HYDROGEOLOGICAL ASSESSMENT OF EARTHEN MANURE STORAGE STRUCTURES IN SOUTHERN MANITOBA

BY

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A Thesis
Submitted to the Faculty of Graduate Studies
In Partial Fulfillment of the Requirements for the Degree of

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THE UNIVERSITY OF MANITOBA

FACULTY OF GRADUATE STUDIES

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CHAPTER 1: INTRODUCTION

1.1 BACKGROUND

North America is experiencing rapid growth in the livestock production industry, particularly as the industry has moved towards more large-scale, intensive operations. In the late 1990's, Manitoba's hog industry experienced 100% growth. Since that time, growth of the industry has continued at a rate of approximately 12% per year. In 1997, over three million hogs were produced in Manitoba, selling for an estimated $1.25 billion. More recent statistics indicate that the province currently produces over 5 million hogs annually, accounting for approximately 16% of Manitoba's total agricultural production (Frontier Centre for Public Policy, 2001; Manitoba Pork Council, 2001). This rapid growth of the industry in Manitoba is in part a result of the removal of grain transportation subsidies (the Crow Rate) in 1995 and an overall increase in the demand for pork, both nationally and internationally.

Despite the economic and socio-economic benefits to the province through the promotion of large-scale hog production, there has been much public opposition to the continued growth of the industry. The main issues of environmental concern often brought up by the public are odour and the management of production wastes (Robert Betcher, Personal Communication, 2000). While odour is primarily an issue for nearby residents, improper waste management practices, particularly in the areas of manure handling and storage, may impact both local and regional surface water and groundwater systems. Wastewater produced by large-scale intensive hog operations typically contains high concentrations of suspended and dissolved organic and inorganic substances, including nutrients and salts. These constituents can cause adverse impacts to groundwater and surface water quality by increasing the concentration of various chemical constituents to provincially and federally unacceptable levels.

The majority of intensive hog operations in the Province use liquid manure systems. These systems are comprised of slotted floors and gutters draining via gravity to a central sump or wet well. The collected liquid is pumped from the sump or wet well to a manure storage structure. Earthen manure storage structures (EMSs) are commonly used in Manitoba to provide long-term storage for wastewater generated by intensive hog operations. These structures are a low-cost alternative to aboveground concrete or fiberglass tanks, and can be easily altered and/or expanded with the addition of new cells to meet the increasing demands posed by an expanding market. EMSs are constructed by first excavating up to 3 metres of *in-situ* materials. Berms are then constructed to heights of up to 3 metres above grade, providing a storage facility of 2 to 5 metres depth. Design capacity of the storage is dictated by the volume of wastewater produced by the operation on a daily basis and the number of days of storage required. Manure storage structures are usually emptied on a yearly basis.

As noted, the primary advantages of EMSs are their low cost relative to above ground facilities and the flexibility to design, construct and expand the EMSs to match the needs and changing needs of the operation. One of the disadvantages of EMSs is that they may allow for migration of contaminants into sub-surface and underlying aquifers. To reduce the potential risk of groundwater contamination, EMSs can be lined with impermeable or low-permeability materials. Typical liners include recompacted clay and/or clay-rich materials, geosynthetic-clay composites, geomembranes or a combination of these materials (Robert Betcher, Personal Communication 2000). However, unlined EMSs are common throughout the Prairie Provinces since thick clay and till aquitards have been thought to adequately impede the downward migration of contaminants (Robert Betcher, Personal Communication, 2000). In Manitoba and the other Prairie Provinces, regulations rely primarily on the properties of underlying aquitard materials and the integrity of recompacted liners to protect underlying groundwater from contamination. However, there is potential for these materials to contain fracture networks which may have an impact on the bulk hydraulic conductivity of the aquitards, by effectively increasing the hydraulic conductivity. Despite the potential for more rapid transport in a fractured aquitard, estimates of transport rates from the facilities during the design and/or evaluation processes are typically based on laboratory estimates of hydraulic conductivity and plug-flow estimates of solute transport.

In 1995, the Livestock Manure and Mortalities Management Regulation (Manitoba Regulation 42/98) was added to Manitoba's Environment Act. The regulation provides requirements for the use, management and storage of livestock manure and mortalities and was designed to ensure environmentally sound handling of production wastes and fatalities. Since inception of Regulation 42/98 in Manitoba, all proposed new or expanded EMSs are subject to a provincial application and permitting procedure and an annual manure management plan must be submitted to the Province. Under the regulation, all EMSs constructed or expanded must be situated in the unsaturated zone and constructed with a minimum 0.2 m recompacted liner. Where less than 5 metres of materials with a saturated hydraulic conductivity of 10^{-7} cm/s separates the EMS base from the top of an underlying aquifer, geosynthetic liners in conjunction with a groundwater quality monitoring program are typically required.

1.2 SCOPE OF THESIS

This study has been undertaken to examine the rates of vertical migration of select nutrients and salts from EMS facilities located in a typical hydrogeologic setting in agro-Manitoba. The work completed included field and laboratory examination of solute migration beneath ten EMS facilities ranging in operational age from 3 to 29 years. Specifically, the objectives of this thesis are as follows.

- Investigate the migration of nutrients and salts from EMS facilities ranging in operational age from 3 to 29 years, including facilities constructed under the current Manitoba permitting policies and those constructed prior to the inception of Manitoba Regulation 42/98. Site locations are shown in Figure 1.1.
- Deduce solute transport parameters (average linear groundwater velocity and dispersion) from the collected field and laboratory data using an appropriate analytical modeling technique for the soil types and hydrogeological conditions encountered.

- Complete preliminary generic predictive modeling using the deduced solute transport parameters and literature published values to evaluate the impact of EMS in a variety of settings and for various construction scenarios.
- Report and discuss the significant findings, including an evaluation of EMS performance, for each of the three objectives provided above.

1.3 THESIS OUTLINE

This thesis is presented in eight chapters. Chapter 1 provides an introduction and sets out the objectives of the work. Chapter 2 outlines select approaches and results for previous studies conducted by other researchers, and provides a summary of the significant findings. Chapter 3 provides an overview of the regional and local physiographic, geological and hydrogeological settings in the study area, including overburden types and physical characteristics at each of the study sites. In Chapter 4, the field and laboratory program is described and the methodology used in these programs is provided. Chapter 5 provides the results obtained from the field and laboratory program, including physical and chemical data obtained from these programs. Chapter 6 outlines the parameter estimation method used and provides the results of the parameter estimates. In Chapter 7, the preliminary generic predictive modeling method is described and the results of a number of simulations are provided. Chapter 8 provides and summary of the significant findings from the work and gives conclusions.

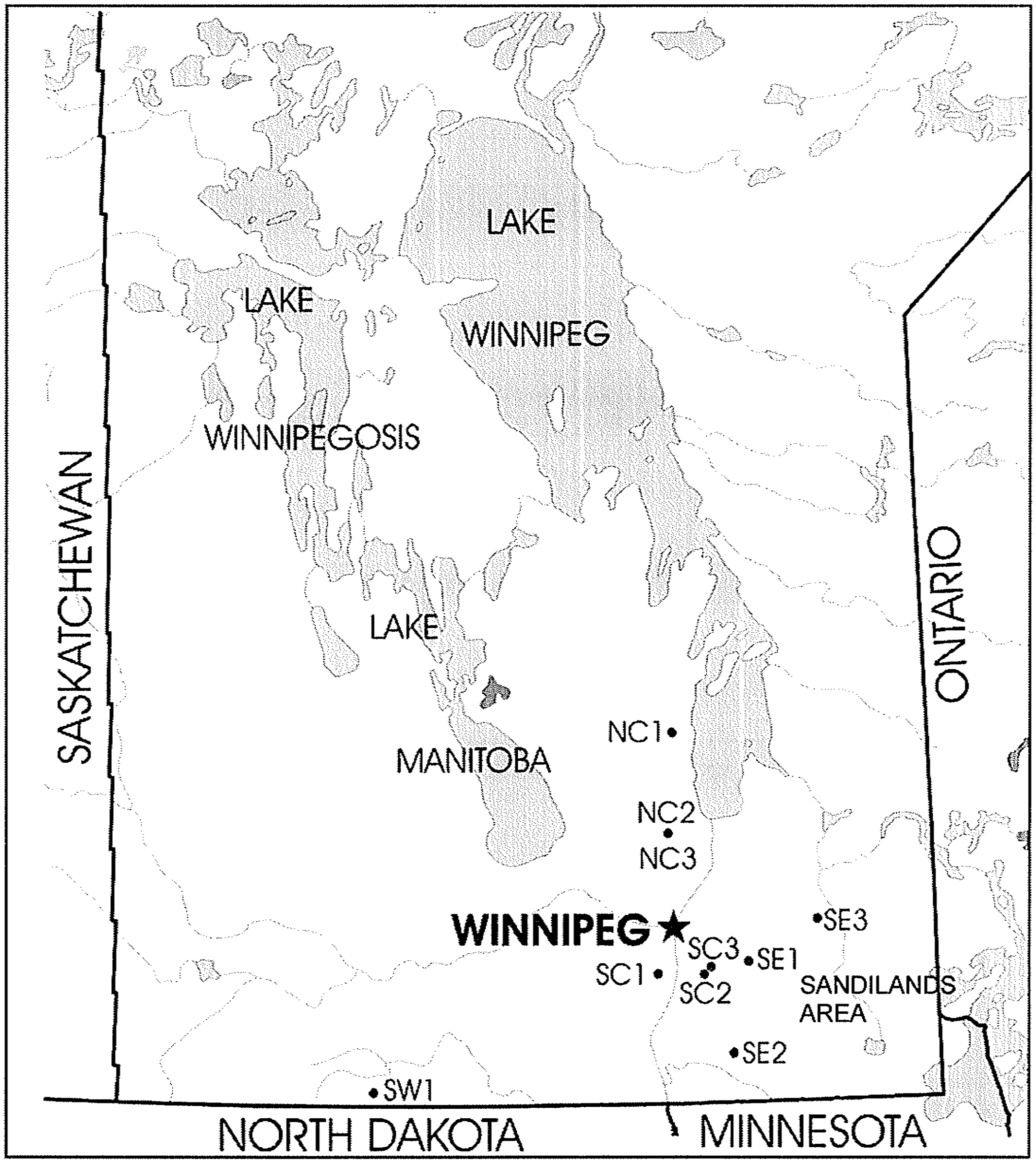


Figure 1.1: Site Locations

Site locations are indicated with a dot on the above figure. Ten EMS sites located in southern Manitoba were examined in this work.

CHAPTER 2: PREVIOUS STUDIES OF SEEPAGE FROM EMS FACILITIES

2.1 STUDY APPROACHES

A number of studies have been undertaken in North America in recent years to examine the potential for groundwater contamination as a result of seepage from EMSs, and results have been extensively reported in the literature. The studies have looked at facilities constructed in a wide range of hydrogeologic settings from EMSs constructed in shallow sand and/or gravel aquifers to EMSs constructed in thick clay-rich aquitards. Results from these studies have been varied with some EMS sites showing significant rates of seepage while others appear to have had an initial rapid rate of fluid loss that reduced quickly with time. These changes in seepage rates over time have been attributed to self-sealing of EMS facilities, a mechanism which has been the focus of much the research conducted to date.

The approaches used to examine seepage from EMSs have been quite variable. Seepage rates from EMS facilities have been estimated using:

- water balance methods to estimate whole lagoon seepage rates (for example, Ham *et al.*, 1997; Ham, 2002);
- groundwater quality monitoring and sampling in piezometers installed adjacent to the EMS to estimate lateral seepage rates and extent (for example, Ciravolo *et al.*, 1979; Ritter, 1983; Miller *et al.* , 1985, Fonstad *et al.*, 1995);

- groundwater quality monitoring and sampling in piezometers installed directly beneath the EMSs to estimate vertical seepage rates and extent (for example, Miller *et al.*, 1985);
- soil sampling from the base and/or inner berms of EMS facilities to examine vertical seepage rates and extent (for example, Miller *et al.*, 1976; Culley *et al.*, 1989; Fonstad *et al.*, 1997; Barrington *et al.*, 1987; Ham, 2002); and,
- laboratory column experiments to examine vertical seepage rates and extent and the nature and impact of the self-sealing mechanism (for example, Culley *et al.*, 1982; Barrington *et al.*, 1987; Maule *et al.*, 1995).

Despite the variability in the approaches used, the common objective in each of the studies has been to quantify seepage from EMS facilities and/or to examine the impacts of seepage from the facilities.

2.2 CHARACTERISTICS OF STORED HOG MANURE

Ham (2002) stated that characterizing waste chemistry is crucial in assessing the environmental risk posed by EMSs and Payne *et al.* (1980) concluded that stored hog wastes have more pollution potential than untreated municipal sewage. In addition, hog manure contains as much as five times the nitrogen concentrations of cattle manure (Ham, 2002). A number of researchers have included a chemical characterization of stored hog wastes and the cumulative results of some of the reported ranges in concentrations are presented in Table 2.1. The table shows the high variability in chemical species concentrations in hog manure from site to site, and between studies.

The effects of dilution, mixing, depth of manure ponding, and time of sampling can all have a significant impact on the concentrations of chemical species within the stored waste (Maule *et al.*, 1996; Ham, 2002). Overall, nitrogen concentrations tend to decrease during the summer as a result of ammonia volatilization (Lorimor *et al.*, 1975). Low concentrations of nitrate-nitrogen suggest the anaerobic character of the EMS at the time of sampling (Payne *et al.*, 1983) is a characteristic typical to most primary EMS cells. Despite variability between sampling events and sites, Ham (2002) reported that minimal vertical stratification of chemical species concentrations occurred within the stored wastewater column, a constancy attributed to wave action and mixing caused by manure addition. However, Ham (2002) and Lorimor *et al.* (1975) also point out that significantly higher phosphorus concentrations tend to occur within the settled sludge layer at the manure-soil interface.

2.3 SELF-SEALING OF EMS FACILITIES

The reduction of fluid loss over time observed in EMSs has been attributed to a self-sealing mechanism, which can effectively reduce soil permeability at or near the manure-soil interface. Self-sealing has been attributed primarily to physical and biological processes which occur at or near the soil-manure interface (Barrington *et al.*, 1983; Barrington *et al.*, 1985; Fonstad *et al.*, 1995). Ham (1998) describes three soil sealing mechanisms which occur as a result of manure ponding: physical clogging of pore spaces by waste products; biological clogging of pore spaces by microbial by-products; and clogging of pore spaces from salt-induced de-flocculation of clays. A number of researchers have concluded that physical clogging of pore spaces by waste products is the predominant self-sealing mechanism. Biological clogging of pore spaces by microbial by-products has been identified as the secondary self-sealing mechanism, and is more predominant in clay-rich soils. Chang *et al.* (1974), Barrington *et al.* (1983) and Barrington *et al.* (1987b) found that the chemical de-flocculation of clay-rich soils, due to manure ponding, plays a negligible role in the self-sealing mechanism, since upon desiccation soil permeability increases to initial values (Chang *et al.*, 1974) and a similar final permeability is achieved in the self-sealing layer, regardless of soil type (Barrington *et al.*, 1983; Barrington *et al.*, 1987b).

The self-sealing layer consists of a surface mat, located at the manure-soil interface, and an underlying zone of clogged soil. The surface mat is rubbery in texture, and is a combination of microbiological slime excretions and manure solids (Barrington *et al.*, 1983). Below the surface mat, pore spaces within the clogged soil zone are occupied by fine organic particles that have penetrated the surface mat (Chang *et al.*, 1974; Lo, 1977; Roswell *et al.*, 1985; Barrington *et al.*, 1987). The entire self-sealing layer is typically less than 20 mm thick in coarse-grained soils, and less than 10 mm thick in fine-grained soils (Barrington *et al.*, 1983; Roswell *et al.*, 1985; Fonstad, 1996; Maule *et al.*, 1999).

While some researchers have chosen to examine the effectiveness of the self-sealing mechanism by estimating whole lagoon seepage rates (for example Davis *et al.*, 1973), the majority of researchers have chosen to examine the effectiveness of the self-sealing mechanism in laboratory column experiments to reduce uncertainty in parameter estimation (Culley *et al.*, 1982; Barrington *et al.*, 1987; Maule, 2000). Much of the research has shown an initial rapid decrease in seepage rates followed by a slower decline to a near asymptotic final value, typically on the order of 10^{-9} m/s (Maule, 2000), regardless of soil type (Culley *et al.*, 1982). Most researchers have reported that self-sealing results in an exponential decrease in seepage rates regardless of soil type with the effects of self-sealing more evident shortly after initial manure ponding (Culley *et al.*, 1982; Roswell *et al.*, 1985; Barrington *et al.*, 1989; Fonstad, 1996; Ham, 1998; Maule *et al.*, 1999). Daniels (1984) reported that self-sealing can cause a reduction in whole lagoon seepage rates, as measured by the water balance method, by as much as three orders of magnitude compared to those predicted by laboratory permeability testing using water as a permeating fluid. Similarly, Maule *et al.* (2000) found that a self-sealing layer ranging from 3 mm to 8 mm thick, measured in laboratory column experiments, can cause a reduction in seepage rates through recompacted earthen liners by two to three orders of magnitude, regardless of soil texture.

Typically, the greatest overall decrease in permeability, as a result of the self-sealing mechanism, occurs in coarse-grained soils (Culley *et al.*, 1982) compared to fine-grained soils due to their initial relatively high permeability. However, the degree of the decline in permeability depends on a number of factors including soil type, duration of manure ponding, manure solids content of the wastewater, and the overall integrity of the self-sealing layer (Barrington *et al.*, 1983; 1985 and 1987a; Fonstad, 1996). Both Culley *et al.* (1982) and Barrington *et al.* (1989) show that manure solids accumulate more rapidly in coarse-grained soils compared to fine-grained soils, hence the more rapid development of the self-sealing layer in sand-rich soils. Smaller pore openings in fine-grained soils reduce organic matter infiltration, hence the slower development of a thinner self-sealing layer (Chang *et al.*, 1974). Manure type and solids content also play a role in the effectiveness of the self-sealing layer and hog manure typically contains smaller organic particles than those found in other types of manure, for example cattle manure. As a result, self-sealing may be more effective in a fine-grained soil ponded with hog manure than the same soil ponded with cattle manure (Barrington *et al.*, 1983).

The integrity of the self-sealing layer can be easily damaged or destroyed as a result of routine operational activities, such as agitation, pumping and solids removal. In addition, desiccation of the self-sealing layer may result in cracking and fissuring of the layer (Chang *et al.*, 1974). The integrity of the self-sealing layer may also be affected by temperatures (Barrington *et al.*, 1987a; Fonstad, 1996; Maule *et al.*, 1999), with higher temperatures being associated with increases in overall seepage rates in laboratory column tests. The higher temperatures are ideal for the fermentation of the manure solids, which results in the release of air bubbles through the surface mat, creating temporary preferential flow paths into underlying soils (Chang *et al.*, 1974; Barrington *et al.*, 1987a; Fonstad, 1996; Maule, 1999). However, Barrington *et al.* (1987b) also note that this same bubbling action is responsible for sedimentation of the fine organic particles which otherwise may be unable to pass through the surface mat into the underlying soils to create the clogged zone. Macropore formation as a result of plant growth can also reduce the integrity of the seal and create preferential flow paths (Ham, 1999). Damage and/or destruction of the self-sealing layer and macropore formation through this layer would result in an increase in overall seepage rates from the EMS.

2.4 VERTICAL SEEPAGE FROM EMS FACILITIES

A number of researchers have examined the vertical movement of chemical species from EMS facilities by removing soil cores from the base and/or side slopes of these facilities. Miller *et al.* (1976), Fonstad *et al.* (1997) and Ham (2002) all examined the movement of chemical species through underlying clay-rich materials using this method. Typically, these researchers have chosen to remove soil cores from the base and/or sidewalls of the facilities in the winter from an ice platform or shortly after the facilities have been emptied for field spreading of manure. Each of the collected cores are sectioned into discrete sub-samples, and analysed for a variety of chemical species.

Miller *et al.* (1976) removed sample cores from the base of four EMS facilities constructed in both fine-grained and coarse-grained soils. They concluded that chemical species migrated to far greater depths in coarse-grained soils compared to fine-grained clay-rich soils and that EMSs should not be recommended in areas of medium to coarse-grained soils. They found that ammonium was elevated to depths of 0.2 to 0.3 m below the EMS base in a fine-grained soil after two years of facility operation, while soluble phosphorus was elevated only in the vicinity of the soil-manure interface. Nitrate was found to be near background levels along the length of the sample cores and suggesting that conditions within the underlying soils were anaerobic. They concluded that in the case of EMSs constructed in fine-grained soils, EMS closure, involving the removal of liquids from the facilities, could pose a potential hazard. Soils underlying the EMS would become aerobic and therefore provide an opportunity for ammonium that had migrated from the EMS into the underlying soils to be converted to nitrate., which would pose an additional risk to any local potable groundwater supplies.

Fonstad *et al.* (1995 and 1997) removed soil cores from the base and inner berms of six EMSs facilities constructed in a variety of soil types ranging from clay-rich to sandy soils. They concluded that EMSs constructed in clay-rich soils significantly reduced chemical species migration potential compared to those constructed in sandy soils. For example, vertical migration of chemical species occurred to depths of less than 5 m in 20 years from an EMS constructed in a clay-rich soil, and occurred to depths of 8 m in 20 years from an EMS constructed in a sandy soil. Their results also showed significantly greater retardation of ammonium in more clay-rich soils. They also concluded that seepage rates can be greater on the sides of the EMS (along the side slopes of the berms) than seepage rate from the base of the EMS, due to dessication resulting from the exposure of the sides of the EMS to freeze-thaw and wetting-drying cycles.

Ham (2002) examined chemical species profiles obtained from cores removed from the base of two abandoned EMSs. The EMSs were 20 and 25 years old at the time of sampling and were constructed in clay-rich soils. He concluded that ammonium and other cations are adsorbed onto clay minerals in clay-rich soils, and that the ammonium build-up zone, located under the EMS and defined as the zone of ammonium sorption beneath the facility as leachate passes through the underlying soils, is typically less than 3 m, while chloride and other anions can migrate to depths of greater than 4 m below the EMS base. His conclusions are consistent with those of Miller *et al.* (1976), and suggests that the greatest risk to groundwater posed by EMS facilities is after closure. According to Ham (2202), EMS site remediation should include excavation and removal of the ammonium build-up zone.

2.5 LATERAL SEEPAGE FROM EMS FACILITIES

A number of researchers have reported lateral seepage from EMSs constructed in sandy materials (for example, Sewell *et al.*, 1975; Ritter *et al.*, 1980; Feng *et al.*, 1992; Westerman et al, 1993). Field cases that examined lateral seepage from EMSs constructed in clay-rich sediments (MacMillan, 2002) are limited. Lateral seepage from EMS facilities has been examined using monitoring well networks and electromagnetic surveys. In general, lateral seepage from EMSs constructed in clay or clay-rich materials have been found to be quite small, while in sandy soils with shallow water tables significant lateral movement can occur.

Sewell (1978) installed a network of monitoring wells downgradient of an EMS facility constructed in sand to sandy clay soils. Groundwater samples were collected from each of the wells six months prior to loading of the EMS facility. After loading, samples were collected on a monthly basis for three years. A rapid increase in chloride and nitrate-nitrogen concentrations was observed in the downgradient wells, located approximately 15 m from the facility, for the first six months after loading; however, the concentrations declined gradually to near background values shortly after. This decline was attributed to self-sealing of the EMS. Sewell (1978) also observed an increase in the water level measured in the wells by approximately 0.5 m after loading began and attributed the increase to water table mounding the in vicinity of the facility.

Ritter *et al.* (1980) installed a monitoring well network of twelve wells around and downgradient of an EMS facility constructed in sandy soils. The sandy soils contained discontinuous clay lenses and/or layers. After 2 years of monitoring, only one of the twelve monitoring wells showed significantly elevated chloride, chemical oxygen demand, and ammonia-nitrogen concentrations. The subject well was located hydraulically downgradient to crossgradient of the EMS facility. Downgradient wells showed little to no impact to groundwater quality as a result of the EMS operation. They concluded that the localized seepage observed in the well was likely due to preferential flow through tree root holes (the site was a former woodlot) and/or animal burrows, and that overall lateral seepage thru the bulk sandy soils was found to be minimal.

Ritter *et al.* (1983) installed monitoring networks around three EMS facilities constructed in sandy soils, and undertook a groundwater monitoring and sampling program over a one to four year period. As a result of elevated ammonia concentrations in downgradient wells, it was concluded that unlined facilities may have a significant impact on groundwater quality, particularly in shallow water table situations. Self-sealing was found to have little effect in reducing the impact. They also concluded that even

EMSs lined with clay-rich materials pose a groundwater contamination risk if the liner is not properly constructed.

Fonstad *et al.* (1995) installed monitoring well networks consisting of seven to ten nested wells around four EMS facilities constructed in silt-rich, clay-rich, and sandy soils. Wells were located from 10 m to 300 m hydraulically downgradient of the facilities. Water levels were monitored in each of the wells and groundwater samples analysed for chloride. Water levels indicated groundwater mounding in the vicinity of the EMSs. Impacts extended laterally to as much as 50 m from the facilities constructed in sandy soils, and little to no lateral seepage was observed in the clay-rich soils.

Brune *et al.* (1988), Ranjan *et al.* (1995) and MacMillan (2002) all used electromagnetic (EM) survey methods to estimate the extent of lateral seepage from EMS facilities. Ranjan *et al.* (1995) points out the electrical conductivity of groundwater increases with the increasing ionic concentrations associated with contamination. This increase in electrical conductivity can be detected by Electromagnetic (EM) survey equipment. Brune *et al.* (1988) used an EM34 to detect lateral seepage from three EMS facilities, all constructed sandy soils, and detected plumes extending out from each of the facilities. They concluded that EM surveys are a useful tool in locating and defining the magnitude of seepage from EMS facilities. Conversely, MacMillan (2002) used an EM31 to survey a total of thirty-five EMSs site in a variety of soil types and concluded that the EM31 is not a reliable indication of seepage from EMS facilities and should not be used as a tool for designing soil and groundwater quality monitoring and sampling programs.

Ranjan *et al.* (1995) used an EM38 to detect lateral seepage from an EMS constructed in clay and concluded that a single survey can not be used to detect lateral seepage since water content, temperature, soil type, and background salinity all impact the EM readings. It should also be noted that the various EM modes evaluate conductivity over different depths since the depth of penetration varies between models. Therefore, the results obtained for a given site would vary based on the EM model used to conduct the survey.

2.6 ESTIMATING SEEPAGE RATES FROM EMS FACILITIES USING MODELS

Over recent years, several modeling studies related to seepage from EMSs have been undertaken. Both generic and calibration studies have been completed in order to estimate seepage rates and characterize contaminant movement from these facilities. In areas of little to no topographic relief, where shallow water tables occur in thick aquitard materials, primarily downward migration of solutes from EMSs may be expected. As a result, researchers studying EMSs situated in the Prairie Provinces have chosen to use a one-dimensional modeling approach to estimate seepage rates from EMS facilities constructed in thick clay/till aquitards (Maule *et al.*, 1996 and Geonet Consulting Ltd., 2003).

Maule *et al.* (1996) simulated both chloride and ammonium transport beneath the center of EMS facilities using a numerical approximation to the advection-dispersion equation of Ogata (1970). Dispersion coefficients for chloride and ammonium were estimated based on the results of laboratory breakthrough tests for chloride and potassium, respectively. Seven representative soil samples were collected from various locations in their Saskatchewan study area. Hydraulic gradient was estimated by assuming that the hydraulic head at the seepage front of the observed seepage profile was zero and that the hydraulic conductivity of the underlying soils was a function of the number and type of sub-soil layers. Model simulations of both chloride and ammonium transport were compared to the measured chloride and ammonium seepage profiles to estimate the hydraulic conductivity of the soils underlying the facilities. Hydraulic conductivity was varied until a reasonable fit was achieved between predicted and observed species migration. It was found that overall the hydraulic conductivities required to produce the observed profiles were much lower than the authors' expectations as gleaned from other studies. The best fit was achieved by reducing the hydraulic conductivity in the upper 0.3 m to 0.6 m of the model to values as much as two orders of magnitude less than expected. Maule *et al.* (1996) concluded that some mechanism, such as self-sealing, was responsible for the decrease in hydraulic conductivity values near the top of the soil profile.

Geonet Consulting Ltd. conducted a generic EMS solute transport study (Geonet Consulting Ltd., 2000). The study focused on a generic site with the purpose of identifying the key mechanisms affecting solute transport through a low-permeability clay aquitard and evaluating the sensitivity of the model to changes in parameter values. One-dimensional solute transport was examined using a homogeneous porous medium approach and a fractured porous medium approach. Parameters were estimated based on both published values and values common in local practice. Consistent with expectations, Geonet Consulting Ltd. found that, under advection-dispersion dominated conditions, the advective term dominates and molecular diffusion has little or no effect on the model outcome. They suggested that under such conditions, hydraulic conductivity, porosity and hydraulic gradient should be quantified in any site investigation. Under diffusion dominated conditions, molecular diffusion and porosity become the key parameters requiring quantification. They also confirmed that a site underlain by a fractured aquitard may behave as an equivalent porous medium and, under such conditions, a porous medium approach can be used to represent one-dimensional solute transport from the EMS. Consequently, consistent with expectations, they found that where fracture spacings are small, and transport down a fracture is slow enough for diffusion into the matrix to equalize concentrations in the matrix and fracture, the resulting transport profile may give the appearance of a homogeneous porous medium profile dominated by advective-dispersive transport.

Table 2.1: Characteristics of Stored Hog Manure

Parameter	Units	CCME Drinking Water Criteria(1)	Reported Ranges in Concentration			
			Ham (2002)(2)	Payne et al (1980)(3)	Ritter et al (1983)(4)	Maule et al (1996)(5)
Nitrate-N	(mg/L)	10 (MAC)	<1.0 - 1.4	--	<0.05 - 4.10	16 - 52
Ammonium-N	(mg/L)	--	180.0 - 3540.0	--	--	980 - 5190
Ammonia-N	(mg/L)	--	--	360 - 550	0.25 - 133	--
Organic N	(mg/L)	--	30.0 - 1190.0	--	3.3 - 202	--
Total Phosphate-P	(mg/L)	--	--	50 - 160	--	--
Total Phosphorous	(mg/L)	--	9.0 - 1307.0	--	2.5 - 63.4	--
Chloride	(mg/L)	250 (AO)	195.0 - 2007.0	--	34.0 - 183	537 - 1330
Potassium	(mg/L)	--	328.0 - 3621.0	--	--	995 - 2645
Magnesium	(mg/L)	--	6.0 - 338.0	--	--	--
Total Suspended Solids	(mg/L)	--	140.0 - 420.0	450 - 1400	--	--
Electrical Conductivity	(mmho/cm)	--	2.7 - 28.0	--	--	--
Chemical Oxygen Demand	(mg/L)	--	1550.0 - 3066.0	1600 - 2400	60 - 1870.0	--
Biological Oxygen Demand	(mg/L)	--	860.0 - 2370.0	290 - 740	--	--

Notes:

-- indicates no guideline yet established and/or parameter not determined

(1) CCME Canadian Environmental Quality Guidelines (Chapter 2: Community Water). 2000; MAC - maximum acceptable concentration (health related); AO - aesthetic objective (non-health related).

(2) Ham, J.M. 2002. Seepage Losses from Animal Waste Lagoons: A Summary of a Four-Year Investigation in Kansas.

Transactions of the ASAE. Vol. 45(4), pp. 983 - 992. Range obtained from the results of twenty sampled EMSs.

(3) Payne, V.W.E Jr., Shipp, J.W. and Miller, F.A. III. 1980. Supernatant Characteristics of Three Animal Waste Lagoons in North Alabama.

Proceedings of Livestock Waste: A Renewable Waste. 4th International Symposium on Livestock Wastes, April 15 - 17, 1980, Amarillo, Texas.

Range obtained from the results of five samples collected from one EMS.

(4) Ritter, W.F. and Chirnside, A.E.M. 1983. Influence of Animal Waste Lagoons on Groundwater Quality. American Society of Agricultural Engineers,

1983 Winter Meeting, December 13 - 16, 1983, Chicago, Illinois. Range obtained from the results of twenty-four to fifty-four samples collected from one EMS.

(5) Maule, C.P. and Fonstad, T.A. 1996. Earthen manure storage study in Saskatchewan: solute movement, how deep and how much?

Proceedings of the Manitoba Swine Seminar, January 1997, Winnipeg, Manitoba. Number of sites/samples not indicated.

CHAPTER 3: STUDY AREA GEOLOGY AND HYDROGEOLOGY

This chapter provides an overview of the regional and local physiographic, geological and hydrogeological settings in the study area, including overburden types and physical characteristics at each of the study sites. These elements are all critical to providing an adequate site characterization.

3.1 PHYSIOGRAPHIC SETTING

The ten sites examined in this work are located in three of Manitoba's distinct physiographic regions: the Manitoba Upland, the Manitoba Lowland and the Precambrian Shield, as depicted in Figure 3.1. The SW1 site (Figure 1.1) is located near the eastern edge of the Manitoba Upland region, which is characterized by rolling hills grading into a series of uplands (Betcher *et al.*, 1995). The NC1 to NC3, SC1 to SC3, SE1 and SE2 sites (Figure 1.1) are located in the Manitoba Lowland physiographic region. This region lies east of the Manitoba escarpment and there is typically gentle to no topographic relief (Betcher *et al.*, 1995). The SE3 site (Figure 1.1) is located within the Precambrian Shield physiographic region. This region is generally characterized by hummocky terrain (Betcher *et al.*, 1995).

3.2 GEOLOGICAL SETTING

3.2.1 Bedrock

The uppermost bedrock units of the Manitoba Uplands region of southern Manitoba consist of Mesozoic shales and siltstones. Paleozoic dolostones and limestones form the upper bedrock units in the Manitoba Lowland region, and Precambrian granitic, metavolcanic, and metasedimentary rocks form the upper bedrock units in the Precambrian Shield region (Render, 1970; Teller and Fenton, 1980; Betcher *et al.*, 1995; Figure 3.2). The sedimentary units dip gently towards the west-southwest at 1 m/km to 10 m/km, forming part of the eastern edge of the Western Canada Sedimentary Basin (Betcher *et al.*, 1995; Figure 3.3), and are underlain by the Precambrian basement which subcrops to the east (McCabe, 1971).

3.2.2 Overburden

Thick glaciolacustrine clays and/or tills overlie bedrock throughout much of southern Manitoba and generally form the surficial deposits at each of the study sites examined (Figure 3.4). These deposits are the result of Wisconsinian glaciation and associated episodes of glacial Lake Agassiz (Grisak and Cherry, 1974). In the Manitoba Upland region, typically 20 m to 40 m of till, outwash and glaciolacustrine sediments overly bedrock, although bedrock outcrops and overburden thicknesses of as much as 250 m do occur in areas within the region. Overburden of the Manitoba Lowland region consists primarily of till of highly variable thickness from 10 m to greater than 100 m. Southwest of Lake Manitoba, glaciolacustrine and/or deltaic deposits overlie the till. Highly variable overburden thickness also occurs in the Precambrian Shield region with glacial and glaciolacustrine sediments often thin or not present at all. However, these deposits can be as much as 100 m thick in some locales (Betcher *et al.*, 1995). Typically, the upper 3 m to 5 m of the surficial overburden unit is weathered, a zone distinguished by oxidation staining, fracturing, and fissuring (Pach, 1994; Robert Betcher, Personal Communication, 2000).

Tills overlie bedrock at all of the sites examined in this work. To the east of the Sandilands area (Figure 1.1), the tills contain predominantly sand-size fractions. These sand-rich tills tend to be low in carbonate content and are derived from the Precambrian rocks of the Canadian Shield. To the west of the Sandilands area, the tills are comprised of a predominantly silt-size fraction with significant clay content. The carbonate content of the silt-rich tills are typically over 50% and are derived from the sedimentary rocks to the northwest (Teller and Fenton, 1980).

At all of the sites examined, except SE2, SW1, and NC1 where it is absent, a surficial glaciolacustrine clay unit, varying in thickness from 3 m to more than 10 m is present. The clays were deposited in Lake Agassiz during the late Pleistocene and early Holocene Epochs, and are composed predominantly of smectite and illite clay minerals (Pach, 1994). Some authors have chosen to divide the clay deposits into two units (for example Pach, 1994 after Day, 1977), an upper late Lake Agassiz deposit and a lower early Lake Agassiz deposit. The early deposit (lower unit) typically has a clay content of over 80% (Pach, 1994 after Last, 1974) and is of high plasticity. The late deposit (upper unit) contains a greater silt-size fraction, resulting in relatively lower clay content and decreased plasticity compared to the lower early deposit. The silt contained in the late deposit often occurs in laminations or varves, which are the result of cyclical depositional environments of Lake Agassiz. The late deposit is often weathered and may be extensively fractured (Pach, 1994). The thicknesses of these units are highly variable depending on location within the greater study area.

3.3 3.3 HYDROGEOLOGICAL SETTING

3.3.1 Hydroclimatology

The climate of southern Manitoba is classified as continental and is characterized by short summers with mean July temperatures of approximately 20°C and long winters with mean January temperatures of approximately –18 °C (Betcher *et al.*, 1995). From late October through to April, mean daily temperatures are typically below 0°C (Betcher, 1986). The mean annual temperature in southern Manitoba is on the order of 3.3°C. Mean annual precipitation varies from 500 mm/yr in the Interlake area (Figure 3.1) to as much as 600 mm/yr in the southeast corner of the province, with 70% of annual precipitation falling as showers between May and October. Potential evapotranspiration is on the order of 425 mm/yr in the Interlake area to 500 mm/yr south of the City of Winnipeg (Betcher *et al.*, 1995).

3.3.2 Drainage

Drainage in the southern half of the province is generally directed to Lakes Winnipeg, Manitoba, and Winnipegosis, which occupy a significant portion of the Manitoba Lowland physiographic region (Betcher *et al.*, 1995). The NC2, NC3, SC1 to SC3, SE1, SE2, and SW1 sites are all located in the Red River Basin, which directs surface water to the Red River and in turn discharges to Lake Winnipeg. The NC1 site is located within the Lake Winnipeg Basin and drainage is directed to the lake, located east of the site. The SE3 site is located at the western edge of the Winnipeg River Basin. Drainage in this basin moves east to the Winnipeg River which in turn, discharges to Lake Winnipeg to the north (Betcher *et al.*, 1995).

3.3.3 Hydrogeology

Bedrock and local overburden aquifers supply much of rural southern Manitoba with water for domestic and agricultural purposes. Throughout the study area, glacial and glaciolacustrine till and/or clay units form semi-confining aquitards. Much of the recharge to local surficial aquifers is from downward flow as infiltration through the overlying aquitards (Keller *et al.*, 1986; Betcher *et al.*, 1995). The lack of topographic relief at each of the sites examined, particularly in the Manitoba Lowlands region, but also locally, indicates that downward flow likely predominates through the aquitard materials, primarily as infiltrating recharge to the underlying aquifers. In areas of greater topographic relief, for example the hummocky areas of the Precambrian Shield physiographic region, local topographically driven flow systems may develop. Local recharge areas are typically associated with topographic highs and discharge areas at topographic lows (Betcher *et al.*, 1995). The location of the piezometric surface is often shallow, varying from above ground surface to depths of more than 3 m in the clays and tills, and is subject to considerable seasonal and yearly variation within this range.

3.3.3.1 Aquifers

Carbonate Bedrock Aquifers

The major aquifer in the Manitoba Lowlands physiographic region of the study area is highly fractured and paleokarstic carbonate bedrock, typically associated with the upper portion of these units (Render, 1970; Betcher *et al.*, 1995). Groundwater in the carbonate bedrock aquifer(s) moves primarily through secondary permeability features such as fractures, joints and solution cavities. These features are locally variable in both distribution and frequency. In the Winnipeg area the upper 15 m to 30 m of the Red River Formation sub-crop, consisting of limestone and dolostone, is the predominant water-bearing unit (Render, 1970). However, significant fracture zones have been encountered at greater depths (Betcher *et al.*, 1995). The carbonate bedrock aquifer(s) form the most extensive aquifer system in the province and has highly variable, but often significant yields of over 1×10^{-2} m^3/s (Betcher *et al.*, 1995; Grasby and Betcher, 2002).

Overall water quality in the carbonate bedrock aquifer(s) is potable in the south, to the east of the Red River, in the north, and to the east of Lakes Manitoba and Winnipegosis. To the west of these locations, the carbonate bedrock aquifers(s) contain high total dissolved solids (TDS), and water quality ranges from brackish to saline (Betcher *et al.*, 1995). From east to west, TDS is known to vary from 240 mg/L in the east to as much as 230 000 mg/L in the west (Grasby and Betcher, 2002).

Precambrian Bedrock Aquifers

In the Precambrian Shield physiographic region, groundwater supplies may be drawn from the fractured zones within the Precambrian bedrock if a suitable overburden aquifer is not present. Groundwater movement in these zones is controlled almost entirely by secondary permeability features such as fractures, joints, and faults. Yields are generally low, with the majority of wells yielding less than 2×10^{-4} m^3/s. However, the variable distribution and frequency of secondary permeability features can cause a significant variability in yields, both locally and regionally (Betcher *et al.*, 1995).

Groundwater quality is typically lower in TDS and potable at depths less than 150 m below bedrock surface. TDS tends to increase with depth, and water becomes increasingly non-potable with depth because of slower overall groundwater movement at greater depths. Areas of higher TDS (on the order of 1000 mg/L) have been observed at shallow depths and have been attributed to geochemical development as recharge water infiltrates through overlying clay-rich overburden (Betcher *et al.*, 1988).

Sand and Gravel Aquifers

Sand and gravel lenses and layers are widely distributed throughout the tills and clays of southern and western Manitoba and may occur as thin unconfined units to underlying units. It is estimated that approximately 35% of the wells drilled in the province are completed in sand and gravel overburden aquifers (Betcher *et al.*, 1995). In the Precambrian Shield physiographic region, sand and gravel aquifers are associated with moraines and glaciofluvial outwash deposits (Teller and Fenton, 1980; Betcher *et al.*, 1995). Sand and gravel aquifers in the Manitoba Upland and Manitoba Lowland physiographic regions are associated with moraines, outwash, deltaic, and beach deposits. The thickness of the units is highly variable along with yields, which are typically on the order of 5×10^{-3} m^3/s (Betcher *et al.*, 1995).

Typically, groundwater quality within the sand and gravel aquifers is potable. TDS concentrations are typically on the order of 300 mg/L to 500 mg/L. However, where the aquifers are overlain by significant clay-rich overburden units, overall water quality declines and TDS concentrations may exceed 1000 mg/L (Betcher *et al.*, 1995).

3.3.3.2 Glacial and Glaciolacustrine Sediments

The clay and till materials overlying bedrock and overburden aquifers in the study area form semi-confining aquitards. Pach (1994) reports field tests of hydraulic conductivity average in the range of 10^{-8} m/s for the tills, while the clay is typically in the range of 10^{-10} m/s (Pach, 1994). These values are consistent with the reported hydraulic conductivities determined in field tests by other researchers, such as Cherry *et al.* (1971) who also report a field estimated hydraulic conductivity of 3×10^{-7} to 3×10^{-5} m/s for the sand and sand-rich tills found in the eastern portion of southern Manitoba.

Generally, there is an assumed ability of clay and till aquitards to protect underlying groundwater supplies from contamination as a result of the high sorption capacity and typically low groundwater velocities (Day, 1977). For example, Woodbury (1997) stated that in the Winnipeg area, the local thick clay and till aquitard was long thought to provide a buffer to the downward migration of contaminants (after Baracos *et al.*, 1983). However, instances of contamination have been discovered at far greater depths than expected (Sabourin, 1989). This movement has been attributed to hydraulically active fracture networks, which can provide a direct connection to underlying aquifers.

Clay and till overburden units in the Prairie Provinces have been divided into an upper weathered zone and a lower unweathered zone (for example, Hendry, 1988 and Pach, 1994). Fracturing and fissuring of both clay and till aquitard materials has generally been associated with the upper weathered zone, often distinguished by brown oxidation staining and extending to depths of up to 10 m below ground surface (Keller *et al.*, 1986). Hendry (1982) reports the weathered zone extending to depths of as much as 18 m in a thick Alberta clayey till. Bulk hydraulic conductivity of clay/till in the weathered zone can be significantly greater than the bulk hydraulic conductivity of the same unit in the unweathered zone. Keller *et al.* (1986) provide evidence that in some cases, the unweathered zone of aquitards can also be extensively fractured, and similar conclusions were drawn by Rowe and Booker (1990), Pach (1994), and Shaw and Hendry (1998). Grisak and Cherry (1974), in their study of glacial till and glaciolacustrine clay of southeastern Manitoba, indicate that fractures in clays and till control the bulk conductivity of these materials rather than intergranular permeability.

3.4 GEOLOGY AND HYDROGEOLOGY OF THE STUDY SITES

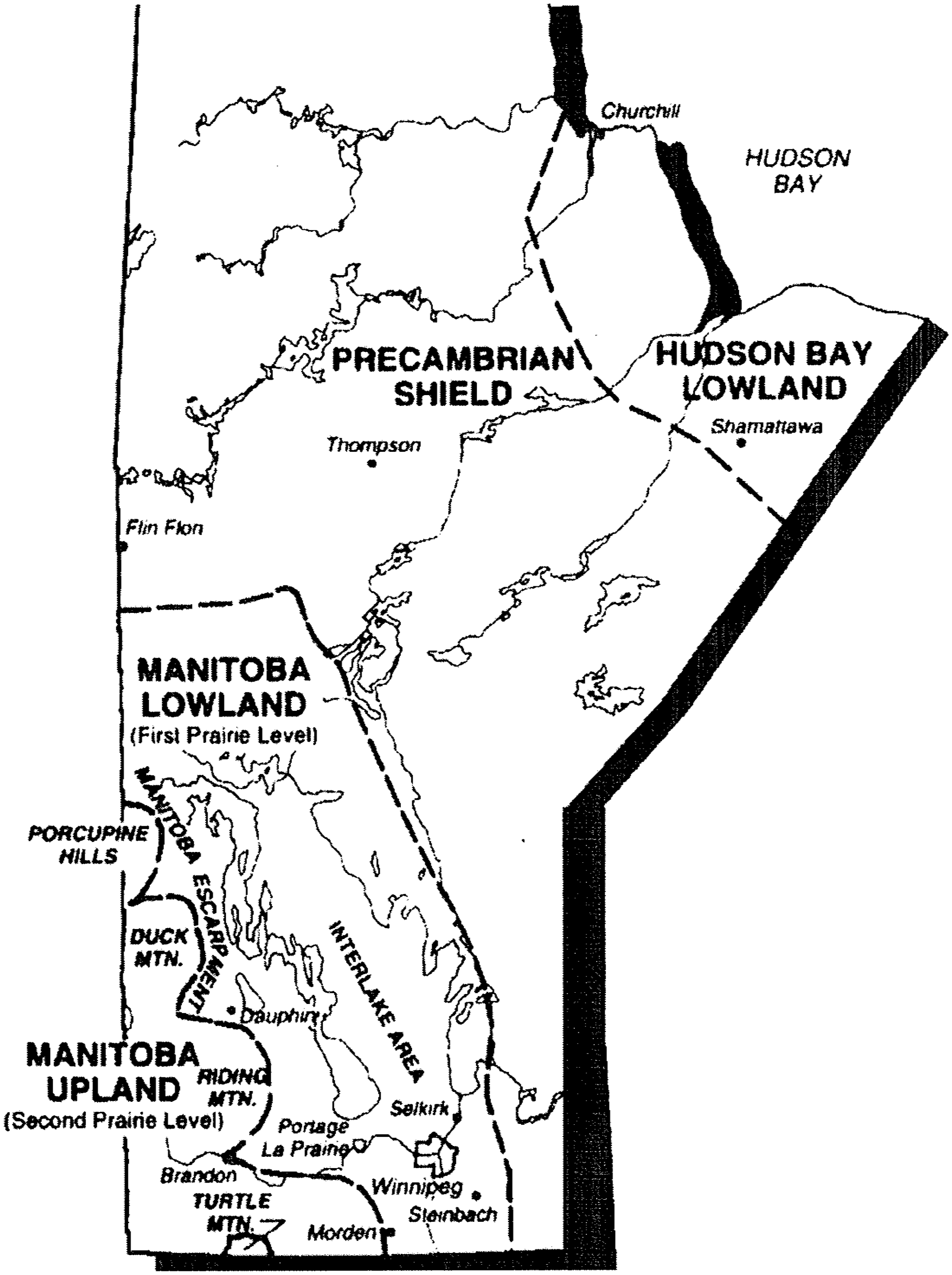
The NC1, NC2, and NC3 sites are underlain by the Ordovician Red River Formation, consisting of dolostone and limestone with argillaceous dolostone and shale interbeds. Overburden in the vicinity of the NC1 site is approximately 35 m thick and consisting of silt till extending from ground surface to the underlying bedrock. Just north of the site, the surficial deposit transitions from silt till to clay. The NC2 and NC3 sites are located in an area with highly variable overburden thickness, varying from 6 m to 22 m. The overburden consists of glaciolacustrine clay underlain by a thin layer of till. The most predominant aquifers in the NC1, NC2, and NC3 areas are upper carbonate bedrock aquifers. Regional groundwater flow at the NC1 site is to the east, towards Lake Winnipeg which is located approximately 15 km east of the site. Regional groundwater flow at the NC2 and NC3 sites is to the southeast, towards the Red River which is located approximately 25 km to the southeast of the sites. The piezometric surface at the NC1 site is located at approximately 10 m below grade, while the piezometric surface at the NC2 and NC3 sites is located at or immediately below grade (Betcher, 1986; GWDrill Database).

The dolostones and limestones of the Ordovician Red River Formation also underlie the SE1, SC1, SC2, and SC3 sites. Overburden in the area of the SE1 site varies from approximately 25 m to 35 m and consists of a 5 m to 10 m thick clay unit underlain by clay till and/or silt till. Overburden near the SC1 site is approximately 20 m to 30 m thick and consists of a significant 8 m to 18 m thick clay unit underlain by clay till and/or silt till. Overburden in the vicinity of the SC2 and SC3 sites is approximately 29 m thick and consists of clay underlain by clay till and/or silt till. The most predominant aquifers in use in the vicinity of the south-central sites are carbonate bedrock aquifers. The regional direction of groundwater flow is to the west-northwest, towards the Red River and the City of Winnipeg. The SE1, SC1, SC2 and SC3 sites are all located within a 40 km radius of the City of Winnipeg. The SE1 site is approximately 30 km west of the Red River, while the SC2 and SC3 sites are approximately 20 km west of the Red River and the SC1 site is located within 2 km of the Red River. The piezometric surface is located at depths ranging from 1 m to 5 m below grade in the south-central area (Little, 1980; GWDrill Database).

The Odanah Member of the Riding Mountain Formation underlies the SW1 site. The Odanah Member bedrock consists of grey to grey-green hard to soft shale. Overlying bedrock is approximately 55 m of interbedded clay till to silt till units and sand and gravel units. A transition in the surficial deposits from sand to till occurs in the vicinity of the SW1 site. A number of drinking water wells in the area are completed in localized sand and gravel units, and the piezometric surface varies from 2 m to 13 m below grade (Sie *et al.*, 1976; GWDrill Database).

Bedrock underlying the SE2 site is the Melita Formation consisting of varicoloured shales, friable quartz sand, hard calcareous quartzose sandstone, and minor limestone. Overburden in the vicinity of the site is approximately 90 m thick and consists of silt till to clay till units interbedded with sand and gravel units of varying thickness. A number of drinking water wells in the area are completed in localized sand and gravel units, and the piezometric surface varies from 1 m to 3 m below grade (Betcher, 1986; GWDrill Database).

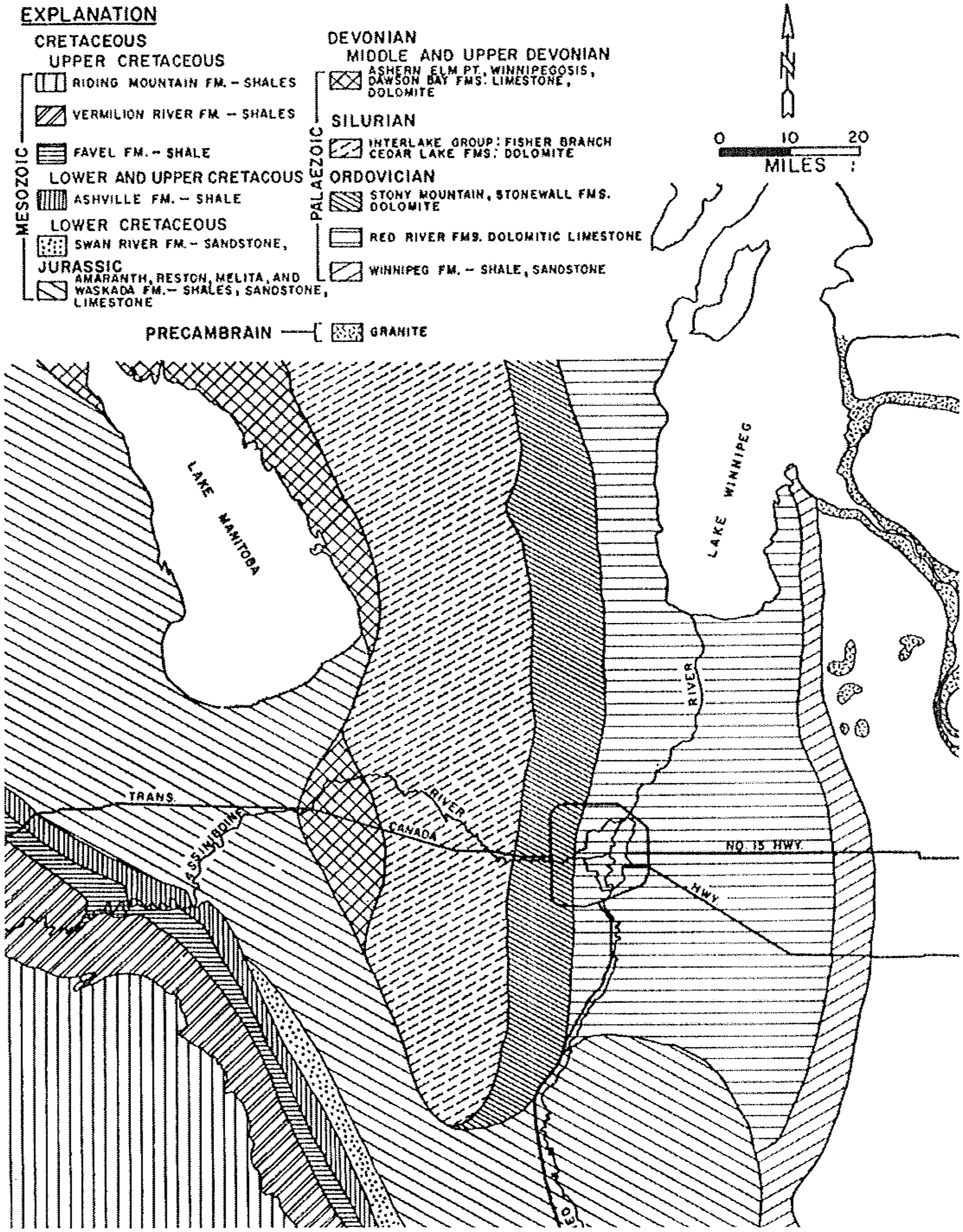
Bedrock underlying the SE3 site consists of Precambrian igneous and metamorphic rocks (Betcher *et al.*, 1995). Overburden in the vicinity of the site is approximately 20 m and consists of clay underlain by clay till. A number of the wells in the area are screened in Precambrian bedrock, and the piezometric surface is approximately 2 m to 5 m below grade in the area (GWDrill Database).



