KTH Architecture and
the Built Environment

ANALYSIS OF THE IMPACT OF ANTHROPOGENIC POLLUTION ON SHALLOW GROUNDWATER IN PERI-URBAN KAMPALA

Robinah Kulabako

May 2005

TRITA-LWR LIC Thesis 2029
ISSN 1650-8629
ISRN KTH/LWR/LIC 2029-SE
ISBN 91-7178-103-X

DEDICATION

This work is dedicted to my parents, husband, children, sisters, brothers and inlaws.

ACKNOWLEDGEMENTS

First and foremost, I would like to express my sincere gratitude to Sida/SAREC for the financial support that has greatly made this research possible.

Iam deeply indebted to my PhD supervisors: Professor Roger Thunvik and Dr. Mai Nalubega for the research discussions, constructive advices, and suggestions during the execution of this research and thesis write-up. You both have enhanced my capacity in more ways than you know, words are simply inadequate to express my sincere gratitude. Additional thanks go to Prof. Emeritus Klas Cederwall for his kind assistance and contribution during the initial stages of this research. I wish to acknowledge Prof. Bijan for the constructive ideas on pursuance of PhD research.

My sincere thanks go to Prof. Leigh Soutter for the guidance and great contribution to the modelling aspects of the research. Am very grateful to Prof. Per Erik Jansson for the further insights into the use of the COUP Model and discussions on the research.

To Joel, Rukia, Rita, Mr. Mukasa, Dauda, Peter^2, David, and Boony, your extra hands and feet did not fail me during field and laboratory activities. Thank you very much for the support. In the same note I wish to thank the residents of Bwaise III who allowed me access to the area and availed land for installation of the monitoring wells. Additional thanks go to my fellow PhD candidates: JB, Umaru, Charles, Herbert and Assumpta for the encouragement that has kept me motivated.

I wish to extend my gratitude to my fellow PhD candidates at LWR, KTH: Kirlna, Ola (and Joanne), Kristofer and the administrators: Britt and Aira who really made me feel at home away from home during my stay in Sweden.

Lastly, my heartfelt thanks go to my family for the spiritual support. I wish to specially thank my husband Moses for his great encouragement and enduring my constant absences. To my children, Fina, Carol, Aldrian and Arnold, I thank you for a good attempt to understand my absence and welcoming me back with open arms.

To God, without you, this truly would not have been!

TABLE OF CONTENTS

LIST OF ABBREVIATIONS

ANOVA	-	Analysis of variance
DO	-	Dissolved oxygen
COUP model	-	Coupled heat and mass transfer model for soil-plant-atmosphere system
EC	-	Electrical conductivity
COD	-	Chemical oxygen demand
BOD_5	-	5-day Biochemical oxygen demand
CEC	-	Cationic exchange capacity
FEMLAB	-	Finite element laboratory
TTCs	-	Thermotolerant coliforms
PTF	-	Pedo-transfer function
NEMA	-	National environment management authority
KUSP	-	Kampala urban sanitation project
KSSMP	-	Kampala sanitation strategy and master plan
KCC	-	Kampala city council
NWSC	-	National water and sewerage corporation
USEPA	-	United States environmental protection agency
GPS	-	Geographical positioning system
TKN	-	Total kjedahl nitrogen
WHO	-	World health organization
WSP	-	Water and sanitation Programme -World Bank
WRMD	-	Water resources management department
DWD	-	Directorate of water development

ABSTRACT

An investigation to assess the anthropogenic pollutant loads, transport and impact on shallow groundwater in one of Kampala's peri-urban areas (Bwaise III Parish) was undertaken. Bwaise III is a densely populated informal settlement with a high water table (<1.5 m) and inadequate basic social services infrastructure (e.g, sanitation, safe water supply, roads, etc).

Field surveys were undertaken to identify, locate and quantify various pollutant sources. Information on the usability and operational aspects of the excreta and solid waste management systems was obtained from consultations with the residents. Water from installed monitoring wells and one operational protected spring and wastewater (sullage) characteristics (quality, discharges for drains and spring, water levels for the wells) as well as soil characteristics (soil stratigraphy, physical and chemical) were determined through field and laboratory measurements. Laboratory batch experiments were undertaken to estimate phosphorus sorption potential of the soils.

The results reveal that excreta disposal systems, solid waste and sullage are the major contributors to shallow groundwater contamination. High contaminant loads from these sources accumulate within the area resulting in widespread contamination. The water table responds rapidly to short rains (48hr) due to the pervious and shallow (<1 m) vadose zone, which consists of mostly organic fill material. Rapid water quality deterioration (increased thermotolerant coliforms, organic content in the form of total kjedahl nitrogen, phosphorus) following rains potentially follows from leaching, desorption and macropore flow. Spatial variation of the water quality in the area is largely related to anthropogenic activities within the vicinity of the well sources. Animal rearing, solid waste dumps and latrines are seen to result in increased localised microbial and organic content during the rains. The spring discharge with high nitrate levels does not respond to short rains suggesting that this source is fed by regional baseflow. The corresponding high microbial contamination in this case is a result of observed poor maintenance of the protection structure leading to direct ingress of contaminated surface runoff. Natural attenuation of contaminants is very limited. Estimated bacteria die-off rates are very low, about $0.01 hr^{-1}$, suggesting a high risk for microbial contamination. The soils still have potential to retain additional phosphorus, whose sorption is largely a function of iron, available phosphorus and moisture content of the soils. This is also seen with the model results in which the phosphorus contaminant plume sticks to the surface irrespective of the rainfall infiltration rates. Simulation results show that continuous heavy intense rains (> 0.25mm/min) result in rapid flooding occurring within 1hr to 2 days. With lower rains, the water table does not rise to the surface, and no flooding takes place.

Protection of the shallow groundwater in the area requires socio-technical measures targeting reduction of pollutant loads within the area as well as a wider spring catchment. Re-protection of the spring, coupled with awareness creation, should be immediately addressed so as to reduce microbial contamination. Community participation in solidwaste management should be encouraged. Resource recovery systems such as composting of the mostly organic waste and use of ecological sanitation toilet systems should be piloted in the area. Successful operation of the systems however depends on continuous sensitisation of the communities.

Key words: Shallow groundwater; Sanitation; Peri-urban; Vadose zone; Anthropogenic; Modelling

1 INTRODUCTION

1.1 Background

This study seeks to characterize the potential pollution sources in an informal settlement (an important element in groundwater protection and management), generate a model to simulate pollutant transport and analyse impacts of anthropogenic pollution on shallow groundwater quality. The case study area is, Bwaise III in Kampala, Uganda.

Increasing urbanisation in the developing countries is greatly associated with rapid expansion of smaller urban centres and peri-urban settlements. These areas contain high levels of concentration of the urban population from both natural increase and rural to urban migration. Much of this growth is unplanned and informal subsequently resulting in environmental degradation in these areas. This is attributed to the increasing generation of anthropogenic waste and inadequate capacity of the municipal authorities to ensure adequate provision for environmental health (Venon, 2002).

Africa, though reported to be the least urbanized continent, is recognized as one where the rate of urbanization is highest (UNEP, 2002). The development and expansion of informal settlements in the peri-urban areas of the cities is widespread. These informal unplanned and unserviced settlements harbour the majority of the urban population. For example, they are the homes for 70% of the population in Dar es Salaam (Mato 2002; Chaggu, 2004), 77% in Blantyre, 80% in Luanda (Palamuleni, 2002) and 60% in Kampala (WSP/NWSC, 2000). They are characterized by, among other things, poorly constructed houses, poor sanitary conditions, lack or inadequate support services and lack of legal status as residential dwellings.

In Uganda, increased urbanization and Industrialization in the recent years, especially in the capital city, Kampala has led to an increase in the city's population and development of informal settlements. The resident urban population of the city is the highest in the country at an estimated 1.2 million with an annual average growth rate of 3.8% (UBOS, 2002). This population size almost doubles during the day since the city serves as a workplace for residents of several nearby areas but who return home in the evening. The informal low-cost settlements in peri-urban areas have a high population density, are located in valley areas with a high water table, predominantly inhabited by the urban poor and lack or have inadequate basic services such as water supply, sanitation (excreta, solid waste, sullage and storm water management) and other infrastructure. Some of these settlements have spilled over into natural wetlands and have contributed significantly to the now common flooding of many residential as well as commercial areas of Kampala during the rainy seasons. Examples of such areas include Bwaise, Katwe, Mulago Nsooba, Lower Namuwongo, Kiswa and Kisenyi.

The capacity of the municipal authorities to provide basic services to meet the needs of the increasing population is currently limited. In Kampala, about 900 tonnes of solid waste are generated daily, of which only about 40% is collected and disposed of by Kampala city council (KCC) while the rest is indiscriminately disposed of (Sikyajula, 2003). Of the total effluent from industrial and domestic sources in the city, less than 6.5% is treated and the rest is discharged untreated (KSSMP, 2004). In the informal peri-urban settlements in the city, the environmental conditions are unhealthy largely due to the poor sanitation in these areas (KCC, 1999; Nuwagaba and Mulogo, 2000). These, coupled with poor land use practices which include wetland destruction, have resulted in unprecedented discharge of large quantities of untreated pollutant loads in both surface water and shallow groundwaters. In addition, storm water channels traversing some of these areas are operating as open sewers and solid waste disposal points, which subsequently discharge into surface water bodies. The quality of the shallow groundwater in these areas is therefore of major concern as majority of residents utilize it for daily activities. Recent studies carried out in these areas suggest a link between the incidence of cholera, acute diarrhoea and use of contaminated protected springs (Howard *et al.*, 2003). In addition, due to poor sanitation, disease outbreaks (malaria, cholera, typhoid, etc) are prevalent especially during the rainy seasons because of flooding (Plan, 2001). These populations therefore, are at great health risk from unsafe water and inadequate sanitation.

Currently there is limited water quality monitoring, data records and analysis of available information. Therefore, the characteristics, which include type, quantities (loads), and sources of this anthropogenic pollution from the peri-urban areas of Kampala, are largely unknown. In addition, there is no basis for assessing the impact of the resultant loads on the subsurface waters in these areas. Consequently, the variability and impacts of pollution as a result of human activity in the different areas are not known. Efforts to mitigate the impacts of this pollution by policy makers are limited by, amongst other reasons: lack of a comprehensive database on pollution from domestic sources, water quality data, financial constraints, land ownership issues, and non-implementation of mitigation measures by local communities.

1.2 Problem statement

In Uganda, the increasing industrialization and urbanization especially in the city of Kampala is not matched with an increase in measures to control pollution resulting from this growth. Many substances emitted into the environment from these anthropogenic activities end up in the water bodies and are increasingly posing a threat to the social, economic and health status of the people as well as to the proper functioning of aquatic ecosystems.

The current sanitation situation in the peri-urban areas of Kampala where the majority (about 60%) of the city's population resides is poor. Though the population in these areas is considered transient, it is increasingly recognized that these settlements are here to stay since they provide the largest source of the workforce necessary for urban growth (Lukman and Doyen, 1996). The limited capacity of the municipal authorities to provide the basic services to these areas has resulted in poor environmental sanitation in these settlements. This has consequently led to contamination of the shallow groundwater aquifers, a source that is heavily relied on by these communities (Howard *et al.*, 2001; Howard *et al.*, 2003). Treatment of this water occurs within the home, if at all. In this context, contamination of these sources clearly poses a direct and immediate public health risk to a relatively higher population. In addition, Town/City authorities lack information on the characteristics of pollution (types, quantities, variation with seasons) and there is no basis for assessing the impact of the resultant loads on the receiving waters.

1.3 Case study area - Bwaise III Parish

Bwaise III Parish is found in Kawempe Division, Kampala District. It is located in the northern part of Kampala City approximately 4 km from the city centre (Fig.1). It is a low-lying area (mostly a reclaimed wetland) with a high water table (<1.5 m) in most of the areas. The parish has an area of 57ha and is divided into six local administrative zones namely: Kalimali, Bokasa, Bugalani, St. Francis, Katoogo and Kawaala (Fig.2). The area, though mainly residential has some economic activity for the low-income residents. This constitutes mostly trading which ranges from cooking and selling food in markets to hawking and retail shops.

The area is largely unplanned with lack of basic services, poor road access and deplorable housing. It has one of the highest population growth rates in Kampala District with an annual average rate of 9.6% more than twice the average city's growth rate with an average population density of about 27,000 persons/km^2 (UBOS, 2002).

1.4 Objectives

The aim of this study is to assess the anthropogenic pollutant loads, impact and transport in Kampala's peri-urban areas with a focus on shallow groundwater.

The specific objectives of the study are:

i) To locate and determine the characteristics (types, loads) of pollution sources in the selected peri-urban settlement

ii) To assess the influences of rainfall and infiltration on pollutant loadings into the shallow groundwater

iii) To determine the attenuation capacity of the soils for selected contaminants

iv) To generate a predictive model for simulation of transport and fate of the studied pollutants in (iii) and impacts on the shallow groundwater quality

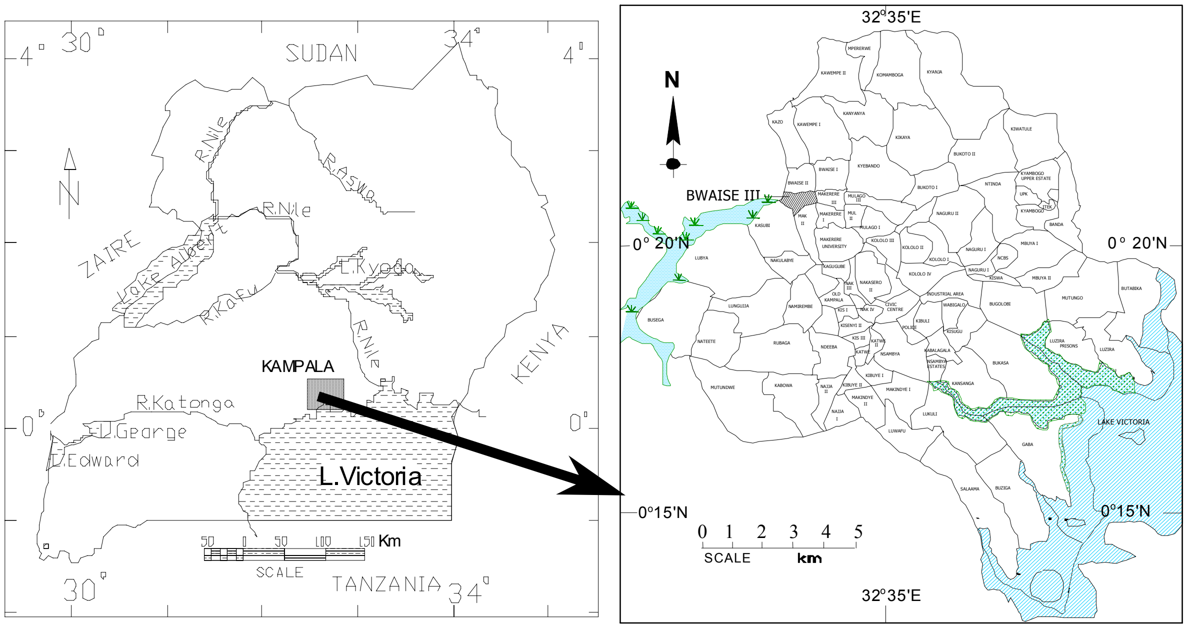


Fig. 1 Map of Uganda showing location of Kampala District (left) and Kampala District map showing the location of Bwaise III, Parish (Right)

1.5 Scope of study

The study focuses on shallow groundwater (<1.5 m below the ground surface) in a selected Peri-urban settlement namely Bwaise III in Kampala as a case study. Pollution studies were limited to solid waste, excreta disposal and sullage/stormwater drain facilities as major contributors to subsurface water contamination in the area. Due to the complexity of the dynamics in the natural environment and spatial heterogeneity of the flow domain, preliminary predictive model formulation focuses on a smaller area within the study area. Contaminant transport studies (adsorption batch experiments) were limited to reactive phosphorus. The latter is observed to occur in high concentrations (>> 0.05mg/l) in both the shallow groundwater and storm/sullage drains.

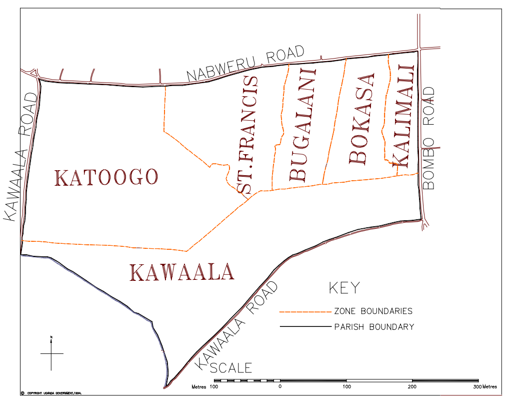


Fig. 2 Administrative zones in Bwaise III Parish

1.6 Thesis layout

The thesis is organized in seven chapters. Chapter one gives the background to the study. The second chapter gives an overview of the groundwater, related hydrogeology situation in Kampala, as well as policy/regulatory and institutional issues. Chapter three gives a more indepth review of the groundwater resource, its vulnerability to pollution (hydrogeological aspects, contaminant transport mechanisms) and chapter four highlights the methodology that was adopted to achieve the objectives of the study. In chapter five and six, the results and discussion are presented respectively. In chapter seven, conclusions and recommendations on potential future research following from the study are given. Appended to the thesis are three papers that reinforce the material in chapters five to six, providing key information on specific issues of the study.

2 The Groundwater Situation in Kampala

2.1 Introduction

In Kampala, where informal settlements make up the majority of the urban neighbourhood, shallow groundwater is still considered a major source of water supply due to its low-cost and relative abundance. Its quality is however, of great concern due to the increasingly large quantities of anthropogenic waste. This chapter describes the groundwater situation in the city of Kampala with a focus on the geology, hydrogeology, water quality and pollution control aspects.

2.2 Topography

Kampala city, in which the study area (Bwaise III parish) is located, is built on a number of hills, the pronounced topography being controlled by differential weathering of various grades of meta-sedimentary bedrock types. The topography results in thin weathered mantles of limited extent (a stripping environment), which produces shallow, localized groundwater flow systems with relatively short residence times. These systems, discharge to springs that form at geological boundaries on the lower slopes of hills and at the edges of lowland swampy areas (ARGOSS, 2002) such as Bwaise. The springs are fed by a combination of base flow and seasonally derived interflow (Barrett *et al.*, 2000).

According to Biryabarema (2001), the low lying areas have comparatively little flat ground but consist of the lower pediment and valley bottoms that are relatively swampy and seasonally flooded. Where topography is very subdued, the valleys are broader and so aggraded that they are choked by silt and carry no surface streams for most of the year, neither are there stream courses. The gradients are so slight that heavy rain usually produces temporary standing water or swamps.

2.3 Geology

The city of Kampala is underlain by a precambrian basement of granite gneisses, Buganda Toro quartizites, schists, phylittes, and amphibolites, and by pleistcene to recent alluvium and lacustrine deposits and soils. The greater part of Kampala is covered with soils derived from weathering of basement rocks. These are brown to red (lateritic), and attain a deeper coloration when found near basic rocks because of higher Fe-contents (Biryabarema, 2001).

The low-lying areas such as Bwaise III, especially the valley bottoms are covered by alluvials sediments of clays, silts, sands and gravels. Pallister (1959) in Biryabarema (2001) noted that most valleys are aggraded with variable thickness of silt or sandy clays with little stratification both laterally and vertically in clay and sand fractions. He noted that a relatively thin dark grey clay layer, 0.3-0.61m thick with much fine organic matter underlies the papyrus of the wet swamps. The clays are of a stiff compact nature and are highly impervious. These are derived from the decomposition of feldspar rocks and formed by sedimentation in the waterlogged valleys. X-ray diffraction studies undertaken on the mineralogy of the alluvial clays of Kampala (Kirabira, 2003 and Biryabarema, 2001) show that these are mostly kaolinites.

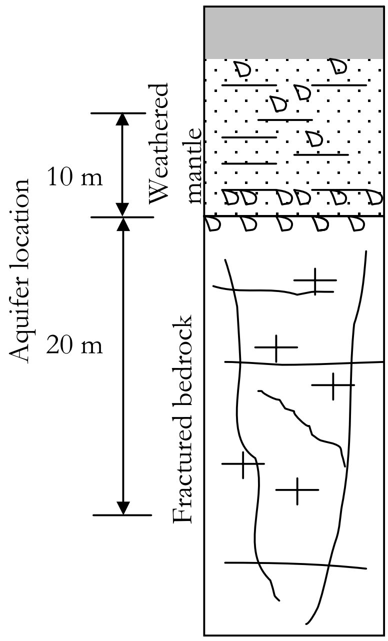


Fig. 3 A typical cross-section through the aquifer system in Uganda *(adapted from Taylor, 2001)*

2.4 Hydrogeology characteristics

The main aquifer type in Uganda, as in much of Sub-saharan Africa and many other tropical regions, is weathered crystalline bedrock. This hydrogeological environment is very extensive though aquifers are often restricted to the upper 30m or less (Morris *et al.*, 2003). In these environments, groundwater is transmitted by fractures in the bedrock and unconsolidated weathered materials that form the overlying mantle or regolith. These form the deep and shallow groundwater aquifers, respectively (Fig. 3). Water level records and geophysical surveys undertaken in regolith aquifers developed in some areas in Mukono, East of Kampala city, reveal an aquifer thickness of at least 10 m, with the aquifer being largely unconfined (Taylor and Barrett, 1999). Groundwater levels within the regolith are usually shallow, at depths usually less than 8m and reducing to less than 5 m lower down the valley slopes (Biryabarema, 2001).

Due to a primarily *insitu* development, mantles of weathered rock constitute the progressive degradation of bedrock materials and therefore produce highly anisotropic aquifers. The weathering dynamics in the unsaturated zone of weathered mantles remain poorly understood. Studies by Taylor and Howard (1999) and Taylor (2000) in ARGOSS (2002) reveal two key characteristics about the weathered mantles:

- ☞ The texture of weathered mantle features a bimodal particle size distribution as sand-sized, primary minerals are progressively weathered to clay-sized, secondary minerals at shallower depths. This has important implications: localised variations in the parent rock matrix give rise to high degree of spatial heterogeneity in weathered mantle lithology; secondary structures of the bedrock such as quartz stringers are translated into the composition of the weathered mantle producing preferential pathways for subsurface flow and contaminant transport.
- ☞ Less aggressive weathering is associated with saturated conditions and the persistence of coarse-grained materials at the base of the weathered mantle. The aquifer in the weathered mantle (shallow groundwater) comprises poorly sorted, muddy sand. This is the most productive horizon within the weathered aquifer system, and is often of regional extent.

The city has a shallow aquifer from the weathered regolith. A typical weathered mantle shallow groundwater aquifer in the study area is underlain by an impermeable layer that constitutes of stiff consolidated clays (Fig. 4).

Recharge to the shallow aquifer is not well understood in part, as rainfall occurs throughout the year. Studies undertaken on the dynamics of groundwater flow (Taylor and Howard, 1998) in Arocha catchment, within the Victoria basin of Central Uganda, where Kampala falls climatologically, suggest that recharge varies considerable across the region. However, it primarily occurs during the two distinct periods of heavier rainfall (Wet seasons: March to May and September to November). The estimated recharge for Arocha catchment based on the soil moisture balance approach supported by stable isotope data and groundwater flow modeling is about 200 $mmyr^{-1}$ and is more dependent on heavy rainfall events (more than 10mm day^{-1}) than the total annual rainfall. Infiltration of precipitation (rainfall) at the soil surface is the principal source of groundwater in the area.

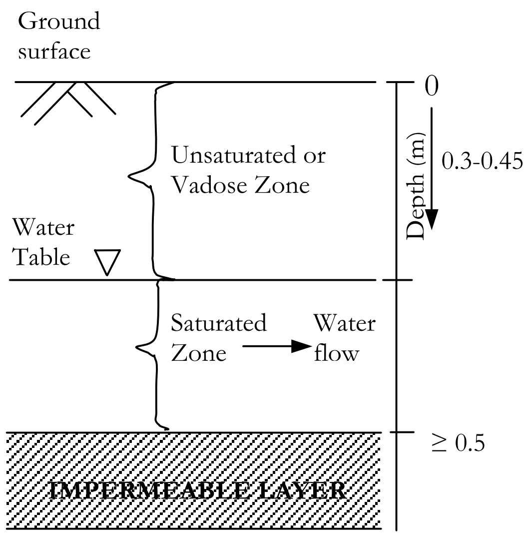


Fig. 4 A typical shallow subsurface water environment in Bwaise III Parish

According to Taylor and Howard (1998), the regolith aquifer does not act independently but is hydraulically connected to the basement aquifer.

Deep groundwater aquifers in the area occur in the underlying crystalline fractured bedrock at depths of at least 30 m (WRMD-DWD, National Groundwater database). Hygrogeological investigations carried out by Kansiime and Nalubega (1999) and Biryabarema (2001) in a similar environment (about 13 km from Bwaise III Parish) involving borehole logs and resistivity measurements encountered deep groundwaters at depths of more than 30 m. Traditionally in Uganda, these have been exploited as water supply sources. Recent studies undertaken in deeper aquifer systems reveal that water yields from the fractured rocks especially in the crystalline basement are lower than in the overlying regolith and sediments by one to two orders of magnitude (Taylor & Howard, 1998). Hence, current efforts focus on development and protection of the regolith and sediments (shallow groundwater aquifer). This is due to the higher yields and lower costs of development under favourable conditions (Taylor and Howard, 2000).

