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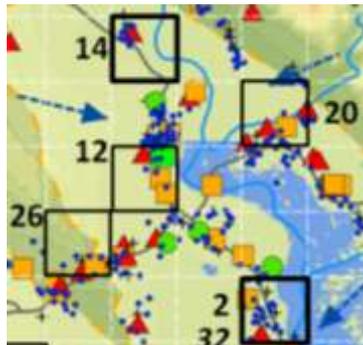
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Graphical abstract

A framework for pragmatic transition from ad-hoc surveys to routinely sampled groundwater-quality network monitoring programmes

Attributes

- ✓ System conceptual model underpins network design
- ✓ *Monitoring Objectives* set to address risks to water quality
- ✓ *Monitoring Potential* of network points scored
- ✓ Network built of priority points to meet objectives
- ✓ Semi-quantitative flexible approach to network design



*Underpinning
Water Safety Plans
& SDG 6 attainment
in the developing world*

Highlights

- Transition to groundwater-quality network monitoring programmes is advocated
- Framework developed for pragmatic groundwater-quality monitoring network design
- Bespoke monitoring objectives scored and prioritised multi-objective network built
- Malawian demonstrated approach provides flexible network-design tool for elsewhere
- Framework development of networks: integral to Water Safety Plans, SGD 6 attainment

**A conceptual model based framework for pragmatic groundwater-quality monitoring network design in the developing world:
Application to the Chikwawa District, Malawi**

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Abstract

Significant need exists in the developing world to transition from occasional groundwater-quality surveys to routinely sampled groundwater-quality network monitoring programmes that provide better safeguard of resources. Networks contribute to the sustainable

management of water resources, are integral to Water Safety Plans, and underpin delivery of Sustainable Development Goal 6. A framework for groundwater-quality monitoring network design is developed that is pragmatic to developing-world needs and its potential application is demonstrated using data from the Chikwawa District – Shire Valley aquifer system in Malawi. The step-wise framework is founded upon a hydrogeological–hydrochemical process-based system conceptual model. The Chikwawa model developed is based upon our interpreted 2012 and archive 2008-9 major-ion survey data; major-ion data often constitute the most easily available datasets in many areas of the developing world. A versatile semi-quantitative approach is adopted which sets bespoke-system ‘Monitoring Objectives’, which are weighted on a scale of 1-5 and then rated against bespoke criteria using a scale of 0-10. This permits development of aggregate ‘Monitoring Potential’ scores at candidate network-point localities. Ideally the process is facilitated by the use of a GIS, although its use is not essential. Monitoring objectives are flexible and typically relate to various perceived risks to groundwater quality; including increasing salinity, anthropogenic activity, etc. The framework, as demonstrated for Chikwawa, allows an incremental build of a prioritised network of points, including a relative estimate of their potential to address the individual monitoring objectives set. The framework methodology is easy to use and adaptable to developing, and indeed developed, world monitoring needs alike. The proposed network for Chikwawa could help pilot transition to a higher resolution national groundwater quality network across Malawi than currently exists. However, attaining spatial monitoring densities suggested, ideally approaching those of European national networks, remains challenging due to the investment required in current infrastructure-capacity alongside the need to develop mechanisms that allow significant running costs to be sustainably met.

Keywords: groundwater quality; groundwater monitoring; monitoring networks; Sustainable Development Goal 6 (SDG 6); Water Safety Plans; Malawi

1 Introduction

Groundwater is increasingly vital to water supply across the developing world. Within Sub-Saharan Africa alone, towards half a billion people rely upon it for basic water needs (JMP, 2012). Advantageously, it is often present in sufficient amount to support proximal water needs without recourse to costly distribution infrastructure (MacDonald and Calow 2009). Moreover, it is usually of suitable quality for both drinking and agriculture, although in some areas, elevated salinity, iron, manganese, fluoride or arsenic can be problematic (Calow et al. 1997; Edmunds and Smedley 2005 MacDonald et al. 2012). However, quantitative groundwater resources data in the developing world are typically lacking, especially groundwater-quality data that are vital to the forewarning of contamination concerns (Foster and Chilton, 2003; MacDonald et al. 2012). Routine monitoring is rare with most available data arising from occasional surveys, or even just sampling at borehole installation perhaps decades ago. Routine operational monitoring and surveillance of drinking-water sources, however, should represent an integral component of 'Water Safety Plans' (WSPs) that aim to 'manage drinking-water quality from catchment to consumer' and ensure provision of water that is safe to drink (WHO, 2005). Our proposed framework methodology that advocates system conceptual model based temporal monitoring of source waters may offer pivotal facilitation of groundwater-resource based WSPs.

In some parts of the developing world, surprising amounts of useful archive groundwater-quality data may exist. Malawi, despite being one of the poorest nations on Earth, has a moderate groundwater-quality archive due to early international development water-resource work (Bath, 1980; Bradford, 1973; Chilton and Smith-Carrington, 1984) subsequently built upon by modern international development-research work (Mapoma and Xie, 2014; Mapoma et al., 2014, 2016; Monjerezi et al., 2011a, 2011b, 2012; Monjerezi and Ngongondo, 2012; Msonda et al., 2007; Pritchard et al., 2008; Sajidu et al., 2008). Datasets (perhaps unpublished or not formally collated) are also held by NGOs (non-governmental organisations) typically arising from large-scale WASH (water and sanitation hygiene) programme activity to address, in particular, Sustainable Development Goal 6 (SDG6) - 'to ensure availability and sustainable management of water and sanitation for all' (<https://sustainabledevelopment.un.org/sdg6>). Datasets relating to such activity may also

be held by the Malawi's Ministry of Agriculture, Irrigation and Water Development (MoAIWD) (termed 'Ministry' herein).

A formal national groundwater monitoring network was unfortunately not established in Malawi under the 1986 National Water Resources Master Plan (Malawi Department of Water, 1986). However, a Ministry of Water Development (2003) report 'Strengthening of the Water Resources Board' that recommended provision of monitoring and statutory provisions for monitoring within Malawi's National Water Policy (2005) (currently in revision) led to MoAIWD initiating a groundwater monitoring network in 2013. This was based upon 35 purpose-drilled monitoring wells in 2009-10 (MoAIWD, 2017a). Unfortunately however, sparse groundwater quality data have been obtained from these wells as periodic sampling was not undertaken due to budgetary constraints (MoAIWD, 2017b).

Whilst it is recognised significant challenges exist, we contend there remains significant need within the developing world to use available archive data to help transition from the status quo of ad-hoc occasional groundwater-quality surveys to routinely sampled groundwater-quality network monitoring programmes operated by government ministry/agency bodies. Networks provide significantly improved safeguard of groundwater resources and their dependencies, should form a cornerstone of WSP implementation and often underpin the formalised statutory acquisition of groundwater quality data used in regulatory decision making (Ward et al., 2004). Networks primarily aim to provide sentinel monitoring of groundwater bodies via a focused, but representative, subset of water points. They aim to establish natural (or influenced) baseline conditions, temporal trends in water-quality and forewarning of contamination concerns (SOGW, 2013, Ward et al., 2004). They should not be seen to replace, but rather compliment, the surveillance monitoring role of more occasional, higher spatial resolution, 'snapshot' surveys of groundwater quality. The regularly monitored temporal network points should, however, act as key tie-in points of a snapshot, linking temporal and spatial monitoring approaches (SOGW, 2013).

Networks may also realise significant spin-off benefits. These include an empowered regulatory body, standardised sampling protocols, improved infrastructure, analytical facilities and procedures, quality assurance - quality control (QA-QC) implementation,

alongside enhanced data reliability, secure archiving, availability and use. They underpin development of more robust WSPs and contribute to attainment of SDG 6 as long-term water management is secured. Establishment of such networks and infrastructure has now become relatively commonplace across the developed world (Belitz et al., 2015; Broers, 2004; Grath et al., 2007; Lesage, 2004; Mendizabal and Stuyfzand, 2009; SOGW, 2013; Ward et al., 2004) and is often facilitated by legislation, notably the Water Framework Directive (WFD) in Europe (Council of Europe, 2000; Grath et al., 2007; Ward et al., 2004).

Whilst our premise is that developing-country establishment of routinely monitored groundwater quality monitoring networks is central to sustainable groundwater resource management, significant inertia exists. This may stem from low prioritisation, a lack of recognition of the importance of monitoring groundwater quality, lack of investment funding (international aid typically does not directly fund this aspect), lack of maintenance income (e.g., water-point licence income) to sustain a network-infrastructure's running costs (incl. chemical analysis), under-resourced regulatory (operational) bodies, absence of political will, extreme event (drought/flood) pressures, and poorly conceived or implemented networks. It is recognised (but is beyond the scope herein) that solutions need to be critically found to these issues that will allow sustained provision of network-based monitoring and practical realisation of network implementation proposed herein.

We outline and demonstrate a 'framework' that allows for pragmatic groundwater-quality monitoring network design. It is intended to be relevant to both developing and developed world contexts, albeit motivated by the apparent needs of the former. Our framework does not purport to be a statistical-based methodology to optimise network point assignment (Esquivel et al., 2015; Kollat et al., 2011; Qin et al., 2016). Rather, it seeks to offer a straightforward, uncomplicated, step-wise framework for network establishment that is versatile and easy-to-use. It aims to use available archive groundwater-quality data/surveys to develop a hydrogeological-hydrochemical process-based system conceptual model understanding that serves as a cornerstone to network design (Ward et al., 2004). We hence demonstrate system conceptual model development for part of the Chikwawa District – Shire Valley aquifer system in Southern Malawi using, not untypical developing-world, groundwater-quality major-ion survey datasets. We demonstrate the utility of the developed framework methodology on our Chikwawa conceptualisation and propose a

rationalised groundwater-quality monitoring network design for that system. Such initiatives form an important contribution to the sustainable management of groundwater resources thereby offering support to the Government of Malawi in their attainment of SDG 6.

2 Methods

2.1 Framework for monitoring network design

At a fundamental level, network design involves the rationalised selection of location points for monitoring that ensure a network is fit for purpose. Initiatives to catalyse network establishment should be uncomplicated, but informed. We propose a straightforward semi-quantitative approach that considers network point placement within the context of prioritised, locally bespoke, monitoring objectives. A step-wise ‘framework’ network-design methodology is outlined in Fig. 1 and explained below with reference to the Chikwawa District demonstration. The framework is not prescriptive, but regarded as flexible, pragmatic and enabling of decision making. It facilitates the development of an evidence-based network, even where data and infrastructure available are perhaps sparse.

2.1.1 System conceptual model development (Steps 1 - 2)

The initiating step involves review of existing groundwater quality datasets (Fig. 1). Ideally these should be predominantly drawn from the sample-point type the network is to comprise. For Chikwawa, acknowledging the aforementioned existing network of dedicated monitoring wells, herein our consideration is based upon a potential network use of hand-pumped supply boreholes that might represent a viable alternative or complementary combined network option. Advantages include village supply boreholes: are regularly used and hence well purged; yield a larger capture zone ‘averaged’ sample; represent the abstracted scale of groundwater supply; are easily sampled; and used in existing groundwater-quality surveys of the area (citations earlier); and form a significant

community focal point and economic resource that is typically valued and protected (Rivett et al., 2018). Moreover, supply boreholes, albeit often large-volume abstractions, are commonly used in national networks for many of the above reasons (Ward et al., 2004).

Archive data underpin the Step 2 development of a process-based system conceptual model that is founded upon the available geological-hydrogeological understanding, especially quantified aquifer parameters, groundwater flow and the identification of system recharge, through-flow and discharge areas including receiving surface waters and abstractions. Such data, if in short supply, should be recognised as priority needs to be obtained alongside any network development. Groundwater chemical-quality data are typically superimposed upon the available flow understanding to generate a hydrogeological process-based – hydrochemical, pollutant, microbiological – transport conceptual model understanding. This conceptualisation is central to the informed deployment of a network. It is regarded as ‘live’, to have uncertainties and to be iteratively developed and adaptive to changes in the environment monitored or perceived. Furthermore, it is to be regarded as a ‘whole system’ conceptualisation drawing upon supporting datasets relevant to monitoring objectives set.

The use of conceptual models to underpin network design has been invoked by, for example, SOGW (2013) in the US and Ward et al. (2004) in the UK. The latter underscore the holistic needs of conceptualisation and associated network development warranted under the European WFD. It is nevertheless recognised that in developing-world contexts data availability to underpin a conceptual model build can be very limited. Even so, what conceptualisation of the system that can be made should be employed and used as a preliminary basis for network design.

