ENVIRONMENTAL IMPACT OF LIVESTOCK

MORTALITIES BURIAL

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ABSTRACT

The objective of this thesis was to determine the potential impact on groundwater quality as a result of the release of leachate from livestock mortality burial for three species of livestock: swine, bovine and poultry. Specific objectives were to:

1. Characterize the chemical composition of leachate in livestock mortality burial pits for three species: bovine, swine and poultry; and

2. Evaluate the potential environmental impact of livestock burial through groundwater transport modelling.

A two part program was followed to achieve these objectives. The first portion involved construction of lined burial pits complete with leachate collection systems. Poultry (1300kg), swine (5900kg) and bovine (9750kg) carcasses were each placed in separate pits and the pits covered with plastic liner material and then approximately one meter of earthen cover. The pits were sampled for leachate chemical analysis at 2 weeks, 1 month, 2 months, 4 months, 8 months, 14 months and 25 months post burial. The second portion involved using the chemical analysis results from first portion and two groundwater modeling software packages (CTRAN and PHREEQC) to characterize the leachate and evaluate the potential this material could have on groundwater resources adjacent to burial pits.

The results indicated that livestock mortality leachate contains, on average, after two years of decomposition, concentrations of 12,600 mg/L of ammonium-N, 34,600 mg/L alkalinity (as bicarbonate), 2,600 mg/L chloride, 3,600 mg/L sulphate, 2,300 mg/L potassium, 1,800 mg/L sodium, 1,500 mg/L phosphorus along with relative lesser amounts of iron, calcium and magnesium. Maximum values for the major ions were up to 50% higher than the average in some instances. The pH of the leachate was near neutral. In comparison to earthen manure storages and landfills, the strength of the leachate was 2-4 times higher.

To properly characterize the leachate chemistry, speciation of the mortality leachate was

performed using PHREEQC. This speciation provided evidence of phosphate compounds precipitating from solution, as well as significant amounts of phosphoric acids (0.03 mol/L). Relatively high concentrations of ammonium sulphate also formed and due to the negative charge, allow for potentially 300 mg-N/L to transport conservatively. In comparison to naturally occurring groundwater, activities of bicarbonate, sulphates, phosphates and other minerals were many orders of magnitude higher than concentrations present in groundwater.

Preliminary simulations were created with two software packages, Geo-Slope CTRAN and PHREEQC to simulate transport of the leachate for three different soil conditions. The Geo-Slope model models a conservative contaminant, while the PHREEQC model involves geochemical speciation and contaminant transport including ion exchange occurring along the pathway. Transport through a low permeable soil ($K=1 \times 10^{-10}$ m/s) was dominated by diffusion allowing unattenuated leachate to transport a distance of approximately three meters in 50 years. The moderately permeable soil situation ($K=1 \times 10^{-9}$ m/s) produced a transport depth of six meters with an approximate concentration of the tracer thirty to forty percent of initial concentration in 50 years. In a highly permeable soil ($K=1 \times 10^{-8}$ m/s), transport reached a depth of 10 meters in 10 years with approximately forty percent of initial concentration. The PHREEQC transport model demonstrated a highly concentrated calcium and magnesium plume forming in front of the ammonium plume suggesting ion exchange and attenuation of ammonium.

In the occurrence of a mass mortality event, regulators in Canada have decided to employ a trench burial system. Trenches could be created using on-the-farm equipment such as backhoes to obtain approximate trench dimensions of 2 m wide and 4 m deep. To assess the impact of multiple trenches and their appropriate spacing, models were created with Geo-Slope CTRAN to evaluate the effects on trench spacing. It was determined through these models that a minimum 10 m separation distance would provide a potential contaminant plume maximum soil contact and no trench-to-trench impact.

To further evaluate the potential impact of livestock burial leachate, mass loading into an aquifer was evaluated for a moderately permeable soil ($K = 1 \times 10^{-9}$ m/s) for a mass mortality event in a 10,000 head feedlot. Disposal consisted of ten 200 m trenches with a 10 m separation distance. Disposal covered 2.2 hectares and provided a mass loading of ammonium to an aquifer 10 m below of 950 kg/year after 50 years and increasing from 50 years until the peak concentration of the plume reached the aquifer. At this loading rate, nitrogen concentrations exceed drinking water standards 10-15 times.

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DEDICATION

This thesis is dedicated to my Mom and Dad for their constant love and support, to my brothers David and Douglas for being the best brothers in the entire world, and to Christopher for pushing me and getting me through this. Without all of you this would not have been possible.

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1   INTRODUCTION

Canada's agricultural industry plays an important role in livestock production throughout the world. According to Statistics Canada (2007), Canada is home to approximately 16.2 million head of cattle, 14.5 million swine, 1.7 million sheep, 625 million chickens and 20.5 million turkeys. Of the 16.2 million cattle, over half reside in Saskatchewan and Alberta. Livestock populations can easily be threatened by foreign disease outbreaks, natural disasters and agro terrorism. Diseases such as avian flu, hoof and mouth, and transmissible spongiform encephalopathy (TSE) also known as bovine spongiform encephalopathy (BSE) or mad cow in cattle, chronic wasting disease in deer and elk and Creutzfeldt-Jakob disease in humans can wreck havoc on entire populations of livestock and wildlife.

A disease outbreak or natural disaster, depending on the location, can cause a large scale catastrophic mortality event. In the case of a disease outbreak, animals within a certain radius would be culled and disposed (SAF 2005). There are many methods currently in place to dispose of mortalities that occur on a regular basis. These include incineration, composting, rendering, and burial (Engel et. al. 2004; SAF 2005). In the case of a large scale mortality event, each of these methods has its advantages and disadvantages. When dealing with carcass disposal due to a disease outbreak, many other factors such as disease transmission and public health, have to be taken into consideration. In doing so, this may eliminate the use of some of the existing disposal practices.

One common method of disposal is incineration, a practice that has been in common use for hundreds of years (Kastner and Phebus 2004). There are three main types of incineration techniques that have been employed to dispose of animal carcass. These include open-air burning, fixed facility incineration and air curtain incineration (Kastner and Phebus 2004). Each technique uses burning techniques to destroy the carcass and poses a fire threat and creates ash by-product. Incineration is a disposal method not suitable for TSE's due to the stability of

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the prion at temperatures less than 850°C (SSC 2003a).

Composting animal carcasses is a common practice by livestock producers to dispose of their routine mortalities. Carcass composting is relatively less capital intensive than incineration and rendering, is a better alternative to burial in areas with shallow water tables, and provides for quick removal and isolation of farm mortalities (Kalbasi et al. 2005). Composting a carcass involves placing it into a carbon rich environment. Decomposition of the animal in the environment by microorganisms produces water vapor, heat, carbon dioxide, and organic residue (compost). The compost can then be applied to agricultural land as a soil amendment. Composting animal carcasses can take 1 to 9 months before usable humus is ready. During the 2004 outbreak of avian influenza in British Columbia, composting was used to dispose of uninfected culled birds, while infected birds were subjected to biological heat treatment and then composted (Engel et al. 2004). Problems associated with using composting for a mass mortality event include management of the compost area, leachate management, maintaining proper temperature and moisture content, maintaining proper aeration, creating an adequate carbon to nitrogen ratio and final disposal of compost (Fonstad et al. 2003). Compost also does not reach temperatures greater than 70°C, therefore it is not suitable for inactivation of many diseases such as TSE's.

Rendering livestock mortalities is the preferred method of disposal (SAF 2005). This method involves recycling carcasses into useful by-products such as meat and bone meal and tallow. In Western Canada, rendering plants are sparse and travel distances to the facilities make it less economical than other disposal options. Trucks are the main method of transporting carcasses to the facilities. The trucks make many stops along the way; therefore they have the potential to spread disease along their travels. In the event of a disease outbreak, rendering may no longer be an option for most producers due to the risk of contamination by the trucks. Rendering will destroy most pathogens due to the temperatures (130°C) achieved during the process; although some diseases may not be destroyed by this process such as TSE's which may then deem rendering unacceptable (SAF 2005).

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Burial is the specific study of this thesis. Livestock mortality burial is a practice that has been used at both small and large scales (Engel et al. 2004). Small scale on-farm burial of normally occurring mortalities is a commonly used method in Saskatchewan (SAF 2005). Requirements for burial pits include: the location of burial in clay or till soils, in areas not subjected to flooding, and at least 2-4 meters between the bottom of the burial pit and a usable water source (SAF 2005). In the event of a disease outbreak or natural disaster, when large quantities of carcasses need to be disposed of, burial has in the past been deemed a reasonable option depending on site conditions (Engel et al. 2004). During the 2001 foot and mouth outbreak in the UK, large scale burial sites were constructed to contain hundreds of thousands of mortalities (MacArthur et al. 2002). Livestock mortality burial is a subject that has not been studied in great detail (CFIA 2006; Glanville 2000; Ritter and Chirnside 1995). With very few studies having been performed, it is concluded that there is a lack of scientific information on the subject (Freedman 2003). Of particular importance is the lack of knowledge of the leachate characteristics that develops in the burial pits from the decomposing animals. It is important to know these characteristics to understand the potential impact of leachate on the surrounding groundwater resources.

Characterization of the chemistry of livestock mortality leachate resulting from burial of livestock mortalities will provide scientists and regulators with the information required to perform risk analysis when considering mortality burial as a management option, either routinely or during a mass mortality situation. Toward the goal of risk analysis, this study also used two contaminant transport software packages to evaluate the potential risk to groundwater posed by the leachate. The objectives of the research are to:

1. Characterize the chemical composition of livestock mortality leachate for three species: bovine, swine and poultry

2. Evaluate the potential environmental impact of livestock burial through groundwater transport modelling.

These objectives were addressed specifically through field experiments and computer models.

2   LITERATURE REVIEW

To achieve the objectives of this study, the assessment of many factors are brought forth. Livestock mortality burial is a common practice among producers for the disposal of normally occurring dead stock. During the event of a mass mortality situation from a disease outbreak, ventilation failure, or other cause of mass mortality, burial has also been deemed a suitable option, however, has not been studied in great detail. To understand the potential threat a mass mortality situation causes, historical mass mortality events are reviewed for livestock as well as human graves from natural disasters and/or war. To then determine the effect on the environment, contaminant transport processes and the evaluation of existing studies involving human cemeteries and mass graves are reviewed as well as current techniques and regulations for livestock mortality burial.

2.1   History of Foreign Disease Outbreaks in Animals

Disease outbreaks among livestock are a major concern to livestock producers. Depending on the disease outbreak, anywhere from a few animals to millions of animals could potentially be culled and disposed. Foot and mouth disease (FMD), bovine spongiform encephalopathy (BSE) and avian flu are among the most devastating to livestock populations, and are among the most documented.

FMD is a highly contagious viral disease that can infect all bovids with cloven hooves (Segarra and Rawson 2001). FMD causes high fevers and blisters of the mouth and feet. Transmission can occur through direct contact with infected animals, contaminated pens or transportation vehicles therefore creating a highly transmissible disease. In the United States, nine FMD outbreaks have occurred between 1870 and 1929. The outbreak of 1924 culled the most animals (170,000). The last outbreak occurred in 1929 (USDA 1998). The United Kingdom has had two major FMD outbreaks in the last fifty years. An outbreak in 1967 required the culling of 442,000 animals and the outbreak of 2001 culled 1,281,271 livestock including cattle, sheep,

pigs and goats (Scudamore et al. 2002). These outbreaks are discussed in greater detail in Section 2.2. The U.K. also had a few localized outbreaks in 2007 resulting in culled herds. In 1997, Taiwan culled 3.8 million swine in their first outbreak since 1929. These swine were disposed of through incineration and burial (USDA 1998). China reported their first outbreak on record in 2005 and has had ongoing outbreaks since (Xinhua 2006). Canada's last FMD outbreak occurred in Saskatchewan in 1952 and has since been eradicated (AARD 2008).

BSE is another, well documented, fatal livestock disease causing neurodegeneration. An outbreak of BSE results in culling of large quantities of animals. In North America, the first recorded BSE case occurred on May 20, 2003 in Alberta and the second in Washington State in December 2003 (Jin et al. 2004). The United Kingdom is the most affected country with more than 179,000 infected cattle recorded, causing the slaughter of approximately 4.4 million cattle during the eight month eradication period between February and September 2001 (DEFRA 2008). In Europe, the Department for Environment Food and Rural Affairs (DEFRA) manages disease control and eradication. Their standards require that the offspring of female BSE cases born within two years of the clinical onset of the disease be culled. They also require the cull of all cohorts which include those animals born in the same herd within a year before or after the BSE case, or those reared with a BSE case (DEFRA 2008). If another epidemic of BSE were to occur, the potential cull of thousands of animals would be required with burial as the preferred disposal method.

Avian influenza outbreaks occur in many regions of the world hundreds of times per year with over 220 million birds affected since 2003 (WHO 2008). North America has had very few outbreaks. The most recent case in North America occurred in Regina, SK causing the death and cull of 48,560 head of poultry on the infected farm and the surrounding 10 km region perimeter (WOAH 2008). Burial was used to dispose of all culled poultry as well as all manure and bedding in the barns. Another major case in the Fraser Valley of British Columbia in the spring of 2004 led to the slaughter of 17 million birds. One major case of avian flu occurred in Vietnam in December 2003 (WHO 2008). A lethal strain H5N1 caused the cull of 50.34 million

poultry. The strain H5N1 is transmissible to humans and has killed over 200 humans since 1997 (WHO 2008). Millions of birds are culled each year throughout the world due to exposure to avian influenza. Documentation of disposal techniques used to dispose of the birds is difficult to find. Due to the high rate of exposure to birds and easy transmission to humans, a severe pandemic is possible in the future.

2.2   Previous Studies Involving Livestock Burial

Livestock burial sites, whether for normally occurring mortalities, or those built for a mass mortality event, pose a threat of contamination to soil and groundwater. Little is known about the leachate composition resulting from a decomposing carcass and therefore little is known about the actual threat involved with a burial site (CFIA 2006). Studies have been performed using experimental burial sites and monitoring the groundwater quality around them (Ritter and Chirnside 1995; Glanville 2000; Environment Agency 2001; Scudamore et al. 2002; MacArthur and Milne 2002; MacArthur et al. 2002;).

Many early investigations focused on monitoring groundwater quality surrounding existing mortality disposal sites (Glanville 2000; Ritter and Chirnside 1995). These methods give a basic analysis of potential contaminant transport into groundwater systems. Most of these studies give a basis for a typical, on-farm, normally occurring mortality situation. Average livestock losses occurring on site are generally not considered a threat in most provinces and states; however, when dealing with a mass mortality event, the sheer number of culled animals to be disposed requires a more careful evaluation of disposal sites and the risks involved (Engel et al 2004; SAF 2005).

2.2.1   Livestock Burial Effect on Shallow Groundwater Quality

The Iowa Department of Natural Resources, working with Dr. Thomas Glanville at the Iowa State University performed two case studies to evaluate impacts in regards to the environment when dealing with livestock burial (Glanville 2000). The results of these studies concluded that the proper disposal of livestock mortalities can be more difficult than manure management due to the rapid breakdown of the animal carcass in the environment. Because it is difficult to store

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carcasses for any long period of time, difficulties arise with preventing disease transmission by rodents, insects and predators and therefore, the animal must be dealt with in a timely fashion to minimize the impact to the environment.

The first case study (Glanville 2000) investigated turkey mortalities located in a poorly drained soil with seasonal high water tables between 0.3 to 0.9 meters. Two pits were constructed one year prior to the study and contained 28,400 kg of turkey mortalities. The bases of the pits were at a depth of 1.8 m. To monitor groundwater quality, 12 monitoring wells were placed within 30 m of the pit, with 9 others within 10 m. These wells were used to monitor changes in groundwater chemistry with time. Groundwater samples were collected monthly for the first 15 months and again at 20 months and 40 months.

Glanville (2000) also studied burial trenches located in well drained, moderately permeable soil. Two trenches were excavated to 1.2 m deep and spaced 2.4 m apart. Six swine carcasses spaced evenly filled each trench. To obtain leachate samples, one of the trenches was lined with PVC sheeting and 100 mm of pea gravel for a lysimeter installation, the other trench was not lined. Monitoring wells were placed within three meters around the trenches to monitor groundwater contamination (Glanville 2000). Total leachate was pumped monthly from a PVC pipe buried vertically at one end of the trench lysimeter. The leachate was measured to examine the mass, concentration, and duration of decay products. Leachate collected was exposed to rainwater during the duration of the experiment and therefore is not a pure leachate example.

For both Glanville (2000) case studies, elevated levels of biochemical oxygen demand (BOD) at 230 mg/L, ammonia-nitrogen (NH_4-N) at 403 mg/L, total dissolved solids (TDS) at 1527 mg/L, and chloride (Cl) at 109 mg/L were found within the burial site and at a distance of 1-2 m downgradient. Results from the lysimeter observed an ammonia concentration of 416 mg/L, chloride of 23 mg/L and 11 mg/L of nitrate. Although chloride concentrations were generally lower than the other contaminants, elevated chloride levels are reportedly the best indicator of

burial-related groundwater contamination (Engel et. al. 2004). Neither case study demonstrated contamination more than a meter or two from the site within the sampling period, but this situation could easily be changed in areas with high water tables and high groundwater velocities. Glanville (2000) concluded that localized contamination may persist for a decade or more in wet soil with a high seasonal water table and low groundwater flow velocity and even in lightly loaded burial trenches constructed in well drained soil, complete decay may take two years or more.

2.2.2   Impact of Poultry Mortality Disposal on Groundwater Quality

To examine the impact of poultry mortality disposal on groundwater quality, Ritter and Chirnside (1995) monitored six existing poultry disposal pits in Delaware. The pits were still active, with approximately 15-25 kg of dead birds added each week. To monitor the potential groundwater contamination, two to three monitoring wells were placed around the pits at a depth of 4.5 m. These pits were located in areas of sandy loam soils and very high water tables, meaning the bottoms of many of the disposal pits were located below the groundwater table for most or all of the year. By using this method of disposal, the potential for groundwater contamination was relatively high. Monitoring of these pits occurred from 1987 to 1990 and samples were drawn every 4-8 weeks and analyzed for ammonia, nitrates, fecal coliforms and fecal streptococci.

This study concluded that ammonia (NH_3-N) concentrations were high in two wells. Of the six pits, three of them demonstrated an increase in ammonia concentrations at a lateral distance of 12 m. In some cases, the ammonia concentrations reached 366 mg/L and streptococci concentrations reached 209 mg/100 mL. In all monitoring wells, fecal coliform and fecal streptococcus concentrations were low and only detected 20% of the time during sampling. Bacterial contamination of groundwater by the disposal pits was reportedly low. They concluded that more groundwater contamination can occur in the event of a large mortality event due to higher mass loading of the burial system.

2.2.3 Mass Burial Sites for Foot and Mouth Disease Outbreak in Scotland

The United Kingdom has experienced two major foot and mouth disease outbreaks, the first in 1967 and the second in 2001. Scudamore et al. (2002) pointed out that the outbreak of 2001 was considered significant due to many reasons. In 1967, cattle populations were similar to that in 2001 at approximately 11-12 million, but sheep populations had increased from around 28 million in 1967 to 40 million in 2001. In 1967, only infected animals and their immediate contacts were culled, unlike in 2001 when all animals in the infected zones were culled. The outbreak of 2001 caused a cull of 1,281,271 livestock including cattle, sheep, pigs and goats. Disposal operations varied by location and included on-farm burial, on-farm pyres, rendering, incineration, mass burial sites and landfills. On-farm burial was limited in many areas due to the requirement of a Groundwater Authorization by the Environment Agency and the need to protect the environment (Environment Agency 2001). Because of the Groundwater Authorization, many sites were found not suitable for burial due to their close proximity to water resources or location in a flood plain (Scudamore et al. 2002).

Within a few weeks of the outbreak, other disposal options such as incineration and rendering soon met their limits and the need for mass burial locations became apparent. At the early stages of designing the mass burial sites, rapid development of designs evolved and changed as construction took place. At the early stages, the pits were a simple hole in the ground and by the end, pits were engineered with sophisticated liners and leachate collection systems which attempted to minimize the risk to groundwater (Scudamore et al. 2002). A total of seven sites were granted a Groundwater Authorization and became mass disposal sites. These included 5 sites in England, one in Scotland and one in Wales, with each site capable of multiple pits, each pit able to hold between 10,000 and 60,000 carcasses.

The most documented and used mass burial site was Birkshaw Forest located near Lockerbie, Scotland (MacArthur and Milne 2002). Approximately 440,000 culled animals were disposed at the site in six disposal pits. Three of the first pits constructed were unlined due to time constraints, but the remaining three were constructed with geo-synthetic clay liners (GCL) and

all pits possessed a leachate collection system (MacArthur and Milne 2002). The burial sites were commissioned in April 2001 and not capped until June 2001. During the first two months of pit filling, rainwater was allowed to enter the pits. To keep pollution at a minimum, leachate was collected and sent to a treatment plant.

MacArthur and Milne (2002) report that due to its high concentration of ammonia and the possibility of it containing the FMD virus, leachate had to be collected and processed. Very soon after disposal, large quantities of relatively high concentrated leachate were generated due to the release of body fluids and rain entering the pits. Animal carcasses consist of proteins, fats and initially, urea. Proteins have a high nitrogen content of approximately 16%, and therefore it was anticipated that there would be a high ammonia-N concentration in the leachate. Initially after one week, the chemical oxygen demand (COD) values were approximately 160,000 mg/L and rapidly fell to approximately 20,000-40,000 mg/L with the ammonia-N rising from 1000-2000 mg/L to 2000-4000 mg/L (MacArthur et al. 2002). Sampling the leachate of the pits was very inconsistent due to exposure to heavy rains. Once capping was complete on each site, a more extensive leachate analysis was completed involving sampling monthly from July 2001 to April 2003 (MacArthur et al. 2002). The results of this leachate analysis can be seen in Table 2.1. While the mean concentrations are relative and somewhat comparable to other leachates, they are diluted by rainwater. Maximum values may be more indicative of the true leachate chemistry.

Surface and groundwater chemistry was monitored closely around all disposal sites. Impacts to streams and rivers were reported at 212 total incidents resulting from leachate spills and contamination along disposal routes, with only one incident of a category 1 which cause major damage to the aquatic ecosystem and 11 category 2 incidents which cause significant damage to the aquatic ecosystem (Environment Agency 2001). Mass burial sites showed very few incidents. Groundwater contamination is continuing to be monitored through boreholes at each site. These boreholes are located at varying distances and depths from the burial sites. No data is currently published from the borehole analysis. According to the Environment

Agency (2001), water companies have found no contamination of public water supplies.

Table 2.1 Leachate chemistry from burial sites in Scotland (MacArthur et al. 2002)

Determinand	# of Samples	Maxima	Minima	Mean	Median	Standard Deviation
pH	199	8	5.9	6.9	6.9	-
Calcium (mg/L)	194	700	2	208	183	132
Iron (mg/L)	193	335	0	52	33	53
Phosphate as P (mg/L)	198	476	1	55	25	77
Ammonia NH3-N (mg/L)	199	19200	28	3294	2700	2702
Nitrate/Nitrite (mg/L)	9	10	0.2	2.1	0.6	3.2
BOD (mg/L)	9	38500	300	12700	11600	12875
COD (mg/L)	199	134200	500	20414	16000	20216
Electrical Conductivity µS	194	45000	5	11210	10985	7435
Alkalinity as CaCO3 (mg/L)	199	88200	152	11935	9400	10233
Suspended Solids (mg/L)	199	5432	10	389	260	486
Dry Residue (mg/L)	108	82400	470	6866	4960	8498

2.2.4   Policy Affecting Livestock Mortality Burial

Currently the Canadian Food Inspection Agency (CFIA) regulates burial site selection. The standards they provide may be superceded by provincial or regional standards. Consultation is required with local authorities before commencement of the burial site. The following sites are excluded for burial: flood prone areas, steep slopes, gravel pit or quarry, and any bedrock outcrops (CFIA 2006). Target soil requirements include a hydraulic conductivity less than $1x10^{-9}$ m/s for a 10 m thick aquitard; 1 m minimum depth from the bottom of the pit to groundwater; 100 m setback from wells, waterways, ponds and creeks; 100 m setback from secondary highways, residences and livestock facilities; and 1 m minimum fill required on top of carcasses (CFIA 2006). In the event of a mass mortality situation in Saskatchewan, Saskatchewan Agriculture and Food (SAF) advised trenches will be used for disposal. Trench dimensions will be approximately 2 m wide by 4 m deep at a length suitable for disposal of all dead stock at the site (Jansen 2006). It may be necessary to construct more than one trench per site. In the event of a mass mortality situation, government officials deem it necessary to use readily available excavation equipment such as backhoes to construct burial sites. Problems arising from using a pit set-up include capping an unstable semi-liquid mass, or constructing a pit too large to fill properly. Logistically, the easiest disposal method would be a trench that on-farm or local excavation equipment could handle. The affect of these trenches to soil and groundwater is

unknown.

2.2.5   Conclusion of Livestock Burial Research to Date

The literature revolving around existing disposal pits does not provide evidence of initial mortality leachate chemistry. Transport of leachate has been documented in studies by Thomas Glanville (2000) and Ritter & Chirnside (1995). It is important to understand the leachate chemistry resulting from burial. By understanding this leachate, proper burial guidelines can be determined.

2.3   Studies from Mass Grave Sites and Cemeteries

For centuries, burial has been a common practice for the disposal of human remains. Whether they occurred from a natural disaster, war, or natural death, cemeteries and mass grave sites exist throughout the world. It can be said that cemeteries are a special kind of landfill with very few research studies pertaining to the hydrogeochemistry of the decay of human remains (Knight and Dent 1995). It is known that human remains decay quickly and liquefy because of their high water content of about 70-75%. This liquification rapidly releases Na^+, K^+, Cl^-, HCO_3^-, NO_3^-, PO_4^{3-}, NH_4^+ and SO_4^{2-} ions (Knight and Dent 1995). As soon as the body liquefies, and enters the soil and groundwater system, it is then affected by factors that affect contaminant transport.

In a study of the Botany Cemetery in Australia (Knight and Dent 1995), salinity was increased close to recent graves and diminished down gradient. As well, concentrations of chloride, nitrate, nitrite, ammonium, orthophosphate, iron, sodium, magnesium and potassium were higher in groundwater near recent graves compared to background waters. The World Health Organization has compiled a list of cemeteries in which groundwater composition has been observed (WHO 1998). A cemetery in Germany showed high concentrations of bacteria, ammonium and nitrate ions in a plume within the cemetery, and their respective concentrations diminished within a short distance of the grave. In Holland another report of a saline plume (2300 uS/cm) consisting of chloride, sulphate and bicarbonate existed beneath the graves (WHO 1998).

Mass fatalities due to natural disasters or during war periods tend to be managed as a mass grave situation (Morgan et al. 2006; Sumathipala et al 2006). The south Asian tsunami disaster in 2004 was one of the largest fatal natural disasters of recent times. In Thailand, Sri Lanka and Indonesia alone, over 200,000 deaths occurred (Morgan et al. 2006). Lambarro, Indonesia contains the largest mass grave holding over 70,000 victims. Most common graves were constructed in existing cemeteries. Both Morgan (2004) and Sumathipala (2006) report that understanding of the effects of location and burial in communal graves and their effect on soil and water are unknown.

2.4   Biochemical Composition of Mammalian Cells

To determine the chemical composition of livestock mortality leachate, it is important to understand the chemical composition of mammals, specifically humans since they are most documented and similar to swine in many ways. Many elements are important constituents for the building blocks of life. Carbon, hydrogen, oxygen and nitrogen are the most essential elements found in living organisms. Out of every million atoms in the body, 993,000 are made up of the major building block elements; carbon, hydrogen, oxygen and nitrogen (Moore et al. 2002). Table 2.2 shows the abundance of elements found in the human body in atoms per million atoms. The table includes the building block elements along with all other major minerals found in the body. There are also trace elements found in the body that only occur in minute amounts. Human biology is not much different than other mammals with swine being the most similar physiologically and biologically (Miller and Ullrey 1987). Because of this similarity, livestock mortality characterizations would be expected to resemble human characteristics.

Hydrogen is one of the four main building blocks of all cells. It is mostly present in the body in the form of water. Water (H_2O) makes up approximately 70% of the body. H_2 can also be found in small amounts in cells.

Oxygen atoms are present in water. O_2 gas is also important during respiration for living

organisms. Oxygen is a component of major biomolecules that make up the body in structures such as carbohydrates, proteins, fats.

Table 2.2 Elements found in the human body (from Moore et al. 2002).

Element	Atoms/Million Atoms in the body	Relative % of Body Weight
Hydrogen	630000	63%
Oxygen	255000	25.50%
Carbon	94500	9.50%
Nitrogen	13500	1.30%
Calcium	3100	0.31%
Phosphorus	2200	0.22%
Chlorine	570	0.06%
Sulphur	490	0.05%
Sodium	410	0.04%
Potassium	260	0.03%
Magnesium	130	0.01%

Carbon is defined as the building block of life. It is found in many forms in the body and helps compose many structures responsible for existence. Carbon is mostly found in the form of bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}).

Nitrogen is found in the body in many forms. One of the most common is cyclic nitrogen compound forming a DNA strand along with a sugar unit and phosphate (Moore et al. 2002). Nitrogen is also a key component in the structure of amino acids and nucleic acids that form proteins in the body. Crude protein makes up approximately 15% of an adult human body (Mitchell et al. 1945) and would be expected for livestock as well. The amino group in most proteins is formed with an H_2N compound (Mitchell et al. 1945). Nitrogen also helps form the peptide linkages of amino acid groups covalently bonded to each other (Moore et al. 2002).

Of the remaining elements found in the body, calcium and phosphorus make up approximately 2.2% of the total body mass. Through various research, it has been determined that calcium makes up an estimated 1.3 to 3.6% of the body and phosphorus 0.63 to 1.2% (Mitchell et al. 1945). The skeleton and teeth comprise 99% of the total calcium and four fifths the total

phosphorus in the body (Sherman et al. 1926). The three main ionic forms of phosphorus found in the body include PO_4^{3-}, HPO_4^{2-} and $H_2PO_4^-$ (Moore et al. 2002). The phosphate ion is the main building block of bones and teeth based on calcium hydroxyphosphate – apatite, for adenosine triphosphate (ATP) and for the phosphate backbone of RNA and DNA as well as other biochemical molecules (Childs 2001). Calcium helps form stable bone material, is a key factor in the formation of cell walls and is important for blood clotting.

Other elements such as potassium, chlorine, sodium, sulphur and iron all have important functions in the body. Potassium is the main cation inside cells and is essential for heart and nerve functions. Sodium is the main extracellular cation and is also important for nerve function. Chlorine is mainly present in the form of hydrochloric acid in the stomach for digestion. Sulphur is present in fats, body fluids, skeletal minerals and is also a key component in determining a protein's tertiary structure (Moore et al. 2002). Iron is the key component of aerobic activity inside the body and is present in relatively small amounts. Iron is located in the haemoglobin and is responsible for carrying oxygen to the body.

Along with the major and minor elements in the body, small amounts of trace elements such as copper, cobalt, mercury, arsenic, etc. also exist. Some of them have a small function in the body; others have not yet had a function discovered. These elements are present in amounts so small, that for mortality leachate, it is expected that they will not be detectable. This could change if an animal's food source has been treated with trace elements to provide them with essential nutrients.

2.5   Groundwater Pollutants and Standards

To evaluate leachate characteristics, it is important to list current regulations regarding quantities of constituents allowed in drinking water. The quality of potable water is determined by the amount of chemicals present in the water, with some being more toxic than others. The depth of soils and subsurface geology through which the water passes through has the greatest effect on the chemicals dissolved in natural groundwater. Table 2.3 shows water quality standards that are not to be exceeded in Canada. To assess the potential risks to

groundwater, Table 2.4 shows a comparison of leachates from four studies; two landfill leachate characterizations (Freeze and Cherry 1979; Thornton et al. 2001), an average concentration of liquid swine manure from (Fonstad 2004), and average mortality leachate values as well as maximum values taken from the mass disposal site in Scotland (Table 2.1) (MacArthur et al. 2002). Table 2.4 shows relatively high concentrations in the leachate that, when allowed to enter groundwater systems, could deem the water unusable for drinking, irrigation and other processes.

Table 2.3 Drinking water quality standards (Environment Canada 2004).

Dissolved inorganic compounds		
Alkalinity	500 (SK)	mg/L
Aluminum	0.2 (WHO)	mg/L
Ammonia	50 (EU)	mg/L
Arsenic	0.025	mg/L
Boron	5	mg/L
Cadmium	0.005	mg/L
Calcium	100 (EU)	mg/L
Chloride	250	mg/L
Chlorine (total residual in chlorine-disinfected water)	>0.5	mg/L
Chlorine (free residual)	>0.1	mg/L
Copper	1	mg/L
Fluoride	1.5	mg/L
Iron	0.3	mg/L
Hardness	800 (SK)	mg/L
Lead	0.01	mg/L
Magnesium	200	mg/L
Manganese	0.05	mg/L
Mercury	0.001	mg/L
Nitrate (as nitrate-nitrogen)	10	mg/L
Nitrate (as nitrate)	45	mg/L
Selenium	0.01	mg/L
Sodium	200 (SK 300)	mg/L
Sulphate	500	mg/L
Sulphide	0.05	mg/L
Total dissolved solids (indicator of the amount of salt in solution)	500 (SK 1,500)	mg/L
Zinc	5	mg/L

Note: (EU) signifies European Union Standards, (SK) represents Saskatchewan Standards, (WHO) represents World Health Organization Standards

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Table 2.4 Landfill and earthen manure storage leachate characteristics.

Parameter	unit	Landfill	Landfill	Manure Storage [a]	Mortality Leachate [b]	Mortality Leachate [c]	DW Standards
		Freeze & Cherry (1979)	Thornton et al. (2001)	Fonstad (2004)	MacArthur et al. (2002)	MacArthur et al. (2002)	Environment Canada (2004)
pH		4-8	7.2	7.6	7	8	-
Alkalinity	mg/L	500-10000	4600	11700	11900	88200	500
Chloride	mg/L	300-3000	1965	13500	-	-	250
Sulphate	mg/L	10-1000	52	700	-	-	500
Calcium	mg/L	100-3000	125	190	200	700	100
Magnesium	mg/L	100-1500	135	96.0	-	-	200
Sodium	mg/L	200-1200	1340	730	-	-	200-300
Potassium	mg/L	200-1000	490	2020	-	-	-
Ammonium-N	mg/L	10-1000	1030	3880	3300	19200	10
Nitrate-N	mg/L	0.1-10	-	3.6	2	10	10
Iron	mg/L	1-1000	14.0	3.30	52	335	0.3
Phosphorus	mg/L	-	-	114	55	476	-

[a]: average concentrations from 7 liquid swine manure storages; [b]: average mortality leachate
[c] maximum mortality leachate

High concentrations of alkalinity can be detrimental to the environment in many ways and also cause unpleasant taste in drinking water. With regards to the contaminants listed in Table 2.4, average alkalinity from mortality leachate is 23 times higher than regulations for drinking water standards while maximum values are 176 times higher (Table 2.3).

Chloride is a conservative ion, meaning that it has no significant oxidation/reduction reactions, does not form important ion complexes and is not significantly absorbed onto mineral surfaces (Hem 1992). Excess chloride in water can cause confusion, weakness and coma when ingested (US EPA 1976). Chloride concentrations exceeding 250 mg/l can cause foul taste and corrosion in water distribution systems due to the ions ability to form soluble salts such as NaCl and KCl (Health Canada 1987).

Sulphate is one of the least toxic anions to humans. Taste thresholds for sulphate begin at around 250 mg/l. Excess sulphate can interfere with disinfection processes as it binds with residual chlorine and can also cause catharsis and gastrointestinal irritation (Health Canada 1994). Sulphate in solution also has the ability to form complex species with preference to divalent or trivalent cations, typically forming $NaSO_4^-$ and $CaSO_4^-$ (Hem 1992).

Calcium is an abundant element in the body and regulates the adsorption of most other elements in the body. Calcium is also the determining factor in water hardness with soft water having less than 75 mg/l as $CaCO_3$ to hard water having more than 300 mg/l as $CaCO_3$ (Health Canada 1987-2). Calcium has only one oxidation state Ca^{2+}. Common ion pairs involving calcium include calcium-phosphate and calcium-carbonate; while calcium is also a strong participant in cation exchange on mineral surfaces (Hem 1992). Concentrations of calcium from animal burial listed in Table 2.4 exceed standards for drinking water by two times while the maximum values are 7 times higher.

