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AN INVESTIGATION OF GROUNDWATER VULNERABILITY WITHIN THE VICINITY OF A LANDFILL: A CASE STUDY OF POMONA LANDFILL, HARARE

By

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To my **Miracle**, mummy loves you so much. A great **Testimony** is coming our way.

DECLARATION

I, **Takudzwa Faith Chihanga**, declare that this research is my own work and additional sources used have been properly and fully acknowledged by means of references. This dissertation has not been submitted before for any degree or examination in any other University.

I am responsible for the research and its articulation alone. In no way do any of the persons mentioned in the acknowledgement bear any direct responsibility for this work.

Signature: ________________________

Date: ________________________

ABSTRACT

Landfills are considered to be one of the major threats to groundwater quality. The study focused on Pomona Landfill in Harare which is the only official active landfill for Harare City Council. The main purpose of this study was to investigate groundwater vulnerability within the vicinity of the landfill. Groundwater and leachate samples were collected and analysed for selected water quality parameters and compared to Zimbabwe Standards of Drinking Water Quality and World Health Organization Guidelines. Groundwater samples were collected from eight points, two located upstream of the landfill and four located downstream of the landfill and two within the landfill. Leachate samples were collected from a leachate trench at the centre of the landfill and from a pond down-gradient where the leachate was drained out by gravity. Samples were collected and analysed, from February 2015 through June 2015 giving a total of thirty-two (32) groundwater samples and eight (8) leachate samples. The Hydrologic Evaluation of Landfill Performance (HELP) Model was used to estimate the quantity of leachate generated. In this study, the DRASTIC Model was also used for a part of Pomona area to generate a small-scale map of groundwater vulnerability to pollution. The results showed that, most parameters (87%) satisfied the Zimbabwe Standards of Drinking Water Quality and the stipulated World Health Organization potable water guidelines except turbidity and iron. The average volume of leachate discharged from Pomona Landfill during the period 1983 to 2014 was 94 486 m^3/year. The average annual leakage from the landfill base was 13% of the average annual total precipitation of 708 140 m^3/year. Four different vulnerability zones were determined, namely low vulnerability (38%), moderate vulnerability (58%), high vulnerability (3%) and very high vulnerability (1%). The current results show insignificant impact of the landfill operations on the groundwater resource. The existing soil stratigraphy at the landfill site consisting of clay and silt-clay is deduced to have influenced natural attenuation of leachate into the groundwater resource. It is however observed that in the absence of a properly designed leachate collection system, uncontrolled accumulation of leachates at the base of the landfill pose potential contamination risk to groundwater resource in the very near future. It is recommended that groundwater be monitored regularly and a properly engineered landfill be constructed.

Keywords: DRASTIC Model, Groundwater quality, HELP Model, Leachate, PCA, Pomona Landfill

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LIST OF SYMBOLS AND ABBREVIATIONS

As…………………………………...Arsenic

BOD………………………………...Biological Oxygen Demand

Cd…………………………………...Cadmium

C.F.U……………………………….Colony Forming Units

Cl^-………………………………….. Chloride

COD………………………………...Chemical Oxygen Demand

Cu…………………………………...Copper

E.C…………………………………. Electrical Conductivity

EMA……………………………….. Environmental Management Agency

FC………………………………….. Faecal Coliforms

Fe……………………………………Iron

HELP………………………………..Hydrologic Evaluation of Landfill Performance

Hg…………………………………...Mercury

MSW………………………………...Municipal Solid Waste

NO_3^- ………………………………...Nitrate

Pb…………………………………….Lead

PCA………………………………….Principal Component Analysis

PO_4^{3-}…………………………………Phosphate

SAZ…………………………………..Standards Association of Zimbabwe

SPSS………………………………….Statistical Package for Social Sciences

SO_4^{2-}………………………………….Sulphate

TDS…………………………………..Total Dissolved Solids

TC…………………………………….Total Coliforms

USEPA……………………………….United States Environmental Protection Agency

WHO…………………………………World Health Organisation

Zn……………………………………..Zinc

CHAPTER ONE: INTRODUCTION

1.0 Background

Large quantities of solid waste are produced daily as a result of human activities (Awaz, 2015). Baily (1990) defines waste as unwanted material arising from household, municipal and commercial facilities, industrial effluents and sludge. The pattern of waste generation is a function of the level of urbanization, industrialization and the economic status of a society (Afolayan, 2012). The quantum of MSW generated is increasing rapidly and it is beyond the assimilation capacity of nature (Agrawal, 2013). The lack of efficient management for solid waste disposal leads to pollution of cities and has adverse impacts on human health and the environment (Yadav and Devi, 2000; Jha et al., 2003). Waste management involves five stages; waste generation, storage, collection, transportation and disposal (Musademba et al., 2011). The composition of solid waste is an important issue in waste management; it affects the density of the waste and is necessary for examining reuse, reduction and recycling of waste and also the appropriate method of waste disposal (Al-Khatib et al., 2010). Municipal Solid Waste (MSW) in developing countries has a much larger proportion of organic waste than in developed countries (World Bank, 2012). Landfills and open dumps are a common municipal solid waste disposal practice and one of the cheapest methods for organized waste management in many parts of the world (Jhamnani and Singh, 2009; Longe and Balogun, 2010). Up to 95% of the total MSW collected worldwide is disposed of in landfills (Ahmed and Sulaiman, 2001; World Bank, 1992).

Waste degradation in MSW landfill is a complex process; once waste is deposited at the landfill (dumpsite) pollution can arise from the migration of both gas and leachate (Al-Khateeb, 2002). In municipal solid waste landfills, both solid and semi-solid wastes are biodegraded anaerobically by microorganisms, producing gas and soluble chemicals that combine with liquid in the waste to form leachate (USEPA, 2009). Leachate can be defined as a liquid that passes through a landfill and contains dissolved and suspended matter from the landfill (Raghab et al., 2013). This leachate accumulates at the bottom of the landfill and percolates through the soil (Mor et al., 2006). The type of solid waste, physical, chemical, and biological activities may determine the quality of leachate (Warith, 2003). The composition of leachate is important in determining its potential effects on the quality of nearby surface water and groundwater (Škultétyová, 2009). Leachate can pollute large amounts of groundwater rendering it unsuitable for use for domestic water (Al-Khateeb, 2002). The cost of cleaning up groundwater contaminated by landfill leachates require large sum of money and technology (Lasisi, 2011).

Globally, in most cities especially in developing countries, solid waste management has become a challenge (Mangundu et al., 2013). Solid waste management has become a major problem in Zimbabwe's towns and cities and the problem is increasing due to urbanization, population growth, industrialization and increased use of non-biodegradable plastics and bottles (Kaseke, 2005). For example, Practical Action Southern Africa (2006) alludes that more than 2.5 million tons of household and industrial waste are produced per annum in urban areas across Zimbabwe with food waste constituting about 70% of the total amount of waste. Per capita solid waste generation in Harare averages 0.481 kg/day and the waste is predominantly biodegradable (Pawandiwa, 2013).

1.1 Statement of the problem

Due to rapid urbanization, groundwater is becoming increasingly vulnerable to pollution from human activities (Aldrick et al., 1999). Landfills have been identified as some of the major threats to groundwater resources (Fatta et al., 1999) especially where they are unlined, as is common in the developing world (Hranova, 2006). Diffuse pollution of surface water in Harare has been studied, and little is known about groundwater pollution in Harare (Love et al., 2006). Since literature has shown that municipal solid waste landfills are often major sources of groundwater contamination (Lee and Jones, 1991; USEPA, 1993; Al-Yaqout and Hamoda, 2003), there is a possibility that Pomona Landfill could be a source of groundwater contamination. Therefore the quality of groundwater in Pomona area, which potentially can be affected by the landfill, must be carefully investigated.

1.2 Justification

The supply of clean water is one of the most important of Southern Africa's concerns, with demand rising at around three per cent per annum (Laisi and Chenje, 1996). Harare, the capital city of Zimbabwe, is facing water quantity and quality problems, with serious pollution of the downstream Lake Chivero (Nhapi, 2009). Increasing populations and an improved quality of life (leading to greater personal water use) have reduced the quantity of water available per person (Xu and Usher, 2006). Urban population increase leads to an increase in the demand of urban groundwater as a source of drinking water (Love et al., 2006). This is also the case with Harare City. Urban groundwater is in fact thought to supply up to half of the world's urban population (Foster, 1999). The impact of leachate on water sources therefore needs to be investigated given the increased groundwater use in Harare.

According to Qasim and Chiang (1994) depending on the composition and extent of refuse and hydrological factors the leachate may be highly contaminated. The knowledge of the composition of leachate helps to ascertain the contamination potential it poses to the immediate ecosystem (D'Souza and Somashekar, 2013). The investigation of impacts of landfill leachate on groundwater is important to the management and disposal of municipal waste.

Landfills are supposed to be sited away from residence because of the inherent environmental nuisance and poor aesthetic value associated with their operations (Kola-Olusanya, 2012). Rapid urbanization has resulted in existing dumping sites originally located at a safe distance outside the municipal boundaries now being increasingly encircled by settlements and housing estates (Schertenleib and Meyer, 1992). Harare is no exception, with approximately twenty thousand (20,000) residents added to the city each year (Zimbabwe National Statistics Agency, 2013). While relatively few scientific studies have been conducted regarding adverse health effects of waste dumps and landfill sites, a study in five European countries found that living near a landfill can raise the risk of having a child with birth defects (such as Downs Syndrome) by as much as 40 per cent (Vrijheid, 2000). Many toxic pollutants released by leachates into the groundwater are not readily removable by the conventional water treatment process, therefore it is essential to carry out an intensive study and monitor the nature and extent of such pollution on groundwater quality.

1.3 Objectives

1.3.1 *Main objective*

The main objective of the study was to investigate groundwater vulnerability within the vicinity of Pomona Landfill in Harare.

1.3.2 *Specific objectives*

The specific objectives were;

1. To analyse groundwater samples for selected water quality parameters in order to check the suitability of the water for potable use.
2. To analyse the leachate composition generated at Pomona Landfill in order to identify compounds in undesirable concentration.
3. To assess the generated leachate quantity using the Hydrologic Evaluation of Landfill Performance (HELP) Model.
4. To assess groundwater vulnerability to contamination using a GIS-based DRASTIC Model.

CHAPTER TWO: LITERATURE REVIEW

2.0 Introduction

Waste management has become an issue of growing global concern as urban populations continue to increase and consumption patterns change (UNEP, 2013). The rate and scale of urbanization has increased in recent decades, with more than half the world's population now living in urban centres (Tacoli, 2012; UNPD, 2012a). The rapid urbanization has seriously aggravated the problem of municipal waste disposal and management (Bhalla et al., 2013). National and municipal governments often have insufficient capacity or funding to meet the growing demand for solid waste management services (Tacoli, 2012). Solid waste management is the single largest budget item for many cities (UN-HABITAT, 2010; World Bank, 2012). In developing countries, open dumpsites are the most common method of disposing of solid waste (World Bank, 2012). Leachate is produced when the waste becomes saturated with water (Marian and Benson, 1999). The leachate generated from solid waste dumps may have the potential to pollute soils and the surrounding water sources (Khan, 2001). The adverse impacts of leachate on the surrounding environment depend on the characteristics of this leachate (Karaca and Bestamin, 2006). Similar contaminants may behave differently in the same environment due to the influence of other constituents in a complex leachate (Futta et al., 1997).

Dumpsites have been linked to many harmful health effects, including skin and eye infections, respiratory problems, vector-borne diseases such as diarrhoea, dysentery, typhoid, hepatitis, cholera, malaria and yellow fever, high blood lead levels and exposure to heavy-metal poisoning (UNEP, 2011). The solid waste dumps, if not managed properly, may cause many types of social and environmental problems, like groundwater pollution, air pollution, soil contamination, odour nuisance and fly nuisance (Zurbugg, 2002). However, the most serious problem is groundwater contamination (Sabahi et al., 2009). Millions of people in the developing world rely heavily on groundwater, mostly through shallow dug wells (Blacksmith Institute, 2015). The U.S. Environmental Protection Agency (1980b) estimates that between 0.1% and 0.4% of usable surface aquifers in the world are contaminated by industrial impoundments and landfills. The nature of groundwater pollution is complicated, imperceptible and its impact is persistent; while its treatment is expensive. Thus, prevention and control of groundwater pollution are principally crucial for its effective management (Tesoriero et al., 1998; Thirumalaivasan et al., 2003; Babiker et al. 2005; Huan et al., 2012; Hallaq and Elaish, 2012; Yin et al., 2012).

In recent years, many studies have been carried out on the assessment of groundwater quality near landfill sites using different approaches and methodologies to find out the level of groundwater pollution, bacterial contamination and the concentration of heavy metals. A number of scholars (e.g. Abu-Rukah and Al-Kofahi, 2001; Mor et al., 2006; Vasanthi et al., 2008; Al-Sabahi et al., 2009; Jhamnani and Singh, 2009; Longe and Balogun, 2010; Akinbile and Yusoff, 2011) have examined possible water contamination around municipal landfills by using the microbiological examination and physicochemical analysis of leachate and groundwater. Results by Al-sabahi et al. 2009 showed that 4 out of 5 boreholes were contaminated, where concentration of physico-chemical parameters were above the standard acceptable levels required for drinking water by Yemen's Ministry of Water and Environment. They therefore concluded that the landfill posed great risk to the environment. Mor et al. 2006 concluded that leachate has significant impact on groundwater quality near the area of Gazipur landfill site, India. Longe and Balogun, 2010 concluded that there was insignificant impact on groundwater underlying Solous landfill site in Nigeria.

2.1 Groundwater resources

Groundwater is the primary source of drinking water for half of the world's population (IAEA, 2014). Worldwide, about 1.5 billion people depend upon groundwater for their drinking water supply (World Bank, 1998; WRI, 1998; UNDP, 2008; UNEP, 2008). The amount of groundwater withdrawn annually is roughly estimated at 600 – 700 km³, representing about 20 % of global water withdrawals (WMO, 1997). Groundwater is a globally important and valuable renewable resource for human life and economic development (Kaur and Rosin, 2011). Groundwater is generally a very good source of drinking water because of the self-purifying properties of soil (Coe, 1970). Even where surface water is abundant, rivers and lakes may be contaminated with disease-causing organisms such as guinea worm or bilharzia. In such cases, groundwater may be an alternative (IAEA, 2014). The dominant role of groundwater resources is clear and their use and protection is, therefore, of fundamental importance to human life and economic activity (Chapman, 2002).

Groundwater is the most important source of water supply in arid and semi-arid regions due to its large volumes and its low vulnerability to pollution when compared to surface waters (USEPA, 1985). Water use in Africa is set to increase markedly over the next few decades as a result of population growth (Vörösmarty et al., 2005).

2.2 Groundwater Contamination

Groundwater pollution is the artificially induced degradation of natural groundwater quality (Rahman, 2008). Any substance added to water that may prospectively temper with its quality; thereby undermining its usage value is referred to as 'water pollution' (Bachmat and Collin, 1990). In contrast with surface water pollution, subsurface pollution is difficult to detect, is even more difficult to control, and may persist for years, decades, or even centuries (Todd, 1980). Contamination of water supplies also creates problems for irrigation and industrial uses, requiring additional processing, which is expensive (UNEP, 2002). Specific locations, where pollution is as a result of human activities; such as discharges from sewage treatment works, industrial wastewater outlets, solid waste disposal sites, animal feedlots and quarries (Figure 2), can be described as point sources (Bartram and Balance, 1996).

Diseases may be contracted through groundwater contamination, and rapidly spread due to groundwater flow mechanism (Afolayan et al., 2012). Water remains the major cause of illness in both developed and developing nations (Baba and Tayfur, 2011; Jones and Watkins, 1985), and good quality water is one of the criteria for a region's socio-economic development. There are four types of contaminant transport mechanisms and these are dispersion, dilution, advection and convection. Contaminants that are dissolved in water are solutes and the water is the solvent and the combination is the solution. As the water flows, the contaminants are transported with the water a process known as advection. As the water flows around the soil particles, it is mixed, a process known as mechanical dispersion. The result is dilution or reduction in the contaminant concentration.

Due to general high population growth and industrialization, greater amounts of domestic and industrial effluents are being discharged, which has led to the pollution of groundwater (Rahman, 2008). There are several types of pollutants that appear to predominate in groundwater such as heavy metals, nutrients, pesticides and other organic chemicals (Sener and Davraz, 2013). Leaching of various pollutants through the vadose zone gives rise to contamination. Leaching processes vary from one location to another (Baalousha 2006; Sener et al. 2009). The intrusion of pollutants from different sources to groundwater alters the water quality and reduces its value to consumers (Melloul and Collin, 1994). In recent years, widespread reports of bacteria, nitrate, synthetic organic chemicals and other pollutants in groundwater had increased public concern about the quality of groundwater (Mahler et al., 1988).

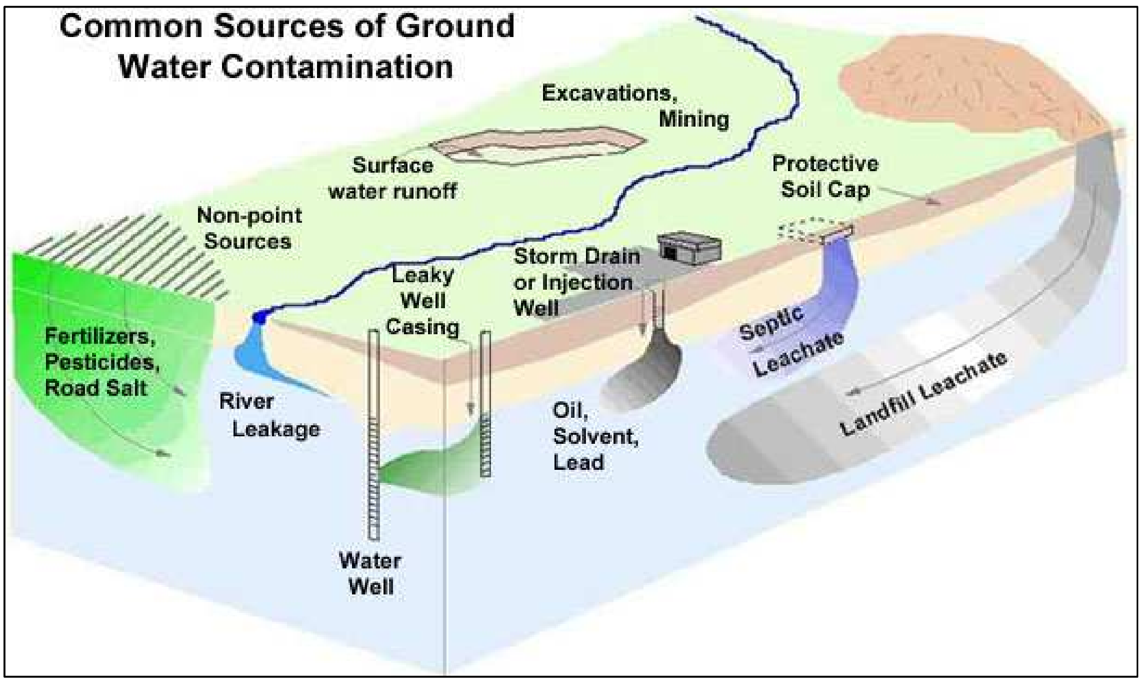

Figure 1: Sources of groundwater contamination (Source: The Groundwater Foundation, 2015)

2.3 Pollution by Landfills

2.3.1 *Landfills*

Sanitary landfilling is an engineered technique of disposing solid waste on land by spreading them in thin layers followed by compacting them to the smallest practical volume before covering them with soil at regular intervals (Brunner and Keller, 1972). Sanitary landfilling involves placing waste in lined pits with appropriate means of leachate and landfill gas control (Alloway and Ayres, 1997; Eludoyin and Oyeku 2010). It is highly recognized as an environmentally and internationally desired technique of solid waste disposal since it minimizes environmental damage and thus eliminates odours (Zamba, 2014). EPA, (2001) in the manual for landfill design indicates that landfills are supposed to have a bottom layer to prevent contaminants from getting into surface and/or groundwater. However, if there is no layer or it is cracked, contaminants from the landfill can make their way into groundwater (Himanshu and Senapati, 2015). The practice of landfill system as a method of waste disposal in many developing countries is rarely practised (Mull, 2005; Adewole, 2009; Eludoyin and Oyeku, 2010). According to Taylor and Allen (2006), waste deposited in landfills or in refuse dumps immediately becomes part of the prevailing hydrological system. Internationally, almost 70% of MSW is disposed of at landfills (OECD, 2001; Zacarias-Farah and Geyer-Allely, 2003). Most disposal sites in Zimbabwe are often categorized as landfills but in technical terms they are not, since they do not have a geo-synthetic liner and leachate collection ponds as per requirements (Zamba, 2014). Masocha, (2004) highlighted that Pomona landfill lacks geo-synthetic membranes to prevent groundwater pollution from leachates hence it is a non-engineered landfill. Landfilling has been widely used as the preferred option for solid waste disposal in large cities such as Harare and Gweru after the realization by the local authorities that crude dumping poses adverse health risks to residents (Zamba, 2014) and contributes to a lower aesthetic value of cities.