2.5 Groundwater sources

The major source of groundwater supply within Kampala is springs within the shallow aquifer. Boreholes are of minor importance for domestic water supply in the city for reasons mentioned in section 2.4. Shallow wells, dug in alluvial clayey sediments are limited in yield due to low soil percolation. Therefore, their occurrence is fairly low. Springs supply about 50% of Kampala's population with the majority of these occurring in high-density settlement areas mainly in the peri-urban (informal settlements). In these areas, they are susceptible to pollution related to anthropogenic activities even when notionally "protected" (KSSMP, 2004). The municipal authority responsible for these sources (inspection, monitoring of the quality) is Kampala City Council (KCC).

A pilot surveillance programme established that there were over 200 protected springs in the city, in addition to the many unprotected springs (Howard *et al.*, 1999 in Howard *et al.*, 2003). The State of Environment Report (1997) gives an estimate of 318 protected springs in Kampala district. Despite the existence of piped water supply infrastructure within the city, the use of protected springs is common in the high-density peri-urban areas. According to KUSP (2004), it is estimated that over 90% of the low-income population with access to protected springs use them for part or all of their domestic water needs including drinking, bathing, washing and cooking, the majority of these being women and children. A situational assessment study also revealed that groundwater sources (protected springs and wells) were a significant alternative water source (in the course of piped water supply interruptions) in these areas (WSP/NWSC, 2000).

Protection of these sources is by constructing a spring box around the eye of the spring, which is then backfilled with a fine gravel or sand and hard core to provide additional filtration. The backfilled area is usually covered by layers of sand (such as lake sand) or finer material (e.g clay) to limit percolation of potentially contaminated surface water (Howard *et al.*, 2001). The whole area is typically fenced and has a ditch to divert surface water away from the source (Fig. 5). The water is collected directly by users from the springs in Kampala.

2.6 Shallow groundwater quality

Previous studies undertaken on the protected springs in Kampala point to widespread faecal contamination (Howard *et al.*, 2003; ARGOSS 2002; Barrett *et al.*, 2000 and Barrett *et al.*, 1998). The risk of virus presence in these sources is probable as pointed out from the limited data on virus indicator detection studies. The findings demonstrate microbiological contamination to be most severe during the rainy (recharge) season and this is most related to short term (24 and 48hr) rainfall in which quick responses are realized.

Values of thermotolerant coliforms (faecal coliforms) $\geq$ 30,000 cfu/100ml have been recorded in these sources with average median values of about 540 cfu/100ml. High

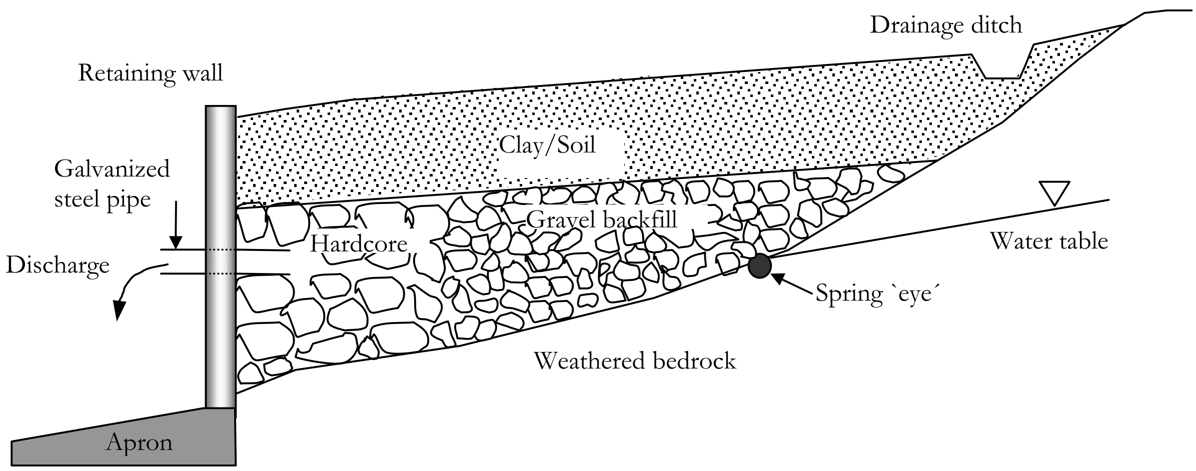


Fig 5 Protected spring design

nitrates (average median values 67 mg/l) and chlorides (average median values of 59 mg/l) have been recorded especially in the high-density areas. This is attributed to the inadequately protected sources by contaminated surface water from different types of sources.

A recent survey undertaken on the quality of about 135 springs within Kampala (KUSP, 2004) reveals that generally all the surveyed springs have a very poor bacteriological water quality well above World Health Organization (WHO) guidelines and national drinking water standards. Only about 10% of the springs sources met the stipulated 0 FC cfu/100ml while values in the range of 10-670 cfu/100ml were detected in the rest of the sources. High nitrite values above the WHO and national standard (3 mg/l; WHO, 2004; NEMA, 1998) were detected in all the springs (range 80-100mg/l). Due to the nature of the anthropogenic activities within the catchment of these sources (residential with very minimal industrial activity), heavy metal concentrations are in trace amounts while organochlorine pesticides were not detected. Most of the springs had acceptable physico-chemical quality (electrical conductivity, turbidity and total hardness). However, the pH is below 5.5 (acidic range) for most of these sources. This survey was undertaken over a short period (mostly during periods of low rains) and hence could not adequately assess the impacts of seasonal variation (including rainfall) on the quality of these sources.

The land use pattern in Kampala (as in many other rapidly urbanizing areas of Sub-Saharan Africa) is complex. Most of the springs have unprotected catchment areas due to over-crowding. Where catchment protection exists, the areas are of limited size, not fenced and hence there is encroachment. The sanitary condition at most of these sources is poor. Protection of catchments of these sources is seen as a complex issue under such circumstances and will involve analysis of both hazards and pathways (Barrett *et al.*, 2000). It is observed that the principal route of contamination of the protected springs is the immediate area surrounding the protection works and that localized sources of pollution are the major contributors to groundwater quality deterioration. Rainfall in this case is the primary climatological control factor, as it will aid both the washing in of contaminants to the backfill area and replenishment of the contaminated surface waters.

The sources of contamination of these shallow groundwaters are noted to result from solid waste dumps, low coverage of excreta disposal facilities (pit latrines), resulting in indiscriminate disposal of faecal matter into the environment (drainage channels, solid waste dumps and surface water), stagnant surface water due to inadequate management of sullage/storm- water runoff, and domestic animals such as goats, cows, pigs and chicken especially in the low income peri-urban settlements.

The drainage and flood aspects of Bwaise III parish from the above-mentioned are therefore a critical factor not only as far as the general living conditions are concerned, but also to the shallow groundwater quality. More than 40% of the parish comprises low lying flood plain that was originally wetland but has in the recent past been encroached on by informal settlements due to pressures of rapid urbanisation. The area is characterized with an irregular network of drainage, about 6900 m in total length (Plan, 2001). It lies within a bigger catchment basin (29.25 km^2), which is one of the major drainage basins of Kampala. Estimated peak runoff for a 10-year return period within the catchment area is 84.4 m^3/s (Lwasa, 1999). Three main stormwater drains flow through the area: Bwaise-Nsooba, Nakamiro and Kiwunya. There are many other secondary and tertiary drains in the area (Fig. 6).

These stormwater drains are generally polluted because they operate as combined sewers and are recipients of solid waste. The direct ingress of the contaminated stormwater during flooding into the poorly protected spring sources has been pointed out as a major source of microbial and organic contamination of these sources in this and similar areas in Kampala (Howard *et al.*, 2003). Studies undertaken on main drains traversing peri-urban settlements in Kampala show that the wastewaters carried by the drains have high organic loads (as high as 350 mg BOD_5/l and 400 mg COD/l) and microbial counts (Thermotolerant counts of orders of magnitude 5 to 6) (Niwagaba, 2002). The self-purification capacity of these surface water drains is greatly impaired by the numerous point and diffuse sources of pollution along their entire length. Previous studies undertaken along the main drain traversing Bwaise III parish (Bwaise-Nsooba) showed a marked increase in the thermotolerant counts (to orders of magnitude of 6), phosphorus (>1.5 mg/l), electrical conductivity (>400 μS/cm) and a reduction in dissolved oxygen (<0.6 mg/l) along the drain as it traversed Bwaise III Parish (Plan, 2001) indicating increased waste disposal into this drain from the area. However, during flooding conditions, the wastewater carried by these drains from the neighbouring areas partly stagnates in the area and hence has a synergistic impact (with the localised sources) on the quality of the shallow groundwater sources and the quantity (as noted from the quick recharge of short term rains) following from the recharge by contaminated flood water (as noted earlier).

2.7 Pollution mitigation measures

To prevent contamination of spring sources in these areas, a lot of effort has been focused on rehabilitation and spring protection works (Howard *et al.*, 2001). As noted in section 2.6, poor operation and maintenance and poor sanitary condition of many of the protected springs are the major causes of contamination, which in part explains why there is continued deterioration of the spring water quality in the high-density peri-urban areas. To improve the quality of these sources therefore requires attention towards preventing pathways into the source developing e.g., sanitary protection of these sources, use of appropriate materials for retention of contaminants, as well as reducing hazards such as improving solid waste and excreta disposal methods.

To protect natural resources, which include both surface and groundwater from pollution, and to ensure a healthy environment, Uganda Government has enacted various policies and legislations. Among these is the KCC Solid waste Management Ordinance, 2000; National Health Policy, 1999; National Environment Statute, 1995 and the Public Health Act, 1964. However from observation, the enforcement of these within the peri-urban areas is very weak hence the current environmental degradation. The communities attribute the latter to inadequate municipal authority service, poverty, absentee landlords, lack of space for construction of excreta disposal facilities and solid waste disposal, as well as poor attitude of the residents associated with their "temporary" resident status (Plan, 2001). The authorities attribute their limited service to lack of financial and human resources and difficulty of enforcement of some of the laws in practice

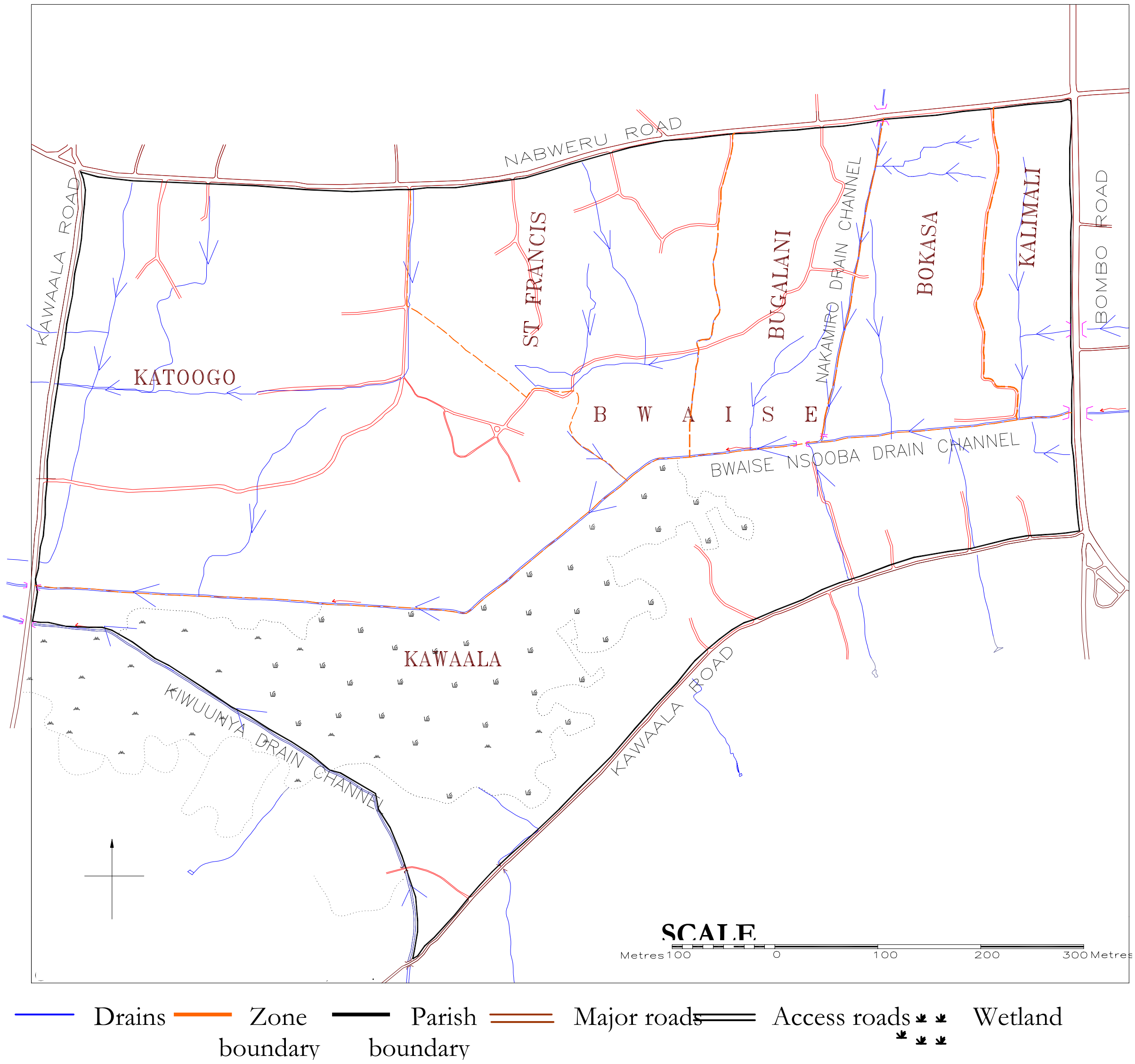


Fig. 6 Drainage network in Bwaise III Parish

(for example, *chapter 269: The Drainage and Sanitation Rules, Sect 59 which prohibits the construction of pit latrines, septic tanks and other works for the treatment, reception or disposal of sewage within 33m of any well, spring or stream of water used by man for drinking or domestic purposes*) (KSSMP, 2004). Besides this, revision of some obsolete laws is currently ongoing but is still an incomplete process.

To the service providers, these areas are considered illegal settlements and hence not catered for (WSP/NWSC, 2000). It is increasingly realized therefore that no single corrective measure can be applied to all the problems associated with the shallow groundwater quality. Hence appropriate solutions to address the existing problems are not only technical but have to be integrated and multi-sectoral. In the case study area socio-technical measures targeted at reducing the actual pollutant loading in the area and subsequently the shallow groundwater as noted earlier (first paragraph section 2.7) should be promoted.

3 Literature Review

3.1 Introduction

In recent years, there has been increasing awareness of, and concern about groundwater pollution in the urban areas in the world. In the developing countries, this results from discharge of inadequately treated or untreated industrial effluents, uncontrolled domestic discharges, diffuse pollution from agriculture, livestock rearing, and various alterations in land use for example settlements in wetland areas. In low-income countries, like Uganda, the situation is noted to be aggravated by the rapid urbanization, which is characterized by inadequate provision of water supply, sanitation, solid waste and drainage infrastructure.

In this chapter, a review of the shallow groundwater resource (widely used in urban areas in developing countries such as Kampala, Uganda), hydrological aspects, flow dynamics and vulnerability to pollution is presented.

3.2 Groundwater

Groundwater is subsurface water that fills voids and permeable geological formations. Shallow groundwater typically occurs at depths of <10-20 m. Like surface waters, the groundwater or subsurface environment is a recipient of pollutants. To effectively solve contamination problems and mitigate threats of contamination, it is important to have a thorough understanding of the factors that govern the transport and eventual fate of subsurface pollutants, which is the subject of this section.

3.2.1 Hydrological aspects of groundwater (quantity and quality)

Groundwater does not exist in isolation, but is an integral component of the hydrological cycle: the endless circulation of water between the oceans, atmosphere and land (Fig. 7). Groundwater aquifers are periodically replenished by precipitation and by surface water percolating down through the soils as shown in Figure 7. However, the degree of replenishment or recharge depends on the soil characteristics (e.g., porosity, hydraulic conductivity and current moisture content), climate and vegetation. Water stored in aquifers is usually in motion, flowing slowly under the influence of gravity, until it discharges into a spring, stream, lake, wetland or the ocean or is taken up by plants or is extracted by wells.

The physical hydrologic processes of precipitation, infiltration, surface runoff, subsurface flow, and stream flow play an important role in the propagation of contaminants generated by human activities in a particular watershed (Fig. 7, also see section 2.6). However, depending on the particular climatic, geologic, topographic and vegetative characteristics of the watershed in question, some of these processes might be negligible (Sergio, 1997).

3.2.2 Typical characteristics of the subsurface zones

In the unsaturated zone (also known as the vadose zone), both air and water are present in the pore spaces between the mineral grains. The uppermost layers of this zone, typically a few tens of centimetres in depth, are traditionally called soil. The latter is characterized by intense biological activity, which entails water and dissolved material uptake by plant roots; bacteria, fungi, and small soil animals; and a variety of natural organic acids are produced from the decay of organic material (dead roots, fallen leaves etc). Precipitation that does not simply runoff the land surface into a surface water body or storm drain, enters the vadose zone (infiltration). It replaces water lost either by plant root uptake and subsequent transpiration or by direct evaporation. If the water content in this zone becomes sufficiently high, the excess water percolates downwards to the water table in a process known as recharge.

The saturated zone is formed by porous material in which all of the pore spaces are filled with water. The water table is defined, as the depth at which pore water pressure equals atmospheric pressure. The location of the water table is determined by digging a hole down into the saturated zone and establishing the maximum level at which water accumulates in the hole in the absence of abstraction (Chapman, 1992). In coarse porous material, the location of the water table very nearly approximates the transition between saturated and unsaturated material; in fine textured porous material, enough water may move upward by capillarity to cause complete saturation of a measurable thickness above the water table (capillary fringe).

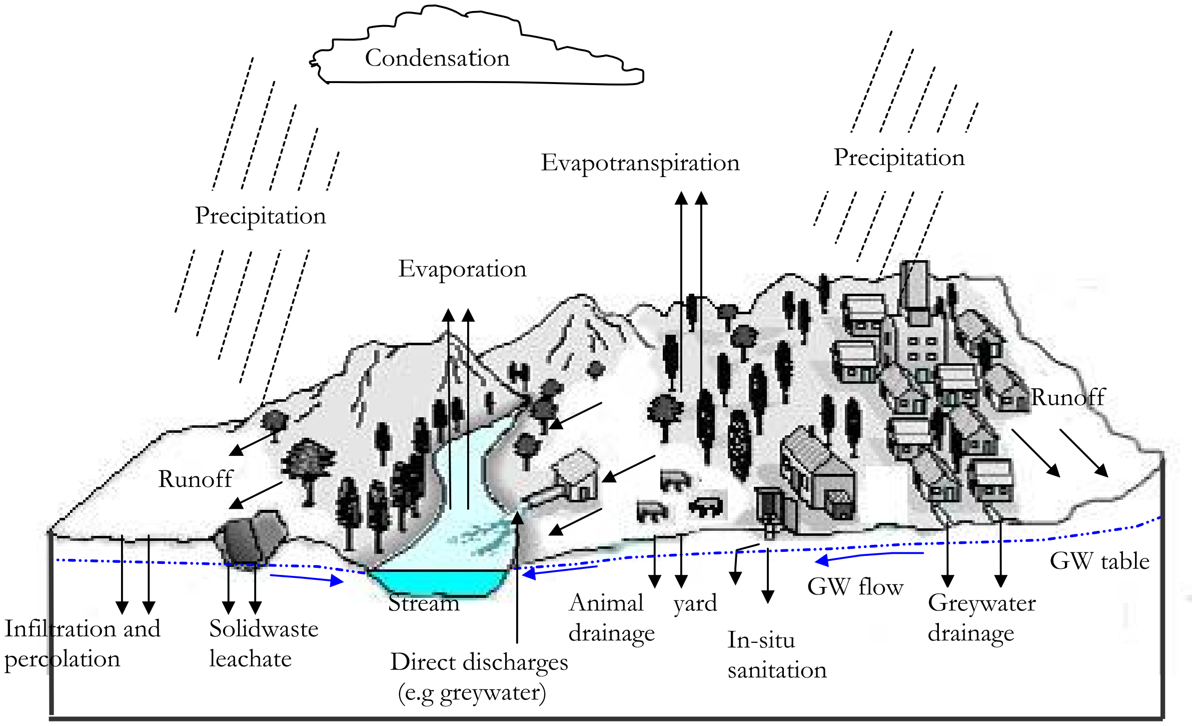


Fig. 7 The hydrological cycle and its role in contaminant propagation *(Modified from Foster et al., 2003)*

According to Lewis *et al.*, (1982) concern about groundwater pollution relates primarily to unconfined aquifers, especially where their vadose zone is thin and their water table is shallow, but may also arise even where aquifers are semi-confined, if the confining aquitards are relatively thin and permeable. This may be attributed to the fact that unconfined aquifers respond rapidly to changes in the rate of infiltration and thus water tables fluctuate according to seasonal changes in the rate of recharge. Shallow groundwater table fluctuations may accelerate the rate of pollutant movement due to interception of descending contaminated infiltration by the rising water table. Consequently where the quality of the groundwater in unconfined aquifers is concerned, full recognition must be given to its relationship to surface-water quality.

3.2.3 Subsurface flow hydraulics

An understanding of the groundwater flow is important inorder to adequately estimate the direction and speed at which contaminants are transported in an aquifer (Dominico and Schwartz, 1990). However, usually it is not possible to directly measure the rate at which groundwater moves through an aquifer, so flow is estimated from known relationships among measurable parameters (Schnoor, 1996).

The media under study comprises of flow in both the unsaturated and saturated zones (Fig. 4). Hence the flow transport model uses a generalization of Darcy" s Law (eq. 1) together with Richard's equation (eq. 2) (Comsol, 2004) to simulate soil-water movement in a vertical profile under rainfall infiltration.

$$\mathbf{q} = -\frac{k}{\mu} k_r \nabla\left(p + \rho_f g D\right) \tag{1}$$

$$\left[C + S_e S\right]\frac{\partial p}{\partial t} + \nabla \cdot \left[-\frac{k_s}{\mu} k_r \nabla\left(p + \rho_f g D\right)\right] = Q_s \tag{2}$$

Where $\mathbf{q}$ is the specific flux (Darcy flux), p is pressure, C is the specific capacity, S_e is the effective saturation, S is the storage coefficient, ρ_f is the fluid density, k_s is the saturated permeability, k_r is the relative permeability, g is the acceleration of gravity, D is the vertical coordinate, and Q_s represents sources and sinks.

The flow of moisture through the unsaturated soil can be calculated given measurements of soil suction head (ψ) at different depths z in the soil and knowledge of the relationship between permeability, k and ψ. These relationships are usually referred to as the unsaturated soil hydraulic characteristics.

Many of the important functions of soil such as buffering, filtering, transport of pollutants, occur in the unsaturated (vadose) zone. Hence soil hydraulic characteristics in this zone are crucial input data for modelling water and solute movement in this zone. However, the ability to predict transport in this zone remains a challenge. Prediction of unsaturated hydraulic characteristics is very problematic (Dexter, 2004). Despite possibility of measuring these characteristics, the measurements are difficult, generally cumbersome, time and labour intensive and costly. To overcome these difficulties, predictive methods that estimate the soil hydraulic properties from easily measured soil attributes have been introduced. These methods infer the hydraulic conductivity function from knowledge of soil water retention data or by an alternative approach, which involves use of pedotransfer functions (PTFs). These two approaches are considered in this study (Appendix, Paper III) and are here reviewed.

Water retention curves

Generally, as the water content of a soil decreases, the pressure head becomes more negative, or alternatively the capillary pressure increases. The relationship between negative pressure head and volumetric water content for a sample is called the water-retention curve. This curve describes how a sample behaves as water is added or removed. In practice, volumetric water content is plotted on an arithmetic scale while the negative pressure head is plotted on either an arithmetic or logarithmic scale. These curves are determined experimentally. They are typically non-linear regardless of how they are plotted. The shape of the water retention curve changes depending on whether the soil is drying (air replacing draining water) or wetting (entry of water to displace the air). The term *hysterectic* is used to describe this effect. In modeling applications, it is common to represent water retention curves using various types of mathematical relationships. The commonly used relationships are the Brooks-Corey (1964) and van Genuchten (1980) equations (eqs. 4 & 5 respectively). These are written in terms of a dimensionless soil moisture content termed the effective saturation (S_e) (Schwartz and Zhang, 2003):

$$S_e = \frac{\theta - \theta_r}{n - \theta_r} \tag{3}$$

where θ is the actual volumetric water content, θ_r is the residual volumetric water content, and n is the porosity. The effective saturation varies between 0 and 1.

$$S_e = \begin{cases} \left(\dfrac{\psi_b}{\psi}\right)^{\lambda}, & \psi < \psi_b \\ 1, & \psi \geq \psi_b \end{cases} \tag{4}$$

Where Ψb is the bubbling or air entry pressure head (L) and is equal to the pressure head to desaturate the largest pores in the medium, and λ is a pore-size distribution index.

$$S_e = \frac{1}{\left[1 + \left(\alpha|\psi|\right)^{\beta}\right]^{\gamma}} \tag{5}$$

Where a is coefficient (1/L), β is the exponent, and $\gamma = 1\text{-}1/\beta$.

