.03 (0–0.09)	0.06 ± 0.03 (0.02–0.12)	$p = 0.7632$; $F_{(2,30)} = 0.2727$

Appendix G. Parametric ANOVA and non-parametric Kruskal-Wallis H test results comparing chemical nutrient variables measured over three decades (1977–1987; 1988–1998 and 1999–2009) at Tonga, Komati (X1H003) Department of Water and Sanitation (DWS) monitoring station located on the Komati River system. Data are presented as mean ± SD and range in parentheses. Common superscripts within rows indicate significant differences (p < 0.05)

Variable	1977–1987 (n = 11)	1988–1998 (n = 11)	1999–2009 (n = 11)	Significance
EC ($\mu S.cm^{-1}$)	220.9 ± 58.45^{a} (142.5–328.9)	280.6 ± 95.35^{b} (180.4–429.6)	394 ± 111.1^{ab} (249.5–591.9)	$p = 0.0004$; $F_{(2,30)} = 10.27$
pH	7.21 ± 0.38^{ab} (6.36–7.69)	7.99 ± 0.33^{a} (7.09–8.25)	8.09 ± 0.08^{b} (7.97–8.23)	$p < 0.0001$; $F_{(2,30)} = 29.09$
TDS ($mg.L^{-1}$)	146.1 ± 38.90^{a} (97.71–221.3)	192.1 ± 65.25^{b} (116.1–288.5)	258.3 ± 78.72^{ab} (169.2–410.7)	$p = 0.001$; $F_{(2,30)} = 8.777$
Na ($mg.L^{-1}$)	21.24 ± 8.14^{a} (10.71–37.31)	28.52 ± 13.84 (12.54–50.06)	39.65 ± 14.52^{a} (19.54–67.19)	$p = 0.0062$; $F_{(2,30)} = 6.051$
Mg^{+} ($mg.L^{-1}$)	8.19 ± 1.84^{a} (5.37–10.79)	10.13 ± 2.50^{b} (7.31–14.10)	14.07 ± 4.13^{ab} (9.48–22.94)	$p = 0.0002$; $F_{(2,30)} = 11.09$
Ca ($mg.L^{-1}$)	9.01 ± 1.48^{a} (7.16–11.75)	11.17 ± 2.74^{b} (7.61–15.33)	15.05 ± 3.85^{ab} (10.73–23.47)	$p = 0.0001$; $F_{(2,30)} = 12.59$
Cl^{-} ($mg.L^{-1}$)	24.31 ± 10.77^{a} (9.01–42.95)	32.78 ± 17.01^{b} (10.91–60.99)	49.59 ± 19.18^{ab} (21.09–90.30)	$p = 0.0031$; $F_{(2,30)} = 7.064$
NO_3^-/NO_2^- ($mg.L^{-1}$)	0.08 ± 0.05^{ab} (0.03–0.17)	0.15 ± 0.06^{a} (0.10–0.30)	0.20 ± 0.08^{b} (0.09–0.33)	$p = 0.0004$; $F_{(2,30)} = 10.26$
SO_4^{2-} ($mg.L^{-1}$)	5.72 ± 3.01^{a} (2.08–11.73)	8.53 ± 3.86 (3.33–15.78)	11.75 ± 4.12^{a} (8.10–22.91)	$p = 0.0025$; $F_{(2,30)} = 7.348$
PO_4^{3-} ($mg.L^{-1}$)	0.01 ± 0.006^{a} (0.003–0.03)	0.02 ± 0.007 (0.009–0.03)	0.02 ± 0.006^{a} (0.008–0.03)	$p = 0.0214$; $F_{(2,30)} = 4.383$
NH_4^+ ($mg.L^{-1}$)	0.05 ± 0.03 (0.02–0.11)	0.04 ± 0.01 (0.02–0.06)	0.05 ± 0.02 (0.03–0.09)	$p = 0.4203$; $F_{(2,30)} = 0.2321$

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