# IMPACT OF MANURE COLLECTION AND STORAGE FACILITIES ON GROUNDWATER IN ALBERTA

August 21, 2007

Government of Alberta

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# ON GROUNDWATER IN ALBERTA

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**August 21, 2007**

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## EXECUTIVE SUMMARY

Improper management of manure and wastewater from confined feeding operations (CFO) can have negative impacts on both the environment and public health. This manure and wastewater has the potential to contribute pollutants to the groundwater environment. Generally, the primary pollutants associated with animal wastes that have the potential to affect groundwater include nitrogen compounds, phosphorus, and organic matter, as well as possibly antibiotics, pathogens, pesticides, and hormones.

In January 2002 the Alberta Provincial Government assumed responsibility for the regulation of CFOs when they amended the Agriculture Operation Practices Act (AOPA). The Natural Resources Conservation Board (NRCB) is the Provincial agency responsible for the administration of the AOPA. The AOPA specifies that the NRCB must administer the permits previously issued by the local health authorities and municipalities for CFOs. Consistent with the regulations at the time, many of the permits were issued without storage unit construction or design standards. The NRCB and Alberta Agriculture and Food are aware that some storage facilities are releasing manure constituents into shallow groundwater resources but are uncertain of the extent and the risk these releases are and could be having on the groundwater environment.

The objective of this project was to complete a comprehensive literature review that assesses the current impact that manure collection and storage facilities used at CFOs have on groundwater quality within the major livestock producing regions of North America, and to identify protocols to monitor the impacts that manure stored at manure collection and storage facilities may be having on groundwater quality and the environment.

This report provides three perspectives on the issues associated with CFOs and groundwater in Alberta: a review of comparative regulations throughout North America, a review of scientific literature on CFOs and groundwater in Alberta and North America, and illustrative groundwater flow and transport modeling of typical CFO environments (as a way of understanding the extent of present and future groundwater impacts). Following these perspectives, key areas for further research are highlighted. The report concludes with recommendations for groundwater monitoring in Alberta pertaining to:

(1) the regulations for liquid manure storage facilities that are deemed to pose a risk to the environment; (2) the performance of liquid and solid manure storage and collection facilities constructed to Alberta standards; and (3) assessment of the impact of seepage on surface and groundwater resources.

The literature review conducted for this report was extensive. In the case of the review of the Alberta data, all available refereed and "grey" literature (non-peer reviewed scientific journal publications) was used. Because associated literature for the rest of North America can be found in disparate sources (e.g., scientific literature, government reports, conference proceedings, web-based reports), efforts focused on the refereed literature with only limited use of the grey literature. A detailed list of key observations from the literature reviews and the numerical flow and transport modeling are presented. Key observations include:

1. Soil and groundwater contamination can occur from CFOs.
2. Hydrogeologic conditions that are sensitive to contamination include sites characterized by coarse grained soils (sands and gravels), shallow unconfined aquifers, or thin natural clay barriers overlying laterally extensive confined aquifers. The last of these conditions could also be seen to include coarse grained soil sites with a constructed engineered clay barrier or liner. Conditions that are more hydrogeologically stable are characterized by thick deposits of fine grained soils with high clay contents, deep and/or confined aquifers, and well designed and engineered waste storage sites.
3. In Alberta, the prevalence of relatively thick, clay till aquitards (fine grained soils) over much of the landscape, and the lack of extensive shallow, aquifer systems suggest that 'hydrogeologically stable' sites should be common.
4. The dominant contaminant in soil and soil pore-water regimes associated with CFOs is $NH_4$.
5. The migration of $NH_4$ can be retarded along the groundwater flow system.
6. In the case of permeable (sands and gravels) media, the retardation of NH4 appears limited. Seepage rates determined from a number of studies, field-based plume

studies, and numerical modeling of contaminant plume migration suggest the contaminants could migrate between 20 and 250 m from the CFO over 100 years.

7. The retardation of NH4 is greater in glacial tills and clays than in sandy media. Data suggest that over 100 years of use the NH4 plumes could migrate between 2 and 10 m from the CFO.
8. Nitrification of NH4 enriched soils (NH4 on the exchange sites) beneath abandoned earthen manure storage lagoon (EMS) sites has been demonstrated. This observation suggests a potentially large reservoir of oxidizable NH4 that may enter the groundwater regime at a later date (e.g., after site closure).

Additional research specific to conditions in Alberta is recommended. Key areas that require research include establishing a series of pilot projects focused on each of the various CFO types (swine, cattle, and poultry). These sites could be situated on representative hydrogeologic conditions in Alberta. The delineation of the extent of contamination requires an understanding of the effects of fracturing on contaminant migration and retardation of $NH_4$. These studies should include determining the controls exerted by fracturing in glacial tills on contaminant transport from CFOs as well as the rates and quantity of sorption (and desorption) of $NH_4$ on soils. In these studies, the migration of known contaminants such as N and poorly understood contaminants such as pharmaceuticals could be investigated.

Research should also be conducted on the effects of the decommissioning of CFOs on groundwater quality because the potential exists for the release and transformation of $NH_4$ on the exchange sites upon decommissioning (resulting from changes in the hydrogeologic and geochemical regimes proximal to the CFO). Under hydrogeochemical conditions resulting from the decommissioning of CFOs, the $NH_4$ on the exchange sites could be released and enter the groundwater regime. As a result, there may be future impact(s) of CFO decommissioning on the environment.

Based on literature reviews and the illustrative groundwater modeling conducted for this report, several observations and comments are made with respect to the existing AOPA and implementing regulations for liquid manure storage facilities deemed to pose a risk to the environment. Although recommendations on aspects of monitoring such as construction and decommissioning of monitoring wells, sampling and analyses (including indicator and baseline parameters) are provided, this report supports the current approach. No specific guidance was provided with respect to the location of monitoring wells because the hydrogeology of each site is unique.

We recommend that the groundwater monitoring protocols described under the existing AOPA for studying the effects of liquid manure storage facilities deemed to pose a risk to the environment could be used to monitor the performance of liquid and solid manure storage and collection facilities constructed to the construction and performance standards specified in the AOPA. Further, we recommend the use of groundwater monitoring wells be augmented with core sample analyses.

Although available data suggest that the fine grained nature of much of the soils in Alberta may limit the extent of contamination in many cases, additional data are required to assess the impact that seepage from manure storage and collection facilities may have on surface and groundwater resources. To do so, a strategy is suggested to quantify the impacts of contaminants on these receptors. Because each study site is hydrogeologically unique, only a conceptual approach was presented. The approach employed at a specific site must consider specific subsurface conditions, and as such, will need to be tailored to suit the hydrogeology of each site. The conceptual study approach is as follows:

1. Identify a suite of potential long-term, high risk and typical (of various hydrogeologic settings) sites for consideration;
2. Using available data and a set of selection criteria, select a smaller number of representative sites for further investigation;
3. Develop site specific investigation plans for these representative sites;
4. Instrument, sample, and analyze data collected from each site; and
5. Determine large-scale implications of CFO seepage on surface and groundwater resources.

# 1. INTRODUCTION AND PROJECT OVERVIEW

## 1.1. Purpose

In January 2002 the Alberta Provincial Government assumed responsibility for the regulation of confined feeding operations (CFO) when they amended the Agriculture Operation Practices Act (AOPA). The Natural Resources Conservation Board (NRCB) is the Provincial agency responsible for the administration of the AOPA. The AOPA specifies that the NRCB must administer the permits previously issued by the local health authorities and municipalities for CFOs. Consistent with the regulations at the time, many of those permits were issued without storage unit construction or design standards. The NRCB and Alberta Agriculture and Food are aware that some storage facilities are releasing manure constituents into shallow groundwater resources but are uncertain of the extent and the risk these releases are and could be having on the groundwater environment.

The objective of this project is to complete a comprehensive literature review that:

- Assesses the current impact that manure collection and storage facilities used at CFOs have on groundwater quality within the major livestock producing regions of North America, and
- Identifies protocols to monitor the impacts that manure stored at manure collection and storage facilities may be having on groundwater quality and the environment.

To assist with the identification of protocols to monitor the impacts of manure located at manure collection and storage facilities under Alberta conditions, “hypothetical” groundwater flow and solute transport models were applied over a range of groundwater and geochemical conditions that were considered representative of those in Alberta. Results of these modeling exercises must, however, be considered illustrative and should not be used to design monitoring strategies since the hydrologic and geochemical conditions of each CFO must be considered unique.

## 1.2. The Problem

Improper management of manure and wastewater from CFOs can have negative impacts on both the environment and public health. This manure and wastewater has the potential to contribute pollutants to the groundwater environment. Generally, the primary pollutants associated with animal wastes that have the potential to affect groundwater include nitrogen compounds, phosphorus, organic matter, and to a lesser extent antibiotics, pathogens, pesticides, and hormones [*c.f.*, San Jose State University Foundation, 2004].

Comparison of water chemistry data associated with CFOs (presented in Chapters 5 and 6) to the maximum acceptable concentration (MAC) suggests that nitrate ($NO_3$) is a contaminant of concern with respect to groundwater associated with CFOs. As established by the Federal-Provincial-Territorial Committee on Drinking Water, the MAC for nitrate in drinking water is 45 mg/L (or 10 mg/L nitrate-nitrogen ($NO_3$-N)) [Guidelines for Canadian Drinking Water Quality, 2006].

The MAC for $NO_3$ in drinking water of 45 mg/L was derived based on the no-observed-adverse-effect level (NOAEL) for infantile methemoglobinemia (cyanosis or "blue baby syndrome") of 45 mg/L. Between 1945 and 1970, some 2000 cases of methemoglobinemia in infants were reported in the world literature [Shuval and Gruener, 1972]. Although the MAC for $NO_3$ is based principally on effects in the most sensitive subgroup (i.e., infants), minimizing exposure of the entire population to nitrate is considered prudent owing to suggestive (equivocal) evidence of an association in several populations between gastric cancer and moderate levels of nitrate in drinking water. The MAC is therefore intended to apply to both children and adults. Recommendations for $NO_3$ in drinking water for mature livestock are commonly < 100 mg/L [Alberta Environment, 1999] with recommendations for young animals similar to those for infants.

On the basis of available data, concentrations of $NO_3$ in waters associated with CFOs are low (data presented in Chapters 5 and 6). These concentrations can, however, be greatly increased via nitrification of elevated concentrations of $NH_4$ often associated with CFOs. On the basis of available data, concentrations of $NH_4$ in waters associated with CFOs often range from a few tens to several thousand mg $NH_4$-N/L (data presented in Chapters 5 and 6).