2.1.2 Monitoring objectives (MOs) and Monitoring Potential (MP) definition (Step 3)

There should be a clear understanding of monitoring priorities in order to define the monitoring objectives (MOs) against which a network’s performance may be evaluated. Monitoring objectives may be set at various levels. A national regulatory body assumes an overarching mandate to provide monitoring that complies with statutory and non-statutory commitments, obtain data to enable better national management of its water resource and environment, and the establishment of baseline conditions, trends in groundwater quality and early warning of groundwater pollution (Ward et al., 2004). Our focus is upon the

development of MOs that more specifically express such overarching objectives at the local regional aquifer system level (or ‘groundwater quality monitoring unit’ in WFD parlance) (Ward et al., 2004). As monitoring fundamentally aims to safeguard a groundwater resource, MOs usually relate to specific risks to resource value. For instance, to monitor encroaching salinity, areas of high resource use, (polluted) surface-water interaction, threat posed by a type of land use, or the quality of ‘fresh’ groundwater expected within key recharge areas.

Step 3 develops bespoke MOs for the conceptualised system, ranked by expert-opinion allocated weights (W_o) varying from 1 to 5, least to the most important. Sub-criteria are then developed for each MO to allow a judged rating (R_o), marked from 0 to 10, of potential attainment of that objective for a particular locality. Expert-opinion, alongside various group consensus methods (Hsu and Sandford, 2007), may be employed to define objectives, weighting and rating values. A Score (S_o) for that MO is then calculated from the product of the weighting and rating allocated:

$$S_o = W_o \cdot R_o \quad \text{Eq. (1)}$$

An overall Monitoring Potential (MP) score is then generated for each assessed locality based on the summation of the individual scores for each MO:

$$MP = \sum S_o = S_{o1} + S_{o2} + S_{o3} + \dots = W_{o1} \cdot R_{o1} + W_{o2} \cdot R_{o2} + W_{o3} \cdot R_{o3} + \dots \quad \text{Eq. (2)}$$

Such a scoring approach is akin to that used by the widely used groundwater vulnerability assessment model, DRASTIC (Aller et al., 1987) and later variants (Panagopoulos et al., 2006; Shirazi et al., 2012). It represents a novel application of such a scoring system to monitoring network design and, as varying subjectivity may be invoked within the ranking and scoring, offers a semi-quantitative approach to network design affording a relative prioritisation of points. With time, collected monitoring programme data may be used to refine the monitoring objectives and rating criteria that may hence become less subjective.

MOs and associated criteria were developed bespoke to the Chikwawa system (Table 1), but illustratively cover a (non-exhaustive) range of common groundwater quality concerns. Criteria rating metrics aim to promote consistency, but should be regarded as guiding and

adaptable within the local context. Alternative MOs can be envisioned though. For instance, ‘MO2 – Monitor hydrochemical-groundwater type to protect supply quality’ could be adapted to a focused objective on meeting drinking water quality regulatory criteria and use ratings based around Water Quality Index (WQI) values (Sener et al., 2017). MOs could also be set to monitor water quality in recharge areas, discharge areas, or areas of high or low (e.g. confined aquifer) groundwater vulnerability where the segregation is based more upon the flow system conceptualised (Ward et al., 2004). As our conceptualisation is underpinned by flow understanding, Table 1 MOs indirectly capture some of these aspects. For instance, higher rating MO2 criteria typically equate to recharge area groundwater quality. Notably, public-health monitoring of groundwater quality could be more explicitly recognised, in say a dedicated MO focused on acute risks. Presently risks associated with acute pathogen incidence (greatest at the start of the rainy season) are incorporated within the more general MO5 to monitor anthropogenic – urbanisation threats.

2.1.3 GIS use (Steps 4 – 5)

Step 4 recommends the use of a GIS (Geographic Information System) to facilitate visualisation and quantitative interrogation of the system conceptual model and network development. Some iterative development of Steps 2-4 (Fig. 1) may occur with population of GIS databases probable within the earlier Steps. The Step 4 GIS multi-layer representation comprises individual data layers displaying spatial variation of relevant system attributes, particularly data layers relevant to MO decision making. ArcMap (version 10.2.2, part of the ArcGIS package (www.arcgis.com), provided accurate spatial plotting and ready comparison of multiple datasets in the Chikwawa case (noting the equivalent could have been undertaken in free GIS software such as QGIS (<http://www.qgis.org/en/site/>)). Relevant single or amalgamated datasets (layers) were selected within the GIS that were bespoke to each MO to allow Rating values and Scores for each MO to be estimated at a geographic grid-cell locality. This was either via enumeration of data within grid cells within the GIS, or via visual inspection of GIS outputs of multi-dataset (layer) spatial map backdrops specific to assessment of each MO. Combination of these scores allows a MP estimate to be made and Step 5 completion. The use of a GIS is not essential, but significantly aids the network development process.

2.1.4 Finalised network design and implementation (Steps 6 - 9)

Step 6 involves monitoring point prioritisation and finalisation of network design. Our demonstrated procedure for Chikwawa was to score an expanding network of possible points; a pragmatic approach that accommodates potential network expansion or contraction, due to say financial constraints. MP scores for points added to the network at a later stage should generally decrease as higher MP score localities have been assigned preferentially. For a balanced network, consideration is given to whether higher weighted MOs are sufficiently addressed already and further network-point allocation should be to lower MP scoring localities, thus addressing under-represented MOs. Hence, the degree to which individual MOs are addressed needs to be considered during network expansion. Graphed bar charts that plot MP values and their breakdown MO scores and plots of cumulative MO score aggregated for the network are demonstrated to provide a convenient metric of the growing network of incrementally numbered points assigned (see later figures).

To allow consideration of spatial coverage afforded by a network, MO7 qualitatively assesses the degree to which an additional monitoring point would further 'Enhance spatial coverage of the monitored domain' (Table 1). Uniquely, compared to other MOs, an assigned score for MO7 depends upon the distribution of points already assigned. Hence if a point is reprioritised within the network order, the scores for MO7 require re-calculation for this and other points. Scoring of MO7 was achieved via visual inspection for the Chikwawa case as 'obvious spatial gaps' were gradually filled within the network, albeit recognising the potential for statistical techniques to be invoked to quantitatively optimise spatial coverage.

Step 7 concerns implementation of the network recognising that iteration may occur due to practicalities faced. Step 8 involves the development to a routinely sampled monitoring network and supporting infrastructure (e.g., analytical laboratories, protocols and data processing, archiving and use). These are best assessed over several years to ensure network sustainability. Finally, Step 9 recognises the necessary feedback required on network performance and iterative update of the system conceptual model and optimisation of the monitoring network and MOs to manage the issues of concern. The detail of these steps is beyond the scope of this paper.

2.2 Case study setting: Chikwawa District, Southern Malawi

The case study area occupies 1500 km², encompassing 30% of the predominantly rural Chikwawa District (Fig. 2). Most of the 500,000 population are subsistence farmers, often living on less than \$0.50 a day with a mean life expectancy of just 45 years (Water for People, 2017). The lower Shire Valley alluvial plain forms a low, gently declining topography with steep granitic escarpment at the eastern valley boundary reaching some 1000 m asl (above sea level). Towards the west, gently rolling topography rises gradually to 500 m asl. The semi-arid climate is characterised by some of Malawi's highest temperatures, lowest rainfall and greatest annual moisture deficits. Lowlands are prone to drought, but also flooding during the intense wet season.

The valley lies within the intensely faulted East African Rift System (Habgood, 1963; Fig. 2). The underlying Basement Complex comprises Precambrian to Lower Palaeozoic high-grade charnockitic granulites and biotite-hornblende gneisses and forms the elevated escarpments. It is unconformably overlain, or faulted against, thick (km-scale) sequences of Karoo sedimentary rocks (Bradford, 1973). The upper Karoo underlies much of the alluvium and comprises grits, (arkose) sandstones, shales, mudstones and marls and tends to be calcite-cemented and indurated; primary porosity is hence low and permeability relates to secondary fracturing or bedding plane enlargements (Smith-Carrington and Chilton, 1983). Some hydrothermal fault rocks are also associated with Karoo boundary faults (Habgood, 1963). The piedmont plain floor comprises large volumes of Quaternary alluvium sediments from fluvial deposition associated with rivers debouching from escarpments (Habgood, 1963). The alluvium is poorly defined (as most boreholes only partially penetrate it), but thickens eastwards up to c. 150 m thickness at its fault-bounded contact with the eastern escarpment (Smith-Carrington and Chilton, 1983). Sedimentary successions comprise highly variable sequences of clays, silts, sands and occasional gravels. Overall, finer-grained sediments predominate with coarser deposits typically closer to the escarpments (Smith-Carrington and Chilton, 1983).

2.2.1 Groundwater

The weathered basement aquifer is typically low-yielding, but nevertheless may widely form locally important supplies with yields dependent upon the degree of local weathering and fracturing (Chilton and Smith-Carrington, 1984; Robins et al., 2013). Whilst the weathered zone is most developed over the plateau areas (c. 15-30 m thick or more), a relatively thick weathered zone often builds up in the Basement Complex around the escarpment-base interface with the alluvium (Smith-Carrington and Chilton, 1983). The low (fractured) porosity storage means a water-saturated thickness of 10 m is necessary for sustained water supply.

The higher yielding, more porous, alluvial aquifer is most productive close to the marginal valley escarpments (Smith-Carrington and Chilton, 1983). Semi-confined conditions may exist when clay horizons are frequent. Towards the escarpment where the alluvium thins, groundwater may be largely derived from the underlying weathered basement in hydraulic continuity. Despite the frequency of clay-silt sediments, the alluvial aquifer still represents the most significant source of groundwater. Yields can be over 10 l/s (Mapona and Xie, 2014; Smith-Carrington and Chilton., 1983), although average test yields in both weathered basement and alluvium have often been reported at around 1 l/s; some low values may relate to poor borehole design (Bradford, 1973; Smith-Carrington and Chilton, 1983). Still, these yields are usually adequate for handpump domestic supplies which only require 0.25-0.5 l/sec (Water Aid, 2013).

Throughout the Chikwawa District, rural village communities are entirely reliant upon groundwater boreholes or gravity-fed spring systems for a safe domestic water supply. Some 3000 groundwater points are presently mapped in the study area of which 52 are spring supplies (see later figure). Recent years have witnessed significant development of village supply wells alongside improved sanitation provision (Water for People, 2017). Most boreholes do not exceed 40-50 m depth and are fitted with surface-mounted (Afridev) hand pumps (Water Aid, 2013).

2.2.2 Groundwater quality monitoring

Hydrochemical studies of the Chikwawa District, tabulated by Mapoma and Xie (2014), confirm natural hydrogeochemical processes often control groundwater quality. The most prominent issue is salinity (Bath, 1980; Mapoma and Xie, 2014; Monjerezi et al., 2011a, 2012). Other hydrochemical or microbiological water quality data are often limited to testing undertaken at well commissioning or in response to perceived water-quality concerns. For the former, MoAIWD guidelines state that each new water point must be tested for microbiological contaminants, salinity and major-ions prior to commissioning for use. The potentially frequent reality is that these tests are not always completed and the borehole is commissioned for use immediately after installation. Where tests are undertaken, results can sometimes be returned weeks or even months after the borehole has already been fitted with a pump and commissioned for use. Moreover, where poor water quality is identified, results are not always appropriately acted upon.

Of the aforementioned national network of 35 purpose-drilled monitoring wells designated for Ministry periodic monitoring of groundwater, only two occur within the Chikwawa District, with only one within the study area at Chikwawa Boma (Fig. 2). This monitoring well was sampled (with a bailer) on just four occasions, all in 2013; a high salinity water quality was evident. These monitoring wells form a subset of the national water quality monitoring network of 195 points, the remainder being surface water or pollution control (effluent) sample points. Analysis is undertaken in Ministry laboratories (3), however, periodic monitoring of water-quality has not transpired. Key issues identified by the Ministry (MoAIWD, 2017a,b) are: monitoring point exact coordinates and registration system are not established, modern computerized database management systems are not developed, the Water Quality Division in the North has no laboratory building, maintenance budgets are insufficient for equipment repair or purchase of reagents for analyses, sampling and analysis activities cannot be undertaken regularly due to budget, transport, and staffing constraints (MoAIWD had a 66% vacancy rate in 2013-14 due to shortfalls in human resources and finances). Whilst our focus is not on these issues *per se*, they clearly represent considerable barriers to network establishment that need to be addressed.

