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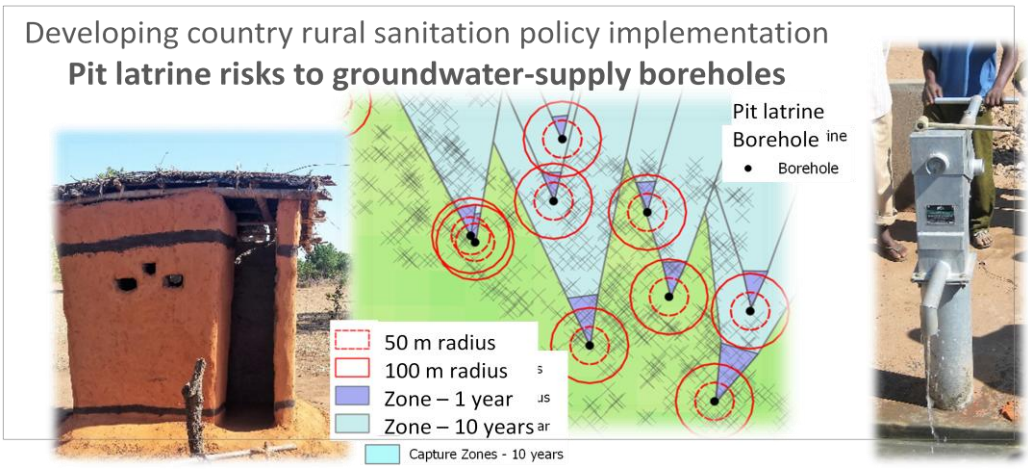
Graphical Abstract



Highlights

- Global rise in both pit latrine sanitation and groundwater supply poses concern
- Groundwater risk due to typical developing-country pit latrine sanitation policies
- Developed risk assessment framework approach pragmatic to regulatory management
- Significance of establishing baseline groundwater quality data shown; a global need
- Low Malawian contamination to date: attenuation effectiveness or emergent problem?

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Risk assessment to groundwater of pit latrine rural sanitation policy in developing country settings

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Abstract

Parallel global rise in pit-latrine sanitation and groundwater-supply provision is of concern due to the frequent spatial proximity of these activities. Study of such an area in Malawi has allowed understanding of risks posed to groundwater from the recent implementation of a typical developing-country pit-latrine sanitation policy to be gained. This has assisted the development of a risk-assessment framework approach pragmatic to regulatory-practitioner management of this issue. The framework involves water-supply and pit-latrine mapping, monitoring of key groundwater contamination indicators and surveys of possible environmental site-condition factors and culminates in an integrated statistical evaluation of these datasets to identify the significant factors controlling risks posed. Our approach usefully establishes groundwater-quality baseline conditions of

a potentially emergent issue for the study area. Such baselines are foundational to future trend discernment and contaminant natural attenuation verification critical to policies globally. Attribution of borehole contamination to pit-latrine loading should involve, as illustrated, the use of the range of contamination (chemical, microbiological) tracers available recognising none are ideal and several radial and capture-zone metrics that together may provide a weight of evidence. Elevated, albeit low-concentration, nitrate correlated with some radial metrics and was tentatively suggestive of emerging latrine influences. Longer term monitoring is, however, necessary to verify that the commonly observed latrine-borehole separation distances (29-58 m), alongside statutory guidelines, do not constitute significant risk. Borehole contamination was limited and correlation with various environmental-site condition factors also limited. This was potentially ascribed to effectiveness of attenuation to date, monitoring of an emergent problem yet to manifest, or else contamination from other sources. High borehole usage and protective wall absence correlated with observed microbiological contamination incidence, but could relate to increased human/animal activity close to these poorly protected boreholes. Additional to factors assessed, a groundwater-vulnerability factor is recommended that critically relies upon improved proactive securing of underpinning data during borehole/latrine installations. On-going concerns are wide ranging, including poorly constrained pit-latrine input, difficulties in assessing in-situ plume natural attenuation and possible disposal of used motor oils to latrines.

Keywords: pit latrine; risk; groundwater quality; sanitation; Millennium Development Goals; Malawi

2. Introduction

A poorly understood threat to the chemical and microbiological quality of groundwater supplies in developing countries is the risk posed by the dramatically increased use of pit latrines for improved sanitation (Graham and Polizzotto, 2013). In response to the Millennium Development Goal on sanitation which targeted improved access levels by 2015 (UN, 2015a), the number of pit latrines is rising globally as populations gain access to improved sanitation under a plethora of water sanitation and hygiene (WASH) programmes (Jain, 2011; UNICEF – WHO, 2015). Pit latrines are the most common low-cost sanitation solution in developing countries and are used by an estimated 1.77 billion people (Graham and Polizzotto, 2013). Sanitation policies in rural areas, alongside some rapidly growing peri-urban areas, are primarily pit latrine based. Such policies may allow districts to cost effectively reach, much sought after, open defecation free (ODF) status and lower exposure risks to faecally-derived, acutely toxic, microorganisms (Cho et al., 2016).

Alongside improved sanitation, improved access to drinking water is also rising globally under WASH programmes. The recently developed Sustainable Development Goals (SDGs) unifies sanitation and sustainable water management under SDG 6 (UN, 2015b). Improved access to water invariably relies upon groundwater sources (Rosa and Clasen 2010). Hence, this twin growth is potentially of concern where groundwater use and pit latrine disposal are located in close proximity. The absence of a physical barrier between latrine-stored excreta and the underlying soil and groundwater (van Ryneveld and Fourie 1997), and the fact that abstracted groundwater is often untreated and infrequently monitored prior to drinking add credence to this concern. There is a pressing need to better understand the connectivity between latrine sources and groundwater supply points and health risks posed at typical rural development scales (BGS, 2002; Bain et al., 2014; Graham and Polizzotto, 2013).

Pit latrine faecal sludge, although produced at low volumetric rates of around 1.5 litre per capita per day, contains not only microbiological pathogens of human-health concern (Bain et al., 2014; Cho et al., 2016; Pedley et al., 2006), but also elevated nitrogenous and carbonaceous organic matter that is persistent due to the confined nature of pit latrines. Pit latrines largely hold, rather than treat the disposed mass (Coetzee et al., 2011). Some mass loss as liquid leachate infiltration is nevertheless expected to occur and enter the underlying soil and groundwater; this forms the migration pathway of concern herein (BGS, 2002). To reduce risks posed, guidelines exist for the minimum separation distance between latrines and groundwater supply points. However, these vary internationally from around 10 to 75 m. National statutory guidelines may not be set (Parker and Carlier, 2009; Section 3.4.2) and, when set, can sometimes be ignored or inadequately regulated. Good underpinning case data can also be sparse and often lack the high spatial resolution ideally required. Furthermore, the associated process-based science is challenging to undertake, has perhaps become dated, or lacks the nuance detail required (Banerjee, 2011; Caldwell and Parr, 1937; Franceys et al., 1992; Graham and Polizzotto, 2013, Howard et al., 2003; Still and Nash, 2002; WaterAid, 2013).

Various African studies have examined soil – groundwater contamination ascribed to pit latrine disposal. These include Verheyen et al. (2009) in Benin, Jacks et al. (1999), Lewis et al. (1980) and Mafa and Vogel (2004) in Botswana, Mzuga et al. (2001) and Okotto-Okotto et al. (2015) in Kenya, Tandia et al. (1999) in Senegal, Still and Nash (2002) and Vinger et al. (2012) in South Africa, Howard et al. (2003) and Nyenje et al. (2014) in Uganda, Chidavaenzi et al. (2000), Dzwairo et al. (2006) and Zingoni et al. (2005) in Zimbabwe and Palamuleni (2002) in Southern Malawi, specifically peri-urban Blantyre. Groundwater contamination - typically discerned from increased total/faecal coliforms, nitrogen species (nitrate, ammonium), chloride and occasionally virus detections when analysed –

appears to largely remain quite close to latrine pits. Distances appear to be typically around 5 to 50 m or so, though it is recognised that case studies may lack spatial resolution to allow confident assessment of distances and discernment of attenuation processes that may limit migration (Banks et al., 2002; Escamilla et al., 2013; Graham and Polizzotto, 2013; Howard et al., 2003; Nichols et al., 1983; Nyenje et al., 2014; Schijven and Hassanizadeh, 2000; Tandia et al., 1999; Wright et al., 2013). Graham and Pilizzotto (2013) conclude from their review that the number of field studies investigating links between groundwater pollution and pit latrine contamination is limited and advocate the need for improved measurement approaches, development of better criteria for locating pit latrines and the examination of a larger set of contextual variables.

Our goal is hence to further the understanding of risks posed to groundwater by pit latrine sanitation policies implemented in a typical developing country, rural, settings. From this position, we aim to develop and demonstrate a pragmatic risk assessment framework approach that may provide for practitioner (regulatory) management of this issue. This has been achieved through study of the Mwanza Valley in Southern Malawi, where development of both groundwater supply and pit latrine sanitation provision has occurred over recent decades and continues apace (Back, 2015; Hinz, 2015; Mackay, 2015). Specific aims were:

- to investigate the potential contamination of supply boreholes from pit latrines within an area subject to continued and recent development of pit latrine and supply borehole infrastructure;
- recognising the study area represents a relatively young problem scenario for the most part, to assess whether the collected data constitute a reasonable baseline against which future influences may be monitored;
- to evaluate the contributing factors to supply contamination incidence, including the statistical evaluation of contextual parameters such as surrounding pit latrine density, borehole infrastructure condition and modelled borehole - groundwater capture characteristics;
- to discuss future contamination risks and safeguard-monitoring recommendations required within the context of growing populations and increased access to simple sanitation systems.

Our developed multi-faceted approach demonstrated involves mapping of supply borehole water points and pit latrine occurrences, questionnaire surveys of water points to allow data collection on local site – environmental conditions and hence assessment of controlling factors, borehole sampling for chemical and microbiological water quality, and quantitative - GIS - statistical data analysis. The latter involved an empirical risk assessment to determine factors significant in controlling latrine risks to water supplies. The approach seeks to be relevant to practitioner (e.g., regulatory body) adoption in developing country settings.

3. Study setting and methods

3.1. Study setting

The Republic of Malawi is landlocked between Tanzania, Zambia and Mozambique. Its resources are under great pressure from a population of 16.83 million growing by 2.8 % per annum (World Bank, 2015) with around 85 % of the people living in rural areas. The semi-arid lowlands of the Chikwawa District in Southern Malawi studied (Fig. 1) are prone to flood and drought conditions that may lead to crop failures and famine conditions. This setting is particularly vulnerable to climate-change influence. Over 80% of Malawi's annual rainfall, some highly intense, occurs between November and April with variation from 700 mm in low-lying parts of semi-arid Southern Malawi to 2,500 mm in highland areas (Ngongondo et al., 2011). Evaporation is elevated in the former (pan evaporation *c*. 1900 mm per annum) due to high monthly average temperatures of 21–30°C for example in the lower Shire valley studied (BGS, 2004).

Malawi is a greatly impoverished nation with over 50 % of its population living below the national poverty line. It currently ranks 174 out of 187 in the 2013 Human Development Index, classifying it as a low-income country (World Food Program, 2015). Most within the predominantly rural Chikwawa District are subsistence farmers, living on less than $0.50 a day with a mean life expectancy of 45 years (Water for People, 2017). Our study was conducted within the Chapananga Traditional Authority (TA) area of Southern Malawi with research undertaken in 2015 at three nested spatial scales: borehole occurrence data were collected and evaluated for the Mwanza River Valley occurring within Chapananga (most of the valley, n = 340); where available both borehole and pit latrine occurrence data were recorded (n = 189); and collection of similar occurrence data alongside borehole groundwater quality sampling (n = 91) were undertaken in the surroundings of Kakoma Health Area jurisdiction sub-area (Mackay, 2015) (Fig. 1 and see Section 3.3.6 for dataset detail).

The topographic relief, main rivers (incl. 2015 flood extent) and underlying geology (Habgood, 1963; Castaing, 1991) are shown in Fig. 1. The Mwanza River occurs within a down-thrown trough that has accumulated a succession of alluvial and colluvial sediments deposited from annual flooding, alongside erosion from exposed weathered Precambrian gneiss bedrock on the valley-side escarpments to the east and the Karoo sedimentary rocks (inter-bedded sandstones, shales, marls) of Permo-Triassic age that outcrop to the west of the valley. The Mwanza valley forms a discrete narrow feature on the western margins of the extensive Lower Shire valley – Chikwawa District

alluvial plain aquifer system that drains towards the Shire River that flows along the eastern margin of the main valley – alluvial aquifer. The Mwanza River periodically ceases to flow in the dry season. During the wet season, however, low relief leads to problematic flooding (Fig. 1). Groundwater head data indicates flow through the sand-rich sediments flows more or less toward and along the direction of river flow (NW to SE) down the escarpment-constrained valley (Monjerezi et al., 2011; and see later figure). Our study focuses upon boreholes occurring in the main Mwanza valley area, predominantly within the alluvial aquifer and some of the Karoo sedimentary units with some minor borehole encroachment into the adjoining and underlying gneiss bedrock (Fig. 1).

3.2. Supply borehole context and survey data acquisition

Chapananga, in common with much of rural Malawi, primarily uses groundwater for water supply with resource development on-going through NGO (non-governmental organisation) – government (Ministry/District) facilitated WASH – drilling programmes. Just prior to our study, an additional 14 village water supply boreholes were installed in the Mwanza Valley (Cheal, 2014). The current distribution and functionality of water supply boreholes and gravity-fed water points (captured spring supplies) were obtained via questionnaire data collected by our partnering NGO Water for People (WFP) in March 2014, March 2015 (after severe flooding) and June 2015 via smartphone-based field surveys with the application Akvo Flow (Akvo, 2017). Dates of borehole drilling provided in the questionnaire returns allowed maps to be produced estimating the development of borehole density throughout the Mwanza Valley spanning some 58 years previous.

Most supply boreholes are 25 to 50 m deep and predominantly draw groundwater from the heterogeneous alluvial valley aquifer, or else Karoo sediments, with possible exception of boreholes towards the valley margins that may be influenced by groundwater draining from the adjoining fractured bedrock. The alluvial sedimentary successions comprise sequences of clays, silts, sands and infrequent gravels. Finer-grained sediments may predominate with the coarser sand-gravel, more permeable, higher yielding aquifer deposits often found closer to the escarpments (Smith-Carrington and Chilton, 1983).

3.3. Pit latrine context and survey data acquisition

In 2008, the Malawian Government adopted the concept of Community-Led Total Sanitation (CLTS), introduced by the NGO, WaterAid. CLTS encourages pit latrine use and elimination of the practice of

open defecation (OD) (CLTS, 2011). Over 2007-11, the number of communities practising OD has been reduced from 11 % to 5.5 %; of the 18,000 households in Chapananga, 50 % have no access to a pit latrine and the rest use mostly simple pit latrines often in disrepair (Hinz, 2015). Regarding the population with latrine access, 4.9 % have access to an improved latrine whilst the remainder use basic pit latrines. We confirm that Chapananga has now (as of 2017) achieved ODF status.

More progressive ecological sanitation (EcoSan) pit latrine variants are available. These take advantage of the abundant nutrients within excreta, generating valuable agricultural resources, alongside reductions in disease risk and waste loading to the environment (Endale et al., 2012; Langergraber and Muellegger, 2004; Mariwah and Drangert, 2011). However, these are yet to be implemented in Chikwawa despite around 10,000 being built elsewhere in Malawi during 2002-10 (Chunga et al., 2016; Morgan, 2010). This is ascribed to concerns over long-term unsustainability once maintenance cost provisions are phased out, alongside a generally negative public perception. Our own discussions (Kalin) in 2015 with the Ministry of Health in Chapananga suggest other reasons may include plentiful fertilizer arising from cattle abundance and that the consequences of increasing pit latrine density are not yet a priority for the Ministry and hence policies are yet to be put in place for EcoSan planning and community engagement. Therefore, the risk from conventional pit latrines remains for the foreseeable future.