The coefficients Ψ_b and λ in the Brooks-Corey equation, and a and β in the van Genuchten equations can be determined by fitting the measured water retention curve with the calculated water retention curves using the unsaturated flow equation.

Hydraulic conductivity for the unsaturated media is not a constant but is strongly dependent on the degree of saturation. The hydraulic conductivity takes on its maximum value when the media is near saturation with a head pressure close to zero. However, when the media dries out (water content reduces), the pores become filled with air, pressure head becomes more negative and the hydraulic conductivity decreases. Approach of the volumetric water content to residual results in a water phase that may not even be continuous through the sample, giving a hydraulic conductivity that is close to zero. Hence in the zone unsaturated, the concept of relative permeability or relative hydraulic conductivity has proven useful in capturing the relationship existing between hydraulic conductivity (*K*) and negative pressure head (-*Ψ*). For the unsaturated media therefore, the relative hydraulic conductivity is defined as:

$$K_r(\psi) = \frac{K(\psi)}{K_s} \tag{6}$$

Where K_r is the relative hydraulic conductivity, which varies between 0 and 1, K_s is the hydraulic conductivity when the medium is saturated, and *K* is the unsaturated hydraulic conductivity. From eqn (7), the product K_rK_s shows that the unsaturated hydraulic conductivity is really some fraction of the saturated hydraulic conductivity. The determinant of this fraction is the –*Ψ*. Following from this, expressions of the relative hydraulic conductivity and unsaturated hydraulic conductivity as a function of the pressure head have been developed. These are for the Brooks-Corey (eqns 8 & 9) and van Genuchten (eqns 10 & 11) models (Schwartz and Zhang, 2003; Jansson and Karlberg, 2004).

Where the coefficients *λ* in eqs. (7 & 8) and α, *β* and *γ* in eqs. (9 & 10) are the same parameters as used in equations 4 and 6 respectively. Parameter *n* in eq. (8) is accounting for pore correlation and flow path tortuosity. The relative hydraulic conductivity may also be expressed as a function of saturation for these models (Eq.11, for the van Genuchten hydraulic model).

$$K_r(\psi) = \begin{cases} \left(\dfrac{\psi_b}{\psi}\right)^{2+3\lambda} , & \psi < \psi_b \\ 1 , & \psi \geq \psi_b \end{cases} \tag{7}$$

$$K = K_s\left(\frac{\psi_a}{\psi}\right)^{2+(2+n)\lambda} \tag{8}$$

$$K_r(\psi) = \frac{\left\{1-(\alpha|\psi|)^{\beta-1}\left[1+(\alpha|\psi|)^{\beta}\right]^{-\gamma}\right\}^2}{\left[1+(\alpha|\psi|)^{\beta}\right]^{\gamma/2}} \tag{9}$$

$$K = K_s\frac{\left(1-(\alpha\psi)^{\beta-1}\left(1+(\alpha\psi)^{\beta}\right)^{-\gamma}\right)^2}{\left(1+(\alpha\psi)^{\beta}\right)^{\gamma/2}} \tag{10}$$

$$K_r(S_e) = S_e^l\left[1-\left(1-S_e^{1/\beta}\right)^{\beta}\right]^2 \tag{11}$$

Where *l* is the pore connectivity and equal to about 0.5 for many soils (Mualem, 1976 in Schwartz and Zhang, 2003). This value produces acceptable results for most coarse-textured soils but not for many medium - and fine textured soils. It is proposed to keep *l* as a free parameter for more flexibility in the fitting of measured data (van Genuchten *et al.*, 1991).

Pedo transfer functions (PTFs)

Use of the water retention curve approach

albeit well validated and sufficiently accurate procedure, still requires a good number of laboratory or field measurements to describe the water characteristic *θ (h)* for a wide range of conditions from saturation to very low values of pressure potential. An alternative approach (indirect method in contrast to direct measurement of hydraulic characteristics), which is receiving increasing attention among soil scientists, modelers and users, is the use of PTFs. A pedotransfer function acts a tool for generating the soil hydraulic characteristics by using a more or less complicated algorithm, with combinations of the soil physical and chemical properties, primarily texture, bulk density, and organic matter (Nunzio and Santini, 1997). These parameters are more easily measurable and hence Wösten (2001) points out that PTFs are in essence predictive functions that translate available soil data into missing soil properties.

According to Wösten *et al*., (1995), PTFs are divided into class and continuous categories. Class PTFs predict the hydraulic characteristics on the basis of texture class of a soil, for example silty clay loam normally in tabulated form while continuous PTFs predict the hydraulic characteristics using for example, the actually measured percentages of clay, silt and organic matter content and additional soil variables such as bulk density via a mathematical relationship. It is noted that generally class PTFs are cheap and easy to use because of it is only the textural class of a soil that has to be determined. However, the obtained accuracy is limited because the approach provides one average hydraulic characteristic for each texture class and yet there may be considerable range of characteristics within a single textural class. Continuous PTFs on the other hand, require relatively costly determination of the exact textural composition of a soil. The predicted hydraulic characteristics in this case, are likely to be more accurate because the texture class can be derived from the textural composition. Wösten *et al*., (1995) points out that application of PTFs cannot exist without the direct methods (i.e field sampling/laboratory measurements) since measurements form the basis of derivation and calibration of predicted hydraulic characteristics.

A considerable number of PTFs has been (and still is) proposed over the years. Most of these PTFs predict Brooks and Corey (BC) or van Genuchten (vG) parameters (as defined in the section on water retention) and have been developed using extensive databases for soils of temperate regions. Suprayogo (2003), on testing of pedotransfer functions developed by Wösten *et al*., (1998) for tropical soils concludes that soil bulk density and clay content were the most important input parameters a PTF user needs to obtain valid results for predicting unsaturated hydraulic characteristics. He pointed out that lack of hydraulic conductivity data is a very serious limitation on testing the PTFs in tropical soils. Generally, there is a dearth of tropical soils data. Nonetheless, Hodnett and Tomasella (2002), propose a PTF for these tropical soils. With this however, they make a key point which is also mentioned in (Soet and Stricker, 2003) that the predictive accuracy varies between PTFs and is affected by factors such as soil, climatic conditions and size of the underlying data set and the applied methodological approach.

There are various soil properties affecting water retention and transport of water and chemicals in the soils. Table 1 is a presentation of the properties used most often because of their availability or because they proved to be most promising ones. In this study, particle size, hydraulic characteristics, organic matter and porosity measurements are considered.

According to Wagner *et al*., (2001), the well known and accepted PTFs for the estimation of unsaturated hydraulic conductivity published over the last 15years are Campbell (1974, 1985), Vereecken *et al*., (1989), Gregson *et al*., (1987), three approaches in Rawls and Brakensiek (1989), Wösten (1997) and Wösten *et al*., (1999). A detailed description of each of these models can be found in Wagner *et al*., (2001). Coefficients in the Brooks and Corey equation (λ, θ_r and Ψ_b) can be estimated using the amount of sand, clay and silt as input. The van Genuchten coefficients are not estimated directly but can be assigned from the Brooks and Corey coefficients as $\alpha = 1/\ \Psi_b$, $\beta = 1+\lambda$ and $\gamma = 1\text{-}1/\ \beta$ or in some models (e.g Wösten, 1997), they are obtained as transformed parameters.

Whereas this approach is cost effective, it must be pointed out that the majority of PTFs are developed from measurements on standard small samples. The soil sample size may affect both average values of soil hydraulic characteristics as well as their spatial distribution (Wösten *et al*., 2001). Although scale effects are documented (as in Nunzio and Santini, 1997), it is still not understood to what extent scale can be incorporated in PTFs. Hence, local application of PTFs developed at the sample scale remains a typical feature of its use.

Whereas it is important to have an understanding of the hydraulic characteristics of the unsaturated zone with respect to water flow, contaminant transport is equally impor-

Table 1: Soil properties often used in pedo transfer functions *(Source: Wösten et al., 2001)*

Particle size properties	Hydraulic characteristics	Morphological properties	Chemical/mineralogical properties	Mechanical properties
Sand, Silt, Clay Very coarse sand, coarse fragments Median or Geometric mean particle size Water-stable aggregates	Water content at -33kPa -1500kPa Reference moisture retention curve	Bulk density Porosity Horizon Structure grade size shape Colour Consistence Pedality	Organic carbon Organic matter CEC Clay type $CaCO_3$ Iron	Penetration Resistance

-tant. This is the subject of the next section. Both these aspects are considered in the modelling effort of this study.

3.3 Contaminant transport

The physical setting of an area usually determines how easily groundwater becomes contaminated if inadequate waste management or improper land use occur. The "physical setting" of a basin includes soil type, characteristics of the subsurface unconsolidated material, depth to bedrock, depth to groundwater, topography, and hydrologic characteristics of the area. The potential for groundwater contamination is determined by land use practices applied to an area in conjunction with the basin's physical setting and source of contamination.

3.3.1 Transport Mechanisms

The extent to which a contaminant moves in the groundwater depends on its behaviour in relation to various processes that encourage transport or serve to retard its movement. The rates at which contaminants applied at the ground surface migrate through the vadose zone and the amounts that are delivered to the saturated zone (shape and speed of contaminant plumes) are therefore determined by these processes and by factors relating to the aquifer materials and contaminant characteristics (Boulding, 1995; Domenico and Schwartz, 1998). This information is of great significance in current modeling efforts, which seek to track the migration of contamination, predict possible fate and migration of contaminants for risk evaluation, evaluate the impact of a remedial action on the groundwater quality as well as design of monitoring networks.

Processes that govern the extent to which contaminants migrate in the subsurface are broadly classified into four categories.

i) Advection: in this process moving groundwater transports the contaminants. This process is the dominant transport mechanism. The advective transport of non-reactive solutes (e.g Chloride, bromide, rhodamine WT dye, electrical conductivity, stable isotopes (^{15}N, ^{13}C), takes into account that flow only occurs in the pores of the geological formation (Xinhao, 1997 and Taylor *et al.*, 2002). This flow termed the seepage velocity (apparent velocity) is given as:

$$v = \frac{q}{n} \tag{12}$$

Where q is the specific discharge and n, the porosity.

The porosity (the ratio of the void volume to the total volume of soil material) can range from 0 to more than 60%. However, it noted that not all of the pores participate meaningfully in the flow of water as well as conservative contaminants hence the concept of effective porosity (ratio of volume of the interconnected interstices to the total volume of the soil or rock). In some cases, the effective porosity can differ significantly from the total porosity (Schwartz and Zhang, 2003).

ii) Dispersion: This is a result of non-uniformity of groundwater flow paths as well as variations in groundwater velocities within the flow paths. Dispersion influences the evolution of contaminants in two ways: a) It dilutes the pollutant and reduces the concentration peaks and b) It accelerates the arrival of the contaminant in relation to the arrival of the same contaminant assumed immiscible (Robins, 1998).

This process also known as mechanical dispersion is usually handled mathematically in the same way as turbulent diffusion and dispersion in surface water; Fick's first law is assumed to apply. In one dimension the dispersion coefficient is often approximated by

$$D = \alpha * v \tag{13}$$

Where D = mechanical dispersion coefficient [L^2/T]; α = dispersivity of the aquifer approximately equal to the median grain diameter (Φ) of the aquifer solids [L]; v = seepage velocity [L/T]

The dispersivity varies from 0.1 to 10m. Field and laboratory tests have indicated that it varies with the scale of the test. However, the most reliable dispersivity values are all at the low end of the range. Large-scale tests have higher α than small laboratory column tests. An approximate value for α is 0.1 times the scale of your system (Fetter, 1999). In 2- or 3-dimensional flow, mixing occurs not only along the axis of flow but also along axes perpendicular to the flow. Longitudinal dispersion (α_L), which occurs in the direction of seepage velocity, is normally larger than dispersion perpendicular to flow (α_T).

iii) Diffusion: This refers to movement of the contaminant from areas of high contaminant concentration to areas of low contaminant concentration (Fick's law, Schnoor, 1996). This spreading of solutes occurs both in the direction of groundwater flow (longitudinal dispersion) and perpendicular to the direction of flow (transverse dispersion).

As mechanical dispersion ($\alpha * v$) and diffusion (D^*) both serve to disperse solutes, the processes are combined mathematically to give what is known as hydrodynamic dispersion (D) (the total Fickian transport coefficient of a pollutant) expressed as:

$$D = \alpha v + D^* \tag{14}$$

It is important to note that mixing by molecular diffusion of pollutants dissolved in pore waters, always occurs even if mechanical dispersion becomes zero as a consequence of no seepage velocity. For most mass transport problems that involve migration of dissolved contaminants in aquifers, the effects of mechanical dispersion are much greater than the diffusive component. Thus the overall dispersion is described in terms of aquifer dispersivity or α values. Only in units with low hydraulic conductivity and small advective velocity will diffusion become important (Schwartz and Zhang, 2003). This may also be important during long periods of no infiltration (Marshall *et al.*, 2000).

iv) Retardation: This principally refers to chemical mechanisms that occur during advection which tend to slow down the rate of contaminant migration. This applies to non-conservative solutes (oxidizable organic matter, nutrients such as nitrates and phosphates, bacteria). Of great importance is the sorption process in which the contaminant is adsorbed onto the subsurface media (soil) and biodegradation/decay during which organic contaminants undergo degradation as a result of microbial activity under aerobic/anaerobic conditions (McCarty and Roberts, 1990; Azadpour-Keeley *et al.*, 1999).

Of particular interest in this study is ortho phosphorus and faecal colifom retardation. However, for this thesis, phosphorus retardation was investigated and hence the review is limited to this.

3.3.2 Phosphorus retention

Soil phosphorus (P) retention capacity is greatly affected by the soil mineralogy and pH. Phosphorus removal processes are noted to be mainly *adsorption* and *precipitation* reactions (Sloot, 1996). Microbial populations in the soil contribute to both mineralization and immobilisation of phosphorus. However, this process plays a minor role in phosphorus regulation especially in areas with high phosphate inputs (Sobehrad, 1997).

To understand the phosphorus removal mechanisms of importance, there is need to clarify some terms. *Adsorption* is the selective accumulation of a chemical at the interface between two phases. The focus in this study is on adsorption at the interface between an aqueous solution and a solid. *Desorption* is the reverse of adsorption; i.e., it is the release of an adsorbed substance to the bulk solution. The substance that adsorbs in this case, phosphorus (from the water) is the *adsorbate* and, the solid at which it binds (the solid being at the solid/liquid interface), is called the *adsorbent*. In principle, adsorption - a surface or interfacial process - can be differentiated from *absorption*, in which a substance is transferred from one phase into the interior of another. However, it is often almost impossible to distinguish between adsorption and absorption, so the two processes are sometimes treated jointly. In such cases, the terms defined above are written as *sorption*, *sorbate*, and *sorbent*, respectively. For the purposes of this study sorption will be adopted. However, it is noted that under high P concentrations or longer periods of time, sorption can also include some precipitation processes where a new solid phase is formed (Börling, 2003)

Phosphorus sorption on the subsurface media is dependent on concentration of the potential adsorbents (containing Al, Fe, Mn, Mg and Ca oxides and their compounds). P may also be removed by humic substances (Al and Fe may form bridges between organic ligands and P ions). However, presence of humic substances (organic carbon), may negatively affect P sorption onto the Fe sorbents present in the subsurface material since these compete with P for Fe. The ability of organic acids to reduce P adsorption was demonstrated by Hue (1991). The particle size distribution of a soil can significantly affect the phosphorus adsorption characteristics. The smaller size fractions of soil, clay and silt have a large surface area and tend to absorb more phosphorus than sand particles. Changes to the soil environment, which may cause changes in the phosphorus flux, include redox potential, pH, dissolved oxygen, salinity, competing anions, elevated nitrate concentrations and changes to the hydrology of the soil system. The pH of water strongly influences P adsorption especially as it controls the solubility of P bound to Fe, Al and Ca. In acid soils, (pH range of 5.8-6.8) P deposition on Fe (III) and Al (III) hydroxides is favoured. In neutral or alkaline soils and sediments, precipitation reactions dominate P removal reactions resulting into the formation of calcium phosphate precipitates (Kansiime & Nalubega, 1999; Mann, 1996).

3.3.2.1 Kinetic models for phosphorus adsorption in soils

The removal of phosphorus by adsorption is normally modeled using isotherm equations. An adsorption isotherm or curve is a graphic representation showing the amount of solute adsorbed by an adsorbent as a function of the equilibrium concentration of the solute (USEPA, 1992). This relationship is quantitatively defined by some type of adsorption isotherm equation. The two most commonly used and simplest of these adsorption isotherm equations are the Freundlich and Langmuir isotherms, which were considered in this study.

The use of a particular model in view of the complexity associated with P adsorption characteristics is specific to the experimental situation under investigation. Phosphorus adsorption characteristics of soils may be modeled in laboratory studies by equilibrium batch experiments to simulate field conditions (Mann, 1996).

The Freundlich Equation

This is given as (USEPA, 1992):

$$\frac{x}{m} = K_f C^{1/n} \tag{15}$$

Where x is the amount of the solute absorbed, m is the mass of the adsorbent, C is the equilibrium concentration of the solute, and K_f (related to the capacity or affinity of the adsorbent) and 1/n (an indicator of the intensity of adsorption or how the capacity of the adsorbent varies with the equilibrium solute concentration) are constants. The 1/n term has no units while the selection of the units for x/m and the equilibrium solute concentration will determine the units of K_f in a given situation.

This equation is frequently used due to its simplicity. It contains two constants both of which are positive value numbers that can be solved statistically when expressed in logarithmic form:

$$\log\left(x/_m\right) = \log K_f + 1/_n \log C \quad (16)$$

Hence the constants K_f and $1/n$ in eq. (15) may be solved via eq. (16) as a simple linear regression,

$$y_i = a + bx_i \quad (17)$$

Where $\log(x/m)_i = y_i$; $\log K_f = a$; $1/n = b$ and $\log C_i = x_i$.

Two major drawbacks reported in using the Freundlich equation are: 1) it cannot be extrapolated with confidence beyond the experimental range used in its construction, and 2) it will not yield a maximum capacity term, which in many cases is a convenient single value number that estimates the maximum amount of adsorption beyond which the soil is saturated and no further net adsorption can be expected (USEPA, 1992).

The Langmuir Equation

There are several Langmuir type expressions that have been widely used to describe adsorption data for solid-liquid systems. The most commonly used expression may be generalized as (USEPA,1992):

$$\frac{x}{m} = \frac{K_l C_{\max} C_{eqm}}{1 + K_L C_{eqm}} \quad (18)$$

Where x is the amount of the solute absorbed, m is the mass of the adsorbent, C_{eqm} is the equilibrium concentration of the solute, and K_L (related to the bonding energy between the adsorbed ion and the adsorbent) and C_{max} (maximum amount that an adsorbent can retain) are constants. The selection of units for x/m and the equilibrium concentration determines the units for C_{max} and K_L.

These equations are frequently used because of their ease of application and like Freundlich equation, they contain only two constants, both of which are positive value numbers. The latter can be statistically solved when eq. (16) is cast in linear form giving rise to the traditional linear and double reciprocal Langmuir eqs. (19 and 20) respectively.

$$\frac{C}{x/_m} = \frac{1}{K_L M} + \frac{C}{M} \quad (19)$$

$$\frac{1}{x/_m} = \frac{1}{K_L MC} + \frac{1}{M} \quad (20)$$

To understand the phosphorus adsorption characteristics of the soil under study, the maximum P adsorption capacity is often compared to the physical and chemical attributes discussed in section 3.3.2.

The Langmuir constants can be obtained from these linear forms using the same technique used to solve the linear form of the Freundlich eq. (16). However, it is recommended that to obtain more accurate values of the isotherm constants, nonlinear regression (nonlinear least squares) should be used than linear regression (USEPA, 1992). These constants when obtained are used in describing the partitioning of the chemicals between soils and water and have been used successfully as input parameters in many models describing the movement of chemicals in soil.

3.3.2.2 Concluding remarks

The adsorption of phosphorus by soils is rather complex. Many factors as mentioned affect the reactions of phosphorus. However, in this study, use is made of adsorption models as a means of assessing the capacity of the soils to retain phosphorus and provide an insight into the likely transport of this contaminant to the shallow groundwater. The adsorption characteristics of the soils are compared to measured physical and chemical soil parameters to explain any relations (in the environment under study) these may have on phosphorus sorption.

The adsorption of phosphorus by soils is rather complex. Many factors as mentioned affect the reactions of phosphorus. However, in this study, use is made of adsorption models as a means of assessing the capacity of the soils to retain phosphorus and provide an insight into the likely transport of this contaminant to the shallow groundwaters. The adsorption characteristics of the soils are then compared to measured physical and chemical characteristics to explain any relations (in the environment under study) these may have on phosphorus adsorption.

Whereas the adsorption models may have some limitations (variation in sorption sites, number of competing ions, changes in pH and ionic strength etc), this study investigates the use of both Freundlich and Langmuir (to obtain a maximum P adsorption capacity) isotherms with a range of P concentrations found in the shallow groundwaters with time, as these help in comparing the suitability of the various soils (in the area under study) to adsorb phosphorus and hence protect the groundwater.

The following section is a summary of the pertinent investigations that one needs to undertake to be in a position to conceptualise the flow dynamics and contaminant transport in the subsurface system.

3.4 Contaminant transport and related hydrogeological investigations

From the aforementioned (previous sections), it is clear that the subsurface movement of potentially polluting substances deposited on the ground surface is strongly influenced by the physical setting of an area if inadequate waste management or improper land use occur. The physical setting in this context includes soil type, characteristics of the subsurface unconsolidated material, depth to groundwater, topography and hydrologic characteristics of the area.

This is of particular importance if one is to understand and be in a position to model contaminant transport within an area. Table 2 highlights the chemical properties that characterize the subsurface and their influence with respect to its attenuation capacity. A summary of the physical processes within the vadose and saturated zones and related applications with respect to contaminant dynamics is given in Table 3. Included are the various methods that can be used to obtain the information on these processes.

From Tables 2 and 3, it can be seen that the physical nature of the subsurface materials and related hydrology does indeed play an important role in contaminant transport while chemical properties of the subsurface influence their retention potential (attenuation) of contaminants. The information generated from the field and laboratory measurements of these aspects constitute the data that is utilized in model formulation. The latter is the subject of Paper III (Appendix) while the next chapter describes the approaches that were used in order to address the objectives of this study.

Table 2 Chemical properties that influence the attenuation potential of the soil *(USEPA, 1995)*

Parameter	Remarks
Cation exchange capacity (CEC)	CEC is the degree to which a soil can adsorb and exchange cations at a given pH. Exchangeable cations are held mainly on the surface of colloids of clay and humus. Hence it is highly dependent on the soil texture and organic matter content.
Organic matter content (%Organic carbon content)	Influences contaminant degradation and chemical transformation mechanisms (dissolution and precipitation, surface adsorption and exchange)
Soil pH	This influences the soils' sorption potential for contaminants (especially polar substances).
Mineral content (cations) and clay type	The relative concentrations of cations (Al, Ca, Mg, Fe, Na etc) in soil solution help determine the degree of adsorption. The adsorption potential of the soils also vary with the clay types

Table 3 Subsurface physical processes, determination methods and application in contaminant transport *(NFESC, 1999; USEPA, 1995; Shwartz and Zhang, 2003)*

Parameter	Methods	Application
Direction and gradient of groundwater flow	Delineate piezometric surface using water table elevations measured from monitoring wells and/or piezocone data; conduct aquifer pumping or slug tests; flowpath modeling	Estimate rate and direction of contaminant migration
Hydraulic conductivity (saturated and vadose zone)	Conduct aquifer pumping or slug tests; run piezocone dissipation tests; conduct vadose zone permeameter and infiltrometer tests; tensiometer tests; evaluate soil moisture content and characteristic curves to determine unsaturated hydraulic conductivity	Estimates rate of contaminant migration; definition of zones of conservative migration
Vadose zone moisture content	Evaluate by use of neutron probe; TDR probe; tensiometer; calcium carbide gas pressure test; direct heating	Estimates vadose zone hydraulic conductivity; microbial moisture requirements
Lithology	Evaluate by field logging; cone penetrometer tests; piezocone strain gauge and pore pressure dissipation tests; structural geology; soil density tests, permeametry; aquifer pumping and slug tests; hydraulic conductivity; piezometric surface mapping	Identification for preferential flow paths, confining layers, and potential for vertical migration
Vadose zone permeability	Measure using permeameters; tensiometers; evaluate soil moisture content and characteristic curves to determine unsaturated permeability	Estimates contaminant migration potential; determine whether or not contaminants can or have migrated from the vadose zone to an aquifer
Precipitation and seasonal water table fluctuations	Measure in the field using monitoring wells; piezocone geophysical properties; consult meteorological data base	Evaluation of variability of contaminant concentrations, variation in flow direction
Infiltration	Hydrograph analysis (rainfall-runoff measurements from a watershed; Infiltrometers (double ring, sprinkler, Tension and furrow)	Estimates the hydraulic loading/flow rate of the contaminants into the subsurface
Grain size distribution	Measure using sieve analyses; hydrometers (sizes <0.075mm)	Estimates porosity, unsaturated and saturated hydraulic conductivity, partition coefficients (in some cases)
Soil bulk density	Measure using cone penetration tests; drive cylinder tests; geophysical properties; sand-cone method; laboratory fluid displacement tests	Estimates groundwater velocity
Soil porosity	Measure laboratory pore volume and specific retention	Estimates groundwater velocity

4 MATERIALS AND METHODS

4.1 Introduction

In this chapter, the methods used to achieve the objectives of the study are presented. These included field surveys and consultations, observation well installation, water and wastewater quality monitoring, water level measurements, rainfall data collection, soil characterization (physical and chemical parameters), laboratory batch experiments and tools used for data analysis.

