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A review of urban groundwater use and water quality challenges in Sub-Saharan Africa

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GROUNDWATER PROGRAMME

OPEN REPORT OR/17/056

A review of urban groundwater use and water quality challenges in Sub-Saharan Africa

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Keywords

Groundwater quality, Urban, per-urban, Sub-Saharan Africa, review.

Front cover

Municipal groundwater supplies for Kabwe, Zambia.[Lapworth DJ]

Bibliographical reference

LAPWORTH, D J, STUART, M E, PEDLEY, S, NKHUWA D C W and TIJANI, M N. 2017. A review of urban groundwater use and water quality challenges in Sub-Saharan Africa. *British Geological Survey Open Report,* OR/17/056. 133pp.

Keyworth, Nottingham British Geological Survey 2018

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Foreword

This report is the published product of a study "Mapping groundwater quality degradation beneath growing rural towns in Sub-Saharan Africa" (NERC grant number NE/002078/1) led by the British Geological Survey (BGS) in collaboration with the University of Surrey, University of Zambia and University of Ibadan. It is an output from Work Package 1 of the project, to review groundwater quality studies in urban and peri-urban sub-Saharan Africa. A related article has recently been published by Lapworth et al. (2017) in the *Hydrogeology Journal*. This study was funded through a Catalyst Grant awarded under the NERC programme "Unlocking the Potential for Groundwater for the Poor–Sub-Saharan Africa (UPGro)". The work is jointly funded by UK's Department for International Development (DFID), Natural Environment Research Council (NERC), and the Economic and Social Research Council (ESRC). The views expressed do not necessarily reflect the UK Government's official policies.

Acknowledgements

The Mr Alex Nwegbu, Director General of the Nigerian Geological Survey Agency (NGSA), and Mr Oyedeji Oluwole, Director of Applied & Engineering Geology (NGSA) are acknowledged for making reports available from urban water quality studies in Nigeria. We also thank Mr Alichi-Alichi (NGSA) for input to the Ibadan case study.

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Large parts of Sub-Saharan Africa (SSA) are experiencing rapid urban population growth. Current projections estimate that by 2030 half the population of SSA will be living in urban areas, and if current trends persist half of these people will reside in slums. According to UN-Habitat, access to safe drinking water for slum dwellers is a major priority (UN-Habitat, 2003). Because of the lack of services, self-provision of water, using groundwater sources, is widespread in slums and is likely to increase as the urban centres expand, putting pressure on the quality and quantity of the groundwater resource.

Over the last three decades there has been a concerted effort to increase access to improved water supply and sanitation across Africa. Within urban, and peri-urban, settings this has led to widespread development of groundwater resources for domestic water supply, mainly through growth in the private groundwater supply sector, and the proliferation of on-site sanitation, largely through the use of pit latrines. Due to the high population densities, latrines are often constructed in very close proximity to wells and springs used for drinking water, and can be a significant source of contamination. The lack of adequate management of household and industrial waste in many urban centres is also a growing concern. This has led to the groundwater resources being put under considerable stress in terms of water quality and in some cases water availability.

The parallel deterioration of the water quality and availability in urban areas in SSA and increases in demand challenge the sustainability of the current situation. This has occurred over the last three decades for most urban centres in SSA (Foster et al., 1996). While there is not the very long historical legacy of urban pollution in many urban centres in SSA, in contrast with Europe and Asia for example, the problem of long term pollutant loading to groundwater is now becoming a reality.

There is currently a limited evidence base of groundwater quality studies in urban centres across SSA with which to inform long term policy on the development of urban and peri-urban groundwater resources. This document provides an overview of urban and peri-urban drinking water and sanitation in SSA as well as urban groundwater development and degradation. The review draws on published and grey literature from across SSA and is both a useful entry point for researchers in this area and is a baseline reference work on the topic of urban groundwater quality in SSA. Detailed case studies from West Africa (Ibadan, Nigeria) and Southern Africa (Lusaka, Zambia) are included in this document. It provides: i) a detailed review of empirical studies assessing groundwater degradation (chemical and microbiological) in urban groundwater across SSA, ii) assesses urban groundwater quality issues in relation to groundwater vulnerability and hydrogeological controls, iii) identifies gaps in the current evidence base regarding groundwater quality and risks to groundwater pollution and human health in urban SSA.

1 Introduction

1.1 URBAN POPULATION GROWTH AND WATER SUPPLY IN AFRICA

This review examines the quality of groundwater beneath peri-urban areas in SSA and the factors that impact upon the quality. Peri-urban is not a well-defined concept and can change depending upon the situation (WHO, 2006). Webster's Dictionary defines peri-urban as "of or relating to an area immediately surrounding a city or town"; effectively the boundary between urban and rural areas. Nyambe et al. (2007) expand the definition of peri-urban to include "unplanned urban settlements developed due to the rapidly increasing population".

Urban and peri-urban settlements are expanding rapidly across large parts of Sub-Saharan Africa (SSA), often at a rate that exceeds the capacity of countries to supply basic services. Over the last three decades there have been several initiatives intended to increase access to improved water supply and sanitation across SSA (Bartram and Cairncross, 2010). A recent example is the UN Millennium Development Goals, which had the target of halving by 2015 the population without sustainable access to improved sanitation from the baseline level of 1990 (JMP, 2008). These have recently been succeeded by the UN Sustainable Development Goals (SDGs), which are intended to promote the continued progress in a broad range of development goals including: Goal 6 to 'Ensure availability and sustainable management of water and sanitation for all'; Goal 9 to 'Build resilient infrastructure…'; Goal 11 to 'Make cities and human settlements inclusive, safe, resilient and sustainable'. Groundwater has a pivotal role to play in achieving progress towards the SDGs.

Groundwater is considered a centrepiece of improved drinking water, and the provision 'safe-supply' in many parts of SSA, and many urban centres in SSA are dependant partly or wholly on groundwater (Showers, 2002). Within the urban and peri-urban context the important role of groundwater has resulted in widespread development of groundwater resources beneath, and in close proximity to, urban centres. However, the high population densities found in urban areas has also led to the proliferation of unimproved and improved sanitation provision largely through the use of pit latrines, often positioned in very close proximity to wells and springs used for drinking water. Furthermore, the absence of adequate management of household and industrial waste in many urban centres is putting additional pressure on groundwater in terms of water quality and water availability.

Future trajectories of growth, both spatially and demographically, of urban centres in SSA and increased demands on groundwater resources mean that urban centres are likely to be more dependent on groundwater in the future. Per-capita usage is also predicted to rise in line with prosperity, putting additional stress on available groundwater resources. Together, these factors challenge the sustainability of current management of water resources in urban areas across SSA. The present situation is compounded by a limited evidence base regarding groundwater quality studies in urban centres across SSA with which to inform long term policy on the use of urban groundwater resources.

1.2 BACKGROUND

Urban and peri-urban populations in SSA are expanding at increasing rates. A study by the UN population fund (UNPF, 2007) estimates that between 2000 and 2030 Africa's urban population will double and become the majority compared to the rural population. Currently, 30% of Africa's population are urbanised (a greater proportion compared to South Asia) and average urban growth rates are approximately 5% (UN, 2005). Although inward migration from rural areas will continue to increase the population or urban areas, the UN predicts that the main growth will come from natural increases in urban populations (UN, 2005). The urban population is also predicted to be distributed in a large number of smaller towns (<200,000) rather than concentrated in large cities. While smaller urban centres may be viewed as more adaptable, from a geographical and policy making standpoint, they often lack the skilled personnel, resources and infrastructure needed to develop in a sustainable way. Current assessments suggest that levels of urban poverty are comparable with those in rural areas, and the proportion of the urban population living in slums in SSA is over 50%, the highest proportion by a factor of two compared to other regions globally (UN-Habitat, 2003).

The Millennium Development Goals (MDGs) originally set out to increase access to safe water supplies, but water safety is difficult and expensive to monitor. To create a viable alternative to safety, the MDG for water was revised to assess the type of facility that was used to supply the water and whether or not it might be considered to be "improved". This has been further refined under the current SDGs. Under the MDG for the purpose of monitoring, an improved source was considered to be safe. Recent work, however, has shown that "improved" and "safe" are not synonymous and that many improved water sources are not safe water sources (Linkov and Seager, 2011). Under the recent SDGs a new definition of '*safe supply*' which includes water quality and access criteria has been introduced. SDGs 6, specifically target 6.1, have defined '*safe managed supply*' as drinking water sources free of microbiological and chemical contamination, with no seasonality and which are supplied at the household level, a '*basic supply*' is defined as an improved source within a 30 min round trip. This is a huge subject in its own right and will not be considered further in this review.

Groundwater is often seen as the most important component of an 'improved' water supply as it is often more reliable than surface water, less vulnerable to pollution and therefore less expensive to treat, and more resilient to climate variability (MacDonald et al., 2011). Many expanding urban areas in SSA are dependent on groundwater for their water-supply (Adelana et al., 2008; Foster et al., 1999). However, despite government and NGO efforts to increase access to safe groundwater, the most vulnerable people still rely on inadequate sources of deteriorating quality.

The change from rural to urban land use causes a dramatic change in groundwater recharge, as well as in water quality and demand (Kittu et al., 1999; Lerner, 1990). This has important implications for resource sustainability and the balance between water supply and growing urban population demand. Compared to Latin America or Asia (Chilton, 1999; Lawrence et al., 2000), there have been few detailed studies that assess urban groundwater resources in SSA (Howard et al., 2003; Taylor et al., 2009). Consequently, there is a limited evidence base on groundwater quality status in SSA with which to inform policy as well as the necessary institutional capacity, facilities and skills (Starkl et al., 2013). Existing studies largely include only basic chemical and microbiological parameters, and have focused on large cities or slums (Wright et al., 2012; Xu and Usher, 2006). There is also a paucity of robustly designed, i.e. statistically representative, studies to draw conclusions from. Given the large number of smaller but rapidly growing rural, groundwater-dependent towns in SSA, the development of effective strategies to protect the resource from contamination is a priority and effective tools for mapping risk to groundwater are needed by a range of stakeholders.

Developing groundwater resources is attractive from an installation and maintenance standpoint, as well as the reduced costs associated with infrastructure and the often minimal requirements for water treatment. However, urban groundwater is easily contaminated from a range of sources and along with increased abstraction rates, this can lead to locally reduced water quality and falling water levels (Cronin et al., 2006; Morris et al., 2000). As population density increases and urban centres expand the increasing pressures on the groundwater resource have a disproportionate impact on poorer inhabitants (Grönwall et al., 2010), who have a greater dependence on shallow unregulated and untreated groundwater sources. Several factors combine to perpetuate these conditions including, inadequate characterisation of the groundwater resource, limited planning and the inadequate supply of piped-water from official institutions. As well as the need for investment and structural reform to the water sector in SSA, there is clearly a very important role for NGO and community-based organisations to help fill the gap and provide local expertise (Allen and Bell, 2011; Allen et al., 2006a). Maintenance is a critical issue for water availability and quality, both in SSA and elsewhere, and has a direct influence on how communities use and relate to a range of different water sources. There are also important cultural factors that control the use and adoption of improved water and sanitation provision, and these may operate on a longer time scales than those of individual projects or policies.

Several studies have shown that faecal contamination of shallow groundwater can be elevated during periods of intense rainfall, increasing the risk of widespread gastrointestinal disease. These observations have important implications for climate change which is expected to change the intensity and frequency of rainfall events (Howard et al., 2010; Howard et al., 2003; Taylor et al., 2009; Taylor et al., 2008; Taylor et al., 2009). In addition to microbiological contamination, and naturally-occurring water quality problems such as fluoride and arsenic, urban aquifers are also at risk of pollution from a large range of contaminants including nitrate, toxic metals, hydrocarbons and micro-organics (Adelana et al., 2008). This is due to inadequate waste management and regulation in new rapidly growing centres, and because groundwater protection is rarely considered a priority during early development (Howard et al., 2006).

The World Health Organization (WHO) Guidelines for Drinking-Water Quality are published with the purpose of protecting public health, however, '*Safe drinking-water, as defined by the Guidelines, does not represent any significant risk to health over a lifetime of consumption, including different sensitivities that may occur between life stages*' (WHO, 2006). Using the WHO Guidelines as a reference many countries in SSA have drafted national standards for drinking water quality. National standards take into account local factors such as geology, geography, social as well as economic factors and as such may differ from WHO guideline values significantly. While standards for drinking water quality may be published at the national level, there are many different approaches across SSA to implementing the standards including, frequently, minimal or no enforcement. The amount and quality of groundwater quality data currently available across SSA is highly variable, and this, to some extent, is a consequence of inadequate implementation and regulation of drinking water quality standards in this region.

1.3 STRUCTURE AND SCOPE OF REVIEW

This report comprises a review of the state of current understanding of groundwater quality beneath urban centres and peri-urban areas in SSA and the issues that contribute to changes in groundwater quality. The report will cover published and grey literature. There is a wealth of information available for large cities but the small rural towns and the rapidly expanding peri-urban zones are much less well characterised. This report provides a much needed synthesis of published groundwater quality studies carried out in urban and growing urban centres in SSA. We will also make a broad assessment of likely future water quality pressures on this critical resource bringing expertise from across SSA and include socio-economic aspects of water supply degradation.

This report reviews and synthesises existing understanding of water quality in urban centres and per-urban centres (including case studies from Southern, East and West Africa) to provide an authoritative assessment of the current state of knowledge of urban groundwater quality in SSA. This document provides a) an overview of the key issues related to groundwater development in an urban context and an entry point for reader new to this topic, b) a systematic review of empirical studies assessing groundwater degradation in urban groundwater in SSA, c) identifies gaps in the current evidence base regarding groundwater quality degradation in urban groundwater. It will also include a review of novel methods of assessing the sources of groundwater pollutant.

Published information was sought predominantly from documents available on the internet on groundwater quality, pollution assessments and supporting data from a wide range of African countries (Figure 1.1). As part of this review a database of literature were compiled of articles, books and reports using Web of Science, PubMed, Google Scholar, ScienceDirect, as well as reference lists available in published documents. Grey literature, available in the BGS Groundwater Archive at Wallingford, UK, and in-country documents were used to inform the case studies from Nigeria, Zambia and elsewhere. The geographical extent of the review covers a large part of SSA and includes the southern parts of Mali, Niger, as well as Ethiopia and South Africa.

Figure 1.1 Countries in Sub-Saharan Africa covered by this review. Source ArcGIS 10.1 data sets (Zhang et al., 1995).

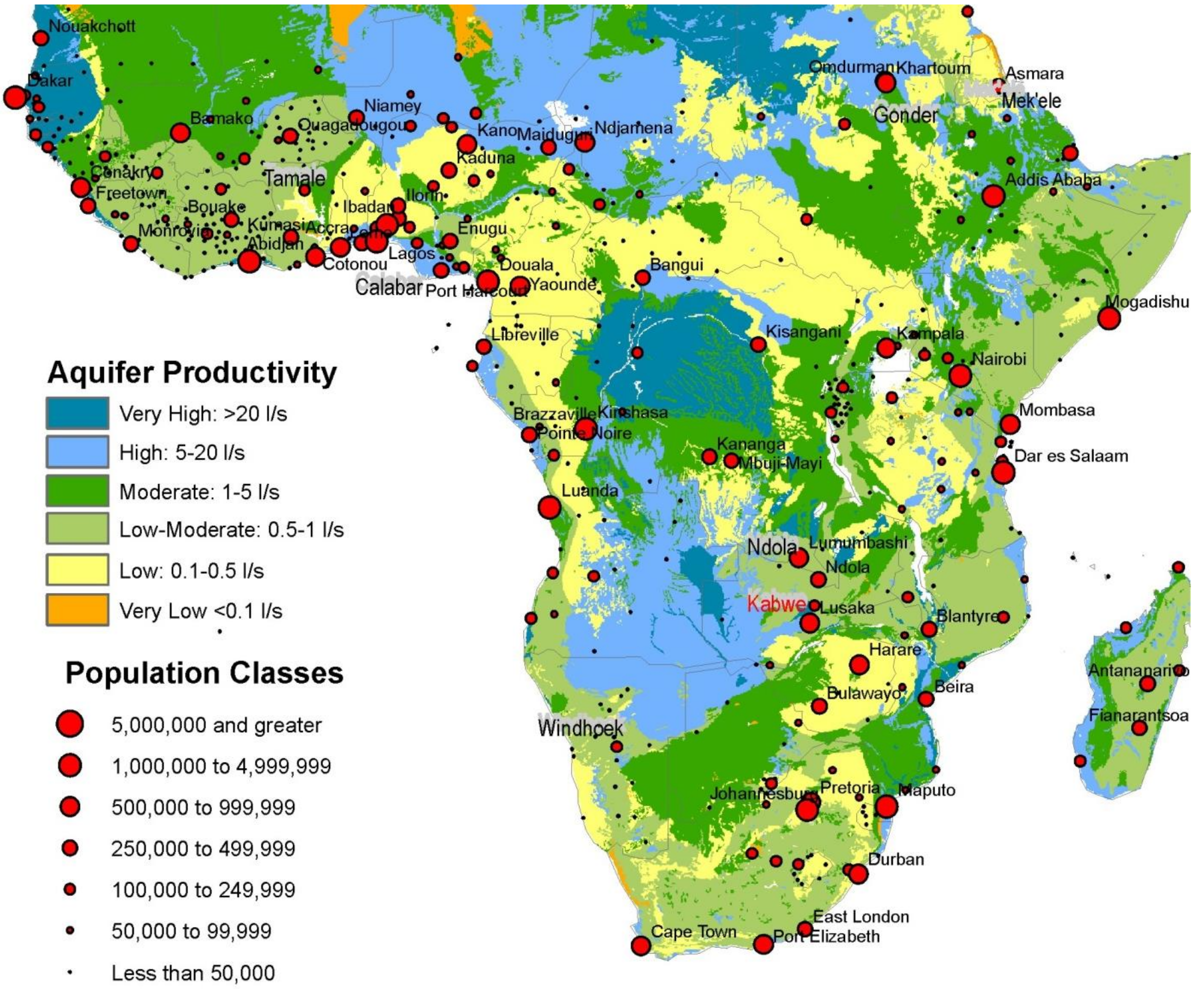

Figure 1.2 Major cities in SSA by population and relationship with groundwater storage Population data from ArcGIS 10.1 data sets (Zhang et al., 1995), aquifer productivity from MacDonald et al. (2012).

Figure 1.2 shows the major cities coloured by population underlain by quantitative groundwater storage maps based on porosity and saturated aquifer thickness (MacDonald et al., 2012). Urban centres with populations >500,000 are displayed and these include most of the locations covered in this review. The population density in Nigeria stands out and Lagos and Kinshasa are the largest cities, although Nigeria also has significant areas of urban sprawl such as Ibadan. It is clear that there is a large concentration of urban centres in coastal regions, and the most densely populated regions are in West Africa and selected parts of East Africa. Major urban centres are underlain by aquifers with a range of estimated aquifer productivity, including many that have low-moderate productivity (0.5-1 l/s) or less. Several large urban centres on the coast of West Africa, from Luanda round to Nouackchott intersect with moderate-high productivity alluvial aquifers; however, these tend not to be regionally extensive with the exception of the Niger Delta aquifer and the Senegal-Mauritania sedimentary basin. The largest and most highly productive aquifers are found in the interior of Africa, in northern Niger, Chad, Mali, Angola and DRC, all of which have low population densities. Figure 1.3 shows the population density and growth in Africa including hot-spots of urban growth in West and East Africa.

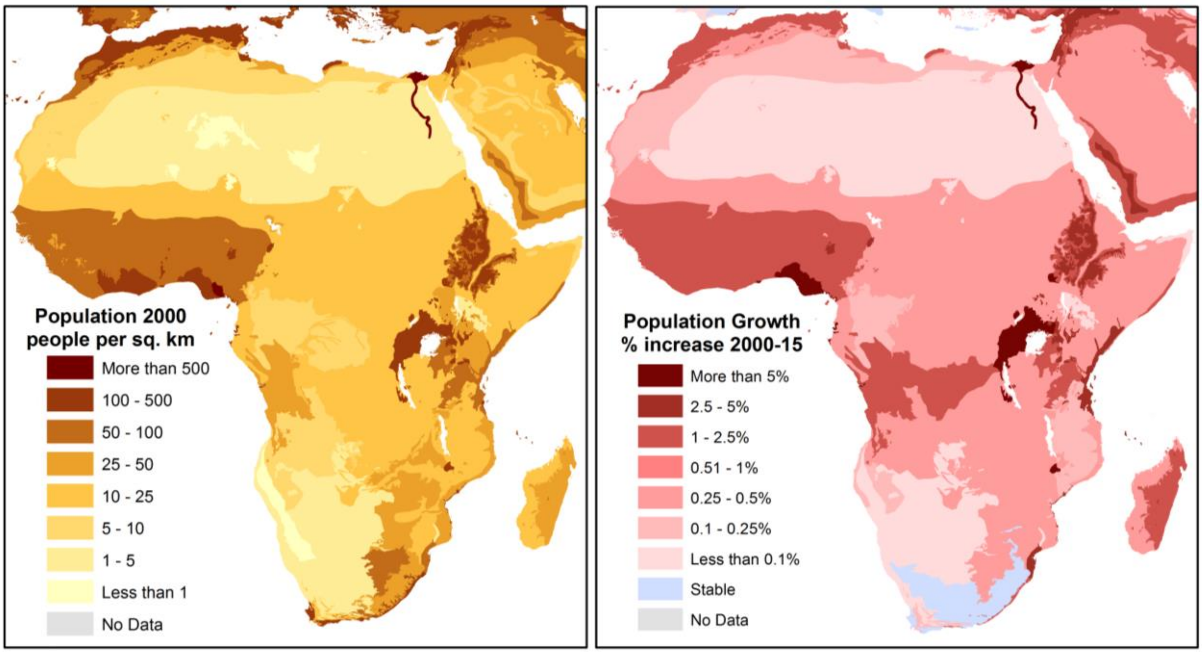

Figure 1.3 Population density and growth hot-spots in Africa (data source: UNEP)

2 Groundwater in urban Sub-Saharan Africa

2.1 THE ROLE OF GROUNDWATER

Groundwater is the critical underlying resource for human survival and economic development in extensive drought-prone areas of south-eastern, eastern and western Africa, especially where the average rainfall is less than 1000 mm/a (Foster et al., 2006). Rainfall tends to be variable year to year, which creates difficulties for managing the quantity and quality of groundwater resources, and this variability is likely to increase with the growing impact of climate change (Carter and Parker, 2009). There are also a number of trans-boundary aquifers which require strategic management (Ashton and Turton, 2009). The proximity of groundwater sources, and the related reduced infrastructure costs, makes urban groundwater an ideal resource to target for development in SSA (Foster et al., 1998; Taylor and Barrett, 1999); however, the susceptibility of groundwater to contamination in urban settings has to date received little attention compared to other regions globally.

In large parts of rural SSA the functioning of infrastructure facilities is dependent on groundwater supplies. Groundwater is used for urban water supply, industrial and tourist development and as a critical resource for the development of mining and other industrial activity (Foster et al., 2006). Demand for groundwater is already high and will continue to grow for the foreseeable future.

Africa's population is increasing rapidly, often exceeding 5% growth per annum. With a four-fold increase in urban population projected over the next 50 years the provision of safe water and food will be a significant challenge (Carter and Parker, 2009). In Africa about one third of the total population lack adequate access to improved water supplies and approximately 70% are not using improved sanitation. The widespread lack of access to these basic services are thought to be the root cause of many diseases that affect Africa (Braune and Xu, 2010).

In a substantial number of large towns groundwater is vital to the continuity of water supply and plays a key role in droughts and other emergencies. Yet in only a few cases has the use of groundwater evolved as part of planned water-supply development; more commonly it has occurred in response to water shortage or service deficiency and often through private initiative (Foster et al., 2006). Groundwater has a vital role in serving the large informal areas that do not have ready access to piped water supplies but this usage is often not formally recorded (Braune and Xu, 2010). In addition, wastewater infiltration in urban areas is a growing concern for groundwater quality (Foster et al., 2006).

While regional scale assessments of groundwater resources have been recently quantified (MacDonald et al., 2012), hydrogeological evidence at the appropriate resolution for managing groundwater resources in many African urban centres is non-existent. There is a lack of appropriate approaches and investment for planning, financing and sustainably using this important resource (Braune and Xu, 2010). Additionally there has been concern over:

- The decline of national institutions responsible for groundwater development, resource administration, groundwater protection and database management, as well as loss of professional personnel since the 1980s (Foster et al., 2006).
- Legislation not catering for community-based arrangements but focussing on centralised government permits at the river basin scale.

2.2 URBAN AND PERI-URBAN DEVELOPMENT

There continues to be a lively discourse regarding population dynamics, urbanisation and urban economies in Africa. One perspective is that rapid urban growth, some fuelled by in-migration from rural areas, has economic benefits (UN, 2005). While the rates of urbanisation may still be overestimated (Potts, 2009), there is a clear consensus that there is undoubtedly significant growth in the urban population in SSA, whether this is through natural growth, in-migration or simply the reclassification of settlements based on somewhat arbitrary thresholds.

Simon (2008) defines the peri-urban zone or rural-urban fringe, which he distinguishes from suburbs, using urban influence, changing land use, access to infrastructure, services and markets, and exposure to urban processes and pollution. He concludes that it is best visualised as a continuum which varies in extent both spatially and temporally. Mpofu (2013) set out the main environmental challenges for urbanisation in Sub-Saharan Africa as crowded living conditions, poor disposal of wastes, inadequate basic infrastructure, pollution of water and the decline of urban green areas. The causes are natural urban population growth, rural to urban migration, poor development control, weak institutions and inadequate financial resources.

2.2.1 Evolution of groundwater development during expansion of urban centres

Foster et al. (1998) summarised the evolution of groundwater in urban settlements and outlines the interdependence of groundwater and urbanisation. A four stage model was proposed:

1) Early settlement:

- Water supply obtained from shallow urban wells and boreholes
- Wastewater discharged to ground
- Pluvial drainage to groundwater or water courses

2) Town-city development:

- Water table lowered beneath city and wells deepened
- Wastewater discharged to ground
- Shallow groundwater in town/city becomes polluted
- Expansion of pluvial drainage to groundwater/water courses

3) City starts to expand:

- Local aquifer beneath city is largely abandoned due to contamination
- Water table rebound occurs in city due to combination of high urban recharge and low pumping rates
- Explore conjunctive use with sources outside of urban centre
- Significant water table decline in per-urban area due to development of well fields on edge of city
- Continued contamination of urban groundwater due to urban recharge

4) Further expansion of city:

- Well-fields unable to cope with increased demand of city
- Rise in costs associated with conjunctive use of water from peri-urban and distant sources
- Steady water table rise in city nucleus leads to problems of flooding and waste water disposal
- Reduced scope for pluvial drainage to groundwater

This generalises the situation regarding the development of urban groundwater use and degradation. In reality there may be significant differences in the progression of urban groundwater development across SSA due to fundamental hydrogeological and geographical differences, as well as contrasting national and regional approaches to planning and investment in the water sector. Groundwater use and development will also depend on socio-economic factors; for example, the least prosperous areas in urban centres often continue to use unimproved shallow groundwater sources for a number of domestic purposes during phase 3-4, not least because of unreliable and unaffordable supplies, but also because these areas are often located where there are shallow groundwater tables and are prone to flooding e.g. Lusaka. The issue of how previously discrete urban centres merge over time to produce very large and essentially continuous sprawling urban areas (e.g. Lagos and Ibadan in Nigeria), and the competition between urban and peri-urban groundwater for irrigation and agricultural activities are two common issues that are not included.

2.2.2 Urban expansion

Many African economies have undergone a period of structural adjustment to their economic programmes with the overall aims of reducing state-based interventions and balancing the books (Briggs and Mwamfupe, 2000). This has resulted, amongst others things, in the privatisation of previously state-owned agencies and liberalisation of planning regulations, potentially allowing investment and expansion of the peri-urban zones of cities.

Briggs and Mwamfupe (1999, 2000) describe the history of urban expansion of Dar-es-Salaam, Tanzania. During the economic crisis of the 1980s, many residents bought land and moved to the edge of the urban area in order to produce their own food. These settlements tended to be along arterial roads to allow some household members to continue to work in the city. The original landowners were displaced and moved further out to cheaper land to continue farming. As the economic situation improved the peri-urban zone was developed for permanent housing and for the expansion of commercial agriculture supplying the city with milk, vegetables and fruit. The deregulation of public transport allowed a system of minibuses and similar to service the interstitial areas. Together with increased private vehicle ownership this has allowed dense settlement at locations away from the arterial roads. More recently a shortage of capital has slowed this process down and a lack of business confidence has restricted the development of large factories or other enterprises. Low income groups have been increasingly excluded from peri-urban land as land has been commoditised and wealthier urban groups have benefitted.

Yankson and Gough (1999) made a study of the urbanisation of Accra, Ghana. This fishing village grew rapidly after becoming the seat of British administration in the late 1800s. The stagnation of the economy in the 1970s and 80s resulted in a breakdown of service provision and the deterioration of existing infrastructure. Since then new residential areas have appeared very rapidly with a fragmented urban structure and lack of development controls. There was a shortage of land in the "fringe zone" for farming and degradation of remaining agricultural land due to increased pressures from urban development. The peri-urban environment has changed with far fewer trees as land was converted first to agricultural use and then to residential. There is pressure on water courses and erosion has increased. Many areas did not have a reliable piped water supply and inhabitants had to use poor quality surface water supplemented by treated water from vendors. None of the houses in the peri-urban area had mains sewerage and many areas used improved pit latrines, with others using pan latrines or what the authors term the "free-range" mode. There was no system for solid waste collection, with many people using empty fields or burning it where possible.

Investment in African cities is often externally driven and may not reflect local conditions or benefit local communities. It can also lead to very polarised socio-economic conditions (Mbiba and Huchzermeyer, 2002). In Mombasa investment has been made in tourist facilities at points on the urban periphery, whilst at the same time the urban area deteriorates (Mbiba and Huchzermeyer, 2002). The remainder of the peri-urban area comprises informal settlements where 30% of the population live below the poverty line. In Zambia the informal peri-urban settlements at main city entrance points are said to be an embarrassment to the authorities, who have made efforts to evict or relocate the residents (Mbiba and Huchzermeyer, 2002). In Lusaka half the population live in illegal or informal settlements. Urban peripheries attract some migrants who do not become town dwellers but inhabit or create peripheral villages for subsistence farming (Swindell, 1988).

Frumkin (2002) described the impact of urban sprawl on water resources in developed countries as increased run off and less recharge, leading to water shortages in one third of US communities. Water quality can be affected in many ways. In developed countries there is better control of point sources such as factories and sewage treatment works but non-point sources remain an issue. These include oil, grease and toxic chemicals from roads and car parks, sediment from improperly managed construction sites and erosion. Suburban development was associated with polyaromatic hydrocarbons, heavy metals, such as zinc and organic waste.

A clear and consistent picture emerges for the peri-urban areas of Kumasi, Ghana (Simon et al., 2004). Common factors are:

- Increased population size due to in migration in search of building land
- Declining area of agricultural land and land under cultivation
- Reduced farm size due to population growth
- Decreased soil fertility close to villages with shortened fallow periods and reduced crop yield
- Increases in facilities and infrastructure – boreholes and dug wells, access roads, schools, pit latrines with some ventilated improved pit latrines (VIP), electricity and piped water for those able to pay
- Increased mining of sand within villages
- Sale of land for housing and commercial development

2.2.3 Urban food security and peri-urban agriculture

As the city expands, the greater proximity and access to the enlarged urban market provides new opportunities to increase peri-urban agriculture and to specialise in higher-value horticultural crops, with both increased risks and returns (Simon, 2008). Urban agriculture occurs behind houses, on roofs, along roadsides, railway lines and under power lines, in the middle of roundabouts and in parks (Cofie et al., 2003). In West Africa, Swindell (1988) records intensely cultivated peripheral zones around capital cities which have developed since the 1930s and extend outwards for some 10 km, e.g. around Brazzaville, Congo; Libreville, Gabon; Bangui, Central African Republic; and Bouaké, Ivory Coast. In the Kinshasa periphery there was considerable pressure on land for speculative building and farming and this has led to a zone of intensive farming stretching into the lower parts of the DR Congo. In cities across West Africa such as Dakar, Bamako, Accra, and Ibadan, urban agriculture provides a significant source of income, supplementing other economic activities (Cofie et al., 2005; Cofie et al., 2003).

In East Africa e.g. in Kumasi and Dar-es-Salaam, home gardening or urban agriculture is very common. In both cases urban growing of staple crops, is largely carried out by women and children to reduce household expenses. Open space production generally supplies cities with fresh leafy vegetables and fruit. Farming in larger spaces provides crops such as maize or sorghum and export crops such as flowers. In convenient locations, e.g. near airports, there may be specialist crops such as pineapples. In Dar-es-Salaam, urban agriculture forms 60% of the informal sector and is the second largest urban employer. In Nairobi and Lusaka farming households produce 20-30% of their food requirements, whereas in poorer Harare and Kampala up to 60% is produced. The next stage in expansion is the loss of cultivatable land to a combination of land sale and degradation, and a reduction in food self-sufficiency and surplus for sale (Simon, 2008).

2.2.4 Broad groups of contaminants and threats to groundwater resources

Urban groundwater resources are at risk of pollution by a large number of different broad groups of contaminants including microbiological pathogens, heavy metals (e.g. lead cadmium, and zinc), macronutrients (nitrogen, carbon, potassium and phosphorus), pesticides and herbicides, and a diverse array of other organic contaminants. High concentrations of naturally occurring contaminants such as fluoride, arsenic and uranium are also of concern, and although these are largely controlled by natural processes, anthropogenic influences can affect the hydrogeological regime and hydrogeochemical environment and hence their mobility and transport in the groundwater system. From a health risk standpoint microbiological pathogens and toxic metals are of particular concern and have been the focus of the majority of published studies to date.

Often excessive and inappropriate amounts of fertilisers and pesticides are used for peri-urban agriculture, contaminating crops, soil and water and causing health problems for farmers (Simon, 2008). Due to shortages of land, water and fertiliser, waste by-products and contaminated water are often used for agriculture in peri-urban areas (Binns et al., 2003). In Kano, Nigeria, water treatment and even water supply are non-existent and poor people cannot afford sanitation. Industrial effluents, particularly tannery wastes containing chromium (Cr) and cadmium (Cd), are a threat to local irrigation water as these are routinely discharged to open drains or water courses without treatment. Where factory or sewage waste are applied directly to land these can also carry viruses and high levels of faecal coliforms. There were also high levels of Pb in unpaved road dust and in dust deposition.

2.2.5 Access to water

The provision of water in densely populated peri-urban settlements is complex, involving a mixture of piped water supplies, self-provision and deliveries by small independent water providers (Sorensen et al., 2015a; Matsinhe et al. 2008). Where consumers make their own provision, the distance to a supply and number of users per facility have an impact on the time spent collecting water and therefore the amount of water collected and used (

Table **2.1**) (Hofmann, 2005). Poorer households often have fewer facilities for storing water in the home (Mweemba et al., 2011) and so will use less water, or spend more time collecting water (towards the left of Table 2.1). At the low end of the scale, adequate volumes for consumption and hygiene cannot be assured. However, as access increases, consumption increases and the volumes of water available for hygiene increases with the result that health concerns may decrease. But the task of collecting water is not evenly distributed across the community. Studies have shown that over 70% of all trips to collect water are made by women and girls over the age of 15 (Trick et al., 2005) with consequent effects on their education and their health.

Hofmann (2005) has assessed the causes and consequences of lack of access to water supply and sanitation. The author concludes that these problems are particularly important in peri-urban areas. Lack of access can be due to:

- The legal system – many policies and regulations do not take account of informal settlements, e.g. on tenure issues, and constitute an obstacle to improved access
- An unfair allocation of resources which do not take account of gender roles, age, ethnicity and disadvantage.

Access to finance can also be a significant barrier to the improved water and sanitation and particularly effects the poorer parts of society (Evans and Trémolet 2010). Evans (2007) outlines some key reasons why poorer parts of urban society are excluded from water and sanitation provision:

- The distance from trunk infrastructure – resulting high unit cost of service provision
- The high cost of self-provision due to scale limitations
- Legal barriers, e.g. land tenure
- Their location in areas which are more technically challenging – e.g. on steep slopes or have a tendency to flood
- They are priced out of formal services due to high connection fees (and corruption)
- They do not have the necessary influence to obtain an official connection through often bureaucratic processes

Increase in amount of water collected Decrease in level of health concern			
No access (<5 L/c/day)	Basic access (~20 L/c/day)	Access within yard level (~50 L/c/day)	Access in the household (~155 L/c/day)
More than 1 km or 30 minutes collection time	Less than 1 km or 5-30 minutes collection time	5 minutes collection time	Optimal access through multiple taps

Table 2.1 Relation between distance to services, collection time and amount of water used (after Hofmann, 2005)

Musingi et al. (1999) showed how the surface water supply for Mombasa has failed to meet the demand due to urban growth. Groundwater quality is compromised by saline intrusion, pit latrines, and septic tanks. In Old Naledi, Gabarone, Botswana, after rezoning of land for residential use, 93% of the population depended on communal standpipes with the remainder having private connections (Gwebu, 2003). Over ninety percent of the population used pit latrines which filled rapidly, frequently overflowed during the wet season, and continually seeped into groundwater affected nearby water resources.

In Dar-es-Salaam, many peri-urban inhabitants could be described as water poor as they lack access to sufficient water of good quality and adequate sanitation (Allen et al. 2006b, see Table 2.2). Customary systems facilitate access to unplanned and high-density low-income housing where groundwater sources are heavily contaminated. As a result, residents are forced to spend a significant part of their income buying water from kiosks, small scale water providers, or other sources. Often, informally supplied water failed to meet minimum standards of quality with impacts on consumer's health. Water is also required for income generating activities such as food vending, poultry and cattle keeping, concrete block making and textile production and lack of water poses a serious risk to livelihoods, and maintains people in poverty (Wright et al., 2013).