Phosphorous (P) is also present in elevated concentrations in waters associated with CFOs. Concentrations of P in waters associated with CFOs often range from a few tens to a few thousand mg Total P/L (data presented in Chapters 5 and 6). P is not included in the Guideline for Canadian Drinking Water Quality [2006]. It is, however, defined as a chronic nutrient in surface waters in Alberta when present in concentrations (as P) greater than 0.05 mg/L [Alberta Environment, 1999] and may be a threat to surface waters at concentrations exceeding 100 parts per billion (μg/L) [US EPA Office of Water, 1997].

Fecal bacteria in livestock waste can contaminate groundwater if waste seeps into nearby wells, causing such infectious diseases as dysentery, typhoid and hepatitis. Organic materials, which may lend an undesirable taste and odor to drinking water, are not known to be dangerous to health, but their presence can suggest that other contaminants are flowing directly into groundwater.

## 1.3. Scope of Work

The scope of work for this project included a review of relevant regulations, guidance, and technical literature relating to the impacts of CFOs on groundwater resources. The project also includes a review of existing groundwater monitoring programs intended to identify impacts from manure collection and storage. Finally, recommendations are provided for groundwater monitoring in Alberta pertaining to: (1) the regulations for liquid manure storage facilities that are deemed to pose a risk to the environment; (2) the performance of liquid and solid manure storage and collection facilities constructed to Alberta standards; and (3) assessment of the impact of seepage on surface and groundwater resources. These recommendations are presented based on a review of existing data, scientific implications of current manure collection and storage practices, other monitoring programs, and hypothetical hydrogeologic modeling.

The literature review conducted for this report was extensive. In the case of the review of the Alberta data, all available refereed and "grey" literature (non-peer reviewed scientific journal publications) was used. Because associated literature for the rest of North America can be found in disparate sources (e.g., scientific literature, government reports, conference proceedings, web-based reports), we focused our efforts on the refereed literature and only used the grey literature to a limited extent.

The scope of work was limited to an analysis of groundwater monitoring at manure storage and collection facilities. Other methods for management of the risk of groundwater contamination, such as unit construction and design standards, the use of tile drains, comprehensive nutrient management plans, siting requirements, and other technical or regulatory requirements designed to limit impacts to groundwater from surface water runoff, were outside the scope of this report.

Further, although the impact of manure spreading is a groundwater contamination issue and extensive reports and papers have been prepared on this issue, it too is outside the scope of this report.

Although the extent of contamination in the post-closure environment is addressed in the literature review and modeling exercises, this report does not comment on post-closure care nor provide recommendations on how to address this issue.

## 1.4. Terminology and Definitions

Definitions of the key terms used in this report were taken or adapted from the AOPA and its implementing regulations [Alberta Standards and Administration Regulation, 2006]. The definition of some terms, such as "livestock", were altered slightly to conform to the scope of this project. Word choices are intended to be consistent throughout this report. For example, the acronym "CFO" is always intended to apply to confined feeding operations as defined by the AOPA.

In the discussion of other jurisdictions, words and phrases may differ from those used in Alberta; these alternative definitions are provided when used. For example, the United States federal regulations apply to "Concentrated Animal Feeding Operations" or CAFOs. In California, the regulations apply to "Animal Feeding Operations" or AFOs. In all other cases, and wherever they appear in this report, the acronyms and descriptions of operations are intended to have the meanings adapted from the AOPA and its implementing regulations as set out below.

**Confined feeding operation (CFO)** means fenced or enclosed land or buildings where livestock are confined for the purpose of growing, sustaining, finishing or breeding by means other than grazing and any other building or structure directly related to that purpose but does not include residences, livestock seasonal feeding and bedding sites, equestrian stables, auction markets, race tracks or exhibition grounds.

**Livestock** means poultry, cattle, or swine.

**Manure** means livestock excreta, associated feed losses, bedding, litter, soil and wash water.

**Manure collection area** mean the floor of a barn, the under-floor pits of a barn, the floor of a feedlot pen and a catch basin where manure collects but does not include the floor of a livestock corral.

**Manure storage facility** means a facility for the storage of manure, composting materials and compost and a facility for composting but does not include such a facility at an equestrian stable, an auction market, a race track or exhibition grounds.

**Groundwater resource** means an aquifer below the site of a confined feeding operation or a manure storage facility that is being used as a water supply for the purposes of domestic use, or if no aquifer exists that is being used as a water supply for domestic use, an aquifer that has sustained yield of 0.76 L/min or more and a total dissolved solids concentration of 4000 mg/L or less as determined by well records, well drilling logs, hydrogeological maps, hydrogeological reports or other evidence satisfactory to an approval officer or the Board or, if there is more than one aquifer that meets these requirements, the aquifer that an approval officer or the Board considers to be the best suited for development as a water supply for the purposes of domestic use.

**Liner** means, with respect to a manure storage facility or manure collection area, a layer constructed out of natural or manufactured materials that restricts the migration of the contents of the manure storage facility or manure collection area.

**Protective layer** means, with respect to a manure storage facility or manure collection area, one or more layers of naturally occurring materials that, individually or in the aggregate, restrict the migration of the contents of the manure storage facility or manure collection area.

**Solid manure** means manure that is 20% or more solid matter and that does not flow when piled.

## 1.5. Organization of Report

The report provides three perspectives on the issues associated with CFOs and groundwater. This includes a review of comparative regulations, a scientific literature review, and illustrative modeling of typical CFO environments as a way of understanding the extent of (present and future) groundwater impacts. The report concludes with recommendations for groundwater monitoring in Alberta pertaining to: (1) the regulations for liquid manure storage facilities that are deemed to pose a risk to the environment; (2) the performance of liquid and solid manure storage and collection facilities constructed to Alberta standards; and (3) assessment of the impact of seepage on surface and groundwater resources.

Chapter 1 describes the goals of the report and the scope of issues to be considered.

Chapter 2 presents a brief overview of the distribution of CFOs in North America and the potential for environmental impacts from those operations.

Chapter 3 summarizes the existing groundwater monitoring program in Alberta and compares that program to other jurisdictions in Canada, the United States and Europe.

Chapter 4 completes the background exploration of the potential impacts of CFO contaminants on groundwater by describing the underlying chemical controls on those impacts.

Chapters 5 and 6 detail the current understanding of groundwater contamination from CFOs in Alberta and in other areas of North America, respectively. This includes an extensive literature review of research including an overview of the specific geology of the primary CFO regions, reported groundwater impacts, and an evaluation of potential conditions that may be consistently present at sites with groundwater impacts. This literature review includes research on livestock, swine and poultry and evaluates conditions in Alberta and other areas of Canada and the United States. The contaminants of concern include $NO_3$, P, as well as pathogens and pharmaceuticals.

Chapter 7 completes the third part of the analysis by presenting groundwater modeling results that could be used as predictive tools to understand and anticipate risks from CFOs under conditions in Alberta. This include modeling of lateral flow of conservative and reactive contaminants in unconfined sand aquifers and unconfined oxidized till.

Chapter 8 provides a summary of the literature and modeling results.

Chapter 9 details areas for further research.

Chapter 10 summarizes and provides recommendations for consideration regarding the monitoring requirements for liquid manure storage facilities that are deemed a risk to the environment.

Chapter 11 presents recommendations for groundwater monitoring protocols to be considered for monitoring the performance of liquid and solid manure storage and collection facilities constructed to the construction and performance standards specified in the Agricultural Operation Practices Act Standards and Administration Regulation.

Chapter 12 outlines a study approach to quantify the impacts of contaminants on receptors.

## 1.6. References

Alberta Environment, 1999, Surface water quality guidelines for use in Alberta, Environmental Assurance Division. Publication No. T/483, http://environment.gov.ab.ca/info/library/5713.pdf, November 1999.

Alberta Standards and Administration Regulation 267/2001 consolidated up to 306/2006.

Guidelines for Canadian Drinking Water Quality, 2006; www.healthcanada.gc.ca/waterquality.

San Jose State University Foundation, 2004, Review of animal waste management regulations: Task 3 report – Comparison of regulations assigned to protect groundwater quality from releases of confined animal facilities, 50 pp.

Shuval, H.I. and Gruener, 1972, Epidemiological and toxicological aspects of nitrates and nitrites in the environment: Am. J. Public Health, 62, 1045.

US EPA Office of Water, 1997, Voluntary Stream Monitoring: A Methods Manual, EPA 841-B-97-003.

## 2. Distribution of Livestock Operations

Statistics Canada conducts an annual inventory of Canadian agricultural operations. According to Statistics Canada [2004], more than three-quarters of Canada's dairy farms and dairy cattle are located in Quebec and Ontario. In addition, Alberta has almost half of the national beef cattle herd, and close to one-third of the beef farms. Alberta, Saskatchewan, Manitoba, and Ontario together account for over 80% of Canada's beef farms and beef cattle. In 1991, the average beef cattle farm had 115 head; 10 years later, the average was 163. Most of the growth was in Alberta, which had 1.8 million more cattle in 2001 than in 1991.

According to Statistics Canada [2004], the number of pigs is increasing: 14.0 million in 2001, up 37% from 1991. A few larger producers started up during the 1990s, some producers expanded, and some smaller operations went out of business. 14,000 fewer farms reported pigs in 2001 than in 1991. The average hog farm nearly tripled in size during that period, to 902 animals. Quebec and Ontario had more than half of all the hogs in Canada.

The United States Department of Agriculture [2001] collected similar statistics. In the United States, the number of potential CAFO operations more than doubled from 1982 to 1997, increasing from about 5,000 to 11,200 (126 percent), or from 1 to 5 percent of all operations. During the same period, the number of animal units (AU) on these farms almost doubled from 9.1 million (30 percent of total confined AU) to 18.0 million (54 percent). Nationally, the average number of AU on each potential CAFO did not increase over the period; the gain in AU on potential CAFO farms was due entirely to the increase in the number of potential CAFO operations. The distribution of potential CAFO farms by animal type underwent substantial change from 1982 to 1997. The share of feedlot beef operations declined from 47 to 17 percent of potential CAFO farms, and swine and poultry experienced growth, from 21 to 39 percent and 24 to 33 percent, respectively. The poultry sector experienced the smallest decline in farm numbers over 1982-97, and again, smaller farms dominated; almost 90 percent of confined poultry farms had fewer than 300 AU. The greatest numbers of confined animals are located in a band from southeastern New Mexico through the Plains States to eastern Nebraska and then eastward through Iowa to the Great Lakes. Other areas with large numbers of confined animals include the Northeast, mid-Atlantic, California's southern Central Valley, western Arkansas, and far Northwest areas. Almost every State has at least 1 county with more than 10,000 animal units.