4.2 Field surveys and consultations

Field surveys were undertaken in Bwaise III Parish to assess the environmental sanitation of the area and to identify and locate the various pollution sources. Record sheets were used to note the source types, location and operational status of the facilities. The latter was obtained from consultations with the residents and local authorities. These included: number of users, desludging frequency and methods (for pit latrines), and management issues (for solid waste dumps). Information on the nature of the solid waste, number of pit latrine stances, nature of sullage drains (lined or unlined, flowing or stagnant) was obtained through surveys.

The location of the pollution sources was geo-referenced using a Geographical Positioning System (GPS) (Garmin etrex vista). The resulting coordinates of the various sources were transferred to a digitized map of the study area, which was obtained from the Department of Surveying, Entebbe, Uganda.

4.3 Monitoring (Observation) well installation

Currently there are no installed boreholes in Bwaise III. The area has only one operational protected spring located in Katoogo zone. Therefore to ascertain flow characteristics,

Fig. 8 Monitoring well locations in the zones of Kalimali and Bokesa in Bwaise III Parish (Note the high density of housing)

temporal and spatial water quality and water level variation of the shallow groundwaters, a total of 16 observation (monitoring) wells were installed in two zones within the Parish (Kalimali and Bokesa) so as to have representative coverage of these zones. These were geo-referenced using a GPS and transferred to a digitised map of the area. The wells were code named MW1 to MW 16 (Fig. 8). The wells were augured using a 6-inch diameter hand augur to depths of between 1.6-2m depending on the water table depth at the time of auguring. This activity was undertaken during the dry season (months of January to February) when the water table was lowest making it easier for installation of the wells. The design and installation were according to USEPA (1995) and current practice of observation well construction in Uganda (consultations with Eng. Callist Tindimugaya, Principal Hydrogeologist, Directorate of Water Development, Entebbe) (Fig. 9).

The wells were developed and purged (until relatively clear water was obtained) during the first week after installation prior to collection of water samples the following week. At 4 selected sites within these zones (at MW3, 6, 8 and 9), 3 wells were installed at distances of about 1.2 m apart down gradient along a presumed flow line so as to assess lateral attenuation potential of the soils.

4.4 Water and wastewater sampling

To assess the seasonal variation of the water quality, water samples were collected from the installed wells weekly during the first dry and rainy seasons (March to September 2003) and thereafter monthly for the following dry and rainy season respectively (October to

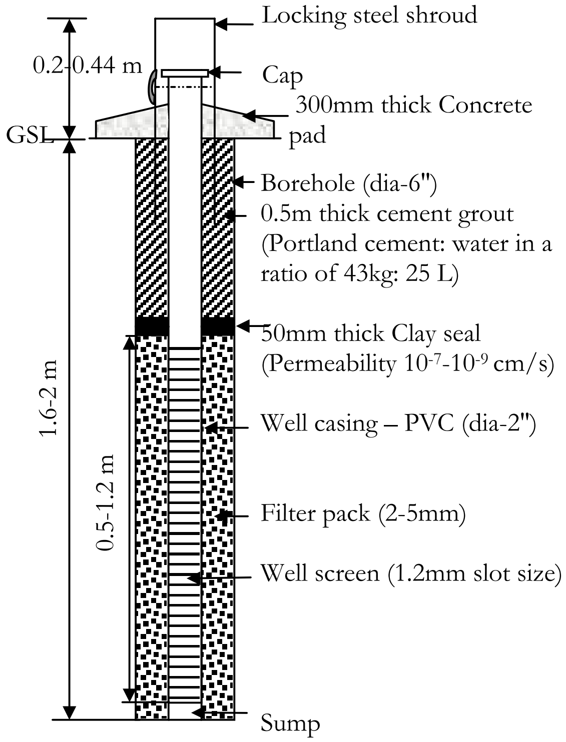


Fig. 9 Monitoring well internal features (left) and installed well (right)

December 2003 and January to December 2004). Water samples were collected by manual pumping with the intake placed in the middle or slightly above the middle of the screened interval. With only one pump available for water sample collection, to achieve decontamination, the water in the pump was discharged before lowering of the intake in the next well. Water was then pumped out for some time so as to clean out the water from the previous well on the inside walls of the pump. Samples were collected after a discharge of about two to four well volumes since the recovery rates were quite fast so as to obtain representative groundwater samples (EPA, 1991). Bottles were rinsed at least twice before sample collection. Samples were collected in 1-litre acid-rinsed and sterilized plastic bottles for chemical and microbiological analysis respectively. The samples were stored in a cool box at 4°C (with ice packs) before delivery to the laboratory for analysis. The latter was undertaken in the Public Health and Environmental Engineering Laboratory of the Department of Civil Engineering, and Animal science laboratory of the Faculty of Agriculture, Makerere University. Prior to storage, the pump was cleaned with distilled water as a way of decontamination in preparation for the next sampling round.

Concomitantly with the above, water samples were collected from an operational protected spring within the area (located in Katoogo zone) to assess the seasonal variation in quality despite the protection against contamination.

To ascertain the characteristics of wastewater carried by the sullage/stormwater drains within the area, wastewater samples were collected from three (3) selected sullage/stormwater drains within the area. These were code named SD1 to SD3. These are secondary drains (recipients of wastewater from tertiary drains within the area) discharging into main drains (Nsooba and Nakamiro drains, Fig. 8). Due to the intermittency of flow in most of these systems, sampling was undertaken regularly from a flowing drain (SD3 within the zone of Kalimali) in the period 2003. This was done weekly during the rainy season (March to April & September) and dry season (July to August) and thereafter monthly until December 2004. To assess the spatial variability of the wastewater carried by these drains with season, samples were also collected monthly from drains SD1 (within the zone of Kalimali) and SD2 (within the zone of Bokasa) from July to December 2004.

Rainwater samples were collected over a short period (April to May 2003) near the Faculty of Technology, Makerere University which is about 1.5km from the study area to assess the contribution (if any) to the pollutant loads to the shallow groundwaters.

4.5 Rainfall data collection

There is no installed weather station in Bwaise III. Rainfall data used in the study was for the Makerere University Weather Station about 2km from Bwaise III Parish. This was collected from the Meteorology Department in Kampala for the period 1991 to 2004 (total daily rainfall records available). Rainfall data for the period 2003-2004 taking into consideration the sampling dates, was used to assess the impact of the rains on water and wastewater quality, spring discharges and water levels in the wells.

4.6 Soil sampling

Information on the nature of the soils at different locations in the area was obtained both qualitatively and quantitatively. For the former, a description of the soil profiles was developed during well installation through observation and "feel" methods. For quantitative measurements, trial pits (about 1.5m wide by 1.5m long and 1m deep) were excavated at locations close to selected monitoring wells for collection of both disturbed and undisturbed soil samples from the different layers using cores for analysis of selected physical and chemical parameters that affect the groundwater flow and transport of contaminants as in section 4.7.2. Soil analysis was undertaken in the laboratories of Soil Mechanics, Faculty of Technology and Soil Science, Faculty of Agriculture at Makerere University.

4.7 Field measurements and Laboratory tests

4.7.1 Insitu-measurements and laboratory water and wastewater quality analysis

Insitu measurements were carried out for the determination of the spring discharge, water levels, Temperature, pH, electrical conductivity (EC), dissolved oxygen (DO) and flow in one of the flowing drains (SD3). These were undertaken at the same period as the water and wastewater sampling activities (section 4.4).

The discharge rate of the spring was determined by the time it took to fill a bucket of a known volume (10.5litres). The average time for three trials using a stopwatch was recorded and used to calculate the rate of discharge from the spring in litres per second. The flow in the drains was estimated using the flotation method.

Water level measurements were taken in each of the wells using a water dip (W7-T-15-50 water level meter-50m flat tape, Soil instruments Inc.) with an audio device before sample collection. At each of the wells, water levels were measured against a mark on the tip of the metallic casing as a measurement reference point (a) whose distance from a marked point on the ground surface (b) was also determined. The water level below the ground surface was determined as (a-b) metres. Electrode probe field meters: WTW TA 197pH/Temperature, HANNA instruments HI 9033 multi range conductivity, and Camlab Handylab OXI oxygen meters were used for measurement of temperature, pH, EC and DO respectively.

Laboratory analysis on the collected water samples was carried out for Thermotolerant (faecal) coliforms, *Faecal streptococci* (on MW1-9 and spring only due to logistic problems), Total Phosphorus, Nitrates, Total Kjeldahl Nitrogen (TKN), Chlorides and Sulphates determination. For wastewater samples, additional analysis was undertaken to determine 5-day Biochemical Oxygen Demand (BOD_5), and Chemical Oxygen Demand (COD). Standard methods for water and wastewater analysis ([1]APHA/AWWF/WEF, 1998) and analytical procedures using the DR/2010 Spectrophotometer as described in [2 & 3]HACH (1997, 1997/8) were followed for the determination of the mentioned parameters (Table 4). Precision and accuracy of the spectrophotometer and the procedures for sample analysis were counter checked by

Table 4 Analytical Methods for chemical and microbiological parameters

Parameter	Methods
Thermotolerant coliforms (TTCs)	Membrane filtration and growth on membrane lauryl sulphate broth (MLSB, Oxoid, UK) [1]
Faecal Streptococci (FS)	Membrane filtration and growth on membrane enterococcus agar (Slanetz and Bartley Agar, Oxoid UK) [1]
Total phosphorus	Ascorbic acid with persulfate digestion[1]
Total Kjeldahl Nitrogen (TKN)	Macro-Kjeldahl[1]
Nitrates	Cadmium reduction[2]
Chlorides	Mercuric Thiocyanate[2]
Sulphates	Turbidimetric (Sulfaver method) [2]
COD	Closed Reflux Colorimetric[1]
BOD_5	BODTrak method[3]

determining the concentration of known standards.

4.7.2 Soil sample analysis

4.7.2.1 Chemical parameters

Chemical parameters determined were pH, Organic matter, available phosphorus, total nitrogen, metallic content (Ca, Na, K, Fe and Mg) and Cation exchange capacity (CEC). Aluminium was not determined due to analytical limitations. Analytical procedures followed were according to Okalebo *et al.*, (2002).

pH - The soil pH was determined from a 1:2.5 soil-distilled water suspension. The mixture was shaken for 30 mins and left to stand for about 20 mins at room temperature before measurement using a pH metre.

Organic matter - The Walkley Black method was used in the determination of the organic matter content of the soils. About 0.3 g of air-dried soil was digested using sulphuric acid followed by titration with potassium dichromate.

Available phosphorus - This was determined by the Bray 1 method. About 2g of air-dried soil were used and extraction undertaken by 14 mls of Bray 1 solution by shaking on a vortex for about 2 minutes. The determination of phosphorus was by readings of absorbencies on a Jenway 6105 spectrophotometer.

Total Nitrogen - This was determined by the Kjeldahl method. About 0.2 g of air-dried soil was digested in a pre-heated digestion block at 360ºC until a clear solution was obtained. Hydrogen peroxide was then added and after cooling, the solution was diluted to 5 mls before distillation.

Metallic content and CEC analysis - These were determined using the leaching method. A 1:20 soil to extracting solution was used. 5g of air-dried soil was put in a leaching column to which 100 mls of the extracting solution (1M ammonium acetate mixed with 96% ethanol in a ratio of 1:1 by volume) was poured. The resultant leachate extracted from the column at a rate of about 2 drops per second was collected and analyzed for Na, K, Ca, and Mg using an atomic absorption spectrophotometer.

Following from the above, for the determination of CEC, 200 mls of sodium acetate was poured onto the soil in the leaching column (to introduce sodium ions onto the free sites of the soils) and allowed to discharge from the column at the same rate as before. The collected leachate was discharged and the excess sodium introduced onto the soil washed with 100 mls of 96% ethanol. The adsorbed sodium was extracted with only ammonium acetate and the leachate discharged at a rate of 2 drops per second. This leachate was collected and analysed for sodium using a flame photometer. The sodium readings (concentrations) obtained give the CEC of the soils.

Extraction for Fe analysis was undertaken with a chelating agent. 40 mls of DTPA extractant (composed of Triethanolamine Diethylenetriaminepentaacetic acid and calcium chloride) were added to 20 g of air-dried soil. The mixture was shaken for 2 hours after which it was filtered and analyzed for Fe using the atomic absorption spectrophotometer.

4.7.2.2 *Physical parameters*

This involved the determination of particle size analysis, specific gravity of the soil particles, bulk density, moisture content, water retention characteristics, and total porosity.

Specific gravity determination of the soil particles -This was determined according to BS1377 (1990). Air-dried soil samples were riffled to about 50 g of the sample. This mass was passed through a 2 mm sieve and the material that passed through was collected and riffled to 5 g samples in duplicates. The latter was oven dried at 105ºC for 24 hrs and then cooled in a dessicator. The determination of the specific gravity was then undertaken on this oven dried mass. This was calculated as:

$$G_s = \frac{W_2 - W_1}{(W_4 - W_1) - (W_3 - W_2)} \qquad (21)$$

Where W_1 is the mass of the empty stoppered density bottle (pycnometer), W_2 is the mass of the pycnometer with soil sample after oven drying, W_3 is Weight of pycnometer filled with water and soil and W_4 is the mass of pycnometer filled with water

The values obtained were used in particle size determination (fine particle size analysis using the hydrometer method).

Particle size distribution **-** Particle size analysis was determined by the wet/dry sieving, and sedimentation hydrometer method according to BS1377, 1975. The hydrometer results for the samples were used in extrapolation of the grading curves into the fine size fractions of the soil samples. The hydrometer type used was ASTM 152H.

Air-dried samples riffled to 50-70 g were used for the hydrometer analysis. The soils were subject to pre-treatment using Hydrogen peroxide (30% v/v) for removal of organic material and hydrochloric acid for removal of calcium compounds/salts (indicated by a soil pH>7 and further, high values of Ca). The mixture was then washed with warm water until no acid reactions on litmus before oven drying and dispersion. The Nomographic chart for solution of Stoke's law was used to obtain the particle diameter, D (mm) using corrected hydrometer readings. The percentage of soil remaining in suspension, K (percentage passing a corresponding D) was calculated from the formula:

$$K = \frac{100G_s}{W_b(G_s - 1)}(R_h + M_t - x) \qquad (22)$$

where W_b is the total dry weight of soil after pretreatment, G_s is the specific gravity of soil particles, R_h is the hydrometer reading corrected for the meniscus reading, M_t is the temperature correction, and x, the dispersing agent correction. The particle size distribution curve obtained for the soil samples was used to deduce the soil texture (percentages of sand (0.075-2.36 mm), silt (0.002-0.075 mm) and clay (<0.002 mm) and hence textural class of the soils using the soil textural triangle. Soil texture was used in estimating soil hydraulic characteristics and assessing its impacts on phosphorus retention (adsorption) according to Mann (1996).

Bulk density - Measurement of this was according to the core cutter Method BS1377, 1975. Undisturbed soil samples were used in the determination of the bulk density. The bulk density was calculated as:

$$wet\ bulk\ density\ (kg/m^3) = \frac{(C_s - C_w),\ kg}{volume\ of\ core,\ m^3}$$

$$dry\ bulk\ density\ (kg/\ m^3) = \frac{C_{sd} - C_w}{volume\ of\ core,\ m^3}$$

Where C_s is the weight of core with the soil sample, C_d is the weight of core with the soil sample after oven drying and C_w is the weight of the empty core. The dry bulk density was used in the determination of the volumetric moisture content for soil water retention measurements.

Moisture content **-** This was determined as:

$$MC\ (\%) = \frac{S_w - S_d}{S_d} * 100$$

Where S_w is the weight of the wet soil and S_d is the weight of the dry soil.

Water retention characteristics

To ascertain the soil hydraulic behaviour, soil water retention measurements for the subsurface in the area were determined using an

Eijkelkamp PF curve standard set (covering the range PF 0-4.2): PF0-2, Sand box, PF 2-2.7, Sand/kaolin box and PF 3-4.2, Membrane apparatus. This data with the help of the COUP model was used to estimate coefficients for the van Genuchten (1980) and Brooks Corey (1964) hydraulic models (section 3.2.3.2), which could then be used for predicting the unsaturated hydraulic conductivity. The obtained soil water retention and the hydraulic conductivity curves were crucial ingredients for predicting water flow and solute transport in the vadose zone (Paper III in the Appendix). For comparative purposes, estimation of these curves was also determined using soil texture (percentages of sand, silt, clay as determined from the particle size distribution analysis) and porosity using the pedo transfer function by Rawl and Brankensiek (1980) in the COUP model.

Undisturbed soil samples were collected near selected monitoring well sites in cores (capacity 100cc, diameter 50mm and height 53mm) in triplicates from the different layers for water retention determinations up to a depth of about 1m. This covered the impermeable layer below the water table (Fig.4). For measurements using the membrane apparatus, semi disturbed samples were collected and stored in polyethlene bags. The disturbed and undisturbed soil cores (covered with plastic lids at both ends) were stored at 4ºC. Preparation of the samples and determination of the volumetric moisture content at the different suction pressures was according to Eijkelkamp operating instructions for the equipment (Eijkelkamp, 2004). Volumetric water contents were determined at suction pressures equivalent to 1, 2.5, 10, 31.6, 63.1, 100, 500 and 2500cm of water column.

Porosity determination -This is an important physical property of soils. The amount of pore space in a soil largely determines how much air and water a given soil can hold and also affects the drainage properties of a soil. The total porosity was measured as the volumetric moisture content at saturation (suction pressure of 1cm of the water column) (Helalia, 1993) of the undisturbed soil cores used for water retention measurements. This was determined in triplicates for each of the soil samples. The obtained values were used in the estimation of the soil hydraulic characteristics in combination with the soil texture as mentioned in the previous section.

The total porosity was then calculated as:

$$Total\ porosity\ = \frac{SC_s - SC_d}{weight\ of\ dry\ soil} * \rho_d$$

Where SC_s is the soil core weight at saturation, SC_d is the soil core weight after oven drying and ρ_d is the dry bulk density of the soil.

4.8 Phosphorus adsorption experiments

To determine the capacity of the soils to retain phosphorus as well as to ascertain whether saturation state of the soils had been reached, phosphorus adsorption batch experiments were undertaken. Microbial reactions were not considered as playing a major role in its removal as high phosphate inputs were encountered in the shallow groundwater within the study area (chapter 5). The experiments were used to derive adsorption isotherm for soils obtained from different profile layers at selected monitoring well sites. Coefficients obtained from the Langmuir and Freundlich adsorption isotherms were then used in model development as presented in Paper III (Appendix).

Soil profiles (about 1m long) were extracted from sites adjacent to selected monitoring wells using fabricated steel cores (about 1.2m long and diameter 0.1m). The extracted soil profiles were air dried at room temperature before being sliced into the different layers. These where then riffled to the required mass, ground and homogenized to a fine powder with a mortar and pestle and then passed through a 2mm stainless steel sieve. The experimental approach followed is illustrated in Fig. 10. Parallel samples (B) were analysed for selected parameters known to affect P adsorption. The analytical tests for determination of these parameters are as described in section 4.7.2.

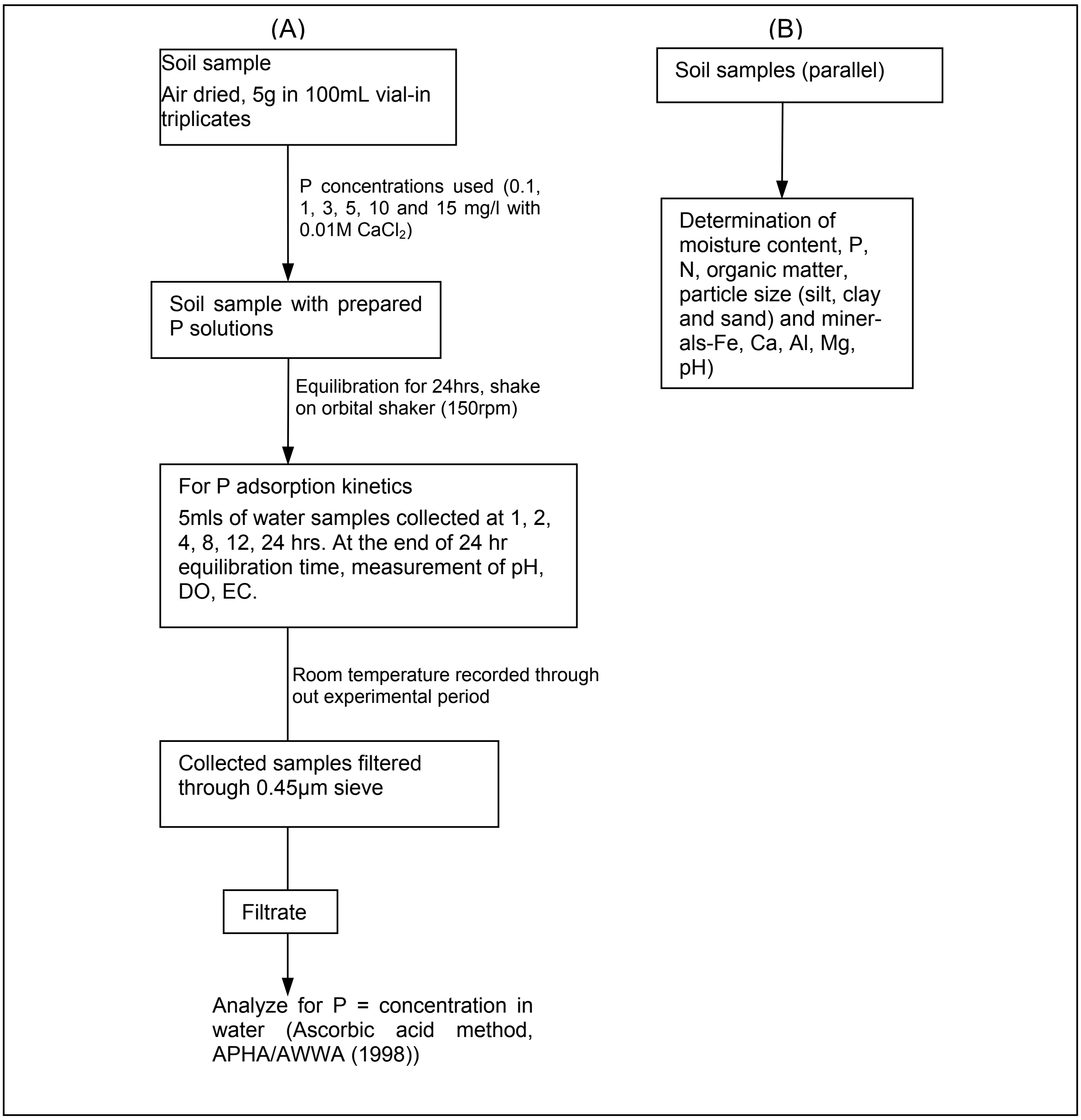


Figure 10 Batch experiment set-ups for P adsorption kinetics and isotherm determination *(after Houba* et al.*, 1995)*

4.9 Data analysis

Several software tools were utilized in the analysis of the data. AUTOCAD was used for spatial location of pollution sources. Slide write was used for development of graphical plots for the analysis of seasonal and spatial water/wastewater quality variations and phosphorus sorption capacity of the different soil layers within the area with time. Comparison of variables was performed using analysis of variance (ANOVA). From the latter differences among means were tested using the standard F-statistics. All the data were first tested for normality and equality of variances using a normal distribution curve plot and Levene's tests. SPSS was used for obtaining coefficients for the Langmuir adsorption isotherm using non-linear regression. Estimation of Freundlich sorption coefficients was by linear regression of the log transformed adsorption values and equilibrium concentrations.

Coefficients of the unsaturated hydraulic conductivity curves of the Brook and Corey (1964) and van Genuchten (1980) unsaturated/saturated flow models were estimated with the COUP[1] model. The model structure is basically a depth profile of a soil and simulates among others, soil water in many types of soils (Jansson and Karlberg, 2004). The estimation of the coefficients was based on the measured water retention data, texture and porosity of the different soil layers.

Model development (Paper III) was undertaken with the Earth Science module in FEMLAB. FEMLAB[2] is a modelling package for simulation of any physical process that can be described with partial differential equations. It features state-of-the-art solvers that address complex problems quickly and accurately, while its intuitive structure is designed to provide ease of use and flexibility. The FEMLAB Earth Science Module consists of a large number of ready-made modelling interfaces for subsurface flow. The module handles transport and reaction of solutes as well as heat transport in porous media (Comsol, 2004).