Iron and magnesium are not considered significantly toxic, but can cause distaste, stain plumbing fixtures and form encrustations on well screens reducing the efficiency of the well (Moody 1990). Iron is most commonly found in water in its ferrous form, Fe^{2+}, but depending on its oxidation state, iron can be present in other forms. Magnesium is commonly present in water in the Mg^{2+} state and has a similar effect in solution as calcium. Magnesium can form complex ions and has again about the same stability in these ions as the calcium equivalents (Hem 1992). Ferrous iron also has the ability to form complexes with organic molecules which are important to photosynthesis and the function of haemoglobin in the blood (Hem 1992). Average iron concentrations from the Scotland mortality leachate in Table 2.4 exceed drinking water standards by 173 times, while the maximum concentration exceeding drinking water standards by 1,117 times.

Sodium is another commonly occurring element and is usually found in the form of sodium chloride. Sodium is a monovalent cation and does not participate in redox reactions and in concentrated solutions, tends to form ion pairs with carbonates and sulphates (Hem 1992). Excess concentrations of sodium in the human body have been known to cause high blood pressure, renal failure and death. Symptoms of sodium poisoning include increased sensitivity, twitching, tremors, oedema and stupor (Health Canada 1979).

Elementary potassium is rare because of potassium's reactive power. Potassium is mainly found in the form of potassium chloride, also known as potash, in which ingestion of 15 grams, is lethal for a human adult (US EPA 1976). Potassium is weakly hazardous in water, but it does spread rapidly, because of its relatively high mobility and low transformation potential; it is also available as an exchangeable cation to exchange sites on soil particles (US EPA 1976). Potassium can also incorporate into clay mineral structures in between crystal layers and once there, cannot be removed by ion exchange (Hem 1992).

Phosphorus is not considered harmful in drinking water, but can cause detrimental effects to the environment when in excess (Freeze and Cherry 1979). Currently no drinking water standard exists for phosphorus content, but the National Tap Water Quality (2008) suggests concentrations exceeding 0.5 μg/L can cause carcinogenic effects in a lifetime of exposure. Average phosphorus in mortality leachate from Table 2.4 exceeds drinking water standards by 110 times, while maximum concentrations exceed the value by 952 times. Small amounts of phosphorous, when added to surface water contributes to accelerated algae and aquatic vegetation growth. Compounds containing phosphorus in groundwater usually possess a negative charge, therefore making them highly mobile and less susceptible to attenuation (Freeze and Cherry 1979). Due to the high mobility, even small amounts of phosphorus allowed to enter a groundwater system that discharges to the surface can have a large impact on the aquatic environment.

Phosphate (PO_4^{-2}), a common form of phosphorus, will strongly bond to low concentrations of oxides Fe, Al and Ca and precipitate from solution, therefore reducing the amount of phosphorus available for transport (Stollenwerk 1996; Weiskel and Howes 1992). Studies by Zanini et al. (1998), Stollenwerk (1996), and Weiskel and Howes (1992) have concluded that in oxic conditions, phosphate transport should not only consider sorption capacity of the soil, but the capacity of formation of phosphate minerals such as varisite ($AlPO_4^.2H_2O$), vivianite ($Fe_3(PO_4)_2^.8H_2O$), rockbridgite ($Fe_5(PO_4)_3(OH)_5$), hydroxyapatite ($Ca_5(PO_4)_3(OH)$), and strengite ($FePO_4^.2H_2O$) among others to precipitate phosphate.

The contaminant of most concern is nitrogen. Relatively high levels of ammonium are shown for all three contaminant sources in Table 2.4 and average and maximum mortality leachate concentrations are 330 to 1,920 times higher in concentration than drinking water standards. The production of ammonium usually occurs from deamination of amino acids during organic waste decomposition (Tatsi and Zouboulis 2002). Nitrate (NO_3^-) is the most stable of all nitrogenous materials; most nitrogenous materials tend to be converted to nitrate in aerobic environments, therefore deeming all sources of nitrogen as potential nitrates. Ammonium can be oxidized to nitrate by chemoautotrophic bacteria when oxygen is present and heterotrophic bacteria when organic carbon are present. In the studies involving livestock manure (Fonstad 2004), large amounts of organic carbon are present, and therefore the oxidation of ammonium will be heterotrophic if oxygen is present in the form of CO_2. The production of nitrate from ammonium showing intermediate states is as follows (Paul and Clark 1996):

$$NH4^+ \rightarrow NH_2OH \rightarrow NOH \rightarrow NO2^- \rightarrow NO3^-$$

Many factors control the rate of nitrification such as pH, aeration, moisture and temperature (Paul and Clark 1996). Nitrification rates tend to decrease in soils with pH of less than 6.0, where high pH values of 8.0 and upwards hinder the conversion of nitrite to nitrate (Paul and Clark 1996). Due to the biological nature of nitrification, temperature is important with an optimum range of 30 to 35°C with slowing of the process occurring below 5°C and above 40°C. Aeration is important due to the bacteria needing O_2. Without proper diffusion of O_2 into the process, nitrification is slowed (Paul and Clark 1996). All of these factors affect the speed in which ammonium will convert to nitrate; the transport of nitrogen from a burial site into groundwater systems is of most concern because of its potential high mobility in groundwater.

When nitrates are ingested into a mammalian body, they are easily reduced to nitrite (NO_2^-) by bacteria and endogenous synthesis. Once reduced to nitrite, rapid absorption into the blood stream occurs, causing methaemoglobinaemia (Health Canada 1987-3). The process of methaemoglobinaemia results in the oxidation of Fe_2^+ in the haemoglobin to Fe_3^+ causing the haemoglobin to become methae-moglobin deeming it unable to release oxygen to the body's

tissues (Health Canada 1987-3). This process is clinically observable at 10% concentration of methae-moglobin by cyanosis. At concentrations of 80% or more methae-moglobin, asphyxia and death occur (Health Canada 1987-3). Infants are the most susceptible to methaemoglobinaemia because of their easily oxidised foetal haemoglobin. Because of this risk, the maximum acceptable concentration for nitrate in drinking water is 45 mg/l or 10 mg/L as nitrogen (Health Canada 1987-3).

2.6   Methods of Contaminant Transport

To analyze the transport of livestock mortality leachate, the methods of contaminant transport from the burial site into the groundwater system below should be discussed. Many factors affect the fate of the leachate in the soil water system. In general, transport of the contaminant is determined by the groundwater gradient, diffusion, dispersion, advection and preferential flow paths (Fetter 1999). Attenuation of the contaminant is important and is affected by ion exchange to solids in the system, adsorption, decay, precipitation and dissolution of the contaminant solution (Freeze and Cherry 1979). Chemical properties of the contaminant can be affected by solubility, vapour pressure, partition coefficient, volatilization, oxidation-reduction behaviour, radioactive decay and biodegradation.

2.6.1   Mass Transport without Attenuation

The fate of livestock burial leachate in the soil water system in respect to mass transport without considering attenuation is controlled by the following factors: diffusion, dispersion, advection, and preferential flow paths. Attenuation factors will be discussed in a later section.

2.6.1.1 Advection

Burial leachate carried along the flow paths with flowing water is termed advective transport. The flow of groundwater and the concentration of the solute control the amount of advective transport. To determine water flow, the average linear velocity (v_x) is used and can be described as the rate of water flux across a cross sectional area of pore space. This pore space is defined as effective porosity (n) which is the pore space water flows excluding noninterconnected and dead end pores (Fetter 1999). Average linear velocity is calculated as

follows:

$$v_{x=-\frac{K}{n}\frac{dh}{dl}}$$

[2.1]

where:

v_x = *average linear velocity (L/T)*

K = hydraulic conductivity (L/T)

n = porosity

dh/dl = hydraulic gradient (L/L)

To then calculate the amount of water flowing into a cross secitional area, flux (q):

q = v_xn

[2.2]

where:

q = flux of water (L/T)

v_x = *average linear velocity*

n = porosity

The change in solute concentration due to one dimensional advective transport with respect to time at any location is defined as (Fetter 1999):

$$\frac{\partial C}{\partial t} = -v_x \frac{\partial C}{\partial x}$$

[2.3]

where:

$v_x = average\ linear\ velocity\ (L/T)$
$dC/dx = concentration\ gradient\ (M/L)$

The method of advection will transport solutes at different rates throughout different types of porous media depending on the characteristics of the geologic materials in the flow path. It is difficult to use only the advection method to determine the resulting distribution of solute because other factors such as dispersion and diffusion aid the process.

2.6.1.2   Diffusion

Diffusion is one common way in which livestock burial leachate solutes could be transported through the unsaturated and saturated zone. Because of the relatively high concentrations expected from mortality leachate, diffusion would speed up transport. Diffusion is defined as solutes moving from an area of greater concentration toward an area of lower concentration (Freeze and Cherry 1979). Diffusion occurs for fluids in motion or for stagnant situations, as long as a concentration gradient exists. Diffusion is the predominant mechanism of transport in low permeability hydrogeologic regimes (Fetter 1999), but given the potential for burial leachate to have concentrations 20 to 30 times the regulatory limit, diffusion could be a significant transport mechanism.   Fick's first law describes diffusion in one dimension as (Appelo 1996):

$$F = -D\frac{dC}{dx}$$
[2.4]

where:

$F = mass\ flux\ of\ solute\ per\ unit\ area\ per\ unit\ time\ (ML^{-2}T^{-1})$
$D = diffusion\ coefficient\ (L^{2}T^{-1})$
$dC/dx = concentration\ gradient\ (ML^{-3}L^{-1})$

Fick's second law describes the diffusion situation where one dimensional concentrations are changing with time and is stated as (Fetter 1999):

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial x^2}$$
[2.5]

where:

$\partial C/\partial t$ is $(ML^{-3}T^{-1})$

Diffusion is also regulated by electrical neutrality of the ions involved with the diffusive process. For example, in a solution containing NaCl, unless the solution contains another negative ion in the region through which Na^+ is diffusing, Na^+ cannot diffuse faster than Cl^- (Fetter 1999). The

coefficient of molecular diffusion can be defined as the coefficient of molecular diffusion for the species in free water multiplied by the tortuosity factor. The tortuosity factor represents a reduction in diffusion due to the presence of a porous medium (Freeze and Cherry 1979).

2.6.1.3   Dispersion

Dispersion is similar to that of diffusion in that it can also be explained by Fick's first law due to the spreading of the concentration front. Mechanical dispersion causes the spreading of concentration by flowing water forced around sediments; in which there are two types of dispersion: longitudinal dispersion and transverse dispersion (Appelo and Postma 1996). Longitudinal dispersion, explains the travel time differences around sediment grains along the flowline. Transverse dispersion is a result of diffusion moving an ion to a different flowline. Dispersion coefficients are therefore a function of velocity and mixing length. Since diffusion cannot be separated from dispersion in groundwater flow, they are combined. Hydrodynamic dispersion coefficient (D_*) is a sum of the longitudinal or tansverse dispersion coefficient and diffusion coefficient. Dispersivity is impossible to measure in the field or laboratory without performing a tracer test. This tracer test can only measure what is observed at the macroscopic scale, but the result assumes that this encompasses what is happening on the microscopic scale defined by individual pore channels, and differences along the flow paths (Freeze and Cherry 1979). Gelhar et al. (1992) made dispersivity observations at 59 sites and estimates that dispersivity can be typically estimated as 1/10 to 1/100 of the travel distance.

2.6.1.4   Advection-Dispersion

Combining previous transport mechanisms, a one dimensional advection/dispersion equation can be formed to evaluate transport of livestock mortality leachate. The derivation of the advection/dispersion equation demonstrates the conservation of mass of solute flux into and out of a small representative volume in one dimension and is shown in Equation 6 (Fetter 1999). The advection/dispersion equation can be expanded to two or three dimensions.

$$\frac{\partial C}{\partial t} = D_* \frac{\partial^2 C}{\partial x^2} - v_x \frac{\partial C}{\partial x}$$

[2.6]

Hydrodynamic dispersion (D_*) will cause the concentration of the solute to decrease with distance from the source and spread will be greater in the direction of groundwater movement than in the direction perpendicular to flow (Fetter 1988). A continuous contaminant source will yield a plume, while a spill will yield a slug of contamination that expands with time as it moves along the flow path as would be expected from a mass mortality burial site. Defining mass transport by the advection/dispersion equation on the basis of D_*, it is possible to explain D_* as the summation of longitudinal dispersion and diffusion. In order to determine which is dominating, Peclet numbers can be calculated. The Peclet number relates mass transport by advection to mass transport by either diffusion or dispersion (Fetter 1999). The Peclet number is defined as:

$$Pe = \frac{v_x d}{D} \qquad\qquad [2.7]$$

where:

v_x = average linear velocity
D = diffusion coefficient in pure water
d = diameter of particles

At low Peclet numbers (less than one), diffusion dominates; at high Peclet numbers (greater than 10), advection dominates the transport process. Depending on flow beneath a burial site, advection or diffusion could potentially dominate. In the event of very low groundwater flows, diffusion will dominate and a plume will form due to high concentration gradients. In the event of higher groundwater velocities, diffusion will have a smaller effect, as advection/dispersion will dominate the transport process.

2.6.1.5 Preferential Flow Paths

Preferential flow paths are important in contaminant transport and should be carefully considered. Cracks, fractured soils, gaps, animal tunnels and holes are all considered preferential flow paths and each can form a rapid route for transport. When contaminants follow preferential flow paths, they are expedited through the path due to higher groundwater velocities along the path causing the contaminant to reach greater depths or horizontal

locations at a faster pace than a contaminant not exposed to a preferential flow path (Freeze and Cherry 1979). These paths still conform to the same processes as other sediments such as advection, mechanical dispersion, molecular diffusion and chemical reactions (Freeze and Cherry 1979). This is important in a livestock mass mortality burial event in Saskatchewan due to typical fractured tills and the trench disposal method provided by the governing body. It is important to determine the equivalent porous media for a site to determine the hydraulic properties present. Barbour (2000) summarized that in small fracture spacings, the concentration profile for transport is similar to that of unfractured porous media.

2.6.2   Attenuation of Contaminants

The fate of livestock burial leachate introduced into the soil is also controlled by the ion exchange capacity of the soil particles, and the soil's ability to adsorb ions, which is then controlled by the porosity of the soil and degree of saturation (Fetter 1999). The surfaces of many sediments have an electrical charge due to isomorphous replacement, broken bonds, or lattice defects (Wayman 1967). Due to this imbalanced electrical charge, they have the ability to adsorb a charged ion to balance the system. Clay minerals in particular have excess imbalanced negative charges in their crystal lattice, therefore they easily adsorb cations. Divalent cations are adsorbed more easily than monovalent cations (Fetter 1999). The soils ability to adsorb is determined by mineralogy, particle size, temperature, saturation, pH, Eh and activity of the ion (Fetter 1988).

The presence of clays increases the ion exchange capacity. The ion exchange capacity is the sum of cation exchange capacity (CEC) and anion exchange capacity (AEC). Although CEC is the most dominant in the uptake of ions, some colloids attract anions such as iron and aluminum oxides. Due to the structure of clay colloids, there is preference in the exchange of cations and that order is shown below (Fetter 1999).

$$Al^{3+} >> Ca^{2+} > Mg^{2+} >> NH_4^+ > K^+ > H_3O^+ > Na^+ > Li^+$$

However, if a soil is flooded with a solution containing a large concentration of one cation, the normal cation exchange order can be reversed (Fetter 1999). The order can also be changed in

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this regard by ionic strength of the solution and relative concentration of each ion (Fonstad 2004).

The amount of solute sorbed onto exchange sites is determined by the concentration of the solute. Sorption isotherms are plots of solute concentration versus amount sorbed onto the soil. Linear adsorption isotherms are commonly used because of their simplicity for mathematical manipulation in the advection/dispersion equation. The slope of the linear adsorption isotherm is known as the distribution coefficient (K_d). This distribution coefficient can then be used to later calculate a retardation factor for the soil.

The advection/dispersion equation can then be modified to include sorption occurring in the transport process (Fetter 1999; Appelo and Postma 1996).

$$D_* \frac{\partial^2 C}{\partial x^2} - v_x \frac{\partial C}{\partial x} = R_f \frac{\partial C}{\partial t}$$ [2.8]

where:

$C = concentration\ in\ solution$

$t = time$

$D_* = hydrodynamic\ dispersion\ (L)$

$v_x = average\ linear\ velocity\ (LT^{-1})$

$R_f = retardation\ factor$

2.6.3   Changes to Chemical Properties of the Contaminant

Chemical properties of the contaminant can be affected by solubility, vapour pressure, oxidation-reduction behaviour, and biodegradation. Once the substance is dissolved in water, its mobility can be determined and can be grouped as follows:

Table 2.5: Mobility of chemicals (Rail 1989).

Extremely Mobile	Chlorides & Sulfates
Somewhat Mobile	Calcium, Magnesium & Sodium
Generally Mobile	Ammonium-N, Silicon, Phosphorus & Potassium
Slightly Mobile	Iron, Chromium, Aluminum & Lead

In the saturated zone, groundwater is assessed as a partial equilibrium system (Fetter 1999). When a contaminant meets the groundwater system, chemical reactions will occur to try and reach equilibrium. Equilibrated reactions form an equilibrium constant (K). For the following reaction the equilibrium constant is found by:

$$aA + bB = cC + dD \qquad [2.9]$$
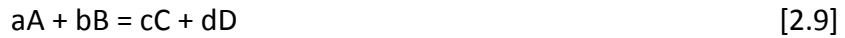
$$K = \frac{(C)^c (D)^d}{(A)^a (B)^b} \qquad [2.10]$$

where:

a, b, c, d = *respective number of moles involved*
A, B, C, D = *molar or molal concentrations*

The above equations are normally only valid for very dilute solutions. When dealing with non-dilute solutions such as livestock burial leachate from burial pits, ions interact electrostatically with each other. Once this begins to happen, an adjusted molar concentration to an effective concentration needs to be determined by using activities instead of concentrations for calculating equilibriums. There are many ways of solving for activity coefficients, including the Debije-Huckel equation for dilute solutions, the Extended Debije-Huckel equation that accounts for the effective size of hydrated ions, the Davis equation used for slightly more concentrated solutions, and the Pitzer equation for concentrated solutions (Appelo and Postma 1996). A simple rule to remember is that activities are usually smaller for multivalent ions than ions with a single charge.

To determine whether or not a mineral is in equilibrium with its solution, the saturation index can be determined. The saturation index is calculated using Eqn. (10).

$$SI = \log_{10}\left(\frac{IAP}{K}\right) \qquad [2.11]$$

where:

SI = saturation index

IAP = product of ion activities

K = equilibrium constant

For an equilibrium situation, the saturation index will be zero, for undersaturation less than zero and for supersaturation greater than zero. This determines whether an element has the potential to precipitate or dissolve in solution. Elements will generally precipitate at and SI greater than one and dissolve at SI less than zero.

The gain or loss of electrons causing elements to change their valence state in a chemical reaction is known as redox reactions or oxidation-reduction reactions (Fetter 1999). This is important for this study due to the potential for relatively concentrated ammonium in leachate solution and its ability to exist in many states. Gaining an electron reduces the charge (reduction), while losing an electron is oxidation. Redox reactions only involve elements that can occur in more than one valence state. Redox reactions are normally slow, but can be sped up using bacterial catalysis (Appelo and Postma 1996). These reactions usually occur when oxidants such as O_2 are introduced into an aquifer containing a reductant, but the addition of organic material can also play an important role (Appelo and Postma 1996). Oxidation potential of a solution (Eh) can be calculated from the Nernst equation (Fetter 1999):

$$Eh = E^o - \frac{RT}{nF} ln \frac{[products]}{[reactants]}$$ [2.12]

where:

Eh = oxidation potential of the aqueous solution in volts

E^o = standard potential of redox reaction in volts, $E^o = -\Delta G_R^o / nF$

$\Delta G_R{}^o$ = Gibbs free energy of the reaction

R = gas constant, 0.00199 Kcal/(mole·K)

T = temperature in Kelvins

F = Faraday constant, 23.06 Kcal/V

n = number of electrons in half reaction

[ ] = product of the activity of products and reactants

The redox conditions in the solution can determine what valance states the ions in solution exist. Depending on the valence state an ion is in, can determine its fate as a contaminant. Positive Eh values generally result in oxidising conditions while negative Eh values generally show reducing conditions. Redox conditions in this study were not determined due to the assumption that the burial pits are in an anaerobic environment, therefore ions such as ammonium should not be oxidized.

Other chemical transformations can also occur. Vapour pressure is related to the solubility of the contaminant in that it indicates the volatility of the compound (Palmer 1996). The transformation of the contaminant from the liquid phase to the vapour phase defines volatility. Due to expected high concentrations of ammonium and organic carbon in livestock mortality leachate, its oxidation could release vapour phases such as nitrogen gas and methane. Biodegradation is another chemical transformation that sometimes occurs in a contaminant.

2.7   Modeling of Contaminant Transport

This work employed the use of two software packages to evaluate the potential impact of mortalities leachate on groundwater resources. The first is PHREEQC (Parkhurst and Appelo 1999) which was used to both determine speciation of the mortalities leachate solution along with calculation of ion activities and saturation indicies and to complete one dimensional transport simulations of the leachate permeating a soil column and being flushed by subsequent groundwater movement. The second software used was Geo-Slope 2004 CTRAN for two dimensional conservative transport modeling.

2.7.1   Geo-Slope 2004 CTRAN

Geo-Slope 2004 CTRAN is also a contaminant transport model based on the solution to the two dimensional advection-dispersion equation (Krahn 2004). CTRAN is based on the finite element technique and uses a Galerkin approach to derive the solution to the finite element equation (Krahn 2004). The finite element technique employs discretization of the area under investigation into elements. Each element is connected by nodes. The nodes are used for calculating the solution to the finite element equation by using integration techniques. CTRAN

is used in this study to provide an conservative transport model to investigate how far livestock mortality leachate could travel in a period of time given typical subsurface soil characteristics.

2.7.1.1 Integration Technique

Concentrations anywhere in the element can be expressed in terms of the nodal concentration

$$C = < N > \{C\}$$
[2.13]

where:

C = concentration in the element

<N> = a matrix of interpolating (or shape) functions

{C} = the vector of nodal concentrations

Using this approach, the finite element equation can be derived and is:

$$R(c) = \theta D \frac{\partial^2 C}{\partial x^2} - q \frac{\partial C}{\partial x} - \lambda K_d \rho_d - \left(\theta + \rho_d \frac{\partial S}{\partial C} \right) \frac{\partial C}{\partial t}$$
[2.14]

where:

R(c) = concentration of finite element

θ = volumetric water content

D = coefficient of hydrodynamic dispersion

*q = flux of water = n * v_x*

K_d = sorption coefficient

λ = decay coefficient

ρ_d = bulk density of the soil

S = adsorption

The solution to the finite element equation is a function of time and the integration is performed by a finite difference approximation scheme (Krahn 2004). The user can either use a Backward Difference Approximation, or a Central Difference Approximation. To solve for nodal concentrations at a certain time increment, CTRAN needs to know the nodal concentrations at the start of the time increment, therefore, the initial conditions of the problem must be known for a solution to the transport equation (Krahn 2004).

2.7.1.2 Boundary Conditions

To solve Equation 14, CTRAN uses boundary conditions to specify parameters in the equation. In basic form, the software is computing for either an initial concentration or flux, and to do this, the model must have a concentration or flux specified at various nodes. The solution can only be obtained if at least one node in the finite element mesh has a specified concentration or flux value (Krahn 2004).

CTRAN setup involves determining the model size and material properties. An initial finite mesh and seepage conditions are created in the corresponding SEEP/W package to provide a solution to the seepage conditions of the model. The CTRAN model inputs parameters such as soil type, water content, hydraulic conductivity, gradient, porosity, diffusion, decay, etc. into the simulation. CTRAN allows boundary conditions to be specified not only at the top and bottom of the model, but anywhere else as well and can be varied throughout the model.

2.8 Leachate Speciation and Transport by PHREEQC

PHREEQC (Parkhurst and Appelo 1999) will be used to characterize the leachate chemistry as well as providing a contaminant transport model including ion exchange. PHREEQC is a computer program available from the U.S. Geological Survey and has the capabilities to provide speciation and saturation indices for chemical solutions based on the user's choice of several thermodynamic databases.

PHREEQC is also used for transport modeling including ion exchange. The basic equation employed by PHREEQC is the advection-reaction-dispersion Equation 2.8 mentioned in Section 2.6.2. Cation exchange equilibria is calculated using PHREEQC's thermodynamic databases which are based on the Gaines-Thomas convention. Due to the flexibility of the software code, other exchange equations can be used. Transport is solved in a finite difference technique, while the chemical reaction term is calculated separately for each time step (Parkhurst and Appelo 1999). The transport simulation involves a 1-D column defined by a series of cells with the same pore volume. Cell length and time steps are defined to provide the velocity of water moving through the cells. A time step is simulated with an upwind scheme shown in Figure 2.1

with concentrations calculated at the cell center.
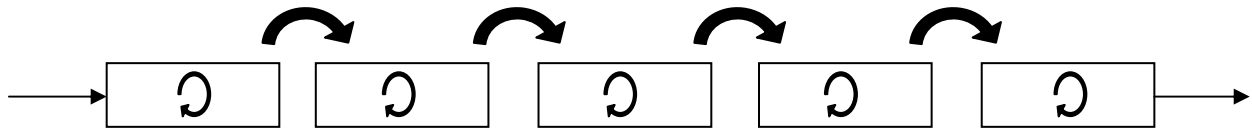
Figure 2.1: PHREEQC transport scheme.

2.9   Summary

The literature on the topic of livestock mortality burial is lacking significant evidence of the actual chemistry of livestock mortality leachate. There have also been very few studies involving contaminant transport of this leachate and the potential affect it could have on groundwater systems. This thesis will determine the chemistry of livestock leachate and its characterization and then produce contaminant transport simulations for three soil conditions to have a basis on which regulatory bodies could develop risk analysis for livestock burial systems.

3   METHODS AND MATERIALS

To properly evaluate the impact livestock mortality leachate can have on burial sites, methods of determining risks are performed. First, it was of importance to know the actual chemistry arising from a livestock burial pit from three species of livestock; poultry, swine and bovine. Characterization of the leachate constituents by performing speciation with PHREEQC was performed to evaluate possible precipitates and ions available for transport. Secondly, to evaluate the risk to underlying groundwater systems, contaminant transport model simulations using Geo Slope CTRAN 2004 and PHREEQC were performed.   A mass mortality disposal event was also simulated to provide a total mass flux into an aquifer 10 meters below the soil surface.

3.1   Leachate Chemistry

Leachate arising from a livestock burial site has not been accurately characterized in the literature. To characterize the chemical composition of this leachate, five burial sites were constructed south of Saskatoon, Saskatchewan. It was also deemed imperative to determine whether or not different species of livestock created a different leachate composition. Three common species of livestock were selected for the study and included bovine, swine and poultry. Each species was analyzed separately to determine its leachate composition.

3.1.1   Burial Site Construction

A total of five livestock mortality pits were constructed for determination of mortality leachate composition. Each pit was excavated to dimensions of 7 x 9 meters and a depth of 2.5 meters with a buffer zone of 20 meters between each (Fig. 3.1). The pits were lined with 40 mil polyethylene and a leachate collection system was located at the bottom center of the pit (Fig. 3.2).
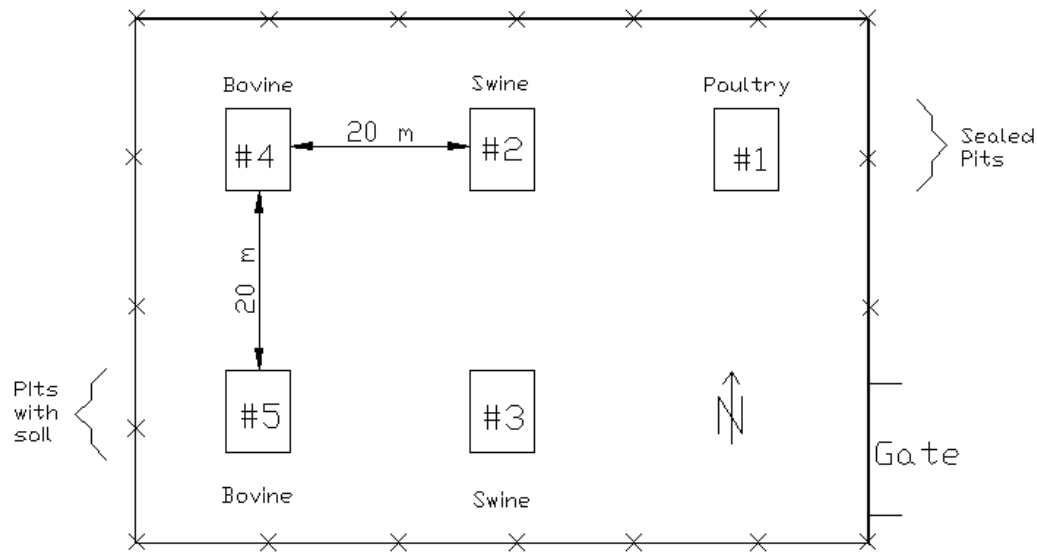
Figure 3.1 Burial site layout.
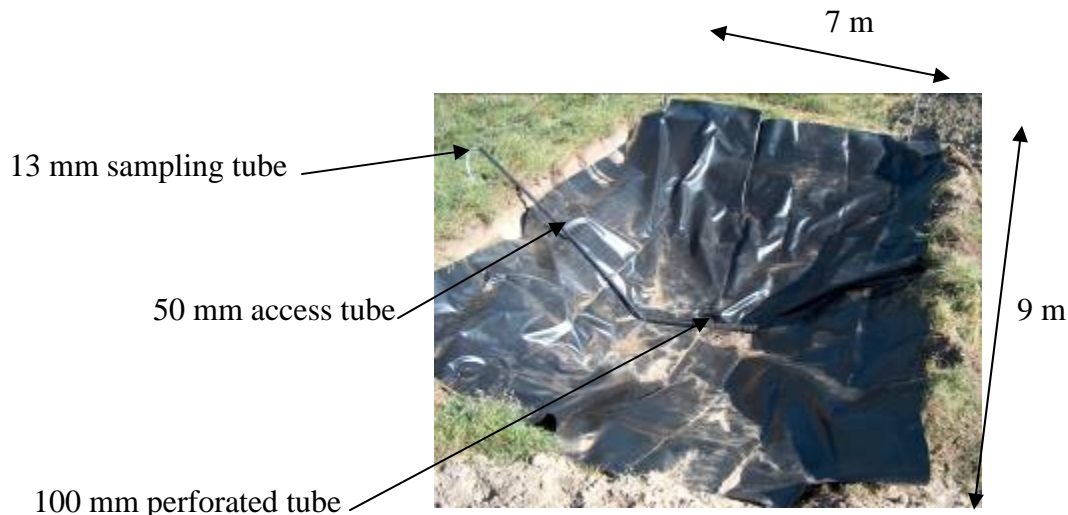
Figure 3.2 Lined burial pit.

3.1.2 Leachate Collection System

Each of the five pits contained a leachate collection system. This system consisted of a 100 mm x 1 m perforated polyethylene pipe covered by a filter sock placed in the bottom center of the pit on top of the polyethylene liner. The 100 mm perforated pipe was attached to a 50 mm ABS conduit pipe that ran to the surface. The ABS conduit had a chamber mounted on the end with a sealed cap. Inside this system of pipes were two 13 mm polyethylene lines. The ends of

these lines were located in the perforated pipe and each end was separated by a distance of 600 mm (Fig. 3.3). This separation distance allowed for mixing of the leachate directly prior to sampling. The ABS conduit with the polyethylene lines ran to the surface for ease of sampling.
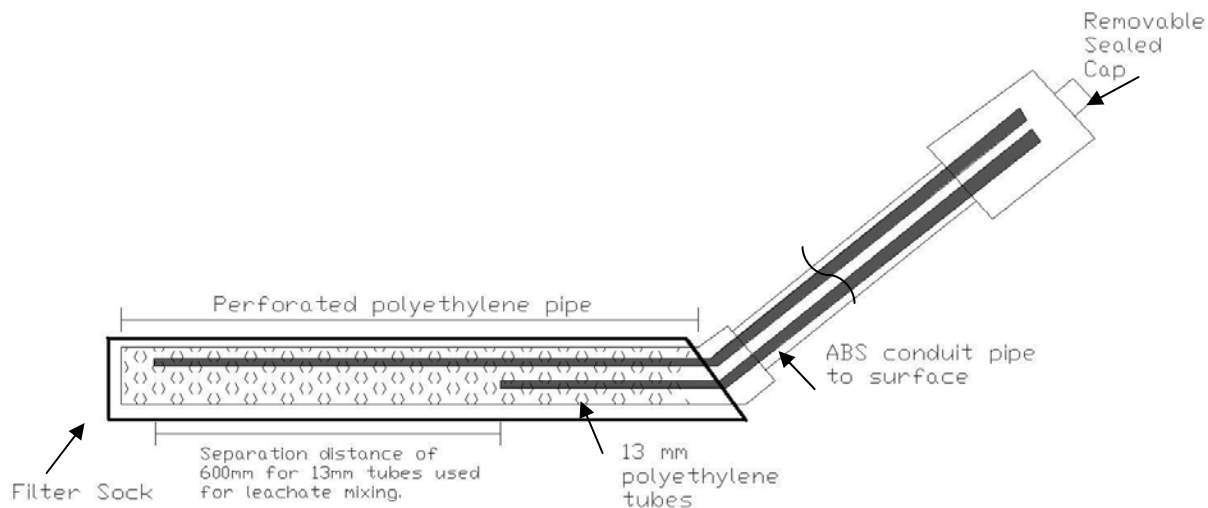
Figure 3.3 Leachate collection system.