2.3 Groundwater-quality surveys undertaken

We utilise two of our groundwater quality surveys to underpin the conceptual model build illustrated. We primarily present our (unpublished) 2012 dataset collected by Fallas (2012) involving 59 boreholes sampled that ultimately yielded 50 points for data analysis. It represents a fairly typical ad-hoc developing world survey of largely major-ion water quality. We also supplement this with a subset of (published) data by Monjerezi et al. (2011a) collected in 2008-9 covering 105 handpump village supply boreholes within and slightly extending the study area across the valley plain (supplementing the 2012 survey that occurred in the wet season where access to remote or flood-influenced areas was difficult within study timeframes). The reader is referred to Monjerezi et al. (2011a) for their field sampling and analysis methods; those used by Fallas (2012) are outlined below.

Samples were collected following standard sampling procedures (APHA (American Public Health Association), 2005: ISO 5667-11:2009) from village boreholes, using 100 ml sterilized polyethylene bottles. Samples were kept refrigerated to 4°C following collection until laboratory analysis at the University of Strathclyde (i.e., were shipped to Scotland). The pH, TDS and EC of the samples were measured using meters in the field. Major cation (sodium, calcium, magnesium, potassium) and anion (chloride, sulphate, Nitrate, fluoride) concentrations were determined by Ion Chromatography (IC) using a Metrohm 850 Professional IC equipment. The bicarbonate content was determined by alkalinity titration with acid following the standard APHA Total Alkalinity method (APHA 1999: 2320 B). Only samples with charge-balance errors falling within $\pm 10\%$ were used for data analysis which resulted in discard of 9 samples. Whilst some other hydrochemical data are available (cited literature, including Monjerezi et al. (2011a)), the conceptual model build focuses upon the major ion data. Major-ion data often constitute the most easily available datasets in many areas of the developing world.

3 Results

Presentation of the results is structured so as to illustrate the stepped implementation of the framework set out in Fig. 1 for the Chikwawa case.

3.1 Survey results and conceptual model development (Steps 1 - 2)

3.1.1 Groundwater quality 2012 survey – summary data

The 2012 survey data compared favourably with national water quality standards for boreholes and protected shallow wells (MBS, 2005) (Fig. 3a; Supplementary Data Table SD1). Standards are only exceeded by 8% of samples for chloride and 14% for sodium and by 6% for EC and TDS (Total dissolved solids). WHO (2017) health-based guideline values are not exceeded, although for sodium 50% of samples exceed noted taste thresholds, while for chloride and sulfate 28% and 6% exceed taste thresholds respectively (Table SD1). The WHO recognises that palatability of water with a TDS <600 mg/l (approximates the study median) is considered good, however, water becomes increasingly unpalatable above a 'brackish' threshold of 1000 mg/l that was exceeded by 28% of samples up to a TDS of 3180 mg/l. These data suggest between c. 28 to 50% of groundwater samples are expected to taste salty. This is primarily driven by elevated sodium and chloride contents, while noting that bicarbonate may constitute a significant proportion of the TDS in samples.

Major-ion dominance of HCO_3^- , Cl^- , Na^+ and $\text{Ca}^{2+} / \text{Mg}^{2+}$ permitted sample grouping into hydrochemical facies type based upon their dominant anion and cation components. Defined hydrochemical facies were Group 1 of Ca/Mg- HCO_3 type, Group 2 of Na- HCO_3 type, Group 3 of Na-Cl type and Group 4 of Ca/Mg-Cl type. A Piper plot illustrates samples were predominantly within Groups 1 to 3 (Fig. 3b). Group mean data indicate the chloride-based Groups 3 and 4 exhibit similar compositions excepting the proportion of Ca-Mg relative to Na (Fig. 3c). Group 1 is most contrasting due to its much lower Cl, SO_4 , Na and TDS contents. Group 2 is characterised by increased Na alongside the lowest Ca-Mg found.

3.1.2 Hydrochemical process controls

Saturation indices were determined for each groundwater sample with respect to key mineral phases (Supplementary Data, Table SD2). Groundwater is predicted to be supersaturated for all groups with respect to the common carbonate minerals calcite and dolomite, but undersaturated with respect to anhydrite, gypsum and halite evaporites.

Hence all waters are more likely to precipitate carbonate minerals and to dissolve evaporites. Halite is the most undersaturated and is more likely to be dissolved than gypsum and anhydrite. Group 2 is most saturated with respect to carbonate minerals and hence most likely to precipitate dolomite and calcite. Group 1 waters are the most undersaturated with respect to evaporites and may dissolve these minerals if present.

Group 1 samples represent the youngest and least mineralised groundwater with high bicarbonate along with calcium, magnesium and some sodium, but low concentrations of chloride and sulfate. These signify silicate weathering of minerals including feldspars, amphiboles, and other ferromagnesian minerals common within the Malawi basement complex; a plot of $\text{HCO}_3^- / \text{Na}$ versus Ca / Na supports this (Fig. 4a). Higher sodium relative to chloride within Group 1 could be attributed to silicate weathering of sodium-rich feldspars common within older escarpment rocks. Further analysis by Monjerezi et al. (2011a) confirms Group 1 (their C3) samples mainly result from H_2CO_3 weathering of aluminosilicate minerals by water super-saturated with CO_2 .

Group 2 groundwaters have decreased Ca/Mg, but increased Na consistent with cation exchange on clay-rich lithologies. Where increase in bicarbonate occurs alongside Na, the latter may originate from further silicate weathering of feldspar minerals. Cation exchange significance is accessed via Fig. 4b plotting of combined Ca + Mg (as meq/l) corrected by subtraction of $\text{HCO}_3^- + \text{SO}_4^{2-}$ (removing the possibility of ions derived from other processes, e.g., silicate weathering, gypsum dissolution) versus combined Na + K corrected by subtraction of Cl (removing the possibility of Na from halite dissolution) (after Monjerezi et al. (2012)). The Group 2 high correlation trend line of gradient -1 supports cation exchange exerts significant control.

Within Group 3, increased Na and Cl, but also SO_4^{2-} relative to other ions, suggest that evaporite mineral dissolution is important. Increased amounts of all ions relative to Group 1 could be indicative of enhanced concentration through evaporation effects on shallow groundwater. Halite dissolution is explored through the Na versus Cl plot (Fig. 4c) that displays a near 1:1 linear Na:Cl (equivalents) relationship for Group 3 samples. Group 4 samples plot in a similar area indicating the significant influence of their high Na - Cl

contents. Group 2 exhibits less correlation that is ascribed to increased sodium from cation exchange; its position between Groups 1 and 3 symptomatic of their mixing.

Surface evaporation of groundwater is accessed via Na/Cl ratio versus TDS plots (Fig. 4d). If evaporation is significant, the ratio remains constant with increased TDS. A ratio near unity may also be ascribed to halite dissolution. Group 3 displays these trends supporting halite dissolution is important with evaporative concentration of Na and Cl influencing groundwater at shallow depths; highest TDS waters may be those most influenced by evaporative concentration. Monjerezi et al. (2011a) conclude similarly, noting also the potential importance of sulfate dissolution from evaporite salts.

3.1.3 Spatial observations and hydrogeological conceptualisation

The spatial distribution of groundwater types and contoured TDS observed in 2008-9 survey (Monjerezi et al., 2011a) reasonably compare to 2012 (Fig. 5). Group 1 comprises Ca/Mg-HCO₃ type, low-TDS groundwater of good quality; Group 2, Na-HCO₃ type groundwater of higher TDS and usually good quality; and Group 3, Na-Cl type groundwater characterised by, high TDS, brackish to moderately saline water. Detailed comparison of the groundwater type occurrence by each survey is provided within the Supplementary Data – Box SD1 and Fig. SD1. The sparser 2012 survey (50 points) reasonably compares to the 2008-9 survey (105 points) acknowledging the limited 2012 monitoring undertaken (and hence data overlap) in the far west and south.

Spatial results require interpretation within the hydrogeological conceptualisation available. A difference of 4 years between the two surveys is not anticipated to cause major changes in groundwater quality in such a low recharge, low gradient, alluvial valley aquifer system. Area recharge was estimated as c. 9% of c. 800 mm annual precipitation by Bradford (1973) (recognising this estimate is old and now influenced by factors such as climate change). Groundwater flow is generally from the upland, higher rainfall/recharge, plain margin - surrounding hard-rock escarpments and expected to laterally cross the valley plain towards the Shire River axis and southern marshland potential discharge areas (Fig. 5). Based on cross-valley hydraulic gradients of c. 0.003 (Monjerezi et al., 2011a), an effective porosity of poorly-sorted (alluvium) sediments of c. 20% and a fine-medium sand hydraulic conductivity of 0.25 m/d, a nominal groundwater velocity of 0.0038 m/d, equivalent to just 1.4 m/yr,

may be calculated for the alluvium. For locally continuous transmissive sands/gravels, velocities may be one to over two orders of magnitude higher. Overall the system is conceptualised as likely low velocity, but with more elevated velocities near the plain margins – escarpment bedrock interface or river systems. High recharge during flooding may also cause increased gradients and flows.

Greatest TDS occurs in the alluvial plain to the immediate west of the Shire River (Fig. 5a,b) and, the southern lowlands and close to the western escarpment contact of the Karoo and alluvium (Fig. 5a). Greater sampling of those predominantly lowland areas in 2008-9 results in the primary contrast in the proportions of water types sampled by each survey; the 2008-9 versus (vs.) 2012 comparison being: Group 1 - 40 % vs. 58%; Group 2 - 40 % vs. 27%; and, Group 3 - 40 % vs. 12%. The 2008-9 survey provides better definition of potential salinity constraints upon resource use (Box SD1).

Group 1 provenance east of the Shire near the plain margins is consistent with recent recharge water of short residence times. Runoff and groundwater discharge from escarpment rock units into valley margin alluvium is probable. Predominant Group 2 occurrence over the western to central plain area is consistent with groundwater occurrence of longer residence times and potentially greater frequency of silt/clay-rich horizons in the central plain offering ion-exchange potential. Predominant Group 3 occurrence to the immediate west of the Shire is consistent with the expectation that enhanced concentration through evaporation effects on shallow groundwater is more prevalent closer to lowland - surface waters (see later figures depicting 2015 flood extent – wetland areas). Evaporite minerals occur within the lowland alluvial aquifer as a result of cyclic flooding and evaporation leading to concentration of precipitated salts within near-surface sediments. Evaporite minerals such as halite are also common within the Karoo and Cretaceous sedimentary aquifers to the immediate west; higher TDS - Group 3 groundwater around the Karoo – alluvium contact likely arises from dissolution of these mineral sources (Monjerezi et al. 2012).

3.1.4 System conceptual model development

Based upon the hydrogeological-hydrochemical analysis above, a process-based system conceptual model is illustrated for the Chikwawa District study area in Fig. 6. Fifteen

numbered processes are indicated, concisely described and their spatial occurrence exemplified (and colour-coded) to help visualise the predominant groundwater hydrochemical type. A schematic cross section illustrates possible vertical flow and process components in recharge–discharge areas and interactions with adjoining geological units to the alluvial aquifer that could result in groundwater quality depth variation. Vertical variation in groundwater quality is increasingly expected as resources become more stressed (abstraction demands, climate change).

More speculative, but reasonably anticipated, processes are included. These include groundwater baseflow discharge to a river (#5) and possible transient reversal (#6) illustrated as pair-based occurrences across the colour-coded groundwater type scheme with low TDS Ca/Mg–HCO₃ discharge east of the Shire River, medium TDS Na–HCO₃ discharge west of the Shire, and high TDS Na–Cl (SO₄) discharge towards the southern lowlands. Additional processes not represented could include increases in more anaerobic iron/manganese-rich waters associated with clay semi-confining conditions and the discharge of deep, thermal, geochemically distinct groundwater along fault contacts (Dulanya, 2006; Msika et al., 2014).

3.2 Network design and monitoring point prioritisation (Steps 3 – 6)

3.2.1 Rationale for network size and density

The conceptual model allows identification and prioritisation of monitoring requirements, including selection of an appropriate number and distribution of monitoring points. Critical factors in making this selection are the known or anticipated spatial variability in groundwater quality (Ward et al., 2004) (albeit recalling the goal is not to provide point density for surveillance snapshot sampling, but rather temporal monitoring) and realisation of representative coverage of the MOs set. The practical, developing-world, reality is that network point numbers and sampling that ultimately transpire may prove to be very much constrained by the sustained financial resource available. Such constraints are already evident for the existing national Malawi water-quality monitoring network (Section 2.2.2).