5 Results

5.1 Pollution source characteristics

The identified anthropogenic pollution sources for shallow groundwaters in the area were excreta disposal facilities, solid waste dumps, sullage/stormwater drains, animal yards, car washing bays, and/or garages (Fig. 11). Judging by the quantities though, it was evident that the major pollutant sources were the excreta disposal facilities (pit latrines), solid waste dumps and sullage/stormwater discharged into unlined drains).

5.1.1 Excreta disposal facilities

Pit latrines are the commonest on-site sanitation systems used in the area (Table 5). The majority of the latrines are elevated because of the high water table. More than 80% of the pit latrines are of the traditional unimproved type and do not meet the basic criteria of hygiene and accessibility to the children and disabled. Subsequently, polythene bags (“flying toilets”) and spaces around the house are used for excreta disposal especially by the children. In these cases, the excreta disposed of ends up in drainage channels and in solid waste dumps.

Poor accessibility to these facilities (due to the dense settlement and inadequate road infrastructure) for desludging by mechanical emptiers and associated costs results in most of the latrine contents being disposed of within the area. This is either into adjacent excavated unlined pits or sullage/stormwater drains adjacent to which they are constructed during the rains. Desludging is carried out at least twice a year for the majority of these systems. Hence all the waste generated in most of the zones accumulates in the envi-

Table 5 Population and excreta disposal facility (pit latrines) distribution in the different zones in Bwaise III

Zones	Kalimali	Bokasa	Bugalani	St. Francis	Katoogo	Kawaala
Land area (ha)	2.36	3.88	4.25	6.71	17.25	16.41
No. of Households	258	451	515	580	1191	225
Settlement density (HH/ha)	109	116	121	86	69	14
Total no. of pit latrines	27	62	39	99	117	64
No. of stances	56	134	77	215	249	129
Estimated no. of users	1595	2610	2180	2870	6150	1305
Users per stance	28	19	28	14	25	10
Class 3 latrines[1]	23 (85%)	59 (95%)	36 (92%)	81 (82%)	95 (81%)	38 (59%)
Estimated no. desludged with a mechanical emptier	-	-	-	17 (17%)	10 (9%)	-

[1]These do not meet the basic criteria of hygiene and accessibility to the children and the disabled persons

1 COUP: Coupled heat and mass transfer model for soil-plant-atmosphere system

2 FEMLAB is a trademark of COMSOL AB, Sweden.

ronment and slowly leaches to the shallow groundwater.

The waste load from the latrines is closely associated with the settlement density or density of pit latrines per hectare, the number of people using each pit, and the geologic conditions (Barrett *et al*., 1999). Hence estimated waste loads from the latrines were based on the number of users and the latters' estimated generation of specific contaminants. Pollutants of health and environment interest associated with waste material from pit latrines are nitrates, pathogens and phosphorus.

It is estimated that each person generates about 2.5kg of nitrogen and 0.4kg of phosphorus per year in Uganda based on national statistics of food supply (Jönsson and Vinerås, 2003). According to Feachem *et al*. (1983), the estimated enterobacteria content based on diet for the Ugandan population-which is largely carbohydrate is on average10^8 cfu per gram of human excreta.

The average daily body waste generation is estimated to fall within the range of 1400-1450 gram per person (Chaggu, 2004). Using an average of 1400 gram per person, the estimated pollutant loads from the pit latrines for the different zones are shown in Table 6.

The loads are highest in the zones of Kalimali, Bokasa and Bugalani largely due to the smaller area extent and high settlement density (Table 5). Irrespective of the zone, the bulk of the generated pollutant loads from the excreta disposal systems is retained in the area and impacts on communities' health as well as the environmental sanitation.

Table 6 Estimated nitrogen (N), phosphorus (P) and bacteria loads from pit latrines in the different zones in Bwaise III

Zone	N (Kg/ha/yr)	P (kg/ha/yr)	Bacteria (x1014 cfu/ha/yr)
Kalimali	1690	270	345
Bokasa	1604	269	344
Bugalani	1282	205	262
St. Francis	1069	171	219
Katoogo	891	143	182
Kawaala	199	32	41

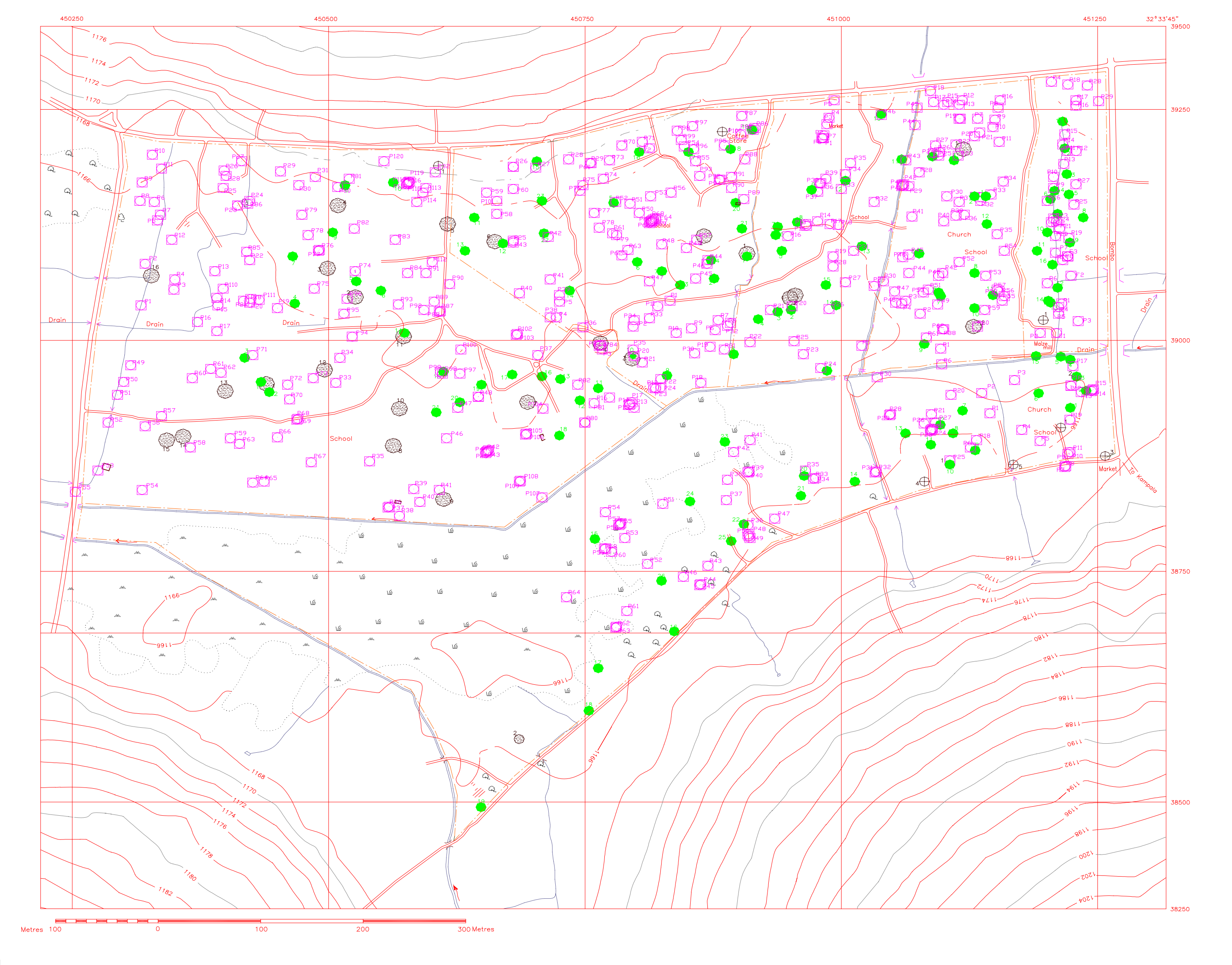
5.1.2 Solid waste dumps

Both biodegradable and non-biodegradable wastes are generated in the area. The biodegradable wastes include food peelings (bananas, cassava and potatoes) generated from homes, commercial eating places, and markets that sell fresh foods while the non-biodegradable include polyethene bags, plastic bottles, plastic bins and other plastic materials used in homes. Other wastes generated are from charcoal sellers, saw dust and timber pieces from carpentry workshops, waste paper and other wastes from institutions like schools.

The area is serviced by one KCC skip, which is located at the Kawempe Division's headquarters, in a neighbouring parish. To avoid the trouble of moving long distances, most of the residents indiscriminately dispose of their waste. The municipal authority has also provided a tractor, which acts as a mobile skip for waste collection 1-2 times a week. This is inadequate and residents who can afford to pay, hire private garbage collectors to remove the waste on agreed days. Efforts by community based organizations and youth groups to improve solid waste management in the area are frustrated by poor attitude (associated with their temporary residential status), non-payment of collection fees by residents that results in a lack of collection equipment (bins, wheel barrows, protection wear etc).

Open dumping is thus the major form of waste disposal in the area. The waste is usually burnt when dry and left to decompose during the wet season. Illegal disposal of waste into stormwater and sullage drains traversing the area is also practiced. Deliberate open dumping is also practiced as a way of wetland reclamation. This has implications for the lithology of the subsurface (section 5.2.4). As mentioned earlier, the dumps are also recipients of human excreta and hence contain pathogens.

As noted above, the bulk of the solid waste material is organic and of household origin. The waste generation rates for the different zones (Table 7) result in a total amount of



Key

Grassy papyrus swamp

Solid waste dumps

Pit latrines

Carwashing bays/garages

Animal feedlots /waste spreading areas

Zone boundaries

Fig. 11 Anthropogenic pollutant source types and locations in Bwaise III

Table 7 Solid waste generation rates for different zones in Bwaise III Parish
(Source: Plan, 2001)

Zone	Average waste generation (m^3/d)	Total daily production (kg/d)*
Kalimali	1.6	720
Bokasa	2.8	1260
Bugalani	3.2	1440
St. Francis	3.6	1620
Katoogo	1.4	630
Kawaala	7.4	3330

*The presented values refer to unrecycled waste with a wet density of 450 kg/m^3

9000 kg/d of waste production in the area. This amount is significant, and as has been observed, is an environment hazard necessitating proper management.

Previous studies in the area show that of the dry wastes generated, 80-90% are biodegradable, 0.4% metal, 2.5% plastics and polyethene bags, 1.1% wood wastes, 1.7% paper and 7.4% others (Nyenje, 2002 and Lwasa, 1999). The estimated phosphorus and nitrogen loads generated from the organic portion of the solid wastes within the area are based on an average composition of 1.1% nitrogen and 0.2% phosphorus of the dry matter content (this ranges between 16-18%, Kasozi et al., 2004), which on average is taken as 17%, and are shown in Table 8.

Table 8 Nitrogen and Phosphorus loads from folid waste for different zones in Bwaise III Parish

Zone	Nitrogen (kg/d)	Phosphorus (kg/d)
Kalimali	1.08	0.20
Bokasa	1.88	0.34
Bugalani	2.15	0.39
St. Francis	2.42	0.44
Katoogo	0.94	0.17
Kawaala	4.98	0.91

5.1.3 Sullage/stormwater drains

As noted earlier, most of the households do not use water borne-sanitation. As a result, most of the sullage water is directly disposed of into the environment via unlined sullage/stormwater drains and open spaces. These drains operate as combined open sewers (Table 10) carrying wastewaters with very high COD, 5-day BOD and thermotolerant values well above the national effluent discharge standards.

Discharges during the dry season were very low (almost stagnant) in the drains with siltation and solid waste disposal. Thus velocity determination using the flotation method was impossible for most of the time. Besides, the flows are intermittent during the dry season. The relatively high discharges realised following the rains are a result of mostly the stormwater runoff. During the wet season, it is noted that a lot of stormwater carried by the main drains collects and stagnants in the area carrying with it lots of pollutants (e.g., thermotolerant coliforms, TKN, total phosphorus, etc) and hence the impact of these drains on the shallow groundwater may actually be greater than envisaged. Not only is there infiltration of contaminants but also a rising water table, which merges with the drain wastewater under flooding conditions.

The stagnation in these systems during periods of no rains results in pollutants gradually leaching into the shallow groundwater. Wastewater carried by the drains during the dry season is more indicative of sullage characteristics while during the wet season it is a combination of sewage, sullage and contaminated stormwater runoff from this and neighbouring areas.

Using the per capita wastewater generation (12 litres per day)[3], population size from each zone (no. of households-Table 5 x household size of 5) and median concentrations during the dry season (Table 10), the estimated loads for the area are obtained (Table 9).

5.2 Shallow groundwater characteristics

In this section, results on the protected spring (Fig. 5) discharges, groundwater levels from the monitoring wells (Fig. 9) and water

[3] Considering that no wastewater enters the sewers, the per capita wastewater generation rate is taken as 80% of the water consumption (Punimia and Ashok, 1999). Daily per capita water consumption rate used is 14litres (water consumption in informal settlements is in the region of 12-15 litres/person/ day, Pers. Comm. Engineer Sonko Kiwanuka, National Water and Sewerage Corporation, Kampala)

Table 9 Estimated total kjedahl nitrogen and phosphorus loads from sullage drains in Bwaise III

Zone	TKN (kg/yr)	Phosphorus (kg/yr)
Kalimali	184	11
Bokasa	321	18
Bugalani	367	21
St. Francis	413	24
Katoogo	850	49
Kawaala	161	9

quality characteristics from these sources as well as rainfall influence on these aspects is presented.

5.2.1 Monitoring well water level variations

Water levels within the monitoring wells varied spatially and temporally (Fig. 12 & 13). The water levels were significantly different at the well locations (P=0.000) and ranged from –1.19 m to 0.22 m. The positive values are indicative of flooding conditions that is, risen water table level above ground. The latter occurred with short-term rains (48hr) ≥ 50 mm.

There is a strongly positive correlation between the short rains and the water table depths at most of the well locations (Table 11). This suggests a quick recharge and highly pervious subsurface material. However, rainwater has very low concentrations of contaminants (Table 12) and hence contributes more towards driving surface contaminants into the shallow groundwaters in the area and dilution of most pollutant concentrations (Tables 10, 13, 14 & 15).

A plot of the average water level isolines (Fig. 14) depicts flow patterns closely related with the groundslope. It suffices to say that the dense housing noted in Fig. 8 also influences the flow. The high water table and hence shorter depth of the unsaturated zone and related soil characteristics (section 5.2.4) greatly influence the attenuation capacity of the subsurface in the area and subsequent water quality characteristics (section 5.2.3).

5.2.3 Spring discharge variations

There is a variation of spring discharge with time (Fig. 15). The discharge ranged from 1.22 to 1.48 m^3/hr (Table 13) and did not vary significantly with season (P=0.087). This implies that the spring has two recharge mechanisms - regional baseflow that sustains the high yields throughout the dry seasons and seasonal interflow during the rains.

Table 10 Seasonal variation of sullage/stormwater quality

WET SEASON	pH	EC µS/cm	Total-P mg/l	TKN mg/l	NO^-_3 mg/l	Cl^- mg/l	TTCs cfu/100ml	COD mg/l	BOD_5 mg/l
Mean	7.58	406	4.92	80.93	8.40	45	1562E4	589	374
Standard error	0.10	34	1.58	27.83	1.15	4	640E4	162	116
Median	7.71	338	1.80	19.73	7.40	39	210E4	320	165
Range	6.93 -8.04	250 -634	0.61 -27.50	2.04 - 470.75	1.30 -20.50	26 - 78	(34 -8400) E4	41 - 1585	27 - 1115
Count	13	13	23	23	17	14	17	12	12
95% CI of the mean	0.21	74	3.27	57.71	2.43	9	1358E4	356	256
Discharge (m^3/s)	0.042 to 0.6 (n = 7)								
DRY SEASON	**pH**	**EC µS/cm**	**Total-P mg/l**	**TKN mg/l**	**NO^-_3 mg/l**	**Cl^- mg/l**	**TTCs cfu/100ml**	**COD mg/l**	**BOD_5 mg/l**
Mean	7.42	545	6.50	111.55	7.10	142	939E4	757	460
Standard error	0.14	32	2.17	30.53	0.71	31	312E4	174	121
Median	7.60	546	1.87	32.58	7.80	72	390E4	914	401
Range	6.58 -7.94	282 - 713	0.74 -33.66	0.60 - 459.48	1.00 - 13.20	33 - 450	(3.9 - 4300) E4	121 - 1482	32 - 1320
Count	13	11	19	21	15	19	17	9	12
95% CI of the mean	0.31	71	4.55	63.69	1.52	66	661E4	401	267
National effluent stds	***6.00 -8.00***	***-***	***10***		***80***	***500***	***10000***	***100***	***50***
Discharge (m^3/s)	9.9E-7 to 0.102 (n = 3, s = 9)								

n= measurements times, s = data set (3 * 3 drains)

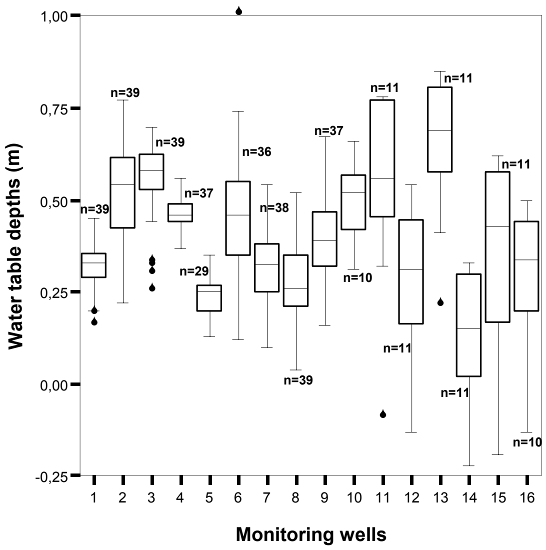


Fig 12 Variation of water table depths (median) at the different monitoring well locations. n = sample size

Table 11 Correlations of the water table depths with short rains at the monitoring well locations (n same as in Fig. 12)

MW	Pearson correlation coefficient between Water table depth and 48hr rainfall	Significance (2 tailed)
1	0.608**	0.000
2	0.317*	0.049
3	0.852**	0.000
4	0.534**	0.001
5	0.364	0.052
6	0.581**	0.000
7	0.490**	0.002
8	0.531**	0.001
9	0.480**	0.003
10	0.555	0.096
11	0.589	0.057
12	0.606*	0.048
13	0.618*	0.043
14	0.524	0.098
15	0.494	0.122
16	0.684*	0.029

** Correlation is significant at the 0.01 level (2-tailed)
* Correlation is significant at the 0.05 level (2-tailed)

However, the low positive correlation between the discharge and short rains (r=0.356, P=0.042) suggests that the spring is primarily fed by regional baseflow from a high storage regolith aquifer. This has implications for contaminant entry into this source, which has a wider catchment including areas outside Bwaise III. The sources of contamination of this source therefore occur over a wider area.

Table 12 Average values ± standard error of the mean and range of water quality parameters of rainwater. (n=4)

Parameter	Mean	Range
Temp (°C)	21.8 ± 0.5	21.3 - 22.3
pH	4.71± 0.70	4.01 - 5.41
EC (μS/cm)	26 ± 9	16 - 44
DO (mg/l)	5.33 ± 0.63	4.70 - 5.95
Total Phosphorus (mg/l)	0.013 ± 0.005	0 - 0.024
Nitrates	2.00 ± 1.26	0.4 - 5.7
TKN (mg/l)	0.53 ± 0.23	0.00 - 1.10
Chlorides (mg/l)	2.53 ± 1.73	0 - 7.6
TTCs (cfu/100ml)	0 ± 0	0-0
FS (cfu/100ml)	2 ± 0	1 - 2

5.2.3 Water quality variation

The shallow groundwater in the area exhibits a variation of quality with season (Tables 13 & 14). The water has very high microbial contamination with median values of thermotolerant coliforms well above national and WHO drinking water guideline values. The spring has nitrate concentrations greater than the recommended values of portable waters as compared to the wells, which contain higher TKN concentrations. Except for the nitrates, the water at the protected spring has relatively lower contaminant values than in the wells.

The quality of the shallow groundwater deteriorates further in the wet season especially with respect to microbial contamination and organic content in the form of TKN. In addition, there is an increase in the nitrate concentrations during the rains implying that the relatively well-aerated rainwaters (Table 12) contribute to mineralization of the organic nitrogen to nitrates. On the other hand, rainfall infiltration results in recharge (as noted from the quick increase in the water levels) that contains contaminants to the shallow groundwater.

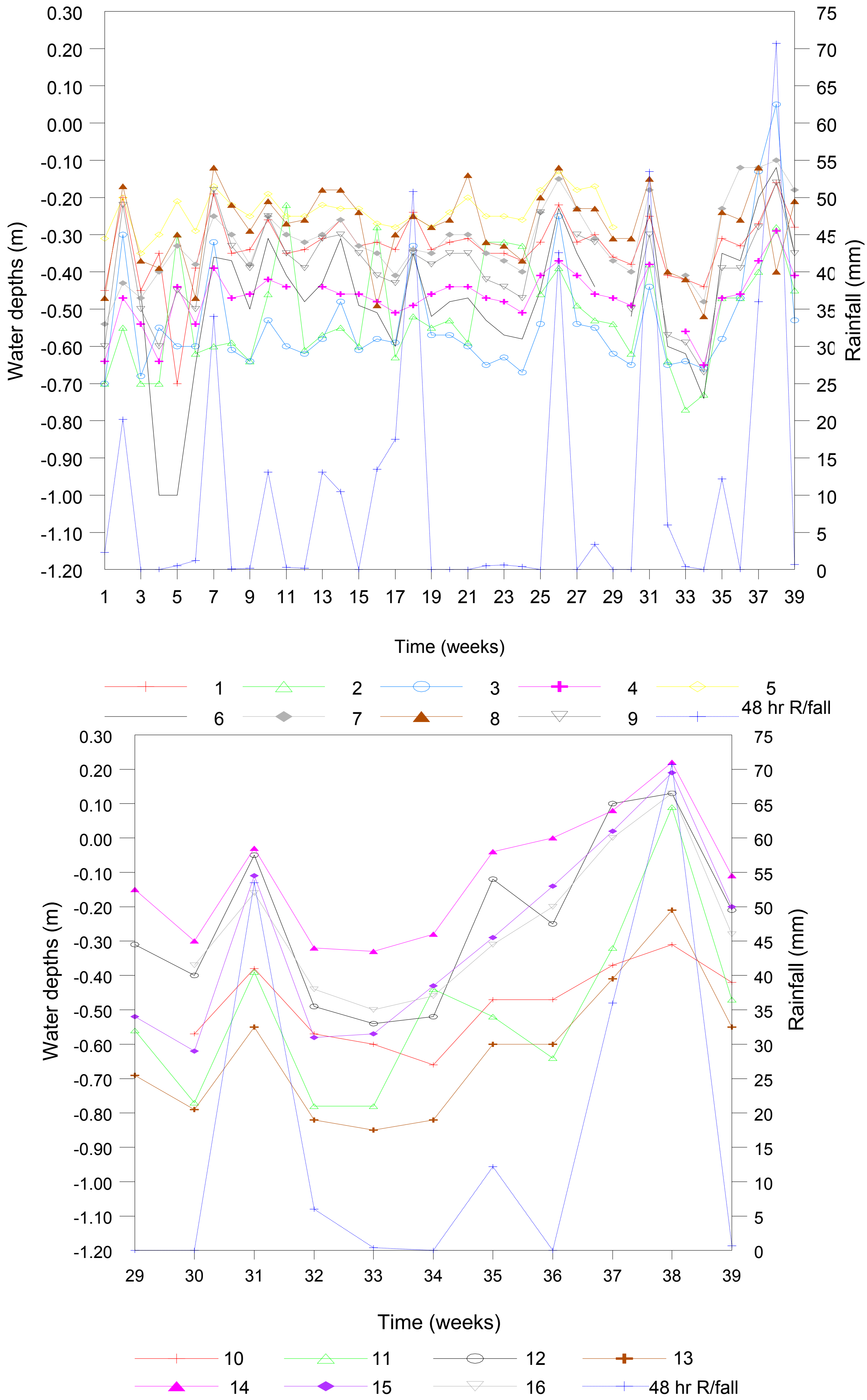


Fig. 13 Water level variations with time and short rains at the monitoring well locations

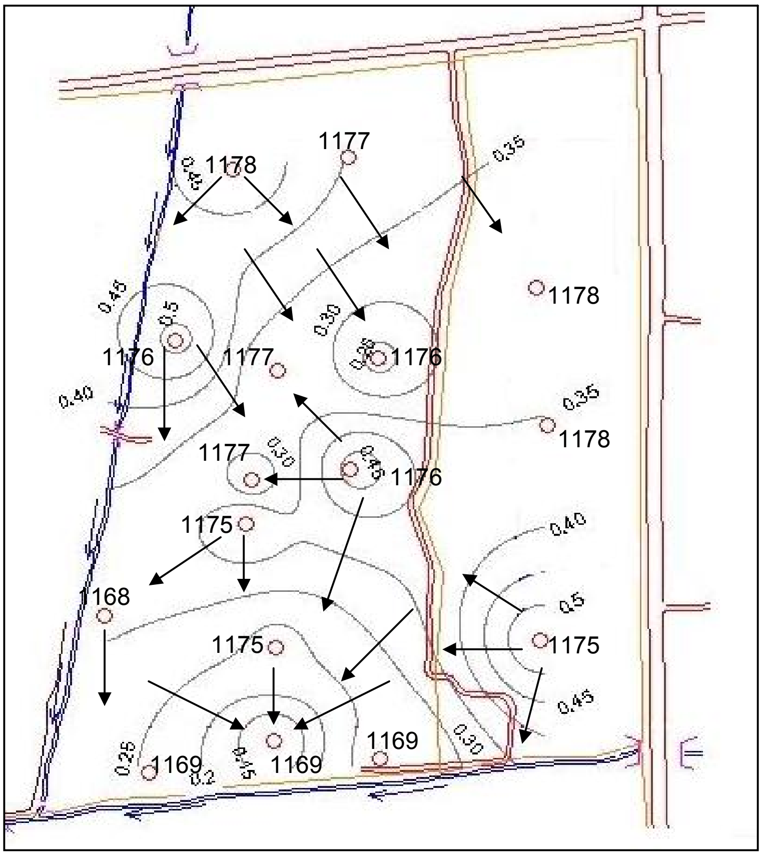


Fig. 14 Average water depth isolines and potential flow patterns. Note the altitude levels (m.a.s.l) at the well locations

The trend for water quality variation with rains is similar for the monitored sites and hence is illustrated by some locations (Fig. 15-17). There is however a variation of the levels of contamination observed at the different sites. As noted earlier, the spring has lower contaminant concentrations as compared to the monitoring wells (Fig. 15-17). This could be attributed to the fact that this source is protected though contaminant entry via local pathways especially of TTCs is evident due to the poor maintenance, and leaching from sources located several metres away.