3.1.3   Burial Pit Loading Procedure

Three of the five pits (one of each species) pits 1, 2 and 4, were completely sealed by placing a 40 mil polyethylene cover above the carcasses to allow a pure leachate sample (Fig.3.4). This cover also contained two 50 mm ABS vent pipes to allow for gas transfer to keep the pits stable. These pits were then capped with soil. The remaining two pits, swine and bovine (pits 3 and 5), possessed the same 40 mil polyethylene liner and leachate collection system in the bottom of the pit, but they did not have a top polyethylene cover. Soil was also layered on top of the polyethylene liner and with the carcasses during burial in pits 3 and 5. The pits were topped with soil excavated from the pit. (Fig. 3.5) To control water infiltration and animal intrusion, all five pits were capped and mounded with approximately 0.9 to 1.2 meters of soil. Due to settlement issues, soil was added or reshaped on the mounds a few months after burial. The pits were located on an area of elevated and well drained land so that surface or groundwater would not influence the results.
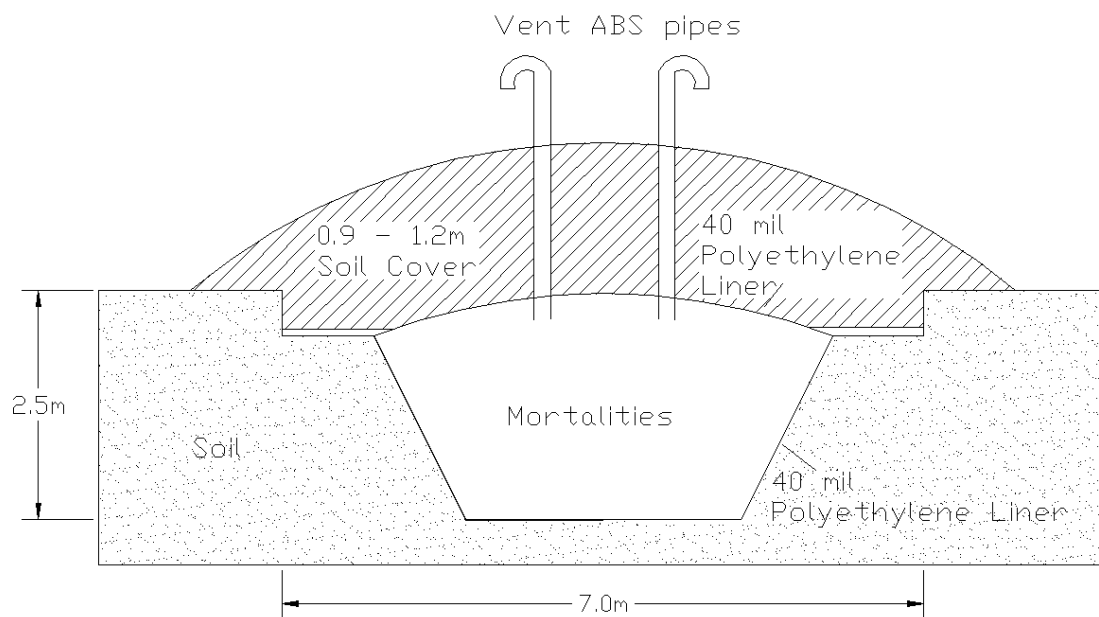
Figure 3.4 Burial pit cross section completely sealed.
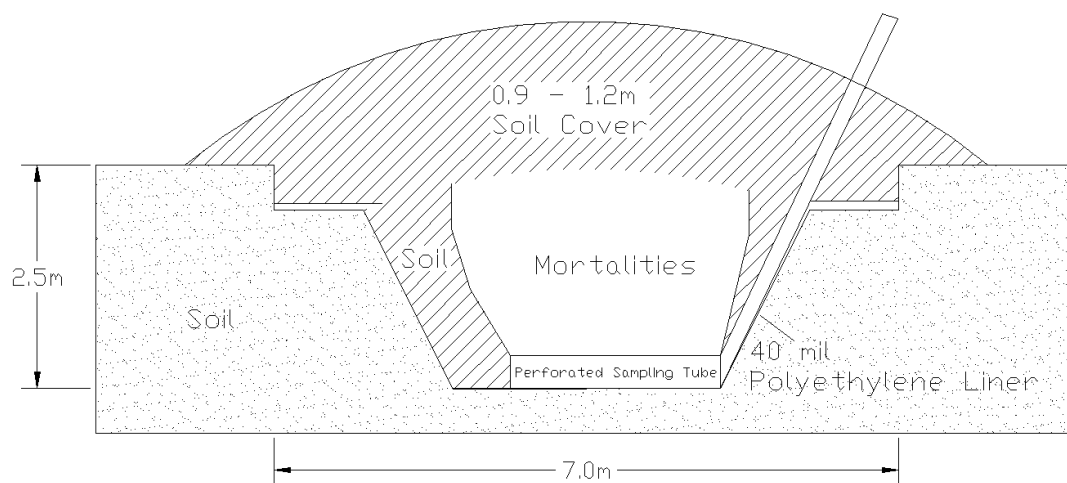
Figure 3.5 Burial pit cross section with soil cover and no top liner.

Pit No. 1 (Poultry) was installed August 4, 2005 and contained 1300 kg of chicken carcasses in an environment sealed from contact with soil and water. The burial pit was excavated with a trackhoe and the polyethylene liner was placed in the pit along with the leachate collection

system. The poultry were then dumped from the rendering truck into the pit. Once poultry were contained within the pit, another polyethylene liner was placed on top and folded inward with the bottom liner to seal the pit. The pit was then capped with a mounded soil cover.

Pit No. 2 (Swine) was installed August 9, 2005 and contained approximately 5900 kg of swine carcasses in a sealed environment. The swine in Pit No. 2 were added to the pit in the same way as the poultry. Pit No. 3 (Swine) also installed August 9, 2005, contained approximately 5900 kg of swine carcass layered with in situ fine sandy clay soil. To layer this pit, the polyethylene liner was placed in the pit and then the leachate collection system was installed. To keep soil from clogging the collection system, a few mortalities were then placed on top of the collection system. Carcasses in this case were slowly added and rearranged with a trackhoe and more soil placed around the carcasses. Once this pit was full, it did not receive a top polyethylene liner but was capped with mounded soil.

Pit No. 4 (Bovine), installed August 9, 2005, contained approximately 3920 kg of bovine carcasses in a sealed environment. To ensure there was no puncture of the polyethylene liner, bovine without horns were used and were placed in the pit using a grapple on the trackhoe, ensuring their hooves were pointed upwards. Pit No. 4 was then topped with a polyethylene liner and finished the same as Pit No 1 and 2. Pit No. 5 (Bovine), installed August 17, 2005, contained approximately 9750 kg of bovine carcasses with in situ fine sandy clay soil. To layer this pit, the polyethylene liner was placed in the pit and then the leachate collection system was installed. To keep soil from clogging the collection system, a few mortalities were then placed on top of the collection system. Carcasses and soil was then added until the pit was full. Once the pit was full, soil was mounded on top with a trackhoe for a cap.

3.1.3   Sampling Methods

Sampling was conducted periodically over a period of two years. Periodic samples were taken to observe the trends in chemical composition as decomposition occurred. Leachate samples were drawn from the pits at two weeks post burial (August 17, 2005) for Pit No. 1 (poultry), and at two weeks for the remaining pits on August 25, 2005. Samples from all pits were then taken

on September 21, 2005; October 20, 2005; November 23, 2005; May 25, 2006; October 26, 2006 and September 11, 2007. Due to winter conditions, samples were not taken between November 2005 and May 2006.

Sampling was achieved by the use of a peristaltic pump connected to each 13 mm line at the ground surface. The purpose of the lines was to provide a mechanism of mixing prior to collecting the leachate sample. The peristaltic pump was used to circulate leachate fluid through the lines for a period of five minutes prior to sampling (Fig. 3.6) thus mixing the fluid collected in the 100 mm chamber in the base of the pit. Once leachate was thoroughly mixed within the perforated chamber, a 1 litre sample was drawn and stored at 4°C for transport.
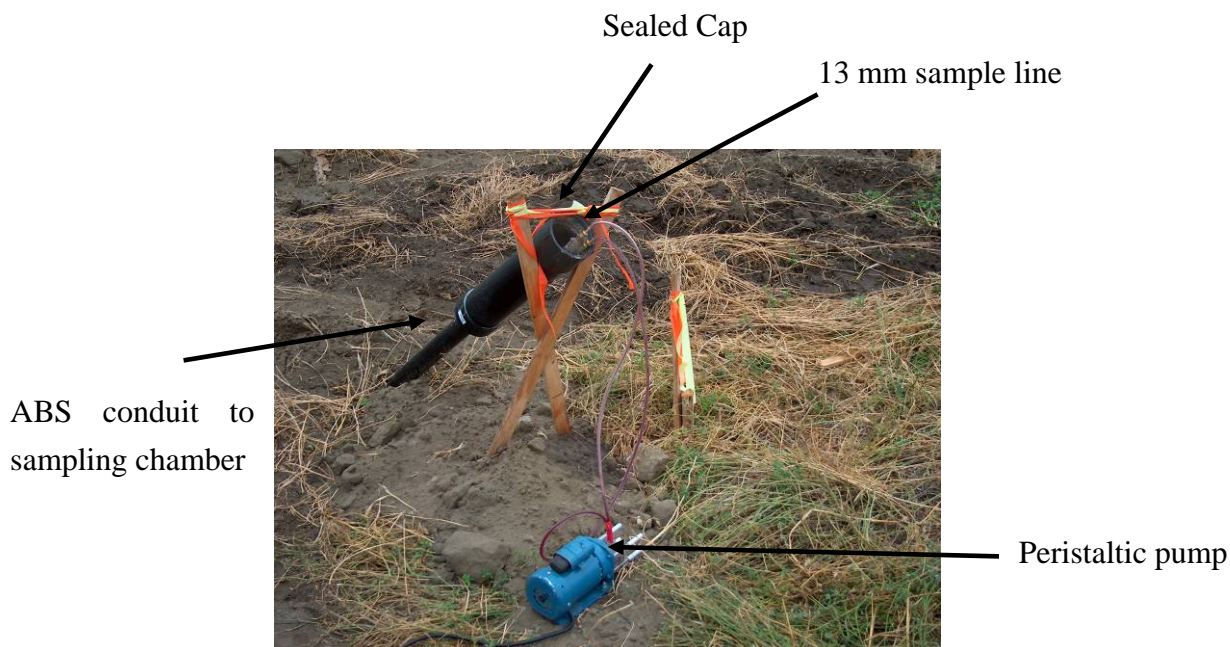
Figure 3.6 Leachate sampling.

Samples were taken to SRC Analytical Laboratories in Saskatoon for analysis. Properties analyzed include: ammonium-N by colorimetry, bicarbonate and carbonate by alkalinity, chloride by colorimetry, hydroxide, pH, specific conductivity, alkalinity by ICP-AES, nitrite-nitrate nitrogen by colorimetry, total kjeldahl nitrogen, inorganic and organic carbon with the Dohrmann Phoenix 8000 Carbon Analyzer, and aluminum, barium, beryllium, boron, cadmium, calcium, chromium, cobalt, copper, iron, lead, magnesium, manganese, molybdenum, nickel,

phosphorus, potassium, silicon, silver, sodium, strontium, sulfate, sulfur, titanium, vanadium, zinc and zirconium by ICP-AES. Analytical methods and detection limits are given in Appendix A.

3.1.4   Leachate Speciation

PHREEQC (Parkhurst and Appelo 1999) was used to help characterize the leachate chemistry. The software calculated species distribution for all ions entered and saturation indices of the related precipitates and partial pressures of gasses. To provide a representation of the element species that can be present in the mortality leachate, an average value from all three species bovine, swine and poultry was used. The thermodynamic database used to calculate the speciation comes from Parkhurst and Appelo (1999) and is titled phreeqc.dat.

3.2   Modeling Methods

Two contaminant transport software packages were used to evaluate the potential impact mortality leachate could have on soil and groundwater systems. Saturated conditions were assumed for all simulations. Two software packages, CTRAN (Krahn 2004) and PHREEQC (Parkhurst and Appelo 1999) were used. The CTRAN model was used to provide a 2-D transport simulation for unattenuated transport conditions as well as the effects caused by pits located within proximity of each other. PHREEQC was used to evaluate the potential attenuation occurring along the transport path and involve 1-D transport.

3.2.1   Geometry and Boundary Conditions

In a meeting with Saskatchewan Agriculture and Food, it was determined that the burial method that would be used for the province in an emergency situation would be a trench 2 meters wide by 4 meters in depth (Jansen 2006). These dimensions for trench burial were used to create the models.

To determine the amount of leachate capable of draining from the trench, the following was assumed based on an average bovine. The average weight of a bovine is 545 kg, in which seventy percent is water (CFIA 2006). The Canadian Food Inspection Agency (CFIA) has determined that 1.5 m^3 is required for burial space of one bovine animal (CFIA 2006). Final

moisture content of the decomposed carcass was further assumed to be 50% or similar to compost. The difference between the initial 70% moisture content and the final moisture content was assumed to be free to leave the pit. Using these assumptions, the amount of free fluid available for drainage can be calculated. These assumptions equate to approximately 1 m^3 of leachate available for drainage per meter length of trench, or a 0.5 meter depth of leachate in the base of a 2 meter wide trench.

3.2.1.1 CTRAN

The CTRAN and coordinating SEEP/W simulation setup is shown in Figure 3.8 with dimensions in meters. The top boundary conditions for each simulation include a finite mass source located at the nodes marked by triangles and include a contaminant depth of 0.5 meters at 15,000 mg/L. The pit bottom elevation is the same as the top elevation of the water table ensuring saturated flow conditions. This software package was also used to simulate whether or not trench spacing had an impact on the zone of influence beneath the burial trench (Figure 3.9). Two trench spacings were simulated; five meters and ten meters spacing. The same concentration of contaminant was used in these simulations and was also considered conservative (it does not react or adsorb with the medium).
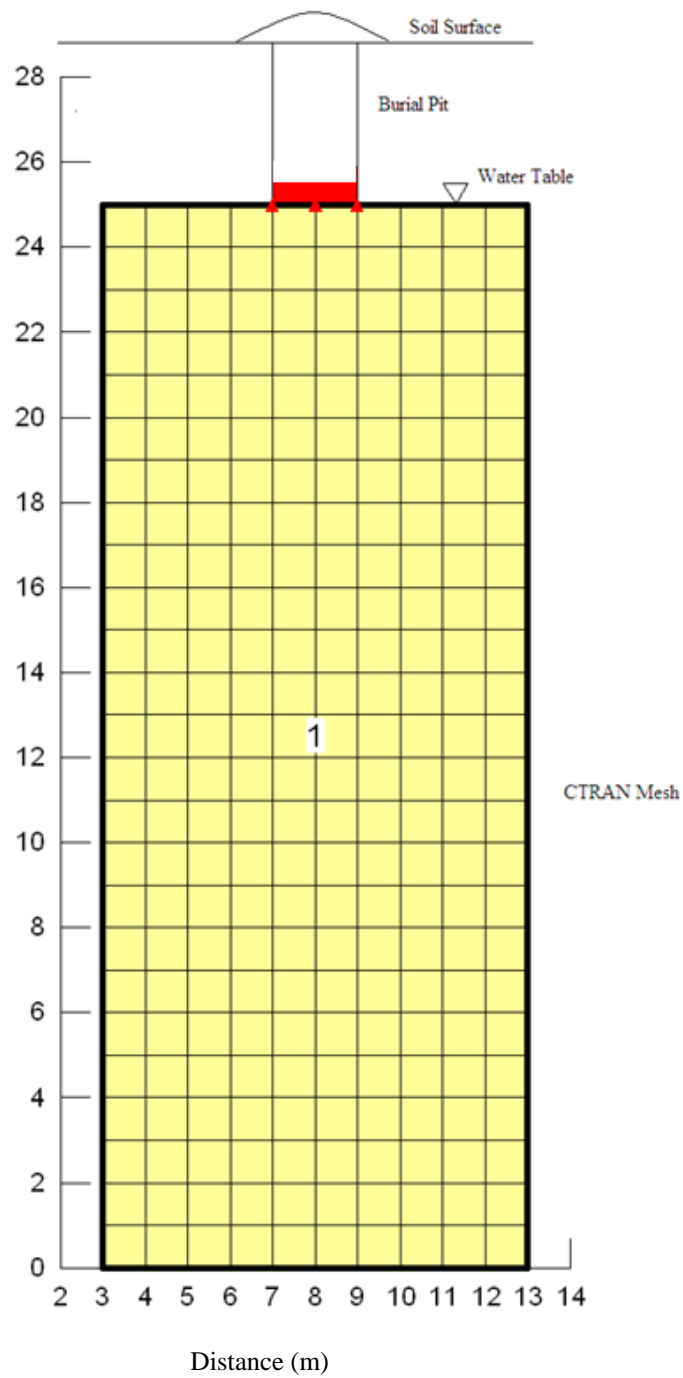
Figure 3.8 CTRAN model dimensions.
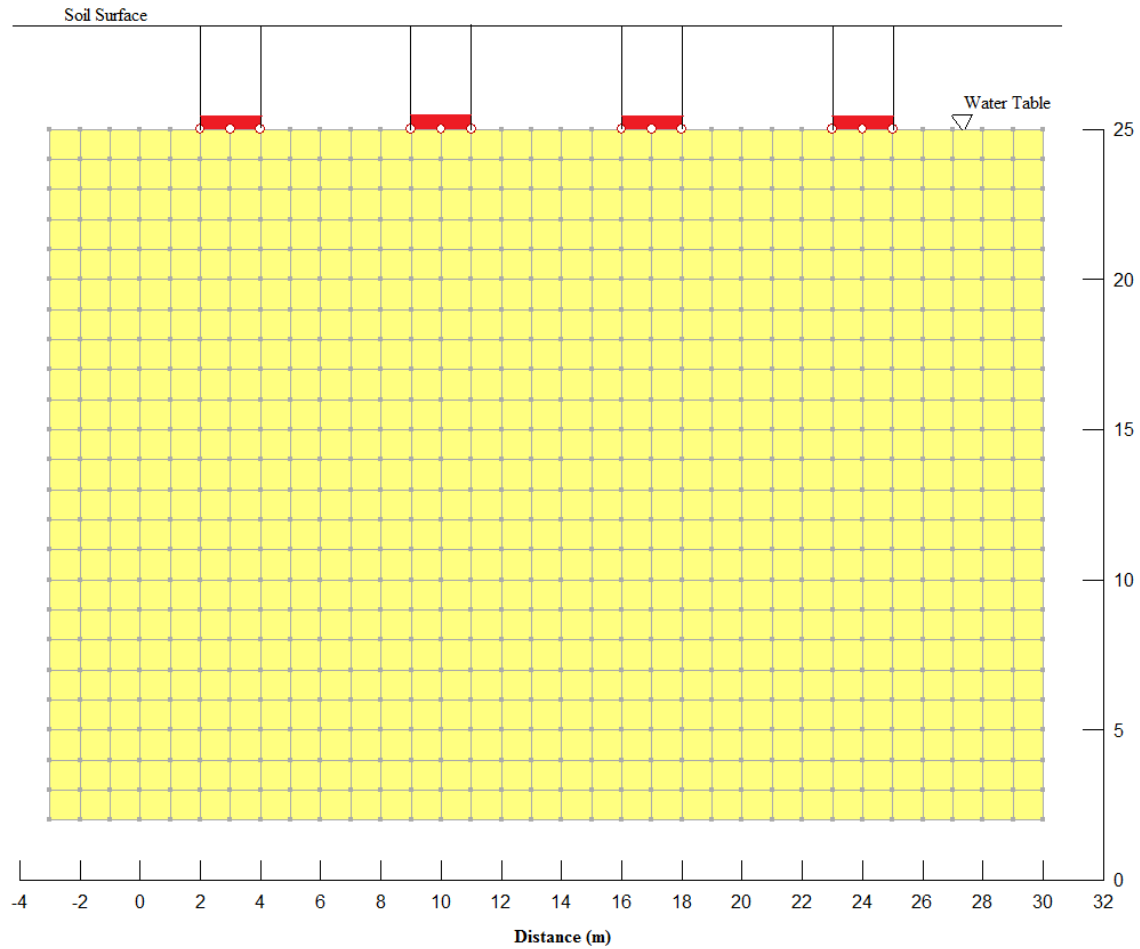
Figure 3.9 CTRAN model dimensions for multiple trenches.

3.2.2 Material Properties

In order to evaluate the potential impact in different soil settings, different situations of soil systems needed to be considered. To predict groundwater flow and solute transport, the following material properties were needed: hydraulic conductivity and gradient, coefficient of molecular diffusion, dispersivity and matrix porosity.

3.2.2.1 Porosity

Porosity for different geologic materials vary over a range, with sands and gravels between 0.2 and 0.5, silts between 0.3 and 0.5, and clays between 0.3 to 0.7 (Freeze and Cherry 1979). A porosity of 0.3 was used for all models due to the possibility of a porosity of 0.3 being present in any soil condition.

3.2.2.2 Coefficient of Molecular Diffusion

The coefficient of molecular diffusion is estimated at 1 x 10^{-10} to 1 x 10^{-9} m^2/s at 25°C for most salts in dilute solutions, but can vary by a factor of three (Robinson and Stokes 1965; Appelo and Postma 1994). Temperature can reduce this value by 50 percent if the temperature is reduced from 25°C to 5°C. The tortuosity factor can range typically from 0.5 to 0.01 for saturated conditions (Freeze and Cherry 1979). Without factoring temperature and tortuosity; 9.5 x 10^{-10} m^2/s, the average diffusion coefficient for most salts was used in all models.

3.2.2.3 Coefficient of Mechanical Dispersion

Dispersivity from section 2.6.1.3, is typically estimated at 1/10 to 1/100 of the travel distance (Gelhar et al. 1992). In a study performed by Woodbury (1997), 0.5 m was used, while Rudolph et al. (1991) used 0.1 m. To represent a typical dispersivity value, for transport distances up to 20 meters, 0.1 m was used in all simulations.

3.2.2.4 Hydraulic Conductivity

The velocity of water through the soil is dependent on the hydraulic conductivity of the soil and the gradient. Hydraulic conductivities of 1x10^{-10} m/s, 1x10^{-9} m/s and 1x10^{-8} m/s were used in the CTRAN models to demonstrate typical hydraulic conductivities for a low permeable, moderately permeable and highly permeable glacial till soil respectively (Freeze and Cherry 1979, Hendry 1982, Keller et al. 1986). Hydraulic gradients range from 0.1 to 0.001 for glacial till (Legget 1976). For the simulations, groundwater velocity was only used in the vertical gradient with a gradient value of 0.01. 0.01 was chosen as the mid-range order of magnitude between typical high and low hydraulic gradients. This translates to groundwater fluxes of 1x10^{-12}, 1x10^{-11} and 1X10^{-10} m/s.

3.2.3   Model Duration

Time spans of 10, 20, 50 and 100 years were simulated to evaluate contaminant movement. The results were then plotted to quantify the difference in transport between each soil condition.

3.2.4 PHREEQC Transport Model

PHREEQC was used to evaluate contaminant transport with attenuation in a 1-D column. The column consisted of 50 cells, each 0.2 m in length for a total column length of 10 meters. Dispersivity and diffusion coefficients were the same as used in the CTRAN model; 0.1 m and 9.5×10^{-10} m^2/s, respectively. Cation exchange was also incorporated into the model with a CEC value of 10 meq/100g. The column was equilibrated with a typical groundwater from Fonstad (2004) shown in Table 3.1. Once equilibration was complete, transport of leachate was performed. The volume of leachate transported is equivalent to 0.14 pore volumes or approximately 0.5 m depth of free solution. To evaluate further transport of the leachate, transport of sequential volumes of groundwater equal to 0.14 pore volumes of the column were completed up to 0.58 more pore volumes.

Table 3.1 Chemical composition of a typical groundwater (Fonstad 2004)

	units	Concentration
pH		7.4
Ca	mg/L	52
Mg	mg/L	25
Na	mg/L	29
K	mg/L	66
NH4-N	mg/L	0.6
Alkalinity	mg/L	253
Cl	mg/L	29
SO4	mg/L	82
P	mg/L	0.37
Fe	mg/L	0.55
Mn	mg/L	0.36
Si	mg/L	6
Sr	mg/L	0.48

3.3 Mass Mortality Event Simulation

To assess total mass loading into the groundwater systems below burial trenches, it is important to simulate a mass mortality event. The simulation calculated total mass loading for a cattle feedlot consisting of 10,000 head. Using transport results for a mod. permeable soil from PHREEQC, mass loading into an aquifer 10 meters below the soil surface was calculated.

4 RESULTS AND DISCUSSION

The results from each burial pit are presented and analyzed in this section. Chemical speciation was also performed in PHREEQC to present a speciation of elements occurring in solution as well as potential minerals dissolving and precipitating. The transport results from CTRAN 2004 and PHREEQC are presented. A simulated mass mortality disposal event was assessed using a 10,000 head feedlot with respect to mass loading on an aquifer at a depth of 10 m below the soil surface.

4.1 Livestock Decomposition Leachate Chemistry Characterization

The major elements found in livestock mortality leachate include ammonium, bicarbonate, calcium, magnesium, iron, potassium, phosphorus, sulphate, sodium, chloride, along with other trace amounts of other elements. The major elements found in the burial leachate will be discussed in the next section. Tables 4.1 – 4.5 show the analytical results for each burial pit per sampling date.

Nitrogen found in ammonium and TKN are of the most concern due to their high concentrations which both ranged from 10,000 to 19,000 mg/L two years after burial. High concentrations of phosphorus ranging from 1000 to 1,900 mg/L and bicarbonate from 40,000 to 59,000 mg/L were also discovered two years after burial.

Poultry leachate concentrations are shown in Table 4.1. Alkalinity concentrations were lowest at 21,100 mg/L at the first sample date and highest at 32,500 mg/L at the last sample date. Chloride concentrations were highest at the first sample date at 3,400 mg/L and lowest on the last sample date at 2,360 mg/L. Ammonium concentrations for poultry peaked on the sixth sample date at 12,700 mg/L, and were lowest at the first sample date at 4,500 mg/L. Nitrate/nitrite nitrogen reached a maximum of 9.1 mg/L on the second sample date. Other ion concentrations such as aluminum, calcium, iron, magnesium, manganese, phosphorus,

Table 4.1 Poultry (Pit No. 1) leachate chemistry.

DATE of SAMPLE		Aug.17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	25700	33200	37000	36000	38800	39000	39600
Carbonate	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Chloride	mg/L	3480	2860	2540	2470	2690	2670	2360
Hydroxide	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
pH	pH Units	6.23	6.2	6.4	6.42	6.44	6.48	6.52
Specific Conductivity	uS/cm	45900	30700	37400	37600	36800	36900	45200
Total Alkalinity	mg/L	21100	27200	30300	29500	31800	32000	32500
Ammonium as Nitrogen	mg/L	4500	4542	9860	11200	8370	12700	10100
Nitrate + Nitrite Nitrogen	mg/L	n/d	9.1	3.7	1.8	n/d	2	4.9
Total Kjeldahl Nitrogen	mg/L	19500	16500	17800	20200	18100	17700	17300
Inorganic Carbon	mg/L	5060	6530	7280	7080	7630	7670	7790
Organic Carbon	mg/L	47000	55000	55000	62000	58000	78000	99000
Aluminum	mg/L	1.5	1.4	n/d	0.8	0.6	1	n/d
Barium	mg/L	0.6	5	0.7	n/d	n/d	0.2	n/d
Beryllium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Boron	mg/L	n/d	n/d	n/d	n	0.2	n/d	0.6
Cadmium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Calcium	mg/L	110	50	60	70	83	90	70
Chromium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Cobalt	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Copper	mg/L	0.6	0.5	0.5	1.3	0.52	1.8	0.4
Iron	mg/L	42	42	28	29	29	18	8
Lead	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Magnesium	mg/L	140	140	110	110	97	80	60
Manganase	mg/L	1.1	1.1	0.8	0.8	0.71	0.6	0.3
Molybdenum	mg/L	n/d	n/d	n/d	n/d	1	n/d	n/d
Nickel	mg/L	n/d	n/d	n/d	0.2	0.07	n/d	n/d
Phosphorus	mg/L	1830	2090	1870	1970	2000	1890	1890
Potassium	mg/L	3100	2500	2300	2500	2600	2100	2400
Silicon, soluble	mg/L	12	16	13	15	14	17	30
Silver	mg/L	0.2	n/d	n/d	n/d	n/d	n/d	n/d
Sodium	mg/L	1900	1600	1800	1800	1600	1500	1600
Strontium	mg/L	n/d	0.1	0.1	n/d	0.09	0.1	n/d
Sulphate	mg/L	4400	4700	3800	4400	4900	3400	3600
Sulphur	mg/L	1500	1600	1300	1500	1600	1100	1200
Titanium	mg/L	0.1	n/d	n/d	n/d	n/d	n/d	n/d
Vanadium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Zinc	mg/L	10	7.3	2	4.1	2.7	3	0.9
Zirconium	mg/L	0.3	n/d	n/d	0.1	0.06	n/d	n/d

n/d: not detectable

Table 4.2 Swine (Pit No. 2) leachate chemistry.

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	34200	36600	40000	40000	47000	47000	51400
Carbonate	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Chloride	mg/L	2855	2700	2570	2490	2360	2650	2140
Hydroxide	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
pH	pH Units	6.47	6.57	6.77	6.82	6.72	6.86	6.66
Specific Conductivity	uS/cm	33200	35500	40400	41900	42800	46600	48700
Total Alkalinity	mg/L	28000	29200	32800	32800	38500	38500	42100
Ammonium as Nitrogen	mg/L	2400	5831	12600	10500	8490	15200	16300
Nitrate + Nitrite Nitrogen	mg/L	25	6.6	7.2	3.6	3.8	4.7	0.83
Total Kjeldahl Nitrogen	mg/L	17100	19000	15100	19200	18800	17200	12000
Inorganic Carbon	mg/L	6730	7000	7670	7870	9250	9250	10100
Organic Carbon	mg/L	44000	48000	48000	54000	56000	68000	70000
Aluminum	mg/L	0.7	2.1	0.9	1	0.5	0.9	n/d
Barium	mg/L	n/d	0.8	0.5	n/d	n/d	n/d	n/d
Beryllium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Boron	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	1.6
Cadmium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Calcium	mg/L	70	30	40	40	65	50	30
Chromium	mg/L	n/d	0.1	n/d	0.1	n/d	n/d	n/d
Cobalt	mg/L	0.1	n/d	n/d	n/d	n/d	n/d	n/d
Copper	mg/L	0.3	1	1.3	1.4	2.7	1.7	0.6
Iron	mg/L	70	37	54	35	29	19	10
Lead	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Magnesium	mg/L	80	30	30	20	20	20	10
Manganase	mg/L	2.6	0.5	0.4	0.3	0.12	0.1	n/d
Molybdenum	mg/L	0.4	n/d	n/d	n/d	0.57	n/d	0.2
Nickel	mg/L	0.3	0.2	n/d	n/d	n/d	n/d	n/d
Phosphorus	mg/L	1720	1380	1870	1520	1540	1540	1460
Potassium	mg/L	2800	2100	3200	2300	2600	2300	2400
Silicon, soluble	mg/L	32	13	15	12	11	12	48
Silver	mg/L	n/d	0.1	n/d	n/d	n/d	n/d	n/d
Sodium	mg/L	2300	1700	2600	2200	1700	1800	1700
Strontium	mg/L	0.2	0.1	0.2	n/d	0.08	n/d	n/d
Sulphate	mg/L	3500	3600	5700	5900	5800	2100	3800
Sulphur	mg/L	1200	1200	1900	2000	1900	690	1300
Titanium	mg/L	n/d	0.1	n/d	n/d	n/d	n/d	n/d
Vanadium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Zinc	mg/L	2.6	2.5	2.9	3.3	2.1	1.8	1.4
Zirconium	mg/L	n/d	0.2	n/d	n/d	n/d	n/d	n/d

n/d: not detectable

Table 4.3 Swine (Pit No. 3) layered with in situ soil leachate chemistry.

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	33900	34600	36200	37100	40300	40600	45900
Carbonate	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Chloride	mg/L	3065	2830	2530	2660	2720	3300	2280
Hydroxide	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
pH	pH Units	6.13	6.16	6.34	6.4	6.42	6.52	6.43
Specific Conductivity	uS/cm	29800	34600	36800	36400	36100	41800	41300
Total Alkalinity	mg/L	27800	28400	29700	30400	33000	33300	37600
Ammonium as Nitrogen	mg/L	4000	5395	10400	9290	7130	10000	12500
Nitrate + Nitrite Nitrogen	mg/L	24	13	5.2	3.3	4.6	6.9	1.2
Total Kjeldahl Nitrogen	mg/L	19300	22000	19700	15400	20200	17100	8840
Inorganic Carbon	mg/L	6670	6310	7120	7300	7930	7990	9030
Organic Carbon	mg/L	68000	59000	56000	66000	59000	75000	64000
Aluminum	mg/L	0.8	1.4	0.7	0.8	0.5	1	n/d
Barium	mg/L	n/d	0.7	0.5	0.1	n/d	n/d	n/d
Beryllium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Boron	mg/L	0.5	n/d	n/d	n/d	n/d	n/d	n/d
Cadmium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Calcium	mg/L	20	100	90	80	100	90	70
Chromium	mg/L	n/d	n/d	0.2	n/d	n/d	n/d	n/d
Cobalt	mg/L	n/d	0.1	0.2	n/d	0.06	n/d	n/d
Copper	mg/L	0.7	1.3	1.6	1.9	1.5	1.6	1.1
Iron	mg/L	58	83	64	60	54	44	21
Lead	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Magnesium	mg/L	10	80	60	50	30	30	20
Manganase	mg/L	0.3	3.3	2.1	2	1.3	1.4	0.4
Molybdenum	mg/L	0.5	n/d	n/d	n/d	0.15	n/d	n/d
Nickel	mg/L	n/d	0.4	0.2	0.2	0.28	0.2	0.2
Phosphorus	mg/L	1170	2150	1860	1820	1800	1850	1580
Potassium	mg/L	2700	2600	2400	2500	2500	2200	2300
Silicon, soluble	mg/L	20	43	41	42	40	47	46
Silver	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Sodium	mg/L	2300	2200	2700	2200	1900	2100	2000
Strontium	mg/L	0.2	0.4	0.3	0.3	0.31	0.3	0.2
Sulphate	mg/L	5500	4600	3700	3900	4700	3100	3100
Sulphur	mg/L	1800	1500	1200	1300	1600	1000	1000
Titanium	mg/L	n/d	0.1	n/d	n/d	0.07	n/d	n/d
Vanadium	mg/L	n/d	n/d	n/d	n/d	n/d	0.1	n/d
Zinc	mg/L	3.9	3.6	3.2	3.1	2.4	2.2	1.5
Zirconium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d

n/d: not detectable

Table 4.4 Bovine (Pit No. 4) leachate chemistry.

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	35100	40700	43100	45600	53400	48700	50100
Carbonate	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Chloride	mg/L	2605	2765	2400	2310	2450	3810	2180
Hydroxide	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
pH	pH Units	6.83	6.81	7.9	7.01	6.98	6.89	6.86
Specific Conductivity	uS/cm	32700	32700	41500	42100	46900	50000	51100
Total Alkalinity	mg/L	28800	33400	35300	37400	43800	39900	41100
Ammonium as Nitrogen	mg/L	5200	4708	13200	13900	10200	13000	19200
Nitrate + Nitrite Nitrogen	mg/L	23	14	2.1	2.6	6.7	3.5	1.1
Total Kjeldahl Nitrogen	mg/L	18300	11000	16000	19400	20100	17600	18400
Inorganic Carbon	mg/L	6900	8010	8480	8970	10400	9580	9860
Organic Carbon	mg/L	43000	46000	47000	57000	60000	68000	75000
Aluminum	mg/L	1.7	1.2	n/d	0.8	0.4	1	n/d
Barium	mg/L	0.3	0.6	0.5	0.2	0.06	n/d	n/d
Beryllium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Boron	mg/L	n/d	0.7	0.9	0.7	0.6	1.2	0.7
Cadmium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Calcium	mg/L	60	30	20	40	38	50	20
Chromium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Cobalt	mg/L	0.1	n/d	n/d	n/d	n/d	n/d	n/d
Copper	mg/L	0.6	1.1	1	1	0.6	0.7	0.5
Iron	mg/L	110	43	46	29	28	15	9.8
Lead	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	0.9
Magnesium	mg/L	30	10	30	20	15	30	10
Manganase	mg/L	0.5	0.3	0.4	0.4	0.19	n/d	0.1
Molybdenum	mg/L	1.8	n/d	0.2	n/d	0.21	n/d	n/d
Nickel	mg/L	0.4	0.1	n/d	n/d	0.1	n/d	n/d
Phosphorus	mg/L	920	1300	1300	1230	1170	1200	1080
Potassium	mg/L	1900	2100	2100	2100	2200	1800	1900
Silicon, soluble	mg/L	29	25	27	26	22	27	29
Silver	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Sodium	mg/L	1600	2000	2700	2200	1900	2000	2000
Strontium	mg/L	0.7	0.3	0.3	0.3	0.26	0.3	0.1
Sulphate	mg/L	3700	4500	6300	6800	4600	1700	2400
Sulphur	mg/L	1200	1500	2100	2300	1500	580	810
Titanium	mg/L	0.2	n/d	n/d	n/d	n/d	n/d	n/d
Vanadium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Zinc	mg/L	3.5	4.2	3.3	2.9	2.3	1.8	1.1
Zirconium	mg/L	0.2	n/d	n/d	n/d	n/d	n/d	n/d

n/d: not detectable

Table 4.5 Bovine (Pit No. 5) layered with in situ soil leachate chemistry.