In developed countries, landfills have historically been the primary method of waste disposal because this method is the most convenient and because the threat of groundwater contamination was not initially recognized (Smith, 2006). Landfilling is an easy and cheaper method of waste disposal (Butt et al., 2008; Longe and Balogun, 2010) than incineration. These attributes have contributed to its extensive use in many parts around the world (Tatsi and Zouboulis, 2002). Municipal solid waste landfills generate leachate that constitutes a pollution source into the environment and water resources (Al-Yaqout and Hamoda, 2003; Butt et al., 2008).

2.3.2 *Leachate generation and composition*

During landfill operation, leachate is generated from the microbiological decomposition of solid urban wastes, infiltration of rainwater through the refuse tips and moisture within the waste in a landfill (Salem et al., 2008). Generally, it is expected that landfills undergo at least four phases of decomposition, (1) an initial aerobic phase, (2) an anaerobic acid phase, (3) an initial methanogenic phase, and (4) a stable methanogenic phase (Christensen and

Kjeldsen, 1995). An additional aerobic or humic phase of decomposition has been proposed (Christensen and Kjeldsen, 1995; Bozkurt et al., 2000). According to Kostova, 2006, the concentration of leachate constituent are in phases namely transition (0 -5 years), acid-formation (5-10 years), methane fermentation (10-20 years) and final maturity (>20 years). The leachate composition and its pollution intensity depend on many aspects such as landfill age, waste ingredient and hydraulic conditions of landfill (Bidhendi et al., 2010). Contamination of groundwater is directly associated with the lifespan of the landfill (Oyiboka, 2014). The ageing of a landfill is accompanied by increased leachate quantity. Even after a landfill has been decommissioned, refuse will continue to decompose (Kjeldsen et al., 2002). An understanding of leachate composition is critical for making projections on the long-term impacts of landfills.

According to Afolayan et al. (2012), leachate formation is the function of the type of waste, season, climate, time and management strategy while its migration and pollution depend on surface water, topography, distance, underlying geology, soil and depth of the land in relation to piezometric level. The issue of distance was further corroborated by Ohwoghere–Asuma and Aweto, (2013), who stated that distance and depth of the sink from the source of leachate had greater impact on the degree and extent of contamination of ground and surface water. They found that wells in the proximity of waste dumps had more concentrations of ions, cations and organic materials than those further from it. The entire decomposition process can take decades, the rate being very much a function of the amount of water that can gain access to the waste. A landfill should reach a final stable non-polluting state within out about 30 years. As time elapses, the produced leachate permeates into groundwater systems leading to change of physical and chemical properties of groundwater (Vasanthi et al., 2008). Longe and Enekwechi, (2007) and Lee et al. (1986) stated that heavy metals such as cadmium, arsenic, chromium have been reported at excessive level in groundwater due to landfill operation. Longe and Enekwechi (2007), report that the volume of leachate depends principally on the size of the landfill, the meteorological and hydrogeological factors and effectiveness of capping. Heavy metals such as lead, mercury, chromium, copper and cadmium, together with household chemicals and poisons can be concentrated in groundwater supplies beneath landfills (Wagner and Rhyner, 1984). Landfills release the widest suite of contaminants: sodium, potassium, ammonia, nitrate, nitrite, chlorides and heavy metals such as iron, cadmium, copper manganese, lead, zinc, mercury and chromium and xenobiotic organic substances such as drugs and food additives (Zhu et al., 1997; Dutova et al., 1999; Christensen et al., 2001).

2.3.3 *Leachate migration*

In an unlined landfill above an aquifer, leachate often accumulates within or below the landfill (Freeze and Cherry, 1979). According to Taylor and Allen (2006), this is due to the production of leachate by degradation processes operating within the waste, in addition to the rainwater percolating. The increased hydraulic head developed increases downward and outward flow of leachate from the landfill or dump. Downward flow threatens underlying groundwater resources. If the source continues to supply the contaminant over a period of time, the distribution of the dissolved contaminant will take a characteristic “plume like” shape (See Figure 4). Groundwater moves slowly and with little turbulence, dilution, or mixing therefore, once contaminants reach groundwater, they tend to form a concentrated plume that flows along with groundwater (Mason, 2015*)*. Lee and Kitanidis (1993) stated that leachate migration from disposal sites can be influenced by the site design, waste type, hydrogeology, geochemistry and climatological conditions. Due to the health impacts caused by landfill leachate, it is very important to estimate its quantity of leachate might reach the groundwater and study the effect of this leachate on groundwater (Alsaibi, 2009). As water moves through the ground, natural processes reduce (or attenuate) the concentration of many contaminants, including harmful micro-organisms. The degree to which attenuation occurs is dependent on the type of soil and rock, the types of contaminant and the associated activity. Attenuation is generally most effective in the unsaturated zone and in particular in the upper soil layers where biological activity is greatest.

2.3.4 *Potential Impacts of Landfills*

Leachate consists of a mixture of organic and inorganic compounds, many of which have a hazardous impact on the environment (Wang et al., 2004). In unlined landfills, the leachate continues to leach into the ground and may contaminate groundwater. Pollutants can escape from improperly designed landfill in a variety of ways. According to Longe and Balogun, (2010), the greatest contamination threat to groundwater comes from the leachate generated from the materials which most often contain toxic substances, especially when wastes of industrial origins are landfilled. Vasanthi et al. (2008) also noted that the produced leachate is normally composed of organic and inorganic compositions. Longe (2007) stated that heavy metals such as cadmium, arsenic, chromium have been reported at excessive levels in groundwater due to landfills operation. The rate and characteristics of leachate production depends on a number of factors such as solid waste composition, cover design, compaction, interaction of leachate with environment and landfill design operation, particle size, degree of compaction, hydrology and hydrogeology of site, age of landfill, moisture and temperature condition, and available oxygen (Longe and Balogun, 2010).

Leachates contain a host of toxic and carcinogenic chemicals, which may cause harm to both humans and the environment (Alslaibi et al., 2011; Laner et al., 2011; Singh et al., 2010). Furthermore, leachate-contaminated groundwater can adversely affect industrial and agricultural activities that depend on well water (Ashraf et al, 2013). Leachate impacts to groundwater may also present danger to aquatic species if the leachate-contaminated groundwater plume discharges to wetlands or streams (Eldridge, 2015.) Leachate then will follow the hydraulic gradient of the groundwater system.

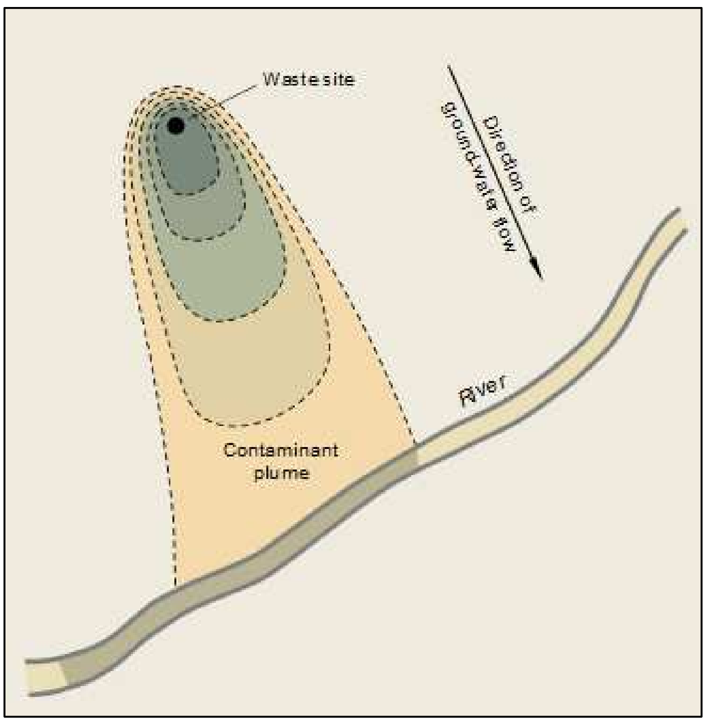

Figure 2: Contaminant plume (Source: USEPA, 1993)

2.4 Groundwater vulnerability assessment

Groundwater vulnerability is one of the key elements in decision making and it is considered in multi-criteria decision making tools in river basins and wastewater management systems (Kholghi, 2010). The tendency and likelihood for general contaminants to reach the water table after introduction at the ground surface is known as groundwater vulnerability (NRC, 1993). The aquifer vulnerability concept mainly involves two particular notions: intrinsic (or natural) vulnerability and specific (or integrated) vulnerability, which have been clearly defined within the European framework (Gogu and Dassargues, 2000; ZWAHLEN, 2004). Intrinsic vulnerability is a term used to define the vulnerability of groundwater to contaminants generated by human activities taking into consideration the inherent geological, hydrological, and hydrogeological characteristics of an area, but is independent of the nature of the contaminants (Daly et al., 2002). Specific vulnerability is used to define the vulnerability of groundwater to particular contaminants taking into consideration the contaminant properties and their relationship with the various components of intrinsic vulnerability (Hamerlinck and Arneson 1998; Doerfliger et al. 1999; Gogu and Dassargues 2000; Varol and Davraz 2010). Different methods such as process-based methods, statistical methods, and overlay and index methods have been developed to evaluate groundwater vulnerability (Tesoriero et al., 1998). In most cases, these methods are analytical tools that try to relate groundwater contamination to land use activities (Bai et al., 2012).

Vulnerability assessment methods divide a geographical area into sub-areas in terms of its susceptibility to groundwater contamination; then, in areas prone to contamination, effective groundwater protection measures should be carried out (Guo et al., 2007). The concept of groundwater vulnerability to contamination was developed by Magrat (1968). Groundwater vulnerability mapping is based on the idea that some areas are more vulnerable to groundwater contamination than others (Piscopo, 2001). Once these areas are identified, they can be targeted by proper land use and intensive groundwater monitoring (Mendoza and Barmen 2006). Groundwater vulnerability assessments are widely used to prevent groundwater contamination as they can provide valuable information for locating vulnerable areas (Antonakos and Lambrakis 2007; Sener et al., 2009). However, groundwater vulnerability is strongly dependent on factors such as depth-to-water, recharge and land use conditions that may change in response to future changes in climate and/or socio-economic conditions (Ruopu and Merchant, 2013).

2.4.1 *Overlay-index methods*

Overlay-index methods rely primarily on qualitative or semi quantitative compilations and interpretations of mapped data (NRC, 1993). They usually involve the overlaying and aggregation of multiple maps. Tilahun and Merkel (2010) noted that these methods are quite effective in determining groundwater vulnerability since they are particularly suitable for use with geographic information systems (Watkins et al. (1996) suggests that groundwater models integrated into GIS can visually represent the spatial aspects of groundwater data as well as execute spatial calculations on data enabling further inferences to be made about susceptible areas. The main advantage is that some of the factors such as rainfall and depth to groundwater can be available over large areas, which makes them suitable for regional scale assessments (Thapinta and Hudak, 2003). They, therefore, constitute the most popular class of methods used in vulnerability assessment. Overlay-index methods are often preferred because the data they require are easily available for regional scale assessments (Jawed et al., 2012). Overlay-index methods are based on assembling information on the most relevant factors affecting aquifer vulnerability (soil type, geologic formation type, recharge.), which is then interpreted by scoring, integrating, or classifying the information to produce an index, rank or class of vulnerability.

In the category of overlay and index-based methods, several approaches have been proposed for developing aquifer vulnerability assessment maps such as GOD (Foster, 1987), IRISH (Beck et al., 1999), AVI (Stempvoort et al., 1993), EPIK (Doerfliger et al., 1999) and DRASTIC (Aller et al., 1987). The advantage of these methods is that they provide relatively simple algorithms to integrate a large amount of spatial information into maps of simple vulnerability classes or indices. The DRASTIC Model is the most popular method of vulnerability assessment (Aller et al., 1985; Al-Adamat et al., 2003; Evans and Myers, 1990; Bedessem et al., 2005; Hamza et al., 2007; Kim and Hamm, 1999; Leone et al., 2009; Piscopo, 2001; Thirumalaivasan et al., 2003; Rahman, 2008).

DRASTIC Model

The DRASTIC Model was developed for the United States Environmental Protection Agency by Aller et al. (1987). The acronym DRASTIC stands for the seven parameters used in the model which are: Depth to- water **(D)**, Net recharge **(R)**, Aquifer media **(A)**, Soil media **(S)**, Topography **(T)**, Impact of vadose zone **(I)**, Hydraulic conductivity **(C)**. Each factor is classified into ranges (continuous variables) or significant media types (thematic data). The rating assigned to each of these ranges indicates their relative importance within each parameter, in contributing to aquifer vulnerability (Babiker et al., 2005).

The DRASTIC Model is a standardized non-subjective method to compare the vulnerability over various hydrological settings. DRASTIC has been used in several places including the USA (Fritch et al., 2000), China (Wang et al., 2012), Jordan (El-Naqa et al., 2006; Al-Rawabdeh et al., 2013) , Iran (Chitsazan and Akhtari, 2009) and South Africa (Lynch et al., 1994). The DRASTIC method is a powerful tool for assessing groundwater vulnerability and is widely used (Rahman 2008; Leone et al., 2009). The data required by DRASTIC Model is easily available which makes it suitable for regional scale assessments (Thapinta and Hudak 2003). In addition, it is relatively simple and includes a high number of input data layers that limits the impacts of errors of the individual parameters on the final result (Zhang et al., 2013). Reliable results have been obtained even for complex areas (McLay et al., 2001).

2.4.2 *Process-based computer simulations*

Process-based methods use simulation models to estimate the contaminant migration but they are constrained by data shortage and computational difficulties (Barbash and Resek, 1996) Process-based models usually require large quantities of data and supplementary information necessary to run mathematical models that form the principal tool of the method. Clearly, such methods are more complicated and thus difficult to apply on a regional scale. Computer models can account for complex physical and chemical processes and at a very detailed scale.

Hydrologic Evaluation of Landfill Performance (HELP) Model

The flow or logic of the input facility of the HELP Model may be viewed as a tree structure. The tree structure is made up of nodes which are the points where new branches are started. The first node is called the trunk, root or parent node, and the terminal nodes of the tree are called leaves. All components (nodes) of the tree structure in the HELP Model are screens that have different functions, with the trunk node being the main menu. Generally, the parameters involved in a hydrological balance are precipitation, surface runoff, evapotranspiration and infiltration.

2.4.3 *Statistical Methods*

Statistical methods incorporate data on known areal contaminant distributions and provide characterizations of contamination potential for the specific geographic area by extrapolation from available data in the region of interest (NRC, 1993). Statistical methods use response variables such as the frequency of contaminant occurrence, contaminant concentration, or contamination probability. These methods are based on the concept of uncertainty, which is described in terms of probability distributions for the variable of interest (NRC, 1993). One possible goal in applying statistical methods to vulnerability assessment is to identify variables that can be used to define the probability of groundwater contamination (Burkart et al., 1999). Typically, one seeks to describe in mathematical terms (function or model) a relationship between water quality and natural and/or human-induced variables in a discrete area. Other statistical approaches, such as principal components analysis, discriminant analysis and cluster analysis, have been used to describe relationships between soil attributes and groundwater vulnerability (Teso et al., 1988; Troiano et al., 1997).

CHAPTER THREE: STUDY AREA

3.0 Location

Pomona Landfill is situated in Harare (Figure 3); 12 kilometres from Harare central business district (Tsiko and Togarepi, 2012). The geographical coordinates for the landfill are 17° 45' 15" South, 31° 5' 11" East. Harare has an urban population of approximately 1.5 million, with a growth rate of 3.2% (ZIMSTAT, 2012). The landfill covers a total area of 10 000 m^2, and has been operational since 1982 (Magadzire, 2005). The landfill is currently the main disposal site for both industrial and domestic solid waste generated in the city following the closure of Golden Quarry Landfill. Pomona Landfill is operated as an open dump instead of a sanitary landfill and has no engineered geo-synthetic liner to prevent water resources pollution by leachate (Tsiko and Togarepi, 2012). The landfill is located in the headwaters of the Gwebi Stream, a tributary of the highly polluted Upper Manyame River (Baldock et al., 1991). To the west of the landfill are residential suburbs which extend to the south and south-east. Nearby residential areas include Hatcliffe Extension to the north, Borrowdale; south east and Pomona Residential Suburb approximately 2 km to the south. Currently, there is a housing project underway 3 km north of the landfill.

3.1 Climate

Harare urban has a tropical continental type of climate, characterized by cold–dry winters and hot–wet summers. There are three main seasons: a warm, wet season from November to March/April; a cool, dry season from May to August (corresponding to winter in the Southern Hemisphere); and a hot, dry season in September/October (DMS, 2015). Mean Annual Rainfall (MAR) is approximately 820 mm, within a range of 440–1220 mm, characterized by high intensities falling between November and April (AQUASTAT, 2003). The mean annual temperature for Harare is in the range 15-20˚C (Mapanda et al., 2005). Minimum sunshine varies from an average of only 2 - 4 hours in the November to February period, when it is mild and humid, to around 7 - 8 hours from May to September, when it is cool and dry (Baldock et al., 1991)

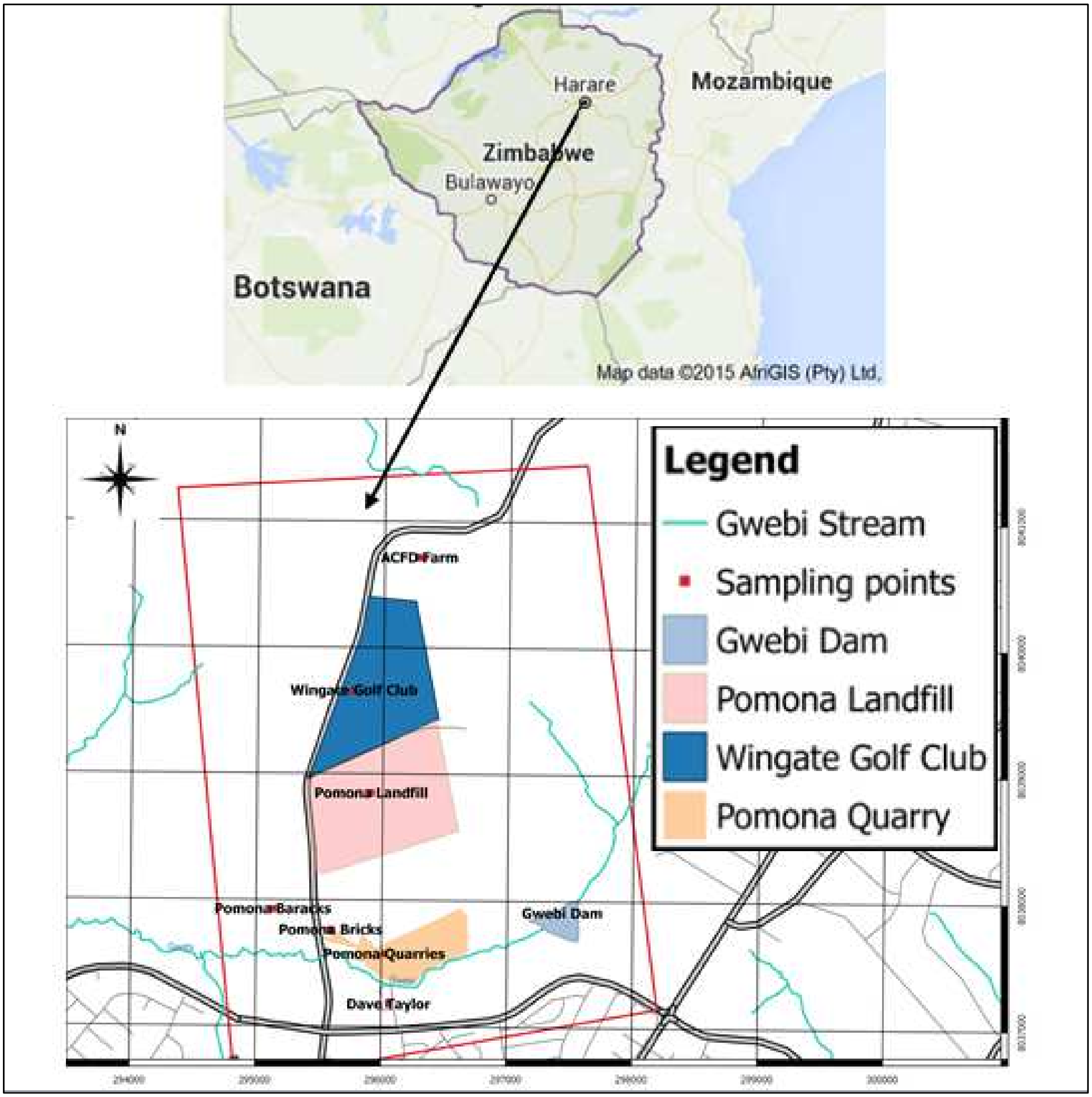

Figure 3: Map showing location of Pomona Landfill

3.2 Solid waste trends in Harare

Per capita solid waste generation in Harare averages 0.481 kg/day and the waste is predominantly biodegradable (Pawandiwa, 2013). Per capita generation is much higher in communities with a higher income. In their 2014 quarterly report, Harare City Council reported that a total of 205,658.80 tonnes of MSW was landfilled in 2014 (WMU, 2014). The waste stream is mainly domestic, market, commercial, industrial and institutional origins (Pawandiwa, 2013). At least 70% of the collected waste is crudely tipped at open dumpsites, (Tsiko and Togarepi, 2012).