From Fig. 16, MW4 has lower contaminant levels as compared to MW 1. This variation may be linked to anthropogenic activities within the well vicinity. At MW1 there are two traditional pit latrines located <1 m upgradient of the well. In addition, the area around the well is a recipient of organic solid waste (food waste, food peelings, banana leaves and wood chippings) as well as waste from bird rearing (chicken house was built over this well). With MW 4, the potential major contaminant sources are pit latrines which are located >5 m upgradient. Besides this well has a relatively deeper water table as compared to MW 1 (Fig. 12 & 18). This implies that pollutant travel times to MW 4 are longer as compared to MW 1 allowing for attenuation to occur. Fig. 17 further highlights the impacts of anthropogenic activities on water quality. A comparison between MW 2 and 5 shows that the latter has lower contaminant levels. This is closely linked to the landuse activitities within the well vicinity. At MW 2, located <0.5 m and 1m upgradient of the well is an excavated pit into which solid waste mostly organic and occasionally children waste is disposed, and unlined sullage drain respectively. At MW 5, a filled and buried pit latrine and an outside bathroom are located about 4 m upgradient of this source. Hence pollutants are potentially subject to attenuation processes such as filtration, sorption, and die off as they travel towards MW 5. Considering that the groundwater flow occurs within a layer of similar characteristics at both these well sites (Fig. 18 & section 5.2.4), the close proximity of anthropogenic pollution sources at MW 2 result in leaching of contaminants and transport of thermotolerant coliforms to the well source.

Table 13 Seasonal variation of protected spring water quality

Season/ Parameter	WET			DRY			WHO drinking Water Stds*
	Median	Range	Count	Median	Range	Count	
Temp °C	24.2	23.9-25.5	15	24.3	24.0-24.8	18	
DO mg/l	2.03	0.76-2.97	15	1.72	1.04-3.50	18	
pH	5.20	4.58-6.28	15	5.10	4.58-5.53	18	6.5-8
EC µS/cm	326	280-372	15	332	272-345	18	
Total-P mg/l	0.00	0.00-0.04	15	0.01	0-0.11	18	
TKN mg/l	1.00	0-262.9	14	1.00	0-5.51	17	
Nitrates mg/l	117.10	24.3-692.6	15	129.75	24.2-144.8	18	50**
Chlorides mg/l	43	28-192	15	39.8	31.0-50.5	18	250
TTCs cfu/100ml	815	29-10000	8	10	0-131	11	0**
FS cfu/100ml	433	6-8300	8	1	0-35	11	
NO^{-3}/Cl^{-} ratio	2.9	0.5-17.8	15	3.18	0.6-4.0	18	
Q m^3/hr	1.37	1.22-1.48	15	1.30	1.22-1.48	18	

Table 14 Seasonal variation of monitoring well quality (For wells 1-16)

Season/ Parameter	WET			DRY			WHO drinking Water Stds*
	Median	Range	Count	Median	Range	Count	
Temp °C	24.2	21.5-27.6	240	24.6	22.1-30.0	350	
DO mg/l	1.10	0.1-2.9	236	0.91	0.08-2.97	291	
pH	7.30	6.42-8.28	241	7.28	6.58-8.32	350	6.5-8
EC µS/cm	926	277-3460	241	1055	323-3420	350	
Total-P mg/l	1.10	0.13-6.65	242	1.05	0.08-13.07	348	
TKN mg/l	8.52	0.7-369.6	242	10.41	0.01-112.00	349	
Nitrates mg/l	6.40	1-779	242	4.90	0-540.40	350	50**
Chlorides mg/l	106	22-660	240	133	20-1000	348	250
TTCs cfu/100ml	126E3	0-26E6	96	8200	0-162E6	198	0**
FS cfu/100ml	154E3	0-28E8	93	3250	0-176E6	198	
NO^{-3}/Cl ratio	0.07	0-3.98	240	0.04	0-6.78	348	

* WHO, 2004
** Based on health reasons

Correlations between selected contaminants at the wells and spring with short-term rains (48hr) (Table 15) shows that rain is a significant driving factor for TTC transport to the shallow groundwater in the area. At some sites (MW 2, 3, 4 and 11) a significant positive correlation exists between nitrate concentrations and short rains. This as mentioned previously is closely linked to the presence of organic nitrogen loads emanating from anthropogenic activities - solid waste disposal (mostly organic), leaching nitrogen from pit latrines, sullage drains and animal rearing (as is the case at MW 3, section 5.2.4.1).

A negative correlation exists (though not significant at some sites) between EC, chlorides or total phosphorus and the short rains suggesting dilution by rainfall infiltration. However from Fig. 15-17 the concentrations of these contaminants increase during periods of low rains (< 5 mm) or no rains (0 mm). This could be attributed to residual contaminated recharge derived from subsurface leaching although this may also be from pit latrines, solid waste dumps or contaminated surface in the drains or open ground.

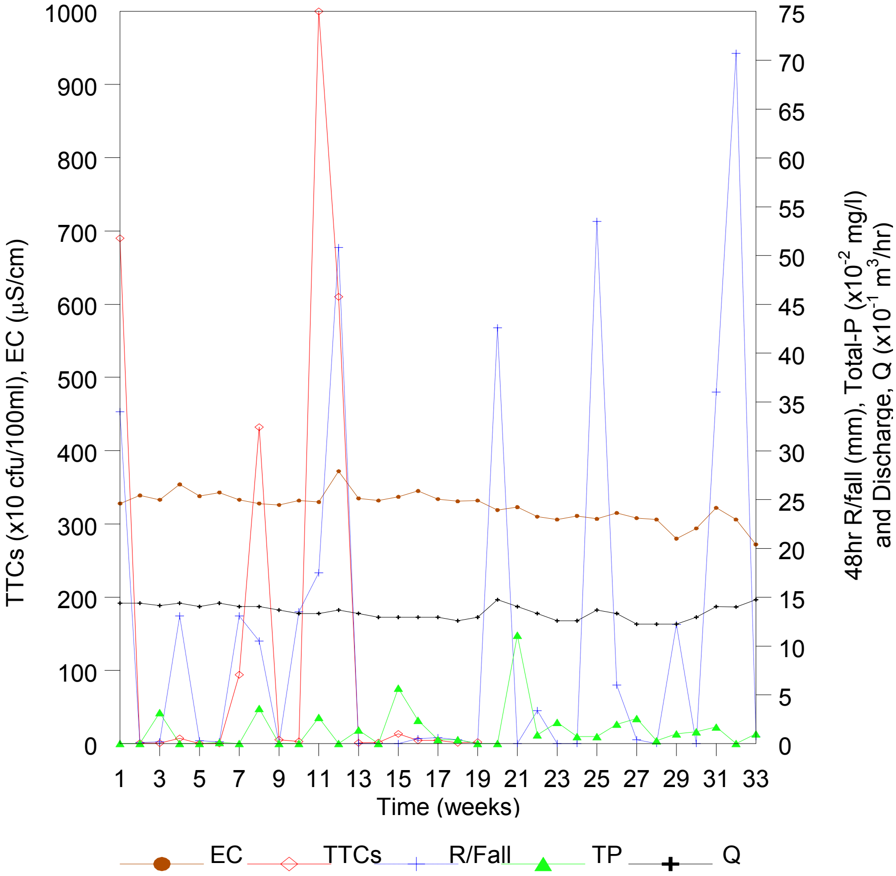


Fig. 15 Variation of selected water quality parameters and discharge at the spring with time and short rains

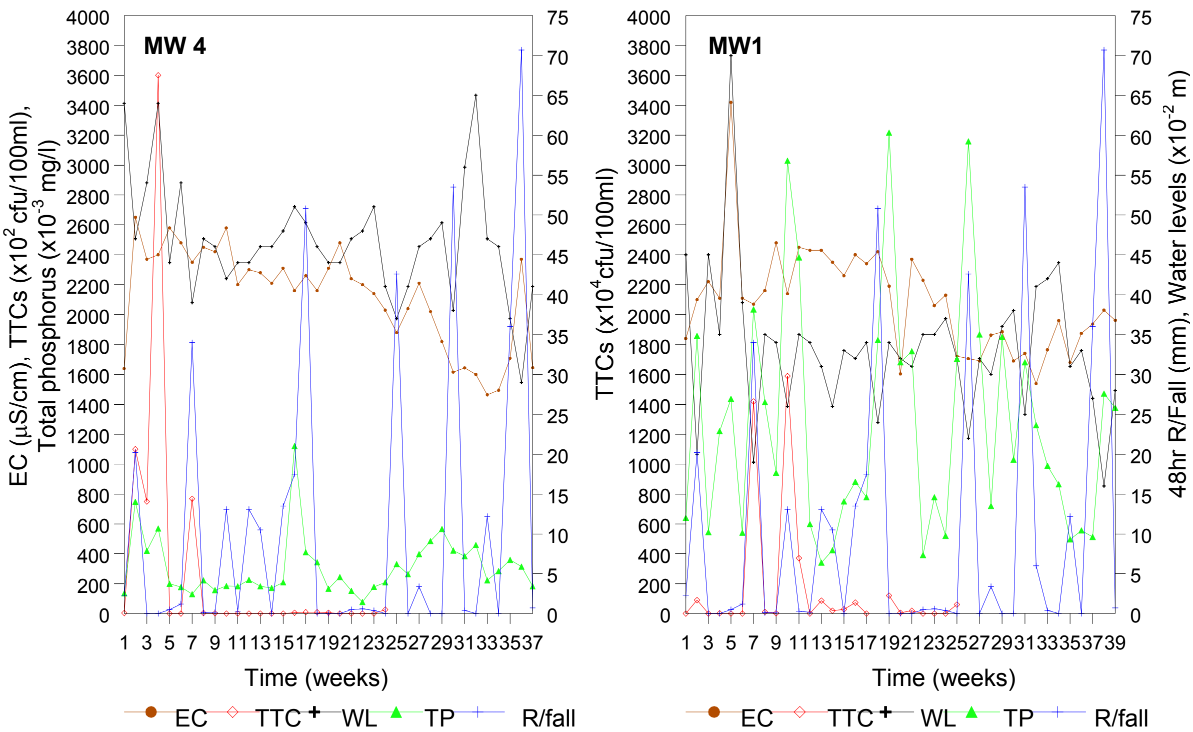


Fig.16 Variation of selected water quality parameters and water levels at monitoring well locations 1 & 4 with time and short rains

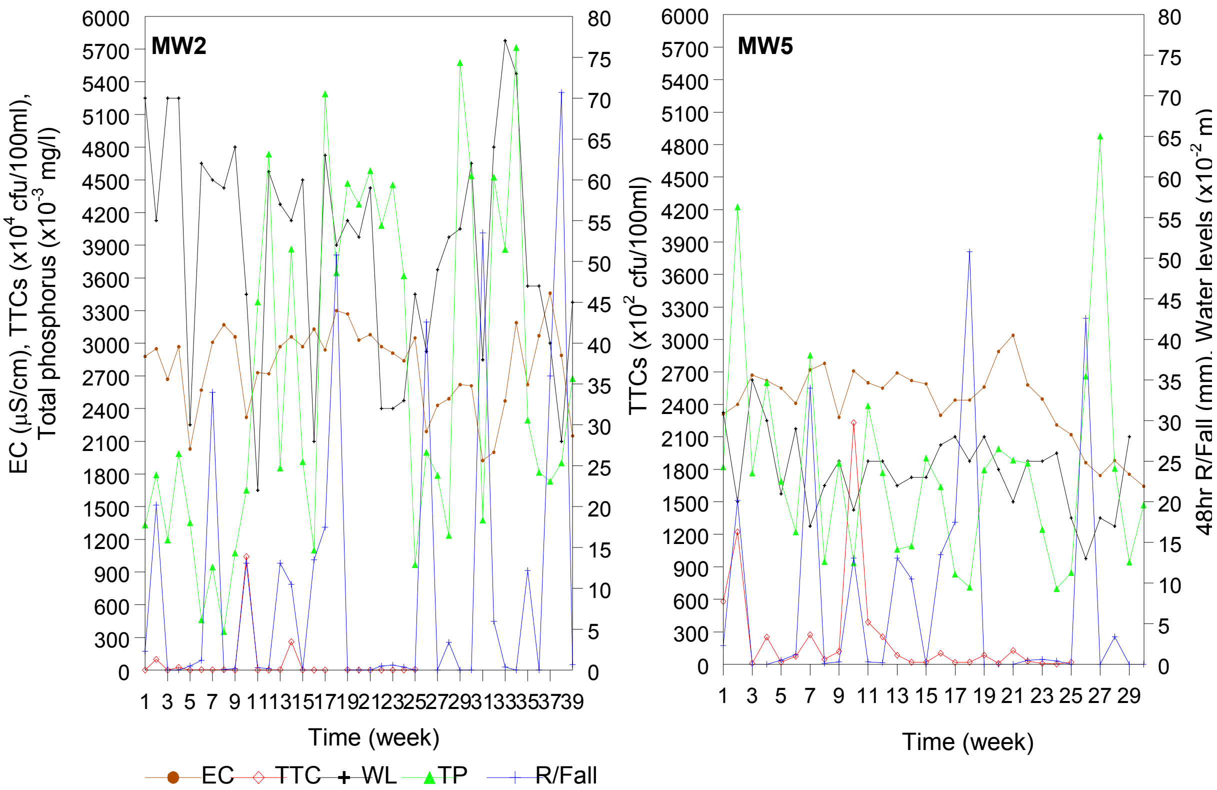


Fig.17 Variation of selected water quality parameters and water levels at monitoring well locations 2 & 5 with time and short rains

5.2.4 Subsurface soil characteristics

The capacity to retain contaminants is dependent on the soil characteristics, among other things. Bwaise III as mentioned previously is a reclaimed wetland and hence the vadose/unsaturated zone is made-up ground in most of the areas. Soil stratigraphy shows that there are largely three distinct layers within the soil profile at most of the sites (Fig. 18). The bottom layer is made up of stiff silts, which are impervious (vertical permeability ≈ 1-3 x 10^{-7} m/s). The fill material in the top two layers contains mostly clayey silty sands with polyethene paper, rusty metals, stones, brick fragments, rugs, and decaying organic material and is pervious (vertical perrmeability ≈ 10^{-3} to 10^{-5} m/s). With reference to the well design (Fig. 9), the sampled shallow groundwater was drawn mostly in the second layer with levels dropping closer to the bottom (or third layer) during prolonged periods of no rains. The soil texture (Table 16) based on particle size distribution curves for soils from selected monitoring well locations for the three common layers show that the first and second layer contains material falling mostly in the sandy fraction (>50% of the coarse fraction is sand size). At sites MW 6 and 12, the fine fraction of the second layer was largely organic material and was burnt out following addition of hydrogen peroxide.

The bottom layer (third layer) on the other hand contains mostly fine-grained soils (>50% passing the 0.075 mm sieve size) with more silt as compared to the clay fraction.

Soil chemical characteristics of the aforementioned 3 layer (1-Top, 2-Middle and 3-Bottom from the groundsurface) (Fig 19 to 21) show that the area is underlain by moderately alkaline soils with high calcium and phosphorus contents (> 4 mg/l and 15 mg/kg respectively, Landon, 1991), which decrease with depth. This could be attributed to anthropogenic influences such as disposal of sullage, human waste and organic solid waste into the subsurface.

Table 15: Pearson correlations of selected contaminants with short rains at the monitoring well and spring locations (n= 25 for TTCs at wells 1 to 9, and between 30-39 for other parameters; n = 11 for wells 11-16)

Source ID	EC	Sig.	Cl-	Sig.	TKN	Sig.	NO-3	Sig.	Total P	Sig.	TTCs	Sig.
MW1	-0.082	0.618	-0.147	0.372	0.205	0.211	0.303	0.061	0.248	0.128	0.758**	0.000
MW2	0.150	0.369	0.357*	0.028	0.172	0.301	0.550**	0.000	-0.162	0.332	0.626**	0.001
MW 3	-0.297	0.070	-0.120	0.480	0.206	0.214	0.341*	0.036	-0.002	0.992	0.741**	0.000
MW 4	-0.031	0.856	0.234	0.176	0.216	0.198	0.519**	0.001	0.179	0.290	0.009	0.965
MW 5	-0.039	0.838	-0.074	0.698	0.164	0.387	-0.035	0.852	0.059	0.758	0.174	0.406
MW 6	-0.220	0.198	0.026	0.883	0.333*	0.047	0.224	0.189	0.011	0.948	0.778**	0.000
MW 7	0.471**	0.003	-0.211	0.203	0.210	0.205	0.154	0.357	-0.247	0.135	0.068	0.747
MW 8	-0.307	0.057	0.018	0.912	0.228	0.163	0.116	0.483	-0.011	0.945	-0.114	0.587
MW 9	0.013	0.937	0.042	0.805	0.216	0.206	0.208	0.217	-0.078	0.648	0.446*	0.026
MW10	-0.732*	0.016	-0.711*	0.021	0.286	0.423	0.356	0.313	-0.141	0.698		
MW11	-0.719*	0.013	-0.658*	0.028	0.230	0.496	0.626*	0.039	-0.270	0.422		
MW12	-0.401	0.221	-0.263	0.435	0.162	0.635	-0.171	0.615	-0.052	0.880		
MW13	-0.551	0.099	-0.298	0.373	0.200	0.556	0.219	0.518	-0.470	0.145		
MW14	0.314	0.347	-0.373	0.258	0.253	0.453	-0.317	0.341	-0.273	0.416		
MW15	-0.508	0.110	-0.585	0.058	0.255	0.450	-0.402	0.220	-0.219	0.518		
MW16	-0.816**	0.004	-0.670*	0.034	0.166	0.648	-0.083	0.820	-0.047	0.904		
Spring	0.029	0.875	-0.019	0.918	0.233	0.232	0.304	0.085	-0.214	0.275	0.575*	0.010

* Correlation is significant at the 0.05 level (2-tailed)
** Correlation is significant at the 0.01 level (2-tailed)

The cationic exchange capacity (CEC) of the soils does not vary significantly between the soil layers (P= 0.857) possibly due to the presence of organic matter in all the three layers and an almost similar pH in the latter. CEC is largely dependent on the soil texture and organic matter content of the soils. In addition it is known to generally increase with soil pH (Heidmann *et al.*, 2005). The CEC values are low and indicative of the dominance of kaolinite clay mineral type (typical CEC range 3-15 meq/100g dry soil, Heidmann *et al.*, 2005). They however, have the potential to retard contaminants such as phosphorus and thermotolerant coliforms (ARGOSS, 2001; Morgan, 1981).

Metal ratios are important in retention of contaminants such as phosphorus (Morgan, 1981). The ratios for the different layers (Table 17) are highest for iron and lowest for magnesium. The implications of this on the studied contaminant (phosphorus) is presented in section 5.2.5.

Table 16 Soil texture characteristics of the different layers. Range is given in parenthesis; n = 11 per layer

Layer/soil fraction (%)	Top	Middle	Bottom
Sand	41±3 (17-58)	33±5 (14-64)	31±4 (14-48)
Silt	29±2 (14-40)	30±5 (5-57)	42±4 (30-57)
Clay	18±3 (10-38)	21±3 (10-41)	27±6 (10-56)

Table 17 Metal ratios for the different layers Range is given in parenthesis; n = 11 per layer

Layer	1	2	3
Ca:P	0.7± 0.1 (0.3-1.5)	0.9±0.2 (0.2-2.7)	0.6±0.1 (0.3-0.8)
Fe:P	3.7±0.9 (1.6±10.8)	13.2±3.4 (1.0±37.2)	13.5±2.8 (3.1±24.3)
Mg:P	0.19±0.01 (0.04-0.2)	0.12±0.01 (0.05-0.2)	0.12±0.03 (0.06-0.3)

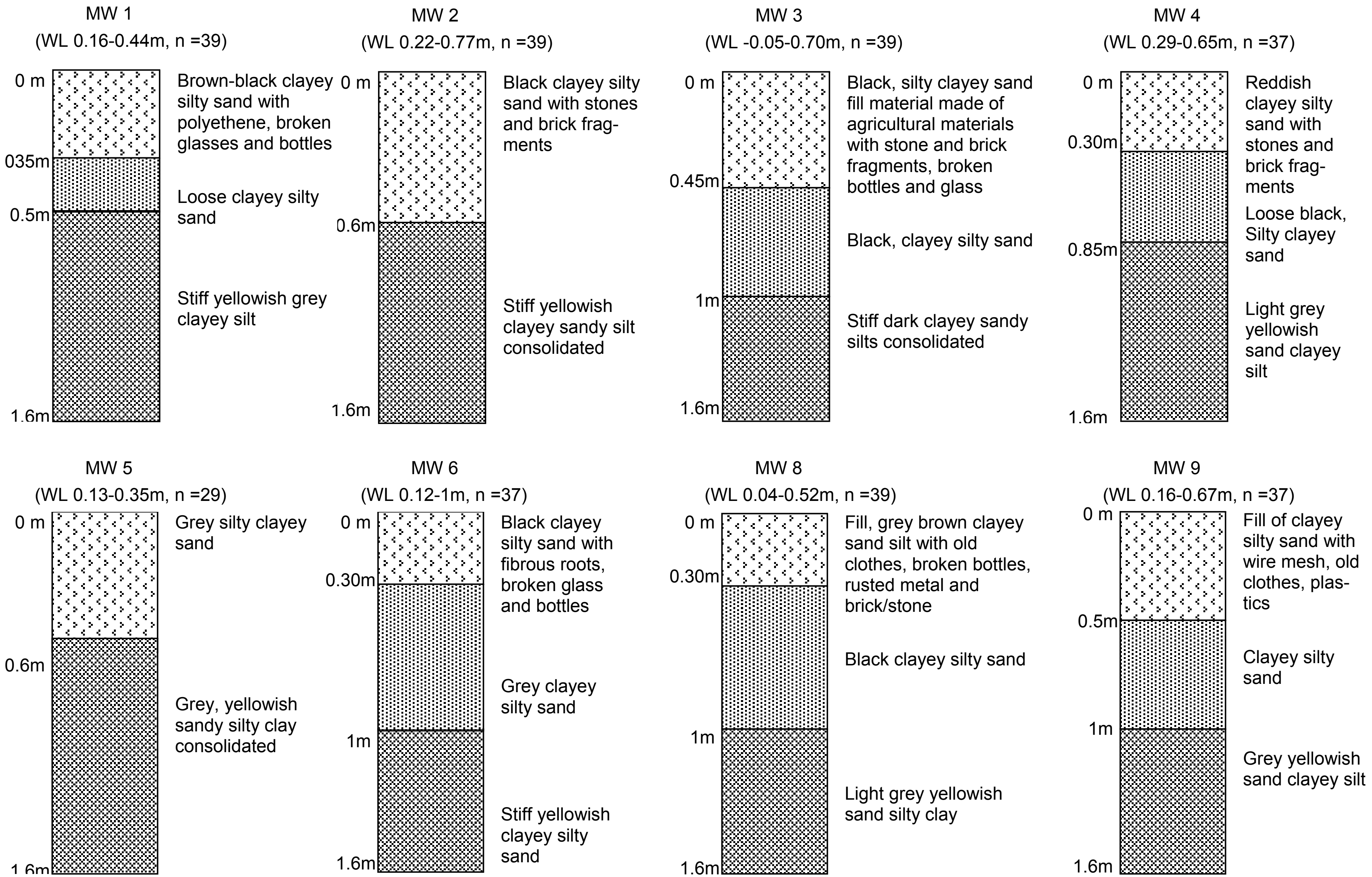


Fig. 18 Soil logs at some monitoring well locations (Soil classification follows from particle size distribution; WL = water level)

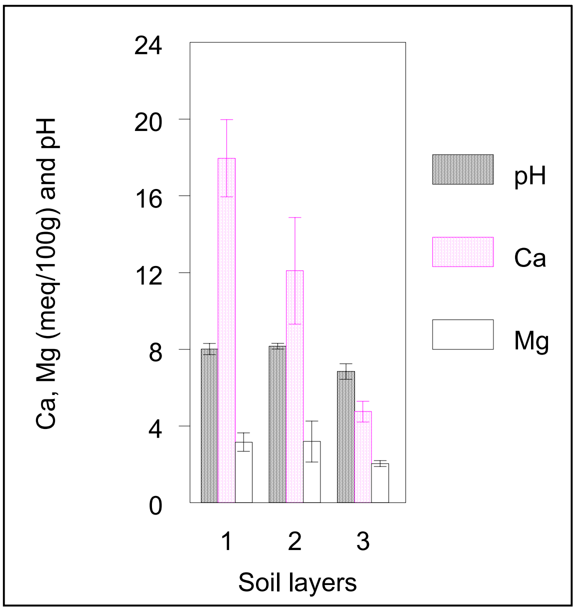


Fig. 19 Variation of mean values (±1 SE bars) of pH, calcium and magnesium content for the 3 different soil layers. (n=11 per layer)

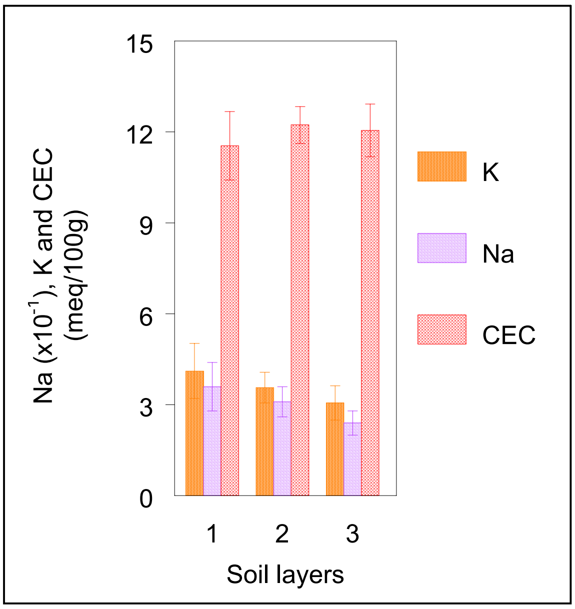


Fig. 21 Variation of mean values (±1 SE bars) of sodium, Potassium and cationic exchange capacity for the 3 different soil layers.

