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Dye tracing of the Lusaka karstified aquifer system: implications towards urban groundwater quality protection

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Abstract Management of groundwater resources requires a large amount of data, coupled with an understanding of the aquifer system behavior. In developing countries, the scarcity in groundwater data has led to aquifers being managed according to rule-of-thumb standards or even abandoned as unmanageable at times. Groundwater quality protection thus has been through prescribed separation distances often without due regard for internal and boundary characteristics that affect response rates of groundwater movement, attenuation of pollutants, and recharge. In this study, we examine the boundary characteristics of the highly vulnerable karst aquifer system in the rapidly expanding city of Lusaka using a dye tracer technique. We investigate the flow dynamics (magnitude and direction) of groundwater using dye tracer dyes (fluorescein and rhodamine) spiked in pit latrines and observed at discharge springs. The results provide irrefutable evidence that pit latrines are a source and a pathway to contamination of groundwater. Dye tracer movement in groundwater was rapid, estimated at 340 and 430 m/day for fluorescein and rhodamine, respectively, through interconnected conduit density. The vadose zone (epikarst) tends to store diffuse recharge before release to the phreatic zone. These rapid groundwater movements render regulatory separation minimum distances of 30 m between abstraction wells and pit latrines/septic tanks in such environments to be an ineffective means of reducing contamination. The policy focus in the protection of groundwater quality should henceforth be on robust sanitation solutions especially for low-income communities that recognize the socio-economic diversity.

Keywords Dye tracing · Groundwater protection · Karst aquifer · Lusaka · Management · Sanitation

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Introduction

Groundwater is considered a safe and reliable source of drinking water in many parts of Africa (Foster, 2022; Foster et al., 2020, 2022). Adelana et al. (2008) identified 49 cities that are heavily dependent on groundwater resources in Sub-Saharan Africa (SSA) especially for drinking water needs. Within the urban and peri-urban contexts, this has led to widespread unprecedented development of groundwater resources. However, high population densities found in urban and peri-urban areas have also led to the proliferation of unimproved sanitation provision largely through the use of septic

tanks and pit latrines (Ahmed et al., 2001), which are often in very close proximity to wells and springs important for domestic use (Lapworth et al., 2017). This has resulted in the groundwater being contaminated with bacteria and nitrates (Reaver et al., 2020) instigating water resources planning and management challenges.

Urban groundwater quality in SSA has been a subject of study mainly focusing on sources and pathways (e.g., Banda et al., 2019), characterization of contaminants (e.g., Sorensen et al., 2015; Twinomucunguzi et al., 2020), impacts of contamination (e.g. Masindi & Foteinis, 2021), and risk assessment (e.g. Lapworth et al., 2017). These studies have shown that karst aquifers, like the ones that underlie African cities such as Dakar, Lagos, Cape Town, and Lusaka, are very vulnerable to groundwater contamination compared to other rock types. This is due to their ability to transmit large quantities of water and contaminants to specific discharge points without the natural filtration that can occur in other porous media. This rapid movement of water also lessens the amount of time available for physical and biogeochemical reactions (e.g., sorption, ion exchange, degradation) to decrease the concentrations of surface-derived contaminants in the subsurface (e.g., Reberski et al., 2022).

Karst aquifers are generally recognized as the most heterogeneous and complex groundwater flow systems (Ford & Williams, 2007; Goldscheider & Drew, 2007). A whole range of hydrogeological media from karst conduits and corrosion-widened fractures to tiny fissures and intergranular porosity within the karst aquifers cause simultaneous presence of contrasting groundwater flows and permeability (Stroj et al., 2020). This condition allows or precludes enhanced vulnerability to retain and spread the contamination accordingly. A detailed understanding of all karst aquifer components, from soil to conduit networks and fissured rock masses in vadose and phreatic zones, is crucial for successful protection and management of karst water resources. Knowledge on flow dynamics in karst aquifers in relation to groundwater quality is still a subject of research and is still poorly understood in SSA.

In Zambia, groundwater quality has deteriorated in the Lusaka karst aquifer as a direct result of increasing human activity on the vulnerable karst geology (Nkhuwa, 1996; Nyambe & Maseka, 2000; De Waele & Follesa, 2003; Museteka & Bäumle, 2009). Bäumle

and Kang'omba (2012) and Chande and Mayo (2019) showed that the shallow groundwater depth (thin vadose zone) makes the Lusaka karst aquifer vulnerable to bacteria, nutrients, and heavy metal contamination from pit latrines/septic tanks, waste ditches, and industrial areas in urban centers. Liddle et al. (2015) suggested that karst aquifers could become vulnerable to contamination due to their excessive/high permeability and porosity that may permit the rapid ingress of surface water and movement within the subsurface. Karen et al. (2019) showed that incidences of fecal contamination in karst aquifers occurred in boreholes as deep as 40–50 m with no link to rainfall events. We postulate that karst aquifer contamination in urban communities could be through pit latrines and septic tanks influenced by permeability and connectiveness of conduits affecting attenuation. The aim of this study was to use dye tracing to investigate the connectedness of the conduits and flow velocities within the karst hydrogeology of Lusaka. Specifically, we want to establish appropriate dye injection dosage to attain breakthrough, map groundwater flow direction, and estimate groundwater velocity. Based on the finding, the study seeks to propose an approach for groundwater protection to enhance water resource planning and management.

Materials and methods

Study area

The study area is located west of Lusaka City as shown in Fig. 1. The area comprises private farm lands and dotted densely populated unplanned residential area lacking amenities and a reticulation system with a population of 36,532 (Ministry of Health, 2019). The area is characterized by a few boreholes with numerous springs mainly utilized for gardening and domestic use. The community heavily depends on groundwater sources for drinking, accessed through springs, shallow wells, and boreholes.