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	20900	38600	40500	41000	53400	56500	58700
Carbonate	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Chloride	mg/L	1775	2200	1830	1630	1940	1870	1800
Hydroxide	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
pH	pH Units	6.49	6.92	7.07	7.28	6.95	7.05	6.92
Specific Conductivity	uS/cm	22300	31200	44000	42600	46000	49300	55400
Total Alkalinity	mg/L	17100	31600	33200	33600	43800	46300	48100
Ammonium as Nitrogen	mg/L	1400	3770	10800	11900	8100	20900	19200
Nitrate + Nitrite Nitrogen	mg/L	22	33	6.9	2.4	4.9	4.5	0.46
Total Kjeldahl Nitrogen	mg/L	17200	13000	14500	16800	19000	20000	19100
Inorganic Carbon	mg/L	4110	7590	7970	8060	10500	11100	11500
Organic Carbon	mg/L	33000	48000	40000	44000	60000	71000	81000
Aluminum	mg/L	1.2	3.5	1.1	0.7	n/d	0.7	n/d
Barium	mg/L	n/d	1.1	0.7	0.5	0.07	0.3	n/d
Beryllium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Boron	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	0.4
Cadmium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Calcium	mg/L	20	60	40	50	64	70	40
Chromium	mg/L	n/d	n/d	n/d	0.1	n/d	n/d	0.1
Cobalt	mg/L	n/d	n/d	n/d	0.1	n/d	n/d	n/d
Copper	mg/L	0.3	0.7	0.6	0.5	0.15	0.4	0.4
Iron	mg/L	46	90	50	32	10	8.6	5.6
Lead	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Magnesium	mg/L	30	10	10	10	12	10	n/d
Manganase	mg/L	0.3	0.5	0.3	0.4	0.24	0.3	n/d
Molybdenum	mg/L	1.3	n/d	n/d	n/d	0.1	n/d	n/d
Nickel	mg/L	n/d	0.3	n/d	n/d	0.12	n/d	0.1
Phosphorus	mg/L	1350	1290	1030	1120	1130	1250	990
Potassium	mg/L	2800	2300	2100	2300	2300	2100	2400
Silicon, soluble	mg/L	10	48	38	40	31	35	54
Silver	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Sodium	mg/L	2100	2000	2100	2100	1700	2000	2000
Strontium	mg/L	n/d	0.7	0.7	0.8	0.5	0.5	0.3
Sulphate	mg/L	4600	4500	5800	8800	5300	1700	2200
Sulphur	mg/L	1500	1500	1900	2900	1800	560	740
Titanium	mg/L	n/d	0.3	0.1	0.1	n/d	n/d	n/d
Vanadium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Zinc	mg/L	2.3	4.6	2.9	13	1.9	1.7	0.8
Zirconium	mg/L	n/d	n/d	n/d	n/d	n/d	n/d	n/d

n/d: not detectable

potassium, sodium, total sulphur, and zinc were all highest at the first two sample dates within one month of burial. Sulphate increased to a concentration of 4,900 mg/L at the fifth sample date and then decreased in concentration. Organic carbon was initially at 47,000 mg/L and increased to 99,000 mg/L at the last sample date.

Table 4.2 consists of leachate chemistry results from Pit No. 2 holding pure swine mortalities. Alkalinity concentrations were lowest at 38,000 mg/L at the first sample date and highest at 42,100 mg/L at the last sample date. Ammonium and organic carbon followed the same trend with 2,400 and 44,000 mg/L on the first sample date and 16,300 and 70,000 mg/L on the last sample date respectively. Concentrations of chloride, nitrate/nitrite, calcium, iron, magnesium, manganese, and silicon were all highest on the first sample date. Total sulphur and sulphate trends differed from the poultry by peaking four months post-burial at 5,900 and 2,000 mg/L respectively. Phosphorus and potassium were highest in the swine mortality leachate on the third sample date with 1,870 and 3,200 mg/L respectively. It is noticeable that higher concentrations except for ammonium, alkalinity and organic carbon, occurred within the first four months of burial for both poultry and swine mortality leachate.

Leachate chemistry results for Pit No. 3 involving swine in a soil layer are shown in Table 4.3. Alkalinity concentrations are consistent with Table 4.1 and 4.2 in that the first sample date showed concentrations of 27,800 mg/L while the last sample date showed the highest concentration at 45,900 mg/L. Ions such as nitrate/nitrite, aluminum, calcium, iron, magnesium, phosphorus, potassium, sulphate, total sulphur and zinc all demonstrated the highest concentrations on the first two sample dates. Sodium peaked at the third sample date with a concentration of 2,700 mg/L. Organic carbon demonstrated a peak value of 75,000 mg/L approximately one year after burial. Chloride concentrations also peaked at 3,300 mg/L one year after burial and then were at the lowest value of 2,280 mg/L two years post-burial.

Bovine mortality leachate chemistry is shown in Table 4.4. The trend of alkalinity concentrations differed somewhat from the poultry and swine mortalities in that

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concentrations of alkalinity were the highest one year post-burial at 43,800 mg/L and lowest on the first sample date at 28,800 mg/L. Sulphate and total sulphur trends were similar to swine mortality leachate and showed their highest value four months after burial at 6,800 and 2,300 mg/L respectively. Elements such as chloride, nitrate/nitrite, aluminum, calcium, iron, magnesium, manganese, phosphorus, silicon, and zinc showed highest concentrations within the first two sample dates. Organic carbon was highest at the last sample date at 75,000 mg/L and lowest at the first sample date at 43,000 mg/L. Ammonium had a low value of 4,708 mg/L at the second sample date, and peaked at 19,200 mg/L on the last sample date.

Table 4.5 shows leachate chemistry results obtained from Pit No. 5 which consisted of bovine mortalities in a soil layer. Similar to results from poultry and swine, alkalinity concentrations were lowest at the first sample date and highest at the last sample date at 17,100 and 48,100 mg/L respectively. Concentrations of chloride, nitrate/nitrite, aluminum, iron, magnesium, manganese, phosphorus, potassium and sodium were highest at the first two sample dates. Sulphate and zinc, like poultry, peaked at the fourth sample date at 8,800 and 13 mg/L respectively. Ammonium concentrations reached 20,900 mg/L after one year post-burial, while organic carbon reached a concentration of 81,000 mg/L at two years post-burial.

4.1.1   Ammonium-N

As the livestock carcass decomposed, ammonium concentrations begin to rise. Figure 4.1 shows the trend over the two year sampling period for concentrations of ammonium. In the early stages of decomposition, up to the first two months, ammonium concentrations were at their lowest around 5,000 mg/L for each species. Ammonium continued in an upward trend for all sample dates except between four and nine months. At two years post burial, bovine had the highest concentration at 19,200 mg/L which is consistent with a maximum ammonium value recorded at the mass mortality burial site in Scotland by MacArthur et al. (2002), with swine following closely at 16,300 mg/L and poultry having the lowest ammonium concentration at 10,100 mg/L.

Since nitrogen is the second most abundant element in the body, the trend of Figure 4.1

approximately shows the breakdown of different cellular systems in the livestock. The first period of sampling between 1-2 months depicts the excretion of components in the digestive system such as urine and manure. The second period between 2-4 months depicts the breakdown of blood and liquids in the animal. Between 4-14 months, it becomes harder to distinguish what is actually happening in the decomposition process. The breakdown of heavy proteins such as muscles and bone marrow is a possible route of more nitrogen appearing in the analysis at those time periods (Vass 2001; Tatsi and Zouboulis 2002).
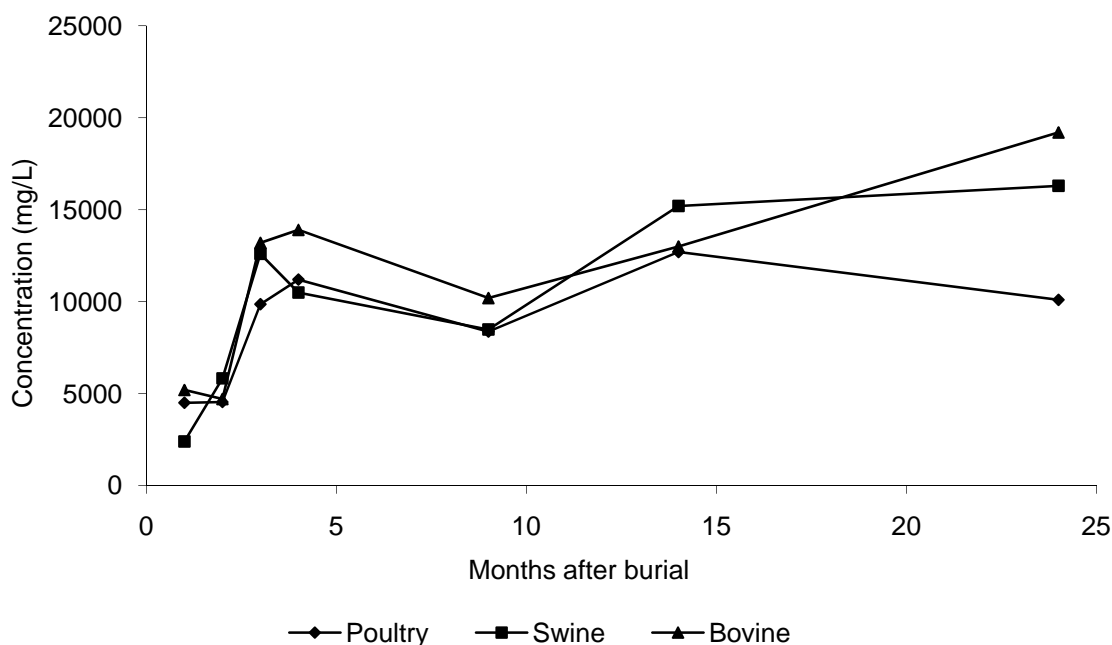
Figure 4.1 Ammonium-N concentrations of pure leachate.

The results between the swine and bovine pits including soil with the carcasses follow the same general trend line. Figures 4.2 and 4.3 are an illustration of those results. The swine layered with soil have slightly lower concentration of ammonium at all sample dates. At two years there is approximately a 5,000 mg/L difference between the concentration of pure swine leachate and that which is layered in soil (Fig. 4.2). For bovine (Fig. 4.3), the trend is not the same as swine, but after two years have reached similar concentrations. There is insufficient data to conclude there is any significant difference in chemical composition between the

capped and non-capped pits. The results would indicate that average ammonium concentrations in livestock mortality leachate could range from 10,000 to 20,000 mg/L. The last 3 sample dates seem to provide a realistic composition of leachate chemistry after decomposition and all averages reported will be over the last three sample dates, although maximum concentrations could exceed these values by as much as 50% indicated by the results. The average ammonium concentration for all species over this period is 12,800 mg/L. This value compares to 3,294 mg/L reported by MacArthur et al. (2002) and is approximately 9,500 mg/L higher.
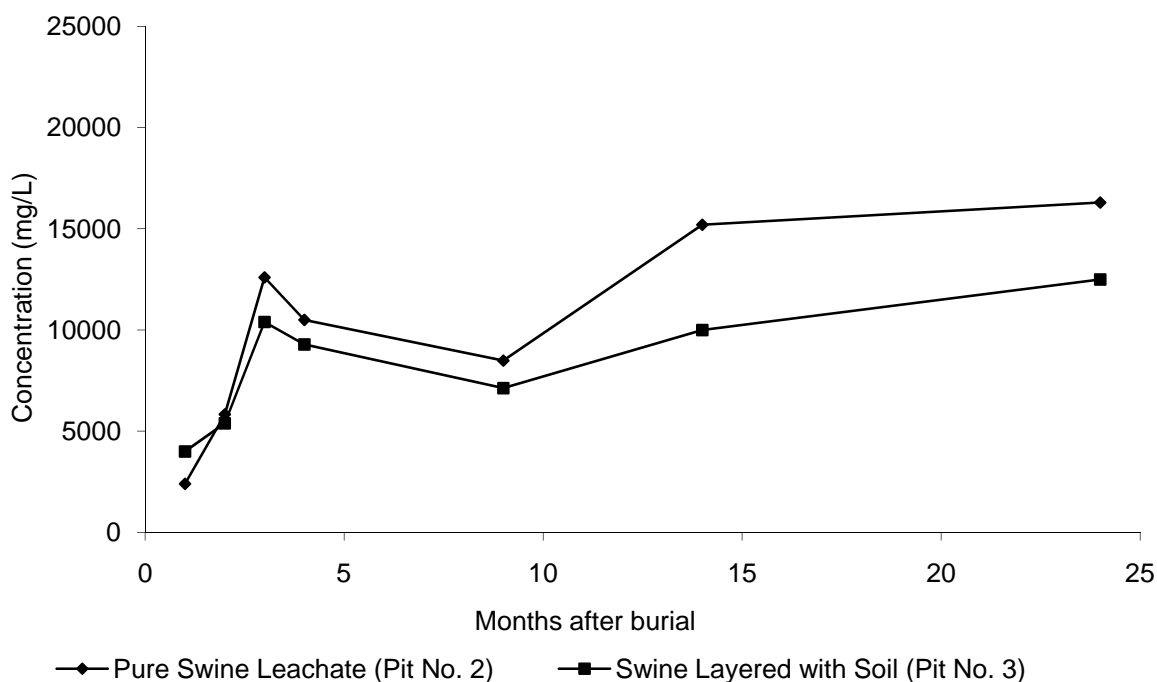
Figure 4.2 Ammonium-N concentrations for swine - pure leachate vs. soil layering.
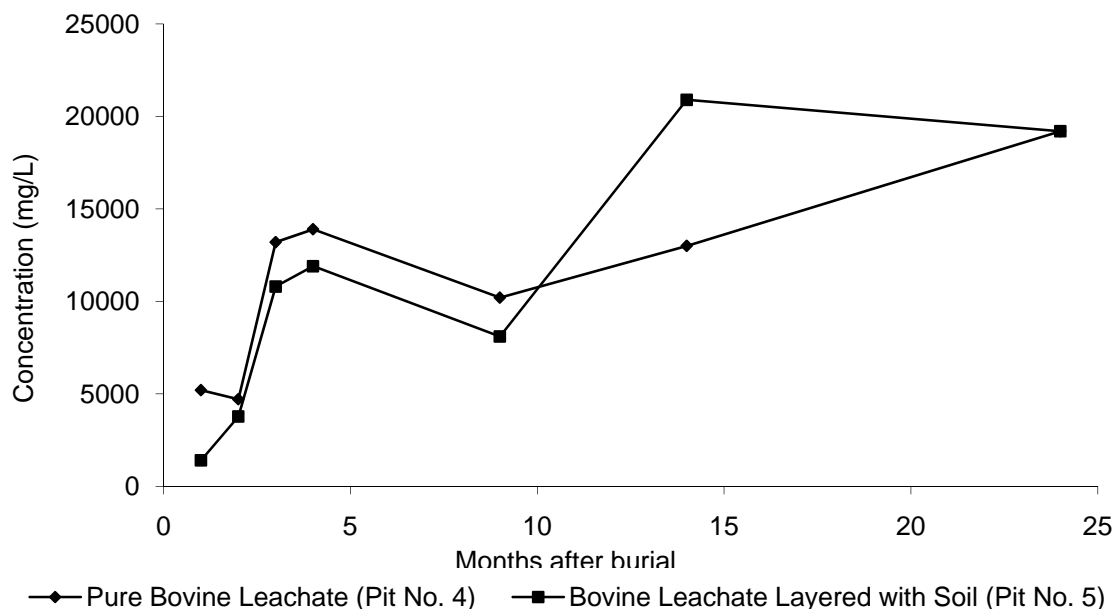
Figure 4.3 Ammonium-N concentrations for bovine - pure leachate vs. soil layering.

4.1.2 Bicarbonate

Bicarbonate (HCO_3^-) values during the sampling period of each species of livestock followed a similar trend (Fig. 4.4). Poultry (Pit No. 1) started with a concentration of 25,700 mg/L and after two years had a concentration of 39,600 mg/L. Swine (Pit No. 2) had a concentration of 34,200 mg/L at the first sample date and ended with a concentration of 51,400 mg/L after two years of decomposition.   Bovine (Pit No. 4) was similar to swine and started at 35,100 mg/L and increased to a peak concentration at nine months at 53,400 mg/L and decreased to 50,100 mg/L after 24 months. Average concentrations of bicarbonate from mortality leachate can be estimated at 46,000 mg/L. Poultry concentrations of bicarbonate could be expected to be up to 10,000 mg/L less than the average, while swine and bovine could be up to 4,000-6,000 mg/L above the average. These values are 3 times the median/average reported by MacArthur et al. (2002) of approximately 14,000 mg/L, but half the maximum reported value of approximately 106,000 mg/L.
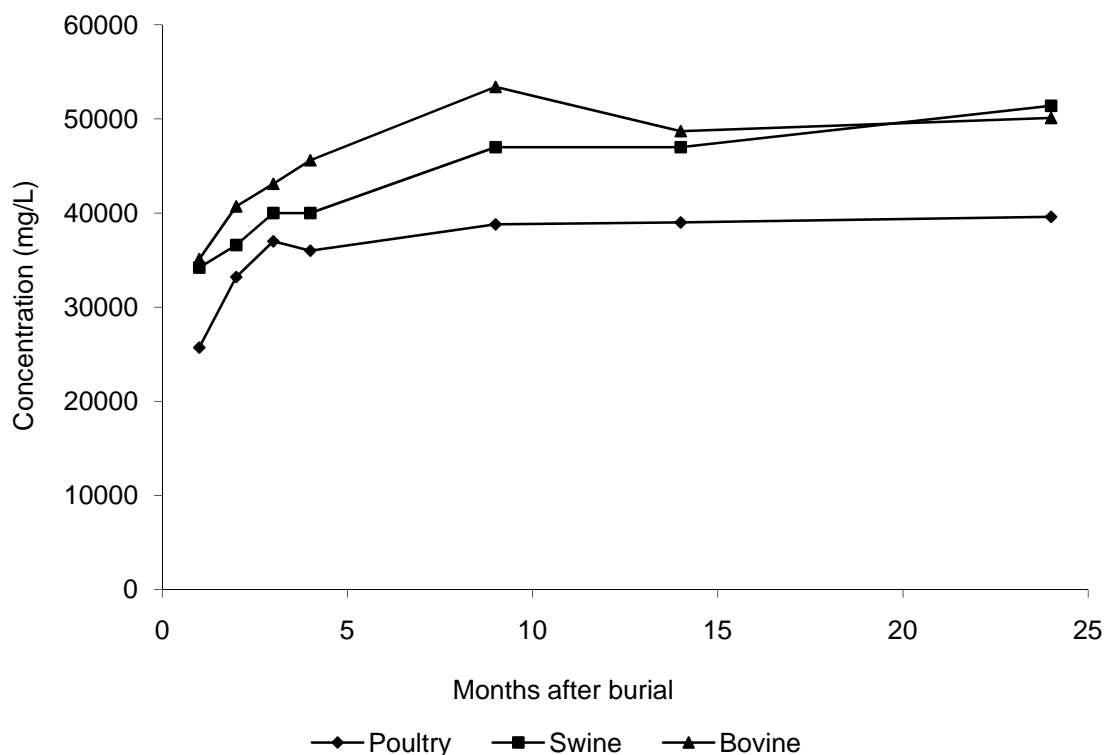
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Figure 4.4 Bicarbonate concentrations of pure leachate.

The results for swine and bovine pits layered with soil are shown in Figures 4.5 and 4.6. At the first sample date, each swine pit had similar concentration values, from the first sample date on, pure swine leachate had a slightly higher concentration than that of the swine layered in soil. By the end of the sampling period, the difference between the two concentrations was 5,500 mg/L or approximately 10%. In the case of the bovine pits, the pure bovine leachate had a higher value on the first sample date than that of the bovine layered in soil. The difference got smaller by the second sample date, and by the ninth month, each pit had the same bicarbonate concentration. From nine months on, the bovine layered in soil had a slightly higher concentration of bicarbonate. The difference at 24 months between the pure bovine leachate and that layered in soil was 8,600 mg/L. Due to the results of swine and bovine layered in soil following opposite paths, meaning swine layered with soil showed lower concentrations than pure swine leachate; and bovine layered in soil showed higher concentrations than pure bovine leachate; without more chemical analysis, the results do not
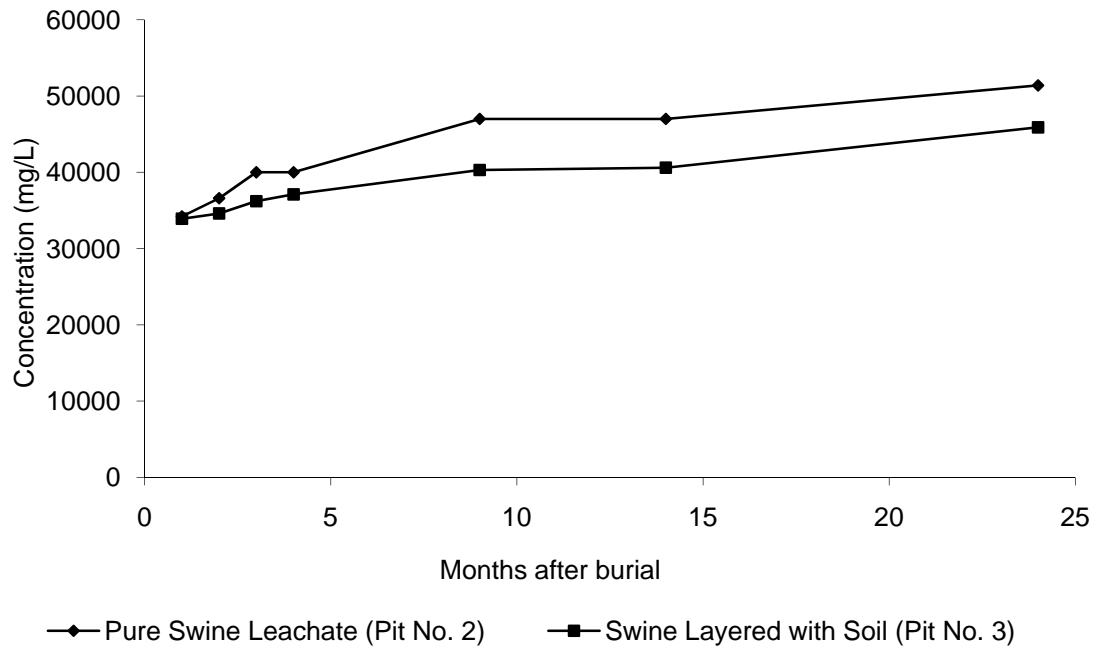
appear to be that different.

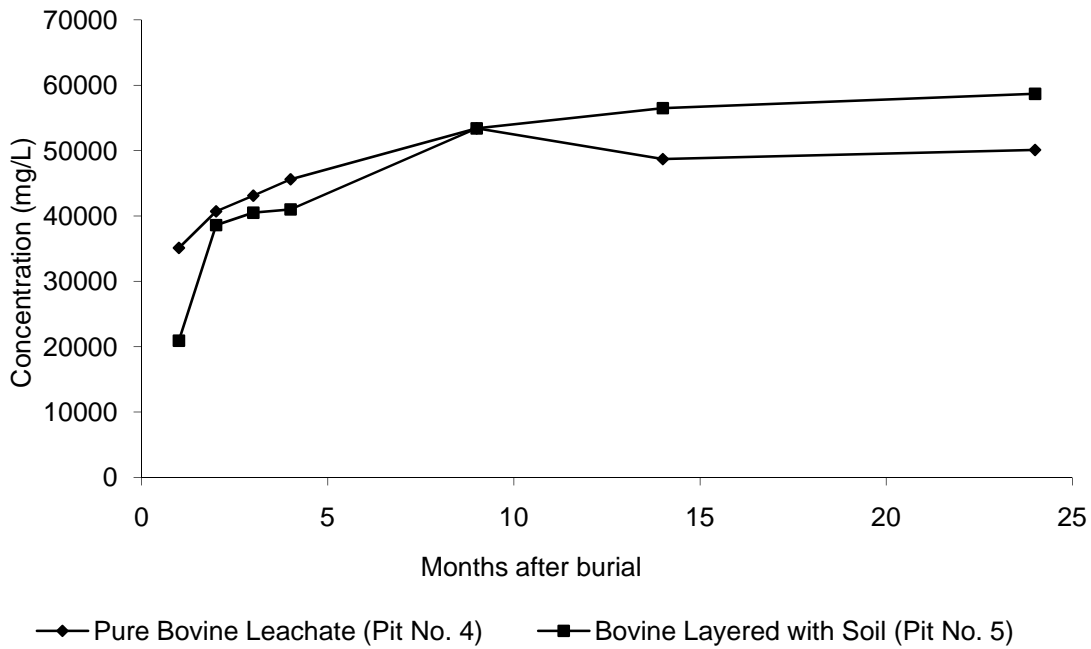Figure 4.5 Bicarbonate concentrations for swine - pure leachate vs. soil layering.

Figure 4.6 Bicarbonate concentrations for bovine - pure leachate vs. soil layering.

4.1.3 Sulphate

Sulphate (SO_4^{-2}) for each species follows the same general trend for the sample period (Fig. 4.7). Poultry at the first sample date started with a concentration of 4,400 mg/L and increased slightly by the ninth month. After nine months, the poultry leachate showed concentrations less than the initial values. By 24 months, the sulphate concentration for poultry was 3,600 mg/L. Swine initially had a concentration of 3,500 mg/L and increased sharply for the first four months to 5,900 mg/L. After four months, the concentrations of sulphate in the swine leachate decreased to 2,100 mg/L at fourteen months and increased slightly to 3,800 mg/L at 24 months. Bovine reached the highest concentration at 6,800 mg/L within four months of burial and after 24 months had a concentration of 2,400 mg/L.   Although there is variance throughout the sample dates, each species seems to be very similar within a year post-burial. An average sulphate concentration to expect in livestock mortality leachate is approximately 3,600 mg/L.

Pit No. 3 and 5 consisting of swine and bovine layered with soil, shows slight differences in sulphate leachate concentrations as compared with the pure leachate (Figs 4.8 and 4.9). Swine leachate concentrations shown in Figure 4.8 have an initial separation of 2,000 mg/L. After the
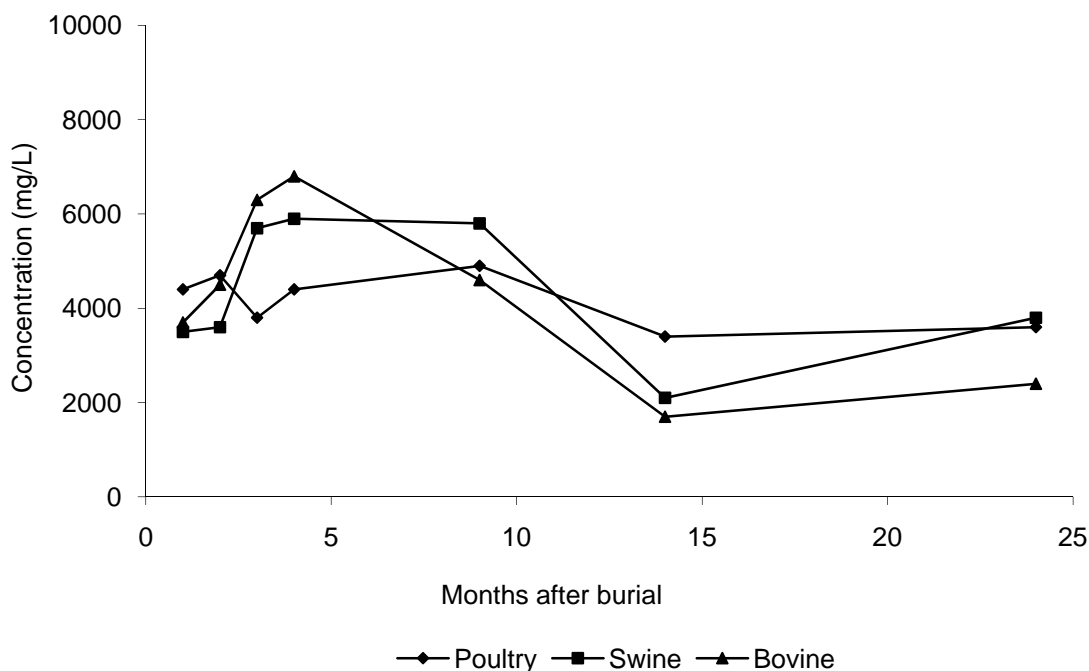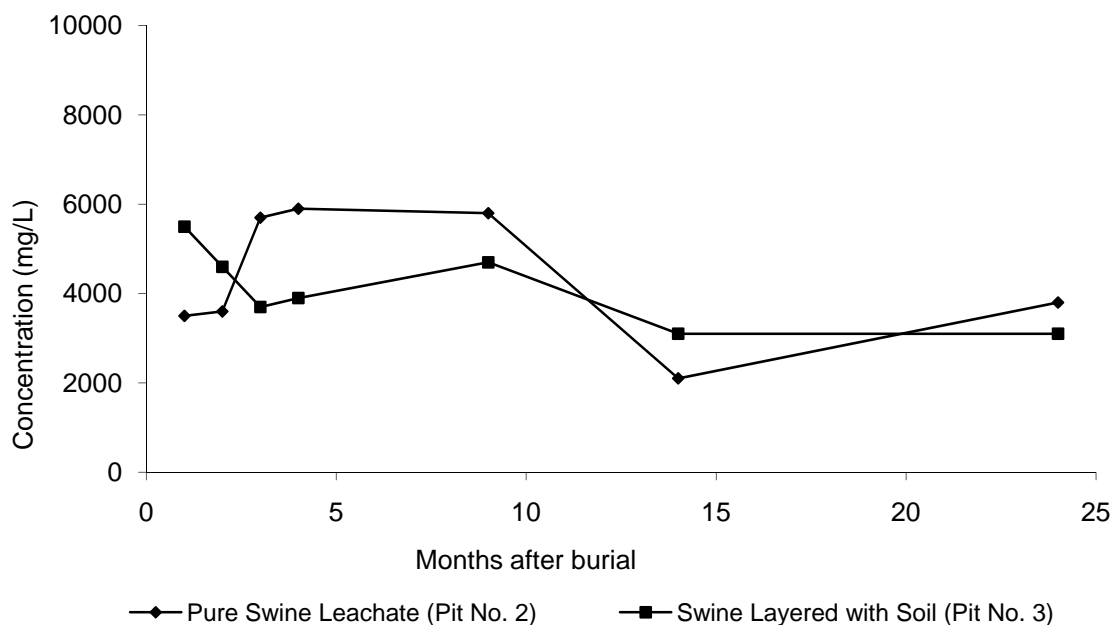

Figure 4.7 Sulphate concentrations of pure leachate.

first sample date, pure swine leachate increased in concentration while the swine layered in soil

decreased in sulphate concentration. Throughout the sample period, the pure swine leachate crossed the trend line of leachate in layered soil three times. By 24 months, pure swine leachate had a higher concentration value at 3,800 mg/L compared to 3,100 mg/L. The two pits of bovine follow each other's trend lines more closely than that of the swine (Fig. 4.9). Initially, the bovine layered in soil had a higher concentration than pure bovine leachate. After four months, this was still the case with a peak concentration of 8,800 mg/L reached by the bovine in layered soil. After 24 months, concentrations in each had decreased to nearly the same value of 2,400 mg/L. Although each species follows a different path, they all seem to end in comparable concentrations.

Figure 4.8 Sulphate concentrations for swine – pure leachate vs. soil layering.
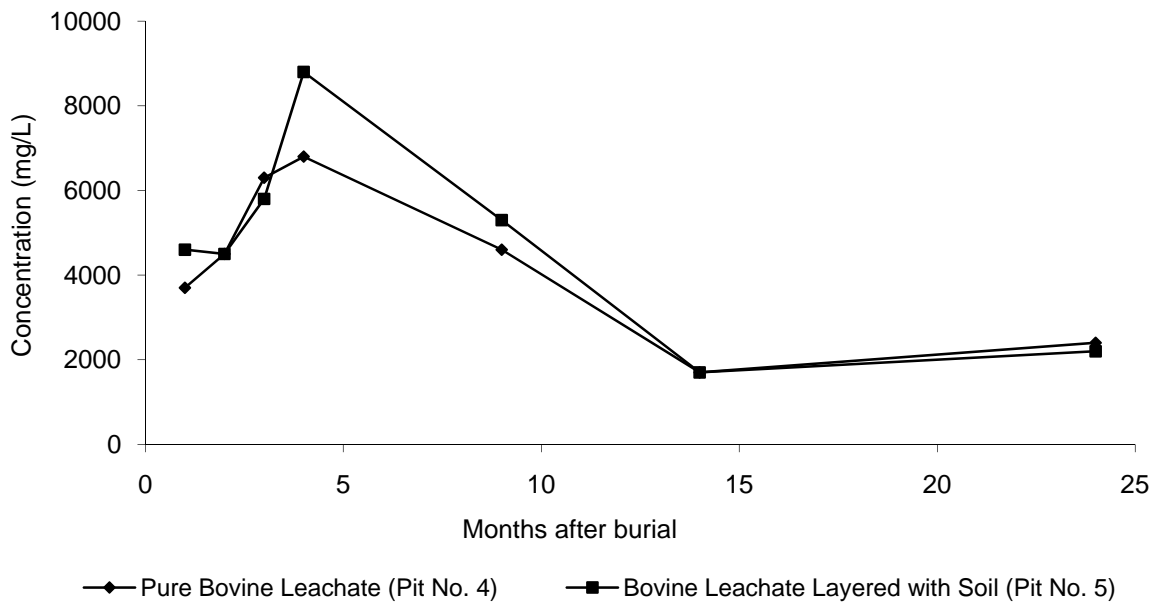
60

Figure 4.9 Sulphate concentrations for bovine – pure leachate vs. soil layering.

4.1.4 Sodium

Each species analyzed presents similar concentrations of sodium for all sample points. Bovine initially had the lowest concentration of sodium. After four months, sodium concentrations in bovine leachate reached 2,200 mg/L. Swine initially had the highest concentration of sodium at the first sample date at 2,300 mg/L. By the fourth month, the concentrations in the swine leachate followed the same trend as bovine with a concentration around 2,200 mg/L. Poultry began with a concentration of 1,900 mg/L, and slightly decreased to approximately 1,600 mg/L for the remainder of the sampling period. Average sodium concentrations expected from mortality leachate are approximately 1,800 mg/L.