3.3 Hydrogeology and Drainage

The Harare region lies on a North/West trending watershed (Baldock et al., 1991). Drainage in the western and south-western sectors is westwards towards Gwebi and Manyame Rivers, which join to flow northwards to the Zambezi. The whole region thus lies to the north of the principal watershed separating the Zambezi and Save drainage basins. Groundwater in the Harare urban area occurs largely in secondary aquifers, with strong stratigraphic and lithological controls on the occurrence (Baldock et al., 1991).

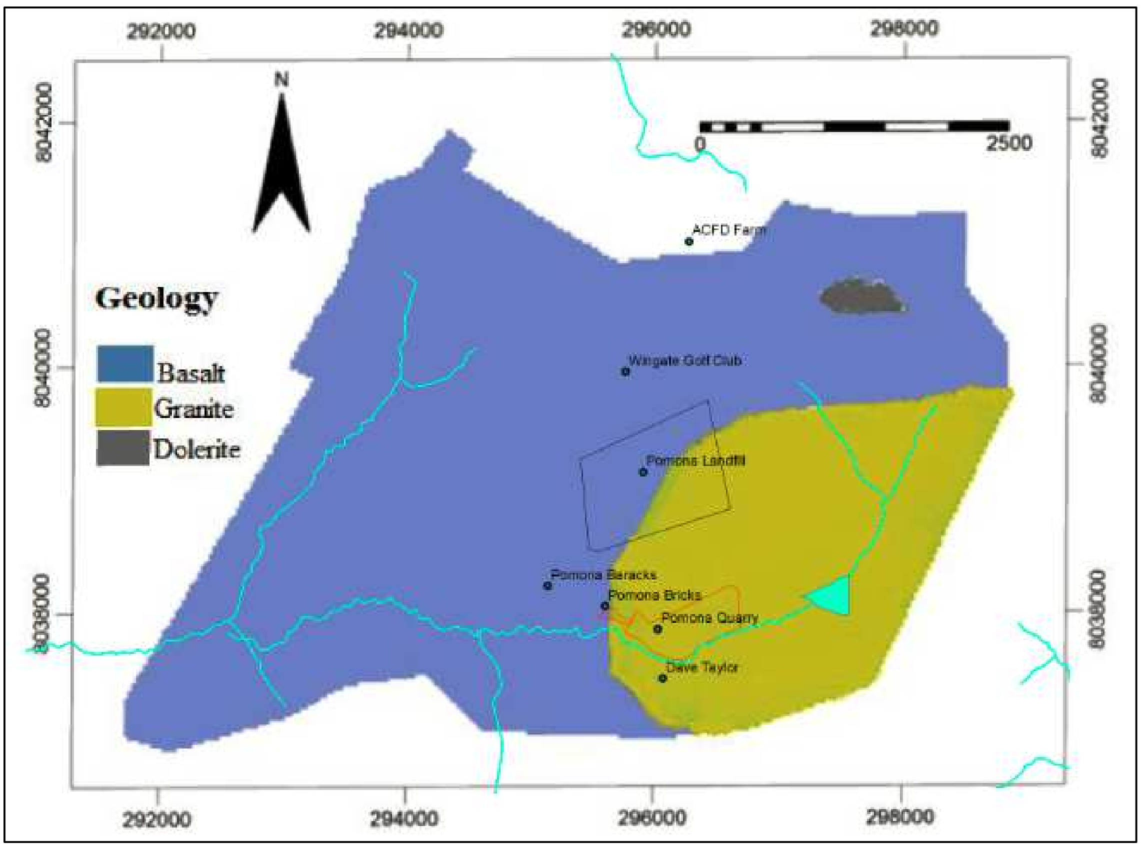

Figure 4: Pomona geology map

The geology across Harare is highly variable, but what is certain is that all rocks are of either igneous or metamorphic origin, which makes most of them massive and crystalline in nature (Broderick, 2012). The geology in Pomona Area is dominated by fractured meta-basalt

(Weaver, 1992), granite and dolerite formations. The meta-basalts are cut by a complex, irregularly-shaped porphyry and granite which occupies an elliptical area of rather flat ground within meta-basalts north of the city and east of the Teviotdale/Alpes Road. The granite is poorly exposed except in Pomona Quarry where it is actively exploited (Baldock et al., 1991). The Alpes lithology is distinctive medium- to coarse-grained quartz-rich granite. Pomona Landfill lies on both basaltic and granite formations (Figure 4).

CHAPTER FOUR: MATERIALS AND METHODS

4.1 Study design

4.1.1 *Selection of study site*

The study was conducted in Harare, the commercial and administrative capital of Zimbabwe. Pomona Landfill is the only official landfill which is currently operational in Harare (HCC, 2014). Harare was purposively selected because of its diversity in terms of economic activities which greatly contribute to the city's waste stream.

4.1.2 *Selection of sampling sites/areas*

Lee and Jones (1991), states that boreholes within close proximity to a landfill could result in groundwater contamination. The Environmental Protection Agency (EPA), 2006 states that the minimum regulated distance of a landfill from residential developments is five hundred meters. Sampling sites were selected relative to their distance from the perimeter of Pomona Landfill.

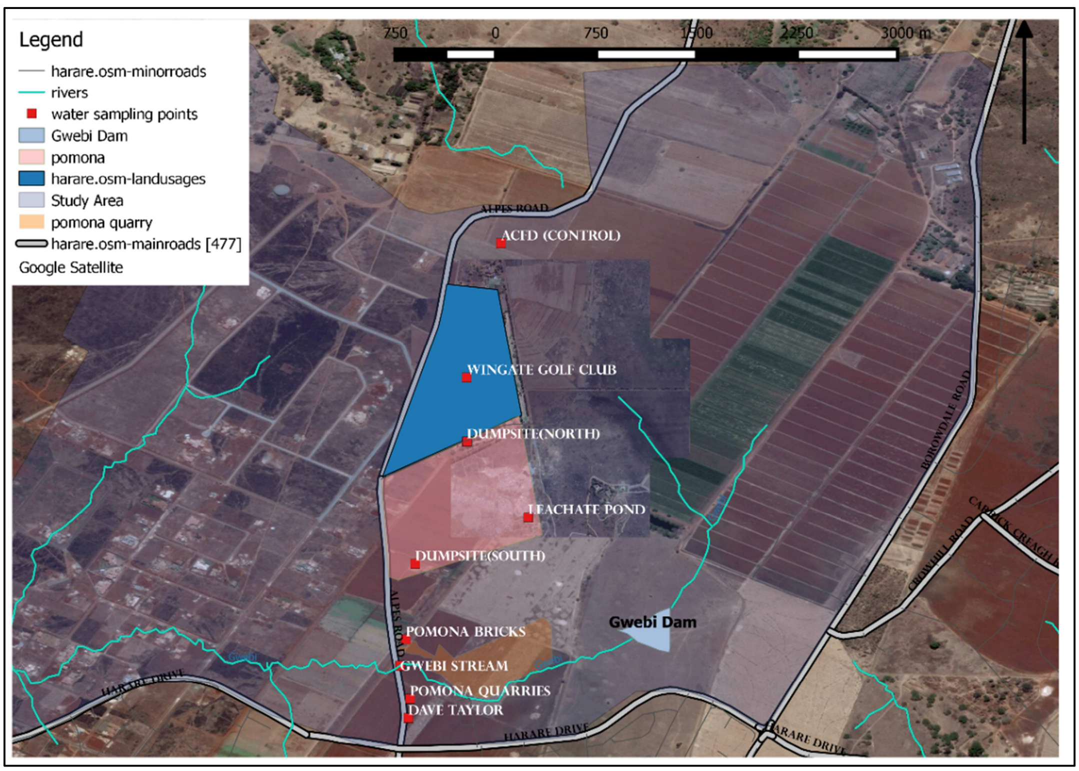

Figure 5: Location of groundwater and leachate sampling points

In order to assess the state of groundwater quality within the vicinity of the landfill, eight groundwater sampling sites and two leachate sampling sites were selected. Two sampling sites were located upstream of the landfill, two within the landfill and four located downstream of the landfill. Amongst all the sampling sites, two boreholes north of the landfill, one being the control and four boreholes south of the landfill were used for drinking purposes amongst other uses. The other two boreholes within the landfill were only for groundwater monitoring purposes and were sampled in order to monitor the extent of contamination. Figure 5 shows the location of the sampling sites.

4.1.3 *Selection of parameters to be analysed*

More than 200 organic compounds have been identified in municipal landfill leachate (Yasuhara et al., 1997; Paxe´us, 2000; Schwarzbauer et al., 2002). Barnar et al. (2006) classified leachate sampling parameters into four groups; physical, organic constituents, inorganic constituents and biological. Parameters for analysis in this study were selected from the four groups. The concentration of the pollutants in leachate and other chemical and physical parameters such as pH and conductivity are commonly used to characterize the leachate (Moreno and Stella, 2011). The organic content of leachate pollution is generally measured by chemical oxygen demand and biochemical oxygen demand (Kamaruddin et al., (2013). Chemical Oxygen Demand (COD) is an important parameter in determining the degree of solid waste decomposition and organic contamination (James, 1977). An excess of Cl^- in water is usually taken as an index of pollution and considered as tracer for groundwater contamination (Loizidou and Kapetanios, 1993). Because chlorides are usually not attenuated by soil and are extremely mobile under all conditions, they have a special significance as the tracer element of leachate plume linking the groundwater (Kumar and Alappat, 2005). Municipal landfill leachates are highly concentrated complex effluents which contain dissolved organic matters; inorganic compounds, such as ammonium, iron, sulphates, chlorides and heavy metals such as cadmium, chromium, copper, lead, zinc; and xenobiotic organic substances such as drugs and food additives (Lee and Jones, 1993 ; Christensen et al., 2001).

Arsenic in groundwater is probably the most serious heavy metal contaminant from landfills and it is carcinogenic (Jasim and Mallikarjuma, 2014). TDS is one of the parameters taken into consideration for licensing discharge of landfill leachate in many countries such as the

U.K. (Koshy et al., 2008). Parameters for characterization of landfill leachate, when monitoring groundwater quality include EC, TDS, Cl^-, NO_3^-, Zn, Cu, Cr, pH. In addition to the above mentioned parameters; for characterization of different phases of landfill development (from aerobic to methanogenic); BOD_5, COD, PO_4^+ were sampled and analysed (NMED Solid Waste Bureau, 2008).

4.1.4 *Methods of sampling and frequency*

The removal of stagnant water from the boreholes was accomplished by purging three well volumes prior to sample collection (EPA, 2003). Since the concentration of the pollutants in groundwater remains fairly stable within a 24 hour period, a single grab sample from an extraction well, batch tank or treatment system is sufficient to determine compliance (Groundwater Sampling Manual, 2015). Sterilized 500 ml glass bottles were used to collect samples for microbiological analysis while 2 litre plastic containers were used for collecting samples for physical and chemical analysis. Taps and handles were disinfected by pouring methylated spirit and lighting with a burner as suggested by EPA, (2003). All this was done so as not to introduce contamination to the samples. Leachate samples were collected from the base of solid waste heaps where the leachate was drained out by gravity (Bhalla et al., 2013). A trench at the centre of the landfill and a pond down-gradient of the landfill were dug to facilitate leachate collection. The containers were rinsed with sample fluids prior to collection to avoid any interference caused by using contaminated containers. At each sampling point containers were labelled. Sampling was conducted over four months giving a total of eight (8) leachate samples and thirty-two (32) groundwater samples.

4.1.5 *Methods of water quality analysis*

Methods of water quality analysis were according to standard methods for examination of water and wastewater specified by American Public Health Association (APHA, 2005). The collected samples were analysed immediately for pH, turbidity, dissolved oxygen (DO) and temperature (that change rapidly with time), whereas others were taken to the laboratory in cooler boxes for analysis. In the laboratory, the samples were preserved at 4˚C until chemical analyses were completed. The time between sampling and analysis was up to a maximum of 6 hours for bacteriological analysis, in order to limit alterations of samples before analyses.

Table 1: Methods of analysis

Parameter	**Method of analysis**
pH	OAKTON Eco Test r pH 1 meter
BOD_5	(Method 5210.B- 5-Day BOD Test using Winkler's Method)
COD	(Method 5220C- Closed Reflux, Titrimetric Method)
Turbidity	Hanna Instrument HI 98703
Chlorides	Argentometric method
Total dissolved solids and Electrical Conductivity	I-50 Lasany Microprocessor
Heavy metals including lead (Pb), mercury (Hg), arsenic (As), copper (Cu)	ICP-AES
Total and faecal coliforms	Membrane Filtration Method

4.2 Determination of leachate quantity

The quantity of leachate generated at Pomona Landfill was estimated using the Hydrologic Evaluation of Landfill Performance (HELP) Model.

4.2.1 *Hydrologic evaluation of landfill performance (HELP) Model*

Regarding the programs designed to perform landfill water balance, HELP – Hydrologic Evaluation of Landfill Performance Model (Schroeder et al., 1994) is the most well-known worldwide. The HELP Model (version 3.07) is the most widely used tool by the United States Environmental Protection Agency (USEPA) to predict leachate quantity and analyse water balance in landfill lining and capping systems (Alslaibi et al., 2013). The use of HELP Model is recommended by the U.S. Environmental Protection Agency (USEPA) and required by most states for evaluating closure design of hazardous and non-hazardous waste management facilities (Manandhar, 2000).

It is a quasi-two-dimensional hydrologic model of water movement across, into, through and out-of landfills (Schroeder et al., 1994). The HELP Model is classified as quasi-two dimensional because several one-dimensional models (percolation vertically, drainage and surface runoff horizontally) are coupled (Berger et al., 1996). HELP generates estimations of runoff amounts, evapotranspiration, drainage, leachate production and leakage from liners. The model uses weather, soil and design data as inputs (Schroeder et al., 1994) as shown in Table 5. Meteorological data from Belvedere Weather Station for the period 1983 to 2014 was entered into the model. The model output covered a period of 31 years by giving yearly values for every parameter that is involved in the hydrological balance.

Table 2: HELP Model input data (Source: Alsaibi, 2009)

Data type	Parameter	Unit	Time Step	Values
Weather data	Evaporative zone depth	cm	–	25
	Maximum leaf area index			1
	Relative humidity	%	Seasonally	
	Average wind speed	km/hr.	–	10.8
	Rainfall data	mm	Daily	
	Temperature Data	˚C	Daily	
	Solar radiation	MJ/m^2	Daily	
Landfill characteristics	Landfill area	m^2	–	10 000
	% of Landfill where runoff is possible	%	–	40%
	Runoff curve number	–	–	88.7
Soil and solid waste data	Layer type and text		–	Vertical percolation
	Layer thickness	in	–	320cm
	Hydraulic conductivity	in/hr	–	0.03
	Porosity, moisture content	vol./vol.	–	
	Field capacity and wilting point	vol./vol.	–	42% & 29.9%
	Recycling ratio	%	–	10

Concepts behind the HELP Model

The HELP Model uses many process descriptions that were previously developed and reported in literature and used in other hydrological models (Alslaibi, 2009; Berger, 2000; Nyhan et al., 1997; Schroeder et al., 1994). For example, runoff modelling is based on the

Soil Conservation Service (SCS) curve number method (Mack, 1995). Potential evapotranspiration is modelled using the modified Penman method (Penman, 1963). Evaporation of interception and surface water is based on the energy balance method, and interception is modelled by a method proposed by Horton (Berger et al., 1996). Vertical drainage is modelled by Darcy's law and saturated lateral drainage is modelled by an analytical approximation to the steady state solution of the Boussinesq equation (Yalçin & Demirer, 2002).

Evaporation from soil, plant transpiration and vegetative growth are extracted and modelled using the methods included in the Simulator for Water Resources in Rural Basins (SWRRB) model (Arnold et al., 1989; Qrenawi, 2006). These processes are linked in a sequential order, starting at the surface with a surface water balance, then evapotranspiration from the soil's profile and, finally, drainage and water routing, starting at the surface with infiltration, proceeding downward through the landfill profile, to the bottom. The solution procedure is applied repetitively for each day as it simulates the water routing throughout the simulation period (Schroeder et al., 1994).

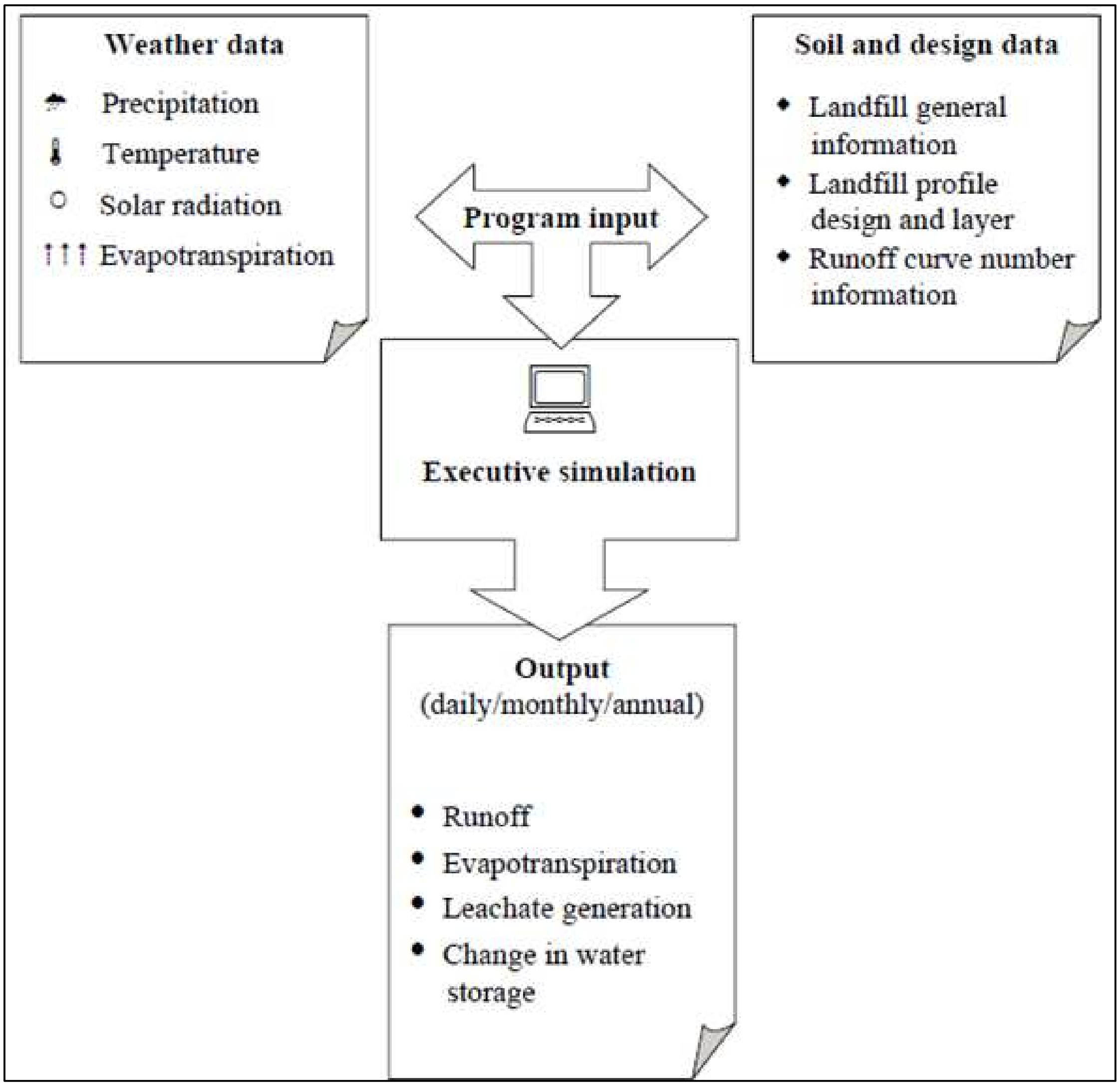

Figure 6: HELP Model flowchart (Source: Wisiterakul, 2006)

Model calibration and validation

The leachate generation value obtained by the HELP Model has a tendency to be overestimated compared to that which is measured in the field. The HELP Model was calibrated using the measured hydraulic conductivity in the field. The Auger-Hole Method was used to measure hydraulic conductivity in the field. A square pit with area of 50 x 50 cm (W x B) and the depth of 60cm was dug. Soil features were observed and recorded. The researcher waited until the pit water level had reached equilibrium with the surrounding water table and water table depth was recorded. A ruler was inserted vertically in a stable position several centimetres into the soil in the base of the pit in one corner where readings could be taken easily. Water was rapidly bailed out from the pit using a bucket. A stopwatch was started immediately after the last bail as the water level on the ruler was being monitored and recorded. The water level should be recorded approximately every 5 seconds for a minimum of 3 minutes or until at least 80% of the pit bail out volume is replaced. However the pit infill rate was slow, so the water level was recorded at 30 minutes intervals to allow accurate measurement .When equilibrium level was obtained the test was repeated in the same pit.