Current (2015) pit latrine data were mapped via WFP smartphone-based field surveys and associated questionnaires using the application Akvo Flow that allowed an estimate of the spatial distribution of at least known latrines. Pit latrine distances to boreholes were calculated from the coordinates using GIS. Mapping covered areas of at least 100 m (and up to 500 m) radius of known borehole – water points, resulting in 189 out of 340 boreholes having pit latrine survey data attached to them (Fig. 1). In the far south-east of the study area, enumerators were unable to undertake mapping of latrines during our study period.

3.4. Groundwater quality survey and sample analysis

A groundwater quality survey was undertaken for the Kakoma Health Area jurisdiction (subsequently termed Kakoma subset) during the June–July 2015 (dry season). Of the 99 boreholes in that area, a total of 91 were sampled and hence high coverage was achieved. Borehole locations were recorded using a Garmin GPS. Two 1 litre polyethylene bottles were filled directly from the borehole handpump (typically Afridev (Water Aid, 2013)). Sampling was typically from boreholes used by local village communities for drinking water and domestic purposes and from those serving various

organisations (e.g., schools, health centres) with daily abstraction rates up to around 5 m^3/d (Schmalfuss, 2014). Regular borehole use caused boreholes to be well purged at sampling. The electrical conductivity (EC) and temperature were measured at each borehole using a Lovibond (Senso Direct, Con200) field probe.

Laboratory methods available in the Ministry's Department of Water Development and Irrigation laboratory are dated compared to a modern laboratory, but based on standard (e.g. ASTM) methods. The acidified bottle of sampled water (2 ml of 37 % HCl) was sub-sampled to determine iron, nitrate (reported as NO_3^- herein) and sulfate via spectrophotometry (UNICO UV 2100) methods and sodium and potassium by flame photometery. The un-acidified bottle of sampled water was analysed for pH (CRISON pH-meter basic 20+) and chloride, carbonate, bicarbonate, calcium and magnesium via titration methods. Turbidity was measured using a DelAgua nephelometric turbidity tube.

Analysis of specific pathogens is complex and expensive. Hence the presence of faecal indictors such as 'Total coliforms' and or *Escherichia coli* (*E. coli*), thermo-tolerant coliform which normally resides in warm environments (~44.5 °C) such as the human intestinal tract (Lawrence et al. 2001) is typically determined. Total coliforms, for the most part, are not harmful to humans, but behave similarly in the environment to many pathogens and hence their use as indicators of possible or impending arrival of disease-causing organisms (Noble et al., 2003). Two microbiological analysis methods were used. For the Filter Membrane method, a 50 ml sample of groundwater collected in a pre-sterilised cup (doused in methanol and set alight) was filtered through a 0.45 µm membrane with a Merck Millipore HAWGO47S6 to capture bacteria on the membrane. This was then placed within a sterilised petri-dish to which lab-prepared media had been added. At the laboratory, petri dishes were incubated at 37 °C for 24 hours and then visually examined for yellow colonies of total coliforms that were then enumerated under magnification.

The Colilert Method involved sterilisation of 250 ml borosilicate glass bottles in a 121 °C autoclave for 30 minutes prior to fieldwork. Bottles were rinsed three times at the borehole and a 200 ml sample obtained. At the laboratory, 100 ml was used to fill large and small sample wells within a Colilert IDEXX Quanti-Tray®/2000 MPN Table (IDEXX, 2015). The tray was then sealed and incubated for 24 hours at 35 °C. A positive test for coliforms appears yellow after incubation and the presence of *E. coli* was confirmed where yellow cells fluoresce under UV light. The numbers of large and small yellow wells on the tray are counted and the Most Probable Number (MPN) determined by reference to the IDEXX Quanti-Tray®/2000 MPN Table.

3.5. Empirical risk assessment approach

An empirical risk assessment approach was used to assess the significance of various factors potentially controlling borehole contamination arising from surrounding pit latrines with the likelihood of contamination being predicted using logistic regression (adapting the approach of Howard et al. (2003)). As several determinants analysed in borehole groundwater samples were potentially indicative of pit latrine contamination - namely microbiologically contamination (by *E. coli* or coliforms), elevated nitrate and elevated chloride - each one of these was selected as a contamination indicator and was assigned a value of either 0 (contamination was not present or below a set threshold) or 1 (contamination was present or above a set threshold). The threshold concentrations for chloride and nitrate were determined based on the observed concentrations in the borehole groundwater quality survey. The parameters that were investigated (later tabulated in the results) were based on either data obtained directly from the questionnaire survey responses on water points and sanitation collated by WFP, or else were calculated parameter values such as pit latrine densities surrounding boreholes computed from these survey data or else from supporting groundwater flow-capture zone modelling (Back, 2015).

The relationship between a categorical response variable, here borehole contamination status, and an explanatory variable, such as borehole age, was hence examined to determine the probability of contamination occurring from each variable parameter. Other categorical (yes or no; allotted values) predictor variables assessed were: the incidence of flooding (as judged by the devastating floods in January 2015 (Fig. 1)); the presence of a water point committee (hence providing improved local borehole management); the presence of unwanted stagnant water around the borehole (that may attract animals, possible indicator of poor management or design); functional permaculture (the engineered use of inadvertent spilt groundwater around the borehole to irrigate local small-scale agriculture); the existence of a protective brick wall (*c*. 1.2 m high) surrounding the borehole (to locally protect the supply); and, the condition of borehole infrastructure (1 – bad; 2 – medium; 3 – good).

Continuous variables analysed as possible risk factors influencing pit latrine contamination of boreholes were: distance to closest pit latrine, radial pit latrine risk assessment approaches (Section 2.6), the age of the borehole, the number of people using the borehole, and, numbers of pit latrines found within modelled 1-year and also 10-year groundwater capture zones of an abstracting borehole (Section 2.7). Much of the above data were obtained from interrogation of the detailed WFP survey questionnaires of the study area water points obtained in 2014–2015 as part of WFP's,

wider geographic and extended purpose, water point questionnaire surveying of the Chikwawa District. All variables and the contamination levels were added into Minitab (Minitab Inc., 2017). The contamination levels were coded as 1 – contaminated and 0 – not contaminated. This binary value was defined as the outcome variable. Subsequently, logistic regressions were run for each risk factor individually in order to evaluate their significance and odds ratios. For the null-hypothesis it was stated that a relationship exists between the contamination and the risk factors.

Consideration was given to collection of further local-scale data, including detailed land-slope (influencing infiltration versus runoff) in the latrine - water point pathway vicinity, supply borehole geological (soil/rock lithology) and hydrogeological (hydraulic conductivity (*K*), current depth to groundwater, etc.) data from borehole-drilling reports and records kept by water-point committees. These data, supplemented by larger scale (hydro)geological-soil map data held by the Malawi Government, would enable local-scale resolution of their influence as factors However, preliminary inspection of the more readily available local data, indicated these appeared to lack in detail, quality assurance and uniformity of information spatially necessary for a rigorous analysis to be developed within the present study timescales. Such data are important for temporal assessment (for instance, depth-to- groundwater variation may account for observed seasonality of contamination (Kostyla et al., 2015)), but especially necessary for spatial estimation of a 'groundwater vulnerability' (to contamination). The latter estimates are often based upon a 'DRASTIC' type of approach or appropriately simplified methodology (Aller et al., 1987; Robins, 2009; Robins et al., 2007; Shirazi et al., 2012; Vías et al., 2005). Development of a groundwater vulnerability based factor is allowed for in the framework approach proposed herein; however, for the study area vulnerabilities are spatially quantified within our on-going work. We note in passing though, the challenges experienced by other workers in developing groundwater vulnerability estimates in the Malawian context (Kanyerere et al., 2012; Robins, 2009; Robins et al., 2007); we comment upon this aspect further in our conclusions.

3.6. Spatial/radial pit latrine risk calculation approaches

To support the empirical risk assessment and the need to consider options to estimate the number of pit latrines potentially interacting with a borehole, several methods have been proposed to project the risk emanating from surrounding pit latrines. Underlying assumptions and calculations differ from method to method and will be explained briefly. The methods share common ground in one aspect in that they all calculate the risk based on radial distances from the abstraction point. The applied radii of assessment are 30, 50, and 100 m, as these distances coincide with suggested

guidelines and may qualify these statements. Their suitability as risk predictors for contamination was also tested in logistic regressions (Section 2.5).

3.6.1. Pit latrine density

This approach, used by Wright et al. (2013) for instance, calculates a pit latrine density simply as the number of pit latrines n_{PL} with a distance to the borehole r_{PL} within a radial area of assessment with a radius r_{assess} (unit: Pit latrine number PL / (Length L)^2, Eq. 1):

$$\text{PL density} = \frac{1}{r_{assess}^2\ \pi} \sum_{r_{\mathrm{PL}}=0}^{r_{\mathrm{PL}}<r_{assess}} n_{\mathrm{PL}} \tag{1}$$

Whilst providing a metric of the overall number of pit latrines surrounding the borehole, the impact of pit latrines very close to a well that may pose a higher risk to the water quality may become obscured by the averaging over the larger radial area used.

3.6.2. Pit latrine reciprocal distance sum

In this approach, the reciprocal distance of all pit latrines within the radius of assessment is summed as follows to give a pit latrine reciprocal distance sum:

$$\text{PL reciprocal distance sum} = \sum_{r_{\mathrm{PL}}=0}^{r_{\mathrm{PL}}\leq r_{assess}} \frac{1}{r_{\mathrm{PL}}} \tag{2}$$

Whilst resulting in a rather intangible number (unit: 1/L, Eq. 2), the estimate does account for higher risk at closer distances. The extreme values are 0, when all pit latrines are located at distances larger than the radius of assessment, or ∞ when a pit latrine is located at zero distance to the well, i.e. are coincident.

3.6.3. Pit latrine loading fraction

Here, the fraction of infiltrating pit latrine leachate within a borehole catchment area, and, in turn, the potential loading to a receptor abstraction borehole was estimated (Eq. 3). A steady-state recharge-abstraction assumption was made whereby spatially uniform natural recharge (RCH – infiltrating precipitation, unit: Length L / Time T) occurring over the surrounding circular assessment

area (L^2) was assumed to undergo radial flow to the borehole and become abstraction. A footprint of pit leachate infiltration A_{PL} to groundwater was then superimposed, assuming each latrine had a footprint loading area of 1 m² (L^2), an infiltration INF_{PL} through the pit of 0.04 m/d (L/T) and that a total number of n_{PL} latrines occurred within the radial catchment considered. The volumetric proportion of pit latrine loading (PL loading) expressed as the flow rate of pit latrine infiltrated water divided by the total flow rate of water recharged/infiltrated, i.e., the areal recharge precipitation flow plus the pit latrine infiltrated component (unit: (L^3/T)/(L^3/T)). The pit latrine loading fraction is hence quantified as:

$$\text{PL loading fraction} = \frac{A_{\mathrm{PL}}\ \mathrm{INF}_{\mathrm{PL}}\ \sum_{r_{\mathrm{PL}}=0}^{r_{\mathrm{PL}} \leq r_{assess}} n_{\mathrm{PL}}}{r_{assess}^2\ \pi\ \mathrm{RCH} + A_{\mathrm{PL}}\ \mathrm{INF}_{\mathrm{PL}}\ \sum_{r_{\mathrm{PL}}=0}^{r_{\mathrm{PL}} \leq r_{assess}} n_{\mathrm{PL}}} \tag{3}$$

A local area annual precipitation of 800 mm and a recharge estimate of 9 % (Bradford, 1973) resulted in a value of 0.0002 m/d for RCH being adopted in our estimates. A typical village borehole abstraction rate of 5 m^3/d equates to the above recharge occurring over a 90-m radius circular area, which was intermediate in the above range of radii used in the other methods.

It should be recognised that under these assumptions for a steady state condition where recharge equates to abstraction, the PL loading fraction will equate to the proportion of pit latrine effluent within the borehole abstraction water (albeit recognising that the contaminant load within the migrating pit latrine infiltrated water may be attenuated prior to reaching the borehole). However, the influence of closer pit latrines may become obscured.

3.6.4. Pit latrine cumulative density

A pit latrine cumulative density approach was used based on summations for 10 m wide ring-shaped zones with radii r_i concentric around a borehole of the individual ring pit latrine densities within the overall radius of assessment (Eq. 4). The innermost interval calculates the pit latrine density for a circle with 10 m radius, while the consequent intervals are 'ring-densities'. The resulting value (unit: PL/L²) provides a cumulative risk estimate that accounts for higher risk at closer distances with the limitation of a precision of 10 m and is calculated as follows:

$$\text{PL cumulative density} = \sum_{r_i=10\,\mathrm{m}}^{r_i \leq r_{\mathrm{assess}}} \left(\frac{1}{r_i^2\ \pi - r_{i-1}^2\ \pi} \sum_{r_{i-1} < r_{\mathrm{PL}}}^{r_{\mathrm{PL}} \leq r_i} n_{\mathrm{PL}} \right)_{r_i = 0, 10, 20, \ldots, 100\,\mathrm{m}} \tag{4}$$

3.7. Estimation of pit latrine occurrence within modelled groundwater capture zones

The assumption of radial borehole interaction (Section 2.6) may be in error as borehole capture zones become more elliptical and biased up groundwater gradient, especially where regional hydraulic gradients and or hydraulic conductivities are increased. Hence, alternative risk factors were evaluated, these being the 'number of pit latrines within' 'one year' and also 'ten year' capture zones around abstracting boreholes. Capture zones were estimated within our supporting numerical groundwater flow modelling work (Back, 2015) that is indicated, in brief, below and in the Supplementary Material (SM).

3.7.1. Groundwater flow model

A groundwater flow model of the Mwanza Valley alluvial aquifer and adjoining Karoo unit was built in MODFLOW-2005 (Harbaugh, 2005) using a ModelMuse interface (Winston, 2009) (noting the model had a wider agenda of use beyond that herein). No-flow boundaries were assumed at the surface-water divide within the Karoo to the south-west and at the alluvium-basement rock contact to the north-east (Figure SM-1). The south-eastern general-head boundary was positioned where the Mwanza Valley opens to the Shire Valley allowing head-dependent discharge to the down-gradient aquifer. Discretisation was to a 90-row by 35-column 500 x 500 m celled domain of grid angle 39° (ModelMuse calculated to improve convergence). Ground surface was defined by a digital elevation model and five model layers used (Figure SM-2). The alluvium was assumed 300-m thick along the north-east (Mwanza fault) boundary with linear interpolation to a 50-m thickness at the opposing south-west valley side. Karoo sediments were assumed 700-m thick with both alluvium and Karoo modelled as dual layers allowing wells to be representatively placed in the uppermost 50-m layer of each. An inserted 10-m low-permeability layer allowed control of leakage between the Karoo and alluvium. The alluvial aquifer interacted with the Mwanza River via riverbed conductance.

Regarding parameterisation, recharge to the alluvium was modeled as 9% of the annual precipitation (Bradford 1973) using local climate data (Chikwawa Boma and Ngabu). These data best represent the lower valley and hence recharge was factored 1.5 times higher for the increased western (Karoo) elevations. *K* was assumed constant over individual units with initial values set at 0.23 m/d for the alluvium equating to a fine sand and 0.027 m/d for the Karoo corresponding to a finer-grained (cemented) sandstone (Back, 2015) with a standard *K* horizontal/vertical anisotropy of 10 assumed. *K* values were used as a fitting parameter with the final value for the alluvium being 1.6 m/d (1.0-2.5 m/d at 95% confidence interval). These, alongside other unit *K* values, were increased over initial expected values and may possibly relate, in part, to the very poorly constrained (possibly high) recharge that is the subject of on-going assessment. The model was calibrated in steady state

(UCODE-2014 (Lu et al., 2014)) using 53 groundwater level observations (50 in alluvium, 3 in Karoo) with simulated versus observed head data compared in Figure SM-3 alongside model water balance data in Table SM-1.