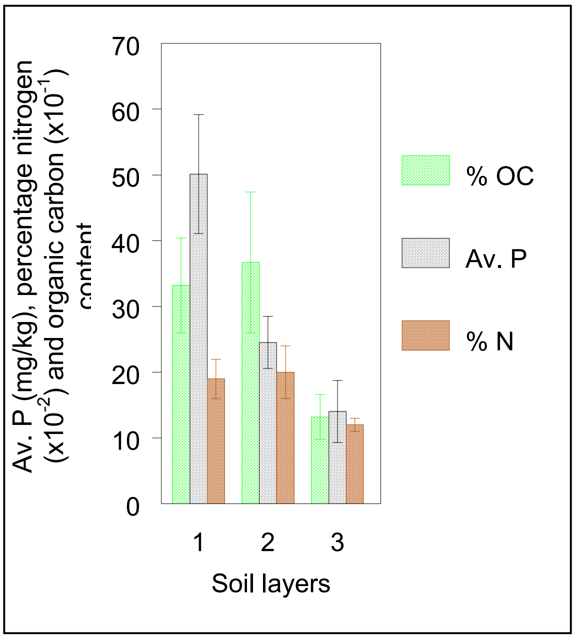


Fig.20 Variation of mean values (±1 SE bars) of available phosphorus, percentage organic carbon and nitrogen content for 3 different soil layers.

5.2.5 Attenuation potential measurements

Field measurements

Results of water quality measurements to ascertain the *in-situ* attenuation potential of the soils from 3 wells (at MW 3, 6, 8 and 9; Fig. 8) installed at a distance of about 1 m apart along a presumed flowline are shown in Fig. (22-25). The water flow was from well 1 downgradient to well 3. With the flow occurring mostly in the middle (second) layer, increase in mean values of electrical conductivity and chlorides (due to the chemical composition of the soil media, Table 18) as conservative tracers show that the flow was towards well 3 at most of the sites.

There are spatial variations of contaminant retention at the different sites. There is a general decrease in TTCs over a distance of 2m at most of the sites except at MW 3 (Fig. 22). This is primarily due to human activities within the well vicinity that is, cow rearing resulting in dung dumping adjacent to the wells. Considering that the flow occurs in the middle layer with an estimated flow of about 10^{-6} ms^{-1} (Paper III), the die-off rates of thermotolerant coliforms calculated from chick's law[4], were 0.008, 0.003 and 0.009 hr^{-1} at MW6, 8 and 9 respectively.

Die-off rates of these organisms under experimental conditions (0.021hr^{-1}, Kansiime and Nalubega, 1999; 0.064hr^{-1}, Gerba, 1985

[4] Die-off rate of faecal coliform bacteria typically follows from a first order relationship described by: ln (C_t/C_o) = -kt where k is the decay constant (hr-1), C_o is the initial faecal coliform concentration at t = 0 and C_t is the faecal coliform concentration at time t (Johnson, 1975)

Table 18 Soil characteristics of the middle layer (layer 2) at the different well locations

Location/ Parameter	MW 3	MW 6	MW 8	MW 9
pH	7.1	8.4	6.7	6.9
Org carbon (%)	6.2	6.8	1.8	3.4
Nitrogen (%)	0.2	0.4	0.1	0.2
Av. Phosp (mg/kg)	25.1	42.5	13.4	20.0
Fe (mg/kg)	336.4	227.6	172.1	63.3
K (meq/100g)	3.3	5.8	1.5	1.4
Na	0.3	0.5	0.2	0.1
Ca	17.0	27.4	8.6	10.2
Mg	4.1	9.0	1.8	1.7
CEC	14.6	14.6	12.8	11.0
Sand (%)	20	14.3	31	36
Silt (%)	5	-	30	32
[illegible]	[illegible]		[illegible]	[illegible]

in ARGOSS, 2002) show that the calculated values are lower. This could be attributed to variations between experimental as well as natural conditions. A range of factors is known to affect the die-off of these organisms under natural conditions. These are reported to be soil composition (presence of organic matter, CEC, soil type, moisture content), temperature, pH, flow rate, and antagonism from soil microflora, (Jamieson *et al.*, 2002). In the Kansiime and Nalubega (1999) study, the experimental temperature ranged between 23-25°C with high dissolved oxygen levels (2.5mg/l). The calculated values on the other hand could be a result of the relatively low dissolved oxygen levels (Table 14) with natural die-off and dilution as the predominating removal processes in the saturated zone (ARGOSS, 2001).

At MW 6 and 8 there is some reduction of phosphorus over a distance of 2m (Fig. 23 & 24). This could be linked to the Fe and Ca content of the soils at these sites, which are are quite high (Table 18). A comparison of the metal ratios shows that Fe:P > Ca:P implying that the retention of phosphorus at these sites may be linked more to the high iron levels providing more sorption sites. Phosphorus increment at MW 9 with distance could be attributed to leaching of this element from the subsurface material possibly due to the low Fe, Mg and Ca content while the trend at MW 3 follows from impacts of animal rearing and hence hardly any attenuation.

For all sites, TKN increases downgradient su-ggesting that there is leaching and slow mineralization of this element. The values of TKN are higher at MW 3 and 6 due to animal rearing and, sullage disposal and organic solid waste disposal respectively in close proximity to these well sites. This influence is also noted from the organic carbon content of the soils at these sites (Table 18), which is higher than at the other sites.

Sites with a high water table (<0.4 m) (Fig. 22-25) appear to have a high electrical conductivity, total phosphorus and TKN content. This could be linked to the inadequacy of the vadose zone depth to allow for contaminant attenuation to occur and suggests therefore that ponding leads to a general deterioration of the water quality in the area.

Phosphorus adsorption

Phosphorus adsorption kinetics had a similar trend for the different layers with adsorption (50-70% of the available P) occurring within 4-8hrs (Fig. 26).

Data fitted to both the Langmuir and Freundlich isotherms for the different layers (L1-Top, L2-Middle and L3-Botton; Fig. 27& 28) shows that the middle (second) layer generally had a higher P sorption than the Langmuir sorption maximum (Cmax) and Freundlich adsorption constant (Kf). This could be attributed to the moderate flows and higher metal ratios in the second layer (Table 17) with iron offering more sorption sites. This could also explain the sorption potential obtained for the third layer (Fig. 28). The discrepancy of the latter with Fig. 27 could be attributed to data fitting. Some of the generated data did not fit the Langmuir sorption isotherm (see note below Table 19). There is however, a positive and significant correlation between the Langmuir adsorption coefficient (KL) and Freundlich adsorption constant (Kf) ($r=0.587$, $P=0.002$) suggesting that both models are useful in describing the adsorption of phosphorus of the soils in the area.

There was a direct correlation between Kf and available phosphorus ($r=0.564$, $P=0.001$), iron ($r=0.488$, $P=0.006$) and moisture content ($r=0.393$, $P=0.032$) of the soil. There was a negative and non-significant correlation between KL and available phosp-

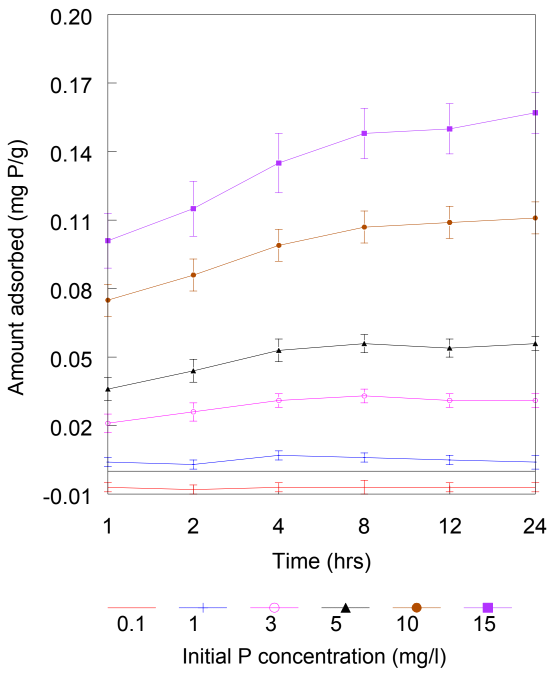


Fig. 26 Phosphorus adsorption kinetics for layer 1 (average values ± 1SE bars, n = 11)

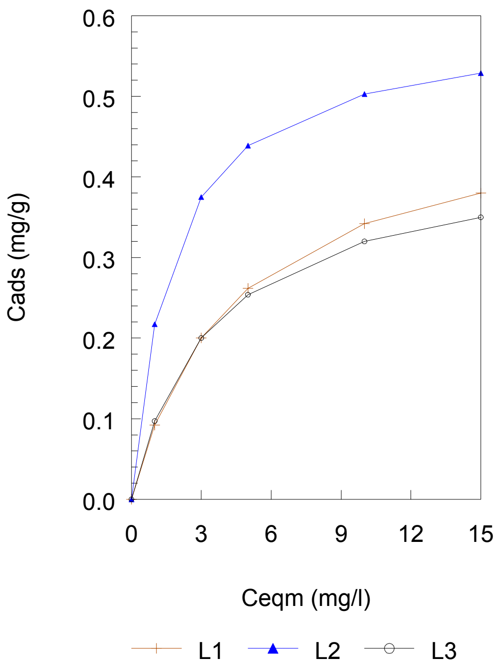


Fig. 27 Fitted phosphorus Langmuir adsorption isotherm for the different layers of soil (average values, n= 9 per layer)

-horus (r=-0.217, P=0.297) and a positive correlation with moisture content (r=0.393, P= 0.032). In both cases, a negative correlation though non-significant existed between Kf and KL with calcium and magnesium. Cmax on the other hand had a positive corre-

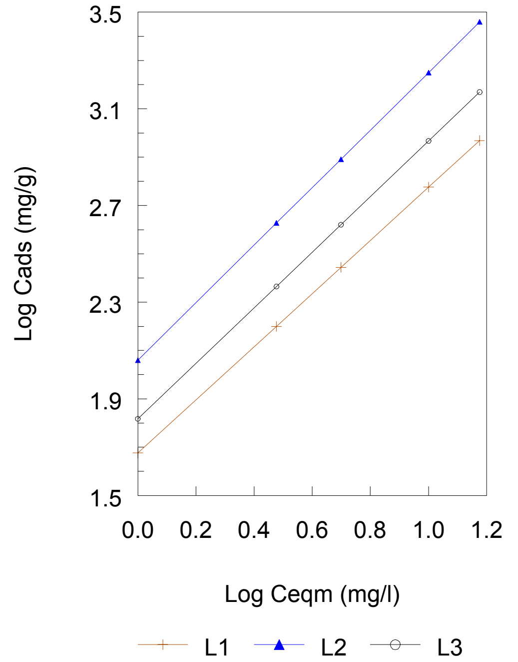


Fig. 28 Fitted phosphorus Freundlich adsorption isotherm for the different layers of soil (average values, n= 11 per layer)

Table 19 Parameter estimates for Langmuir-sorption isotherm (average values±standard error) for the different layers.[1] Ranges are given in parenthesis.

Layer/ Coeff.	Cmax (mg/g DW)	KL (l/mg)	r^2
Top	0.49±0.10 (0.23-1.26)	0.23±0.07 (0.01-0.72)	0.95 (n = 9)[2]
Middle	0.59±0.17 (0.18-1.78)	0.58±0.20 (0.01-1.95)	0.97 (n = 9)
Bottom	0.43±0.09 (0.22-0.74)	0.29±0.08 (0.03-0.60)	0.97 (n = 9)

[1] Experimental conditions (room temperature, °C = 24.8±0.05; pH = 7.12±0.04 and EC, µS/cm = 811±4.87 (± SE, n=186)

[2] Out of a total of 33 samples, data for 6 samples did not fit the Langmuir sorption isotherm

Table 20 Parameter estimates for Freundlich-sorption isotherm (average values±standard error) for the different layers.[1] Ranges are given in parenthesis.

Layer/ Coeff.	Kf (ml/g DW)	N (=1/n)	r^2
Top	47.37±8.62 (3.65-109.4)	1.10±0.09 (0.57-1.61)	0.80 (n = 11)
Middle	114.71±22.69 (17.06-218.78)	1.19±0.13 (0.60-2.08)	0.80 (n = 11)
Bottom	65.55±10.94 (36.06-130.32)	1.15±0.21 (0.47-2.33)	0.78 (n = 11)

[1] Experimental conditions as above

lation with available phosphorus (r=0.476, P=0.016) and negative non-significant correlations with calcium and iron soil contents. For all cases, there was a poor direct correlation between with either soil pH, sand, silt or clay soil fractions. This implies that the phosphorus sorption potential of the soils in the area may be largely a function of the available phosphorus, iron and moisture content.

6 General Discussion

Bwaise III from the aforementioned (Chapter 5) is an area with a porous and limited vadose zone that is mostly made-up ground. The topography of the area is such that flooding following heavy rains is a common phenomenon. Liquid and solid waste management practices in Bwaise III are grossly poor resulting in poor environmental sanitation. The latter impacts on the health of the communities with malaria, diarrhoea, typhoid and cholera epidemics being common especially during the rain season (Lwasa, 1999).

The inadequacy of excreta disposal facilities, unsanitary practice of discharging latrine contents into open drains, solidwaste dumping as a means of wetland reclamation and stagnant sullage waters in unlined drains contributes to high pollutant accumulation in the area. This subsequently leads to the deterioration of the shallow groundwater quality. It is reported though that the impact of solid waste dumps on the microbiological quality of the groundwater is likely to be more localized than widespread (ARGOSS, 2001). Wetland reclamation following from solidwaste disposal has implications for the lithology (source of soil organic matter content) and subsequent contaminant transport to the shallow groundwater through leaching and as a result of rainfall infiltration. There is a great need for proper waste management systems in the area.

The shallow water table and hence limited but porous vadose zone (<1 m) implies that groundwater is very vulnerable to contamination. The rapid recharge following short rains is indicative of a highly permeable vadose zone. This zone extends over the top and part of the second layer, which have a medium to high permeability. The third layer has a very low permeability according to the permeability degree classification by Head (1982), which explains the constant saturation conditions in this zone. The presence of the impervious bottom layer therefore (with deeper groundwater occurring at least 30 m, section 2.4, Chapter 2) suggests that this not only retards pollutants but also groundwater recharge into the deep aquifers. The top layers on the other hand lead to not only a faster recharge of the shallow aquifer but also rapid surface contaminant transport to these systems. This could be attributed to the fact that the top layers are mostly man-made with high potential of macropore flow occurrence as a result of waste disposal (Fig. 29). The vadose zone consequently offers limited attenuation of contaminants and in certain cases, a likely contaminant source through leaching.

The discharge of the spring does not exhibit a marked change with rains. According to Girish and Varun (2004), spring water discharge fluctuations are primarily due to variations in the amount of rainwater that is able to infiltrate to the ground and recharge the groundwater. However, in this case the non-responsiveness of the spring discharge as well as to adjacent potential pollutants following

Fig. 29 Trial pit at MW 15 location exposing the profile. Note the presence of foreign material which bring about macropore flow

the rains suggests that there is hardly any mix between the "old" and "new" water from rainfall infiltration in this system. This implies that the spring is primarily fed by regional baseflow. The latter possibly explains the high nitrate concentrations (Table 13) and nitrate: chloride ratios observed throughout the monitoring period in the discharge suggesting continuous contamination.

According to ARGOSS (2002), relatively high nitrate to chloride ratio (>1) is indicative of contamination of faecal origin while a close relationship between the nitrate and chloride concentrations on a regional scale indicates a dominant source for both species. High nitrate to chloride ratios in the spring discharge are > 1 (Table 13) implying contamination of faecal origin while the absence of a direct correlation between the nitrate and chloride concentrations during dry and wet seasons suggests that multiple sources of contamination exist for both species. Hence protection measures for the spring should target the potential pollutant sources (including pit latrines, solid wastes etc) located within the wider spring catchment and not only the adjacent areas. However, it is noted that reduction of the loads from potential nitrate sources do not result in a rapid decrease of this element in shallow groundwaters with high concentrations. In addition, denitrification as a potential attenuation process is particularly ineffective (ARGOSS, 2001). This implies that there may be a need to plan for alternative sources of water supply such as piped-water systems in the immediate future so as to safe guard communities' health. Nitrates in concentrations above 50 mg/l is known to cause *methaemoglobinaemia* especially in infants under three months of age while levels upto 300 mg/l are associated with gastric cancer (WHO, 2004).

The high TTCs observed in the spring following short-term rains (Fig. 15) are a result of local ingress of contaminated surface waters. The latter are known to be a major threat to the microbial quality of shallow groundwater in Kampala (Howard *et al.*, 2003). The poor maintenance of the protected spring as evidenced from the loss of the cut-off drain and damage of the protection wall (around the "roof" of the spring), (Fig. 30) was seen to contribute to direct ingress of contaminated stormwater runoff from the area upgradient of the spring and uphill neighbouring areas (not within Bwaise III Parish) into this source through the sides of the damaged protection wall and "roof" of the spring. This is suggestive of the fact that proper maintenance of the protected spring structure plays a significant role in reducing localised microbial contamination.

Monitoring well water quality implicates widespread contamination of the shallow groundwater in the area. This may be attributed to multiple sources of contamination, which are considered more important than pathways in explaining the widespread contamination (Howard *et al.*, 2003). The variation in spatial water quality observed with season is largely related to anthropogenic activities within the wells' vicinities and water table depth. During the rains (when the water had high contaminant levels), the groundwater table occurred mostly in the top layer (and at times above the ground surface during flooding) and close to or within the bottom layer during the dry season (Fig. 18). This points to the fact that the low flows in the bottom layer increase the contact time between the water and soil media to allow for attenuation to occur. Flooded conditions on the other hand lead to a deterioration of the water quality due to the potentially macropore flow of the infiltrating water resulting in inadequate retention for contaminant attenuation to occur. Sites with adjacent landuse activities: animal rearing, solidwaste dumping, sullage drains and pit latrines contributed had increased microbial, nitrates and/TKN and phosphorus concentrations in the water especially during the rains. This implies that protection of shallow groundwaters should include proper waste management.

The relatively low contamination levels exhibited by the spring (except for the nitrates) as compared to the wells (Table13 & 14) could be attributed to the nature of the flow regimes in the two systems and influence of anthropogenic activities on these systems. Where seasonal interflow does not play a

Fig. 30 Protected spring within the study area. Note the damaged spring protection wall and loss of cut-off drain, which contribute to direct entry of surface contaminants into the spring backfill during the rains

significant role as is the case with the spring, the potential for contaminant attenuation exists due to the travel times involved with the regional baseflow except for nitrates which is conservative under relatively aerobic conditions (ARGOSS, 2002). On the other hand, the wells have a significant seasonal interflow whose contribution primarily influences the quantity as well as the quality of the water. Due to the porous nature of the subsurface, these kinds of systems appear to be highly vulnerable to pollution from anthropogenic activities in close proximity especially following heavy rains. Protection of such systems would entail not only reduction of the pollutant loads but also reduction of the permeability of the top layers.

Natural attenuation potential of the soils in the area is limited by human activities, which lead to sporadic pollutant discharges over the area as well as an influenced subsurface a noted from the porous as well as chemical nature of the materials. Of consequence therefore, phosphorus and microbial removals over short distances (about 2 m) are limited while leaching of TKN occurs through out. The latter may be a result of overloading of the system, which has very low dissolved oxygen levels (Table 14). Phosphorus leaching could be attributed to the dominance of preferential flow paths through the soil profile, which limit the contact of the solute with the subsoil and/or the organic soils with low P sorption capacity compared to the phosphorus in the soil and a shallow water table (Börling, 2003). In this case the subsurface waters have P concentrations high enough to cause environmental problems such as eutrophication (levels>>0.05 mg/l, minimum level required for eutrophication to occur in freshwater bodies, Wetzel, 1983). The calculated low bacteria die-off rates corresponding to a travel time of 23 days (< 25 days travel time) suggests significant risk of the shallow groundwater from microbial contamination. with potential to trigger outbreaks of water borne diseases. Hence to ensure very low risk to microbial contamination (> 50 days travel time), which offers greater confidence that the water will meet WHO guidelines (ARGOSS, 2001) would require pollutant sources to be located at least 4 m from a water source.

Laboratory phosphorus sorption experiments show that all the three soil layers still have a potential to retain additional phosphorus loads. This is related to the metal ratios with iron and calcium seen as providing more sites for P sorption. Magnesium on the other hand occurs in very low concentrations and hence its impact on P sorption is minimal. A comparison between the Langmuir sorption maximum values (Table 13) and those obtained for sediment material in an inhabited natural wetland receiving municipal and industrial wastewaters (0.6-1.6 mgP/gDW, Kansiime and Nalubega, 1999) show that these values are lower. This could be a result of the complex landuse pattern in Bwaise III, a habited wetland as well as the nature of the particle sizes. In the above study, the sediment material was mostly silty clays (>75% passing 0.075 mm sieve) while in this case, coarse soil fractions (sands) predominate the top layers. The smaller size fractions of soil, clay and silt have a large surface area and tend to adsorb more phosphorus than sand particles (section 3.2.2).

Available phosphorus, iron and moisture content of the soils largely influence its phosphorus sorption potential as seen from the correlations. This implies that management strategies for minimizing transport of this element to the shallow groundwaters may target reduction of phosphorus loads in the soil system or wetland reclamation using soils with iron such as locally available lateritic soils which should be well compacted so to ensure moderate flows and adequate contact time for P sorption to occur.

7 CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusions

The aim of this on-going study is to assess the anthropogenic pollutant loads, transport, and impact on shallow groundwater in one of Kampala's peri-urban areas (Bwaise III Parish). By using a combination of field surveys, field and laboratory measurements as well as laboratory experiments, an insight into the above has been obtained and follows:

- A variety of potential sources of groundwater contamination exist in Bwaise III. These are scattered over the area. The major types in view of the estimated annual quantities are excreta disposal systems (41,500 kgN, 6,690 kgP; and 8,546 x 10^{14} TTCs), solid waste dumps (4,910 kgN and 890 kgP) and sullage (2296 kgTKN and 132 kgP).
- There is widespread contamination of the shallow groundwater from the above contaminants in the area as evidenced by the water quality from the protected spring and monitoring wells. Animal rearing, solid waste dumps and latrines are seen to result in increased localised microbial and organic content (total kjedahl nitrogen) during the rains.
- The shallow groundwater in the area does not conform to the National and WHO drinking water standards with very high nitrates and thermotolerant coliform counts, and low pH values in the case of the protected spring.
- Short rains result in increased TTCs, organic (TKN and/or NO^-_3), and total phosphorus contaminant levels in the waters. Rainfall infiltration is a significant driving force for TTCs into these systems.
- The highly permeable and shallow vadose zone with high organic matter content offers limited attenuation of contaminants especially during the rains. Natural attenuation is very slow as evidenced from the low thermotolerant die-off rates and minimal phosphorus retention. This is

highly interfered with by sporadic disposal of anthropogenic-generated wastes.