The climate is mild and sub-tropical (James et al., 2009) with three seasons: a hot dry season from August to October with temperatures between 26 and 38 °C; a rainy season from November to April with temperatures between 27 and 34 °C; a cool dry season from May to August with temperatures from 13 to 20 °C. The area lies in region II of the ecological
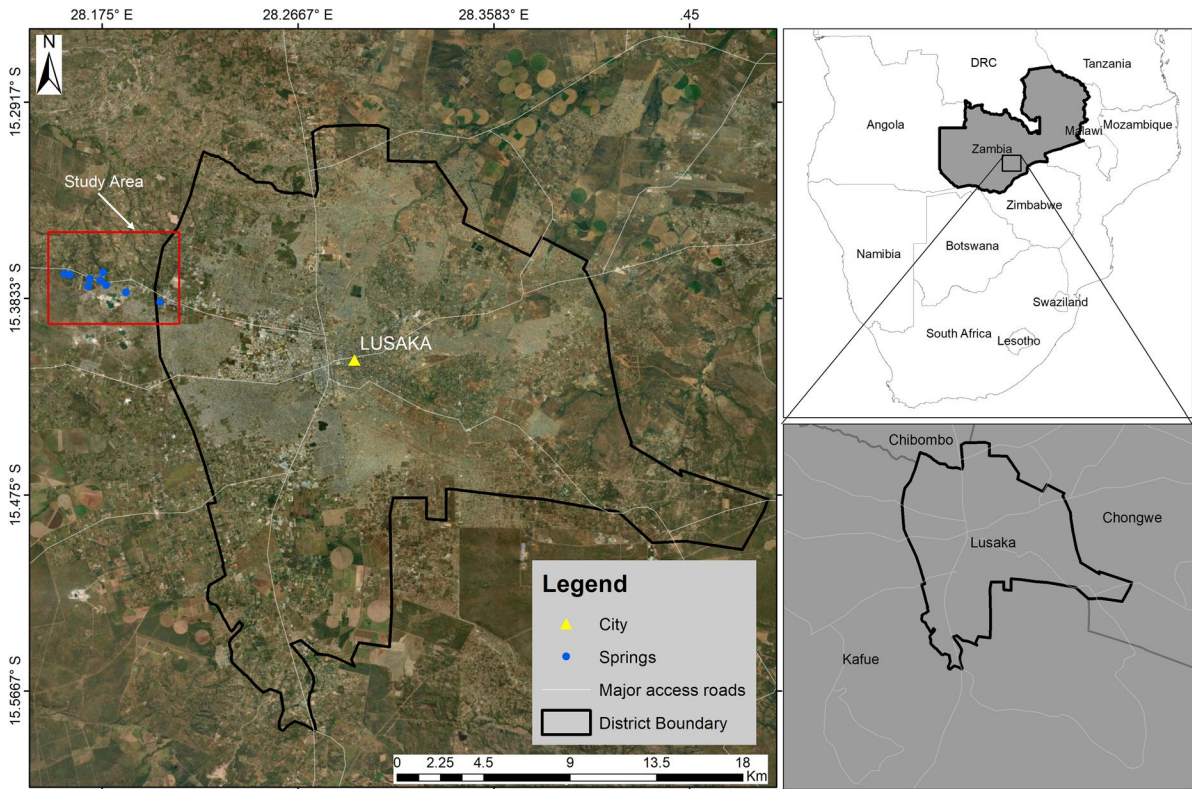
Fig. 1 The location of the study area, Laughing Waters, Lusaka, Zambia

regions of Zambia with average rainfall of between 800 and 1000 mm.

Geology and hydrogeology of the study area

Lusaka lies topographically on a plateau of about 1280 m above sea level (Bäumle & Kang'omba, 2012). It is underlain by a thick sequence of metasedimentary rocks of the Katanga Supergroup (Nkhuwa, 1996). Geologically, the lowest geological unit is the Chunga Formation (Fig. 2) that comprises crystalline rock of low permeability. The Cheta Formation then overlies the Chunga that comprises mainly of schist and quartzite but with some more permeable dolomitic limestone, exploited to a limited extent as a water source (Nyambe & Maseka, 2000). The Lusaka Formation is the uppermost with a dolomitic limestone and marble unit. With its complex interconnective of dissolution conduits that follow joint set planes and sheet fissures, it is the primary aquifer for Lusaka. It has a cutter and pinnacle epikarst surface topography extending to a depth of 25 m (Nkhuwa, 2003). Groundwater abstraction from the Lusaka aquifer has led to a progressive decline in the water table elevation and is suggested to be overexploited (Mpamba et al., 2008). It has generally shallow water table depths from 6 to 15 m.