Matsinhe et al. (2008) describe a similar picture for Maputo, Mozambique. Here water delivery to large parts of the peri-urban area depends on informal service provision. Household waters sellers and small independent providers cater for over 20% of unconnected households, with the remainder from private wells and neighbour to neighbour water resale. Jiménez et al. (2012) assessed the role of sustainable urban water distribution and measured the efficiency of a development project in Wukro Town, Ethiopia as a case study. This used the distance to a service connection as a benchmark. The relationship between stakeholders is critical to effective contribution to sustainability.

2.2.6 Links between culture, gender and sanitation in urban SSA

Improved WASH in urban SSA has obvious and well documented sanitary benefits as well as more subtle and poorly documented non-sanitary impacts and benefits (McMichael, 2000). Non-sanitary impacts are briefly reviewed by Pearson and McPhedran (2008), these include enhanced security, education and the dignity of disabled people, this is particularly important for women and girls as well as poor communities (Mweemba et al., 2011). Pearson and McPhedran (2008) note that this is an area of research that has received little attention, and there have been few studies in SSA. New cultural norms take a long time to be adopted, and the WASH sector is by no means immune from this (Paterson et al., 2007). Many projects have found that the up-take and use of improved sanitation and drinking water facilities have been slowed or inhibited by a number of factors, the fact that changes in cultural norms work on the generational time scale while projects often work on the sub-decadal timescale remains a huge challenge. While the adoption of new norms is generally thought to be more rapid in urban settings, understanding these issues and how to enhance early adoption of new cultural norms where possible is critical to the future success of improved access to water and sanitation in SSA.

Table 2.2 Peri-urban water supply in Dar-es-Salaam (after Allen et al., 2006b)

Provider	**Policy driven practice**	**Needs-driven practice**
Public sector	Piped network and public standpipes Wells and boreholes Provision by tanker	Water kiosks
Private sector	Buying from licensed tankers Buying packaged water (cans, bottles and sachets)	Buying from informal tankers Private vendors drawing from own well or piped connection and sold through pushcarts and bicycle vendors
Community		Rainwater harvesting Water theft Clandestine connections Private wells and boreholes Piped network and taps run with NGO support Borehole and kiosks run by the community

3 Governance of water supply and sanitation in peri-urban areas

3.1 INTRODUCTION

Rapidly growing, unplanned peri-urban areas are not served effectively by centralised systems and are characterised by a lack of infrastructure. At the same time the peri-urban interface is associated with both rural and urban features and consists of highly heterogeneous and rapidly changing socio-economic groups (Allen et al., 2006a). This diversity means that the needs of local populations and producers of water and sanitation services are also diverse and change over time. The identification of these needs can be more complex than in either urban or rural areas due to the particular mix of newcomers and long-established dwellers, and also because farming, residential and industrial land uses often coexist. Current practice, as proposed for example by Montgomery and Elimelech (2007), is to recommend a decentralised approach that relies on household water treatment and sanitation technology. Closas et al. (2012) also argue for a more integrated and sustained approach to water and sanitation in African cities using the principles of Integrated Urban Water Management, which can encourage innovative solutions to water supply problems. Montgomery and Elimelech (2007) illustrate the high child mortality in Sub-Saharan Africa relative to other areas. Sources which meet their definition of 'improved' include a household connection, borehole, protected dug well, protected spring or rainwater collection. Improved sanitation includes connection to a sewer or septic tank, or VIP latrines.

Closas et al. (2012) categorise a number of large African cities on how well they perform against several urban water management indicators (Table 3.1): urbanisation challenge (growth rate and percentage informal area); solid waste management (percentage collected and disposed in controlled sites); water resources availability (including runoff and baseflow); water supply service (percentage coverage, water consumption, percentage billed); sanitation service (percentage with access to improved sanitation and of wastewater treated); and flood hazard (frequency). For the cities within the scope of this review, relative to the total, the issues are urbanisation and water supply services.

In many countries, peri-urban areas generally lie outside the coverage of formal networked water and sanitation systems, which are, in most cases, restricted to a relatively small metropolitan core. Part of the reason for this is that many peri-urban settlements develop outside existing formal regulations, affecting their formal right to these basic services. Within this environment, complex systems of water delivery flourish including self-supply from hand dug wells on the householder's plot; collection from local wells, springs and surface water; collection from stand-pipes; and purchase from mobile water vendors. However, if adequate land policies and official control procedures are in place, the goal of improving access to water and sanitation by the peri-urban poor should not necessarily require formal land or housing tenure, but might instead focus on collective land rights and responsibilities for paying for these basic services (O'Hara and Shanahan, 2006).

Isunju et al. (2011) discussed the reasons for the lack of progress on water supply and sanitation to meet the Millennium Goals. They set out the reasons as:

- Lack of prioritisation of sanitation
- Inadequacy of public funds
- Lack of appropriate technical solutions
- Shared responsibilities

Table 3.1 Urban water management indicators for selected cities in Sub-Saharan Africa (after Closas et al., 2012)

City	Urbanisation challenge	Solid waste management	Water resources	Water supply service	Sanitation	Flood hazard
Accra	1	NA	2	2	1	1
Brazzaville	1	NA	3	0	0	1
Dar-es-Salaam	3	1	1	2	2	1
Douala	1	3	3	0	1	0
Harare	3	1	1	1	1	0
Ibadan	1	1	1	0	1	1
Kampala	1	1	1	2	3	3
Kano	0	0	1	0	1	2
Kinshasa	2	NA	3	1	1	1
Kumasi	3	1	0	0	3	0
Lagos	2	1	1	1	3	1
Lusaka	2	1	1	1	1	0
Maputo	0	2	1	0	2	1
Yaoundé	3	1	3	1	1	0

Note: Urbanisation and flood hazard are pressures and scored oppositely to the other factors which reflect how well the city copes. 0=below average, 1=average, 2=above average, 3=well above average

3.2 WATER SUPPLY

Niemczynowicz (1999) summarises the problems of water management associated with the majority of urban areas. These are storm water management, drinking water supply and consumption, water for sanitation, wastewater for irrigation and recycling of wastewater nutrients, water for urban agriculture and water to recharge depleted aquifers. But there are other factors that contribute to the problem of delivering of water supplies, including a high rate of population growth, a lack of investment in water supply infrastructure, and the limit posed by the availability of water resources. Added to this within the delivery systems are high levels of unaccounted for water, wastage, low tariffs, and poor billing, which all contribute to poor service delivery (Mwendera et al, 2003). Some analysts suggest that a stable regime is required to tackle these problems, often not found in developing countries (Van der Bruggen et al. (2010).

Showers (2002) examine urban-rural water linkages in African cities. Many cities obtained water from groundwater during the 1970s and, except for North Africa and the Sahara, relied on sources that were nearby. By the 1990s this pattern had changed, the percentage dependency on groundwater had declined (from 73% to 54% from the example studies), many cities had developed a water “deficit” and water supplies were brought in from much further afield. Lusaka, for example, began obtaining supplies from the Kafue River, 45 km from the city.

Economic pressures and macro-level water resources management have contributed to the slowdown of overall growth in water usage to sustainable levels in many parts of Sub-Saharan Africa (Mwendera et al., 2003). However, in poorer areas communities are not managing to secure adequate supplies to allow them to live healthy lives. Mwendera et al. (2003) suggest that equity in resource allocation and the implementation of water demand management (WMD) can go a long way towards optimal water use. Under WMD water price should reflect water quality (Niemczynowicz, 1999).

In rural and peri-urban areas communities need to be at the forefront of their own water development activities, to be able to select appropriate technology and to be provided with operation and maintenance skills (Nkhuwa, 2009). A number of authors set out mechanisms for improvement of supply to poor urban dwellers. Mwandu Siyeni (2008) assessed if provision of water supply to the peri-urban areas of Lusaka could be achieved through the partnership between the water utility and small scale water providers. The two providers were found to have complementary strengths which when combined would enhance service provision.

3.3 SANITATION

Improved sanitation is a key component of the Millennium Development Goals, but its targets for access will not be met in most countries of SSA (Isunju et al., 2011). Sanitation brings both health related and non-health related outcomes. Isunju et al. (2011) argue that providing sanitation in slum areas is more complex than is generally recognised. Factors that are often ignored are land tenure, social structure, demand and the drivers for this, the segmented nature of the sector and on the provision processes, and the lack of clarity on the definition of improved sanitation; indeed, in many countries of SSA there is no legal definition of basic sanitation on which to develop any rights-based approach to provision (Payment and Locas, 2011). In slum areas sanitation coverage is often lower than the average for urban areas (Ashbolt et al., 2001; Gleeson and Gray, 1997).

Von Münch and Mayumbelo (2007) assessed sanitation improvement options in three peri-urban areas of Lusaka. Unlined pit latrines were the most common sanitation provision observed, despite the use to boreholes and shallow wells for water supply. They shortlisted VIP latrines with downstream processing and an ecological option with urine-diversion, dehydrating toilets also with down-stream processing as the best options for improving sanitation. Converting these options into sustainable sanitation provision is not straightforward as the demand for sanitation in these areas is often low. Stimulating demand for sanitation is a significant challenge for workers in the water and sanitation sector and has attracted the attention of researchers and NGOs. Demand for sanitation has been described as a social and behavioural process that goes through a number of stages and that an appreciation of this process can inform the use of social marketing tools (Leclerc et al., 2001).

4 Urban sources of groundwater pollution

4.1 INTRODUCTION

Groundwater contamination can occur whenever there is a source releasing contaminants to the environment (Sililo et al., 2001) (Table 4.1). The major sources of groundwater pollution include:

- Municipal (sewer leakage, sewage effluent, sewage sludge, urban runoff, landfill, latrines, septic tanks);
- Agricultural (leached salts, fertilisers, pesticides, animal wastes);
- Industrial (process waters, water treatment, plant effluent, hydrocarbons, tank and pipeline leakage); and
- Mining (solid wastes and liquid wastes) activities.

Table 4.1 Main sources of urban groundwater pollution in South Africa with some of their characteristics (adapted from Sililo et al., 2001)

Category	Source	Main pollutant	Potential impact
Municipal	Sewer leakage, septic tanks and latrines	Nitrate, minerals, organic compounds, viruses and bacteria	Health risk to users, eutrophication, odour and taste
	Sewage effluent and sludge		
	Storm water runoff	Bacteria and viruses, oils, grease, asbestos, heavy metals	Health risk to water users
	Solid waste disposal	Inorganic minerals, organic compounds, heavy metals, bacteria and viruses	Health risk to users, eutrophication, odour and taste
	Cemeteries	Nitrate, viruses and bacteria	Health risk to water users
Peri-urban agriculture	Livestock wastes	Nitrate, ammonium, viruses and bacteria	Health risk to water users
	Pesticides	Toxic/carcinogenic compounds	Health risk to water users
	Fertilisers	Nitrogen, phosphorus	Eutrophication, health risk to water users
Industrial	Process plant and effluent	Organic compounds, heavy metals	Toxic/carcinogenic compounds
	Industrial solid waste	Inorganic minerals, organic compounds, heavy metals, bacteria and viruses	Health risk to users, eutrophication, odour and taste
	Leaking storage tanks	Hydrocarbons, heavy metals	Odour and taste
	Chemical transport	Hydrocarbons, chemicals	Toxic/carcinogenic compounds
	Pipeline leaks		
Atmospheric deposition	Vehicle emissions	Acidic precipitation	Acidification of groundwater and toxic leached heavy metals
	Coal-fired power stations		
Mining	Tailings and stockpiles	Acid drainage	

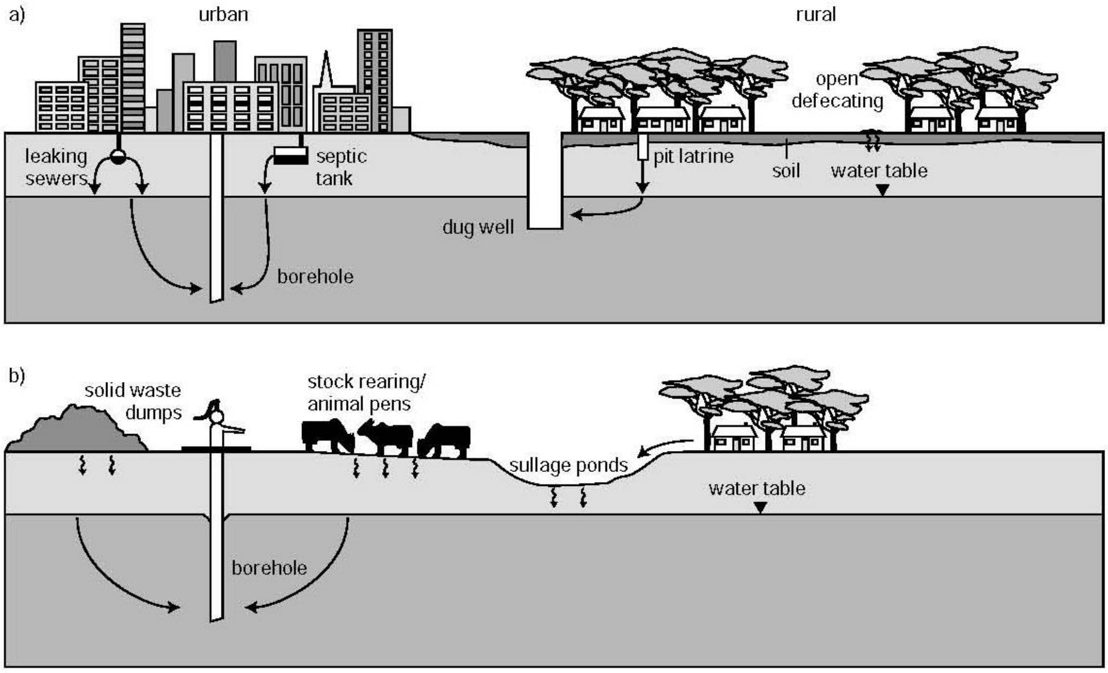

Figure 4.1 Sources of faecal pollution within urban a) and rural b) settings (from Lawrence et al., 2001)

Lawrence et al. (2001) describe the various sources of faecal pollution in different settings and the pathways of these pollutants into groundwater (Figure 4.1). This simplified representation is valid for many groundwater contaminants and in peri-urban areas all of these sources of pollution may occur together. However, the figure does not show the processes of attenuation in the system that will reduce the concentration of contaminants reaching the water table. For microorganisms in faecal and other waste materials, the main barrier to their movement into groundwater is the unsaturated zone. Once into groundwater, however, a complex interaction of other physical, chemical and biological factors control the survival and mobility of the microorganisms (Mesquita et al., 2013) and influence the distance that they can travel from the source.

The causes of water quality contamination may be separated into those related to the production of contaminants at the source and those which govern their delivery into the water environment (Pegram et al., 1999). In peri-urban environments they are primarily caused by infrastructure and services which are not adequate for the settlement characteristics, are poorly managed by the provider and /or poorly managed by the community.

4.2 SANITATION

4.2.1 Sanitation in peri-urban areas

Sanitation facilities in peri-urban areas largely take the form of on-site technologies such as pit latrines and pour-flush toilets. Reticulated systems are also an important source of contamination but these have very limited coverage overall in urban areas in SSA. On-site sanitation technologies, are often the cheapest and most appropriate forms of sanitation in rural and in urban areas. However in peri-urban areas the ground conditions give rise to poor drainage and risk of contaminated drinking water sources (Paterson et al., 2007). High population densities result in a high liquid load. There are a number of current options:

- A simple pit latrine consists of a seat or squatting hole over a pit, in which the human waste collects.

- A VIP latrine has the addition of a screened vent pipe which extends from the pit to eliminate odours and flies.
- A pour-flush toilet is similar to a cistern-flush toilet but with a shallower U-bend so that the toilet can be flushed by manually pouring a small amount (2–3 L) of water into the toilet pan. For on-site disposal, the toilet is connected to a pit.
- Urine-diverting toilets. Sometimes referred to as EcoSan toilets, urine diverting systems separate the urine and the faeces in different compartments. The urine can be diluted and used as a liquid feed for plants; the faeces is digested for several months after which it can safely be used as a manure. Despite widespread marketing of EcoSan systems in Africa, the technology has a limited application.

For all these toilet types, apart from the Ecosan toilet, the pit is normally designed so that liquids disperse into the surrounding soil, while the solids accumulate and decompose over time, and can be safely removed after a few years for disposal or re-use on agricultural land. In practice, emptying pits is a considerable challenge in many peri-urban areas due to the difficult access for vacuuming trucks and other emptying machines, and the very mixed nature of the pit contents (often including rags and metal objects). As a result, the pit contents tend to accumulate and provide a constant source of contamination to groundwater.

Paterson et al. (2007) suggest that a simplified (or condominial) sewerage system where a shallow possibly small diameter network of pipes takes the sewage elsewhere to be treated/disposed may be appropriate. This approach, using communities to install network, has been pioneered with some success in Pakistan by the Orangi Pilot Project (Gavini et al., 1985). For Bwaize III region of Kampala, Uganda, Katukiza et al. (2010) present a technology selection method that takes into account social acceptance, technological applicability, economic and institutional aspects and health and environment benefit. These were a septic tank system, biogas toilet, compost pit and urine diversion dry toilet.

Parkinson and Tayler (2003) advocate a decentralised approach to wastewater treatment in peri-urban areas in low-income countries. Options include anaerobic treatment, waste stabilisation ponds and constructed wetlands.

4.3 PERI-URBAN AGRICULTURE

The pattern of peri-urban agriculture has common features across the continent, which is illustrated by examples of peri-urban cropping in Dar-es-Salaam, in the East, and Accra, in West Africa, shown Tables 4.2 and 4.3.

The intensification and expansion of agriculture brings many benefits and Jacobi et al. (2000) state that due to urban and peri-urban agriculture Dar-es-Salaam is not short of food. Similarly in Zimbabwe, the yields from urban areas is greater than the yields from rural areas, where fertiliser use is limited (Brazier, 2012). However, there are also associated risks to water resources. These can come from fertilisers, either synthetic or manure, and pesticides. Furthermore, if irrigation is not managed adequately there can also be risks of salinisation.

Animal wastes provide an important resource for agriculture, yet Harris et al. (2001) report that around Kumasi, Ghana, although 80% of poultry manure was used in farming, the remainder can be dumped and burnt by the roadside and in Uganda poultry manure is mixed with brewery waste and used as cattle feed. In Niamey, Niger, peri-urban agriculture is characterised by fruit and vegetable production including cabbage, lettuce, tomato, carrot, onion, zucchini, sweet pepper, hot pepper, eggplant, French beans, melon, cucumber, cassava root, maize and strawberry (Andres and Lebailly, 2011). During the hot season (March-May), market gardeners grow hot pepper, zucchini and cucumber. During the rainy season (June-September), they cultivate gumbo, melon and beans. During the dry season (October-February), they grow lettuce, cabbage, tomato, sweet pepper, beetroot, celery, carrot, and parsley. Fertiliser use can be high and comprises manure from the slaughter house or from local breeders and is supplemented with synthetic nitrogen, phosphorus and potassium (NPK) (Graefe et al., 2008).

A survey of fertiliser use for urban and peri-urban agriculture in Namibia, (Dima et al., 2002), where water is a limiting factor, showed that the most common type of fertiliser was digested human solid waste. Other sources were compost, household waste, inorganic fertilisers and fresh cow dung. Most fertiliser was applied once per year.. The main pests were corn cricket, American bollworm, spiders, aphids and fungal attacks. Use of pesticides was very limited and the commonest method of control was hand picking. In Mekelle, Tigray, Ethiopia, urban and peri-urban agriculture is supported by an Urban Agricultural Office which provides hand dug wells for irrigation, technical support on seeds planting and fertilisers, and planting material and seedlings (Ashebir et al., 2007). Several crops can be grown each year and both staples and vegetables are grown. The majority of farmers used both chemical and organic fertilisers.

In Bamako, Mali and Ouagadougou, Burkina Faso, the disposal of household waste also poses a challenge (Eaton and Hilhorst, 2003). In both cities, waste is produced at an average of 0.6-0.7 kg/person/day. The composition of this varies considerably seasonally, with an increase in the amount of sand and dust during the dry season. Together with plastic, paper, metals and textiles, these inorganic components need to be separated out to allow safe recycling of organic matter. In both cities an informal private sector has established itself to collect waste. Organic solid waste is used for peri-urban agriculture by agreement, sometimes illicit, with waste disposal operatives. Farmers remove large inorganic objects and spread the waste on fields before the onset of the rainy season. The waste has an organic content of about 11%, nitrogen of about 0.3% and phosphorus 0.16%. Applications appear to be mainly on cereal crops.

Table 4.2 Peri-urban cropping in Dar-es-Salaam (after Jacobi et al., 2000)

Zone	**Crop type**	**Example**	Comment
High density area gardens (15-20% of houses typically 270 m^2)	Green leafy vegetables	Sweet potato, cow pea, cassava and pumpkin	Water limiting factor
Low density area gardens (4000 m^2)			Bigger plots often with tap
Community gardens	Diverse		
Urban area (65% of houses)	Livestock production	Cattle fed on public land and cut grass, poultry both extensive and intensive	Growth of dairy cattle
Open space, unoccupied plots and river valleys	Market orientated leafy vegetables	Chinese cabbage	Important in dry season when gardens water limited
Peri-urban	Livestock	Dairy cattle	
Peri-urban typically 2 ha with 0.6 ha under vegetables and fruit	Green and other vegetables, fruit, staples	Maize, rice, cooking bananas, cassava, sweet and hot pepper, eggplant, okra,	Being gradually swallowed up by urban area

Table 4.3 Peri-urban cropping in Accra, Ghana (after Asomani-Boateng, 2002)

Zone	Crop type	Example	Fertiliser	Comment
Household	Staples and fruit	Maize, plantain, cassava and cocoyam, pineapples, mangoes, paw paw, orange, coconut palm oil	Cow manure and chicken droppings	
Vacant space including banks of streams and drains	Vegetables	Cauliflower, lettuce, cabbage, carrots, sweet peppers, French beans, peppers, beetroots, herbs, okra, peppers, tomatoes, eggplant and green leafy vegetables (ademe, ayoyo, gboma, busanga)	Cow manure and chicken droppings Synthetic fertilisers	Streams and drains prone to flood
Peri-urban	Fruit	Pineapples, mangoes, paw paw, orange, coconut and palm oil	Cow manure and chicken droppings Synthetic fertilisers	

4.4 WASTEWATER IRRIGATION AND SOIL AMENDMENT USING HOUSEHOLD AND HUMAN WASTE

The impact of wastewater irrigation has been well documented in several newly industrialised countries. For example in Mexico, there was a large study for DFID on irrigation using the effluent from Mexico City (CNA et al., 1998). This scheme had been operating for many years with progressive extension of the irrigated area and a large rise in groundwater levels. Almost three quarters of the public supply groundwater sources in the main area exceeded the limit for nitrate-N (11.3 mg/L), about half for alkalinity and about one third for chloride. Most of the heavy metals were retained in the soil. Surveys of microbiological quality found between one third and one half of supplies were positive for faecal coliforms, with some >50 cfu/100mL. Levels fluctuated widely over time. Enterovirus was detected at three of the four sites where it was analysed and hepatitis and rotavirus were detected in springs in the area. In León, Mexico, irrigation with effluent containing a mixture of domestic and tannery effluent had also increased groundwater levels (Chilton et al., 1998). Shallow groundwater was impacted mainly by salinity with concentrations of chloride of over 500 mg/L. Chromium from tannery effluent appeared to be almost all precipitated in sediments and lagoons of the wastewater distribution system and groundwater concentrations were not elevated above the background (Stuart and Milne, 2001).

In Zambia, research on wastewater irrigation in the Mufulira and Kafue areas has focused on heavy metals in soils and crops (Kapungwe, 2013). Crops were irrigated using both sewage and, in some areas, industrial wastewater containing copper mining effluent (Marshall et al., 2004). Crops were shown to be contaminated with cadmium, copper, lead and zinc. This was similar to results from Zimbabwe (Muchuweti et al., 2006). Mayeko (2008) showed that the market gardeners of Kinshasa, Congo were exposed to chemical and microbiological contamination from the use of wastewater for irrigation. Water contained both heavy metals and microbiological contamination. Kulabako et al., (2009) and Stevens et al., (2003) investigated the use of novel, multi-stage, home-made trickle-filters, to treat kitchen and used bathing waters for use in household irrigation. The final effluent from these treatment systems was found to be suitable for small-scale urban agriculture.

Keraita et al. (2003) reported the impact from the use of urban wastewater irrigation in and around Kumasi, Ghana. Due to inadequate waste treatment capacity surface and shallow groundwaters are being impacted by large volumes of untreated and partly treated waste water from the city. High nutrient concentrations (N and P) and microbiological counts were found in stream water receiving runoff from contaminated irrigation water as well as crops in Kumasi market which are a serious risk for consumers. In per-urban Kano, northern Nigeria, contamination of waters and soils on land used for food production is such that Binns et al. (2003) question the long term sustainability of urban agriculture due to environmental and toxicological concerns. Economic pressures have also made it difficult to gain access to farmland and many farmers have diversified in order to survive (Maconachie and Binns, 2006).

In Addis Ababa, Ethiopia, untreated wastewater is discharged directly to surface water which is used for irrigation by some farm households. Weldesilassie et al. (2009) found that 88% of households reported that they benefited from wastewater as for some this was the only means for survival. The safe use of this resource was therefore considered valuable. However, Weldesilassie et al. (2011) found that the cost of treating the health impact from worm infections alone made the use of poor quality water more expensive.

Cofie et al. (2005) studied the use of human waste for urban agriculture in Tamale, Northern Ghana. Farmers used faecal sludge for cultivation of cereals, such as maize, sorghum and millet, sometimes combined with farmyard residues or chemical fertilisers. Sludge was discharged during the dry season to the field surface or to large pits and left to dry before spreading. Estimated nutrients applied were 455 kg/ha N, 61 kg/ha P and 121 kg/ha K as well as 1183 kg/ha additional organic carbon. Problems include odour, unacceptability of crops to the public and health problems in workers.

In Africa urban refuse comprises 50-90% organic material, and includes kitchen waste, food leftovers, rotten fruit and vegetables, leaves, crop residues and animal excreta and bones (Asomani-Boateng and Haight, 1999). Most African countries have traditionally used such organic material as a soil improver. Promoting reuse on a large scale as a response to disposal problems requires decentralising of planning, waste separation, composting facilities and landuse planning. There are recognised health risks to both farmers and produce consumers, as well as threats to the environment in reuse due to the high faecal content of waste.

4.5 INDUSTRY

In newly industrialised countries the rate of industrialisation has been fast and the environmental problems more acute as natural attenuation processes have not yet had time to make an impact on environmental recovery (Morris et al., 2003). Additional problems can occur in developing countries where regulations controlling waste disposal are not adequately enforced, control measures are not sufficient, and the resources to monitor discharges are non-existent or inadequate (Tallon et al., 2005).

Chukwu (2008) evaluated the impact on groundwater of abattoir waste in Minna, Nigeria, in two wells. Groundwater had elevated TDS and SS, pH, and low dissolved oxygen. Sangodoyin and Agbawhe (1992) studied the pollution from abattoirs in the Ibadan area, Nigeria in both ground and surface water. Effluents consisted of a slurry of suspended solids, fat, blood, scraps of tissue and soluble material generally discharged to local streams without treatment. They also identified elevated phosphates from detergents in washdown water. In general groundwater appeared to be affected by effluent with increases in TDS.

Table 4.6 show the characteristics and composition of waste effluents from large and small industries in Nigeria which discharge mainly to surface water (e.g. Adebayo et al., 2007; Kanu and Achi 2011; Taiwo et al., 2010).

Olayinka and Alo (2004) studied the impact of textile effluents on groundwater in parts of Lagos, Nigeria. These effluents had high BOD (100-390 mg/L) and COD (204-2000 mg/L) and pH (10-12 for one of the plants) and were highly coloured. Some wells within 25 m of the discharge were affected by elevated TDS and oxygen depletion due to high BOD TDS whilst those further away were not measurably affected. As a result of the contamination, groundwater wells close to the discharge had been abandoned by local residents.

Table 4.4 Types of waste effluents generated by selected industries in Nigeria (after Kanu and Achi, 2011)

Type of waste	Type of industry
Oxygen- consuming	Brewery, dairy, distillery, packaging, pulp and paper mill, tannery, textiles
High suspended solids	Brewery, coal washing, iron and steel, distillery, pulp and paper mill, palm oil mill
High dissolved solids	Chemical plant, tannery, water softening
Oil and grease	Laundry, metal finishing, oil field, petroleum refinery, tannery, palm oil mill
Coloured	Pulp and paper mill, tannery, textile dying, palm oil mill
Acid	Chemical plant, coal mine, iron and steel, sulphite pulp
Alkaline	Chemical plant, laundry, tannery, textile finishing mill
Hot effluent	Bottle washing, laundry, power plant

Table 4.5 Industrial pollutants (after Kanu and Achi, 2011)

Industry	Compounds found in receiving waters
Pharmaceutical and personal care	Antibiotics, lipid regulators, anti-inflammatories, antiepileptics, tranquilizers, and cosmetic ingredients containing oil and grease
Soap and detergent	Alkyl sulphates, high BOD & COD, oil and grease
Paper	Sugars and lignocelluloses,
Fertiliser	Ammonium-nitrogen, urea, nitrate-nitrogen, orthophosphate-phosphorus
Textile –sizing and desizing	Starch, waxes, carboxymethyl cellulose(CMC), polyvinyl alcohol (PVA), wetting agents, , fats, waxes, pectins
Textile – bleaching and mercerising	Sodium hypochlorite, Cl_2, NaOH, H_2O_2, acids, surfactants, $NaSiO_2$ sodium phosphate, cotton wax
Textile – dying and printing	Dyestuffs urea, reducing agents, oxidizing agents, acetic acid, detergents, wetting agents, pastes, urea, starches, gums, oils, binders, acids, thickeners, cross-linkers, reducing agents, alkali
Brewing	Carbohydrates and nitrogen
Tanning	Cr, oxidising agents, Cl, fats
Palm oil milling	Carbohydrates and nitrogenous compounds giving high BOD & COD, oil, fatty acids, low pH

Chukwu (2008) evaluated the impact on groundwater of abattoir waste in Minna, Nigeria, in two wells. Groundwater had elevated TDS and SS, pH, and low dissolved oxygen. Sangodoyin and Agbawhe (1992) studied the pollution from abattoirs in the Ibadan area, Nigeria in both ground and surface water. Effluents consisted of a slurry of suspended solids, fat, blood, scraps of tissue and soluble material generally discharged to local streams without treatment. They also identified elevated phosphates from detergents in washdown water. In general groundwater appeared to be affected by effluent with increases in TDS.

Table 4.6 Industrial effluent quality in Nigeria (after Adebayo et al., 2007 and Taiwo et al., 2010)

Parameter	Lagos (textile)	Lagos (brewery)	Kaduna (mixed)	Port Harcourt (mixed)
Temperature (C)	27.6	30.3	30	
pH	7.6	4.8		
Conductivity (mS/cm)	761	1157		
Alkalinity (mg/L)	767	445		
Nitrate (mg/L)			4.0	362
Ammonium (mg/L)			1.0	
Phosphate (mg/L)			1.0	836
Total hardness (mg/L)	1233	4083		
Oil and grease (mg/L)	20	0	7	2343
BOD (mg/L)	534	1352	300	4374
COD (mg/L)	850	2253	1800	
H_2S (mg/L)	17.2	130	0.6	

4.6 MANAGED AQUIFER RECHARGE (MAR) AND FUEL STORAGE

The recharge of groundwater using treated wastewater was studied in an area of Addis Ababa (Abiye et al., 2009). Treated wastewater from the Kaliti plant was discharged to the Little Akaki River overlying the Akaki wellfield. The work showed that the soil was able to remove most contaminants from infiltrating wastewater and that MAR would be a feasible method to improve groundwater resources. This is likely to be particularly relevant in more arid urban zones in SSA.

Sources of pollution at a typical petrol station in Nigeria include leakage from underground storage tanks, spills during loading and other operations and dumping of waste, commonly in shallow pits (Nganje et al., 2007). In a survey of petrol stations and mechanics workshops around Calabar, Nigeria found concentrations of total hydrocarbons and total polyaromatic hydrocarbons to be higher than the WHO drinking water guideline values in groundwater. These were found by factor analysis to be associated with poor yard practice and waste management.

4.7 SOLID WASTE DISPOSAL

4.7.1 Solid waste characteristics

Urban solid waste creates large environmental problems in Africa; the generation of waste has increased considerably in the last 3 decades (Yhdego, 1988). To an extent, this is a consequence of the growth in urban populations, but the problem is compounded by the lack of resources and infrastructure to cope with growing amount of waste leading to uncontrolled disposal into water courses and other convenient areas (Carrillo et al., 1985; Yhdego, 1995). In Dar-es-Salaam it was reported that over 80% of the wastes produced were left in open pits, streets, markets or storm water drainage channels (Yhdego, 1995). In Tanga, Tanzania, all types of solid waste were disposed in an old partially flooded sand quarry close to a residential area (Mato, 1999). Wastes may be treated partially by uncontrolled burning**.** Scavenging is widespread and uncontrolled. There is evident pollution of both ground and surface water.

Table 4.7 Waste generation rate and origins (from Guerrero et al., 2013)

Country	GDP (US$)	Year of study	City	Waste arriving at disposal site	Waste generation rate (kg/capita/day)
Ethiopia	344	2009	Addis Abba	H,O,M,I,S	0.32
Kenya	738	2009	Nakuru	H,O,C,M,A,I, S	0.50
Malawi	326	2009	Lilongwe	H	0.50
South Africa	5786	2009	Pretoria	H,O,C,M,S	0.65
South Africa	5786	2009	Langeberg	H,O,C,M,A,I,S	0.65
South Africa	5786	2009	Emfuleni	H C,I	0.60
Tanzania	509	2010	Dar-es-Salaam	H,O,M,A,I, S	0.50
Zambia	985	2010	Lusaka	H,O,C,A,I,S	0.37

H=household, O=offices & schools, C=construction, M=healthcare, A=agriculture, I=industry, S=shops

In many African countries the collection of waste is haphazard and inefficient (Yhdego, 1988). In addition, hazardous and non-hazardous wastes are often disposed of without separation (Carrillo et al., 1985) creating a risk to the health of the local population, the workers who collect the waste and those who make a living by scavaging from waste disposal sites. The main sources of solid waste in the urban areas of Tanzania are domestic, commercial activities, industries, streets and markets. The composition ofwastes is primarily vegetables and other putrescible matter with very high moisture content. (Yhdego, 1995). Guerrero et al. (2013) found a similar pattern for a number of cities in SSA, see Table 4.7.

Leachate quality varies through the lifetime of a landfill and after its closure (Klinck and Stuart, 1999). During the early stages leachate is acidic and high in volatile fatty acids and pathogens. It may also contain mobilised heavy metals, ammonium and organic carbon. As waste degradation progresses conditions become anaerobic and the methanogenic phase is initiated. The majority of the remaining organic compounds are high molecular weight and leachate is characterised by low BOD. Ammonium remains high but falling redox potential immobilises many metals as sulphides. The health hazards of poor waste disposal are long-established. Klinck and Stuart (1999) list human faecal matter; industrial waste; decomposition products from the waste, inorganic macro-components, heavy metals, dissolved organic matter expressed as COD or TOC including methane and volatile fatty acids and anthropogenic organic compounds; smoke from waste burning including polyaromatic hydrocarbons and dioxins. Stuart and Klinck (1998) provide indicative leachate quality for a range of landfills from newly developing countries, including where waste is periodically burnt.

4.7.2 Disposal practices

In many countries of SSA, the disposal of waste is poorly regulated and enforced. As a result, substantial quantities of waste are disposed of illegally and without any consideration of the human and environmental health consequences (Carrillo et al., 1985). In areas of Nairobi, Kenya, dump sites were selected for convenience, not appropriateness based on environmental risks (Henry et al., 2006). Figure 4.2 shows the changes in municipal solid waste generation and collection capacity for Nairobi, Kenya, between 1972-2004. In Addis Ababa, Ethiopia, 56% of households surveyed deposited their waste in to plastic bags, 19% just dumped it in open spaces, waterways and around their home and 6% burnt it causing significant air pollution (Mazhindu et al., 2012).

Table 4.8 Components of industrial solid waste in Dar-es-Salaam in order of rate of generation (after Mbuligwe and Kaseva, 2006)

Solid waste component	Rate (t/a)	Major source
Glass cullets	13903	Breweries, distilleries, soft drink manufacturers
Bottle caps	9389	Soft/alcoholic drink manufacturers
Grain bran	4717	Grain millers
Spent grain and yeast	3638	Breweries
Packaging materials	2194	Soft drinks, breweries, distilleries, vegetable oil refining etc
Spent bleaching earth	1889	Food, beverages, vegetable refining
Plastics and rubber	1884	Plastic and rubber industries
Dust/soil	603	Grain millers, paint manufacturing
Peelings and crushed spent seeds	481	Fruit and vegetable canning
Waste/discarded paper	295	Pulp and paper industries
Scrap metal	252	Fabricated and basic metal industries
Sludge	140	Food, beverages, vegetable refining, paint manufacturing
Sand and discarded tiles	43	Tile manufacturers
Rejected plastic bottles	36	Bottled drinking water manufacturers, pharmaceutical industry
Foam	7.1	Foam mattress manufacturers

However, there are some notable exceptions. Industrial solid waste in Dar-es-Salaam is stored in open air piles, bins, masonry enclosures and silos (Mbuligwe and Kaseva, 2006). Waste is partially segregated since it tends to be stored near the industry type that produced it. Some edible oils, paper and pulp, glass, plastics and batteries are formally segregated as there is some recycling. About 60% is collected and transported by the municipal authorities. The main components are shown in Table 4.8.