- The area is underlain by a thick impervious stiff sandy clayey silt layer, which retards not only pollutants but also groundwater recharge into the deep aquifers. Hence the poor residents who have no access to piped water are left with mostly shallow and heavily contaminated groundwater sources.
- The soils in the area have the potential to retain further additional phosphorus loads following from the batch experiments. The prediction of this sorption potential is largely a factor of available phosphorus, iron, and moisture content of the soils.
- A conceptual model of a small cut-out section in the area has been developed. Simulations from this model show that short-term heavy rains (> 0.25 mm/min) result in rapid rise of the water to the surface within 1hr to about 2 days considering continous infiltration while with lower rains (< 0.25 mm/min), the water table does not reach the surface and there is no flooding.
- Phosphorus simulations with an initial concentration of 15 mg/l for the above rainfall infiltration rates with sorption results in the contaminant plume sticking to the surface. This implies that the soils still have the capacity to retain phosphorus as evidenced from the sorption coefficients.

7.2 Recommendations

- The operational spring in the area must be protected to safe guard communities' health. Spring re-protection measures should target restoration of the cut-off drain, repair of the damaged protection wall and regular maintenance. This will greatly reduce incidences of microbial contamination observed following the short rains. As an immediate short-term measure, the communities should be urged to boil their water before drinking it.
- Protection of the spring from contamination (such as high nitrate levels) should target reduction of pollutant loads over a wider catchment. Creation of protection zones as a mitigation measure would be impractical in the densely populated peri-urban settings due to the nature of the socio-economic conditions. However, with the high nitrate concentrations, consideration should also be given to provision of piped water. In addition, communities should be encouraged to harvest rainwater, as this would not only have the added advantage of an alternative source of water supply, but also reducing the contaminated surface runoff in the area.
- Measures aimed at reducing the actual pollutant loading in the area as well as the wider spring catchment should be promoted. Sources of great significance to be considered are pit latrines, solid waste, sullage drains and animal rearing activities. Identification of appropriate measures should include consultations with community, municipal authorities, policy makers, planners etc. This is noted to contribute to the sustainability of the chosen and implemented systems (Parkinson & Kevin, 2003).
- Ecological sanitation dry toilet systems should be piloted in the area especially at household level. In Jacks *et al.*, (1998), it is pointed out that these systems have been implemented in India in areas with a high groundwater table. In Kampala, there are ongoing plans under the Sida funded Ecosan Project managed by the municipal authority (Kampala City Council) to implement these units in urban and peri-urban areas. These systems have the advantage of space economy, resource recovery and minimal impact on the environment including groundwater. For the successful operation of these units, there should be continuous sensitisation of the community on the operational aspects as well as a support mechanism for reusing the materials as appropriate
- Privatization of solid waste management and community participation should be encouraged to improve waste collection. Reuse of this organic waste for example as compost and food peelings (especially

banana peelings) should be encouraged. This however requires communities to separate their waste at the source.

- The drainage network of the area is haphazard necessitating planning to prevent stagnation. The drains should be lined and solid waste disposal into them prohibited. Natural wetlands currently receiving these wastewaters should be gazetted and encroachment restricted by the National Environment Management Authority.

7.3 Areas for further research

Inorder to understand better the hydrogeology of the shallow groundwater sources, origin and transportation of contamination in the area as well as identifying pollutant mitigation measures the following further studies are proposed:

- Seasonal wastewater quality monitoring of the main sullage/stormwater drains traversing the area to determine the pollutant loads transported to the area as well as any contribution from the area into the drains and subsequently down-gradient areas. Specifically, total organic carbon (TOC), chemical oxygen demand (COD), biochemical oxygen demand (BOD), total phosphorus and microbial loads should be determined. Analysis of surface runoff in the area should also be undertaken as a measure of the quality of the infiltrating contaminated recharge.
- It is not well known at what rate the surface contaminants are transported to the shallow groundwaters during the rains. Hence to provide information on this, rainfall-water table response studies using a data logger should be undertaken for continuous measurements over some period of time (before, during and some hours after the end of the rains). This information is useful for model calibration as well as validation.
- Column experiments should be carried out to provide an understanding of contaminant removal potential of the vadose zone. This information would be useful in model validation.
- There is a dearth of knowledge on the shallow groundwater flow in the area. Pumping tests should be undertaken at some sites to provide information on this. This is particularly useful information for hydrodynamic modelling and contaminant in these systems.
- Macro-pore flow is suspected to contribute to the fast transport of contaminants to the shallow groundwater during the short rains. It would be useful for tracer tests to be undertaken to provide information on water flow paths.
- Mitigation measures should be identified for the area. These should consider socio-economic, cultural, legal/policy, financial and institutional aspects.

REFERENCES

APHA /AWWA/WEF (1998). *Standard methods for the examination of water and wastewater, 20th Edition*. American Public Health Association, American Water Works Association and Water Environment Federation Publication. Washington D.C, USA.1220p.

ARGOSS (2001). *Guidelines for Assessing the Risk to Groundwater from On-site Sanitation*. British Geological Survey Commissioned Report, CR/01/142. National Environmental Research Council, London. 97p.

ARGOSS (2002). *Assessing Risk to Groundwater from On-site Sanitation: Scientific Review and Case Studies*. British Geological Survey Commissioned Report, CR/02/079N. National Environmental Research Council, London. 105p.

Barrett, M., Howard, G., Pedley, S., Taylor, R., and Nalubega, M., (1998). *A comparison of the extent and impacts of sewage contamination on urban groundwater in developed and developing countries*. In: Chorus, I., Ringelband, U., Schlag, G., and Schmoll, O., (Edited) Water, Sanitation and Health World Health Organisation Water Series. IWA Publishing. pp 179-185.

Barrett, H.M., Nalubega, M., Howard, G., Taylor, G.R., and Pedley, S., (1999). *The impact of on-site sanitation on urban groundwater quality*. In: Miriam Fendekovà and Marian Fendesk (Edited) Proceedings of XXIX International Association of Hydrogeologists Congress-Hydrogeology and Landuse Management. IAH Publication.pp 335-340.

Barrett, M.H., Johal, K., Howard, G., and Pedley, S., (2000). *Sources of faecal contamination of shallow groundwaters in Kampala*. In: Sililo, O., and Rotterdam, O., (Edited) Groundwater: Past Achievements and Future Challenges, Proceedings of the XXX IAH congress on groundwater, Cape Town, South Africa. AA Balkema Publishers, Rotterdam. pp 691-696.

Barrett, M.H., and Howard, A.G., (2002). *Urban Groundwater and Sanitation-Developed and Developing Countries*. In: Howard, K.W.F., and Israfilov, R. G., (Edited) Current Problems in Hydrogeology in Urban areas. Kluwer Publishers, Netherlands. pp 39-56.

Biryabarema, M., (2001). *Groundwater contamination and the role of geologic strata in retaining pollutants around waste disposal sites in Greater Kampala*. PhD Thesis, Makerere University, Kampala-Uganda. 310p.

Boulding, R.J., (1995). *Practical Handbook of Soil, Vadose zone and Groundwater contamination: Assessment, Prevention and Remediation*. Lewis Publishers, USA. 728p.

Börling, K., (2003). *Phosphorus sorption, accumulation and leaching. Effects of long-term inorganic fertilization of cultivated soils*. PhD, Thesis, Swedish University of Agricultural Sciences, Department of Soil Sciences, Uppsala, Sweden. 88p.

BS (1975, 1990): *Methods of test for soils for civil engineering purposes*. Parts 2 & 5. British Standard Institution, London.

Chaggu, J.E., (2004). *Sustainable Environmental Protection Using Modified Pit-Latrines*. PhDThesis, Wageningen University, The Netherlands. 141p.

Comsol (2004). *Earth Science Module Model Library, FEMLAB Version 3.1*. COMSOL AB Publication, Stockholm, Sweden. 280p.

Dexter, A.R., (2004). *Soil physical quality Part III: Unsaturated hydraulic conductivity and general conclusions about S-theory*. In: Geoderma **120**: 227-239.

Domenico A. Patrick and Schwartz W. Franklin (1998). *Physical and Chemical Hydrologeology. 2nd Edition*, John Wiley & Sons, New York, N.Y. 528p.

Eijkelkamp (2004). *Operating Instructions for sandbox (08.01), Sandbox/Kaolin (08.02) and -Membrane apparatus (08.03)*. Eijkelkamp Agrisearch Equipment, The Netherlands.

Fetter C.W (1999). *Contaminant Hydrogeology, 2nd Edition.* Macmillan Pub. Co. 500p.

Gaspà, M.M., (2004). *Use of hydrochemical, microbiological and physical monitoring to determine contamination mechanisms of springwater discharging from a deeply weathered regolith aquifer, Kampala-Uganda.* Master Thesis, Centre of Hydrogeology, University of Neuchâtel, Switzerland. 80p.

Girish, C.S.N., and Varun, J., (2004). *Rainfall and spring discharge patterns in two small drainage catchments in the western Himalayan mountains, India.* In: The Environmentalist. Kluwer Academic Publishers, The Netherlands. **24**: 19-28.

HACH (1997). *DR/2010 Spectrophotometer Handbook (Model 49300-22).* Hach Company, USA. 824p.

HACH (1997/8). *BODTrak^{TM} Instrument Manual. Edition 1, Rev.5.* Hach Company, USA. 50p.

Heidmann, I., Christl, I., Leu, C., and Kretzchmar, R., (2005). *Competitive sorption of protons and metal cations onto kaolinite: experiments and modelling.* In: Journal of Colloid and Interface Science, Elsevier Science Publishers B.V. **282**: 270-282.

Helalia, A.M., (1993). *The relation between soil infiltration and effective porosity in different soils.* In: Agricultural Water Management. Elsevier Science Publishers B.V. **24**: 9939-47.

Hodnett, M.G., and Tomasella, J., (2002). *Marked differences between van Genuchten soil water retention parameters for temperate and tropical soils: a new water-retention pedo-transfer function developed for tropical soils.* In: Geoderma Elsevier Science B.V. **108**: 155-180.

Houba, V.J.G., Van der lee, J.J., and Novozamsky, I., (1995). *Soil Analysis Procedures (Soil and Plant Analysis, Part 5B). A series of Syllabi, 6th Edition.* Department of Science and Plant Nutrition, Wageningen Agricultural University, The Netherlands. 262p.

Howard, G., Mutabazi, R., and Nalubega, M., (2001). *Rehabilitation of protected springs in Kampala, Uganda.* In: Pickford, J (Edited) Water, sanitation and hygiene: challenges of the millennium, Proceedings of the 26th WEDC Conference, Dhaka, Bangladesh. Water, Engineering and Development Centre (WEDC), Loughborough University, UK. pp: 20-23.

Howard, G., Pedley, S., Barrett, M., Nalubega, M., and Johal, K., (2003). *Risk factors contributing to microbiological contamination of shallow groundwater in Kampala, Uganda.* In: Water Research. Elsevier Science Ltd. **37**: 3421-3429.

Hue, N.V., (1991). *Effects of organic acids/anions on P sorption and phytoavailability in soils with different mineralogies.* Soil Science **152**:463-471.

Jacks, G., Sefe, F., Carling, M., Hammar, M., and Letsamao, P., (1999). *Tentative Nitrogen Budget for Pit Latrines – Eastern Botswana.* In: Environmental Geology **38 (3)**, pp 199-203.

Jamieson, R.C., Gordon, R.J., Sharples, K.E., Stratton, G.W., and Madani, A., (2002). *Movement and persistence of faecal bacteria in agricultural soils and subsurface drainage water: A review.* In: Canadian Biosystems Engineering, **44**:1.1-1.9.

Johnson D., J., (1975). *Disinfection, water and wastewater.* Ann Arbor Science Publishers, Inc., Michigan, U.S.A. 425p.

Kansiime, F., and Nalubega, M., (1999). *Wastewater treatment by a Natural Wetland: the Nakivubo Swamp, Uganda. Processes and Implications.* PhD Thesis, Wageningen University/International Institute for Infrastructural, Hydraulic and Environmental Engineering, Delft, The Netherlands. A.A. Balkema Publishers, Rotterdam, The Netherlands. 326pp.

Kasozi, N.J., Sabiti, E.N., and Bareeba, F.B., (2004). *Effect of feeding varying levels of banana peels supplemented with maize bran, cotton seed cake and Gliricidia sepium on the performance of lactating diary cows.* Paper presented at the Regional Symposium on Livestock Revolution Challenges and Opportunities, 15-18 March 2004, Nairobi, Kenya. 18p.

KCC, (1999). *A study on the Sanitation Quality Gap in the Central Division, Kampala City Council, Vol.V-Report Highlights, conclusions and interventions/recommendations and emerging issues.* The Public Health Department, Kampala City Council. 30p.

Kirabira John Baptist (2003). *Characterisation of Ugandan raw minerals for fire bricks before and after sintering*. Licentiate thesis, KTH, Stockholm. ISRN KTH/MSE-03/60-SE MEK/AVH. 125p.

Kookana, R.S., and Naidu, R., (1998). Vertical heterogeneity in soil properties and contaminant transport through soil profiles. In: Dillon Peter and Simmers Peter (Edited) International contributions to Hydrogeology, series 18 Shallow groundwater systems. Balkema Publishers, The Netherlands. pp 15-27.

KSSMP (2004). *Sanitation strategy and Master plan for Kampala city. Master Plan, Vol.2- Main Report*. Government of Uganda, Ministry of Water for Lands and Environment, National Water and Sewerage Corporation. 240p.

KUSP (2004). *Surveillance of spring water sources in Kampala District*. First Interim Report by Chemiphar (U) Ltd for Kampala Urban Sanitation Project (KUSP), Kampala City Council. 50p.

Landon, J.R., (ed.) (1991). *A Handbook for soil survey and agricultural land evaluation in the tropics and subtropics*. Longman Scientific and Technical Group Ltd, Hong Kong. 474p.

Lewis W. J., Foster, S.D.S., and Drasar S.B., (1982). *The Risk of Groundwater Pollution by On-Site Sanitation in Developing Countries, A Literature Review*. International Reference Centre for Wastes Disposal (IRCWD) Report No. 01/82, Duebendorf, Switzerland.79p.

Lukman Salifu and Doyen Jean (1996). *Urban Sanitation: The Need for a Strategic Approach*. In: Volume II, Papers & Presentations, Reform of the Water Supply & Sanitation Sector in Africa.

Lwasa, S., (1999). *The impact of Drainage and Solid waste Management on Environmental Quality in Unplanned Settlements in Kampala: A Case Study of Bwaise III*. Masters Degree Thesis in Land Use and Regional Development Planning, Department of Geography, Makerere University, Kampala-Uganda. 102p.

Marshall, D.J., Shimada, W.B., and Jaffe, R.P., (2000). *Effect of temporal variability in infiltration on contaminant transport in the unsaturated zone*. In: Journal of Contaminant Hydrology. Elsevier Science B.V Ltd. **46**: 151-161.

Stumm, J., and Morgan, J.J., (1981). *Aquatic chemistry, An introduction emphasizing chemical equilibria in natural waters, second edition*. John Wiley & Sons Inc., New York. 780p.

Mato, R.R.A.M., (2002). *Groundwater Pollution in Urban Dar es Salaam, Tanzania: Assessing Vulnerability and Protection Priorities*. PhD Thesis, Eindhoven University of Technology, The Netherlands. 134p.

McCarty, P.L., and Roberts, P.V., (eds) (1990). *Contaminants in the Subsurface Environment*. In: Water Science and Technology. Elsevier Science B.V. Ltd. **22** (6): 1-6.

Morris, B.L, Lawrence, A.R.L, Chilton, P.J.C, Adams, B., Calow, R.C., and Klinck, B.A., (2003). *Groundwater and its susceptibility to Degradation: A Global Assessment of the Problem and Options for Management*. Early Warning and Assessment Report Series, RS.03-3. United Nations Environment Programme, Nairobi, Kenya. 126p.

NEMA, (1997). *Environmental standards and preliminary environment impact assessment for water quality and discharge of effluent into water/land*, 1st Edition. National Environment Management Authority, Kampala-Uganda. 90pp.

Niwagaba, B.C., (2002). *An assessment of point source pollution and its effect on the self-purification capacity of Nakivubo channel, Kampala*. MSc. Dissertation, Makerere University-Kampala, Uganda. 97p.

Nunzio, R., and Alessandro, S., (1997). *Effectiveness of using pedo-transfer functions to quantify the spatial variability of soil water retention characteristics*. In: Journal of Hydrology. Elsevier Science Ltd Publishers. **202**: 137-157.

Nuwagaba, A., and Mulogo, M.E., (2000). *Kampala Urban Sanitation Project (KUSP) Final Report.* Submitted to Kampala City Council October 2000. 20p.

Okalebo, J.R., Gathua, K.W., and Woomer, P.L., (2002). *Laboratory methods of soil and plant analysis: A working manual, 2nd Edition*. Tropical Soil Biology and Fertility Programme. Sacred Africa Publishers, Nairobi, Kenya. 88p.

Palamuleni, G.L., (2002). *Effect of sanitation facilities, domestic solid waste disposal and hygiene practices on water quality in Malawi's urban poor areas: a case study of South Lunzu Township in the city of Blantyre*. In: Physics and Chemistry of the Earth. Elsevier Science Ltd Publishers. **27**: 845-850.

Parkinson, J., and Tayler, K., (2003). *Decentralized wastewater management in peri-urban areas in low-income countries*. In: Environment and Urbanization: Rural-urban transformations. International institute for Environment and Development (IIED) Publication, London, UK. **15 (**1): 75-89.

PLAN, (2001). *Feasibility Study Project Preparation Bwaise III-Kawempe Division. Field Report on Community, Environment, Sanitation and Solid waste Management, Drainage and Flood Aspects*. COWI Consults for Plan International Uganda. 137p.

Robins, N.S., (1998). Recharge: The Key to Groundwater Pollution and Aquifer Vulnerability. In: Robins, N.S (ed.*) Groundwater Pollution, Aquifer Recharge and Vulnerability*. Geological Society, London, Special Publications. 130: 1-5.

Schnoor L.J., (1996). *Fate and Transport of Pollutants in Water, Air and Soil.* John Wiley & Sons, Inc., USA. 704p.

Schwartz W.F., and Zhang, H., (2003). *Fundamentals of Groundwater. Wiley International Edition*. John Wiley & Sons, Inc., Newyork, USA. 592p.

Sergio E.S., (1997). *Hydrology for Engineers, Geologists and Environmental Professionals, An integrated treatment of surface, subsurface and contaminant hydrology*. Hydroscience Inc., USA. 480p.

Sikyajula, E., (2003). *An assessment of the current solidwaste management practices in Kampala with respect to sustainability*. BSc. Civil Engineering Dissertation, Makerere University, Kampala, Uganda. 64p.

Sloot, J., (1996). *Nitrification, Sediment phosphorus sorption and sediment nutrient release in Nakivubo swamp*. BSc. Thesis for the award of degree in Environmental Hygiene from Wageningen Agricultural University, The Netherlands. 35p.

Soet, M., and Stricker, J. N. M., (2003). *Functional behaviour of pedotransfer functions in soil water flow simulation.* In: Hydrological Processes. John Wiley & Sons, Ltd. **17**: 1659-1670.

Suprayogo, D., (2003). *Report on use of existing data to test "pedotransfer functions" for tropical soils-Appendix II*. In: Support for an Agroforestry Model user group, Final Technical Report. Forestry Research Programme, Project R7936/ZF0149, DFID. Indonesia.

Taylor, G.R., and Howard, W.F.K., (1998). *The dynamics of groundwater flow in the regolith of Uganda*. In: Dillon, P., and Simmers, I., (Edited) International contributions to hydrogeology, Series 18, Shallow Groundwater Systems. Balkema publishers, The Netherlands. pp 97-113.

Taylor, R., (2001). *Groundwater pumping and tracing tests:the unconsolidated, weathered gneiss aquifer of Iganga Town in Uganda*, Research Report for WRMD Workshop. 29p.

Taylor, R., and Barett, M., (1999). *Urban groundwater development in sub-saharan Africa*. In: Pickford John (Edited) Proceedings of 25th WEDC conference on integrated development for water supply and sanitation. Addis Ababa, Ethiopia. Water, Engineering and Development Centre (WEDC), Loughborough University, UK. pp 203-207.

Taylor, R.G., and Howard, K.W.F (2000). *A tectonic geomorphic model of the hydrogeology of deeply weathered crystalline rock: evidence from Uganda*. In: Hydrogeology Journal. Springer Verlag. **8**: 279-294.

UBOS (2002). 2002 *Uganda population and housing census-provisional results*. Uganda Bureau of Statistics, Kampala-Uganda.

UNEP (2002). *Africa Environment Outlook. Past, Present and Future Perspectives*. United Nations Environment Programme Publication. Earth Print, Nairobi. 448p.

Van Genuchten, M.Th., Leij, F.J., and Yates, S.R., (1991). *The RETC Code for quantifying the hydraulic functions of unsaturated soils*. Robert S. Kerr Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, ADA, Oklahoma 74820. EPA/600/2-91/065. 117p.

Vernon, H., (2002). *Urbanization in developing countries*. In: World Bank Research Observer, Oxford University Press, UK. **17** (1): 89-112.

Wagner, B., Tamawski, V.R., Hennings, V., Müller, U., Wessolek, G., and Plagge, R., (2001). *Evaluation of pedo transfer functions for unsaturated soil hydraulic conductivity using an independent data set*. In: Geoderma. Elsevier Science B. V. Ltd. **102**:275-297.

Wetzel, R.G., (1983). *Limnology, 2nd Edition*. Saunders College Publishing, Florida, USA. 767 p.

WHO, (2004). *Guidelines for Drinking Water Quality, 3rd Edition, Vol. 1 Recommendations.*World Health Organization, Geneva. 366p.

Wösten, J.H.M., Finke, P.A., and Jansen, M.J.W., (1995). *Comparison of class and continuous pedotransfer functions to generate soil hydraulic characteristics*. In: Geoderma. Elsevier Science Ltd Publishers. **66**: 227-237.

Wösten, J.H.M., Pachepsky, Y.A., and Rawls, W.J., (2001). *Pedotransfer functions:bridging the gap between available basic soil data and missing soil hydraulic characteristics*. In: Journal of Hydrology Elsevier Science B.V. **251**: 123-150.

WSP/NWSC, (2000). *Identification of Management Options for Improved Water and Sanitation Services in Informal Settlements in Kampala, Situation Assessment Report*. For Water and Sanitation Programme and National Water and Sewerage Corporation by Aqua Consults Consulting Engineers, Kampala. 140p.

OTHER REFERENCES

Foster Stephen, Garduño Hector, Kemper Karin, Tuinhof Albert, Nanni Marcella and Dumars Charles (2003). *Briefing Note 8: Groundwater quality protection: defining strategy and setting priorities*. Sustainable Groundwater Management: Concepts and Tools-Groundwater Management Advisory Team (GW-MATE) Briefing Note Series. World Bank Group, Global Water Partnership Associate Program. Published by The World Bank, Washington D.C., USA. 6p. www.worldbank.org/gwmate.

Jansson, P-E, and Karlberg, L., (2004). *Coupled heat and mass transfer model for soil-plant-atmosphere systems*. Royal Institute of Technology, Department of Civil and Environmental Engineering, Stockholm. 435p.

ftp://www.lwr.kth.se/CoupModel/CoupModel.pdf

Mann, R.A., (1996). *Phosphorus removal by constructed wetlands: Substratum adsorption*. PhD Thesis, Faculty of Science and Technology, University of Western Sydney (Hawkesbury), Austrailian digital theses program. 283p.

http://library.uws.edu.au/adt-NUWS/public/adt-NUWS20030801.081135/

NFESC, (1999). *Natural Attenuation General Data Guide*. User's Guide UG-2035-ENV. Naval Facilities Engineering Service Center , Port Hueneme, California 93043-4370. 33p http://enviro.nfesc.navy.mil/erb/erb_a/restoration/technologies/remed/comb_mech/UG-2035.pdf

Sobehrad Susan (1997). *Phosphorus cycling in the estuarine environment*. Research paper. http://bellnetweb.brc.tamus.edu/phosphor.htm.

EPA (1991). *Compendium of ERT Groundwater Sampling Procedures*. EPA/540/P-91/007. U. S. Environmental Protection Agency, Washington, DC. 63p.

http://www.epa.gov/region09/qa/pdfs/fieldsamp-ertsops.pdf.

Azadpour-Keeley, A., Russell, H.H., and Sewell, W.G., (1999). *Microbial Processes Affecting Monitored Natural Attenuation of Contaminants in the Subsurface*. United States Environmental Protection Agency Publication, EPA/540/S-99/001. 18p.

http://clu-in.org/download/remed/microbial.pdf.

USEPA (1992). *Batch -Type procedures for estimating soil adsorption of chemicals*. Technical Resource Document, EPA/530/SW-87/006-F, United States Environmental Protection Agency (USEPA), Washington, DC. 116p.

http://www.epa.gov/cgi-bin/clariton/clhtml/pubtitleOSWER.html.

USEPA (1995). *Technical Guidance Manual for Hydrogeological Investigations and Groundwater Monitoring*. Ohio, Environment Protection Agency, USA. 41p.

http://www.epa.state.oh.us/ddagw/Documents/chapter03HYDRO.PDF

APPENDIX - PAPERS