After calibration, the leachate quantity was estimated using default values of hydraulic conductivity in the model. The predictions of any model achieve validity through field results. In arid and semi-arid areas there are few published field results of landfill performance. This is the case with Pomona Landfill. The model was validated by comparing the model predicted percolation to the estimated leachate quantity using the simple water balance method. This was due to lack of field monitoring data from previous years to evaluate seepage from the existing landfill.

Water Balance Method

The simple water balance method was used to predict moisture movement within the landfill. Several mathematical models have been developed which attempt to predict leachate production from the knowledge of basic hydrological factors (Lema et al., 1988). The prediction of leachate quantity was performed using a simple mass (Canziani and Cossu, 1989):

$\mathbf{L_p = P - R_{off} - E}$……………………………………………………………Equation 1

Where:

Lp …... Leachate production   P ….. Rainfall

Roff ….. Run-off   E ….. Evapotranspiration

As a first approximation, the quantity of leachate produced may be regarded as proportional to the volume of water percolating through the landfilled waste (Lema et al., 1988). Reduction of the quantity of water entering the tip is therefore of great importance in reducing the rate of leachate generation. Nevertheless, the advantages of decreasing the water input must be carefully balanced against the disadvantage of possible reduction in the rate of landfilled waste decomposition (stabilization). Leachate production has also been found to be greater whenever the disposed refuse is less compacted, since compaction was found to reduce the filtration rate (Lema et al., 1988). The calculation of hydrological balance was performed following the rational method of Canziani and Cossu (1989). In this method, the water content of landfilled solid wastes is not taken into account. It is also assumed that there is not any other water inflow into the landfill from natural aquifers. Monthly variations in the run-off coefficient were obtained from related literature studies (Canziani and Cossu, 1989). Surface run-off *R* (expressed in mm water) was calculated by applying the simple equation:

R = C x P x k…………………………Equation 2

Where:

C is the selected run-off coefficient (0.30)

P the monthly precipitation (rainfall, expressed in mm water).

k is the monthly variation of run-off coefficient

The available water for infiltration is the difference between precipitation and surface run-off. It has been assumed that the initial water content of the soil from January to May and from October to December, which is available for evapotranspiration, is 40 mm water. Therefore, the total available moisture for leachate production is the sum of the initial water content of the soil and water available for infiltration. The climatic data were obtained from the Meteorological Services Department of Zimbabwe. The remaining available moisture is therefore the difference between the total available moisture and the actual evapotranspiration values. Hence, percolation was finally estimated as the difference between the remaining available moisture and the initial soil moisture.

4.3 Groundwater vulnerability assessment

The DRASTIC Model was used to assess groundwater vulnerability within the vicinity of Pomona Landfill. It is defined as a mappable unit with common hydrological characteristics which control groundwater pollution (Aller et al., 1987). Each DRASTIC factor was assigned a DRASTIC weight ranging from 1 to 5 (Table 3) and a rating, typically from 1 to 10 (Klug, 2009). The weights and rates of the original DRASTIC Model parameters are presented by Aller et al. (1987). The numerical ratings and weights, which were established using the Delphi technique (Aller et al., 1987), are well defined and are used worldwide.

Table 3: DRASTIC Model parameter rating and relative weight (Modified from Aller et al., 1987)

Parameter	**Range**	**Rating**	**Description**	**Relative Weight**
Depth to water (**D**) (m)	0-5 5-15 15-30 30-50 50-75 75-100 >100	10 9 7 5 3 2 1	Refers to the depth to the water surface in an unconfined aquifer. Deeper water table levels imply lesser chance for contamination to occur. Depth to water is used to delineate the depth to the top of a confined aquifer.	5
Net recharge **(R)** (mm)	0-2 2-4 4-7 7-10 >10	1 3 6 8 9	Indicates the amount of water per unit area of land which penetrates the ground surface and reaches the water table. Recharge water is available to transport a contaminant vertically to the water table, horizontal with in an aquifer.	4
Aquifer media **(A)**	Weathered met./igneous Sand and gravel Basalt	4 8 9	Refers to the consolidated or unconsolidated medium which serves as an aquifer. The larger the grain size and more fractures or openings with in an aquifer, leads to higher permeability and lower attenuation capacity, hence greater the pollution potential.	3
Soil media **(S)**	Clay loam Sandy loam Sandy Clay loam	3 6 7	Refers to the uppermost weathered portion of the vadose zone characterised by significant biological activity. Soil has a significant impact on the amount of recharge which can infiltrate into the ground.	2

Topography **(T)** (slope %)	0-2 2-6 6-12 12-18 >18	10 9 5 3 1	Refers to the slope of the land surface. It helps a pollutant to runoff or remain on the surface in an area long enough to infiltrate it.	1
Impact of vadose zone **(I)**	Silt/clay Sand and gravel Basalt	3 8 9	Is defined as unsaturated zone material. The significantly restrictive zone above an aquifer forming the confining layers is used in a confined aquifer, as the type of media having the most significant impact.	5
Hydraulic conductivity (m/day) **(C)**	1-100 100-300 300-700 700-1,000 1000-2,000 >2,000	1 2 4 6 8 10	Refers to the ability of an aquifer to transmit water, controlling the rate at which groundwater will flow under a given hydraulic gradient.	3

Preparation of the maps usually involves overlaying several thematic maps of selected physical factors that have been chosen to depict vulnerability (Mohammed, 2011**)** as shown in the DRASTIC Model flowchart (Figure 10). For the assessment of groundwater vulnerability to contamination of Pomona area, the DRASTIC Model (Aller et al., 1987) and a geological information system, Integrated Land and Water Information System (ILWIS) version 3.0 were used to produce the vulnerability map. The seven sets of data layers were digitized and converted to raster data sets that were processed in ILWIS. DRASTIC is based on four assumptions (Al-Zabet, 2002): 1) the contaminant is introduced at the ground surface; 2) the contaminant is flushed into the groundwater by precipitation; 3) the contaminant has the mobility of water; and 4) the area evaluated is 0.4 km^2 or larger (Secunda et al., 1998).. The final vulnerability map is based on the DRASTIC index (DI) which is computed as the weighted sum overlay of the seven layers using Equation 3 as suggested Aller et al., (1987):

$$\boldsymbol{DI = Dr\,Dw + RrRw + ArAw + SrSw + TrTw + IrIw + CrCw} \text{.........Equation 3}$$

Where:

DI - DRASTIC Index for a mapping unit

r -rating

w -weighting

D, R, A, S, T, I, and **C** are the seven parameters

Once the index is calculated, susceptible areas can be classified; the bigger the DRASTIC index, the greater its susceptibility to pollution (Corniello et al., 1997; Hua et al., 2011; Al-Rawabdeh et al., 2013).

4.3.1 *Depth-to-Water Table (**D**)*

The depth-to-water table parameter was derived from field measurements of water levels using a water level indicator. The indicator consists of a probe, a cable with laser-marked graduations, and a cable reel. The probe was lowered into the borehole and when the probe came in contact with the water surface the LED illuminates giving a beeping sound. Depth-to- water level was measured from graduations on the cable. The depth to water table from ground level point information was interpolated to derive the depth to groundwater table. The borehole location vector layer was prepared based on the GPS survey and the spatial distribution map of water table was obtained through Inverse Distance Weighting (IDW) interpolation technique which was in terms of raster image.

4.3.2 *Net recharge (**R**)*

Recharge water is available to transport a contaminant vertically to the water table and horizontally within the aquifer (Aller et al., 1987). The greater the recharge, the greater the potential for ground-water pollution (Aller et al., 1987). This recharge value was then grouped into a range of values that are given a rating for use in the final DRASTIC calculation. By applying Equation 2 (Piscopo, 2001) to the study area, the ratings for recharge were calculated as shown in Table 6.

$$\boldsymbol{Recharge\ value} = \boldsymbol{Slope}\,\% + \boldsymbol{Rainfall} + \boldsymbol{Soil\ permeability} \ldots\ldots\ldots\text{..Equation 4}$$

4.3.3 *Aquifer media (**A**)*

The aquifer media parameter was prepared using a subsurface geology map, (Harare Geological Map A, Bulletin No. 94) from the Zimbabwe Geological Survey Department and drilling reports for some of the boreholes. The ratings assigned as per DRASTIC Model to the aquifer media parameters are given in Table 3. Based on the point data the spatial variation map for effective infiltration was prepared.

4.3.4 *Soil Media (**S**)*

Various soil types have the ability to attenuate or retard a contaminant as it moves through the soil profile. The attenuation character of soil media varies widely depending on the soil texture and with regard to the different type of contaminants. The soil media parameter was prepared using the geological map (Harare Geological Map A, Bulletin No. 94) from the Zimbabwe Geological Survey. The soil media types were then assigned ratings from 1 to 10 as per DRASTIC model (Table 3) with sandy loam assigned a rating of 8.

4.3.5 *Topography (**T**)*

Digital Elevation Models (DEMs) can be used to derive a wealth of information about the morphology of a land surface (U.S. Geological Survey, 1987). The slope of Pomona area was extracted from the digital elevation map shown in Figure 19. The Slope tool from the Surface toolset in the Spatial Analyst extension was used to generate the slope of Pomona area. The tool used the 30-meter DEM to calculate a grid layer showing per cent slope. The slope layer was then reclassified to rating values according to the per cent ranges recommended in the DRASTIC Model (Table 3).

4.3.6 *Impact of Vadose Zone (**I**)*

This parameter is one of the most significant parameters in vulnerability assessment and hence it has a weight of 5 (Table 3). The vadose zone consists of the material existing as the surface soil, as well as the bedrock layers without a holding capacity for groundwater. The impact of pollution on the vadose zone is measured based on the thickness, porosity, and permeability of all material within the vadose zone. The ratings are assigned as per the influence of the least impervious material, taking into account all types of material toward the surface. According to Aller et al., (1987), the vadose media for an unconfined aquifer system is the same as the aquifer media. In other words, the DRASTIC methodology allows any standard geological map depicting the distribution of lithological units, to be used as a measure of the impact of the vadose zone.

4.3.7 *Hydraulic Conductivity (**C**)*

Hydraulic conductivity values for various rock types have been proposed by Domenico and Schwartz, (1990). These were the values used in this study (Appendix 1).

4.3.8 *Drastic Vulnerability Index Map (**DI**)*

When all thematic maps had been registered and geo-referenced, they were on-screen digitized to create point, segment, and polygon maps of the different geographical entities. The Drastic Vulnerability Index (DI) map was prepared based on the overlay process using reclassified input parameters with desired weights in the raster calculator tool of GIS using Equation 3. The natural breaks method available in ILWIS was used to capture the natural grouping of ratings. The natural breaks classification method considers visually, logical and subjective aspects to grouping data sets hence the choice of the method. One important purpose of natural breaks is to minimize value differences between data within the same class. Another purpose is to emphasize the differences between the created classes.

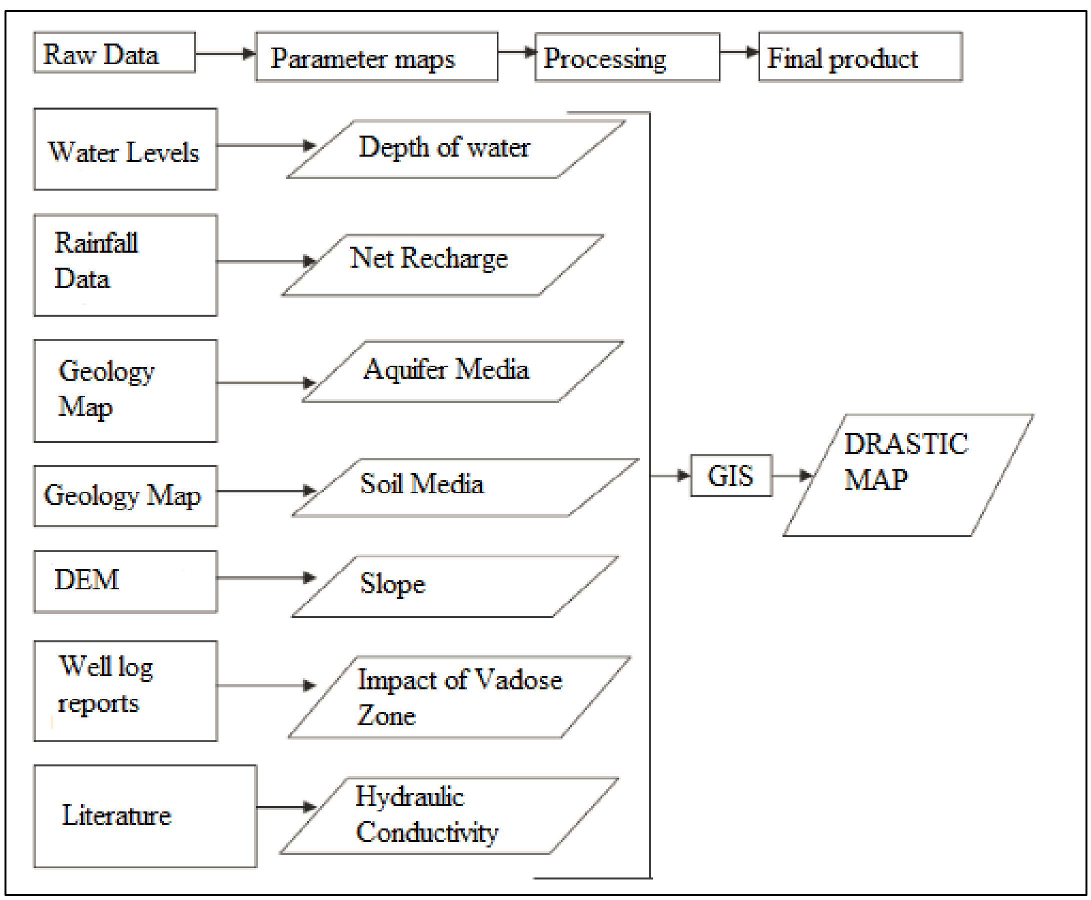

Figure 7: Drastic Model flowchart (Source: Rahman, 2008)

4.3.1 *Data analysis*

The data collected was tabulated and analysed using Statistical Package for Social Sciences (SPSS) software package version 22. The data was presented in the form of range, arithmetic mean, standard deviation. In order to extract the most important parameters in assessing variation in groundwater quality, Principal Component Analysis (PCA) was used. Principal Component Analysis (PCA) method is a statistical analysis method which can reduce the dimensions of multi-index data (Zhang, 2010). First, it changes initial random vectors related to its components into new random vectors which are not related to its components any more by means of orthogonal transformation. Secondly, the variance is considered to be the measurement of information and then the dimension of high variant space is lowered, which make the calculating process much easier (Lu and Li, 2006). Because of these advantages, PCA was applied to evaluate groundwater quality. PCA is used to find a few comprehensive indexes, which have great influence on water quality, by studying the internal structure of correlation matrix of initial variable (Jin and Chen, 2010).

Statistical differences between the means of leachate and groundwater samples were compared using *t*-test at *p*-value ≤ 0.05 (Van Belle et al., 2004). The independent t-test, also called the two sample t-test or student's t-test, is an inferential statistical test that determines whether there is a statistically significant difference between the means in two unrelated groups. The independent samples t-test is a parametric test. Following extraction of the principal components having great influence on groundwater quality using PCA, an independent t-test was carried out to compare the means of leachate and groundwater samples at 95% confidence level.

Groundwater quality results were compared to Zimbabwe Standards for Drinking Water Quality (SAZ, 2007) and World Health Organization Guidelines (WHO, 2004). The Rank function in Excel was also used to analyse the percentage of results that exceeded the national and international standards.

CHAPTER FIVE: RESULTS AND DISCUSSION

5.1 Leachate quality

Table 4 presents the summary of descriptive statistics results for the leachate samples collected from Pomona Landfill. Results for leachate samples were compared to Effluent and Solid Waste Disposal Regulations SI6 2007 (EMA, 2007). SI6 regulates the disposal of waste (solid waste and effluent), and uses polluter pays principle through licensing which is according to the following four classes:

- **Blue** – in respect of a disposal which is considered to be environmentally safe.
- **Green** - in respect of a disposal that is considered to present a low environmental hazard
- **Yellow -** in respect of a disposal which is considered to present a medium environmental hazard and,
- **Red -** in respect of a disposal that is considered to present a high environmental hazard.

The regulations provide the water quality standards in which the effluent should be discharged into the environment.

Table 4: Descriptive statistics for characteristics of leachate samples from the leachate pond

	Leachate Pond						Effluent & Solid Waste Disposal Regulations SI6 (EMA, 2007)				
							Blue		Green	Yellow	Red
	Range	Mean	Std Deviation	Std Error	COV	%	Sensitive	Normal			
pH	7.2-8.4	7.8	0.5	0.3	0.1	7	6.0 - 7.5	6.0 - 9.0			
Turbidity	55-121	96	31	15.5	0.3	32	<5	<5	*	*	*
EC	5360-6150	5645	364	181.9	0.1	6	<200	<1000	<2000	<3000	<3500
TDS	2720-3070	2944	264	132.0	0.1	9	<100	<500	<1500	<2000	<3000
TC	0-200	200	0	0	0	0	*	*	*	*	*
FC	35-80	57	22	11.2	0.4	40	*	*	*	*	*
Chloride	650-2793	1904	1068	533.8	0.6	56	<200	<250	<300	<400	<500
COD	265-430	353	89	44.3	0.3	25	<30	<60	<90	<150	<200
BOD_5	38-64	46	12	6.1	0.3	27	<15	<30	<50	<100	<120
Nitrates	48-65	59	7.4	3.7	0.1	13	<10	<10	<20	<30	<50
Arsenic	0.0-0.0	0.0	0.0	0.0	1.3	128	<5	<5	*	*	*
Zinc	0.1-0.2	0.2	0.1	0.0	0.4	37	<0.3	<1	<2	<5	<8
Iron	11-Aug	9.8	1.3	0.7	0.1	14	<1	<1	<2	<3	<5
Copper	0.0-0.5	0.2	0.2	0.1	0.8	81	*	*	*	*	*

Table 5: Descriptive statistics for characteristics of leachate samples from the leachate trench.

	Leachate Trench						**Effluent & Solid Waste Disposal Regulations SI6 (EMA, 2007)**				
							Blue		**Green**	**Yellow**	**Red**
	Range	**Mean**	**Std Deviation**	**Std Error**	**COV**	**%**	**Sensitive**	**Normal**			
pH	7.1-8.1	7.7	0.4	0.2	0.1	5.6	6.0 - 7.5	6.0 - 9.0			
Turbidity	83-257	174	93	46.3	0.5	53.3	<5	<5	*	*	*
EC	5320-5650	5568	126	62.9	0	2.3	<200	<1000	<2000	<3000	<3500
TDS	3800-3980	3928	86	42.8	0	2.2	<100	<500	<1500	<2000	<3000
TC	0-200	200	0	0	0	0	*	*	*	*	*
FC	55-85	72	12	6.2	0.2	17.4	*	*	*	*	*
Chloride	870-1570	1155	242	120.8	0.2	20.9	<200	<250	<300	<400	<500
COD	274-368	308	42	20.8	0.1	13.5	<30	<60	<90	<150	<200
BOD_5	94-129	61	10	4.9	0.2	16.2	<15	<30	<50	<100	<120
Nitrates	82-102	91	9	4.5	0.1	9.9	<10	<10	<20	<30	<50
Arsenic	0.0-0.0	0	0	0	2	200	<5	<5	*	*	*
Zinc	0.18-0.31	0.3	0.1	0	0.2	22.9	<0.3	<1	<2	<5	<8
Iron	12-16	14	1.8	0.9	0.1	13.4	<1	<1	<2	<3	<5
Copper	0.2-0.4	0.3	0.1	0	0.3	31.4	*	*	*	*	*

Descriptive statistics for the characteristics of the leachate samples collected from Pomona Landfill site are presented in Table 4 and 5. The mean pH values were found to be 7.8 and 7.7 for the leachate pond and leachate trench respectively. Fatta et al., (1998) observed that the initial period of leachate formation is characterised by very low pH values and later with higher pH values at the methanogenic phase. Leachate is generally found to have pH between 4.5 and 9 (Christensen et al., 2001). It can be noted that leachate is alkaline and the biochemical activity at Pomona Landfill was in its final stage and the organic load was biologically stabilized. pH values for both the leachate trench and leachate pond fall in the blue normal category which is considered safe with respect to wastewater disposal.