Direct recharge is predominantly from the Lusaka Forest Reserve, southwest of Lusaka, an area undergoing rapid deforestation and settlement development. Substantial localized recharge has also been reported from sources such as overflowing sewers, adding to the public health risk (Karen et al., 2019). Recharge to the groundwater during the wet season from November to April averages 186 mm/year, which is approximately 27% of the annual rainfall (De Waele & Follesa, 2003; Nyambe & Maseka, 2000). Groundwater flow direction is generally southeast
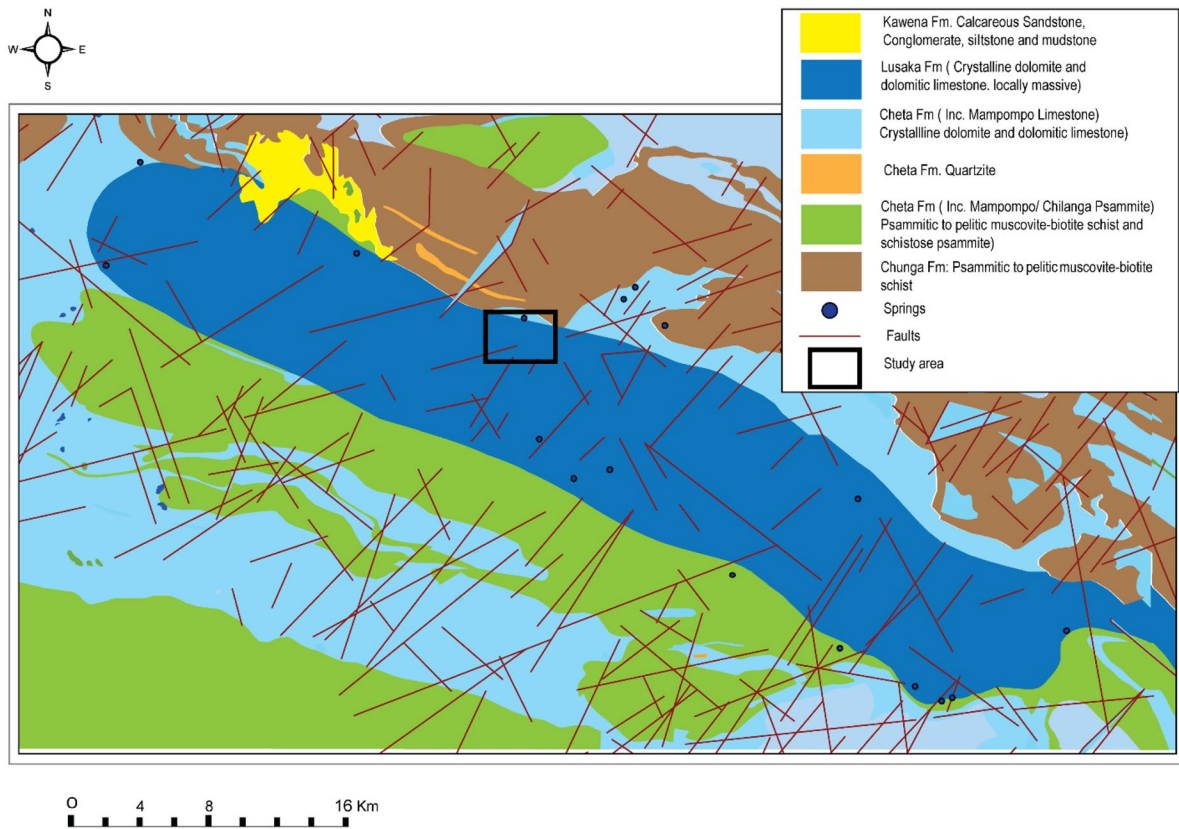
Fig. 2 Generalized geology of the Lusaka area (Bäumle & Kang'omba, 2012) and the fault distribution based on Landsat imagery, 1972

to northwest through the Lusaka area but tends to be in northeast and southwest orientation in some parts (Museteka & Bäumle, 2009; Mpamba et al., 2008; Von Hoyer et al., 1978). These general, large-scale flow directions, however, are not definitive in a karstic system and thus require detailed assessment as they are based on limited groundwater observations. Groundwater discharge is on the schist–dolomite contact through intermittent springs and in low-lying swampy areas (Museteka & Bäumle, 2009). Using the underlying geology, soil type, and soil thickness, Bäumle and Kang'omba (2013) classified groundwater contamination vulnerability classes for the Lusaka area and its environs. Most of the area associated with the Lusaka Formation (Fig. 2) was designated as extremely vulnerable to groundwater pollution. Of great concern is that pit latrines and drinking water sources may be directly connected via highly conductive karst conduits, a situation which can lead to

bacterial and nitrate contamination and the spread of waterborne diseases (Reaver et al., 2021).

Dye tracing

Field setup

A reconnaissance survey was undertaken to identify potential dye-injection points (pit latrines) and observation points within the study area. The selection of the dosing points was done assuming that the springs (Fig. 3) represent focused discharge points for the groundwater therefore dosing points should be placed upstream (Benischke, 2021). The general methodological approach undertaken was qualitative over a 10-week maximum period. The observation in the dry season was done in between the third week of September and second week of November 2018, while the observations for wet season was between second
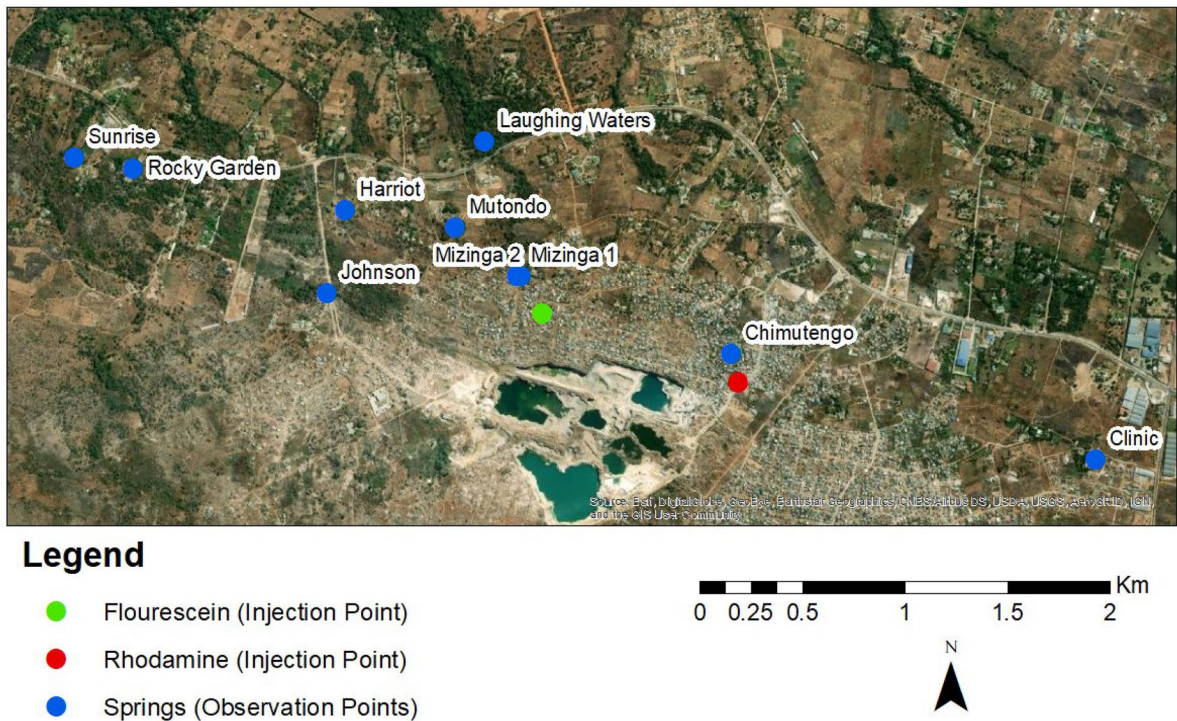
Fig. 3 Location of trace dosing points and recovery springs in the study area.