The model reasonably, but not exactly, represented the groundwater head (water table) and flow regime observed to be down the Mwanza Valley with flows locally modified towards river reaches (Fig. 2). Simulated heads within the valley alluvium tended to be a little over-, rather than under-estimated. The observed data shown (Monjerezi et al., 2011; Sehatzadeh, 2011), however, are not a perfect indicator of the flow field. Although both utilise similar data sources, they exhibit local differences in contouring of the water table and river interaction. These data were mostly obtained from well installations spanning many years (1973-2008) and hence cannot be regarded as a point-in-time temporal snapshot, rather a temporally merged representation. Assumptions on the river interaction (e.g., if river stage is contoured) are influential. Our simulated and observed head contours of Sehatzadeh (2011) exhibited flows towards the Mwanza River, whereas the observed Monjerezi et al. (2011) contours do not exhibit a river interaction. Both could be correct in different seasons of high and low groundwater table and, or assumptions made on riverbed conductance. Cognisant of the above uncertainties, the Fig. 2 simulated flow field may be taken to provide a reasonable platform upon which to simulate well capture zones.

3.7.2. Simulation of borehole catchment zones

Borehole catchment zones (US EPA, 1994; Kunstmann and Kastens, 2006) were delineated from the steady-state flow model via release of reverse flow-field tracking particles from each of the 154 extraction wells (74 in Kakoma subset) simulated. Individual well abstraction rates assumed were 5.1 m^3/d based on preliminary field investigation data (Schmalfuss, 2014). In order to be conservative, cognisant of regional flow direction uncertainty, dispersive spread of simulated advected particles was subsequently calculated adopting conventional dispersion assumptions (Gelhar et al., 1992; Pang et al., 2004) thereby allowing relatively discrete, but wider, capture zones to be generated. The dispersed capture zone areas, corresponding to 10 years of groundwater travel time to boreholes, were delineated and pit latrines occurring within those areas enumerated as a risk factor to be considered in the risk assessment. A 1-year, very local, capture zone was estimated and is most relevant where contaminant migration is anticipated to be attenuated, for instance microbiological contaminants or perhaps nitrate or ammonium. A 10-year zone (maximum travel distance 744 m) was judged very conservative, an improbably long travel time for pathogen migration, but not for more conservative solutes also appreciating faster flow zones are locally probable within heterogeneous alluvium.

4. Results

4.1. Framework approach developed

The framework approach developed in the course of the research is illustrated in Fig. 3 and demonstrated herein. It meets the objective of providing a simple framework approach to the assessment of pit latrine sanitation risk to groundwater supply points that may be implemented by practitioners in developing country contexts in Malawi and elsewhere using the methodologies outlined in Section 2. It draws together the diligent mapping of water supply points and surrounding pit latrines, basic hydrochemical monitoring of key groundwater contamination indicators, use of questionnaire surveys to provide data on possible environmental-site context controls culminating in an integrated statistical evaluation of the obtained datasets to determine the significant risk-controlling factors. The approach illustrated is followed with the exception of the development of a groundwater vulnerability factor (as discussed in Section 2.5).

4.2. Historical development of borehole and pit latrine infrastructure

The development in borehole installations from 1968 to 2015 shown in Fig. 4 exhibits marked periods of activity ascribed to increased WASH projects run by NGOs. There has been significant activity since 2011 with new boreholes being drilled in areas with, and without, existing boreholes. Not all installed boreholes may remain functional though for various reasons, including salinisation, poor design or installation and maintenance issues. Some 38.5 % of installed boreholes were estimated as not functional in 2015 based upon survey returns.

Fig. 4 also displays the parallel growth in pit latrine installations. The recent pit latrine survey by WFP (June 2015) estimated that some 4833 latrines were in use alongside 1961 pit latrines that were full and no longer used within the study area. The survey recorded whether or not the current pit latrine was a replacement and how many filled or abandoned pit latrines were present. Of current latrine pits, 2752 were the first and 1759 replacements (with 322 unknown). With an average pit fill-up rate of 3.9 years (Malawi Government - MoIWD, 2008) and the number of pervious pits, it can be estimated that 57 % of the pit latrines were built after 2011, 31 % in the period 2007-2011, and only very sparse pit latrine development took place in the early 2000's and late 90's.

Pit latrine numbers and density increase towards the populated areas of the central valley area around Timbenao and the confluence of the Ngona and Mwanza (Fig. 4, inset). The densities in the groundwater quality survey area of Kakoma subset are amongst the highest. Fewer pit latrines are found in more remote areas due to a combination of lower populations requiring fewer latrines and also CLTS may be yet to reach those communities.

4.3. Groundwater quality – assessment of pit latrine contamination indicators

4.3.1. TDS and groundwater type

Discernment of pit latrine influence is unlikely from total dissolved solids (TDS) data alone; however, increased TDS alongside changes in groundwater type may provide some supporting evidence. A TDS mean of 1684 $\pm$ 1722 mg/l (n = 91) and median of 966 mg/l were observed indicating that TDS was moderately elevated, with 26 % of samples exceeding Malawi's 2000 mg/l standard (MBS, 2005). The World Health Organisation (WHO, 2011) recognises that water palatability is generally considered good for TDS <600 mg/l (28 % of boreholes) and becomes increasingly unpalatable above a 'brackish' TDS threshold of 1000 mg/l. This was exceeded by 49 % of boreholes with 9 % over 5000 mg/l. TDS values are commensurate with other Chikwawa District studies (Monjerezi et al., 2011, 2012; Mapoma and Xie, 2014).

The distribution of four allocated groundwater group types based on their major ion relative dominance (Hinz, 2015) is shown together with TDS data in Fig. 5a. Group G1 of 31 % occurrence and mean TDS of 2040 mg/l has a Ca-(Mg)-HCO3 composition and infers aluminosilicate weathering to be influential along valley margins. G2, the most dominant group at 36 %, has a Na-mixed cation-HCO3 composition of low mean TDS of 1120 mg/l. It occurs on the north bank somewhat set back from the river. Consistent with Monjerezi et al. (2012), a combination of aluminosilicate weathering, cation exchange and precipitation of carbonates and clays is probable. G3 samples of Na-(Ca, Mg)-HCO_3-Cl composition and mean TDS of 2380 mg/l tend to occur near the river, but dispersed within the other water types. They appear to be a mixture of G2 and G4 waters.

Group G4 represents brackish or saline waters of Na-Cl composition and high mean TDS of 4630 mg/l. Their occurrence, somewhat clustered, is essentially restricted to the north bank. Elevated sulfate suggests that dissolution of both gypsum and halite evaporates could be an important. Contributing processes may include: shallow groundwater evaporation near rivers exacerbated by flooding-drying cycles; deposited evaporite dissolution from palaeo-lacustrine

environments; and, (fault-based) intrusions of mineralised groundwater in the Karoo and Cretaceous Lupata formations (Monjerezi et al., 2012).

It is improbable, as suspected, that pit latrine contributions can be distinguished from TDS data alone. Likewise, complexity of groundwater types means that perturbation of hydrochemical types is unlikely to be manifest from latrine inputs. Hence, the data value is largely one of hydrochemical-flow regime conceptualisation that underpins the more specific tracer evaluation.

4.3.2. Chloride

Chloride is a useful tracer as it is conservative and able to migrate (with dispersion) at advecting groundwater velocities without attenuation loss (Nyenje et al., 2014). However, there are many anthropogenic sources of chloride alongside its potential natural dissolution from rock minerals. Groundwater chloride observed in the Kakoma subset is shown in Fig. 5b, with symbols used to further differentiate higher chloride G3 and G4 groundwater groups from low chloride G1 and G2 groups. Provisionally, elevated chloride in G3 and G4 are predominantly ascribed to natural dissolution of minerals and would appear to largely preclude chloride use as a pit latrine tracer in the area. Prospects of success with chloride as a tracer are likely limited to areas where just G1 and G2 are prevalent.

Examination of Na:Cl ratio data could perhaps provide a further tool to differentiate pit latrine and natural chloride. Our analysis (not shown) demonstrated a 1:1 ratio was approximately followed by G3 and G4 samples as anticipated for halite dissolution dominated waters. Concentrations were so elevated, however, that latrine chloride contributions may form a limited component and hence ratio changes, if occurring, are likely limited. G2 samples, although at lower concentrations, exhibited a considerable amount of scatter in the ratio values that would likely make pit latrine plume differentiation problematic. G1 type groundwater, however, offer the advantage of both low concentration and a relatively uniform ratio trend towards the Na side of the 1:1 ratio and may be favourable for plume differentiation where pit latrine input ratio are contrasting.

Overall chloride use as a pit latrine tracer is problematic due to the locally elevated and varied chloride naturally present. Where elevated chloride occurs, supporting evidence of other pit latrine contamination indicators would be required to confirm source apportionment of observed chloride (or part of) to latrine inputs.

4.3.3. Nitrate

Nitrate is often used as an indicator of potential faecal contamination due to elevated nitrogen content within excreta. However, it was only encountered at very low concentrations throughout Kakoma subset to a maximum of only 2.79 mg/l (as NO_3^-) with a mean of just 0.52 $\pm$ 0.49 mg/l. Moderate clustering of higher nitrate occurred around the central to northern area (Fig. 5c) with the most frequently encountered elevated nitrate contamination occurring in the south-east of the study area. Being furthest down the valley, ground elevations and depths to groundwater are likely lowest here and groundwater potentially more vulnerable. Low nitrate was frequently encountered towards the western alluvial margin-Karoo and provisionally ascribed to fresher recharge groundwater of decreased anthropogenic influence in this more sparsely populated area. Observations are consistent with similarly low nitrate reported across Chikwawa (Monjerezi et al., 2011). Low occurrence is ascribed in part to widespread, but low intensive, agriculture in contrast to nitrate pervasive in European groundwater attributed to many decades of nitrogen-based fertiliser application (Durand et al., 2011; Rivett et al., 2007).

In addition to land-use constraints, low nitrate could arise from its attenuation under anaerobic conditions where denitrification results in ultimate degradation to nitrogen gas (Nyenje et al., 2014; Rivett et al., 2008). Whilst beneficial in mitigation of pit latrine impacts, it limits the use of nitrate as a conservative tracer of pathway connectivity to where aerobic conditions prevail. Nitrogen may also be present in a reduced form within a pit latrine setting; ammonium will initially form via ammonification of nitrogen-rich organic matter prior to being oxidised (nitrification) via nitrite to nitrate. Where ammonium persists, cation exchange, particularly in more clay-based strata, will cause its transport to be retarded and restricted to near-source occurrence, recognising the potential for ammonium oxidation and release as mobile nitrate if aerobic conditions return (e.g., latrine input abatement. This leads to ammonium rarely being used as a primary indicator of latrine-borehole impacts compared to nitrate (Graham and Polizzotto, 2013).

Whilst ammonium analysis was unavailable in the present study, heterogeneous reducing conditions were evidenced by moderate total iron (Fe) concentrations in boreholes sampled at 0.30 $\pm$ 0.22 mg/l to a maximum of 0.9 mg/L (n=89). A plot of these iron data versus nitrate is shown in Fig. 6 with summary statistics of occurrence above and below an arbitrary 0.5 mg/L Fe elevated concentration threshold (equivalent to the 80th percentile). Whilst much of the data occur within the low nitrate - low iron quadrant, the plot indicates that where nitrate is elevated (1-3 mg/L) then iron concentrations are low (< 0.5 mg/L Fe) and is consistent with more aerobic plume conditions prevailing allowing greater nitrate mobility. Also, where iron is elevated (0.5-0.9 mg/L Fe), then

nitrate is low, often below 0.5 mg/L. This is consistent with more reducing anaerobic plume conditions and decreased nitrate concentrations may have potentially arisen from denitrification, nitrate being (thermodynamically) preferentially used over iron as an electron acceptor (Rivett et al., 2008). Reducing conditions may also favour nitrogen occurrence as ammonium. Anaerobic/anoxic conditions are more probable where unsaturated zones are limited (high water table-flood conditions); significant labile organic matter occurs, e.g., the main body of a latrine leachate plume; and, increased low-permeability silt/clay horizons (not uncommon in this predominantly finer-grained alluvial system (Smith-Carrington and Chilton, 1983)) that lead to more prevalent (semi)-confined aquifer conditions.

Fig. 6 may serve as a useful baseline plot (of easily obtainable data) against which emergent latrine-plume impacts with time could be assessed. Decreased occurrence with time of samples plotting within the low nitrate - low iron quadrant may be expected concurrent with increased high nitrate - low iron sample occurrence if aerobic plumes prevail, or else high iron - low nitrate occurrence for anaerobic plumes. Further supporting data are required, however, to resolve if nitrate attenuation is actually occurring, the controlling processes involved and the discrimination of latrines as the nitrogen source. This is most likely to be realised via isotope techniques (Anornu et al., 2017; Aravena and Robertson, 1998; Matiatos, 2016; Puig et al., 2017; Varnier et al., 2017). Of particular interest to evaluate would be the anticipated isotopic enrichment of $\delta^{15}N$ and $\delta^{18}O$ occurring as denitrification causes nitrate depletion along a groundwater flow path (Aravena and Robertson, 1998). Use of these and other isotopes, alongside enhanced biogeochemical sampling, may further help to discriminate denitrification and other N-cycle reaction types and associated electron donor/acceptor controls. The above serves to illustrate that nitrate use as a tracer requires careful consideration where anaerobic conditions may prevail and the attendant uncertainties should be recognised.

4.3.4. Microbiological contamination

Of the 91 boreholes sampled, 13 tested positive for microbiological contamination. All boreholes were tested using the filter membrane analysis, but only 34 using the Quanti-Tray enumeration procedure. 7 borehole samples detected the presence of *E. coli* and 7 were found to contain coliforms present, with one borehole testing positive for both *E. coli* and coliforms. Coliform colony counts ranged from 3 to 25 cfu/50 ml whilst the MPN of *E. Coli* ranged from 1.1 to 4.1 MPN/100 ml with the exception of a single, very elevated, outlier of 524.7 MPN/100 ml. Further confirmation,

through repeat sampling would be advisable of these detection data. However, within the data analysis that follows, these boreholes are provisionally classified as microbiologically contaminated.

The spatial distribution of the microbiological detections is shown in Fig. 5d. Whilst some local pairs of borehole occurrences occurred, the distribution overall is rather sporadic with microbiological detections occurring at both high and low nitrate concentrations (comparing to Fig. 5c). The latter, perhaps counter intuitive, could potentially arise where microbiological source – receptor connectivity to a pit latrine (or other source of microbiological contamination) occurs but where nitrate is attenuated through locally high dissolved organic matter loading from a latrine.

4.3.5. Pit latrine contamination thresholds

The results do not demonstrate a particularly obvious preferred indicator tracer of latrine source-borehole receptor connectivity in the study setting. Chloride is likely influenced by natural mineral dissolution, nitrate concentrations are very low and possibly subject to variable attenuation (Fig. 6 providing a line of evidence) and microbiological contamination relatively infrequent and, at preliminary inspection, generally not related to elevated nitrate (or chloride) occurrences. This, however, does not preclude assignment of threshold concentrations indicative of potential contamination by surrounding pit latrines still being made to evaluate if correlations of an assigned potentially contaminated borehole occur and are significantly controlled by factors investigated in the empirical risk assessment. Sets of provisionally 'contaminated boreholes' to be evaluated in the assessment were based on the pragmatic thresholds assigned below:

- 'elevated chloride' where concentrations exceeded the 80th percentile, >490 mg/l (17 out of 85 boreholes)
- 'elevated nitrate' where concentrations exceeded the 80th percentile, >0.73 mg/l (18 out of 87 boreholes)
- 'microbiologically contaminated' where positive microbiological detection was observed (13 out of 91 boreholes).