Figures 2.1 to 2.3 provide an overview of the locations of livestock density on farmland in Alberta (Figure 2.1 and 2.2) and the adjacent provinces of Saskatchewan (Figure 2.2) and British Columbia (Figure 2.3). The majority of CFOs in Alberta are located in a corridor between Lethbridge and Edmonton. Three areas of concentrated CFOs exist in Alberta: around Lethbridge, between Calgary and Edmonton, and north of Edmonton. In Saskatchewan, the distribution is relatively uniform across the southern part of the province. The dominant concentration of CFOs in British Columbia is in the Vancouver-Abbotsford region.

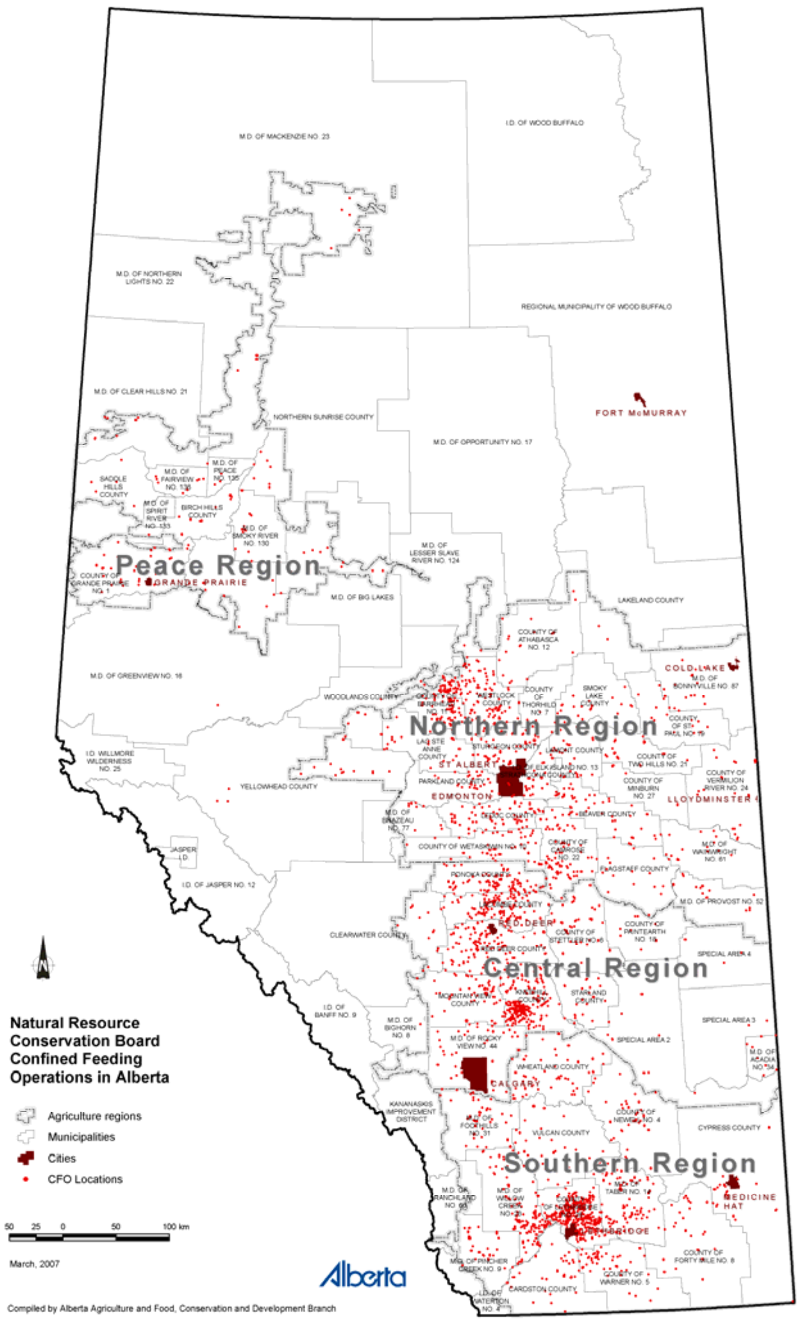


**Figure 2.1. Confined feeding operations in Alberta [Alberta Agriculture and Food, 2007].**

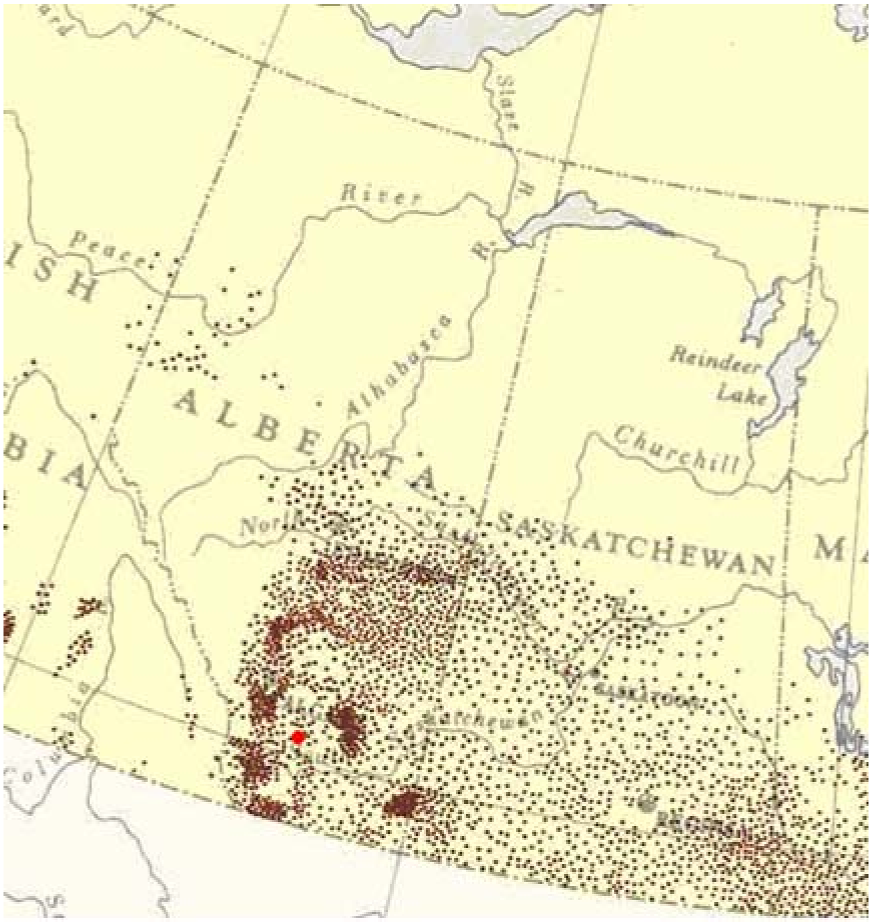


**Figure 2.2. Livestock Density on Farmland Alberta and Saskatchewan (from T. Fonstad).**

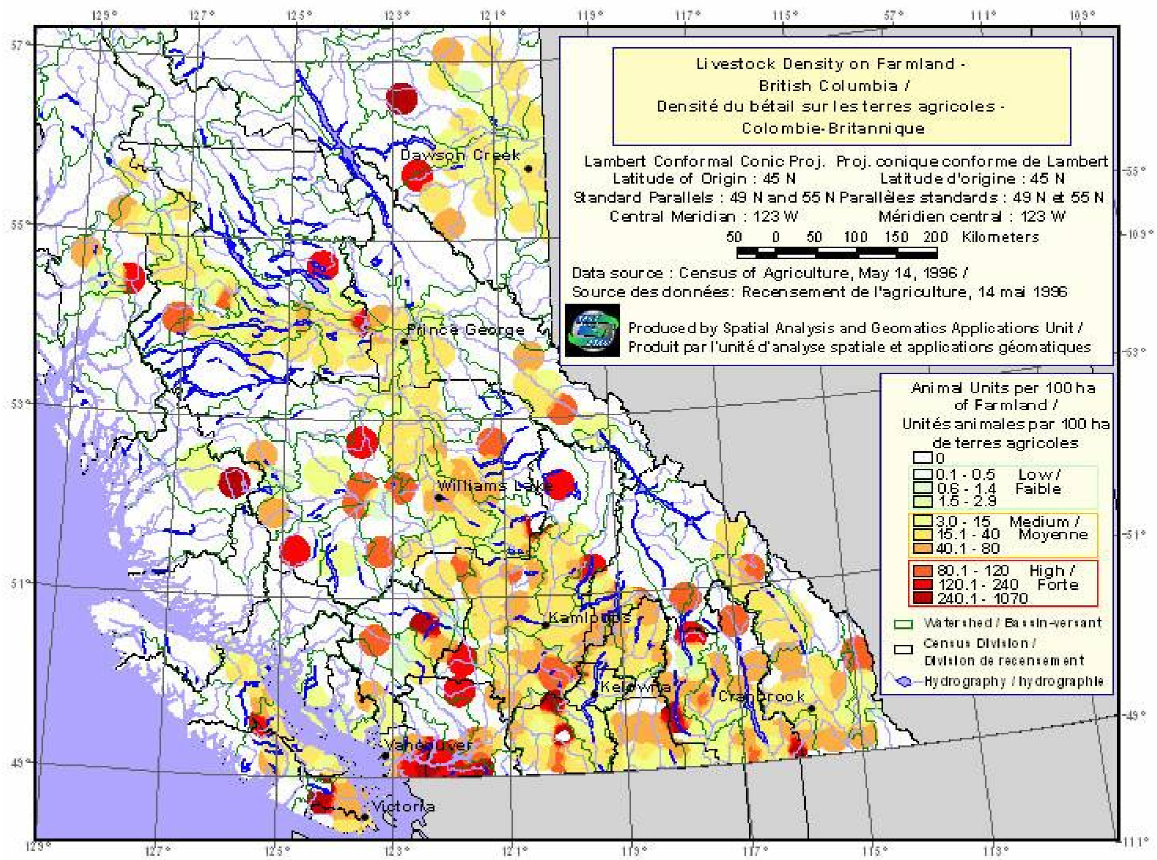


**Figure 2.3. Livestock Density on Farmland in British Columbia [Statistics Canada, 1996].**

## 2.1. References


Alberta Agriculture and Food, 2007, Conservation and Development Branch, March.

Statistics Canada, 1996, Census of Agriculture.

Statistics Canada, 2004, Canadian Agriculture at a Glance 96-325-XPB, June, 2004.

United States Department of Agriculture (USDA), 2001, Confined Animal Production and Manure Nutrients (Chapter 3), 2001-06-26, http://www.ers.usda.gov/publications/aib771/aib771e.pdf.