week of March and second week of April 2019. Dye tracing was done using fluorescein and rhodamine. Fluorescein is a reddish-brown powder that turns vivid yellow-green in water. It is photochemically unstable and its concentration decreases at a pH of less than 5.5 (Mull et al., 1988). However, fluorescein has moderate to high detection, and generally has a low sorptive tendency and diffusity (Benischke, 2021). Rhodamine, on the other hand, is highly detectable with a low sorptive and diffusivity capacity. It turns vivid orange-pink when in water and is photochemically stable. Dye-recovery sites (springs) were selected based on the assumption that they will intersect groundwater flow from the injection points (pit latrines). Furthermore, we also considered if the spring was perennial and accessible throughout the year.

Dye receptors were placed in all of the area springs. They were first placed in the springs and sampled before any dye injection to establish background dye signals. The receptors were made of wire mesh envelopes measuring about 8×10 cm containing two tablespoons of activated charcoal (Fig. 4). Sampling was done in both the wet and dry season. The quantity of dye used in the injection into pit latrines (Fig. 5) was estimated based on the estimated flow conditions, dilution, and distance from the injection point to the most remote recovery point. A study conducted by Field (2003) has recommended several formulae for estimating quantities of injection. However, these formulae are based on a number of parameters such as average discharge, distance between injection and sampling place, expected time of the first detection and the peak of a dye breakthrough, desired peak concentration, specific coefficients for the type of tracer, and other parameters. Furthermore, these formulas were developed by the authors from experience and no one can cover all the various actual conditions during a test.

In this study, we used the approach recommended by Benischke (2021) summarized as "as much as necessary, as little as possible." It was judiciously decided that 300 mg fluorescein and 958 mg rhodamine would be dosed in the pit latrines.

The initial charcoal receptors were deployed to measure background data and presence of compounds

Fig. 4 A Activated coconut charcoal with strong affinity for fluorescence. **B** Receptor containing activated charcoal made of wire mesh used in the study.
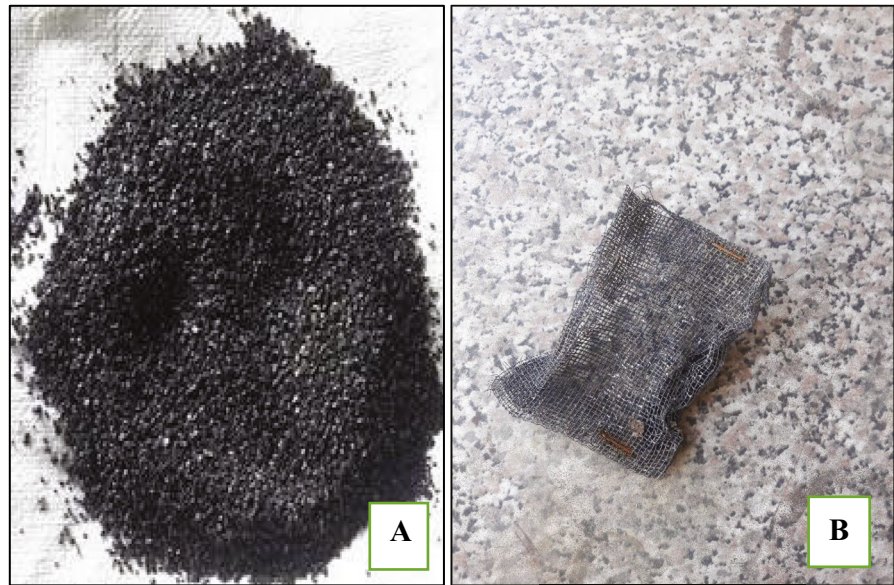
that fluoresce at similar wavelengths as rhodamine and fluorescein. These background receptors were left in place for 4 days prior to any dye injection. This was done to detect any presence of dyes and their concentrations. New receptors were replaced at the recovery points just after dye injection to detect the dye breakthroughs and taken to the laboratory for measurement and replaced with new receptors every 4 days.

Laboratory analysis

Once collected, the receptors were kept in clean Zip-loc® plastic bags which were labeled with the time collected, site number, date of placement, and

collection. The receptors were kept in a cool place out of the light. An AquaFluor® handheld fluorimeter was used to analyze the samples. This instrument has the capacity to measure both rhodamine and fluorescein by switching channels. The minimum detection ranges for fluorescein and rhodamine was 0.4 ppb with linear range of 1–400 ppb. Before analysis, the AquaFluor® fluorimeter was calibrated using blank samples and known standards of the dyes. An eluent composed of propanol (5 parts), distilled water (3 parts), and ammonia hydroxide (2 parts) was prepared as described by Mull et al. (1988). On collection of receptors, 5 g of the charcoal was removed from the mesh and placed in a beaker. Then, 10 mL of the

Fig. 5 A Dosing of fluorescein and **B** dosing of rhodamine in pit latrines upstream of observation springs

eluent was poured into the beaker to immerse the charcoal. The elution was then allowed to stand for 1 h after which the eluent was poured into a cuvette for analysis. This was repeated three times to ensure replication of results for quality assurance. A detailed description of the method can be found in Currens (2013). The tracer concentration results were then used to plot breakthrough curves to evaluate tracer transport in the karst aquifer system.