The mean TDS values were found to be 2944 and 3928 mg/L for the leachate pond and leachate trench respectively. The amount of TDS reflects the extent of mineralization and a higher TDS concentration can change the physical and chemical characteristics of the receiving water (Al-Yaqout and Hamoda, 2003). Total Dissolved Solids values for both the leachate trench and leachate pond fall in the red category which is considered to present a high environmental hazard with respect to wastewater disposal. The increase in salinity due to increases in TDS concentration increases toxicity by changing the ionic composition of water (Umar et al., 2010).

Electrical Conductivity (EC) values of leachate ranged from 5320 µS/cm to 6150 µS/cm. The mean EC values were found to be 5645 and 5568 mg/L for the leachate pond and leachate trench respectively. The values of EC in leachate samples indicate the presence of inorganic material. EC values of leachate falls in the red category of EMA regulations stating that values greater than 3500 µS/cm present a high environmental hazard.

Chloride values of leachate ranged from 650 mg/L to 2793 mg/L. The mean Chloride values were found to be 1904 and 1155 mg/L for the leachate pond and leachate trench respectively. Chloride values for both the leachate trench and leachate pond fall in the red category which is considered to present a high environmental hazard with respect to wastewater disposal. Chloride is a conservative contaminant and therefore poses serious threat to groundwater.

Concentrations of NO_3^- ranged from 48 mg/L - 102 mg/L in the leachate samples. The mean NO_3^- values were found to be 59 mg/L and 91 mg/L for the leachate pond and leachate trench respectively. Nitrate values for both the leachate trench and leachate pond fall in the red category which is considered to present a high environmental hazard with respect to wastewater disposal. Nitrates are conservative contaminants as they are not affected by biochemical processes and natural decontamination processes taking place inside the landfill as well as their infiltration into the vadose zone (Fatta et al., 1999). This explains why nitrates are potential threat to groundwater pollution.

The BOD_5 values of leachate ranged from 12mg/L to 129 mg/L and COD values ranged from 265mg/L to 430mg/L. The presence of BOD_5 and COD indicates high organic strength of leachate. The value of BOD_5 in leachate tends to indicate the maturity of the landfill (Ohwoghere–Asuma and Aweto, 2013) which typically decreases with time (Qasim and Chiang,1994). These values obtained for both BOD_5 and COD are comparable to those obtained by Christensen et al., (2001) as normal range for typical landfill leachate.

The BOD_5/COD ratio for leachate was in the range 0.09 - 0.21. BOD_5/COD ratio indicates the age of the landfill as portrayed by Curi et al., (1994); they reported that ratios vary from 0.4 to 0.6 for young landfills and 0.05 to 0.2 for matured landfills. A decrease in BOD_5 and COD is often reported with the increase in age of the landfill. For stabilized leachates, COD generally ranges between 5000–20,000 mg/L (Kurniawan et al., 2006). The BOD_5/COD ratio provides a good estimate of the state of the leachate and this ratio for young leachate is generally between 0.4 - 0.6 (Rivas et al., 2004). During the methanogenic phase, the organic strength of the leachate is reduced by methanogenic bacteria such as methanogenic archaea and the concentration of Volatile Fatty acids (VFAs) also declines which results in a ratio of BOD_5/COD less than 0.1 (Kurniawan et al., 2006 ; Rivas et al.,2004). COD values of leachate were above the permissible standard limit. They fall in the red category of EMA regulations stating that values greater than 200 mg/L present a high environmental hazard.

For heavy metals, iron values ranged from 8 mg/L to 16 mg/L with mean values 10 mg/L and 14 mg/L for the leachate pond and leachate trench respectively. This result concurs with Al-Yaqout and Hamoda (2003) who reported that iron is a common metal in municipal landfill leachate and is responsible for the reddish-brown colour of leachate that may change the groundwater colour. Cr (0.1 mg/L), Cu (0.2 mg/L) was also present in the leachate samples. Concentration of heavy metals in a landfill is generally higher at earlier stages because of higher metal solubility as a result of low pH caused by production of organic acids (Kulikowska and Klimiuk, 2008). As a result of decreased pH at later stages, a decrease in metal solubility occurs resulting in rapid decrease in concentration of heavy metals (Harmsen, 1983).

5.2 Groundwater Hydrology

A water table contour map shows the elevation and the configuration of the water table at a certain datum. Hydraulic head was calculated by subtracting the static water level from the surface elevation. The map was prepared by plotting the absolute water levels of all observation points of equal water table elevation. This water table contour map (Figure 6) is an important tool in groundwater investigations as one can derive from it the gradient of the water table and the direction of the groundwater flow. Generally, the pattern of groundwater flow follows the topography with seasonal variations in water levels characterized by rising water levels during the wet months and declining during the dry months (Usman and Lar, 2013). In the study area, the groundwater is flowing in a general north to south and southwest direction (Figure 6).

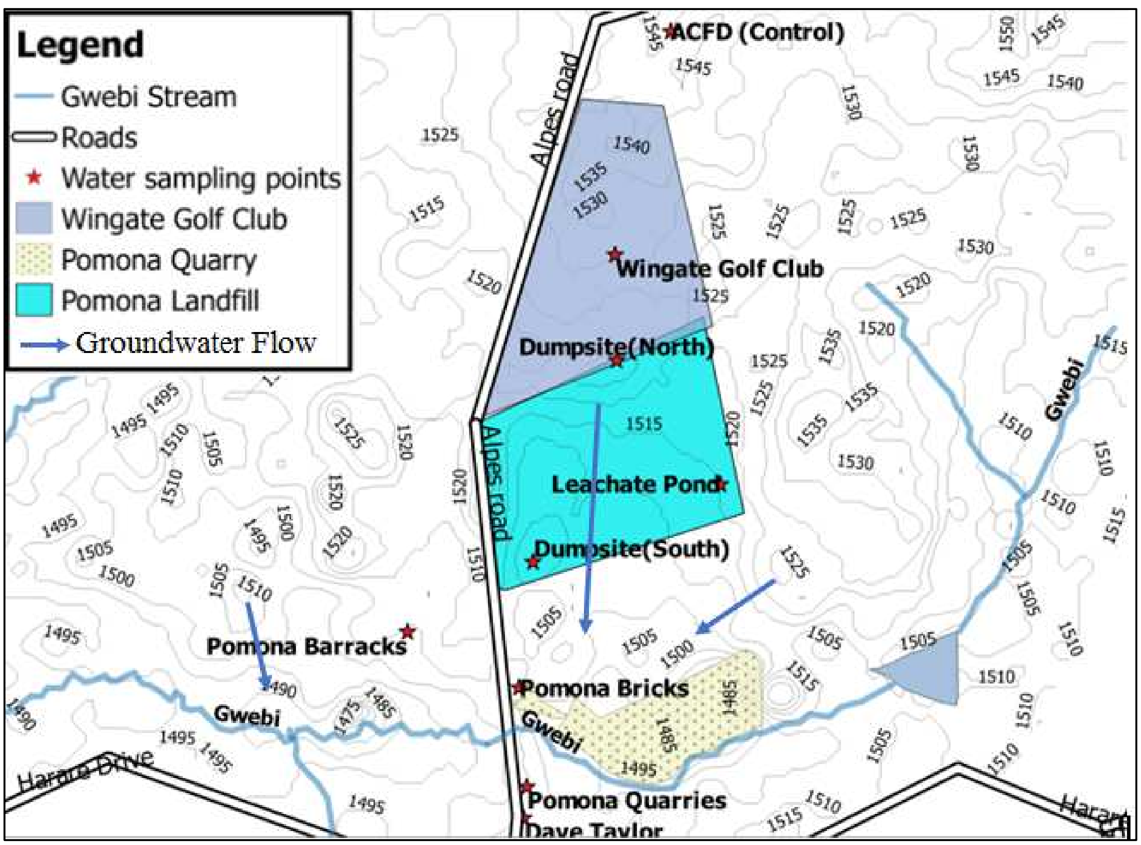

Figure 8: Water table contour map

5.3 Groundwater quality

Table 6 presents the summary of analytical results of the groundwater samples collected in Pomona area. Results for groundwater samples were compared to Zimbabwe Standards of Drinking Water Quality (SAZ, 1997) and World Health Organization guidelines (WHO, 2011).

Table 6: Groundwater quality results

Parameters	Range	Min	Max	Mean	Std Deviation	Std Error	COV	%
pH	6.3-8.2	6.3	8.2	7	0.5	0.1	0.1	8
Turbidity	0.09-63	0.09	63	7.28	14.8	2.6	2	203
Electrical Conductivity	192-716	192	716	398	167	29.6	0.4	42
Total Dissolved Solids		97.3	570	219	108	19.1	0.5	49
Total Coliforms	0-80	0	80	26	33	5.7	1.3	126
Faecal Coliforms	0-80	0	80	20	35	6.2	1.7	174
Chloride	13-165	13	165	55	49	8.7	0.9	90
COD	108-756	108	756	302	176	31	0.6	58
BOD5	13-43	13	43	28	8	1.4	0.3	27
Nitrates	0.05-45	0.05	45.05	12.76	14.9	2.6	1.2	116
Iron	0-1	0	1	0.37	0.28	0	0.8	76
Zinc	0-0.2	0	0.2	0.06	0.05	0	1	95
Arsenic	0-0.01	0	0.014	0.001	0.002	0	4.2	421
Copper	0-0.2	0	0.23	0.04	0.06	0	1.4	137

Groundwater quality variability

Table 7 presents the summary of analytical results from Principal Component Analysis (PCA).

Table 7: Eigen value and contribution rate of every component

Component	Eigen Value	% of Variance	Cumulative %
1	9.6	80	80
2	0.6		
3	0.5		
4	0.4		
5	0.3		

One principle component was extracted and rotated using the varimax normalization (Kaiser, 1960). The cumulative contribution rate of the first principal component was up to 80% as shown in Table 7 which is quite good and can be relied upon to identify the main sources of variation in the hydrochemistry. The first component reflects most information implicated in initial data, thus we use the first principal component to evaluate groundwater quality. PCA 1 has eigenvalue of 9.6 and shows total variance of 80%. The first principal component has great load on pH, Cl^-, Turbidity, EC, TDS, TC, NO_3^-, BOD_5, Fe, Cu, As, Cr, and Zn. The range of these loads is 0.5 to 0.8 (Appendix 3). TDS reflects the impurity concentration; the higher the value of TDS, the larger the impurity concentration in groundwater (Jin and Chen, 2011). COD reflects the contamination status of organics. All these variables indicate the comprehensive pollution conditions to some extent. Total Coliforms in this component can be viewed as an indication of how groundwater is vulnerable to contamination by harmful microorganisms.

5.3.1 *pH*

Table 6 summarizes the ranges, averages and standard deviations at each sampling point for the four sampling campaigns. The pH values for groundwater were in the range 6.3 – 8.2 with a mean of 7.0±0.5 (n=32). All the results were within the minimum pH limit of 6.0 and the maximum limit of 9.0 prescribed by SAZ as shown in Figure 11, however 12.9 % were below the minimum acceptable limit of 6.5 by WHO guidelines. Among all the sampling points in Pomona area, 59.6% had acidic water and 40.4 % had alkaline water. There were no significant pollution differences for pH on comparing the pollution levels between boreholes used for drinking purposes and leachate samples (p=0.05, df =6).

Highly acidic water may result in corrosion parts of borehole pump, causing the possible release of iron, lead, or copper into the water while a low pH may discolour the water and give it a bitter taste. Longe and Balogun (2010) assessed groundwater quality near a municipal landfill in Lagos, Nigeria and found that pH was acidic in nature. They concluded that this nature of Lagos groundwater is characteristic of the coastal groundwater whose pH is primarily controlled by its hydrogeological setting (Longe et al., 1987). It can be concluded that leachate is not contributing to pH levels in groundwater in and around Pomona.

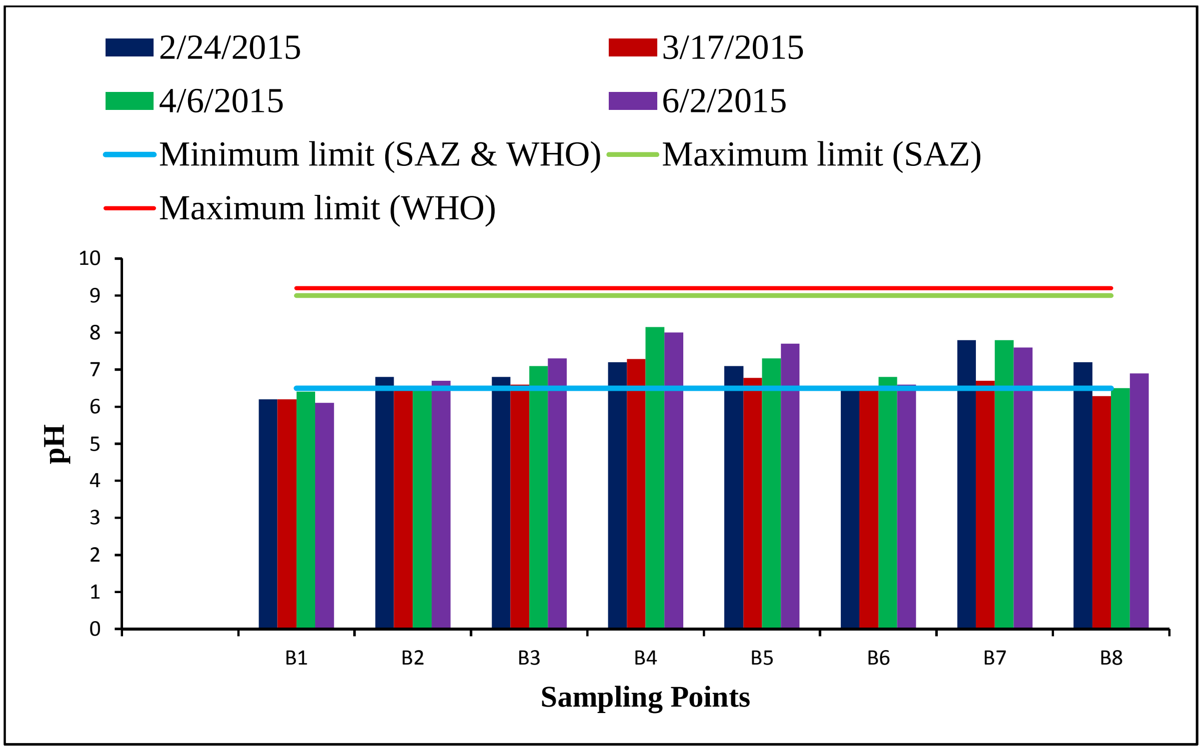

Figure 9: pH variation among sampling points

5.3.2 *Chlorides*

The concentration of chlorides ranged from 13 mg/L to 165 mg/L with a mean value of (55 ± 49) mg/L (Table 6). Results for chlorides for all the boreholes were below SAZ maximum limit and WHO guideline of 300 mg/L and 250 mg/L respectively as shown in Figure 12. High chloride concentration may impart saline taste thereby rendering the water not fit for portable uses. There were significant pollution differences in Cl^- levels between borehole samples (B1, B3, B4, B5 and B8) and leachate samples. These results demonstrate the significant contribution by landfill leachate to these boreholes. An excess of Cl^- in water is usually taken as an index of pollution and considered as tracer for groundwater contamination (Loizidou and Kapetanios, 1993). Potential sources of Cl^- in groundwater is likely to originate from food scraps and pet wastes discarded in landfills and from natural sources such as rainfall, the dissolution of fluid inclusions (Bhalla et al., 2013). It was therefore concluded that landfill leachate has an effect on groundwater quality.

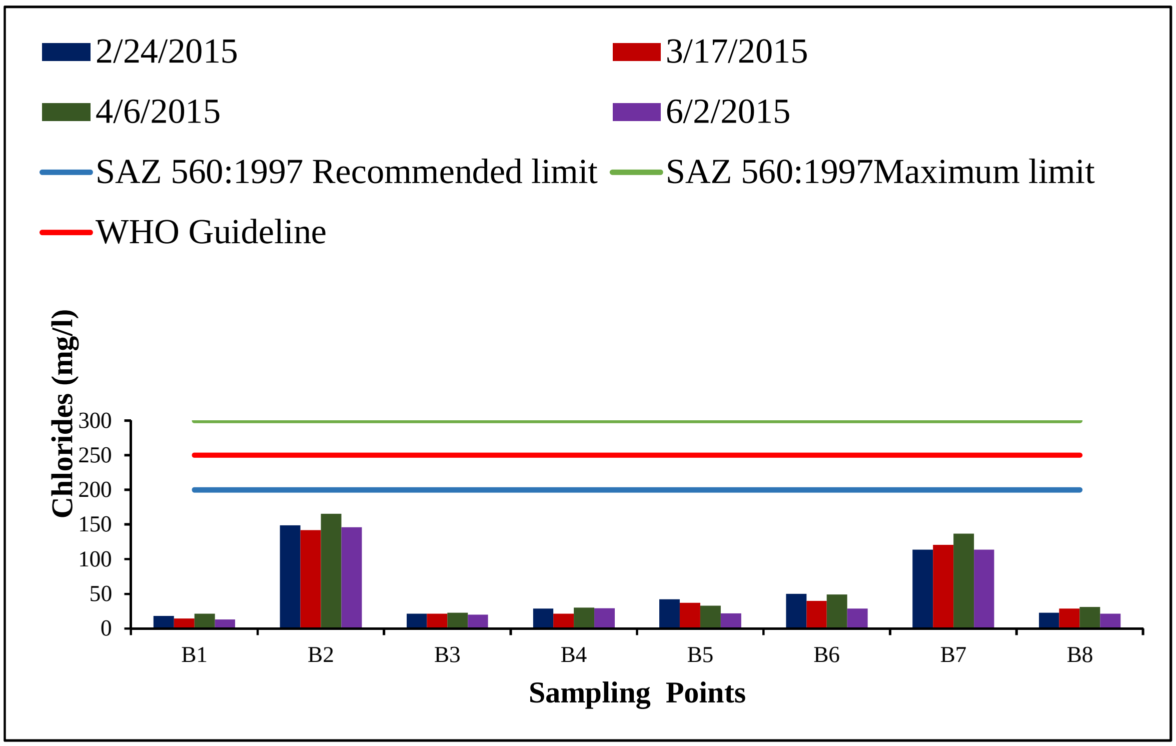

Figure 10: Chlorides variation among sampling points

5.3.3 *Turbidity*

Turbidity was measured in the field at each sampling point for the four sampling campaigns. Turbidity values ranged from 0.09 to 63 NTU with a mean value of (7.3 ±15) NTU. Turbidity results are presented in Figure 13. For all the four sampling campaigns, 23.3 % exceeded the SAZ and WHO recommended and maximum permissible limits of 5 NTU. Figure 13 shows that B7 and B8; the two monitoring wells located within the landfill exceeded the permissible limit. Turbidity affects the acceptability of water to consumers. In drinking water, the higher the turbidity level, the higher the risk that people may develop gastrointestinal diseases. This is because contaminants like viruses or bacteria can become attached to the suspended solids. There were significant pollution differences (p=0.05, df =6) in turbidity levels between leachate samples and all the groundwater samples.