# 3. OVERVIEW OF REGULATORY FRAMEWORK

## 3.1. Alberta

### 3.1.1. Overview

As noted in Chapter 1, CFOs in Alberta are regulated by the NRCB under the authority of the AOPA. The AOPA defines a CFO as a:

> "fenced or enclosed land or buildings where livestock are confined for the purpose of growing, sustaining, finishing or breeding by means other than grazing and any other building or structure directly related to that purpose but does not include residences, livestock seasonal feeding and bedding sites, equestrian stables, auction markets, race tracks or exhibition grounds." [Alberta AOPA, 2007; s.1]

The requirements of the act are implemented through the Standards and Administration Regulation (306/2006) [2007] (hereafter "the Regulations"). Various guidance documents have also been developed to "provide further clarification and direction" including *Leak Detection and CFOs* (TG 2004-01; NRCB [2004a]) and *Concrete Manure Liner Guidelines* (TG-2004-02; NRCB [2004b]). These guidelines were developed after consultation with industry and the public.

Under the existing regulatory program, all new and expanding CFOs and certain manure storage facilities and collection areas are subject to approval or registration requirements [Regulations, 2007; ss. 13, 14]. Generally, CFOs or manure storage facilities that existed on January 1, 2002 would be "deemed to have been issued an approval, registration or authorization under the Act" [Regulations, 2007; s. 18.1]. The NRCB is responsible for enforcing any prior approval, registration or authorization issued by any other jurisdiction and for issuing authorizations for any new or expanding facility.

### 3.1.2. Groundwater Monitoring Wells

Two provisions of the Regulations relate to groundwater monitoring at CFOs. Under Section 7(1), a manure storage facility or a manure collection area must be constructed more than 100 m from a spring, 100 m from a water well, and 30 m from a common body of water. The siting requirements for springs and water wells do not apply if the owner or operator demonstrates than an aquifer from which the spring rises or into which the water well is drilled is not likely to be contaminated by the facility and, if required by the approval officer or the Board, implements a groundwater monitoring program at the site [Regulations, 2007; s.7.2].

Under Section 18, if the NRCB determines that a risk to the environment exists, the approval office may require the owner or operator of a liquid manure storage facility to install and maintain a leakage detection system for the liquid manure storage facility consisting of at least one monitoring well upgradient of the facility and at least 2 monitoring wells downgradient from the facility. The wells must be monitored on a schedule determined by the NRCB. The Section 18 requirements are the focus of this review.

As noted, Section 18 of the Regulations authorizes the NRCB to impose groundwater monitoring requirements in certain cases. According to TG 2004-01 *Leak Detection and CFOs* [NRCB, 2004a], the upgradient and downgradient monitoring wells required under s. 18 of the Regulations should be sited as directed by the AOPA and/or as directed by a qualified professional (e.g., geologist, hydrogeologists, engineer) and approved by the NRCB. Construction of the wells must be overseen by the qualified professional and the NRCB must be notified prior to any changes to the system.

### 3.1.3. Monitoring

The s. 18 CFO groundwater monitoring program consists of two phases. In the first phase, Baseline Monitoring must be conducted twice during the first year of operation. This sampling is recommended to be conducted in the spring and fall or as directed by the NRCB. One set of samples should be taken prior to the operation of the unit. In the second phase, Indicator Monitoring should be conducted annually or biennially as directed by the NRCB on the same schedule (spring or fall) as the Baseline Monitoring. If the Indicator Monitoring demonstrates an increase in any of the parameters, the NRCB will determine what additional evaluations or supplemental monitoring should be conducted. Monitoring results must be reported as specified in the approval, registration or authorization or as directed by the NRCB.

***Table 3.1. Monitoring Parameters (TG 2004-01; NRCB [2004a]).***

| Monitoring Parameters | Baseline | Indicator | Supplemental |
|---|---|---|---|
| Potassium ($K^+$) | √ | √ | |
| Chloride ($Cl^-$) | √ | √ | |
| Nitrate + Nitrite ($NO_3^- + NO_2^-$) | √ | √ | |
| Total Kjeldahl Nitrogen (TKN) | √ | √ | |
| Total Dissolved Phosphorous (TDP) | √ | √ | |
| Electrical Conductivity (EC) | √ | √ | |
| Sodium ($Na^+$) | √ | | |
| Calcium ($Ca^{+2}$) | √ | | |
| Magnesium ($Mg^{+2}$) | √ | | |
| Bicarbonate ($HCO_3$) | √ | | |
| Sulphate ($SO_4^{-2}$) | √ | | |
| Dissolved Organic Carbon (DOC) | √ | | |
| pH | √ | | |
| Total Dissolved Solids | √ | | |
| E. coli | | | √ |
| Total coliforms | | | √ |

## 3.2. Groundwater Quality Monitoring Programs for CFOs in Other Jurisdictions

### 3.2.1. Quebec, Ontario, British Columbia, Saskatchewan, Manitoba

Neither Quebec nor Ontario currently requires groundwater monitoring for CFOs. In Saskatchewan, intensive livestock operations may be required to implement a water quality monitoring program as a special condition of the government's approval of the project. In Saskatchewan, monitoring is only required in situations where specific risks exist (such as a shallow aquifer) [Saskatchewan Agricultural Operations Act, 1995].

British Columbia provides that “agricultural waste must not be directly discharged into a watercourse or groundwater” but has not implemented specific regulations for groundwater monitoring [s. 11, British Columbia Agricultural Waste Control Regulation, 321/2004].

In Manitoba, the province may require the operator of a manure storage facility to install monitoring wells or may require monitoring if the Director determines that the storage, handling and management of livestock manure in the agricultural operation is causing or would likely cause pollution of surface water, groundwater or soil. The Director will approve a sampling plan, and results collected in accordance with that plan must be submitted annually [s. 6.1, Manitoba Livestock Manure and Mortalities Management Regulation 42/98]. Manitoba has also issued a Monitoring Well Sampling Information Sheet for Manure Storage Facilities that sets out suggested sampling protocols (i.e., number of bottles, well purging procedures, etc.).

### 3.2.2. United States Federal Regulations

In the United States, the United States Environmental Protection Agency (EPA) regulates environmental emissions and generally may delegate that authority to the states. The EPA has implemented regulations to address pollution caused by what it has defined as Concentrated Animal Feeding Operations (CAFOs). A CAFO is defined by the EPA to mean agricultural operations where animals are kept and raised in confined situations for at least 45 days in a 12-month period, and there is no grass or other vegetation in the confinement area during the normal growing season [United States 40 Code of Federal Regulations (CFR), 2007; Part 122.23(b)]. The federal regulation also states that no animal feeding operation is a CAFO if such animal feeding operation discharges only in the event of a 25-year, 24-hour storm event [40 CFR, 2007; Part 122 Appendix B].

The EPA initially issued national effluent limitations guidelines and standards for feedlots on February 14, 1974 [39 Fed Reg 5704, 1974] and permitting regulations on March 18, 1976 [41 Fed Reg 11458, 1976]. In February 2003, EPA issued revisions to these regulations [68 Fed. Reg. 7176, 2003]. The 2003 CAFO rule required the owners or operators of all CAFOs to seek coverage under a discharge permit, comply with technical design and construction standards as applicable, and prepare a nutrient management plan. The final rule did not include groundwater monitoring. The EPA believes that the rule:

> "is expected to reduce nitrate levels in private drinking wells by reducing the rate at which manure is spread on cropland, thus reducing the rate at which pollutants will leach through soils and reach groundwater...Based on [US Geological Survey data],

> EPA estimates that 9.2 percent of households that currently rely on private wells with nitrate concentrations exceeding the [Safe Drinking Water Maximum Contaminant Level (MCL)] will have these concentrations reduced to levels below the MCL because of the effluent limitation guidelines for Large CAFOs.” [68 Fed Reg 7176, 7241, 2003].

### 3.2.3. Various State Regulations

Selected state regulatory programs were reviewed to identify groundwater monitoring programs for CFOs. These include the high plains states (Kansas, Iowa, Nebraska, North Dakota, Oklahoma, South Dakota, Montana, and Minnesota) and sample large producer states (California, Missouri, North Carolina, New Mexico and Texas.) More than 20 percent of the CFOs in the United States are located in Iowa, Kansas, Nebraska and Missouri. These states have 985 or 56% of the beef cattle CFOs in the country, and 1,131 or 29% of the hog CFOs. Kansas, Nebraska, and Iowa are second, third, and fifth, respectively, in fed cattle production in the nation; 44% of the country's cattle are fed in these states.

California has approximately 2,200 dairies with an average size of about 700 milk cows. Several hundred feedlots, poultry operations, and other animal feeding operations also exist in the State. The exact number of CFOs in California is unknown but is estimated at between 1,000 and 1,200.

North Carolina was also selected for review because of the historical significant environmental impacts from CFOs in that state. North Carolina also led the nation in identifying the scope of the issues associated with CFOs, and in developing comprehensive regulations.

As noted, the EPA may authorize states to implement federal water quality programs. The authorized states were required to adopt regulations similar or more stringent than the federal regulations by 2005. As of the date of this report, all states are authorized with the exception of New Mexico, Alaska, Idaho, New Hampshire, and Massachusetts. The federal regulations apply in the non-authorized states.

Generally, most state programs reviewed are similar in scope to the federal program. In IA, KS, MN, MT, ND, SD, and TX, the states impose requirements for permitting (typically construction and operation permits are required), for construction design of the manure management facilities, and for manure management or manure application plans. In the states that adopted programs similar to the federal regulations, no groundwater monitoring is required although stringent regulations are in place relating to surface water runoff and discharges to surface water that might impact surface and groundwater. Other states, as summarized below, impose some form of groundwater monitoring.

### 3.2.4. California

Water quality in California is regulated on both a state and regional basis. Most livestock operations are dairy, and most are located in the Central Valley Region. In response to the recent remand of several aspects of the federal regulations, the Central Valley Region is developing a general Waste Discharge Requirements (WDR) Order to regulate the dairies. The WDR Order is essentially a state permit, and a tentative version of the order is under consideration as of December 2006. The Order is expected to include groundwater monitoring, however the current draft version of the permit only includes the following language:

> "D. Discharge of waste at existing milk cow dairies shall not cause the underlying groundwater to be further degraded, to exceed water quality objectives, unreasonably affect beneficial uses, or cause a condition of pollution or nuisance. The appropriate water quality objectives are summarized in the Information Sheet, which is attached to and part of this Order, and can be found in the Central Valley Water Board's Water Quality Control Plan for the Sacramento and San Joaquin River Basins (4th Ed.) and the Water Quality Control Plan for the Tulare Lake Basin (2nd Ed.) [California Regional Water Quality Control Board Central Valley Region, 2006].