Source: Betcher *et al.*, 1995

Figure 3.1: Physiographic Regions of Manitoba

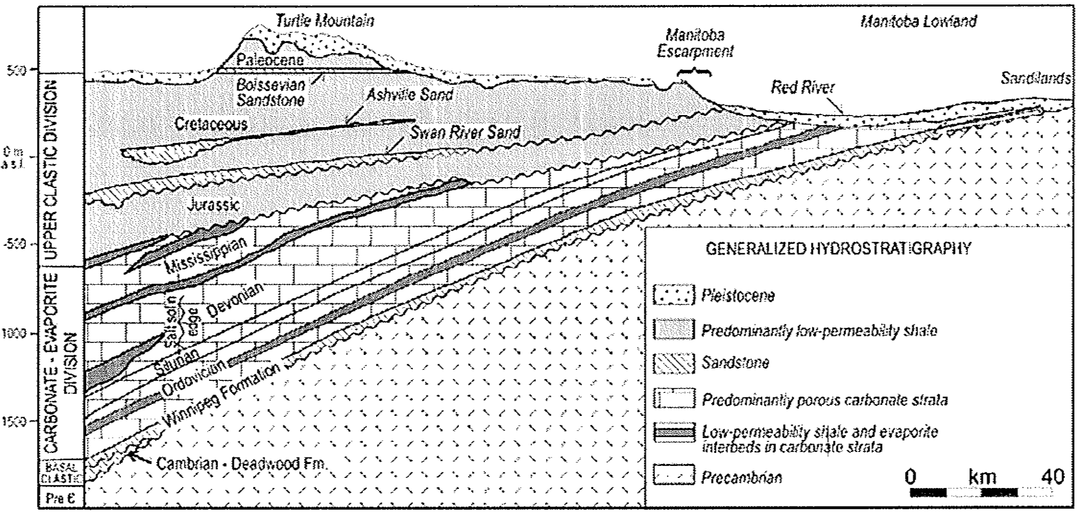
The sites examined are located within the Manitoba Upland, Manitoba Lowland and Precambrian Shield physiographic regions.



Source: Render, 1970

Figure 3.2: Simplified Surficial Bedrock Units

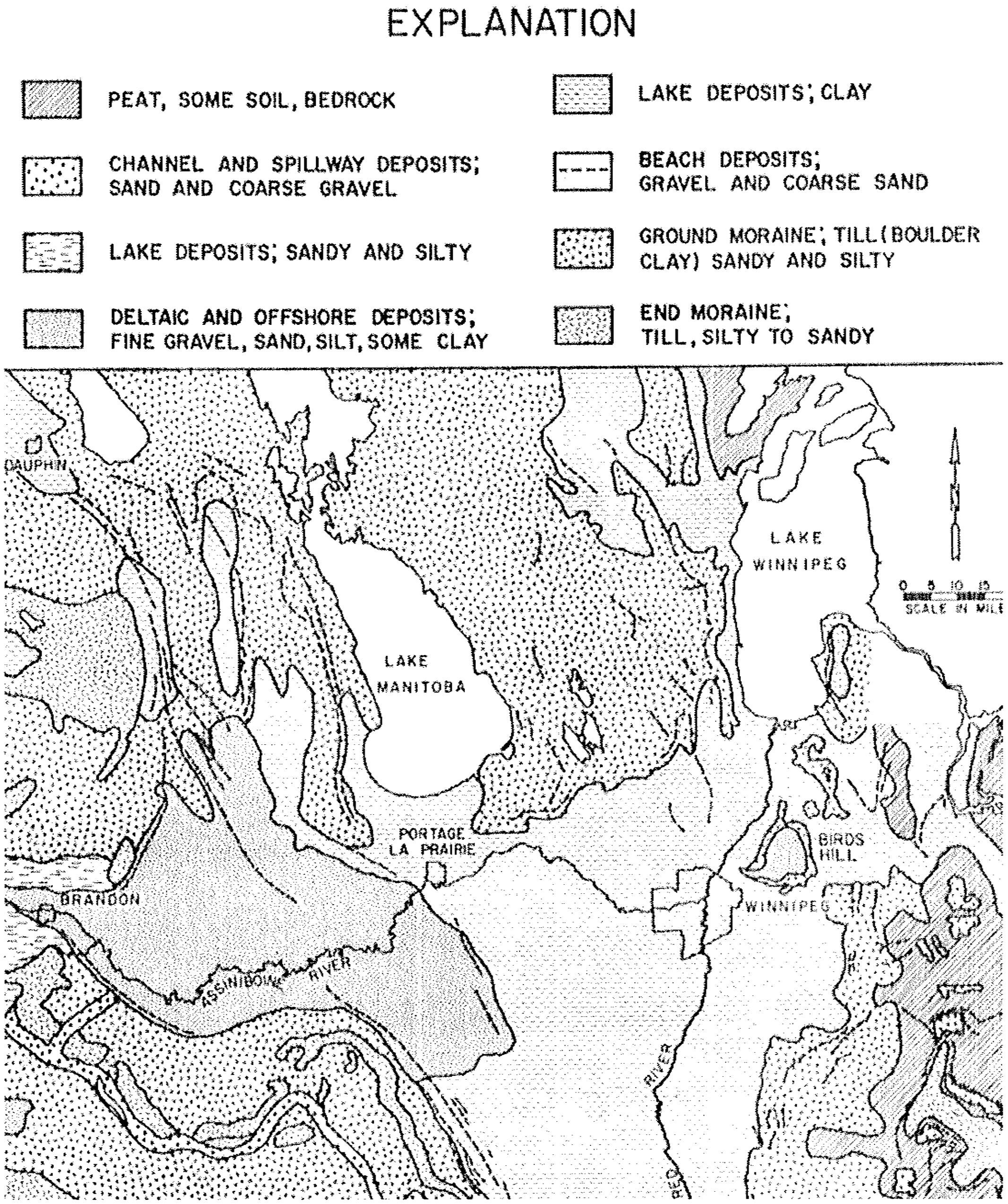
The majority of the sites examined are underlain by carbonate bedrock. To the east of the Winnipeg formation subcrop, surficial bedrock units consist of Precambrian igneous and metamorphic rocks (Betcher *et al.*, 1995).



Source: Grasby *et al.* (2002) after Simpson *et al.* (1987)

Figure 3.3: Hydrostratigraphic Cross-section

East-west transect along Manitoba-Minnesota/North Dakota border, from the Sandilands area in the east to the Saskatchewan border in the west, showing generalized hydrostratigraphy.



Source: Render, 1970

Figure 3.4: Surficial Deposits

Surficial clays in the central regions of Manitoba were deposited from Lake Agassiz. The clays typically contain a significant silt size fraction and are underlain by clay and silt tills.

CHAPTER 4: FIELD AND LABORATORY METHODOLOGY

4.1 FIELD PROGRAM

The extent of the field program undertaken at each of the ten study sites was contingent on the degree of cooperation offered by the facility owner and/or operator and accessibility for drilling. As a result, the extent of the field program varied from site to site. The activities undertaken as part of the field program for each of the ten sites are summarized in Table 4.1. A full program included:

- collection of multiple EMS sample cores;
- collection of a background sample core;
- installation of a piezometer nest;
- groundwater monitoring and sampling;
- wastewater sampling; and,
- test pit excavation and fracture mapping.

At the time the first EMS sample cores were collected from the NC1, NC2, NC3, SC1, SC2, SC3, SE1, SE2, SE3 and SW1 sites, the observed seepage profiles represented 4, 19, 29, 9, 20, 20, 14, 3, 14 and 18 years of facility operation, respectively (see Table 4.1). Secondary cores collected at the NC1, SC2 and SE2 sites represented seepage profiles developed at that location after 7, 21 and 6 years of facility operation, respectively. Three EMS sample cores were collected from different locations within the

SC3 EMS base during the same sampling event, each of which represents approximately 20 years of facility operation.

4.1.1 EMS and Background Continuous Sample Core Collection

Both EMS and background sample cores were collected using a split spoon sampler through hollow stem augers using track-mounted and backhoe-mounted drill rigs. The exception to this standard core collection method was the first EMS sample core collected from the base of the NC1 EMS, which was collected by hand-driving Shelby tubes from the base of the EMS just after the stored wastewater had been removed. Sample cores were collected to assess the impact of the EMS on underlying soils and groundwater.

EMS sample cores were generally collected during the winter from an ice platform. A track-mounted rig was used to collect continuous EMS sample cores where ice thickness on the storage was greater than 0.6 m (Figure 4.1). Where ice thickness were less than 0.6 m and/or where steep side-slopes made access impossible, a backhoe-mounted rig was used (Figures 4.2 a, 4.2 b, and 4.2 c). In both cases, a hole was first drilled through the ice and the hollow stem augers were then lowered into the hole until they met resistance at the base of the storage. Plastic was wrapped around the drill bit to ensure liquids did not enter the augers. Once at the base, augers were advanced 0.3 m to 0.6 m below the EMS base, in order to seal the base of the augers into the underlying sediments, and create a casing through which samples could be collected. Where leakage of wastewater into the augers was anticipated once the plastic seal was broken, particularly in low to non-cohesive materials, a layer of granular bentonite was placed in the hole to ensure reliability of the seal and minimize contamination. Split spoon samples were collected from the base of the EMS to a final predetermined depth, which was based on plug-flow estimates of seepage rates, or until refusal was encountered.

Background sample cores were collected at a location 50 m to 100 m from the outer edge of the EMS berms, in an area where there was no history of fertilizer application and that was estimated to be beyond any seepage influences presented by the EMS facility. Background cores were collected through hollow stem augers using split spoon samplers and a track-mounted drill rig to a depth consistent with the maximum sampling depth of the EMS sample cores (Figure 4.3). As set out in Table 4.1, background cores were collected from seven of the ten sites (NC1, NC2, NC3, SC2, SC3, SE2 and SE3).

Each split spoon sample collected from the EMS and/or background hole was sectioned into 0.05 m to 0.15 m sub-samples, pared and sealed in air-tight plastic bags. Split spoon samples, collected from below the base of the EMS, were typically sectioned into 0.05 m sub-samples near the top of the soil profile and 0.15 m sub-samples at depths where the soils were unlikely to be impacted by the EMS. This ensured greater detail of concentration of chemical species at shallow depth in order to capture where changes in chemical concentration occurred. All soil sub-samples were placed in a cooler and stored with ice packs until transported to a commercial laboratory in Winnipeg, Manitoba for chemical analysis. Each of the EMS holes was back-filled with a granular (pellets) swelling bentonite seal on completion.

4.1.2 Piezometer Nest Installation

Each background hole was completed as a piezometer nest, consisting of two to three piezometers. Piezometers were constructed from 25 mm inside diameter polyvinyl chloride (PVC) pipe. The base of the piezometer was fitted with a screen of approximately 0.6 m length, consisting of no. 10 slot size (0.254 mm) machine-slotted PVC screen. Screened intervals were sheathed with a geotextile filter sock to reduce the migration of fines into the piezometer. The annulus of each piezometer nest was back-filled with clean graded silica sand from the base of a given piezometer to 0.1 m to 0.3 m above the slotted screen. The remainder of the annulus was back-filled with granular swelling bentonite seal and drill cuttings. The annulus between any two piezometers was filled with granular swelling bentonite seal to reduce the likelihood of hydraulic short-circuiting between piezometers within the installation. End plugs and lockable aboveground steel casings were installed on each of the piezometer nests. A typical installation schematic is provided in Figure 4.4.

Piezometer nests were installed at the NC1, NC2, NC3, SC2, SC3, SE2 and SE3 sites. In general, one piezometer was screened at a location coincident with the approximate base of the EMS and one piezometer was screened at a location below the base of the EMS. Piezometer nests were installed to provide information on shallow background groundwater quality, background vertical hydraulic gradients and a means of estimating in situ hydraulic conductivity.

Prior to any head measurements, recovery testing or water quality sampling, the piezometers were developed by purging until dry. When possible, piezometers were developed directly after construction to remove any sediment in the piezometer and surrounding annulus. Otherwise, piezometers were developed shortly after installation after sufficient water levels had developed in the installation. Purging of piezometers was completed using either a dedicated inertial lift pump, consisting of Waterra-type tubing and a foot valve, or a dedicated disposable bailer. Water levels were allowed to re-equilibrate over a nine-month period before any water level measurements were made. Piezometers were again purged of any standing water prior to groundwater quality sampling using the dedicated equipment and allowed to recover to a sufficient level to facilitate sampling. This ensured that only fresh formation water was sampled.

4.1.3 Groundwater Monitoring

Groundwater levels were measured a minimum of three times during the course of a one-year period, to establish an estimate of the seasonal water table location and the natural vertical hydraulic gradient at the site. Water levels were established as a depth below the top of the piezometer casing using a water level meter, from a fixed location at the top of the casing. Water level readings are considered to be accurate to ± 0.001 m.

The water level meter was decontaminated between piezometers using distilled water rinses to prevent cross-contamination.

The vertical hydraulic gradient between any two piezometers within a single piezometer nest was calculated as follows:

$$[4.1] \quad i_z = \frac{h_s - h_d}{\Delta L}$$

where i_z is the average vertical hydraulic gradient between the mid-point of the screened interval of any two piezometers, h_s is the water level in the shallow piezometer, h_d is the water level in the deep piezometer and ΔL is the distance between the mid-point of the screened interval of the two piezometers (Freeze and Cherry, 1979). A positive vertical gradient represents downward groundwater flow and a negative vertical gradient represents upward groundwater flow.

4.1.4 Recovery Testing

Falling head tests were conducted in each of the piezometers at the NC1, NC2, NC3, SE3, SC2, and SE2 sites, to provide an estimate of the in situ hydraulic conductivity of the main geologic units of interest at the sites. The number of falling head tests (slug tests) conducted at each of these sites is provided in Table 4.1. Prior to the start of the test, the static water level was recorded using a water level meter. At more distant sites, where slow recovery was anticipated, a 34.5 kPa (5 psi) pressure transducer and data logger was used to record the recovery of water levels.

Slugs were constructed from 12.5 mm diameter PVC pipe, 0.6 m in length, filled with sand and sealed (Figure 4.5). The displacement volume of the slug was established prior to testing and seals were checked to ensure integrity during the test. The slug was introduced into the piezometer, inducing a near instantaneous known displacement of water, and water levels were recorded at increasing time intervals until at least 63% recovery of the static water level was achieved. For sites where the pressure transducer was used to record the recovery of water level, a solid slug was attached to and introduced simultaneously with the pressure transducer. Both the water level meter and pressure transducer were decontaminated between piezometers using distilled water rinses.

Results of the falling head tests were analysed according to the method of Hvorslev (1951), which is based on the time required for water levels to recover to equilibrium after the introduction or removal of a known volume of water. The method estimates the hydraulic conductivity of geological materials in the immediate vicinity of the sand pack, and is calculated as follows:

$$[4.2] \quad K = \frac{r^2 \ln(L/R)}{2LT_o}$$

where r is the inner-radius of the piezometer, L is the screen or sand-pack length, R is the radius of the annulus and T_o is the time required for the piezometer to recover 63% of the equilibrium water level. The equation is applicable for cases where $L/R > 8$, where the piezometer is not fully penetrating and where the screened interval and/or sand pack is installed within the saturated zone (Fetter, 1994). In cases were the sand pack or screen lies partially in the unsaturated zone, the method will overestimate the hydraulic conductivity of the saturated porous medium.

The method of Hvorslev (1951) may cause an overestimate in hydraulic conductivity since it assumes that the specific storage of the medium is zero. In addition, the method assumes that the medium is homogeneous. In the case of fractured materials, this assumption may not be valid and may lead to inaccuracy of the estimate (Pach, 1994 after Palmer and Paul, 1989).

4.1.5 Groundwater Sampling

Groundwater samples were collected from each piezometer installed at the NC1, NC2, NC3, SC2, SC3, SE2 and SE3 sites. The samples were collected in order to provide shallow background groundwater quality in the areas studied. Dedicated disposable bailers or Waterra-type inertial lift pumps were used for sampling activities in the piezometers. All groundwater samples were placed directly into the appropriate laboratory supplied container and stored in a cooler with ice packs for transport to a commercial laboratory. Groundwater samples being submitted for laboratory metals analysis were field filtered using 0.45 μm filters prior to placing the sample in the laboratory supplied container. All groundwater samples were transported to a commercial laboratory in Winnipeg, Manitoba for chemical analysis.

4.1.6 Wastewater Sampling

Wastewater samples were collected from each of the EMS facilities examined in this work. The samples were collected in order to provide estimates of source concentrations of the chemical species of interest. Samples were collected from the surface of the EMS facilities by immersing a clean laboratory sample container below the surface of the stored liquids and then opening the container and allowing it to fill. All wastewater samples were stored in a cooler with ice packs until transported to a commercial laboratory in Winnipeg, Manitoba for analysis.

4.1.7 Fracture Mapping

In the fall of 2001, two test pits were excavated at the SC2 and the NC1 sites for the purpose of fracture mapping. The test pits were designed in accordance with Province of Manitoba standards (Guidelines for Excavation Work, Workplace Safety and Health Division, Manitoba Labour and Immigration), and the original pit design is provided in Figure 4.6. The design provided for three vertical faces located from a depth of 2 m to 3 m below ground surface. Actual pit excavation varied somewhat from the original design. Both the SC2 and NC1 test pits were excavated with 1:1 side slopes to a depth of 2 m below ground surface and four vertical faces extending to a depth of 3 m below ground surface.

Although significant slickenside structures were noted in the SC2 test pit, no fracturing was evident on the vertical faces of the pit, and smearing of the sidewalls was evident during the excavation (Figure 4.7). Favourable excavation conditions and minimal smearing allowed all four vertical faces to be mapped in the NC1 test pit (Figure 4.8 a). Prior to fracture mapping, vertical faces were scraped by hand to reduce the effect of smearing caused by the backhoe during excavation and to expose fracture faces for strike and dip measurement (Figure 4.8 b and 4.8 c).

Strike and dip of major joint sets at the NC1 test pit vertical faces were recorded in the field, as well as spacing between the individual joints of a set. Observations regarding the potential oxidation state of the fractures, precipitates on fracture faces and fracture continuity were recorded. In addition, observations were recorded regarding material transitions, depth of root penetration and degree of weathering with depth within the test pit.

4.2 LABORATORY PROGRAM

Laboratory chemical analysis included soil samples sectioned from the EMS and background sample cores, water samples removed from the piezometers, and wastewater samples removed from the EMS facilities. These samples were submitted to a commercial laboratory in Winnipeg, Manitoba for analysis. Additional analyses included determination of grain size distribution, and hydraulic conductivity, analyses conducted at both the Manitoba Highway's Soils Laboratory and the University of Manitoba, Department of Civil Engineering's Geotechnical Laboratory.

4.2.1 Soil Chemical Analysis

Chemical analyses of soil sub-samples were conducted by EnviroTest Laboratories and Norwest Laboratories, both of Winnipeg, Manitoba. The analyses were conducted using either saturated paste analyses or 1:2 dilution methods to determine the mass of a chemical species per unit mass of dry soil. Prior to commencing with the analyses, the soil sub-samples were dried and ground in a pulverizer. During the drying process, gravimetric moisture content was determined. Deionized water was then added to the sample. For the saturated paste analysis, deionized water was added until the sample reaches saturation, forming a thick paste. In the case of the 1:2 dilution analysis, two parts deionized water was added to one part soil, by volume. Samples were then filtered, and the filtrate was analysed for the chemical species of interest. Generally, good correlation is achieved between the two methods, although the saturated paste analysis is typically considered more accurate (Paul Routledge, EnviroTest Laboratories, personal communication 2003)

The results of the analyses both represent the total amount of a chemical species in the pore fluids and that which has desorbed from the solid soil surfaces in the sub-sample. Chemical concentrations measured for each sub-sample included: chloride (Cl);

ammonium-nitrogen and/or ammonia-nitrogen (NH_4-N and/or NH_3-N, respectively); nitrate+nitrite-nitrogen (NO_3+NO_2-N); phosphate-phosphorus (PO_4-P); and potassium (K).

4.2.2 Soil Chemistry Profile Construction

Soil chemistry profiles, or seepage profiles, were constructed by associating the laboratory results for a particular sub-sample with a single depth. The single depth was calculated as the average of the depth from which the sub-sample was collected below grade or below the base of the EMS.

4.2.3 Pore Fluid Extraction and Analysis

Pore fluids were extracted from selected sub-samples of EMS cores obtained from the SC3 site. Pore fluids were extracted by placing the sample under a vacuum pressure of approximately 1034 kPa (150 psi) and allowing fluids to drain through a maximum filter size of 1.6 μm. The filtrate was collected and submitted to EnviroTest Laboratories of Winnipeg, Manitoba for analysis of routine water quality parameters, major cations, major anions, nutrients and select metals. The analysis included the determination of Cl, NH_4-N and/or NH_3-N, NO_3+NO_2-N, PO_4-P, and K concentrations.

4.2.4 Groundwater Analysis

Groundwater samples from each of the piezometers of the piezometer nests installed at the sites outlined in Section 4.1.5, were submitted to EnviroTest Laboratories for analysis of routine water quality parameters, major cations, major anions, nutrients and select metals.

4.2.5 Wastewater Analysis

Wastewater samples from each of the EMS facilities outlined in section 4.1.6, were submitted to EnviroTest Laboratories or Norwest Laboratories for analysis of routine water quality parameters, major cations, major anions, nutrients and select metals.

4.2.6 Grain Size Distribution Analysis

Representative soil samples from the NC1, SC1, SC2 and SW1 sites were submitted to the University of Manitoba Geotechnical Laboratory or the Manitoba Highways Soils Laboratory for grain size distribution analysis. The analyses were conducted in general accordance with ASTM D for coarse-grained soils and ASTM D 422 for fine-grained soils. The results were used as a check on soil classifications and/or descriptions, and to provide a general indication of the hydraulic conductivity of the soil.

4.2.7 Hydraulic Conductivity Analysis

Triaxial hydraulic conductivity tests were conducted in general accordance with ASTM D 5084-90. The tests provide a laboratory estimate of the saturated hydraulic conductivity of the geological material tested. Representative soil samples from the NC1, SC2 and SE2 sites were submitted for triaxial hydraulic conductivity testing.

Soil specimens were prepared by trimming then installed in the apparatus. Once installed, specimens were allowed to saturate over a period of approximately seven days. After saturation, the sample was then permeated using distilled water under a constant deviator stress of 28 kPa (4 psi). Inflow and outflow quantities were recorded until the flow through the specimen was constant. The resulting estimate of saturated hydraulic conductivity was scaled to a temperature of 20°C to account for the decrease in water viscosity with increasing temperature.

4.3 QUALITY ASSURANCE/QUALITY CONTROL MEASURES

A number of quality assurance/quality control measures were established during the field program. These measures included: cleaning augers between holes; paring soils samples collected to ensure soil from the appropriate depth was analysed and that potential surface contamination was removed; using dedicated sampling equipment to collect groundwater samples from each piezometer; decontaminating water level meters between piezometers; collecting samples in clean laboratory supplied containers which were stored on ice, in coolers after collection; and delivering samples to the laboratory within acceptable holding times.

Table 4.1 Summary of Field and Laboratory Work Conducted

Site	Background Core Collected and Analysed	No. of EMS Cores Collected and Analysed	Grain Size Distribution Analysis Conducted	Laboratory Hydraulic Conductivity Analysis Conducted	Piezometer Nest Installed	No. of Water Level Measurements	No. of Slug Tests Conducted	Groundwater Sample Collected and Analysed	No. of Wastewater Sample Collected and Analysed	Test Pit Excavation and Fracture Mapping	EMS Age – EMS Sample Core Collection (in years)
NC1	Y	2	Y	Y	Y	6	6	Y	2	Y	4 & 7
NC2	Y	1	N	N	Y	6	7	Y	1	N	19
NC3	Y	1	N	N	Y	3	4	Y	1	N	29
SC1	N	1	Y	N	N	N	N	N	N	N	9
SC2	Y	2	Y	Y	Y	3	3	Y	2	Y	20 & 21
SC3	Y	3	N	N	Y	N	N	Y	1	N	20
SE1	N	1	N	N	N	N	N	N	N	N	14
SE2	Y	2	N	Y	Y	6	4	Y	3	N	3 & 6
SE3	Y	1	N	N	Y	3	2	Y	1	N	14
SW1	N	1	Y	N	N	N	N	N	N	N	18

Notes:

Y	indicates that the activity was conducted
N	indicates that the activity was not conducted or not applicable

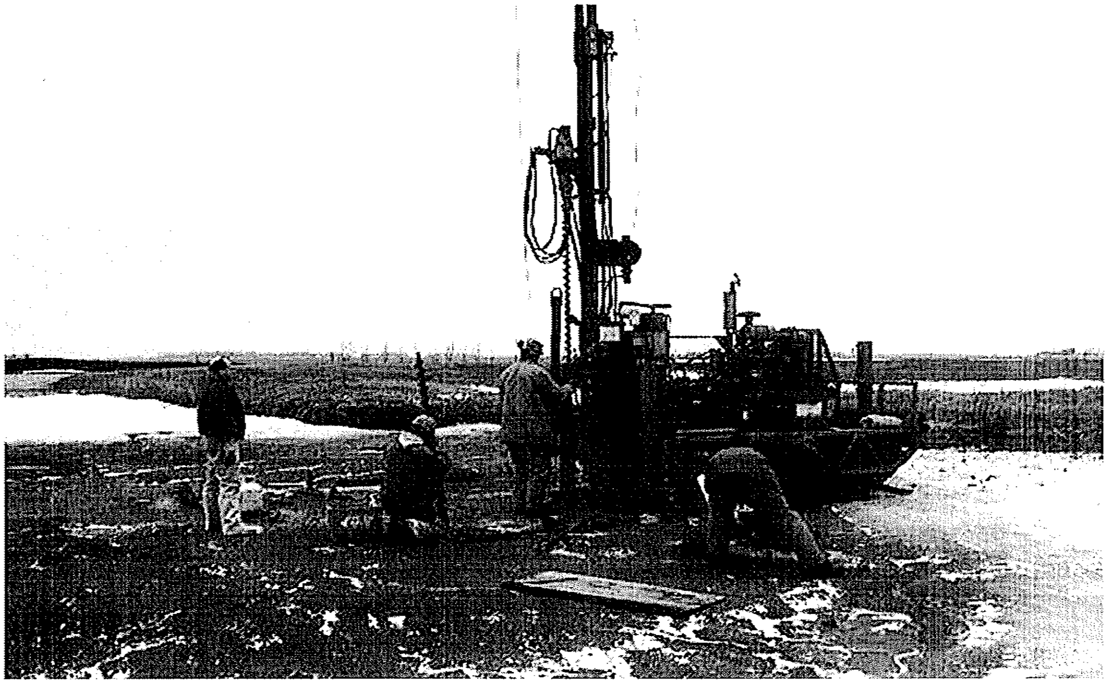


Figure 4.1: Track-mounted Drill Rig

EMS sample core removal from an ice platform using a track-mounted drill rig equipped with hollow stem augers.

Figure 4.2 a: Backhoe-mounted Drill Rig

EMS sample core removal using a backhoe-mounted drill rig equipped with hollow stem augers.

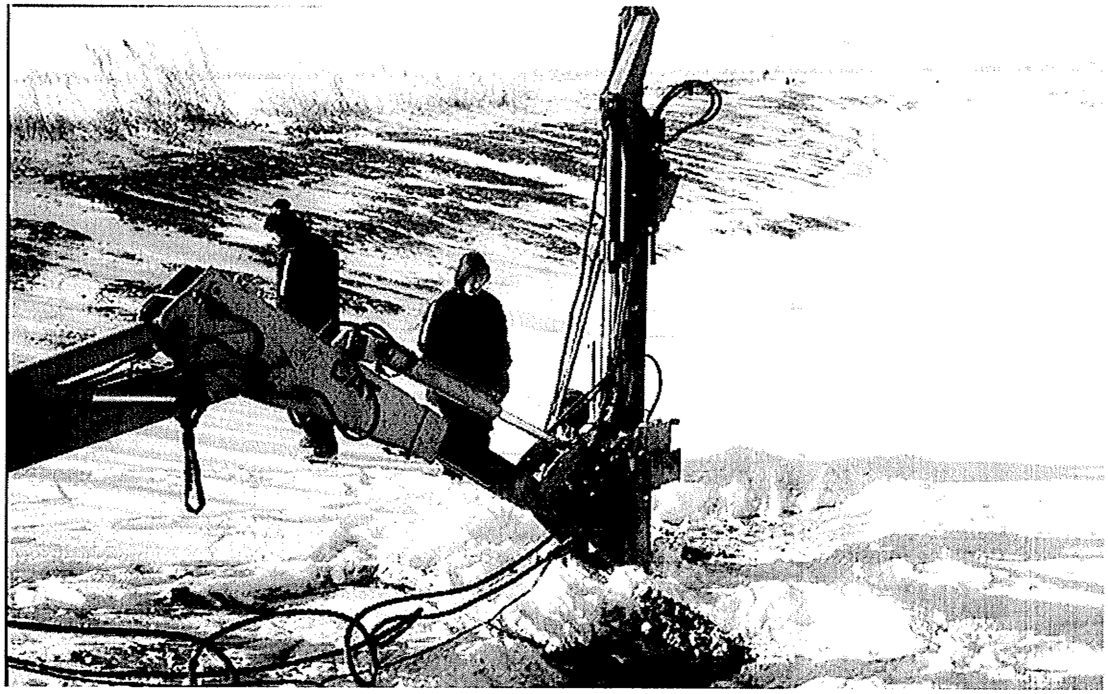


Figure 4.2 b: Backhoe-mounted Drill Rig

EMS sample core removal using a backhoe mounted drill rig equipped with hollow stem augers. Ice thicknesses of less than 0.6 m on this EMS necessitated the use of the backhoe-mounted rig.

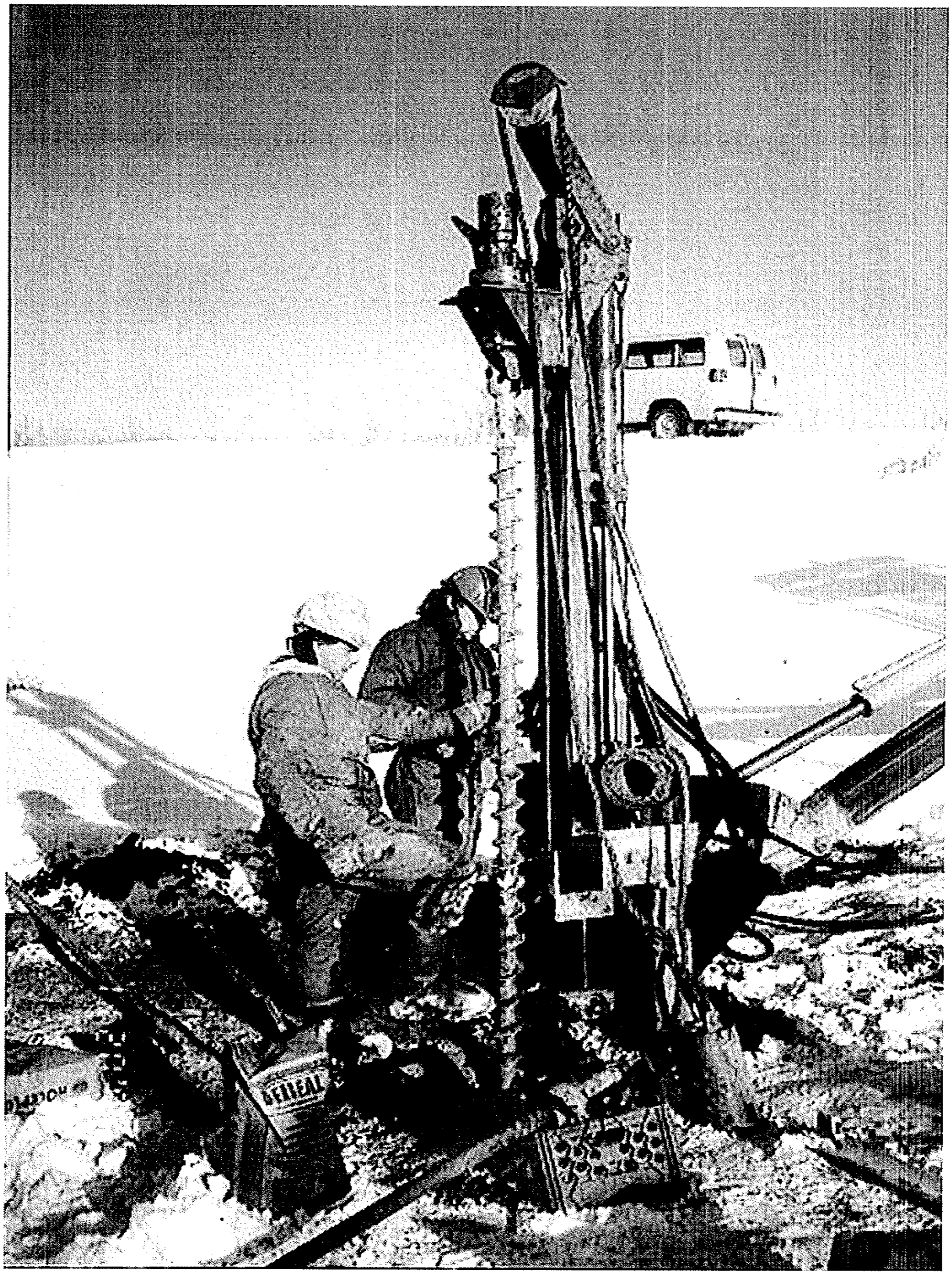


Figure 4.2 c: Backhoe-mounted Drill Rig

EMS sample core removal using the backhoe-mounted drill rig. Solid stem augers were used to ream out hole after sloughing to facilitate continuous core collection.

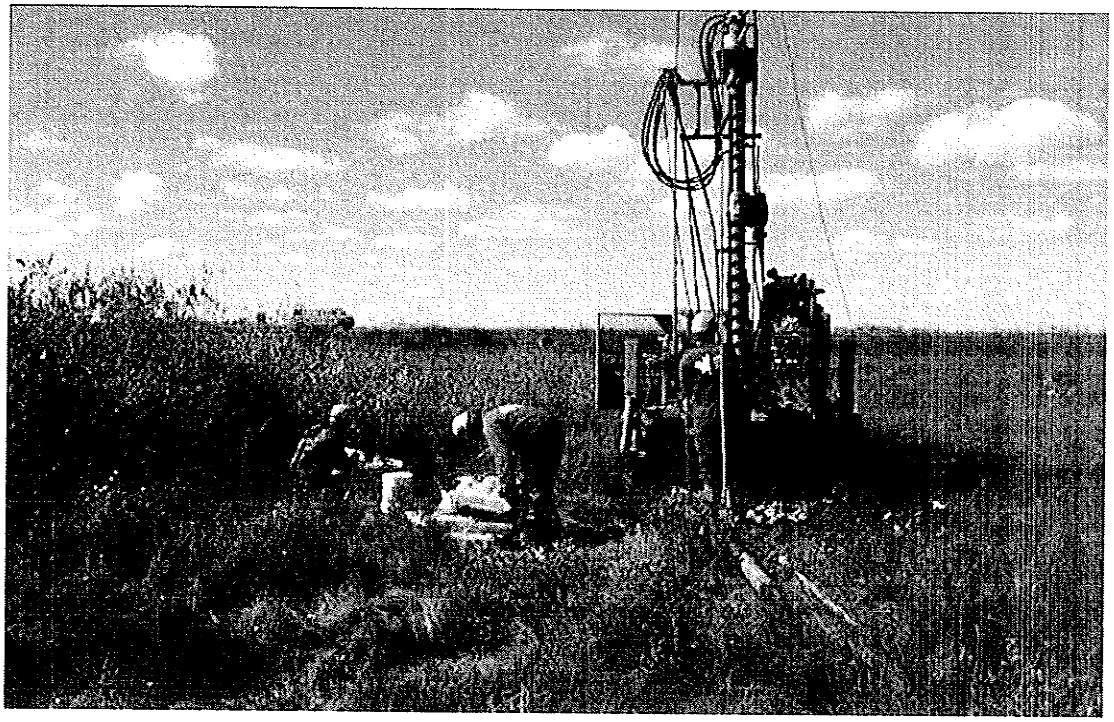


Figure 4.3: Track-mounted Drill Rig

Background sample core removal using a track-mounted drill rig equipped with hollow stem augers.

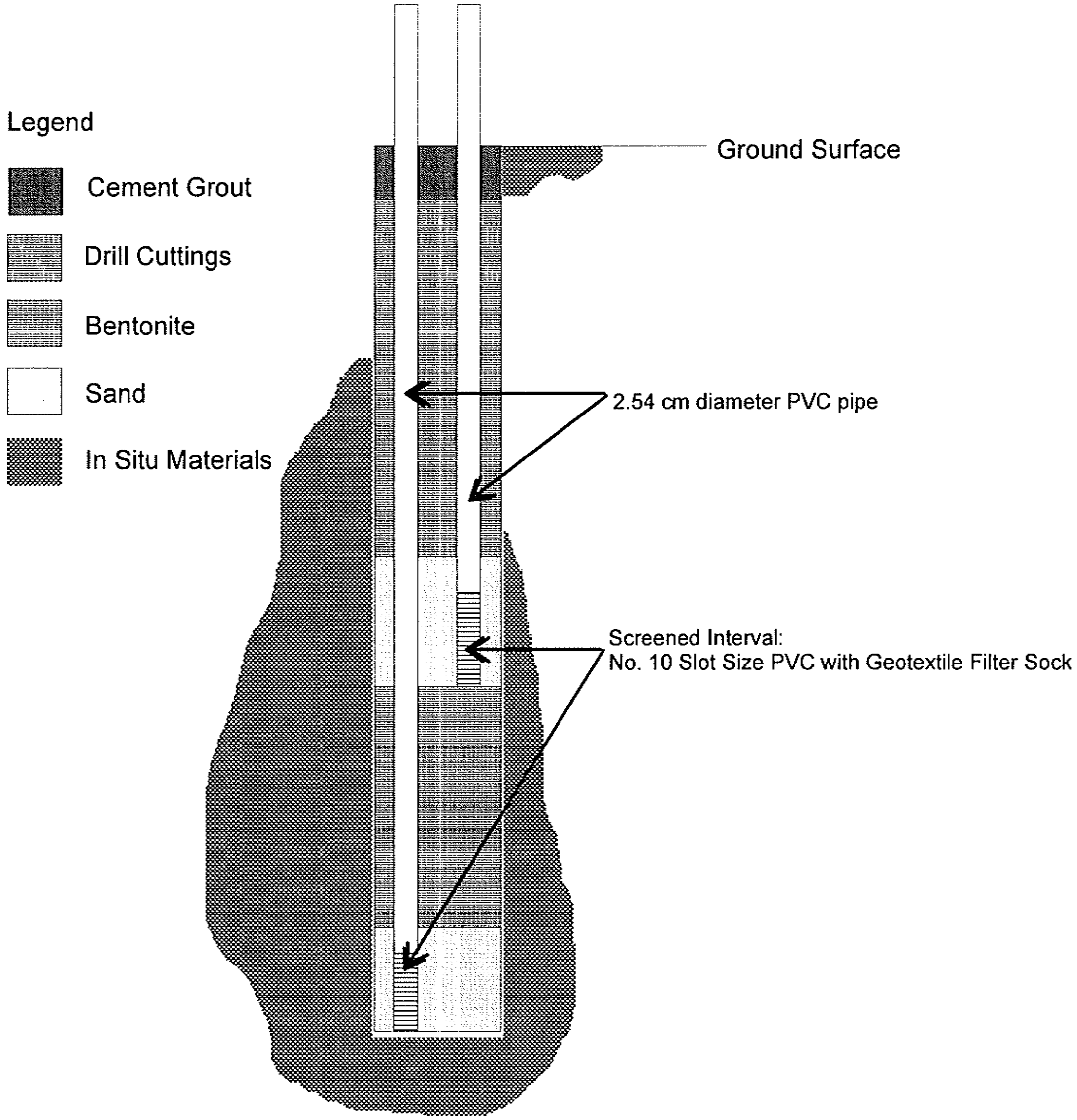


Figure 4.4: Typical Piezometer Nest Installation Schematic

Piezometer nests were installed in each of the background holes. Each piezometer nest consisted of two to three piezometers.

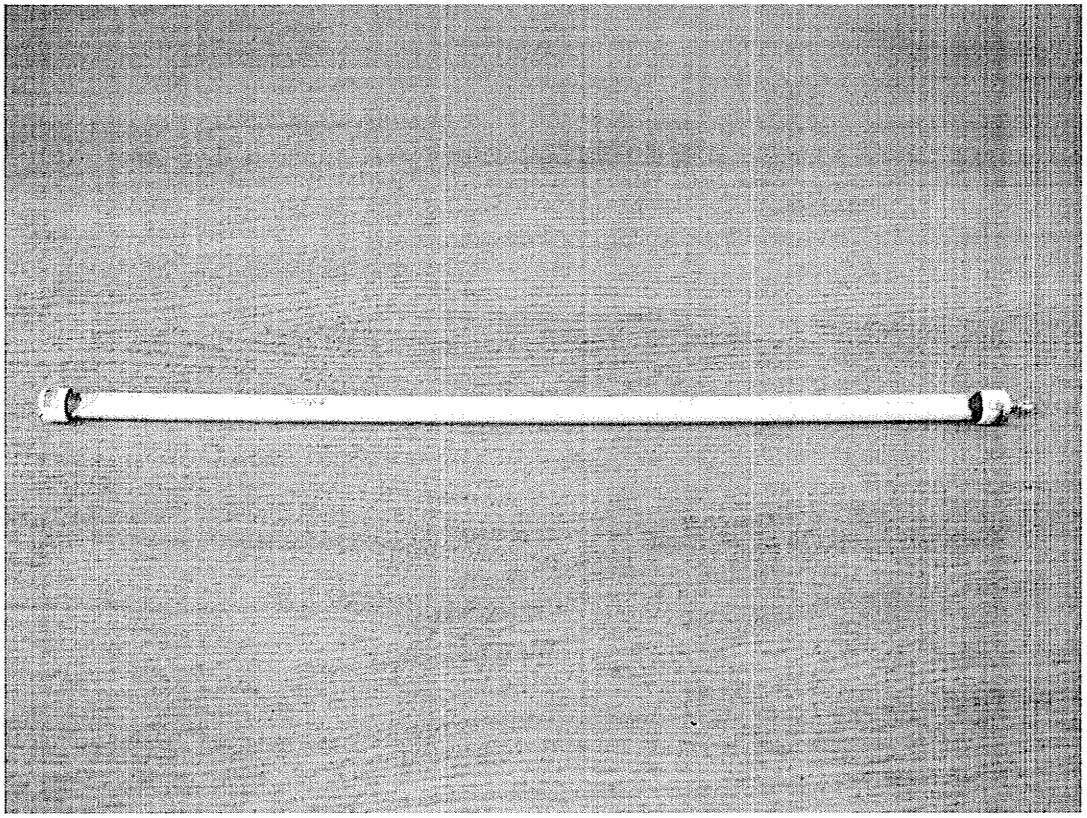
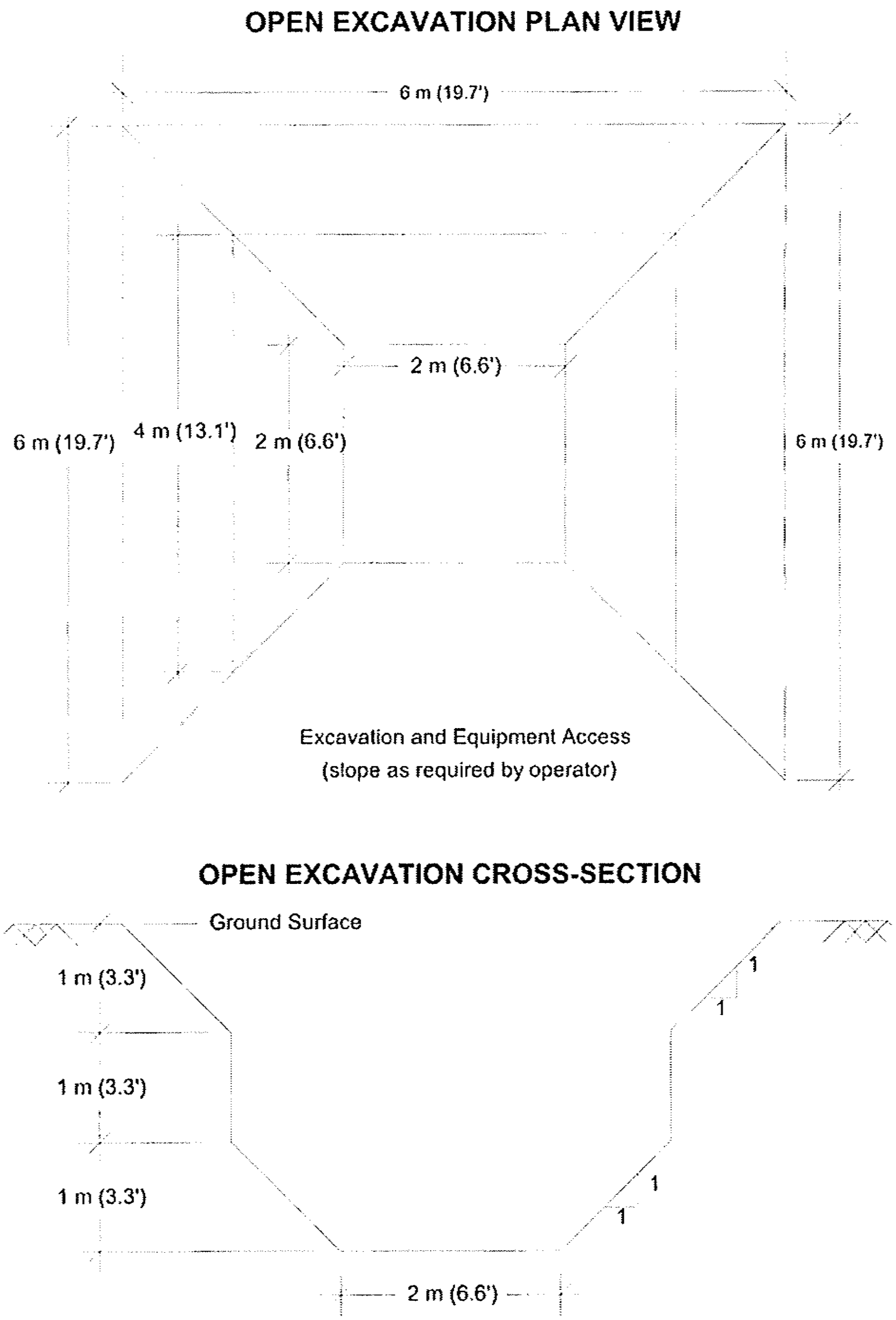


Figure 4.5: Polyvinyl Chloride (PVC) Slug

Slugs were constructed using sand-filled and sealed 1.25 cm PVC pipe. The slugs were used to conduct falling head tests in individual piezometers to estimate in situ hydraulic conductivity.



OPEN EXCAVATION PLAN VIEW

OPEN EXCAVATION CROSS-SECTION

Figure 4.6: Test Pit Design

Test pits were designed in accordance with provincial safety standards. Modifications to the test pit construction occurred in the field as a result of site specific conditions.

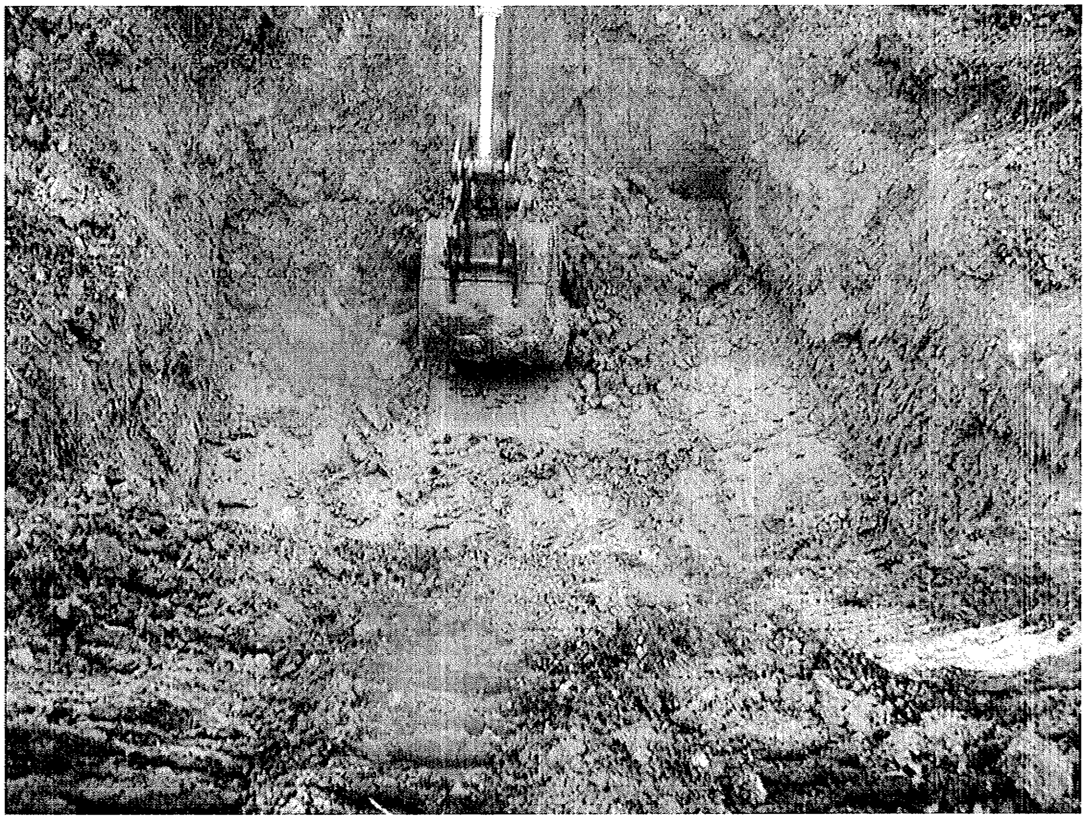


Figure 4.7: SC2 Test Pit Excavation

The SC2 test pit was excavated in clay. No significant fractures were evident on the vertical faces of the pit.

Figure 4.8 a: NC1 Test Pit Excavation

Excavating the NC1 Test Pit, located approximately 100 m east of the outer berm of the EMS facility.

Figure 4.8 b: NC1 Test Pit

After excavation of the test pit, vertical faces were scraped clean using a spatula to reduce smearing and expose fractures and other geological features.

Figure 4.8 c: NC1 Test Pit

Examining test pit walls for fracturing.

CHAPTER 5: FIELD AND LABORATORY RESULTS

5.1 HYDROGEOLOGICAL SITE CHARACTERIZATION

Geological and hydrogeological site characterization for each of the sites is discussed in detail below. Interpreted site stratigraphy from the base of the EMS to the top of the underlying aquifer is set out in Table 5.1. Table 5.2 includes the results of the grain size distribution analyses for the predominant soil types encountered in sample cores collected from the NC1, SC1, SC2, SE2, SE3 and SW1 sites. Estimated depths to water and falling head test results for the NC1 to NC3, SC2, SE2 and SE3 sites are set out in Tables 5.3 and 5.4, respectively.

5.1.1 NC1 Site

The NC1 site is underlain by approximately 35 m of till with the piezometric surface of the underlying carbonate aquifer reportedly located approximately 6 m below grade (Betcher, 1985; GWDrill Database). The site is underlain by dolomitic limestone of the Red River Formation (Betcher, 1985). A number of surrounding water wells use the upper fractured zone of the formation as a water supply (GWDrill Database). Regional groundwater flow in the area is to the east-southeast and the estimated horizontal gradient in the underlying aquifer is on the order of 1×10^{-3} m/m (based on data presented by Betcher, 1985).

Till was the predominant soil type encountered during the drilling program. The till transitions from light brown to grey at a depth of approximately 5 m below grade. The results of a grain size distribution analysis for the material collected from depths of 0.3 m and 1.1 m below the EMS base indicate that the underlying materials range from a silty clay, consisting of 48% clay, 41% silt and 11% sand by weight, to a clayey silt, consisting of 38% clay, 51% silt, 10% sand and 1% gravel by weight. The EMS facility was constructed with a clay-rich liner and as a result, the later (clayey silt) material is likely most representative of the underlying native till. Fractures were noted throughout the sample cores collected from the site.

Based on the piezometric data collected from the background nested piezometer installation over the period of March 2001 to October 2001, the water table fluctuated from 0.6 m below grade in July 2001 to 2 m below grade in October 2001. It should be noted that the background hole was located within 20 m of the north berm of the EMS facility. Falling head recovery testing was conducted on three occasions in both the shallow and deep piezometers installed in the background hole. Hydraulic conductivity ranged from 9.7×10^{-9} m/s to 4.8×10^{-8} m/s in the shallow piezometer and from 1.1×10^{-9} m/s to 4.6×10^{-8} m/s in the deep piezometer. Both the piezometers were screened in the underlying silt till.

A test pit was completed to a depth of 3 m below grade at a distance of approximately 30 m from the east berm of the EMS facility. Vertical fractures were noted in the test pit at a depth below approximately 1 m (to the final depth of exploration of 3

m) in a matrix consisting of hard, cemented and oxidized light brown till. Two predominant joint sets were observed, one trending north-south and another trending east-west. The fracture spacing for each joint set was measured and the average fracture spacing between each observed fracture, for both joint sets was on the order of 45 mm. Fractures were observed to have a halo of approximately 5 mm of reduced (grey) material perpendicular to the fracture plane.

5.1.2 NC2 Site

The NC2 site is underlain by clay followed by till, based on water well logs for the area. The overburden thickness in the area is highly variable, ranging from 6 m to 22 m. The piezometric surface is reported to be located at or very near ground surface artesian conditions have been reported (Betcher, 1985; GWDrill Database). Bedrock underlying the site consists of the dolomitic limestone of the Red River Formation. A number of surrounding water wells use the upper fractured zone of the formation as a water supply (GWDrill Database). According to a GWDrill Well Information Report for a well drilled on the subject property, carbonate bedrock is located at a depth of 7 m below grade. The direction of groundwater flow in the underlying carbonate bedrock aquifer is to the southeast under an estimated gradient of 9×10^{-4} m/m (Betcher, 1985).

Silty clay and till were the predominant soil types encountered during the drilling program undertaken at the site. The silty clay extended to a depth of 2.4 m below grade in the background hole. The silty clay unit was not encountered in the EMS hole, the top of which was approximately 1.8 m below grade. Silt inclusions, thin silt lenses/layers and no to some fine sand were encountered in the upper silty clay unit. The till generally consisted of clay and silt to silt trace clay, trace to some sand and trace to some gravel. The till became more silt-rich with depth and transitioned from light brown to grey at a depth of approximately 5 m below grade in the background hole and the EMS hole. No fractures were noted in either of the sample cores collected from the site.

Based on the piezometric data collected from the background nested piezometer installation over the period of February 2001 to October 2001, the water table fluctuated from 0.8 m below grade in August 2001 to 2.8 m below grade in February 2001 at the location of the background hole. The background hole is located approximately 100 m east of the EMS. Falling head recovery testing was conducted on three occasions in both the shallow and deep piezometers installed in the background hole. Hydraulic conductivity ranged from 2.4×10^{-7} m/s to 3.7×10^{-7} m/s and 6.8×10^{-7} m/s to 1.5×10^{-6} m/s in the deep piezometer. Both the shallow and the deep piezometer are screened in the underlying till unit.

5.1.3 NC3 Site

The NC3 site is located approximately 300 m south of the NC2 site and geological and hydrogeological conditions for the area interpreted from the available map series and databases is consistent with that discussed for the NC2 site (Section 5.1.2).

According to a GWDrill Well Information Report for the NC3 property, carbonate bedrock is located at a depth of 6.7 m below grade.