Whilst not ignoring the significance of such constraints, we present a notional case for consideration that provides a spatial density of points commensurate with regional monitoring densities within national networks implemented by regulatory bodies in the developed world. This approach, whilst perhaps idealistic within a developing-world context, may nevertheless serve as a reference 'benchmark' network to which points may be added or removed based on the network's potential to address the critical factors identified above, conditioned by budgetary constraints. The approach parallels that demonstrated by Ward et al. (2004) in the Environment Agency's (for England and Wales) development of national networks that statutory comply with the European WFD.

We propose a benchmark monitoring point density of one monitoring point per c. 33 km² for the Chikwawa aquifer system that is intermediate within the range of minimum spatial densities recommended for monitoring aquifers in England of one monitoring point per 25 km² for unconfined major aquifers, per 35 km² for confined major aquifers, and per 50 km² for Minor aquifers (Chilton and Milne, 1994; Ward et al., 2004). As the study area comprises a total area of 1440 km² of which 1140 km² (79%) is judged viable for point allocation (excludes areas of national park or high flood risk – wetland areas of Fig. 6 that do not contain water points), a benchmark mean-area coverage of 33 km² per point requires a network of 35 points. Further rationale for this network size includes: it could potentially be sampled with around 1 week of field effort for a core suite of determinants used by Malawi's national network (Table SD3); it likely will provide an informed geographic coverage (based on findings from our 50-point 2012 survey, recognising more judicious placement of points based upon the system conceptual model); and, it potentially may provide a balanced coverage of the seven MOs set with a reasonable point to MO ratio of 5:1. It is recognised, however, that yet further increased densities may be warranted where greater spatial groundwater quality variability is expected, notably complex urban environments.

Selection is also required of the grid cell (pixel) resolution necessary for decision-making on the spatial allocation of network points. For Chikwawa, a uniform 2-km spaced grid was applied, each 2-km by 2-km cell being a candidate location to host a network point. This generated 286 cells within the viable point area from which 35 cells were to be selected to generate the benchmark network. This constitutes an available to selected cell ratio of 8:1

which is considered practical (compared with say a total of 1140 cells of 1 km² and a choice ratio of 32:1, considered excessive). Nevertheless this grid allocation is recognised as a fairly arbitrary starting point that could be varied in later iterations of network development (and informed by monitored data).

3.2.2 Development of proposed monitoring network

The proposed monitoring network is shown in Fig. 7. The network is built upon addressing the MOs set (Table 1), the system conceptual model developed (Fig. 6) and its underpinning GIS database. The GIS-generated (multi-layer) map underlay illustrated within Fig. 7 invokes elements of the conceptual model and depicts several of the datasets that were used to rate and score the specific MOs set. Examples include: observed colour-coded groundwater types measured in the 2008-9 and 2012 surveys that contributed to evaluation of MO1, MO2 and MO4; the occurrence of water points contributing to MO3 primarily and informing MO5 as an indicator of populated areas – potential anthropogenic threat; the occurrence of rivers – flooded areas underpinning MO6 assessment primarily, but also MO1 as areas of higher salinity potential. MO5, to monitor anthropogenic-urbanisation threat to the groundwater resource was also informed by the occurrence of road networks and villages shown in Fig. 7. It was anticipated that use of the numbers of pit latrines could also inform MO5 (or potentially an additional MO that could be set relating to monitoring microbiological pathogen risks) (Back et al., 2018), however, whilst some 13,800 pit latrines are mapped in the area (Supporting Data Fig. SD2), that dataset is incomplete across the study area. Hence MO5 was awarded moderately increased scores where mapped high latrine density was very apparent, but cognisant that this potential threat could not be evaluated in the areas yet to be latrine mapped.

It is important to recognise that as a user determines ratings, their integrative expert judgment should be used invoking the system conceptual model understanding, ideally extending beyond the simplification schematically captured (Fig. 6). For example, recognising assessment of MO1 on salinity is not only a key issue for cells identified with Group 3 groundwater type, but potentially for cells (with no groundwater data), in lowland or flood-prone areas or cells influenced by urban-anthropogenic sources. Consideration of groundwater flows should also be appropriately invoked, for instance noting the potential

influence of up-gradient cell features on the cell under consideration for a monitoring point. The framework should hence be used, or adapted (including enhanced GIS manipulation in due course), to facilitate and automate such expert input.

The proposed, incrementally built, monitoring point locations are shown as priority numbered boxed cells in Fig. 7. Hence if only 5 monitoring points were to be ultimately used, the network would comprise the shown points 1 to 5; if 10 network points, then points 1 to 10 etc. Such an incrementally numbered prioritised network allows for rationalised expansion or contraction of a network size. The network was iterated to the final shown selection of 35 points. This includes points 31 to 35 that are proposed shallow/deep monitoring point pairings with priority points 1 to 5. These points are hence intended to provide vertical data at high-priority localities.

Iterative development (re-ordering point priority) of the growing network was aided by graphically plotting individual network point MP estimates and a cumulative score for individual MOs for the overall network. Fig. 8 plots these for the network shown in Fig. 7. Fig. SD3 (Supporting Data) illustrates the sensitivity to weighted and non-weighted MOs. MP scores approximately decrease as the network grows due to higher scoring localities being preferentially allocated. The anomalous low points in the declining trend of MP scores (Fig. 8a) are typically due to points that provide coverage of good quality (Group 1) groundwater that may have low scores on the key salinity issue MO1 and other scores such as MO5 on anthropogenic impact monitoring. Rebound to higher MP scores for points 31 to 35 is due to the shallow/deep-point pairing with points 1 to 5 positioned at high MP score localities.

The cumulative score for individual MOs aggregated over the network (Fig. 8b) informs the prioritisation and balance achieved across the various MOs as the number of points and network density increase with a corresponding reduction in the mean area per point (reaching 33 km² for 35 points). The influence of the weighting is best seen in the cumulative plots (compare spread of profiles in Fig. 8b with more bunched profiles in Fig. SD3). These plots illustrate that a reasonable balance was achieved across the MOs set. The accelerated early rise and higher values of more highly ranked MOs accords with their prioritisation with lower gradients (levelling) of profiles once that MO is largely addressed by the network and priority is given to addressing other MOs. A decision was made to use

points 31 to 35 to resolve depth variation as it was judged from Fig. 8 that the network of 30 points had already addressed all MOs with reasonable balance and achieved a satisfactory coverage equating to a mean area of 38 km² per monitoring point commensurate with values for regional-national network deployment in Europe discussed earlier.

3.3 Potential implementation (Steps 7 – 9)

The proposed Chikwawa network represents a monitoring option for consideration with spatial densities that are commensurate with European national monitoring frameworks, or perhaps even better when it is recognised system conceptualisation may aid reduction of densities compared to original national benchmarks (Ward et al., 2004). Potential implementation and sustained periodic monitoring at this relatively local regional scale, whilst a possible option does, however, require consideration of its relative importance nationally. The number of points, 35, is fortuitously identical to the number of monitoring wells within the present entire national network of which only one network site is located within the study area at Chikwawa Boma.

Proposals within the recently revised National Water Resources Master Plan (MoAIWD 2017a,b) are for an increase of up to a total of approximately 90 groundwater monitoring network points. Whilst the study area alluvial aquifer system represents an important high storage strategic groundwater resource, it could only realistically be perhaps expected to attract 3 to at most 5 of this revised total 90 points. This is clearly far below the numbers proposed by the 35-point network illustrated in Fig. 7. The versatility of the framework is that network points 1 to 3 (or 5) could be selected for a much reduced down network, although in practice it is suggested that points 1 to 10 should be reconsidered and potentially reordered to allow a further optimised selection of relatively few points.

It should be recalled, however, that there should be good linkage and compliment between temporal network monitoring and surveillance, higher spatial resolution, snapshot survey monitoring of groundwater quality with common network points to both. It is hence proposed if high frequency (perhaps quarterly) temporal monitoring of points is limited, as is likely the case for Malawi, the 35-point network shown in Fig. 7 could still provide a

rationalised selection of points comprising a core priority network used in more occasional surveillance snapshots (perhaps annual) that are the core priority maintained network points within even more occasional surveillance snapshots (perhaps taken every 5 years) that aim to sample many more additional points and achieve higher spatial resolution. Such strategic, nested sampling, would be best accomplished with supply wells rather than purpose installed monitoring wells (that, with low use, would be prone to loss, vandalism etc.). It is recommended, if the framework approach is being used in this way, with a significant leaning towards providing a basis for spatial surveillance monitoring, that spatial statistical analysis is later employed using monitored data collected over the first 1 to 6 years (hence including 2 detailed spatial snapshots) to allow further optimisation of core network points selected and their appropriateness for temporal and spatial monitoring and delivery of monitoring objectives (Esquivel et al., 2015; Kollat et al., 2011; SOGW, 2013).

It is recognised many of the challenges faced by Malawi are likely prevalent across the developing world. Moreover, it is acknowledged that network establishment even in the developed world has been found to be challenging (Ward et al., 2004; SOGW, 2013). Regional leading to national-scale implementation of monitoring networks in developing countries certainly constitutes a significant undertaking. It is proposed, however, that the framework set out herein, focusing on developing pilot networks within the study area and perhaps several other regions containing important, but contrasting groundwater systems may help contribute a viable, technically informed, pragmatic pathway to the delivery of sustained regional and national groundwater-quality monitoring network establishment in Malawi.

4 Conclusions

A step-wise framework for groundwater-quality monitoring network design is outlined and demonstrated within a developing world context. The setting and weighting of monitoring objectives bespoke to a monitored groundwater system and the rating and scoring of the monitoring potential of localities against bespoke criteria offers a flexible and semi-quantitative approach to the development of a network design within a facilitating, but not necessarily required, GIS. The methodology provides a straight-forward, uncomplicated,

framework for network establishment that is versatile and easy-to-use. It may be pragmatically adapted to not only other developing world, but also developed world, contexts. It is anticipated that the core framework approach of setting, weighting, ranking and scoring of monitoring objectives can be flexibly developed within more complex GIS frameworks furthering or automating the calculation of rating values and, or algorithms providing a more statistical basis for point selection and rating evaluation.

The approach is underpinned by a process-based hydrogeological-hydrochemical system conceptual model that allows expert input to the design. It promotes utilisation of available, typically sporadic, groundwater quality survey data. For instance, we utilise for the Chikwawa District demonstration case, major-ion hydrochemical survey data that likely represent the default minimum dataset potentially available in many developing world localities. The study illustrates the development of a conceptual model from these essential hydrochemical data and how these may be used to feed into monitoring objectives that are not only pertinent to evolution of the natural hydrochemistry of the system, but also a baseline against which increasing anthropogenic impacts may be identified and monitored.

The framework successfully realised its ambition to provide a pragmatic and rationalised network design within the Malawian study setting. The proposed Chikwawa network, with complementary extension to other areas, could help pilot the transition to a higher resolution national groundwater quality network across Malawi than currently exists. Attaining the spatial monitoring densities suggested that approach those of national networks in Europe is, however, challenging due to the need to obtain the significant investment that is required in current infrastructure and technical capacity and also the development of funding sustainable funding mechanisms that allow a network's high running costs to be met long term. Whilst such high densities of points may not be viable in developing countries for frequent temporal monitoring, proposed networks at such densities may still form a rationalised core of network points for more occasional surveillance snapshot monitoring, as well as candidate points for any increases later made in a temporal network.

It is hence anticipated that the framework approach proposed will help contribute to the much needed developing-world transition to sustained groundwater quality monitoring

network programmes that vitally underpin Water Safety Plans and ensure the improved long-term safeguard of developing-world groundwater resources and hence contribute to attainment of SDG 6 in the developing world.

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Table 1. Monitoring objectives and valued rating criteria used for Chikwawa District.