Recognising too the observed occurrence of high iron with low nitrate that is potentially a result of denitrification (Fig. 6) for which the iron plume is a secondary indicator of a pit latrine plume presence, the contaminated boreholes assessment also evaluated:

- 'elevated iron' where concentrations exceeded the 80th percentile, >0.5 mg/l (18 out of 89 boreholes).

4.3.6. Resulting segregation of datasets for statistical analysis

Segregation of datasets is adopted to maximise analysis from the available water quality and pit latrine occurrence data. A radial pit latrine risk calculation (Section 2.6) was undertaken for 189 boreholes distributed throughout the valley (white and green points in Fig. 1). Boreholes tested for microbiological contamination in Kakoma subset totalled 91, of which 77 were subject to pit latrine risk calculations (green in Fig. 1) and 14 were not (due to an absence of latrine inspection data, blue in Fig. 1). Of the 91 samples, a subset of 13 samples was microbiologically contaminated and 77 were microbiological contamination free. At the same time, 87 boreholes were tested for nitrate, 73 were subject to calculations and 14 were not. Of these 87 boreholes, 18 had elevated nitrate and 68 low nitrate concentrations. Analysis of these various subsets was undertaken. With some opportunity loss due to partial latrine mapping data for some boreholes, the end result is that the following subsets of boreholes are later analysed to evaluate borehole-surrounding pit latrine relationships:

- Mwanza Valley borehole (not in Kakoma subset), unknown microbiological (n = 112) / unknown nitrate (n = 116) contamination (blue graph line in later figures)
- Kakoma subset borehole – microbiological (n = 65) / nitrate (n = 61) contamination free (green graph line)
- Kakoma subset borehole – microbiological (n = 12) / nitrate (n = 12) contamination (red graph line)
- All boreholes (total of the above), n = 189 (black graph line)

4.4. Empirical risk assessment

4.4.1. Summary of parameters - statistical analysis results

Table 1 summarises the parameters used in the empirical risk assessment-statistical analysis to evaluate the significance of the various factors potentially controlling borehole contamination due to pit latrines. Parameter values are populated from our questionnaire survey responses, groundwater quality survey data and supporting numerical flow model-capture zone study. As per Section 3.3.6, the sample size available to each factor assessment varies depending on whether the assessment primarily draws upon the smaller Kakoma subset and associated groundwater quality data or the greater Mwanza valley – pit latrine incidence data. The assessment below initially considers correlations with the various metrics of pit latrine occurrence relating to radial distance introduced

in Section 2.6, followed by correlations with pit latrines encountered in modelled groundwater capture zones and finally assessment of the range of other environmental-site condition factors (Table 1). The statistical analysis results obtained from the logistic regression analysis are summarized in Table 2, where the significant parameters (p-value < 0.05), odds ratios and confidence intervals determined are indicated (complete tabulations of statistical analysis results are provided in the Supplementary Material – Tables: SM-2 for chloride and microbiological contamination, SM-3 for nitrate and SM-4 for iron). These results are discussed within their relevant sections below.

4.4.2. Radial-based pit latrine occurrence metrics and correlations

Examination of the relationship between the closest pit latrine to a borehole and the microbiological and nitrate contamination observed is respectively shown in the cumulative profiles of Figs. 7a and 7b. Examining initially the 'All data' (black) profiles (identical in each plot) indicates the median closest pit latrine distance is 37.9 m with 25th and 75th percentiles at 28.9 and 57.9 m respectively. The percentile curve increases most rapidly over the 30-40 m radial interval with 28.9 % of the sample population within this radial interval and 38.7 % in the 30-50 m interval.

These closest-distance data compare to international borehole-pit latrine minimum separation distance guidelines of 15 m suggested by the WHO (Franceys et al., 1992), 30 m for Haiti (Reed, 2010) and also suggested for disaster response projects (Sphere project, 2011) with more conservative guidelines of 50 m suggested by WaterAid (WaterAid, 2013) and 75 m by South Africa (Still and Nash, 2002). Parker and Carlier (2009) summarise national guideline minimum distances of 15 m for Bangladesh, 25 m for Burkina Faso, 30 m for Ethiopia, 50 m for Ghana, 3 or 10 m (depending on water table depth being greater or less than 2 m) for India, 15 m for Mali, 50 m for Uganda and 30 m (per sanitation guidance) and 50 m (per well drilling guidance) for Mozambique, with, at 2009, Madagascar, Nepal, Nigeria, Pakistan, Papua New Guinea, Tanzania, Timor Leste and Zambia having no set guidelines. Whilst Malawi identified a specific strategy to determine a minimum allowable distance from a groundwater source to pit latrines within its National Water Policy (Malawi Government – MoIWD, 2007), this document does not specify a distance. However, the National Sanitation Policy indicates, within its definitions, latrines should be at least 30 m from a groundwater source or surface watercourse (Malawi Government – MoIWD, 2008). In practice, it seems distances adopted in Malawi vary between 30 m (potentially used by Ministry of Health staff) and 50 m (potentially used by Ministry of Water Development staff). Such distances do indeed appear consistent with the observed distribution of Fig. 7.

Regarding the influence of potential contamination of boreholes, the Fig. 7 profiles for the microbiologically contaminated (or not) and elevated nitrate (or not) subsets are fairly comparable to the All data and N/A profiles ('not available' dataset for which water quality data were not obtained), but with some indication that the elevated nitrate contamination profile (Fig. 7b) exhibits marginally increased percent occurrence at shorter separation distances. This points to a possible influence of latrine proximity upon increased, albeit low concentration, nitrate. This would be consistent with emergent latrine pit-borehole connectivity in its infancy.

Examining further simple radial based data, preliminary assessment of the degree of chloride and nitrate contamination observed with the numbers of pit latrines enumerated within radii of 30, 50 and 100 m is illustrated in the Fig. 8 plots. Increased concentrations with greater numbers of pit latrines are not obvious. Whilst chloride at 30 m and nitrate at 30 m and 50 m exhibit the increasing, albeit very slight, trends anticipated there is significant data scatter and consequently R^2 values are extremely low, at 0.016 or less.

Extending from the simple radial assessment above, Fig. 9 displays box-plots for the four-alternative radial/spatial pit latrine occurrence metrics proposed. The plots again cover 30 m, 50 m and 100 m radial-based assessment areas with points plotted for elevated nitrate and microbiological contamination (or not) and the larger N/A dataset. Corresponding mean and median data are indicated in Table 1 and analysis estimates provided in Table 2. The data show a trend of higher metrics values being obtained for the elevated versus the non-elevated nitrate concentration in the boxplots. This trend is confirmed in logistic regression data, which shows correlation for the 100-m assessment radius for all metrics, and for the 50-m assessment radius for all except one method, but not for 30 m. Hence, at larger influence radii assessed the proposed metrics reliably predict elevated nitrate concentrations exhibiting correlation with latrine densities, although the absolute concentration may still be small. This points to the aggregate loading of the bulk of pit latrines at 30 m or more distance being important and that nitrate attenuation (with some variable occurrence possible based on the Fig. 6 observed nitrate-iron data) is insufficient to prevent nitrate loading to wells occurring from distant sources, i.e., latrines within the 30- to 100-m radial interval. Temporal monitoring over several years at least would be required to establish if the observed present low load of nitrate to wells, i.e., the established 'baseline', increases as would be anticipated for an emergent problem detected in its infancy. This would be a reasonable conceptualization given the recent growth in pit latrine use in the study area.

Such correlations at medium to large radial distance were not manifest for microbiological contamination incidence which is consistent with the above conceptualization when it is recognised

that attenuation is typically much more significant for microbiological contaminants compared to nitrate. Lower trending metric values for microbiologically contaminated boreholes and negative correlations (odds ratios < 1) were found for microbiological contamination, particularly for two of the 50-m metrics. This would suggest that higher metric values, i.e., greater pit latrine loading, were associated with lower microbiological contamination risk. This observation may be explained by significant attenuation of microbiological contamination associated with the aggregate bulk of latrines at 30 m, or more, distance causing correlations observed for more mobile nitrate not to be apparent for microbiological contamination, at present. Furthermore, the lack of correlation of microbiological contamination incidence with pit latrine loading tentatively points towards alternative sources other than pit latrines being responsible for the observed microbiological contamination of boreholes.

Chloride did not display any obvious metrics correlation which is consistent with the above conceptualization in that whilst chloride may migrate conservatively from surrounding pit latrines (alongside nitrate), its presence is likely masked by elevated natural background chloride concentrations already present in the study area. Likewise, elevated iron occurrence failed to display any metrics correlation of significance. The above analysis and conceptualization arising illustrates the importance of considering a range of pit latrine contamination tracers and selection of radial-based latrine-loading estimates in order to provide an effective baseline against which future impacts of ongoing pit latrine development may be evaluated. The analysis does not indicate one metric being more suitable than others, however, weight-based methods that would better accentuate the impact of closer to medium distance latrines which may potentially emerge as key metrics as contamination scenarios mature. The recommendation is hence to include a variety of metrics to provide a flexible baseline methodology and build a weight-of-evidence approach to evaluate potential contamination risk emergence with time.

4.4.3. Correlation with pit latrine incidence within modelled capture zones

The above radial-based approaches are only valid if flow to an abstraction is approximately radial. When regional hydraulic gradients are pronounced and aquifers are transmissive, resulting in abstracted groundwater largely being drawn from up-gradient, then a model-based capture zone recognising such gradients is preferred. This is because the drawback of water from even moderate distances down gradient is unlikely unless the reverse conditions to the above apply, i.e. low regional gradients, less transmissive units alongside high abstraction rates (improbable with handpumps).

Fig. 10 illustrates an example area of modelled 1-year and 10-year capture zones and their associated interaction with surrounding pit latrines. Zone lateral discreteness inevitably may cause some latrines to be just within, or just beyond the modelled zone. Zone orientations are very sensitive to the regional flow-field incorporated within the model, which may variably represent the local reality (see Fig. 2 discussion). Although the dispersion approach included within the methodology allows some fuzziness of the lateral boundary, flow regime uncertainties may result in a proximal pit latrine in reality causing borehole contamination to be missed by a discrete capture zone simulated that would be accounted for by a radial-based method of sufficient distance. This possibly accounts for the Table 2 result that significant correlations were not found for nitrate or microbiological contamination with pit latrine incidence within either the 1-year or 10-year capture zones, but were found for nitrate using some of the radial methods.

No correlations at all with radial or capture zone based metrics were found for the elevated iron concentration subset. Further checking of alternative sampled borehole subsets based around the Fig. 6 nitrate – iron plot data was also undertaken of elevated iron with low nitrate (a “candidate anaerobic plume” sample subset) and elevated nitrate with low iron (a “candidate aerobic plume” sample subset). Both failed to reveal any correlations with the radial area, capture zone or other metrics tested in Table 2. Hence at the present time, it is not possible to correlate elevated iron or those combined data candidate plume type occurrences with pit latrine sources.

Fig. 10, however, remains graphically illustrative of the potential risks, even threat, posed by the wealth of pit latrines in the general vicinity of wells. Construction of more local-scale models parameterized with higher resolution local data (perhaps often not available or at least not collated) is endorsed to substantiate the predictions of our regional-scale modelling, particularly where latrine densities are fairly high and, or contamination is perhaps emergent. The enormous numbers of pit latrines within many of the simulated 10-year capture zones may appear somewhat disturbing and does certainly illustrate the critical need for natural attenuation of contaminants to be effective to prevent impacts. Although offset by dilution with increasing latrine distance from boreholes, it may be anticipated that conservative, non-attenuated, migration of chloride (under any conditions) and nitrate (under aerobic conditions) would eventually give rise to gradually increasing concentrations of these contaminants at boreholes, particularly those able to interact with significant numbers of latrines nearby. This would endorse the need for baseline datasets to evaluate such trends over time. Baselines may importantly allow some confirmatory identification of boreholes more vulnerable to latrine loading as shown by rising chloride or nitrate (not attributable to other non-

latrine sources) and highlight needs to potentially target these for more regular monitoring of acute pathogen risks to ensure that the microbiological attenuation presumed is adequate.

4.4.4. Correlation with environmental-site condition factors

The only two significant variables (p-value < 0.05) in the causing of microbiological contamination were the number of people using a borehole and whether or not the water point had a protective wall installed around the well with p-values of 0.04 and 0.01, respectively. Despite elevated nitrate concentrations exhibiting some correlations with the radial distribution of pit latrine metrics, significant correlations were not observed with any of the other factors investigated.

The odds ratio indicates that for every 50 people more withdrawing water at a water point the borehole is 1.07 times more likely to be microbiologically contaminated. 'A wall in place' means that the borehole is 5.9 (calculated from 1/0.17) times less likely to be contaminated. The 95 % confidence intervals indicate a 95 % certainty that the likelihood of the borehole not being contaminated with a wall in place is between 1.2 (from 1/0.83) and 25 (from 1/0.04) times higher compared to not constructing a wall.

The other variables do not have a significant influence on the contamination of the borehole. A relationship for a 'water point committee in place' could not be calculated, as quasi-separation of the datasets prevails (only 1 out of 79 boreholes has no water point committee). Although not significant, a trend to a lower likelihood of contamination is associated with the implementation of permaculture (the development of a borehole garden utilising spilt abstracted groundwater to primarily help fund water point maintenance via garden-produce sales (Vitari and David, 2017)). However, only the construction of a wall around the borehole and the number of people using a borehole has a significant influence on the microbiological contamination of boreholes.

Integrating the above, it is reasonably hypothesised that the majority of pit latrines are at sufficient distance (perhaps only just in many cases given the high incidence in the 30–40 m interval) to allow microbiological die-off and that correlation with radial pit latrine occurrence metrics only occurs with nitrate, albeit at low concentrations, that is likely more mobile (tending to conservative under aerobic conditions) and arises from the often local prevalence of latrines. It may be inferred from the observed correlations with site condition factors that alternative local microbiological sources and pathways could exist, notably including short-circuiting along a poorly sealed borehole annulus (Knappett et al.2012). A protective wall will more effectively safeguard against this pathway as animals and other hazards are kept away. With increased numbers of borehole users, the more likely

it is that contamination takes place around the borehole; a hectic environment may cause high spillages, attract animals and cause localised contamination. Existing permaculture, an indicator of careful water-point management, has the potential to prevent localised contamination; critically it should remove the accumulation of stagnant ponds that may become contaminated by animal faeces and locally infiltrate and potentially pollute the borehole source that caused the pond. Whilst conjectural, the above may reasonably account for the specific findings.

5. Conclusions and relevance

Risks to groundwater supplies posed by current sanitation policies that are often pit-latrine based in many developing countries are shown within the Malawi study context to be challenging to assess, but critically important to consider moving forward. The assessment framework demonstrated (Fig. 3) is pragmatic, includes a range of collectable datasets, integrative and is implementable by practitioner bodies such as regulators managing a jurisdiction. The approach is seen to be particularly useful in the vital establishment of baseline conditions of what is likely an emergent issue in many developing countries. Baselines are fundamental to future trend monitoring and verification of pit latrine contaminant natural attenuation pivotal to the long-term viability and success of sanitation policies. Increasing population and life expectancy, development pressures and sheer numbers of latrines and wells underscore the on-going need for effective approaches to assess and manage the impact of latrines upon groundwater resources.