Silty clay and till were the predominant soil types encountered during the drilling program undertaken at the site. The silty clay extended to a depth of 3 m below grade in the background hole. The silty clay was also encountered in the EMS hole and extended to a depth of 1.8 m below grade. The silty clay generally consisted of silt and clay with no to some sand and no to trace organics. Silt inclusions and thin silt lenses/layers were encountered in this upper unit. The till generally consisted of clay and silt to silt trace clay, trace to some sand and trace to some gravel. The till became more silt-rich with depth and transitioned from light brown to grey at 5.5 m below grade. No fractures were noted in either of the sample cores collected from the site.

Based on the piezometric data collected from the background nested piezometer over the period of July 2001 to October 2001, the water table fluctuated from 0.7 m below grade in August 2001 to 2.2 m below grade in October 2001. This nest was located within 20 m of the north berm of the former EMS facility. Falling head recovery testing was conducted once in the shallow piezometer and three times in the deep piezometer. Hydraulic conductivity ranged from 2.2×10^{-6} m/s to 1.5×10^{-5} m/s in the deep piezometer. A sand seam was encountered within the screened interval of the deep piezometer during drilling of the background hole and as a result, the hydraulic conductivity obtained does not likely reflect the hydraulic conductivity of the till unit underlying the site. The hydraulic conductivity was estimated to be 5.7 x 10^{-9} m/s in the shallow piezometer which was screened in the underlying till.

5.1.4 SC1 Site

According to a GWDrill Well Information Report for a water well located on the SC1 property, till underlies a 10 m thick clay unit and extends to 25 m below grade and is followed by carbonate bedrock. A 1 m thick sand layer was reported in the hole at a depth of 21 m below grade. The piezometric surface in the well was reported to be at a depth of 2.7 m below grade in 1984. Other wells in the area report that the piezometric surface is located 2 m to 3 m below grade. The site is underlain by the limestone and dolostone of the Ordovician Red River Formation. A number of surrounding water wells obtain their water supply from the upper fractured zone of the formation (GWDrill Database). Regional groundwater flow in the carbonate aquifer in the area is to the east, towards the Red River (Betcher *et al.*, 1995). Based on data presented by Kennedy (2002), the estimated horizontal gradient in the aquifer, in the vicinity of the SC1 site, is on the order of 2 x 10^{-3} m/m.

Clay with trace silt and trace sand was the predominant soil type encountered during the drilling program undertaken at the site, and transitioned from grayish brown to grey at a depth of approximately 5.3 m below grade. Grain size distribution analyses were conducted on samples of clay collected from 2.2 m and 4.1 m below grade. The results indicate that the underlying material consists of 94% to 96% of clay, 3% to 5% silt and

1% sand by weight. No background hole, therefore no piezometer installation, was completed at the site. Fractures were noted along the EMS sample core to a depth of approximately 4 m below grade.

5.1.5 SC2 Site

The SC2 site is underlain by approximately 10 m of clay followed by 20 m of till with the piezometric surface reportedly located approximately 3 m below grade (Little, 1980; GWDrill Database). The site is underlain by dolostones and limestones of the Ordovician Red River Formation (Little, 1980). A number of surrounding water wells use the upper fractured zone of the formation as a water supply (GWDrill Database). Regional groundwater flow in the carbonate aquifer underlying the SC2 site is to the northwest (Betcher *et al.*, 1995). The estimated horizontal gradient in the aquifer is on the order of 1.4×10^{-2} m/m (based on data presented by Kennedy, 2002).

Clay was the predominant soil encountered during the drilling program undertaken at the site. The clay generally consisted of clay, trace to some silt, no to trace sand and trace organics. The clay transitioned from brown to grey at a depth of approximately 5 m below grade. The results of a grain size distribution analyses for a sample of clay obtained from a depth of 6 m below grade indicates that the material consists of 90% clay, 9% silt and 1% sand by weight. Minor fractures were noted in the upper 1.2 m of the second EMS sample core collected from the site. Fractures did not appear to be present in the first sample core or the background sample core materials.

Based on the piezometric data collected from the background nested piezometer installation over the period of July 2001 to August 2001, the water table is located approximately 2.5 m below grade at the location of the background hole. The background hole was located approximately 50 m northwest of the north berm of the EMS facility. Falling head recovery testing was conducted once in the shallow piezometer, on two occasions in the medium piezometer and once in the deep piezometer. The hydraulic conductivity in the shallow piezometer was estimated to be 6.2×10^{-9} m/s and hydraulic conductivity estimates in the medium and deep piezometers were 1.8×10^{-9} m/s and 8.5×10^{-9} m/s, respectively.

A test pit was completed to a depth of 3 m below grade at a distance of approximately 100 m north of the north berm of the EMS facility. Significant smearing of the walls of the test pit occurred during excavation. No fractures were observed in the clay in either the sidewalls or the base of the excavation, although a number of slickensides were observed, which may increase the overall hydraulic conductivity of the clay.

5.1.6 SC3 Site

The SC3 site is underlain by approximately 26 m of overburden materials consisting of approximately 19 m of clay underlain by till. The piezometric surface is reported to be located approximately 5 m to 9 m below grade in the area (GWDrill

Database). The site is underlain by the limestone and dolostone of the Ordovician Red River Formation (Little, 1980). A number of surrounding water wells utilize the upper fractured zone of the formation as a water supply (GWDrill Database). Regional groundwater flow in the underlying aquifer is to the northwest with a deduced horizontal gradient of 1.4 x 10^{-2} m/m (Betcher *et al.*, 1995; Kennedy, 2002).

Clay was the predominant soil type encountered during the drilling program undertaken at the site. The clay generally consisted of clay, which was described as silty to containing trace quantities of silt. A background hole was completed approximately 50 m from the EMS facility. According to a GWDrill Well Information Report for a water well located on the property, a till unit underlies the clay and extends from 19 m to 26 m below grade and is underlain by carbonate bedrock. The clay transitions from brown to grey at a depth of 5.5 m below grade. The piezometric surface was reported to be at a depth of 6.1 m below grade in 1967 and 7.5 m below grade in 1997. No fractures were noted along the EMS or background sample cores.

5.1.7 SE1 Site

According to a GWDrill Well Information Report for a water well located on the SE1 property, a 2 m thick till unit underlies 6 m of clay and is followed by over 5 m of sand and gravel. The well is reportedly screened in the sand and gravel unit from a depth of 11 m to 13 m below grade. The site is underlain by the limestone and dolostone of the Ordovician Red River Formation (Little, 1980; GWDrill Database). A number of surrounding water wells utilize the upper fractured zone of the formation as a water supply (GWDrill Database). The piezometric surface in the well located on the SE1 property was reported to be at a depth of 2.1 m below grade in 1990 and the piezometric surface for the SE1 area in general is often 2 m below grade (GWDrill Database). Although insufficient information is available to estimate the groundwater flow direction and gradients within the aquifer, the regional groundwater flow direction in the underlying carbonate aquifer is to the northwest under and estimated gradient of 1.4 x 10^{-2} m/m (Betcher *et al.*, 1995; Kennedy, 2002).

Clay was encountered to a depth of 2.8 m below the EMS base during the drilling program undertaken at the site and transitioned to a silty/sandy clay till from 2.8 m to 3.6 m below the EMS base followed by saturated sand and gravel from 3.6 m below the EMS base to the final depth of exploration (4.6 m below the EMS base). Till was encountered at 4.6 m below the EMS base. No background hole, therefore no piezometer installation, was completed at the site. No fractures were noted in the sample core during the drilling program.

5.1.8 SE2 Site

The SE2 site is underlain by approximately 46 m of till, with significant sand to silty sand/sandy silt interbeds. The piezometric surface is reported to be located approximately 1.5 m below grade in the area. The site is underlain by the shale, sandstone and minor limestones of the Melita Formation. A number of surrounding water wells utilize the localized sand and/or gravel overburden units as a water supply (GWDrill Database). According to a GWDrill Well Information Report for a water well located on the subject property, the facility draws water from a sand and gravel unit commencing 46 m below grade. The well is screened from 45 m to 52 m below grade. Groundwater flow and gradients within the localized sand and gravel aquifers can be highly variable and strongly influenced by stratigraphy. Insufficient information is available to estimate groundwater flow direction or gradients within the underlying sand and gravel aquifer.

Till was the predominant soil type encountered during the drilling program undertaken at the site. The till transitioned from light brown to grey at a depth of approximately 5 m to 6 m below grade in the background and EMS holes. The results of a grain size distribution analysis conducted on a sample of till collected during the drilling program indicated that the till consists of 17% clay, 48% silt, 31% sand and 4% gravel by weight. No fractures were noted along the sample cores collected from the site.

Based on the piezometric data collected from the background nested piezometer installation over the period of July 2001 to September 2001, the water table was 0.1 m to 0.7 m below grade. The background hole was located approximately 75 m east of the east berm of the EMS facility. Due to a problem with the shallow installation construction, falling head recovery testing could only be conducted in the deep piezometer. The estimated hydraulic conductivity of the till, based on the results of the testing, ranged from 2.5×10^{-7} to 2.6×10^{-7} m/s.

5.1.9 SE3 Site

The SE3 site is underlain by silty sand to a depth of 1 m followed by just over 5 m of clay and then followed by 9.5 m of clay-rich till. Precambrian bedrock reportedly underlies overburden at the site. The piezometric surface is reported to be located approximately 2.4 m below grade in the area. A number of surrounding water wells use the upper zone of fractured bedrock as a water supply (GWDrill Database). Groundwater flow direction and gradient in the underlying bedrock aquifer is primarily fracture dominated, and can vary significantly. Insufficient data and/or information is available to estimate either groundwater flow direction or horizontal gradient in the underlying aquifer.

Clay was the predominant soil type encountered during the drilling program and was encountered over an interval of 1 m to 6.1 m below the EMS base, while clayey sand was encountered over the interval of 0 m to 1 m below the EMS base. The clay transitioned from brown to grey at a depth of approximately 2 m below the EMS base, and may be indicative of the location of the water table (either at the time of the drilling program, or prior to construction of the EMS facility at the SE3 site location. The result of a grain size distribution analysis for the clay encountered indicates that the clay contains 78% clay, 18% silt and 4% sand by weight. No fractures were noted along the sample cores collected from the site.

Based on the piezometric data collected from the background nested piezometer installation over the period of August 2001 to October 2001, the water table is located approximately 0.6 m to 1.1 m below grade at the location of the background hole. The background hole was located approximately 50 m northwest of the northwest berm of the EMS facility. Falling head recovery testing was conducted in both the shallow and deep piezometers installed in the background hole. The estimated hydraulic conductivity of the clay, based on the results of the testing in the shallow installation (screened in the clay), ranged from 5.7×10^{-8} to 3.1×10^{-9} m/s (average 3.0×10^{-8} m/s). The estimated hydraulic conductivity of the underlying till, based on the result of the testing in the deep installation (screened in the till), was 7.0×10^{-8} m/s.

5.1.10 SW1 Site

According to a GWDrill Well Information Report for a water well located on the property, a 3 m thick sand and/or gravel unit underlies 1.5 m of clay and is followed by 13 m of till. A number of surrounding water wells, as well as the SW1 property water well, utilize the localized sand and/or gravel overburden units as a water supply (Sie and Little, 1976; GWDrill Database). The well reported in the database for the subject property is screened in a significant sand and gravel unit from a depth of 23 m to 26 m below grade. The piezometric surface in the well was reported to be at a depth of 3.0 m below grade in 1988. As noted earlier, groundwater flow and gradients cannot be established using available information and data for the localized underlying sand and gravel aquifer.

The EMS had been abandoned (liquids removed) and backfilled prior to drilling on the site and as a result, clay fill was encountered from ground surface to a depth of 1.8 m below grade. The clay fill was underlain by 0.2 m of organics, representing the location of the former EMS base. Sandy clay till was encountered from the former EMS base (2 m below grade) to a depth of 3.5 m below grade and was underlain by 0.5 m of saturated sand. The sand unit contained significant organics, was saturated and had a strong manure odour, indicating contamination by a source of manure, likely the former EMS facility located on the site. Underlying the sand unit at a depth of 4.0 m below grade was a sandy clay till. Grain size analyses were completed for two samples of sandy clay till and found that the material contained 32% clay, 28% to 31% silt, 35% to 36% sand and 2% to 4% gravel by weight. No fractures were noted in the sample core during the drilling program. No background hole, therefore no piezometer installation, was completed at the site.

5.2 SOURCE CHARACTERIZATION

Table 5.5 provides the concentration of select chemical species in wastewater contained in the ten EMS facilities examined in this work. Wastewater stored in the facilities contained generally high concentrations of nutrients, salts and dissolved metals. The CCME Canadian Environmental Quality Guidelines (CEQG, 2000) are shown for comparative purposes only. The results of the wastewater analysis obtained in this work are consistent with the values listed in Table 2.1 (see Chapter 2). It should b noted that wastewater in the abandoned EMS facilities would be diluted with precipitation.

Concentrations of chloride and potassium show a high degree of variability, ranging from a low of 24.0 mg/L and 37.5 mg/L, respectively, in the abandoned NC2 EMS facility to a high of 2300 mg/L and 1480 mg/L, respectively, in the operational SC1 EMS facility (Table 5.5). Sodium concentrations show less variability and are generally lower than chloride concentrations, with a minimum value (24.1 mg/L) obtained in the abandoned NC2 EMS facility, and a maximum value (834 mg/L) obtained in the operational SC3 EMS facility. A high degree of variability is also observed in the ammonia-nitrogen and ammonium-nitrogen concentrations, which vary from 10.7 mg/L in the abandoned NC2 EMS facility to 3973 mg/L in the SC1 facility. Nitrate+nitrite-nitrogen concentrations vary from 0.03 mg/L in the NC2 EMS facility to 412 mg/L in the operational NC1 EMS facility. Overall, nitrate+nitrite-nitrogen concentrations are typically in the range of 0.1 mg/L to 1.0 mg/L. Phosphate concentrations vary from 27.2 mg/L to 393 mg/L. It should be noted that all samples were collected by hand from locations along the edge of the inner berm near the surface of the stored wastewater, thus the variability in chemical species concentrations within the EMS is unknown.

5.3 SOIL QUALITY

5.3.1 Background Sample Core Chemistry Results

Concentrations of chloride, ammonia, nitrate+nitrite, phosphate, potassium and sulphate in the background cores collected from the NC1, NC2, NC3, SC2, SE2 and SE3 sites are summarized in Table 5.6. The concentrations typically showed some degree of spatial variability, both within the individual cores and between sites. The greatest spatial variability observed in concentrations in individual background cores was noted at the NC3, SC2 and SC3 sites, and the variability for each of these three sites is discussed below.

At the NC3 site, the background core showed evidence of potential manure impact from a depth of approximately 1.0 m to 4.0 m below ground, with elevated concentrations of chloride, ammonia, nitrate+nitrite and sulphate occurring within this interval (Figure 5.1). The elevated concentrations may be the result of:

- lateral contaminant migration from the EMS facility (located 25 m to the south of the background core location);
- past overflow and/or spillage of stored wastewater into the low-lying area surrounding the facility;
- past farming practices on site prior to installation of the EMS; or
- other deposition of animal wastes in the vicinity of the background core.

Elevated phosphate concentrations were noted in the upper 1.0 m of the background profile, with a maximum concentration of 806 mg/kg, and support the later two possibilities, since phosphate is an indicator of manure impact, but is not particularly mobile in the subsurface and tends to be strongly sorbed to soil and organics. Potassium concentrations in the NC3 background core decreased from 1320 mg/kg at the top of the profile to an average of 76.2 mg/kg from a depth of 3.3 m to the final depth of exploration (7.2 m below grade).

The SC3 background core also shows evidence of manure impact from ground surface to a depth of approximately 2.5 m below ground, with elevated concentrations of chloride, ammonia, nitrate+nitrite and phosphate occurring within this depth interval (Figure 5.2). The elevated concentrations may be the result of:

- past overflow and/or spillage of stored wastewater into the low-lying area surrounding the facility; or
- other deposition of animal wastes in the vicinity of the background core.

The elevated nitrate+nitrite concentrations and reduced ammonium concentrations in the upper 2 m of the profile would indicate that conditions in the upper portion of the profile may be aerobic, facilitating the nitrification of ammonium to nitrate.

The greatest variability in chloride concentrations in the background cores was observed in the SC2 core, with chloride concentrations increasing from 36.0 mg/kg near the top of the soil profile to as much as 634 mg/kg near the bottom of the profile (10.5 m below ground surface; Figure 5.3). This trend of increasing concentration with depth was not noted in the other chemical species analysed in the SC2 background core. Except for sulphate concentrations, the concentrations of the other species were similar in value to those observed in the background cores collected from the other sites. This leads to the hypothesis that the increase in chloride concentration with depth is a naturally occurring phenomenon in the area, due to past or current salinity in the underlying carbonate aquifer, and not the result of contamination from the EMS facility. This theory would explain the increasing chloride concentration with depth as a result of flushing of saline water in the soil pores with fresh infiltrating water from the surface.

Due to the variability observed at the above mentioned sites, all of the NC3 background core results and the SC2 background core chloride concentrations results are not included in the following discussion. Typical average background chloride concentrations ranged from 10.8 mg/kg (in the NC1 background core) to 28.7 mg/kg (in the SE3 background core). Average ammonia concentrations varied from 3.6 mg/kg (in the NC2 background core) to 17.1 mg/kg (in the SE2 background core). Average nitrate+nitrite concentrations varied from 2.2 mg/kg (in the SE2 background core) to 16.8 mg/kg (in the NC2 background core). Average phosphate concentrations in the background cores ranged from 0.4 mg/kg to 7.1 mg/kg, for the NC2 and SE2 background cores, respectively. Average potassium concentrations ranged from 63.4 mg/kg in the SE2 background core to 400 mg/kg in the SC2 background core. Sulphate concentrations showed the greatest spatial variability within the background cores with average concentrations ranging from 50.6 mg/kg to 3612 mg/kg in the NC1 and SE3 background cores, respectively. Within individual samples cores, for example the SC2 and SE3 background cores, sulphate concentrations ranged from as much as 20.6 mg/kg to 6180 mg/kg and 32.4 mg/kg to 8400 mg/kg, respectively. At both sites, peak values occurred at approximately 3.0 m depth, with decreasing concentrations with depth and towards ground surface. This trend was not observed in the remaining background cores examined.

5.3.2 EMS Sample Core Chemistry Results

The following sections provide the results and a discussion of the results, obtained from examination of the soil quality profiles obtained from the EMS sample cores for each of the sites. Concentrations are compared to the background concentrations obtained from either the soil quality profiles constructed from the background samples cores (where available) or from the lower portions of the EMS core soil quality profiles, when constant values are reached and maintained with depth. Due to the variability in background concentrations between sites, background concentrations have been established for most sites as discussed, and these site-specific background concentrations, set out in Table 5.6, are compared to the EMS sample core chemical results for that specific site.

5.3.2.1 NC1 EMS Sample Cores

Two sample cores were collected from the base of the NC1 EMS facility, one after four years of facility operation and one after seven years of operation. The sample cores were collected from two separate locations within the inner side slopes of the berms of the secondary cell.

After four years of facility operation, chloride concentrations appear elevated to depths exceeding about 1.4 m below the EMS base (Figure 5.4), when compared to the average background chloride concentration of 10.8 mg/kg. Ammonium concentrations are elevated to depths of 0.4 m below the EMS base (Figure 5.5), when compared to the average background ammonia concentration of 4.8 mg/kg. The depth of vertical migration of chloride cannot be established because the sampling did not extend more than1.4 m below the EMS base, however concentrations are elevated at the final depth of sampling, indicating the vertical migration of contaminants originating from the NC1 EMS facility occurred to a depth of 1.4 m or greater. Nitrate+nitrite and phosphate concentrations are consistent with background concentrations (averaging 2.7 and 6.5 mg/kg, respectively) along the length of the sample core, with the exception of the 0.05 m sub-sample, in which the nitrate+nitrite is elevated with a concentration of 139.5 (Figure 5.5). This would indicate that the production of nitrate+nitrite and the vertical migration of phosphate is not occurring to an appreciable degree beneath the NC1 EMS at the location sampled. It should be noted that the four year EMS sample core was collected during the initial stages of the sampling and analytical program development, and as a result, was analysed for only limited chemical species. The analytical program at that time did not include potassium and sulphate concentration determinations.

Chloride, potassium and sulphate soil quality profiles for the seven year NC1 EMS sample core are shown on Figure 5.6. After seven years of facility operation, chloride concentrations appear consistently elevated to a depth just over 1.0 m below the EMS base, with chloride concentrations exceeding the average background concentration of 10.8 mg/kg. Chloride concentrations also appear to be elevated within the 2.5 m and 2.7 m sub-samples, with concentrations of 19.0 and 17.2 mg/kg, respectively, and may be indicative of potential contaminant migration to that depth. Potassium concentrations, on the order of 80 mg/kg, are consistent with the range of background concentrations for the NC1 site provided in Table 5.6 and with the average background concentration of 99.0 mg/kg. Sulphate concentrations show a high degree of variability along the entire EMS sample core length and tend to increase with depth from a low of 6.0 mg/kg, near the top of the profile (within the 0.06 m sub-sample), to a high of 43.6 mg/L, at the bottom of the profile (within the 3.6 m sub-sample). These measured sulphate concentrations within the 7 year EMS sample core are generally consistent, although slightly lower, than the expected background concentrations set out in Table 5.6. Ammonium, nitrate+nitrite and phosphate soil profiles are provided on Figure 5.7. Ammonium concentrations appear elevated in the upper 1 m of the profile, with concentrations ranging from 11.0 to 44.0 mg/kg, when compared to the average expected background concentration of 4.8 mg/kg. Nitrate+nitrite and phosphate concentrations are consistent with background concentrations along the entire profile, confirming that the production of nitrate+nitrite and the vertical migration of phosphate is not occurring to an appreciable degree through the sub-surface materials underlying the NC1 site.

Overall, the results indicate that seepage from the EMS has impacted the potentially fractured, underlying silt-rich till to a depth of greater than 1.4 m below the

EMS base after four years of facility operation (at the location of the first EMS sample core), and to a depth of potentially 2.5 m below the EMS base after 7 years of facility operation (at the second EMS sample core location). The results also indicate that the underlying silt-rich till soil has capacity for ammonium sorption, as evidenced by the reduced vertical movement of ammonium (0.4 m and 1.0 m) when compared to the movement of conservative chloride (greater than 1.4 m and as much of 2.5 m) after 4 years and 7 years, respectively of facility operation. After seven years of facility operation at the NC1 site, the ammonium build-up zone appears to extend to a depth of 1.0 m below the base of the facility, at the location of the second (7 year) EMS sample core, with maximum observed concentrations nearly 10 times the expected average background ammonium concentrations. The results also show that the production of nitrate+nitrite and the vertical migration of phosphate are minimal through the underlying silt-rich till sediments underlying the NC1 site. It should be noted that although the 4 year and 7 year EMS sample cores were collected from different locations at the base of the EMS facility and a direct comparison can only be made if it is assumed that the conditions at the specific location apply to the entire base of the facility, it would appear that vertical migration of contaminants has continued at a consistent rate between sampling events. This continued vertical contaminant migration can only be confirmed with the ammonium concentration results, for which the final depth of migration is captured within the two profiles. After 4 years of facility operation, ammonium has migrated from the facility to a depth of 0.4 m below the EMS base and after 7 years of facility operation, ammonium originating from the NC1 EMS facility has migrated to a depth of 1.0 m below the EMS base, assuming the two sampling locations can be compared directly.

5.3.2.2 NC2 EMS Sample Core

One sample core was collected from the base of the NC2 EMS facility 19 years after facility operation commenced. The NC2 EMS facility had been abandoned for approximately 3 years at the time the EMS sample core was collected from the base of the facility. The facility was not backfilled on abandonment, and although the majority of liquids had been reportedly pumped from the facility at the time of closure, organics and some liquids had been left. Liquids contained within the NC2 EMS facility at the time of sample core collection and sampling program were a mixture of the remaining wastewater and organics and intercepted precipitation.

Chloride, potassium and sulphate soil profiles are shown on Figure 5.8. Chloride concentrations appear elevated along the entire length (4.5 m) of the NC2 EMS sample core (compared to the average background concentration of 12.4 mg/kg) and range from 55.4 mg/kg to 125 mg/kg. Potassium concentrations appear elevated in the upper 0.5 m of the profile (when compared to the average background concentration of 162.0 mg/kg) with concentration of 998.0 to 330.0 mg/kg reported within the upper 0.5 m. Sulphate concentrations range from a low of 6 mg/kg near the top of the profile to as much as 86.4 mg/kg near the base of the profile, and appear to increase with depth. This trend of increasing sulphate concentration with depth may indicate that flushing of naturally higher background sulphate concentrations (average background concentration estimated to be 69.2 mg/kg) from the underlying sediments is occurring as low sulphate concentration fluids from the EMS migrate down through the underlying sediments. The sulphate concentration in the stored liquid from the NC2 EMS facility was measured to be 24.0 mg/L, and was much lower than the sulphate concentrations measured in wastewater collected from the other sampled EMS facilities (see Table 5.5).

Ammonium concentrations appear elevated to a depth of 2.5 m below the EMS base when compared to the average background concentration set out in Table 5.6, and also appear elevated in the lower 0.5 m of the profile (Figure 5.9). This may indicate the presence of ammonium contamination within the shallow aquifer underlying the site. Nitrate+nitrite and phosphate concentrations are consistent with background concentrations along the entire length of the sample core, except for the phosphate concentration at the top of the profile, in the 0.1 m sub-sample and the nitrate+nitrite concentrations over the sampling interval from 2.2 to 2.5 m below the EMS base. All analysed chemical species were also found to have elevated concentrations over the 2.2 m to 2.5 m sampling interval, except for phosphate which remained at background levels. The peak values within this sampling interval may be the result of contamination of the samples during the sampling program or the preferential movement of contaminants, originating from the overlying EMS or another source, through a higher permeability zone intersection at the EMS sample core location between the depths of 2.2 m and 2.5 m below the EMS base. No secondary permeability features or potentially higher permeability sediments were noted over this interval.

Overall, the results indicate that seepage from the NC2 EMS has impacted the underlying silt-till to a depth of greater than 4.5 m, 19 years after facility operation commenced. The NC2 site ceased producing hogs and operating 3 years prior to the removal of the EMS sample core. No additional production wastewater had been added to the EMS facility since operation ceased and the remaining stored liquids have since been diluted with precipitation. This fact is not evident in the measured concentration profiles, and there is no significant reduction in concentrations of the analysed chemical species near the top of the profile, although it is likely that limited ponding of liquids has resulted in a reduced vertical hydraulic gradient and hence reduced rates of vertical flow from the facility since abandonment. The results indicate that the silt-rich tills encountered at this site have the capacity for ammonium sorption, with the extent of vertical ammonium migration from the facility to approximately 2.5 m. Ammonium concentrations in this zone are significant with concentrations as much as 10 times that of the average expected background ammonium concentration. Nitrate+nitrite production and vertical phosphate migration are minimal to nil beneath the NC2 EMS site. The lack of nitrate+nitrite production within the ammonium build-up zone underlying the NC2 EMS is likely the result of maintained anaerobic conditions beneath the EMS by allowing fluids to remain in the NC2 EMS since operation ceased.

5.3.2.3 NC3 EMS Sample Core

The NC3 EMS sample core was collected from the base of the EMS, within the side slopes of the inner berms, approximately 29 years after the facility operation commenced, and 14 years after the facility had ceased operating and the EMS was abandoned. As was the case with the NC2 site, abandonment of the facility was simply the cessation of any further wastewater inputs. No information was available as to whether or not the NC3 EMS was pumped of any stored liquids as part of the abandonment.

Chloride, potassium and sulphate concentration profiles are shown in Figure 5.10. Chloride concentrations appear elevated to the base of the profile (5.1 m), when compared to the average background concentration of 30.2 mg/kg. The peak measured chloride concentration was 300 mg/kg at a depth of 0.7 m below the EMS base. Chloride concentrations were generally measured to be on the order of 125 mg/kg along the NC3 EMS sample core profile. Potassium concentrations vary from a maximum of 2820.0 mg/kg at the top of the profile, to a minimum value of 45.5 mg/kg at the base. The average background potassium concentration was estimated to be 88.3 mg/kg, and based on this background value, vertical migration of potassium from the facility had occurred to a depth of 2.3 m below grade at the time the EMS sample core was removed. Sulphate concentrations appear to be elevated within the upper 0.3 m of the profile, when compared to the average background concentration of 79.9 mg/kg, with a peak concentration of 453.0 mg/kg occurring at the top of the profile, within the 0.1 m sub-sample. Ammonium, nitrate+nitrite and phosphate concentration profiles are shown graphically on Figure 5.11. Ammonium concentrations are elevated in the upper 1.8 m of the profile, with a peak concentration of 1720.0 mg/kg measured within the 0.4 m sub-sample. This would indicate that the ammonium build-up zone extends to a depth of 1.8 m below grade at 29 years after facility operation commenced, including 14 years of abandonment. Nitrate+nitrite concentrations are typically on the order of 2.0 mg/kg to 8.0 mg/kg and are lower than the average background concentration of 33.7 mg/kg provided in Table 5.6. This would indicate that the lower end of the background concentration range (3.5 mg/kg; see Table 5.6) may be a more appropriate background value than the average calculated value. Phosphate concentrations are consistent with background concentrations at depths greater that 0.3 m below the EMS base, indicating limited vertical migration of phosphate from the abandoned NC3 EMS. The maximum phosphate concentration (155 mg/kg) occurs in the first sub-sample, collected from an average depth of 0.1 m.

After 29 years since operation commenced at the NC3 site, vertical contaminant migration, as determined from the measured chloride concentrations, extends to a depth of greater than 5.1 m. The silt-till underlying the site shows capacity for ammonium sorption as evidenced by the vertical migration of ammonium to a depth of only 1.8 m, compared to the conservative chloride migration of greater than 5.1 m below the EMS base. Conditions under the NC3 EMS facility have likely remained anaerobic, since there does not appear to be any production and/or vertical migration of nitrate+nitrite occurring in the underlying soils, although significant potential exists, in the form of available ammonium, for the production of nitrite and/or nitrate should aerobic conditions occur beneath the EMS facility. Vertical migration of both potassium and sulphate appear to have occurred from the facility, with the depth of movement being 2.3 m for potassium and 0.3 m for sulphate. No significant movement of phosphate has occurred from the NC3 EMS facility. As was the case with the NC2 EMS, no additional production wastewater had been added to the EMS facility since operation ceased and the remaining stored liquids have since been diluted with precipitation. This probable dilution was not evident in the measured concentration profiles from the NC3 EMS sample core, and there is no significant reduction in concentrations of the analysed chemical species occurring near the top of the profile. As was the case with the NC2 EMS, it is likely that limited ponding of liquids has resulted in a reduced vertical hydraulic gradient and hence reduced rates of vertical flow from the facility since abandonment.

5.3.2.4 SC1 EMS Sample Core

One sample core was collected from the base of the SC1 EMS facility, within the side slopes of the inner berms, approximately nine years after facility operation commenced. No background core was collected from the site, and consequently background concentrations could only be estimated from the EMS sample core analytical results.

Chloride concentrations vary from 309 mg/kg to 608 mg/kg in the SC1 EMS sample core (Figure 5.12). No background concentration can be estimated from the chloride profile, and as a result, the depth of conservative chloride migration from the SC1 EMS facility cannot be established from the available data. However, the chloride concentrations do appear to be high along the entire profile, with a general decreasing trend occurring with depth. The concentration profiles of ammonium, nitrate and phosphate are shown on Figure 5.13. Ammonium concentrations are clearly elevated in the upper 0.4 m of the profile, with a maximum of 3445 mg/kg occurring in the 0.1 m sub-sample, indicating migration of ammonium from the facility to a depth of 0.4 m after 9 years. The background ammonium concentration appears to be on the order of 8.0 mg/kg, as estimated from the lower 4.6 m of the soil quality profile. Nitrate concentrations are slightly elevated in the upper 0.2 m of the profile, with concentrations on the order of 10.0 mg/kg to 20.0 mg/kg, indicating some nitrate production and/or vertical migration from the SC1 EMS facility. Below 0.2 m, nitrate concentrations are on the order of 4.0 mg/kg, which are considered to be representative of background concentrations. Phosphate concentrations are also elevated in the upper 0.2 m of the profile, with a peak concentration of 626 mg/kg occurring in the first sub-sampling interval, compared to the apparent background concentration of 4 mg/kg, as estimated over the average sub-samples intervals from 0.3 m to 1.8 m. The SC1 EMS sample core was collected during the initial stages of the sampling and analytical program development, and the program did not include the determination of potassium and sulphate.

As noted previously, a background core was not collected as part of the program completed at the SC1 site. Although estimates of ammonium, nitrate and phosphate background concentrations could be determined from the SC1 EMS sample core, background chloride concentrations could not be established. An estimate of the extent of vertical conservative chloride concentration from the SC1 EMS cannot be reasonably made from this limited data, although chloride concentrations did appear to be high along the entire profile (5.0 m). The extent of migration of ammonium, nitrate and phosphate is clear from the results of the EMS sample core analysis. Ammonium appears to have migrated to a depth of 0.4 m below the EMS base over the 9 years that the SC1 EMS has been operating. Although both nitrate and phosphate are elevated in the 0.1 m sub-sample, the production and/or migration of these two chemical species is limited to this first sub-sample.

5.3.2.5 SC2 EMS Sample Core

Two cores were removed from the base of the SC2 EMS facility, one after 20 years of facility operation and one after 21 years of facility operation. The sample cores were collected from within the side slopes of the SC2 EMS facility, from two different locations.

After 20 years of facility operation, chloride concentrations beneath the facility ranged from 130 mg/kg to 260 mg/kg, with one anomalous concentration of 420 mg/kg occurring at an average depth of 4.4 m below the EMS base (Figure 5.14). Since background chloride concentrations tend to increase with depth naturally beneath the site, the depth of chloride migration from the SC2 EMS facility is difficult to estimate. No distinct trends in chloride concentrations with depth are apparent from the profile. Ammonium, nitrate and phosphate concentrations are shown on Figure 5.15. Ammonium concentrations are clearly elevated to a depth of approximately 1.0 m below the EMS base, when compared to the average background concentration provided in Table 5.6 of 10.5 mg/kg. The peak concentration of ammonium within the ammonium build-up zone is 598.0 mg/kg. This limited movement of ammonium over 20 years of facility operation, demonstrates the low permeability nature of the underlying clay soils and the potentially high capacity of the underlying clay soils for ammonium sorption. Both nitrate and phosphate concentrations are similar to background values along the entire length of the sample core, indicating no production and/or vertical migration of nitrate and phosphate from the facility at the location of the sample core. The 20 year EMS sample core was collected during the initial stages of the sampling and analytical program development; consequently potassium and sulphate were not included in the analysis.

Figure 5.16 shows the chloride and potassium profiles constructed for the 21 year SC2 EMS sample core. Sulphate was not included in the sampling program at the time the 21 year SC2 EMS sample core was collected. Chloride concentrations appear highly variable and are elevated in the upper 6.5 m of the profile, with concentrations declining to lower values at depths greater than 6.5 m below grade. Consistent with this decrease in chloride concentrations with depth, potassium concentrations also appear to decline to low values at depths of 6.5 m or greater. These two chemical species may provide an indication of the extent of conservative contaminant migration at the location of the 21 year SC2 EMS sample core. Peak chloride and potassium concentrations are 1160 mg/kg and 464 mg/kg, respectively, and occur within the upper 4 m of the profile. Ammonium, nitrate+nitrite and phosphate concentration profiles are shown on Figure 5.17. These profiles are generally consistent with background concentrations, with some minor variability observed. The lack of elevated ammonium in the top of the profile is not consistent with the 20 year SC2 EMS sample core results or the results obtained from other study sites. Consequently, it is considered likely that the upper portion of the sample core was lost when seating the hollow stem augers during the sample core collection.

Overall, the trend of increasing chloride concentration with depth in the background core and high chloride concentrations observed along the entire length of the 20 year sample core, make an estimate of the extent of vertical chloride migration from the facility difficult. However, the 21 year EMS sample core shows unexpected results, including a decline in the chloride concentrations, coincident with a decline in the potassium concentrations, at a depth of 6.5 m below grade. These results may indicate that after 21 years of facility operation, the extent of vertical conservative contaminant migration is on the order of 6.5 m below grade. The ammonium concentration profiles between the 20 and 21 year sample cores are also inconsistent, with the ammonium build-up zone extending to a depth of 1.0 m in the 20 year EMS sample core and not appearing in the 21 year EMS sample core. As an explanation to this inconsistency, it is possible that the upper portion of the 21 year EMS sample core was lost during seating of the hollow stem augers. Comparison of the potential extent of vertical conservative contaminant migration (based on the potassium and chloride profiles from the 21 year EMS sample core) of 6.5 m, and the extent of vertical ammonium migration (based on the ammonium profile from the 20 year EMS sample core) of 1.0 m, the underlying clay soils show significant capacity for ammonium sorption. Consistent with results obtained from the other study sites, there is little to no production and/or vertical migration of nitrate and/or phosphate from the EMS, either in the 20 year or the 21 year EMS sample cores.

5.3.2.6 SC3 EMS Sample Core

Two sample cores were removed from two different locations within the side slopes of the inner berms of the SC3 EMS facility, twenty years after operation commenced. It is important to note that the background core collected from the site showed evidence of manure impact and that the average background concentrations reported in Table 5.6 may not be representative of the actual background concentrations. This was discussed in detail previously, in Section 5.3.1.

Soil quality profiles for the 20 year SC3 EMS sample core #1 are shown on Figures 5.18 and 5.19. Chloride concentrations along sample core #1 typically range from 22.0 mg/kg to 362 mg/kg, and are elevated in the upper 1.9 m of the profile, when compared to the expected average background concentration of 86.0 mg/kg (see Table 5.6). Ammonium concentrations are generally elevated in the upper 1.2 m of the profile, reaching constant values on the order of 35 mg/kg below a depth of 1.2 m below the EMS base. These constant values would indicate that the calculated average background ammonium concentration obtained from the background sample core results of 18.0 mg/kg may be an underestimate of the background concentrations at the sample core location. Overall, nitrate+nitrite and phosphate concentrations are on the order of 15 mg/kg and 10 mg/kg along the entire length of the 20 year SC3 EMS sample core #1, which are likely consistent with background concentrations. This would indicate limited to no vertical migration and/or production of phosphate or nitrate+nitrite from the SC3 EMS. Potassium and sulphate concentrations were not determined in the SC3 EMS sample cores.

Soil quality profiles for SC3 EMS sample core #2 are shown on Figures 5.20 and 5.21. Chloride concentrations along sample core #2 typically range from 142 mg/kg to 599 mg/kg, and appear to be elevated along the 4.8 m length of the sample core, when compared to the expected average background concentration of 86.0 mg/kg. However, examining the trend of constant chloride concentration on the order of 165 mg/kg from a depth of just over 2.2 m to the final depth of sampling may indicate that the background chloride concentration at the location of sample core #2 is better represented by a value on the order of 165 mg/kg. In which case, the extent of vertical migration of conservative chloride from the SC3 EMS at the sample core #2 location would be on the order of 2.2 m below the EMS base after 20 years of facility operation. This result would be consistent, qualitatively, with the result obtained from sample core #1. Ammonium concentrations are generally elevated in the upper 0.8 m of the profile, with a peak concentration of 1190.0 mg/kg. Nitrate+nitrite concentrations are consistent with apparent background values in the upper 0.7 m of the profile and become elevated over the interval of 0.7 m to 3.6 m, which may indicate a past nitrate+nitrite loading from the SC3 EMS facility or past and/or present aerobic conditions beneath the EMS facility allowing for the nitrification of ammonium to nitrate and the subsequent vertical migration of nitrate+nitrate beneath the EMS facility. The peak nitrate+nitrite concentration is 840.4 mg/kg and occurs at a depth of just under 1.0 m below the EMS base, just below the ammonium build-up zone. Phosphate concentrations are elevated in the upper 0.7 m of the profile, with a peak concentration of 110 mg/kg, and decline to apparent background concentrations of less than 10 mg/kg, on the lower end of the expected range set out in Table 5.6.

Overall, the results of the chloride analysis for the two SC3 EMS sample cores are reasonably consistent, with the extent of vertical migration of chloride from the facility being 1.9 m and 2.2 m at the sample core #1 and sample core #2 locations, respectively. The extent of the ammonium build-up zone in the two sample cores shows some variability, with the elevated ammonium concentrations extending to a depth of 1.2 m below the EMS base in sample core #1 and to a depth of 0.8 m in sample core #2. Comparison of the extent of vertical chloride migration to the extent of vertical ammonium migration at the two sample core locations indicates that the underlying clay soils show some capacity for ammonium sorption. The more rapid decline to background concentration in the second sample core may be due, in part to the presence of nitrate+nitrite below this zone, and could indicate the presence of aerobic conditions and the conversion of ammonium to nitrate at the location of sample core #2. Quantitatively, the concentrations of comparable chemical species between the two sample cores appear to be reasonably consistent between locations. Although limited production and/or vertical movement of nitrate+nitrite and phosphate are apparent in sample core #1, nitrate+nitrite production and migration has occurred to a depth of just over 3 m below grade in sample core #2.

5.3.2.7 SE1 EMS Sample Core

The SE1 EMS sample core was collected fourteen years after facility operation commenced. The core was collected from a single location within the side slopes of the inner berms. No background sample core was collected from the SE1 site and consequently, apparent background concentrations were estimated from the EMS sample core analytical results.

Soil quality profiles for chloride, ammonium, nitrate and phosphate are shown on Figures 5.22 and 5.23, respectively. Potassium and sulphate determinations were not part of the analytical program at the time the SE1 EMS sample core was collected. Chloride concentrations are typically on the order of 120 mg/kg along the entire length of the EMS sample core. Peak chloride concentrations of 393 and 276 mg/kg occur over the 0.5 m to 0.7 m sampling intervals and could be indicative of contamination during the sampling program or of the preferential movement of contaminants, originating from the overlying EMS or another source, through a higher permeability zone intersection in the EMS sample core location between the depths of 0.5 and 0.7 m below the EMS base. No secondary permeability features or potentially higher permeability sediments were noted over this interval. Ammonium concentrations are elevated in the upper 1.7 m of the profile with values generally declining from a high of 268 mg/kg at the top of the profile, to a low of 28.6 mg/kg near the base of the profile. Anomalously high ammonium concentrations also occur over the 0.5 m to 0.7 m sampling intervals, with values of 2522 and 1179 mg/kg, respectively. Since the depth of conservative chloride migration cannot be established from the available data, no comment can be made as to the capacity of the underlying clay soils for ammonium sorption. Nitrate concentrations are consistent with background values along the entire length of the sample core. Phosphate concentrations are slightly elevated in the upper 0.4 m of the profile, with a peak concentration of 43.4 mg/kg. These later results would indicate limited vertical phosphate migration and no nitrate production and/or vertical migration from the SE1 EMS facility.

5.3.2.8 SE2 EMS Sample Core

Two sample cores were collected from the base of the SE2 EMS facility, one after three years of facility operation and one after six years of facility operation. The sample cores were collected from two separate locations within the inner side slopes of the berms of the SE2 EMS facility secondary cell.

After three years of facility operation, chloride concentrations appear elevated, with a peak value of 64.1 mg/kg, to depths of approximately 1.0 m below the EMS base (Figure 5.24), when compared to the average background concentration of 20.2 mg/kg (Table 5.6). Ammonium concentrations are elevated to a depth of approximately 0.2 m below the EMS base (Figure 5.25), when compared to the average background ammonium concentration of 17.1 mg/kg. The peak ammonium concentration measured in the 3 year EMS sample core is 132.0 mg/kg. Nitrate+nitrite and phosphate concentrations are consistent with background concentrations of 2.2 and 7.1 mg/kg, respectively along the length of the sample core, although the apparent background phosphate concentration observed at the location of the 3 year EMS sample core, appear to be better represented by a value on the order of 2 mg/kg. Potassium and sulphate were not included in the sub-sample analysis at the time that the 3 year SE2 EMS sample core was collected.

Chloride, potassium and sulphate soil quality profiles for the six year SE2 EMS sample core are shown on Figure 5.26. After six years of facility operation, chloride and potassium concentrations appear elevated to a depth of approximately 0.5 m below the EMS base, much less than expected given that vertical chloride migration in the 3 year EMS sample core extended to a depth of 1.0 m below the EMS base. The peak potassium concentration is 189 mg/kg and occurs at the top of the profile and the peak chloride concentration in the upper 0.5 m interval has a value of 64.0 mg/kg. The peak chloride concentrations measured in the 3 year and 6 year EMS sample cores are consistent, with values of 64.1 and 64.0 mg/kg consistently. Sulphate concentrations generally increase with depth from a low of 6 mg/kg near the top of the profile to a high of 46.8 mg/kg near the base of the profile. Chloride, potassium and sulphate concentrations are all elevated within the 1.5 m sampling interval with concentrations of 118 mg/kg, 59.5 mg/kg and 46.8 mg/kg, respectively. This may be the result of contamination of the sub-sample during the sampling program or of the preferential movement of contaminants, originating from the overlying EMS or another source, through a higher permeability zone intersection in the EMS sample core location in the vicinity of the depth of the 1.5 m sub-sample. As in previous cases where anomalous peak values have occurred, no secondary permeability features or potentially higher permeability sediments were noted in the sampling interval. Ammonium, nitrate+nitrite and phosphate soil profiles are provided on Figure 5.27. Ammonium concentrations appear elevated in the upper 0.4 m of the profile, with a peak ammonium concentration of 146.0 mg/kg and is consistent with the peak ammonium concentration measured in the 3 year EMS sample core of 132.0 mg/kg. As was the case with the 3 year EMS sample core results, nitrate+nitrite and phosphate concentrations are generally consistent with background concentrations along the entire profile.

Overall, the results between sampling times and locations are relatively consistent, with the exception of the chloride concentration profiles. Examining the 3 year EMS sample core profiles, the underlying silt-rich till and/or sand shows capacity for ammonium sorption. This is evident when comparing the depth of vertical chloride migration of 1.0 m in the 3 year EMS sample core, to the depth of vertical ammonium migration of 0.2 m in the same sample core. Comparing the depth of vertical migration of chloride and ammonium in the 6 year EMS sample core (0.5 m and 0.4 m, respectively), would indicate limited ammonium sorption capacity in the soils at the location of the 6 year EMS sample core. Variation in the extent of vertical migration and sorption capacity between the 3 year and 6 year EMS sample cores at the site is likely the result of smaller scale heterogeneities in the underlying silt-rich till, which notably contained silt and sand lenses and/or layers. As with many of the previous sites, nitrate+nitrite production and vertical migration of phosphate through the underlying sediments are nil to limited and are not considered to be a concern under the existing site conditions at the time of sampling.

5.3.2.9 SE3 EMS Sample Core

One sample core was removed from the base of the SE3 facility, within the side slopes of the inner berms. The core was removed approximately fourteen years after facility operation commenced.

Chloride, potassium and sulphate profiles are shown on Figure 5.28. Chloride concentrations show some variability along the entire length of the EMS sample core, however they are generally elevated in the upper 2.1 m of the profile, compared to the expected average background chloride concentration of 28.7 mg/kg (Table 5.6). This would indicate that the extent of conservative contaminant migration through the clay soils underlying the SE3 site is 2.1 m after 14 years on continued facility operation. The peak chloride concentration is 84.8 mg/kg, and occurs in the 0.3 m sub-sample. Potassium concentrations are elevated in the upper 4.9 m of the profile, compared to the average background concentration, as determined from the background sample core, of 306.0 mg/kg. The peak potassium concentration occurs in the 0.2 m sub-sample, with a value of 1280 mg/kg. Sulphate concentrations generally increase with depth along the profile, with peak concentrations on the order of 750 mg/kg occurring near the base of the profile. Ammonium, nitrate+nitrite and phosphate soil quality profiles are shown on

Figure 5.29. Ammonium concentrations are elevated in the upper 0.7 m of the profile when compared to the average expected background concentration of 9.0 mg/kg (Table 5.6), with a peak ammonium concentration of 712 mg/kg occurring in the 0.3 m sub-sample. Comparison of the extent of conservative chloride migration (2.1 m) to the extent of ammonium migration (0.7 m) indicates that the underlying clay soils have some potential capacity for ammonium sorption. After 14 years of facility operation, the ammonium build-up zone is contained within the upper 0.7 m of the underlying sediments. Nitrate+nitrite and phosphate concentrations are consistent with background concentrations along the entire length of the sample core, except the phosphate concentration in the 0.2 m sub-sample which is elevated with a concentration of 37.0 mg/kg, compared to the average expected background concentration, determined from the background sample core, of 3.3 mg/kg. As found at the majority of the other study sites, limited to no vertical phosphate migration or nitrate production and/or migration is occurring beneath the EMS facilities examined.

5.3.2.10 SW1 EMS Sample Core

One sample core was removed from the base of the SW1 facility, within the side slopes of the inner berms. The core was collected approximately eighteen years after facility operation commenced. No background core was collected from the site, and consequently background concentrations could only be estimated from the EMS sample core analytical results.

Chloride concentrations are elevated in the upper 3.2 m of the profile with peak concentrations occurring at the top of the profile, and a spike in concentration occurring between the average sub-sample depths of 1.6 to 2.1 m below the EMS base (Figure 5.30). Background chloride concentrations, based on the results of the EMS sample core analyses for the lower 3.2 to 6.0 m of the profile, appear to be on the order of 10 mg/kg. The peak chloride concentration is 251 mg/kg, and occurs in the top (0.1 m) sub-sample. The chloride profile results indicate that after eighteen years of continued facility operation, conservative vertical contaminant migration through the interbedded clay-rich till and sand sediments had occurred to a depth of 3.2 m below the EMS base. Elevated ammonium, nitrate+nitrite and phosphate soil quality profiles are shown on Figure 5.31. Ammonium concentrations are elevated in the upper 2.6 m of the profile, with a peak concentration of 1430 mg/kg occurring in the upper 0.1 m of the ammonium concentration profile and a secondary peak occurring at a depth of 2.2 m below the EMS base. Comparison of the extent of vertical chloride migration of 3.2 m to the vertical extent of ammonium migration of 2.6 m indicate that the underlying sandy clay-rich till and sand soils underlying the site have some limited capacity for ammonium sorption. The secondary peaks in the chloride and ammonium concentrations occurring at depth along the profile was the location of a notably contaminated sand layer intersected during drilling. Nitrate+nitrite and phosphate concentrations are consistent with background concentrations (estimated to be on the order of 5 mg/kg and 8 mg/kg, respectively) along the entire length of the sample core, except the phosphate concentration in the upper 1.2

m of the profile which range from a peak of 890 mg/kg at the top of the profile to 12.4 mg/kg at a depth of 1.2 m below the EMS base.

5.3.3 Pore Water Chemistry

Pore fluids were extracted from select sub-samples of the SC3 EMS sample core #1, and from the 1.2 m sub-sample collected from the SC3 background core. The results of the analysis of filtrate from the sub-samples were compared to results of analysis of bulk soil for the same sub-sample. The results were compared in order to determine if concentrations in mg/kg could be converted to concentrations in mg/L using gravimetric moisture content, and adequately represent actual pore water concentrations. In order to complete the parameter estimations outlined in Chapter 6, converted chloride concentrations in mg/L developed from a bulk analysis of the soil were used. The comparison was done to justify the applicability of the conversion method and the use of the converted chloride concentrations. The computed pore water concentrations shown in Table 5.7 represent the chloride concentrations in mg/L obtained using the result of the saturated paste and/or 2:1 dilution method results and divided by the gravimetric moisture content on a dry basis to obtain an estimate of the pore water concentration within the given sub-sample. The extract pore water concentrations shown in Table 5.7 represent the concentrations of chloride obtained from the analysis of pore fluids. The results are shown graphically for sample core #1 in Figure 5.32.

In all cases, except the 1.42 m depth sub-sample of sample core #1, computed pore water concentrations exceed the analytical pore water concentrations obtained from the pore fluid extraction and subsequent analysis by factors varying from 1.26 to 2.02. In the case of the 1.42 m depth sub-sample, the computed pore water concentration fell below the analytical pore water concentration by a factor of 0.905. The variations between the computed pore water concentrations and the analytical pore water concentrations range from 9% to 101%. The R^2 correlation coefficient between the analytical and computed pore water concentrations along the SC3 sample core #1 (Figure 5.33) is 0.8953. The variation between the computed and measured pore water concentrations is likely the result of the saturated paste and/or 2:1 dilution method, which may cause some of the chloride naturally bound to soil, and not occurring in the pore water, to be released. This results in higher computed chloride concentrations compared to measured pore water chloride concentrations.

5.4 GROUNDWATER QUALITY

Groundwater samples were collected from each of the piezometer installations on one occasion. The results were analysed for a number of parameters including ammonia-N, nitrate+nitrite-N, phosphorus-P, electrical conductivity, pH, chloride, potassium, sodium and sulphate. The results of these analysis obtained for the NC1 to NC3, SC2, SE2 and SE3 site are provided in Table 5.8, together with the CCME Canadian Environmental Quality Guidelines (CEQG; 2000), which are shown for comparative purposes only.

Ammonia-N concentrations ranged from 0.05 mg/L in the deep NC2 piezometer to 1.14 mg/L in the shallow SE2 piezometer. These values likely represent typical background ammonia-N concentrations in shallow groundwater at the site.

Nitrate+nitrate-N concentrations range from a low of 0.01 mg/L in the SC2 medium piezometer to 25.0 in the NC3 shallow piezometer, which exceeded the CCME CEQG for nitrate+nitrite-N of 10 mg/L. All other groundwater samples were below the guideline for nitrate+nitrite-N, and likely represent typical background concentrations in groundwater at the sites. Elevated nitrate+nitrate-N concentrations in groundwater may be indicative of impacts originating from the EMS or other agricultural/anthropogenic sources in the area.

Phosphorus-P concentrations in shallow groundwater at the sites ranged from 0.007 mg/L to 0.028 mg/L. This range is considered to be consistent with background phosphate-P concentrations in shallow groundwater at the sites.

Electrical conductivity ranged from 703 μmhos/cm in the SE2 deep piezometer to 5660 μmhos/cm in the SC2 medium piezometer. Elevated values are associated with high concentrations of chloride, potassium, sodium and sulphate (salts), and are particularly elevated in the piezometers at the NC2, NC3, SC2 and SE3 sites. Together with elevated nitrate+nitrite, chloride, potassium, sodium and sulphate, electrical conductivity may indicate some impact to shallow groundwater as a result of migration of fluids from the EMS facilities or impacts to shallow groundwater as a result of other agricultural/anthropogenic activities in the area. Elevated chloride at the SC2 site appeared to be naturally occurring, as evidenced by the increasing chloride concentrations within the background core, with depth below grade. This high level of chloride would cause elevated electrical conductivity within the shallow groundwater at the site.

Chloride concentrations range from less than the laboratory method detection limit (< 10 mg/L) to 1120 mg/L in the SC2 deep piezometer. The chloride concentration of 279 mg/L measured in groundwater collected from the NC3 shallow piezometer exceeds the CCME CEQG for chloride of 200 mg/L and may indicate impact to shallow groundwater as a result of subsurface lateral liquid migration from the EMS facility or other impacts resulting from agricultural/anthropogenic activities in the area. Chloride concentrations in the SC2 shallow, medium and deep piezometers were 702 mg/L, 856 mg/L and 1120 mg/L, respectively. These results are consistent with the increasing chloride concentrations observed with depth at the site, particularly in the background sample core.

Potassium concentrations varied from 3.37 mg/L in the SE3 shallow piezometer to 12.0 mg/L in the SC2 medium piezometer. These values likely represent typical background potassium concentrations in shallow groundwater at the sites.