Rating	Criteria
MO1 - Monitor key water quality issue – increasing salinity (weight = 5)	
0 - 2	Low / zero threat of salinity perceived / background data
3 - 4	Salinity threats appears low - distant / only suspected
5 - 6	Evidence of salinity up-gradient, but relatively distant
7 - 8	Within or somewhat down-gradient of known high salinity
9 - 10	Immediately down-gradient of known high salinity
MO2 - Monitor hydrochemical-groundwater type to protect supply quality (weight = 5)	
0 - 2	Hydrochemical type well known, but supply quality is poor, resource write off?
3 - 4	Hydrochemical type unknown, but supply quality suspected poor and low value
5 - 6	Hydrochemical type and supply quality variable, but mainly good and to protect
7 - 8	Hydrochemical type poorly known, but suspect good supply quality to protect
9 - 10	Hydrochemical type well known and known high supply quality to protect
MO3 - Monitor high groundwater resource use area (weight = 4)	
0 - 2	Area has only isolated water points
3 - 4	Area has relatively sparse water-point density
5 - 6	Area has moderate water-point density relative to regional surroundings
7 - 8	Area has high water-point density relative to regional surroundings
9 - 10	Area contains near maximum water-point density for entire study area
MO4 - Monitor probable concentration transients / heterogeneity (weight = 4)	
0 - 2	Region has homogeneous hydrochemical type, low transients, unknown
3 - 4	Area fairly homogeneous hydrochemical type, low transients
5 - 6	Heterogeneous hydrochemical type / probable concentration transients
7 - 8	Transients/heterogeneity understanding via local well pairing
9 - 10	Transients/heterogeneity understanding via shallow / deep pairing
MO5 - Monitor anthropogenic – urbanisation threat to resource (weight = 3)	
0 - 2	Rural, very sparse villages, very few water points, remote
3 - 4	Mainly rural villages sparse, few water points, minor roads, quite remote
5 - 6	Moderately urbanised - road(s), moderate village / well density
7 - 8	Highly urbanised area up-gradient, moderate urbanisation locally
9 - 10	Highly urbanised locally
MO6 - Monitor groundwater - surface-water interactions (weight = 3)	
0 - 2	Surface water absent or very remote or largely disconnected to groundwater
3 - 4	Region has some more distant surface water
5 - 6	Area has surrounding more distant surface water
7 - 8	Notable surface water occurrence, possible groundwater interaction, springs
9 - 10	Significant surface water occurrence, probable groundwater interaction, springs
MO7 - Enhance spatial (or depth) coverage of monitored domain (weight = 4)	
0 - 2	Limited spatial / domain gain relative to monitoring in place
3 - 4	Minor spatial / domain gain relative to monitoring in place
5 - 6	Usefully extends monitoring, but in less key areas
7 - 8	In-fills useful area devoid of monitoring / useful boundary to define
9 - 10	In-fills key region devoid of monitoring; defines key boundary/areas

FIGURES

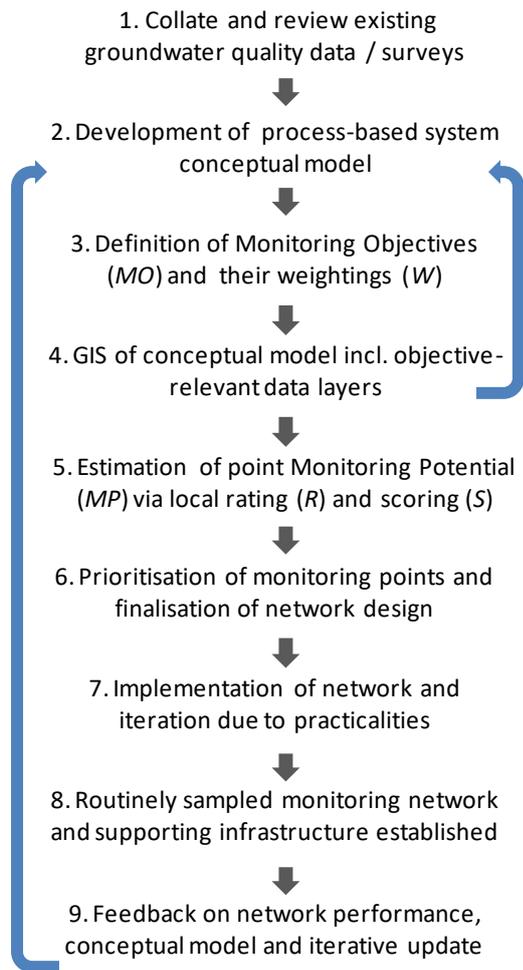


Fig. 1. Framework for Groundwater-quality monitoring network design.

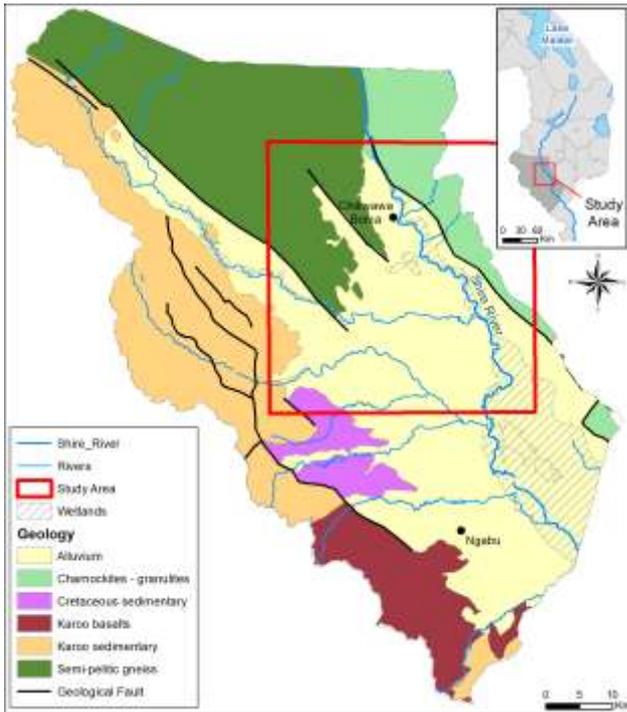


Fig. 2. Chikwawa District study area, Southern Malawi: surface water, solid geology and alluvium (after Geological Atlas of Malawi, 1st Ed. Sheet I (Nsanje), Geological Survey of Malawi).

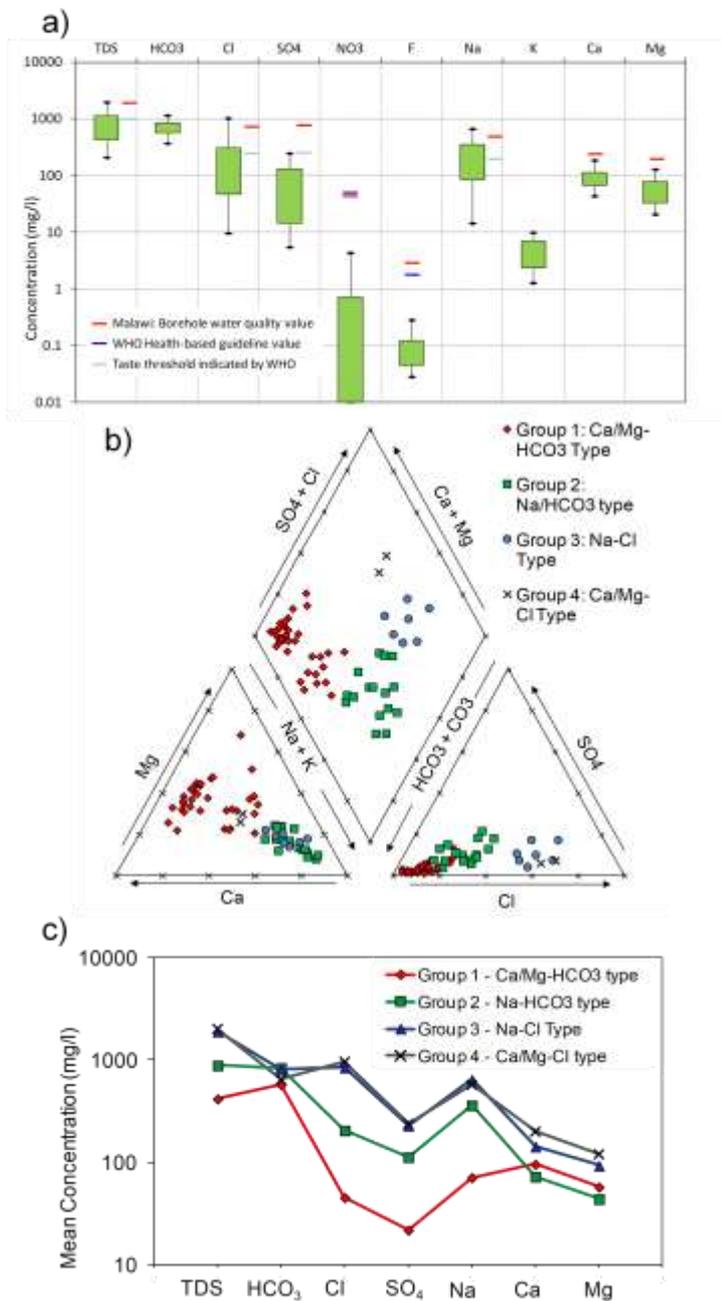


Fig. 3. Groundwater 2012 survey data (n = 50): (a) major-ion occurrence; hydrochemical groundwater type analysis compared to Malawi groundwater standards (MBS, 2005) and WHO health-based guideline values and taste threshold estimates (WHO, 2017): (b) major ion mean concentrations, (c) Piper plot.

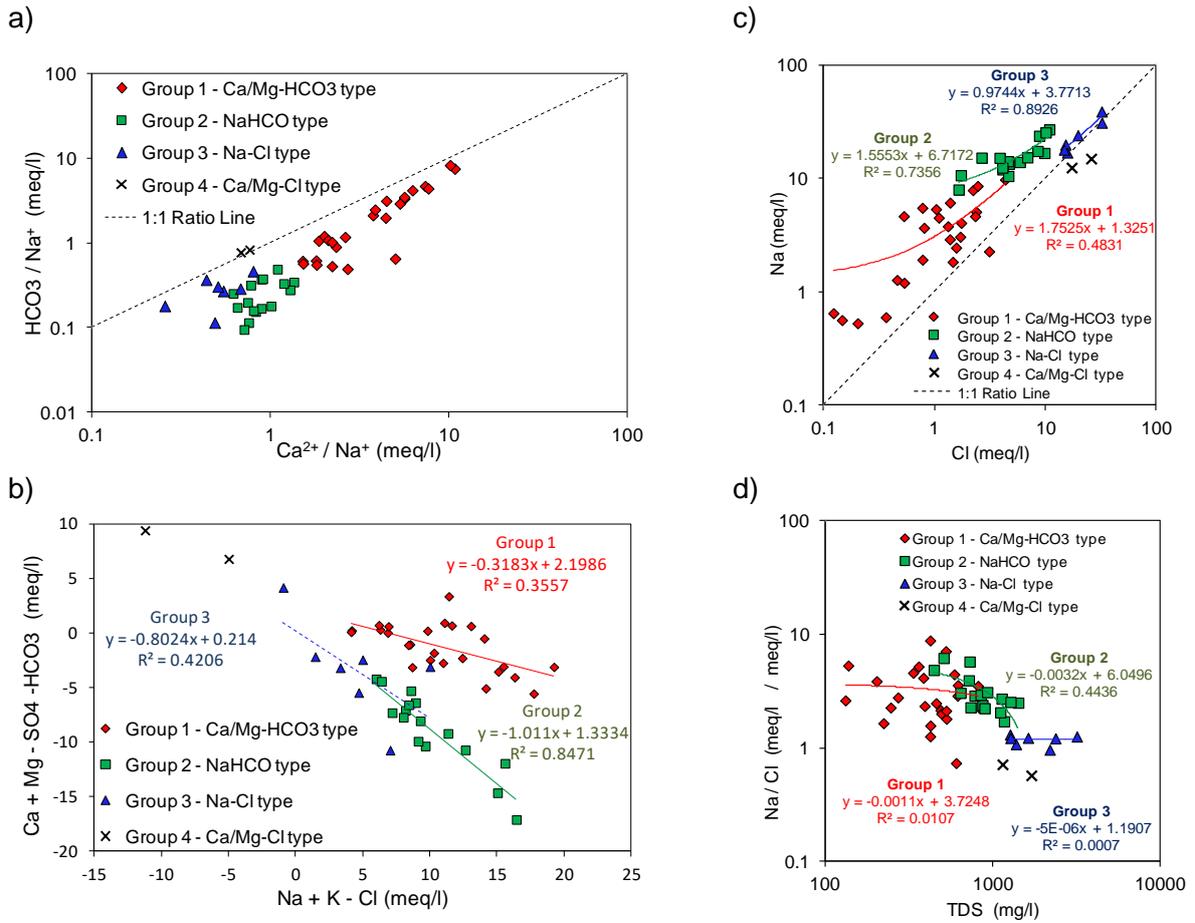


Fig. 4. Groundwater 2012 survey: a) plot of Na-normalised HCO₃ versus Na-normalised Ca²⁺ to assess silicate weathering; b) plot of (Ca²⁺ + Mg²⁺) - (SO₄²⁻ - HCO₃⁻) against (Na⁺ + K⁺) - Cl⁻ to assess cation exchange; c) plot of Na versus Cl indicating the significance of evaporite dissolution and/or evaporative concentration; d) plot of Na/Cl ratio versus TDS.