Establishing so-called ‘pollutant linkages’ between latrine sources and receptor groundwater points poses significant challenge. Similarly, proving the sufficiency of latrine contaminant natural attenuation occurrence in groundwater critical to human-health safeguard is equally challenging and may be contributory to the wide international range in guidelines on safe latrine – water-point separation. Investigative resources are invariably limited in developing countries, groundwater monitoring is often restricted to supply borehole receptors and hence the migration pathway remains unevaluated, the emergent problem may be near imperceptible as plumes gradually grow, the typical latrine contaminant tracers used, although complementary, have individual drawbacks, and the discrimination of latrine sources and contaminant natural attenuation process occurrence is difficult and requires advanced (e.g., isotopic) analytical tools. These are all illustrated to be issues within this Malawian study and expected to be globally relevant to developing country contexts elsewhere.

Incorporation of a groundwater vulnerability factor is recommended within the overall framework methodology presented (and the subject of study area future work looking to use data from recent drilling programmes). However, as illustrated by vulnerability assessments elsewhere in Malawi (Robins 2009; Robins et al., 2007; Kanyerere et al., 2012), assessments may be fairly onerous. Down-scaling the data-intensive regional approach to catchment scales is challenging. The simpler vulnerability assessment scorecard technique developed from DRASTIC principles by Robins (2009), whilst a more qualitative, subjective and site-specific approach designed to be amenable to the African (sub-)catchment and distributed rural village scale, still extensively relies upon data from well-documented borehole drilling programmes. Improvements in the systematic securing and archive availability of geological log, groundwater level/parameters and soil-type data from both borehole and latrine installation (WASH) programmes is vitally required in Malawi and critically underpins effective groundwater vulnerability assessment work.

Specific management concerns and research needs hence identified for Malawi, and expected to have applicability elsewhere, include:

- a lack of agreed, science-based, guideline values for minimum separation distances to be implemented between pit latrines and water points;
- the potential for new pit latrines to be dug in the vicinity of old ones in an uncontrolled manner potentially closer to water points;
- contaminant mass loading from pit latrines to groundwater being poorly constrained;
- policy reliance upon pathogen attenuation and die-off that is also poorly constrained, particularly within the changed hydro-biogeochemical environment associated with latrine inputs;
- widespread increase in nitrogen loading to groundwater from both increased latrine and agricultural sources and spread of potentially mobile nitrate;
- typically limited (financial) resources to undertake appropriate routine groundwater monitoring, non-ideal reliance upon receptor supply wells for sentinel monitoring and an absence of pathway monitoring at local scales to resolve controlling processes;
- latrines likely form, low lateral dispersion, i.e. thin, groundwater plumes that are difficult to monitor, i.e. easily missed (see septic tank examples of Robertson et al. (1991))
- the standard latrine contaminant tracers (used herein) are problematic and other supporting more diagnostic tracers are required; for example, Robertson et al. (2016) use artificial sweetener acesulfame (ACE) to estimate the proportion of nitrate in groundwater samples

apportioned to septic tank wastewater discharges (presupposing the consumption of ACE is sufficient or will increase in developing countries to serve as a viable tracer);

- gaining proof of in-situ contaminant plume natural attenuation is challenging, especially in heterogeneous, fast-flow, high-risk environments and sophisticated analytical approaches (e.g., isotopes) are ideally required;
- adequate advance collation of data (e.g., during drilling programmes) to underpin groundwater vulnerability factor estimation;
- effective management of future well placements relative to pit latrines taking into consideration probable groundwater flow directions;
- the need for improved chemical-microbiological water quality analysis laboratory facilities and training – this would include what is regarded as standard (in developed countries), as well as advanced, techniques; and
- risks yet to be considered – for instance, anecdotal evidence that used engine oil is put into pit latrines to suppress smells.

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Supplementary Material

Supplementary Material may be found at >>> Journal to provide web link

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Table 1: Summary of parameters from questionnaires, groundwater survey data and flow modelling for statistical analysis

		Mean	Median	Yes	No	Sample size
Nitrate / mg L^{-1}		0.50	0.37			87[a]
Elevated nitrate concentration (> 0.73 mg L^{-1})				18	69	87[a]
Chloride / mg L^{-1}		375.1	153.0			85[a]
Elevated chloride concentration (> 490 mg L^{-1})				17	68	85[a]
Iron / mg L^{-1}		0.30	0.24			87[a]
Elevated iron concentration (> 0.5 mg L^{-1})				18	69	87[a]
Microbiologically contaminated				13	78	91[a]
Flooded in January 2015				17	56	73[a]
Water point committee				78	1	79[a]
Stagnant water				60	18	78[a]
Permaculture				24	45	69[a]
Wall in place				34	45	79[a]
Infrastructure				Bad (1): 24, Medium (2): 33, Good (3): 22		79[a]
Age of borehole		7.1	4.0			79[a]
Users		500	400			79[a]
Latrines in 1 year capture zone		0.7	0			74[a]
Latrines in 10 year capture zone		13.3	6.5			74[a]
Distance to closest latrine		59.8	37.9			189[b] (77[a])
Pit latrine density	30 m	$1.7 \cdot 10^{-4}$	0			189[b] (77[a])
	50 m	$2.8 \cdot 10^{-4}$	$2.6 \cdot 10^{-4}$			189[b] (77[a])
	100 m	$3.1 \cdot 10^{-4}$	$2.6 \cdot 10^{-4}$			189[b] (77[a])
Reciprocal distance sum	30 m	0.03	0			189[b] (77[a])
	50 m	0.07	0.05			189[b] (77[a])
	100 m	0.17	0.14			189[b] (77[a])
Loading fraction	30 m	0.029	0			189[b] (77[a])
	50 m	0.050	0.048			189[b] (77[a])
	100 m	0.057	0.048			189[b] (77[a])
Cumulative density	30 m	$4.4 \cdot 10^{-4}$	0			189[b] (77[a])
	50 m	$1.1 \cdot 10^{-3}$	$7.1 \cdot 10^{-4}$			189[b] (77[a])
	100 m	$2.8 \cdot 10^{-3}$	$2.1 \cdot 10^{-3}$			189[b] (77[a])

[a]Kakoma subset; [b]Chapananga data set (n=189; n=77 within Kakoma subset)

Table 2: Significant parameters, odds ratios and confidence intervals from logistic regression analysis

Response	Parameter (* significant)	*p*	OR	LCI	OCI	Unit of change

Elevated chloride concentration	no correlations due to high background signal						
Microbiologically contaminated	Wall in place*		0.011	0.17	0.036	0.829	No-Yes
	Users*		0.042	1.07	1.0007	1.1433	+50
	PL density	30 m	0.756	0.89	0.41	1.93	+0.001
		50 m	0.087	0.47	0.16	1.40	+0.001
		100 m	0.375	0.84	0.56	1.27	+0.001
	PL reciprocal distance sum	30 m	0.660	1.37	0.35	5.35	+0.1
		50 m*	0.047	0.08	0.00	1.59	+0.1
		100 m	0.682	0.62	0.06	6.51	+0.1
	PL loading fraction	30 m	0.591	1.27	0.54	3.02	+0.1
		50 m*	0.049	0.25	0.05	1.25	+0.1
		100 m	0.725	0.78	0.19	3.19	+0.1
	PL cumulative density	30 m	0.841	0.95	0.57	1.57	+0.001
		50 m	0.109	0.66	0.36	1.23	+0.001
		100 m	0.404	0.90	0.70	1.17	+0.001
Elevated nitrate concentration	PL density	30 m	0.089	3.08	0.86	11.0	+0.001
		50 m*	0.010	7.34	1.57	34.2	+0.001
		100 m*	0.018	11.1	1.48	82.3	+0.001
	PL reciprocal distance sum	30 m	0.309	1.32	0.79	2.23	+0.1
		50 m	0.063	1.49	0.98	2.27	+0.1
		100 m*	0.037	1.34	1.02	1.76	+0.1
	PL loading fraction	30 m	0.107	1.99	0.88	4.52	+0.1
		50 m*	0.011	3.48	1.3023	9.31	+0.1
		100 m*	0.020	4.40	1.25	15.5	+0.1
	PL cumulative density	30 m	0.092	1.40	0.96	2.06	+0.001
		50 m*	0.016	1.44	1.07	1.93	+0.001
		100 m*	0.014	1.26	1.05	1.52	+0.001
Elevated iron concentration	no correlations						

*Significant parameters, [a]OR – Odds ratio, [b]LCI – Lower confidence interval (95%), [c]OCI – Upper confidence interval (95%)

FIGURE CAPTIONS

Fig. 1. Study area depicting the Mwanza Valley with inset showing the Kakoma Health Area sub-area where groundwater quality sampling was undertaken.

Fig. 2. Groundwater model simulated steadystate water table for the Mwanza valley alluvial aquifer system compared to observed head data re-plotted from Monjerezi et al. (2011) and Sehatzadeh, (2011).

Fig. 3. Framework for assessment of pit latrine sanitation risk to groundwater-supply points. The italicised (blue) text items were not implemented herein, but are recommended where data are available.

Fig. 4. Historical development of groundwater supply points and pit latrines within the Mwanza Valley.

Fig. 5. Surroundings of Kakoma Health Area 2015 groundwater quality survey: a) TDS (with elevation contour lines), b) chloride, c) nitrate, d) microbiological contamination detections.

Fig. 6. Kakoma Health Area 2015 groundwater quality survey: plot of observed nitrate versus total iron (with summary statistics relative to an arbitrary threshold concentration of 0.5 mg/L Fe).

Fig. 7. Cumulative percentile plots of surveyed borehole water points versus distance to closest pit latrine for overall datasets, unknown contamination for which water quality data were not available (N/A) (Chapananga dataset) and (a) microbiological contaminated and (b) elevated nitrate subsets for Kakoma subset.

Fig. 8. Bivariate plots of the numbers of pit latrines within varying radial distances of boreholes shown versus observed borehole chloride or nitrate concentrations.

Fig. 9. Box-plots for the four alternative radial/spatial pit latrine occurrence metrics proposed over shown radial assessment areas with Kakoma subset plotted for elevated nitrate and microbiological contamination (or not) and the larger 'not available' (N/A) dataset for which water quality data were not obtained (Chapananga dataset).

Fig. 10. Example area of modelled 1-year and 10-year capture zones and their associated interaction with surrounding pit latrines.

Fig. 1

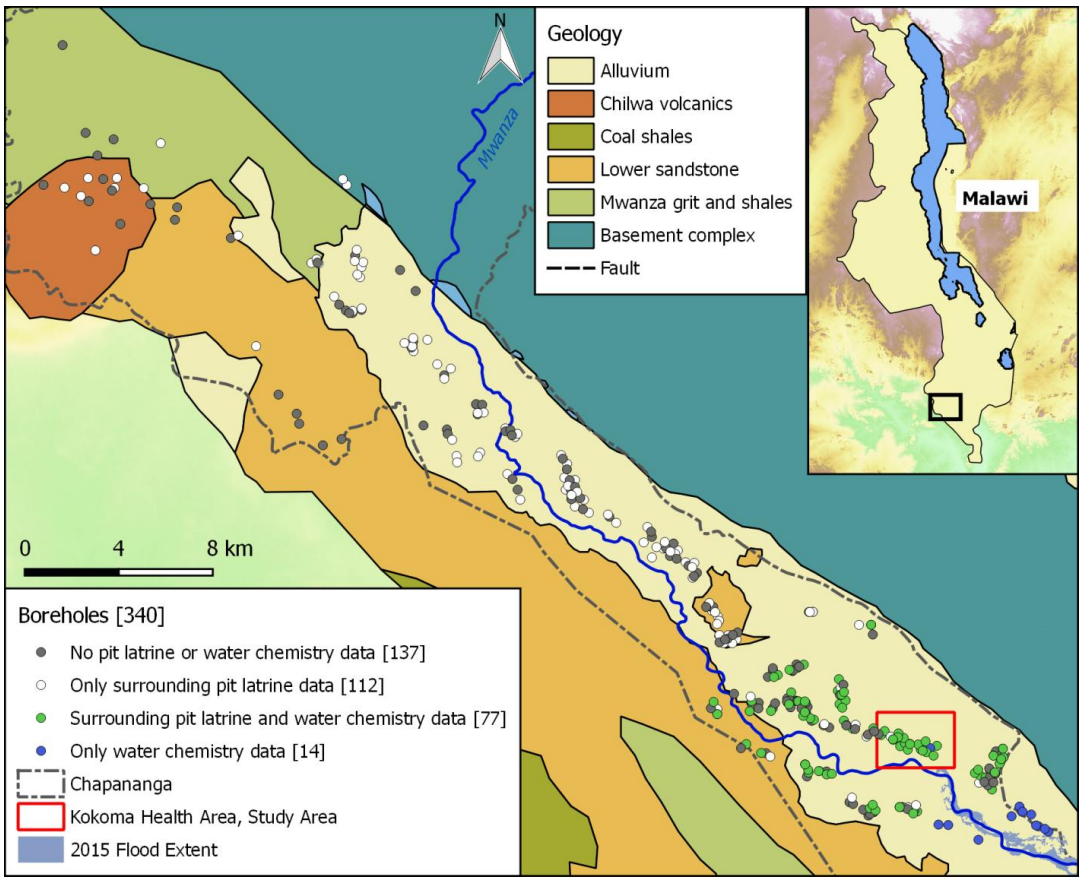

N
Geology
Alluvium
Chilwa volcanics
Coal shales
Lower sandstone
Mwanza grit and shales
Basement complex
Fault
Mwanza
Malawi
0
4
8 km
Boreholes [340]
No pit latrine or water chemistry data [137]
Only surrounding pit latrine data [112]
Surrounding pit latrine and water chemistry data [77]
Only water chemistry data [14]
Chapananga
Kokoma Health Area, Study Area
2015 Flood Extent

Fig. 2

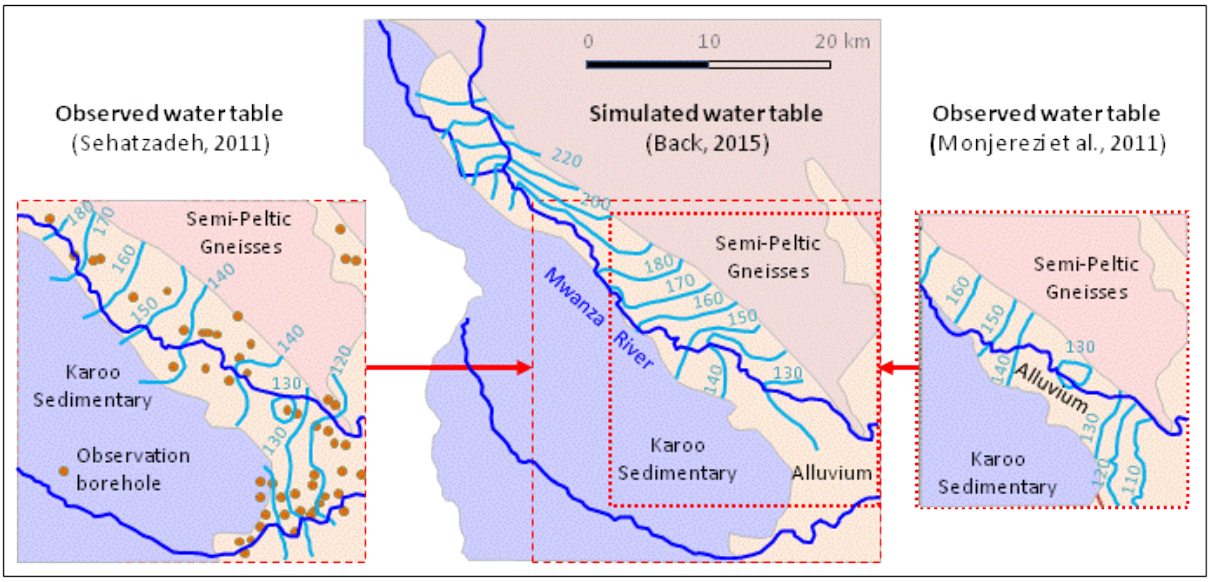

Observed water table
(Sehatzadeh, 2011)
Simulated water table
(Back, 2015)
Observed water table
(Monjereziet al., 2011)
0
10
20 km
Semi-Peltic
Gneisses
Karoo
Sedimentary
Observation
borehole
Mwanza
River
Alluvium
180
170
160
150
140
130
120
220
200
110