### 3.2.5. Nebraska

The Nebraska regulations present the most comprehensive regulatory program of the state programs reviewed. In Nebraska, the Department of Environmental Quality may require groundwater monitoring for any large concentrated animal feeding operation based on a site-specific review. The following information will be considered in that review:

- The materials and methods used in the construction of the facility;
- The size of the animal feeding operation;
- Depth to groundwater;
- Type of soils;
- Type of consolidated or unconsolidated sediments above and below the water table;
- Local and regional use of ground water for drinking water and other beneficial uses; and
- Other criteria, including but not limited to location of nearest public water supply wells, use of local Rural Water District, and location of on-site wells [N.A.R. Title 130, Ch. 13 (001)]

Groundwater monitoring may be required for any small or medium animal feeding operation if any one of the following has occurred:

- A spill or non-permitted release from the facility;
- The Department determines that percolation from the facility exceeds the allowable percolation rate; or
- Any other circumstance that the Department determines may impact groundwater quality [N.A.R. Title 130, Ch 13 (002)]

The applicant may request reconsideration and may provide information such as geologic logs and static water levels in existing on-site wells [N.A.R. Title 130, Ch 13 (003)].

If groundwater monitoring is required, facilities must have a minimum of three monitoring wells, one upgradient and two downgradient, constructed in accordance with state well construction standards. The Department may approve alternative methods for monitoring the groundwater, sediments, and rocks above or below the water table including direct push techniques, or lysimeters [N.A.R. Title 130, Ch 13 (004)].

The applicant must prepare a Sampling and Analysis Plan for approval by the Department. Sampling must include nitrate, chloride, and ammonia and depth to water prior to sampling [N.A.R. Title 130, Ch 13 (005)]. The Department may also require water level measurements at a frequency adequate to establish seasonal groundwater flow directions.

Additional groundwater monitoring and investigation may be required in the event that contaminant concentrations exceed background, there is a discharge from the facility, the Department determines that percolation from the facility exceeds the allowable percolation rate, or any other circumstances that the Department determines may impact groundwater quality [N.A.R. Title 130, Ch 13 (008)].

### 3.2.6. Oklahoma

EPA and Oklahoma share jurisdiction for the regulation of CFOs. Under Oklahoma law, Licensed Managed Feeding Operations must install a leak detection system or monitoring wells in accordance with criteria approved by the state. All waste retention structures must have "sufficient numbers of groundwater monitoring wells upgradient and downgradient in the direction of groundwater flow." All monitoring well locations shall be approved by the state on a case-by-case basis. No monitoring well shall be installed more than one hundred and fifty (150) feet from the crown of the outer berm.

All new monitoring wells must be drilled through the first aquifer encountered, but need not extend more than fifty (50) feet below the bottom of the waste retention structure. One downgradient monitoring well shall be drilled to the first aquifer encountered or the first impermeable layer, but need not extend more than one hundred (100) feet below the bottom of the waste retention structure. All monitoring wells shall be drilled and completed by an Oklahoma Water Resources Board licensed monitoring well driller.

If no groundwater is encountered during the drilling operation, the bore hole shall be left open for at least forty eight (48) hours but not more than thirty (30) days for the aquifer to recharge the bore hole. Thereafter, the bore hole shall be either developed into a monitoring well or plugged according to Oklahoma Water Resources Board requirements.

All new monitoring wells shall meet the following minimum requirements:

(i) A minimum of two (2) inch diameter PVC casing shall be used with a sealing cap on the bottom.

(ii) The casing shall consist of minimum SDR-21 rated casing with a minimum SDR-21 rated factory screen in the saturated zone, or the bottom ten (10) feet if no groundwater is encountered.

(iii) Perforated zone shall be gravel or sand packed originating at the bottom of the screen and extending to two (2) feet above the top of the screen, and otherwise as appropriate for the installation.

(iv) Bentonite shall be placed in the annular space of the well above the gravel or sand pack for an interval of at least two (2) feet to form an impermeable seal.

(v) A cement grout or a mixture of bentonite and cement shall be placed above the bentonite seal to prevent seepage from entering behind the pipe and causing hydrologic connection.

(vi) At least the top ten (10) feet of the annular space shall be filled with type A cement.

(vii) A concrete apron, minimum of four (4) inch thickness and two (2) feet from the casing shall be installed at the surface to prevent seepage of rain water into the bore hole. The apron shall be sloping away from the casing to avoid percolation of rain water.

(viii) A lockable protective cap shall be placed on top of the casing, which shall be a metal protective casing extending two (2) feet above the concrete apron and one (1) foot into the apron. The well shall remain securely capped and locked at all times, except during sampling events.

Groundwater monitoring wells shall be sampled at least annually for electrical conductivity, pH, ammonium-nitrogen, nitrate-nitrogen, total phosphorus, and fecal coliform bacteria. [O.A.R 35:17-3-11(e)(6)].

### 3.2.7. North Carolina

North Carolina implements permitting and manure management regulations that are similar to the federal program.  However, North Carolina also has a separate groundwater protection program that can be used to impose groundwater monitoring where determined to be appropriate [15A NCAC 02H].

According to the North Carolina Division of Water Quality, Aquifer Protection Section, within the past five years, approximately twelve CFOs instituted groundwater monitoring. The monitoring did not display significant changes to groundwater attributed to the waste management activities and the monitoring programs were discontinued.  Those requirements were imposed through the following permit language:

> Groundwater Monitoring
>
> Monitor wells (MW-1 through MW-3) shall be sampled every March, July and November for the parameters listed below. The depth to water in each well shall be measured from the surveyed point on the top of the casing. The measuring points (top of well casing) of all monitoring wells shall be surveyed relative to a common datum. The water level shall be measured and recorded prior to purging the wells. The pH shall be measured and recorded prior to sampling for the remaining parameters:
>
> - Nitrate ($NO_3$-N)
> - Total Ammonia Nitrogen ($NH_3$-N)
> - Chloride
> - Total Dissolved Solids (TDS)
> - Fecal Coliform
> - pH (field)
> - Potassium
> - Water Level
> - Sodium
>
> Monitor wells (MW-1 through MW-3) shall be sampled upon permit renewal, for the parameters above and the additional parameters that follow. Any of the following

parameters that exceed state groundwater quality standard limits upon initial or renewal sampling must be added to the above list for routine monitoring:

- Copper
- Zinc
- Total Phosphorous

If concentrations for any single parameter, or any combination of parameters,  exceed state groundwater quality standards in any single monitor well, or any combination of monitor wells, for three consecutive monitoring periods, then monitor wells (MW-1 through MW-3) shall be sampled thereafter every March, July and November for all of the parameters listed above.

Any laboratory selected to analyze parameters must be certified by the Division of Water Quality (DWQ) for those parameters required.

### 3.2.8. New Mexico

In New Mexico, EPA and the New Mexico Environment Department (NMED) share jurisdiction for regulating CFOs.  New Mexico has not been delegated the waste water discharge permitting program and therefore the EPA issues CFO permits in accordance with the federal regulations in coordination with the state.  In addition to the federal permit, the state implements groundwater quality requirements through the Environment Department Groundwater Quality Board (GWQB).

The New Mexico regulations provide general prohibitions against unauthorized discharges in excess of stated limits [20.6.2.3100 et. seq. NMAC]. The GWQB currently only regulates dairy operations and does not impose requirements on feedlot operations. The following description of the New Mexico program therefore applies only to dairy operations.

Prior to discharging, an operator must install groundwater monitoring wells at the facility. Typically, according to GWQB staff, one well is located upgradient of the facility for purposes of establishing regional conditions, one is located within 50 feet downgradient of the lagoons, and one is located within 50 feet downgradient of the land application areas. Additional wells may be required based on site-specific conditions.  Prior to installation, the

GWQB must approve the locations of the wells. Groundwater monitoring wells must be completed in accordance with *Monitoring Well Construction and Abandonment Guidelines* [NMED, 2007], and construction and lithologic logs for the wells shall be submitted within 30 days of well completion.

All monitoring wells shall be surveyed to a common permanent benchmark, and depth to water must be measured to determine groundwater flow direction and gradient. Results must be submitted to the GWQB within 30 days of completion of the survey. If the survey indicates that the monitoring wells were not installed downgradient of the intended sites, well replacement or additional wells may be required.

Groundwater monitoring wells must be sampled prior to discharging and on a quarterly basis thereafter. Prior to sampling, depth to water shall be measured in each well, and groundwater shall be analyzed for nitrate-nitrogen ($NO_3$-N), total Kjeldahl nitrogen (TKN), Chloride (Cl), and total dissolved solids (TDS). Depth to water measurements and groundwater analytical results are due to the GWQB on a quarterly basis.

The GWQB requires quarterly, semi-annual, and annual monitoring reports. Reports submitted on a quarterly basis typically include groundwater monitoring analytical results and depth to water measurements in each well. Semi-annual reports include process wastewater analytical results, monthly meter readings of discharges to and from the lagoon, a log of all metered releases to each land application area, Land Application Data Sheets, and chemical fertilizer application logs. Annual reports include soil test analytical results and plant material results if required.

## 3.3. Research Institutions

The Natural Resources Conservation Service (NCRS) (formerly Soil Conservation Service) is a branch of the United States Department of Agriculture. This agency has as its primary mission to provide outreach and technical assistance to the agriculture industry and does not issue enforceable technical regulations. The agency has developed numerous guidance documents for the operation of manure storage and collection facilities and effective nutrient management but has not prepared guidance that recommends or describes groundwater monitoring.

## 3.4. European Union

Generally, the goals for groundwater quality are established by the European Union (EU) and implemented by the various Member States. For the EU as a whole, the main source of nitrogen input to agricultural land is mineral fertilizer, with livestock manure a close second. However, the situation varies considerably from one country to another. For example, in 1995, mineral fertilizers accounted for 50% or more of total nitrogen input in Denmark, Germany, Greece, France, Luxembourg, Finland and Sweden. In Belgium and the Netherlands, livestock manure was responsible for more than 50% of nitrogen inputs.

The EU's Nitrates Directive was introduced in 1991 with two main objectives: to reduce water pollution by nitrates from agricultural sources and to prevent further pollution. Specifically, the objective of the Community Directive on Nitrates [91/676/EEC] is to reduce water pollution caused or induced by nitrates from agricultural sources. The directive involves: monitoring of water quality in relation to agriculture; designation of nitrate vulnerable zones; establishment of (voluntary) codes of good agricultural practice and of (obligatory) measures to be implemented in action programs for the nitrate vulnerable zones. For these zones, the directive also establishes a maximum limit of nitrogen from livestock manure that can be applied per hectare of 170 kg N/ha per year.

Codes of good agricultural practice cover such activities as application periods, fertilizer use near watercourses and on slopes, manure storage methods, spreading methods and crop rotation, and other land management measures. Action programs must include obligatory measures concerning periods of prohibition of the application of certain types of fertilizer, capacity of manure storage vessels, limitations to the application of fertilizers (on steep slopes; to water-saturated, flooded, frozen or snow-covered ground; near water courses), as well as other measures set out in codes of good agricultural practice.