The disposal of medical waste presents challenges such as the spread of contagious diseases and impact to water resources (Nkhuwa et al., 2008). An inventory of the medical waste from health centres in Lusaka, Zambia found:

- Infectious waste, such as bandages, swabs and disposable equipment
- Pathological waste, such as tissue, organs, blood and body fluids
- Sharp equipment including needles
- Pharmaceutical waste
- Radioactive waste
- Other, such as kitchen, bed linen, paper

The majority of this waste was disposed in refuse pits (paper, plastic, kitchen waste), and placenta pits (treated with sulphuric acid to create more space). Some waste (drugs, sharps, swabs etc) was sent for incineration, but the cost of incineration and inadequate temperatures achieved by the incinerators meant that potentially incineratable wastes were found in open pits.

Nkhuwa et al. (2008) showed that for the health centres studied in Lusaka local groundwater quality was compromised by coliforms, TDS and high COD, but this could not necessarily be separated from other local sources in the area. In a study in Owerri, Nigeria, Arukwe et al. (2012) found solid waste disposal to be a source of a range of emerging groundwater contaminants including phthalate plasticisers, polycyclic musks, bisphenol A and UV filters.

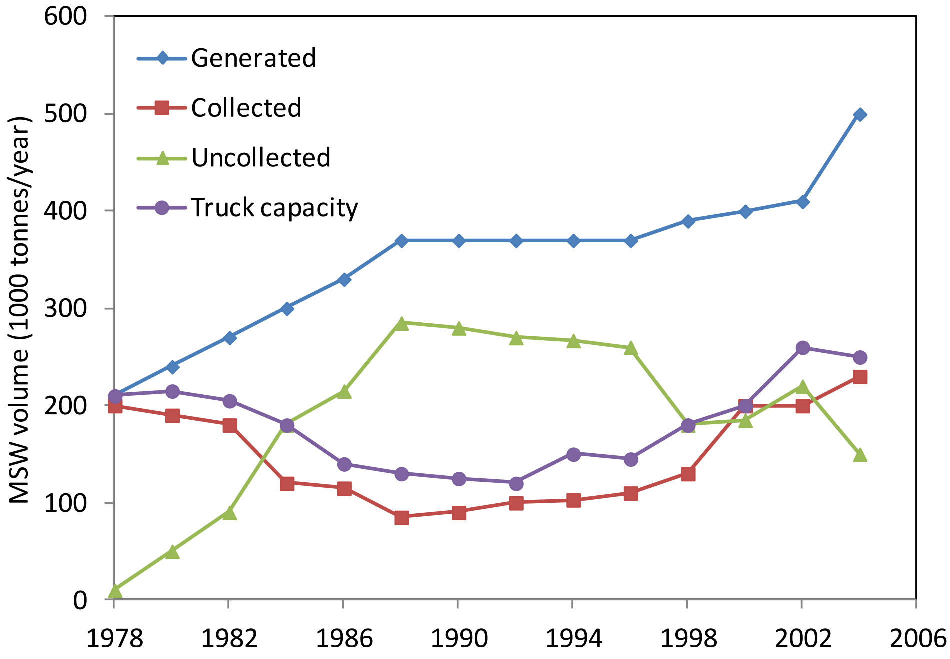

Figure 4.2 Municipal solid waste (MSW) generation and collection in Nairobi, Kenya (after Henry et al., 2006)

4.8 CEMETERIES

Although cemeteries have the potential to pollute groundwater, there have been very few published studies that have assessed this potential. The number and variety of pollutants is extensive, ranging from the mineral, organic and biological decomposition products from the bodies, to the chemicals used for embalming the bodies and treating and preserving the coffins. Engelbrecht (1998) studied a municipal cemetery site in the Western Cape, South Africa. This was situated on unconsolidated sands in the same geological succession as the local aquifer. Groundwater was sampled using specially drilled well points and analysed for a range of microbiological parameters, inorganic species and organic carbon. Groundwater was extremely polluted compared to water in the surrounding area with elevated concentrations of potassium, ammonium, TON, organic carbon, phosphate and a higher pH. Elevated concentrations of faecal coliforms, *Escherichia coli*, faecal streptococci and *Staphylococcus aureus* (an organism that colonises the skin and nasal passages of humans) were also measured. The impact of pathogens from cemeteries on the underlying groundwater has also been reported by Trick and co-workers (Trick et al., 2005).

Their study of a working cemetery in the UK detected pathogens in the groundwater even with an unsaturated zone thickness of between two and three metres. However, the vulnerability the groundwater to pathogen contamination from the unsaturated zone will be dependent on the soil composition, and so it is difficult to extrapolate from this report to other environments. A more recent study of soil samples taken from a cemetery in Guateng, South Africa, identified elevated concentrations of minerals and metals that could pose a risk to human health if they were transferred into the groundwater (Jonker and Olivier, 2012). These studies show the importance of cemeteries as a source of contamination to groundwater, and highlight the risks to drinking water sources in peri-urban areas of SSA where on-plot burial is occasionally practiced (e.g. Zume 2011).

5 Impact of mining

5.1 INTRODUCTION

This is a specific problem to particular parts of SSA which have a legacy of mining and problems of poor waste management, limited urban planning and rapid urban and peri-urban expansion. This can result in many informal settlements on or close to mine waste tips with potential for soil and water quality degradation and significant associated health risks for local residents.

5.1.1 Copper Belt

The sulphide Zn–Pb–Cu ore deposits of the Central African Copperbelt in the Democratic Republic of Congo and Zambia are mostly found in deformed shallow marine platform carbonates and associated sedimentary rocks of the Neoproterozoic Katanga Supergroup (Kampunzu et al., 2009). Economic ore bodies, that also contain variable amounts of minor Cd, Co, Ge, Ag, Re, As, Mo, Ga, and V, occur mainly as irregular pipe-like bodies associated with collapse breccias and faults as well as lenticular bodies subparallel to bedding.

Kipushi, in the Democratic Republic of the Congo, and Kabwe in Zambia are the major examples of carbonate-hosted Zn–Pb–Cu mined deposits with important by-products of Ge, Cd, Ag and V in the Lufilian Arc, a major metallogenic province famous for its world-class sediment-hosted stratiform Cu–Co deposits (Kampunzu et al., 2009).

5.1.2 Gold Belt

In the Migori Gold Belt, Kenya, gold occurs in quartz veins within mafic volcanics (Ogola et al., 2002). Mining was done on a large-scale up to the time of independence and thereafter reverted to artisan miners. This tends to be unregulated and managed with lack of awareness of metal poisoning issues. Panning is carried out along the river profiles and there is visible evidence of pollution from colouration and siltation. There are elevated concentrations of lead, mercury and arsenic in rivers and stream sediments.

5.1.3 Impacts

Mining and opencast workings can impact the environment and human health via a variety of chemical and physical routes (Morris et al., 2003) summarised in Figure 5.1. Human exposure can occur by accidental ingestion of mine wastes or waste contaminated soils by hand-to-mouth transmission, inhalation of dusts blown from tailings or waste piles, inhalation of gases or atmospheric particulates generated by smelting or roasting, and consumption of vegetables that take up metals from waste-contaminated soils or that accumulate mine waste dusts on their leafy parts (Plumlee and Morman, 2011). Nwankwo and Elinder (1979) measured cadmium, lead and zinc concentrations in soils and leafy vegetables close to a lead-zinc smelter at Broken Hill, near Kabwe, Zambia. The highest Cd concentrations were found in spinach, cabbage and rape.

5.2 GROUNDWATER CONTAMINATION

Sources of groundwater contaminants can be mine drainage, acid mine drainage (AMD), tailings lagoons and waste rock dumps (Morris et al., 2003). The routes to groundwater are shown in Figure 5.1. Mine waste piles can contain clay-sized particles to boulder sized blocks of unmineralised or partly mineralised rocks (Plumlee and Morman, 2011). Tailings contain ore, host rocks, and alteration minerals that were ground to $<500\mu m$ particles, and are often enriched in non-economic iron sulphides, crystalline silica, and alumina-silicates. AMD forms when oxygenated rain or groundwater oxidizes iron sulphides exposed in mine workings, mine wastes, or tailings impoundments. AMD can contain high levels of potentially toxic metals (e.g. lead, cadmium, nickel) and metalloids (e.g. arsenic, antimony) leached from the ores.

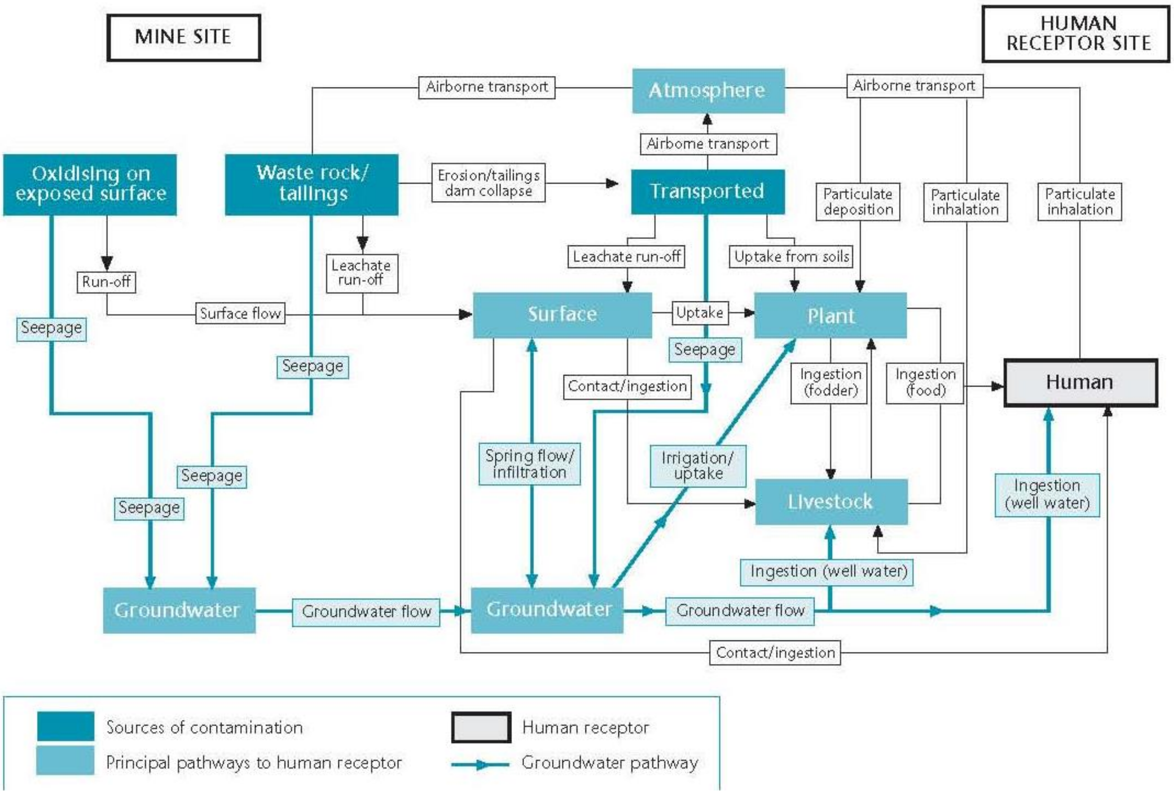

Figure 5.1 Main pathways of mine contamination to human receptors including the groundwater pathway (after Klinck and Stuart, 1999)

Post mining, secondary minerals form by the weathering of mine wastes, and these can include soluble salts formed by the evaporation of AMD. These salts commonly contain high levels of iron, other metals, and acid, and easily dissolve when they come into contact with water (Plumlee, 1999). Calcines and particulates produced by roasting or smelting of sulphide ores include water-soluble chlorides, oxides and sulphates of the ore metals; weathering can transport the metals into iron or manganese oxides. Smelter slag contains metal-enriched glass, residual sulphides, and metal chlorides, sulphates, and oxides.

Contamination of drinking-water supplies by mine wastes, AMD, and mineral processing solutions can introduce low pH, sulphate, calcium and magnesium derived from AMD, and its interaction with the aquifer matrix can introduce iron, manganese and a range of other heavy metals. Water can also be contaminated with asbestos, and other fibrous silicate minerals and by processing chemicals such as cyanide (Plumlee and Morman, 2011).

Mining of nickel-copper-cobalt ore at Selebi-Phikwe, Botswana, started in 1974. The area is underlain by high-grade metamorphic rocks with a 30-m weathered zone. Seepage from a tailings dam was identified as a major source of groundwater pollution (Schwartz and Kgomanyane, 2008). Seepage water had a pH in the range 1.7 to 2.8 and contained high concentrations of sulphate (5680 mg/L), nickel (6230 µg/L), copper (1860 µg/L) and cobalt (410) µg/L. Groundwater has a relatively low pH (generally <6) and contains relatively high concentrations of calcium and magnesium. Most of the heavy metal content is therefore scavenged within 500 m by iron and manganese oxides, but the sulphate is not attenuated.

Investigation of the impact of acid mine drainage in Selebi-Phikwe, showed that shallow soils had reacted with the leachate at the surface (Shemang et al., 2003). This had led to relatively conductive shallow layers containing sulphate, from pyrite oxidation, and Cu and Zn enhanced above background levels.

5.3 ZAMBIAN COPPERBELT

The Zambian Copperbelt has a high proportion of tailings impoundments, residue heaps, and extensive ore deposits close to high-density informal settlements (von der Heyden and New, 2004). Seepages from active and decommissioned tailings deposits pose a substantial threat to groundwater that may be used as a source of domestic water by the local residents.

At a site in the Chambishi catchment von der Heyden and New (2004) identified a high solute plume, possibly derived from a tailings impoundment either by dry deposition of tailings dust or, more likely, by leaching from the tailings material. The plume was 500-700 m down-gradient of the tailings dam and characterised by high concentrations of sulphate, calcium, magnesium, cobalt, nickel and zinc in groundwater (Table 5.1). Within the main plume heavy metals were low, thought to be due to the precipitation of hydroxides and sulphides and sorption to organics and clays. While pH remained buffered in the impoundment and groundwater the plume was assessed as not posing a major risk. In an assessment of current and post-closure pollution potential from mining (Limpitlaw and Smithen, 2003) identify a wide area that has been impacted by copper mining in the upper Kafue area.

Branan (2008) assessed Kabwe to be one of the ten most poisonous places on earth due to contamination of soils by heavy metals (e.g.Ikenaka et al., 2010; Nakayama et al., 2011; Tembo et al., 2006). Branan (2008) reports studies carried out in Kabwe that indicate tens of thousands of residents are suffering from severe lead poisoning.

Table 5.1 Summary of concentrations of metals in solid tailings and groundwater in Chambisi, Zambia (after von der Heyden and New, 2004)

		Average concentration (range) (mg/L)		**Average concentration (range) (μg/L)**			
Type		**Al**	**Fe**	**Co**	**Cu**	**Ni**	**Zn**
Solid tailings		35,000	63,000	17,000	21,000	1700	2500
Ground-water	Tailing	0.032 (0.02-0.05)	2.05 (1.68-2.42)	24.7 (23.8-25.7)	10.1 (1.7-18.5)	7.5 5.4-9.6)	65.4 (39.7-91.1)
	Plume western	0.03 (0.02-0.05)	5.09 (2.19-7.83)	124 (<50-206)	12.6 (10.0-24.0)	8.1 (4.7-16.0)	79.3 (33.0-133)
	Plume eastern	0.03 (0.02-0.04)	4.16 (0.22-3.68)	9.7 (7.0-10.0	7.8 (3.0-11.3)	4.7 (2.7-5.9)	46.2 (16.5-83.6)
	Back-ground	0.107 (0.09-0.12)	0.352 (0.00-0.52)	6.6 (3.3-9.9)	15.1 (13.0-17.3)	3.0 (1.5-4.5)	23.9 (12.8-35.0)

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6 Groundwater vulnerability and risk assessment

6.1 URBAN RECHARGE

To date very few studies have focussed on urban recharge processes in Africa. Urbanisation affects both the quantity and quality of underlying aquifer systems by (Morris et al., 2003):

- Radically changing patterns and rates of aquifer recharge
- Initiating new abstraction regimes
- Adversely affecting quality

Recharge patterns can be affected by modifications to the natural infiltration system, by changes in natural drainage (for example by less permeable surfaces), and by any water supply network. Water may also be imported from outside of the city (Morris et al., 2003) adding to the volume of water that will recharge local groundwater. The net effect can be a rise in the total volume of recharge. This is most pronounced where on-site sanitation or amenity watering is important, and in arid and semi-arid climates where natural recharge is enhanced. Higher groundwater levels can impede drainage and result in groundwater flooding.

Borst et al. (2013) calculated a water balance for Nablus, Palestine where wastewater is discharged untreated into neighbouring wadis. This indicated that recharge by wastewater was as much as 50% of that from precipitation and that nitrogen pollution was about 60% of the agricultural load.

There are methods available to help to identify water inputs to groundwater. Vázquez-Suñé et al. (2010) proposed a series of tracers to estimate the proportion of water of different origins in the Barcelona, Spain, city aquifers. A preliminary hydrological model is required to choose appropriate compounds. They used a combination of major ions, Cl, SO_4, residual alkalinity, pollution indicators, total N, B, F, Zn, and isotopes $\delta^{34}S$, $\delta\,^{18}O$, $\delta^{2}H$ and a method employing mixing ratios. Their analysis identified leakage from water supplies and the sewage network, rainfall, runoff and surface water infiltration as the main sources of recharge to the Barcelona City aquifers. Kumar et al. (2011) used tritium and stable isotopes to identify recharge zones and sources of water in Delhi, India. They found that groundwater was being recharged by surface water in the dry season and also by precipitation in the monsoon.

6.2 AQUIFER VULNERABILITY

In recognition of the importance of protecting groundwater resources from contamination, techniques have been developed for predicting which areas are more likely than others to become contaminated as a result of human activities at the land surface. Once identified, areas prone to contamination can be subjected to certain use restrictions or targeted for greater attention. The fact that some areas are more likely than others to become contaminated has led to the use of the terminology “groundwater vulnerability to pollution”. Some authors view it as an intrinsic characteristic of the subsurface matrix. Others have associated vulnerability with the properties of individual contaminants or contaminant groups, or specific set of activities at the land surface (Sililo et al., 2001).

The vulnerability of groundwater to pollution depends upon:

- The time of travel of infiltrating water
- The contaminant attenuation capacity of the soil and geological materials through which the water and contaminants travel.

Some assessments take into account the soil zone. The soil is a biologically active zone and many pollutants can be attenuated. For pollutants which are surface spread or applied, such as pesticides or artificial and organic fertilisers, this can be significant. It may also be important for above-ground latrines and solid waste disposal.

Sililo et al. (2001) set out a methodology for rating properties of the soil in Southern Africa as part of their assessment of groundwater vulnerability. In conclusion to their study Sililo and co-workers note the importance of soil properties in determining groundwater vulnerability; for example, in Windhoek, Namibia, the thin soils developed on the underlying Kuiseb schist and amphibolites and fracturing of the bedrock make the aquifer particularly vulnerable. Similarly Conboy and Goss (2000) found that Zimbabwean soils and geological settings did not offer much protection possibly due to deeply weathered joints and fractures. Mapani and Schreiber (2008) assessed the vulnerability of groundwater to a number of urban pollutants including pesticides, oil spills, toxic chemical spills, solid waste dumping, septic tanks and fertiliser applications. Soil properties also have been recognised as important for groundwater vulnerability in the karst Kanye wellfield, SE Botswana (Alemaw et al., 2004).

The unsaturated zone represents the next line of defence against contamination reaching the groundwater and needs to be fully considered in evaluation of risks both in terms of thickness and travel time. Flow in the unsaturated zone is predominantly vertical but there can be spreading on less permeable layers. Natural flow rates in the unsaturated zone of almost all formations do not generally exceed 0.2 m/d in the short term, and less when averaged over longer periods. Water flow and pollutant penetration rates in fractured formations may be an order of magnitude higher (Lawrence et al., 2001). Rock type, degree of consolidation and fracturing are key factors, especially for pathogens. In the saturated zone, attenuation will continue but may be at a lower rate because water moves more rapidly. In this zone, dispersion and dilution play an important role in reducing contaminant concentrations.

Early experiments using lithium bromide tracers in weathered basement geology in Botswana to model the movement of contaminants from a pit latrine to a borehole showed rapid transport times and good recovery, implying fracture flow with little diffusion (Lewis et al., 1980). Taylor et al. (2009) carried out a study in the weathered basement in Zimbabwe using *E.coli* bacteriophage and forced gradient solute tracer experiments. The tracer was largely unrecovered, and detection at the pumping well show that groundwater flow velocities exceed that of inert solutes and are consistent with statistically extreme flow pathways.

Table 6.1 shows typical permeability values for aquifer materials. Sands and gravels which have large well-connected spaces between the grains make good aquifers whereas silts and clays which also have large porosity but little interconnection transmit water very poorly. Fractured rocks transmit water very readily. Table 6.2 shows typical ranges for transmissivity values for both the fractured basement and weathered regolith from SSA. These typically vary by at least three orders of magnitude spatially and vertically (Taylor and Barrett, 1999). The greater the subsurface travel time the greater the opportunity for contaminant attenuation. Aquifer vulnerability can be defined into four broad classes (Table 6.3). Extreme vulnerabilities are associated with highly fractured aquifers which offer little chance for contaminant attenuation. The likely vulnerabilities of a range of broad categories of aquifer types are shown in Table 6.4.

Many large cities abstract water from thick sedimentary sequences e.g. the Karoo aquifers in South Africa, Tanzania, Mozambique, Malawi and Zambia and the regional Iullemeden aquifer system in Mali, Niger and Nigeria. These are often layered with complex flow patterns. Deeper groundwater reserves may be several thousand years old and of naturally good quality; however they may be mineralised at depth. Mountain valley sediments can be thick and also variable in permeability and are similar to the above. Laterite soils are widespread across a large part of SSA. The base of laterite soils have low vertical but high lateral permeability which means that contaminants can be transported significant distances (Bonsor et al., 2013). The variety of aquifer types and properties means that source identification and tracking can be challenging in these environments.

Table 6.1 Permeability values for different rock types (from Lawrence et al., 2001)

Lithology	Range of likely permeability (m/d)
Silt	0.01–0.1
Fine silty sand	0.1–10
Weathered basement (not fractured)	0.01–10
Medium sand	10–100
Gravel	100–1000
Fractured rocks	Variable, 10s or 100s possible

Table 6.2 Transmissivity values for fractured basement and weathered regolith (from Taylor and Barrett, 1999)

T range (m^2/d)	Location (reference)
Fractured crystalline rock	
5-60	Botswana (Buckley and Zeil, 1984)
0.8-90	Zimbabwe (Houston and Lewis, 1988)
0.07-250	Uganda (Howard and Karundu, 1992; Taylor and Howard, 1998, 1999)
Weathered regolith	
0.2-20	Malawi (Chilton and Foster, 1995)
0.04-170	Uganda (1996 (Taylor and Howard, 1996);(Taylor and Howard, 1999)
1-60	Zimbabwe (Chilton and Foster, 1995)

Table 6.3 Aquifer vulnerability classes (from Lawrence et al., 2001)

Vulnerability class	Definition
Extreme	Vulnerable to most water pollutants with relatively rapid impact in many scenarios
High	Vulnerable to many pollutants except those highly adsorbed and/or readily transformed
Low	Only vulnerable to most persistent pollutants in the very long-term
Negligible	Confining beds present with no significant groundwater flow

Weathered basement aquifers occur over large areas of Africa. These have no primary porosity with water being present in both the weathered and fractured layers. The thickness of the weathered layer controls the time of travel and hence the vulnerability (Figure 6.1).

The karst dolomites which underlie Lusaka and cities in the Copperbelt, such as Kabwe, are extremely vulnerable, particularly with the shallow water levels typical of this area. There have been a number of approaches to assessing karst aquifers. Andreo et al. (2006) used the combination of an intrinsic vulnerability map and contaminant properties to prepare specific vulnerability maps for faecal coliforms and petroleum compounds (BTEX) and applied this to a karst case study area in Southern Spain.

Table 6.4 Pollution vulnerability of principal hydrogeological environments (from Lawrence et al., 2001)

Hydrogeological environment		Travel time to saturated zone	Attenuation potential	Pollution vulnerability
Thick sediments associated with rivers and coastal regions	Shallow layers Deep layers	Weeks-months Years-decades	Low-high High	High Low
Mountain valley sediments	Shallow layers Deep layers	Months-years Years-decades	Low-high Low-high	Low-high Low-high
Minor sediments associated with rivers	Shallow layers Deep layers	Days-weeks	Low-high	Extreme
Windblown deposits	Shallow layers Deep layers	Weeks-months Years-decades	Low-high High	High Low
Consolidated sedimentary aquifers	Sandstones Karstic limestones	Months-years Days-weeks	Low-high Low	Low-high Extreme
Weathered basement	Thick weathered layer (>20m) Thin weathered layer (<20m)	Weeks-months Days-weeks	High Low-high	Low High

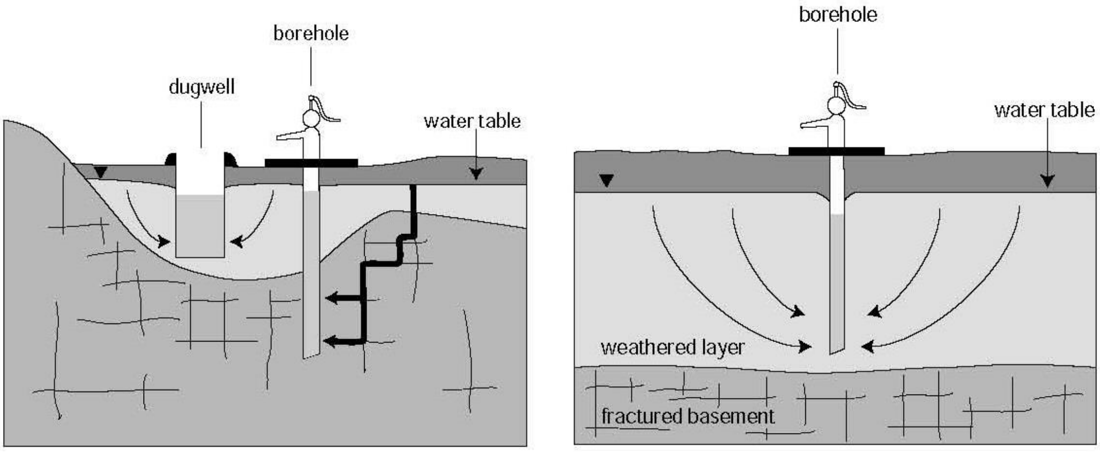


Figure 6.1 Water movement to wells and boreholes in weathered basement aquifers: a) thin weathered layer with fast movement in fractures; b) thick weathered layer (from Lawrence et al., 2001)

Intrinsic vulnerability was derived by two similar methods:

- The PI method which uses the **P**rotection factor related to all the layers between the ground surface and the soil and the degree by which this is bypassed by karst features, such as swallow holes, the **I**nfiltration factor.
- The COP method which uses the **C**oncentration of flow (the surface conditions that control water movement to zones of rapid infiltration), **O**verlying layers and **P**recipitation (both quantity and intensity).

These methods both classified large areas of the Sierra de Líbar as having elevated vulnerability, with the PI method giving greater areas as very high due to the epikarst development. These classifications were validated using tracer tests and hydrographs. Specific vulnerability was assessed by modification of the O factor to taken pollutant attenuation into account.

Aquifer vulnerability mapping was used as a planning tool and applied to the coastal sand aquifer at Calabar, Nigeria, where uncontrolled disposal of domestic, industrial and agricultural wastes have caused groundwater contamination (Edet, 2013). The DRASTIC method used seven parameters (depth to groundwater table, net recharge, aquifer media, soil media, topography, influence of vadose zone and hydraulic conductivity), to produce vulnerability maps. Documented nitrate concentration in hand-dug wells and boreholes are in agreement with vulnerability zones. A recent study undertaken by BGR within the Lusaka region used the DRASTIC method to map aquifer vulnerability. Tilahun and Merkel (2010) also used DRASTIC to assess and map groundwater vulnerabilities in areas of Dire Dawa, Ethiopia. The study successfully identified area of low, medium and high vulnerability.

Ibe et al. (2001) assessed the vulnerability of the Owerri aquifer in southeast Nigeria in order to develop a protection strategy. They compared the results of several models, including GOD, Siga, Legrand and DRASTIC. This confirmed the vulnerable nature of the sandy and gravelly sequences which underlie the city of Owerri and the area to the southwest.

GIS based vulnerability models: e.g. GOD, LeGrand and DRASTIC

The GOD model was developed by Foster (1987) and uses a combination of ratings from three factors; groundwater confinement, overlying strata and depth to groundwater. The LeGrand model (LeGrand, 1964) uses a similar rating method

These earlier models formed the basis of parameter weighting and rating methods of which the most commonly used model is DRASTIC. DRASTIC was developed by Aller et al. (1987) and uses seven parameters; depth to groundwater, net recharge, aquifer material, soil material, topography, retardation in vadose zone and hydraulic conductivity. A DRASTIC vulnerability index is obtained by computing linear combinations of the rating and weights for each factor.

6.3 LOCAL PATHWAYS

As well as moving through the body of the aquifer contamination can occur via pathways resulting from the design and construction of the supply or its deterioration with time (Figure 6.2)

Localised contamination is a very common cause of the decline in quality of groundwater supplies, and is frequently illustrated by the rapid rise in the concentration of contaminants following rainfall events (Ishii and Sadowsky, 2008). It can occur either where contaminated water:

- Is in direct contact with the head-works of boreholes, wells and springs and where pathways exist that allow this to mix with the water supplies
- Has infiltrated into the sub-surface in the close vicinity of a borehole, well or spring moves along fast horizontal pathways to the supply (Lawrence et al., 2001).

6.3.1 Rapid pathways in high risk terrains

Figure 6.2 summarises the key pathways in high risk settings such as fractured basement terrains with lateritic soil, highlighted in orange, including surface and subsurface pathways for migration of pollutants from sources to receptors. Very rapid horizontal pathways exist in the shallow tropical soil zone (5), which may be laterally extensive, providing transmissivities in excess of 300 m^2/day. Rapid vertical pathways also exist due to the presence of natural macro-pores e.g. from burrows and tree roots (6), which can reach significant depths in places.

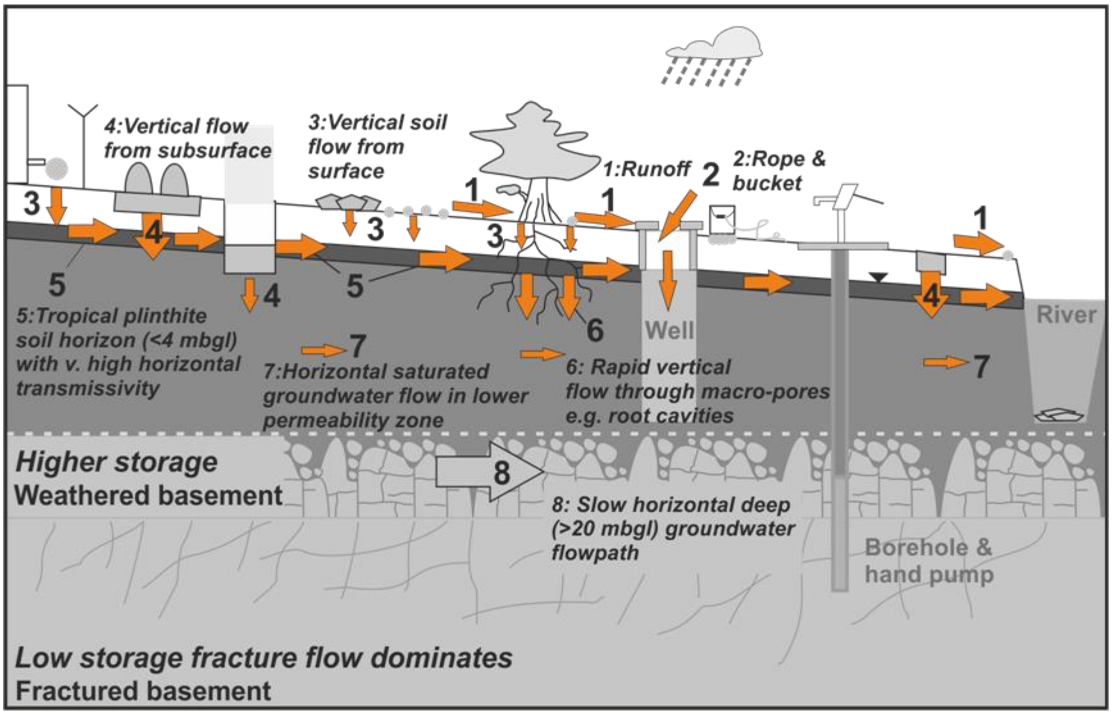


Figure 6.2 Pathways for local contamination into groundwater supplies in high risk basement settings (after Lapworth et al., 2015)

Combined, these more rapid pathways make shallow wells and spring sources particularly vulnerable to contamination and are increased during high water table conditions or when soil infiltration capacity is exceeded. Horizontal saturated groundwater flow, both in the lower permeability horizon above the weathered basement (7) and in the weather basement and fractured basement (8) is a pathway which can affect deeper groundwater sources such as boreholes. These pathways are slower and longer and provide the greatest attenuation potential for hazards.

In areas with red tropical soils and laterites groundwater flow exhibits extremely high permeability characteristics, i.e. very rapid transient pathways may operate for short periods of time and show sudden changes in permeability. The combination of high rainfall and the prevalence of these types of tropical soils suggest that a significant part of Sierra Leone, and neighbouring regions, may be susceptible to these types of extreme hydraulic flow conditions. This, combined with the fact that diffuse open defecation is widespread, cast doubt on the simplistic use of single minimum separation distances from particular hazard sources, and requires further investigation.

Surface pathways include surface runoff (1) which can contaminate surface waters and poorly constructed wells, bypass pathways for contamination of well and spring collectors by ropes, buckets used to draw water (2). Shallow sub-surface pathways include vertical soil flow from surface (3) and subsurface sources (4) where there is hydraulic continuity, e.g. from a liquid discharge or from a buried source such as a pit latrine, cemetery or buried waste.

Localised contamination will result where:

- Potential contaminating activities are not excluded from the vicinity of the headworks;
- Sanitary protection measures employed in the headworks are insufficient;

- The design and construction of a groundwater supply is inadequate.

A summary of these factors is shown in Table 6.5:

Table 6.5 Pathways and indirect factors influencing contamination of groundwater sources (from Lawrence et al., 2001)

Source type	Pathway factor	Contributing factors to contamination
Protected spring	• Eroded backfill or loss of vegetation cover • Faulty masonry	• Lack of uphill diversion ditch • Lack of fence • Animal access close to the spring
Borehole	• Gap between riser and apron • Damaged apron	• Lack of diversion ditch • Lack of wastewater drain • Animal access to borehole
Dug well	• Lack of headwall • Lack of cover • Use of bucket and rope • Gap between apron and well-lining • Damaged apron	• Lack of diversion ditch • Lack of wastewater drain • Animal access to dugwell • Uncontrolled use

Dug wells are one of the lowest-cost forms of water supply. They are particularly vulnerable to contamination as it is difficult to ensure that the lining of the top layers is impermeable. The most common way of collecting water from open wells is the use of rope and bucket technology (Fig 6.6). Ropes and buckets are often left on the ground and are an important route for groundwater pollution by faecal coli forms (Cronin et al., 2006). When large diameter wells are abandoned, due to alternative sources or changes in groundwater tables, they are often then used to dispose of household waste material and become a subsequent source of contamination in the sub-surface

Figure 6.3 Rope and bucket used to draw water

6.4 PROTECTION

Groundwater protection is complex in Africa, with many small sources being used, compared to the fewer larger sources characteristic of Europe (Robins et al., 2007). Nevertheless, some areas lend themselves to conventional zoning of land-use, such as the dolomites underlying Lusaka. In Lusaka many urban supply sources now have 1-km protection zones around them. A key recharge zone is the Lusaka Forest Reserve with land-use increasingly dominated by small-holdings with increased runoff and reduced recharge. Schemes such as 'aquafarms' are being proposed to protect and enhance recharge for municipal supplies in towns such as Kabwe, Central Province, Zambia.

Demlie et al. (2008) used both major ion chemistry and stable isotopes to assess groundwater occurrence and flow in a heavily urbanised area of Ethiopia, including Addis Ababa. They were able to identify three flow systems: shallow controlled by steep topography, intermediate with some recharge from the shallow system, and deep, getting meteoric recharge and interconnected with overlying water through faulting. Understanding these complex flow patterns is key to protecting groundwater.

In many countries well protection zones are defined using deterministic models based on Darcian flow assumptions. Taylor et al. (2004) reviewed evidence from field studies to determine the effectiveness of well protection zone methodologies. They concluded, citing evidence from natural and introduced microbiological tracers,that statistically extreme flow velocities must be considered, and that bulk macropore flow and presumed pathogen survival times (30-50 days) are not consistent with field observations. This is of particular relevance in fissured aquifers e.g. crystalline basement aquifers, as well as the karstic dolomite aquifers, such as those found in Zambia and at the base of lateritic soils which are widespread in SSA (Bonsor et al., 2013).