Fig. 3

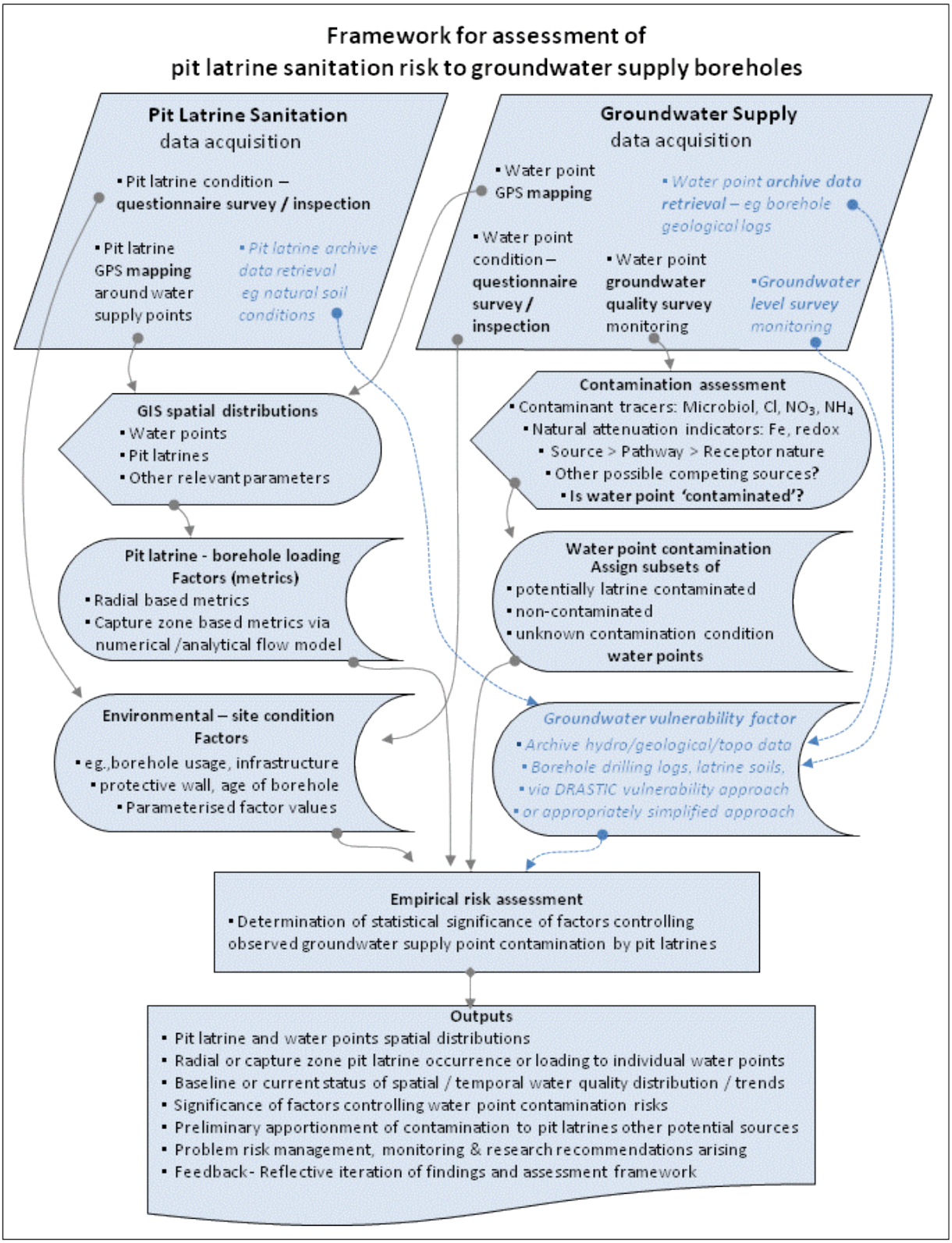

Framework for assessment of
pit latrine sanitation risk to groundwater supply boreholes
Pit Latrine Sanitation
data acquisition
▪ Pit latrine condition – questionnaire survey / inspection
▪ Pit latrine GPS mapping around water supply points
▪ Pit latrine archive data retrieval eg natural soil conditions
Groundwater Supply
data acquisition
▪ Water point GPS mapping
▪ Water point condition – questionnaire survey / inspection
▪ Water point groundwater quality survey monitoring
▪ Water point archive data retrieval – eg borehole geological logs
▪ Groundwater level survey monitoring
GIS spatial distributions
▪ Water points
▪ Pit latrines
▪ Other relevant parameters
Contamination assessment
▪ Contaminant tracers: Microbiol, Cl, NO3, NH4
▪ Natural attenuation indicators: Fe, redox
▪ Source > Pathway > Receptor nature
▪ Other possible competing sources?
▪ Is water point 'contaminated'?
Pit latrine - borehole loading Factors (metrics)
▪ Radial based metrics
▪ Capture zone based metrics via numerical /analytical flow model
Water point contamination Assign subsets of
▪ potentially latrine contaminated
▪ non-contaminated
▪ unknown contamination condition water points
Environmental – site condition Factors
▪ eg.,borehole usage, infrastructure
▪ protective wall, age of borehole
▪ Parameterised factor values
Groundwater vulnerability factor
▪ Archive hydro/geological/topo data
▪ Borehole drilling logs, latrine soils,
▪ via DRASTIC vulnerability approach
▪ or appropriately simplified approach
Empirical risk assessment
▪ Determination of statistical significance of factors controlling observed groundwater supply point contamination by pit latrines
Outputs
▪ Pit latrine and water points spatial distributions
▪ Radial or capture zone pit latrine occurrence or loading to individual water points
▪ Baseline or current status of spatial / temporal water quality distribution / trends
▪ Significance of factors controlling water point contamination risks
▪ Preliminary apportionment of contamination to pit latrines other potential sources
▪ Problem risk management, monitoring & research recommendations arising
▪ Feedback- Reflective iteration of findings and assessment framework

Fig. 4

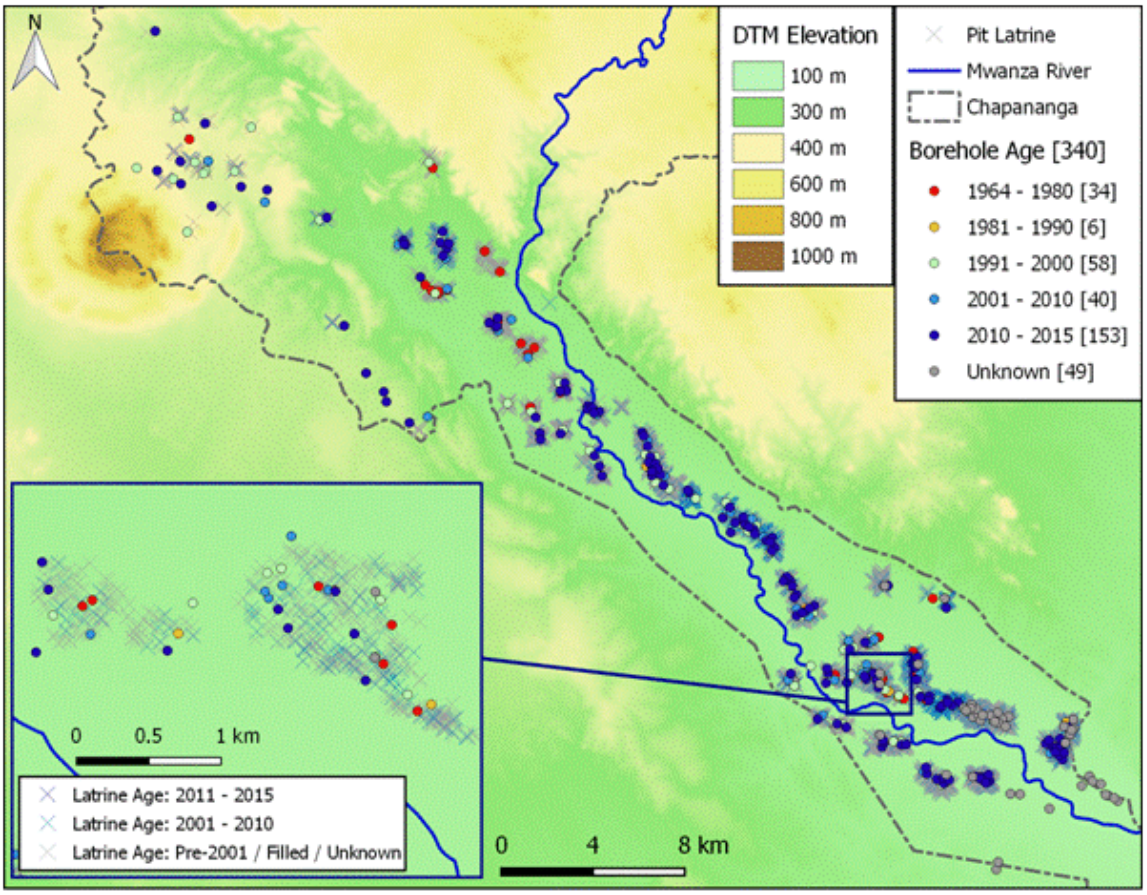
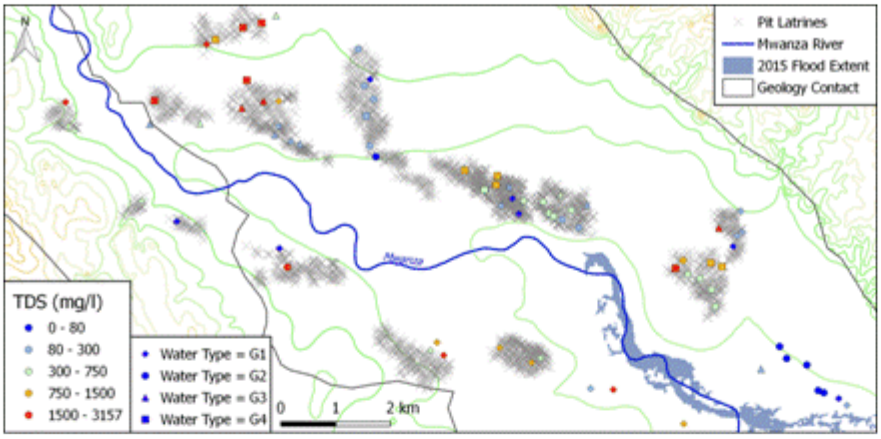
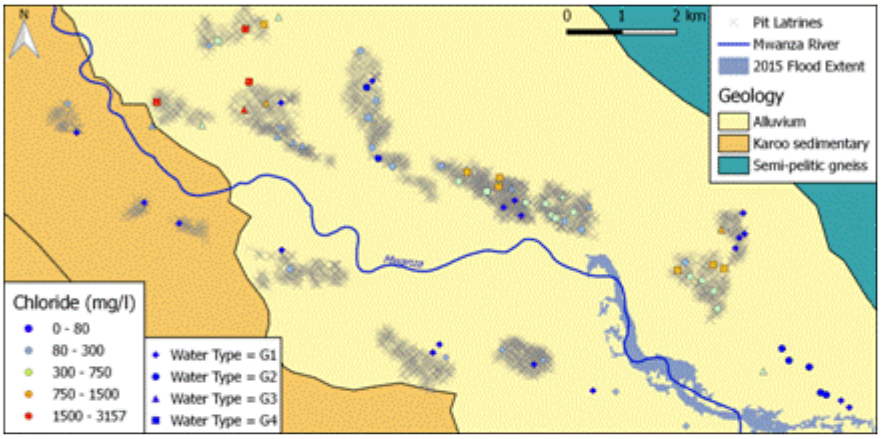

N
DTM Elevation
100 m
300 m
400 m
600 m
800 m
1000 m
Pit Latrine
Mwanza River
Chapananga
Borehole Age [340]
1964 - 1980 [34]
1981 - 1990 [6]
1991 - 2000 [58]
2001 - 2010 [40]
2010 - 2015 [153]
Unknown [49]
0 0.5 1 km
Latrine Age: 2011 - 2015
Latrine Age: 2001 - 2010
Latrine Age: Pre-2001 / Filled / Unknown
0 4 8 km

Fig. 5

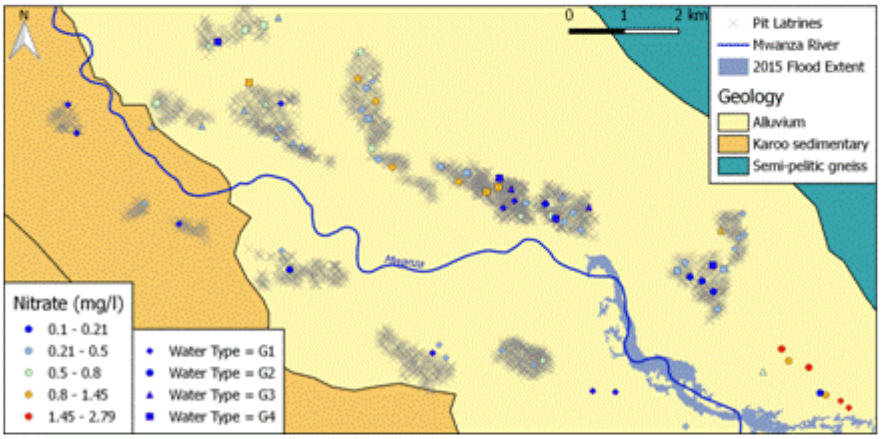
a)


N
Pit Latrines
Mwanza River
2015 Flood Extent
Geology Contact
Mwanza
TDS (mg/l)
0 - 80
80 - 300
300 - 750
750 - 1500
1500 - 3157
Water Type = G1
Water Type = G2
Water Type = G3
Water Type = G4
0
1
2 km

b)


N
0
1
2 km
Pit Latrines
Mwanza River
2015 Flood Extent
Geology
Alluvium
Karoo sedimentary
Semi-pelitic gneiss
Mwanza
Chloride (mg/l)
0 - 80
80 - 300
300 - 750
750 - 1500
1500 - 3157
Water Type = G1
Water Type = G2
Water Type = G3
Water Type = G4

c)


N
0
1
2 km
Pit Latrines
Mwanza River
2015 Flood Extent
Geology
Alluvium
Karoo sedimentary
Semi-pelitic gneiss
Mwanza
Nitrate (mg/l)
0.1 - 0.21
0.21 - 0.5
0.5 - 0.8
0.8 - 1.45
1.45 - 2.79
Water Type = G1
Water Type = G2
Water Type = G3
Water Type = G4

d)