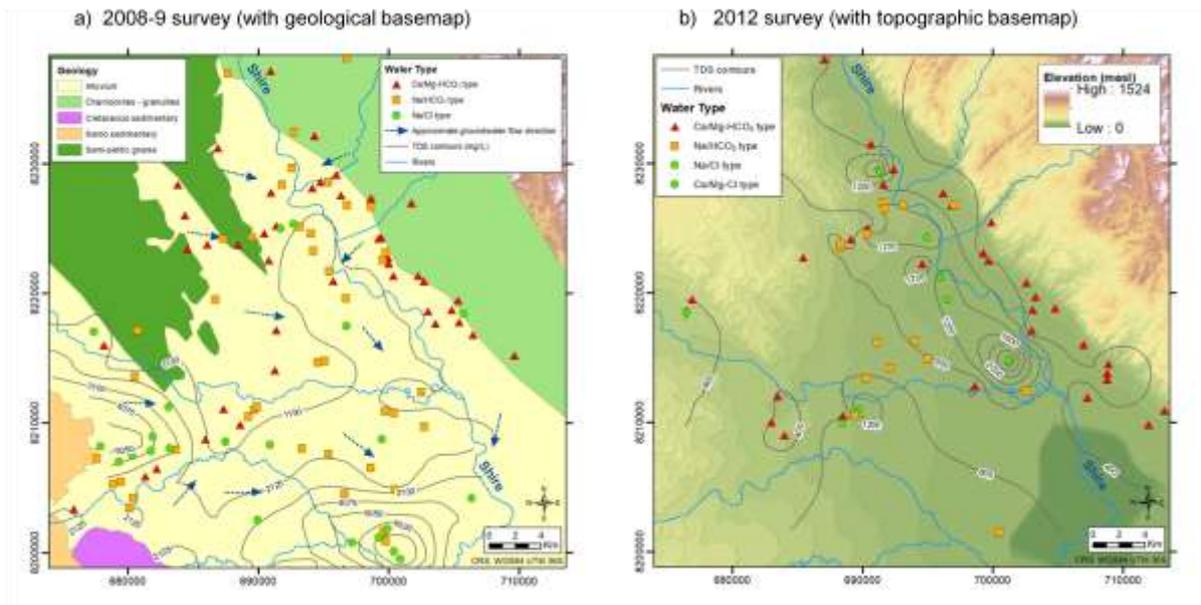
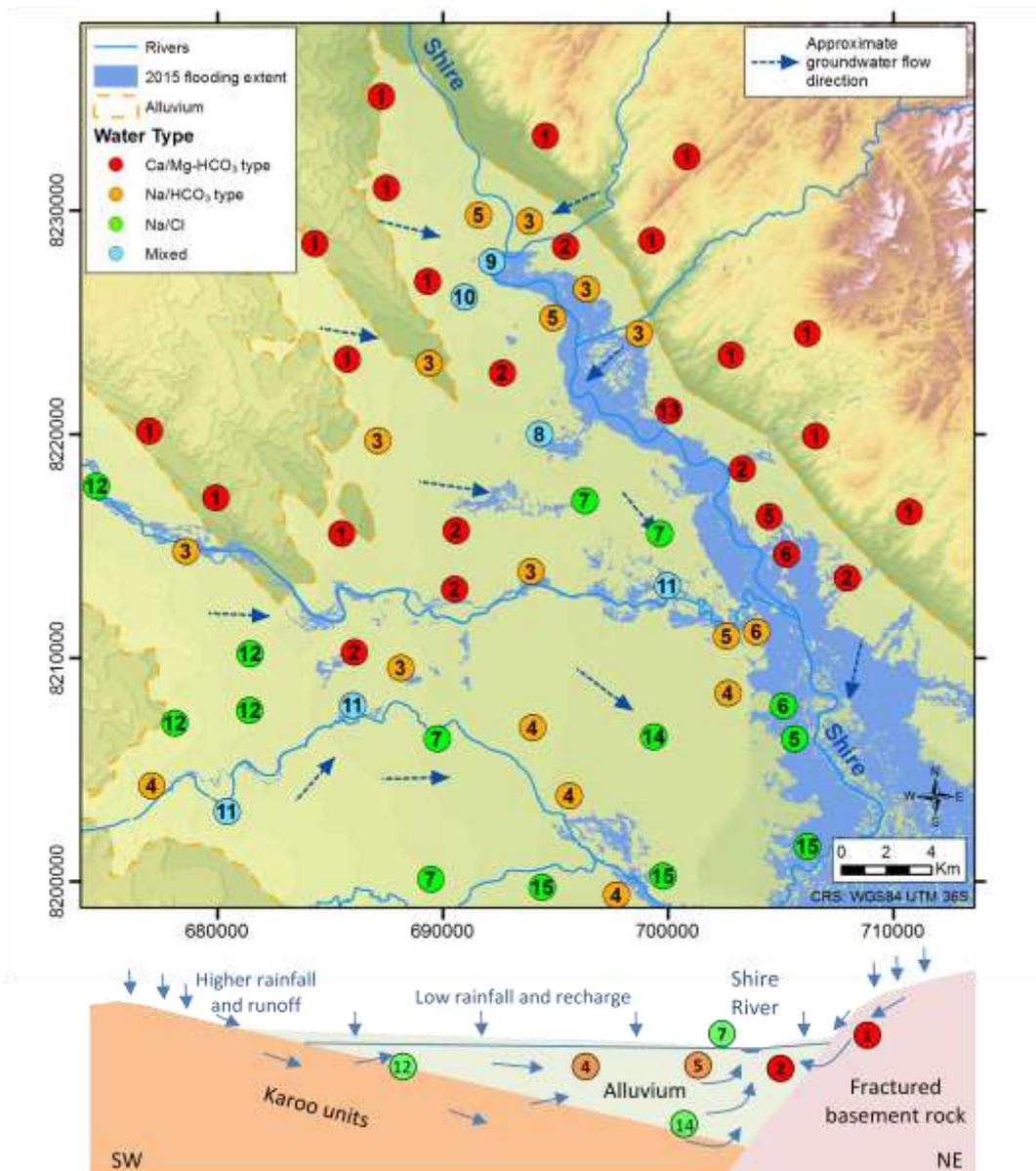


Fig. 5. Spatial distribution of hydrochemical groundwater group types and (inverse-distance) contoured TDS (mg/l) for: a) 2008-9 survey of Monjerezi et al. (2011a) [their Figure 4b]; b) 2012 survey. Groundwater type plotting in the figure reasonably assumes, based on similarity of median TDS and composition data, that for the clusters developed from the hierarchical cluster analysis of Monjerezi et al. (2011a), their Cluster C1 equates to the 2012 survey Group 3 (Na-Cl type), C2 to Group 2 (Na-HCO₃ type) and C3 to Group 1 (Ca/Mg-HCO₃ type) groundwater discussed in the manuscript. Indicated groundwater flow directions are estimated from Monjerezi et al. (2011a) hydraulic head data.



1. Fractured rock infiltration - discharge to alluvial aquifer, aluminosilicate rock dissolution low TDS, Ca/Mg-HCO₃
2. Groundwater flow from escarpment / plain aquifer margins towards river network – aluminosilicate dissolution, young water, low TDS, Ca/Mg-HCO₃
3. As 2, but with increased ion exchange due to clay minerals, mixed Ca/Mg-HCO₃ and Na-HCO₃
4. As 3, yet older groundwater with increased ion exchange due to increased clays/age, mainly Na-HCO₃, SO₄ higher TDS
5. Groundwater baseflow discharge to river / wetlands
6. Groundwater discharge to river, but with temporary flow reversals due to high river stage / flooding or seasonality
7. Water table in lowland close to ground surface prone to evaporation, flood-drying cycles, high TDS, NaCl waters
8. River influent leakage and recharge of water table at depth of river water quality, seasonally variable TDS/ composition
9. River influent leakage and recharge of water table lowered due to possible over abstraction or seasonal conditions
10. Heterogeneous groundwater quality due to possible urban influence, near-river, near-geological contacts complexities – all water types
11. Heterogeneous quality groundwater due to mixing of water types from different localities in complex flow regimes – all water types
12. Dissolution of evaporite (halite, gypsum) beds in adjoining sandstones with discharge into main aquifer, high TDS unless diluted – Na-Cl(SO₄)
13. Short residence time - transmissive, permeable aquifer with high flow rates and usually younger water Ca/Mg-HCO₃ tendency
14. Long residence time – less permeable units (fine sands /clays), shallow hydraulic gradients, older Na-SO₄-Cl waters
15. Similar to 14 in groundwater discharge to wetland

Fig. 6. Hydrogeological-hydrochemical conceptual model for northeast Chikwawa District.

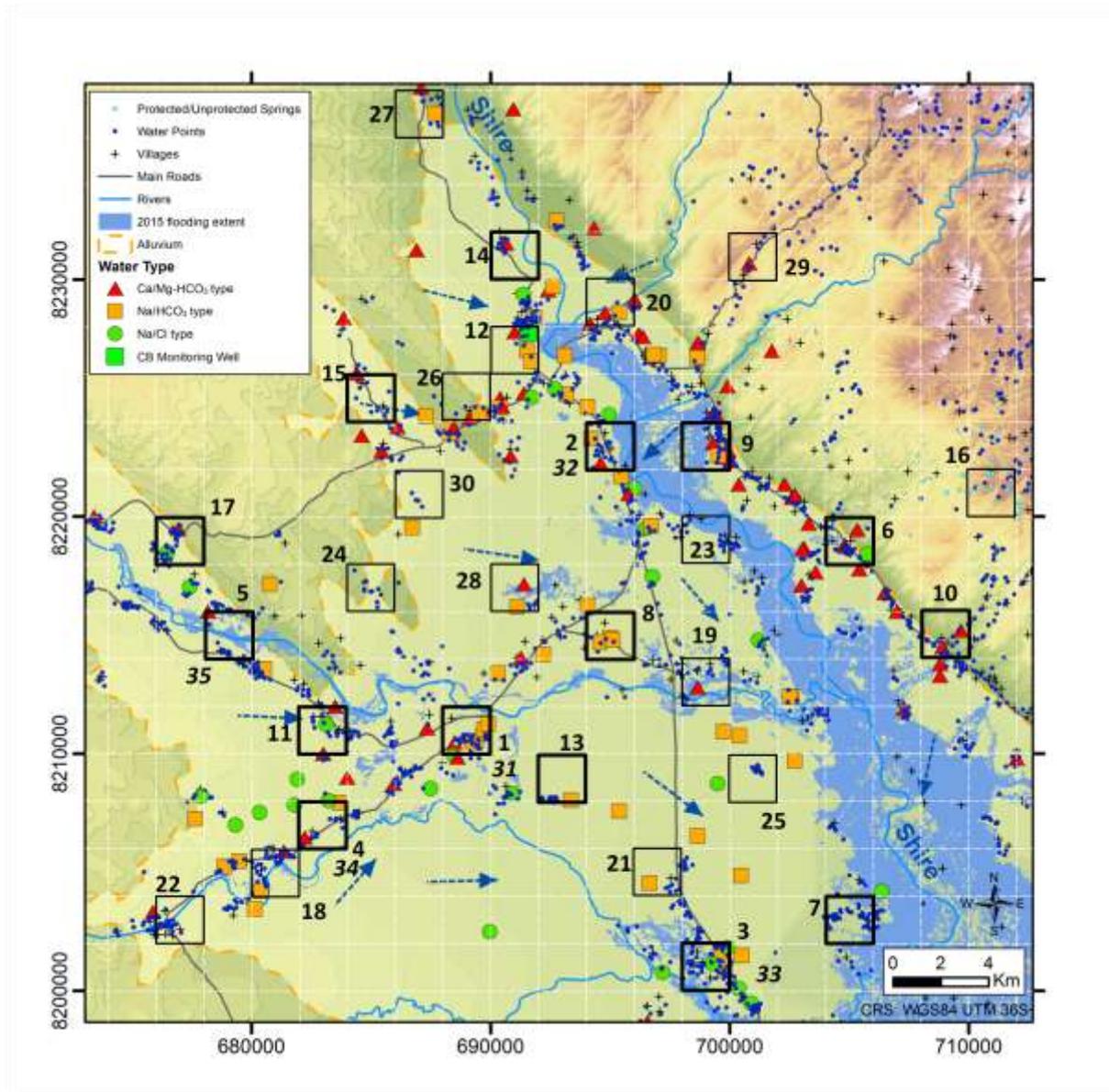


Fig. 7. Northeast Chikwawa District proposed monitoring network with prioritisation of points 1 to 30 allocated to 2 km grid square localities. Points 31 to 35 are proposed shallow/deep monitoring point pairings with priority points 1 to 5. The single current monitoring well at Chikwawa Boma (CB) is shown. The topographic legend of Fig. 5b applies.