Frind et al. (2006) developed a methodology for deriving a protection zone for individual wells in a complex multi-layered aquifer. This assessed both the intrinsic vulnerability of the well and the impact of potential sources giving a quantitative risk assessment for water managers. Groundwater protection priorities for Dar-es-Salaam, Tanzania, were established using an empirical model (Mato, 2007). This used five factors to derive a protection score:

- Water quality – rating system developed using nitrate concentrations
- Aquifer yield – rated to provide a public supply (>40 m^3/hour) as the top yield
- Vulnerability – developed using DRASTIC coupled to GIS,
- Use value of water – highest in areas where no piped supply and lowest in areas where groundwater is not used.
- Landuse – highest ease of implementing measures in urban/fringe areas. And lowest in unplanned and industrial areas.

The results indicated areas of the built-up city centre and other areas with a sewerage system had the highest priority for groundwater protection.

In Iganga and parts of Kampala, Uganda, analysis of data from protected springs, wells and boreholes in weathered basement rocks, showed that boreholes were less vulnerable to contamination than wells (Howard et al., 2002). The results indicate that a horizontal separation of 10 m between sanitation facilities and groundwater sources was adequate even in the rainy season. Nsubuga et al. (2004) looked at the effectiveness of protection of a number of springs used to provide water for settlements without a piped, treated supply in Kampala, Uganda. Both high and low income areas were assessed. The protection in place was shown to be ineffective with ammonium, nitrate, faecal coliforms and faecal streptococci detected at over the guideline levels for potable water. Concentrations were higher in the high-density settlements. Pit latrines and animal wastes were found within 5 m of the springs. Furthermore, these springs were vulnerable to a rapid deterioration in water quality immediately following rainfall (Ishii and Sadowsky, 2008), probably as a result of surface contamination accessing the water supply though the highly degraded spring protection (Figure 6.4)

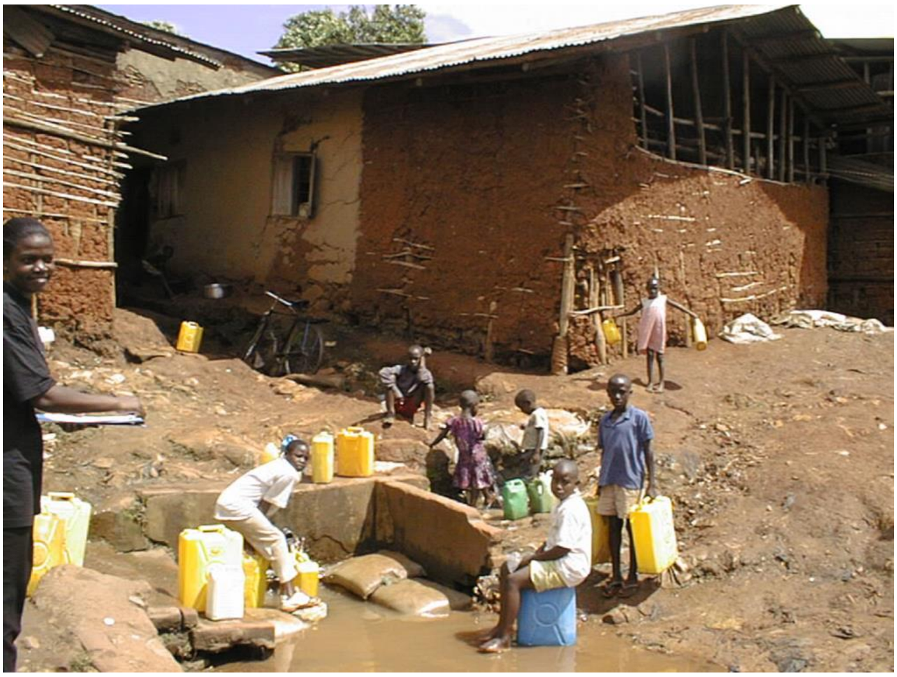


Figure 6.4 Highly degraded spring protection in Kampala, Uganda

Edet et al., (2012) collated results from systematic field sampling and analysis in Calabar, Nigeria, were integrated using descriptive statistics; correlation matrices; bivariant plots; geochemical modelling; and a mixing model to gain insight into the hydrogeochemical processes. The dominant processes controlling groundwater chemistry were found to be silicate weathering, cation exchange and human activity (waste disposal).

Kreamer and Usher (2010) set out the steps for improving groundwater protection in SSA by mirroring the strategies already adopted by more-industrialised nations, namely:

- Define acceptable risk for African populations and ecosystems.
- Establish numerical, heath-based guideline values (beyond World Health Organization guidelines).
- Initiate risk-based remediation predicated on improved site characterization.
- Create hydrogeological and water quality data storage systems that are accessible, electronic, and versatile.
- Formulate a common vision on monitored natural attenuation and technical impracticability.
- Encourage proactive management, leak detection systems, and early remedial action beyond “emergency” response.
- Strengthen natural protected areas.
- Consider implementation of guidance and strategies from other countries.

6.5 RISK ASSESSMENT

Risk assessment needs to demonstrate the presence of hazards:

- Source hazard
- Pathway hazard by which the risk is transmitted
- Receptor which is harmed.

Trauth and Xanthopoulos (1997) described a process in which the impairment of groundwater quality due to non-point sources of pollution in urban areas could be assessed, using Karlsruhe as an example.

This included:

- Design of groundwater quality monitoring network
    - Sampling point design
    - Spatial distribution
    - Sampled depth
- Determination and characterisation of the catchment areas-
    - Groundwater model to determine recharge area – groundwater recharge
    - Anthropogenic pollutant emission risk- landuse
- Establish rules to link landuse and groundwater contamination on the basis of the observed data

They found a number of problems with this type of approach. In urban areas the different types of land use lie close together, and these change with time. The model did not sufficiently account for decreased recharge in paved areas and also did not deal very well with leaking sewers, and particularly the additional recharge from exfiltration

Cissé Faye et al. (2004) assessed the vulnerability of the aquifer underlying the Thiaroye area of Dakar, Senegal, to pollution from urban development and other land uses and related these to aquifer characteristics using GIS. Water quality data were grouped according to landuse features. They found nitrate to be associated with urban areas where the water table was shallow and the aquifer was oxygenated. In contrast nitrate was very low in uninhabited areas, particularly reforested zones.

Love et al. (2004) used factor analysis to develop separate chemical signatures for uncontaminated water, agricultural activities, mining activities and potentially sewage inputs using two examples from Zimbabwe. Using the example of an iron ore mine the analysis found:

- Ca, Mg and HCO_3 representing the dolomitic water signature
- K and NH_4 representing fertiliser or livestock manure
- Na, Cl and SO_4 associated with the main mine dumps and workshops.

Whereas at a sewage disposal works serving Harare they found:

- Cr inversely associated with PO_4, Pb and Ni
- High NO_3, PO_4, minor Fe- sludge and effluent related
- Fe, Ni due to possible geological reason

Orebiyi et al. (2010) showed that high values of colour, turbidity, nitrate, iron, manganese lead, total suspended solids, phosphate, bacteria and total coliforms were related to pollution hazards in urban areas. In peri-urban areas the levels of these contaminant was lower. Robins (2010) propose an approach for the evaluation of vulnerability to pollution of African basement aquifers which uses a tick-box approach for field use for a list of parameters (

Table 6.6). This assessment can be adapted to focus on the parameters important in the area of concern by weighting the score. It excludes parameters which are difficult to evaluate in the field, such as unsaturated zone properties.

There have been a number of approaches to risk assessing karst aquifers. Mimi and Assi (2009) described a two-stage assessment using GIS to make a risk assessment of an aquifer in Ramallah district of Palestine:

Hazard mapping:

- Definition and inventory of hazards – divided into infrastructural, industrial and agricultural activities
- Hazard data requirements – assessing harmfulness of each type of hazards – nature, location, characterisation, quantification

- Rating and weighting of hazards – allocation of value between 10 and 100

Risk mapping:

- Product of intrinsic vulnerability and hazard map.

Using this assessment, Mimi and Assi (2009) identified the main hazards as urban areas with leaking sewers, unsewered villages, wastewater discharge to surface water, solid waste disposal, petrol stations, quarrying and stone cutting and industries including pharmaceuticals, dairy, textiles, detergents and soft drinks. Overlaying this on vulnerability suggested that only 1% of the area was at high risk and 4% at moderate risk.

Ducci (1999) used a GIS-based risk mapping scheme for an area of southern Italy using a similar combination of a hazard and a vulnerability map and combined these with a third layer representing the socio-economic value of the resource.

6.5.1 Sanitary risk assessment

Sanitary risk inspections have been described by the WHO (WHO, 1997) as:

> "*A sanitary inspection is an on-site inspection and evaluation by qualified individuals of all conditions, devices, and practices in the water-supply system that pose an actual or potential danger to the health and well-being of the consumer. It is a fact-finding activity that should identify system deficiencies—not only sources of actual contamination but also inadequacies and lack of integrity in the system that could lead to contamination.*"

Surveys are carried out using a series of simple questions, specific for the water source, designed to identify the common hazards and pathways to contamination that may be present at a small water supply, such as a hand-dug well, or protected spring. Normally, the survey is done at the same time as the water is sampled for analysis of faecal indicator bacteria and the combination of the two results provides a basis for the design of remedial actions to protect the source. However, the results of the surveys can be used on their own, and because they are simple to administer the WHO suggest that surveys should take precedence over analysis (WHO, 1997).

A sanitary risk assessment for open dug wells in the Maldives showed that only 6.4% of wells sampled met the WHO Guidelines for faecal coliforms (Barthiban et al., 2012). The most important factor was the separation between the well and any latrines. Due to the vulnerability of the hydrogeological setting, it was not possible to establish a safe distance. In contrast, Wright and co-workers could not find any correlation between thermotolerant coliforms and sanitary risk score from a survey of 263 wells surveyed over a six year period in Kisumu, Kenya (Wright et al., 2013); however, there was a significant correlation between the nitrate and chloride concentrations and the density of pit latrines in a 100 metre radius of the sample point.

In Bangladesh, tube wells in flood-prone areas were found to be vulnerable to contamination by faecal coliforms. This was poorly correlated with sanitary risk factors, but was better predicted by a history of inundation. In northern Mozambique, there was a high risk of faecal microbiological contamination related to local as opposed to aquifer pathways. The principal factors were stagnant water on and around the wellhead, loose base of the handpumps, cracks at the base of the handpumps, and buckets and rope that can become contaminated (Godfrey et al., 2005).

A more substantial approach for making a risk assessment of private water supplies has been introduced into the UK and is being implemented in England and Wales by local environmental health departments with the support of the Drinking Water Inspectorate (http://www.privatewatersupplies.gov.uk/private_water/21.html; accessed 01/08/2014). However, the principles are the same as the simpler sanitary survey and the results are used to inform the management of water supplies.

Table 6.6 Vulnerability scorecard for weathered basement aquifers (after Robins, 2010)

Parameter	**Indicator**	**Score**
Topography	Flat	0
	With hollows	3
Depth to water	<5 m	3
	5-10 m	2
	>10 m	1
Clay zone thickness	Absent	3
	<2 m	2
	>2 m	0
Degree of weathering	Shallow < 5 m	1
	Thick > 5 m	3
Vegetation and land use	Sparse cover	2
	Farmland	0
	Livestock	1
Laterite present	Absent	0
	Patchy	1
	Continuous	2
Human dependence	High	2
	Other sources	2
	None	0
Polluting activities	Mining	3
	Fuel dumps	2
	Livestock	1
	Few	0
Total		

6.6 RISK MANAGEMENT

Tait et al. (2004) addressed the issue of borehole location in urban areas. The Borehole Optimisation System (BOS) was developed to predict groundwater quality at a potential new sites and to use this to determine the best sites for future development. This was a data-intensive model which integrated a Modflow catchment zone probability model with a GIS-based landuse model, and contaminant information to inform a probabilistic pollution risk model for a user defined borehole. This was applied to Nottingham as a case study. It made assumptions about steady-state abstraction and continuous pollution sources making it perhaps unsuitable for an evolving urban area.

Mapani (2005) assessed risks to groundwater quality and their mitigation for Windhoek, Namibia. Particular source hazards considered were cemeteries, sewage pipe leaks, chemical and petroleum spills, and amenity pesticides. As described in Section 6.2 (Sililo et al., 2001) this aquifer is very vulnerable since the soil cover is thin and groundwater flow is through fractures. In order to manage the risk, Mapani (2005) recommends that:

- A minimum of 20 cm soil cover is maintained over surface expression of faults
- Recharge areas should be protected from further development
- Monitoring of petrol stations, landfills, cemeteries and the sewage system

Pegram et al. (1999) set out the key factors for characterising settlements which can be used to guide the identification of priority problems. These include:

- Water quality problem, priority water quality effects on health, ecology or water treatment, associated water quality problems, critical receiving water
- Settlement character - activities, infrastructure
- Institutional arrangements
- Socio-economic conditions

Using examples from urban centres in Nigeria, Ojo (1995) discuss the reduction of health risks from drinking water. They focus on the control and disposal of solid wastes and the need for public awareness of risks for groundwater and surface water protection.

Risks to health from use of small private supplies can also be managed by implementation of WHO water safety plans. Mahmud et al. (2007) assess their implementation in an area of Bangladesh. These comprised baseline assessments of water quality, sanitary condition and hygiene practices and implementation of actions, such as relocating pit latrines or a level of household-scale water treatment. In the UK, this approach is being used effectively to monitor and manage the quality of private water supplies.

6.7 GROUNDWATER MANAGEMENT

Groundwater has many advantages and is a significant water resource for SSA which can be developed for low capital expenditure compared to large surface water schemes, is generally good quality, has large storage and capacity, can buffer climate extremes and can be developed in close proximity to urban centres, again keeping infrastructure cost down. However, the resource is not without potential quantity and quality problems, and therefore must be managed effectively if it is to be used sustainably in the long-term. This includes assessing requirements for human use such as drinking water and irrigation as well as maintaining base flow for groundwater dependant ecosystems such as wetlands and rivers. There are different dimensions of sustainability including societal, economic and environmental considerations – sustainability is only achieved when there is a discourse between all three dimensions (Gauthier and Archibald, 2001).

Morris et al. (2003) distinguish between two contrasting philosophical standpoints for groundwater management, the ‘technical’ view proposes an incremental approach based on reinforcing existing institutions to address hydrogeological problems, and the ‘holistic’ approach in which coping strategies and technical measures are used together to resolve management issues. The holistic view has its roots in the idea that existing institutions and management systems are not adequate to resolve current and future groundwater problems and that difficult choices may need to be made regarding how the ultimately finite resource is used. The basis of effective groundwater management responses are outlined by Morris et al. (2003):

- Awareness of the status of groundwater (both quality and quantity)
- Understanding of the hydrogeology to be able to identify options to remedy the problem
- Water laws and rights in place that are accepted and clear
- Surveillance, to monitor adherence to regulation
- Awareness in governmental planning and society of the importance of groundwater

It is true to say that these are rarely all in place, and certainly water rights and laws can be difficult and unclear especially with regards to ownership, i.e. landowner vs common pool ownership Morris et al. (2003). The lack of basic data sets to underpin groundwater management options should not be underestimated in many parts of Africa.

Alkhaddhar and Hepworth (2001) set out a ten-point plan for managing groundwater in Dar-es-Salaam resulting from investigation of groundwater quality at a number of sites:

- Coordination of monitoring/sampling
- Hydrogeological investigation and modelling
- Establishment of protection zones
- Quantification and characterisation of current aquifer exploitation
- Promotion of alternative ways to ensure sustainable abstraction
- Promotion of groundwater protection
- Investigation and promotion
- Generation of funding to support activities
- Facilitation of effective communication
- Education and training

Guidance on the mitigation of groundwater pollution in developing countries is set out by Foster and Vairavamoorthy (2013). For sanitation they recommend a more integrated approach to urban water supply, mains sewerage provision, and urban land use to avoid persistent and costly problems, especially where local aquifers are providing the municipal water supply. Public administrations and water service providers can employ a number of simple measures to improve groundwater sustainability (e.g. Drangert and Cronin, 2004; Foster et al., 2010). These could include:

- Prioritising recently urbanised districts for sewer coverage to protect good quality groundwater and/ or limiting the density of new urbanisation served by in-situ sanitation to contain groundwater nitrate contamination.
- Establishing groundwater source protection zones around all utility waterwells that are favourably located to take advantage of parkland or low density housing areas.
- Ensuring availability of ‘nitrate-dilution capacity’ by securing a stable source of high quality supply for blending.
- Involving residents in wastewater quality improvement by seeking cooperation on not discarding unwanted chemical products to toilets or sinks, and avoiding the use of particularly hazardous community chemicals.

Foster et al (2010) state that much more effort is needed to change attitudes towards wastewater reuse and associated energy and nutrient recovery, which can contribute positively to urban groundwater management. New technologies that promote wastewater as a resource need to be tailored to conditions in low-income countries, including low-cost membrane systems, hybrid natural and constructed wetlands and eco-sanitation, which separates urine from faeces and recovers both for reuse. This reduces the subsurface contaminant load. But large scale retro-installation in existing dwellings is not straightforward and it is not well suited for cultural groups who use water for anal cleansing.

Where there is significant industrial activity interspersed with public utility and private domestic wells, it is essential to carry out groundwater pollution surveys and risk assessments. Fuel storage facilities, chemical plants, paint factories, metallic and electronic industries, dry-cleaning establishments, leather tanneries, timber treatment, and waste tips can all discharge mobile, persistent, and toxic chemicals with potential to contaminate groundwater and thus need to be closely monitored. The intensity of subsurface contamination is not necessarily a function of the size of industrial activity. Often small, widely-distributed enterprises use considerable quantities of toxic chemicals and pose a major threat since they operate outside the formal registers and environmental controls.

Groundwater pollution surveys and risk assessments should be commissioned by the public health, environmental, or water resource agencies, in close liaison with water service utilities, using recommended protocols (Foster et al., 2002, reprinted 2007). A typical survey would involve the following steps:

- A systematic survey of existing and past industrial activity to assess the probability of different pollutant types contributing to subsurface contaminant load.
- A groundwater pollution hazard assessment considering the interaction between the subsurface contaminant load and local aquifer pollution vulnerability.
- A detailed groundwater sampling and analysis programme with the analytical parameters being guided by the above survey.

The results of such scientific survey and assessment work should guide policy by:

- Introducing pollution control measures including better constraints on handling and disposal of industrial effluents to reduce groundwater pollution risk.
- Increasing quality surveillance for selected utility wells and/or progressive investment to replace wells considered at greatest risk of serious pollution.
- Advising and warning private domestic well users of potential pollution risks, imposing use constraints, and in extreme cases forcing closure of wells.
- Designing a long-term focused groundwater monitoring programme to improve water quality, surveillance and security.

7 Groundwater quality studies in urban areas

7.1 INTRODUCTION

Urbanisation processes are the cause of extensive but essentially diffuse pollution of groundwater by nitrogen and sulphur compounds, salinity as well as pathogenic bacteria, protozoa and viruses (Morris et al., 2003). Household attitudes to hazards posed by drinking water can enhance quality problems with poor water treatment, types of drinking water vessels, hand washing practices, perceptions of safe water quality using only visual parameters (normally clarity of the water), and knowledge on waste disposal practices (Kioko et al., 2012).

This section firstly reviews a wide range of water quality studies in urban and peri-urban settings in SSA (n=44) assessing the impacts of urbanisation on groundwater quality, these are summarised in Table 7.1. Summary statistics, across all 44 studies for more commonly analysed water quality indicators have been summarised by in Figure 7.1. The second section then reviews case studies (n=16) focussed on the impact of on-site sanitation, principally pit latrines, on groundwater quality, these are summarised in Table 7.2. Targeted studies focused on non-sanitary sources of contamination such as industries, historical mining legacy and waste dumps/landfills are shown in Table 7.3. Key details and conclusions from these studies are summarised and critically reviewed. A hand full of case studies include both specific assessments of impacts of pit latrines as well as broader environmental hygiene considerations in spring catchments and well capture zones and are included in both Table 7.1 and 7.2.

Overall, compared to other regions globally there have been relatively few studies carried out in SSA. The review draws on studies published in books, research articles and reports, and it is recognised that these have been published for a range of purposes. With this in mind, the studies can be categorised into two broad groups. The first being short case-studies presenting data from a limited number of groundwater sites (n<20), limited temporal resolution as a single survey or use only basic chemical indicators (n<3). The second being examples which either draw from larger data sets or include both chemical and microbiological indicators or have greater temporal resolution.

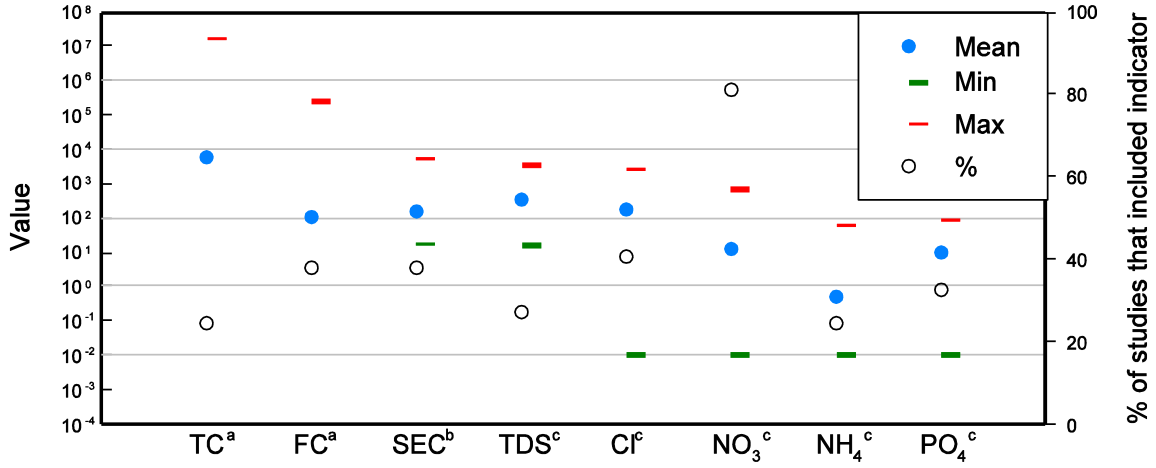


Figure 7.1 Summary statistics for common water quality indicators across 44 studies included in Table 7.1. Units for values on x-axis (log scale): a=counts/100mL, b=µs.cm^{-1} and c=mg/L

Table 7.1 Groundwater quality surveys in Sub-Saharan urban and peri-urban areas (n=44)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*	Sampling time frame	Conclusion and sources of contamination	Reference
Dakar, Senegal	Quaternary	Wells (56)	NO_3 0-122	July-October 1997	Nitrate contamination from point-source seepage in urban areas	Cissé Faye et al. (2004)
Conakry, Guinea	Volcanic rocks, fissured	Wells (69)	**Fountains** TC 0-39 FC 0-28 FS 0-2 $NO_3$0-2.5 $NH_4$0-0.05 Cl 1.2-1.8 F 0-0.05 Turb. 0.9-6.1 **Mod.wells** TC n.d FC 370-1x10^5 FS 90-9x10^3 NO_3 2-46 NH_4 0.06-7 Cl 17-130 F 0-0.16 Turb. 1-70 **Trad. wells** TC n.d FC 50-2 x10^5 FS 150-2 x10^4 $NO_3$7-51 NH_4 0.01-8 Cl 8-284 F 0.0.38 Turb. 1-63	Dry season April-May 1994	Widespread contamination by nitrate and FC linked to poor sanitation and well construction	Gélinas et al. (1996b)
Various. Ivory coast	Basement	Boreholes (230)	NO_3 mean 69 Cl mean 24	1981 and 1982	High nitrate (up to 200 mg/L) linked to domestic pollution and deforestation	Faillat (1990)
Bolama City, Guinea Bissau,	Sandy soils and Cenozoic – Modern sediments	Wells (28)	Turbidity 1-26, mean 6.5 SEC 27-326, mean 136 TC 0-23000, mean 2306 FC 0-5000, mean 410 Fecal Enterococci 0-850, mean 74 NO_3 0.9-55.3, mean 16.6 NH_4 0.01-1.37, mean 0.11 NO_2 0.03-0.13, mean 0.04 Cu, Fe, Cr, As,	July 2006	80% of wells contaminated with FC linked to widespread use of PL	Bordalo and Savva-Bordalo (2007)
Cotonou, Benin	Quaternary to mid Pleistocene sandstone	Dug wells in upper aquifer in densely populated area (379)	SEC 320-1045 Mn 0.06-0.19 NO_3 10.4-118 PO_4 <0.05-21.6 SO_4 3.14-86.3	May 1991, August 1991 and April 1992	High P and K concentrations in upper aquifers linked to anthropogenic pollution	Boukari et al. (1996)
Kumasi, Ghana	Precambrian Basement	Hand-dug wells (10)	TDS 6-230, mean 113 NO_3 0-0.968, mean 0.16 PO_4 0.67-15, mean 7.8 TH 8-103, mean 54 TC and EC <20	N/A	Water quality survey showed that water quality parameters were within WHO drinking water guideline values	Nkansah et al. (2010)
Kumasi, Ghana	Precambrian Basement	Borehole and wells in peri-urban communities (9)	Fe 0.001-0.955 Mn 0.018-0.238 Pb 0.005-0.074 TC 3-16.8×10^6	Monthly between Dec 2000 and Jan 2001	Poor quality overall, contamination linked to proximity to PL and refuse tips as well as livestock	Obiri-Danso et al. (2009)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*	Sampling time frame	Conclusion and sources of contamination	Reference
			FC 1.5-4.37×10^4 Enterococci 1.3-53.5			
Niamey, Niger	Basement	Wells (20)	NO_3 1.4-162, mean 28 $\delta^{15}N$ >+12 ‰	Monthly between July 1989 and April 1990	Contamination found even in deeper sites up to 10 mg/L. Source of loading ($\delta^{15}N$) linked to rapid urbanisation (latrines) and deforestation.	Girard and Hillaire-Marcel (1997)
Ilesha, Nigeria	Basement	Wells (86)	Mean results: NO_3 35 Cl 34 SO4 2.8	Single survey	Evidence of anthropogenic impact on water quality degradation	Malomo et al. (1990)
Benin City, Nigeria	Quaternary to mid Pleistocene sandstone	Boreholes and open wells (6)	Pb 0.03-0.25 Zn 0.98-7.19 Cr 0.02-1.1 Cd Nd-0.23 FC 4600-240000 FS 600-35000	Single survey	Elevated Pb, Cr, Cd and Zn attributed to indiscriminate waste disposal and FC occurrence linked to PL, soak-always and septic tanks	Erah and Akujieze (2002)
Calabar, Nigeria	Tertiary to recent sands and gravels	Existing wells (20)	BOD 0.06-4.09, mean 1.72 N 0.09-3.5, mean 2.15 Cl 0.1-1, mean 0.45 FC 0.75-4.32, mean 1.86	N/A	FC, nitrate and Cl had a positive correlation with urbanisation	Eni et al. (2011)
Ibadan, Nigeria	Basement, banded gneiss and schist	Existing wells (N/A)	TSS 159-186.6, mean 174 Cl 1.1-10, mean 5 TC 2300-9200, mean 5120	Dry season	Gross pollution of groundwater attributed to poor well construction, PL and waste management	Ochieng et al. (2011)
Lagos, Nigeria	Alluvium over sedimentary	Urban wells (18)	TDS 79-1343, mean 514 TH 24-289, mean 110 (as $CaCO_3$?) Na 8-274, mean 79 NO_3 0.05-1.51, mean 0.4 Pb 0-1.9, mean 1.6 Zn 0-4.2 mean 0.3	Survey August to October 2004	Sources of contamination included sanitation, textiles, pharmaceuticals, food, tanneries, motor industry	Yusuf (2007)
Ibogun, Pakoto, Ifo, Ogun State,Nigeria	Cambrian basement geology and weathered regolith	Dug wells, communities of 5000-20,000 people (20)	TDS 100-2200 TH 6-246 NO_3 0.8-88 TC 0-0.6 (cfu x10^5) FC 0-0.2 (cfu x10^5) FS 0-0.7 (cfu x10^5)	July-August 2009	Water quality standards for nitrate, FC, FS not met for significant proportion of wells	Adelekan (2010)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*	Sampling time frame	Conclusion and sources of contamination	Reference
Surulere, Lagos, Nigeria	Alluvium over sedimentary	Wells and boreholes in a middle class area (49)	Al 1-99 µg/L Cd 1-98 µg/L Pb 1-24 µg/L	July 2009	Pb and Cd above WHO drinking water standards in >30% of sites	Momodu and Anyakora (2010)
Abeokuta, Nigeria	Basement igneous and metamorphic	Shallow wells including sanitary survey (40)	All bacterial count>20 Maximum 800 EC+PA+SAL	December 2005	Shallow groundwater is highly contaminated with bacteria. Sources include pit latrines, livestock and solid waste	Olabisi et al. (2008)
Urban & peri-urban area, Abeokuta, Nigeria	Basement igneous and metamorphic	Shallow wells (76)	**Urban** TDS 402 TH 30.3 NO_3 12.02 PO_4 0.21 Pb 0.25 Zn 0.12 TC 10500 **Peri-urban** TDS 263 TH 31.7 NO_3 10.7 PO_4 0.03 Pb 0.19 Zn 0.09 TC 10000	Dry season	Mean values for Pb, nitrate EC and TC > WHO standards. Trading, textiles, transport, cottage industries, pit latrines Generally higher in dry season	Orebiyi et al. (2010)
Peri-urban area, Abeokuta, Nigeria	Basement igneous and metamorphic	Hand-dug wells (25)	TDS 50-270, mean 163 NO_3 2.97-40.7, mean 17.6 NH_4 0-0.59, mean 0.11 PO_4 12-86 µg/L , mean 46 TH 12-210 , mean 106 (as $CaCO_3$?)	Rainy season 2008	Direct surface run off into wells is suggested as possible contamination source	Taiwo et al. (2011)
Warri River plain, Delta, Nigeria	Alluvial Benin formation	Boreholes near WW treatment plant	TDS 16-81 COD 0.4-44.4 NO_3 0.3-1.2 Fe 0.05-0.15	2 year sampling campaign	River infiltration, municipal wastewater, agriculture, oil industry	Ibe and Agbamu (1999)
Warri River plain, Delta, Nigeria	Quaternary and older sedimentary sequences	Dug wells	Fe 0.32-2.75 Pb 0.058-0.443 Ni 0.008-0.188 V 0-4 Cr 0-9 Cd 0.75-8.5 Zn 0-1.8	N/A	Pb, Ni exceed WHO standards. Sources include Warri River, settlement, refinery. Highest values in village 3 km from refinery	Aremu et al. (2002)
Calabar, Nigeria	Lacustrine deposits overlying gravels	Boreholes in low income area (8)	SEC 52-321, mean 149 µS/cm NO_3 0.23-10.63, mean 2.3 NO_2 0.001-0.027, mean 0.008 NH_4 0-5.18, mean 0.88 PO_4 0.007-0.122, mean 0.012 TC 0-7, mean 2.1	Wet and dry season July-Dec 1999	Pollution in low income area linked with pit latrines and shallow waste dump	Ugbaja and Edet (2004)
Masaka, Nigeria	Cretaceous sandstone and clay	Dug wells, high density (12)	TDS 528-935 NO3 44.5-92.5 Alk 67-179	Samples taken in wet season but not during	WHO standards exceeded for a range of contaminants including nitrate, TDS, Cr,	Alhassan and Ujoh (2011)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*	Sampling time frame	Conclusion and sources of contamination	Reference
			Cl 41-118 Fe 0.085-0.199 Cr 0.005-0.0126 Total bacteria 25900-78400	rainfall events to avoid contamina-tion from surface runoff	Cd and TC. High density settlement with shallow water table	
Yaounde, Cameroon	Basement	Springs and wells in high density area (> 40)	For groundwater SEC 18.2-430, mean 87 FC 60% >100 FS 5%>100	-	Groundwater's in high density zones show significant degradation (chemical and microbiological), linked to PL	Ewodo et al. (2009)
Douala, Cameroon	Alluvium over Pliocene sand and gravel	Springs , wells and boreholes (72)	SEC 25-362 NO_3 0.21-94.3 FC 0-2311	-	High levels of FS indicative of contamination from PL, related to age and density of settlement	Takem et al. (2010)
Kinshasa, DR Congo	Alluvial and sedimentary sequences	Wells including sanitary survey	**Dry season** TDS 180-450 NO_3 76-118 PO_4 0.53-4.6 TH 110-149 Pb 0.04-0.09 Cd 0.13-0.20 **Wet season** TDS 200-710 NO_3 97-198 PO_4 3.6-14.6 TH 17-52.5	-	Latrines, metal works, solid waste dump	Vala et al. (2011)
Mekelle, Ethiopia	Mesozoic sediments	Wells, springs and boreholes (100)	SEC 542-5300 TDS 330-3454 NH_4 0.01-2.38 NO_3 0.21-336 Cl 5.76-298 F 0-1.27 PO_4 0.001-0.58	N/A	Highly variable water quality indicative of a range of redox zones and sources of contamination	Berhane and Walraevens (2013)
Bahir Dar, Ethiopia	Weathered and fractured Alkaline Basalt	Dug wells and protected pumps in inner, middle and outer zones (8)	**Middle and inner city** TDS 20-600 NO_3 0.18-57.2 NH_4 0-12 Cl 46-270 FC 93% of sites, mean 1.5 log cfu EC 80% sites mean 1.4 log cfu **Outer city** TDS 20-70 NO_3 0.08-8.8 NH_4 0-12 Cl 0-40	Sampling over a 5 month period 2006/2007	Groundwater contamination linked to population density and urbanisation. All dug wells and boreholes had microbiological contamination in excess of WHO/EU standards. Dug wells had significantly higher FC.	Goshu and Akoma (2011) Goshu et al. (2010)
Addis Ababa, Ethiopia	Volcanics	Boreholes and springs (9)	Alk 8-41 NO_3 0.72-35	Various	The authors made a link between the surface water	Abiye (2008)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*		Sampling time frame	Conclusion and sources of contamination	Reference
			NO_2 <0.01 COD 6.8-41 Cl 6.8-28 PO_4 <0.03-0.1 F 0.22-0.72 As 0.2-0.63 Pb 4.6-25 SEC 300-1200 TC 0-34000			quality and groundwater quality. Major sources of contamination identified were domestic waste, and industrial pollution from textile industry and petrol stations	
Addis Ababa, Ethiopia	Volcanics	Springs and boreholes (10)	Zn 0.87-146 Ni 0.31-0.98 Cu 0.44-1.82 Pb 4.3-56.2 Cd <0.1-0.2 Co <0.1-0.12		2002	Geogenic sources of heavy metals is the likely sources of groundwater contamination in this setting due to high heavy metal concentrations in soils and rocks	Alemayehu (2006)
Addis Ababa, Ethiopia	Volcanics	Springs and wells (63)	Ni 2-152 µg/L Pb <1 Co 0.5-165 As <3 Zn <20-2100 Cu 1.5-164 Cd 0.3-12.3 Cr 18.2-214		Februrary-March 2004, July to September 2005	Urban area, leaching from polluted soils.	Demlie and Wohnlich (2006)
Timbuktu, Mali and Lichinga, Mozambique	Quaternary/ Basement gneiss-granite complex	Hand dug wells: Timbuktu(31), Lichinga (159)	**Timbuktu** SEC 221-2010 NO_3-N 35 med Cl 500	**Lichinga** SEC 220 med NO_3 5.6 med Cl 13.5	Timbuktu September 2002 to May 2003 Lichinga, April 2002-August 2004	Contamination of groundwater sources from on site sanitation traced using N:Cl	Cronin et al. (2007)
Kampala, Uganda	Weathered Basement	Wells and springs	**High density** NO_3 mean 67 Cl mean 59 TC mean 14	**Low density** NO_3 mean 22 Cl mean 21 TC mean 544	Contrasting hydrological conditions	Significantly higher contamination in high density regions compared to low density	Barrett et al. (1998)
Kampala, Uganda	Weathered Basement	Springs (25)	TtC (FC) FS BLD-23000		Monthly between September 1998-March 1999	Evidence of rapid recharge to springs following rainfall. Local environment hygiene and improved sanitary completion shown to be more important than	Howard et al. (2003)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*		Sampling time frame	Conclusion and sources of contamination	Reference
						on-site sanitation for spring protection	
Kampala, Uganda	Weathered Basement	Monitoring wells (16)	**Dry season** SEC 272-345 P BDL-0.11 N BDL-5.5 NO_3 24-144 Cl 31-50.5 TC 0-131 FC 0-35	**Wet Season** SEC 280-372 P BDL0.04 N BDL-263 NO_3 24-692 Cl 28-192 TC 29-10000 FC 6-8300	2003: weekly March-May and September in dry season, and June to August, wet season.	High population density with pit latrines and livestock sources identified. Microbiological water quality deterioration after heavy rainfall	Kulabako et al. (2007)
Lusaka, Zambia	Dolomite	Wells and streams in intensely urbanised area (9)	Broad hydrochemistry suite including key water quality variables: SEC 200-710 NO_3 <0.1-43 NH_4 <0.25-3.5 Cl 4.6-36 PO_4 <0.1-4 B <1-10 As <0.2-0.49 Pb 0.14-0.67 Hg <0.4-13		July 2001	Values for nitrate and Hg were in excess of WHO standards on some occasions. Poor sanitation and solid waste disposal implicated.	Cidu et al. (2003)
Lusaka province, Zambia	Dolomite	Boreholes and wells (28)	Limited inorganic and organic suit, no microbiology		N/A	Nitrate concentrations in GW significantly higher than surface water and compared to copperbelt. Poor sanitation implicated.	Nachiyunde, Kabunga et al. (2013)
Lusaka, Zambia	Dolomite	Private and public boreholes (N/A)	Alk 124-564 NO_3 0.03-39 NO_2 0.002-42 NH_4 0.08-60 Cl 42-102 TC 1-TNTC FC 21-TNTC BOD 2-69 COD 9-320		Various: 1995-2000	Hydrochem, microbiology and incidence of cholera outbreaks compiled to show the rapid deterioration of GW sources associated with poor sanitation	Nkhuwa (2003)
South Lunzu, Blantyre, Malawi	Precambrian Basement	Borehole, springs and dug well (9)	**Dry season** SEC 210-330 Cl 21-35 Fe 0.1-0.8 FC 0-5200 FS 0-640	**Wet season** SEC 306-383 Cl 14-29 Fe 0.4-0.7 FC 0-11,000 FS 0-7000	Wet and dry season on two occasions	Groundwaters highly contaminated due to poor sanitation and domestic waste disposal. 58% of residence use traditional PL	Palamuleni (2002)

Area	Geology	Sample sites (n)	Selected results for TTC, TC, FC, (mg/L) for inorganic chemistry*		Sampling time frame	Conclusion and sources of contamination	Reference
Southern Malawi	Weathered basement	Shallow wells	NO_3 0-2 NH_4 detectable most samples TC None 0, 50% >50 cfu		Wet and dry season (26)	Contamination levels higher during wet season	Pritchard et al. (2008)
Tamatave and Foulpointe, Madagascar	Weathered basement and unconsolidated sediments	Boreholes (53)	FC 73%>0, 55% 0-10, 54%>10 NO_3 4.4-35, mean 23 Pb 1-215, mean ca. 5		Single study	Widespread drinking water contaminated with FC and concerns over Pb from pump materials	MacCarthy et al. (2013)
Dodomo, Tanzania	Basement	Wells, boreholes and springs	NO_3 0.01-331 mean 41 Cl 5.4-1104 mean 238 SO_4 0.01-748 mean 83 SEC 301-4740		Single study	High nitrate (>100 mg/L)common in groundwater. Likely sources identified as sewage effluents	Nkotagu (1996)
Bagamaoyo, Tanzania	Unconsolidated sediments	Shallow wells and boreholes	Entovirus detected in 1% Rotavirus 4% E. Coli virulence genes 42% Human specific Bacteriodales marker 4% Turbidity		Single study	The occurrence of E. coli VG in source water was important for transmission. Significant correlation with turbidity	Mattioli et al. (2013)
Epworth, Harare, Zimbabwe	Granite	Wells and boreholes, transect of formal and informal zones (18)	NO_3 0-30, mean 11 PO_4 0-27.2, mean 3.03 FC 0-2, mean 0.75 (cfu x10^4)		Survey carried out with duplicate sampling	Pit latrines, faecal coliforms in older and informal trading areas, urban agriculture, home industries and commercial areas	Zingoni et al. (2005)
Bulawayo, Zimbabwe	Granite and greenstone	Boreholes, residential, commercial and industrial (32)	**Dry season** SEC 610-1210 NO_3 6-8.2 PO_4 1.7-4.6	**Wet season** SEC 180-1450 NO_3 0.1-1 PO_4 0.5-1.8	Seasonal study	Municipal landfill. N highest in residential, P in commercial and SEC in industrial and residential	Mangore and Taigbenu (2004)

SEC-specific electrical conductivity, TDS- total dissolved solids, TH-total hardness, , BOD-biological oxygen demand, COD-chemical oxygen demand, FC-faecal coliforms, EC- E. Coli, TC-total coliforms, FS-faecal streptococcus. Microbiological units as cfc/100 mL unless stated otherwise, TNCT-too numerous to count, BDL-below detection limit.