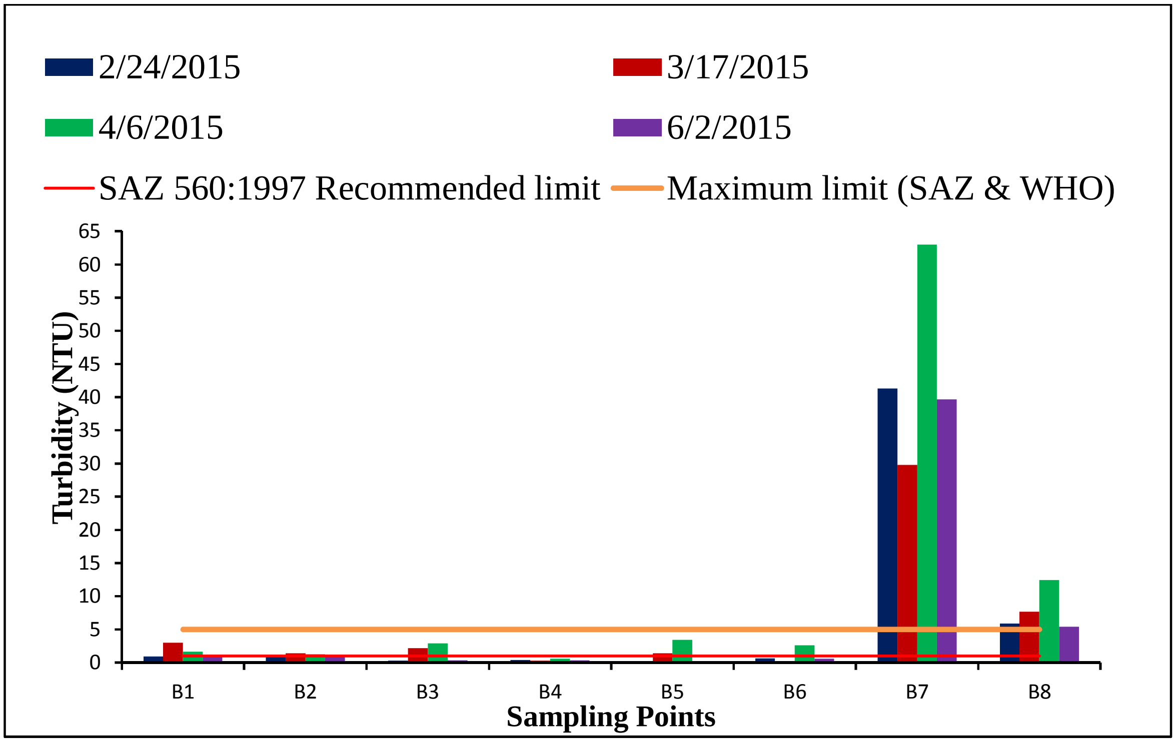

Figure 11: Turbidity variation among sampling points

5.3.4 *Nitrates*

Nitrate values ranged from 0.05 mg/L to 45 mg/L with a mean value of (13 ± 15) mg/L (Table 6). Among all the sampling points, 37.9 % were above the SAZ recommended limit of 10 mg/L. B1 and B6 exceeded the SAZ recommended limit of 10 mg/L (Figure 14). For B1, this could be attributed to application of fertilizers since B1 is within a farm and B6 attributed to landfill leachate since it is located within the landfill. Natural levels of nitrate in groundwater may be enhanced by municipal and industrial wastewaters including leachate from waste disposal sites and sanitary landfills. High nitrate concentrations thus have detrimental effects on infants less than three to six months of age. Nitrate reduces to nitrite which can oxidize haemoglobin (Hb) to methaemoglobin (metHb), thereby inhibiting the transportation of oxygen around the body (Chapman, 1992, Lee and Jones-Lee, 1993; Sabahi et al., 2009). Groundwater in drinking water wells in two suburbs of Ibadan and Lagos were found to have very poor water quality, including unacceptable concentrations of nitrate and ammonia, ascribed to local waste disposal sites (Ikem et al., 2002). Groundwater quality around Pomona Landfill may be said to be highly polluted.

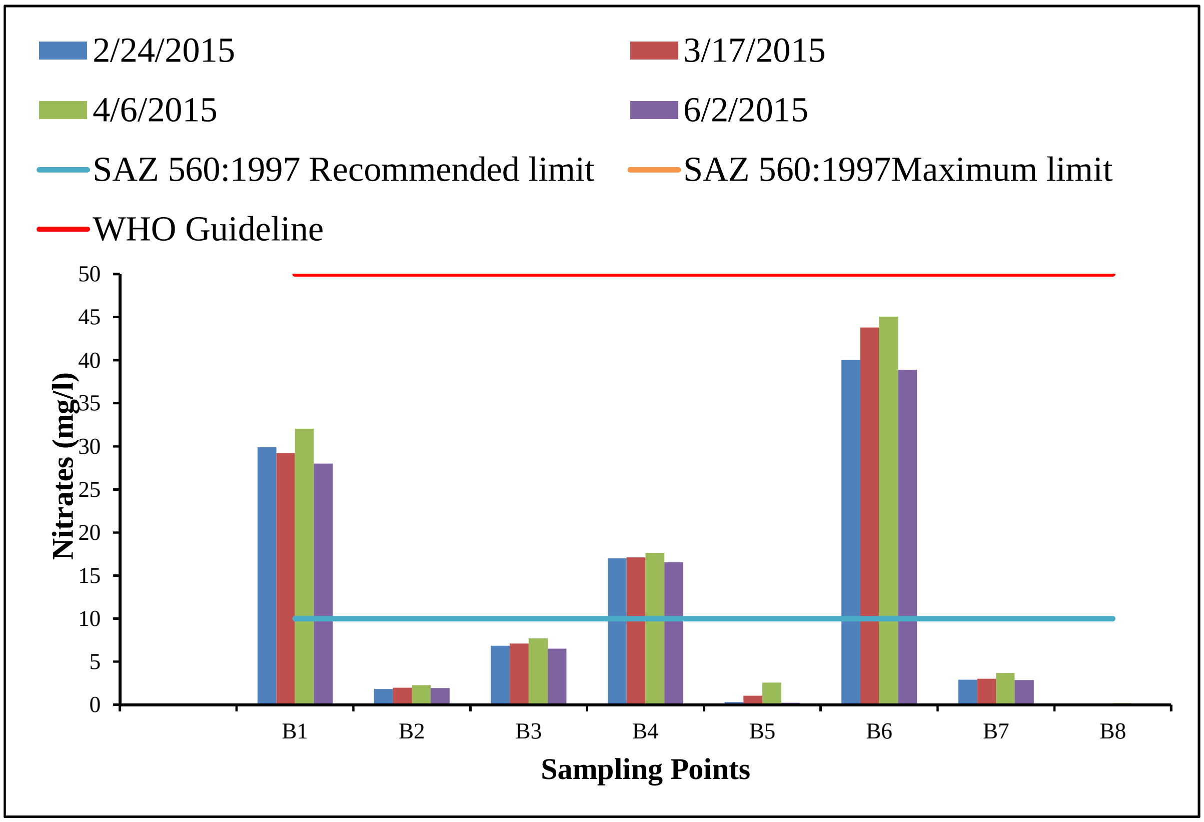

Figure 12: Nitrates variation among sampling points

5.3.5 *Total dissolved solids*

TDS values ranged from 200 mg/L to 570 mg/L with a mean value of 113 mg/L. Longe and Balogun (2010) found comparable results from landfills located in Ibadan and Lagos respectively. In addition, the work of Olaniya and Saxena (1977) has established measurable high level of TDS concentration as an indication of contamination of groundwater near refuse dumpsite. The concentration of Total Dissolved Solids (TDS) indicates the nature of water quality and or its salinity. Improperly lined landfills may lead to increased total dissolved solids concentrations in groundwater. It was therefore concluded that groundwater around Pomona is highly polluted from landfill leachate

Conductivity values ranged from 192 µS/cm - 716 µS/cm; with a mean of (398±167) µS/cm for all the sampling points (Table 6). Almost all the boreholes were within the SAZ recommended and maximum limits of 700 µS/cm and 3000 µS/cm. The range of values found in the study area are comparable to a study carried out by (Agrawal et al., 2011) in Raipur town, India with mean values between 124 µS/cm and 320 µS/cm. Conductivity can be regarded as a crude indicator of water quality for many purposes, since it is related to the sum of all ionised solutes or total dissolved solid (TDS) content. There were significant pollution differences (p=0.05, df =6) in conductivity levels between leachate samples and all the groundwater samples. It was concluded that groundwater samples collected within and near the landfill site contain soluble salts as a result of leachate percolation into the groundwater.

5.3.6 *Total Coliforms*

Total coliforms were too numerous to-count (TNTC). Table 6 presents results for all the sampling points for Total coliforms exceed 0 cfu/100ml SAZ and WHO guideline value (WHO, 2011). The results show a wide variation for all the sampling points for all the sampling dates as the sampling period progressed. B7 and B8 however are the most contaminated since the monitoring wells are located within the landfill. There were significant pollution differences (p=0.05, df =6) in total coliform levels between leachate samples and all the groundwater samples indicating leachate pollution. The presence of coliforms in drinking water indicates that other disease-causing organisms (pathogens) may be present in the system and this implies that the water is not safe for human consumption.

5.3.7 *Heavy Metals*

Table 8: Heavy metals variation among sampling points

	Sampling Points							
Parameter	B1	B2	B3	B4	B5	B6	B7	B8
Chromium	ND	ND	ND	ND	ND	ND	ND	ND
Iron	0.28	0.44	0.60	ND	ND	0.35	0.78	0.59
Zinc	0.03	0.09	0.14	ND	0.01	0.09	0.07	0.07
Arsenic	ND	ND	ND	ND	ND	ND	ND	ND
Copper	0.04	0.19	0.02	ND	ND	ND	0.07	0.05

[1] ND: Not Detected; the detection limit was 0.01 mg/L

Results for heavy metals ranged from Not Detected (ND) to 0.79 mg/ L (Table 8). The detection level was 0.01 mg/ L. From all the sampling points, Fe concentrations at B4 and B5 were found to be below SAZ and WHO limits and above the limit at all other sampling points (0.28 mg/L - 0.78 mg/L). Cu, Pb, Hg, Cd, Cr, As and Zn concentrations were found to be below SAZ and WHO limits at all sampling points. Iron (Fe) occurs naturally in the environment however when it exceeds the natural levels Fe gives colour to water which makes it undesirable to use for laundry, as it stains the clothes. According to Bjerg et al., (1955) and Christensen et al., (2001) heavy metals from landfill do not constitute a groundwater pollution problem due to the strong attenuation (resulting from dilution, dispersion, biodegradation, irreversible sorption and radioactive decay) of these metals in the landfill itself or due to the type of surrounding soil. Other studies have reported the impact of leachate on groundwater, probably due to the fact that the attenuation process was not as effective (Bjerg et al., 1995).

Longe, et al., (2007) found in groundwater near a municipal landfill in Nigeria, high concentrations of heavy metals such as Cr, Cd and Cu and this does not tally with the present study. In another study conducted in India, different physical and chemical parameters were

1

measured and moderately high concentrations of certain metals such as Iron and Zinc were found (Mor et al., 2006). Metals have the affinity to be absorbed by clayey soil (Mor et al., 2006; Longe and Enekwechi, 2007). Their absence in the groundwater samples can be attributed to the sub-surface geology of the site which consists of clay.

5.4 Leachate quantity results using HELP Model

The results that were obtained through the application of HELP Model are shown in Table 9. In this study result indicated that precipitation and evapotranspiration has the most influenced on leachate generation increase and decrease, respectively. 82% of annual precipitation isn.t percolated into Semnan landfill due to evapotranspiration. HELP Model simulations were indicated that the maximum and average value of leachate height above barrier layer is 36 and 3mm, respectively. HELP was run using 22 years (1981 – 2002) of daily climatic data for the study area. The landfill was modelled using three layers (from bottom to top), the barrier soil layer, the compacted solid waste layer and the soil cover layer, the soil used for cover is silty soil with sand that is available at the site. Results of the simulation using HELP including annual leakage/percolation, the average head on the top of the barrier soil layer and volume of leakage through the barrier soil layer are presented in Table 9.

Table 9: Water balance parameters for years 1983-2014

Parameters	**%**	**mm**	**m^3**
Precipitation	100	708	708 000
Runoff	6	43	43 000
Evapotranspiration	63	442	442 000
Percolation/Leakage through the landfill base	13	94	94 000
Change in water storage	18	128	128 000

The average annual leakage from the landfill base was 13% of the average annual total precipitation. Evapotranspiration was 63%, surface runoff was 6% and change in water storage was 18%. The evaporation and leakage through the landfill's base were dominant factors in water balance in landfill cover. As found in literature, for other disposal sites in Europe (Hjelmar, 1989; Ehrig, 1991), the yearly leakage from the base of the site, after the final capping, usually fluctuates between 22% and 50% , evapotranspiration between 30% and 70% and surface runoff between 25% and 40%. These quantities are depended on the local climatic conditions and on the design characteristics of the disposal site. Fatta et al. (1999) found the yearly leakage from the base of the site to be 42.8% of the average annual total precipitation for 33 years. The difference between the results obtained by Fatta et al. (1999) and the current study can be attributed to the fewer number of years entered in the model and also different climatic conditions and landfill design characteristics. The relationship between rainfall and cumulative leachate generation from the landfill indicate that high amounts of leachate were generated over the rainy season normally between October and December of each year. This could mainly be based on initial moisture content of MSW, decomposition of waste and precipitation in this period.

5.5 Groundwater vulnerability assessment

The DRASTIC Model was used in this study to perform a specific vulnerability assessment in the Pomona area.

5.5.1 *Depth-to-Water Table (**D**)*

The water levels from eight boreholes were used to derive the depth-to-water table parameter. The area around Pomona Landfill had water table values ranging from 0.5- 40 m. A high rating of value 9 was assigned to low depth-to-water table areas (see Table 3). A greater part of the study area is highly vulnerable in terms of depth-to-water and contributes 96% of the total area. The area is characterised with the brown and red colour on the depth-to-water table parameter map as shown in Figure 15.

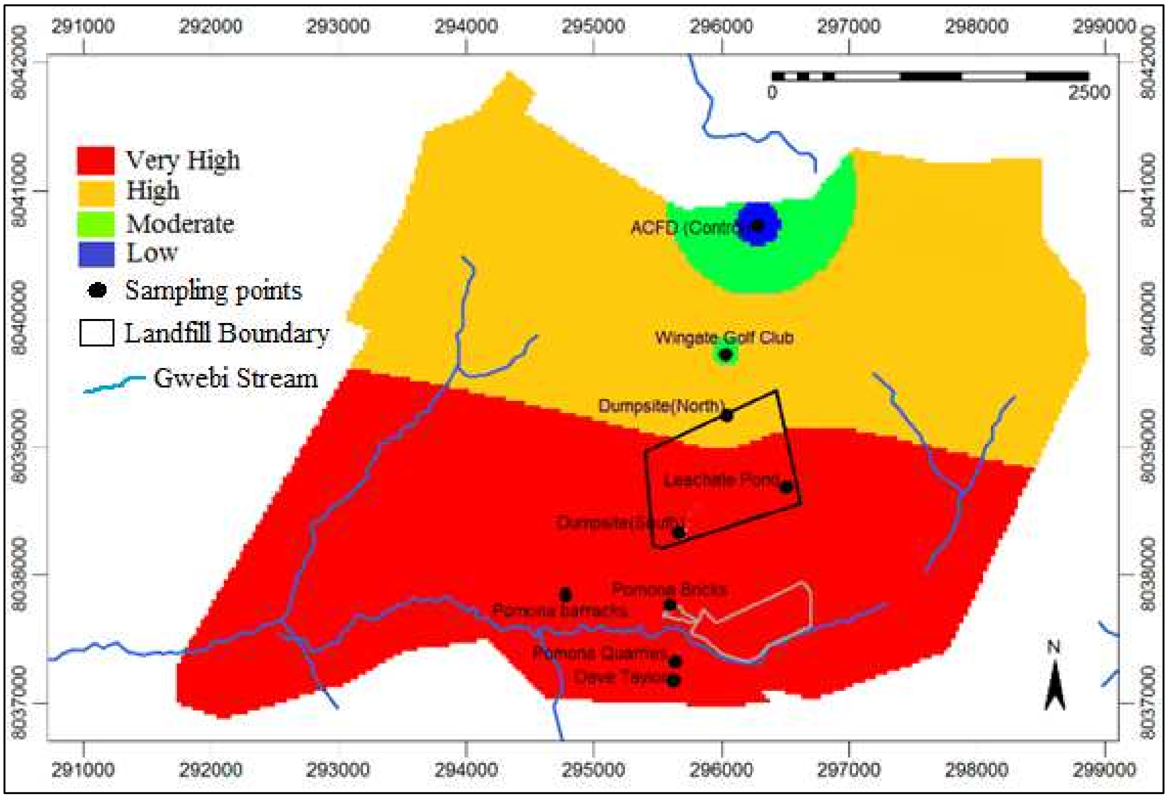

Figure 13: Depth-to-water table parameter map

Recharge values were generated using equation 2 (Piscopo, 2001).The recharge values for Pomona area range from 172 - 176 mm. A rating value of 6 was assigned to the high recharge values. 100% of the study area is highly vulnerable in terms of net recharge. The area is characterised with the brown colour on the net recharge parameter map shown in Figure 16.

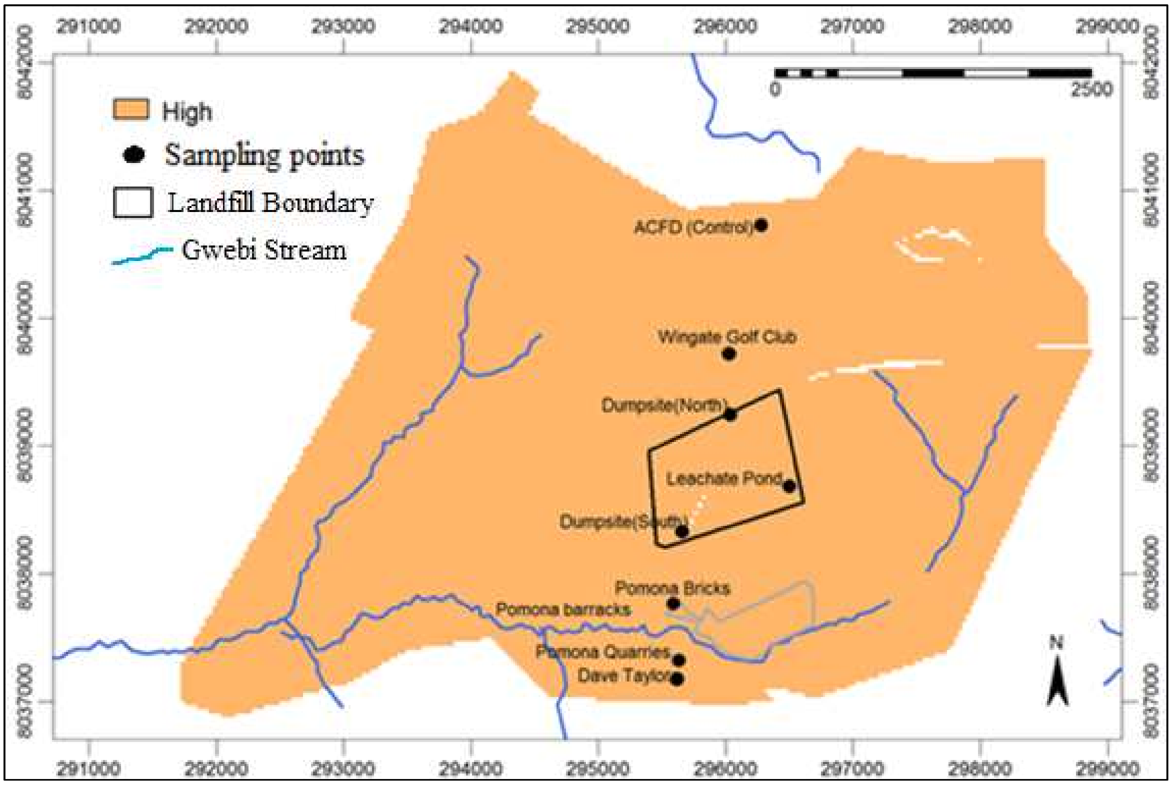

Figure 14: Net Recharge parameter map

The aquifer media in the study area is classified as weathered metamorphic igneous, sand / gravel and basalt. Basalt was given a high rating of value 9 and contributes 26% high vulnerability of the study area is highly vulnerable in terms of aquifer media. The area is characterised with the brown colour as shown in Figure 17.

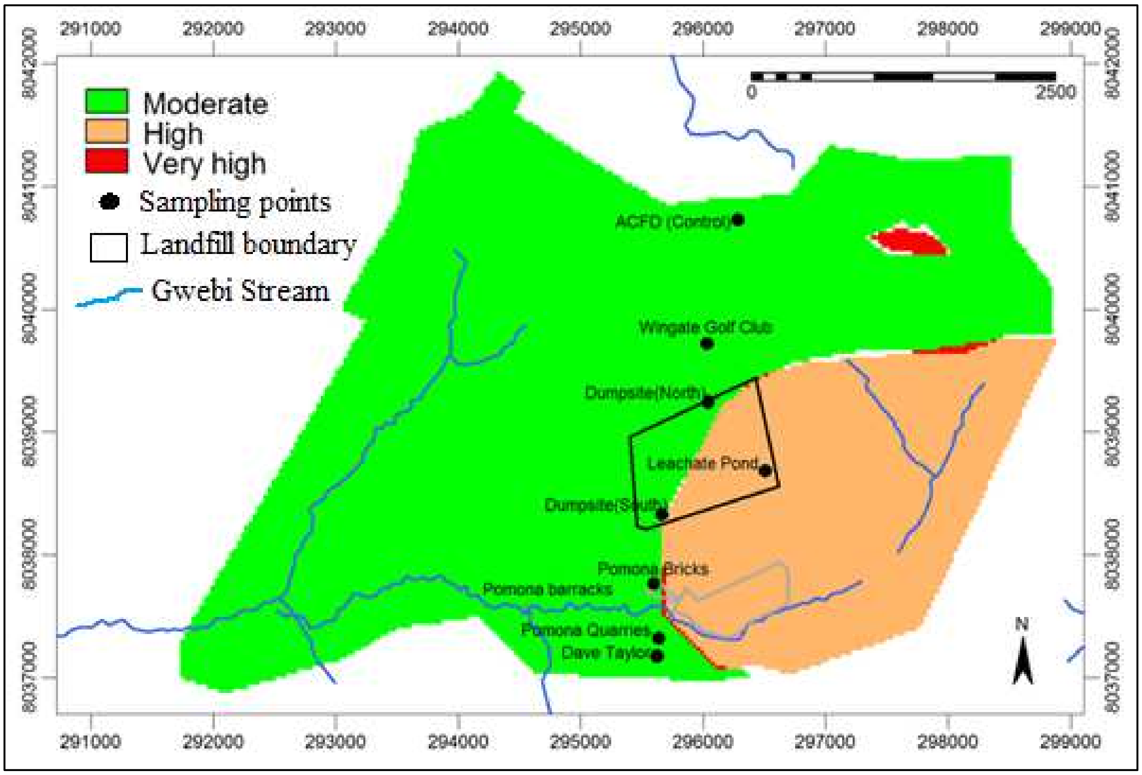

Figure 15: Aquifer media parameter map

5.5.2 *Soil Media (**S**)*

The soil available in the study area was categorized into three texture ranges namely, clay loam, sandy clay loam, and sandy loam. The soil layer was then reclassified to rating values according to the soil types recommended in the DRASTIC model. Sandy loam was the most vulnerable and clay loam, the least vulnerable. Sandy clay loam covers the greater part in the study area which is highly vulnerable in terms of soil media and contributes 99% of the total area. The area is characterised with the brown colour on the soil medium parameter map is shown in Figure 18.