To calculate the average travel time using the measured concentrations per recovery site, it can be determined using two methods, namely, first by calculating the concentration against the distance from the dosing point. Second, a formula is employed to calculate time of travel using concentrations (Eq. 1) (Benischke, 2021; Chapra, 2008) as below:

$$t = \frac{\sum_{i=0}^{n-1} \left(C_i t_i + C_{i+1} t_{i+1} \right)_{(t_{i+1}-t_i)}}{\sum_{i=0}^{n-1} \left(C_i + C_{i+1} \right)_{(t_{i+1}-t_i)}} \qquad (1)$$

where t = average travel time, C_i = concentration (mg/L), t_i = discrete time when sampled (min), n = number of weeks, and I = batch numbers.

Average groundwater velocity was then calculated by dividing the distance from the dosing pit latrine to the recovery points with the average time.

Results

The breakthrough curves of all the observation springs are shown in Fig. 6a, b. Fluorescein concentrations observed were generally much higher than rhodamine. Given that observed background concentrations for fluorescein and rhodamine were 150 and 26 mg/L, respectively, springs such as the Clinic, Laughing Waters, and Rocky Garden, which have predominately background concentration values, can thus be considered as no breakthroughs observed. Sunrise spring recorded a maximum peak concentration in the second week of observations and was the first spring to have above background concentrations. Simultaneous maximum peak values were observed at Harriot, Chimutengo, Mizinga 1, and Mizinga 2 springs after 4 weeks as shown in Fig. 6. Mutondo Spring peaks after 4 weeks, likely because the tracer transits through Mizinga 1 and Mizinga 2 before it appears. Some of the breakthrough curves appear to

have a double peaking (such as Chimutengo and Mizinga 1, with the short distance from injection points) probably due to the tracer being stored in the vadose zone (epikarst) before release to the phreatic zone.

The average travel time results using Eq. (1) for both tracers range between 20 and 30 days (Fig. 7). The travel times in Fig. 7 are significantly different with a P value of 0.338 ($P < 0.500$) between the groups. The arrival time indicates that rapid transmittance of the tracer likely by radial flow provided the simultaneous arrival.

In the wet season, the breakthrough response was significantly diluted arising from diffuse rainfall recharge response. Based on the dry season breakthrough curves, peak concentration of 790.8 mg/L of fluorescein and 179 mg/L rhodamine at Sunrise Spring (furthest western observation point) were observed just 8 days after dosing. However, the eastern furthest spring at the local Clinic (Fig. 3) peaked twice at 4 and 9.5 weeks at maximum peak of 106 mg/L for fluorescein and 28 mg/L of rhodamine. The breakthroughs at Harriot and Mutondo springs were spread out peaking between 4 and 5 weeks (Fig. 6) indicating a more diffuse flow. Using maximum concentrations observed of the tracers (790.8 mg/L fluorescein and 293.1 mg/L rhodamine), groundwater velocities were calculated to be 340 and 430 m/day, respectively.