Sodium concentrations range from 57.7 mg/L in the NC3 deep piezometer to 653 mg/L in the SC2 medium piezometer. Exceedances of the CCME CEQG for sodium were measured in the NC2 shallow piezometer (357 mg/L), the NC3 shallow piezometer (243 mg/L), the SC2 shallow, medium and deep piezometers (574 mg/L, 653 mg/L and 512 mg/L, respectively), and the SE3 shallow and deep piezometers (303 mg/L and 200 mg/L, respectively). Elevated sodium concentrations, together with elevated levels of other indicator parameters, may indicate impact to shallow groundwater as a result of subsurface lateral liquid migration from the EMS facility or other impacts resulting from agricultural/anthropogenic activities in the area.

Sulphate concentrations exceeded the CCME CEQG for sulphate in the NC2 shallow, the NC3 shallow, SC2 and SE3 piezometers. The greatest sulphate concentrations occurred in the SC2 shallow and medium installations with concentrations of 2410 mg/L and 1750 mg/L respectively. The remaining exceedances were between 570 mg/L and 972 mg/L. Exceedances of the CCME CEQG for sulphate coincided with exceedances of the CCME CEQG for sodium. Although often associated with impacts from EMS facilities, elevated sulphate concentrations in groundwater often occur naturally in shallow groundwater in southern Manitoba (Robert Betcher, Personal Communication, 2001).

Based on the results of the analysis of shallow groundwater at each of the sites, it appears that impacts exist at the NC3 site as a result of agricultural/anthropogenic activities in the vicinity of the piezometer, however it should also be noted that this conclusion is based on a single sampling event and would have to be confirmed with additional sampling events. This may or may not be the result of fluid migration from the EMS facility and is postulated due to elevated concentrations of nitrate+nitrite-N, chloride and sodium within the shallow groundwater underlying the site. It should be noted that the background sample core did show some evidence of manure impact, and that the piezometer was located in a former cattle pasture area. Chloride, sodium and/or sulphate concentrations were found to exceed the CCME CEQGs at the NC2, SC2 and SE3 sites, however other indicator parameters, such as nitrate+nitrite-N are not elevated, indicating that the elevated concentrations may be representative of natural background levels in shallow groundwater in the area. Overall, based on the limited groundwater sampling program, there do not appear to be any significant impacts measured in the monitoring wells installed at a number of the study site examined in this work, indicating that for the most part, lateral migration from the facilities, in the direction of the background monitoring wells is limited.

Table 5.1: Interpreted Site Geology

Site	Depth (m) [(1)]		Material Description
	From	To	
NC1	0	33	SILT (TILL) – clayey to trace clay, trace to some sand, trace to some gravel
	33	> 33	CARBONATE BEDROCK (not sampled)
NC2	0	7	SILT (TILL) – and clay to trace clay, trace to some sand, trace to some gravel
	7	> 7	CARBONATE BEDROCK (not sampled)
NC3	0	1.8	CLAY – silty, no to trace sand, no to trace organics
	1.8	6.7	SILT (TILL) – to trace clay, trace to some sand, trace to some gravel
	6.7	> 6.7	CARBONATE BEDROCK (not sampled)
SC1	0	8	CLAY – no to trace silt, no to trace sand
	8	19	TILL (not sampled)
	19	20	SAND (not sampled)
	20	23	TILL (not sampled)
	23	> 23	CARBONATE BEDROCK (not sampled)
SC2	0	8	CLAY – silty to trace silt, trace sand
	8	28	TILL (not sampled)
	28	> 28	CARBONATE BEDROCK (not sampled)
SC3	0	18.6	CLAY – trace to some silt
	18.6	25.9	TILL (not sampled)
	25.9	> 25.9	CARBONATE BEDROCK (not sampled)
SE1	0	2.8	CLAY – trace to some silt
	2.8	3.6	CLAY (TILL) – silty, sandy, trace gravel
	3.6	4.6	SAND – and gravel
	4.6	6	TILL (not sampled)
	6	> 6	SAND AND GRAVEL (not sampled)
SE2	0	15.4	SILT (TILL) – and to trace sand, trace to some gravel, no to trace organics
	15.4	19.2	SAND (not sampled)
	19.2	23.2	TILL (not sampled)
	23.2	27.7	SAND (not sampled)
	27.7	32.3	TILL (not sampled)
	32.3	35.3	SAND (not sampled)
	35.3	46	TILL (not sampled)
	46	> 46	SAND AND GRAVE (not sampled)
SE3	0	1	SAND – silty, trace to some clay
	1	6.1	CLAY – no to trace silt, no to trace sand
	6.1	15.8	CLAY (TILL) – sandy, silty, no to trace gravel
	15.8	> 15.8	PRECAMBRIAN BEDROCK (not sampled)
SW1	0	1.5	CLAY (TILL) – sandy
	1.5	2	SAND
	2	15.7	CLAY (TILL) – sandy
	15.7	24.5	SAND (not sampled)

Notes:

(1) Depth reported/interpreted as depths below EMS base;

Interpreted site geology based on combined field results and literature.

Table 5.2: Results of Grain Size Distribution Analyses

Site	**Material Description** [(1)]	**Percent (%) of Size Fraction** [(2)]			
		Gravel	**Sand**	**Silt**	**Clay**
NC1	SILT - and clay, some sand, trace gravel	1	10	51	38
SC1	CLAY – trace silt, trace sand	0	1	3 – 5	94 – 96
SC2	CLAY – trace to some silt	0	1	9	90
SE2	SILT – sandy, some clay, trace gravel	4	31	48	17
SE3	CLAY – some silt, trace sand	0	4	18	78
SW1	SAND – clayey, silty, trace gravel	2 – 4	35 – 36	28 – 31	32

Notes:

(1) Soil description terminology: “and” – 35% to 50%; adjective – 20% to 35%; “some” – 10-% to 20%; and “trace” > 0% to 10%;

(2) Gravel size fraction refers to grain size diameter of < 62 mm; sand size fraction refers to a grain size diameter of < 2 mm diameter; silt size fraction refers to a grain size diameter of < 0.063 mm; and clay size fraction refers to a grain size diameter of < 0.002 mm.

Table 5.3: Estimated Depth to Water Table

Site	Monitoring Date	Depth to Groundwater Table (m) [(1),(2)]	Average Depth to Groundwater Table (m) [(1),(2)]
NC1	March 23, 2001	1.0	1.2
	July 9, 2001	0.6	
	August 29, 2001	0.9	
	September 27, 2001	1.7	
	October 12, 2001	2.0	
NC2	February 19, 2001	2.8	1.9
	July 6, 2001	1.1	
	August 9, 2001	0.8	
	September 13, 2001	2.1	
	September 26, 2001	2.1	
	October 23, 2001	2.4	
NC3	July 5, 2001	1.2	1.4
	August 9, 2001	0.7	
	October 26, 2001	2.2	
SC2	July 4, 2001	2.5	2.5
	July 17, 2001	2.5	
	August 2, 2001	2.5	
SE2	July 9, 2001	0.2	0.4
	July 13, 2001	0.5	
	August 2, 2001	0.2	
	August 28, 2001	0.1	
	September 12, 2001	0.6	
	September 28, 2001	0.7	
SE3	August 13, 2001	1.1	0.8
	August 28, 2001	0.6	
	October 18, 2001	0.7	

Notes:
(1) Depth to water based on piezometric elevations measured in piezometers;
(2) All depths reported relative to ground surface.

Table 5.4: Results of Falling Head Recovery Tests

Site	Piezometer Screened Interval (m)[1]		Material Description	Average Hydraulic Conductivity (m/s)	Number of Tests Conducted
	From	To			
NC1	3.99	5.28	Till	3.4×10^{-8}	3
	7.86	8.95	Till	1.6×10^{-8}	3
NC2	3.81	4.72	Till	2.9×10^{-7}	3
	6.70	7.62	Till	9.3×10^{-7}	3
NC3	3.66	4.77	Till	5.7×10^{-9}	1
	6.40	7.37	Till	7.8×10^{-6}	3
SC2	1.98	3.05	Clay	8.5×10^{-9}	1
	5.94	6.71	Clay	6.2×10^{-9}	2
	10.36	11.12	Clay	1.8×10^{-9}	1
SE2	5.79	6.97	Till	2.6×10^{-7}	3
SE3	3.35	4.27	Clay	3.0×10^{-8}	2

Notes:
(1) All depths reported relative to ground surface.

Table 5.5: Results of Stored Manure Analysis

Parameter	Units	CCME Drinking Water Criteria (1)	Site									
			NC1	NC2	NC3	SC1	SC2	SC3	SE1	SE2	SE3	SW1
Ammonia-N	mg/L	-	179 – 433	10.7	-	-	882 – 1940	-	215	49.5 – 974	672	-
Ammonium-N	mg/L	-	-	-	599	3973	-	2560	-	1194	1552	330
Nitrate+nitrite-N	mg/L	10 (MAC)	0.26 – 412	0.03	-	-	0.29 – 0.71	< 0.1	0.17	0.77 – 2.06	0.27	-
Phosphorus-P	mg/L	-	24.7 – 178	27.2	-	-	70.1 – 78.1	210	170	32.9 – 393	102 – 119	40.0
Electrical Conductivity	umhos/cm	-	3350 – 6830	802	-	2720	9110 – 14500	2240	5380	2420 – 9820	7870	-
pH	pH units	6.5 – 8.5	7.77 – 8.19	7.74	6.90	7.40	7.76 – 7.77	6.50	6.80	7.98 – 8.16	7.96	-
Chloride	mg/L	250 (AO)	228 – 448	24.0	244	2300	515 – 1190	1100	880	< 100 – 542	456 – 471	250
Potassium	mg/L	-	362 – 473	37.5	839	50.0	858 – 1250	1480	144	371 – 917	818 – 1075	320
Sodium	mg/L	200 (AO)	159 – 167	24.1	240	190	449	834	313	382 – 478	156 – 239	120
Sulphate	mg/L	500 (AO)	309 - 456	24.0	1198	270	186 - 736	658	42.0	159 – 628	190	450

Notes:

(1) Canadian Council of Ministers for the Environment (CCME) Canadian Environmental Quality Guidelines (Chapter 2: Community Water), 2000; MAC – maximum acceptable concentration (health related); AO – aesthetic objective (non-health related).

- indicates no guideline established and/or analysis not conducted.

Table 5.6: Summary of Background Sample Core Concentrations

Site	Statistics	Concentration (mg/kg)					
		Chloride	**Ammonia**	**Nitrate+nitrite**	**Phosphate**	**Potassium**	**Sulphate**
NC1	Range	7.2 – 20.0	1.3 – 9.8	1.8 – 8.9	6.2 – 15.5	72.1 – 243	6.0 – 430
	Average	10.8	4.8	2.7	6.5	99.0	50.6
	Median	10.0	1.9	2.2	6.2	85.1	18.2
NC2	Range	3.0 – 22.0	0.4 – 10.5	3.5 – 85.9	0.1 – 4.0	31.9 – 448	24.0 – 213
	Average	12.4	3.6	16.8	0.4	162	69.2
	Median	12.8	2.3	8.9	0.1	114	67.2
NC3	Range	13.6 – 75.8	1.0 – 10.2	3.5 – 138.2	0.1 – 806	59.6 – 1320	26.4 – 226
	Average	30.2	3.5	33.7	88.3	312	79.9
	Median	27.5	2.0	15.1	1.5	145	69.6
SC2	Range	36.0 – 634	4.3 – 18.4	1.8 – 12.0	0.5 – 12.4	220 – 568	20.6 - 6180
	Average	333	10.5	5.8	3.8	400	1414
	Median	331	9.1	5.8	3.1	402	1155
SC3	Range	31.0 – 353	6.0 – 48.0	14.0 – 329	7.0 – 121	-	-
	Average	86.0	18.0	99.0	50.0	-	-
	Median	67.0	17.0	68.0	23.0	-	-
SE2	Range	11.0 – 46.2	3.5 – 44.2	1.8 – 15.5	6.2 – 12.4	21.6 – 120	8.3 – 543
	Average	20.2	17.1	2.2	7.1	63.4	53.0
	Median	18.0	17.6	1.8	6.2	66.5	36.3
SE3	Range	5.6 – 77.4	1.8 – 15. 8	2.7 – 12.4	0.1 – 7.0	84.4 – 461	32.4 – 8400
	Average	28.7	9.0	5.8	3.3	306	3612
	Median	19.2	7.6	5.3	3.0	368	2860

Table 5.7: Comparison of Extracted and Computed Pore Water Chloride Concentrations

Sample Core No.	Sample Interval Depth (m)			Concentration in Soil (mg/kg)	Moisture Content (%)	Pore Water Concentration (mg/L)		Variation (%)
	From	To	Average			Computed	Extracted	
1	1.32	1.52	1.42	212	28.8	736	813	9
1	1.83	2.03	1.93	1.1	30.2	349	277	26
1	2.24	2.44	2.34	76	33.8	264	145	82
1	2.74	2.90	2.82	42	34.5	146	72.4	101
1	3.20	3.35	3.28	22	34.6	76	48.9	56
1	3.81	3.96	3.89	24	36.9	83	43.6	91
3	1.02	1.32	1.17	66	22.1	299	258	16

Table 5.8: Summary of Select Chemical Species Concentrations in Groundwater Collected from Piezometer Installations

Parameter	**Units**	**CCME Drinking Water Criteria** [1]	**NC1**		**NC2**		**NC3**		**SC2**			**SE2**	**SE3**	
			Shallow	**Deep**	**Shallow**	**Deep**	**Shallow**	**Deep**	**Shallow**	**Medium**	**Deep**	**Deep**	**Shallow**	**Deep**
Ammonia-N	mg/L	-	0.08	0.27	0.13	0.05	0.13	0.11	0.12	0.77	0.99	1.14	0.17	0.48
Nitrate+nitrite-N	mg/L	10 (MAC)	0.30	0.04	5.98	0.27	25.0	0.04	0.37	0.01	0.04	0.03	0.61	0.02
Phosphorus-P	mg/L	-	0.018	0.012	0.018	0.018	0.07	0.015	0.014	0.028	0.023	0.007	0.027	0.018
Electrical Conductivity	umhos/cm	-	868	900	2490	1540	3040	853	5500	5660	4400	703	2080	2490
pH	pH units	6.5 – 8.5	7.59	7.60	7.52	7.41	7.25	8.22	7.48	7.24	7.36	7.66	7.20	7.09
Chloride	mg/L	250 (AO)	< 10	< 10	56	46	279	33	702	856	1120	< 10	17	19
Potassium	mg/L	-	4.25	4.46	6.21	7.07	11.0	9.66	7.94	12.0	9.26	4.51	3.37	6.10
Sodium	mg/L	200 (AO)	67.0	77.4	357	81.8	243	57.7	574	653	512	44.7	303	200
Sulphate	mg/L	500 (AO)	71	93	570	175	639	81	2410	1750	716	23	598	972

Notes:

(1) Canadian Council of Ministers for the Environment (CCME) Canadian Environmental Quality Guidelines (Chapter 2: Community Water), 2000; MAC – maximum acceptable concentration (health related); AO – aesthetic objective (non-health related).

- indicates no guideline established and/or analysis not conducted.

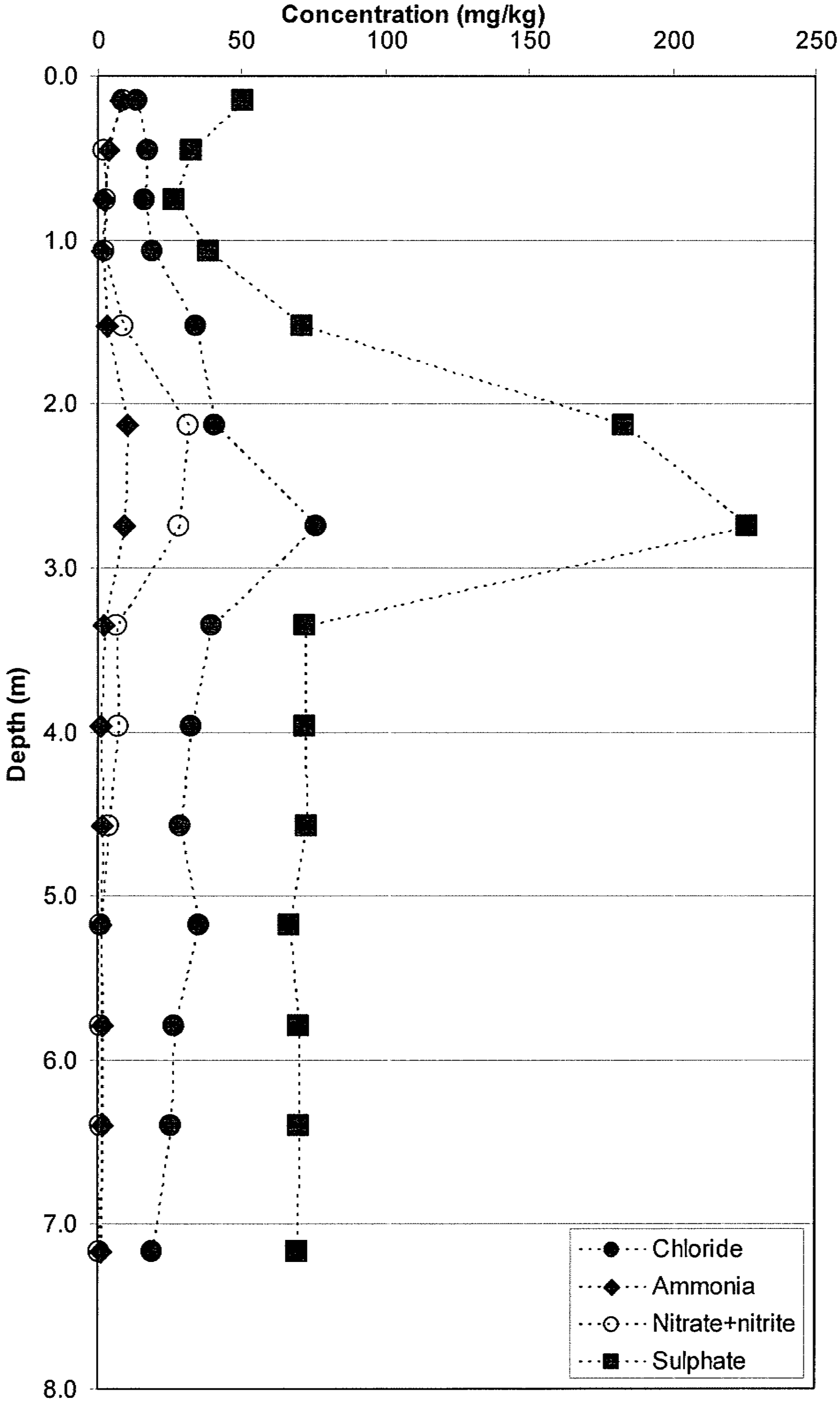


Figure 5.1: Soil quality profile for the NC3 background sample core.

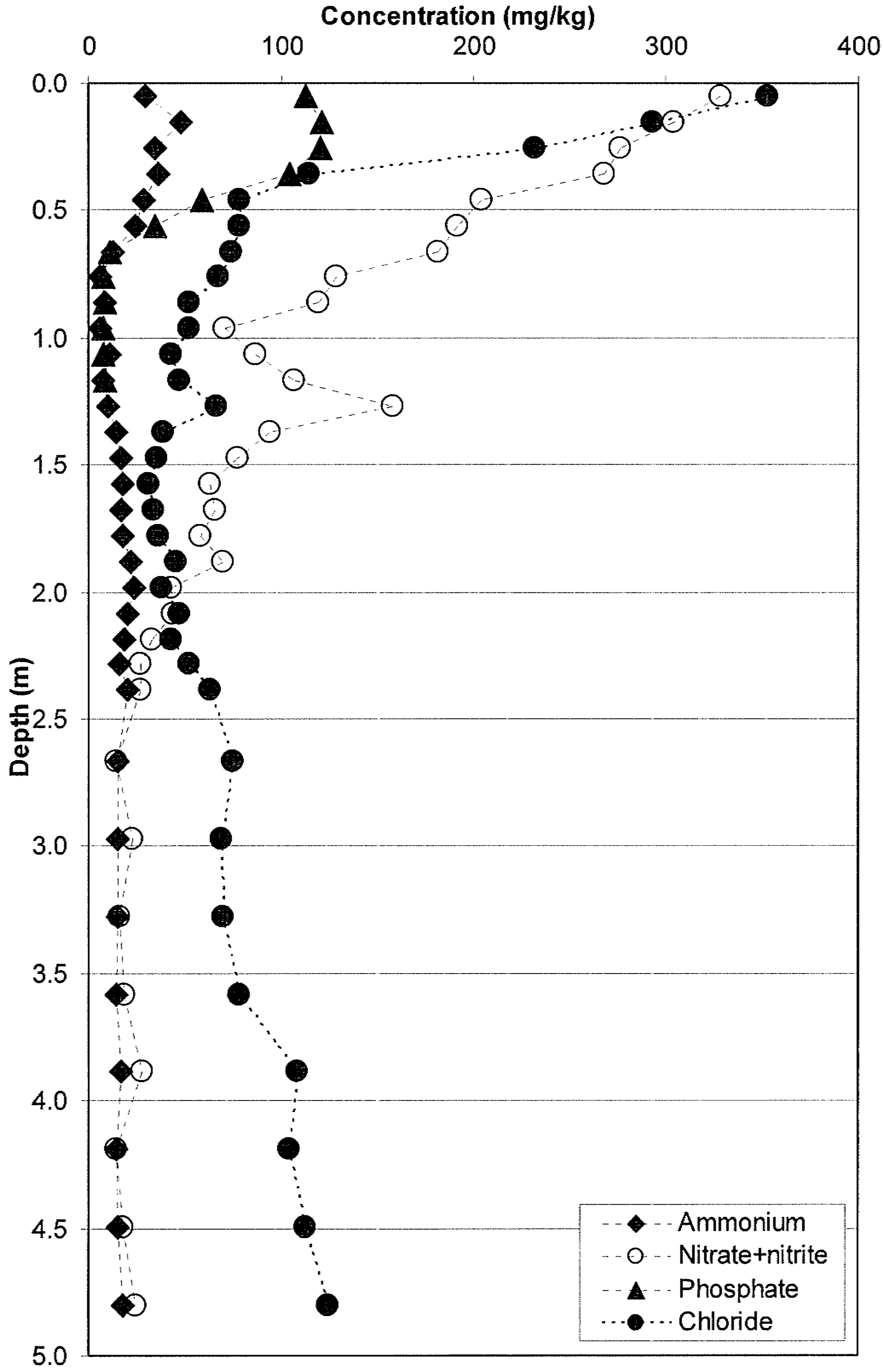


Figure 5.2: Soil quality profile for the SC3 background sample core.

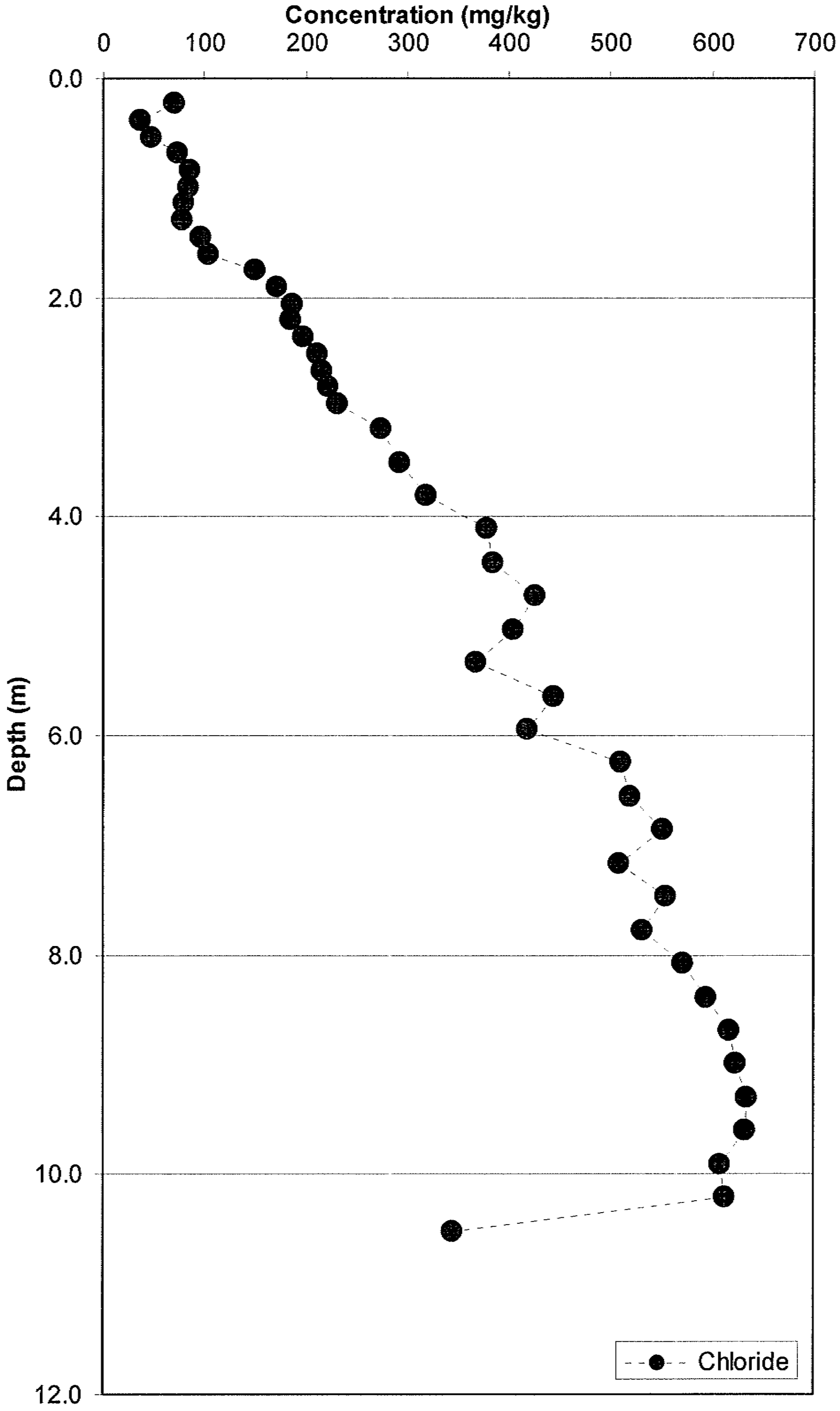


Figure 5.3: SC2 background sample core chloride concentrations.

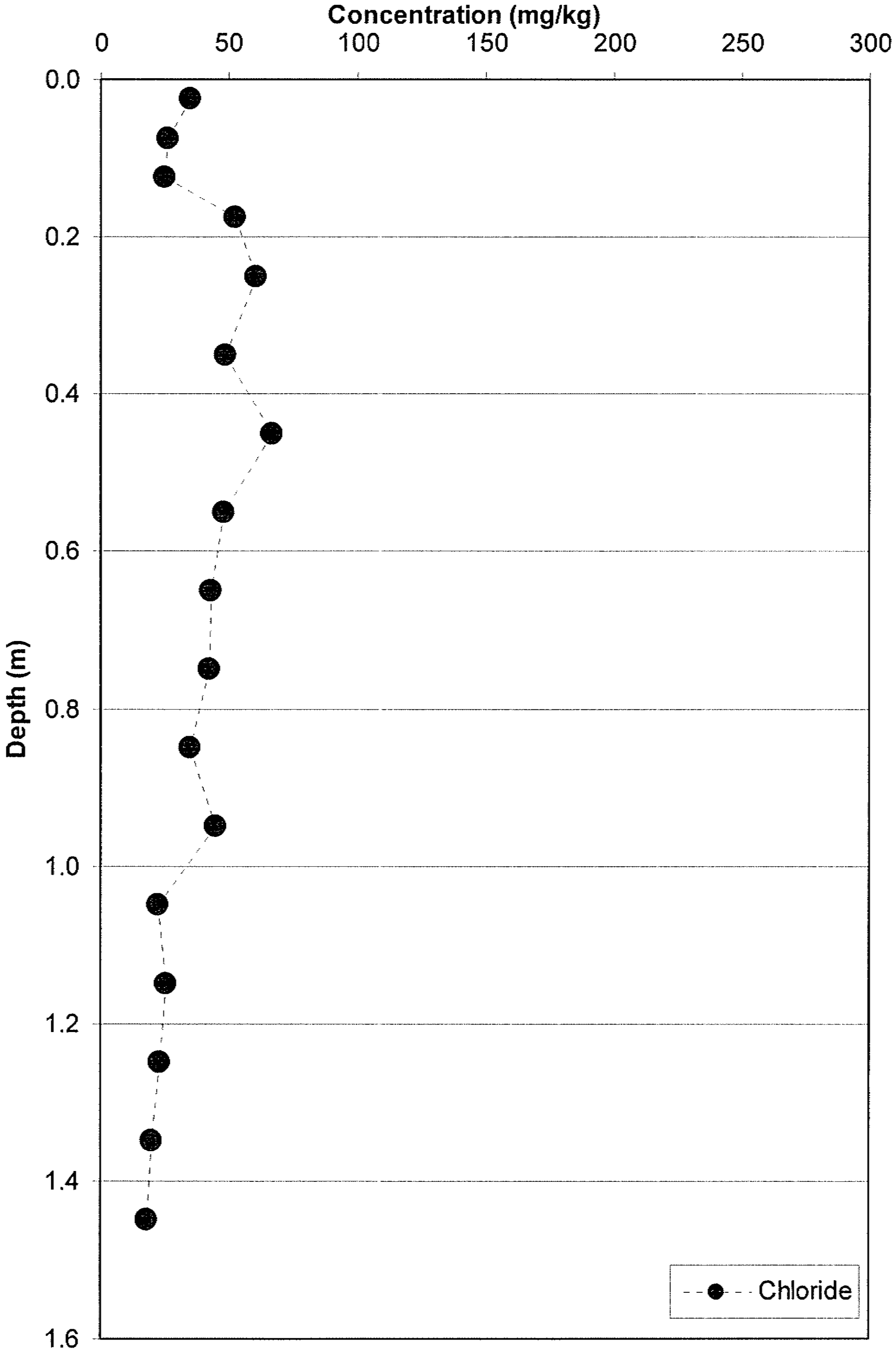


Figure 5.4: Soil quality profile (salts) for the NC1 4 year EMS sample core.

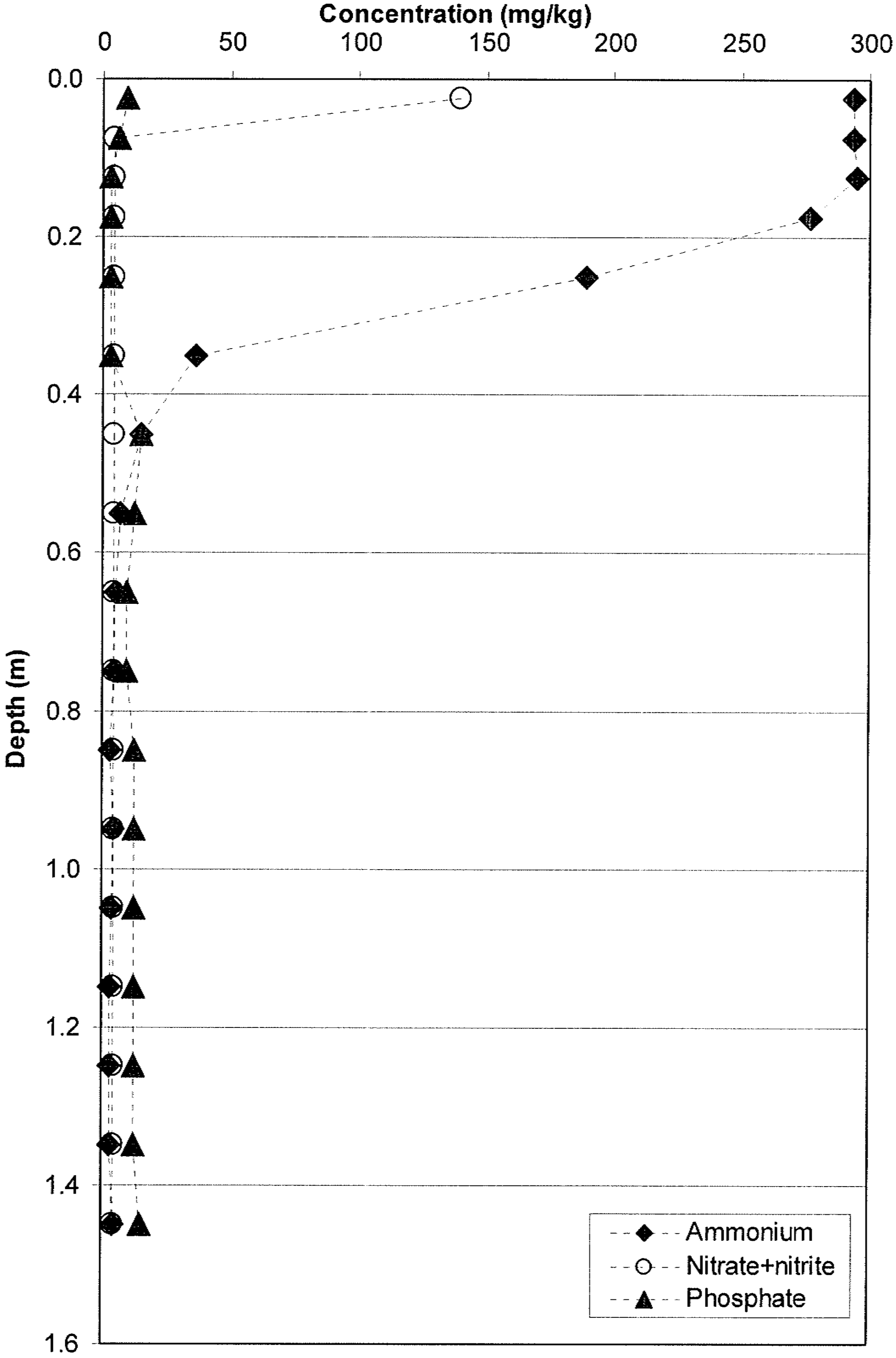


Figure 5.5: Soil quality profile (nutrients) for the NC1 4 year EMS sample core.

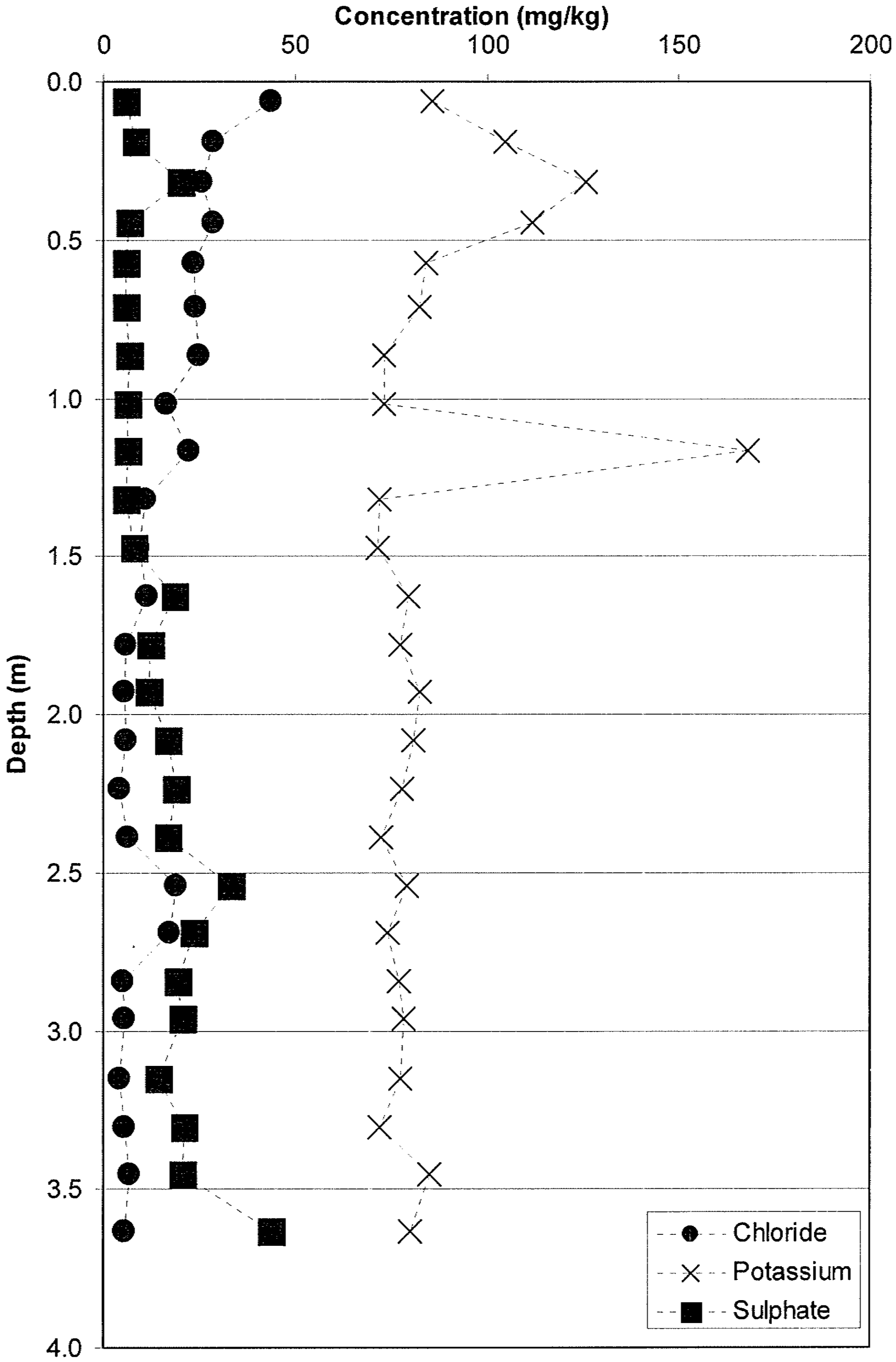


Figure 5.6: Soil quality profile (salts) for the NC1 7 year EMS sample core.

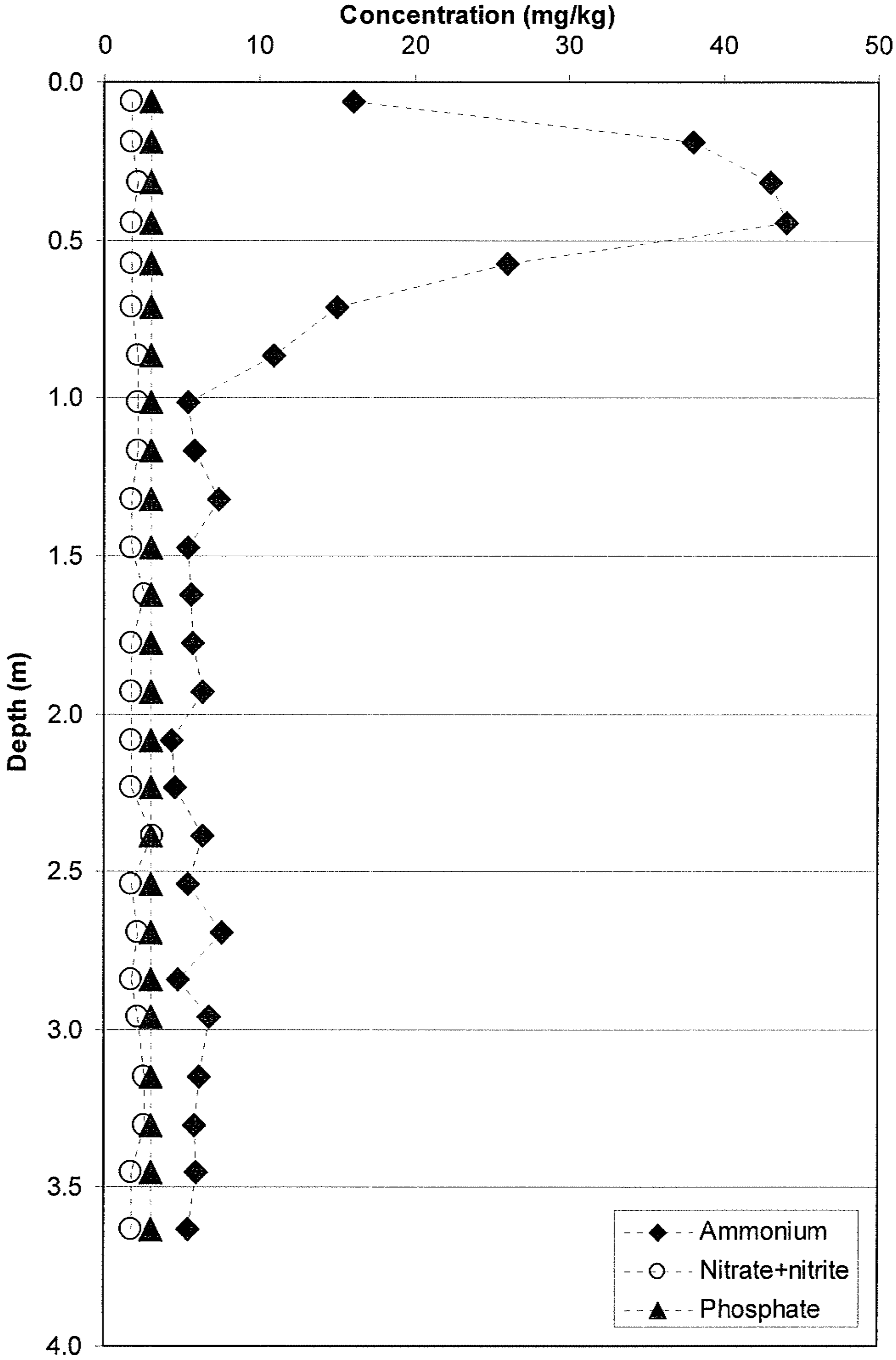


Figure 5.7: Soil quality profile (nutrients) for the NC1 7 year EMS sample core.

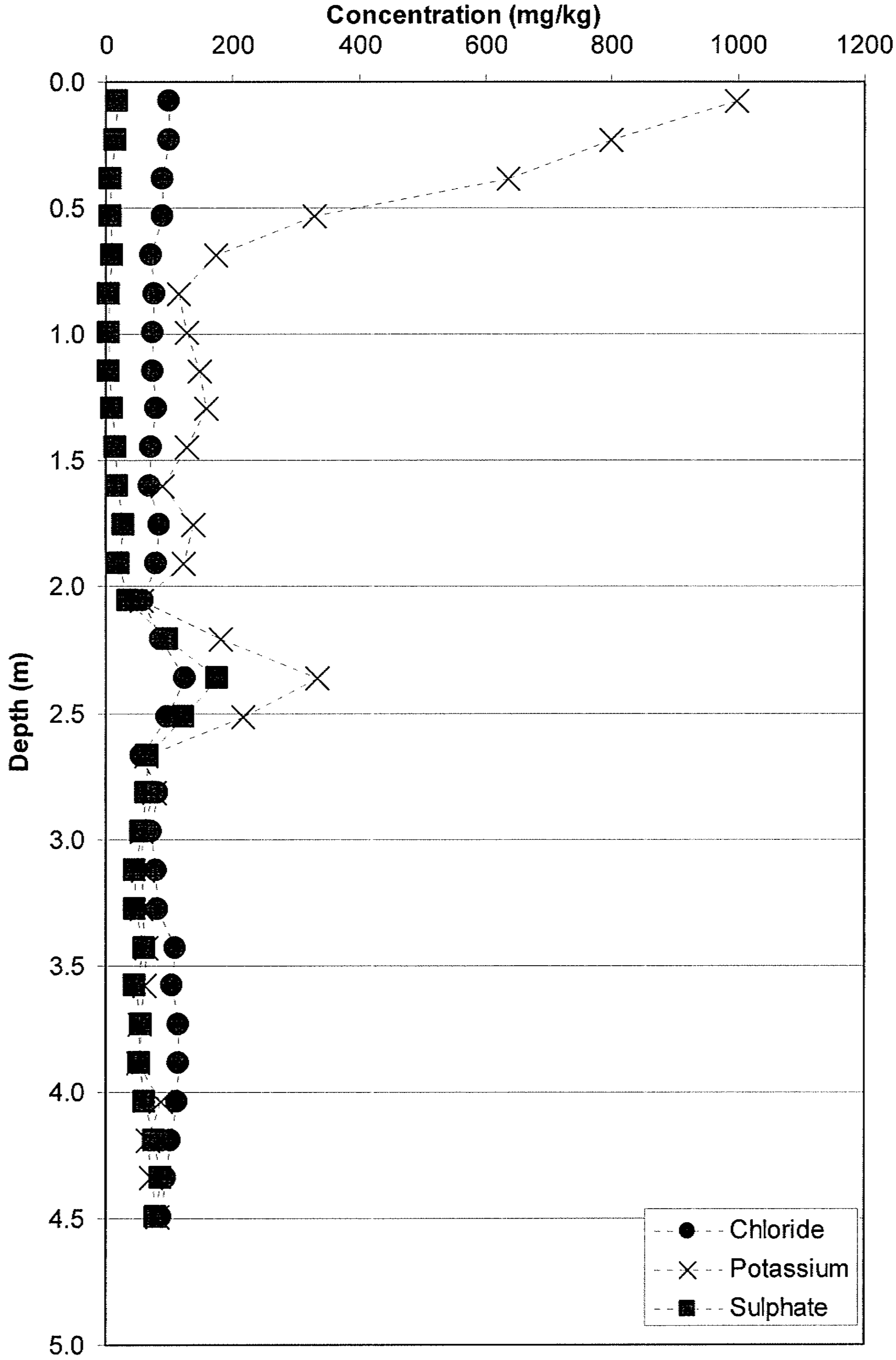


Figure 5.8: Soil quality profile (salts) for the NC2 19 year EMS sample core.

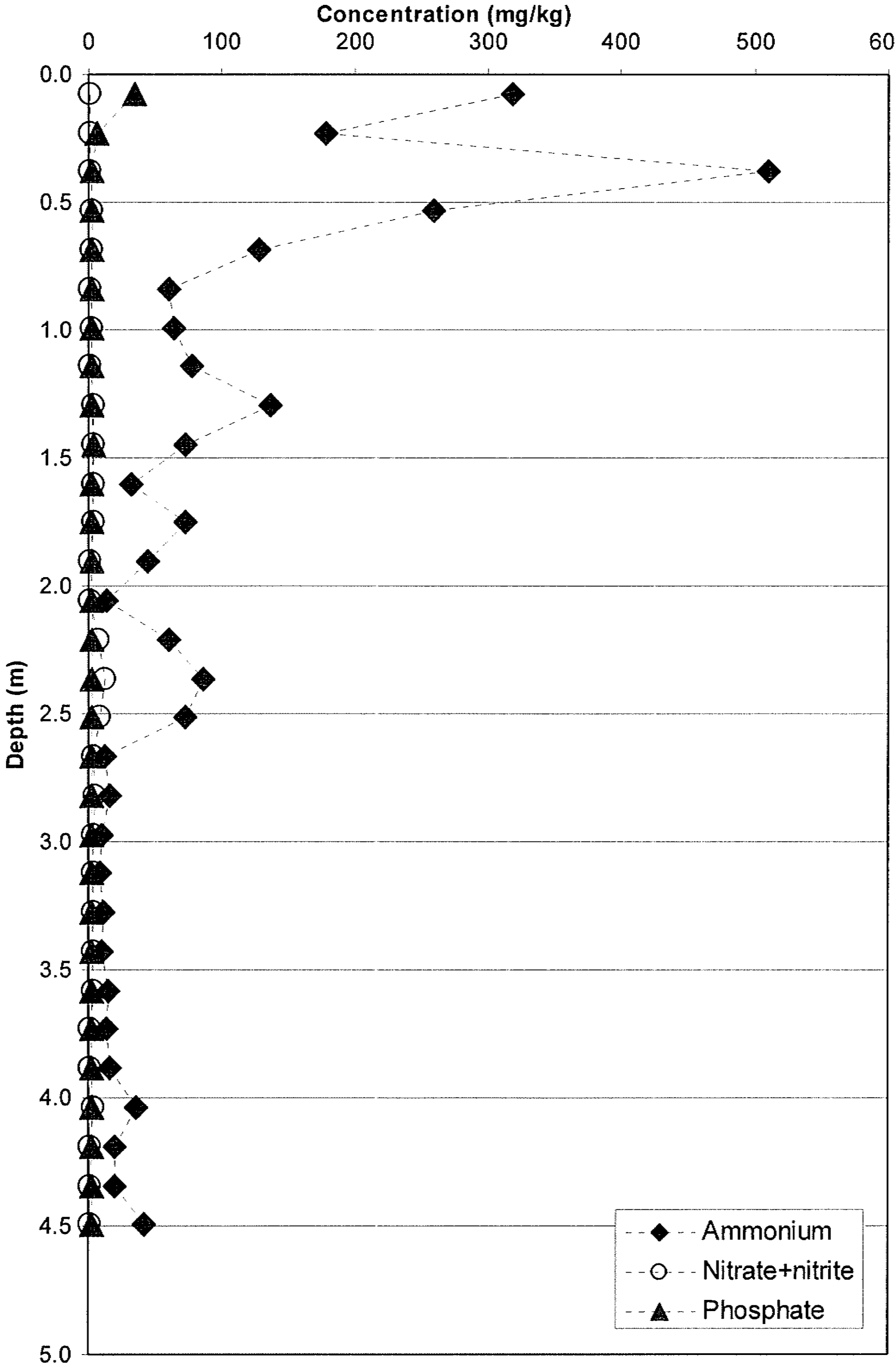


Figure 5.9: Soil quality profile (nutrients) for the NC2 19 year EMS sample core.

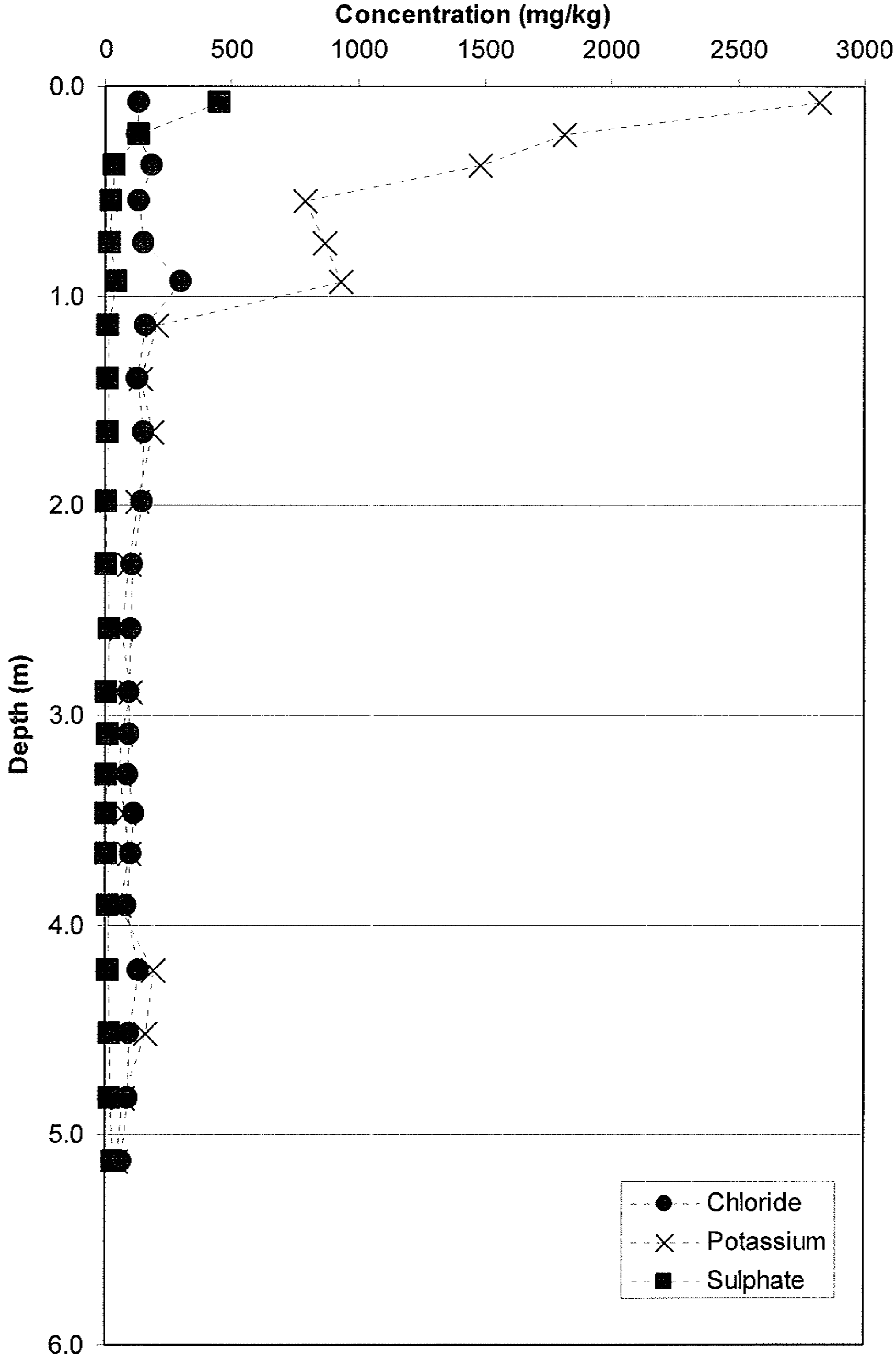


Figure 5.10: Soil quality profile (salts) for the NC3 29 year EMS sample core.

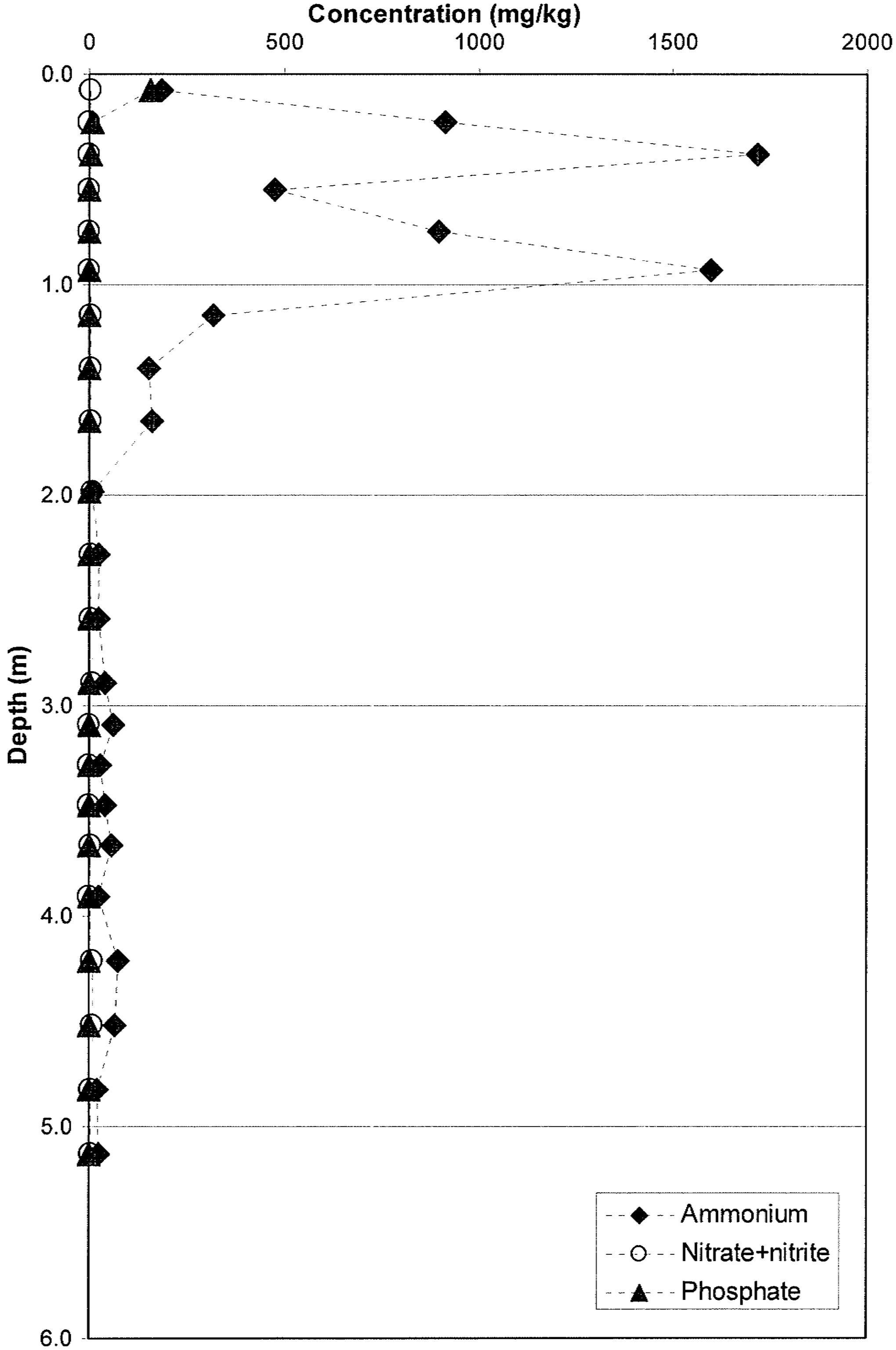


Figure 5.11: Soil quality profile (nutrients) for the NC3 29 year EMS sample core.