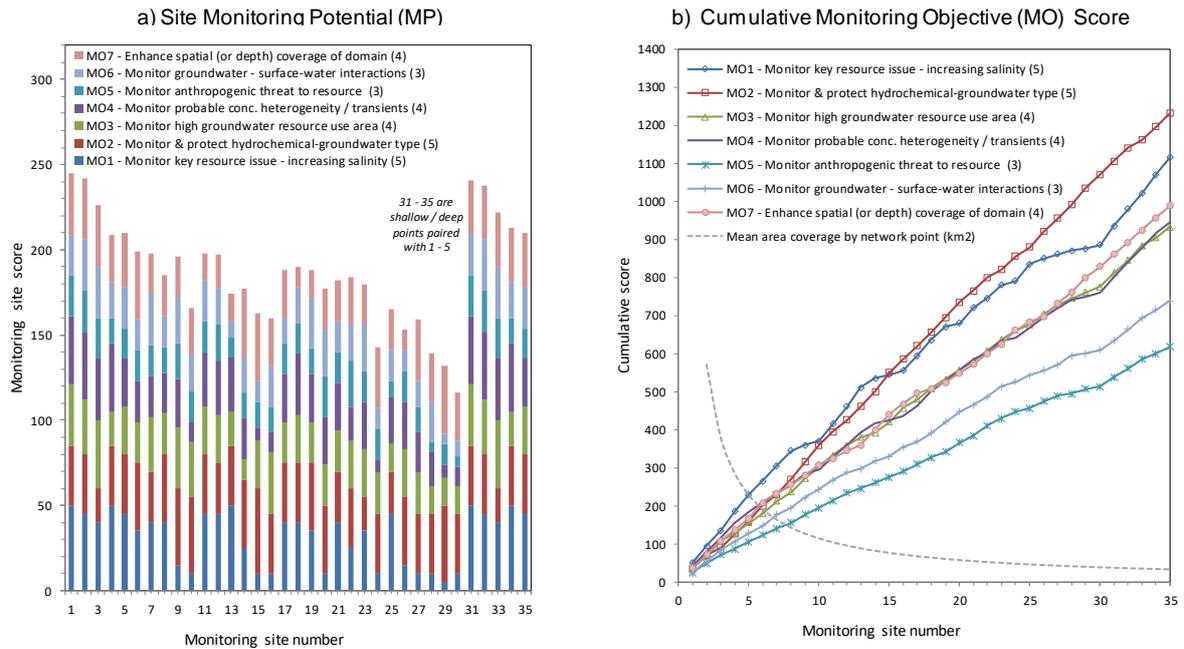


Fig. 8. Plots of: (a) individual network point Monitoring Potential (MP) estimates; and, (b) cumulative Monitoring Objective (MO) overall network Scores with increasing network size based on proposed Chikwawa network points shown in Fig. 7 (the mean area (km²) coverage profile uses the y-axis values with units of km²).

SUPPLEMENTARY DATA

FOR

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CONTENTS

Table SD1. Summary of 2012 survey groundwater sample hydrochemical data.

Table SD2. Mean, min and max values of determined saturation indices for each groundwater type in the 2012 survey.

Box SD1. Local sub-area comparison of the 2008-9 and 2012 Groundwater quality surveys, including

Fig. SD1. Spatial distribution of hydrochemical groundwater group types and contoured TDS (mg/l) for: a) 2008-9 survey of Monjerezi et al. (2011), b) 2012 survey of Fallas (2012).

Table SD3. Suite of water quality determinants analysed by the Water Quality Services Division of the Department of Water Resources, Ministry of Agriculture, Irrigation and Water Development.

Fig. SD2. Mapped occurrence of sanitation facilities (typically pit latrines) in the study area (as of November 2017).

Fig. SD3. Comparison of proposed Chikwawa network site Monitoring Potential (MP) and cumulative (with increasing network size) Monitoring Objective (MO) scores for assumptions of equal and prioritised weighting of MOs (Table 1 of main manuscript).

Table SD1. Summary of 2012 survey groundwater sample hydrochemical data (n = 50) compared to Malawi Standards (MS) (2005) ‘Standard for water delivered from Boreholes and Protected Shallow Wells’ (MS733:2005) and WHO (World Health Organisation) health-based guideline values and taste threshold estimates where relevant (WHO, 2017). The percentage of samples exceeding these criteria is indicated in parentheses.

Parameter	Range	Mean	Med'n	Malawi Standard MS733:2005 (% exceeding)	WHO Health-based guideline value (% exceeding)	WHO Est. Taste threshold (% exceeding)
pH	7.2 -8.3	7.6	7.6	6.0 – 9.5 (-)	N/A	
EC (µS/cm at 25°C)	262 - 6360	1653	1259	3500 (6 %)	N/A	
TDS (mg/l)	131 - 3180	827	629	2000 (6%)	N/A	1000* (28%)
Bicarbonate (mg/l)	247 - 1359	722	700	N/A	N/A	
Chloride (mg/l)	4.3 - 1342	239	90	750 (8%)	N/A	250 (28%)
Sulfate (mg/l)	3.4 - 500	85	57	800 (-)	N/A	250 (6%)
Nitrate (NO ₃ , mg/l)	N/D – 15.5	1.1	0.03	45 (-)	50 (-)	
Fluoride (mg/l)	0.01 - 0.64	0.11	0.08	6 (-)	1.5 (-)	
Sodium (mg/l)	12.1 - 1095	261	211	500 (14%)	N/A	200 (50%)
Calcium (mg/l)	34.7 - 227	99	96	250 (-)	N/A	
Magnesium (mg/l)	15.6 - 166	62	54	200 (-)	N/A	
Potassium (mg/l)	0.07 – 42.2	5.5	4.2	N/A	N/A	

* unpalatable water ‘brackish’ threshold

N/A denotes ‘not applicable’ as standard not defined for parameter

Table SD2. Mean, min and max values of determined saturation indices for each groundwater type in the 2012 survey. Saturation indices (log. of ion activity product (*IAP*) / solubility product (*K_{sp}*)), were determined using the geochemical model WATEQ4F (Ball and Nordstrom, 1991) for each groundwater sample with respect to key mineral phases. Groundwater is assumed to be in mineral equilibrium if the saturation index is within a range of -0.05 to +0.05, undersaturated if the value is below -0.05 and supersaturated if the value is above +0.05 (Merkel and Planer-Friedrich, 2008).

		Saturation Indices (log IAP/Ksp)			
		Calcite (CaCO ₃)	Dolomite CaMg(CO ₃) ₂	Gypsum (CaSO ₄ ·2H ₂ O)	Halite (NaCl)
Group 1	min	0.20	0.47	-3.02	-8.34
	max	1.01	2.91	-1.54	-3.02
	mean	0.72	1.58	-2.40	-6.89
Group 2	min	0.59	1.31	-2.21	-6.57
	max	1.09	2.37	-1.42	-5.28
	mean	0.92	1.96	-1.85	-5.84
Groups 3/4	min	0.67	1.60	-1.95	-5.41
	max	0.96	2.01	-1.05	-4.53
	mean	0.81	1.78	-1.46	-5.06

Box SD1

Local sub-area comparison of the 2008-9 and 2012 Groundwater quality surveys

Local sub-area comparison of the 2008-9 (Monjerezi et al., 2011) and 2012 (Fallas, 2012) groundwater quality surveys is made below referring to specific localities (labelled [A], etc.) shown on Fig. SD1.

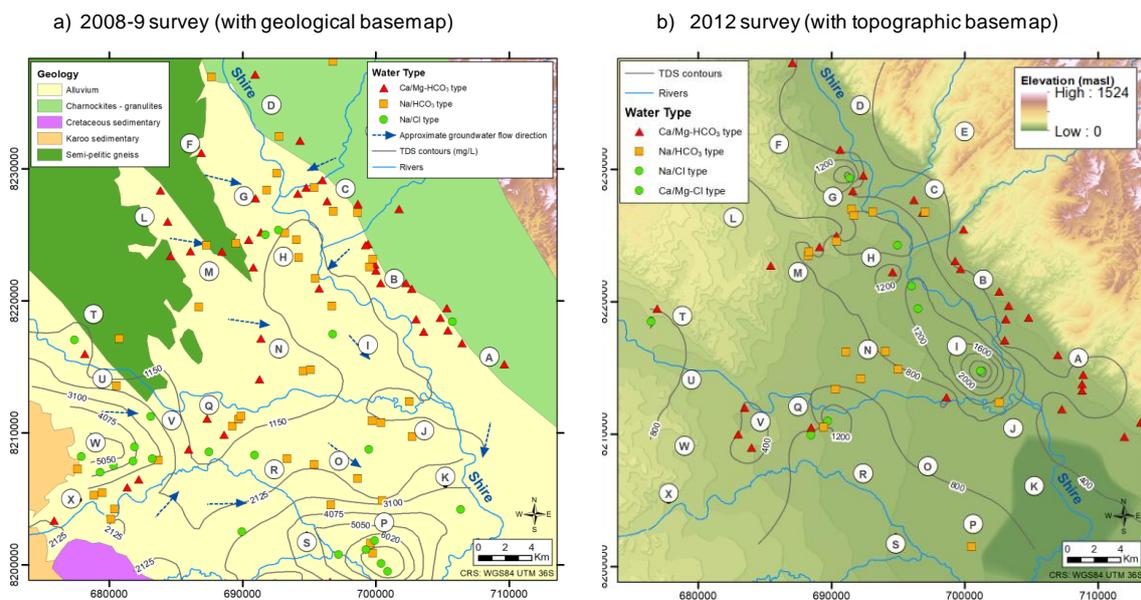


Fig. SD1. Spatial distribution of hydrochemical groundwater group types and contoured TDS (mg/l) for: a) 2008-9 survey of Monjerezi et al. (2011), b) 2012 survey of Fallas (2012). Groundwater type plotting in Fig. SD1 reasonably assumes, based on similarity of median TDS and composition data, that for the clusters developed from the hierarchical cluster analysis of Monjerezi et al. (2011a), their Cluster C1 equates to the 2012 survey Group 3 (Na-Cl type), C2 to Group 2 (Na-HCO₃ type) and C3 to Group 1 (Ca/Mg-HCO₃ type) groundwater discussed in the manuscript. * See Box SD1 footnote

Given that only a few years separates these surveys, major changes in groundwater quality type or TDS should not be expected in this system. The comparison provides more local area detail of controlling processes (summarised in the main manuscript) and informs on the local adequacy of the sparser 2012 survey (50 points) to reproduce the findings of the 2008-9 survey (105 points). It endorses the value of multiple surveys to a conceptual model build.

There is reasonable TDS comparison along both sides of the Shire River with the >1000 mg/l area comparable. However, the 2012 survey sparseness towards the west plain margin and southern areas, whilst detecting some evidences of increased salinity, fails to detect salinity over 2000 mg/l TDS. Examining the eastern margin of the plain in areas [A – C], the 2012 survey predominantly detects the low TDS Group 1 water, but whilst picking up some hint of Group 2 water around Area [C] does not have sufficient well density to detect the fairly mixed Group 1 and 2 composition

evident in the 2008-9 survey. That survey more conclusively confirms Group 1 quality in the rock escarpment [E].

Immediately west of the Shire, near [F] and [G], samples within and closest to the western escarpment better confirm the provenance of Type 1 groundwater. Within the more urbanised Chikwawa town area [G], both surveys observe heterogeneous quality with boreholes sampling each of the three groups in this urban, near-river, near-geological contact setting. South of Chikwawa [H – I], both surveys observe high salinity although the Group 3 Na-Cl character is more evident in the 2012 survey. Between [I] and [J] both surveys have three wells that exhibit contrasting results and suggestive of heterogeneous water quality. Further south at [J] to [K], data is only available for the detailed survey where an increase in salinity is expected towards the lowland - marshlands area.

North of the central plain area [L], the detailed 2008-9 survey conclusively confirms Group 1 groundwater associated with the hard-rock escarpment that the 2012 survey also suggests. Around [M] and continuing to [N], both surveys indicate the emerging significance of Group 2 groundwater that continues, per the detailed survey only, to assume greater significance further south through [O – P]. With increasing TDS at [P] and [S] in the south lowland - near-river - likely shallow groundwater - setting, there is again evidence for transition to expected Group 3 character.