For Pit No. 3 and 5, sodium concentrations in pure swine leachate and that of swine layered in soil follow the same general trend line over the sample period. Swine layered in soil showed slightly higher concentrations of sodium over the sample period and after 24 months, showed a difference of only 300 mg/L. In the case of pure bovine leachate and bovine layered in soil, they also follow each other quite closely. At 14 months, both sample sets had equal values.
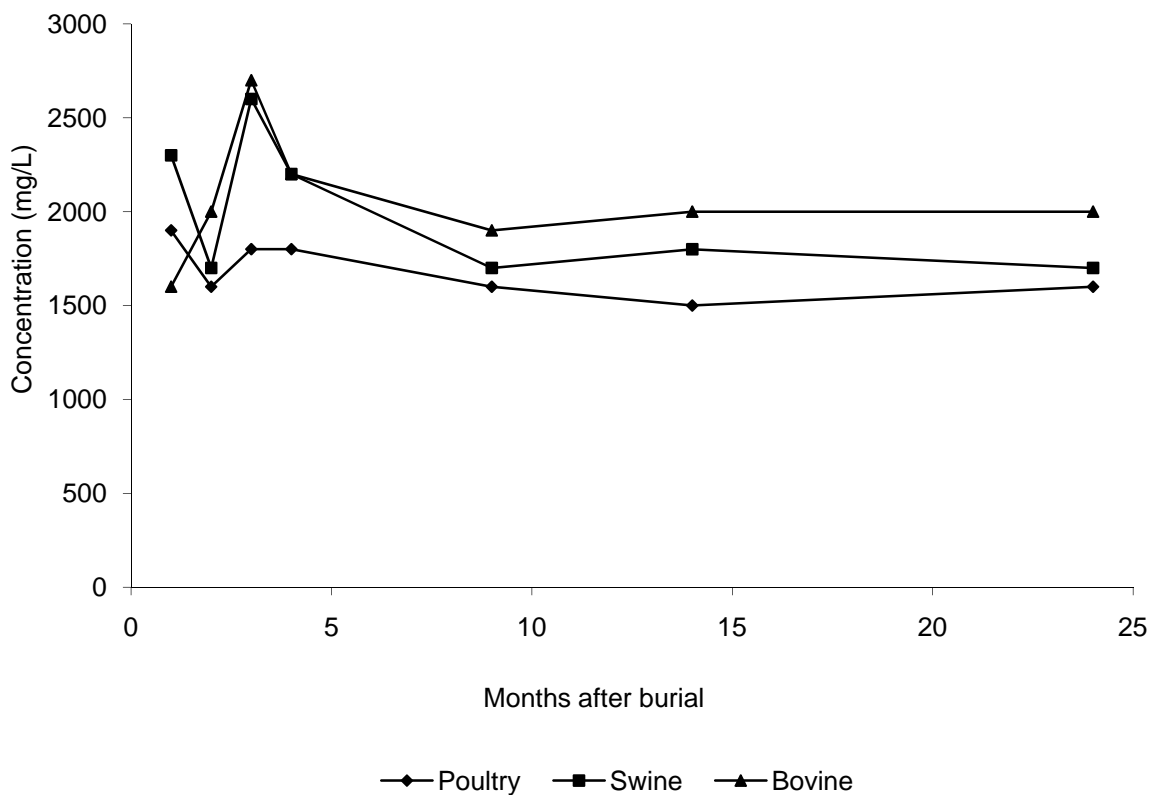
61

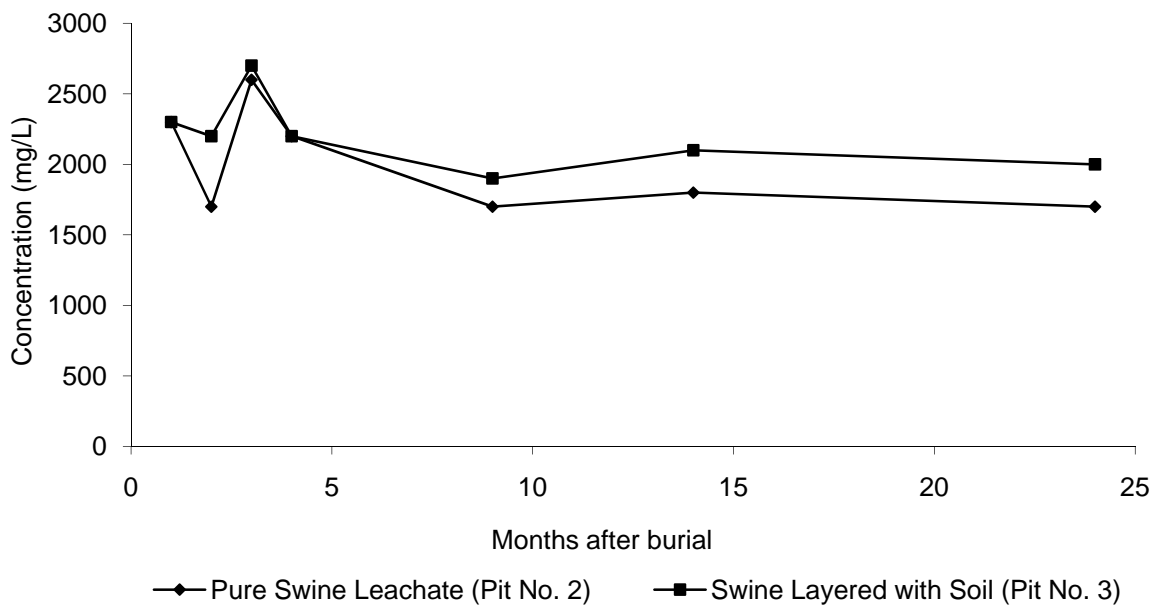Figure 4.10 Sodium concentrations pure leachate.

Figure 4.11 Sodium concentrations for swine – pure leachate vs. soil layering.
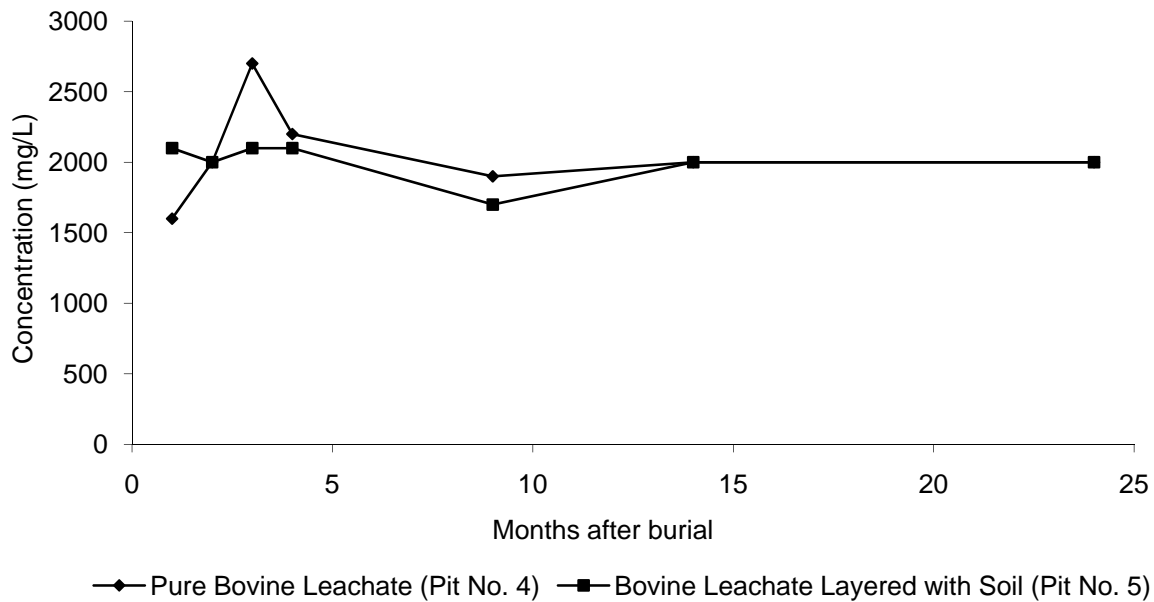
Figure 4.12 Sodium concentrations for bovine – pure leachate vs. soil layering.

4.1.5 Chloride

Chloride concentrations for each species generally follow the same trend and have similar concentrations for all sample dates (Fig. 4.13). Bovine reached the highest concentration at fourteen months with a chloride concentration of 3,810 mg/L. The results follow slightly different paths to the same end therefore are probably not statistically different. An average chloride value for livestock decomposition leachate was calculated as 2,600 mg/L.

Chloride concentrations for pure swine and swine layered in soil (Fig. 4.14) follow the same general trend. For the case of pure bovine leachate and bovine layered in soil (Fig. 4.15), the initial separation in concentrations of chloride was nearly 900 mg/L. After 24 months, pure bovine leachate had a higher concentration than that of bovine layered in soil for the entire sampling period. Due to the results of swine and bovine layered in soil following opposite paths, meaning swine layered with soil showed higher concentrations than pure swine leachate; and bovine layered in soil showed lower concentrations than pure bovine leachate; without more chemical analysis, the results appear to be not that different.
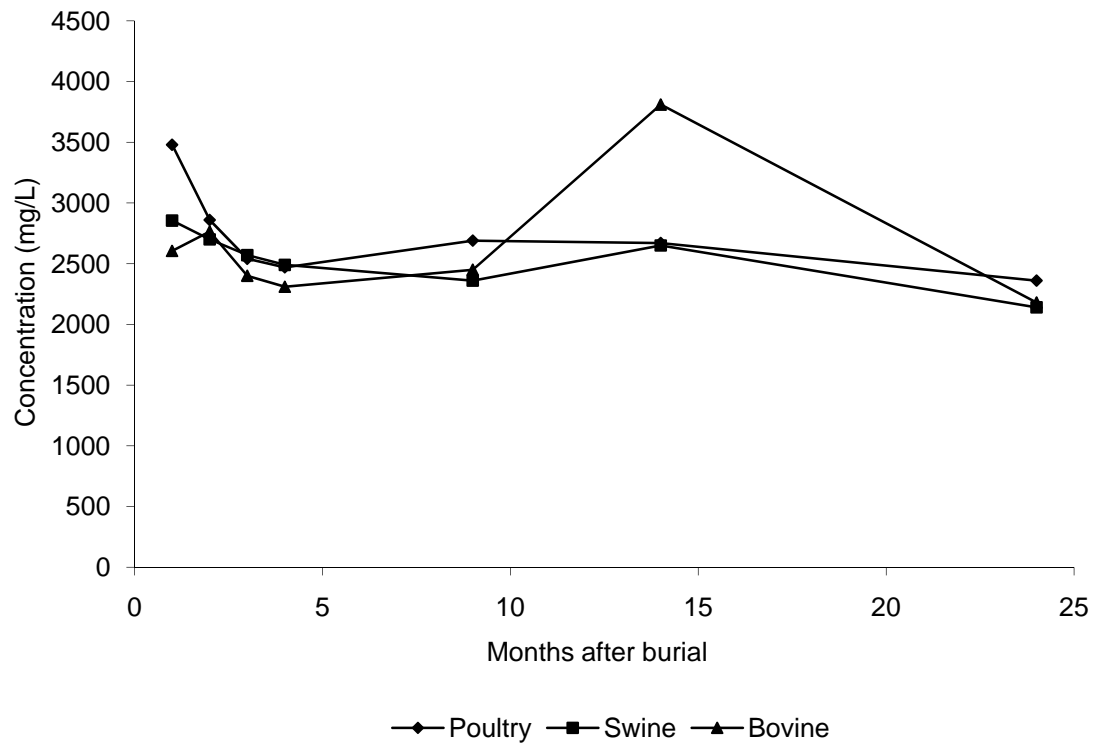
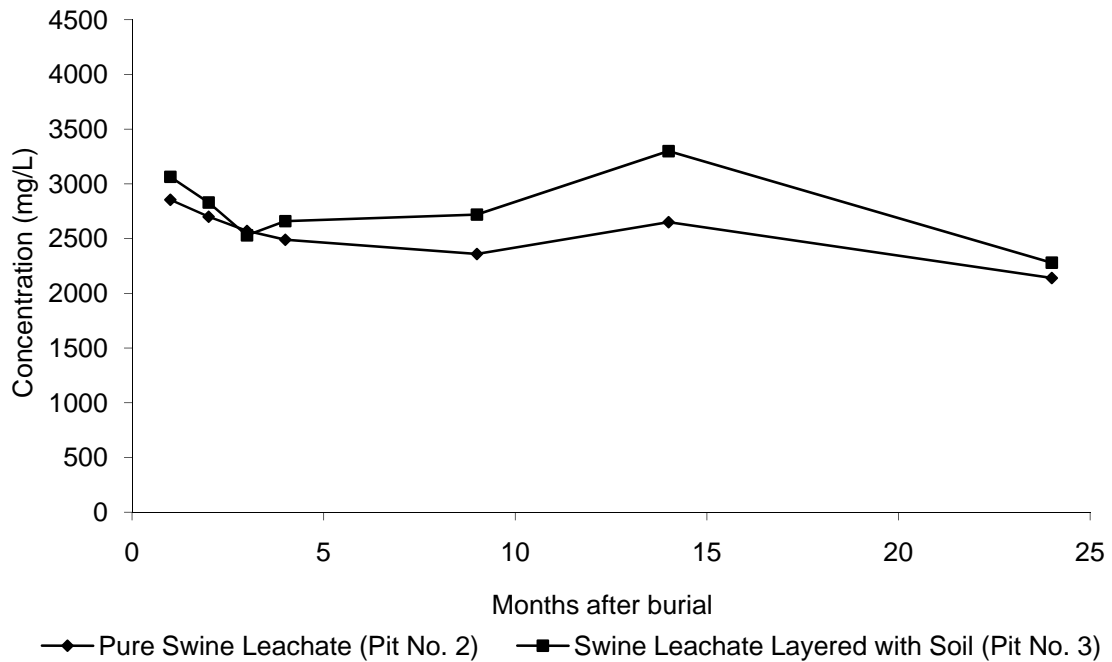Figure 4.13 Chloride concentrations of pure leachate.

Figure 4.14 Chloride concentrations for swine – pure leachate vs. soil layering.
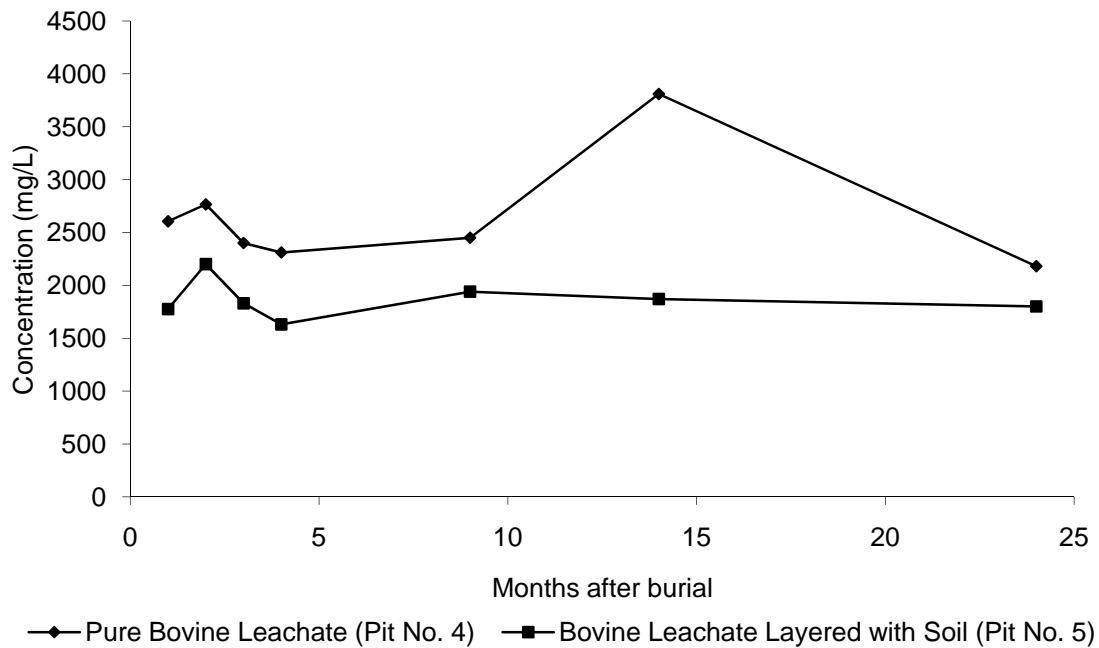
Figure 4.15 Chloride concentrations for bovine – pure leachate vs. soil layering.

4.1.6 Potassium

Potassium concentrations for each species did not fluctuate much during the 24 month sample period (Fig. 4.16). During the entire sample period, bovine had the lowest average concentration and poultry and swine had a slightly higher concentration. The breakdown of proteins could be a major source of potassium production in the leachate (Vass 2001). The average potassium value for the three species during the last three sample periods is 2,300 mg/L, with bovine approximately 25% lower than swine and poultry. Little effect was caused by soil layered in the swine and bovine pits (Fig. 4.17 and 4.18).
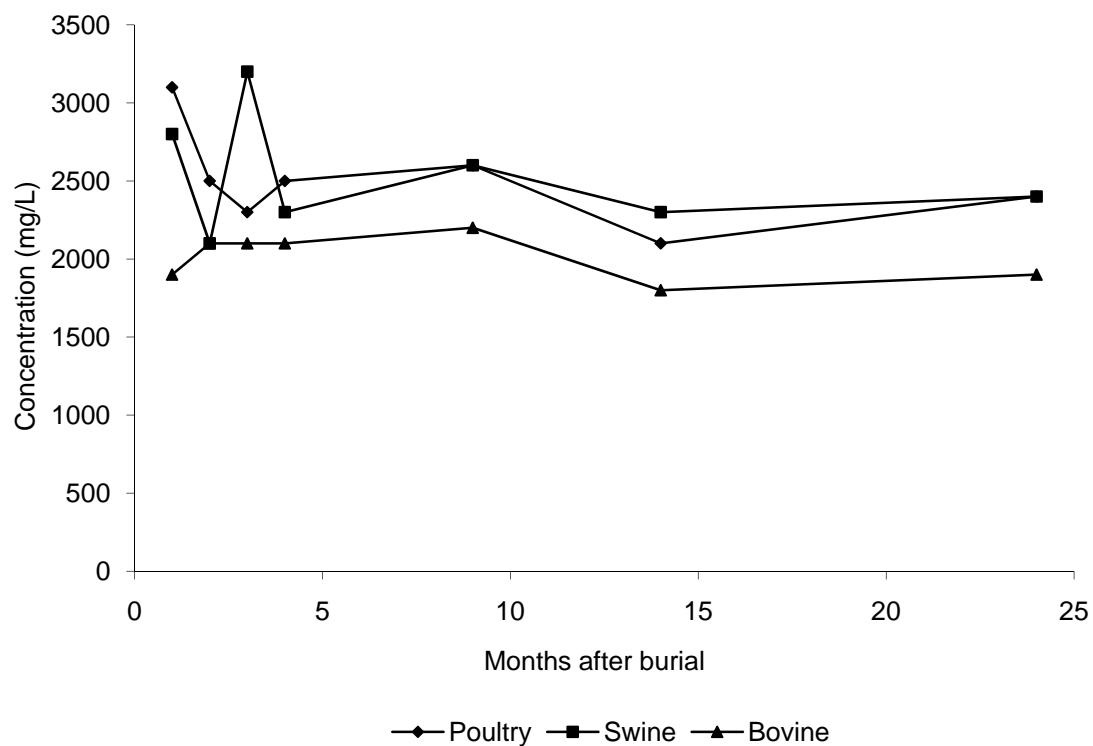
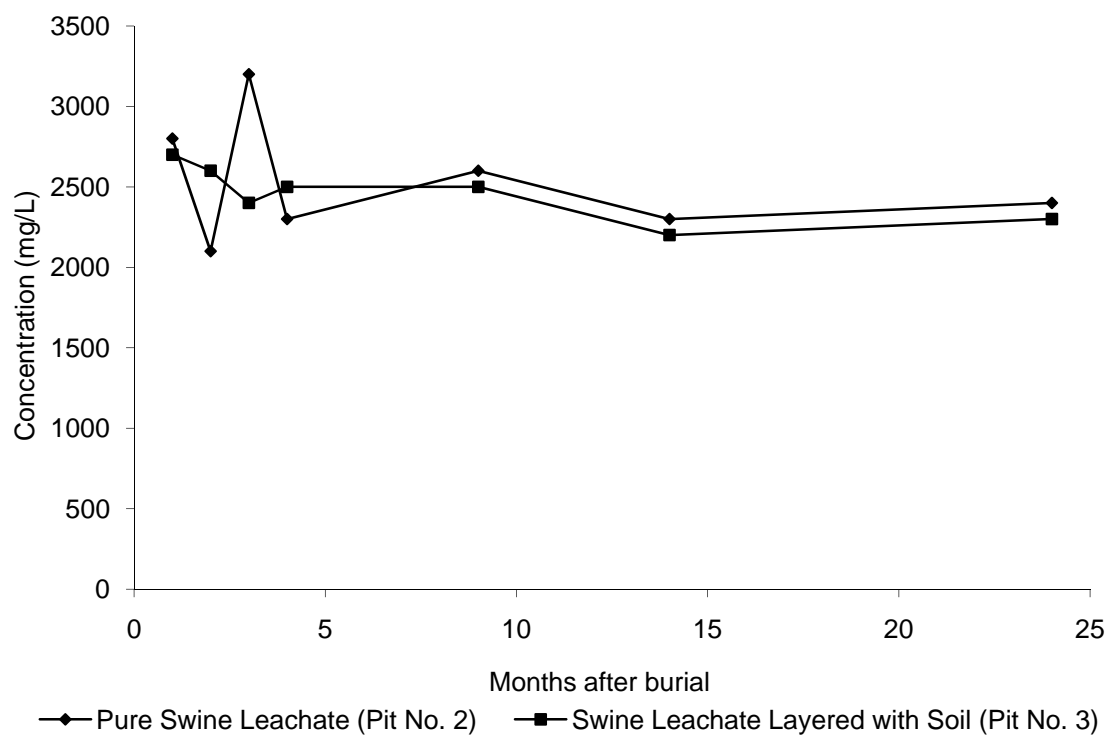Figure 4.16 Potassium concentrations of pure leachate.

Figure 4.17 Potassium concentrations for swine – pure leachate vs. soil layering.
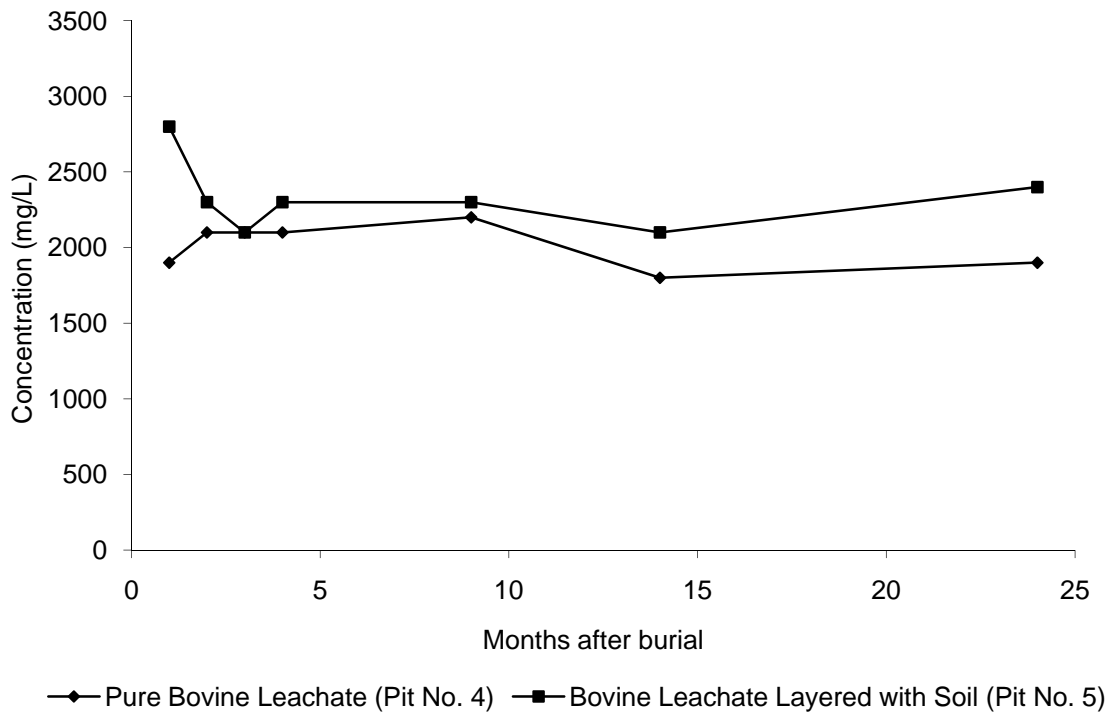

Pure Bovine Leachate (Pit No. 4) ━■━ Bovine Leachate Layered with Soil (Pit No. 5)

Figure 4.18 Potassium concentrations for bovine – pure leachate vs. soil layering.

4.1.7 Phosphorous

Phosphorous concentrations for each species generally follow the same trend of slight fluctuations in early sample dates to a levelled value by fourteen months. Poultry contained the highest amount of phosphorous with a maximum value of 2,090 mg/L. Swine had a maximum value of 1,870 mg/L and bovine the lowest at a maximum value of 1,300 mg/L. Differences in the values could possibly be explained by diet and biological attributes of the species. Poultry, in general have significantly smaller frame than bovine and swine, and due to the large amount of feathers have more protein structures that include phosphorous molecules. As for pure swine leachate and swine layered in soil, each sample date follows the same trend. Bovine also follow the same basic trend and are not statistically different. Average phosphorus concentrations range between 1,200 (poultry) and 1,800 mg/L (bovine). These values are significantly higher than those reported by MacArthur et al. (2002) who reported an average value of 55 mg/L and a peak concentration of 476 mg/L.
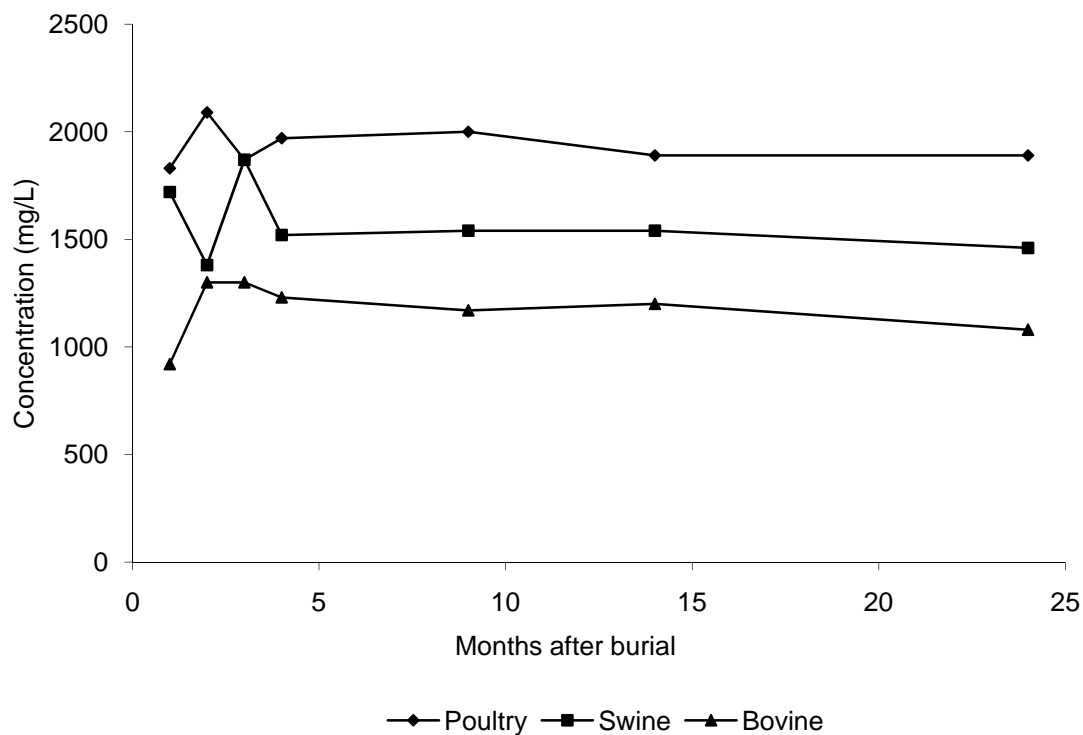
Figure 4.19 Phosphorus concentrations of pure leachate.
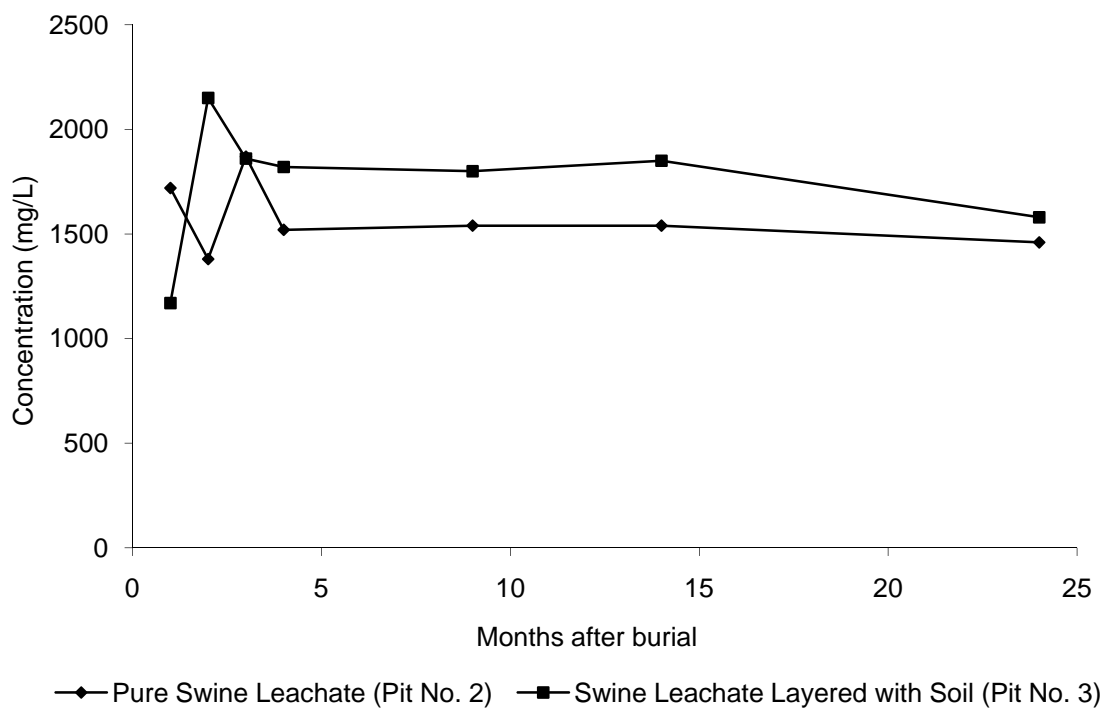
Figure 4.20 Phosphorus concentrations for swine – pure leachate vs. soil layering.
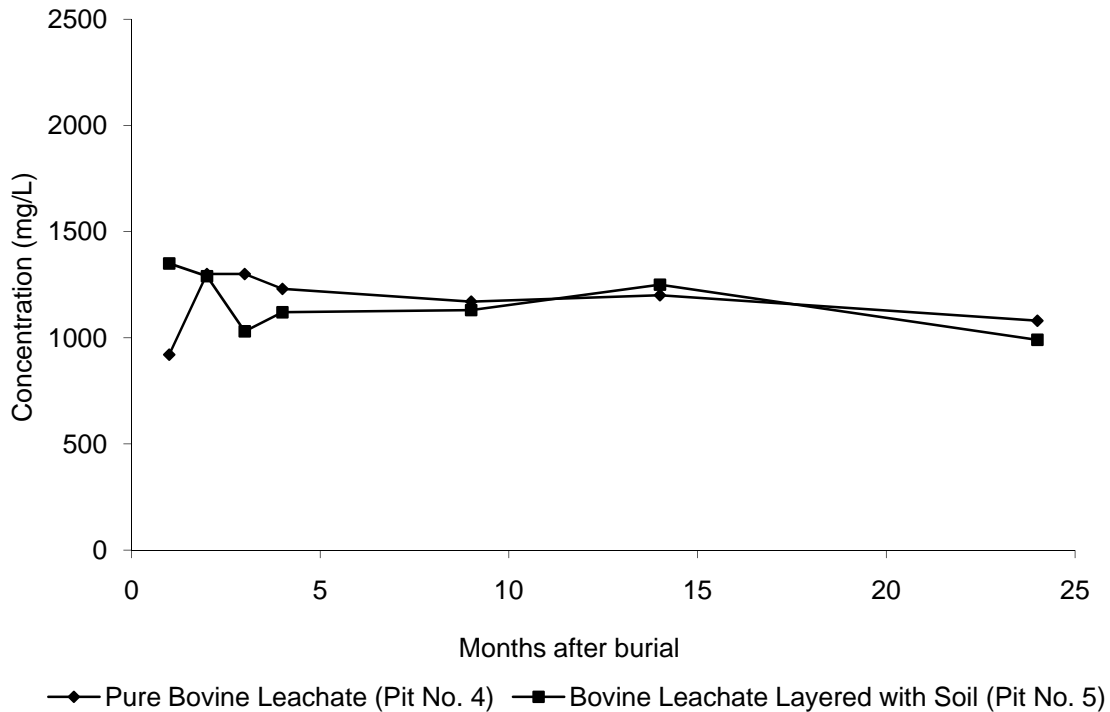
Figure 4.21 Phosphorus concentrations for bovine – pure leachate vs. soil layering.

4.1.7 Average Concentrations Present in Mortality Leachate

Due to the early rapid release of fluids from the livestock such as urine, digestive juices and blood, higher concentrations for most ions are not achieved until 6-9 months post-burial. If livestock are buried in a soil with low hydraulic conductivity, a solution will be present for long periods of time, allowing the accumulation of higher concentrations due to the decomposition of the animal carcass. From this, the expected concentrations of livestock mortality leachate should be averaged over the last three sample dates i.e. 9-24 months post burial. Table 4.6 demonstrated average concentrations expected per species and a total average which could be expected. Total alkalinity or bicarbonate is likely the value with the most uncertainty due to difficulties involved with titration of the original leachate solutions.

Table 4.6 Average mortality leachate concentrations per species and total.

	units	Poultry	Swine	Bovine	Total Average
Bicarbonate	mg/L	39133	48467	50733	46100
Chloride	mg/L	2570	2380	2813	2600
pH		6.5	6.7	6.9	6.7
Total Alkalinity	mg/L	22500	39700	41600	34600
Ammonium as Nitrogen	mg/L	10400	13300	14100	12600
Nitrate + Nitrite Nitrogen	mg/L	2.3	3.1	3.8	3.1
Inorganic Carbon	mg/L	7697	9533	9947	9100
Organic Carbon	mg/L	79000	65000	68000	71000
Aluminum	mg/L	0.5	0.5	0.5	0.5
Calcium	mg/L	81	48	36	60
Copper	mg/L	0.9	1.7	0.6	1.1
Iron	mg/L	18	19	18	20
Magnesium	mg/L	79	17	18	40
Manganase	mg/L	0.5	0.1	0.1	0.2
Phosphorus	mg/L	1927	1513	1150	1500
Potassium	mg/L	2400	2400	2000	2300
Silicon, soluable	mg/L	20	24	26	20
Sodium	mg/L	1600	1700	2000	1800
Sulfate	mg/L	3970	3900	2900	3600
Sulfur	mg/L	1300	1297	963	1200
Zinc	mg/L	2.2	1.8	1.7	1.9

4.2 Leachate Characterization

The characterization of livestock mortality leachate, will aid regulators and scientists in their efforts to properly assess sites suitable for livestock burial and the parameters involved in doing so. To determine the important characteristics of mortality leachate, thermodynamic speciation was performed on average concentrations of constituents present in the leachate samples, evaluation of potential leaching, as well as comparison to existing known contaminants.

4.2.1 Chemistry Significance by Speciation

In order to assess chemical species present in the leachate sample, speciation was performed using PHREEQC (Parkhurst and Appelo 1999). This analysis helped to estimate what chemical species are present along with their respective activity in solution. Average concentrations over the last three sample points of the burial pits were used as initial concentrations for speciation shown in Table 4.7. Speciation of the mortality leachate chemistry produced the following

results shown in Table 4.8. The calculated charge balance by PHREEQC on this solution is approximately eight percent. By adjusting alkalinity the charge balance percentage could be lowered to less than one percent. Due to the nature of titration for determining alkalinity, it is possible for these concentrations in particular to have errors. The speciation was then compared to a speciation of naturally occurring groundwater from Fonstad (2004). Groundwater concentrations used in this simulation originate south of Saskatoon in a glaciolacustrine deposit typical of Saskatchewan and are shown in Table 4.7.

Table 4.7 Average mortality leachate and groundwater concentrations (Fonstad 2004) used in PHREEQC.

Input Parameter	Units	Mortality Leachate	Groundwater
pH		6.7	7.4
Ammonium	mg/L	12600	0
Alkalinity	mg/L	34600	253
Sulphate	mg/L	3600	82
Sodium	mg/L	1800	29
Chloride	mg/L	2600	29
Potassium	mg/L	2300	66
Phosphorus	mg/L	1500	0.37
Iron	mg/L	20	0.55
Calcium	mg/L	60	52
Magnesium	mg/L	40	25
Silicon	mg/L	20	6
Zinc	mg/L	2	0
Copper	mg/L	1	0

The results for phosphorus are consistent with that of Moore et al. (2002) showing that phosphorus is mainly present in the body in the form of phosphoric acids and at this pH, it is proportioned approximately 60% and 40% and hydrogen phosphate and dihydrogen phosphate respectively. These phosphoric acids are then attracted to cations Fe^{2+}, Ca^{2+}, Mg^{2+}, K^+ and Na^+ consistent with Stollenwerk (1996); and Weiskel and Howes (1992). The mortality leachate also shows oversaturation of hydroxyapatite and vivianite consistent with Zanini et al (1998) as well as Stollenwerk (1996); and Weiskel and Howes (1992) (Table 4.8). Phosphorus present as phosphate compounds allows for unattenuated transport of phosphorus as well as ions bound to the phosphate due to the negative charge on the ion. Phosphoric acids are corrosive and

available for unattenuated transport on their own, but are able to easily complex with metals in the soil therefore advancing the transport of metals through the system (Stollenwerk 1996). Sulphate is found mostly in the form of ammonium sulphate, and sulphate. It also forms ion complexes with Ca^{2+}, Mg^{2+}, K^+ and Na^+ consistent with Hem (1992). Sulphate complexes such as KSO_4^- and $NaSO_4^-$ allow for unattenuated transport of potassium and sodium as well as sulphate. Nitrogen is found commonly in the leachate as NH_4^+, with some ammonium sulphate salts forming along with small amounts of nitrate. Due to the decomposition of livestock mortalities, nitrogen is leached in its most reduced state, ammonium, and without exposure to oxidizing conditions; the majority of nitrogen will remain as ammonium. Ammonium sulphate concentrations in solution are approximately 2,400 mg/L which is approximately 300 mg-N/L; the negative charge on this ion allows for the transport of unattenuated nitrogen. Chloride in leachate is present only in the form of Cl^-, consistent with Ham (1992). Calcium in this solution is present as calcium-bicarbonate, calcium-phosphates, and calcium-sulphates, as well as some free calcium consistent with Hem (1992). Iron is mostly present as ferrous iron and the majority is complexed with phosphates and bicarbonate; while magnesium is complexed with bicarbonate, sulphate and phosphate with small amounts of free magnesium. Eighty percent of sodium in solution is available as free sodium, while the remainder is complexed with bicarbonate, sulphate and phosphate consistent with Hem (1992). Potassium is mostly present as free potassium, with small amounts of ion complexes with sulphate and phosphate.