As of 2006, all the Member States have transposed the Directive, set up a monitoring network, drawn up a code of good practice and designated vulnerable zones (except Ireland). The impact of the Directive's implementation will only be felt in a few years' time, though it is believed that positive results are occurring in some regions.

On 23 October 2000, the "Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy", or the EU Water Framework Directive (WFD), was adopted. This comprehensive water quality Directive requires that all surface waters and groundwaters within defined river basin districts must reach at least 'good' status by 2015. It will do this for each river basin district by:

- Defining what is meant by 'good' status by setting environmental quality objectives for surface waters and groundwaters;
- Identifying in detail the characteristics of the river basin district, including the environmental impact of human activity;
- Assessing the present water quality in the river basin district;
- Undertaking an analysis of the significant water quality management issues;
- Identifying the pollution control measures required to achieve the environmental objectives;
- Consulting with interested parties about the pollution control measures, the costs involved and the benefits arising; and
- Implementing the agreed control measures, monitoring the improvements in water quality and reviewing progress and revising water management plans to achieve the quality objectives.

To implement the WFD, on September 19, 2003 the European Commission adopted a proposal for a new Directive to protect groundwater from pollution [COM(2003)(550)]. Based on an EU-wide approach, the proposed Directive introduced, for the first time, quality objectives, obliging Member States to monitor and assess groundwater quality on the basis of common criteria and to identify and reverse trends in groundwater pollution. The EU policy on groundwater is that:

> "The presumption in relation to groundwater should broadly be that it should not be polluted at all. For this reason, setting chemical quality standards may not be the best approach, as it gives the impression of an allowed level of pollution to which Member States can fill up. A very few such standards have been established at European level for particular issues (nitrates, pesticides and biocides), and these must always be adhered to. But for general protection, we have taken another approach. It is essentially a precautionary one. It comprises a prohibition on direct discharges to groundwater, and (to cover indirect discharges) a requirement to monitor groundwater bodies so as to detect changes in chemical composition and to reverse any anthropogenically induced upward pollution trend. Taken together, these should ensure the protection of groundwater from all contamination, according to the principle of minimum anthropogenic impact."

The European Commission adopted a proposal for a new Directive to protect surface water from pollution on 17 July 2006 [COM(2006)(397 final)]. The proposed Directive, which is required to support the WFD, will set limits on concentrations in surface waters of 41 dangerous chemical substances (including 33 priority substances and 8 other pollutants) that pose a particular risk to animal and plant life in the aquatic environment and to human health.

A groundwater monitoring technical guidance document was adopted to assist in the achievement of the goals the WFD and the implementing Directives [2000/60/EC Guidance Document No. 15, 2006].

Given the recent nature of the directives and guidance in the European Union, how the new groundwater policies will be implemented in the Member States is not clear, and additional discussions regarding implementation are continuing within those States.

## 3.5. Comparison of Groundwater Monitoring Programs to Alberta

Generally, most of the jurisdictions in the United States (state and federal) rely on design and construction criteria, siting restrictions, and nutrient management plans as a way to manage environmental risk from manure storage and collection. According to some agency sources in the United States, this decision is partly political, since imposing groundwater monitoring on agricultural operations would not likely be popular political policy, and partly practical since how nitrates from CFOs should be distinguished from other sources of nitrates in mixed-use agricultural areas is not clear. The primary reason may be that the prevailing statutory authority for the protection of water quality in the United States by the EPA authorizes regulation of “point source discharges” to surface water, and does not specifically include the protection of groundwater resources except groundwater that is hydrogeologically connected to surface water. Regardless of the political basis for the choice in policy, the result is that the majority of jurisdictions in the United States choose to manage the risk of contamination to surface and groundwater from manure by controlling surface runoff and through construction design standards, siting and operational criteria rather than by groundwater monitoring.

For those states that do require groundwater monitoring, the programs fall into two categories: programs that specifically call for a monitoring well program (i.e., “one up and two down”), and those that simply prohibit discharges to groundwater and have the discretion to impose monitoring on a site-by-site basis as the performance criteria for achieving that prohibition. For states such as New Mexico and California, the divergence with EPA policy appears to be based on scientific studies that demonstrate existing impacts to groundwater. In both states, however, agency staff speculate that current impacts might be the result of historical construction designs that were not as stringent as current requirements.

With regard to the EU, the primary difference in regulatory programs is reflected in the political attitude toward groundwater quality. As noted, the EU has taken the position that no impact on groundwater is acceptable and therefore, all regulatory strategies are developed with that policy in mind. In North American, health standards such as Maximum Concentration Limits (MCLs) for drinking water and ecological standards have been developed, which indicate that some levels are acceptable. This is also reflected in the development of the technical construction standards for constructed and natural liners, which allow for and anticipate some amount of hydraulic conductivity. However, a review of the policy and regulatory strategies is outside the scope of this project.

One other important point of comparison is that in some jurisdictions such as Alberta and New Mexico, the details of the groundwater monitoring program are set out in technical guidance.  In various states, such as Nebraska and Oklahoma, the monitoring program is incorporated into enforceable regulations.

Generally, state groundwater monitoring programs are similar to that adopted in Alberta. Most require upgradient and downgradient monitoring of specific constituents and use the upgradient well to establish background conditions.  New Mexico is unique in that the state requires only one downgradient well and specifies the minimum distance to the manure storage or collection facility.  New Mexico also uses the upgradient well to provide information about upstream potential sources since many of their dairy operations are located in close proximity, many on adjacent properties.  Some states, such as Oklahoma and Nebraska, incorporate well construction standards in the regulations instead of relying on well construction guidance.

***Table 3.2. Comparison of Various Groundwater Monitoring Programs at CFOs***

| Jurisdiction | Monitoring Wells/Location | Monitored Constituents | | Frequency | | Other |
|---|---|---|---|---|---|---|
| | | **Baseline** | **Indicator** | **Baseline** | **Indicator** | **Supplemental** |
| Alberta | 1 upgradient<br>2 downgradient | Potassium ($K^+$) Chloride ($Cl^-$)<br>Nitrate + Nitrite ($NO_3^-$+$NO_2^-$)<br>Total Kjeldahl Nitrogen (TKN)<br>Total Dissolved Phosphorus (TDP)<br>Electrical Conductivity (EC)<br>Sodium ($Na^+$)<br>Calcium ($Ca^{+2}$)<br>Magnesium ($Mg^{+2}$)<br>Bicarbonate ($HCO_3$)<br>Sulphate ($SO_4^{-2}$)<br>Dissolved Organic Carbon (DOC)<br>pH<br>Total Dissolved Solids (TDS) | Potassium (K+)<br>Chloride (Cl-)<br>Nitrate + Nitrite ($NO_3^-$ +$NO_2^-$)<br>Total Kjeldahl Nitrogen (TKN)<br>Total Dissolved Phosphorus (TDP)<br>Electrical Conductivity (EC) | Twice during first year (spring and fall) | One or two year intervals (spring or fall) | Total coliforms<br>E. coli |

| Jurisdiction | Monitoring Wells/Location | Monitored Constituents | | Frequency | | Other |
|---|---|---|---|---|---|---|
| | | Baseline | Indicator | Baseline | Indicator | Supplemental |
| Nebraska | 1 upgradient<br>2 downgradient | NA | Nitrate<br>Chloride<br>Ammonia | NA | As approved | Water level measurements |
| North Carolina | 1 upgradient<br>2 downgradient | NA | Nitrate ($NO_3$-N)<br>Total Ammonia Nitrogen ($NH_3$-N)<br>Chloride<br>Total Dissolved Solids (TDS)<br>Fecal Coliform<br>pH (field)<br>Potassium<br>Water Level<br>Sodium | NA | March, July, November | Copper<br>Zinc<br>Total Phosphorous |
| New Mexico | 1 upgradient<br>1 downgradient within 50 feet | $NO_3$-N<br>TKN<br>Cl<br>TDS | $NO_3$-N<br>TKN<br>Cl<br>TDS<br>Water level | Prior to discharge | Quarterly | Remediation plan if exceed limits |

| Jurisdiction | Monitoring Wells/Location | Monitored Constituents | | Frequency | | Other |
|---|---|---|---|---|---|---|
| | | Baseline | Indicator | Baseline | Indicator | Supplemental |
| Oklahoma | Sufficient wells in direction of groundwater flow, wells within 150 feet of berm, upgradient to first aquifer or 50 feet. Downgradient to first aquifer or 100 feet. May install leak detection with prior approval. | NA | EC<br><br>pH<br><br>Ammonium-nitrogen<br><br>Nitrate-nitrogen<br><br>Total Phosphorus<br><br>Fecal coliform<br><br>Water level | NA | Annually | Regulations specify well construction standards. |

## 3.6. Closure Requirements for CFOs

### 3.6.1. Canada

As noted, materials in the soil may serve as a source of impacts to groundwater. Therefore, when assessing the need for management of risk from manure management, evaluating the impacts from closed or out of service manure storage or collection areas is also important.

In Alberta, the Regulations state that the owner operator "must remove the manure, composting materials and compost from the land or buildings within one year, or a shorter or longer term set by an approval officer, an inspector or the Board" [Regulations, 2007; s. 21 267/2001]. No additional guidance on the extent of removal that would be appropriate is provided in the regulations or Alberta technical guidance. Given that under certain conditions closed units will continue to pose a threat to groundwater, considering what appropriate closure criteria would be in order to minimize this long term risk is important. A summary of the closure requirements in other jurisdictions and a comparison to the Alberta program is provided below.

In Quebec, the regulations do not expressly prescribe closure requirements; however the regulations do state that "[a manure] pile must be completely removed and reclaimed or eliminated, in accordance with section 19, within 12 months after the date the pile is created" [Q-2, r.11.1]. S. 19 states that all livestock waste must be removed or eliminated. In Ontario, a person who owns or controls a permanent outdoor confinement area shall ensure that manure is removed from the confinement area at least once a year or more frequently if the accumulated manure may produce an adverse effect (as described in subsection 18 (3) of the Act.) [O. Reg. 267/03, s. 60 (2)].

Manitoba requires the owner or operator to submit a closure plan if the manure storage facility has been out of operation for one year [S. 6.2 (42/98)]. The Manitoba regulations do not specify closure requirements. The regulations of Saskatchewan and British Columbia are silent on the subject of decommissioning or closure of manure management units.