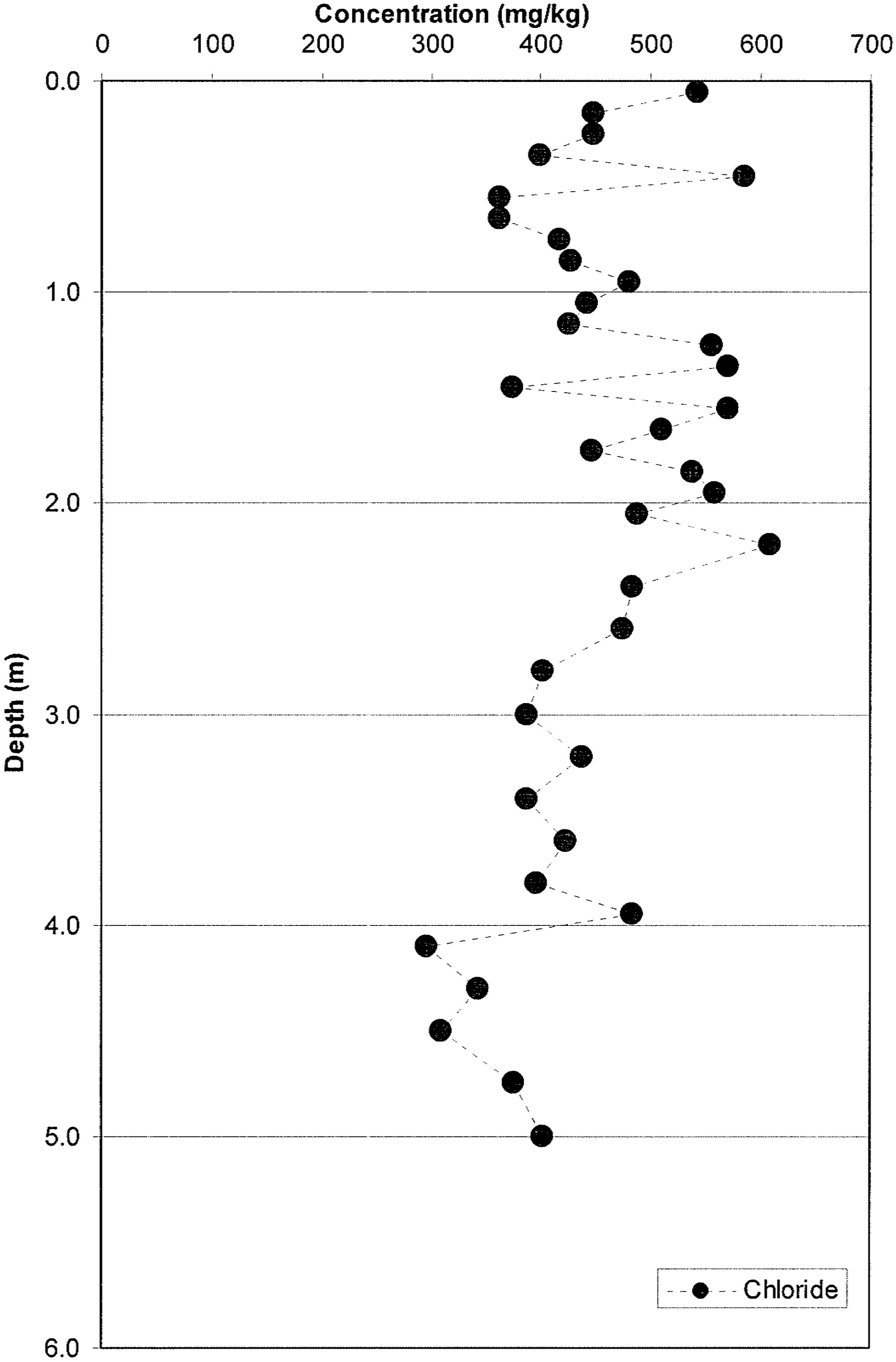


Figure 5.12: Soil quality profile (salts) for the SC1 9 year EMS sample core.

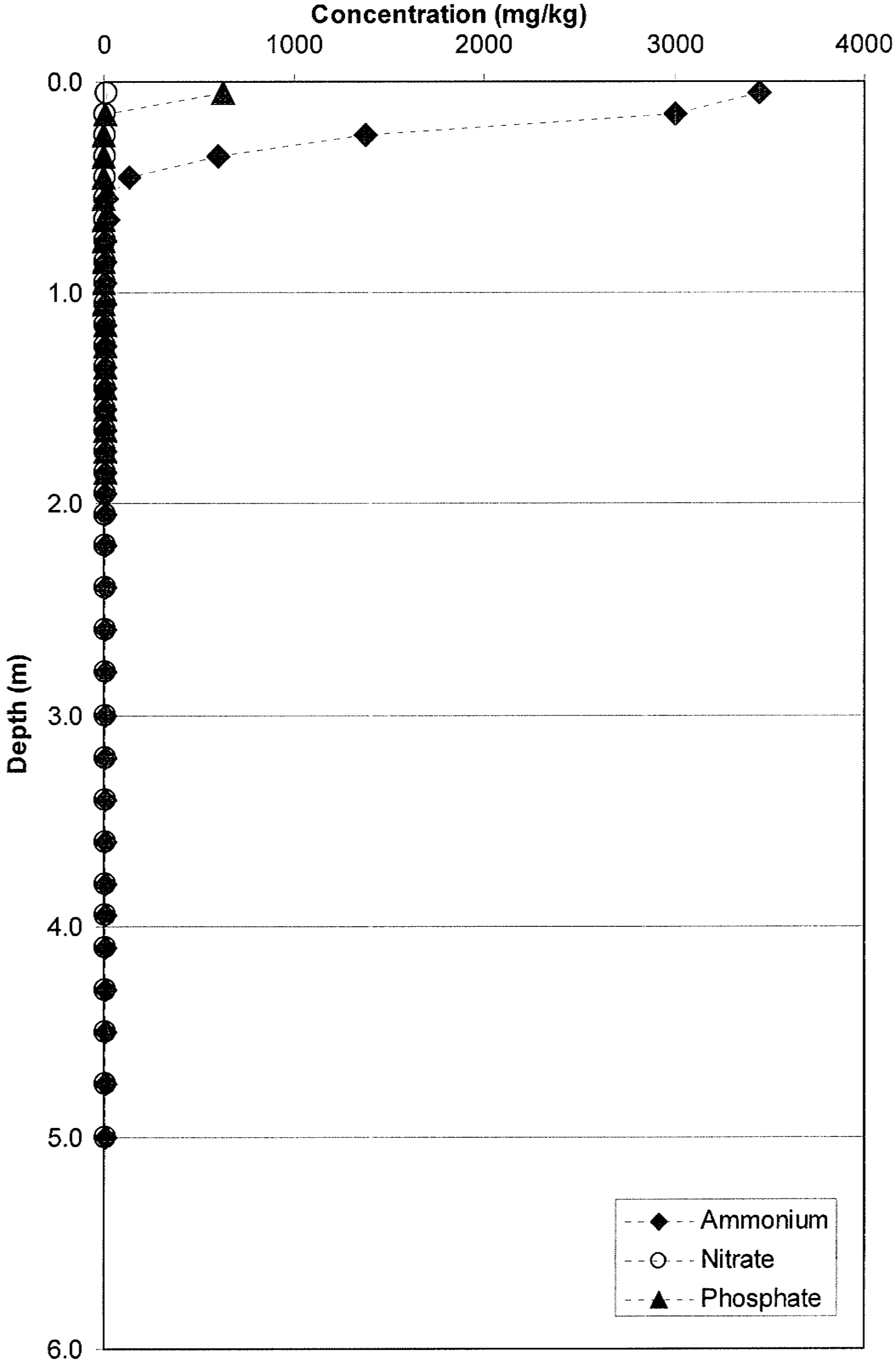


Figure 5.13: Soil quality profile (nutrients) for the SC1 9 year EMS sample core.

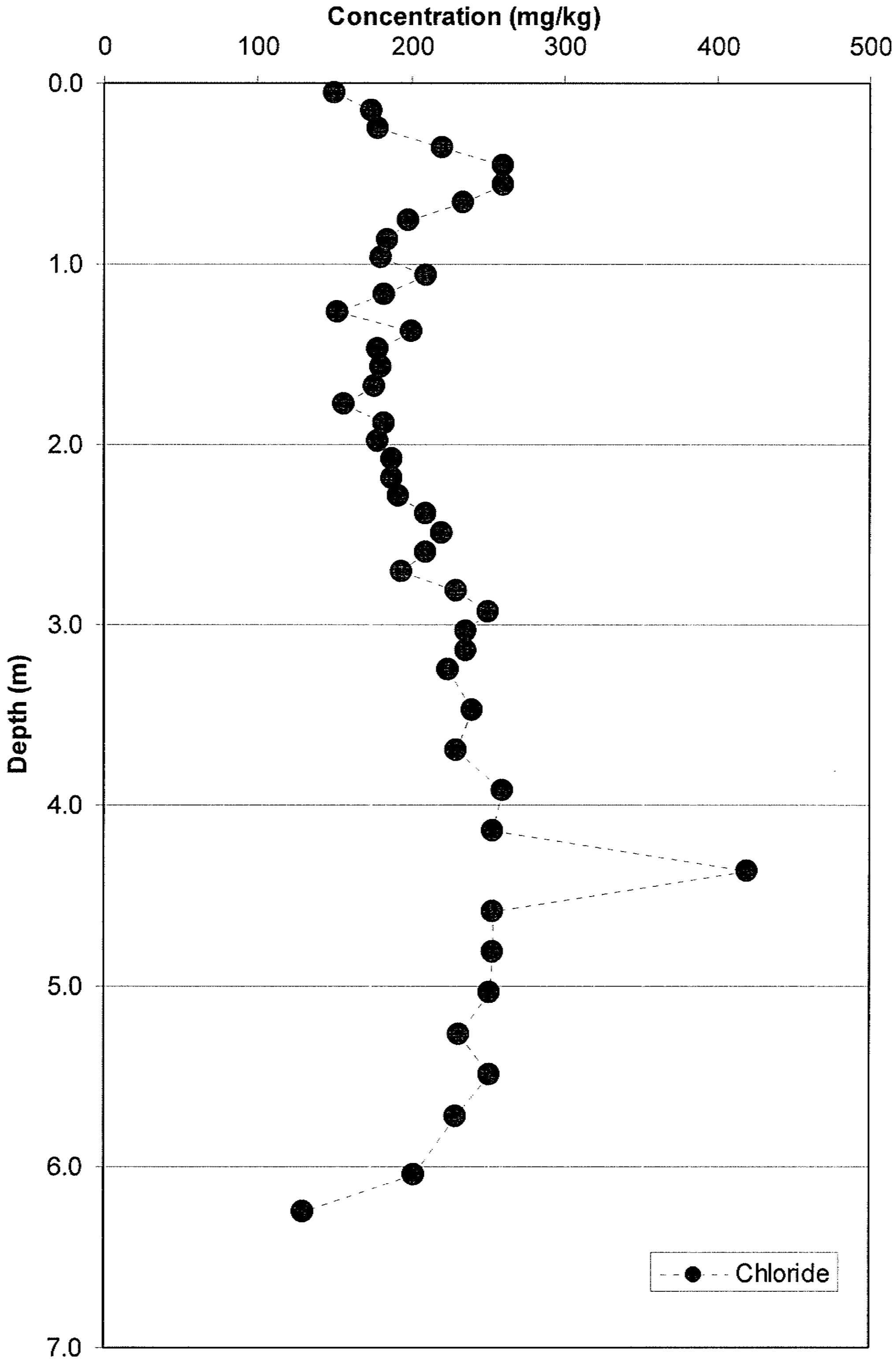


Figure 5.14: Soil quality profile (salts) for the SC2 20 year EMS sample core.

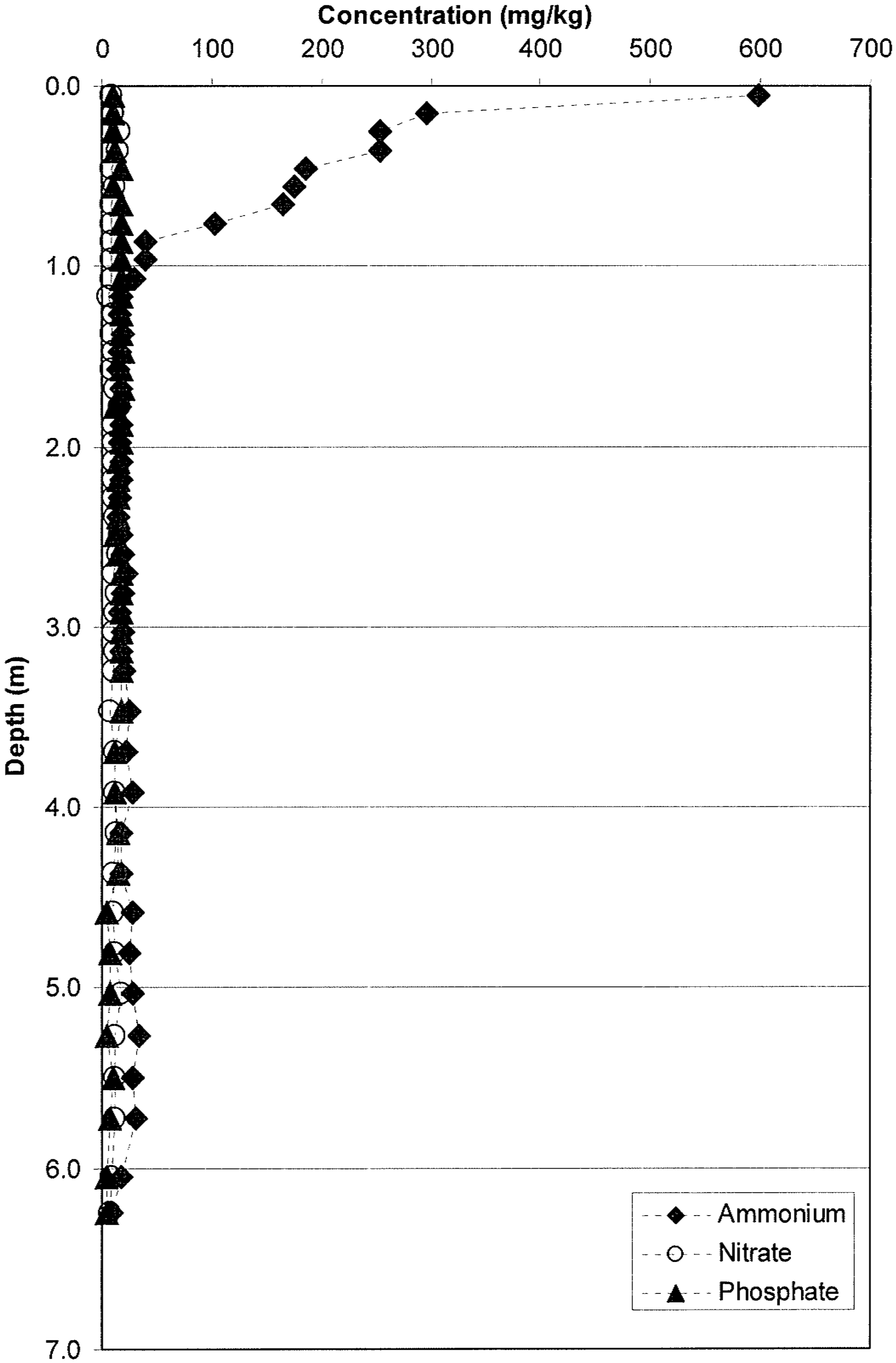


Figure 5.15: Soil quality profile (nutrients) for the SC2 20 year EMS sample core.

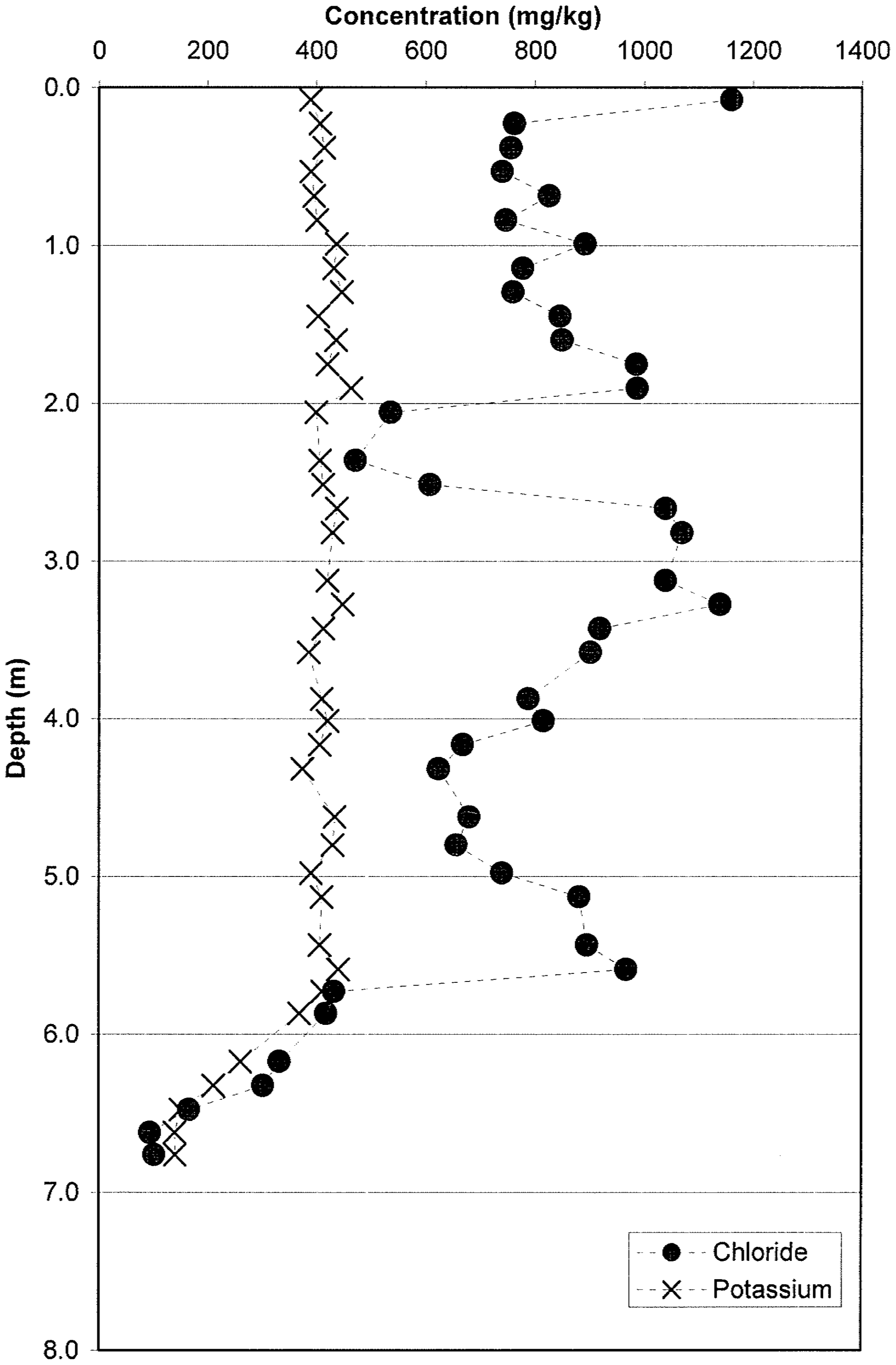


Figure 5.16: Soil quality profile (salts) for the SC2 21 year EMS sample core.

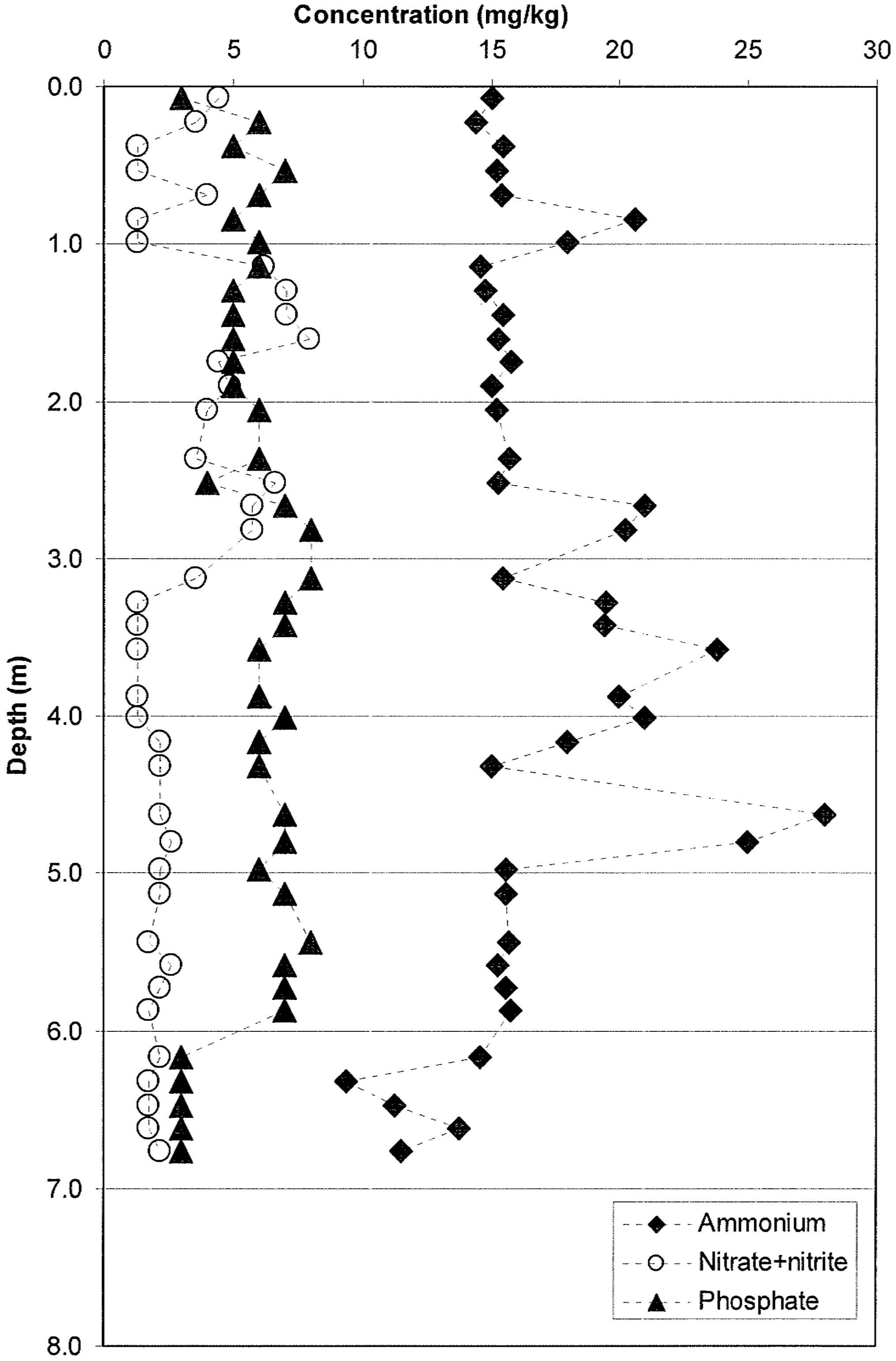


Figure 5.17: Soil quality profile (nutrients) for the SC2 21 year EMS sample core.

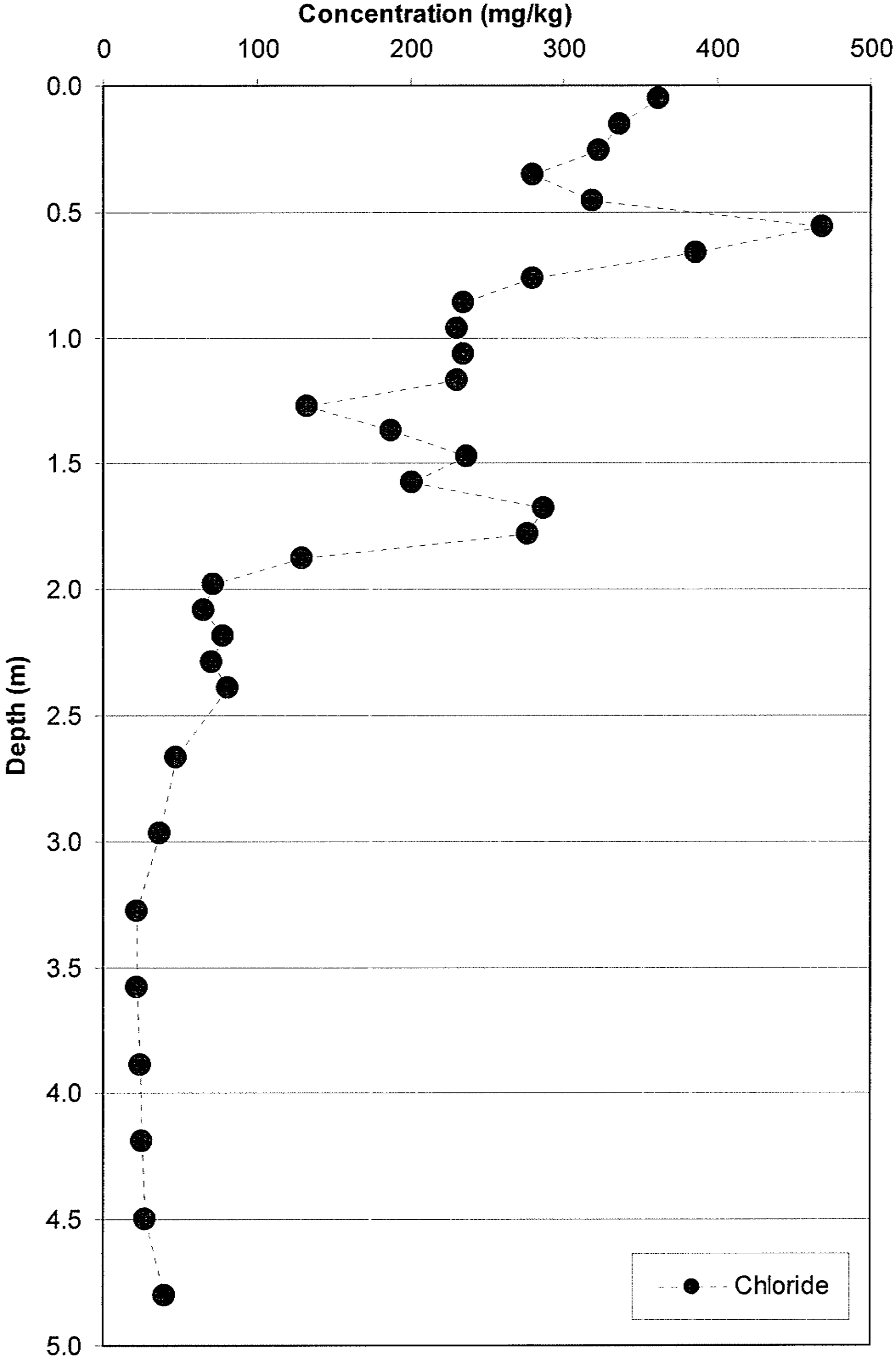


Figure 5.18: Soil quality profile (salts) for the SC3 20 year EMS sample core #1.

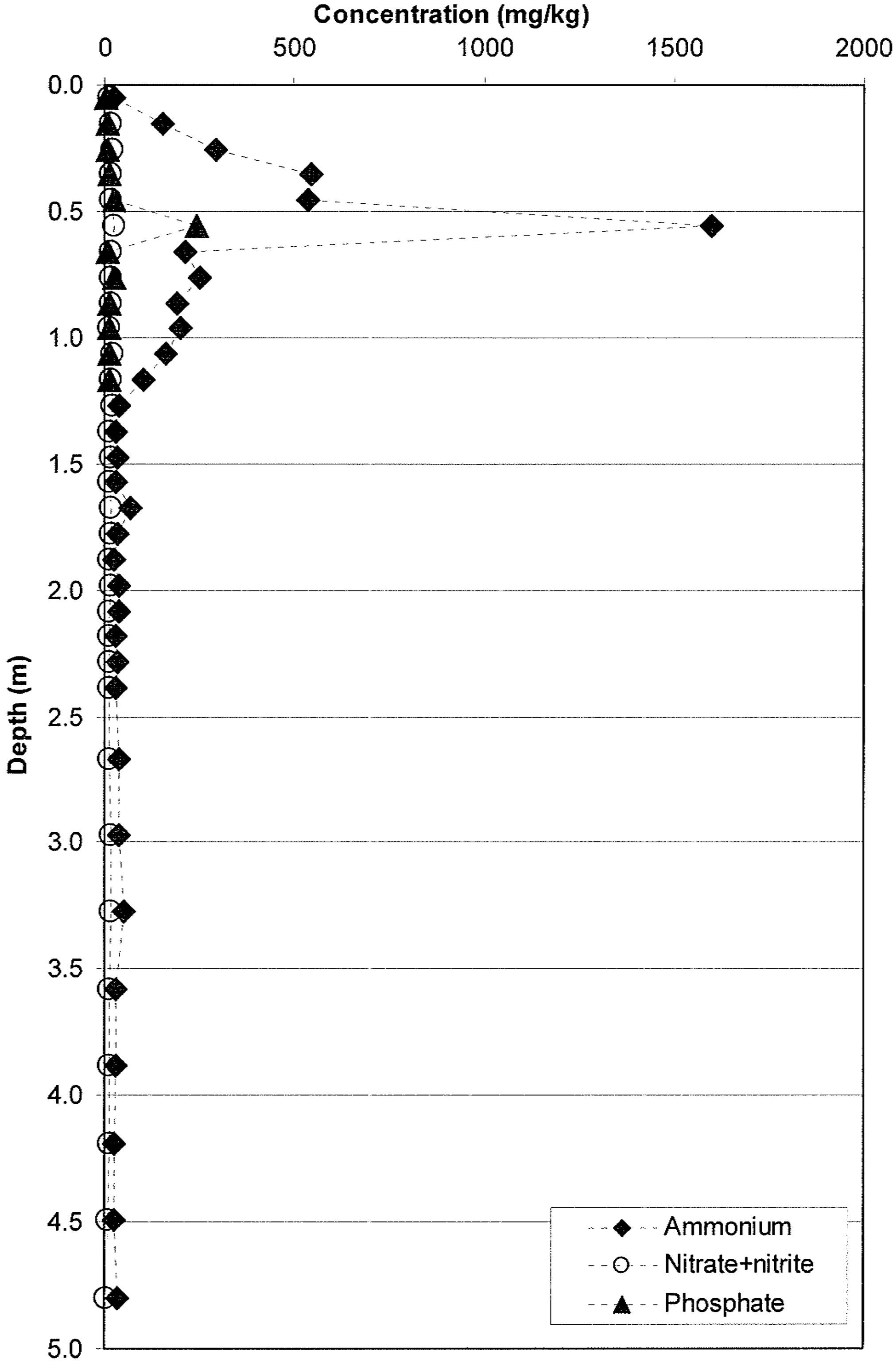


Figure 5.19: Soil quality profile (nutrients) for the SC3 20 year EMS sample core #1.

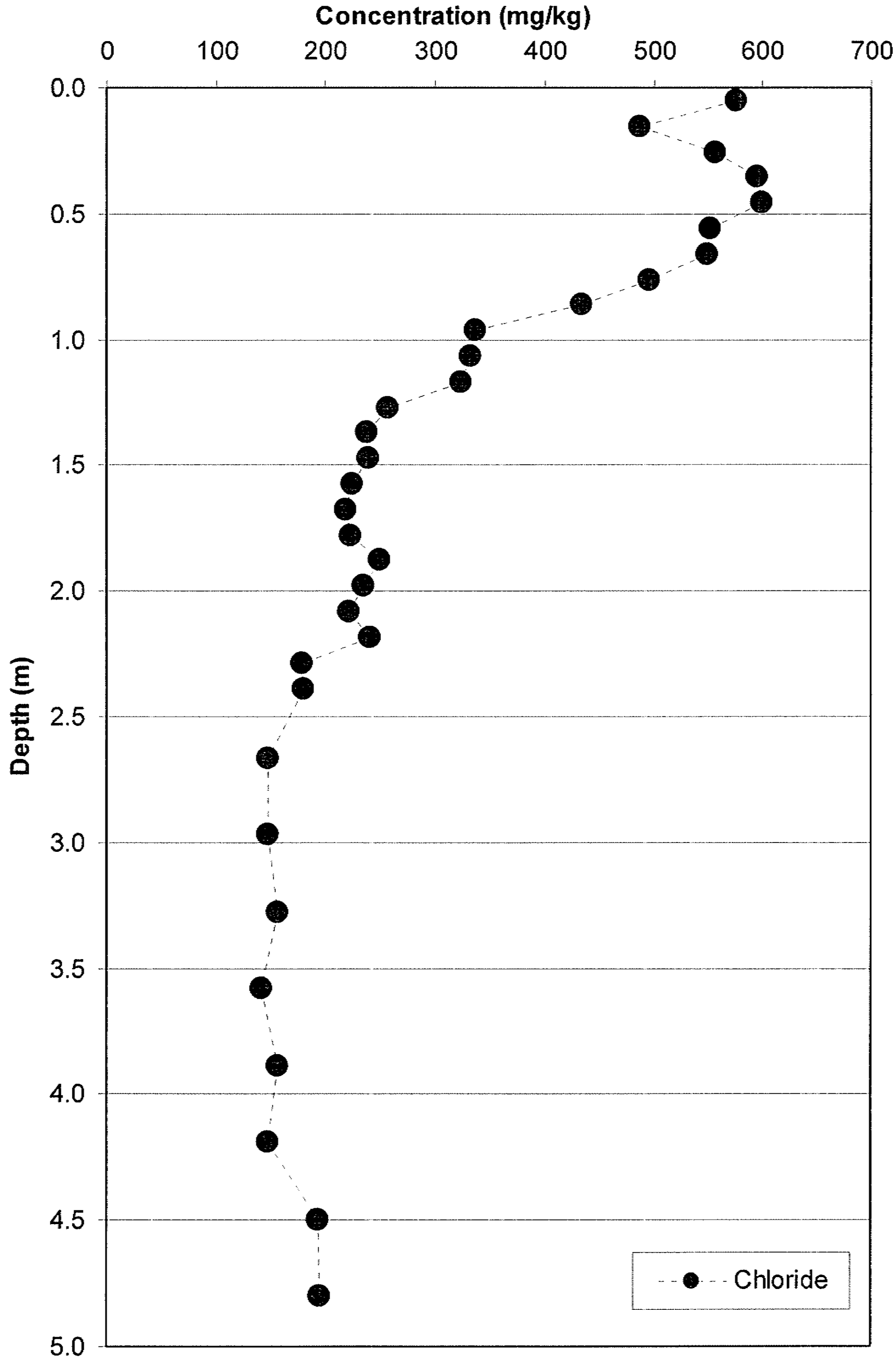


Figure 5.20: Soil quality profile (salts) for the SC3 20 year EMS sample core #2.

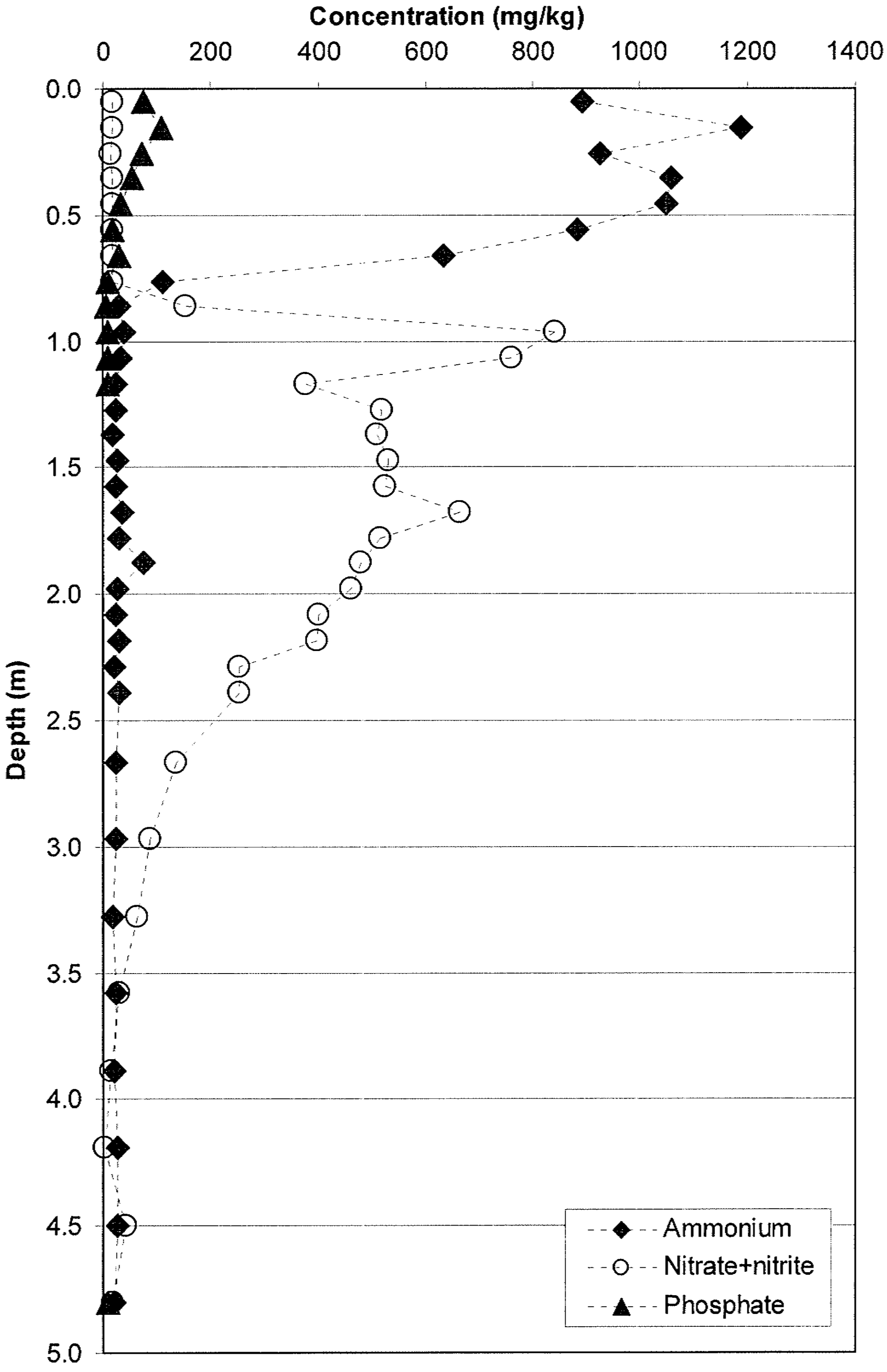


Figure 5.21: Soil quality profile (nutrients) for the SC3 20 year EMS sample core #2.

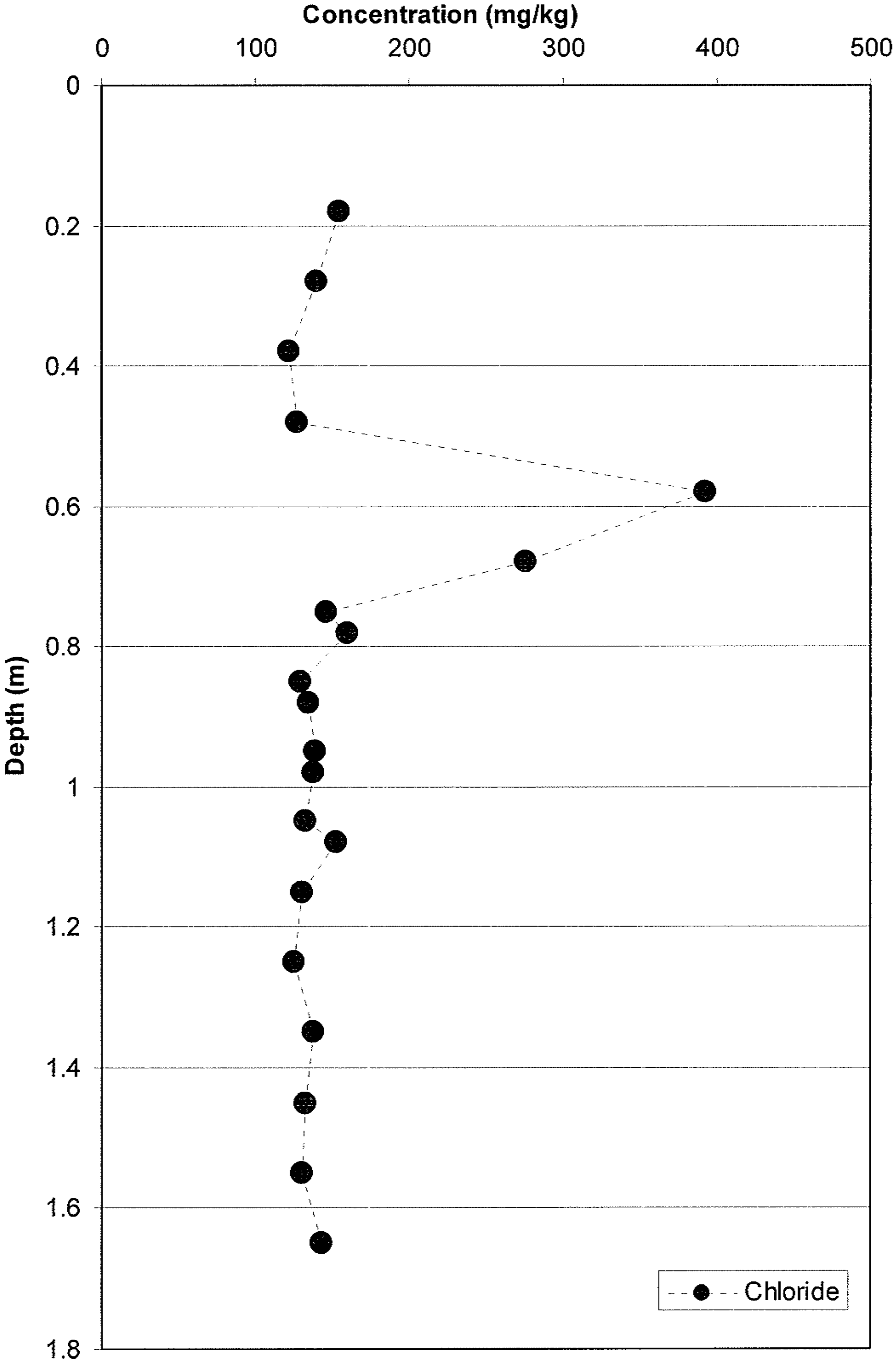


Figure 5.22: Soil quality profile (salts) for the SE1 14 year EMS sample core.

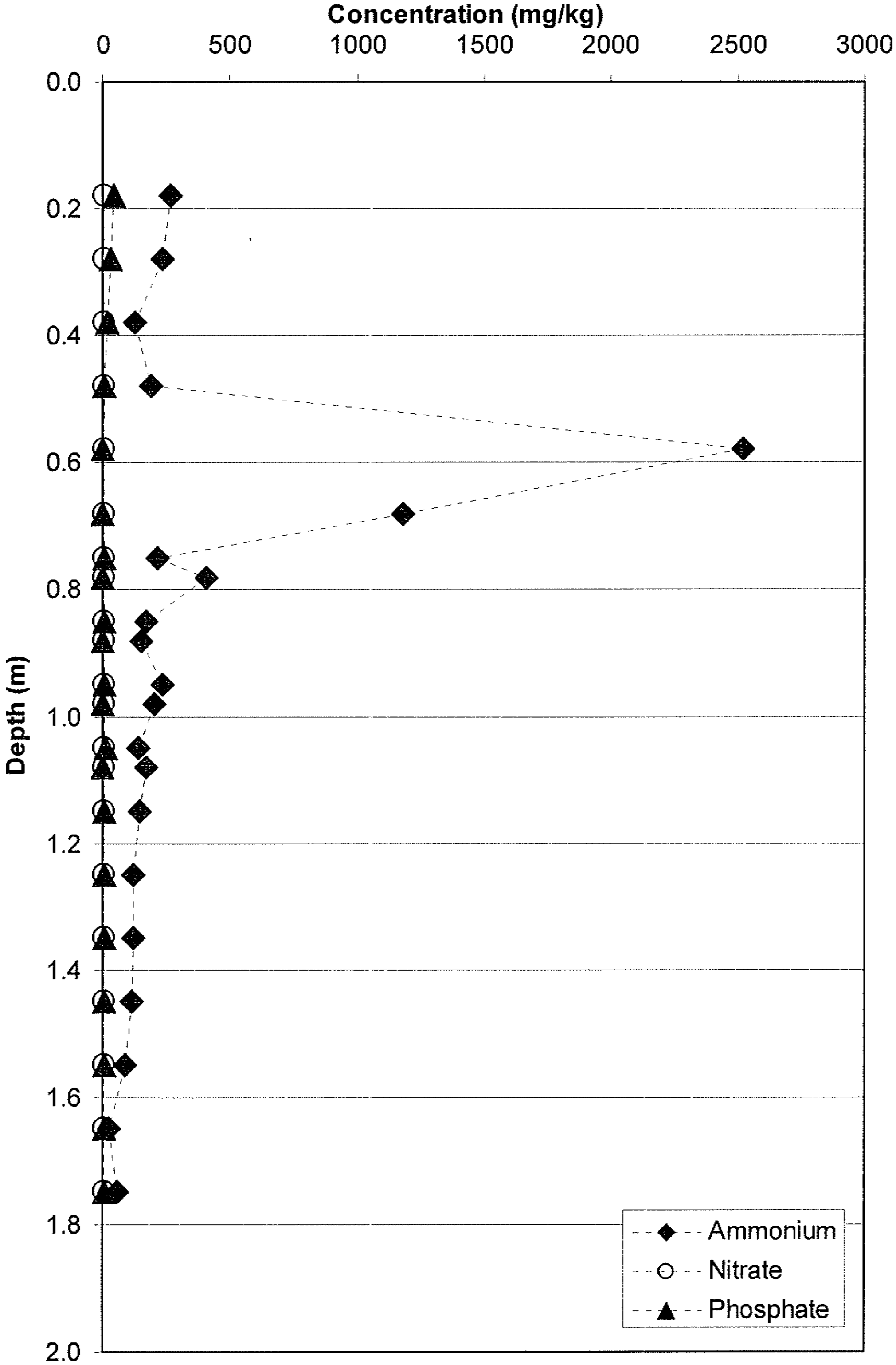


Figure 5.23: Soil quality profile (nutrients) for the SE1 14 year EMS sample core.

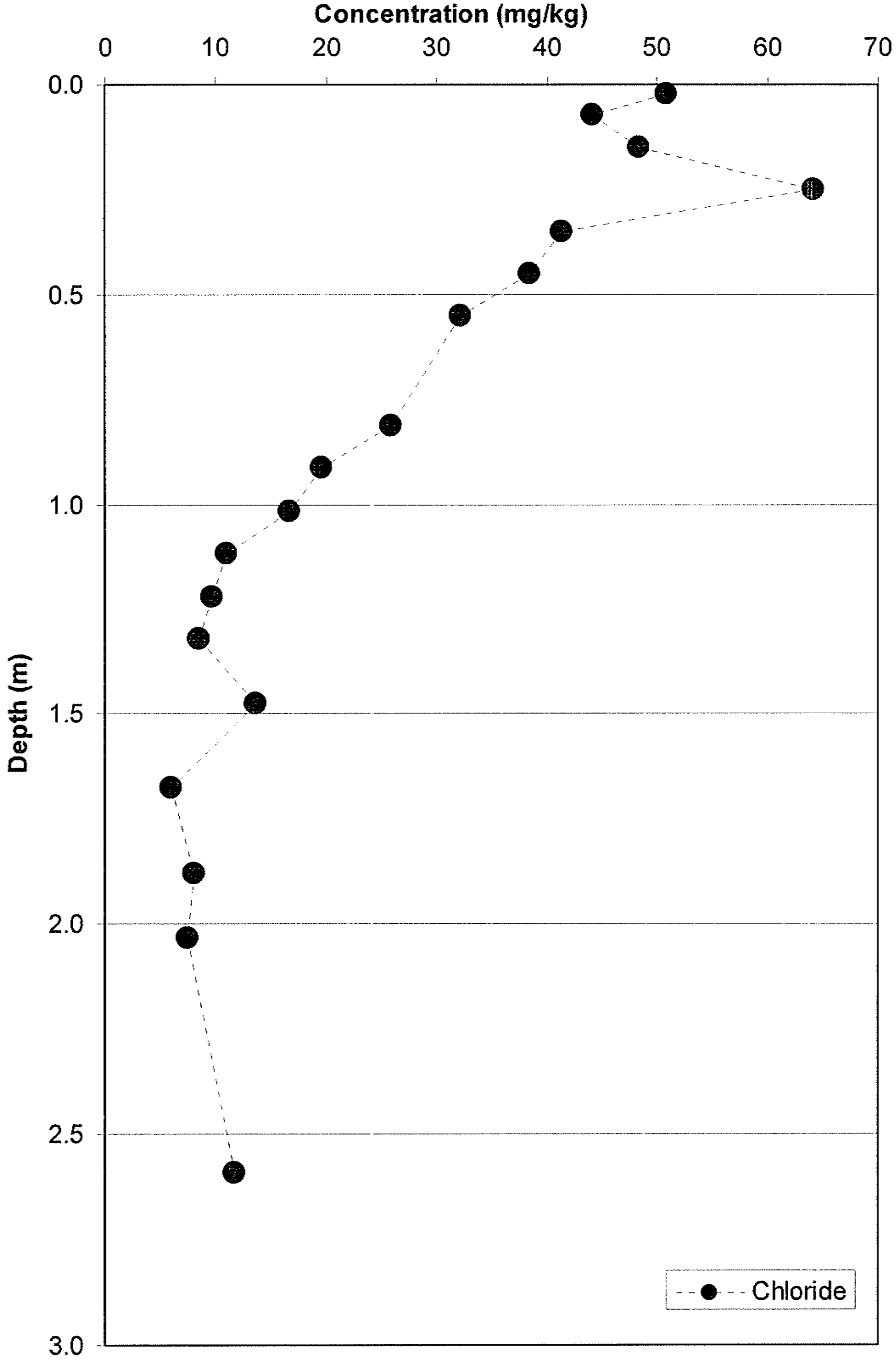


Figure 5.24: Soil quality profile (salts) for the SE2 3 year EMS sample core.

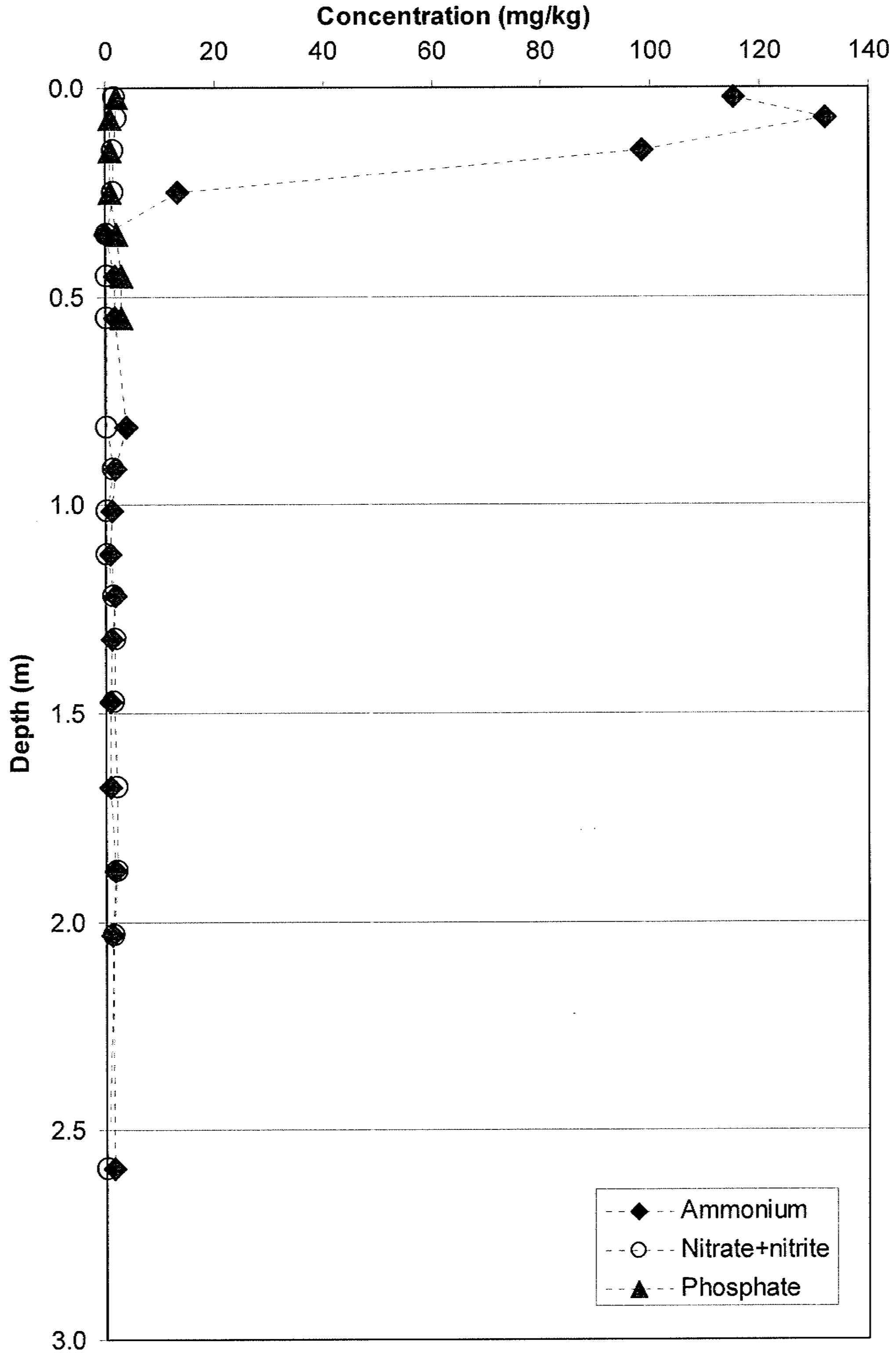


Figure 5.25: Soil quality profile (nutrients) for the SE2 3 year EMS sample core.