It is observed that the speciated concentrations when back calculated to mg/L are at slightly higher values than the input. This is explained due to the assumption that the density of the leachate is the same as pure water, 1000 kg/m^3. It appears that this assumption is incorrect, therefore causing the calculations to show slightly higher concentrations. It also could be explained by the activity of the water molecules in solution. Due to the solution's high ionic strength, many of the water molecules are bound with the ions in solution forming their hydrated radii. Due to this phenomena, PHREEQC reports values in molality (mol/kg solution), therefore the concentration of active water in that kg of solution is less than 1 kg/L.

Table 4.8 PHREEQC Speciation of averaged livestock mortality leachate chemistry concentrations.

Species	Molality	mg/l	Activity	Species	Molality	mg/l	Activity
C(4)	**8.67E-01**			MgH2PO4+	6.68E-05	8.1	5.33E-05
HCO3-	6.89E-01	41997	4.50E-01	MgSO4	6.38E-05	7.7	8.10E-05
CO2	1.65E-01	7256	2.09E-01	**N(-3)**	**9.56E-01**		
NaHCO3	1.02E-02	859	1.30E-02	NH4+	9.34E-01	13080	4.88E-01
CaHCO3+	9.03E-04	91	5.90E-04	NH4SO4-	2.07E-02	2260	1.65E-02
MgHCO3+	8.67E-04	74	6.91E-04	NH3	1.10E-03	19	1.39E-03
CO3-2	5.80E-04	35	1.06E-04	**Na**	**8.32E-02**		
Ca	**1.59E-03**			Na+	7.15E-02	1644	5.13E-02
CaHCO3+	9.03E-04	91	5.90E-04	NaHCO3	1.02E-02	859	1.30E-02
Ca+2	4.15E-04	17	1.03E-04	NaSO4-	8.49E-04	101	6.77E-04
CaHPO4	1.72E-04	23	2.18E-04	NaHPO4-	4.86E-04	58	3.88E-04
CaSO4	4.25E-05	5.8	5.40E-05	**P**	**5.15E-02**		
CaH2PO4+	4.11E-05	5.6	3.28E-05	HPO4-2	2.99E-02	2869	3.87E-03
CaCO3	1.44E-05	1.4	1.83E-05	H2PO4-	2.01E-02	1944	1.25E-02
Cl	**7.80E-02**			NaHPO4-	4.86E-04	58	3.88E-04
Cl-	7.80E-02	2763	4.66E-02	KHPO4-	3.47E-04	47	2.77E-04
Fe(2)	3.81E-04			MgHPO4	2.97E-04	36	3.77E-04
FeHCO3+	2.57E-04	30	2.04E-04	CaHPO4	1.72E-04	23	2.18E-04
FeHPO4	5.52E-05	8.4	7.01E-05	**S(6)**	**3.98E-02**		
FeH2PO4+	3.56E-05	5.4	2.84E-05	NH4SO4-	2.07E-02	2362	1.65E-02
Fe+2	2.23E-05	1.2	4.54E-06	SO4-2	1.73E-02	1660	2.63E-03
FeCO3	9.08E-06	1.1	1.15E-05	KSO4-	8.50E-04	115	6.78E-04
FeSO4	1.67E-06	0.3	2.13E-06	NaSO4-	8.49E-04	101	6.77E-04
K	**6.25E-02**			MgSO4	6.38E-05	7.7	8.10E-05
K+	6.13E-02	2391	3.67E-02	CaSO4	4.25E-05	5.8	5.40E-05
KSO4-	8.50E-04	115	6.78E-04	**Si**	**3.54E-04**		
KHPO4-	3.47E-04	47	2.77E-04	H4SiO4	3.53E-04	34	4.49E-04
Mg	**1.75E-03**			**Zn**	**3.25E-05**		
MgHCO3+	8.67E-04	74	6.91E-04	Zn(CO3)2-2	1.81E-05	3.4	7.30E-06
Mg+2	4.38E-04	11	1.31E-04	ZnHCO3+	1.09E-05	1.4	8.67E-06
MgHPO4	2.97E-04	36	3.77E-04	ZnCO3	2.54E-06	0.3	3.23E-06

The activities of the ions in solution vary by one to two orders of magnitude. Important ion complexes in Table 4.8 are plotted in Figure 4.22 on a log scale to show activities for each species present in mortality leachate and compared to species present in a sample solution of groundwater from Fonstad (2004). Ammonium and bicarbonate are an order of magnitude higher than activities such as chloride, potassium and sodium in mortality leachate. In livestock mortality leachate, ions such as ammonium and bicarbonate have the highest activities and make up 97% and 80% of their respective complexes. With respect to activities of species

found in groundwater, bicarbonate activity is two orders of magnitude higher. The representative sample of groundwater used did not contain any ammonium. For the remaining ions shown in Figure 4.22, all are one to five orders of magnitude higher than natural groundwater. It is interesting to note the activities of ammonium sulphate and phosphoric acids are similar, but in regards to concentration, there is less ammonium sulphate in solution.
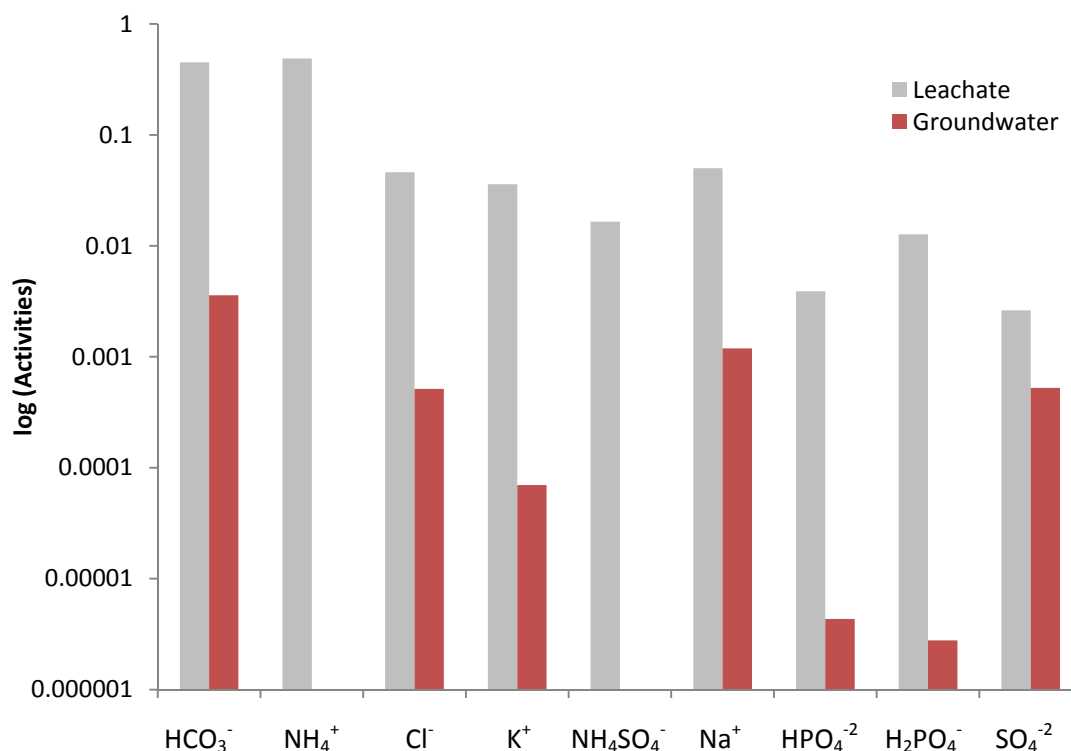

Figure 4.22 Activities of selected ions in livestock mortality leachate compared to groundwater.

Comparing livestock mortality leachate with groundwater in respect to molalities present in solution, leachate is higher by orders of magnitude (Fig. 4.23). The transport of ammonium will be attenuated by the soil, but the transport of ammonium sulphate may be unattenuated. The attenuated transport of ammonium is still important and a threat due to its relatively high initial concentration. If ammonium sulphate in mortality leachate travels through the soil, it has the potential to carry 300 mg/L of nitrogen unattenuated, still well above regulated limits for nitrogen in drinking water. Molalities of chloride, potassium and sodium are 1.5 to 2.5 orders of magnitude higher than concentrations occurring in the groundwater sample and exceed regulated drinking water standards by 11 and 8 times respectively. Potassium will likely

be attenuated while chloride and sodium are not as likely to be attenuated. Comparing Fig. 4.22 to Fig. 4.23 it is interesting to note the plot of molalities for phosphoric acid and sulphate are an order of magnitude higher than their subsequent activities plot, meaning their effective concentration in solution is greater.
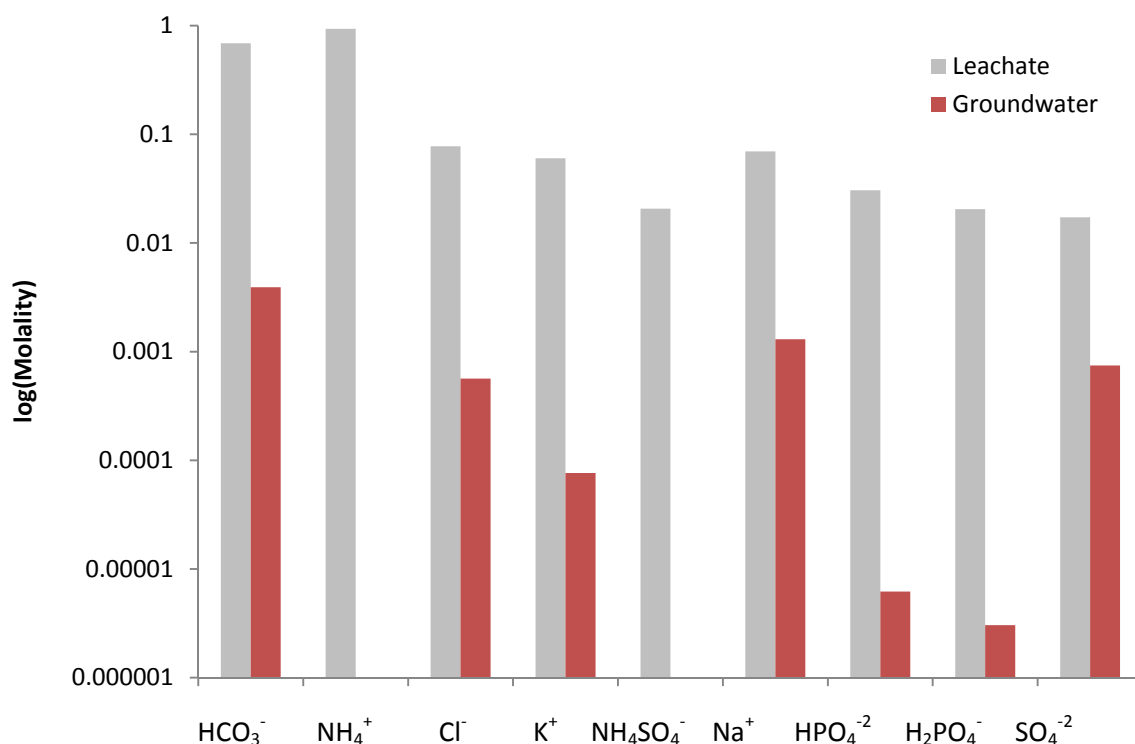

Figure 4.23 Molalities of important ions in livestock mortality leachate compared to groundwater.

4.2.2 Chemistry Significance and Comparison

Livestock mortality leachate is relatively highly concentrated compared to other contaminants such as hog manure, and landfill leachate. Figures 4.24 (mg/L) and 4.25 (meq/L) show the difference between the results of this study to other potential contaminants such as earthen manure storages, landfill leachate, and the Scotland burial site study whose concentration values were taken from Tables 2.1 and 2.4.

The results of this study show that ammonium concentrations range from 2 to sometimes 4

times higher than hog manure and landfills. The highest concentration exceeds drinking water standards over 400 times. The threat occurs with this concentration when it reaches oxic conditions causing the ammonium to oxidize and form nitrate (Paul and Clark 1996). It is significant that the maximum ammonium concentration values found in this study match those values obtained from the mortality burial site in Scotland. Alkalinity in livestock mortality leachate is 60 times higher than drinking water standards and also exceeds concentrations in hog manure and landfill leachate by 20,000 mg/L. Alkalinity concentrations found at the Scotland site has a maximum value of 88,200 mg/L but average values were at approximately 12,000 mg/L. Average alkalinity concentrations from this study are nearly 20,000 mg/L higher than averages from Scotland. Sodium concentrations exceed drinking water standards by ten times and is also higher in concentration than hog manure and landfill leachate. Sulphate occurring in livestock mortality leachate is approximately 6-7 times higher than concentrations occurring in hog manure and landfill leachate and exceeds drinking water standards by approximately nine times. Phosphorus is not regulated in drinking water standards, but it can be detrimental to aquatic environments even in small amounts. Potassium concentration in livestock mortality leachate exceeds concentrations of hog manure and landfill leachate. Calcium and magnesium present in the leachate is less than hog manure and landfill leachate but calcium exceeds drinking water standards by 47 times, while magnesium is less than drinking water standards and is therefore not a threat. Chloride exceeds drinking water standards by ten times and is also higher in concentration than hog manure and landfill leachate. Chloride will transport conservatively and mostly unattenuated. Iron is similar for livestock mortality leachate and landfills, but can exceed drinking water standards by 120 times. Iron in solution is likely to complex with other ions or be adsorbed by the soil particles.
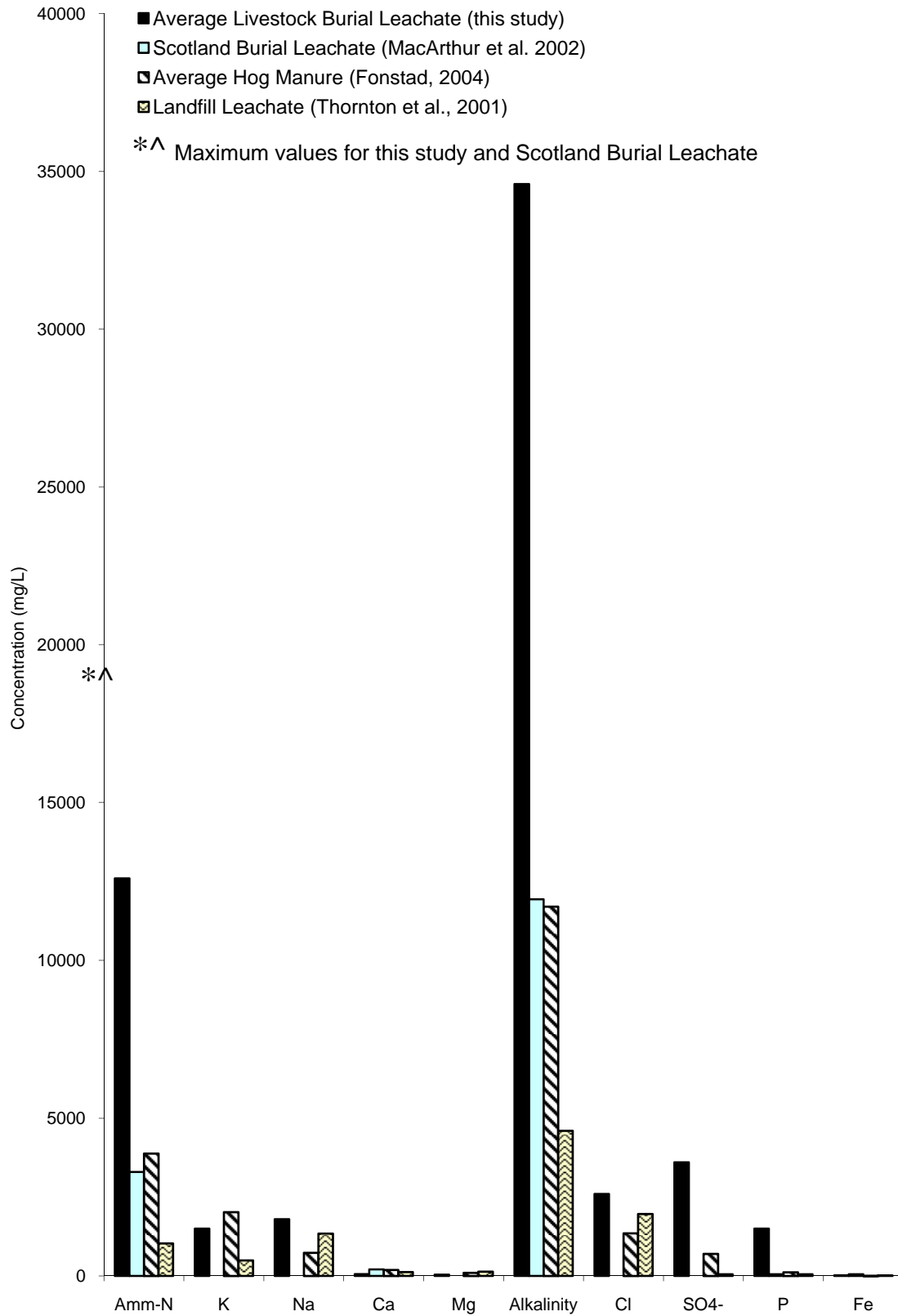
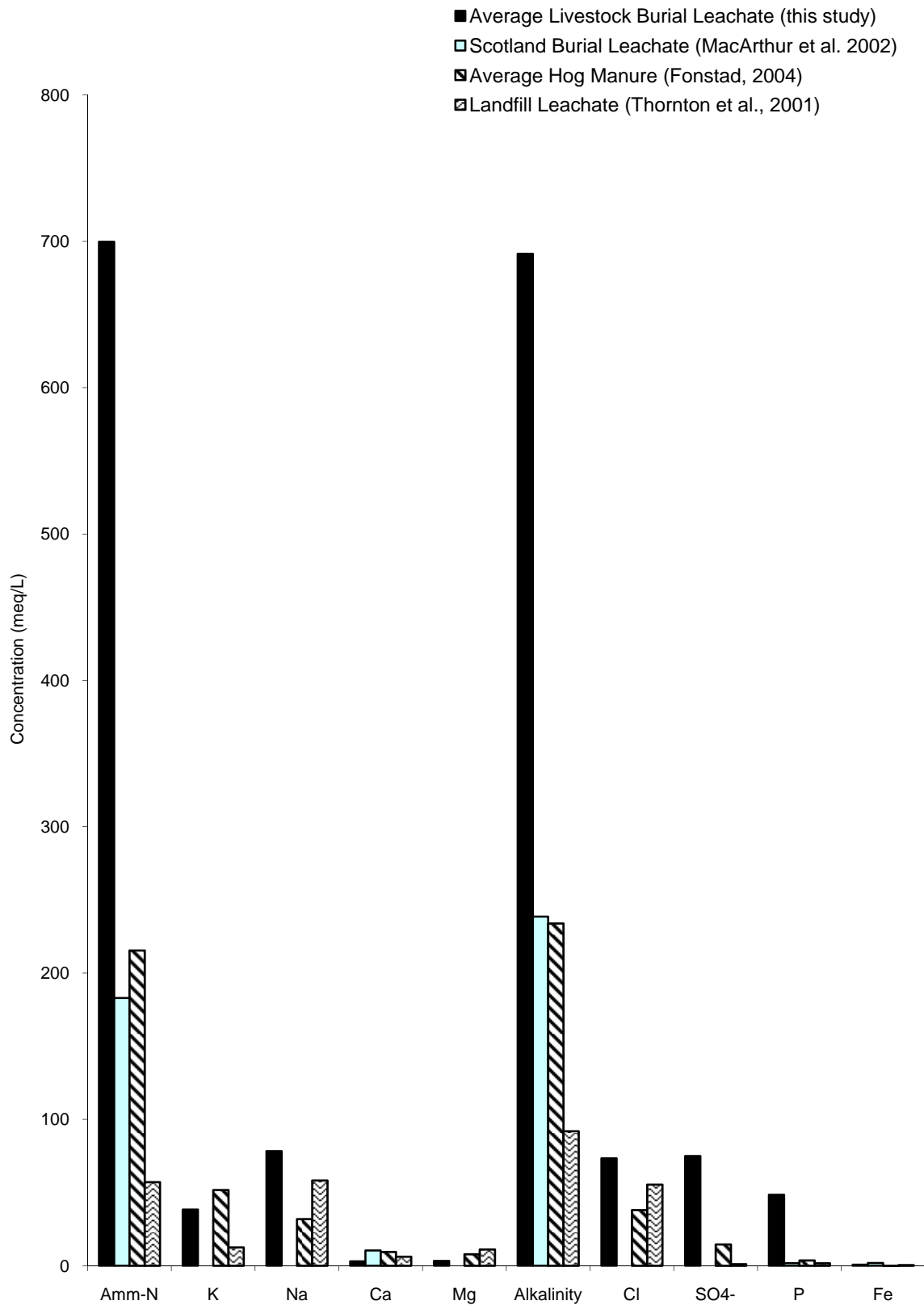Figure 4.24 Leachate concentration comparisons (mg/L).

Figure 4.25 Leachate concentration comparisons (meq/L).

4.3   CTRAN Results

This section presents the results found using the CTRAN modeling software. This exercise was done to illustrate the impact of the relatively high concentrations of ions present in the mortalities leachate. The Geo-Slope software was used to give graphical representations of a simple transport model.   Concentration profiles are presented as a ratio of computed concentration versus initial concentration (C/Co). The initial concentration was assumed to be an average ammonium-N value of 15,000 mg/L with no attenuation or adsorption included in the model.   Due to the variance in averages between species shown in Table 4.6, a round number of 15,000 made for easier analysis and evaluation.   Boundary conditions were specified in section 3.2.1.2. Three models were created to simulate transport through three soils defined earlier as low permeable, moderately permeable and permeable soil.

The concentration profiles for a low permeable soil (Fig. 4.26) show that at 50 years, 10% of the initial concentration has reached a depth of 3 meters. The concentration profile for a moderately permeable soil shown in figure 4.27 depicts 10% of the initial concentration will reach a depth of 10 meters in 50 years. The permeable soil simulation shown in Figure 4.28, demonstrates a relatively high transport velocity of the contaminant. At ten years, the contaminant has gone just as far as it had in 50 years in a moderately permeable soil. The highly permeable soil allows for rapid advection and dispersion causing wide horizontal and vertical spread. This is noticed later on in Figure 4.31. Due to its ability to spread horizontally, the concentrations at depth are lower than that of the moderate and low permeable soils.
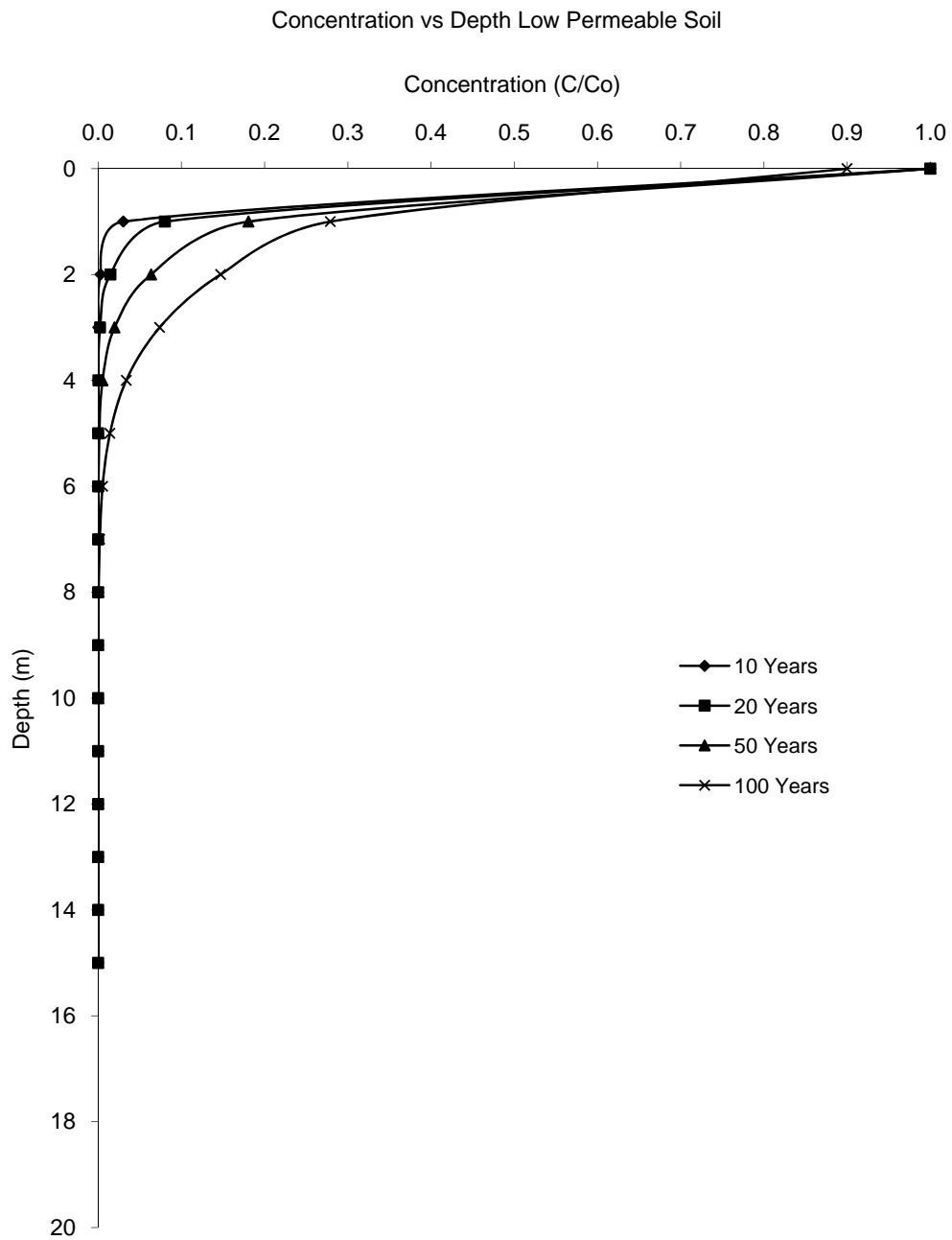
Figure 4.26 Concentration profile using CTRAN for a low permeable soil up to 100 years.

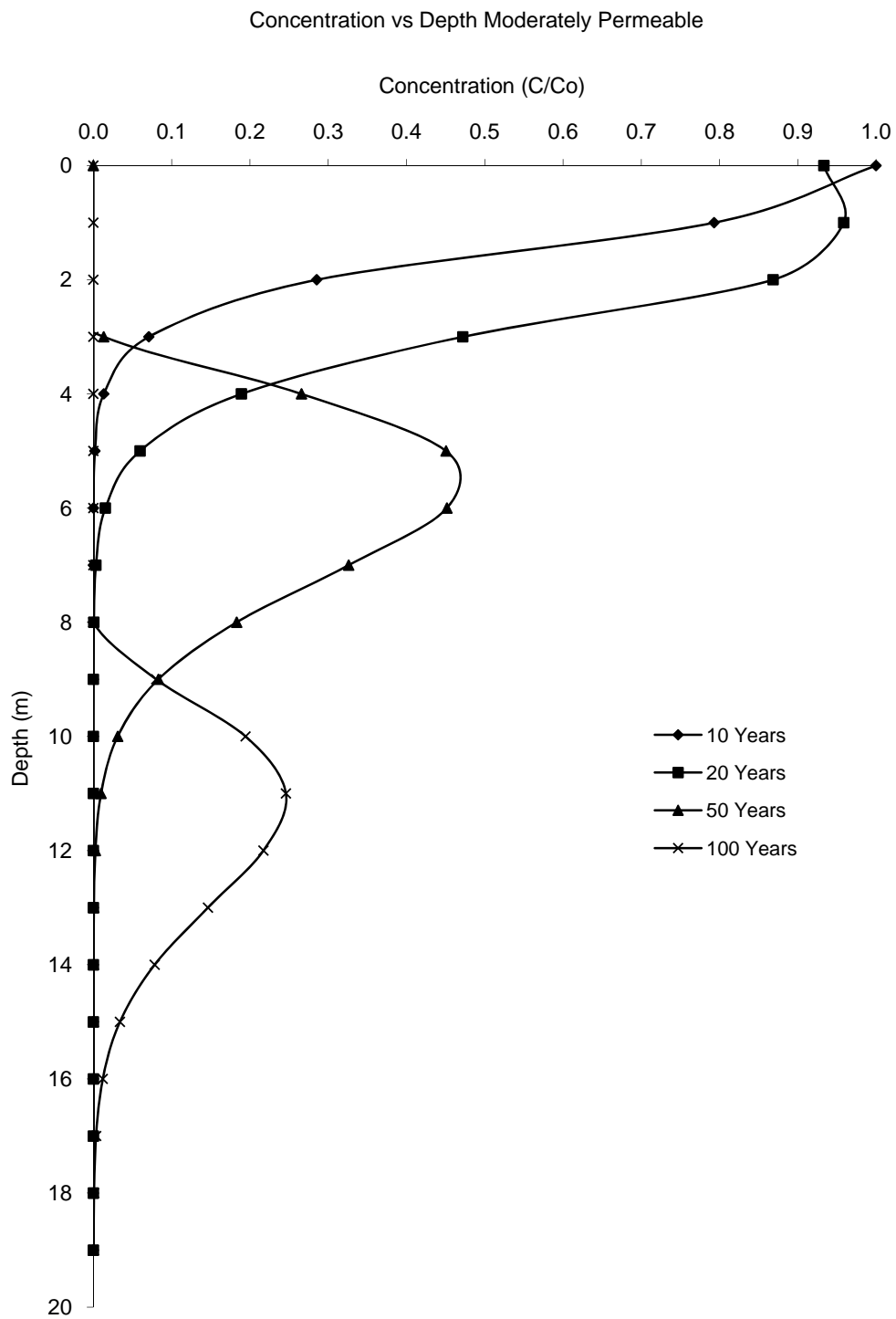Concentration vs Depth Moderately Permeable

Figure 4.27 Concentration profile using CTRAN for a moderately permeable soil up to 100 years.
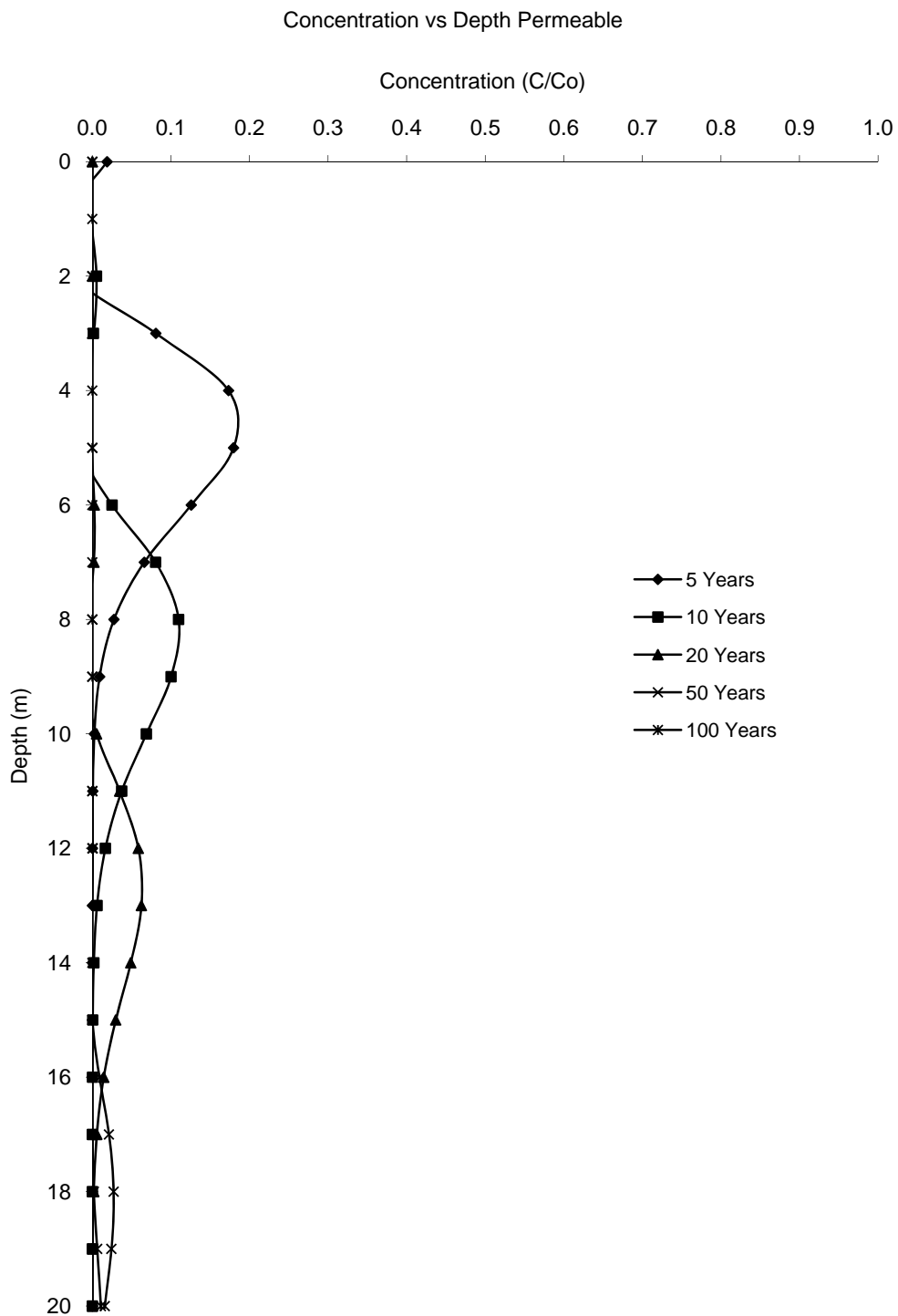
Figure 4.28 Concentration profile using CTRAN for a permeable soil up to 100 years.