### 3.6.2. United States

In selected American jurisdictions, guidance for appropriate closure practices has been provided in detail. These include *Closure of Earthen Manure Structures* [National Center for Manure and Animal Waste Management, 2001], *Conservation Practice Standard 360-Closure of Waste Impoundments* [National Resources Conservation Service, 2001], and *Closure of Lagoons and Earthen Manure Storage Structures* [Texas Cooperative Extension, 2002]. In Nebraska, when the operations have ceased, the owner or operator is required to:

- "Remove all accumulated manure, litter, and process wastewater, including any sludge and sediment; follow agronomic practices including the sampling and testing of any wastes removed; and dispose in an agronomic manner; and
- Continue groundwater monitoring, as required, unless the Department has vacated the monitoring requirement. If the groundwater monitoring requirement has been vacated, monitoring wells must be properly abandoned." [N.A.R. 130. 11.010]

In New Mexico, the GWQB requires that upon closure, the facility remove all manure from the corrals and apply it to land application areas or transfer it offsite. Lagoons must be emptied of process wastewater and solids, and liners must be perforated or removed. The facility must backfill all lagoons to blend with surface topography and prevent ponding. All groundwater monitoring wells shall be monitored for two years following closure, and, if sampling shows groundwater standards are exceeded, the permittee must submit an abatement plan to the GWQB. The abatement plan shall include a site investigation to define the source, nature and extent of contamination and a proposed abatement option. Site investigation and abatement option shall be consistent with the requirements and provisions of Sections 20.6.2.4101, 20.6.2.4103, 20.6.2.4106, 20.6.2.4107, and 20.6.2.4112 NMAC. If groundwater quality remains below standards during post-closure monitoring, the permittee may request approval to plug the monitoring wells according to NMED Monitoring Well Construction and Abandonment Guidelines and terminate the discharge permit. Additionally, the facility must perform closure requirements in accordance with Natural

Resources Conservation Service Standard 360, Closure of Waste Impoundments [NRCS, 2001].

In Kansas, the regulations require the development of waste-retention lagoon or pond closure plan under certain conditions [K.A.R. 28-18-14(e)]. Generally, the plan must include a description of the maintenance, deactivation, conversion, or demolition of all waste-retention lagoons or ponds or the closure of any waste retention lagoon or pond by:

- Removing the berms, and levelling and revegetating the site to provide erosion control;
- Leaving the structure or structures in place for use as a freshwater farm pond or reservoir;
- Retaining the structure or structures for future use as a part of an animal waste management system;
- Other method approved by the agency.

The plan must also describe the plugging of any water or groundwater monitoring wells at the confined feeding facility [K.A.R. 28-18.16(e)].

In Iowa, the owner of an open feedlot operation who discontinues the use of the operation shall remove and land-apply in accordance with state law all manure, process wastewater and open feedlot effluent from the open feedlot operation structures as soon as practical but not later than six months following the date the open feedlot operation is discontinued. The owner of a CFO shall maintain compliance with all requirements in the CFO's waste discharge permit until all manure, process wastewater and open feedlot effluent has been removed and land-applied pursuant to the CFO's Nutrient Management Plan [I.A.C. 65-011(7)].

In Minnesota, the owner of an animal feedlot or a manure storage area is responsible for closure and within one year after stopping operations is required to "remove and land apply manure and manure-contaminated soils from manure storage areas and animal holding areas." As soon as practicable after removal, the owner or operator is required to reduce soil nitrogen by growing alfalfa, grasses, or other perennial forage for at least five years. Within 60 days after final closure, the owner or operator must submit a certified letter to the commissioner or county feedlot pollution control officer stating that the animal feedlot or the manure storage area was closed according to the regulations. The letter must identify the location of the animal feedlot or the manure storage area by county, township, section, and quarter section [M.R. 7020.2025].

Oklahoma is unique among the states reviewed in that the state requires financial assurance for closure of livestock waste impoundments. Any person who is licensed to operate an animal feeding operation with a liquid animal waste management system must furnish evidence of financial ability to comply with the requirements for closure of retention structures and other waste facilities [OK Stat. 9-209.1(A)].

The Oklahoma regulations also provide that if for any reason the facility ceases to function for a period of 24 months or by action of the Board is ordered to cease operations, the owner is responsible for proper closure of all waste retention structures. The owner must submit a closure plan including at a minimum:

(1) The sequence of closing process including but not limited to handling of waste retention structure wastewater, solids, and handling and safe disposal of bottom sludge;

(2) Demonstrate the availability of sufficient land area for land application of the liquid, solid, and sludge component of the waste retention structure; and

(3) Provide a copy of a written estimate, in current dollars, of the cost of hiring an independent third party to decommission each waste retention structure.

Post closure monitoring must be conducted for a period of at least three (3) years.

Closure of all retention structures shall commence within six (6) months and be completed within one (1) year of cessation of operations. Closure shall be in accordance with a closure plan approved by the Department. Liquid contents of a waste retention structure may be pumped out and land applied according to Department rules. Solids from the waste retention structure shall be removed and disposed of in an environmentally safe manner.  Sludge from the bottom of the waste retention structure shall be removed without compromising the integrity of the liner. Sludge may be land applied according to Department rules [O.A.R. 35:17-3-25].

## 3.7. Comparison of Closure Requirements to Alberta

As noted, the Regulations in Alberta state that the owner operator must remove certain manure and other materials within one year or as specified by the NRCB [Regulations, 2007; s. 21 267/2001].  Other Canadian jurisdictions generally require removal within one year of accumulation or sooner if a risk of adverse effects exists.  No Canadian jurisdictions provide specific closure performance standards.

Several sources of specific closure performance standards exist in the United States and several states impose specific closure activities.  In several states, groundwater monitoring continues after closure for a specified period of time or until the closure performance standard is demonstrated to be met.  Only one jurisdiction imposes financial assurance requirements for closure.  Table 3.3 summarizes general closure requirements.

*Table 3.3. Summary of Closure Requirements*

| Jurisdiction | Performance Standards | Closure Plan Required | Post Closure Groundwater Monitoring | Closure Trigger |
|---|---|---|---|---|
| Alberta | Remove manure, composting material and compost | No | No | Within one year of ceasing operations |
| Quebec | Remove or eliminate livestock waste | No | No | Removed once per year or as directed |
| Ontario | Remove annually or if having adverse effect | No | No | At least annually |
| Manitoba | Case-by-case | Yes | No | Out of operation one year |
| US Agricultural Extension Services | Issue technical guidance for closure | | | |
| Nebraska | Remove manure, sludge, sediments | No | Yes | When operations cease |
| New Mexico | Remove manure, liners | Yes | Yes | When closed |
| Kansas | Removal of materials, revegetate | Yes | Plug at closure | At closure |
| Iowa | Remove manure | No | Comply with permit until closed | 6 months after discontinue use |
| Minnesota | Remove manure and revegetate | No | No | Within 1 year of stopping operations |
| Oklahoma | Remove all manure | Yes with cost estimates | Yes. | 24 months after cease operations |

## 3.8. References

### 3.8.1. United States- Regulations

The following are citations to the state regulatory programs reviewed. State regulations that do not include groundwater monitoring programs are not cited in the text; however, all regulatory programs reviewed are included on the reference list.


California - California Code of Regulations, Title 27, Division 2, Chapter 7, Subchapter 2, Article 1. http://www.ciwmb.ca.gov/Regulations/Title27/ch7s2345.htm#Article1

California – Draft Water Discharge Requirement Order for Existing Milk Cow Dairies, California Regional Water Quality Control Board Central Valley Region (December 2006).

Iowa – 567 I.A.C. 65.1 et. seq. (455B) http://www.legis.state.ia.us/Rules/Current/iac/567iac/56765/56765.pdf

Kansas – K.A.R. 28-18-1 et. seq http://www.kdheks.gov/feedlots/prop_regs/2007/Regs_KS_Register_March_2007.pdf

Minnesota – M.R. 7020.0205 et. seq. http://www.revisor.leg.state.mn.us/bin/getpub.php?pubtype=RULE_CHAP&year=current&chapter=7020

Missouri – 10 CSR 20-6.300 et.seq. http://www.sos.mo.gov/adrules/csr/current/10csr/10c20-6c.pdf

Montana - A.R.M. 17.30.1300 et seq. http://deq.mt.gov/dir/legal/Chapters/CH30-13.pdf

Nebraska – N.A.R. Title 130 Chap 1 et. seq. http://www.deq.state.ne.us/

New Mexico – 20.6.2 NMAC http://www.nmcpr.state.nm.us/nmac/parts/title20/20.006.0002.htm

North Carolina - NC Administrative Code 15A NCAC 02H Section .0200 http://h2o.enr.state.nc.us/admin/rules/2H.0200.pdf

North Dakota – Rules and Regulations for the Control of Pollution from Certain Livestock Enterprises, 33.16.01 (large operations) and 33.16.03.1 (small and medium operations). http://www.health.state.nd.us/WQ/AnimalFeedingOperations/AFOProgram.htm

Oklahoma – Oklahoma Concentrated Animal Feeding Operations Act, OK Rev. Stat. Title 2, Chapter 1 Article 9.200 et. seq. and O.A.R. 35:17-3-11 et. seq. http://www.oar.state.ok.us/oar/codedoc02.nsf/frmMain?OpenFrameSet&Frame=Main&Src=_75tnm2shfcdnm8pb4dthj0chedppmcbq8dtmmak31ctijujrgcln50ob7ckj42tbkdt374obdcli00_

South Dakota – Art. 74:52:01 et seq. http://www.legis.state.sd.us/rules/rules/74/52/7452.doc

Texas – 30 T.A.C. Ch 321 Subchapter B et. seq. http://info.sos.state.tx.us/pls/pub/readtac$ext.ViewTAC?tac_view=5&ti=30&pt=1&ch=321&sch=B&rl=Y

United States – Code of Federal Regulations, 2007. Title 40: Protection of the Environment, part 122 http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?type=simple;c=ecfr;cc=ecfr;sid=311c9c517fd851459d4868e219573f20;idno=40;region=DIV1;q1=122;rgn=div5;view=text;node=40%3A21.0.1.1.12

United States – National Effluent Limitations Guidelines and Standards for Confined Animal Feeding Operations, 39 Federal Register 5704 (February 14, 1974).

United States – National Pollutant Discharge Elimination System Permitting Regulations for Confined Animal Feeding Operations, 41 Federal Register 11458 (March 18, 1976).

United States – Revised National Pollutant Discharge Elimination System Permitting Regulations for Confined Animal Feeding Operations, 68 Federal Register 7176 (February 12, 2003).