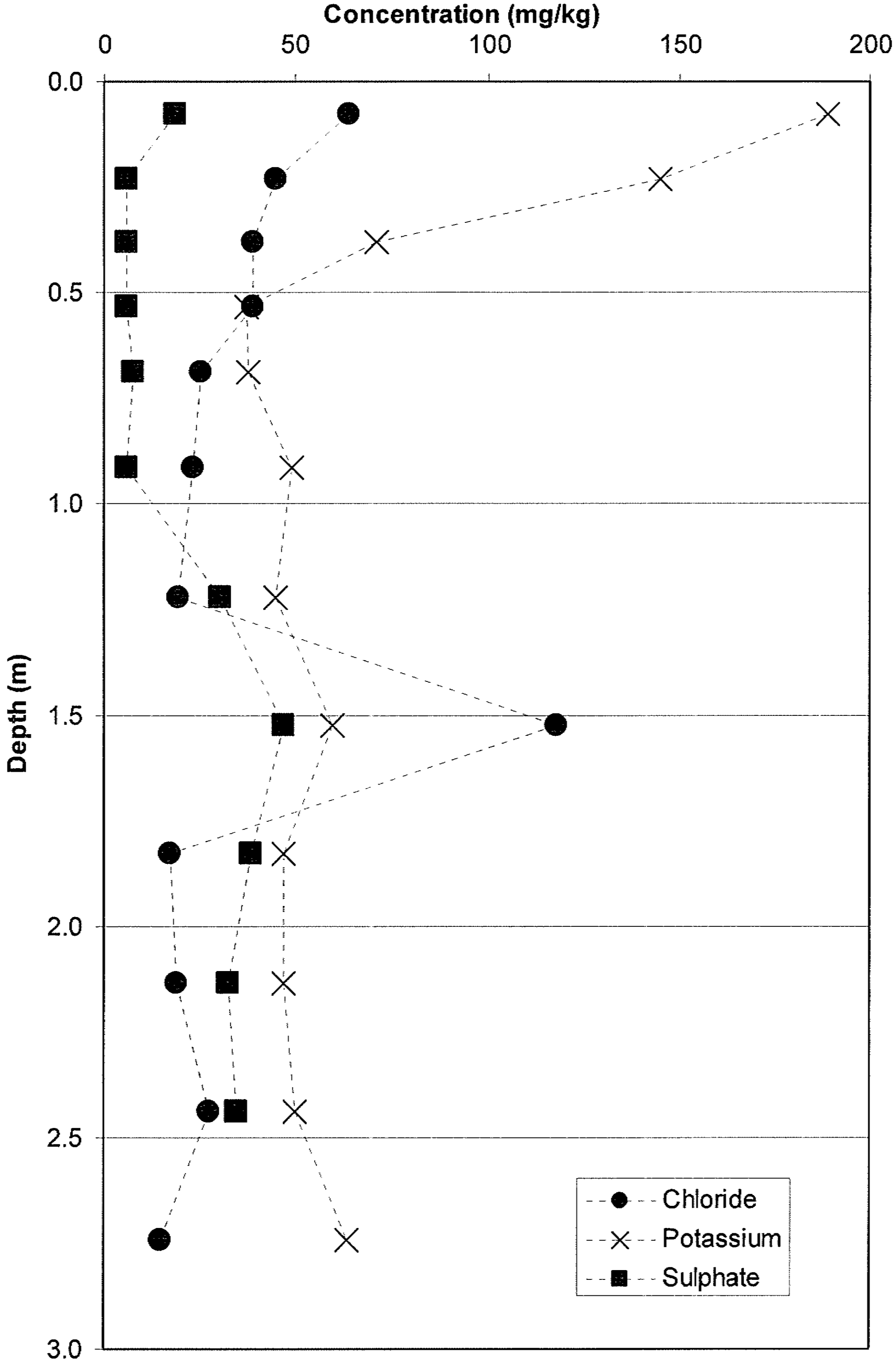


Figure 5.26: Soil quality profile (salts) for the SE2 6 year EMS sample core.

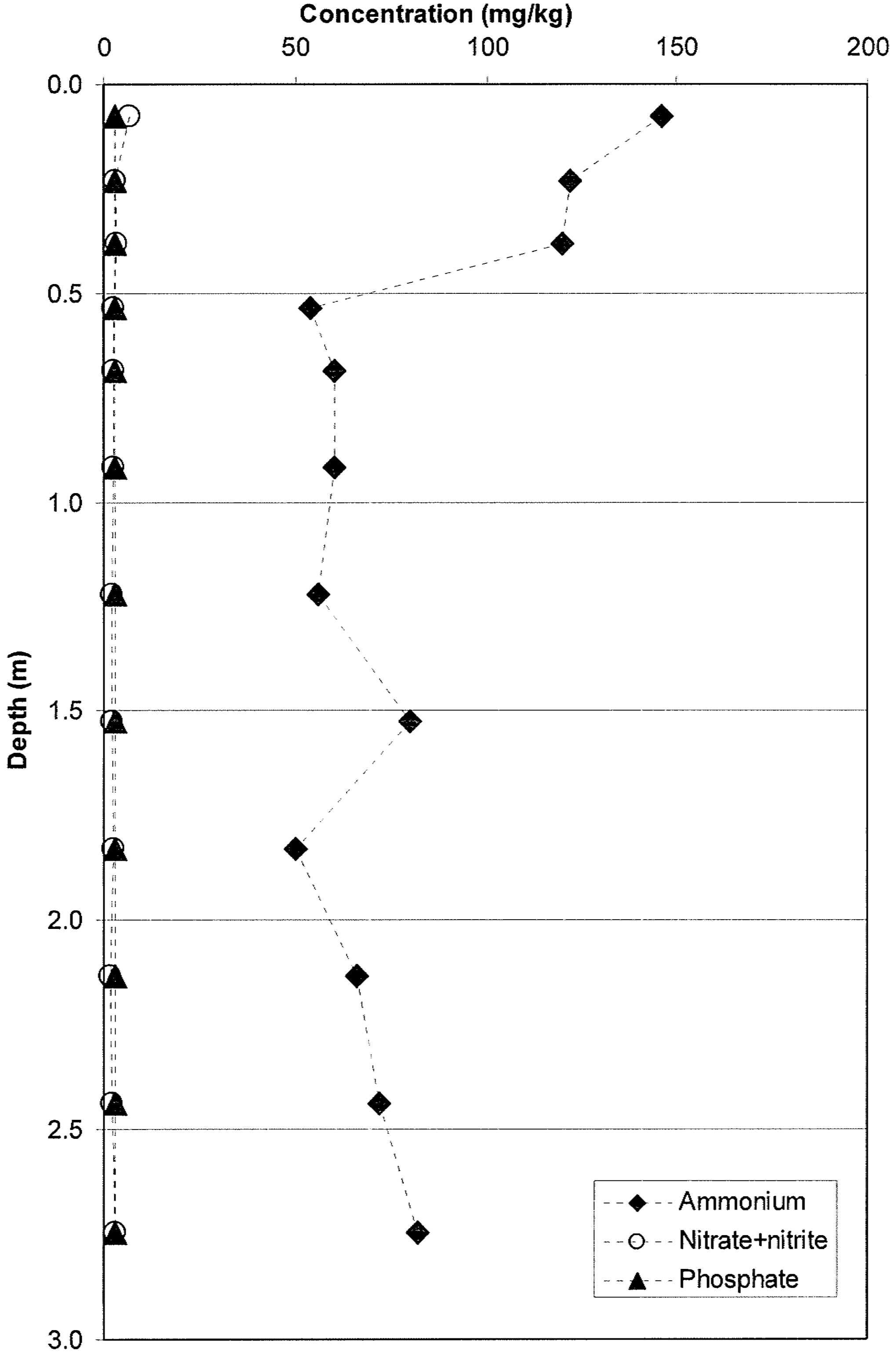


Figure 5.27: Soil quality profile (nutrients) for the SE2 6 year EMS sample core.

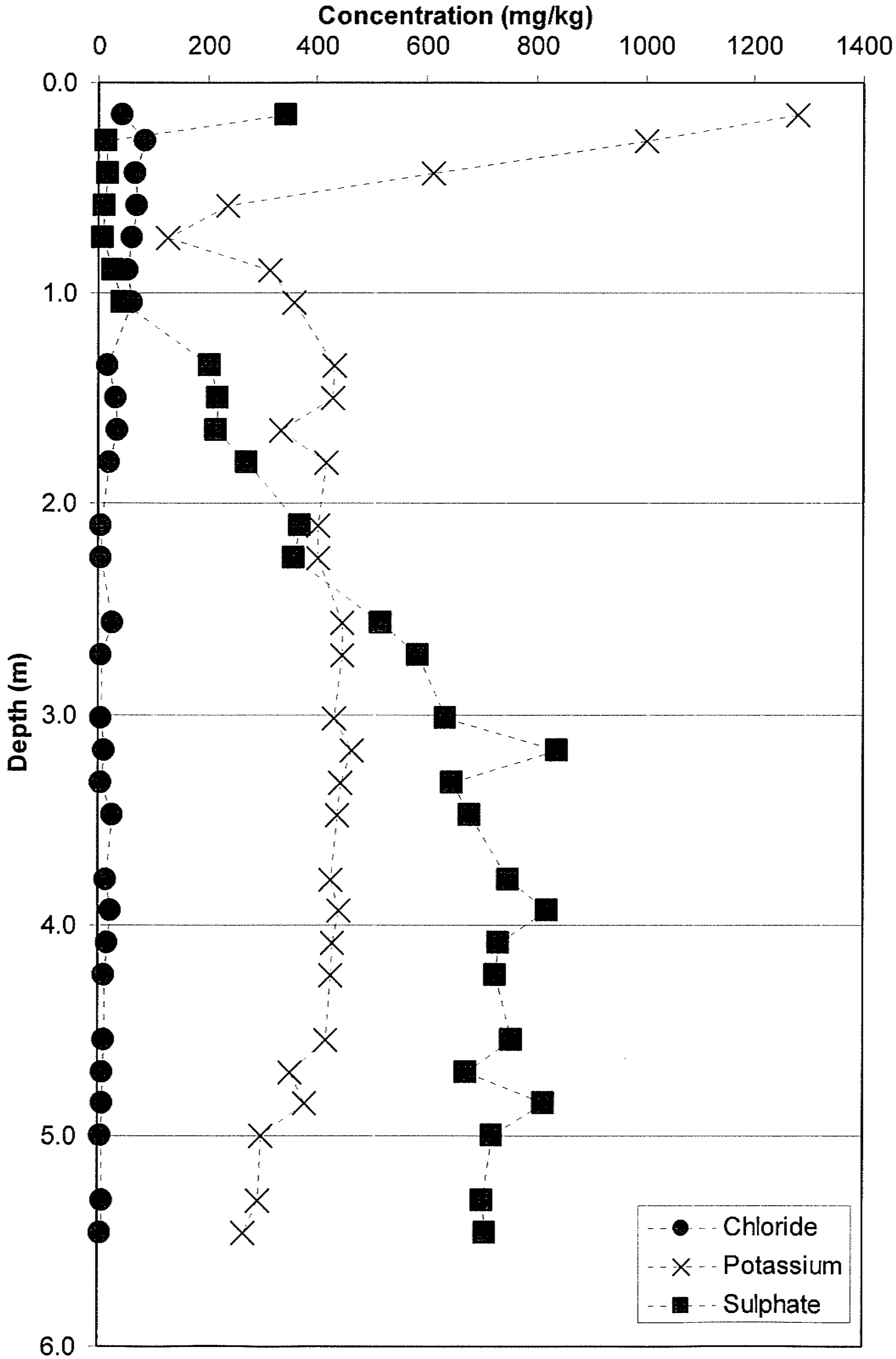


Figure 5.28: Soil quality profile (salts) for the SE3 14 year EMS sample core.

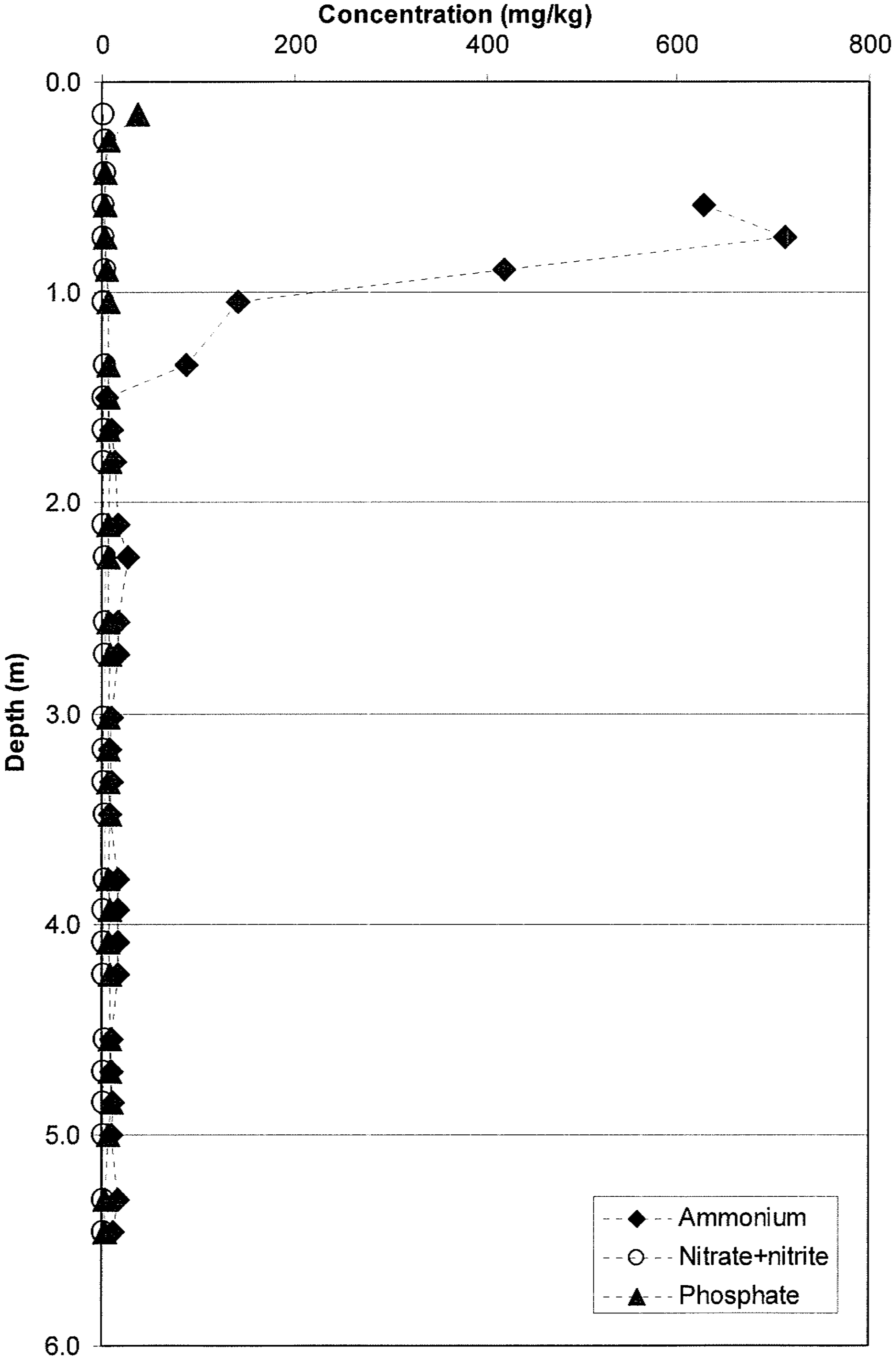


Figure 5.29: Soil quality profile (nutrients) for the SE3 14 year EMS sample core.

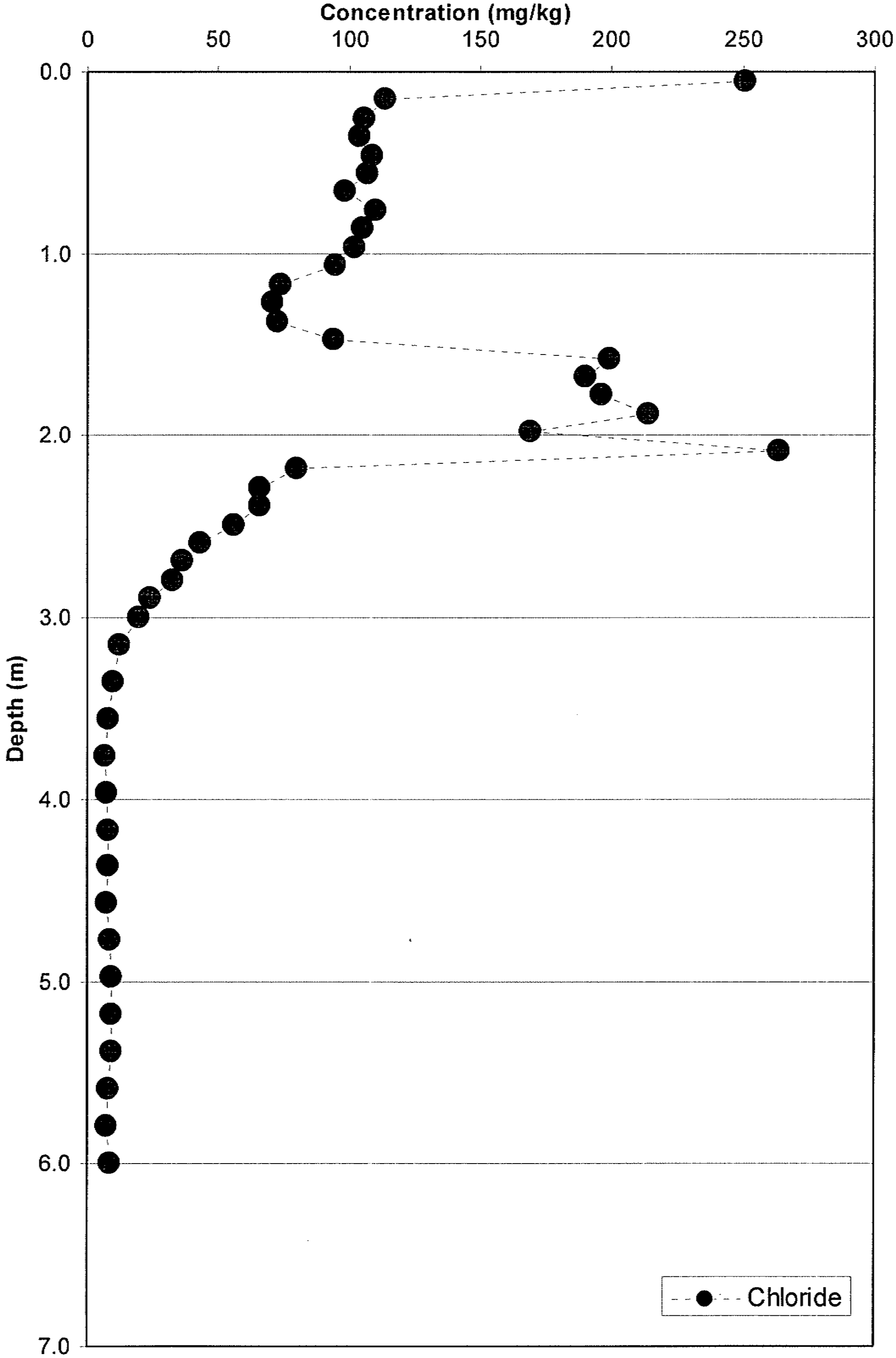


Figure 5.30: Soil quality profile (salts) for the SW1 18 year EMS sample core.

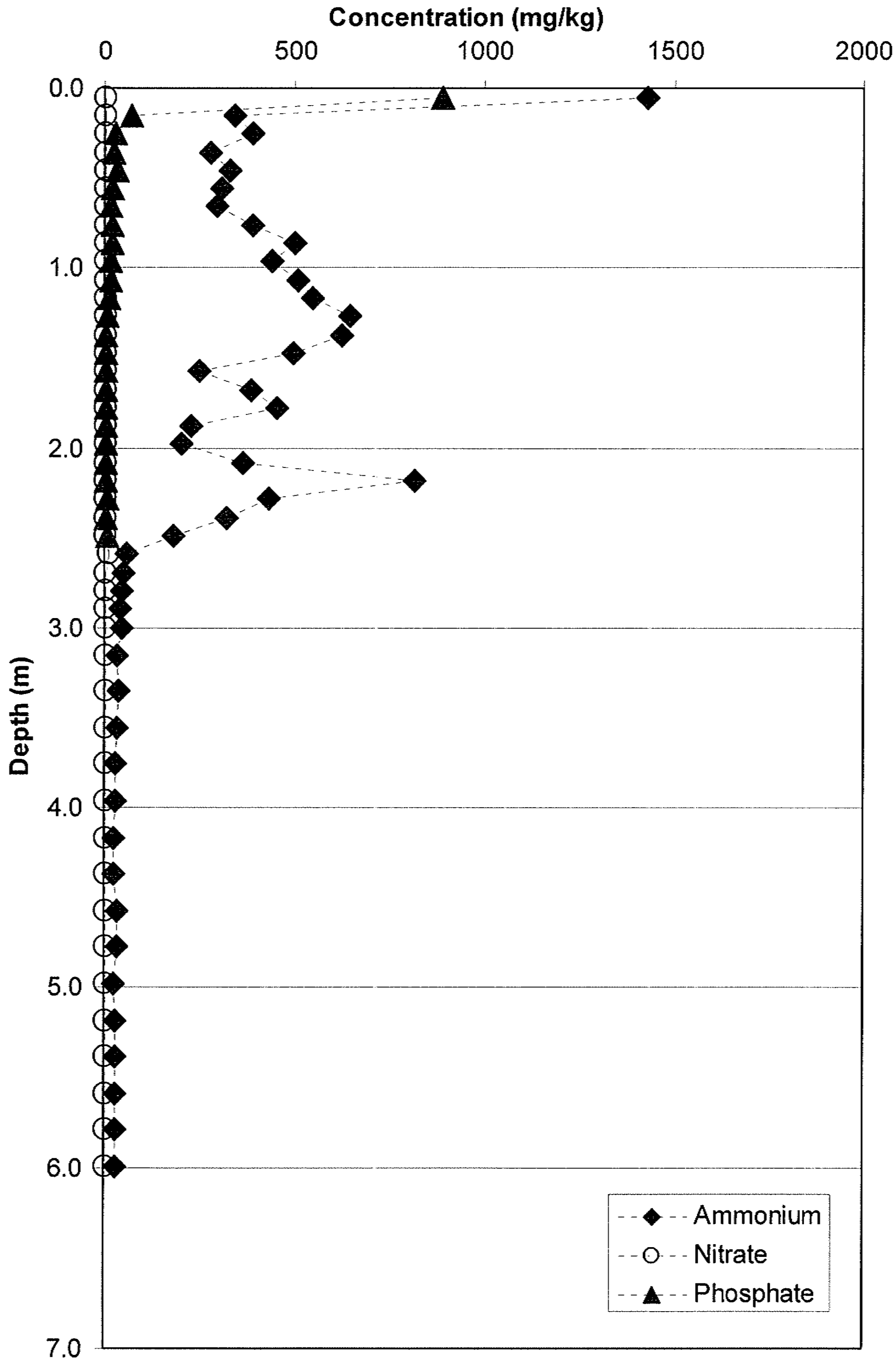


Figure 5.31: Soil quality profile (nutrients) for the SW1 18 year EMS sample core.

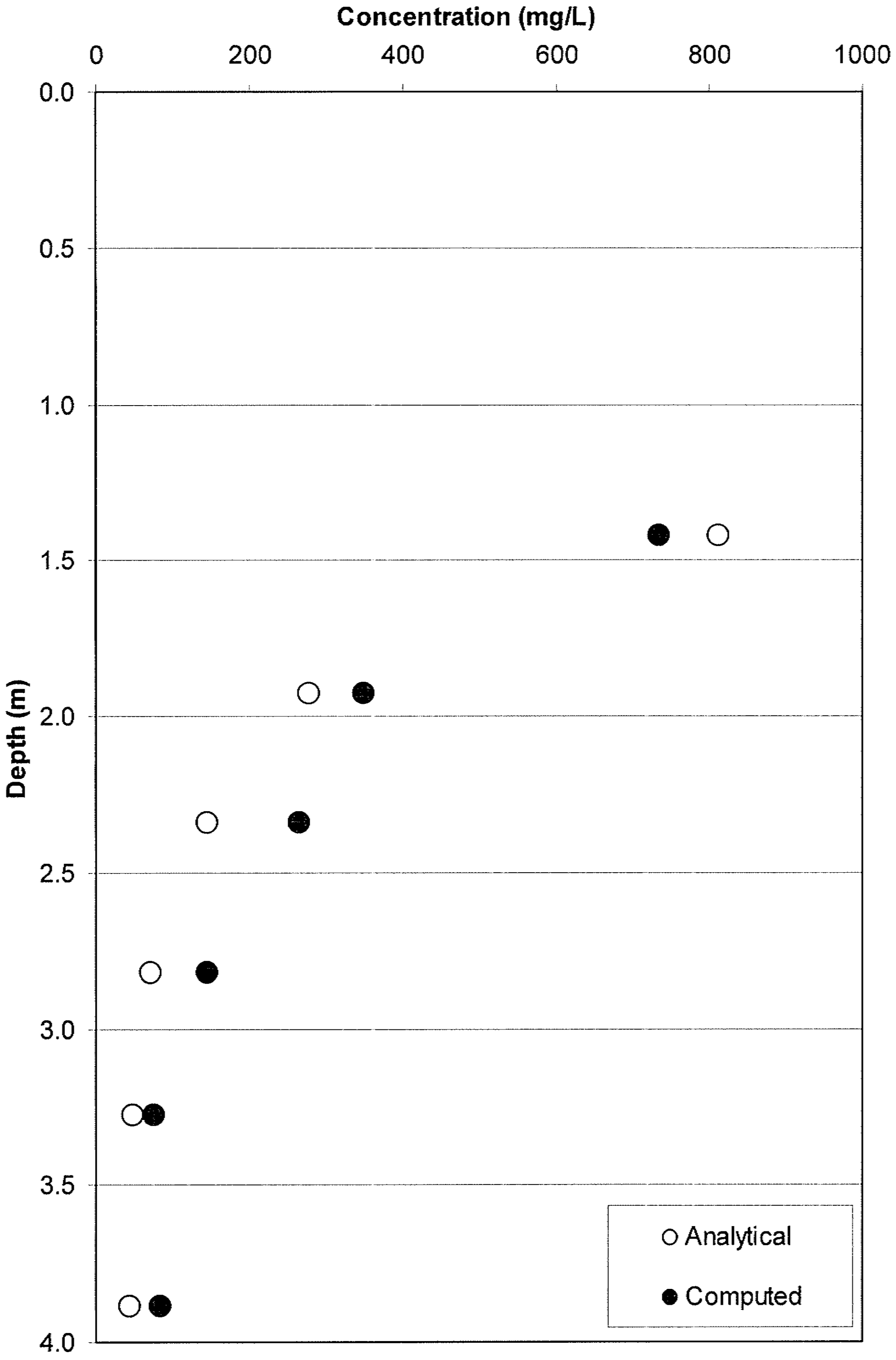


Figure 5.32: Analytical and computed pore water chloride concentrations.

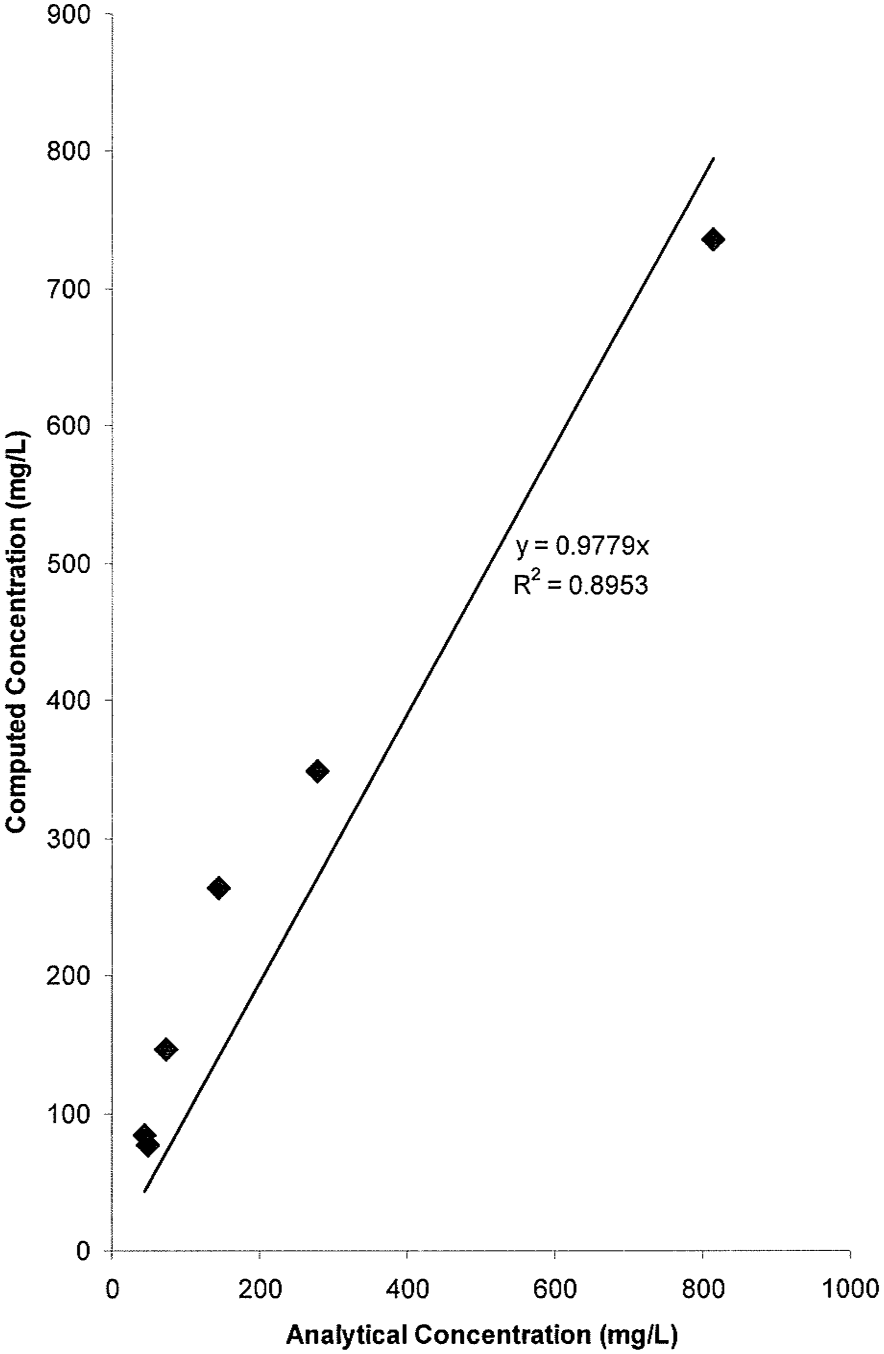


Figure 5.33: Scatter plot showing analytical vs. computed pore water concentrations.

CHAPTER 6: PARAMETER ESTIMATION

Using the measured conservative chloride concentration profiles and other measurements obtained from the field and laboratory analysis, estimates of parameters governing the migration of a conservative contaminant from the EMS facilities can be deduced through inversion using the Ogata (1970) analytical solution to the one-dimensional advection dispersion equation. These parameter estimates, developed from actual field cases will theoretically provide the most realistic estimate of these values at the field scale. The deduced parameter values can then be used to complete predictive modeling of EMS impact in various site settings and under various conditions.

6.1 OVERVIEW OF THE PARAMETER ESTIMATE TECHNIQUE

Parameters required for predictive modeling were estimated by fitting the one-dimensional advection dispersion equation to an effective chloride seepage profile. The effective chloride seepage profiles were constructed by subtracting average background chloride concentrations from total reported chloride concentrations for each sub-sample in a given EMS sample core. The Ogata (1970) solution of the one-dimensional advection dispersion equation, was fit to the observed chloride seepage profile to provide estimates of average linear velocity, longitudinal dispersivity and effective diffusion. These estimates are used in Chapter 7 to conduct predictive modeling of EMS impact for various site conditions.

6.2 CONSTRUCTING OBSERVED CHLORIDE SEEPAGE PROFILES

In order to obtain a seepage profile resulting from the EMS source alone, background chloride concentrations were subtracted from the measured chloride concentrations. This was done once the chloride concentration for each relevant sub-sample had been converted from mg/kg to mg/L using the measured moisture contents for the sub-sample. Equation 6.1 was used to estimate chloride concentrations in pore water:

$$[6.1] \quad A = \frac{B}{w} - E$$

where A is the estimate of chloride concentration resulting from seepage from the EMS alone (referred to in the text that follows as the "effective" chloride concentrations), B is the measured chloride content in the sub-sample, w is the gravimetric water content measured in the sub-sample and E is the average background chloride concentration calculated for a subject site.

It is important to note that the Ogata solution does not consider a variation in initial species concentration with depth, as was observed at each of EMS sites examined in this study. As a result, a single background chloride concentration for each EMS that is representative of the location was taken as the average chloride concentration along the background sample core or the lower portion of the EMS sample core.

At seven of the eleven sites examined, background cores were collected from a location proximal to the EMS facility, but laterally distant enough to be considered unaffected by impacts produced by the facility. The location of the background cores were also selected to minimize the chance of encountering impact associated with past fertilizer application or any other agricultural activities which may have occurred on the property. Background cores were collected from the NC1, NC2, NC3, SE2, SE3, SC2, and SC3 facilities. Background concentrations could also be reasonably estimated from the NC1(2), SE2(1), and SE3 EMS cores where a constant chloride concentration was reached and maintained at depth along the core.

In the case of the NC1(2), SE2(1), and SE3 sites, the chloride concentrations from the background core tended to exceed the background chloride concentration observed in the lower portion of the EMS cores examined. Within these lower portions apparent background concentrations have been reached. Figure 6.1(a) shows the effective and predicted seepage for the SE3 EMS sample core where the effective concentration was developed using an average background chloride concentration estimated from the background sample core analytical results. In this case, the average background chloride concentration developed from the background core of 90.9 mg/L appears to be an overestimate of the actual background chloride concentration in the sediments beneath the facility. This conclusion is based on the consistent negative chloride concentrations which are produced by Equation 6.1 below the maximum depth of chloride migration from the overlying EMS facility (just under 2.0 m below the base of the EMS). In Figure 6.1(b), the effective chloride profile was developed using an average background chloride concentration estimated over the depths of

2.1 m to 5.5 m of 27.9 mg/L. Using this estimate of average background chloride concentrations produces zero and near zero values below the maximum depth of chloride migration from the facility of 2.0 m, and hence a better estimate of the background chloride concentration at the location of the EMS sample core. Similar results were found for the NC1(2) and SE2(1) cases, where the background cores yielded an average background chloride concentration of 78.8 mg/L and 230.8 mg/L, respectively, and the lower 1.8 m to 3.6 m and 1.1 m to 2.6 m of the EMS cores (below any apparent impacts from the EMS facilities) yielded estimated average background chloride concentrations of 49.5 mg/L and 121.9 mg/L, respectively. The lower values estimated from the lower reaches of the EMS sample cores are considered a more reasonable estimate of background chloride concentration at the location of the EMS sample core.

6.3 DEVIATIONS FROM EXPECTED EFFECTIVE CHLORIDE SEEPAGE PROFILES

The chloride seepage profiles for the two SC2 EMS sample cores (collected after 20 years and 21 years after facility operation commenced) show deviations from profiles observed in EMS cores collected from the other sites examined. In particular, for the other sites examined, no distinct increasing trends with depth were noted in the background chloride concentrations. In the case of the SC2 site, background chloride concentrations, as measured in the background sample core, showed a distinct increasing trend with depth (Figure 6.2). This trend was not noted in the other chemical species analysed in the background core, and as a result, appears to be naturally occurring and not indicative of contamination beneath the site at the background sample core location. A number of studies have addressed the issue of high TDS groundwater in the carbonate bedrock aquifers in southern Manitoba, including Grasby and Betcher (2002). The higher than expected background chloride concentrations with depth at this site may potentially be attributed to saline groundwater in the underlying aquifer. Concentrations of chloride in the overburden pore water at the location of the background sample core would therefore be expected to approach a constant value at depth which would theoretically be consistent with the chloride concentration in the underlying aquifer.

At depths of 2 m below grade, corresponding with the base of the facility, the background chloride concentration (measured in the background sample core), once converted to a mg/L value, was on the order of 340 mg/L. Depths of 8 m and 9 m below grade in the background core correspond to the final sampling depth for the two EMS sample cores collected from the SC2 site (6.7 m). At these depths, background chloride concentrations were on the order of 900 mg/L and 1000 mg/L, respectively. A reasonably uniform increase in background chloride concentration with depth was observed at this site and depicted in Figure 6.2.

The background chloride concentrations at depths of 2 m and 8 m below grade (0 m and 6 m below the EMS base) exceed the chloride concentrations observed at these depths in the 20 year SC2 EMS sample core (Figure 6.3). This may be the result of a flushing effect caused by lower chloride concentrations in the infiltrating fluids originating from the EMS which are lower than those occurring naturally in the underlying pore water. This would cause a reduction in the background chloride concentrations beneath the EMS.

The second EMS core removed from the site (21 years after facility operation commenced) suggests higher input chloride concentrations which decline to a value somewhat less than the background concentrations expected at the final sampling depth of 7 m below the EMS base (approximately 9 m below grade; Figure 6.4). Based on these results, it is not possible to make any conclusions about the validity of the background core data, or the potential flushing effect which may be occurring with chloride beneath the SC2 EMS. Due to this uncertainty, parameter estimation using data for this site is not considered reasonable using the Ogata (1970) solution and as a result, was not conducted.

6.4 ONE-DIMENSIONAL TRANSPORT IN A POROUS MEDIUM

In one-dimension, classic advective and dispersive mass transport for a conservative species in a fully saturated homogeneous porous medium is described as follows (after Bear, 1972):

$$[6.2] \quad \frac{\partial C}{\partial t} = D_z \frac{\partial^2 C}{\partial z^2} - v_z \frac{\partial C}{\partial z}$$

where C is the concentration of the conservative chemical species originating from the source, D_z is the longitudinal hydrodynamic dispersion coefficient, z is the vertical depth below source, and v_z is the average linear groundwater velocity. Bear (1972) defines the coefficient of hydrodynamic dispersion in the longitudinal direction as:

$$[6.3] \quad D_z = \alpha_z v_z + \tau D^*$$

where α_z is the longitudinal dispersivity coefficient, and v_z is the average linear groundwater velocity in the z direction, τ is the tortuosity, and D^* is the coefficient of effective diffusion.

Ogata (1970) provides the following analytical solution to the governing partial differential equation given in equation [6.1], which is given as follows:

$$[6.4] \quad C(z,t) = \frac{1}{2}C_o\left(erfc\left[\frac{z - v_z t}{2(D_z t)^{1/2}}\right] + \exp\left[\frac{vz}{D_z}\right] erfc\left[\frac{z - v_z t}{2(D_z t)^{1/2}}\right]\right)$$

where t is time and C_o is the source concentration. Boundary conditions used to arrive at the solution presented in equation [6.3] are as follows:

$$[6.5] \quad C(z = 0, t > 0) = C_o(t)$$

$$[6.6] \quad C(z = \infty, t > 0) = 0$$

where the source concentration as a function of time, $C_o(t)$, is assumed to be constant. The initial condition imposed to derive the solution is expressed mathematically as follows:

[6.7] $C(z > 0, t = 0) = 0$

This initial condition necessitates the removal of background concentrations from the observed chloride seepage profile if a direct comparison is to be made.

Assumptions inherent in the approach include that of steady-state flow and transport in a fully saturated, homogeneous porous medium. Actual conditions at the EMS facilities would not be conducive to steady-state flow due to changing fluid levels in the EMS on a seasonal basis. However, the model uses an estimate of the greatest potential vertical gradients between the base of the EMS facility and the underlying aquifer. As a result, parameter values obtained using this method will provide only a conservative or worst-case estimate. Similarly, it is expected that concentrations of various chemical species within the lagoon will also fluctuate on a seasonal basis, including chloride, hence more variability in the concentration profile is expected.

The one-dimensional Ogata (1970) approach also assumes hydrodynamic dispersion is Fickian, with dispersion occurring in the longitudinal direction only. Given that the EMS seepage problem is three-dimensional, dispersion will occur in the transverse directions as well, causing a reduction in the chloride concentrations at a given point, compared to those produced by the one-dimensional model at the same point. As a result, the one-dimensional approach will be most appropriate near the centre of the facility, where horizontal gradients are the least, and will be less likely to reproduce actual conditions under the lagoon at a given time and point towards the edges of the facility.

The approach also assumes that the gradient distribution between the base of the EMS facility modeled and the underlying aquifer is linear, and that the potentiometric surface measured in the nearest water well represents the head at the top of the aquifer directly beneath the facility. Since no data was available to better estimate the head distribution beneath the EMS facilities modeled, this was considered to be the most reasonable approximation. In fact, we would expect a non-linear distribution of hydraulic gradient beneath the facility, with the greatest gradient near the base of the facility and reducing with depth below the facility.

6.5 PARAMETER RANGES USED FOR ESTIMATES

Maximum and minimum parameter constraints for average linear groundwater velocity, longitudinal dispersion and effective diffusion were utilized to obtain the best-fit parameter estimates that minimized the root mean square (RMS) error between the predicted chloride concentration profiles and the constructed chloride concentration profiles. Each constraint is discussed below and was selected based on literature reported or field values obtained.

Vertical hydraulic gradients used were estimated based on the potential range of head in the EMS facility, typically from 0 m to 2 m above the base of the facility, and the recorded head in the underlying aquifer, as estimated from the nearest water well record to the site (GWDrill Database) and any available potentiometric surface mapping available from the Province of Manitoba. The head distribution across the underlying overburden was assumed to be linear. The range of vertical hydraulic gradients therefore varied from site to site depending on the reported hydrogeological conditions.

Porosities were estimated based on a range of values set out in Freeze and Cherry (1979). Porosities for clay typically range from 0.4 to 0.7, porosity for silt and silt-rich materials range from 0.35 to 0.5 and porosity for sand range from 0.25 to 0.5. For sites underlain by more than one material type, the selected range in porosity must encompass the range in porosity for each of the materials examined, and porosities used in the model must reflect effective values. Since the assumption was made that the underlying sediments were fully saturated, then it is considered reasonable to assume that the porosity of the underlying sediments is reflected by the volumetric water content of the soil. Based on this argument, for the sediments encountered in this work, the volumetric water contents, and hence the effective porosities, ranged from 0.1 to 0.3, with the lower values associated with sand and silt-rich materials and the upper value associated with clay-rich materials.

Ranges of hydraulic conductivities at the sites examined were based on literature published values and from the results of *in-situ* hydraulic conductivity testing conducted as part of this research. For clay overburden, hydraulic conductivities were estimated to range over 1×10^{-9} m/s to 1×10^{-11} m/s and for till overburden, hydraulic conductivities were estimated to range from 1×10^{-7} m/s to 1×10^{-9} m/s (Pach, 1994). These values are also consistent with those found from recovery tests conducted as part of this work, at sites where monitoring wells were installed (see the results provided in Chapter 5). The recovery testing results indicated bulk conductivities in the till overburden ranging from

7.8×10^{-6} m/s to 5.7×10^{-9} m/s and in the clay overburden ranging from 3.0×10^{-8} m/s to 1.8×10^{-9} m/s.

Longitudinal dispersion is typically considered to be a function of scale and is subject to considerable uncertainty. For the purposes of this work, longitudinal dispersion was limited to values of 0.01 m to 0.1 m, corresponding to a scale of between approximately 1 m and 10 m (Gelhar *et al.*, 1992).

The coefficient of effective diffusion is dependent on the properties of a given solute and the porous medium. For chloride and other major ions in groundwater, the coefficient of molecular diffusion has an approximate range from 0.03 m^2/year to 0.06 m^2/year at 25 degrees Celsius and a range of approximately 0.015 m^2/year to 0.03 m^2/year at 5 degrees Celsius (Freeze and Cherry, 1979). Assuming a reduction in the overall range of molecular diffusion coefficient of 0.5 due to the effect of the porous medium, then the range of effective diffusion coefficients would be on the order of 0.008 m^2/year to 0.03 m^2/year. For the purposes of this work, the coefficient of effective diffusion was limited to values lying between 0.03 m^2/year and 0.003 m^2/year.

6.6 PARAMETER ESTIMATION METHOD

Model calibration involves the variation of uncertain parameters, within expected or typical ranges, until a fit is obtained which meets a predetermined calibration criterion (Spitz and Moreno, 1996). The definition of a calibration criterion ensures that the calibration process is subjective. Typically, the criterion used to define a successful calibration between observed and model-predicted results is that the RMS does not exceed 10% of the difference between the maximum and minimum concentrations observed.

RMS is computed as follows:

$$[6.8] \quad RMS = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(C_{i,observed} - C_{i,predicted})^2}$$

where n is the number of data points, $C_{i,observed}$ is the observed concentration at point i and $C_{i,predicted}$ is the predicted concentration at point i.

The significant scatter in the observed data for this work necessitated that the typical calibration criterion be relaxed and fits to be evaluated on both a quantitative and qualitative basis. Calibration was defined as the model solution which qualitatively appeared to best-reproduce the observed profiles and generated the minimum RMS. In order to do this, results were obtained by using Microsoft EXCEL's built in solver function which uses the linear simplex method to find the optimum solution which produces a minimum RMS value between the model-predicted and observed profiles. Constraints on the solution are those indicated in Section 6.3.

6.7 PARAMETER ESTIMATION RESULTS

The results of the parameter estimation are provided in Table 6.1 and on Figures 6.5 to 6.15 for each of the modeled study sites. In general, average linear groundwater velocities were found to range from 0.05 m/year to 0.10 m/year in the underlying clays and from 0.1 m/year to 0.27 m/year in the underlying tills. Longitudinal dispersion was found to be on the order of 0.01 m to 0.04 m in the clay and 0.02 m to 0.1 m in the till. Coefficients of effective diffusion were estimated from range from 0.003 m^2/year to 0.03 m^2/year.

The parameter estimation results show that while average linear groundwater velocity was typically within the predefined minimum and maximum limits (0 m/year to 1 m/year), longitudinal dispersion and effective coefficient of diffusion are often at the limits of the range, indicating that the predefined ranges set for these two parameters may be to narrow. In order to evaluate to the impact of the constraints on the solution, the models were re-run for the NC1(2), NC2, NC3, SC1, SC3(2), SE1, SE2(1), SE2(2) and SE3 profiles. Each of these profiles had an estimated longitudinal dispersion value and/or an effective diffusion coefficient value which was at the upper or lower constraint values set. The constraints for these two parameters were expanded to range between 0.001 m and 1 m and 0.0003 m^2/year and 0.3 m^2/year for longitudinal dispersion and the effective diffusion coefficient, respectively.

In general, the fits between predicted and observed chloride concentrations show qualitatively good agreement, and although scatter in the observed data is not reproduced by the simple method used to make the parameter estimations, the general trends and magnitudes with depth are adequately reproduced. The scatter in the data is likely the result of variable source loading during normal operation of the EMS facilities, including changing quantities of stored wastewater and variable chloride concentrations within the source. In addition, heterogeneities within the material may also cause some of the observed scatter.

The parameter estimation results using the expanded constraint values are provided in Table 6.2. In the case of the NC1(2), NC2, NC3, SC1, SC3(2), SE2(1) and SE3 profiles, expansion of the constraint range shows the a more loosely constrained problem produces similar results, with the new estimates of longitudinal dispersion and effective diffusion coefficient being just outside of the former pre-defined range. This would suggest that the range, as defined previously, does provide reasonable constraints on the problem. The exception would be the results for the SE1 and SE2(2) profiles, which show parameters which are still tending towards the values of the problem constraints. Based on this result, it would appear that for these two profiles, there is too much scatter in the observed data to obtain a reasonable solution to the problem using the chosen method.

The sensitivity of the solution to changes in parameter values was examined for the SE3 site. Table 6.3 shows the percent change in the correlation coefficient resulting from a change in input average linear groundwater velocity, hydrodynamic dispersion, and effective diffusion coefficient values. The sensitivity of each parameter was examined by changing the given parameter and holding the remaining parameters constant at their respective "best fit" values. This was completed for each of the uncertain parameters. Based on the analysis for the SE3 profiles, changes in average linear groundwater velocity have the greatest impact on the correlation coefficient between the predicted and observed chloride concentration profiles. Changes in effective diffusion have the least impact on the fit between predicted and observed chloride concentrations. The sensitivity analysis suggests that chloride migration is an advection-dominated process.

In general, the results of the sensitivity analysis support the importance of proper site characterization, in particular the importance of having a good understanding of the bulk hydraulic conductivity, porosity and hydraulic gradient when making predictions related to contaminant transport times and distances.

Table 6.1: Summary of Parameter Estimation Results

Parameter	Site (Core)										
	NC1(1)	**NC1(2)**	**NC2**	**NC3**	**SC1**	**SC3(1)**	**SC3(2)**	**SE1**	**SE2(1)**	**SE2(2)**	**SE3**
T (y)	4	7	19	29	15	20	20	9	3	6	14
Co (mg/L)	285	285	600	1075	830	900	1300	1100	350	350	300
v (m/y)	0.2700	0.1025	0.1243	0.1611	0.3204	0.0823	0.0520	0.2361	0.2700	0.2541	0.0744
α (m)	0.0185	0.1000	0.0100	0.1000	0.1000	0.0398	0.0135	0.1000	0.0199	0.1000	0.0276
D^* (m^2/y)	0.0047	0.0300	0.0202	0.0300	0.0300	0.0049	0.0030	0.0300	0.0030	0.0300	0.0030
RMS	74.3	49.6	96.4	293.7	129.1	126.7	127.5	150.7	62.6	156.0	37.2
$10\%(\Delta C_{obs})$	41.3	41.6	111.9	122.3	59.2	100.7	162.8	56.8	54.2	70.7	40.9
R^2	0.5886	0.7878	0.5194	-0.0201	0.0978	0.6968	0.9068	-0.0039	0.8689	0.2560	0.9371
y =	1.0412x	1.0244x	1.0178x	0.8459x	0.9742x	0.9671x	1.0019x	0.9615x	1.0294x	1.0139x	1.1382x

Notes:

$10\%(\Delta C_{obs})$ indicates 10 percent of the maximum measured change in concentration observed in the site sample core.

Table 6.2: Summary of Parameter Estimation Results (Expanded Constraints)

	Site (Core)								
Parameter	**NC1(2)**	**NC2**	**NC3**	**SC1**	**SC3(2)**	**SE1**	**SE2(1)**	**SE2(2)**	**SE3**
T (y)	7	19	29	15	20	9	3	6	14
Co (mg/L)	285	600	1075	830	1300	1100	350	350	300
v (m/y)	0.1007	0.1243	0.1620	0.3204	0.0520	0.2623	0.2700	0.2541	0.0744
α (m)	0.1027	0.0100	0.0999	0.0999	0.0131	1.0000	0.0199	0.0010	0.0215
D* (m^2/y)	0.0339	0.0202	0.0303	0.0299	0.0026	0.3000	0.0030	0.1325	0.0027
RMS	49.6	96.4	293.7	129.1	127.3	146.6	62.6	152.9	37.1
10%(ΔC_{obs})	41.6	111.9	122.3	59.2	162.8	56.8	54.2	70.7	40.9
R^2	0.7879	0.5194	-0.0131	0.0978	0.9073	-0.0048	0.8689	0.2847	0.9364
y =	1.0206x	1.0178x	0.8447x	0.9742x	1.0002x	0.9830x	1.0294x	1.0104x	1.1332x

Table 6.3: Results of Sensitivity Analysis

% Change in Input Parameter	Input Parameter		
	v (m/y)	α_L (m)	D*
-50%	-20.570 %	-0.130 %	-0.330 %
-25%	-4.620 %	-0.040 %	-0.040 %
25%	-3.850 %	0.001 %	0.001 %
50%	-12.880 %	0.000 %	-0.030 %

Notes:

Values on table are % change in R^2 from best-fit result for the SE3 site ($R^2 = 0.9371$);

Best-fit input parameters for the SE3 site were: v = 0.0744 m/year; α = 0.0276 m/year and D* = 0.003 m/year.

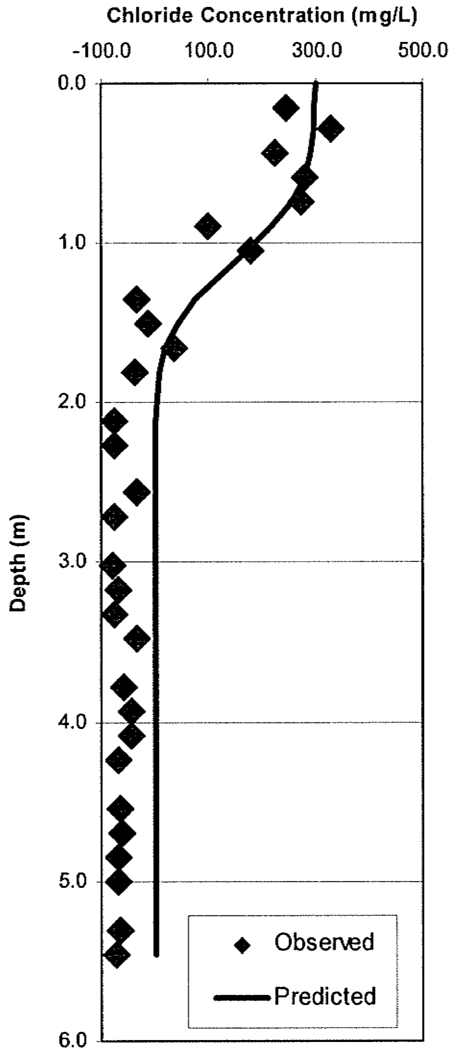


Figure 6.1(a): Observed vs. predicted seepage for the SE3 EMS sample core.
Case 1: Background concentration estimated from background sample core (90.9 mg/L average chloride concentration).

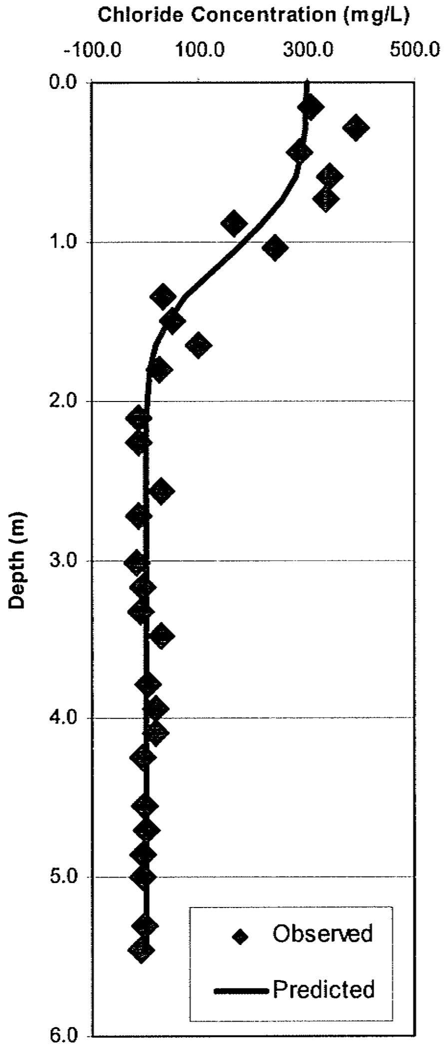


Figure 6.1(b): Observed vs. predicted seepage for the SE3 EMS sample core.
Case 2: Background concentration estimated from lower 2.1 m to 5.5 m of the EMS sample core (27.9 mg/L apparent background concentration).