Table 7.2 Studies investigating groundwater contamination from pit latrines (n=16)

Region/Country (rural/urban)	Subsurface conditions	Sample sites (n)	Water quality parameters	Sampling time frame	Conclusion	Reference
Kamangira, Zimbabwe (rural)	Sandy soils	Installed test wells (17)	Ammonium, nitrate, turbidity, pH, Conductivity, TC, FC	Feb-May 2005	Low FC >5m from PL, N conc. usually below WHO standards	Dzwairo et al. (2006)
Epworth, Zimbabwe (urban)	N/A	New and existing wells (28)	Na, Zn, Cu, Fe, phosphate, nitrite, TC, FC	N/A	Elevated N and Coliforms in most of study area	Zingoni et al. (2005)
Epworth, Zimbabwe (urban)	Fine sandy soils	Installed wells	Nitrogen, sulphate, coliforms	2-8 week intervals 1998-1999	Rapid reduction in Coliforms, S and N 5-20 m from PL	Chidavaenzi et al. (2000)
Lusaka, Zambia (urban)	Dolomite	Existing wells (?)	Nitrate, Cl, FC	November 2003, March 2004, October 2004	Greatest FC loading from PL and other waste sources in wet season and dilution of N pollution	Nkhuwa (2006)
Dakar, Senegal (urban)	Fine-course sands	Existing wells (47)	Broad hydrochemistry, FC	July and November 1989	Nitrate strongly linked to PL proximity	Tandia et al. (1999)
NW Province, South Africa (rural)	N/A	Existing wells (9)	Ammonium, nitrate, nitrite	June-July	High contamination <11 m from PL	Vinger et al. (2012)
Mbazwana, South Africa (urban)	Sands	Installed test wells (5)	FC and nitrate	Bimonthly 2000-2002	Low nitrate (<10 mg/L) and FC (<10/100mL) >1m from PL	Still and Nash (2002)
Bostwana, Mochudi/Ramotswa (rural)	Well-poorly drained soils	Existing wells (>60)	P, N, stable isotopes and Cl	N/A	Variable N leaching from PL	Lagerstedt et al. (1994)
Botswana (rural)	Clayey soils and fissured geology	Existing well and observation well (2)	Broad Hydrochemistry, *E. coli*	October-February 1977	Contamination of wells near latrine with E. Coli and nitrate	Lewis et al. (1980)
Various, Benin (rural)	N/A	Existing wells (225)	Andenovirus, rotavirus	Wet/dry season 2003-2007	Viral contamination n linked to PL proximity	Verheyen et al. (2009)
Langas, Kenya (urban)	N/A	Existing wells (35)	TC,FC	January-June 1999	97% wells positive for FC, 40% of wells >15m from PL	Kimani-Murage and Ngindu (2007)
Kisumu, Kenya (urban)	N/A	Existing wells (191)	FC, nitrate, Cl, F	1998 to 2004	Density of PL within a 100 m radius was significantly correlated	Wright et al. (2012)

Region/Country (rural/urban)	Subsurface conditions	Sample sites (n)	Water quality parameters	Sampling time frame	Conclusion	Reference
					with nitrate and Cl but not FC	
South Lunzu, Blantyre, Malawi (urban)	Precambrian Basement	Borehole, springs and dug well (4)	SEC, Cl, Fe, FC,FS	Wet and dry season on two occasions	Groundwaters highly contaminated due to poor sanitation and domestic waste disposal. 58% of residents use traditional PL	Palamuleni (2002)
Uganda, Kampala (urban)	Karstic geology	Installed wells and spring (17)	Conductivity, pH, P, nitrate, Cl, FC and FS	March-August 2003, weekly and monthly	Widespread well contamination linked to PL and other waste sources	Kulabako et al. (2007)
Uganda, Kampala (urban)	Karstic geology	Springs (4)	FC, FS., nitrate, ammonium	Wet and dry season for 5 consecutive weeks	Widespread contamination from PL and poor animal husbandry, both protected and unprotected sources unfit for drinking	Nsubuga et al. (2004)
Uganda, Kampala (urban)	Karstic sediments	Springs (25)	FC, FS	Monthly September 1998-March 1999	Spring contamination linked to local environmental hygiene and completion rather than on-site sanitation	Howard et al. (2003)

PL = Pit latrine, FC = Fecal coliform, TC = Total coliform, FS = Fecal strep. Concentrations in mg/L unless otherwise stated.

Table 7.3 Studies focused on impacts of non-sanitary sources on groundwater quality

Area/Country	Geology	Sample sites	Results (mg/L)		Sources	Reference
Akure, Nigeria	Basement	Boreholes in landfill vicinity	TDS 18-342 TH 136-140 NO_3 30-61 Fe 0.9-1.4 Pb 0-1.21 Zn 0-2.3 Cr 0-0.25		Landfill values decrease with distance 50-100 m	Akinbile and Yusoff (2011)
Ojota, Lagos, Nigeria	Sedimentary	10 boreholes, 10 dug wells	SEC 68-3030, mean 584 µS/cm Fe 0-21.4, mean 4.23 Cu 0-33, mean 0.02 Pb 0-14.8, mean 2.4 Zn 0-0.23, mean 0.04		Industrial areas and landfill, Sites within 2 km radius of landfill affected.	Oyeku and Eludoyin (2010)
Igando, Lagos, Nigeria	Sedimentary	Wells 10-375 m from landfill	TDS 3-23, mean 9.0 NO_3 17.4-60.5, mean 38.5 NH_4 0.12-0.3, mean 0.22 PO_4 7.07-15.12, mean 10.7		Municipal landfill	Longe and Balogun (2009)
Ibadan, Nigeria	Basement	Soil and groundwater	Cd 0.01 Cr, Pb, Co, Ni not detected		Municipal refuse dumps	Adelekan and Alawode (2011)
Ilorin, Nigeria	Basement		Colour, turbidity over WHO limit EC 161-731 TH 37-153 TC 1600->1800		Industrial estate	Adekunle (2009)
Dar-es-Salaam, Tanzania	Sedimentary, Sandstones	Wells up and down gradient	**Dry Up gradient** Mn 0.03 Fe 0.07 FC (cfux10^4/100 mL) 1.5 EC 5,400 SO_4 76 **Dry Down gradient** Mn 0.02 Fe 0.12 FC (cfux10^4/100 mL) 3.4 EC 6,500 SO_4 49	**Wet Up gradient** Mn 0.00 Fe 0.12 FC (cfux10^4/100 mL) 0.7 EC 5,000 SO_4 35 **Wet Down gradient** Mn 0.05 Fe 0.24 FC (cfux10^4/100 mL) 3.7 EC 5,300 SO_4 72	Solid waste disposal	Kassenga and Mbuligwe (2009)
Lokpaukwu, Lekwesi and Ishiagu, Nigeria	Shales and igneous intrusions	Springs and open dug wells	**Dry season** TDS 25-3150 Cl 0-30 NO_3 –N 0.04-0.74 SO_4 0-33.6	**Wet season** TDS 33-11,126 Cl 2.1-1155 NO_3 –N 0.04-0.68 SO_4 1-381	Mining	Ezekwe et al. (2012)

Area/Country	Geology	Sample sites	Results (mg/L)		Sources	Reference
			Fe 0-3.98 mg/L Mn 0-0.21 mg/L Pb BDL Zn 0-0.06 mg/L Cd 0-0.258 mg/L	Fe 0-5.07 mg/L Mn 0-0.82 mg/L Pb 0-0.24 mg/L Zn 0-1.07 mg/L Cd 0-0.196 mg/L		
Lusaka and Copperbelt, Zambia	Sedimentary, Dolomite	Surface and groundwater	As 0-0.506, mean 0.009 Cr 0-0.089, mean 0.01 Cu 0-0.270, mean 0.012 Mn 0-10.4, mean 0.369 Ni 0-0.698, mean 0.015 Pb 0-0.094, mean0.003 Zn 0-1.21, mean 0.75,		Mining –Mn, Cu and Ni correlated	Nachiyunde, K et al. (2013)

It is clear from looking at the studies in the published literature that there have been particular concerns related to groundwater quality in southern Nigeria, which account for about 30% of the published studies. These studies are located near Lagos and Abeokuta in the south west, the Delta area in the south and Calabar, and there are additional concerns over pollution from the oil industry in parts of southern Nigeria. Other notable examples of urban areas that have a number of studies include Lusaka in Zambia, Dakar in Senegal, Addis Ababa in Ethiopia and Kampala in Uganda.

7.2 IMPACTS OF URBANISATION

7.2.1 Chemical and physical indicators of groundwater quality degradation

7.2.1.1 PHYSICAL INDICATORS

Total dissolved solids (TDS) or specific electrical conductivity (SEC) are the most commonly applied physical water quality indicators in groundwater studies and are often used in combination with more specific indicators such as dissolved chemistry or microbiology (see Table 7.1). They have a major advantage of being field methods, which are easy to use and versatile, enabling the user to carry out a crude assessment of water quality rapidly and with minimal cost. The baseline quality of groundwater, with relatively low total dissolved solids (TDS), makes TDS a good indicator of contaminant loading. Many studies also use hardness or field alkalinity in the same manner, as these are also indicative of contaminant loading and have many of the benefits outlined above. Geophysical methods such as electrical resistivity tomography (ERT) are also valuable for characterising subsurface contamination from landfills and sites with historical industrial pollution as well as saline intrusion induced by abstraction (Loke et al., 2013; Martínez et al., 2009). However, these are costly systems to deploy there are only a limited number of studies that have used these techniques for assessing groundwater contamination in urban SSA, with the majority of the studies being from South Africa and Nigeria (Akankpo, 2011; Ehirim et al., 2009; Silliman et al., 2010).

7.2.1.2 NITRATE AND CHLORIDE

Nitrate and chloride are the most widely used water quality indicators of anthropogenic pollution. Nitrate data has been reported in over 80% of the groundwater studies summarised in Table 7.1. Relatively simple sample preservation and analysis required makes these parameters attractive for initial water quality screening. Nitrate concentrations ranged from Below Detection Level (BDL) to >500 mg/L (as NO_3), although typical maximum concentrations were generally below 150 mg/L. The WHO guideline value for nitrate is 50 mg/L as NO_3. The WHO has not published a health-based guideline for chloride, but suggests that concentrations over 250mg/l can give rise to a detectable taste.

Both tracers have been used in a broad range of geologic and climate zones to investigate pollution from on-site sanitation, waste dumps, as well as urban agriculture (Table 7.3). Nitrate concentrations show a high degree of variability both within studies and between studies that have been reviewed. Two principle factors that affect nitrate occurrence are firstly the prevailing redox conditions in groundwater, and secondly the residence time and vulnerability of the groundwater body. There are several examples of low nitrate groundwater in Table 7.1 which show evidence of faecal contamination (Gélinas et al., 1996a; Mwendera et al., 2003; Nkhuwa, 2003) which has implications for the potential for denitrification in shallow groundwaters. Nitrate has been used successfully to characterise urban loading to groundwater from a range of sources including pit latrines (Cissé Faye et al., 2004), landfills (Ugbaja and Edet, 2004; Vala et al., 2011) and applied to look at impacts on groundwater quality across different population densities (Goshu and Akoma, 2011; Goshu et al., 2010; Orebiyi et al., 2010). There are other sources of N loading to groundwater in growing urban areas including the impact of deforestations, and these have been delineated using N:Cl ratios and in a few examples by using $\delta^{15}N$ analysis (Faillat, 1990).

A series of geochemical transformations can occur in water with a high carbon loading with a progressive decline in redox potential, leading sequentially to the removal of nitrate by denitrification, the mobilisation of manganese and iron and the reduction of sulphate (Gooddy et al., 1999). Borehole mixing processes can cause dilution and overall low nitrate concentrations while still having significant microbiological contamination. Lagerstedt et al. (1994) and Cronin et al. (2007) have successfully used NO_3:Cl to fingerprint different sources of urban and peri-urban pollution in groundwaters in SSA. This has a certain appeal due to its simplicity; however, prevailing redox conditions and mixing processes need to be considered when using this approach. Many studies have effectively used nitrate in combination with other basic physical indicators such as SEC or TDS and turbidity to assess contamination and map areas of high and low pollution.

7.2.1.3 Ammonium and phosphate

It is evident from the literature that only a minority of case studies (ca. 20%) contain data for NH_4 and close to 30% contain data for PO_4. In part this is due to the more involved analytical procedures for NH_4, the high detection limits for PO_4 by ion chromatography and the fact that these parameters need to be analysed rapidly after sampling to ensure valid results. The WHO have not published health-based guidelines for ammonium and phosphate, however P is often the limiting nutrient in the aquatic environmental and therefore concentrations >20 μg/L are considered high in surface water bodies.

Both species are closely associated with contamination from pit latrines and leaking sewer systems. Examples of ammonium and phosphate contamination from the cities of Lusaka, Abeokuta, Calabar and Makelle are shown in Table 7.1. (Berhane and Walraevens, 2013; Cidu et al., 2003; Taiwo et al., 2011; Ugbaja and Edet, 2004). Ammonium concentrations in urban groundwater range from BDL-60 mg/L, although most case studies had maximum concentrations below 10 mg/L. The highest concentrations were reported in Lusaka, Zambia where karstic limestone aquifer which underlies much of the city and very rapid transport times in the groundwater are implicated. Both indicators do not behave conservatively in soils and groundwater, NH_4 is positively charged and therefore has a strong affinity for negatively charged surfaces such as clays, for this reason, as well as microbiological processing, attenuation is particularly high in the soil zone.

Phosphate concentrations range from BLD-86 mg/L, although very few studies report values >20 mg/L. Phosphate has very limited mobility in the subsurface and has a strong affinity to iron oxy-hydroxides as well as carbonates, background concentrations are usually low, e.g. <0.2 mg/L, concentrations in urban groundwater are also usually low unless there is either a very high loading or very rapid groundwater flow for example in fractured basement or karstic limestone (Cidu et al., 2003; Nkansah et al., 2010; Zingoni et al., 2005).

7.2.1.4 Trace elements

Overall relatively few studies have characterised trace element contamination in urban groundwater, and studies published to date usually report results for only a handful of elements (e.g. Fe, Mn, Pb, Zn, Ni, V, Cr, Cd). This is in part due to the cost and access to suitable analytical facilities in SSA for multi-element analysis by ICP-MS or AES, and the relatively poor detection limits for some single element methods.

Studies investigating trace element concentrations have tended to be focused on non-sanitary sources, either the effect of mining (Ikenaka et al., 2010; Nachiyunde, Kabunga et al., 2013; von der Heyden and New, 2004) or waste dumps (Momodu and Anyakora, 2010; Yusuf, 2007). The main findings from these studies are summarised in Table 7.1 and there are further examples in Table 7.3. Low concentrations of Cd (0.13-0.2 μg/L) and Pb (0.04-0.09 μg/L) were reported by Vala et al. (2011) in a study in Kinshasa, DRC, where groundwater was contaminated from waste dumps. Aremu et al. (2002) report high Pb (0.4 mg/L) and moderate Cr (9 μg/L) and Cd (8.5 μg/L) in groundwaters from the Warri River plain in Nigeria, contamination from industry is cited as the source of contamination. von der Heyden and New (2004) reported elevated concentrations of Co, Ni and Zn down flow gradient of tailings, however concentrations were found to be below WHO drinking water quality standards in all cases, this is likely due to sulphide precipitation and natural attenuation processes.

There have been three notable studies in Addis Ababa, Ethiopia, that have considered natural geogenic contamination within an urban context (Abiye, 2008; Alemayehu, 2006; Demlie and Wohnlich, 2006). These studies report elevated concentrations of Zn, Ni, Pb, Cd, Co and Cr all above WHO drinking water quality standards.

7.2.2 Microbiological indicators of groundwater quality degradation

The importance of concentrating efforts on the control of microbial contamination of drinking water rather than chemical contamination is expressed in the WHO Guidelines for Drinking-Water Quality (WHO, 2011):

> *"The most common and widespread health risk associated with drinking water is microbial contamination, the consequences of which mean that its control must always be of paramount importance."*

Until relatively recently there has been a widely held misconception that groundwater was safe to drink with the minimum of treatment. The overlying soil layers were believed to provide a barrier to the transport of surface contamination to the groundwater, and that pathogenic microorganisms in particular would be removed by filtration or inactivated by sunlight, desiccation, and predation. Although this assumption may be accurate in many cases, it cannot be universally applied: it is particularly dangerous assumption when managing shallow groundwater sources of the type widely used in SSA. Hand-dug wells are widespread in urban and rural areas of SSA and are frequently poorly constructed, inadequately protected and badly maintained, making them vulnerable to contamination from pollution sources on the surface and underground. The microbiological quality of the water in these sources is often poor due to contamination by human and animal excreta.

Although the contamination of water by faeces can be measured by a number of different chemical parameters, it is a common and widespread practice to use bacterial indicators of faecal contamination as the best representative of the presence of pathogens. Of the 44 studies listed in Table 7.1, 38 have included one or more microbiological indicators in the suite of parameters that have been tested. The most widely used group of indicator bacteria in these studies is the coliform group, which includes *E.coli*, but some authors, for example Gelinas et al (1996) and Bordalo and Savva-Bordalo (2007) have also included faecal streptococci as an alternative indicator of faecal contamination. This section will briefly describe the ideas behind the use of faecal indicator bacteria, and then list the main characteristics of the two groups of indicators used in the 44 studies listed above.

7.2.2.1 INDICATOR CONCEPT

The main source of pathogen contamination in water is faeces. Human and animal faeces contains a high microbial load with a very diverse range of species (Leclerc et al, 2001) that are derived from the normal flora of the gut. Many of the bacteria in the gut are either very difficult to culture, or cannot be cultured with the techniques currently available, so estimating the bacterial load in faeces is extremely difficult. Using methods that detect genetic material, O'Hara and Shanahan (2006) have estimated between 10^{11} and 10^{12} bacteria in one gram of colonic content (60% of the faecal mass). Most of these bacteria are harmless saprophytes that colonise the gut and aid digestion, but faeces also contains pathogenic microorganisms, and it is these that are of concern when they get into water. Table 7.4 shows the range of pathogens that have been isolated from groundwater under the different categories of bacteria, viruses and protozoa, and the diseases that they are known to cause.

Table 7.4 Pathogenic microorganisms of concern in groundwater (from: Pedley et al., 2006)

Organism	**Disease**
Viruses	
Coxsackievirus	Fever; pharyngitis; rash; respiratory disease; diarrhoea; haemorrhagic conjunctivitis; myocarditis; pericarditis; aseptic meningitis; encephalitis; reactive insulin-dependent diabetes; hand, foot and mouth disease.
Echovirus	Respiratory disease; aseptic meningitis; rash; fever.
Norovirus	Gastroenteritis
Hepatitis A	Fever; nausea; jaundice; liver failure
Hepatitis E	Fever; nausea; jaundice; death
Rotavirus A and C	Gastroenteritis
Enteric adenovirus	Respiratory disease; haemorrhagic conjunctivitis; gastroenteritis.
Calicivirus	Gastroenteritis
Astrovirus	Gastroenteritis
Bacteria	
Escherichia coli	Gastroenteritis; haemolytic Uremic Syndrome (HUS, enterotoxic *E.coli*)
Salmonella spp.	Enterocolitis; endocarditis; meningitis; pericarditis; reactive arthritis; pneumonia; typhoid fever; paratyphoid fever.
Shigella spp.	Gastroenteritis; dysentery; reactive arthritis.
Campylobacter jejuni	Gastroenteritis; Guillain-Barrè syndrome
Yersinia spp.	Diarrhoea; reactive arthritis
Vibrio cholera	Cholera
Protozoa	
Cryptosporidium spp.	Diarrhoea
Giardia lamblia	Chronic diarrhoea

Ideally, drinking water quality monitoring strategies would include a screen for the major pathogens causing disease, but currently there are methodological problems that make this approach impractical. As shown in Table 7.4 there are a lot of different pathogens and each one has its own method of analysis, which can be slow and expensive. Furthermore, many of the pathogens in the three main categories cannot be cultivated using standard laboratory methods. New methods of analysis that detect nucleic acids, and which can be applied to the detection of any organism, are being introduced, but they are not yet widely available and are expensive and technically demanding. A further constraint on the direct detection of pathogens for routine monitoring is that their occurrence in water is often seasonal, with their presence coinciding with the pathogen circulating in the population (Payment and Locas, 2011).

To overcome the current limitations of testing directly for pathogens in water, microbiologists have identified a small number of enteric microorganisms, universally present in faeces, that are not themselves pathogens but which can be used to indicate the potential risk of pathogens being present in the water. These microorganisms are referred to as faecal indicator organisms. The characteristics of an ideal indicator are (Payment and Locas, 2011):

- Should be absent in unpolluted water and present when the source of pathogenic microorganisms of concern is present.
- Should not multiply in the environment.
- Should be present in greater numbers than the pathogenic microorganisms.
- Should respond to natural environmental conditions and water treatment processes in a manner similar to the pathogens of concern.
- Should be easy to isolate, identify, and enumerate.
- Should be inexpensive to test thereby permitting numerous samples to be taken.
- Should not be pathogenic microorganisms (to minimise the health risk to the analyst).

The coliform group of bacteria, which includes *E.coli*, and faecal streptococci (enterococci), are the most widely used groups of faecal indicator bacteria; however, none of them fulfil all of the characteristics of an ideal indicator and there are widely reported limitations to their use. Despite the negative publicity faced by coliforms and other faecal indicators, they are likely to continue being used in water quality monitoring programmes for the foreseeable future. This is reflected in the conclusions of a recent review of faecal indicator studies by Payment and Locas (2011) who state:

> "*Fecal indicators are the best predictor of potential risk, but their concentrations rarely correlate perfectly with those of pathogens. Thus, bacterial indicators can predict the probable presence of pathogens in water, but they cannot predict precisely the level of occurrence*."

In the following sections we describe the basic characteristic of the coliform group of faecal indicators, and faecal streptococci. More comprehensive reviews of faecal indicators have been published (Ashbolt et al., 2001; Gleeson and Gray, 1997; Leclerc et al., 2001; Mesquita et al., 2013; Tallon et al., 2005).

7.2.2.2 Total Coliforms

This is a diverse group of bacteria that share several biochemical and physiological properties. They are all Gram negative (a simple staining procedure that divides bacteria into two groups - Gram positive and Gram negative – that is determined by the cell wall structure), rod-shaped organisms. They will grow in the presence of bile salts (chemicals excreted into the gut by the gall bladder) and utilise lactose as a sugar source, producing acid and gas as metabolic by-products. They are oxidase negative and do not produce spores or other sub-cellular structures that are highly resistant to environmental stress. More recently they have been grouped according to having the enzyme B-galactosidase, which is involved in the metabolism of lactose and can be detected using a colour producing substrate. They grow at 37^{o} C and are facultative anaerobes, which mean that they can grow in the presence and absence of oxygen using two different metabolic systems.

The total coliforms include several genera of bacteria. From early on it was discovered that several of these genera contain bacterial species that can grow in the environment in the absence of faecal contamination. Thus the isolation of total coliforms from water did not necessarily mean that the water was contaminated by faeces. In a an analysis of over 1000 strains of Enterobacteriacae from drinking water, 51% were not of faecal origin (Gavini et al., 1985). Their role as faecal indicators, and their potential significance as a measure of health risk, was thus undermined and recently many countries have reduced their significance in the list of water quality indicators (Stevens et al., 2003).

Tables 7.1 to 7.3 above list several studies that have included total coliforms amongst the suite of microbiological parameters that have been used to measure the quality of water from a variety of sources. But given the widespread occurrence in the environment of species belonging to the total coliform group (Leclerc et al., 2001), the significance of their presence in shallow groundwater in peri-urban areas is questionable.

7.2.2.3 THERMOTOLERANT COLIFORMS AND *ESCHERICHIA COLI*

Thermotolerant coliforms (sometimes referred to as faecal coliforms) are a subgroup of the total coliform group. They are differentiated from the total coliform group by their ability to grow at 44ºC. The thermotolerant coliforms are much more indicative of faecal contamination than the total coliform group; however, this group still contains some genera of bacteria, particularly *Klebsiella, Enterobacter* and *Citrobacter* that can be isolated from environmental sources in the absence of faeces (Tallon et al 2005).

The method for quantifying thermotolerant coliforms is relatively straight forward because it only requires incubation at 44ºC to select for the organisms; no further selective growth steps or confirmation steps are performed on the isolates. In many laboratories in developing countries this is the limit of the resources that are available, and several of the more well-known field test kits (for example, Potalab, Delagua and Paqualab) can only be this specific. But there is a stronger justification for using TTC as an indicator of faecal contamination in drinking water than total coliforms. *E.coli* is the predominant species of the TTC group and constitutes over 90% of the species isolated from faeces (Tallon et al, 2005), and as much as 99% of TTC isolated from shallow groundwater in peri-urban areas (Howard et al, 2003). It is the only species in the coliform group that is exclusively associated with a faecal source (animal and human) in temperate waters. The association is not quite so clear in tropical waters, with the results of some studies suggest that *E.coli* can be isolated from pristine waters (Carrillo et al., 1985; Ishii and Sadowsky, 2008) and from certain industrial discharges that have not been exposed to faeces (Gauthier and Archibald, 2001). Nevertheless, *E.coli* is recommended as the sole indicator bacteria of recent faecal contamination (Tallon et al, 2005).

The predominance of *E.coli* in the TTC group supports the assumption of faecal contamination in drinking water based on the TTC result alone. However, further confirmation of TTC isolates as *E.coli* proves the presence of faecal contamination. *E.coli* has the general characteristics of TTCs but, in addition, synthesises the enzyme β-glucuronidase, which has been used as a target for the development of simple, relatively rapid, specific and sensitive detection methods (Tallon et al, 2005). To date, these methods based on specific enzyme detection have not been used widely in developing countries due to the cost of equipment and the availability of reagents. But recent developments have created low-cost packaging for the test that removes the need for equipment (for example see: http://www.aquagenx.com/) and makes it more appropriate for use in low-income settings.

7.2.2.4 ENTEROCOCCI

Faecal streptococci (sometimes referred to as enterococci or faecal enterococci) are not used as widely as the coliform group in SSA as an indicator of faecal contamination. Of the studies listed in Table 7.1, eight report the use of enterococci.

Enterococci are between ten-fold and a thousand-fold less numerous in human faeces than *E.coli*, (Gleeson and Gray, 1997; Stevens et al., 2003) but are generally more numerous than *E.coli* in faeces from herbivores. Enterococci survive longer in water than *E.coli* and can be used to indicate older contamination sources. Furthermore, there is no evidence to suggest that enterococci can replicate in the environment, even in tropical regions where some authors have reported the growth of *E.coli*.

Several major studies have been done during the last 30 years to determine the relative significance of *E.coli* and enterococci as an indicator of health risk, rather than as an indicator of faecal contamination. The majority of these studies have concentrated on the health effects of sea-bathing and have reached the conclusion that enterococci are a better predictor of the risk of ill-health than *E.coli (*for example Kay et al., 1994) and they have now been introduced as the indicator of choice in several countries. The relative significance of *E.coli* and enterococci as an indicator of health risk in drinking water is less well established; however, Howard et al (2003) conclude from a study of springs in Kampala, Uganda, that faecal streptococci may be more useful as an indicator relevant to public health.

7.3 IMPACTS FROM PIT LATRINES

In-situ sanitation, largely in the form of pit latrines, is the dominant cause of microbiological contamination and a major cause of nutrient loading to water resources in SSA. This is a very well-studied area and a worldwide review has been published recently by Graham and Polizzotto (2013). The main findings from studies carried out in SSA have been collated in Table 7.2 and are summarised below along with other studies specifically targeting contamination from pit latrines.

7.3.1.1 Microbiological contaminants

Human faeces harbour a large number of microbes, including bacteria, archaea, microbial eukarya, viruses, protozoa, and helminths (Graham and Polizzotto, 2013). In the context of this review there have been no studies that have assessed protozoa or helminths, which exhibit little movement in groundwater due to their size (Lewis et al., 1982). The characteristics of microorganisms and the aquifer and soil environment that affect microbial transport and attenuation in groundwater are shown in Table 7.4.

Microorganisms have been assumed to be rapidly attenuated after excretion but recent studies with viruses suggest that water quality may be impaired for a considerable length of time. Using a mixture of routine culture methods and genetic detection methods, Charles and coworkers detected viruses over 300 days after they were introduced in simulated groundwater systems (Kay et al., 1994). A number of approaches have been used to define the quantities and transport distances of latrine-derived microbial contaminants. The majority of these have been culture-based studies of faecal bacteria; there has only been one study of viruses related to pit latrines (Verheyen et al., 2009).

Attenuation of microbes is likely to be dependent on the hydrological conditions both in terms of water levels and recharge rate and permeability of the aquifer, and is highly variable (section 7.2.2). Dzwairo et al. (2006) found faecal and total coliforms greatly reduced beyond 5 m from pit latrines in Zimbabwe, whereas Still and Nash (2002) found faecal coliforms to be attenuated to <10 cfu/100 mL after 1 metre in Maputaland, KwaZulu-Natal. In Abeokuta, Nigeria, Sangodoyin (1993) found coliform attenuation to be correlated both with distance from the source and with the depth of the groundwater well. In Epworth, Zimbabwe, groundwater contamination was higher in the dry season rather than in the wet, with coliforms detected up to 20 metres from the pit (Chidavaenzi et al., 1997). In Benin, Verheyen et al. (2009) found a positive association for detection of viruses in water sources with at least one latrine within a 50 m radius. They postulated that during the wet season viruses were transported in shallow groundwater whereas in the dry season contamination was likely to be from surface water.

In an informal settlement in Zimbabwe, Zingoni et al. (2005) found detectable total and faecal coliforms in over 2/3 of domestic boreholes and wells. In the area 75% of households used pit latrines and there were also informal trading areas. In Langas, Kenya, Kimani-Murage and Ngindu (2007) found that 50% of wells were within 30 m of a pit latrine and that all shallow wells were positive for total coliforms with 70% >1100 mpn/100 mL; however, in Kisumu Wright and co-workers failed to find a significant correlation between the levels of thermotolerant coliforms in water sampled from shallow wells and the density of pit latrines (Wright et al., 2012).

Table 7.4 Factors affecting transport and attenuation of microorganisms in groundwater (from Mesquita et al., 2013)

Characteristics of the microorganism	Aquifer/soil (environment) properties
Size	Groundwater flow velocity
Shape	Dispersion
Density	Pore size (intergranular or fracture)
Inactivation rate (die-off)	Kinematic/effective porosity
(Ir)reversible adsorption	Organic carbon content
Physical filtration	Temperature
	Chemical properties of groundwater (pH etc.)
	Mineral composition of aquifer/soil material
	Predatory microflora
	Moisture content
	Pressure

7.3.1.2 Chemical contaminants

The chemicals of greatest concern from excreta disposed in on-site sanitation systems are regarded to be nitrate, phosphate and chloride. Pin-pointing specific sources is challenging as nitrate may be derived from numerous sources including plant debris, animal manure, solid waste and fertilisers. A common approach has been to compare areas that are similar but have different latrine densities. In an informal settlement in Zimbabwe, Zingoni et al. (2005) demonstrated that the highest nitrate concentrations were associated with the highest population and pit latrine density. A similar pattern was observed in Senegal and South Africa. (Tandia et al., 1999; Vinger et al., 2012). Studies in the peri-urban areas of Kisumu, Kenya have shown that the density of latrines within a 100m radius of the sources was significantly correlated with nitrate levels (Wright et al., 2012). In eastern Botswana the build-up of nitrogenous latrine effluent in soils and downwards leaching resulted in nitrate concentrations of above 500 mg/L (Lewis et al., 1980). The highest concentrations are found downstream of areas with high latrine use (Vinger et al., 2012). Direct measurements are sparse but Graham and Polizzotto (2013) estimate lateral travel distances of 1-25 m for pit-latrine derived nitrate. Sangodoyin (1993) found that nitrate concentrations were not related to distance from the source in Abeokuta, Nigeria.

Chloride is typically transported with minimal retention and frequently tracks nitrate (e.g. Lewis et al., 1980) unless subsurface conditions promote denitrification. Ammonium does not tend to accumulate in groundwater near latrines but can accumulate and persist in anaerobic conditions and when the water table intersects the base of the latrine pit (Ahmed et al., 2002; Dzwairo et al., 2006). Other contaminants include potassium, sulphate and DOC.

7.4 IMPACTS FROM NON-SANITARY SOURCES

Table 7.3 summarises studies focused on the impact of waste dump sites, industrial activity and mining on groundwater quality. There are only a handful of case studies which have characterised the impacts of waste dumps and industry in urban areas and these are dominated by examples from Nigeria. Examples of contamination from historical mining activity come from Nigeria (Ezekwe et al., 2012) and Zambia (Nachiyunde, Kabunga et al., 2013).

Oyeku and Eludoyin (2010) investigated contamination in wells and boreholes from industrial and waste dumps in Ojota, Nigeria. Sites within a 2 km radius were found to be affected with elevated concentrations of Fe, Cu, Pb and Zn as well as high SEC. In Akure, Nigiera, boreholes in the vicinity of a landfill were found to be contaminated, with high nitrate (36-61 mg/L), Pb and Zn

(Akinbile and Yusoff, 2011). In Lagos, Longe and Balogun (2009) found elevated concentrations of nitrate (mean 38.5 mg/L), phosphate (mean 10.7 mg/L) and ammonium (mean 0.2 mg/L) as well as high TDS (3-23 g/L) in wells between 10-400 m from a municipal landfill. Kassenga and Mbuligwe (2009) characterised seasonal and up/down gradient water quality in wells in the vicinity of a solid waste dump in Dar-es-Salam, Tanzania. Higher FC counts were found in sites down gradient ($3.4\text{-}3.7\times10^4$) compared to up gradient ($1.5\text{-}0.7\times10^4$). Higher FC counts were also found during the wet season compared to dry season down gradient of the dump site. Lower SEC as well as SO_4 and Fe were found up gradient and in the wet season, implying dilution from rainfall and recharge and contamination from the dump site.

Elevated concentrations of Pb (0.24 mg/L), Cd (0.25 mg/L) and Zn (1.07 mg/L) were found in wells in the vicinity of mining activities in Nigeria (Ezekwe et al., 2012). A study by Nachiyunde, Kabunga et al. (2013) included sites from the Copperbelt in Zambia and found overall low contamination levels in groundwater with no trace elements above WHO drinking water guideline values. Von der Heyden and New (2004) also carried out a detailed study in the Zambian Copperbelt of the impact of mine tailings on groundwater quality. They found that there was only local effect from the tailings and that concentrations of Co, Ni and Zn were below WHO drinking water quality standards.