Discussion

The results show that the breakthrough curves had multiple variations in the concentration peaks. The shapes and magnitudes of curves are influenced by the amount of dye injected, velocity of flow, mixing characteristics within the flow system, and discharge diluted by non-dyed waters (Cunningham et al., 2004; Mull et al., 1988; Quinlan, 1982). The fluorescein and rhodamine dyes, which both have adsorptive behavior to mimic contaminant movement (Sabatini & Austin, 1991), indicated a breakthrough within 2 to 4 weeks of tracing at Sunrise Spring 3.3 km away from the dosing points followed by Rocky Garden, Harriot, Mutondo, Laughing Waters, Johnson, and Clinic. Although Laughing Waters is nearer, the spring had earlier dried up in the dry season. Both dyes were detected in considerable concentrations that were not visible to the eye. The fluorescein breakthrough at Sunrise Spring arrives the quickest and with the highest peak indicating relatively
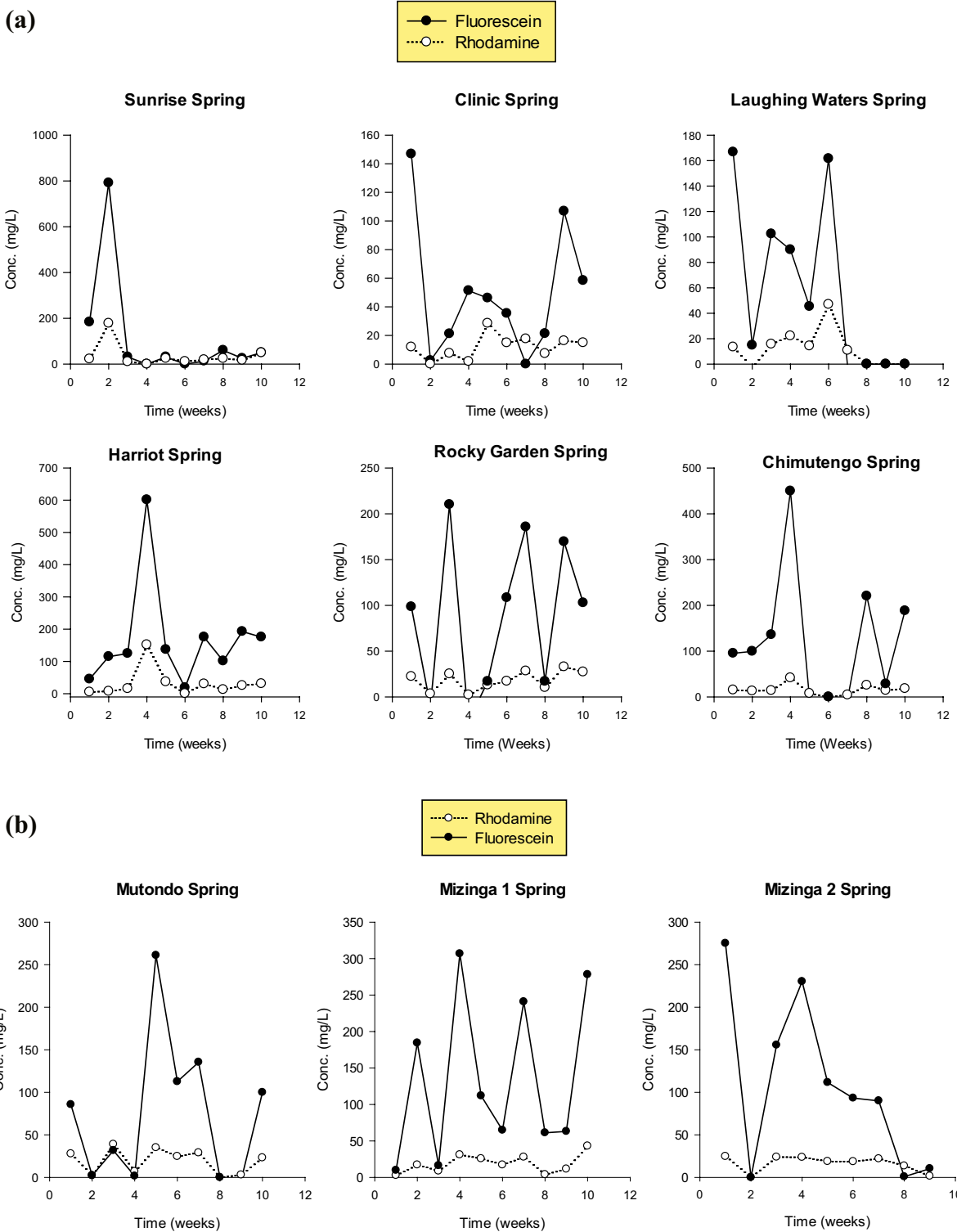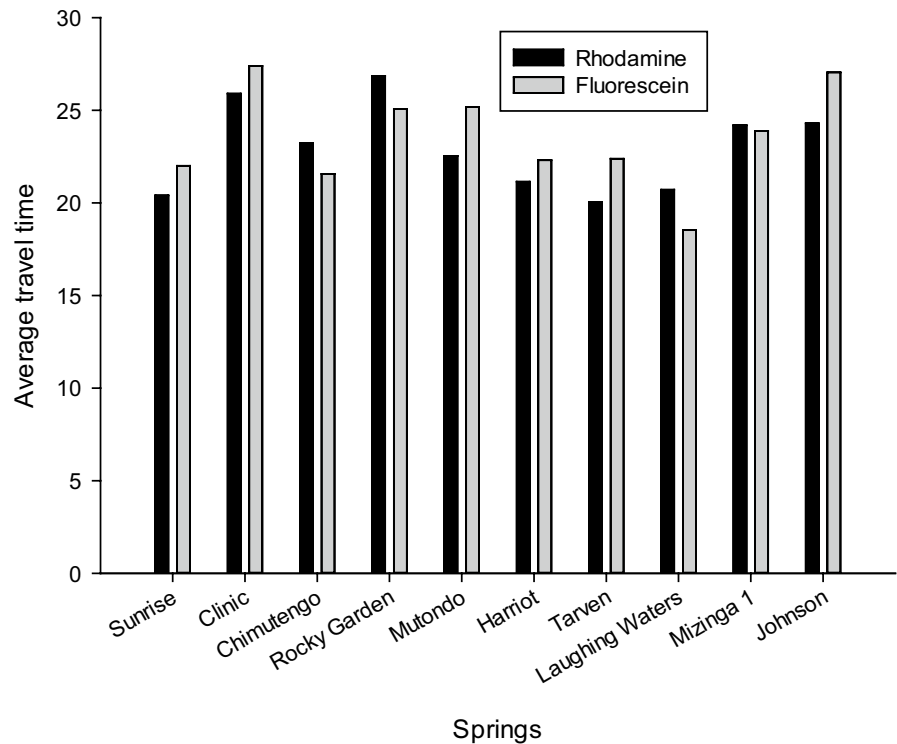
Fig. 6 **a**, **b** The dye breakthroughs at the observation springs shown in Fig. 3

Fig. 7 The average calculated travel time during the dry season for rhodamine and fluorescein at the observation springs in Laughing Waters area, Lusaka, Zambia


rapid advective transport. This is likely due to anthropogenic activities around the area, specifically the active quarry mining where large volumes of water are abstracted, diverting the flow. The shape of curve is skewed to the left before which concentrations reduced to background values of which we interpreted as incomplete flush-out of the dye, bearing in mind that samples collected were discrete water sample measurements rather than dye receptor which integrates over the time the receptor was in the water. This incomplete flush-out of the tracer demonstrates a rapid response to large flow volumes (focused channelized flow) and fast velocity of this aquifer system (Green et al., 2006). Mean groundwater velocity was estimated to be 340 and 430 m/day using fluorescein and rhodamine maximum breakthrough curves. This agrees with tracer test results on the Walkerton karst aquifer which had karst velocities from 320 to 480 m/day (Green et al., 2006). Dye loss may be due to multiple flow paths, entrapment of the dye within the aquifer, and sorption of the dye by rock/soil (Benischke, 2021).