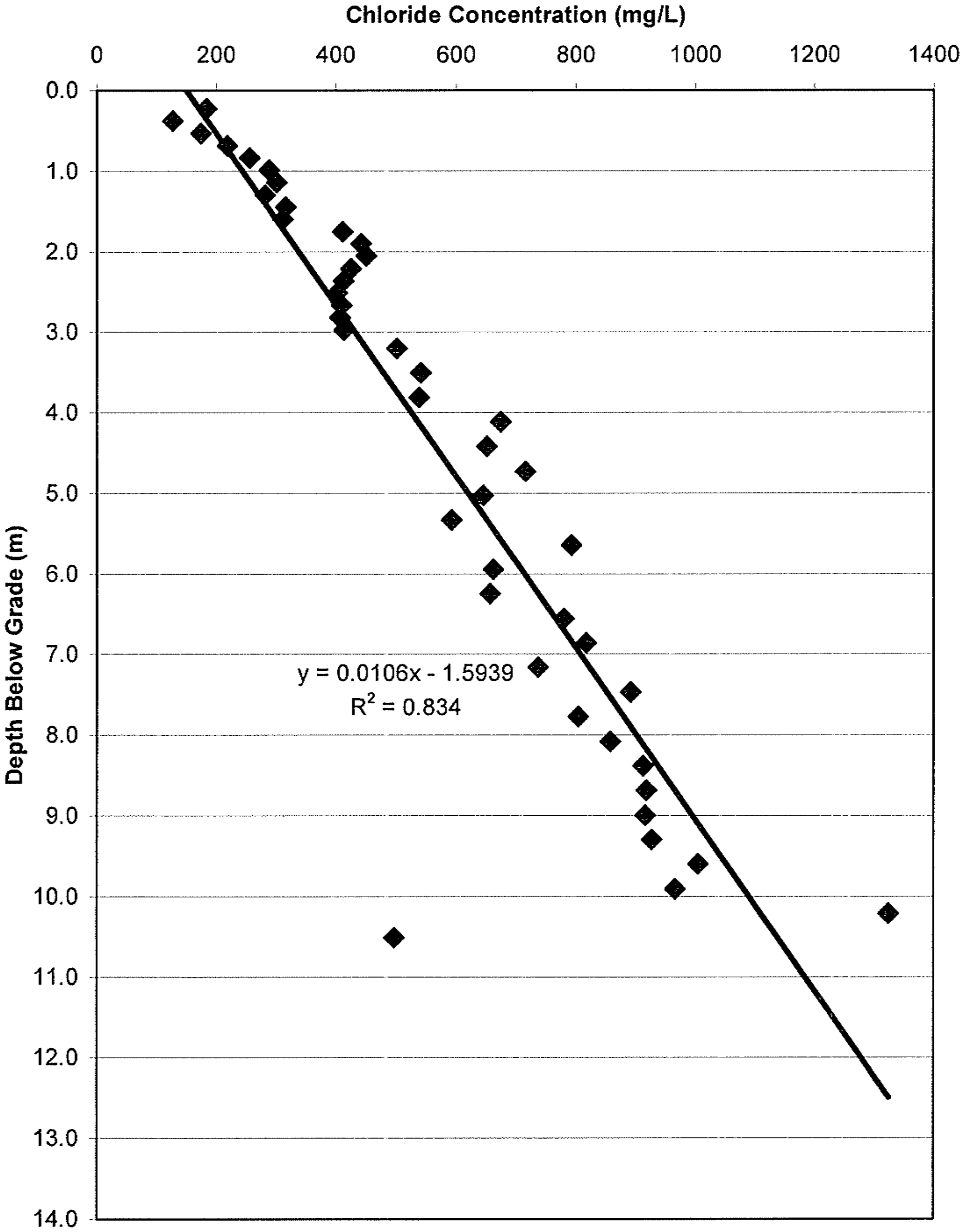


Figure 6.2: Background chloride concentration profile for the SC2 site. The trend of increasing chloride concentrations was observed for both the mg/kg and converted mg/L chloride concentration profiles.

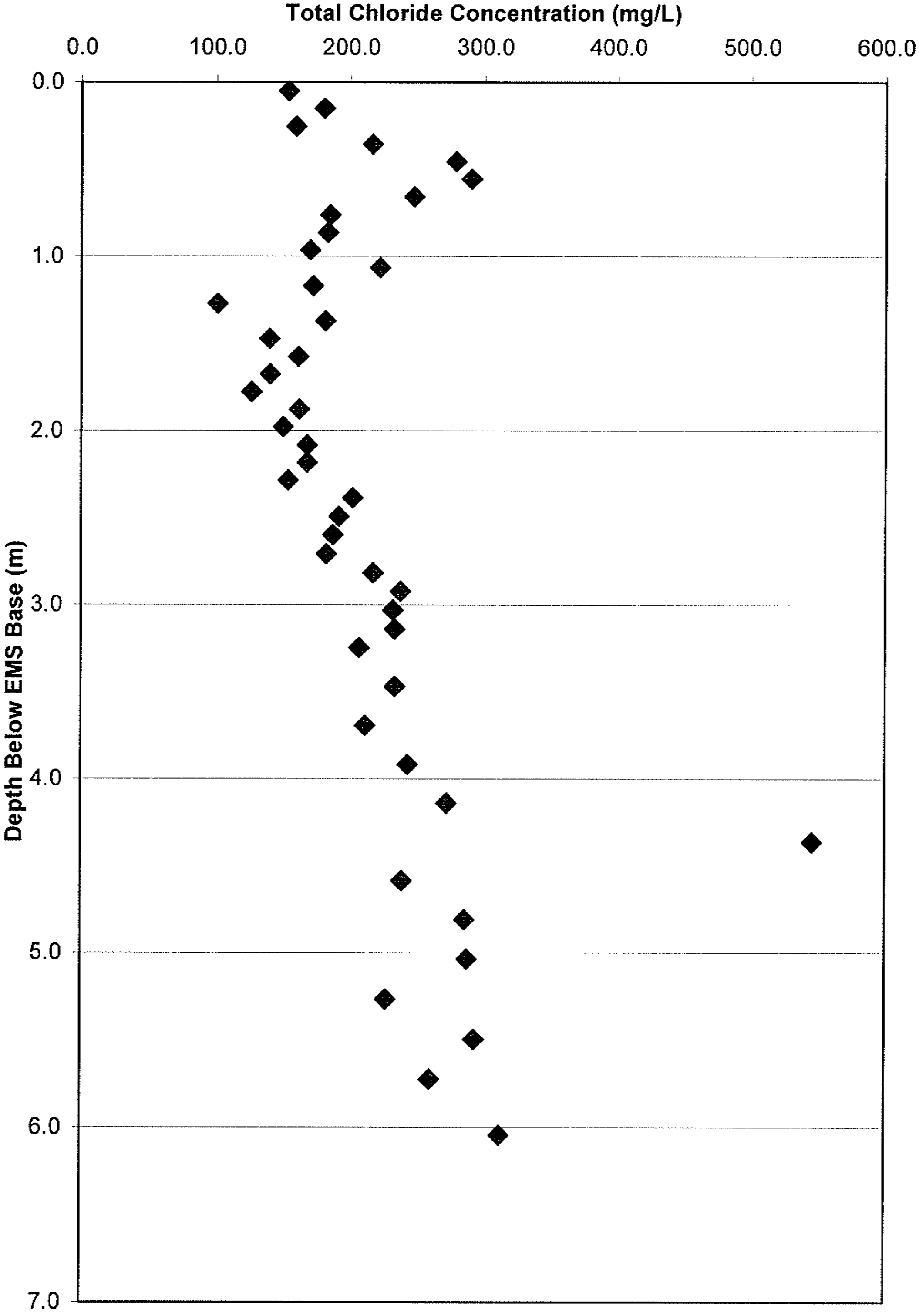


Figure 6.3: Total chloride concentration profile from the 20 year SC2 EMS sample core.

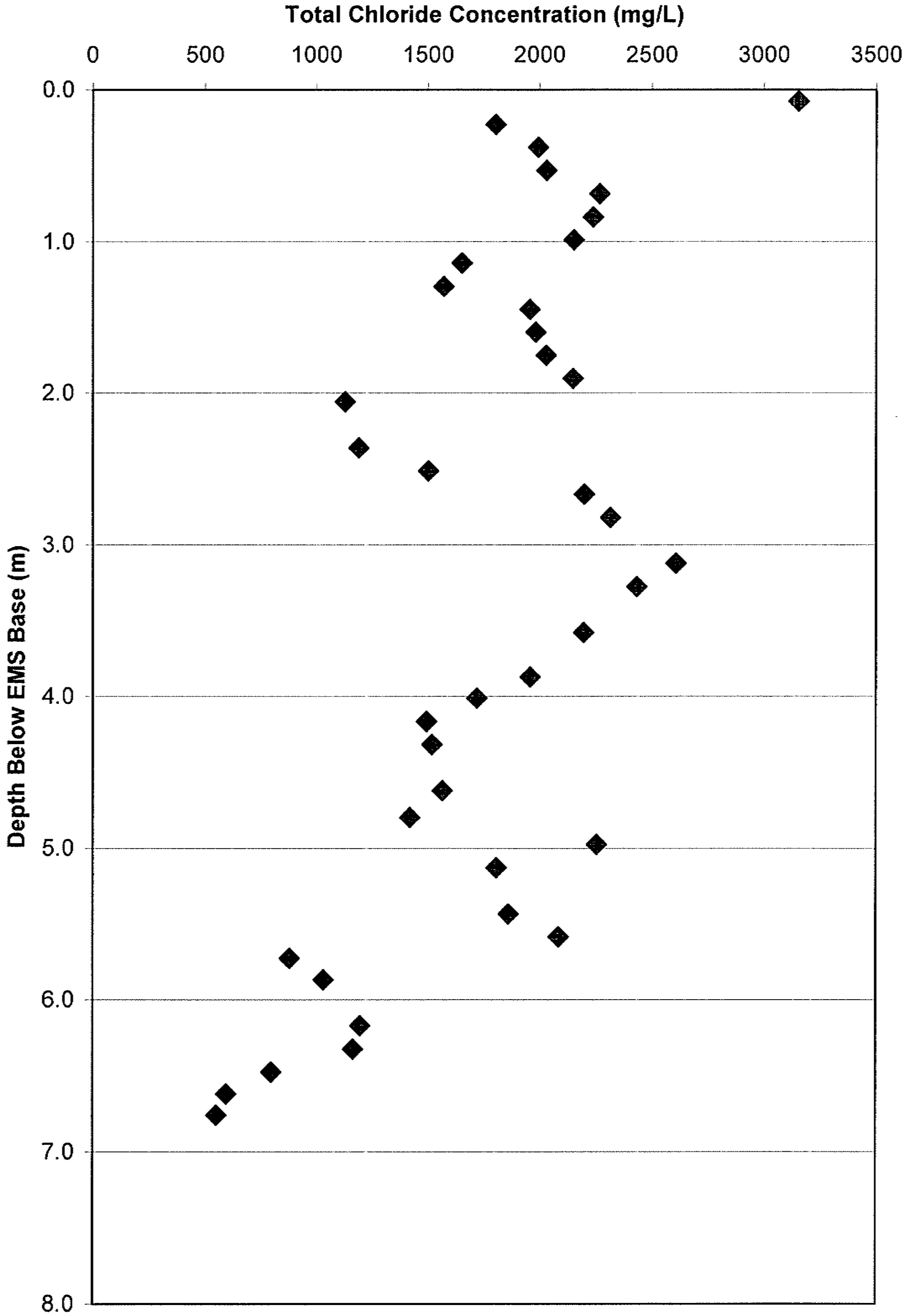


Figure 6.4: Total chloride concentration profile from the 21 year SC2 EMS sample core.

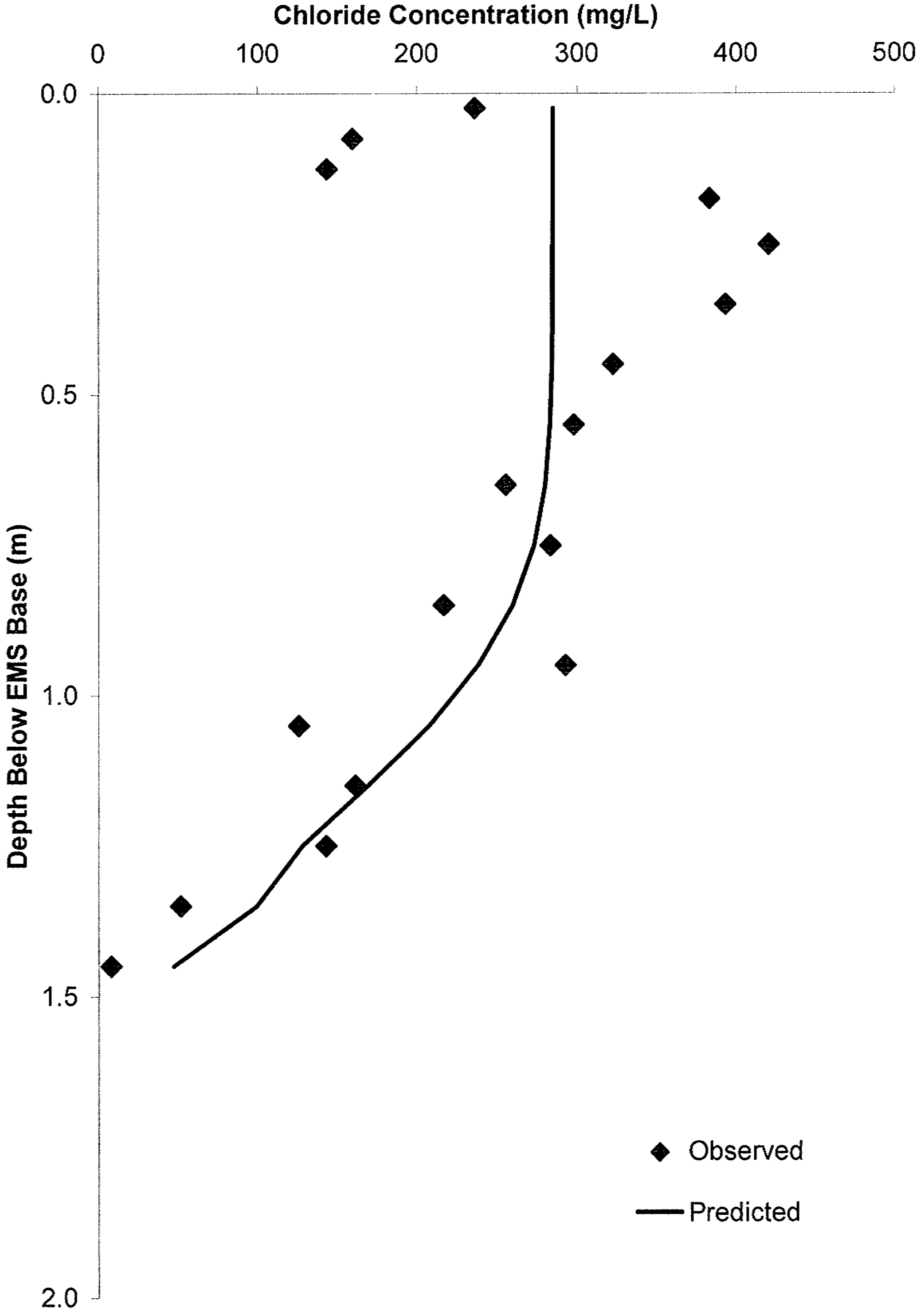


Figure 6.5: Observed vs. predicted seepage for the NC1(1) EMS sample core.

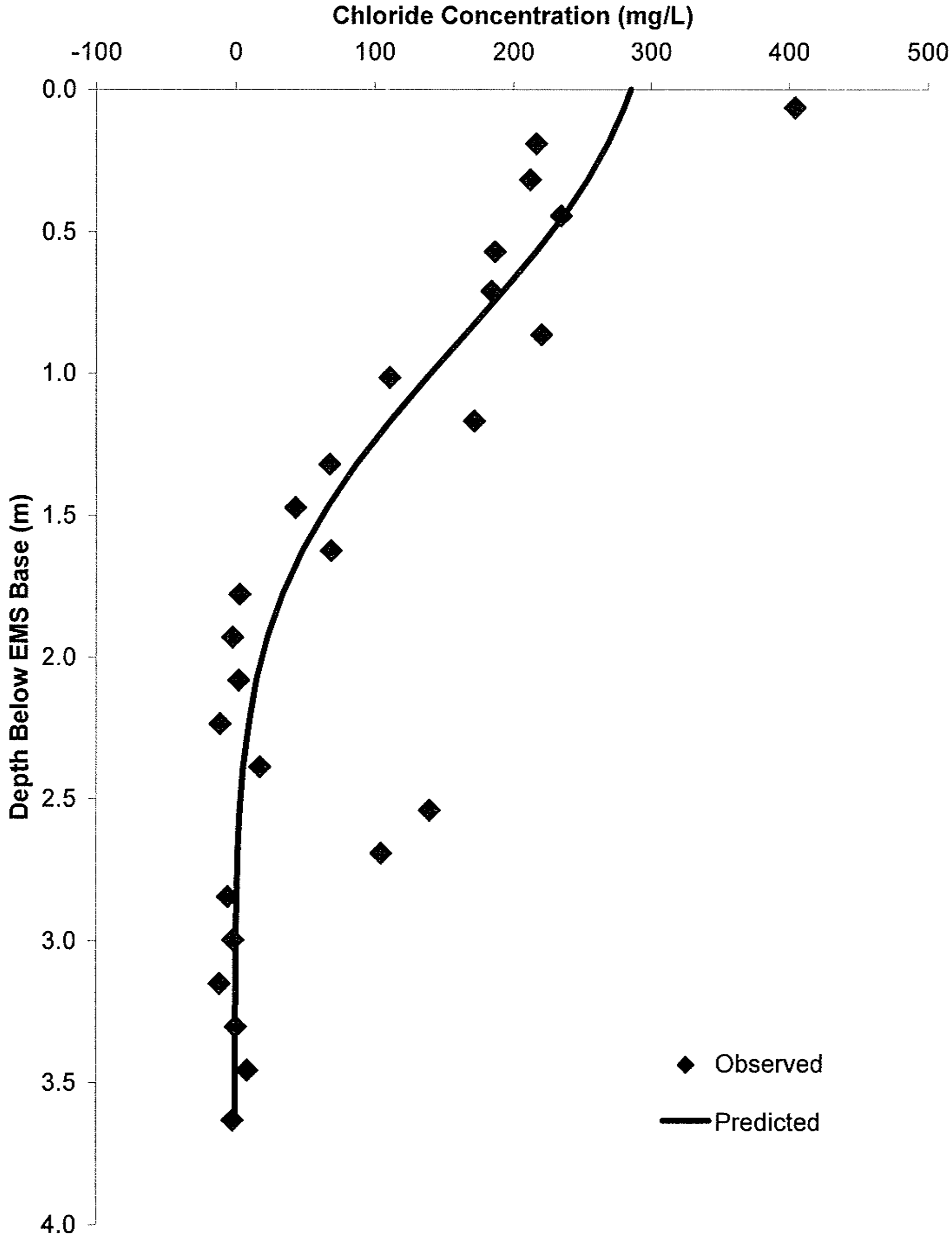


Figure 6.6: Observed vs. predicted seepage for the NC1(2) EMS sample core.

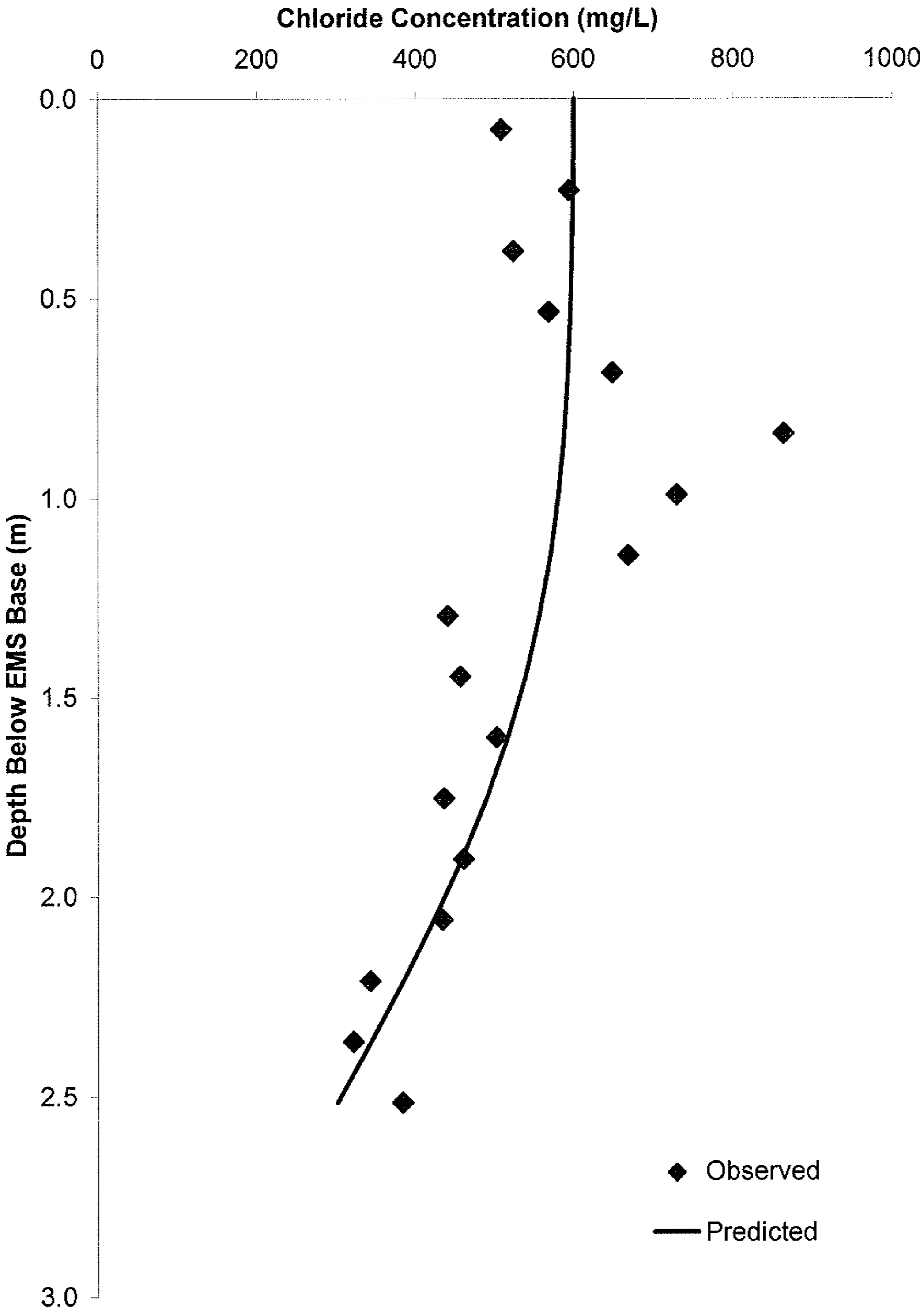


Figure 6.7: Observed vs. predicted seepage for the NC2 EMS sample core.

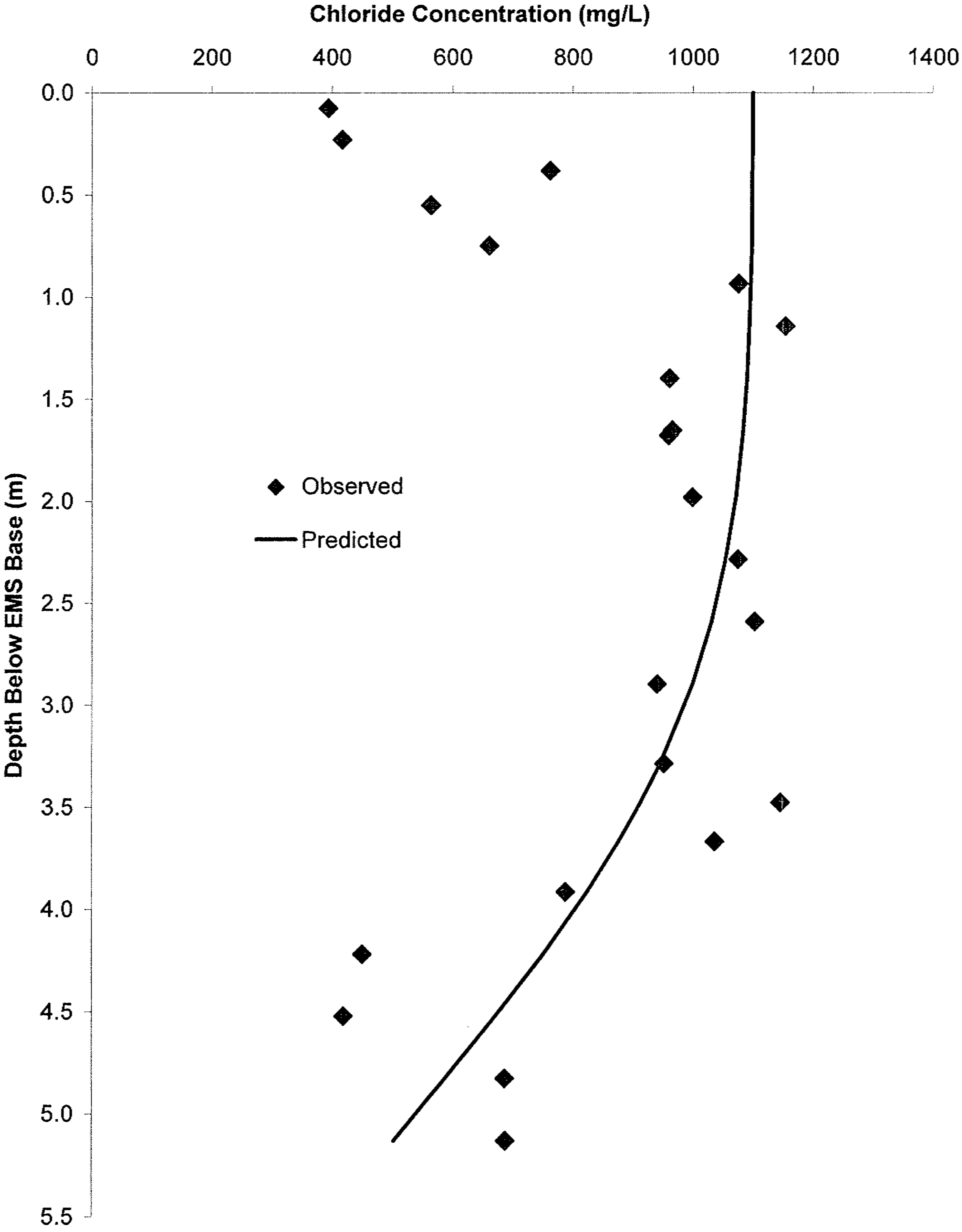


Figure 6.8: Observed vs. predicted seepage for the NC3 EMS sample core.

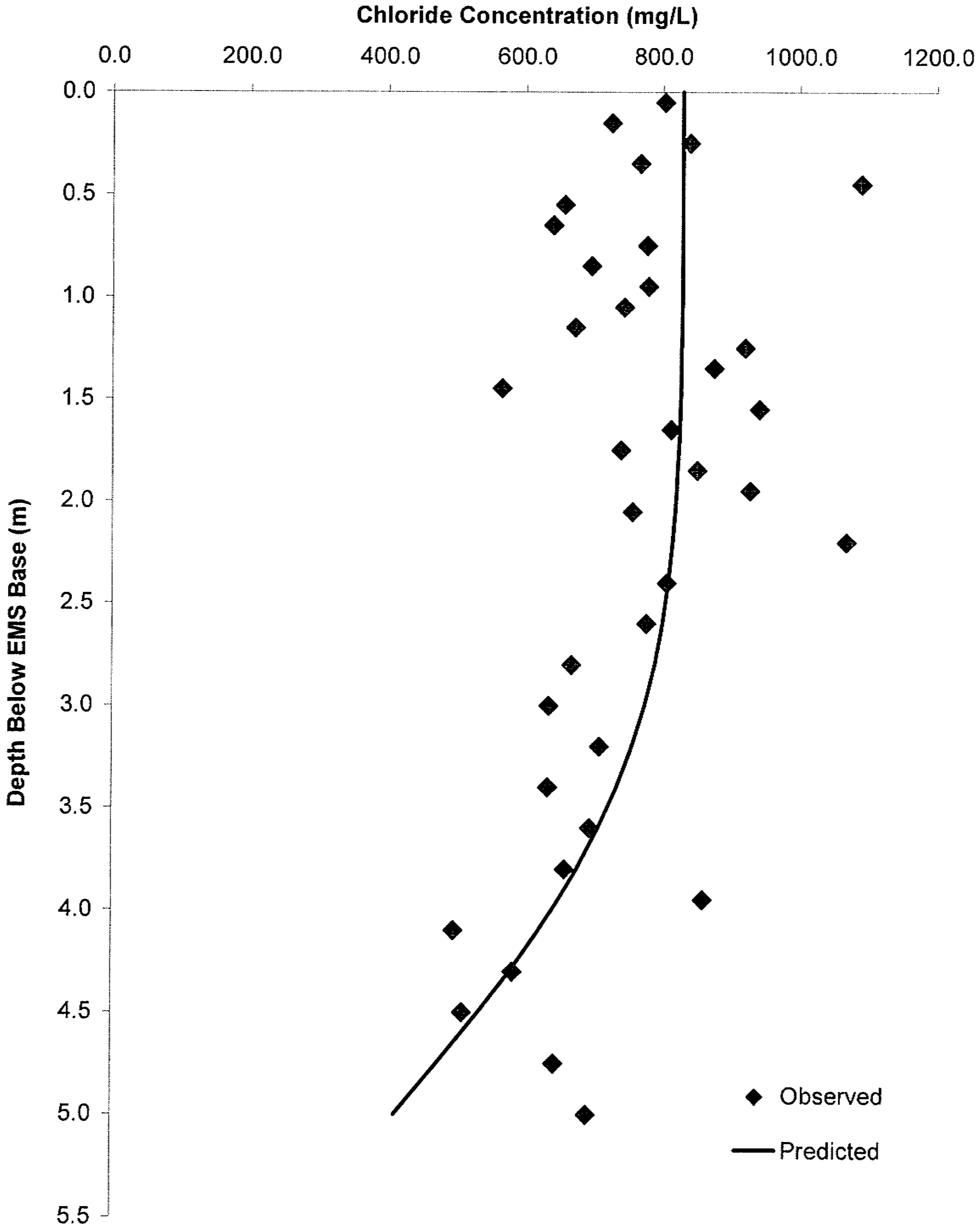


Figure 6.9: Observed vs. predicted seepage for the SC1 EMS sample core.

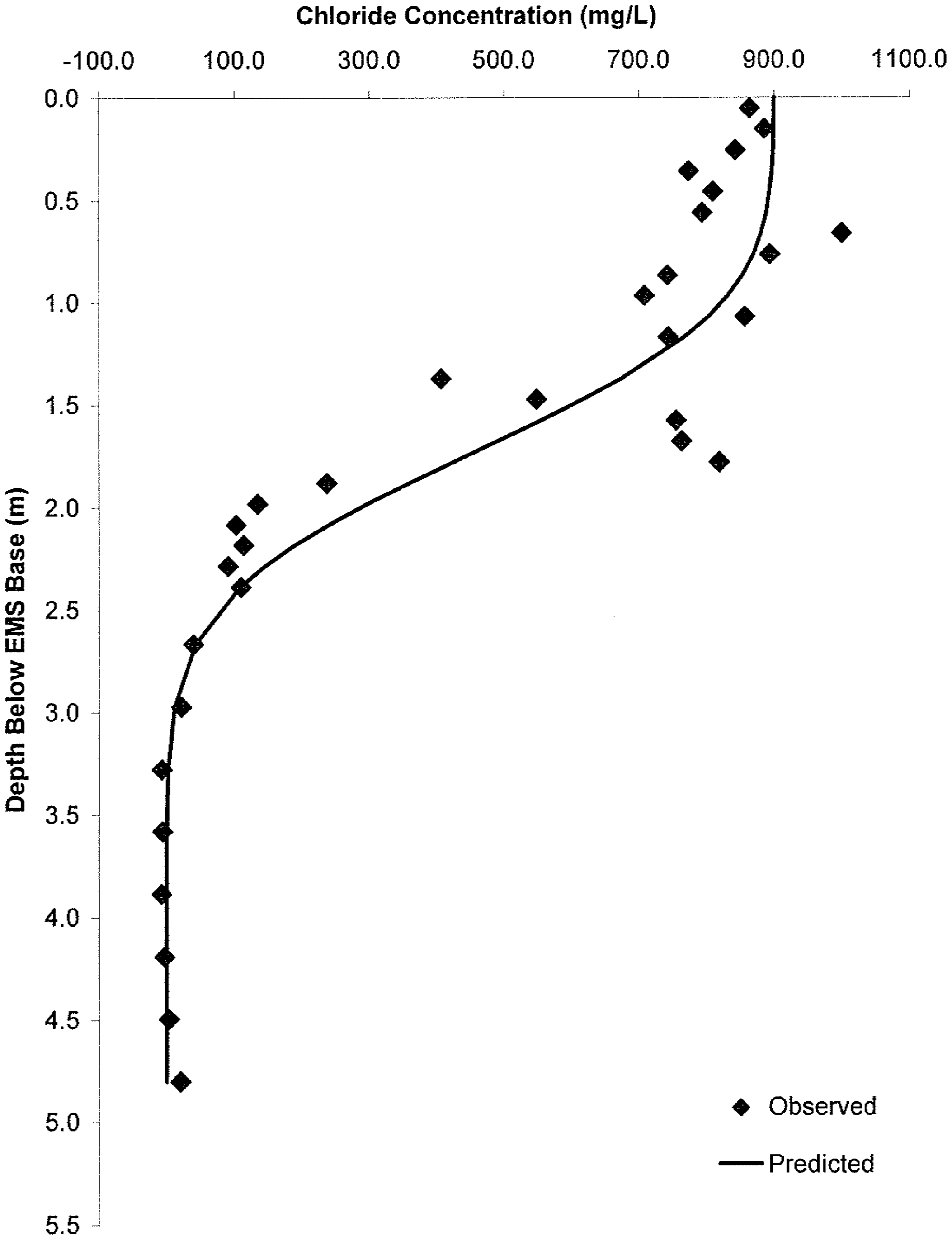


Figure 6.10: Observed vs. predicted seepage for the SC3(1) EMS sample core.

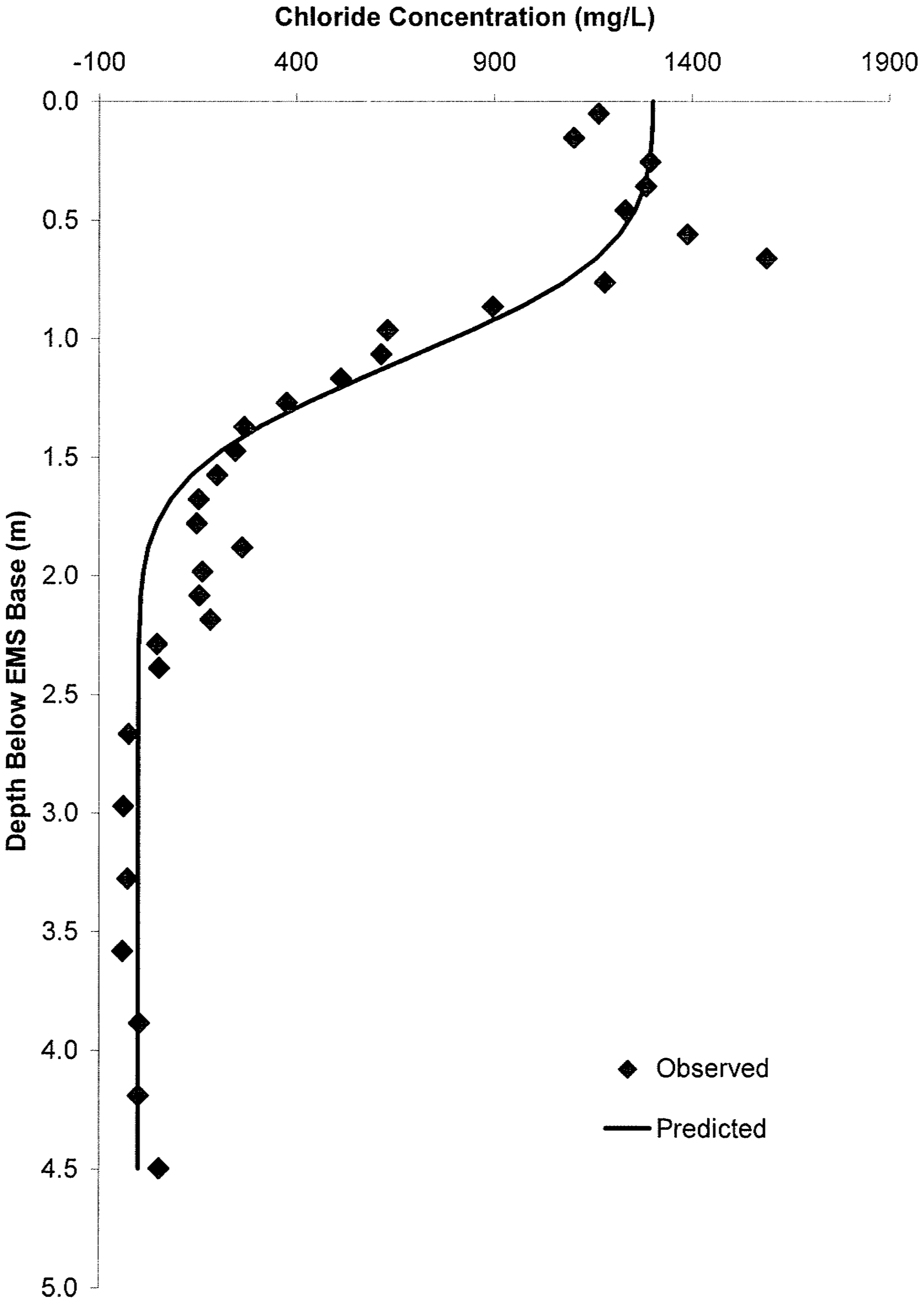


Figure 6.11: Observed vs. predicted seepage for the SC3(2) EMS sample core.

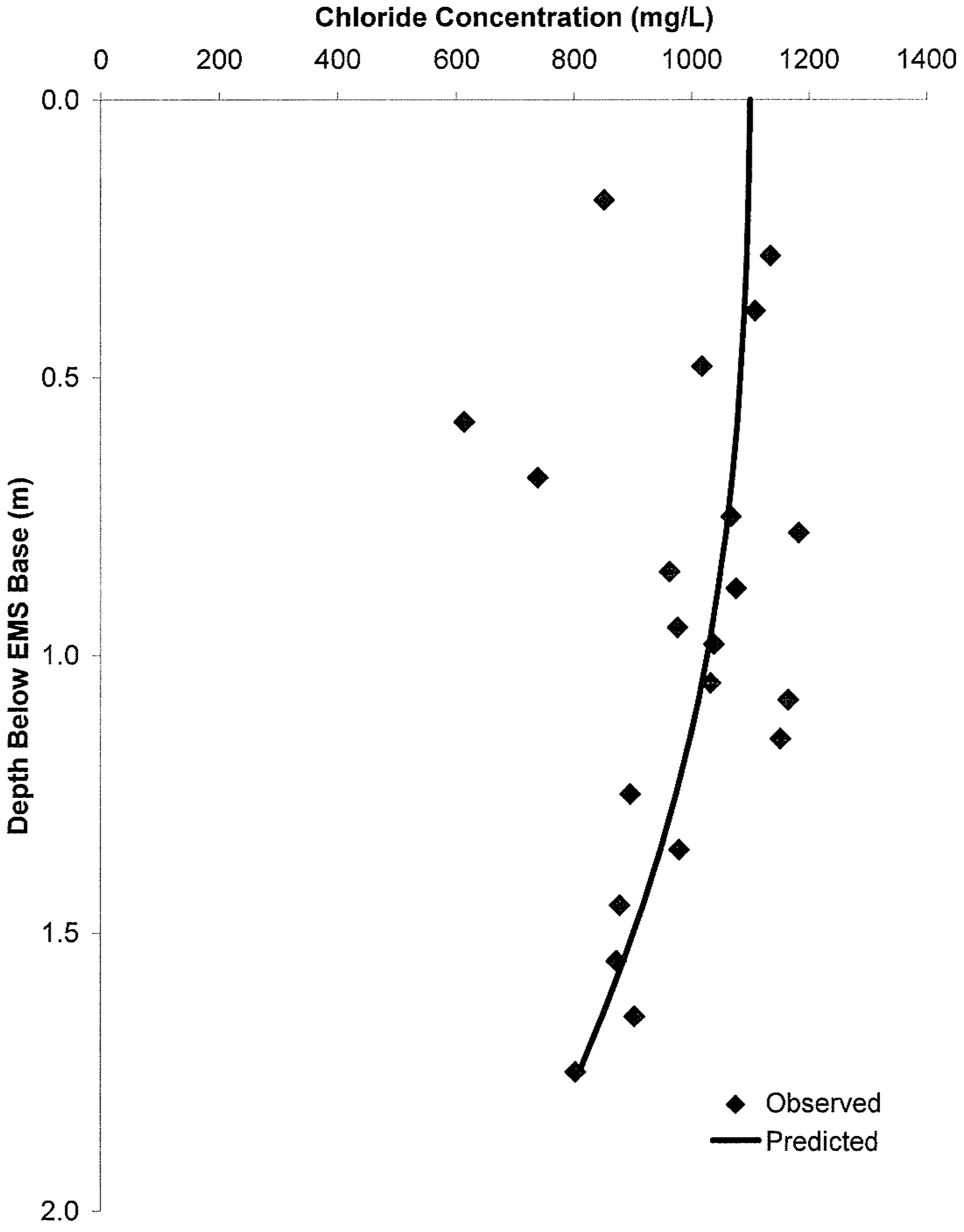


Figure 6.12: Observed vs. predicted seepage for the SE1 EMS sample core.

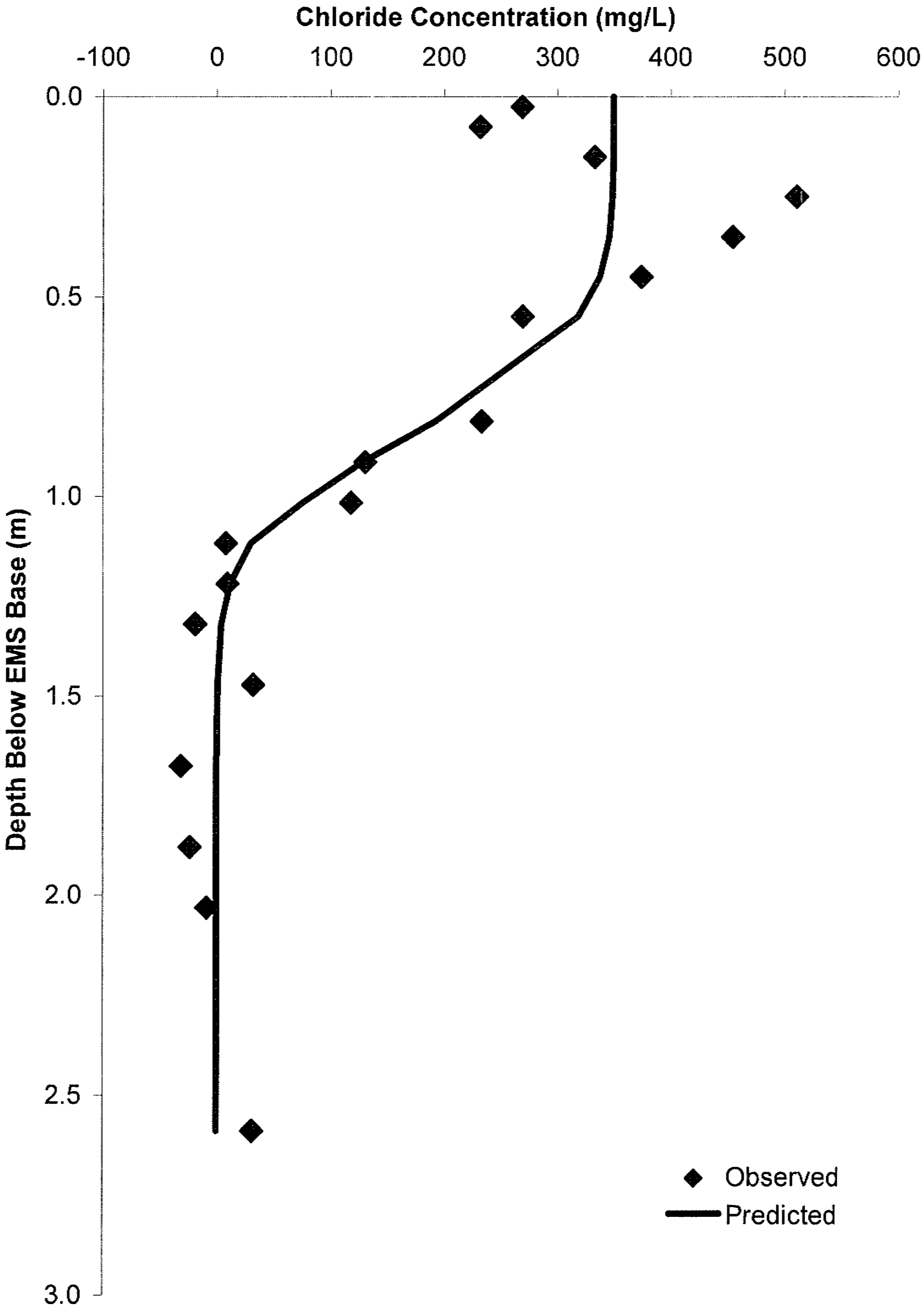


Figure 6.13: Observed vs. predicted seepage for the SE2(1) EMS sample core.

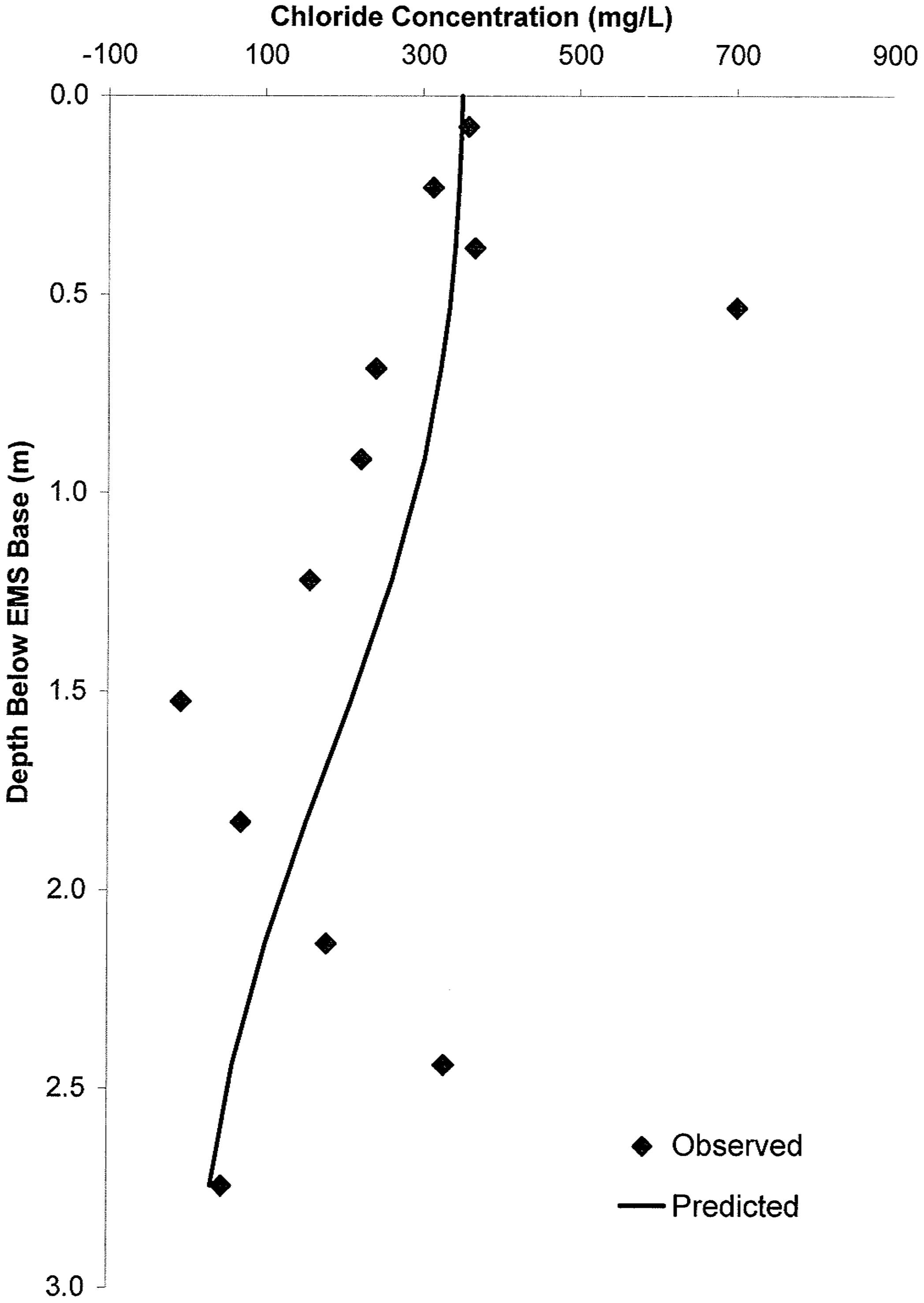


Figure 6.14: Observed vs. predicted seepage for the SE2(2) EMS sample core.

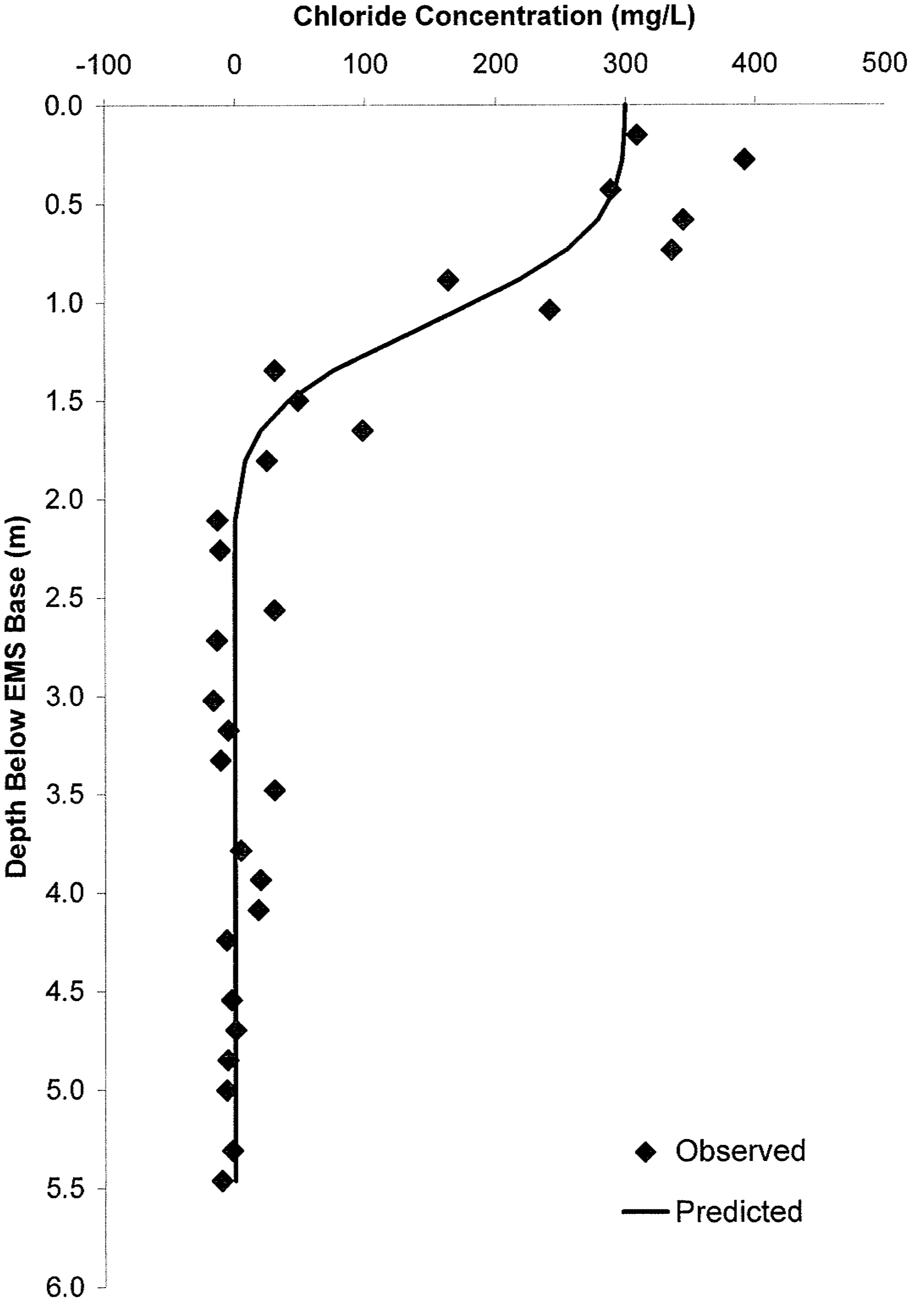


Figure 6.15: Observed vs. predicted seepage for the SE3 EMS sample core.

CHAPTER 7: GENERIC PREDICTIVE MODELING

In this section, the parameter estimates obtained in Chapter 6 using the one-dimensional analytical solution to the advection dispersion equation, are used to predict the impacts on underlying aquifers for a number of generic cases. These cases include examinations of the following scenarios:

- the impact of liner and liner type in a till aquitard;
- the impact of sand layer(s) within a till aquitard;
- the impact of liner on a till-sand layered aquitard system;
- the impact of liner thickness in a till-sand layered aquitard system; and
- the impact of aquitard thickness and type.

In addition, the sensitivity of the solution for the homogeneous aquitard case, to changes in aquifer velocity, is also examined. Where available, the parameter estimates established in Chapters 5 and 6 are used in the estimate of material properties used in the simulations. Specifically, these include field measured and estimated parameters including hydraulic conductivity, porosity and hydrodynamic dispersion coefficient.

It is important to note that the generic modeling results provided in this chapter are preliminary and the predicted results should be used with caution due to assumptions inherent in the approach and the selection of input parameters.

The predictions are made using MIGRATEv9, a two-dimensional finite-layer solute transport model, which provides a semi analytical solution to the two-dimensional advection dispersion equation, and allows for variable velocities between strata (Rowe *et al.*, 1995). The purpose of the generic predictive modeling work is to: examine the impact of vertical migration of contaminants from an overlying EMS on an underlying aquifer for a selection of the typical hydrogeological settings encountered in this work; and, and assess impacts within an underlying aquifer, downgradient of a hypothetical facility.

7.1 MODELING METHODOLOGY

Generic predictive modeling is conducted using the MIGRATEv9 software package. Solutions can be obtained for single conservative or non-conservative solutes. The solution allows for multiple strata, however each stratum is considered homogeneous. Darcy velocity in each stratum is constant (Rowe *et al.*, 1995). The expectation in an operating EMS facility is that Darcy velocity in the underlying strata would vary depending on the depth of wastewater ponding within the facility, which varies seasonally due to precipitation inputs, waste inputs and facility pumping. A constant Darcy velocity, based on average to maximum liquid levels within the hypothetical EMS facility, will provide a conservative estimate of the vertical chloride migration rates.