7.5 SEASONAL TRENDS IN GROUNDWATER QUALITY

There are relatively few studies that have undertaken regular water quality monitoring or have carried out seasonal comparisons. A study by Howard et al. (2003) is one notable example where detailed seasonal monitoring of microbiological indicators was carried out over a twelve month period to characterise the risks factors for spring contamination in Kampala, Uganda. Significantly higher contamination was observed after rainfall events and there was strong evidence that rapid recharge of the shallow groundwater causes a rapid response in spring quality (Ishii and Sadowsky, 2008).

Median and maximum results for FS, SEC and NO_3 from four studies that assessed seasonal effects by carrying out sampling at the same sites in both the wet and the dry season are summarised in Figure 7.2. Higher maximum FS counts are found in the wet season compared to the dry season for studies in Uganda (Kulabako et al., 2007) and Malawi (Palamuleni, 2002). Higher maximum SEC are observed in all three case studies in Figure 7.2 during the wet season, however median values are comparable. Changes in nitrate show a mixed picture with higher maximum concentrations in two studies from Uganda and DRC (Kulabako et al., 2007; Vala et al., 2011) during the wet season, while in the case study from Zimbabwe (Mangore and Taigbenu, 2004) lower maximum values are found. Median values for nitrate are lower in the wet season for both the Uganda and Zimbabwe case studies, which may indicate a dilution effect, while the higher maximum concentrations may be explained as a result of a pulse of contaminants at the start of the rainy season and or evaporative effects during the dry season.

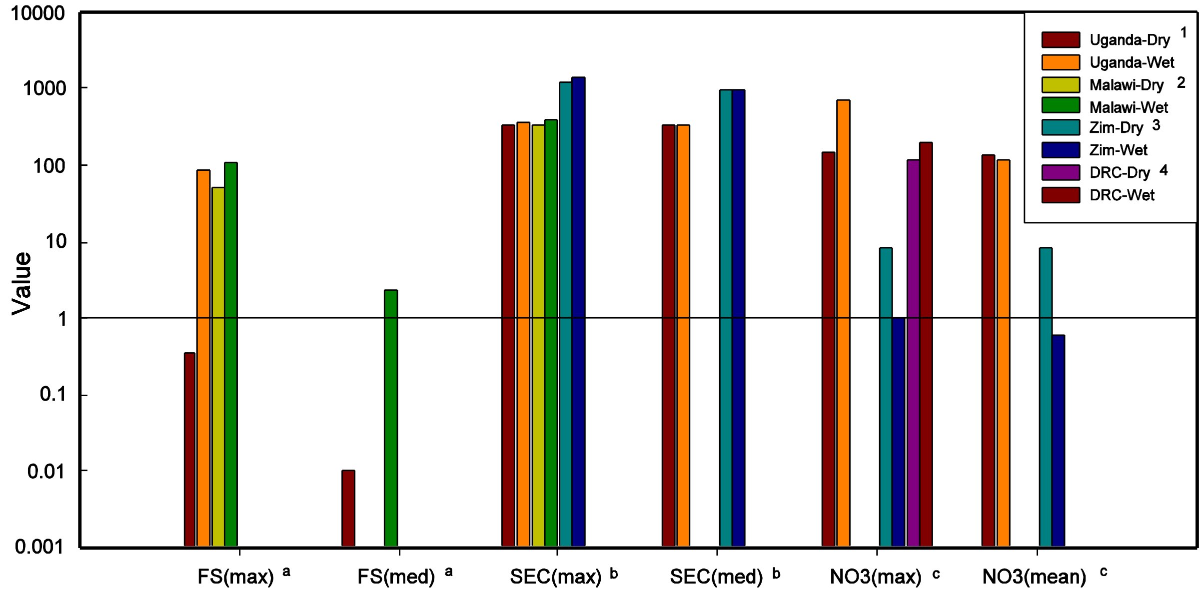


Figure 7.2 Comparison of groundwater quality during contrasting rainfall regimes. X-axis: a=cfu/100mL, b= μS.cm^{-1}, c= mg/L. Key: 1=Kulabako et al. (2007), 2=Palamuleni (2002), 3=Mangore and Taigbenu (2004) and 4=Vala et al. (2011). Some data for median and maximum values are not available from the literature.

Understanding seasonal trends in nitrate are complicated by the changes in redox conditions, particularly in low lying areas which are prone to flooding in the wet season which are not uncommon in SSA, e.g. Lusaka, Zambia. These may shift from an oxidising regime in during low water levels which retains NO_3 to a reducing regime where denitrification can take place during inundation (Sanchez-Perez and Tremolieres, 2003; Spalding and Exner, 1993).

7.6 COMPARISONS BETWEEN DIFFERENCE WATER SOURCES

The results from Table 7.1 show water quality data from a range of different sources. Open and unlined wells are consistently of poorer quality compared to lined or ‘improved’ wells (e.g. Godfrey et al 2005; Sorensen et al., 2015a; 2015b). In some studies springs have been found to be better quality compared to boreholes (e.g. Takem et al. 2010) and others cases the trend is reversed (e.g. Palamuleni 2002) or both sources were found to have comparable levels of contamination by FC (e.g. Abiye 2008). It is important to note that many of these studies contained very few observations for each source type and generalisations should be treated with caution. However, together they form a more compelling body of evidence.

There is some evidence that the water quality of wells may be affected by usage rates, i.e. with fewer groundwater sources being relied on towards the end of the dry season there is greater risk of contamination, e.g. from materials used for drawing water, especially for unimproved sources (Godfrey et al. 2006). For boreholes this contamination pathway is generally not a major risk factor and this supports the generally better quality found in these types of sources. The lower storage volume of shallow boreholes compared to wells may also be an important factor as this type of contamination can be more rapidly flushed out.

Improved wells do not generally exhibit the same level of gross contamination observed in traditional wells and springs. However, in the majority of studies, wells (both improved and unimproved), are found to have water with unacceptable levels of contamination with faecal coliforms by WHO standards (and typically > 100 cfu/ 100 mL) in at least some part of the year and often throughout the year. With perhaps the exception of highly karstic settings for microbiological and nitrate the following order of water source quality (best to worst) was found as follows: boreholes >> improved wells = springs > traditional wells.

8 Lusaka case study

8.1 SETTING

Lusaka is the capital of Zambia with a rapidly-growing population. The 2010 national census reports the population of Lusaka as 1.7 million, with an average annual growth rate between 2000 and 2010 of 4.9% (Abujafood, 2013). The population of the Lusaka Province is c. 2.2 million; an increase of 0.7 million since 2000. Population growth is from both migration from rural areas but also from the Copper Belt. The majority of the population in Lusaka, up to 70%, live in peri-urban settlements, both formal and informal. Low–income, densely populated settlements lie to the north, north-west and south of the central district. These generally lack well-functioning water and electricity supplies, sewerage systems and solid waste collection (Grönwall et al., 2010).

Nkhuwa (2003) reported a high incidence of gastro intestinal diseases, such as diarrhoea, dysentery and cholera between 1997 and 1999. In this period infant mortality in the poorest households was as likely in the most developed province of Lusaka as the poorest area in Zambia (Madise et al., 2003). Grönwall et al. (2010) also reported cholera outbreaks in the city.

Nyambe et al. (2007) state that Zambia is one of the most urbanised countries in southern Africa with 40% of the population living in urban areas. This is concentrated in unplanned settlements or peri-urban areas of Lusaka and the Copperbelt towns of Ndola and Kitwe. Common features of these settlements are overcrowding, poor water supply and sanitation coupled with high rate of unemployment and disease. There are 372 peri-urban areas in Zambia of which only 51% are legalised. Residents are supplied with water through domestic connections, communal taps and water kiosks (Nyambe et al., 2007). Sharma et al. (2005) evaluates the sustainability of handpumps in rural and semirural areas of Zambia.

8.2 HYDROGEOLOGY

Lusaka is underlain by a thick sequence of Pre-Cambrian Katangan metasedimentary rocks (Figure 8.1). The northern part of the city, where the formal settlements are, is underlain by strongly folded and faulted Ridgeway Schist and Matero Quartzite. Some groundwater is available from carbonate sequences in the schists. Below this are high-yielding karstified limestone/dolomitic marble of the Lusaka Dolomite Formation, which out-crops in the southern part of the city. Yields are highest in the top 30 m or so of the strata, where fissures are well developed. These aquifers provide a significant proportion of the water supply to the city of Lusaka, especially from karstic sections, where boreholes yield up to 35–50 l/s (BGS, 2001).

Groundwater flow is from SE to NW. The water table is relatively shallow, typically 6 to 15 m below ground surface, but can be as shallow as 0.5 m. Water levels are declining in some places, ascribed to over abstraction, which leads to shallow wells drying-up during the dry season, particularly in the schist and quartzite areas. In other areas, abstractions mitigate the annual risk of flooding (Von Münch and Mayumbelo, 2007; De Waele et al., 2004; Mpamba et al., 2008).

Yields from boreholes in the limestones are high, but because of the karst nature of the aquifer, the success rate is less than 50%. Low yields and borehole failures in the schists and quartzites are attributed to poor local drilling practice (Grönwall et al., 2010). Typical borehole depths are around 50–70 m below ground level.

The Lusaka terrain shows a complete lack of surface water drainage due to the karst nature of the aquifer and, as such, rain and wastewater drain rapidly to groundwater with little natural attenuation (Nkhuwa, 2006).

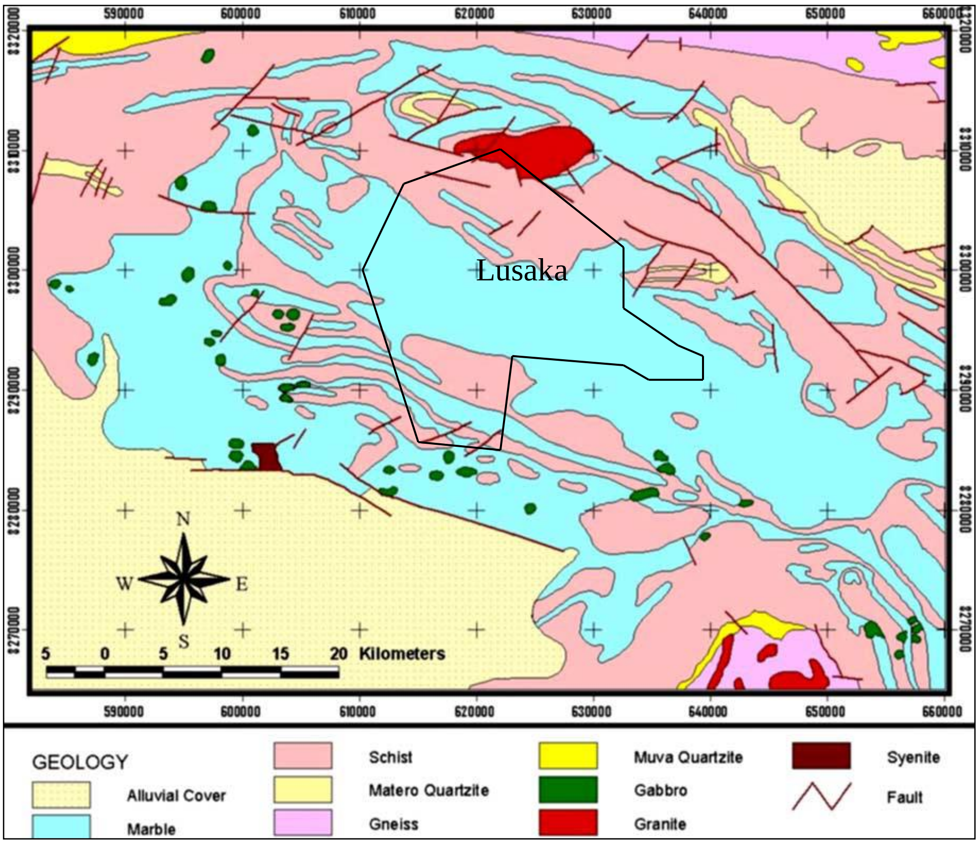


Figure 8.1 Geological map of the Lusaka area (Source: Nkhuwa (2003))

8.3 WATER SUPPLY

The city derives about 70% of its total water supply, which amounted to some 240,000 m^3/day in 2004, from groundwater with the remainder coming from the Kafue river (De Waele et al., 2004). It is estimated that between 80,000 and 150,000 m^3 of groundwater are also abstracted daily from the ground by private boreholes and shallow wells (De Waele et al., 2004; Nyambe et al., 2007). In 2010, it was estimated that there might have been 1900 registered private boreholes and at least as many unregistered ones. In addition, there are a large number of shallow hand-dug wells in backyards. Figure 8.2 gives the spread of boreholes across the Lusaka landscape in the late 1990s (Nkhuwa, 2003).

Most residents in high-density settlements lack a connection to the reticulated supply. The only options, when/where available, are communal taps, community based schemes using NGO boreholes, rare public taps, public hand-pumps and water kiosks, where water may be purchased from the water company or from private borehole owners. As such, shallow hand-dug wells are common in these areas of Lusaka, most of which are open and lack any stone or brick lining. These give moderate to good yields since water is drawn in relatively small quantities. Little, if any, of the water from these private sources is treated before use (Grönwall et al., 2010). Industries and commercial farms in and around Lusaka also use groundwater.

UN Habitat (2010) assessed that almost all households in Lusaka, including those in the urban area, have access to safe drinking water (Figure 8.3), but this may be a long distance away and may necessitate spending long hours in queues.

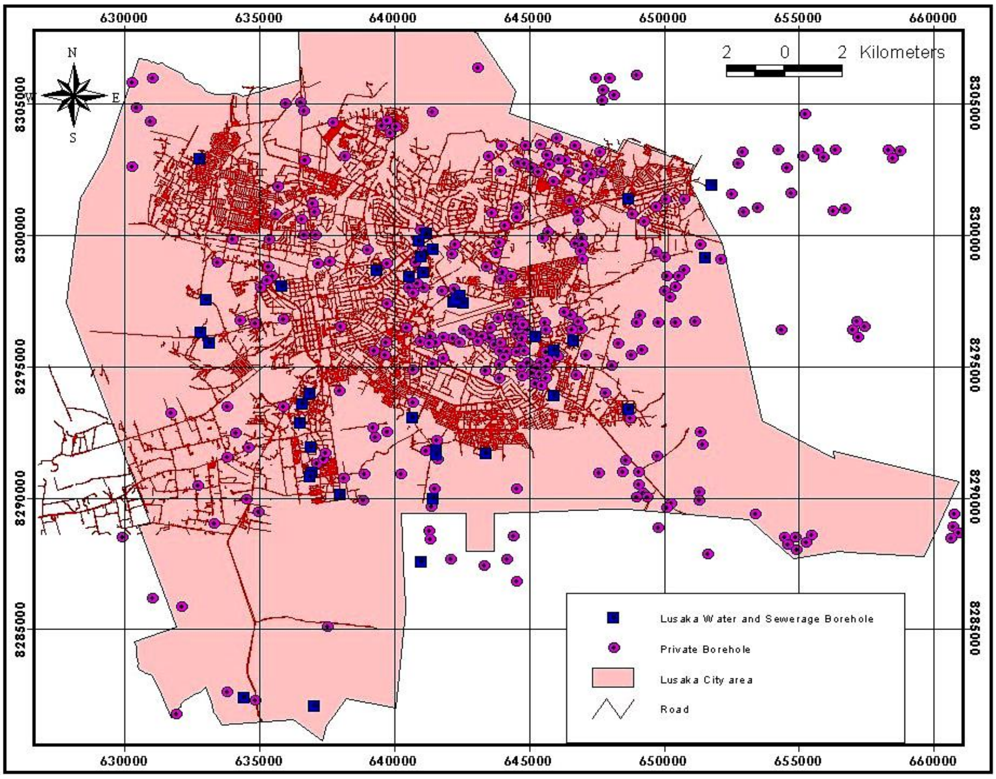
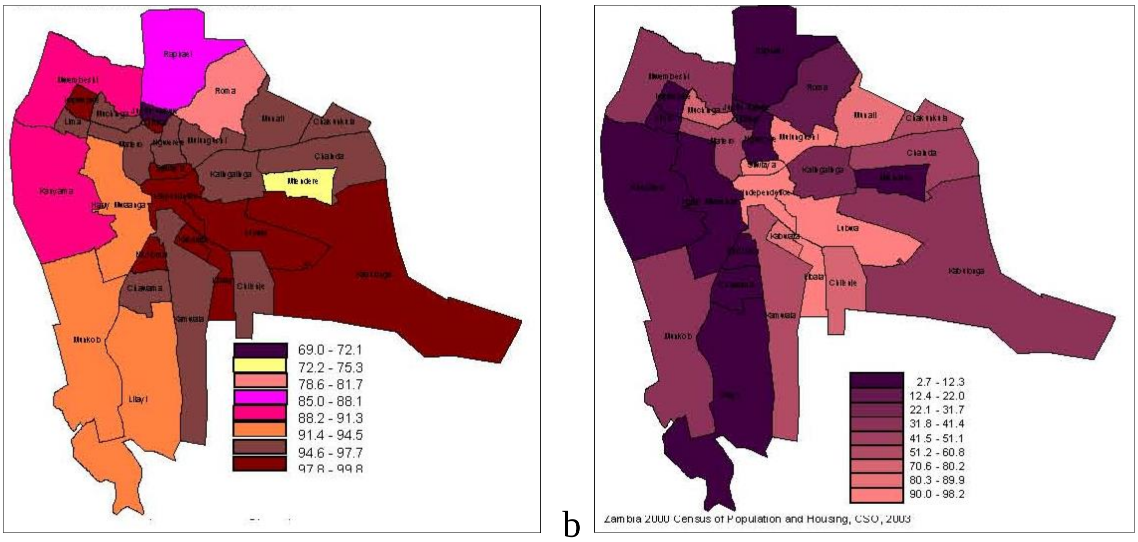


Figure 8.2 Distribution of the water supply authority and private boreholes in Lusaka in the mid-late 1990s (modified after Nkhuwa (2003))



a b

Figure 8.3 Households in the Lusaka urban area with access to: a) safe drinking water; b) sanitary means of excreta disposal (from UN Habitat, 2010)

8.3.1 Sanitation

The lack of a sewerage system means that sanitation provision in the peri-urban and up-coming low-density areas is generally left to the initiative of the residents, the majority of whom use on-site pit latrines and septic tanks, respectively, that they dig within their plot boundaries (von Münch and Mayumbelo, 2007). The pits are covered with soil once they are full. The liquid fraction of the excreta percolates into the ground and ultimately reaches the groundwater. To avoid these pit latrines filling up quickly, the practice has been to dig them quite deeply, 4-6 m, which results in less attenuation of the leachate to groundwater (Nkhuwa, 2003).

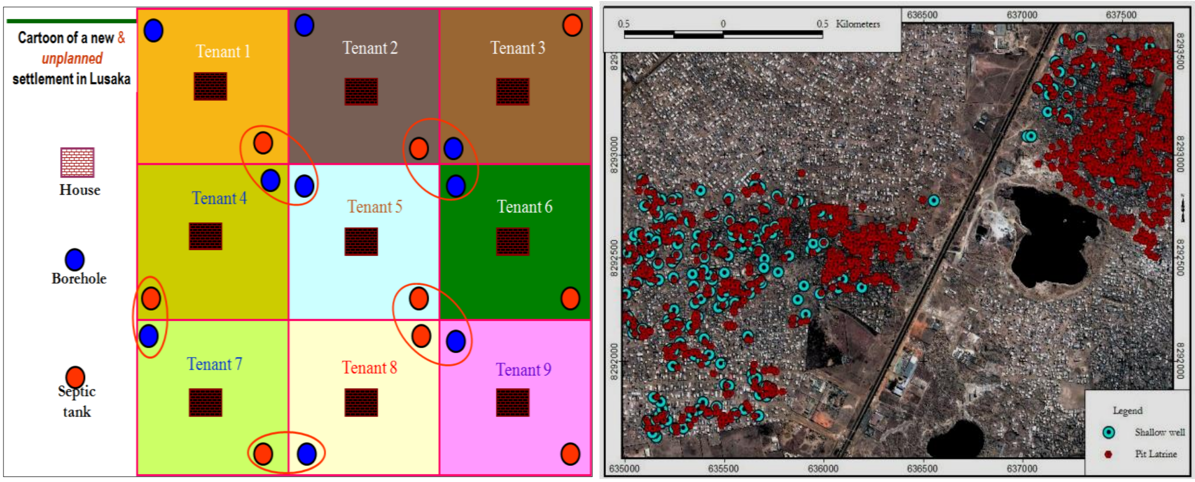


Figure 8.4 Proximity of water supply sources to on-site sanitation facilities in low- and high-density settlements of Lusaka. (Nkhuwa, unpublished (left) & 2006 (right))

There are some VIP latrines, but many are of rudimentary construction, unlined or are shared. People may also use dried up wells in the dry season (von Münch and Mayumbelo, 2007). Many latrines and septic tanks are typically close to water supply wells (Figure 8.3). The sanitation coverage (ratio of population with access to adequate sanitation) is quite low - only 17% (Bäumle et al., 2010). This poses great risk to the quality of groundwater.

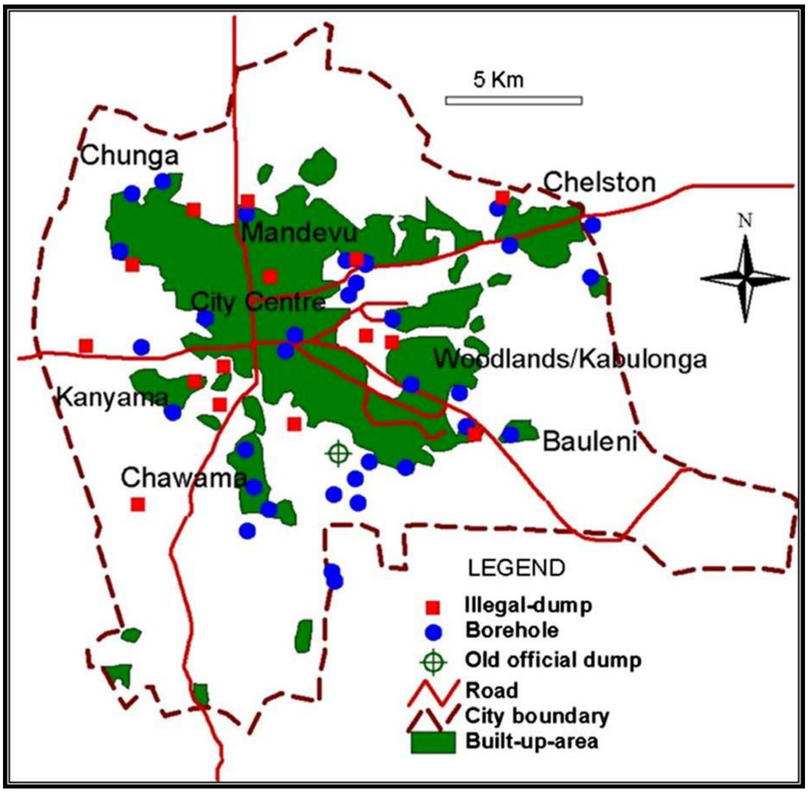


Figure 8.5 Locations of waste disposal sites and public supply boreholes in Lusaka (Nkhuwa, 2003)

8.4 OTHER SOURCES

Lusaka produces about 1.1 tonnes/day of solid waste, of which only about 10% is collected (De Waele et al., 2004). This is transported away from the city to landfill. The remainder of the solid wastes are often dumped between houses or dumped in uncontrolled sites on the dolomite, perhaps filling old quarries (Cidu et al., 2003). Waste dumped in these features consisted of, among others, old car bodies, oils and various forms of infectious/hazardous clinical wastes (Nkhuwa, 2003) with some medical waste also being disposed of inappropriately in refuse pits (Nkhuwa et al., 2008). UN Habitat (2010) assessed that solid waste disposal is very poor across Lusaka with the potential for the spread of disease.

8.5 GROUNDWATER QUALITY

Limited baseline data available suggest possible high fluoride in some parts of Lusaka (BGS, 2001). Bäumle et al. (2010) showed that there are still areas with water quality unaffected by human activity, in which SEC is less than 800 μS/cm and nitrate, chloride and sulphate below 10 mg/L. Groundwater was still considered acceptable for domestic use after treatment in 2004 (De Waele et al., 2004). Under the prevailing pH (median = 7.0, min = 5.8, max = 8.0) in the calcareous geological environment, potentially toxic heavy metals like lead, cadmium or arsenic as well as iron or manganese tend to form hydroxyl- and carbonate complexes which are insoluble and can therefore not be found in the water. Thus, concentrations of these cations are generally well below the WHO limit in groundwater. Nachiyunde et al. (2013) also assessed both nitrate and sodium to be widespread and prominent in high density residential areas. Table 8.1 gives a summary of the inorganic water quality in different parts of Lusaka undertaken by different researchers between 2003 and 2013.

In one large-scale groundwater study in 1978 (von Hoyer et al., 1978) there was bacteriological and chemical pollution in all the water samples tested. The study also revealed that both biological and chemical content of the water varied with season and borehole locality. Exceptionally high values were recorded during the rainy season and in areas that are reliant on pit latrines and septic tanks. Further, chemical and bacteriological investigations conducted in 1996 also revealed considerable and variable levels of pollution in the same areas (Nkhuwa, 1996). Isotopic studies conducted by Nkhuwa and Tembo (1998) indicated that the greatest susceptibility to pollution of the aquifer occurs mainly in the rainy season (November–April) when recharge to the groundwater store occurs. This study also showed that this process is especially exacerbated during periods of continuous and prolonged rainfall.

In the 1990s many boreholes had high concentrations of total and faecal coliforms. All data reported by (Nkhuwa, 2003) were positive and 28% had total coliforms TNTC (Table 8.2). In 2008 and 2010, only one third of sampled boreholes were below the Total Coliform limit of MPN=20 under the ZDWS, whereas elevated concentrations of Escherichia coli occurred much less frequently (Bäumle and Nick, 2011). Nitrate concentrations were found to be very high in many boreholes and often exceeded the Zambian Drinking Water Standards (ZDWS) limit of 44 mg/L. While the large production public supply boreholes of LWSC exhibit nitrate concentrations below the ZDWS, boreholes for the local supply of peri-urban high density settlement areas showed considerably higher values, with some exceeding 100 mg/L. Groundwater in the vicinity of medical waste disposal sites (Nkhuwa et al., 2008) showed variable levels and evidence of pollution dependent on clinic practices (Table 8.2).

Table 8.1 Inorganic water quality in Lusaka

Supply	NO_3 (mg/L) Mean (Range)	NH_4 (mg/L)	Cl (mg/L)	SEC (µS/cm)	Other (µg/L)	Reference
Buckley	13	0.08	52		Alkalinity 361	Nkhuwa (2003)
John Laing	39.5	59.7	102		211	
Kamanga	10.3	10.3	42.3		564	
Woodlands	0.03	0.9	90.1		124	
Wells and surface water, Lusaka	16.8 (0-43)	0.99 (<0.25->4)	14.7 (4.2-36)	570 (200-860)	Hg <0.4-13	Cidu et al. (2003); (De Waele and Follesa, 2003; De Waele et al., 2004)
Affected areas			123	1450		(Bäumle et al., 2010)
Lusaka province	19.1 (0-128)	3.09 (0.17-6.49)	19.2 (0.63-73)			Nachiyunde, K et al. (2013)

Table 8.2 Microbiological water quality in Lusaka

Supply/Area	SEC	COD	Total Coli (counts/100mL)	Faecal coli (counts/100mL)	Reference
Near medical clinics	669 (506-1060)	52 (48-64)	20 (0-58)	14 (0-45)	Nkhuwa et al. (2008)
Chawama	NA	58 (9-320)	19 (1-TNTC)	21, TNTC	Nkhuwa (2003)

8.6 AQUIFER MANAGEMENT

This has involved groundwater monitoring and sampling network, thematic mapping, development of groundwater information system, capacity building and awareness raising (Bäumle et al., 2010; Bäumle and Nick, 2011). An outline is shown in (Figure 8.6).

Kang'omba and Bäumle (2013) make the following recommendations for managing groundwater

- They assessed the current level of groundwater abstraction as sustainable but to safeguard supplies by continuing conjunctive use of surface water and to transfer some large abstraction to areas 5-30 km from the city.
- To improve sanitation, including biogas systems, and dry toilets, focussing first on highly vulnerable areas
- To protect groundwater by a zoned approach of land-use restrictions and to improves these zones using tracer tests.
- To make the city council responsible for water and sanitation and to provide resources for this

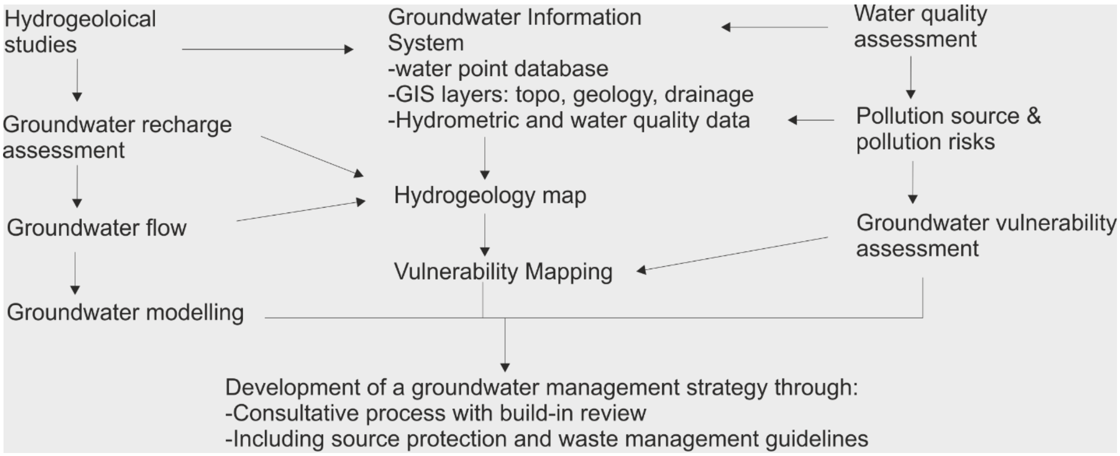


Figure 8.6 Simplified flowchart showing the investigation program towards the development of a groundwater management strategy for Lusaka (after BGR, 2013)

9 Ibadan case study

9.1 SETTING

With a total estimated population of 160 million Nigeria is the most populous nation in Africa with a large number of cities and smaller growing urban centres. Notable among these urban centres are Lagos, Kano and Ibadan which are also among the top hundred largest cities in the world. With the reality of poor landus/infrastructural planning, the attendant pressures on land and infrastructure translate into socio-and environmental problems in most urban centres in Nigeria.

This section is an appraisal of the environmental setting of Ibadan metropolis, south-western Nigeria, with particular reference to the impacts of urbanization on water resources. The overall intent is to assess the main anthropogenic drivers/sources of groundwater contamination related to the need for integrated urban planning, water resources and waste management.

A number of published and unpublished hydrochemical data were collated and assessed in respect of data quality protocols, including a database of 40 hydrochemical samples from Tijani and Onodera (2005), a database of 77 hydrochemical samples from NGSA (2010) and a data base of 57 hydrochemical samples from Tijani and Diop (2011). In addition, a database of 50 microbiological profiles and a number of other published and unpublished literatures were also incorporated.

9.1.1 Location and spatial extent of Ibadan

Ibadan metropolis, located in the south-western part of Nigeria, is the largest pre-colonial city in Nigeria, and Sub-Saharan Africa. It is 128 km northeast of Lagos and 530 km southwest of Abuja, the Federal capital, and is a prominent transit point between the coastal region of the south and the areas to the north in the extreme western portion of the country. The total land area of Ibadan metropolis is 3,123km^2 15% of which represents the urban old city centre while the remaining 85% represents the surrounding peri-urban areas. Administratively, Ibadan metropolis is made up of eleven (11) Local Government Areas (LGA) five (5) of which encompassed the core traditional areas of the city (i.e. Ibadan North, Ibadan North-East, Ibadan North-West, Ibadan South-West, Ibadan South-East) while the remaining six (6) constitute the surrounding peri-urban settlements (i.e. Akinyele, Egbeda, Ido, Lagelu, Ona-Ara and Oluyole) (see Figure 9.1).

Furthermore, the spatial analysis of spectral signatures of Landsat satellite imageries of 1972, 1984 and 2006 by Afolayan (2010) as presented in Table 9.1, revealed that the land-use of Ibadan has changed dramatically, reflecting an expanding (but essentially unplanned) development of urban areas. The rapid expansion of the coverage of urban and sub-urban areas from about 24% in 1972 to 66% of the land-use area in 1984 can be attributed to the transformation of the former rural and vegetated areas in the periphery of the city (such as Lalupon, Alakia, Olodo, Ogbere, Apata, Odo-Ona, Podo, Akanran, Bode-Igbo and Moniya) into the peri-urban entities to form the Ibadan metropolis.

Table 9.1 Dynamics of Urban Land-use Change in Ibadan metropolis

Category of Land Use (in %)	1972	1984	2006
Urban	5	14	15
Peri-Urban	19	52	53
Water	13	8	5
Rural / Vegetation	63	26	17

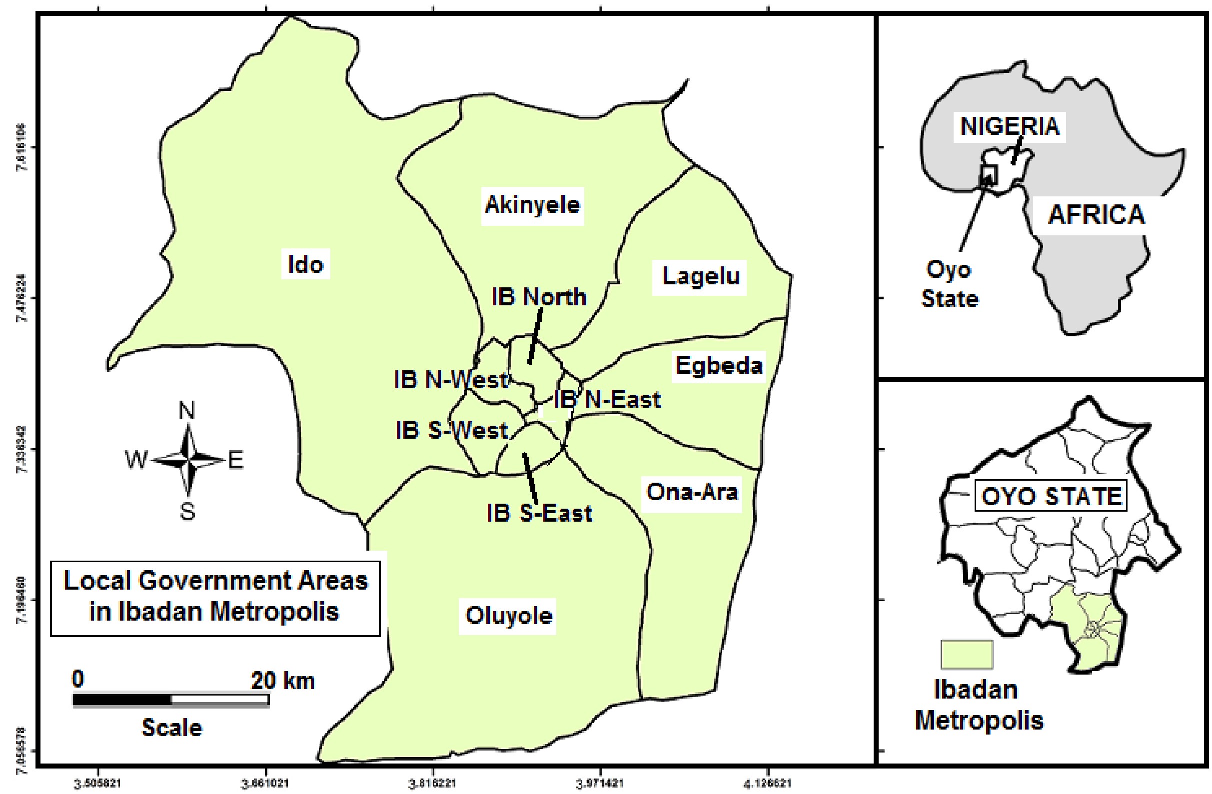


Figure 9.1 Location map of Ibadan Metropolis

The highlighted remarkable urbanization between 1972 and 1984 can be attributed to the oil boom era and the attendant rural to urban migration that prompted drastic utilization of land resources, especially in terms of building houses and erecting facilities for businesses (Ajayi and Abegunrin, 1994). The increase in population density of 586 person/km^2 in 1991 to 816 person/km^2 in 2006 is a consistent with the observed spatio-temporal changes in the land-use pattern of Ibadan metropolis. The increasing population of the metropolis and its suburbs as well as the outward residential mobility of people from the city to the suburbs are said to be the forces responsible for the merging together of the city with its former peri-urban zones to become the Ibadan metropolis (Udo, 1994).

9.1.2 Demography and socio-economic settings

Ibadan is the capital city of Oyo State, Nigeria. It is the largest metropolitan geographical area and is the third largest in Nigeria in terms of population, after Lagos and Kano, with a population of 2.5 million (Shemang, 1990). Ibadan was founded in 1829 and was initially occupied by immigrants, who moved into the city in search of security from intertribal wars (Wright and Burgess, 1992). Ibadan had since grown to be a big metropolis characterized by rapid remarkable growth with a population of 100,000 in 1851, to 175,000 in 1911 and 745,448 in 1952 during the colonial era. Subsequently, the population rose to 1,141,677 in 1963 at a growth rate of 3.95% per annum. In 1991 the population rose to 1.8, and then to 2.5 million in 2006 (Shemang, 1990). With a projected annual population growth of 4%, the population of Ibadan metropolis can be put at well over 3.3 million inhabitants today.