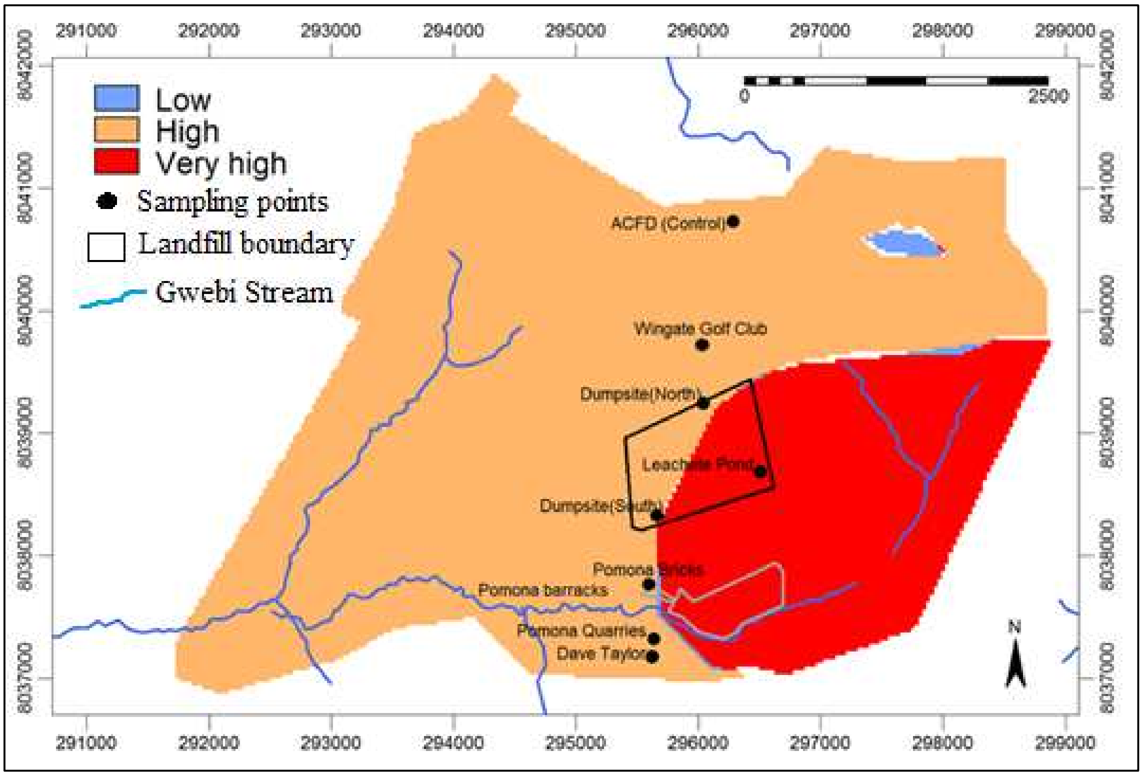

Figure 16: Soil media parameter map

5.5.3 *Topography (**T**)*

The topographical layer displays a gentle slope (0–6%) over most of the study area which has been assigned the DRASTIC ratings of 5, 9, and 10 (Table 3). A greater part of the study area is lowly vulnerable in terms of topography contributing 3% of the total area. The topography parameter map is shown in Figure 20.

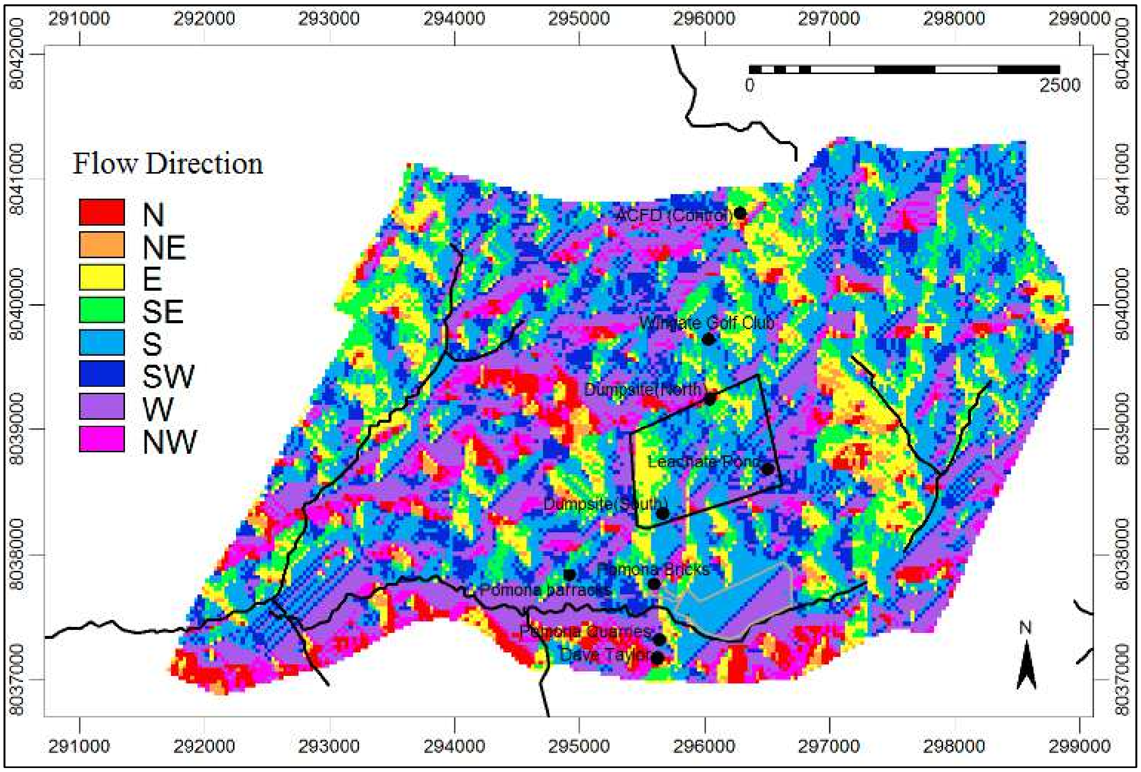

Figure 17: Pomona Digital Elevation Map (DEM)

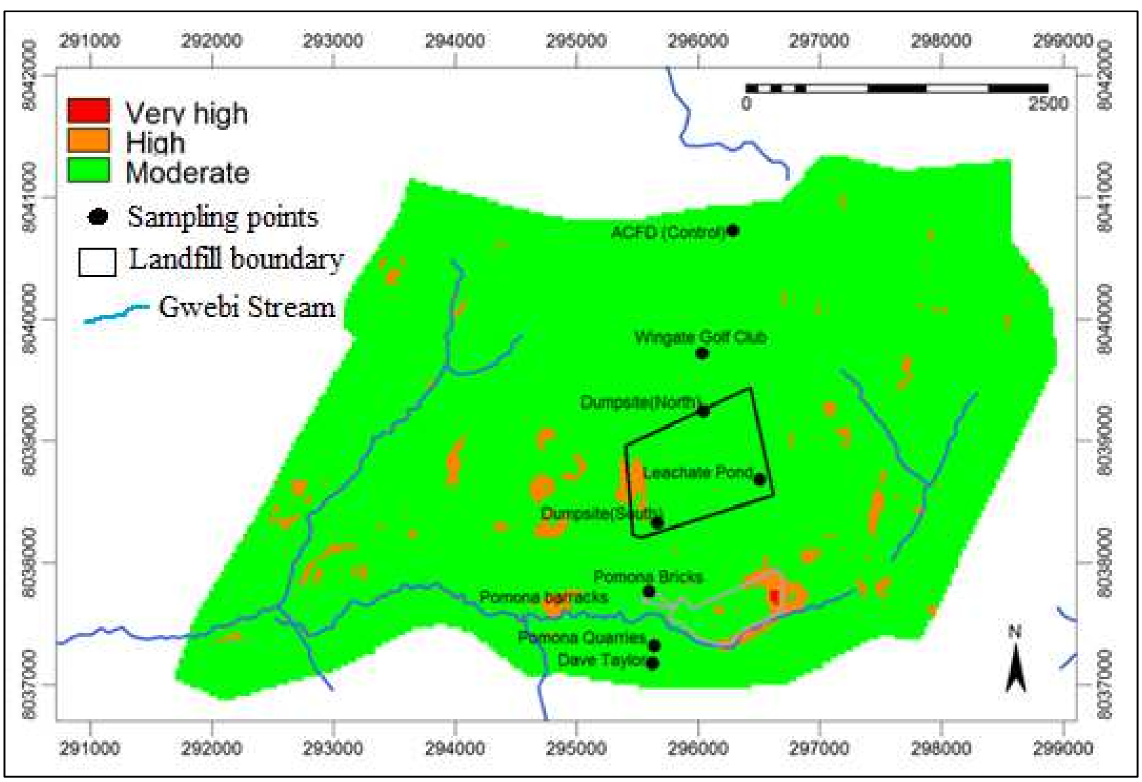

Figure 18: Topography parameter map

5.5.4 *Impact of Vadose Zone (**I**)*

The basaltic soils were assigned a high rating value (8), the sand and gravel was assigned moderate rating value (6) while the lowest rating value (3) were assigned to the silt/clay. 32 % of the total area is highly vulnerable characterised with the brown colour on the impact of the vadose zone parameter map is shown in Figure 21.

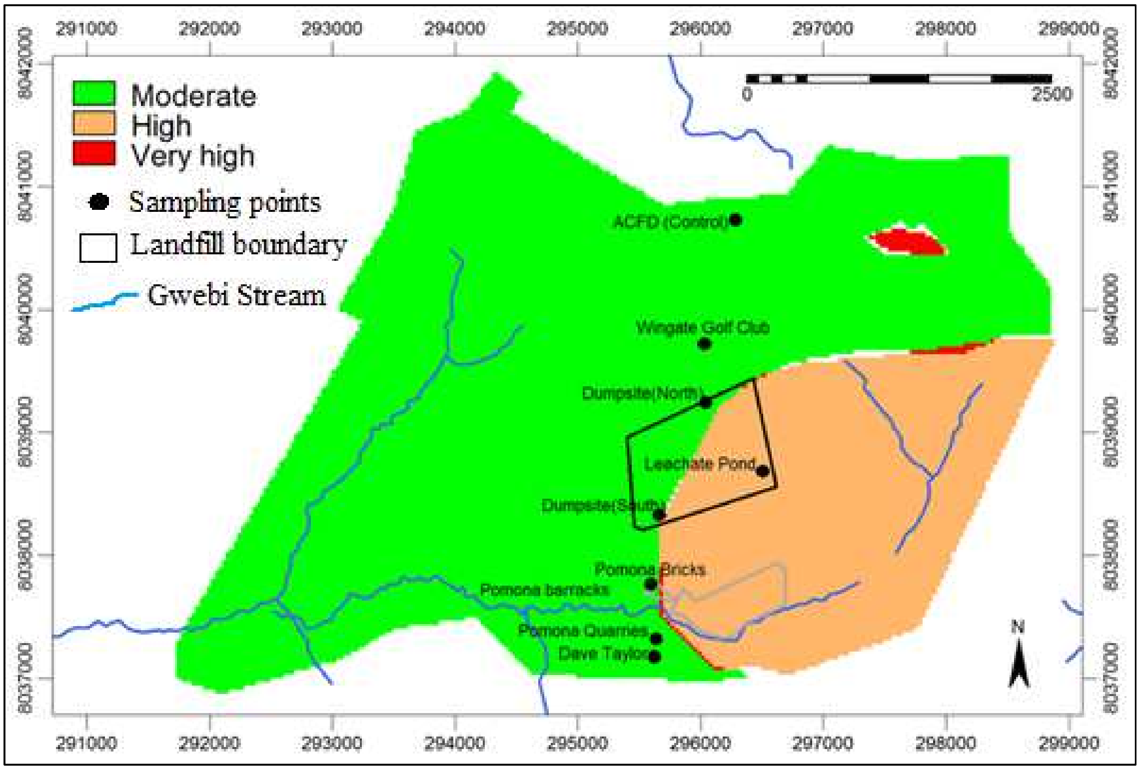

Figure 19: Impact of the vadose zone parameter map

5.5.5 *Hydraulic Conductivity (**C**)*

Hydraulic conductivity values for various rock types have been proposed by Domenico and Schwartz, (1990). There are three main rock types in the study area basalt, granite and dolerite having hydraulic conductivity values of 0.01 m/day, 0.001 m/day and 0.0012 m/day, respectively. Moreover, we used the established map of aquifer lithology as a base to estimate the values of hydraulic conductivity using (Rodrìguez et al., 2001). The study area had hydraulic conductivity values ranging from 0.01 m/day to 1.21 m/day and they were all assigned a rating value of 1; indicating low vulnerability. The resulting hydraulic conductivity map is shown in Figure 22.

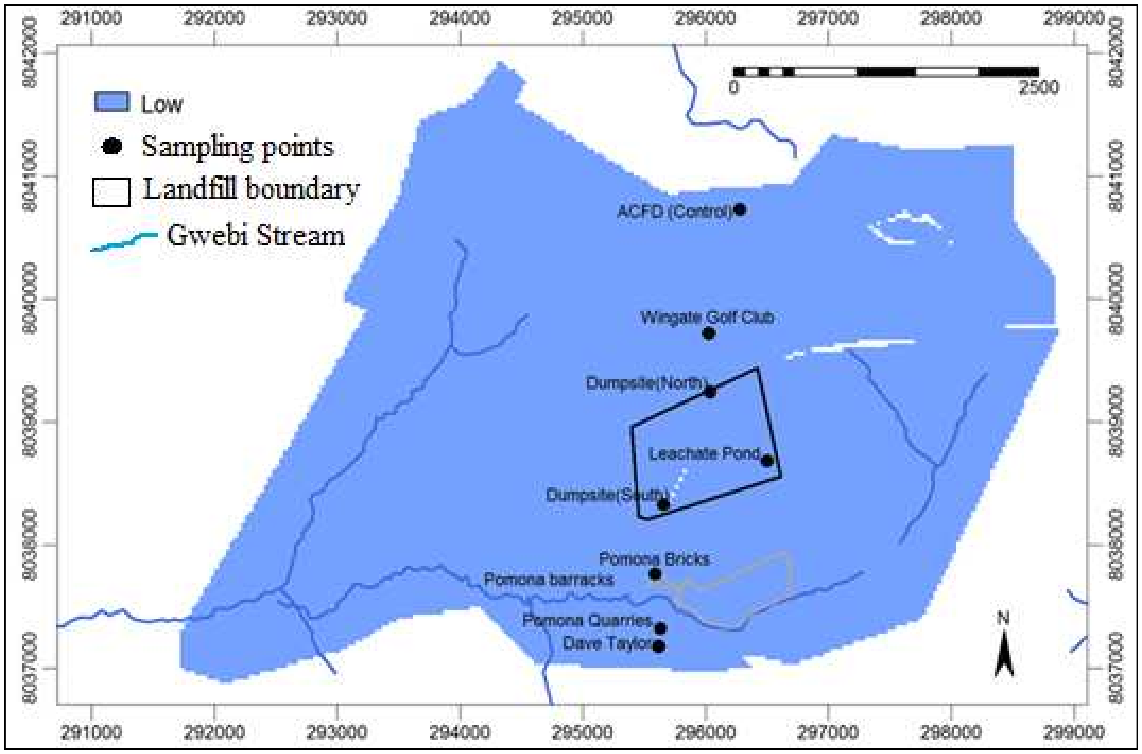

Figure 20: Hydraulic conductivity parameter map

5.5.6 *Drastic Vulnerability Index Map (DI)*

The established generic DRASTIC Index map shows four vulnerability classes: low, moderate, high and very high vulnerability as per Natural Breaks (Jenks) classification method (Figure 23). DI map shows four vulnerability classes: low (38%), moderate (58%), high (3%) and very high vulnerability (1%). High vulnerability falls in the farming area. The landfill area is highly vulnerable in terms of depth to water table, net recharge, soil media and partly aquifer media. In terms of Impact to vadose zone, the most significant parameter the landfill lies in moderately vulnerable area. Highly sensitive areas are not within the landfill

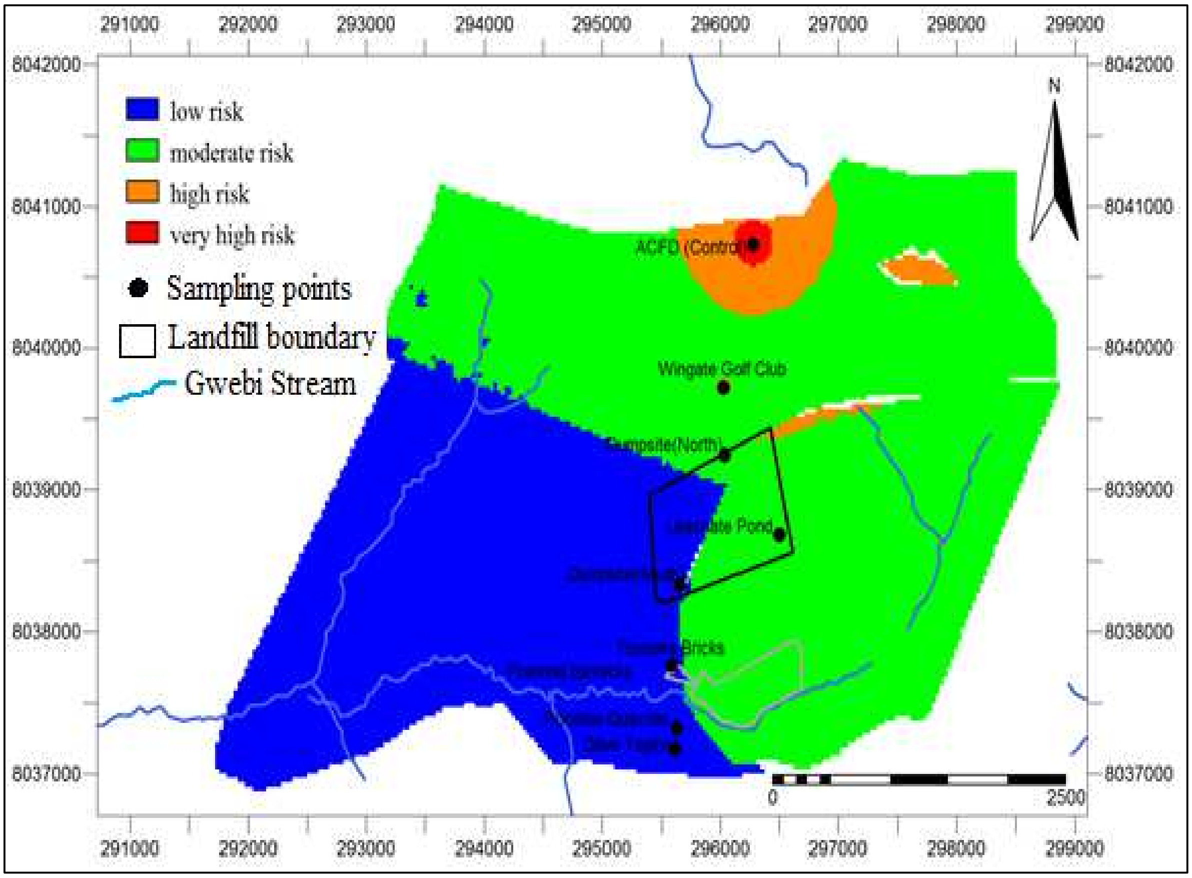

Figure 21: Vulnerability map

The study area's vulnerability was classed as low, medium and high according to data obtained from hydrogeological investigations. High vulnerability lies at African Centre for Fertiliser Development (ACFD) which is a farming area. High vulnerability in this area is attributed to depth-to-water-table. The landfill area is highly vulnerable in terms of depth to water table, net recharge, soil media and partly aquifer media. In terms of Impact to vadose zone, the most significant parameter the landfill lies in moderately vulnerable area. However the highly sensitive areas are not within the landfill. It can be concluded that of the total area (22 km²), 38% lies in the low vulnerability area, 58% in the moderate vulnerability area, 3% in the highly vulnerable areas and 1% in very high vulnerable areas.