The near – escarpment western margin of the alluvial plain [T – X] was only sparsely sampled in the 2012 survey and provides a misleading characterisation suggesting mostly Group 1 low TDS groundwater. The detailed survey shows a high TDS Na-Cl Group 3 groundwater extending from the escarpment along the southern tributary. Monjerezi et al., (2011) attribute this salinity to dissolution of halite and gypsum evaporite salts within the adjoining sedimentary Karoo and Cretaceous Lupata sandstones followed by mineralised seep from these units into the relatively thin eastern alluvial aquifer likely prone to such influences. Evidence of a more Na-Cl groundwater is only very tentative in the 2012 survey at [T] and Group 4 samples at [Q] that have Group 3 similarity. These observations reveal the significance of increased sampling at the plain-aquifer margins.

The reader is referred to the main manuscript for the development of the above detail into the system conceptual model.

* Monjerezi et al. (2011) undertake a hierarchical cluster analysis (HCA) and principal components analysis (PCA) of their entire major ion dataset (247 samples over a greater area) with a resultant HCA classification to three main clusters: C1 of dominant composition Na-Cl and median TDS of 3436 mg/l; C2 of dominant composition Na-HCO₃ and median TDS of 966 mg/l; and, C3 of dominant composition Ca/Mg- HCO₃ and median TDS of 528 mg/l. These compositions compare to our 2012 survey groups and we reasonably assume Cluster C1 corresponds to Group 3, C2 to Group 2 and C3 to Group 1. This permits direct comparison of the data with similarly coloured symbols in Fig. SD1 and manuscript figures. TDS values are comparable (the 2012 survey median TDS data Group 3 at 1624 mg/l, Group 2 at 857 mg/l and Group 1 at 422 mg/l compared to above, although our somewhat lower value for the Na-Cl median occurs due to the greater sampling of lowland areas predominantly in the south with increased TDS by Monjerezi et al. within and beyond our study area. This is also apparent in terms of proportions of the water types sampled by each survey in the actual study area, the 2008-09 versus (vs.) 2012 comparison being: Group 1 - 40 % vs. 58%; Group 2 - 40 % vs. 27%; and, Group 3 - 40 % vs. 12% (indicated in the manuscript).

Table SD3. Suite of water quality determinants typically analysed by the Water Quality Services Division of the Department of Water Resources within Malawi's Ministry of Agriculture, Irrigation and Water Development (MoAIWD).

Determinant

pH
Electrical Conductivity (EC)
Total Dissolved Solids (TDS)
Carbonate
Bicarbonate
Chloride
Sulfate
Nitrate
Fluoride
Sodium
Potassium
Calcium
Magnesium
Total Iron
Manganese
Turbidity
Suspended Solids (SS)
Total Hardness (as CaCO₃)
Total Alkalinity (as CaCO₃)
Copper
Phosphate
Dissolved Oxygen (DO)
Chemical Oxygen Demand (COD_{cr})
Biochemical Oxygen Demand (BOD₅)
Faecal coliforms
Faecal streptococci

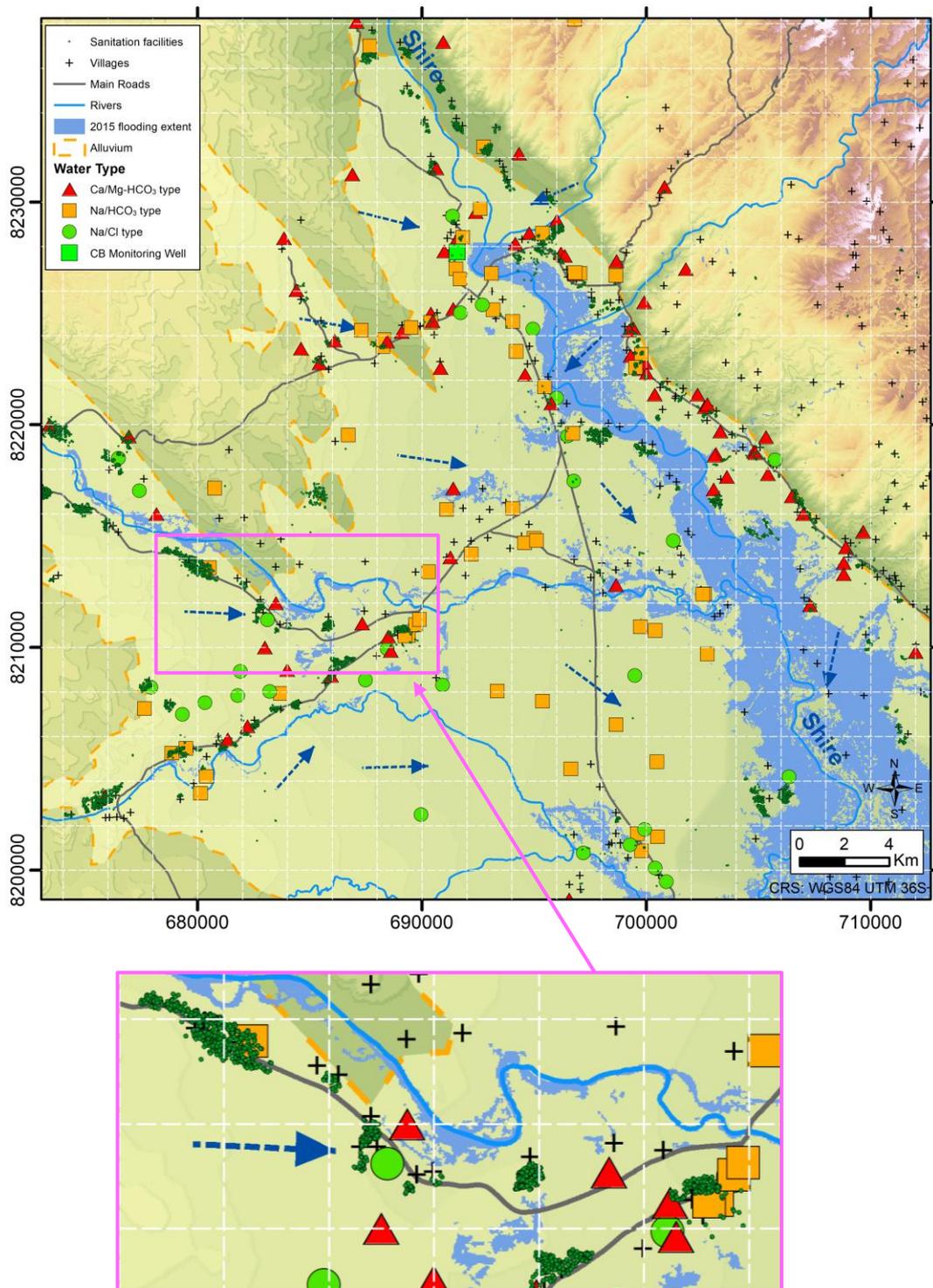
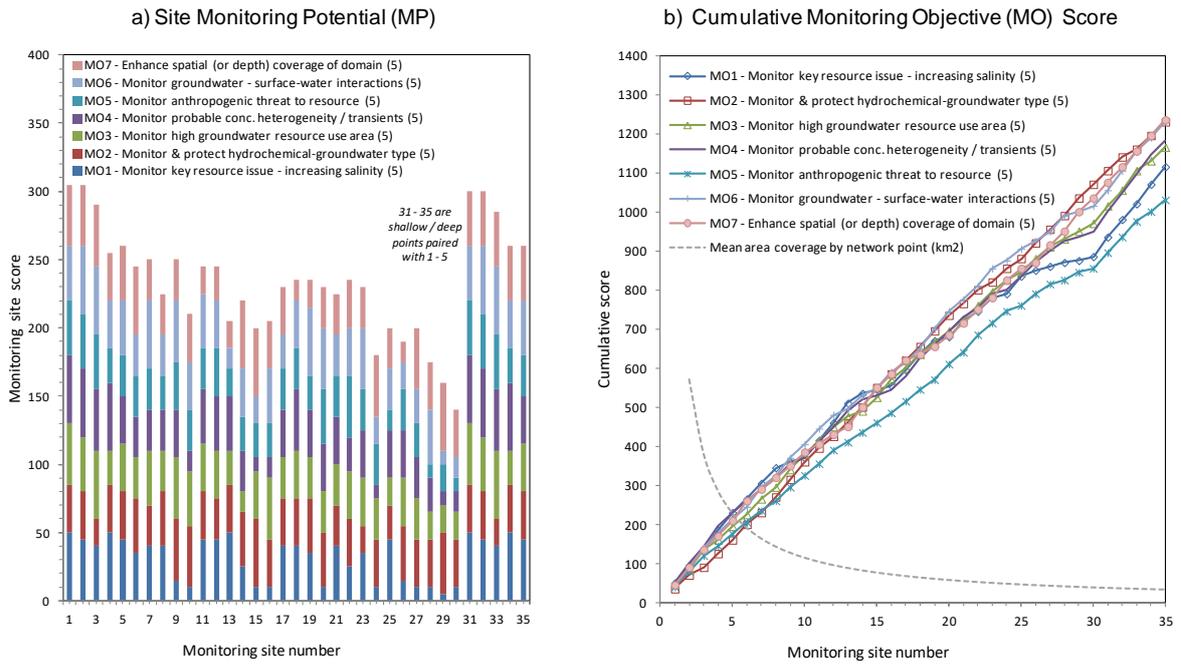


Fig. SD2. Mapped occurrence of sanitation facilities (typically pit latrines) in the study area (as of November 2017). Survey mapping is typically undertaken within a 100 m radius of a waterpoint, however, not all areas have been mapped and hence the dataset is incomplete. It is nevertheless indicative of the high pit latrine densities that may occur in some mapped areas. Such data may inform assessment of monitoring objectives relating to monitoring of anthropogenic (MO5) or microbiological pathogen risks.

Equal weighting



Prioritised weighting

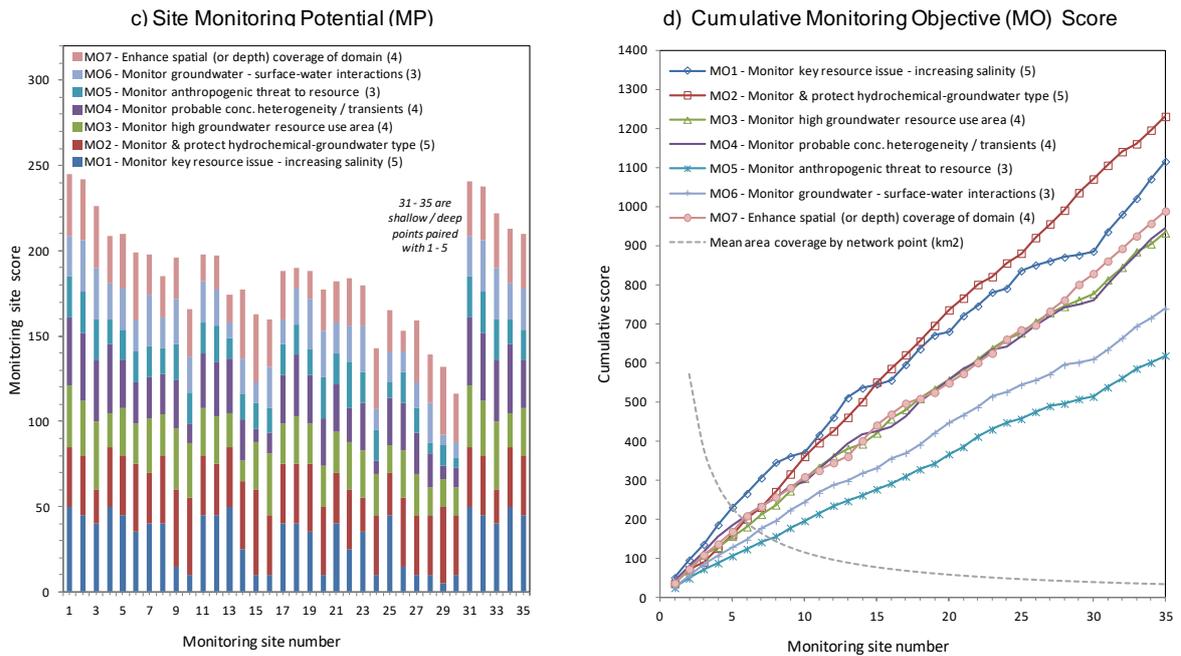


Fig. SD3. Comparison of proposed Chikwawa network site Monitoring Potential (MP) and cumulative (with increasing network size) Monitoring Objective (MO) scores for assumptions of equal and prioritised weighting of MOs (Table 1 of main manuscript). Plots (c) and (d) are also included in the main manuscript as Fig. 8.

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