Udo (1994) reported that the population density of the urban area increased by 9.5% while that of the surrounding rural area increased by 100% within a period of 15years (1991-2006). A situation that can be attributed to the movement of population towards the peri-urban area, consequent to the rapid urbanization of the core area and industrialization of the peri-urban centre where lands are readily available for industrial development. The economic activities of Ibadan include commerce, handicrafts, small- and medium-scaled manufacturing and service industries as well as agriculture mainly in the surrounding peri-urban areas. A number of the farming activities are on part-time basis aimed at augmenting earnings from other jobs while the predominant crop production in Ibadan is staples such as cassava, maize and vegetables such as spinach, okra, tomatoes and pepper. Due to its strategic location in southwestern Nigeria, Ibadan is an important commercial centre with market square or stalls at virtually every street and corner in the traditional core area and the inner suburbs of the city. Within the city there are many daily neighbourhood markets and a number of periodic (8^{th}-day or 3^{rd}-day) markets.

The challenges of the demographic and social settings are summarised by (Onibokun and Kumuyi, 1996) and attributable to:

- Unplanned growth and inefficiently managed land use planning over the years
- Poorly managed waste disposal system, poorly constructed dumpsites and drainage systems
- Grossly inadequate public utilities and social infrastructural services such as poor and inadequate housing/sanitation, poor/inadequate water supply and attendant environmental contamination problems

9.1.3 Environmental Setting

The traditional core neighbourhood, as the oldest part of the city, is a high density area occupied mainly by residents of the city and centred on the famous Mapo Hill with characteristic rusty red roofs and unplanned buildings (Figure 9.2). There are hardly any gaps between the buildings, a situation causing serious ventilation and accessibility problems among others. Where roads are available, these are narrow and usually without drainage gutters and walk ways, thus constituting challenges to household waste and sewage collection, with attendant environmental implications.

Furthermore, like many urban centres in developing countries, poor land-use planning, lack of adequate water supply, lack of proper sewage, and waste disposal systems also characterize the Ibadan metropolis. Consequently, many households, especially within the congested central portion of the city lack toilet and waste disposal facilities while most rely on in-house hand-dug (shallow) wells for their domestic water supplies. As a result, direct discharge of sewage water and dumping of domestic wastes/refuse into the drainage channels are common practices (Tijani and Onodera, 2005). Since late 1970s, many of the municipal facilities have declined to almost zero functionality. Top on the list of those municipal services that have seemed to fail most strikingly is waste collection and disposal followed by the public water supply system, the coverage of which had not been extended beyond the pre-1970 limit. The service is frequently inadequate, resulting in refuse generated remaining uncollected, and with large parts of the drainage systems blocked by refuse dumps (Figure 9.3a and b).

Figure 9.2 Views of central Ibadan highlighting the unplanned urban scenario

Figure 9.3 Images of waste dumps and blockage of drainage systems (a & b) and unprotected household dug-wells (c).

Due to improper management of wastes (through the use of household pit toilets and bathrooms), contaminants from human excreta and urine seep into shallow weathered basement aquifers and thus contaminate many households' hand-dug wells that are not well protected (e.g. Figure 9.3c). Poorly managed waste disposal systems, as a result of poorly constructed dumpsites and drainage systems as well as waste and sewage management, remains one of the most pressing and unresolved issues that portends negative impacts in terms of groundwater contamination.

9.1.3.1 SANITATION AND WASTE MANAGEMENT

The overcrowded and unplanned nature of the city centre is characterized by the lack of adequate basic environmental infrastructural services such as water supply, proper sanitation, solid waste disposal sites, good drainage and good roads. Inadequate collection and disposal of household/municipal wastes is a typical problem. It has been reported that most municipal governments spend 20 – 50 % of their available operational budgets for solid waste services and around half of the urban households benefit from collection services (Udo, 1994).

In 1998, the Oyo State government, in its continuous efforts to rid Ibadan of indiscriminate dumping of wastes on the road side and other undesignated areas, commissioned four waste dumping sites (Awotan, Ajakanga, Amuloko and Lapite) which are currently active. Nonetheless, most municipal wastes that are collected, in the Ibadan metropolis, end up in open dumps site or open-air incineration sites. A significant portion of household waste also ends up in the drainage systems through the uncontrolled disposal of both solid and liquid wastes into open drains and along roads sides (Figure 9.4). This poses a threat to both surface water and groundwater quality, and flooding as well as serious health hazards in form of waterborne diseases that are common in the city, particularly in the older traditional indigenous areas. Such situations forces the inhabitants to spend appreciable portions of their low income and time on improving their personal health, with adverse consequences for general economic well-being (Onibokun, 1989).

Figure 9.4 Images of waste dumps on road-sides (a), blockage of storm drains (b) and a poorly managed public toilet (c).

In addition, Akintola and Agbola (1989) projected the amounts of liquid waste for 1990 and 1995 at 113.7 million and 126.5 million litres, respectively. A number of industries (chemical, paper and poultry among others) make private arrangement for the disposal of their waste, without adequate monitoring to ensure proper environmental safeguards. Also, according to the National Population Commission (Shemang, 1990), only 18.47% of households in Oyo State have a water closet linked to a reticulated system, 32.73% use pit latrines and 37.13% use open defecation. (Awosika, 2008).

Consequently, the lack of the most basic waste management services in overcrowded low income neighbourhoods is said to be a major contributor to the high morbidity and mortality among the urban poor (Udo, 1994). For example, mortality rate of children (under the age of five) through diarrheal disease is high (about 50%), largely as a result of poor sanitation, contaminated drinking water and associated problems of food hygiene (World Bank 1993), as is the case in many other countries in SSA.

9.1.3.2 WATER SUPPLY

Water supply still poses a serious problem in both the urban and peri-urban parts of Ibadan metropolis. The Ibadan metropolis accounted for nearly two thirds of the total domestic water supplies in the whole Oyo State since mid-1980s. Consequently, Oyo State in recent years has embarked on a programme of rehabilitation of the major waterworks serving the Ibadan metropolis. These include Asejire and Eleyele waterworks as well as Osegere water scheme through a loan from the African Development Bank (ADB). The impacts of these projects has not been felt by the wider population in Ibadan, due to the fact that the water supply networks are only supplying 7% of the households, mainly in the central area of the city (Ajayi and Abegunrin, 1994).

The total output of 200 Ml/d (i.e. Eleyele with 27ml/d, Osegere with 13Ml/d and Asejire Phases I&II with 80Ml/d each) was envisaged following the ADB assisted upgrade. However, this is dwarfed by the projected water demand of >600 Ml/d for Ibadan metropolis by the beginning of the 21st century (Udo, 1994). Therefore, sources of water for a large proportion of the Ibadan metropolis still remain the streams, springs, ponds, dug-well, shallow boreholes and harvested rainwater as highlighted in Table 9.2.

Table 9.2 Access to water used for drinking in Ibadan Metropolis.

S/No.	Sources of Water Supply	No. of Households	Percentage
1	Pipe-borne Inside Dwelling	34,348	2.75%
2	Pipe-borne Outside Dwelling	50,912	4.41%
3	Tanker Supply/ Water Vendor	28,833	2.31%
4	Borehole	85,895	6.88%
5	Hand-dug Well	695,720	55.74%
6	Rain Water	103,800	8.32%
7	River/ Stream/ Spring	204,891	16.41%
8	Pond/ Dugout /Lake	10,063	0.81%
9	Others	33,733	2.70%
	TOTAL	1,248,105	100%

Data source: Udo (1994)

In summary, the problems and challenges posed by the rapid population growth of Ibadan are immense. Prominent among these are the inadequate infrastructure including potable water supplies and waste management. Environmental degradation is widespread, including contamination of drainage channels and shallow groundwater systems, due poor waste disposal and management systems.

9.1.3.3 PHYSICAL GEOGRAPHY AND DRAINAGE

The main physiographic features of the city is characterized by undulating terrain which consist of quartzite ridges and inselbergs of gneiss that run approximately in northwest – southeast direction. These ridges are surrounded by the adjoining plains and valleys while the ridges are characterized by peaks such as Mapo, Mokola and Aremo in the central part of the city with elevation range of 160 to 275 metres above sea level.

In terms of hydrology, the metropolitan area of Ibadan is drained by two main rivers, the Ogunpa River and its tributaries drain the eastern parts of the metropolis, while the Ona River and its tributaries drain the western parts of the metropolis. The drainage systems are dendritic, characterized by unmodified stream channels that run in a southerly direction through much of the Ibadan metropolis (Figure 9.5).

River flows are irregular during the dry season, and the considerable influence of population pressure have resulted in built-up areas encroaching river banks. Consequently, direct discharge of household waste, as well as refuse dumps at various points along stream channels, are common. These constitute obstructions to flow and hence pose a constant danger of flooding during the peak of the rainy season within the metropolis, a situation that resulted in catastrophic flooding event in August 1980 and lately the devastating flood of 25th August 2011.

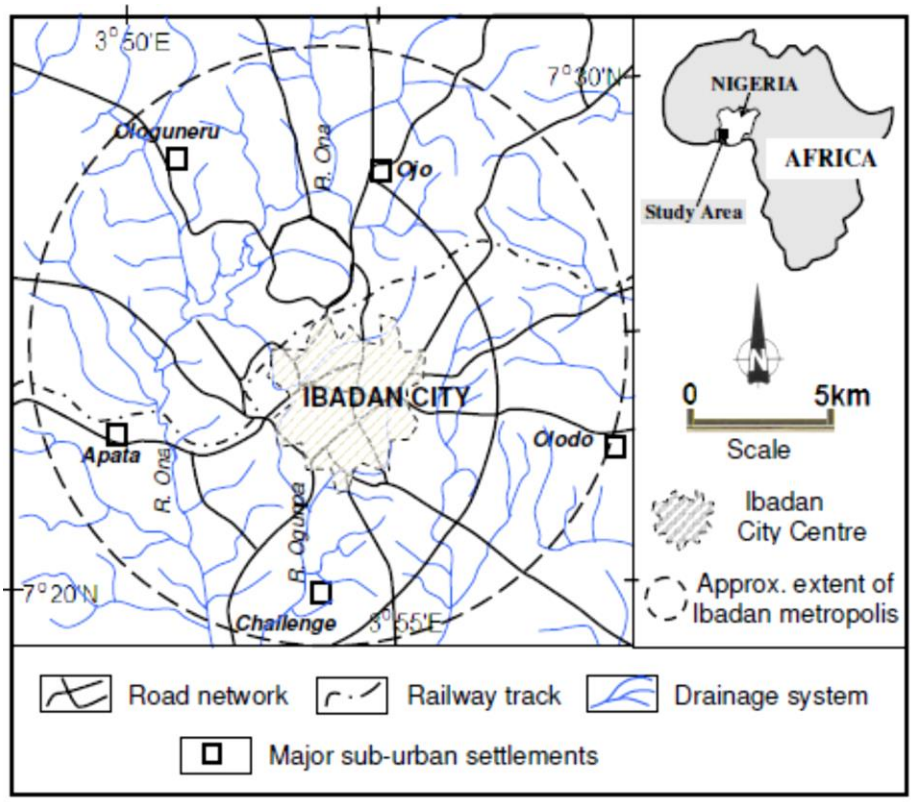


Figure 9.5 Drainage Map of Ibadan Metropolis

9.1.3.4 CLIMATE, VEGETATION AND SOILS

The Ibadan metropolis and its environs are characterized by tropical humid climates with two distinct seasons: the wet season, which occurs between March and October with an average annual rainfall of about 1250 mm and dry season from November to February (Ileoje, 1987). During the wet season, the area is under the influence of the moist maritime south-west monsoon winds from Gulf of Guinea, while the dry season is characterized by dry dust-laden and northeast–southwest trade winds from the Sahara desert. The distribution of rainfall is bimodal with the two peak regimes i.e. May-June with average monthly rainfall of 190mm and September-October with average monthly rainfall of 170 mm, separated by a period of lower precipitation in August (Ileoje, 1987).

The mean annual temperature is 26 ^{o}C with a minimum of 21 ^{o}C while the average temperature for rainy and dry seasons is 27^{o}C and 30^{o}C respectively. Humidity is relatively high for most parts of the year (except during the peak of the dry season) with annual average greater than 80 percent.

The natural vegetation was moist deciduous tropical rainforest with thick undergrowth (Eduvie, 2006). However, due to population pressure and the attendant human impact, most of the surrounding peri-urban forest vegetation has been replaced by secondary forests and derived savanna for field crops such as maize, yams and cassava. There exists two cropping seasons in the area as a result of the bimodal rainfall pattern observed. The first is from late April to early August while the second and shorter one is between early September and November. The dominant field crop production system is that of rotational bush fallow with a long fallow period of 8-10 years (in the past) which has been reduced to 2-4 years due shortage of farmland and intensification of the farming system due to the rapid population growth.

As expected the soils in and around Ibadan metropolis were formed from the underlying crystalline basement rock units (Awosika, 2008) under moist semi-deciduous forest cover (Hopkins, 1965). The soils were mapped and classified into four soil associations and series by Smyth and Montgomery (1962). These are (i) Iwo, (ii) Okemesi, (iii) Egbeda and (iv) Mamu soil associations. The classification, according to Awosika (2008) is largely based on soil parent materials and the soil associations represent the weathered soil/ mottled soil layer (pedolith) over the main bedrock types in Ibadan metropolis.

The soils of the Iwo association were formed from the granite gneisses, those of Okemesi and Egbeda from quartzite and schist while those of Mamu were formed from the pegmatites. Due to the relative resistance to weathering of the schist quartzites, the associated Okemesi soil group is usually shallower; it contains a high proportion of coarse sands, and particularly those that occur in the upper and middle slope portions of the catena. The soils belonging to the Iwo and Mamu association derived from granite gneiss and pegmatite are fairly clayey with a characteristics humus layer, especially over the granite gneiss. The Iwo and Egbeda soil association are the most extensive in the eastern part of the region while those of the Mamu association occur mainly in the south of Ibadan city.

Generally, the soils have low nutrient-holding capacities as reflected in their general low cation exchange capacities that usually vary between 5.0 and 12.0 mili-equivalent per 100 grams of dry soil in the top 20cm of the profile, due to prominently kaolinitic nature of the associated clay minerals (Moorman and Van De Wetering, 1985)

9.2 GEOLOGY AND HYDROGEOLOGY

Geologically, Ibadan metropolis and environs are underlain by Precambrian basement rocks, which comprise of crystalline igneous and metamorphic rocks mostly quartzite and quartz-schist of meta-sedimentary series and the migmatites complex comprising of banded gneiss, augen gneisses, granite-gneiss, and variably migmatized biotite-hornblende gneiss with intruded pegmatites, quartz veins, aplites and dolerite dykes (Burke et al., 1976) (Figure 9.6).

In general, the minerals in the rock types are quartz, biotite, muscovite, iron oxide and plagioclase feldspar in various percentages. The rocks have undergone various episodes of tectonism depicted by foliation with characteristic alternating light and dark coloured bands (Jones and Hockney, 1964) while major lineament trends are commonly in NNE-SSW or N-S directions. In fact, most rock types stated above are covered in most places by weathered regoliths, but outcrop in few places.

Weathering provides avenues for water percolation and forms weathered regolith aquifers. These are generally discontinuous with groundwater occurring in localised disconnected (Tijani and Diop, 2011) regolith aquifers under unconfined to semi-confined conditions while trough geophysical measurements. Olayinka et al. (1999) showed that the regolith can be up to 60 m thick.

9.2.1 Hydrogeologic setting and hydraulic characteristics

The hydrogeology of an area depends on the bedrock geology, structure and climate of the area while the underlying geological formation structures determine the types of aquifer as well as the nature of recharge (Jones and Thornton, 2003; Lewis, 1987). However, the climatic and hydrologic situations do determine the amount and rate of recharge of the aquifer (Olayinka and Yaramanci, 1999). From a hydrogeologic point of view, availability of groundwater in crystalline basement rock settings, like Ibadan metropolis, depends on the mineralogy of the rocks, which in turn dictates the degree of fracturing and weathering. Hence, significant aquifer units can develop within the weathered overburden and deeper fractured bedrock zones. In these zones, the groundwater resources depend on the depth of the water level and the relative thickness of the weathered horizon. Usually, deeper (thicker) weathering profiles are an indirect indication of good groundwater potentials. Nonetheless, due to the complex interactions of the various factors affecting weathering, water-bearing horizons may not be present at all at some locations.

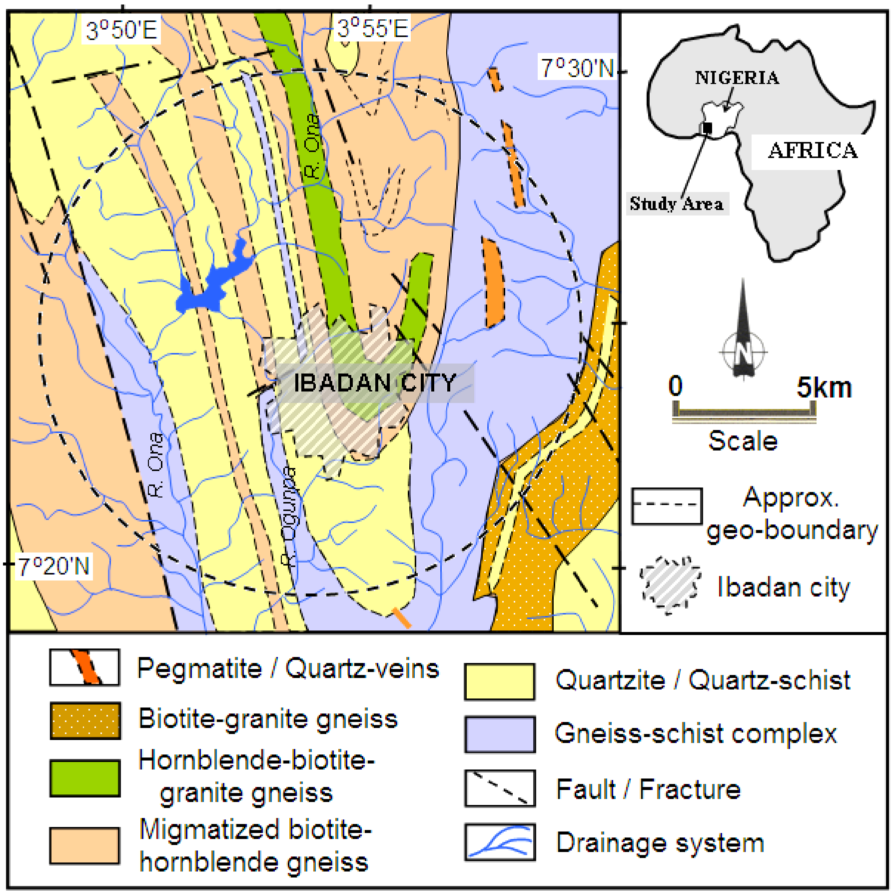


Figure 9.6 Geological Map of Ibadan metropolis and environs

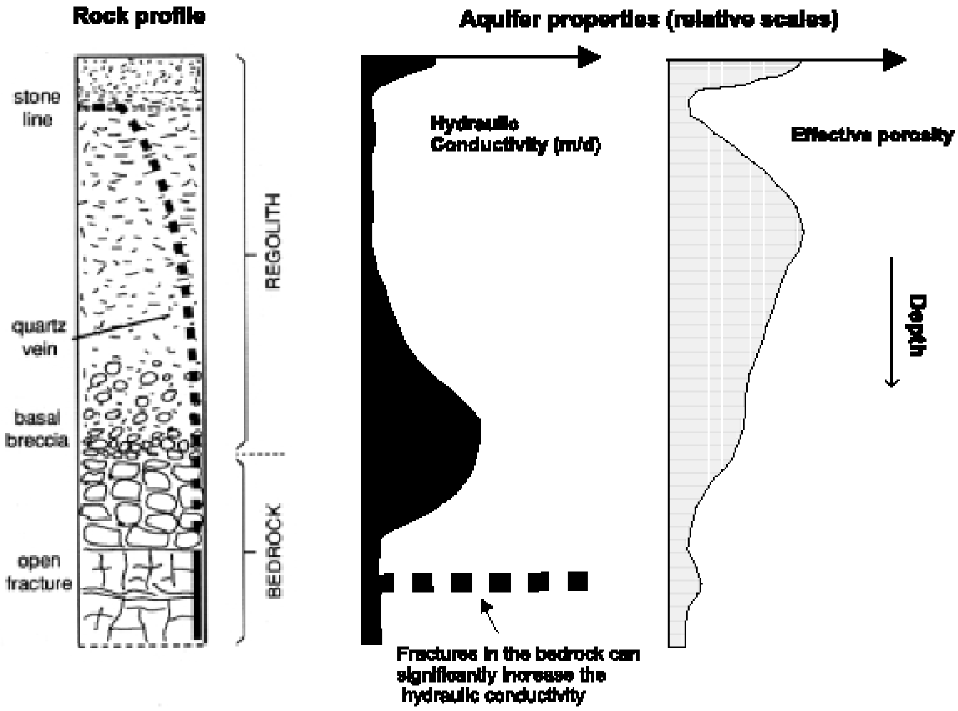


Figure 9.7 Porosity and permeability variations with depth in the weathered basement (source: Chilton and Foster (1995)

In Ibadan the tropical humid setting does enhance active weathering with resulting thick and generally well developed weathered profiles. Consequently, the weathered zone can be as much as 60m thick, but more commonly in the range of 20 – 30m. Below this zone the rock becomes progressively less weathered and more consolidated until fresh fractured bedrock is reached (Figure 9.7). In such profiles, it has been observed that porosity generally decreases with depth while permeability, however, has a more complicated relationship, depending on the extent of fracturing and the clay content of the weathered overburden (Chilton and Foster, 1995). Usually beneath the soil zone, the rock is often highly weathered and clay rich, therefore permeability is low. The saprock zone representing the base of the weathered zone, near the fresh rock interface consists of fractured rock, and is often permeable allowing free movement of water (Chilton and Foster, 1995). Wells or boreholes that penetrate this horizon can usually provide sufficient water to sustain a hand pump. Deeper fractures within the basement rocks are also an important source of groundwater, particularly where the weathered zone is thin or absent. These deep fractures are tectonically controlled and can sometimes provide appreciable water supplies (Laniyan et al., 2013).

9.2.1.1 WEATHERED BASEMENT AQUIFERS OF IBADAN

As mentioned earlier, Ibadan is underlain by Precambrian basement rocks, which comprise of crystalline igneous and metamorphic rocks mostly granite-gneiss, quartz-schist, augen-gniess, pegmatite intrusions and variably migmatized biotite-hornblende gneiss. Assessment of previous studies revealed varied weathered profiles over the different bedrock units within Ibadan, suggesting influence of rock types and mineralogy on the extent of weathering and in essence the occurrence of groundwater. A schematic presentation of such profiles for granite-gneiss, pegmatite and quartz-schist settings (Piper, 1944) indicate thickness variability of weathered basement rocks (Figure 9.8).

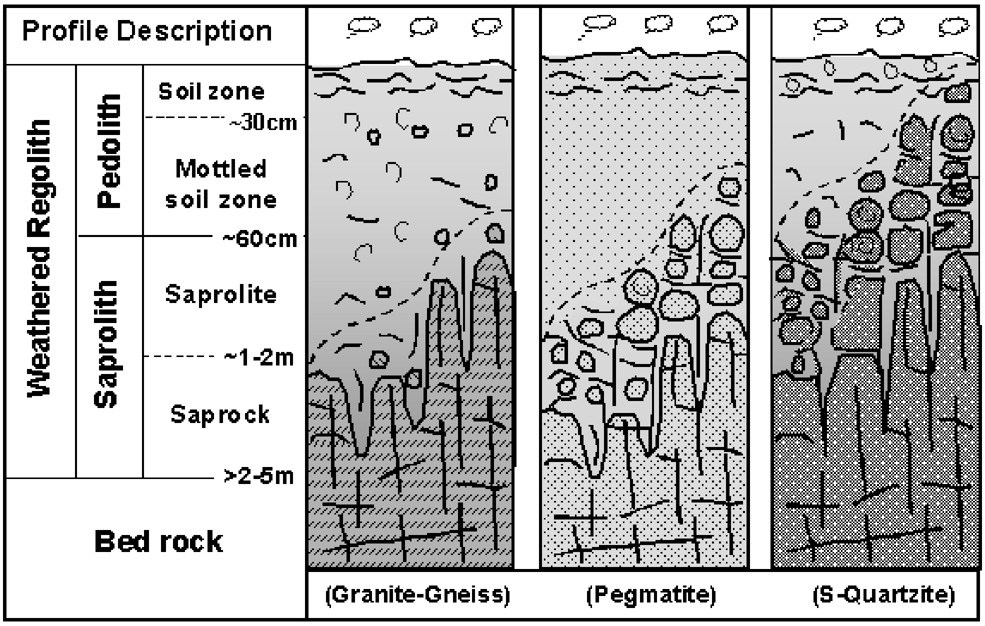


Figure 9.8 Weathering profile over different bedrock units within Ibadan metropolis

For Ibadan, studies have revealed relatively poor aquifers with low groundwater potential, high groundwater yield are said to be found areas where thick overburden overlies fracture zones (Moormann and Greenland, 1980). Recent studies indicated that yields from wells in areas underlain by quartzite and quartz-schist are much higher than areas with rock types such as: granites, migmatites and gneisses. This is due to the higher occurrences of fissures in the former rock types with enhanced transmissivities and permeabilities in contrast to the areas underlain by the latter rock types (Aweto, 1994; Tijani et al., 2010).

Two aquifer units have been identified, the weathered overburden unit which usually sustain shallow hand-dug wells (5–15mbgl) and fractured saprock units that sustain shallow boreholes (30–60mbgl) (Aweto, 1994). More detailed evaluation of hydraulic data from pumping tests of a number of boreholes within Ibadan metropolis were undertaken by Aweto (1994) and Tijani, et al., 2010. As presented in Table 9.3, the depth of wells within the different geologic settings range from 12.12 – 68m (mean 47.7m) for augen gneiss, 21.6 – 60m (mean 36.2m) for banded gneiss and 60 – 87m (mean 66.4m) for schistose quartzite setting (Table 9.3). This variability in well depths signifies the geologic control and differences in the extent of weathering. Nonetheless, the thicknesses are within the range that can be anticipated for shallow boreholes in a basement terrain aquifer (Tijani, 1994; Uma and Kehinde, 1994; Edet and Okereke, 2005).

Depths to water level in the boreholes vary widely from <1m to more than 10m but generally less than 15m below ground surface in the different geologic settings with quartz-schist bedrock exhibiting relatively deeper water level. This facilitates the development of hand-dug wells and shallow boreholes for domestic water supply. Saturated thickness in the boreholes shows a wide variation of 8.02m (in gneiss) to 81.58m (in the quartz schist setting).

The measured yields of the tested boreholes revealed relatively low values between 30m^3/d – 138m^3/d, and mean 80m^3/d (Table 9.3) with higher yields in the augen gneiss (range 30-138.2 m^3/d; mean 82.87m^3/d). Banded gneiss sustained yields between 31-88.6 m^3/d; mean 55.9 m^3/d) while quartz-schist aquifer have yield of 62 – 86.4m^3/d. High yields typify boreholes that intersect fractured bedrock with shallow weathered overburden, while low yield boreholes are associated with thick weathered regolith and no prominent fractured intersection (Tijani et al., 2010).

Table 9.3 Summary of borehole inventory and evaluated hydraulic properties of the basement aquifer in Ibadan metropolis

Parameters	Banded gneiss (N = 6)			Augen gneiss (N = 13)			Quartzite (N = 5)		
	Min.	Max.	Mean	Min.	Max.	Mean	Min.	Max.	Mean
Well depth (m)	21.6	60.0	36.2	12.1	68.0	47.7	60.0	87.0	66.4
Depth to WL (m)	0.94	5.9	3.4	0.75	11.2	4.67	3.07	11.3	6.1
Drawdown (m)	2.2	26.4	13.7	3.3	29.4	11.3	15.1	34.7	28.9
Sat. thickness(m)	16.4	54.9	30.7	8.02	62.8	42.2	48.7	81.6	62.7
Yield (m^3/d)	31.0	88.6	55.9	30.0	138.2	88.7	62.8	86.4	75.4
Sp. Capacity (m^2/d)	2.92	7.9	5.6	1.25	37.8	13.0	1.86	17.5	5.23
Trans. (m^2/d)	1.10	4.16	2.8	0.76	27.2	7.25	0.41	10.6	2.71
Sensitivity, Cc	0.24	2.0	0.7	0.25	0.7	0.43	0.17	0.42	0.28
Hydr. Cond.(m/d)	0.07	0.15	0.1	0.02	0.5	0.2	0.01	0.18	0.05

Source: Alichi (2006)

In addition, the evaluated transmissivity T values for the three rock types ranged from 1.1 – 4.16m^2/d (mean 2.79m^2/d) for banded gneiss, 0.76 – 27.16m^2/d (mean 7.25m^2/d) for augen gneiss, and 0.41 – 10.55m^2/d (mean 2.71m^2/d) for the quartzitic rock. These imply that quartz-schist and banded gneiss settings show lower potential when compared to the augen gneiss. Nonetheless, the majority of the boreholes (47%) revealed values ranging between 1.0 and 5m^2/d which are said to be typical of the weathered basement aquifers (Offodile, 1983) while 29% exhibit transmissivity in excess 5m^2/d which is indicative transmissive fractured zones. This variation is a clear indication of variability and localized nature of the basement aquifer as also dictated by bedrock and the degree of weathering.

The hydraulic conductivity was estimated from evaluated transmissivity and saturated thickness of the aquifer with values of 0.01 – 0.18m/d for banded gneiss and quartz-schist compared to 0.2-0.5m/d for augen gneiss (Table 9.3). These values cut across the various ranges of likely hydraulic conductivity for weathered granite and metamorphic rocks (Halford and Kuniansky, 2002) and are indicative of regolith aquifers with generally low permeability. Specific capacity generally, gives a better indication of aquifer performance than yield since it also reflects aquifer transmissivity and saturated thickness (Mace, 2000; Uma and Kehinde, 1994). Specific capacity of the tested boreholes range from 1.3 – 37.8m^2/d; mean 13.8m^2/d (Table 9.3) for augen gneiss. Banded gneiss and quartz-schist have specific capacity of 2.9 – 7.9 m2/d (mean 5.6 m^2/d) and 1.86 – 17 m^2/d (mean 5.2 m^2/d) respectively.

9.3 ASSESSMENT OF GROUNDWATER QUALITY DEGRADATION

In the following sections, selected hydrochemical and water quality studies are reviewed and summarized for the purpose of highlighting the groundwater quality scenarios in Ibadan.

9.3.1 Microbiological contamination and hydrochemistry

Adetunji and Odetokun (2011) assessed the impact of septic tanks on the bacterial quality of dug wells. All the wells sampled had high total coliform counts (2.29±0.67 log cfu/mL). There were no significant differences in the bacterial counts between covered and uncovered wells. The mean distance (8.93±3.61m) of wells from the septic tanks was below the limit (15.24 m or 50 ft) set by the United State Environmental Protection Agency (USEPA). Ifabiyi (2008)investigated relationships between water quality and well depth, and Amidu and Olayinka (2006) looked at impact of a septic tank plume.

Oloruntoba and Sridhar (2007) assessed the bacteriological quality of drinking water from wells, spring, borehole, and tap sources together with that stored in containers by urban households in Ibadan during the wet and dry seasons. Results showed that majority of households relied on wells, which were found to be the most contaminated of all the sources. At the household level, water quality significantly deteriorated after collection and storage as a result of poor handling. Furthermore, there was significant seasonal variation in E. coli count at source (P=0.013) and household (P=0.001). A summary of inorganic water quality studies carried out across Ibadan are presented in Table 9.5.

Table 9.4 Microbiological water quality in Ibadan

Supply	SEC/TDS	Total Coli (counts/100mL)	Faecal coli (counts/100mL)	Reference
Dug wells SE LGA	TDS 174	Max=5,120		Ochieng et al. (2011)
Boreholes residential areas		73%	E coli 18%	Abiola (2010)
Dug wells across city	SEC 561	Total heterotrophs 600 (20-3500)	26 (1-200)	Tijani and Diop (2011)
Dug wells		TC 2.29±0.67 log cfu/mL		Adetunji and Odetokun (2011)

Table 9.5 Inorganic water quality in Ibadan

Supply	NO_3 (mg/L) Mean (Range)	NH_4 (mg/L)	Cl (mg/L)	SEC (µS/cm)	Alkalinity and other mg/L and (µg/L)	Reference
Dug wells across city	30.1 (3-178)		39.5 (3-128)	561 (85-1572)	HCO3 134 (15-532) mg/L	Tijani and Diop (2011)
Boreholes residential areas					Pb 4.9 µg/L	Abiola (2010)
Dug wells SE LGA			5	TDS 174	Alkalinity 48.2 mg/L	Ochieng et al. (2011)
Dug wells SE LGA	8.01		136		Alkalinity 108 mg/L Fe 0.56 mg/L	Ifabiyi (2008)
Orita-Aperin waste site	16.3	1.67	68.9	TDS 349	Fe 4.6 mg/L Cr 0.07	Ikem et al. (2002)
Waste disposal sites					Cd 1	Adelekan and Alawode (2011)
Auto-mechanic villages					Cu 4.55-9.02	Adelekan and Abegunde (2011)
Lead battery factory, Wofun	4.31	0.32		0.7	Fe 0.57 Cu 1.22 Ni 0.52 Pb 1.05 Cd 0.003 all mg/L	Dawodu and Ipeaiyeda (2007)

Sangodoyin and Agbawhe (1992) monitored slaughter house pollution in adjacent wells. Slaughter house waste generally has a high COD and TDS. Groundwater approximately 250 m from the abattoir site was found to be unsatisfactory as a raw water source for drinking purposes.

Results from a study characterising the hydrochemistry of shallow groundwater compared to surface water from Ibadan metropolis by Tijani and Onodera (2005) are summarized in Table 9.6. The major ion chemistry in groundwater samples are within the WHO limits with the exception of NO_3, the main critical quality index, and associated slight elevated Na and Cl concentrations. Nitrate concentrations range from 17.2 to 412 mg/L (average 112.3 mg/L) indicate contamination of the shallow groundwater system. Nonetheless, a closer look at the nitrate concentrations reveal that all the sampled dug-wells have NO_3 concentrations of >15 mg/L, while about 50% of them have NO_3 concentrations of >50 mg/L above the WHO recommended limits.

Dug-wells with depths of <5 m have variable EC ranging from 100 to 2000 µS cm^{-1}, while those with depth of >5 m (are characterized by EC less than 1,000 µS cm^{-1} and revealed a general tendency of decreasing EC with increasing depth. This is consistent with the fact that the deeper wells (>5 m) have NO_3 concentrations of less than 50 mg/L, compared with shallow wells (<5 m) with variable NO_3 concentrations of 20 to 410 mg/L. Field observations revealed that NO_3 contamination is related to inputs from domestic wastewaters and leachates from household septic tanks and pit latrines. Hence it can be concluded that shallow wells (<5m deep) are characterized by high TDS and high NO_3 concentrations suggesting the impacts of the infiltrating contaminants through the loose weathered regolith materials. However, the relatively deeper wells (>5–14 m deep) are characterized by low TDS and low NO_3, indicating fresh groundwater from fractured saprock units, relatively free from infiltrating pollutants from the upper loose weathered regolith unit.

In addition, hydrochemical assessment of 50 shallow groundwater samples from Ibadan metropolis as presented by Tijani and Diop (2011) and summarized in Table 9.7. This study revealed that most of the major ionic parameters are within the acceptable limits of WHO (1994) and the Nigeria's regulatory standards (NAFDAC) for drinking water quality. Ca and Na dominate major cations with average concentrations of 31.2 mg/L and 20.2 mg/L respectively while the major anions are HCO_3 and Cl with average concentrations of 133.8mg/l and 39.5 mg/L respectively.

Table 9.6 Summary of the hydrochemical data for surface and groundwater systems Tijani and Onodera (2005).

Parameter	Groundwater (N=40)			*WHO Standard	Surface water (N=40)			MWR+
	Range	Mean	SD§		Range	Mean	SD	
DWD (m)	3.2–14.1	6.2	2.8	–	–	–	–	–
Temp. °C	27.0–30.5	28.7	0.80	Variable	25.7–37.4	30.5	2.5	Variable
pH	5.9–8.3	7.4	0.62	6.5–9.5	5.9–8.9	7.4	0.9	Variable
EC (µS cm^{-1})	105–1679	586.7	405.6	400–1480	164–1878	821.0	433.9	Variable
TDS	66–1063	373.5	265.4	500–1000	103–1188	516.7	277.9	90.0
Ca	0.8–132.0	23.9	30.2	75–200	2.0–72.6	30.9	18.7	15.0
Mg	0.8–41.1	15.0	12.3	50–150	2.0–31.1	14.7	7.0	4.1
Na	6.2–204.4	41.2	44.3	20–200	17.8–383.4	92.1	89.4	6.3
K	0.5–86.4	16.4	18.8	10–12	7.3–178.5	41.9	42.0	2.3
Fe	0.01–4.4	0.37	0.84	0.3–1.0	0.03–23.9	1.8	3.8	0.5
HCO_3	34.0–100.0	65.3	13.5	Variable	38.0–118.0	67.1	17.7	58.0
Cl	21.0–84.0	49.2	18.0	250–600	25.0–150.0	74.8	33.2	7.8
SO_4	13.0–45.0	29.9	8.53	250–400	10.0–49.0	29.3	12.0	11.0
NO_3	17.2–412.0	112.3	112.0	25–50	22.8–366.0	104.3	95.1	1.0

§ SD = Standard deviation. *WHO Standard, 1993. DWD = Shallow dug-well depth in metres.
MWR+ = Mean world river (from Hem, 1985: Martin & Maybeck, 1979), values in mg/L, unless otherwise stated

The measured depth range of 2.1 – 15.5m (av. 8.1 m) for the sampled dug-wells alongside the measured TDS of 54 – 1,055 mg/L (av. 373 mg/L) are clear indication of low mineralized shallow groundwater system with limited circulatory history typical of a weathered crystalline basement setting. This is consistent with the finding of Tijani and Onodera (2005) as well as other similar studies elsewhere (Tijani, 1994, Tijani and Abimbola, 2003).