The flow direction of the groundwater was generally in the south-east to north-west direction, which is similar to the preferential orientation of geological structural lineaments (Simpson et al., 1963; Nkhuwa, 1996, Baumle & Kang'omba, 2009). Figure 8 shows a rose diagram of lineaments from the Lusaka dolomite and river channels, demonstrating a preferential north-west to south-east orientation. These lineaments are a result of tectonic banding which were likely pathways used to facilitate solution weathering and thus karstication (Baumle & Kang'omba, 2009). The dye tracer given breakthroughs in the various breakthroughs suggests that groundwater flows in multiple directions.

A further examination of the flow path based on arrival time of the tracers shows a complex movement of groundwater (Fig. 9). This flow path was probably due to the complex fracture/conduit connective in the subsurface. Nkhuwa (1996) suggests that intensely folded carbonate rocks of the Lusaka Dolomite have suffered extreme differential dissolution, resulting in the development of a system of subterranean conduits and solution channels. The fracture density shown in Fig. 2 was thus even more intense than what was mapped using Landsat imagery.

Fig. 8 Preferential north-west to south-east orientation of lineaments from the Lusaka aquifer extracted from Landsat imagery and river channels modified from Baumle and Kang'omba (2009)
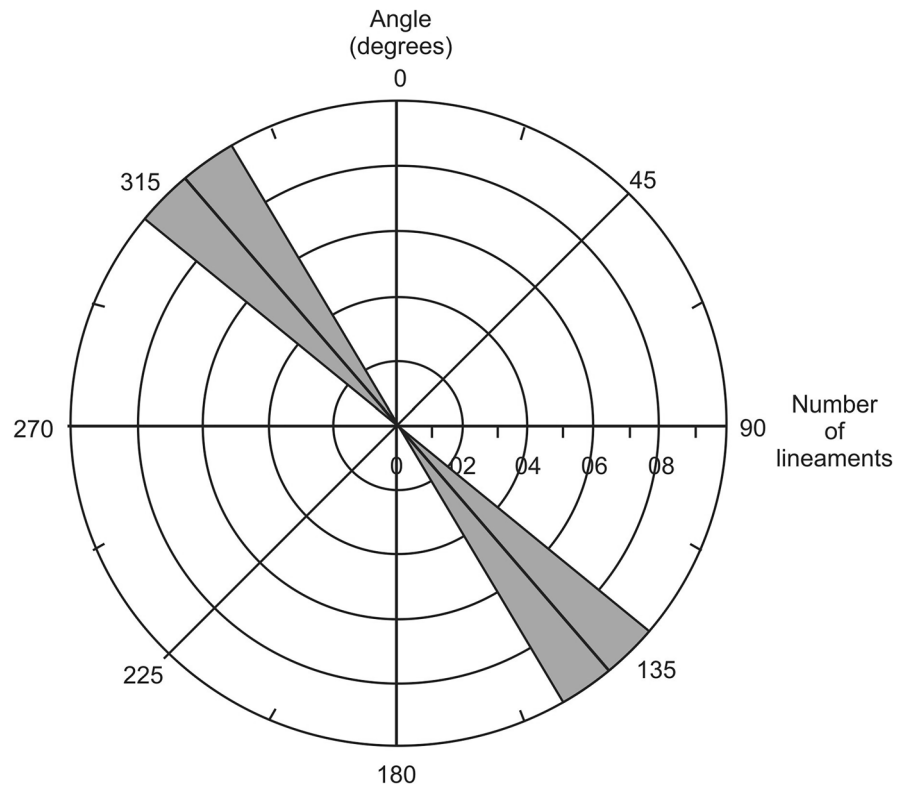
Implications for groundwater quality protection

The vulnerability of karst aquifer systems is exacerbated in rapidly developing cities such as Lusaka through informal settlements with little to no access to sewage reticulation (Kihila & Balengayabo, 2020). Pit latrines and septic tanks have been promoted to support improved sanitation to the peri-urban and new urban settlement communities; however, their increased number and usually poor construction and maintenance result in increased groundwater contamination (Nyenje et al., 2013; Templeton et al., 2015; Gaye & Tindimugaya, 2019; Foster, 2022). Urban groundwater quality protection must aim to prevent the mixing of waste water with the fresh portable groundwater. One immediate management approach to the prevention of well head contamination, typically attributed to poor drilling practices and a lack of technical supervison of borehole installation, would be the construction of a sanitary seal according to the local geology conditions (Takavada et al., 2022). Poor construction of the sanitary seal effectively leads to contamination by recently recharged groundwater mixing with the contamination plume (pathogens) in

the near-surface aquifer (Banks et al., 2021; Ferrer et al., 2020; Hynds et al., 2013). Based on the foregoing, effective grouting and sealing off of the shallow horizons in the design of borehole is therefore critical.