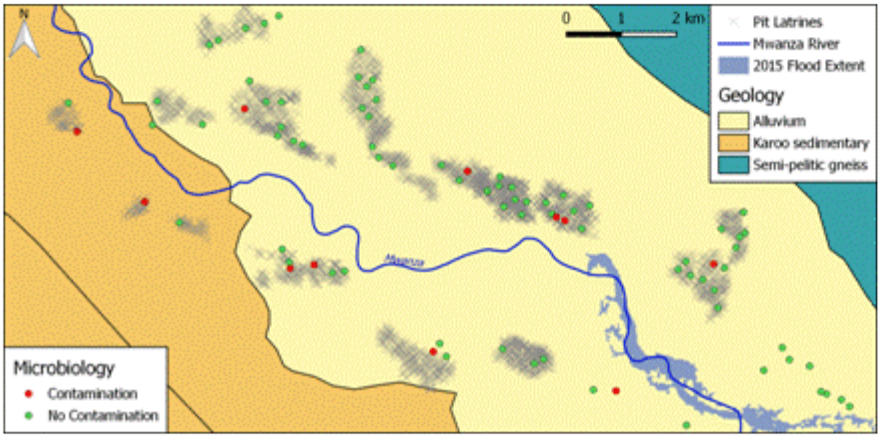
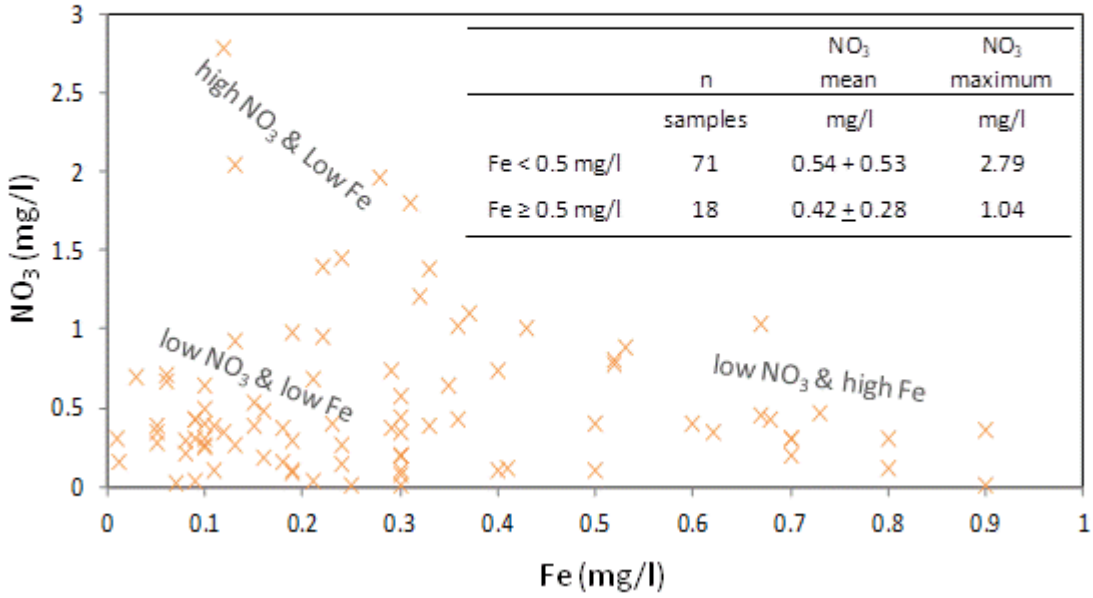

N
0
1
2 km
Pit Latrines
Mwanza River
2015 Flood Extent
Geology
Alluvium
Karoo sedimentary
Semi-pelitic gneiss
Mwanza
Microbiology
Contamination
No Contamination

Fig. 6



	n	NO_3 mean	NO_3 maximum
	samples	mg/l	mg/l
Fe < 0.5 mg/l	71	0.54 + 0.53	2.79
Fe ≥ 0.5 mg/l	18	0.42 ± 0.28	1.04

Fig. 7

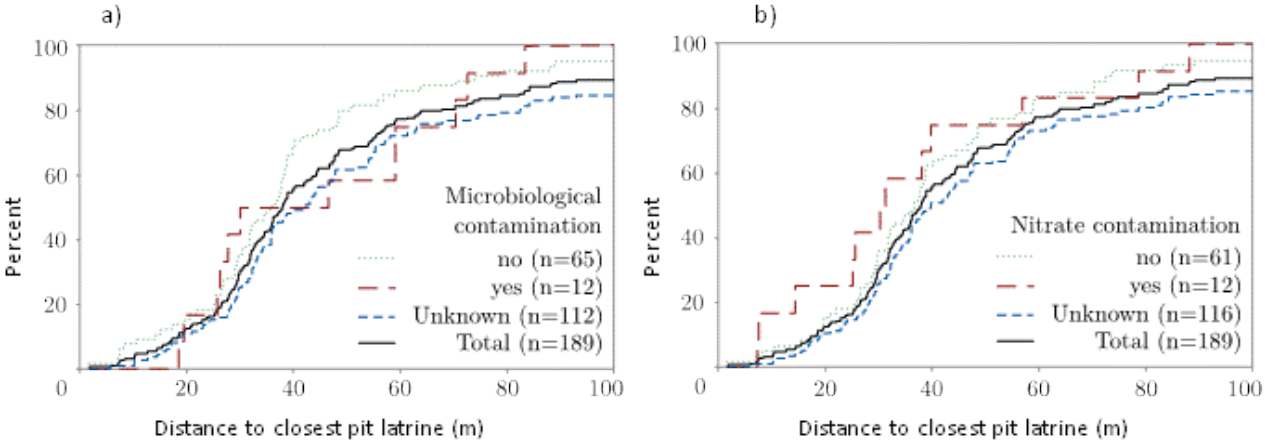

a)
Percent
Distance to closest pit latrine (m)
Microbiological contamination
no (n=65)
yes (n=12)
Unknown (n=112)
Total (n=189)
b)
Percent
Distance to closest pit latrine (m)
Nitrate contamination
no (n=61)
yes (n=12)
Unknown (n=116)
Total (n=189)

Fig. 8

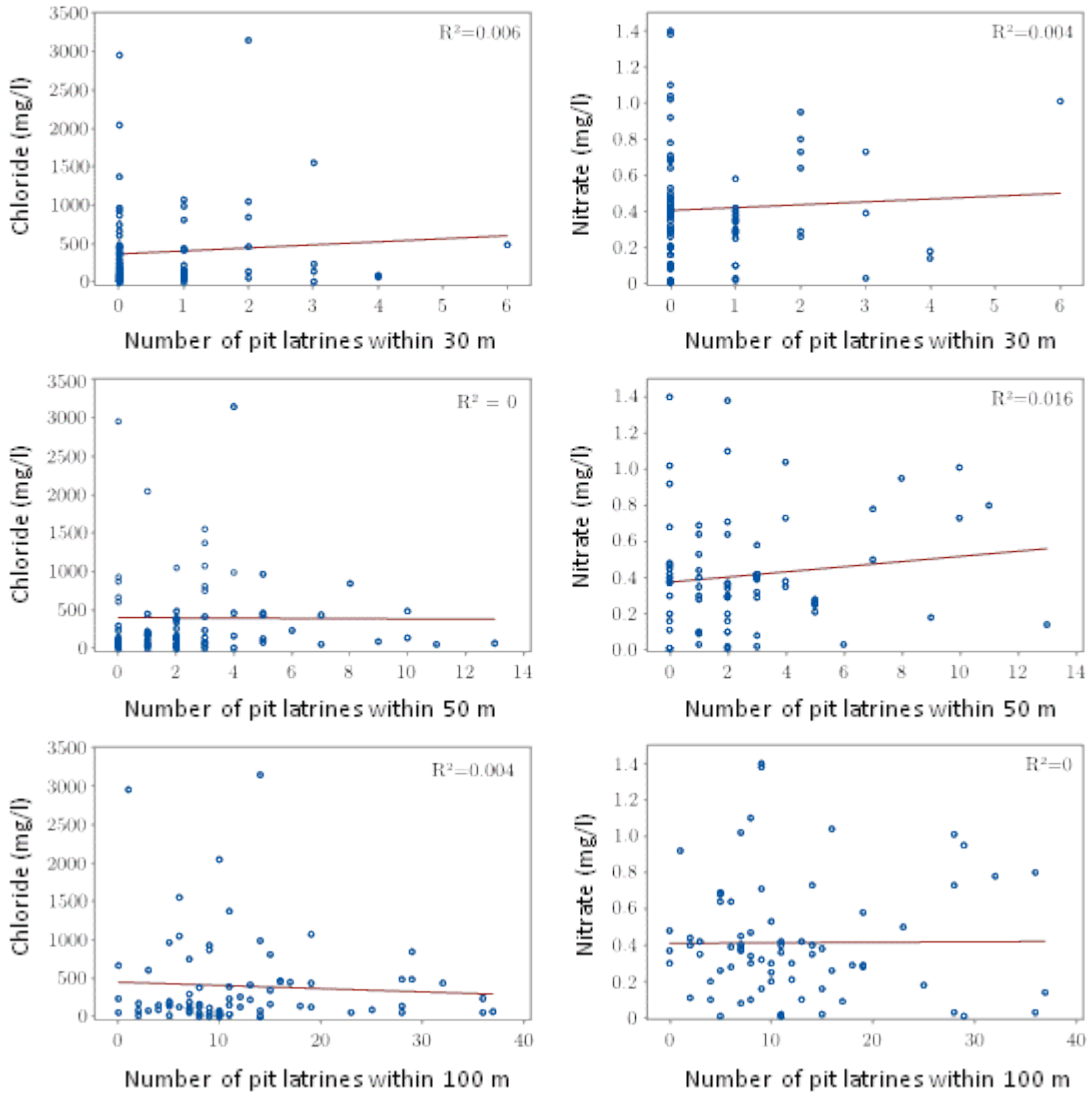

Chloride (mg/l)
R²=0.006
Number of pit latrines within 30 m
Nitrate (mg/l)
R²=0.004
Number of pit latrines within 30 m
Chloride (mg/l)
R² = 0
Number of pit latrines within 50 m
Nitrate (mg/l)
R²=0.016
Number of pit latrines within 50 m
Chloride (mg/l)
R²=0.004
Number of pit latrines within 100 m
Nitrate (mg/l)
R²=0
Number of pit latrines within 100 m

Fig. 9

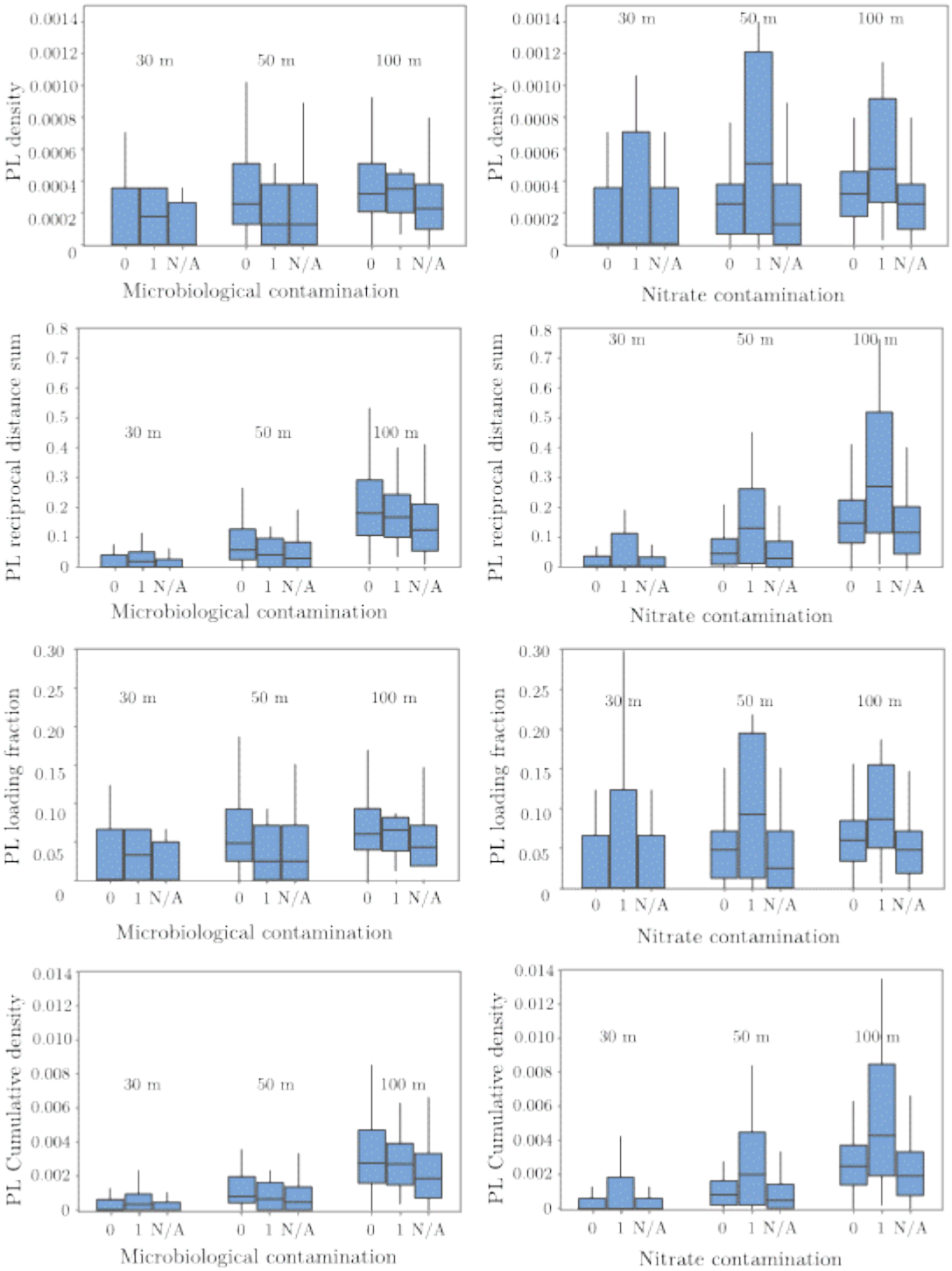
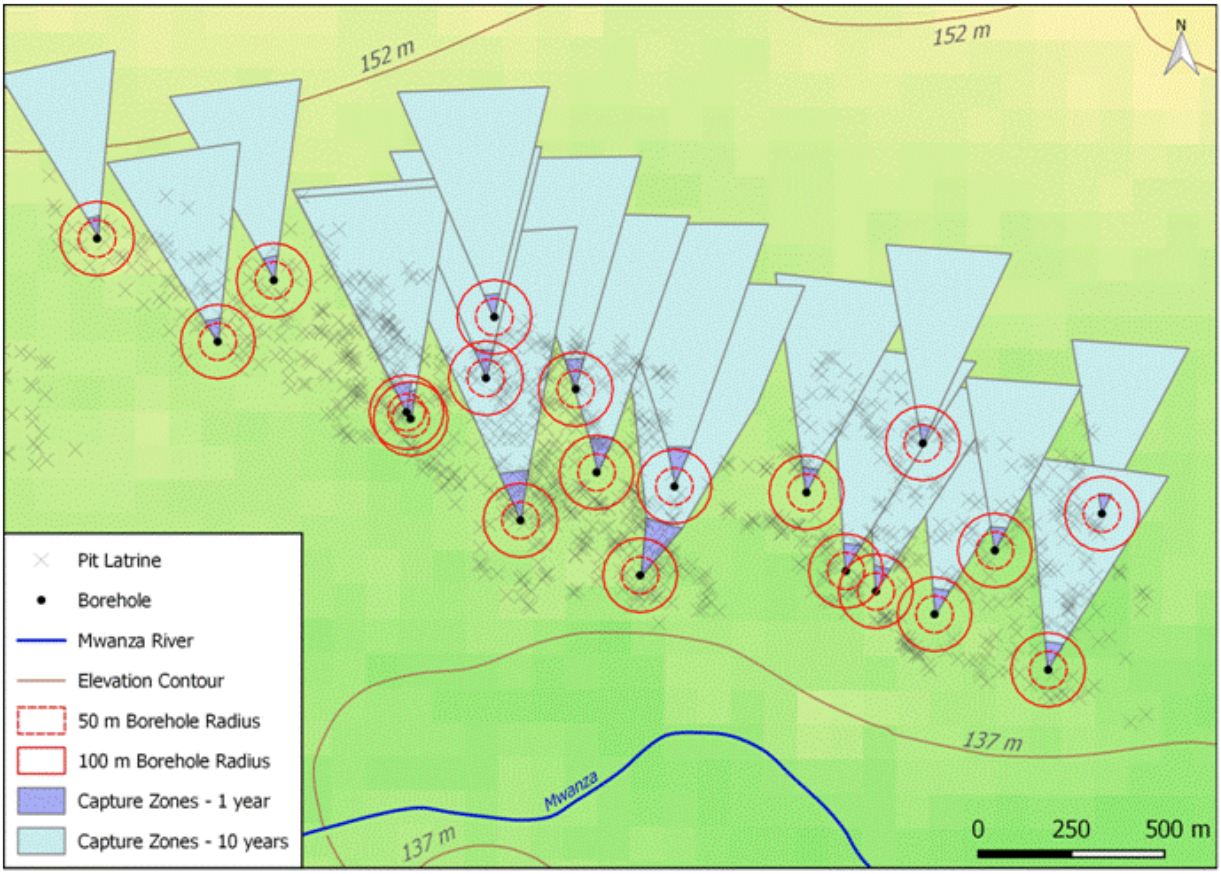