Nonetheless, a confirmation of the anthropogenic impact on the shallow groundwater system as highlighted earlier is reflected by the microbial contamination of most of the water samples with total heterotrophic bacteria count (TBC) in the range of 20 – 3,500 (CFU/100ml). Also the presence of coliform bacteria (*E. Coli*) in some of the water samples (1-200 MPN/ml), most of which are also incidentally characterised by high NO_3 concentrations and point to impacts of human waste inputs from in-house septic/soak-away pits.

Further indication of water contamination is also revealed by the evaluation of chemical analyses results of both water and stream sediment samples at 40 locations undertaken along the stream channels of Ona-Ogunpa-Ogbere drainage system that drains major parts of Ibadan metropolis. From the summary of results (see Table 9.6), the TDS for the surface water system in Ibadan metropolis is generally low with average value of 516.7 mg/L. This is in line with results from catchments underlain by similar, low solubility Precambrian Basement Complex rocks.

However, isolated locations within the stretches of the sampled drainage network with TDS >700 mg/L also found to have higher NO_3 concentration ranging between 80 and 366 mg/L. This can be clearly attributed to the discharge of untreated domestic/municipal sewage water as well as refuse dumps into the drainage channels.

Table 9.7 Summary of the hydrochemical data for groundwater system in Ibadan city Tijani and Diop (2011).

Parameter	Min.	Max.	Mean	Median	WHO* Standard	NAFDAC Standard
	(N=50)					
Temp °C	25.5	29.8	27.8	27.7	Variable	
pH	5.8	7.9	6.8	6.7	6.5-9.5	6.5–8.5
EC (µS/cm)	85.3	1,572	561.6	465.2	480-1480	1000
TDS mg/l	54.4	1,055	372.8	302.1	1000	500
TH	13.8	437.7	153. 9	129.2	100-500	
Ca	4.4	80.6	31.2	28.0	75-200	75
Mg	0.88	65.2	18.3	13.5	50-150	200
Na	0.23	107.3	20.3	12.1	20-200	200
K	0.3	19.0	7.1	6.7		
Fe	bdl	0.28	0.05	0.02	0.3-1.0	3.0
HCO_3	15.3	532.0	133.8	109.0	Variable	100
Cl^-	3.2	128.2	39.5	20.1	200-600	100
SO_4	bdl	69.1	21.4	16.1	200-400	100
NO_3	3.3	177.5	30.2	20.9	25-50	10.0
TBC	20	3500	600.2	300		
CC	10	200	25.7	5.0		

* TH= Total Hardness (mg/$CaCO_3$); CC= Total coliform counts (MPN/mL) and TBC= Total bacterial counts (CFU/100mL), values in mg/L, unless otherwise stated

A plot of the NO_3 and EC along the stretches of the drainage (Figure 9.10) also confirmed the urban anthropogenic influence on the surface water quality as the NO_3 peaks coincide with those of EC within the stretches of the urban centres. In addition, the NO_3 concentrations are considerably above 100 mg/L in the populated urban section of the metropolis compared to the suburb or peri-urban areas located at the upstream and downstream sections (see Figure 9.9). The field observation revealed that the spatial disposition of the NO_3 and EC peaks coincided with sections of the stream channels that receive direct discharge of domestic wastewaters and/or refuse dumps. This also further confirmation of the influence of untreated household/municipal effluents on the urban drainage networks within the populated old city centre and a reflection of poor land-use and waste management services in the Ibadan metropolis.

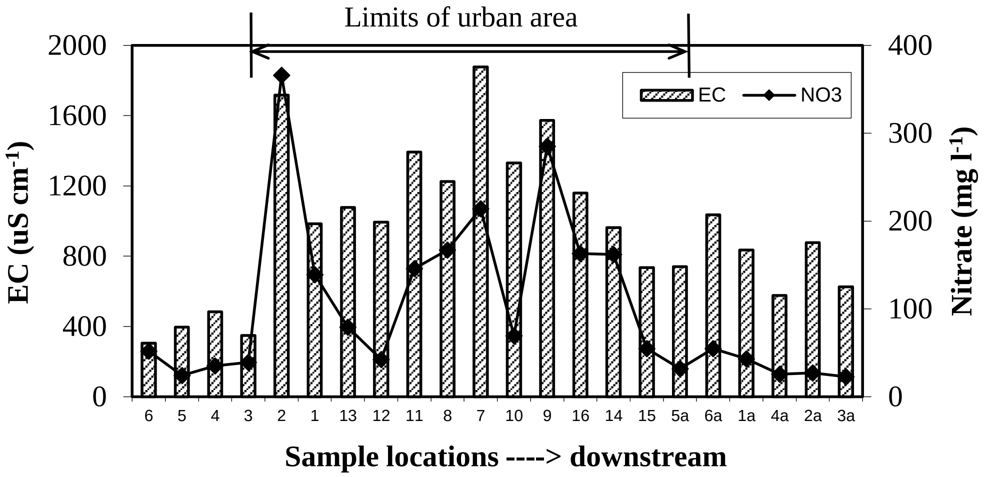


Figure 9.9 Profiles of EC and NO_3 of the water in the drainage system within the urban stretches of Ibadan metropolis

Table 9.8 Summary of the hydrochemical data for groundwater system in Ibadan city (source: NGSA 2006).

Parameters	Min.	Max.	Mean	Median	WHO Std.	NAFDAC Std.
	(N=77)					
Temp °C	24.8	29.6	26.3	25.8	Variable	
pH	4.8	8.0	6.8	7.0	6.5–9.5	6.5–8.5
EC(μS/cm)	102.3	3321.0	490.9	343.0	900–1200	1000
TDS	53.2	676.0	228.6	170.7	500–1500	500
Ca^{2+}	6.85	89.04	32.56	27.40	75	75
Mg^{2+}	2.08	104.00	22.21	16.64	50-150	200
Na^{+}	0.90	40.40	23.97	26.20	200–250	200
K^{+}	0.30	93.60	30.58	29.10	200	
HCO_3^-	1.15	590.03	162.42	157.41	100–500	100
Cl^-	2.00	262.00	66.15	50.00	200–250	100
SO_4^-	3.00	210.00	56.88	43.00	250–500	100
NO_3^-	0.00	4.40	0.47	0.03	10–50	10.0
TC	2.00	98.00	28.6	20.00		
TTC	1.00	82.00	7.85	4.00		

TTC= Thermo-tolerant coliform counts (MPN/mL) and TC= Total coliforms counts (CFU/100mL). Values in mg/L unless otherwise stated.

Further assessments of major ion chemistry and groundwater quality of Ibadan metropolis were undertaken late 2006 by Nigerian Geological Survey Agency (NGSA) including a hydrochemical database of 70 observations, these are summarised in Table 9.6. Like other studies, the major cations revealed similar concentration trends with average of 32.6 mg/L and 22.2 mg/L for Ca^{2+}

and Mg^{2+} respectively, while Na^+ and K^+ ions revealed average concentrations of 24 mg/L and 30.6 mg/L respectively. Bicarbonate dominated the anions with an average concentration of 162.4 mg/L followed by Cl^- with an average value of 66.2 mg/L while SO_4^{2-} and NO_3 have average values of 56.9 mg/L and 0.50 mg/L respectively.

While the concentrations of Cl ions can be attributed to the occasional disinfection of the some of the dug-wells with chlorinated products, the generally low NO_3 concentrations (0.01- 4.4 mg/L), compared to the other studies, is perhaps due to the timing of the sampling during the peak of the dry season. The implication is that the low NO_3 concentration can be attributed to natural attenuation by de-nitrification process within the weathered regolith, as well as lack of recent rain-induced recharge and vertical leaching of the household waste from latrines and soak-away pits into the underlying groundwater system during the dry season. Nonetheless, the total bacterial count of 2-98 CFU/100ml and coliform count of 1-82 MPN/mL (See Table 9.8) is also a clear indication that the impacts of poor household waste disposal and management on the groundwater systems in Ibadan metropolis.

9.3.2 Trace metal contamination

To assess the trace metals contamination within Ibadan metropolis, the studies of Tijani et. al., (2004) and Tijani and Onodera (2005) with respect to contamination of selected trace metals in the groundwater and drainage systems within Ibadan were also evaluated and summarized in Table 9.7 and 9.8. As presented in Table 9.7, for the groundwater system, trace metals such as Cu, Pb, Zn and Cd are either below the detection limits and/or occur at low concentrations below the recommended WHO limits for drinking water standards. However, As and Hg are reported in detectable concentration with average values of 1.89 and 0.38 mg/L, respectively, compared to the recommended WHO limits of 0.01 mg/L and 0.001 mg/L, respectively, an indication of impacts of untreated cosmetics-loaded household waste waters.

Table 9.9 Summary of the trace metal concentrations in groundwater system in Ibadan city

Para-meters	Groundwater (N=40)			*WHO Standard	Surface water (N=40)			MWR+
	Range	Mean	SD§		Range	Mean	SD	
As	0.1–3.8	1.89	0.87	0.01	0.20–3.30	1.70	0.79	0.001
Cu	0.001–0.02	0.008	0.005	2.0	0.001–0.03	0.010	0.006	0.003
Pb	0.01–0.23	0.01	0.06	0.01	0.010–0.58	0.089	0.110	0.003
Hg	0.1–0.8	0.32	0.23	0.001	0.20–0.40	0.300	0.100	0.001

Zn and Cd are below the detection limits; hence are not reported here. § SD = Standard deviation.
*WHO Standard, 1993. MWR+ = Mean world river (from Hem, 1985: Martin & Maybeck, 1979).
Values in mg/L unless otherwise stated.

Like the groundwater system, only As and Hg are in elevated concentrations, with average values of 1.7 mg/L and 0.35 mg/L, respectively, in the surface waters of the urban drainage systems in Ibadan metropolis (see Table 9.7). The apparently low concentrations of most of the analysed trace trace/heavy metals in the surface water were attributed to possible preferential partitioning into the sediment phase of the drainage system. This assertion was supported by the relatively higher concentrations of trace metals in the stream sediments (see Table 9.8) compared to the stream and surface waters.

Table 9.10 Summary of trace metal concentrations in the stream sediments and environmental quality indices

Metals	Stream sediments (N=40)					AF_{gw}	AF_{sw}
	Min.	Max.	Mean	*EF*	$R_{tot/ads}$		
As	0.30	6.80	3.22	2.22	–	188.9	1695
Cd	0.06	0.27	0.12	0.44	–	–	–
Cu	3.10	44.60	9.20	–0.36	10.0	0.01	3.2
Pb	2.50	702.	36.46	0.35	27.9	9.8	29.7
Zn	55.7	115.6	31.64	0.53	5.7	–	–
Hg	2.0	11.20	5.75	70.9	–	316.7	300
Fe	56.6	7598	35667	0.49	1519	1.23	6.0

EF, Enrichment factor; $R_{tot/ads}$, Ratio of total to adsorbed metal concentration;
AF_{gw}, Anthropogenic factor for groundwater system;
AF_{sw}, Anthropogenic factor for surface water system. Values in mg/L unless otherwise stated.

The concentration of trace/heavy metals in the stream sediments are about 5–10 orders of magnitude higher than those measured in the water phase of the drainage system within Ibadan metropolis. Fe, Cu, Pb and Zn are found to be the most abundant in the analysed stream sediments with concentrations of 3.1–44.6 mg/L Cu (average 9.7 mg/L), 2.5–702.5 mg/L Pb (average 36.5 mg/L) and 5.7–115.6 mg/L Zn (average 31.6 mg/L), respectively. However, Hg, As and Cd are in relatively lower concentration, with average values of 5.8 mg/L, 3.2 mg/L and 0.12 mg/L, respectively. Nonetheless, it should be noted that the above concentration trends do not reflect the respective degree of contamination; rather the metal contamination index will depend on the reference threshold (background) values (Tijani et al., 2004). By and large, the variability of concentrations of these metals within the stretches of drainage channels (like NO_3 in the water column) suggests local anthropogenic input sources through domestic and municipal sewage effluents at various points along the drainage channel.

Therefore, further assessments using contamination indices with respect to environmental bioavailability of the trace metals revealed that the ratio of the total to the adsorbed concentration (Rtot/ads) are 0.24 (Zn), 0.17 (Cu) and 0.09 (Pb) (see Table 9.8). These was interpreted to mean that about 30% of Zn, 20% of Cu and 12% of Pb are in adsorbed form as bio-environmental available portion which can be released back into the water phase in response to changes in the physico-chemical conditions. Hence, it can be concluded that the relatively low concentrations of the trace metals (Cu, Pb, Zn, Cd, As and Hg) in the water column compared to the stream sediments are indications of partition between the water and sediment phase, while the proportions of adsorbed concentration in the stream sediments are potential contamination sources for the water column (Tijani, et al., 2004; Tijani and Onodera, 2005).

Further assessment of the quality status and level of trace metal contaminations in water and sediment samples of the urban drainages in Ibadan metropolis was also undertaken by Tijani and Onodera (2005) using enrichment factor (EF) and anthropogenic factors (AF) (see Table 9.8). Based on the assessment and for the groundwater system, the estimated AF values with respect to the WHO standards, are generally <1.0 for Cd, Cu and Zn, implying no contamination or anthropogenic inputs, while for As, Hg, Pb and Fe the estimated AF are considerably >1.0, suggesting enrichment or contamination above the recommended WHO limits. However, for the surface water system, with the exception of Cd and Zn which are below detection limits, other trace/heavy metals (Cu, Pb, As and Hg) have AF values of >1.0 suggesting contamination or enrichment above the level of mean world river (MWR) and WHO limits. Hence it can be concluded that despite the absolute low concentrations of the analysed trace metals in both surface and groundwater system, there is evidence of slight enrichments of As, Hg, Pb, Cu and Fe relative to the WHO and MWR reference standards.

For the stream sediments, most of the analysed trace metals have enrichment factor (EF) of <1.0, except for As and Hg with values of 2.3 and 70.9, respectively, with respect to the corresponding background values of the granitic crystalline bed rock units in Ibadan metropolis. This is also a reflection of the contamination of As and Hg associated with the stream waters, as mentioned earlier. While the source of As was attributed to anthropogenic activities and dumping of wastes/refuse into the stream channels, the sources of Hg (as a constituent of medicated soaps and cosmetics materials) was attributed to inputs from discharge of untreated domestic/household waste waters into the stream channels.

9.3.3 Hydrochemical Characterisation of the Groundwater System

Natural water chemistry is said to be dependent on a number of processes including precipitation, mineral weathering and evaporation-crystallization (Didymus, 2012). However, effects of these controlling processes on different chemical species vary as other processes such as cation exchange, anthropogenic contamination and mixing process may also exert some considerable influence on the groundwater chemistry. Hence, the prevailing chemical character of any groundwater system is usually not only a function of the character of recharging water but also a function of the interaction with the aquifer system during subsurface flow (Tijani and Abimbola, 2003). As part of the review of the groundwater contamination in Ibadan metropolis, hydrochemical characterization of the chemical data base using Piper trilinear diagrams were employed as presented in Figure 9.10a and b. In general, there are three main water types namely;

a) Ca-Mg-HCO_3, at times with significant component of alkali metals as Ca-Mg-(Na)-HCO_3
b) Ca-Mg-(Na)-SO_4, with occasional significant chloride component as Ca-Mg-(Na)-Cl and
c) Na-(K)-HCO_3 water types.

This is an indication of the fact that the natural chemical composition of the groundwater is influenced by rock–water interaction mostly controlled by CO_2-charged mediated weathering and dissolution of silicate minerals as represented by Ca-(Mg)-HCO_3 waters, while ion exchange processes in represented by Na-(K)-HCO_3 waters.

The overall assessment highlights the geogenic and anthropogenic controls on shallow groundwater system in the study area. The evolution of such water types can be related to the weathering and alteration of calcium- and magnesium- rich minerals within the bedrock units mediated by CO_2-charged infiltrated rain waters. This observation is also supported by Gibbs diagram (Figure 9.11) indicating the dominance of weathering – dissolution process in respect of the hydrochemical evolution of groundwater shallow basement crystalline aquifer of Ibadan metropolis.

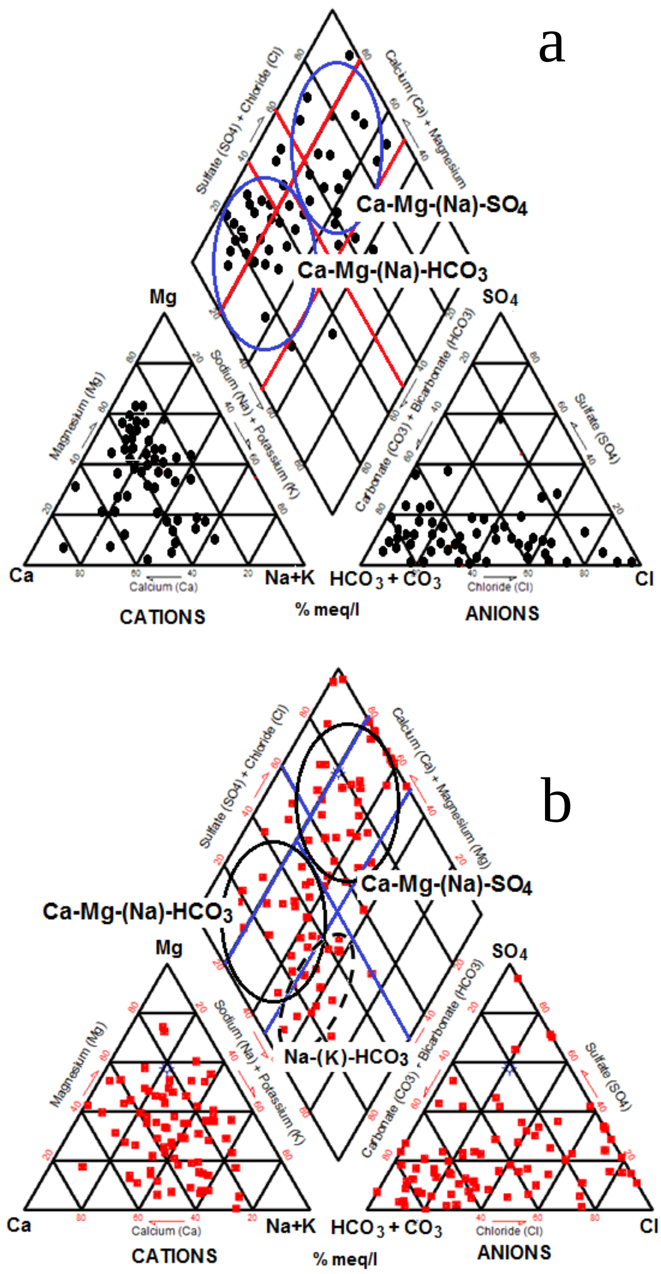


Figure 9.10 a) Piper diagram plot of the different hydrochemical facies for groundwater system in Ibadan metropolis (Source: Tijani and Onodera (2005); Tijani and Diop (2011)). b) Piper diagram plot of the different hydrochemical facies for groundwater system in Ibadan metropolis (Source: Unpublished NGSA Report, 2006)

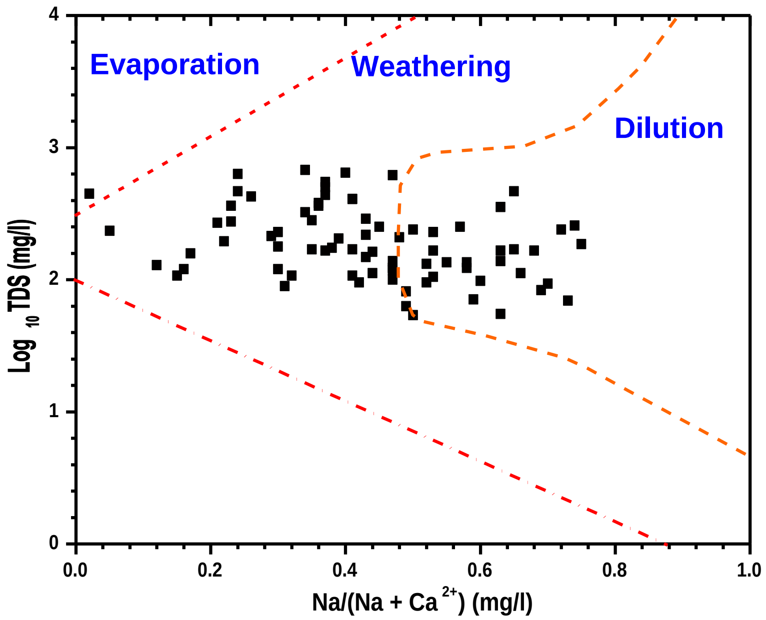


Figure 9.11 Gibbs diagram showing evolution of groundwater in Ibadan metropolis

Factor analysis has been commonly used in a number of hydrogeochemical studies (Briz-Kishore and Murali, 1992; Gupta and Subramanian, 1998; Lambrakis et al., 2004; Subbarao et al., 1996). The R-mode factor analysis was employed to investigate the factors controlling the hydrochemical characteristics of the groundwater systems in Ibadan metropolis. Such analysis provides features that allow interpretation of large data sets (Jayakumar and Siraz, 1997) and allows the identification of variables that are characterized by a similar process using the factor loadings and eigenvalues. As part of further evaluation of the hydrochemistry, the database (n=70) was subject to analysis using principal component analysis (PCA). PCA was performed on the groundwater data in order to better understand their interrelationships, probable source of the major ions and to explore the reduction of the experimental variables. From the results of the factor analysis, three (3) main factors with eigenvalue >1 and all accounting for about 76% of the data matrix were extracted and presented in Table 9.9.

Table 9.11 Variable loading within each factor component/group

PCA (Eigenvalue)	*pH*	*EC*	*Ca*	*Mg*	*Na*	*K*	*Fe*	*HCO_3*	*SO_4*	*Cl*	*NO_3*	*TDI*
F-1 (5.56)	<u>0.41</u>	0.30	<u>0.78</u>	<u>0.78</u>	0.28	0.40	0.16	<u>0.92</u>	<u>0.66</u>	0.19	-0.01	<u>0.86</u>
F-2 (2.35)	<u>0.53</u>	<u>0.83</u>	0.35	0.06	0.31	0.33	<u>0.80</u>	0.16	0.17	0.32	<u>0.77</u>	<u>0.43</u>
F-3 (1.82)	-0.55	0.23	0.20	<u>0.41</u>	<u>0.79</u>	<u>0.60</u>	0.25	-0.10	0.37	<u>0.72</u>	0.26	0.18

The first factor with eigenvalue of >5 which represented about 46% of the data matrix was interpreted as the dominant factor controlling the groundwater chemistry. Furthermore, within each of the factor groups, only chemical variables with loading of at least 0.4 were considered significant members of the respective factor group (see Table 9.9). Further interpretations in respect of each factor are summarized below:

- Factor 1: dissolution-weathering factor which is loaded in respect of pH, Ca, Mg, HCO_3, SO_4 and TDI. This association strongly represents geogenic influence of the bedrock geology on groundwater which is a reflection of the observed Ca-(Mg)-HCO_3 type.

- Factor 2: anthropogenic leaching factor (loaded in respect of NO_3, EC and TDS) which is a reflection contamination through household waste pits and
- Factor 3: Alkali-exchange factor (loaded in favour of Na, K, Mg and Cl) as a reflection of the observed Na-(K)-HCO_3 exchange water type.

9.3.4 Synthesis of Groundwater Contamination Sources

Like many urban centres in developing countries, the environmental setting of Ibadan metropolis is characterized by poor land-use planning, lack of adequate water supply, lack of proper sewage, and waste disposal systems. Consequently, many households, especially within the congested central portion of the city lack toilet and waste disposal facilities while most rely on in-house hand-dug (shallow) wells for their domestic water supplies. As a result, direct discharge of sewage water and dumping of domestic wastes/refuse into the drainage channels are common practices. In summary, the likely causative factors responsible for deterioration of groundwater quality in the study area are improper disposal of waste, increased anthropogenic activities arising from population explosion in the city and limited planning and inadequate infrastructure. Based on the foregoing hydrochemical assessments and evaluations, a conceptual framework of contamination sources and mechanisms with respect to groundwater quality degradation or contamination in Ibadan metropolis are graphically summarised in Figure 9.12.

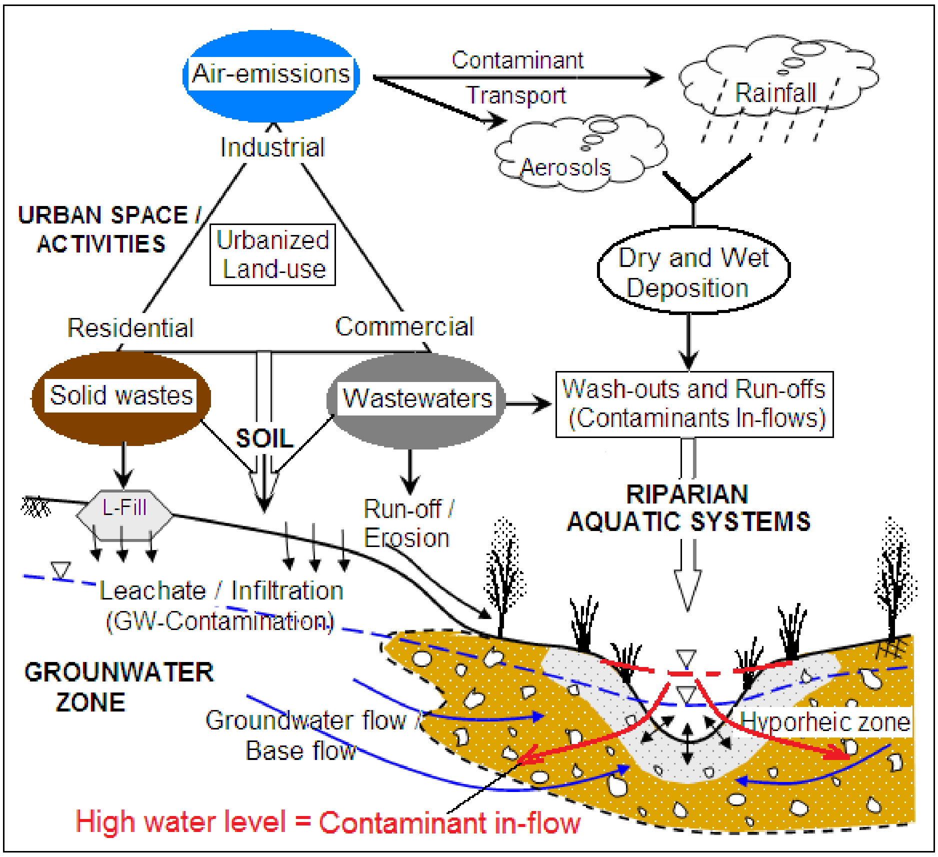


Figure 9.12 Conceptual framework of groundwater quality degradation sources and mechanisms in the Ibadan metropolis

The main drivers of the groundwater quality degradation come from the urban land use dominated by residential commercial and industrial activities (Figure 9.13). These activities lead to generation of solid and liquid wastes, and poor waste management leads directly to high environmental inputs of chemical and microbiological contaminants. The environmental consequence in terms of quality degradation of groundwater quality through direct infiltration of wastes and leachates on one hand and direct discharge into the riparian aquatic systems are presented. This has obvious heath implications for users of shallow wells for drinking water and domestic purposes, and also has implications for groundwater dependant ecosystems and surface water bodies.

9.4 SUMMARY

Major and trace element hydrochemistry of shallow groundwater, surface water and stream sediments from an urbanised drainage catchment, with respect to anthropogenic activities, were evaluated and discussed as a case study of surface and groundwater contamination in a typical growing African urban setting with basement geology. From this study, it is evident that urbanisation coupled with lack of proper waste disposal system has considerable influence on the water quality of urban surface and groundwater bodies with obvious health and environmental implications.

Groundwater samples are within the permissible limits of WHO and NAFDAC standards, with the exception of NO_3, the chemical characterisation reveal a geogenic control on the occurrence of most of the major cations through weathering-induced water-rock interactions characterized by Ca-Mg-(Na)- HCO_3 water type with subordinate Na-(Ca)-HCO_3 exchange water type. Overall the hydrochemical evaluation indicated weathering, dissolution and dilution as the principal mechanism responsible for the hydrochemical evolution of groundwater in the study area. PCA analysis identified an additional anthropogenic factor with high loadings (>0.7) for NO_3, Fe and SEC.

Urban surface and shallow groundwaters were typically contaminated with high NO_3. The NO_3 contamination in the shallow groundwater can be attributed to leaching from household septic tanks/soak-aways and rubbish pits. The observed total coliform count (TC) and presence of TTC counts above WHO standards, alongside elevated NO_3 concentrations, is a clear indication of anthropogenic controls on the groundwater quality. This is a consequence of poor and indiscriminate disposal of household waste waters and unsanitary conditions of latrines and soak-away pits within the populated areas of Ibadan metropolis.

The study shows that the concentrations of all the analysed heavy metals Pb, Hg and As in the surface water are slightly higher and suggest contamination compared to the mean composition of world rivers, while the respective concentrations of Pb, Cu, Hg and As in sediment phase revealed an overall enrichment and contamination relative to the background value in the granitic bedrock units.

Evaluation of bioavailability and partitioning of the trace metals revealed that, about 30% of Zn, 20% of Cu, 12% of Pb and <1% of Fe were in adsorbed form compared to the respective total metal concentrations. The implication is that those adsorbed portions are potential contamination sources for the surface water due to possible remobilization and release into the water phase in response to possible changes in the physico-chemical condition of the drainage system.

Finally, there is the need to recognise that the urban development issues facing Ibadan metropolis, and other cities in Nigeria, have and important social and health implications (Udo, 1994) as well as long term implications for surface water and groundwater dependant ecosystems. There is the need for adequate financing of urban infrastructures and the institutional arrangement for delivery of urban services, especially drinking water supplies, solid waste and waste water managements. To date the majority of studies have focussed on major ion chemistry and microbiological parameters. There is a knowledge gap regarding the extent of contamination and potential human and environmental exposure to specific pathogens, trace elements and organic contaminants in urban settings.

10 Factors for risk mapping

10.1 PRELIMINARY ASSESSMENT FROM LITERATURE REVIEW

From each of the chapters the key hazard factors have been extracted to inform the risk mapping process. These have been grouped into source-pathway-receptor categories (Table 10.1) with other local area factors such as population density. From the literature review a preliminary assessment of the importance of the various hazard factors can be made and these are shown as primary and secondary factors in the table.

10.2 FACTORS TO BE INFORMED BY LOCAL CONDITIONS

Whilst Table 10.1 sets out the key factors it does not always prioritise them or attempt to score them. The assessment can be informed by use of a risk assessment matrix (Figure 10.1). For a contaminant or a suite of contaminants, the various hazard factors are summed along the x-axis and a measure of the contamination along the y-axis. If the hazard factors selected are appropriate and/or correctly weighted the data will plot in the greyed areas of the diagram. Data which plot in the white areas can inform the weighting of the risk factors. In reality a scatter of data will still be obtained with some data points with excessive or unaccounted hazard factors.

The details of these will be provided from the project case study in Kabwe, Zambia and will underpin the risk mapping tool (Stuart et al., 2015) and this topic is covered in more detail in this accompanying report. There are a number of hazard details which do not appear to be covered in the review, e.g.:

- The critical separation between the source and the water supply in different settings.
- The role of borehole depth
- Relative importance of hazard factors

There may be other hazards that may be important which prove not to have been not covered by the summary in Table 10.1 at all which include:

- Socio-economic factors related to ownership, governance and legal frameworks;
- Other gaps in the knowledge base such as local understanding of recharge mechanisms.

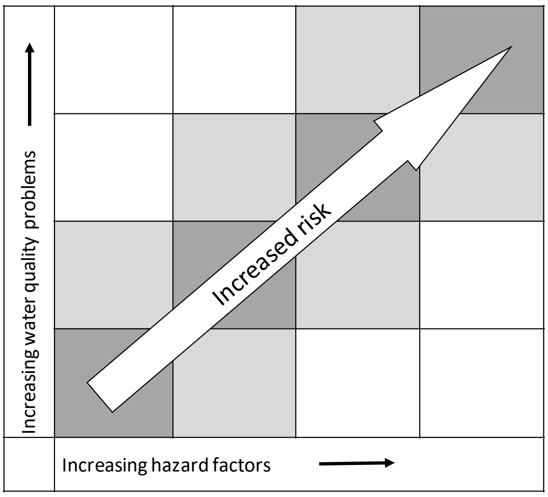


Figure 10.1 Risk assessment matrix

Table 10.1 Examples of risk mapping hazard factors that need consideration

Hazards	Category	Primary factors	Secondary factors
Regional considerations		Population density Land use category Relief/slope Abstraction regime	Managed aquifer recharge
Aquifer vulnerability		Lateritic soils Shallow groundwater Surface flooding Karstic limestones River sediments	Plinthosols and lateritic horizons Weathered basement conditions
Contaminant sources	Municipal/ household	Septic tanks Latrines Solid waste disposal Amenity pesticides	Sewers Sewage effluent Storm water runoff Cemetery Abbatoir
	Peri-urban agriculture	Livestock wastes Wastewater irrigation	Pesticides Fertilisers Soil amendment
	Industrial	Process plant effluent –textiles, pharmaceuticals, detergents Industrial solid waste disposal Fuel stations/leaking storage tanks Chemical transport and pipelines	
	Hospitals and treatment centres	Hospital waste discharge Medical waste disposal	
	Mining	Tailings and stockpiles	
Pathways	Aquifer vulnerability	Lateritic soils Shallow groundwater Karstic limestones River sediments	Weathered basement Windblown sediments
	Local/headworks pathways	Eroded or devegetated spring backfill Faulty masonry Gap between borehole riser and apron Damaged borehole apron Lack of dugwell headwall Lack of well cover Use of bucket and rope Gap between apron and well lining Damaged well apron Propensity to flood	Lack of diversion ditch Lack of wastewater drain Lack of fence Animal access Uncontrolled use
Receptors	Groundwater use Surface water	Private unmonitored supply Public unmonitored supply	Water treatment not installed No safety plan

11 Current knowledge gaps and future research

Knowledge gaps reflect the complexity of the urban groundwater system and the associated large costs of characterising the system fully. They also reflect the lack of research that has been undertaken in SSA compared to other more developed parts of the world. Systematic groundwater quality surveys with rigorous methods and large sample sizes are lacking for SSA and it would be fair to say that the research effort to date has largely focussed on water availability in SSA rather than water quality. The focus of future research should be targeted at understanding the sources, pathways and survival of pathogens in groundwater, particularly in vulnerable settings with rapid horizontal and vertical pathways.

Quantifying and fingerprinting source terms

There are a wide range of different industries across SSA which will have a correspondingly broad range of effluent quality and method of disposal. Detailed information on the groundwater impact of many of these is completely lacking. Within the literature there has been a focus on understanding faecal sources from pit latrines, and a notable paucity of studies that have investigated the impact of multi-point faecal contamination due to limited sanitation coverage in many urban and peri-urban centres in SSA.

Understanding rapid pathways in the subsurface

Important knowledge gaps remain in the characterisation of the aquifer pathway term. It is unlikely that for many urban areas of SSA there will be a wealth on detail on the characteristics of the local aquifer particularly in terms of permeability and local flow direction. Given the important role rapid horizontal (and vertical) pathways in tropical soils have in the migration of contaminants in the subsurface, and their widespread occurrence in this region, and Africa more generally, this is a key topic that warrants further investigation. Research focused on understanding the factors controlling the high failure rates (hydrogeological or otherwise) of shallow groundwater sources in the dry season would be beneficial.

There is a need to understand the resilience of groundwater quality to changes in rainfall patterns and recharge processes, including flooding, due to regional and seasonal changes in climate patterns as well as the risk of water point failure due to limited waste management in this region and their implications long-term for water resource policy and management. Tracing and quantifying residence times and pathogen occurrence in the subsurface, including in shallow groundwater systems as well as deeper systems is key to making a robust assessment of the vertical separation required between sources of pollution and groundwater points.

Microbial source tracking and anti-microbial resistance (AMR) in the environment

New techniques such as molecular marker methods (e.g. Mattioli et al., 2012; Sorensen et al 2015c) for fingerprinting pathogens, fluorescence sensors for rapidly mapping microbiological contamination of water sources (e.g. Sorensen et al., 2015a), and attention on type/depth of water point may help resolve key sources and pathways for contamination of groundwater points in this region. Very few studies using molecular tracking techniques have been undertaken in SSA to look at different sources of faecal contamination. AMR is a growing concern and very few studies have assessed anti-microbial resistance and anti-microbial residue occurrence in the environment in Africa.

Pathogen survival

There is still relatively little known about pathogen survival in the subsurface and much of the literature does not consider environmental conditions that are relevant for SSA. Much more work, including laboratory and field based studies are needed to quantify pathogen survival in the subsurface. Relatively little is known about the occurrence and mobility of enteric viruses in shallow groundwater systems in SSA compared to bacterial pathogens.

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