CHAPTER SIX: CONCLUSIONS AND RECOMMENDATIONS

6.0 Conclusions

1) The concentrations of Cl^-, NO_3^-, Fe, Zn in groundwater samples fall below SAZ limits and WHO guidelines indicating an insignificant impact of leachate on groundwater. It is however observed that in the absence of a properly designed leachate collection system, uncontrolled accumulation of leachates at the base of the landfill pose potential contamination risk to groundwater resource in the near future.
2) The composition of the leachate under study showed a significant range of various pollutants that pose pollution to groundwater. Parameters like; EC, TDS, Cl^- and NO_3^- fall in the red category of EMA regulations which is considered to present a high environmental hazard with respect to wastewater disposal. The organic load appeared to be quite high and the low BOD_5/COD ratio confirmed the fact that the landfill is mature.
3) The application of Hydrologic Evaluation of Landfill Performance Model showed that the yearly leakage from the base of the landfill is 56% of the total annual precipitation. The average volume of leachate discharged from Pomona Landfill during the period 1983 to 2014 was around, 79,000 m^3/year. It was also concluded that the leakage from the landfill base is greatly influenced by precipitation.
4) Pomona Landfill falls both in the low and moderately vulnerable area. Out of the total area (22 km^2), 38% lies in the low vulnerability area, 58% in the moderate vulnerability area, 3% in the highly vulnerable areas and 1% in very high vulnerable areas.

6.1 Recommendations

1) It is therefore recommended that boreholes within the vicinity of the landfill used for drinking purposes be monitored regularly. Monitoring wells must be sited and located on the most appropriate position for easier detection of contaminants in drinking water.

2) It is also recommended that the landfill be decommissioned and a properly engineered landfill constructed.
3) It is recommended that groundwater modelling be adopted by relevant authorities.
4) It is further recommended that specific vulnerability index maps be used as screening tools to spotlight trouble spots and not as an alternate for detailed site-specific analysis. As detailed site specific analysis is costly, these assessments can be used as tools, which identify the zones of concern and as a tool to identify the need for a detailed assessment into such zones of concern.

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APPENDICES

Appendix 1: Hydraulic conductivity values for various rock types

The following tables show representative values of hydraulic conductivity for various unconsolidated sedimentary materials, sedimentary rocks and crystalline rocks (Domenico and Schwartz 1990).

Unconsolidated Sedimentary Materials	
Material	**Hydraulic Conductivity (m/sec)**
Gravel	$3x10^{-4}$ to $3x10^{-2}$
Coarse sand	$9x10^{-7}$ to $6x10^{-3}$
Medium sand	$9x10^{-7}$ to $5x10^{-4}$
Fine sand	$2x10^{-7}$ to $2x10^{-4}$
Silt, loess	$1x10^{-9}$ to $2x10^{-5}$
Till	$1x10^{-12}$ to $2x10^{-6}$
Clay	$1x10^{-11}$ to $4.7x10^{-9}$
Unweathered marine clay	$8x10^{-13}$ to $2x10^{-9}$

Sedimentary Rocks	
Rock Type	**Hydraulic Conductivity (m/sec)**
Karst and reef limestone	$1x10^{-6}$ to $2x10^{-2}$
Limestone, dolomite	$1x10^{-9}$ to $6x10^{-6}$
Sandstone	$3x10^{-10}$ to $6x10^{-6}$
Siltstone	$1x10^{-11}$ to $1.4x10^{-8}$
Salt	$1x10^{-12}$ to $1x10^{-10}$
Anhydrite	$4x10^{-13}$ to $2x10^{-8}$
Shale	$1x10^{-13}$ to $2x10^{-9}$

Crystalline Rocks	
Material	**Hydraulic Conductivity (m/sec)**
Permeable basalt	$4x10^{-7}$ to $2x10^{-2}$
Fractured igneous and metamorphic rock	$8x10^{-9}$ to $3x10^{-4}$
Weathered granite	$3.3x10^{-6}$ to $5.2x10^{-5}$
Weathered gabbro	$5.5x10^{-7}$ to $3.8x10^{-6}$
Basalt	$2x10^{-11}$ to $4.2x10^{-7}$
Unfractured igneous and metamorphic rock	$3x10^{-14}$ to $2x10^{-10}$

Appendix 2: Water quality test results for all sampling campaigns

C1: Sampling Campaign 1 **C2**: Sampling Campaign 2

C3: Sampling Campaign 3 **C4**: Sampling Campaign 4

LEACHATE POND					
		C1	**C2**	**C3**	**C4**
Parameter	**Units**	**24-Feb-15**	**17-Mar-15**	**28-Apr-15**	**2-Jun-15**
pH		7.3	7.4	8.4	8
Dissolved Oxygen	%	16	41	93	16
Temperature	T˚C	14	27	29	27
Turbidity	NTU	121	88	55	119
Electrical Conductivity	µS/cm	5360	6150	5670	5400
Total Dissolved Solids	mg/l	2730	3070	3256	2720
Total Coliforms	No/100ml	TNTC	TNTC	TNTC	TNTC
Faecal Coliforms	No/100ml	80	35	40	71
Chloride	mg/l	2793	1383	650	2790
COD	mg/l	430	288	265	428
BOD_5	mg/l	39	38	42	64
Nitrates	mg/l	62	59	48	65
Phosphates	mg/l	3.1	2.0	1.9	2.9
Sulphates (SO_4^{2-})	mg/l	364	659	475	350
Mercury	mg/l	0	0	0	0
Arsenic	mg/l	0.0	0.0	0.0	0.0
Cadmium	mg/l	0.0	0.0	0.0	0.0
Zinc	mg/l	0.2	0.2	0.1	0.2

Lead	mg/l	0.0	0.0	0.0	0.0
Iron	mg/l	11	9.7	8.0	10.6
Copper	mg/l	0.2	0.5	0.0	0.2
Chromium	mg/l	0.1	0.1	0.0	0.1

LEACHATE TRENCH					
		Campaign 1	**Campaign 2**	**Campaign 3**	**Campaign 4**
Parameter	**Units**	**24-Feb-15**	**17-Mar-15**	**28-Apr-15**	**2-Jun-15**
pH		7.7	7.1	7.9	8.1
Dissolved Oxygen	%	49	25	48	46
Temperature	T˚C	36	36	34	26
Turbidity	NTU	257	83	105	250
Electrical Conductivity	µS/cm	5620	5380	5650	5620
Total Dissolved Solids	mg/l	3974	3800	3956	3980
Total Coliforms	No/100ml	TNTC	TNTC	TNTC	TNTC
Faecal Coliforms	No/100ml	75	55	85	71
Chloride	mg/l	1198	1453.5	870	1100
COD	mg/l	298	274	368	290
BOD_5	mg/l	57	60	75	52
Nitrates	mg/l	95	86	82	102
Phosphates	mg/l	5	3	3	4
Sulphates(SO_4^{2-})	mg/l	859	1046	965	850
Mercury	mg/l	0	0	0	0.01
Arsenic	mg/l	0	0	0	0.01
Cadmium	mg/l	0	0	0	0.025
Zinc	mg/l	0.23	0.28	0.31	0.18
Lead	mg/l	0.28	0.2	0.07	0.32
Iron	mg/l	16	13	12	14
Copper	mg/l	0.35	0.2	0.18	0.3
Chromium	mg/l	0.1	0.17	0.03	0.08

		C1	C2	C3	C4	C1	C2	C3	C4	C1	C2	C3	C4
		24-Feb	**17-Mar**	**27-Apr**	**2-Jun**	**24-Feb**	**17-Mar**	**27-Apr**	**2-Jun**	**24-Feb**	**17-Mar**	**27-Apr**	**2-Jun**
Parameter	**Units**	**BH1**				**BH2**				**BH3**			
		ACFD				**Wingate**				**Pomona Bricks**			
pH		6.4	6.32	6.4	6.5	6.8	6.5	6.5	6.7	6.8	6.6	7.1	7.3
Dissolved Oxygen	%	59	64.8	43	55	60	69.3	51	59	71	51	64.6	65
Temperature		21.3	22.2	22.6	21.1	25.1	24	22.9	22	25.6	27.4	26.1	25.8
Turbidity	NTU	0.91	2.99	1.64	1.05	1.17	1.4	1.27	1.1	0.27	2.16	2.9	0.32
Electrical Conductivity	$\mu S/cm$	230	226	220	227	641	658	716	638	254	255	261	250
Total Dissolved Solids	mg/l	115	119	119	110	300	334	354	301	127.3	129.5	130	125
Total Coliforms	No/100ml	6	5	0	8	6	12	8	5	3	8	2	0
Faecal Coliforms	No/100ml	0	0	0	0	2	0	0	0	0	1	0	0
Chloride	mg/l Cl	18	14.2	21.1	13	148.89	141.8	165.1	146	21.27	21.27	23	19.7
COD	mg/l	200	216	214	182	756	396	745	648	108	396	405	191
BOD_5	mg/l	26	31.5	33.6	23.45	24.23	34	37	23.1	27.2	39.3	42	24.93
Nitrates	mg/l NO_3^-	29.88	29.2	32.05	28	1.8	1.99	2.27	1.91	6.85	7.1	7.68	6.51
Phosphates	mg/l PO_4^{3-}	0.18	0.12	0.1	0.16	0.6	0.91	1.4	0.71	0.6	0.67	0.71	0.58
Sulphates	mg/l SO_4^{2-}	2200	2262	2270	2210	2100	2154	2205	2000	1980	2000	2015	1985.8
Lead	mg/l Pb	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Mercury	mg/l Hg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Chromium	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Iron	mg/l Fe	0.24	0.32	0.3	0.22	0.23	0.62	0.45	0.2	0.5	0.53	0.67	0.48
Cadmium	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Zinc	mg/l	0.02	0.04	0.04	0	0.1	0.09	0.17	0.08	0.11	0.14	0.2	0.09

Arsenic	mg/l As	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Copper	mg/l Cu	0.01	0.03	0.05	0	0.19	0.17	0.23	0.11	0.03	0.04	0.06	0

		C1	C2	C3	C4	C1	C2	C3	C4	C1	C2	C3	C4
Parameter	**Units**	**24-Feb**	**17-Mar**	**28-Apr**	**2-Jun**	**24-Feb**	**17-Mar**	**28-Apr**	**2-Jun**	**24-Feb**	**17-Mar**	**28-Apr**	**2-Jun**
		BH4				**BH5**				**BH6**			
		Pomona Quarry				**Pomona Barracks**				**Dave Taylor**			
pH		7.2	7.29	8.15	8	7.1	6.78	7.3	7.7	6.5	6.5	6.8	6.6
Dissolved Oxygen	%	61	68	84.2	80.1	67	68.8	67.2	56.2	61	32	58	51
Temperature	T˚C	25.8	26.1	27.1	25.4	24.5	21.5	24.3	22.7	23.1	24.2	23.9	22.4
Turbidity	NTU	0.43	0.27	0.61	0.32	0.1	1.39	3.44	0.09	0.65	0.1	2.59	0.6
Electrical Conductivity	µS/cm	433	429	456	430	360	384	398	366	376	382	398	365
Total Dissolved Solids	mg/l	211	215	222	205	216	416	570	188	193.6	191.5	210	190
Total Coliforms	No/100ml	0	8	2	6	25	30	15	20	9	4	0	3
Faecal Coliforms	No/100ml	0	0	1	0	0	0	0	0	1	0	1	0
Chloride	mg/l	28.36	21.3	30.15	29.3	42	37	33	21.8	49.63	40	49	28.4
COD	mg/l	144	216	341	165	200	175.2	165.8	265	180	576	467	214
BOD_5	mg/l	26.4	33.8	35	23.73	18.52	20.7	26.8	13.2	19.12	22.9	27	16
Nitrates	mg/l	17	17.11	17.6	16.52	0.32	1.04	2.56	0.2	40	43.79	45.05	38.9
Phosphates	mg/l	1.01	1.64	1.77	0.9	0.01	0.02	0.14	<0.01	1.03	1.52	2.01	0.94
Sulphates	mg/l	1200	1230.77	1292	1198	1989	2000	2010	1950	1100	1180	1200	1098.7
Lead	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Mercury	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Chromium	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Iron	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	0.36	0.32	0.45	0.37
Cadmium	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Zinc	mg/l	ND	ND	ND	ND	ND	0.01	ND	ND	0.01	0.06	0.1	ND
Arsenic	mg/l	0.002	0	0.014	0	0.001	0.002	ND	ND	ND	ND	ND	ND
Copper	mg/l	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

		C1	C2	C3	C4	C1	C2	C3	C4
Parameter	**Units**	**24-Feb**	**17-Mar**	**28-Apr**	**2-Jun**	**24-Feb**	**17-Mar**	**28-Apr**	**2-Jun**
		BH7				BH8			
		Landfill (Northern BH)				**Landfill (Southern BH)**			
pH		7.8	6.7	7.8	7.6	7.2	6.29	6.5	6.9
Dissolved Oxygen	%	25	35.5	40	51	67.2	78.5	101	80.1
Temperature	T˚C	26.2	17.6	26.4	24.1	22.6	21.3	24.2	22.2
Turbidity	NTU	41.3	29.8	63	39.7	5.88	7.66	12.4	5.4
Electrical Conductivity	µS/cm	635	629	648	630	212	216	192.1	209
Total Dissolved Solids	mg/l	306	313	332	300	136	109.6	97.3	129.8
Total Coliforms	No/100ml	80	80	80	80	80	80	80	80
Faecal Coliforms	No/100ml	80	80	80	80	80	80	80	80
Chloride	mg/l	113.44	120.53	136.8	113.44	23.03	28.36	31.1	21.3
COD	mg/l	396	216	367	405	171.2	180	202	165
BOD_5	mg/l	19.38	29.17	33.09	20	32.1	39.8	43	29.88
Nitrates	mg/l	2.9	3.02	3.67	2.86	0.07	0.1	0.18	0.05
Phosphates	mg/l	1.2	1.27	2.03	1	0.6	0.67	0.84	0.61
Sulphates	mg/l	450	453.85	460.04	445	458	462	470	460.35

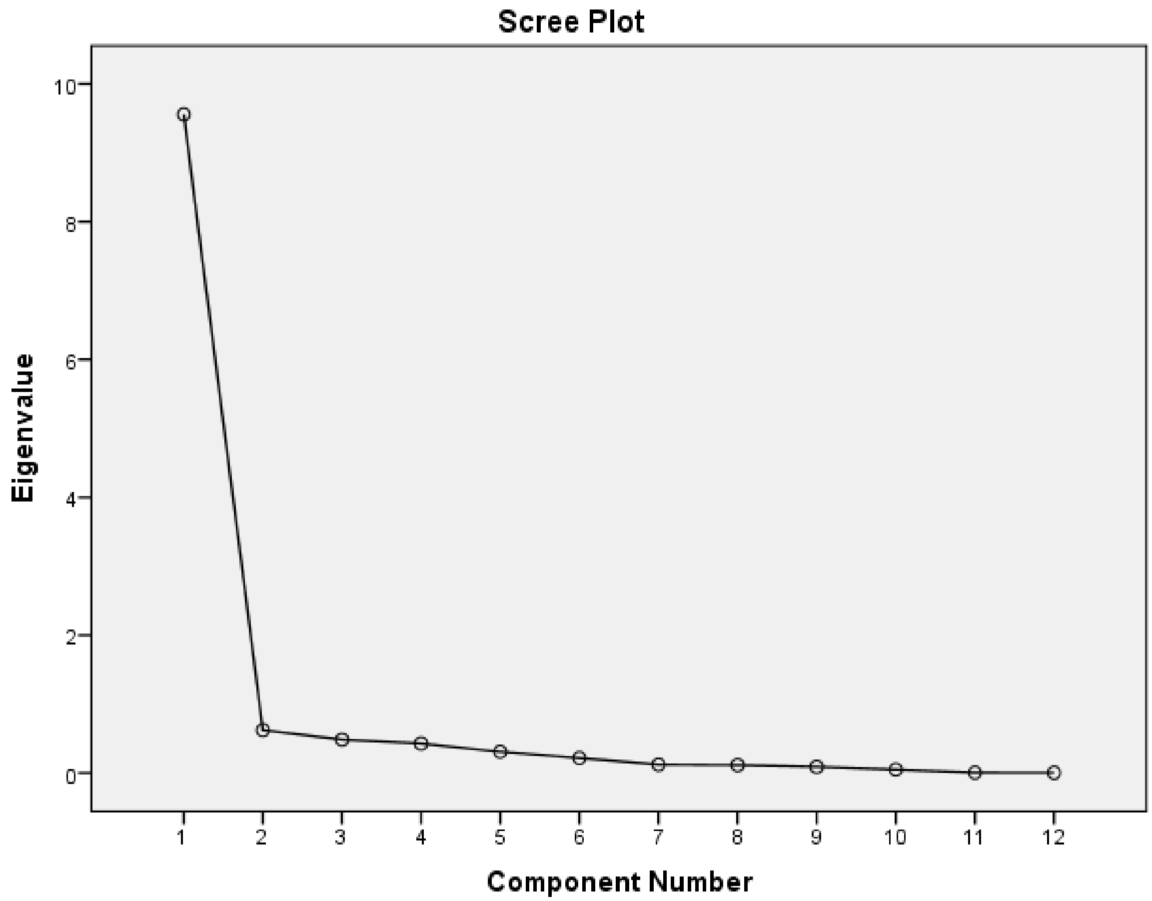
Lead	mg/l	ND	ND	ND	ND	ND	ND	ND	ND
Mercury	mg/l	ND	ND	ND	ND	ND	ND	ND	ND
Chromium	mg/l	ND	ND	ND	ND	ND	ND	ND	ND
Iron	mg/l	0.71	0.6	1	0.8	0.51	0.6	0.8	0.46
Cadmium	mg/l	ND	ND	ND	ND	ND	ND	ND	ND
Zinc	mg/l	0.06	0.08	0.1	0.05	0.04	0.07	0.1	0.05
Arsenic	mg/l	ND	ND	ND	ND	ND	ND	ND	ND
Copper	mg/l	0.07	0.06	0.09	0.05	0.04	0.05	0.071	0.03

Appendix 3: PCA Results

Anti-image Matrices

		Turbidity	EC	TDS	TC	Cl^-	$NO3^-$	Zn	As	Cu	BOD5	$PO4^+$	Cr
Anti-image Correlation	Turbidity	.759[a]	.525	-.071	-.402	-.227	.155	.336	-.652	-.468	.338	-.638	.600
	EC	.525	.758[a]	-.772	-.407	-.770	-.099	.306	-.033	-.570	.280	-.313	.538
	TDS	-.071	-.772	.816[a]	.066	.781	.141	-.199	-.529	.398	-.115	.109	-.238
	TC	-.402	-.407	.066	.903[a]	.082	.411	-.054	.115	.330	-.246	.125	-.216
	Cl^-	-.227	-.770	.781	.082	.766[a]	.104	-.255	-.238	.379	-.131	.195	-.566
	$NO3^-$	.155	-.099	.141	.411	.104	.912[a]	.120	-.434	.227	.049	-.322	.074
	Zn	.336	.306	-.199	-.054	-.255	.120	.861[a]	-.130	-.518	-.406	-.305	.200
	As	-.652	-.033	-.529	.115	-.238	-.434	-.130	.862[a]	.012	-.259	.312	-.424
	Cu	-.468	-.570	.398	.330	.379	.227	-.518	.012	.787[a]	.002	.227	-.339
	BOD5	.338	.280	-.115	-.246	-.131	.049	-.406	-.259	.002	.891[a]	-.254	.301
	$PO4^+$	-.638	-.313	.109	.125	.195	-.322	-.305	.312	.227	-.254	.855[a]	-.420
	Cr	.600	.538	-.238	-.216	-.566	.074	.200	-.424	-.339	.301	-.420	.798[a]

a. Measures of Sampling Adequacy(MSA)



Communalities		
	Initial	**Extraction**
Turbidity	1.000	0.808
EC	1.000	0.930
TDS	1.000	0.934
TC	1.000	0.834
Cl^-	1.000	0.747
NO_3^-	1.000	0.743
Zn	1.000	0.678
As	1.000	0.972
Cu	1.000	0.643
BOD_5	1.000	0.667
PO_4^{3-}	1.000	0.813
Cr	1.000	0.790
Extraction Method: Principal Component Analysis.		