The graphical interface of the CTRAN software package provides cross sectional profile plots. Results for low, moderate and permeable soils are shown in Figures 4.29 to 4.31. The models for low permeable soil show minimal transport of the contaminant after 10 years. Even after 100 years, the plume has only reached 4 meters at a concentration of 500 mg/L which is approximately 3% of the initial concentration. For a moderately permeable soil, (Fig. 4.30) the contaminant plume reaches approximately 6 meters with a concentration of 6,000 mg/L in a 50 year time period, which is approximately 40% of initial concentration. In the permeable soil, 20% of the initial concentration has transported approximately ten meters after only ten years. Horizontal transport is also displayed on the plots, with the low permeable soil having the smallest horizontal bulb with an 8 meter diameter from the center of the pit to each side after 100 years and the permeable soil having the largest with an 8 meter diameter after only ten years.
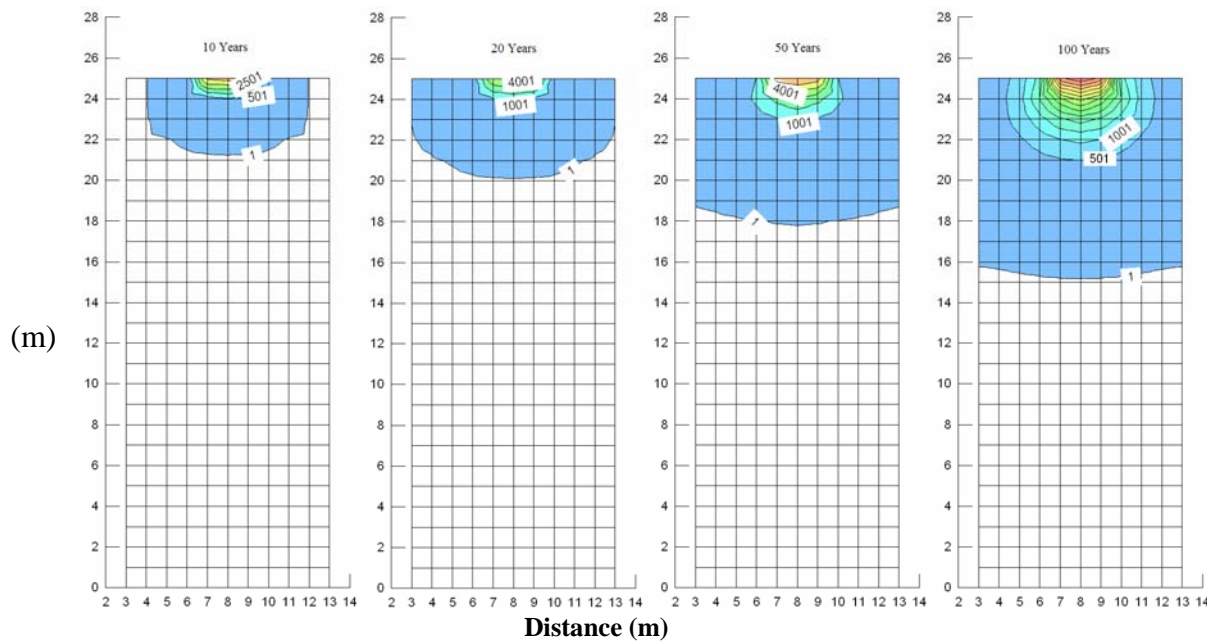

Figure 4.29 Low permeability soil, soil cross sectional transport view provided by CTRAN.
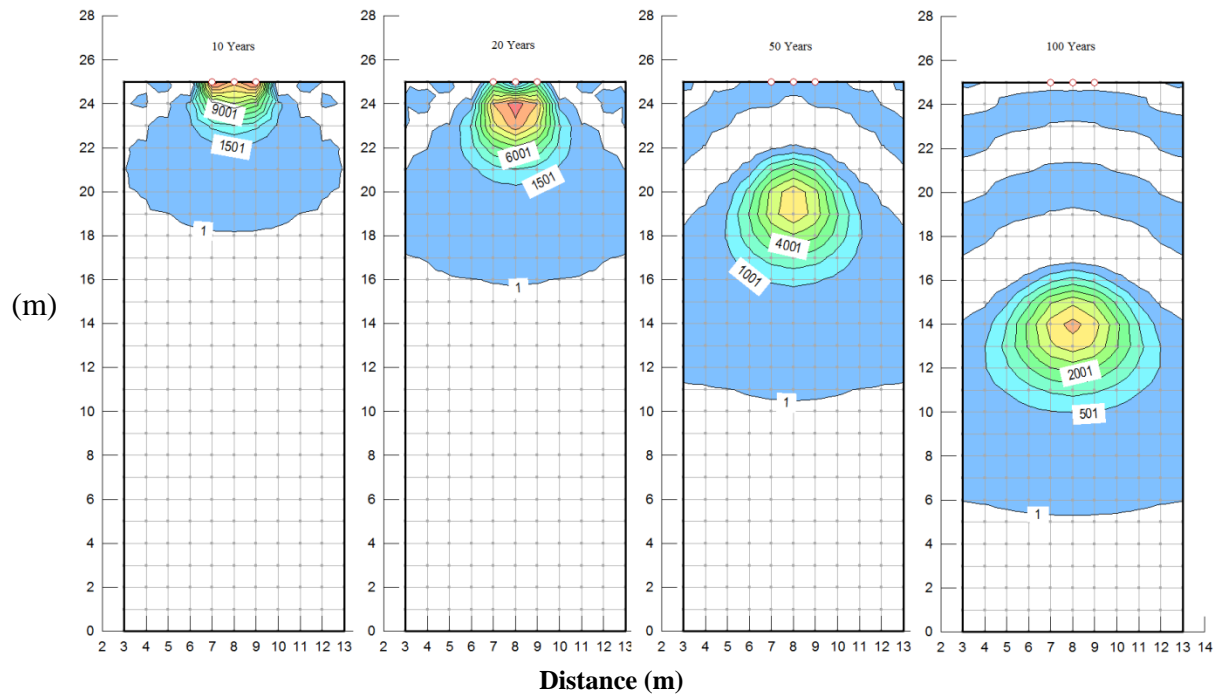
Figure 4.30 Moderately permeable soil, cross sectional transport view provided by CTRAN.
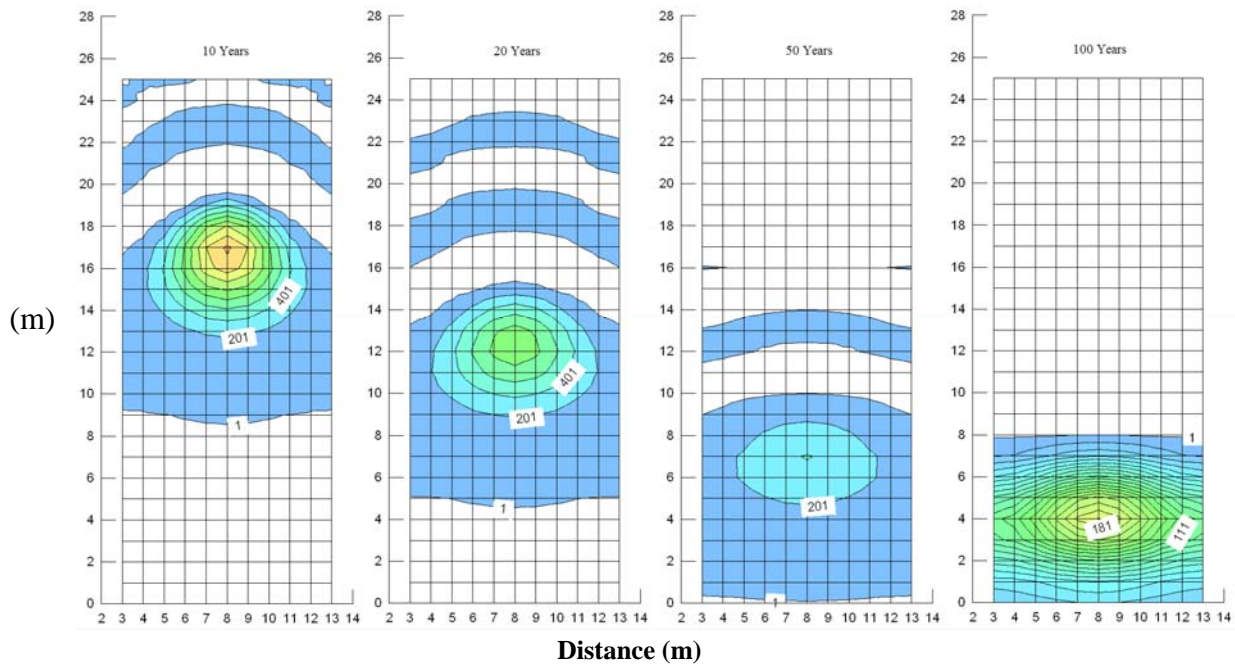

Figure 4.31 Highly permeable soil, cross sectional transport view provided by CTRAN.

Simulations were also created to evaluate the impact of multiple 2 meter wide trenches side by side. Boundary conditions used were the same as those for the moderately permeable soil simulation. Two spacings were used in these models. Figure 4.32 shows four trenches, spaced five meters apart with fifty years of transport. Figure 4.33 shows ten meter spacing for fifty years of transport. For the five meter spacing scenario, within fifty years, each burial trench's leachate is combining with the trench beside it. A one meter thick zone of approximately 2,000 mg/L (13% of initial concentration) exists in-between the pits after fifty years. Surrounding that zone is a concentration of 1,000 mg/L (7% of initial concentration), while the bottom of the joint plume of all the burial trenches is connected with a concentration of approximately 500 mg/L. The risk associated with joining plumes determines the zone of influence of the burial site, and the goal would be to eliminate the possibility of these plumes joining. Figure 4.33 demonstrates ten meter spacing between burial trenches. A concentration of 1 mg/L or 0.006 percent of initial concentration joins each burial trench. At fifty years, the 500 mg/L zone is still separated by a distance of 4 meters. In order to expose the burial leachate to its maximum potential for adsorption onto soil particles, a significant separation distance of at least 10 meters should be in place.
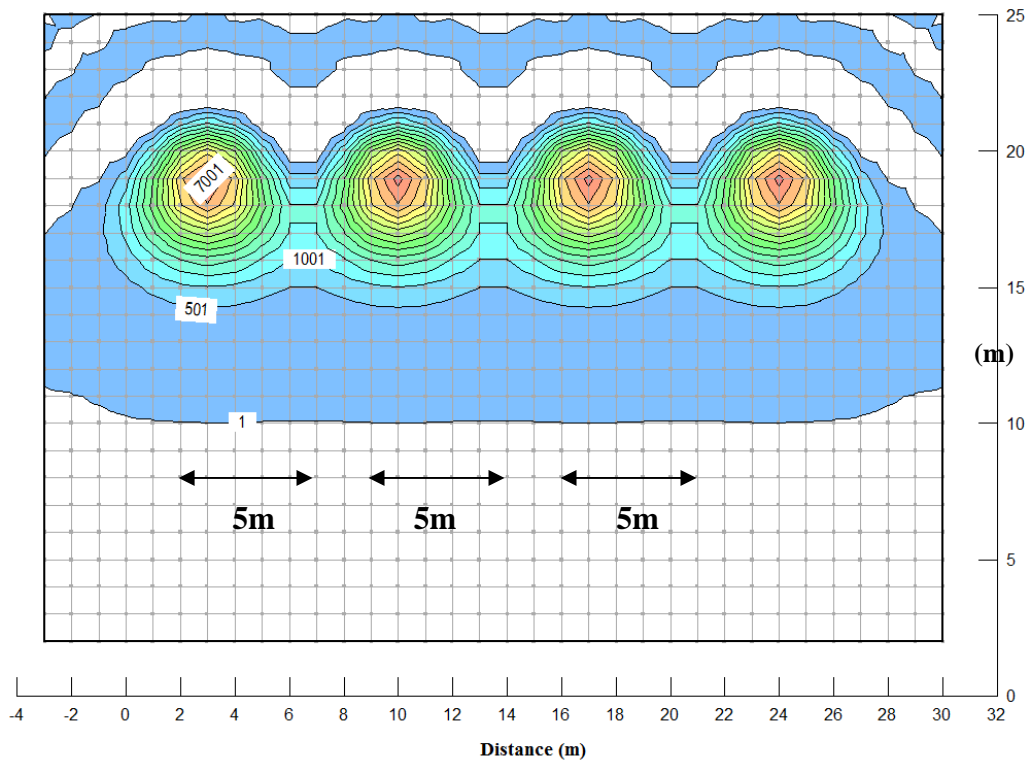
Figure 4.32 Multiple burial trenches spaced five meters apart, 50 years of transport.
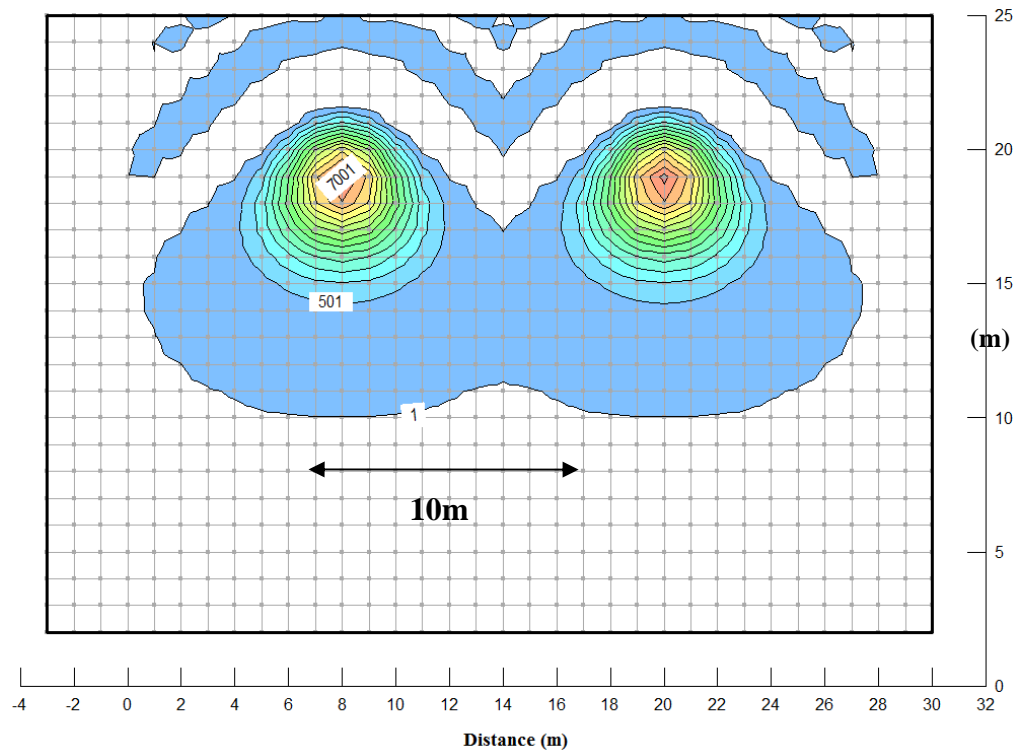

Figure 4.33 Multiple burial trenches spaced ten meters apart, 50 years of transport.

4.4   PHREEQC Transport Results

In order to evaluate potential geochemical reactions and ion exchange occurring beneath the burial trench during transport, PHREEQC was used to simulate transport of livestock mortality leachate through a column of moderately permeable soil. The transport simulation involves a 1-D column defined by a series of cells with the same pore volume. The 10 meter column was equilibrated with groundwater from Table 4.7 initially and 0.14 column pore volumes of livestock mortality leachate concentrations were added to the column. This is an equivalent value equal to approximately 0.5 m depth of fluid. Once this occurred, groundwater was added back on top of the leachate in 0.14 pore volume increments up to 0.56 pore volumes. The results at each pore volume increment can be seen in Figures 4.34-4.38.

Figure 4.34 shows an initial concentration front of calcium and magnesium leading the plume due to cation exchanges occurring with ammonium on the exchange sites.   This salt concentration front is approximately one meter ahead of the ammonium front. Bicarbonate, sodium and chloride are showing an unattenuated transport.   This trend continues once groundwater is put back on top of the mortality leachate. Figure 4.35 shows 0.14 pore volumes of groundwater allowed to transport after the leachate.   This figure shows ammonium retardation and rising concentrations in the calcium/magnesium salt front.   Bicarbonate concentrations are slightly decreasing; this could be caused by bicarbonate ions precipitating with calcium and magnesium. Figure 4.36 shows 0.28 pore volumes of water added to the column.   The salt front of is now slightly decreasing in concentration.   The attenuation of ammonium is still occurring and bicarbonate is also decreasing in concentration.   The same is occurring in figures 4.38-4.38. At 0.56 pore volumes of groundwater infiltration, concentrations of bicarbonate exceed drinking water standards 30-50 times, calcium concentrations exceed standards by 65 times, magnesium concentrations exceed standards by 17 times, and chloride exceeds by 6 times.
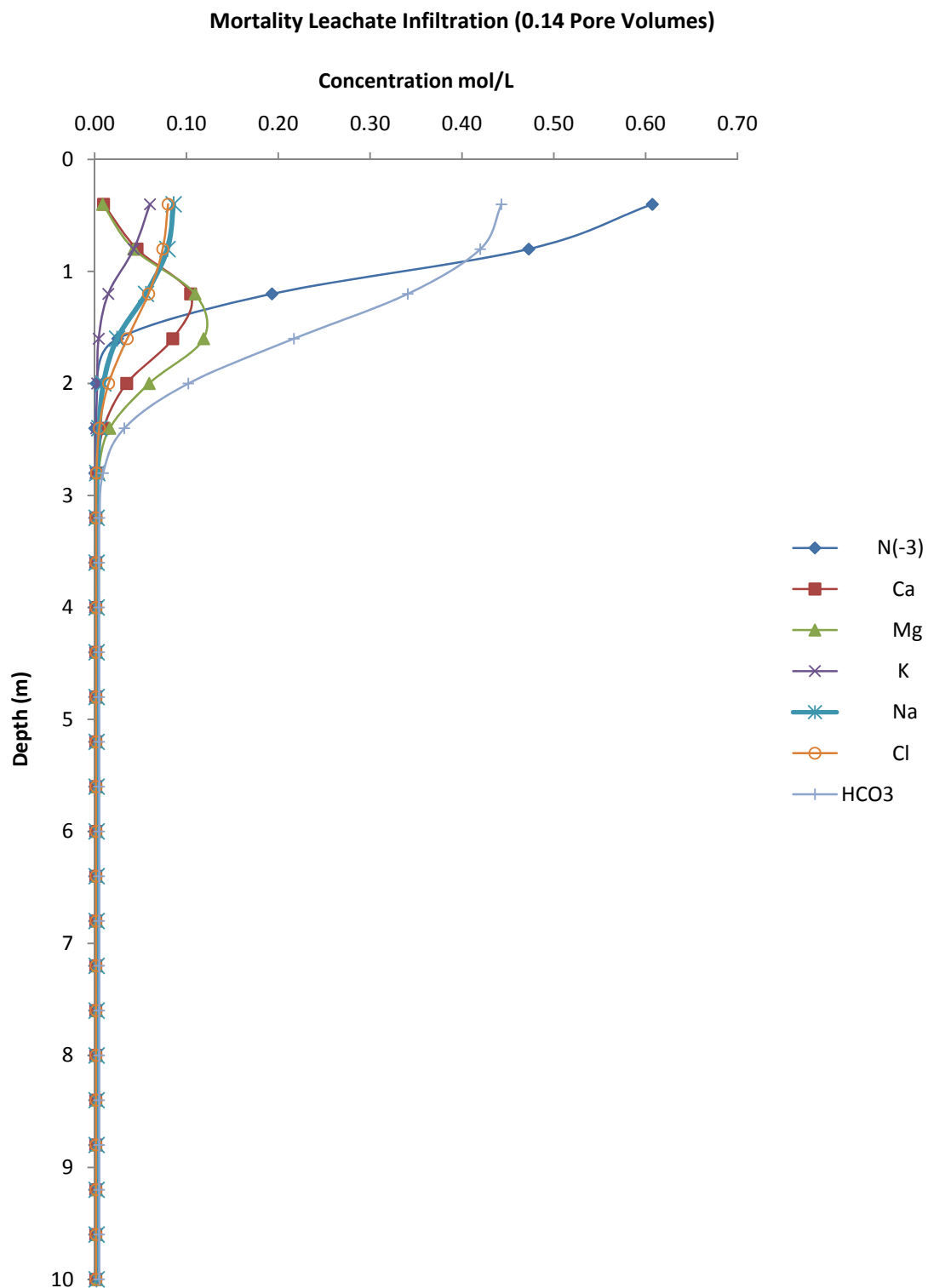
Figure 4.34 Mortality leachate 0.14 pore volumes infiltration into column.

Figure 4.35 Groundwater 0.14 pore volumes infiltration into column on top of leachate.

Figure 4.36 Groundwater 0.28 pore volumes infiltration into column on top of leachate.

Groundwater Recharge Infiltration (0.42 Pore Volumes)
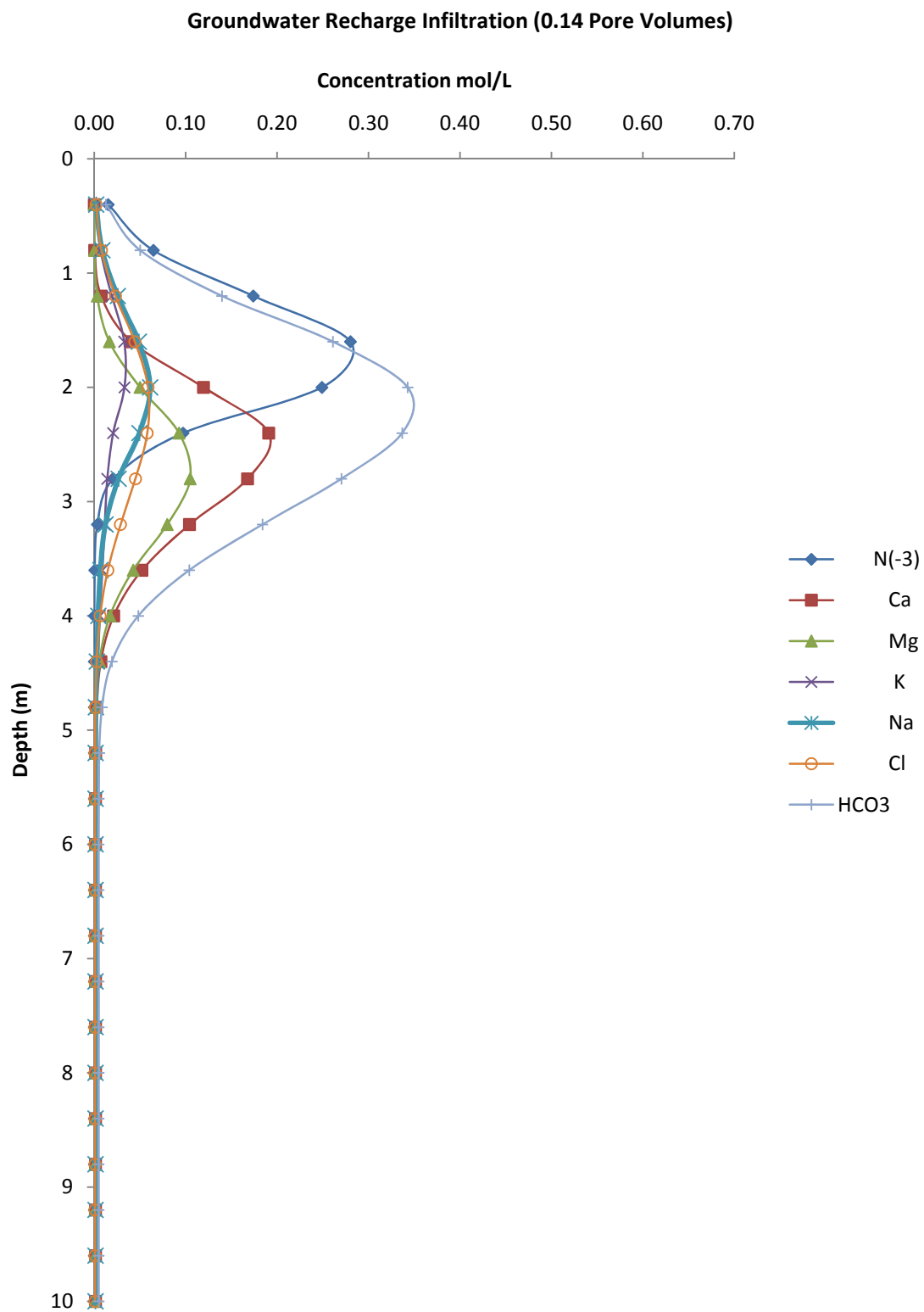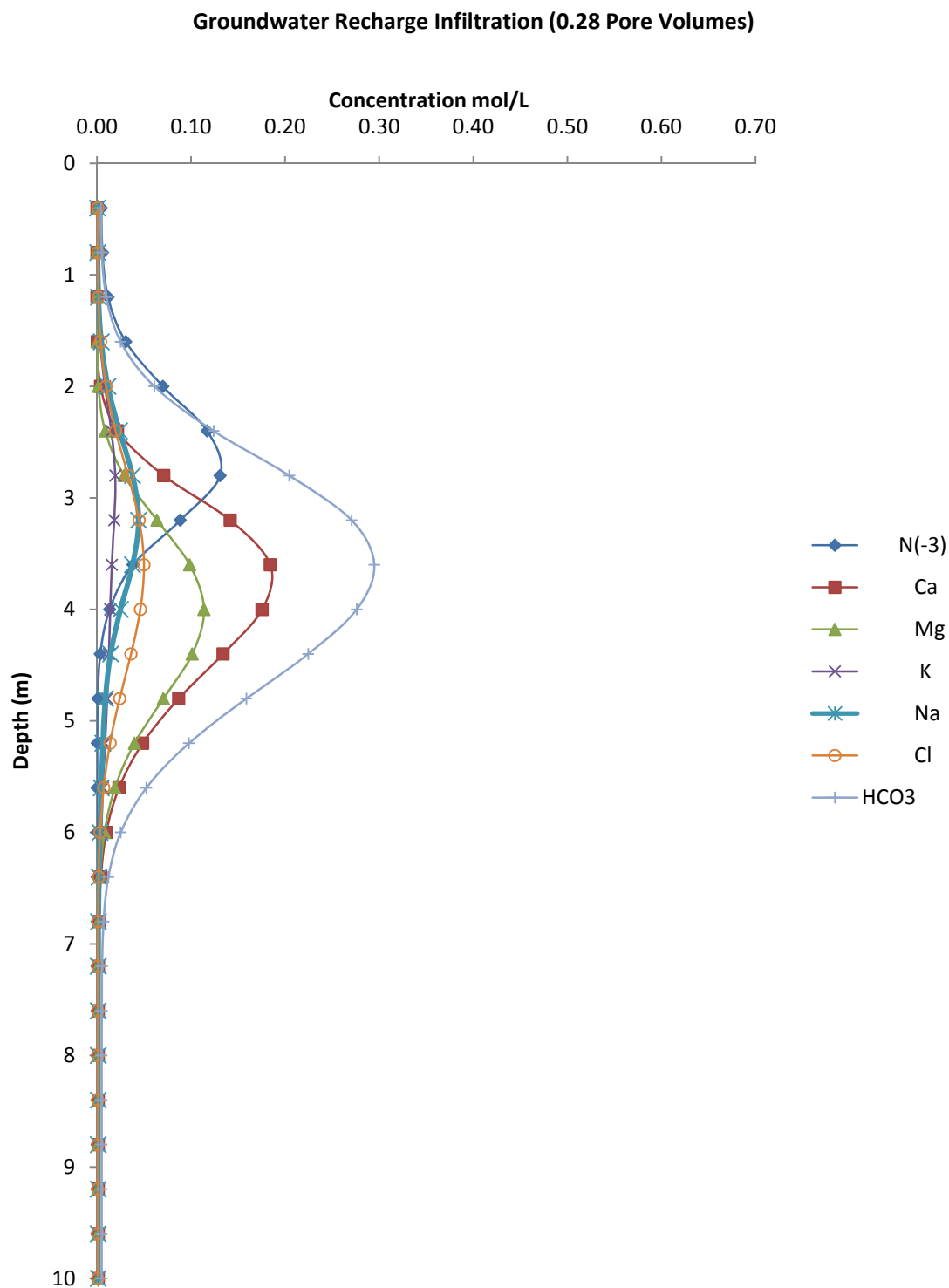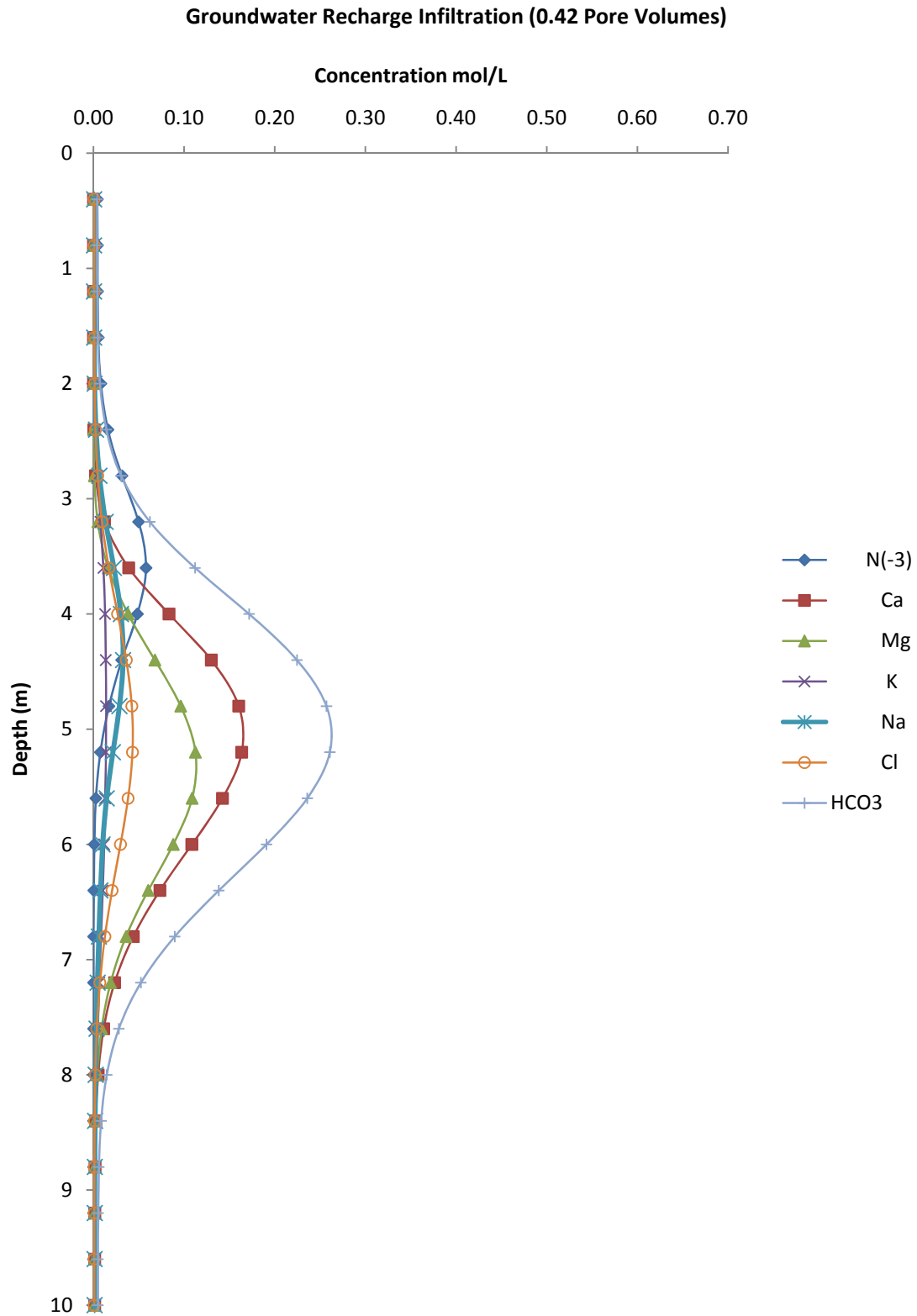
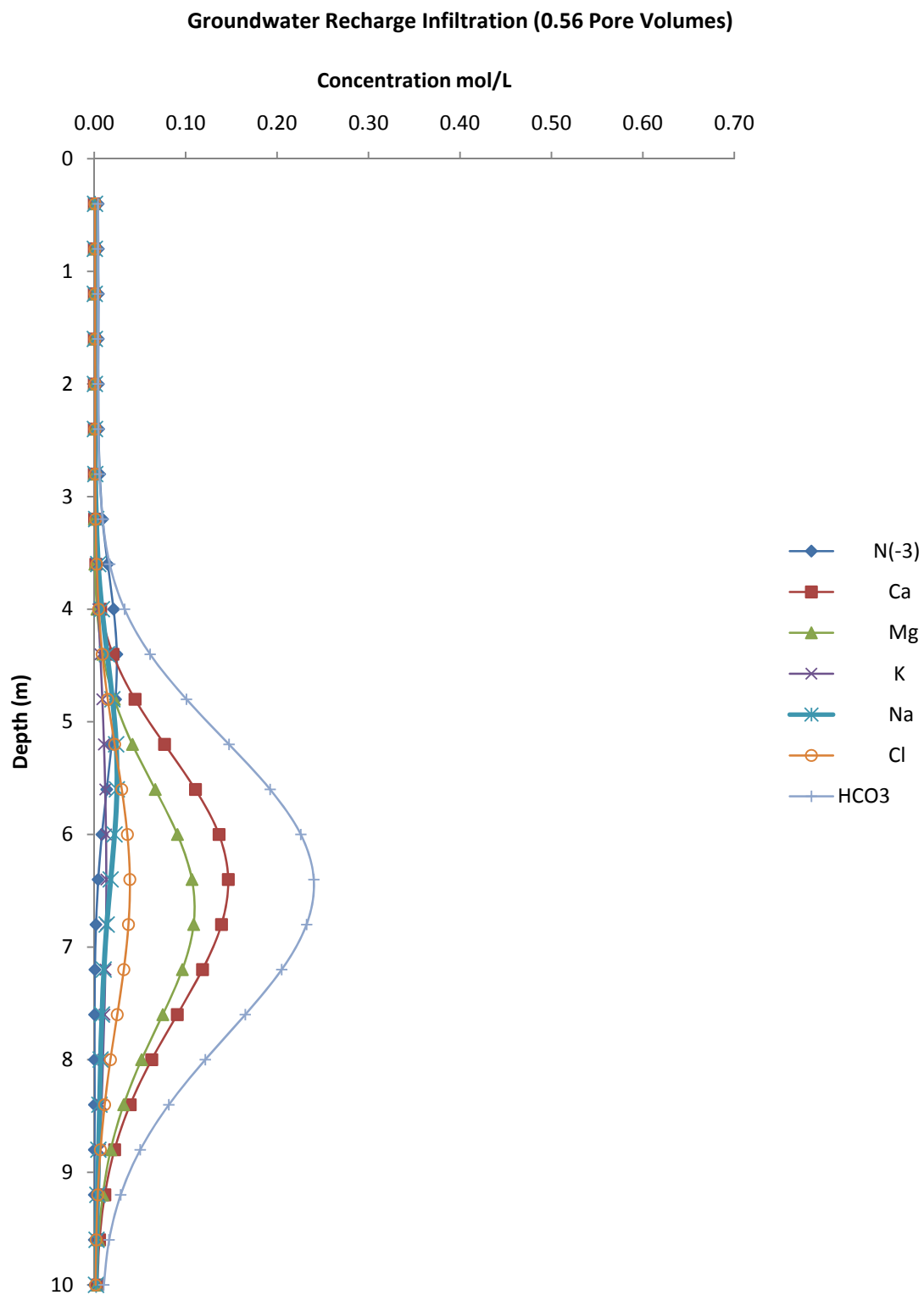Figure 4.37 Groundwater 0.42 pore volumes infiltration into column on top of leachate.

Figure 4.38 Groundwater 0.56 pore volumes infiltration into column on top of leachate.

To observe the plume separation of NH4, Na, K, Ca, and Mg, their respective distances from the chloride plume were calculated and shown in Table 4.9. Chloride was used for comparison due to its conservative nature. Ammonium initially is retarded by a distance of 0.5 m at 0.14 PV recharge; by 0.56 PV recharge, its retardation distance is 2 m behind chloride. Sodium follows the same type of retardation, except less distance. Potassium is initially retarded, but by 0.56 PV recharge, its plume is located at the same location as the chloride plume. Due to the occurrence of cation exchange, calcium and magnesium lead the chloride ion. Due to high ammonium concentrations, Ca and Mg have been replaced on soil exchange sites and are put into solution. Calcium initially is approximately 0.25 m ahead of the chloride plume, by 0.56 PV recharge, it is at the same location as the chloride plume. Magnesium initially is 0.5 m ahead of the chloride plume and remains ahead throughout the duration of the model.