PL density
PL reciprocal distance sum
PL loading fraction
PL Cumulative density
30 m
50 m
100 m
0 1 N/A
Microbiological contamination
Nitrate contamination

Fig. 10


152 m
152 m
N
Pit Latrine
Borehole
Mwanza River
Elevation Contour
50 m Borehole Radius
100 m Borehole Radius
Capture Zones - 1 year
Capture Zones - 10 years
Mwanza
137 m
137 m
0
250
500 m

Science of the Total Environment

Supplementary Material

for

Risk assessment to groundwater of pit latrine rural sanitation policy in developing country settings

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Contents

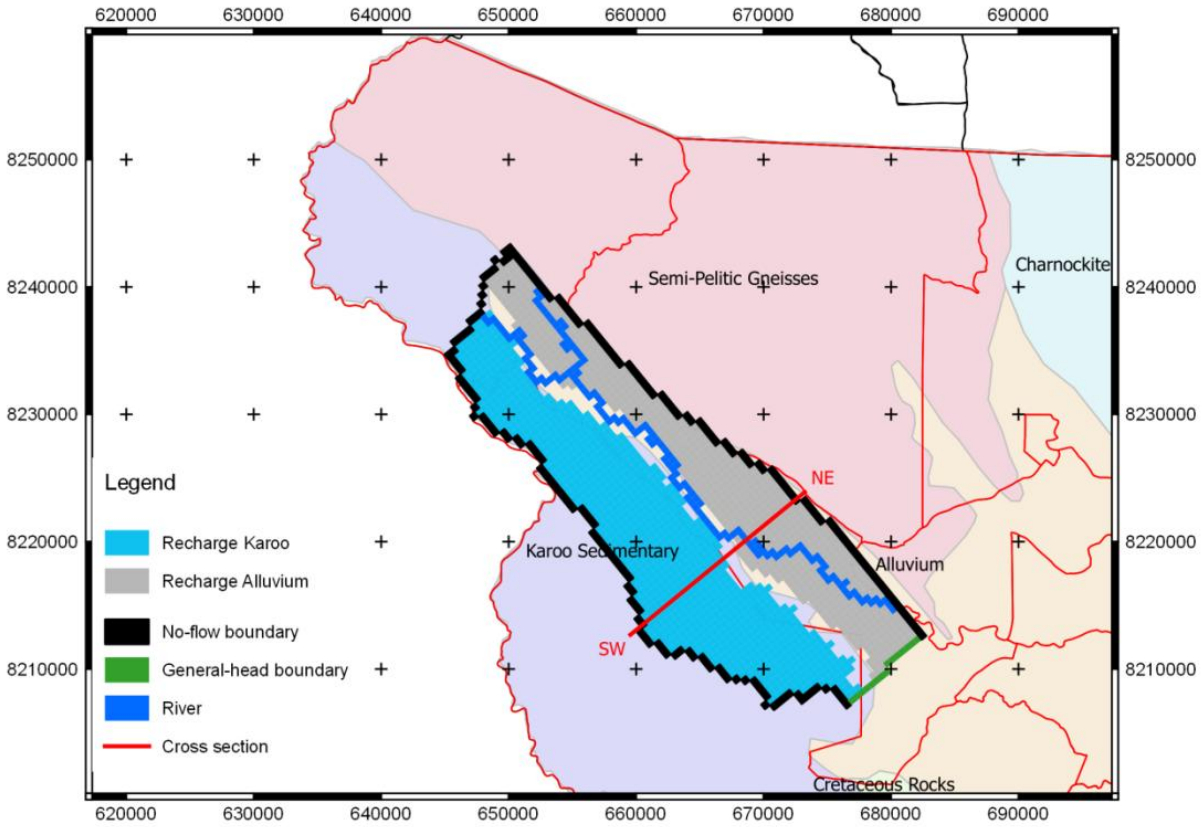

Figure SM-1. Groundwater model geometry and boundary conditions (Back, 2015).

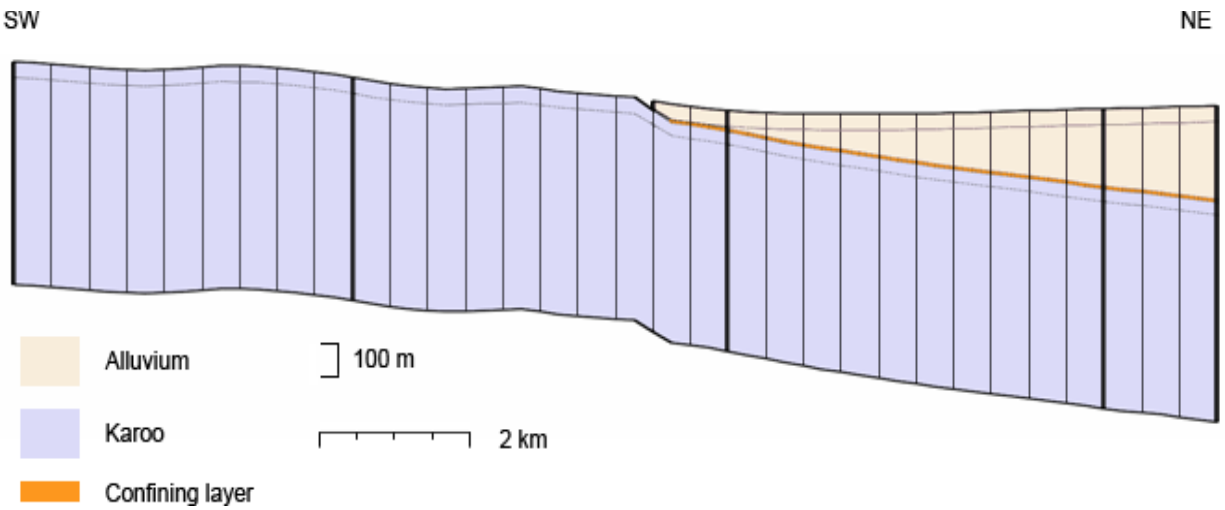

Figure SM-2. Groundwater model cross section.

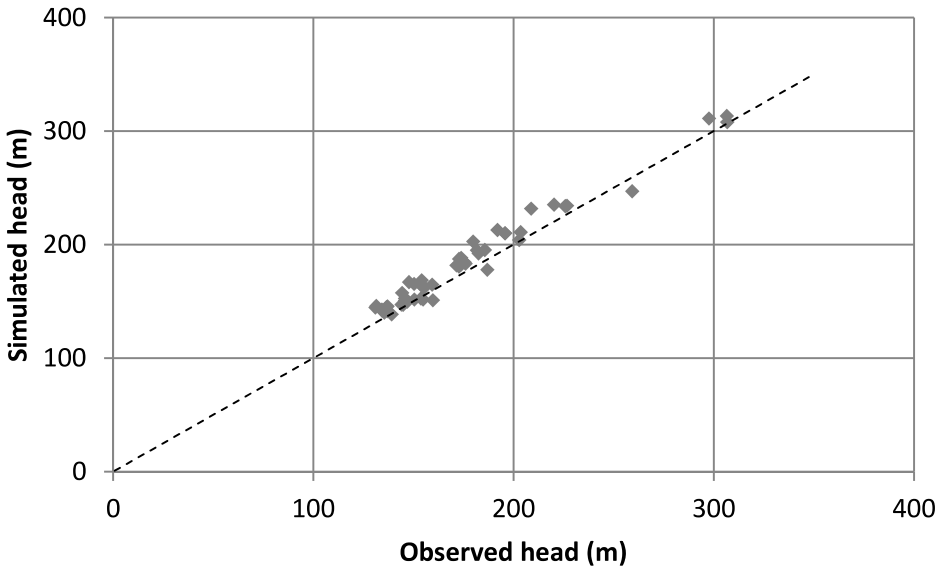

Figure SM-3. Groundwater model callibration: simulated versus observed head values.

Table SM-1. Groundwater model water balance. The groundwater model simulated water balance is tabulated below. River leakage and recharge compose the inflow into the system, with 70% of the influx coming from recharge. The vast majority of the outflow goes into river leakage, whereas the head dependent boundary in the south and the abstraction from wells exert little influence.

		Flow rates (m^3/d)
IN	River leakage	$5.08 \cdot 10^4$
	Recharge	$1.22 \cdot 10^5$
	Total in	$1.73 \cdot 10^5$
OUT	Wells	$7.85 \cdot 10^2$
	River leakage	$1.70 \cdot 10^5$
	Head dependent boundaries	$2.13 \cdot 10^3$
	Total out	$1.73 \cdot 10^5$

Table SM-2. Odds ratios and confidence intervals from logistic regression analysis for chloride and microbiological contamination.

Response	Parameter (* significant)		*p*	OR[a]	LCI[b]	UCI[c]	Unit of change
Elevated chloride concentration	no correlations due to high background signal						
Microbiologically contaminated	Wall in place*		0.011	0.17	0.036	0.83	No-Yes
	Users*		0.042	1.07	1.00	1.14	+50
	Flooded in 2015		0.540	0.61	0.12	3.12	No-Yes
	Water point committee		Quasi-complete data separation				No-Yes
	Stagnant water		0.597	0.70	0.19	2.57	No-Yes
	Permaculture		0.312	0.50	0.12	2.03	No-Yes
	Infrastructure		0.454	0.41	0.10	1.67	1-2[d]
				1.61	0.14	7.26	2-3[d]
	Age of borehole		0.693	1.02	0.94	1.10	+1 year
	Latrines in 1 year capture zone		0.238	0.63	0.19	2.11	+1
	Latrines in 10 year capture zone		0.579	0.99	0.95	1.03	+1
	Distance to closest latrine		0.971	1.00	0.99	1.01	+1 m
	PL density	30 m	0.660	1.37	0.35	5.35	+0.001
		50 m*	0.047	0.08	0.00	1.59	+0.001
		100 m	0.682	0.62	0.06	6.51	+0.001
	PL reciprocal distance sum	30 m	0.756	0.89	0.41	1.93	+0.1
		50 m	0.087	0.47	0.16	1.40	+0.1
		100 m	0.375	0.84	0.56	1.27	+0.1
	PL loading fraction	30 m	0.591	1.27	0.54	3.02	+0.1
		50 m*	0.049	0.25	0.05	1.25	+0.1
		100 m	0.725	0.78	0.19	3.19	+0.1
	PL cumulative density	30 m	0.841	0.95	0.57	1.57	+0.001
		50 m	0.109	0.66	0.36	1.23	+0.001
		100 m	0.404	0.90	0.70	1.17	+0.001

*Significant parameters, [a]OR – Odds ratio, [b]LCI – Lower confidence interval (95%), [c]UCI – Upper confidence interval (95%), [d]1 – bad, 2 – medium, 3 – good

Table SM-3. Odds ratios and confidence intervals from logistic regression analysis for nitrate contamination.

Response	Parameter (* significant)		*p*	OR[a]	LCI[b]	UCI[c]	Unit of change
Elevated nitrate concentration	Wall in place		0.234	2.13	0.61	7.46	No-Yes
	Users		0.226	0.93	0.80	1.08	+50
	Flooded in 2015		0.577	0.64	0.12	3.29	No-Yes
	Water point committee		Quasi-complete data separation				No-Yes
	Stagnant water		0.368	0.53	0.14	2.04	No-Yes
	Permaculture		0.618	1.44	0.35	5.98	No-Yes
	Infrastructure		0.826	1.40	0.30	6.56	1-2[d]
				1.18	0.28	5.03	2-3[d]
	Age of borehole		0.473	1.03	0.95	1.11	+1 year
	Latrines in 1 year capture zone		0.123	1.22	0.95	1.56	+1
	Latrines in 10 year capture zone		0.231	1.02	0.99	1.05	+1
	Distance to closest latrine		0.382	0.99	0.97	1.01	+1 m
	PL density	30 m	0.089	3.08	0.86	10.97	+0.001
		50 m*	0.010	7.34	1.57	34.24	+0.001
		100 m*	0.018	11.06	1.48	82.34	+0.001
	PL reciprocal distance sum	30 m	0.309	1.32	0.79	2.23	+0.1
		50 m	0.063	1.49	0.98	2.27	+0.1
		100 m*	0.037	1.34	1.02	1.76	+0.1
	PL loading fraction	30 m	0.107	1.99	0.88	4.52	+0.1
		50 m*	0.011	3.48	1.30	9.31	+0.1
		100 m*	0.020	4.40	1.25	15.54	+0.1
	PL cumulative density	30 m	0.092	1.40	0.96	2.06	+0.001
		50 m*	0.016	1.443	1.07	1.93	+0.001
		100 m*	0.014	1.26	1.05	1.52	+0.001

*Significant parameters, [a]OR – Odds ratio, [b]LCI – Lower confidence interval (95%), [c]UCI – Upper confidence interval (95%), [d]1 – bad, 2 – medium, 3 – good

Table SM-4. Odds ratios and confidence intervals from logistic regression analysis for iron contamination.

Response	Parameter (* significant)		*p*	OR[a]	LCI[b]	UCI[c]	Unit of change
Elevated iron concentration	Wall in place		0.816	1.14	0.37	3.56	No-Yes
	Users		0.736	1.01	0.94	1.09	+50
	Flooded in 2015		0.71	1.29	0.35	4.82	No-Yes
	Water point committee		Quasi-complete data separation				No-Yes
	Stagnant water		0.863	1.13	0.28	4.62	No-Yes
	Permaculture		0.091	0.28	0.06	1.41	No-Yes
	Infrastructure		0.583	1.94	0.44	8.52	1-2[d]
				1.07	0.29	3.97	2-3[d]
	Age of borehole		0.764	1.01	0.94	1.09	+1 year
	Latrines in 1 year capture zone		0.155	0.63	0.23	1.72	+1
	Latrines in 10 year capture zone		0.897	1.00	0.97	1.03	+1
	Distance to closest latrine		0.757	1.00	0.99	1.01	+1 m
	PL density	30 m	0.121	2.66	0.78	9.07	+0.001
		50 m	0.903	0.90	0.17	4.78	+0.001
		100 m	0.396	0.38	0.04	3.84	+0.001
	PL reciprocal distance sum	30 m	0.471	1.22	0.73	2.02	+0.1
		50 m	0.846	1.05	0.67	1.64	+0.1
		100 m	0.696	0.94	0.68	1.29	+0.1
	PL loading fraction	30 m	0.118	1.89	0.86	4.13	+0.1
		50 m	0.918	0.95	0.34	2.65	+0.1
		100 m	0.427	0.59	0.15	2.29	+0.1
	PL cumulative density	30 m	0.181	1.30	0.89	1.88	+0.001
		50 m	0.543	1.10	0.82	1.48	+0.001
		100 m	0.878	0.98	0.81	1.20	+0.001

[*]Significant parameters, [a]OR – Odds ratio, [b]LCI – Lower confidence interval (95%), [c]UCI – Upper confidence interval (95%), [d]1 – bad, 2 – medium, 3 – good