Existing and planned boreholes may be at risk of contamination if sufficient stand-off distance between the sanitation unit and borehole is not provided for. However, there is no international consensus on the stand-off distance (Graham & Polizzotto, 2013; Rivett et al., 2022); furthermore, uncertainties arise from the hydrogeological circumstance and the type of pathogen (bacteria, virus, and protozoa) compounded with climate change. In Zambia, to protect the groundwater supply in karst aquifers, borehole regulations were developed which prescribed a minimum distance of 30 m as stand-off distance. Based on the groundwater velocities observed in this study, groundwater contaminants can travel 30 m in as little as 0.088 days (~2 h). The prescribed minimum distance therefore provides very little lead-time, and due to economic, social, and technical conditions, the reality is that it is impractical to implement (Templeton et al., 2015). Consequently, these karst aquifers are still vulnerable as the threat
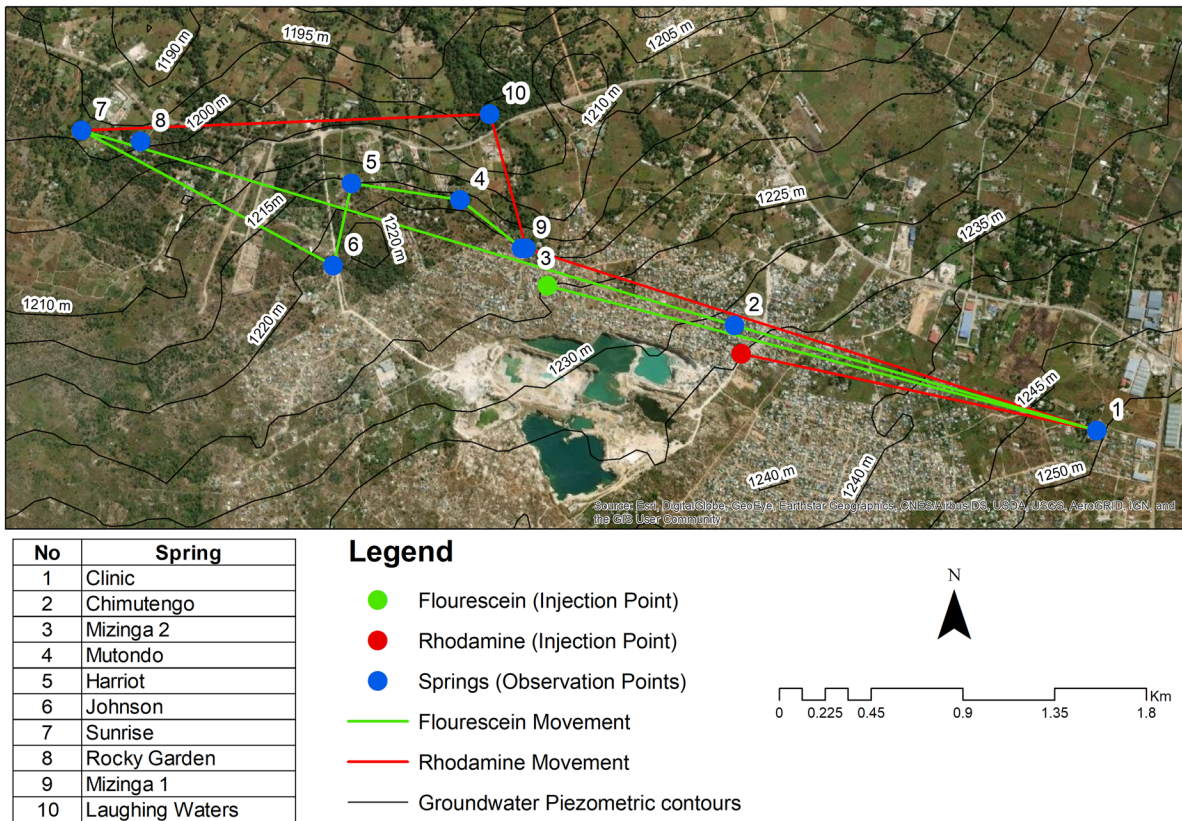
Fig. 9 The springs (recovery points) and pathways for rhodamine and fluorescein dyes depicting a complex flow

to contamination can still extend to distant areas. The problem is exacerbated by poor groundwater monitoring networks and the fact that many settlements are highly populated with housing units in close proximity. These challenges are not unique to Zambia but common in lower-income countries such as Uganda, Ghana, and Tanzania (Twinomucunguzi et al., 2020). There was, therefore, a need for more adaptive approaches that recognize constraints in such settings.

In most hydrogeological environments, some layering or stratification is present and this normally results in significant differences between the horizontal and vertical hydraulic conductivity (ratio of K_v/K_h varies from 1 to 0.01 or less). This, in turn, can lead to long travel times for groundwater to flow downwards from the surface of the water table to depth. Lawrence et al. (2001) proposed Eq. (2) for the calculation of travel time:

$$t = 2.984 \frac{n}{Q} \left(d \sqrt{\frac{K_h}{K_v}} \right)^3 \qquad (2)$$

where t = travel time from water table to well intake, n = kinematic porosity, K_h/K_v = ratio of horizontal to vertical hydraulic conductivity, d = depth of well intake, and Q = pumping rate.

Based on Eq. (2), the vertical separation between the water table and the screen of a borehole can provide an important time delay allowing significant reduction in the risk of microorganisms arriving at the screen. The depth to the water intake (screens) must therefore be considered in the construction and design.

Furthermore, in situ sanitation units (such as latrines, cesspits, and septic tanks) can provide an adequate service level for excreta disposal under favorable conditions, but also always exert a major influence on

groundwater recharge and quality (Foster, 1999; Foster et al., 2022). Groundwater pollution can also be reduced by a modification of sanitation units by retrofitting or deploying dry (eco-sanitation) units, in which urine is separated from feces, with both being recycled. Retro installations might be expensive and perhaps this might be more feasible for new installations. Frequent emptying of sanitation units and their sitting in places with deeper groundwater level must be encouraged. Innovations must be targeted at creating demand for sanitation waste. Value-added products such as pellets for fertilizer or manure will create opportunities for the low-income population. These interventions will require an enabling environment with standards and guidelines supported by appropriate legislative frameworks (Dasgupta et al., 2021).

Conclusion

The following are conclusions of this study:

1. The Lusaka aquifer tends to store diffuse recharge in the vadose zone (epikarst) before release to the phreatic zone;
2. Groundwater is rapidly transported through complex conduits (karsts) that are well connected with average travel time estimated at 340 and 430 m/day based on fluorescein and rhodamine maxima breakthrough curves, respectively; and
3. Groundwater flow direction was in multiple directions in a radial manner. This is facilitated by the complex geology.

The study recommends that urban groundwater protection in vulnerable aquifer systems of developing countries requires innovative approaches that require value addition to waste products especially from sanitation in the long term. Other interventions would include effective grouting and sealing off shallow aquifer horizons, increased depth between the groundwater intake and water, and modification in the design of in situ sanitation units such as dry (eco-sanitation) units.

Availability of data and material   The datasets and material generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declarations

Conflict of interest   The authors declare no competing interests.

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