Table 4.9 Plume Separation Distances from Cl^- Ion.

| | Distance from Cl^- peak (m) | | | |
	0.14 PV	0.28 PV	0.42 PV	0.56 PV
NH_4^+	-0.5	-0.9	-1.5	-2
Na	-0.15	-0.25	-0.7	-0.8
K	-0.4	-0.75	-1.1	0
Ca	0.25	0.15	0	0
Mg	0.5	0.4	0.2	0.2

* PV = pore volume

Not only are these basic forms of ions of concern, the speciation and complexing occurring during transport can cause problems as well. Livestock mortality leachate is initially oversaturated with respect to hematite, vivianite, dolomite, hydroxyapatite and calcite; while the groundwater is oversaturated with respect to hematite and hydroxyapatite; and undersaturated with vivianite (Figure 4.39 and Table 4.10). The initial transport of the mortality leachate into the soil-water system provides oversaturation with respect to pyrite, hydroxyapatite, sphalerite, dolomite, hematite, etc. After the leachate has fully infiltrated the column and groundwater was added back on top, the system then became undersaturated with

respect to vivianite and hematite. The dissolution of hematite and vivianite frees up iron to precipitate as pyrite, while the dissolution of vivianite also frees phosphate to then be available to precipitate as hydroxyapatite. Due to relatively high concentrations of phosphorus in mortality leachate, precipitation of phosphate minerals is possible. The mixing of sulphate with the groundwater causes it to precipitate as pyrite, mackinawite and sphalerite. Carbonates within the system can precipitate aragonite, removing some calcium from solution.
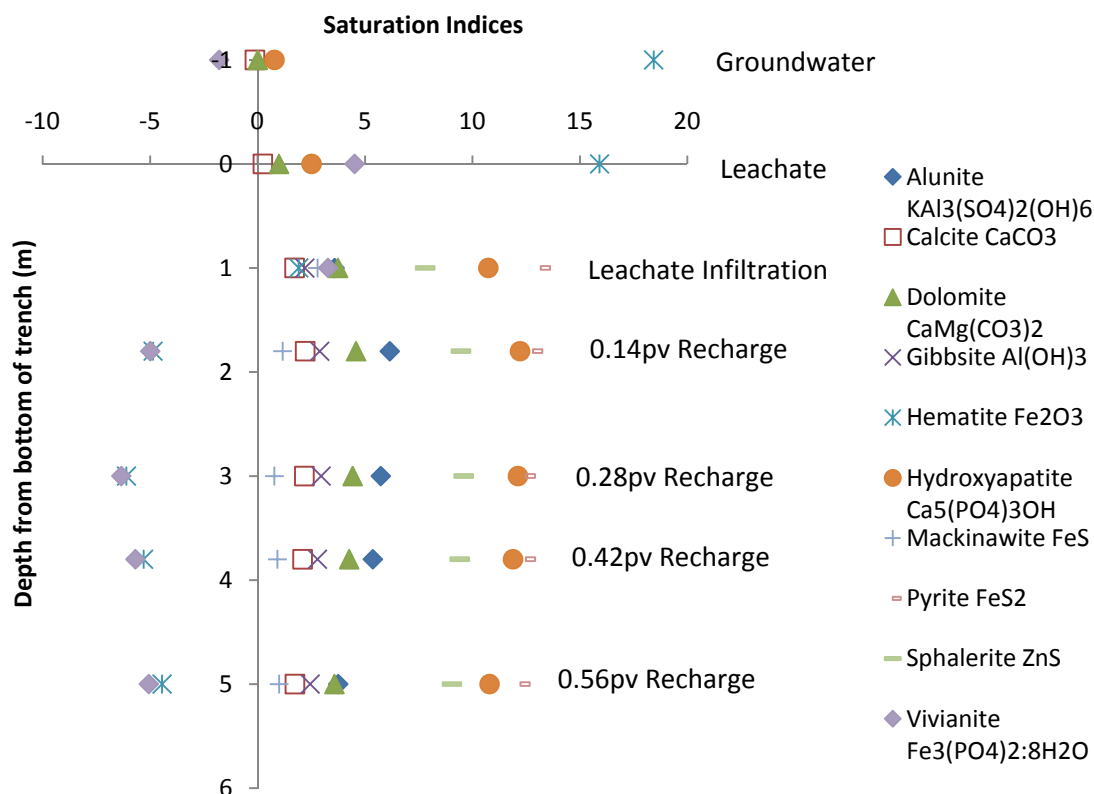

Figure 4.39 Saturation indices of select mineral phases as a function of depth pore volume infiltration.

Table 4.10 PHREEQC saturation indices.

Phase		Groundwater Depth (m)	Pure Leachate	Leachate Infiltration 1	0.14 PV GW 1.8	0.28 PV GW 3	0.42 PV GW 3.8	0.56 PV GW 5
Alunite	KAl3(SO4)2(OH)6			3.59	6.16	5.74	5.37	3.76
Anhydrite	CaSO4	-2.08	-2.20	-0.66	-0.55	-1.01	-0.82	-1.11
Aragonite	CaCO3	-0.12	0.25	1.73	2.22	2.18	2.1	1.74
Calcite	CaCO3	0.02	0.40	1.88	2.36	2.33	2.24	1.89
Dolomite	CaMg(CO3)2	0.01	1.00	3.75	4.59	4.43	4.27	3.58
FeS(ppt)	FeS			2.05	0.43	0.06	0.2	0.28
Gibbsite	Al(OH)3			2.2	2.89	2.96	2.79	2.45
Goethite	FeOOH	8.23	6.95	-0.03	-3.43	-4.05	-3.64	-3.21
Gypsum	CaSO4:2H2O	-1.84	-2.01	-0.42	-0.33	-0.79	-0.6	-0.88
Halite	NaCl	-7.65	-4.17	-5.83	-4.77	-4.57	-4.81	-5.26
Hematite	Fe2O3	18.44	15.92	1.92	-4.86	-6.1	-5.3	-4.44
Hydroxyapatite	Ca5(PO4)3OH	0.79	2.52	10.74	12.22	12.12	11.89	10.8
Jarosite-K	KFe3(SO4)2(OH)6	-0.62	0.65	-22.7	-32.4	-34.91	-33.54	-32.83
Mackinawite	FeS			2.79	1.17	0.79	0.93	1.01
Melanterite	FeSO4:7H2O	-6.7	-5.45	-5.8	-8.81	-9.72	-9.24	-9.07
Pyrite	FeS2			13.18	12.82	12.48	12.49	12.24
Siderite	FeCO3	-0.12	1.81	1.23	-1.37	-1.84	-1.66	-1.57
Smithsonite	ZnCO3		-0.75	-1.74	-1.05	-1.01	-1.16	-1.51
Sphalerite	ZnS			7.8	9.47	9.6	9.41	9.05
Sulfur	S			-1.57	-0.3	-0.25	-0.39	-0.71
Talc	Mg3Si4O10(OH)2	-3.19	-6.64	-2.01	-1.68	-2.21	-2.28	-2.99
Vivianite	Fe3(PO4)2:8H2O	-1.78	4.52	3.28	-4.99	-6.34	-5.68	-5.06

The 22 most abundant species of ions are shown in Figure 4.40; ammonium and bicarbonate being the most abundant cation and anion. The top nine most abundant species shown in Figure 4.41 include, ammonium, bicarbonate, chloride, calcium, calcium bicarbonate, sodium, potassium, magnesium and magnesium bicarbonate. To demonstrate the remaining abundant species, Figure 4.42 plotted on a smaller concentration scale shows the variance in those ions dominated by phosphoric acid and sulphate compounds. During the transport intervals, ammonium decreases in concentration and is four percent of initial concentration after 0.56 pore volumes of fresh groundwater transport. Sodium and potassium also show significant drops in concentration as well as phosphoric acid. Increases in sulphate concentrations can be attributed to the high amount of sulphate in the leachate as well as the concentration of sulphate present in the groundwater. Because of increasing sulphate concentrations to match background conditions, it can be expected that sulphate minerals such as $CaSO_4$ and $MgSO_4$ will

increase in concentration until the sulphate concentration has stabilized. Increases in calcium and magnesium concentrations can be attributed to ion exchange, as ammonium is trading places with these ions on soil exchange sites. Due to the release of these ions, their concentrations in solution are increasing as well as they are binding with other ions such as phosphoric acid, bicarbonate, and sulphate.

The speciation results of the leachate, show potential threats from ammonium transport due to the formation of ammonium sulphate ($NH_4SO_4^-$) and phosphoric acid and sulphate compounds in solution. Figure 4.43 shows ammonium suphate concentrations at depth during transport at each interval. Initially the concentration is increasing, up to 0.28 pore volumes of recharge. From then on, the ammonium sulphate concentrations are decreasing. Due to ion exchange, ammonium in solution is competing with exchange sites and in doing so is replacing calcium and magnesium. With a reduced concentration of ammonium along the flow path, there is less competition for the sulphates to complex with the ammonium ions.
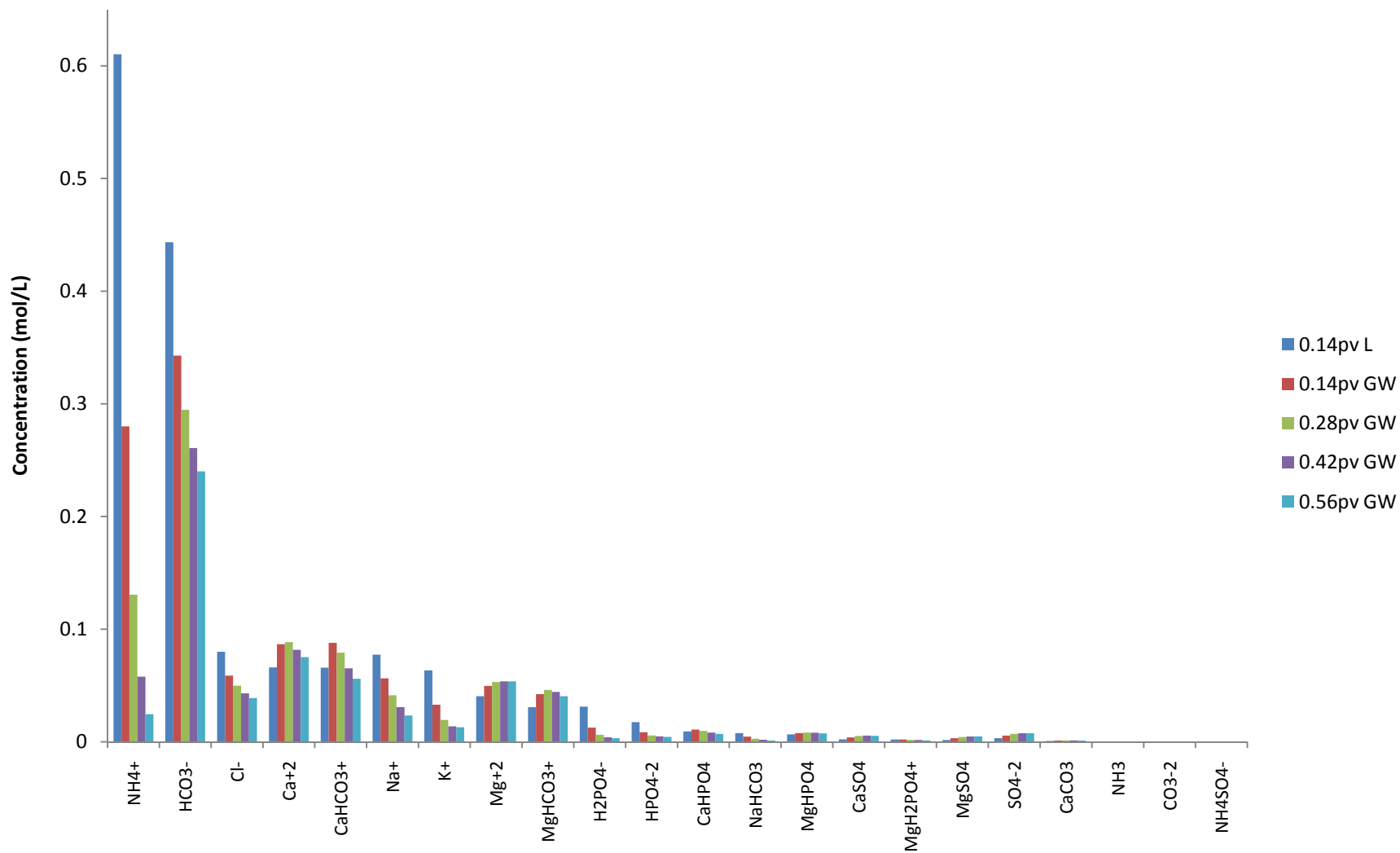
Figure 4.40 Concentration of abundant species during transport intervals predicted by PHREEQC.
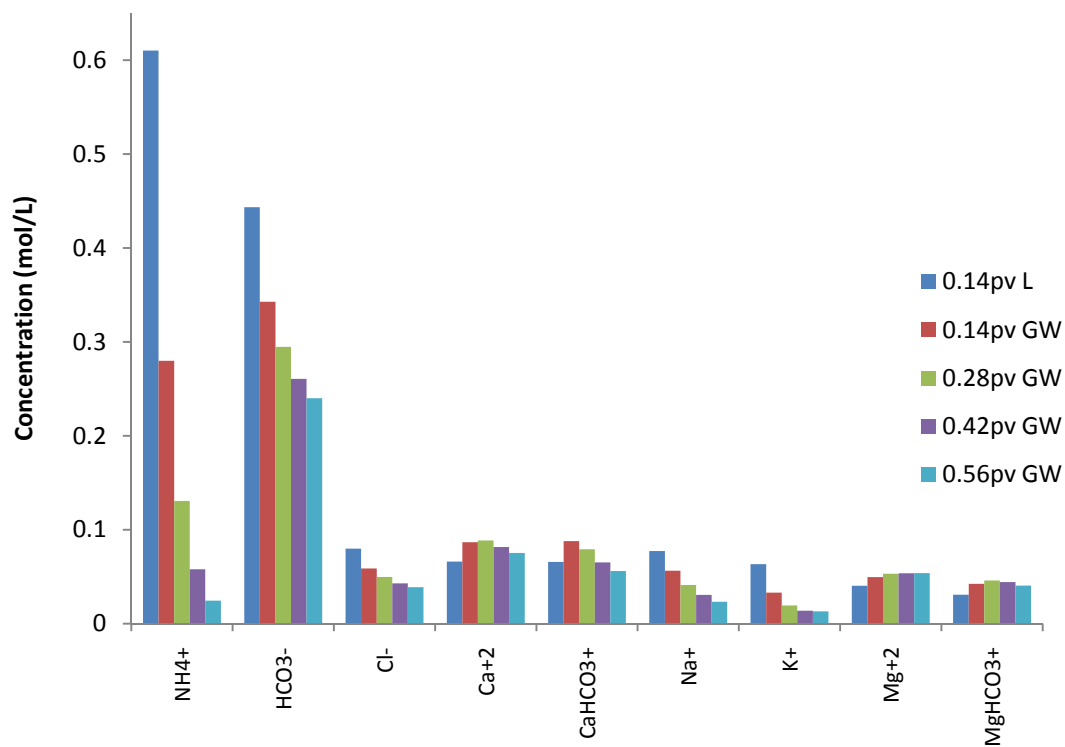
Figure 4.41 Concentration of 9 most abundant species during transport.
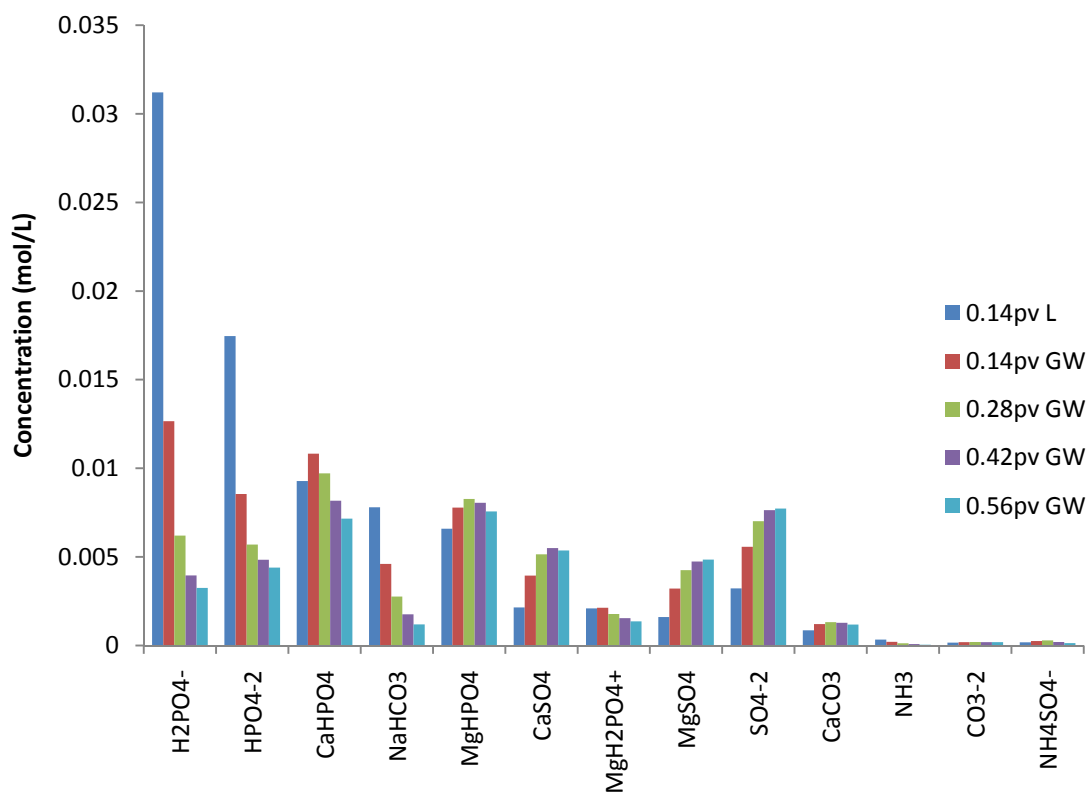
Figure 4.42 Concentration of remaining most abundant species during transport.

Phosphoric acid is another concern due to its corrosivity. Figure 4.44 shows the phosphoric acid transport through the column. Phosphoric acid has the tendency to bind to iron, magnesium, potassium and sodium. Some of these compounds will transport unattenuated, others will not, depending on their charge. The decreasing concentration of phosphoric acid demonstrates its ability to bind with other minerals.

Ammonium Sulphate Concentrations During Transport
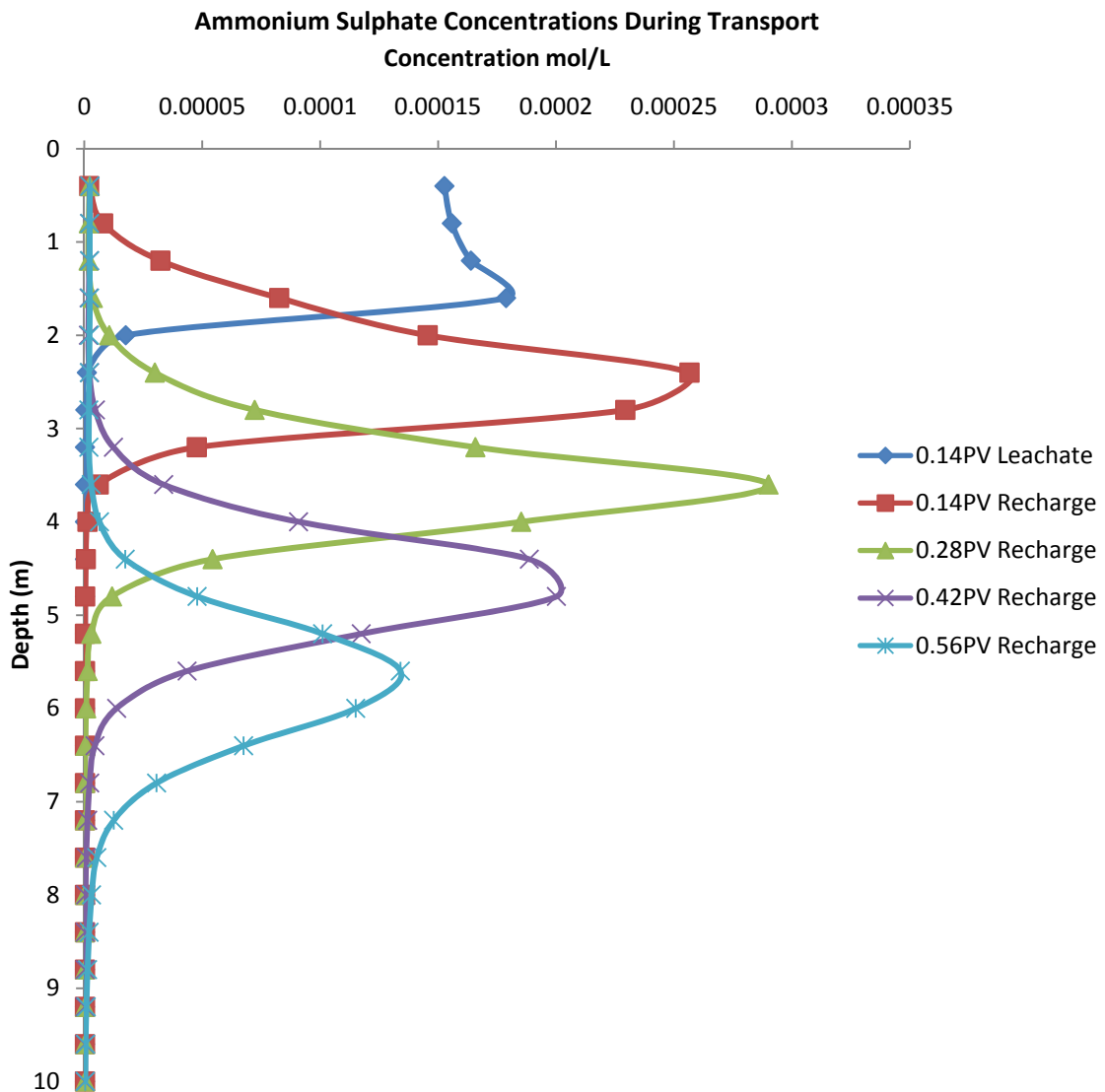
Figure 4.43 Ammonium sulphate concentrations during transport.

Phosphoric Acid Concentrations DuringTransport
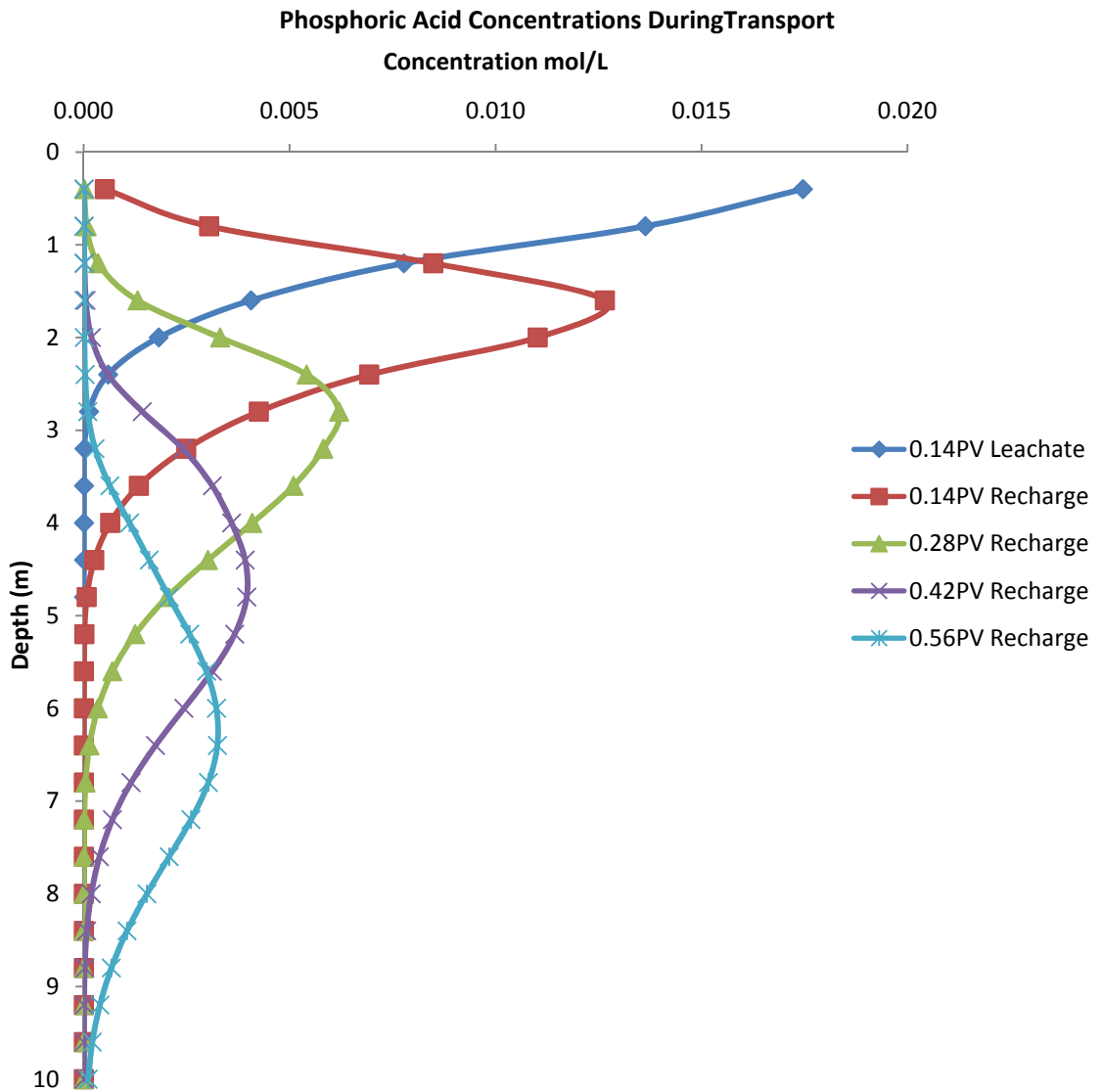
Figure 4.44 Phosphoric acid concentrations during transport.

4.5   Model Significance and Limitations

Each model generated from CTRAN demonstrates a worst case scenario for three soil conditions. Convergence issues occurred in the integration steps for the model when calculating for a highly permeable situation.

Using software based on the two-dimensional contaminant transport equation to predict contaminant transport only gives an estimate of the situation at hand. Real life

conditions such as non-homogeneous soils and horizontal velocity gradients were not modelled in this thesis due to their complexity. These models also do not consider infiltration of rainwater, drought conditions, or other directions of water flux. The CTRAN package is capable of simulating these properties if they are known for the modelled time period.

The PHREEQC model used a standard database of equilibrium thermodynamics. Errors in this model could occur depending on temperature, redox conditions, charge balance, etc. The transport model provides a good estimate of exchange reactions occurring along the flow path. A more in-depth approach would involve evaluating a real-life site location and modifying the exchange coefficients to match soil conditions along the flow path.

4.6   Mass Mortality Event Simulation

In the event of a mass mortality situation on a 10,000 head cattle feedlot, disposal by burial would be extensive. Using CFIA (2006) standards requiring 1.5 m^3 of burial space for one bovine; one meter of a 4 m by 2m trench would hold 5.3 bovine. To dispose of 10,000 head, approximately 1,876 meters of trench would be required. The best scenario for burial would be ten trenches at 200 meters of length with ten meter spacing in-between trenches. The total area required for disposal of these 10,000 bovine would then be 2.2 hectares (5.43 acres). For further analysis, an aquifer is present at 10 meters below the soil surface, allowing for 6 meters of transport. The amount of leachate available for transport would then total approximately 1876 m^3. To calculate total mass available for transport at the burial site, concentrations were used from Table 4.6. Mass loading into the simulated aquifer after fifty years as well as of total mass available for each element is shown in Table 4.11. At 50 years in a moderately permeable soil, approximately three to four percent of the initial concentration of ammonium is reaching the aquifer. Effects of mass loading on this aquifer may not be seen in the short term i.e. less than 50 years, but effects will be seen in the long term i.e. greater than 50 years. Although mortality burial trenches are

a slug-type contaminant, the total mass of nitrogen entering the system is many times greater than recommended values and should be carefully monitored.

Table 4.11 Total mass available in leachate and mass loading in 10 m aquifer at 50 years in a till soil for 10,000 head bovine simulation.

Constituent	Total Mass (kg)	Aquifer Loading (kg/year) @ 50 years
Ammonium	23675	947
Alkalinity	64910	32455
Sodium	3292	691
Chloride	4858	2769
Potassium	4230	1269
Phosphorus	2870	1492
Calcium	103	11948
Magnesium	71	4438

5   CONCLUSIONS

The first objective of this thesis was to determine the chemistry of livestock mortality leachate for three species: bovine, swine and poultry. Models were created to simulate a worst-case scenario of leachate transport for three soil conditions; a low permeable, moderately permeable and permeable situation. Characterization of the leachate through chemical speciation by PHREEQC was important to evaluate species and their activities, as well as dissolution and precipitation of ions present in solution as well as providing a comparison of the leachate to a typical Saskatchewan groundwater. A reactive transport model using PHREEQC presented results for a moderately permeable soil with a CEC of 10 meq/100g.

Livestock mortality leachate contains on average (Table 4.6) a concentration of 12,600 mg/L of ammonium, 34,600 mg/L alkalinity, 2,600 mg/L chloride, 3,600 mg/L sulphate, 2,300 mg/L potassium, 1,800 mg/L sodium, 1,500 mg/L phosphorus, along with small amounts of iron, calcium and magnesium. The pH of the leachate was near neutral. In comparison to earthen manure storages and landfills, the strength of the leachate was relatively higher.

Speciation of the leachate by PHREEQC provided estimates of phosphate compounds precipitating from solution, and phosphoric acids available for unattenuated transport. Due to the relatively high concentration of ammonium and sulphates, significant concentrations of ammonium sulphate form and due to the negative charge, will allow unattenuated transport of nitrogen. Activities of ammonium and bicarbonate, sulphates, phosphates and other minerals are many orders of magnitude higher than concentrations present in naturally occurring groundwater.

The remaining objectives were to create preliminary simulations with two software packages, CTRAN and PHREEQC to simulate transport of the leachate for three different soil settings. Transport through a low permeable soil caused the unattenuated leachate to transport a

distance of approximately three meters in 50 years. The moderately permeable soil situation produced a transport depth of six meters with an approximate concentration of 30-40% of initial values for unattenuated ions. For a highly permeable soil, transport reached a depth of 10 meters in 10 years with approximately 40% of initial values of unattenuated contaminant. The PHREEQC transport model used a CEC of 10meq/100g and produced results for the moderately permeable soil consistent with the Geo-Slope simulation. PHREEQC demonstrates the cation exchange occurring along the flow path will produce a highly concentrated magnesium and calcium salt front.

For a mass mortality event occurring on a 10,000 head feedlot located in a moderately permeable soil, mass loading to an aquifer 10 m below surface after 50 years of transport would yield 947 kg/year of ammonium and a calcium/magnesium salt front inducing 11,948 kg and 4,468 kg respectively into the aquifer. Mass loading concentrations would continue in subsequent years until the highly concentrated contaminant bulb infiltrated the aquifer. At this loading rate, nitrogen concentrations exceed drinking water standards by 10-15 times.

The results of this thesis should aid in determining burial regulations by providing a basic understanding of the composition of leachate arising from a mortality disposal site. The worst case scenario models can provide a basic guideline for similar soil types along with total mass loading on a per volume basis.

6   RECOMMENDATIONS FOR FURTHER RESEARCH

Due to the strength of livestock mortality leachate, the option of burial locations should be carefully determined. Further research would be helpful to determine the fate of the bacteria or viruses causing the diseases in the livestock and whether or not it is transported through the underlying soil. Research on existing burial sites should be conducted to study how far the leachate has travelled since burial and how much has been attenuated by the soil particles. Other questions arising from this research pertain to the stability of the chemicals over time.

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APPENDIX A – SRC Analytical Laboratory - Leachate Analysis Methods

LEACHATE ANALYSIS

Ammonia

To analyze for ammonia, the water automated colorimetric method was used. Ammonia reacts with hypochlorite and phenol in the presence of a catalyst to form idophenol, which gives the sample a blue color. The Cobras Fara II automated centrifugal analyzer measured the absorbance of light in the solution at 630 nm which is proportional to the concentration. The detection limit is 0.01 mg/l. Samples with concentrations greater than 2.0 mg/l must be diluted.

Nitrate and Nitrite

Colorimetric determination with a continuous flow analyzer was used to determine Nitrate and Nitrite in the leachate. Nitrate is reduced to Nitrite with hydrazine sulphate and the nitrite is determined by diazotizing with sulphanilamide and coupling with N-(1-naphthyl)-ethylenediamine dihydrochloride to form a highly coloured azo dye which is measured colorimetrically at 529 nm. Nitrite was analyzed without the reducing agents using the colorimetric technique. The detection limit is 0.01 mg/l as N. Samples with concentrations above 2 mg/l must be diluted.

Chloride

Chloride was analyzed using the mercuric thiocyanate colorimetric method on the Cobras Fara II centrifugal analyzer. The thiocyanate ion (SCN-) is displaced from mercuric thiocyanate by chloride. The liberated thiocyanate ion reacts with ferric ion to form the coloured complex ferric thiocyanate. The concentration is proportional to the original chloride concentration. The concentration is measured by measuring the absorbance at 480 nm. The detection limit was 1 mg/l. Samples with concentrations greater than 500 mg/l must be diluted.

Organic Carbon

Organic carbon in the leachate was analyzed using the Dohrmann Phoenix 8000 carbon analyzer. Aqueous samples were treated with phosphoric acid to release inorganic carbon as CO_2. The remaining organic carbon in the sample was reacted with sodium persulphate and UV light to release the organic carbon as CO2. The CO2 was measured with a non-dispersive infrared detector. The detection limit was 0.2 mg/l for both Total Organic Carbon and Dissolved Organic Carbon. Samples with concentrations greater than 20 mg/l must be diluted.

Calcium, Magnesium, Potassium, Sodium and Sulphate

Several elements in the leachate were determined using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) using the TJA-IRIS Advantage Analyzer. The plasma was a stream of argon gas ionized by an applied radio frequency field. Sample aerosols were injected into the plasma which subjected the atoms to temperatures of 6000 to 10,000 K. The high temperature ionized a high percentage of atoms in the aerosol and reduced chemical interferences. The emission spectra of the ionized atoms were optically measured in a computer controlled spectrometer. The light emitted from the plasma was focused onto the entrance slit of and Echelle Optical System. A prism and diffraction grating disperse the light which was focused on a Charge Injection Device (CID) detector. The detector monitored all configured wavelengths in a computer controlled environment. Upper detection limits for these elements are as follows: Calcium 50 mg/l, Magnesium 100 mg/l, Potassium 20 mg/l, Sodium 15 mg/l and Sulphate 300 mg/l.

Metals

Several elements in the leachate were determined using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) using the TJA-IRIS Advantage Analyzer. The plasma was a stream of argon gas ionized by an applied radio frequency field. Sample aerosols were injected into the plasma which subjected the atoms to temperatures of 6000 to 10,000 K. The high temperature ionized a high percentage of atoms in the aerosol and reduced chemical interferences. The emission spectra of the ionized atoms were optically measured in a computer controlled spectrometer. The detection limits and upper limits for this procedure are shown in Table A1.

Table A1: Detection Limits of Various Metals

Metal	Upper Limit (mg/L)	Detection Limit (mg/L)
Al	70	0.005
Ba	30	0.001
Be	10	0.001
B	100	0.001
Cd	20	0.001
Cr	100	0.001
Co	100	0.001
Cu	60	0.001
Fe	50	0.001
Pb	35	0.002
Mn	30	0.001
Mo	50	0.001
Ni	60	0.001
P	50	0.01
Si	100	0.01
Ag	50	0.001
Sr	10	0.001
Ti	30	0.001
V	100	0.001
Zn	30	0.005
Zr	70	0.001

Table A2: Summary of detection limits (units of mg/L)

Inorganic Chemistry	Detection Limit	Upper Limit
Bicarbonate	1	
Carbonate	1	
Chloride	1	500
Hydroxide	1	
pH	0.07	
Alkalinity	1	
Ammonia-N	0.01	2
Nitrate-Nitrite as N	0.01	2
TKN	0.05	2
Inorganic Carbon	0	
Organic Carbon	0.2	20
Sulfate	10	300
S	5	
K	0.1	20
Ca	0.1	50
Al	0.005	70
Ba	0.001	30
Be	0.001	10
B	0.001	100
Cd	0.001	20
Cr	0.001	100
Co	0.001	100
Cu	0.001	60
Fe	0.001	50
Pb	0.002	35
Mn	0.001	30
Mo	0.001	50
Ni	0.001	60
P	0.01	50
Si	0.01	100
Ag	0.001	50
Sr	0.001	10
Ti	0.001	30
V	0.001	100
Zn	0.005	30
Zr	0.001	70