Input requirements to the program when modeling the movement of a conservative species such as chloride, include upper and lower boundary conditions, as well as Darcy velocity, effective dispersion coefficients and effective porosity for each soil layer. For the cases examined, constant concentration boundary conditions are used for both the upper and lower boundaries. The upper boundary represents the base of the EMS facility being simulated and the bottom boundary represents the top of the underlying aquifer. Although there is an expectation that input chloride concentrations would fluctuate within the EMS on seasonal and operational stage basis, depending on liquid waste composition (as discussed previously), which may vary with feed type, etc., and amount of dilution, the selection of a constant value is considered reasonable to assess long-term impacts.

7.2 PARAMETER SELECTION AND ASSIGNMENT

Estimates of Darcy velocity in the vertical direction (q_v) and longitudinal hydrodynamic dispersion (D_L; see Equation 6.3 in Chapter 6), can be calculated from the approximated parameters described in Chapter 6. The parameter estimates were used to select parameters representing the till and clay materials underlying the hypothetical EMS facility for the generic modeling cases discussed below. To represent other materials modeled, such as geosynthetic clay liners and sand, published literature values were use. A summary of the parameter ranges deduced in Chapter 6 are provided below.

- Average linear groundwater velocities were found to range from 0.05 m/year to 0.10 m/year in the underlying clays and 0.10 m/year to 0.27 m/year in the underlying tills.
- Longitudinal dispersivity was found to be on the order of 0.01 m to 0.04 m in the clay and 0.02 m to 0.1 m in the till.

- Coefficients of effective diffusion were estimated to range from 0.003 m^2/year to 0.03 m^2/year.

The parameters used in the predictive modeling scenarios, for all material types examined in this section, are provided in Table 7.1. For each case examined, transverse hydrodynamic dispersion was estimated to be 10% of the longitudinal hydrodynamic dispersion (Freeze and Cherry, 1979). The selected hydraulic conductivities were consistent with the average values found to produce a reasonable range from the predictive modeling section, with the exception of the sand and GCL values which were based on published literature values (Freeze and Cherry, 1979, and Lake, 2000, respectively). Effective porosity values selected and (shown in Table 7.1) were based on published values of effective porosity and longitudinal dispersivity was selected based on the problem scale for the overburden sediments (Freeze and Cherry, 1979; Gelhar *et al.*, 1992).

It should be noted that in all cases involving multiple strata, Darcy velocity in the vertical direction is calculated using the harmonic mean to establish an effective hydraulic conductivity for flow perpendicular to stratification (e.g. Freeze and Cherry, 1979) and a hydraulic gradient used of 0.02 m/m is maintained for all cases and was based on the average calculated hydraulic gradient between the base of the EMSs examined in this work and the underlying aquifers. This allows for a direct comparison between multiple cases, to examine the effect of varying scenarios. In nearly all the cases, the overburden thickness is constant at 7 m thick, to ensure that the head conditions at the base of the EMS and the top of the aquifer are consistent between cases.

For each of the generic simulations, the underling aquifer was assumed to have an average horizontal groundwater velocity of 10 m/year (Kennedy, 2002) and a coefficient of hydrodynamic dispersion in the longitudinal direction of 1 m^2/year (Gelhar *et al.*, 1992).

Chloride is modeled in the generic prediction to provide a worst-case estimate of impacts resulting from the EMS source. Chloride is a conservative anion which is not sorbed or biodegraded as it moves through sediments. The source boundary condition imposed on the problem was a constant chloride concentration of 1000 mg/L for all times examined. This chloride concentration is within the expected ranges of chloride concentrations measured in EMS facilities (see Table 2.1 and Table 5.2). The initial condition and bottom boundary condition imposed on the problem was a zero chloride concentration at the initial time zero. Not incorporating background concentrations within the problem allow for an evaluation of the impacts to the underlying sediments and aquifer resulting from the EMS source alone.

The MIGRATEv9 program requires the selection and input of four parameters for the inversion of the Laplace transform, these are TAU, N, SIG and RNU. The default value of 7 for TAU and 0 for SIG were selected, the values of RNU and N were calculated for each of the problems based on the total problem thickness, the Darcy velocity and the minimum diffusion coefficient. FINE was chosen for the numerical Gauss Integration parameter for all cases examined. The numerical stability of the model outputs was checked by examining the impact of changes to the number of sub-layers divisions and the numerical Gauss Integration parameters. The numerical stability check was conducted by running the case C1 model for four additional analyses, selecting MEDIUM and WIDE Gauss Integration parameters and by reducing the number of sub-layers by one-half and increasing the number of sub-layers by a factor of 2. The results of the sensitivity analysis is provided in Appendix A, and the results show that the solution obtained for case C1 using the input integration parameters and sub-layers is numerically stable.

7.3 CASE DEFINITIONS AND SIMULATION RESULTS

A number of cases were examined to investigate the impact of liner types and thickness, the importance of site characterization, and to examine the sensitivity of the model output to changes in parameter values. The cases examined would be consistent with typical hydrogeological settings in agro-Manitoba, particularly those representing minimum overburden thicknesses (for example the NC2 and NC3 sites). The results of the generic simulations are described for each of these cases below. In all cases, the simulated EMS facility is defined by a base width of 100 m, a top width of 110 m and side slopes of 1:2. For all cases examined, a vertical hydraulic gradient of 0.02 was selected to represent conditions beneath the facility and facilitate sensitivity analysis. It should be noted that the cases discussed in Section 7.3.5, and examining the impact of aquitard thickness, do not maintain the constant gradient for all cases (in particular the C23AG and C25AG cases), but maintain the head drop between the bottom of the hypothetical EMS and the top of the underlying aquifer. This hydraulic gradient value was used for each case analysed in an attempt to directly compare the impacts on the underlying aquifer. Additionally, unless stated otherwise, a 3 m thick aquifer is present at a specified depth below the EMS facility. It is assumed that the aquifer groundwater flux is sufficiently large compared to the vertical flux entering this layer, thus, a similar aquifer groundwater velocity can be anticipated for each case examined.

For the purposes of the following discussion, the contaminant front is defined at that point along the profile where the chloride concentration is equal to one-half the source concentration.

7.3.1 Impact of Liner and Liner Type in a Till Aquitard

Cases C1, C1L and C1GCL examine the impact of a hypothetical EMS facility on the underlying soils and aquifer. In case C1, an EMS facility overlying 7 m of till followed by a 3 m thick aquifer is simulated. Case C1 simulates a facility constructed with no recompacted clay liner. In the C1L and C1GCL cases, facilities of the same dimensions and situated in the same hydrogeological setting as case C1 are simulated, however these facilities examine the impact of installing a 1 m thick clay liner and a 0.015 m thick geosynthetic clay liner overlying till, respectively. The GCL thickness was based on typical thicknesses found in manufacturer specifications. In Case C1L, 1 m of recompacted clay overlies 6 m of till; and in Case C1GCL, a 0.015 m thick geosynthetic clay liner overlies 6.985 m of till.

Figure 7.1 shows the chloride concentration with depth below the centerline of the hypothetical EMS facility 30 years after operation commenced, corresponding to a typical EMS facility lifespan. After 30 years, the contaminant front in case C1 is located at a depth of 5.0 m below grade. As discussed in Chapter 1, 30 years is considered a reasonable estimate of the maximum operational lifespan of many EMS facilities in Manitoba. The installation of a 1 m thick clay liner (case C1L) results in a contaminant front location of 0.7 m below the EMS base and no impact detected below a depth of 3.5 m. Case C1GCL also results in a significant reduction in the contaminant front position at 30 years, with a contaminant front position of 1.8 m below the EMS base and no impact apparent below 5 m below the base of the facility.

The reduction in contaminant front movement between the C1, C1L and C1GCL cases is due to the reduction in overall effective hydraulic conductivity calculated using the harmonic mean. The addition of the 1 m thick clay liner reduces the effective hydraulic conductivity of the overburden by a factor of 4.7 and the addition of the 0.015 m thick GCL reduces the effective hydraulic conductivity of the overburden by a factor of 3.1, which would have the greatest overall impact on migration beneath the facilities, and would cause a proportional reduction in the Darcy velocity beneath the hypothetical EMS. Hydrodynamic dispersion beneath the hypothetical facility is also reduced where the liner or GCL has been substituted for the case C1 till material, and as a result, there would be a reduction in the extent of movement due to the presence of these materials.

Figure 7.2 show the chloride concentration with distance from the centerline of the hypothetical EMS facility 30 years after operation commenced. At 30 years, in the unlined case (case C1), elevated chloride concentrations occur within the underlying aquifer and extend to a distance of approximately 800 m from the centerline of the facility. The peak chloride concentration of 10 mg/L occurs downgradient, at a distance of 110 m from the upgradient edge of the facility (55 m from the EMS centerline). In both the lined cases (cases C1L and C1GCL), no detectable impact is present in the underlying aquifer at 30 years after operation commenced. For the C1 case, chloride concentrations increase from 6 mg/L to 10 mg/L beneath the EMS as a result of continuous inputs of chloride as the flowline along the top of the aquifer moves through the aquifer, beneath the facility.

7.3.2 Impact of Sand Layer(s) within a Till Aquitard

Cases C1SAND and C1SAND2 examine the impact of sand layer at depth within a till, underlying a simulated EMS facility. These two cases are compared to case C1 to examine the impact of a sand layer within the aquitard on chloride migration within the subsurface. In case C1SAND, a 1 m thick sand layer is present at a depth of 3 m below the base of the EMS; and in case C1SAND2, two-1 m thick sand layers are present; one layer at a depth of 3 m below the EMS base and one at a depth of 5 m below the EMS base.

The contaminant front in case C1SAND occurs 1.0 m deeper than case C1 at 30 years after the simulation commenced, and the contaminant front in case C1SAND2 occurs 1.8 m deeper than case C1 (Figure 7.3). However, it is important to note that the location of the contaminant front in this case would be influenced by the underlying aquifer. The increase in the contaminant front movement between the C1, C1SAND and C1SAND2 cases is due to the increase in overall effective hydraulic conductivity calculated using the harmonic mean. The addition of a single 1 m thick sand layer causes an increase in the effective hydraulic conductivity of the overburden by a factor of 1.2 and the addition of two sand layers, totaling 2 m thick, causes an increase in the effective hydraulic conductivity of the overburden by a factor of 1.8. The increase in the effective hydraulic conductivity, as was the case previously, would have the greatest overall impact on migration beneath the facilities, and would cause a proportional increase in the Darcy velocity beneath the hypothetical EMS. Hydrodynamic dispersion beneath the hypothetical facility is also increased where the sand layers have been substituted for the case C1 till material, and as a result, there would be an increase in the extent of movement due to the presence of the sand.

The difference between cases C1, C1SAND AND C1SAND2 is most notable when examining Figure 7.4, which shows the concentration of chloride with distance from the EMS centerline at 30 years after the simulation commenced. Peak concentrations occur on the downgradient side of the facility at a distance of approximately 100 m from the EMS centerline (corresponding with the edge of the facility). The peak concentrations are on the order of 10 mg/L, 40 mg/L and 185 mg/L for cases C1, C1SAND and C1SAND2, respectively. In addition, detectable chloride concentrations extend to distances of about 500 m, between 500 and 1000 m, and greater than 1000 m for cases C1, C1SAND and C1SAND2, respectively.

Figure 7.5 and 7.7 and 7.6 and 7.8 show time series plots of the concentration versus depth and concentration versus distance for the C1 and C1SAND2 cases, respectively. For case C1, detectable chloride concentrations appear in the underlying aquifer for the 30 year simulation. Peak concentrations in the aquifer increase from approximately 10 mg/L at 30 years, to 90 mg/L and 125 mg/L at 50 and 100 years, respectively. Additionally, elevated concentrations extend downgradient over 100 m past the EMS centerline (Figure 7.6). Conversely, for case C1SAND2 elevated chloride concentrations appear in the underlying aquifer for the 15 year simulation. At 30 and 50 years after facility operation commences, peak concentration in the aquifer increase from just under 120 mg/L to approximately 185 mg/L, respectively. For the 30 and 50 year simulations these peak concentrations occur on the downgradient side of the EMS

Comparing Figures 7.6 and 7.8, it is apparent that the impact on the aquifer is reduced for the case where there is lower advective and diffusive flux into the aquifer from the EMS, as expected. In all cases, at 30 years after EMS operation commenced, chloride concentrations are below the CCME guideline for chloride in drinking water of 250 mg/L at the downgradient edge of the EMS.

7.3.3 Impact of Liner in a Till-Sand Layered Aquitard System

Cases C1SANDL and C1SAND2L examine the impact of installing a liner for the case of a till containing 1 m thick sand layer(s) as described in Section 7.3.3 above, should an EMS be constructed in such a setting. These two cases are compared to the cases where no liner was installed (C1SAND AND C1SAND2). In case C1SANDL, a 1 m thick clay liner is installed at the base of the facility, followed by 2 m of till, 1 m of sand followed by an additional 3 m of till. In case C1SAND2L, a 1 m thick clay liner is installed at the base of the facility followed by 2 m of till, 1 m of sand, 1 m of till, 1 m of sand followed by 2 m of till.

The contaminant front in the lined case (case C1SANDL) occurs at a depth of approximately 0.7 m compared to a depth of 6 m in the unlined case (C1SAND). Similarly, the contaminant front in case C1SAND2L occurs at a depth of approximately 0.8 m compared to a depth on the order of 6.8 m in the unlined case (C1SAND2). Overall, the installation of a liner in these two cases results in a reduction in the movement of the contaminant front by a factor of approximately 8 (Figure 7.9). In such situations, the installation of a liner will provide a significant reduction in the migration of chemical species from the facilities, due to the reduced advective flux through the sediments and the reduced dispersive flux through the liner materials compared to the unlined cases.

Figure 7.10 shows the concentration of chloride with distance from the EMS centerline at 30 years after the simulation commenced. Peak concentrations occur on the downgradient side of the facility at a distance of approximately 110 m from the upgradient edge of the EMS, corresponding with the downgradient edge of the facility. The peak concentrations are on the order of 40 mg/L and 185 mg/L for cases C1SAND and C1SAND2, respectively. Elevated chloride concentrations extend to a distance on the order of 500 m to 1000 m downgradient of the facility in the C1SAND case, to a distance of greater than 1000 m downgradient of the facility in the C1SAND2 case. For the lined comparative cases, no impact is detected in the aquifer at 30 years after operation commenced. This would indicate that the installation of a 1 m thick lower permeability (clay) liner has a significant impact on reducing chloride migration from the facility, where sand layer(s) are present in the overburden materials.

7.3.4 Impact of Liner Thickness in a Till-Sand Layered Aquitard System

The analysis completed in section 7.3.3 showed the potential benefits of installing a 1 m thick low-permeability clay liner to a till-sand layered aquitard system. The following analysis assesses the impact of liner thickness on the migration of chloride into the sub-surface beneath a facility underlain by a till unit containing two 1 m thick sand layers.

Cases C1SAND2L-15, C1SAND2L-30, C1SAND2L-60 and C1SAND2L cases have a clay liner thickness of 0.15 m, 0.3 m, 0.6 m and 1 m thick, respectively, and the facility is underlain by 2.85 m of till, 2.7 m of till, 2.4 m of till and 2 m of till, respectively, followed by 1 m of sand, 1 m of till, 1 m of sand and 2 m of till.

The impact of increasing liner thickness on the movement of the contaminant front is illustrated in Figure 7.11. The location of the contaminant front is reduced by factors on the order of over 1.5, 2, 3 and 8 for case C1SAnD2L-15, C1SAND2L-30, C1SAND2L-60 and C1SAND2L, respectively, when compared to the unlined case (C1SAND2). It is apparent, that the thicker the clay liner installed, the lower the advective and diffusive flux of chloride from the hypothetical EMS into the underlying sediments.

Figure 7.12 shows the concentration of chloride with distance from the EMS centerline at 30 years after the simulation commenced. Again, peak concentrations occur on the downgradient side of the facility at a distance of 110 m from the upgradient edge of the EMS. The presence of liners varying from 0.15 m to 1 m in thickness result in a reduction of chloride impacts in the underlying aquifer, when compared to the unlined case presented in Figure 7.10 (C1SAND2). However, only liner thicknesses of 0.3 m or greater will facilitate near non-detectable chloride concentrations in the aquifer 30 years after facility operation commences. It should be noted that even a relatively thin recompacted clay liner of 0.15 m has benefit over no liner, with peak concentrations for unlined case on the order of 180 mg/L compared to peak concentrations for the 0.15 m clay lined case of approximately 10 mg/L,

7.3.5 Impact of Aquitard Thickness and Type

Figure 7.13 shows the concentration with depth beneath the centerline of a simulated EMS facility. As noted previously, case C1 simulates an EMS site underlain by 7 m of till followed by a 3 m thick aquifer. Case C27 simulates an EMS site underlain by 7 m of clay followed by the same 3 m thick aquifer. In cases C23AG and C25AG, the simulated sites are underlain by 3m and 5 m, respectively, of clay followed by the same 3 m thick aquifer.

For the three clay cases (C23AG, C25AG and C27), the contaminant front is located at a depth of approximately 0.4 m below grade after 30 years of facility operation. The progression of the contaminant front for the 3 m, 5 m and 7 m clay cases show only slight variation, with approximately 0.1 m of additional shift downward in the contaminant front for the 3 m thick case, compared to the 7 m thick case. Overall, compared to the 7 m till case (C1), the location of the contaminant front position at 30 years is reduced by a factor of over 10. This is expected since chloride migration will be reduced through clay aquitards compared to till due to the lower advective flux and lower dispersive flux in the clay compared to till.

In each of these four cases, no chloride impacts are detectable in the underlying aquifer at 30 years after operation of the facility commenced and it is apparent that the clay overburden cases (case C23AG, C25AG and C27) will result in a significant delay in breakthrough.

7.4 DISCUSSION OF RESULTS

The results presented in Section 7.3.1, which examine the effect of installing two different types of liners, a 1 m thick clay liner and a 0.015 m thick GCL, suggest that the installation of a clay or GCL liner results in a delay in breakthrough, or first instance of impact, of chloride into the aquifer. In addition, it is observed that within the expected operational lifespan of the facility (30 years), minimal impact will be observed in the underlying aquifer should either a 1 m thick clay liner or a 0.015 m thick GCL be installed. Without the installation of a liner system, some degree of impact will be observed in the underlying aquifer, although the impacts will be below the CCME guidelines for chloride in drinking water.

Examination of the impact of sand layer(s) within a till deposit indicates that the presence of these units can have a significant effect on the position of the contaminant front and whether any impacts will be detected in the underlying aquifer during the typical operational lifespan of an EMS facility. Examination of these cases also serves to illustrate the importance of proper site characterization, through a detailed sub-surface investigation during the initial facility feasibility and design stages. The presence of sand layer(s) in the underlying overburden, should they be undetected, may result in an increase in peak concentrations by a factor of 18 (in the case of two 1-m thick lenses and/or layers) in the underlying aquifer and elevated chloride concentrations extending to as much as 1 km downgradient of the facility during this period, when compared to the case of a homogeneous till. Should the presence of these layer(s) be detected through site characterization, liner systems may be installed to delay chloride breakthrough into the aquifer. As shown, chloride concentrations were not elevated in the underlying aquifer after 30 years of operation for a facility constructed with a 1 m thick clay liner. Results indicate that liners of thickness 0.3 m or greater will result in minimal elevated chloride concentration in the underlying aquifer to less than 1 mg/L after 30 years of continuous facility operation (for the selected till thickness).

The results of the generic simulations indicate that there is a significant reduction in conservative contaminant movement when sites are underlain by clay rather than till. This reduces the impact on aquifers underlying these aquitards. In addition, there is little change in the position of the contaminant front after 30 years of continuous operation if the site is underlain by a thinner clay aquitard as opposed to a thicker clay aquitards. As little as 3 m of clay separating the base of the facility and the underlying aquifer will ensure no impacts are observed in the underlying aquifer after 30 years of facility operation.

The above discussion assumes that the overburden materials separating the base of the facility from the underlying aquifer are intact with no secondary permeability features such as fractures of fissures. Should the underlying aquitard be fractured, there is potential for significant impacts to occur in the underlying aquifer after 30 years of continuous facility operation. It should also be noted that the results of the generic modeling discussed in this chapter are preliminary and should be used and interpreted with caution.

Table 7.1 Parameter Values Used for Generic Predictive Modeling

Material	**K (m/s)**	**n_e**	**α_L (m)**	**D* (m^2/year)**
Clay	2×10^{-9}	0.40	0.1	0.003
Till	5×10^{-8}	0.20	0.1	0.0165
Sand	5×10^{-4}	0.15	0.1	0.03
GCL(1)	5×10^{-11}	0.40	Negligible	0.01

Notes:

(1) Geosynthentic Clay Liner (GCL) properties obtained from estimates set out in Lake, 2000.

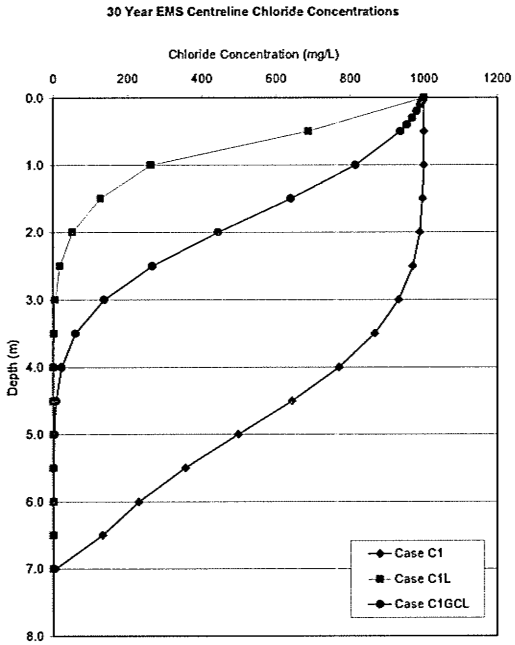


Figure 7.1: Chloride concentrations beneath EMS centerline 30 years after facility operation commenced for the C1, C1L and C1GCL simulation cases.

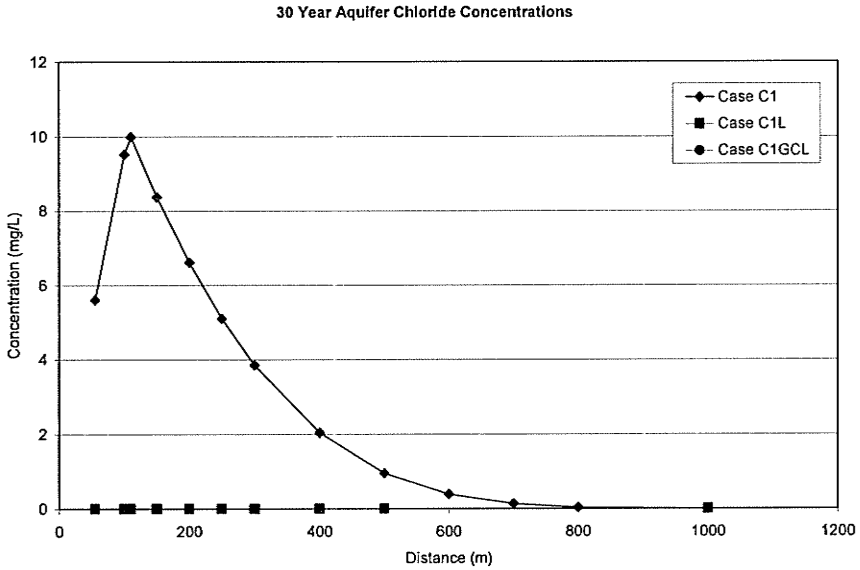


Figure 7.2: Chloride concentrations at top of aquifer with distance from EMS upgradient edge 30 years after facility operation commenced for the C1, C1L and C1GCL simulations.

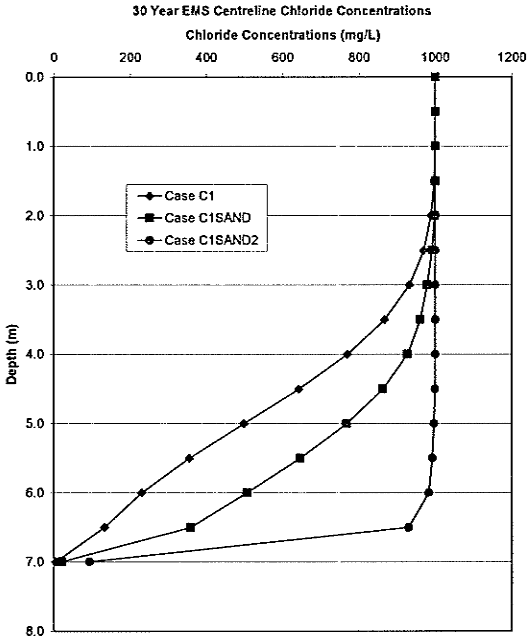


Figure 7.3: Chloride concentrations beneath EMS centerline 30 years after facility operation commenced for the C1, C1SAND and C1SAND2 simulations.

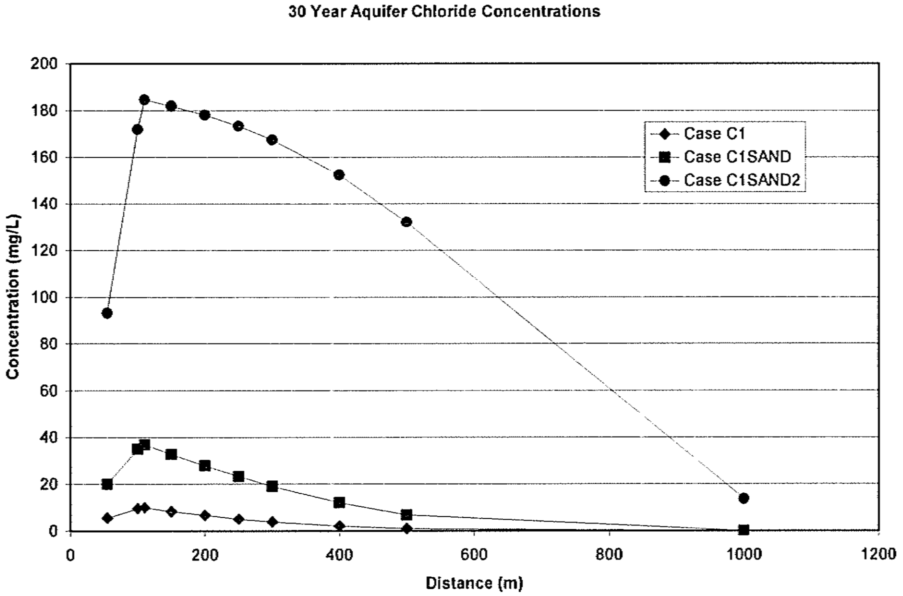


Figure 7.4: Chloride concentrations at top of aquifer with distance from EMS upgradient edge 30 years after facility operation commenced for the C1, C1L and C1GCL simulations.

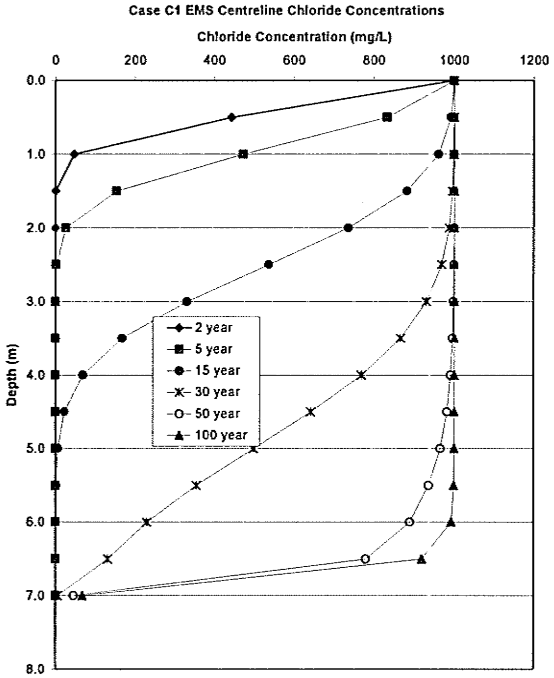


Figure 7.5: Chloride concentrations beneath EMS centerline for the case C1 simulation, 2, 5, 15, 30, 50 and 100 years after facility operation commenced.

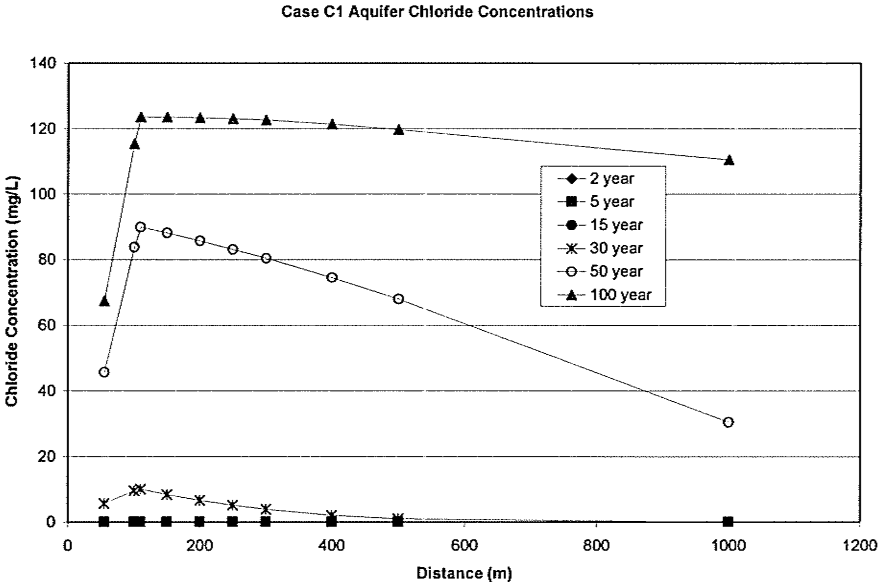


Figure 7.6: Chloride concentrations at top of aquifer with distance from the upgradient edge of the EMS, 2, 5, 15, 30, 50 and 100 years after facility operation commenced for the C1 simulation.

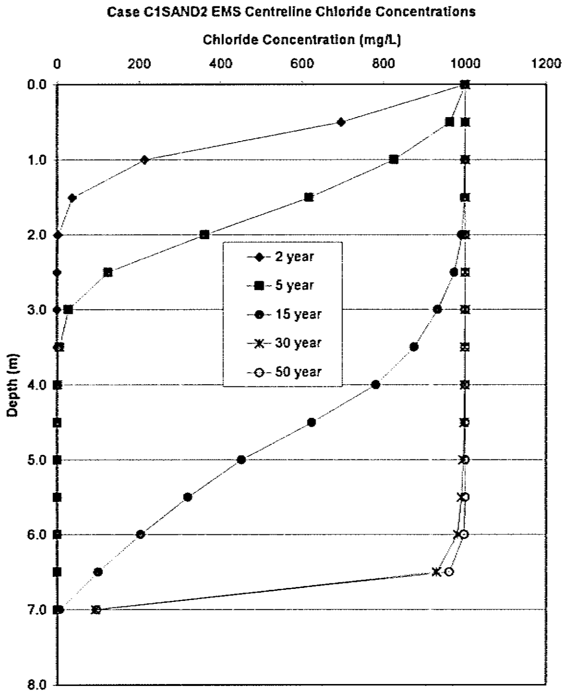


Figure 7.7: Chloride concentrations beneath EMS centerline for the case C1SAND2 simulation, 2, 5, 15, 30 and 50 years after facility operation commenced.

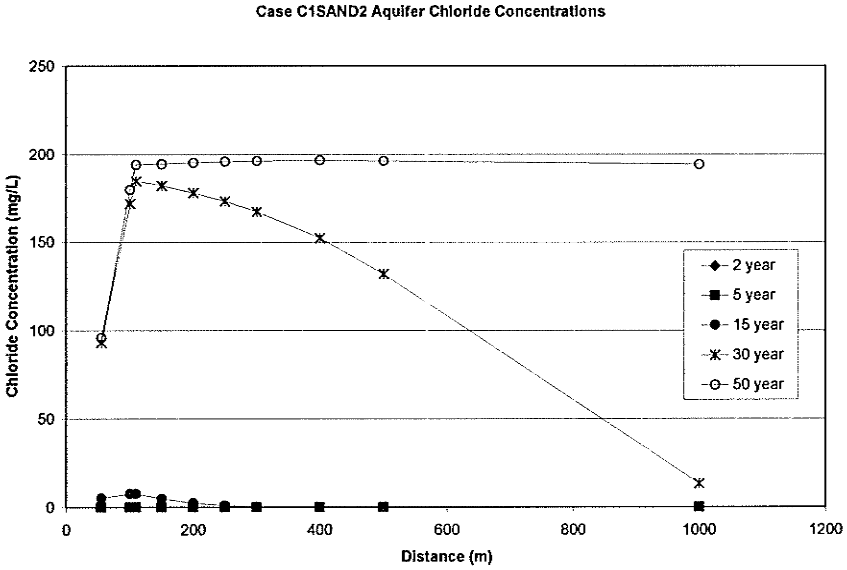


Figure 7.8: Chloride concentrations at top of aquifer with distance from the upgradient edge of the EMS, 2, 5, 15, 30 and 50 years after facility operation commenced for the C1 simulation.

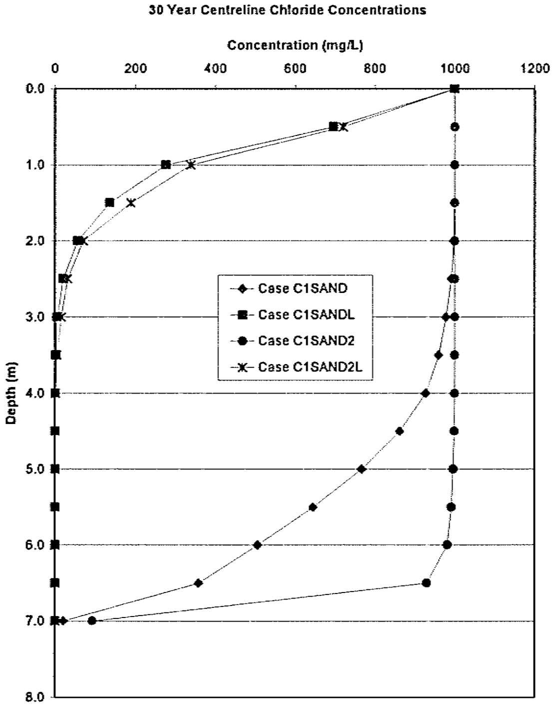


Figure 7.9: Chloride concentrations beneath EMS centerline after 30 years of facility operation for the C1SAND, C1SAND2, C1SANDL and C1SAND2L simulations.

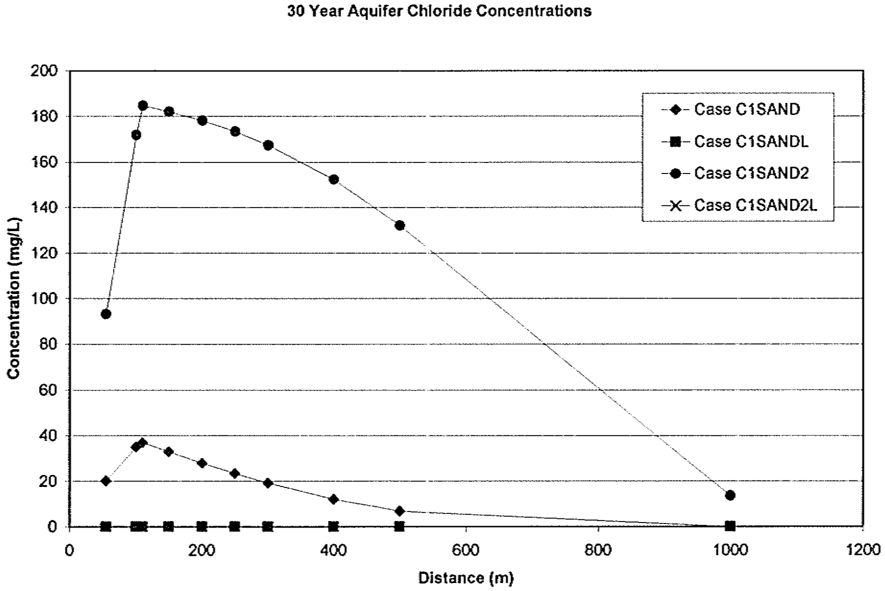


Figure 7.10: Chloride concentrations at top of aquifer with distance from EMS upgradient edge 30 years after facility operation commenced for the C1SAND, C1SAND2, C1SANDL and C1SAND2L simulations.

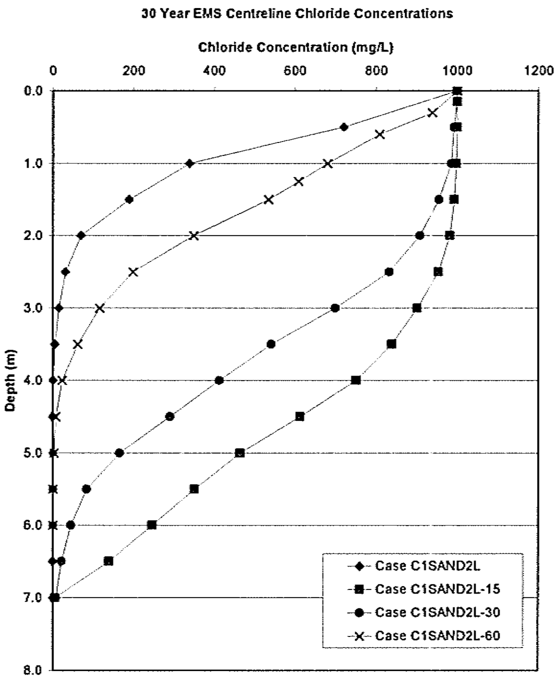


Figure 7.11: Chloride concentrations beneath EMS centerline after 30 years of facility operation for the C1SANDl, C1SANDl-15, C1SANDL-30 and C1SANDL-60 cases.

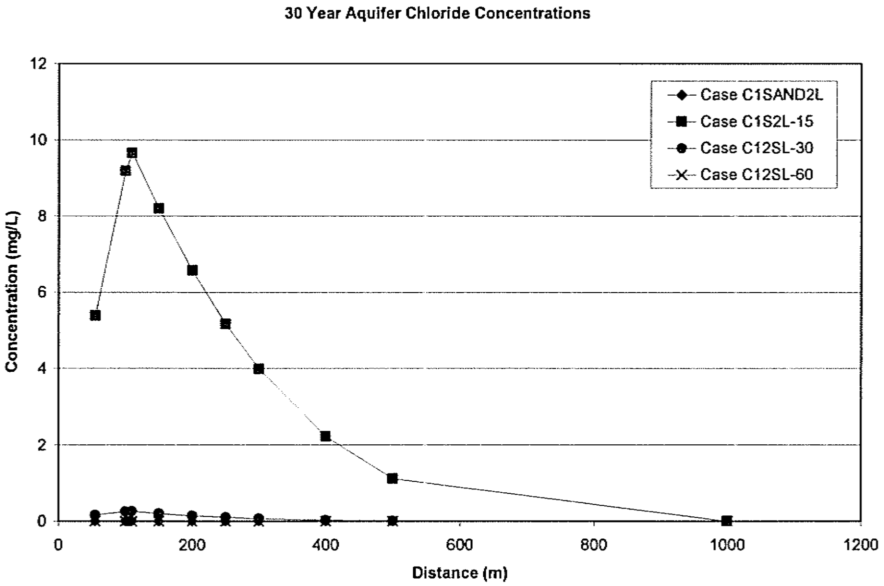


Figure 7.12: Chloride concentrations at top of aquifer with distance from EMS upgradient edge 30 years after facility operation commenced for the C1SANDl, C1SANDl-15, C1SANDL-30 and C1SANDL-60 cases.

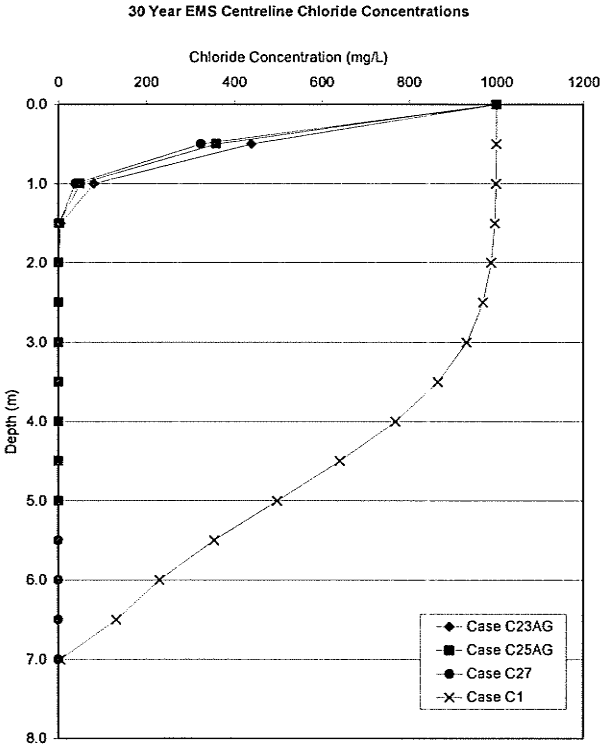


Figure 7.13: Chloride concentrations beneath EMS centerline after 30 years of facility operation for the C27, C23AG, C25AG and C1 simulations.

CHAPTER 8: CONCLUSIONS AND RECOMMENDATIONS

The results of this study indicate that seepage is occurring from EMS facilities located in typical hydrogeological settings encountered in agro-Manitoba. Transport of contaminants associated with this migration may reach underlying aquifers, particularly where overburden thicknesses are thin and heterogeneities, such as fractures or higher permeability layers, exist. This conclusion was established based on the examination of sample cores collected beneath ten EMS facilities, chemical analysis of soil pore water, and the result of one-dimensional solute transport modeling used to estimate parameters effecting the migration of chloride, a conservative contaminant, from the facilities. Further preliminary generic predictive modeling, suggests that the installation of clay liners and geosynthetic clay liners have the potential to reduce the extent of conservative contaminant migration beneath the facilities by as much as a factor of eight within a typical EMS operating lifespan, particularly where higher permeability sand layers are present in the aquitard materials. Overall, the generic predictive modeling results suggest that the effect of the liner reduces contaminant impact on underlying aquifers during the lifespan of an EMS facility compared to cases where there is no liner, even where the facilities are underlain by low permeability native soils, such as the clays and tills encountered in this work. However, it should be noted that the generic predictive modeling conducted at this point is preliminary and further modeling is recommended.

The results of the field study also show that ammonium migration is occurring beneath all of the EMS facilities examined, and that ammonium is generally contained within the upper 2 m of the overburden sediments, while elevated chloride was present to depths of as much as 6 m below the EMS facilities. The lack of nitrate+nitrite beneath the facilities would serve to indicate that nitrate production is not occurring beneath any of the facilities examined in this work, even those which had ceased operation years prior to removal of the sample core. The lack of nitrate+nitrite beneath these facilities is considered to be due to the presence of ponded water after closure which has maintained anaerobic conditions in the underlying sediments.

Based on the significant findings of this work, as outlined above, a number of recommendations are made.

- Adequate site characterization, including the identification of any heterogeneities such as sand layers, is critical. Generic modeling results show that the presence of sand layers can cause a significant increase in the extent of conservative contaminant migration and also allow for the accumulation of contaminant mass.
- The installation of a clay liner or geosynthetic clay liner can significantly reduce the extent of conservative contaminant migration from EMS facilities. Liners should be considered in all sites, even where the EMS is to be underlain by relatively low permeability materials such as clays or silt and/or clay-rich tills. It should be noted that the potential for damage to the liner during emptying of the facilities, by equipment, dessication and/or cracking is significant and should also be considered and adequate protection provided.
- Proper closure of an EMS facility should include an evaluation of the vertical extent of the ammonium build-up zone beneath the facility to be abandoned. Proper closure should include the removal of the ammonium-rich sub-soils underlying the facility, or

alternatively a plan which includes the contingencies for the continuation of anaerobic conditions beneath the former facility.

- Multi-dimensional contaminant transport modeling should be completed which considers the potential for fracturing of the overburden sediments encountered in this work.

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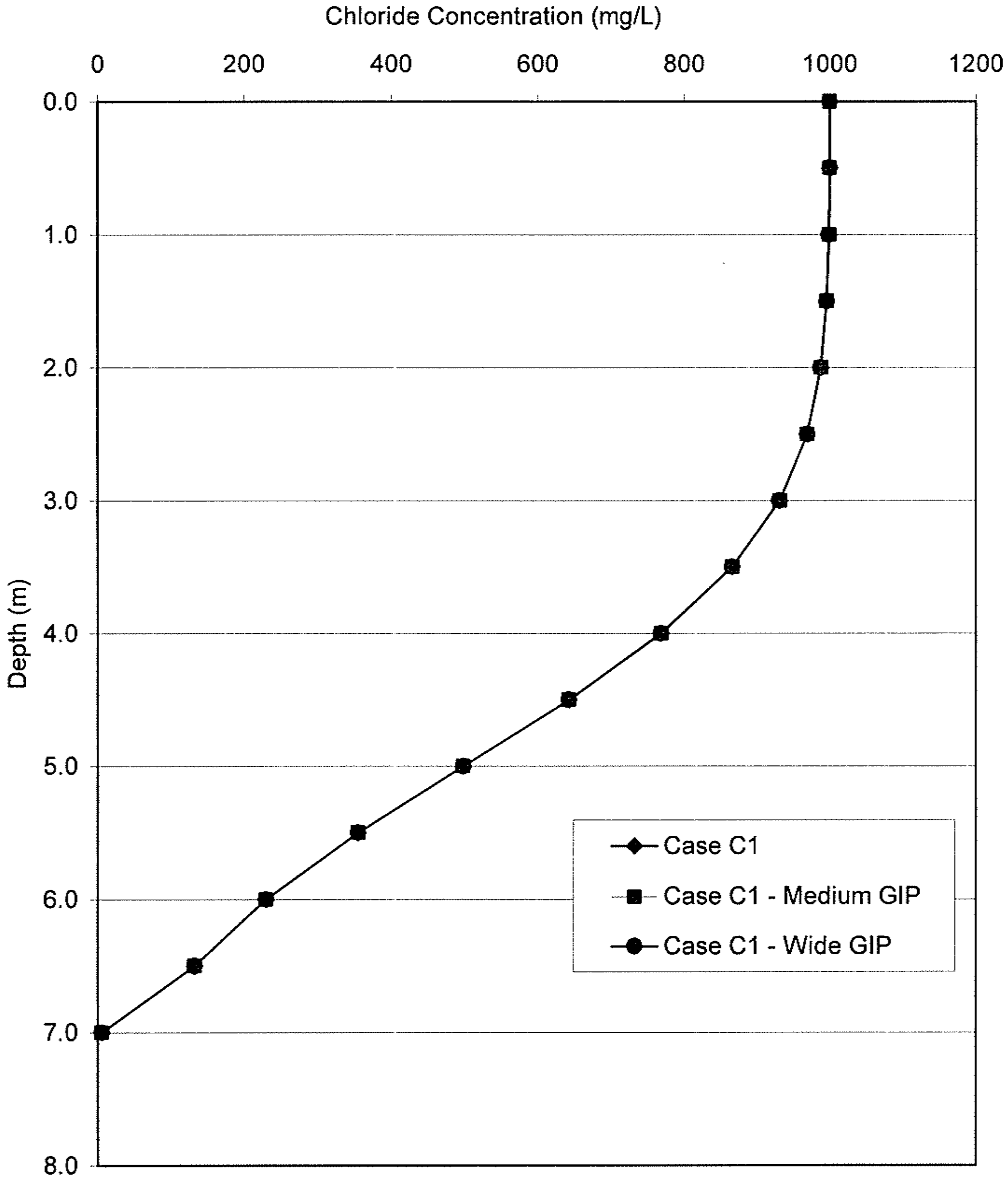
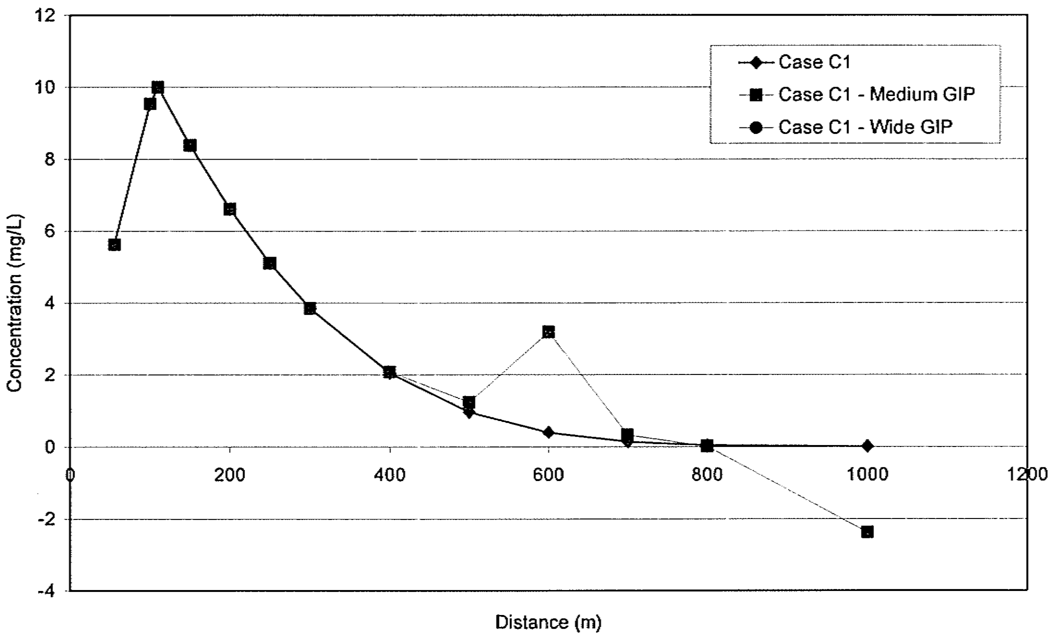
APPENDIX A

SENSITIVITY ANALYSIS FOR NUMERICAL STABILITY CHECK

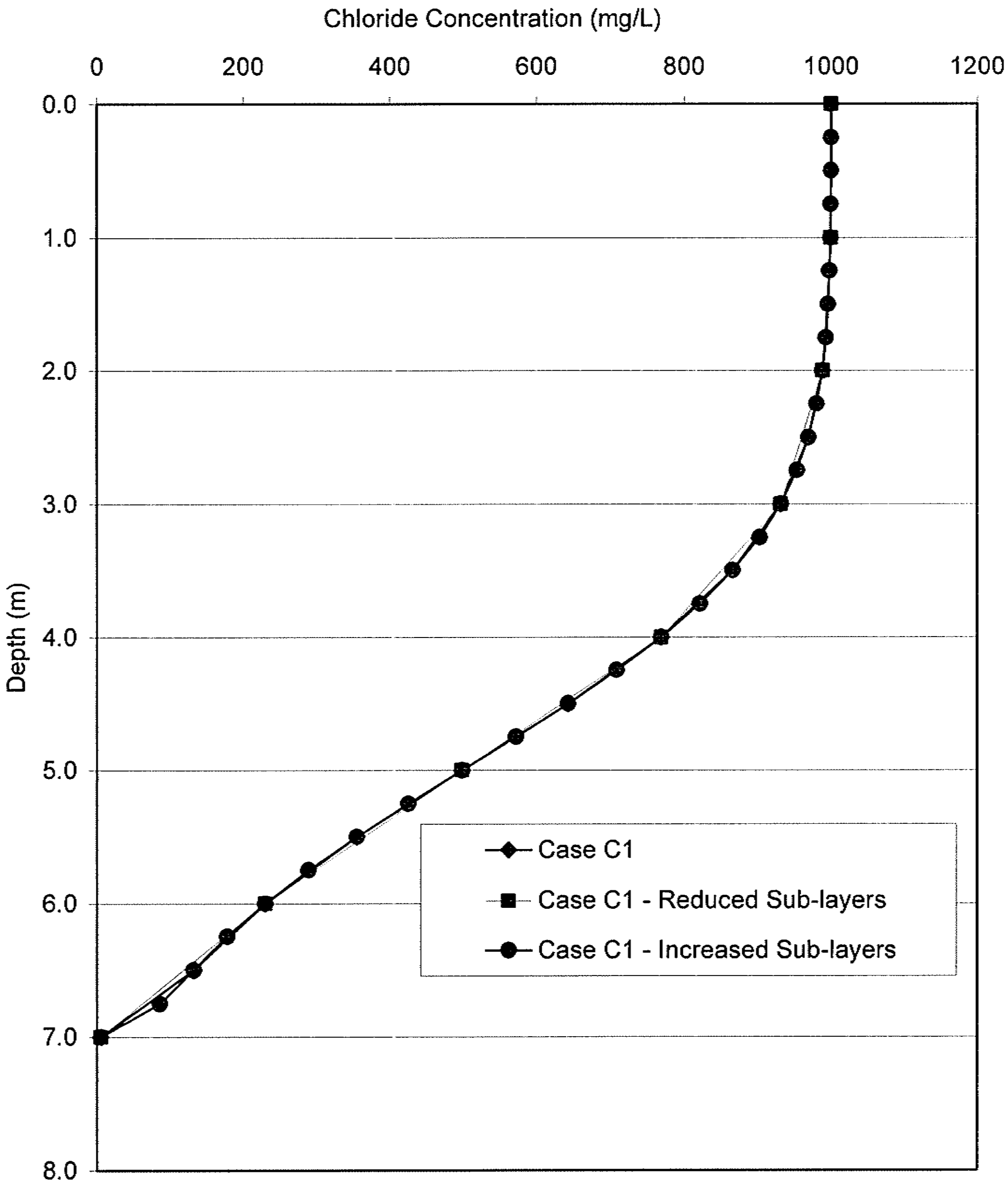
30 Year EMS Centreline Chloride Concentrations

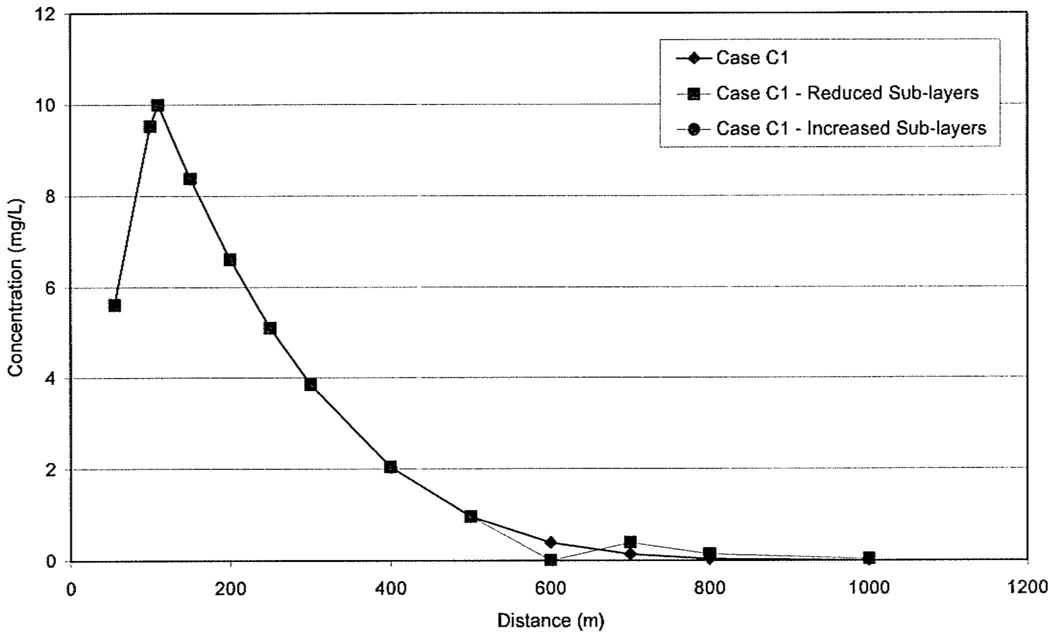


30 Year Aquifer Chloride Concentrations



30 Year EMS Centreline Chloride Concentrations




30 Year Aquifer Chloride Concentrations