### 3.8.2. Canada – Regulations

Alberta – Agricultural Operation Practices Act (A-7 RSA 2000), 2007. http://www.qp.gov.ab.ca/documents/Acts/A07.cfm?frm_isbn=0779742621

Alberta – Standards and Administration Regulation (306/2006), 2007. http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

British Columbia - Agricultural Waste Control Regulation, (B.C. Reg. 321/2004) http://www.qp.gov.bc.ca/statreg/reg/E/EnvMgmt/131_92.htm

Manitoba - Livestock Manure and Mortalities Management Regulation (42/98) http://web2.gov.mb.ca/laws/regs/pdf/e125-042.98.pdf

Ontario - Nutrient Management Act, 2002 - O. Reg. 267/03 http://www.e-laws.gov.on.ca:81/ISYSquery/IRL814E.tmp/36/doc

Quebec - Agricultural Operations Regulation, 2001 (Q-2, r.11.1) http://www2.publicationsduquebec.gouv.qc.ca/dynamicSearch/telecharge.php?type=3&file=/Q_2/Q2R11_1_A.htm

Saskatchewan – Agricultural Operations Act, 1995 (A-12.1) http://www.qp.gov.sk.ca/documents/English/Statutes/Statutes/A12-1.pdf

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Directive concerning the protection of waters against pollution caused by nitrates from agricultural sources. 91/676/EEC (December 12, 1991) Amended by: Regulation (EC) No 1882/2003 of the European Parliament and of the Council of 29 September 2003 http://europa.eu.int/eur-lex/lex/Notice.do?val=172969:cs&lang=en&list=195740:cs,172969:cs,&pos=2&page=1&nbl=2&pgs=10&hwords=&checktexte=checkbox&visu=#texte

Directive of the European Parliament and of the Council on the Protection of Groundwater Against Pollution (COM(2003)550) (September 19, 2003). http://eur-lex.europa.eu/LexUriServ/site/en/com/2003/com2003_0550en01.pdf

Directive on the EU Water Framework Directive, (2000/60/EC) (October 23, 2000).

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### 3.8.4. Technical References

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New Mexico Environment Department (NMED), 2007, Ground Water Monitoring Well Construction and Abandonment Guidelines http://www.nmenv.state.nm.us/gwb/New_Pages/docs_policy/MW_guidelines.pdf

Texas Cooperative Extension, 2002, Closure of Lagoons and Earthen Manure Storage Structures, Publication B-6122. http://tcebookstore.org/tmppdfs/18206363-B6122.pdf

# 4. Geochemical Controls on CFO Contaminants in Groundwaters

Geochemical reactions in the subsurface can control the concentrations of many dissolved species and thus their migration in groundwaters. Because an understanding of these controls is necessary to characterize the fate of CFO leachate in groundwater, this Chapter describes the geochemical controls on nitrogen and phosphorus, which are the key aqueous phase contaminants present in CFOs. Because dissolved organic carbon (DOC) is a control on geochemical reactions that influence the N species and is present in high concentrations in CFO waters, the geochemical controls on DOC are also presented here.

## 4.1. Nitrogen in Groundwater

The most common N contaminant identified in groundwater is dissolved N in the form of $NO_3$, or nitrate. Nitrate pollution of groundwater is an increasing problem throughout North America and Europe and is a major concern for water supplies and surface waters.

Nitrate in groundwater is derived from various point and non-point sources. These sources include but are not limited to CFOs. To evaluate the impact of N contamination from these facilities on groundwater and surface water receptors requires an assessment of the impact of chemical transformations within the groundwater regime on the form(s) and concentrations of the N species in the groundwater both over time and space.

Nitrogen in groundwater near a CFO can be derived from a number of sources including: transport from the manure storage facility (MSF), transport from the CFO (excluding the MSF), the application of manure to fields proximal to the CFO, the application of inorganic N fertilizers proximal to the CFO, and natural occurring [*c.f.,* Hendry et al., 1984]. Unlike conservative solutes in groundwater (e.g., halogens), N is also subject to many possible transformations depending on environmental conditions in the subsurface. These transformations include redox and exchange reactions, and are summarized below.

### 4.1.1. Nitrogen Sources and Distribution

Although $NO_3^-$ is the dominant form of N in groundwaters, N can also exist in the dissolved state as ammonium ($NH_4^+$), ammonia ($NH_3$), nitrite ($NO_2^-$), nitrogen ($N_2$) and nitrous oxide ($N_2O$). Nitrogen can occur in oxidation states ranging from –III ($NH_4^+$) to +V ($NO_3^-$), and its reduction series can be written as:

$$NO_3^- \rightarrow NO_2^- \rightarrow NH_4^+ \rightarrow NH_4^+ \rightarrow N_{2(g)} \qquad [4.1]$$

Microorganisms facilitate many N transformations in groundwaters at normal temperatures and pressures [Fujikawa and Hendry, 1991; Paul and Clark, 1996].

The multiple valence states in which N can exist in groundwaters indicate that the distribution of N species is controlled by redox processes. Species in the redox series presented above also contain $H^+$, which further suggests the distribution of some of the N species are also controlled by pH. The influence of both redox and pH on the distribution of N in groundwaters is reflected in redox diagrams (cross plots of Eh or pe vs. pH) for N. These diagrams show $NO_3$ is stable in oxic groundwaters while $NH_4$ and $NH_3$ are stable in anoxic groundwaters. These diagrams also show $NH_3$ is stable relative to $NH_4$ above pH 9.2. As most groundwater environments have a pH of less than 9.2, the stable reduced species in most groundwaters is $NH_4$. The two most important overall reactions from a microbiological standpoint are *denitrification* and *nitrification*. These reactions are discussed below.

### 4.1.2. Ammonium

The positive charge on $NH_4$ causes it to readily adsorb to clays and organic matter. As a result, as much as 50% of the total N in subsurface horizons is fixed within interlayer portions of clays. Because of its ability to strongly adsorb to clays and organics in soils and geologic media, $NH_4$ is often used to displace other cations on the exchange complexes of soils and geologic media in order to determine their Cation Exchange Capacities (CEC).

The process of nitrification results in the oxidation of $NH_4$ to $NO_3$ by heterotrophs, and can be represented by:

$NH_4^+ + 2O_2 = NO_3^- + 2H^+$ [4.2]

In this reaction, bacteria use labile organic carbon as their energy source. In many natural groundwaters, sufficient labile organic carbon exists to support heterotrophic nitrification [Korom, 1992], although most nitrification occurs in the soil zone (above the water table) where the presence of organic carbon and $O_2$ are not limiting. Because nitrification is a biological process, temperature affects nitrification rates (30 to 35$^o$C is optimum). Nitrification, although slow below 5$^o$C, occurs under snow cover in many soils [Paul and Clark, 1996]. Nitrification rates are also dependent upon pH with optimum values occurring between 6.6 and 8.0; rates typically decrease below pH 6.0 and become negligible below 4.5. High pH values inhibit the transformations of nitrite to nitrate [Paul and Clark, 1996].

### 4.1.3. Nitrate

Nitrate is a negatively charged ion and, as such, it does not readily adsorb to soil organic matter or clays and is mobile in oxic groundwater environments.

The main mechanism to reduce $NO_3$ concentrations in groundwaters is denitrification. Denitrification can be accomplished by heterotrophic bacteria that use labile organic matter as an electron donor:

$5CH_2O + NO_3^- \rightarrow 2N_{2(g)} + 4HCO_3^- + CO_2 + 3H_2O$ [4.3]

Dissimilatory reduction of $NO_3$ is preferentially used by microorganisms when dissolved $O_2$ levels decrease to less than about 0.2 mg/L [*c.f.,* Trudell et al., 1986; Hendry et al., 1983]. As a result, denitrification should occur as long as the groundwater is oxygen deficient (i.e., under anoxic conditions), and should not be attenuated in $O_2$ rich groundwaters. This is exemplified by the persistence of $NO_3$ derived from poultry manure throughout the oxygenated Abbotsford-Sumas aquifer, British Columbia [Wassenaar et al., 2006; Wassenaar and Hendry, 2007].

The process of denitrification results in the reduction of $NO_3$ to $N_{2(g)}$ by bacteria through a complicated pathway involving intermediaries like $NO_2$. When these intermediaries are found in groundwater (not often, and when present they are in low concentrations), they confirm the presence of ongoing nitrate reduction. Denitrification is not a reversible process.

Rates of denitrification in groundwater from lab and field studies were reported by Korum [1992] with a range of 0.12 to 3.1 mg-N/L per day (44 to 1132 mg-N/L/year) with an average rate of 0.86 mg-N/L per day. Temperature affects denitrification exponentially above 15 to 20$^{o}$C and linearly below 15 to 20$^{o}$C [Paul and Clark 1996]. Fujikawa and Hendry [1991] used the rates of $N_2O$ production to measure rates of denitrification below the water table in a fractured glacial till. Trudell et al. [1986] used the increase in $HCO_3^-$ to estimate the rate of denitrification in a shallow sand aquifer.

Although denitrification is common in groundwater systems, Hendry et al. [1984] showed, using ion and Eh measurements, that nitrate can exist in isolated aerobic enclaves at depths below the water table in fractured glacial till deposits of southern Alberta for very long periods of time.

Denitrification can occur without microbial conversion but is exceedingly slow. Most denitrifying bacteria function best at pH 6 to 8 and the rates of denitrification slow below pH 5, although this process can still be significant.

Dissimilatory $NO_3$ reduction to $NH_4$ although possible in groundwaters normally plays a subordinate role [Appelo and Postma, 2002]. However, $NH_4$ produced by dissimilatory $NO_3$ reduction to $NH_4$ can be nitrified if redox conditions become favorable [Korum, 1992].

Autotrophic denitrification can occur via the oxidation of inorganic compounds of $Fe^{2+}$, $HS^-$ and $Mn^{2+}$ (which serve as electron donors) [Stumm and Morgan, 1996; Korum, 1992; Appelo and Postma, 2002] and can play a role in the removal of nitrate from groundwaters. This process is often not fully examined in many groundwater studies [Korum, 1992]. An example of autotrophic denitrification is nitrate reduction by pyrite oxidation, and is described by:

$$5FeS_2 + 14NO_3^- + 4H^+ \rightarrow 7N_2 + 5Fe^{2+} + 10SO_4^{2-} + 2H_2O \qquad [4.4]$$

and

$$10Fe^{2+} + 2NO_3^- + 14H_2O \rightarrow 10FeOOH + N_2 + 18H^+ \quad [4.5]$$

The energy yield from sulfide oxidation is greater than from Fe (II) oxidation, so incomplete pyrite oxidation yields $Fe^{2+}$ rich environments [Appelo and Postma, 2002]. Other oxidation reactions involve elemental sulfur and partially reduced sulfur compounds [Paul and Clark, 1996]:

$$5S + 6KNO_3 + 2H_20 = 3N_2 + K_2SO_4 + 4KHSO_4 \quad [4.6]$$

and

$$5K_2S_2O_3 + 8\ KNO_3 + H_20 = 4\ N_2 + 9\ K_2SO_4 + H_2SO_4 \quad [4.7]$$

Although autotrophic denitrification can occur if organic carbon is present, as is the case in many CFO environments (see Chapters 5 and 6), organic carbon should be the preferred electron donor as it yields more energy than these inorganic compounds [Appelo and Postma, 2002].

In summary, the dominant forms of N in groundwaters are $NO_3$ and $NH_4$. The two most important overall reactions that control the presence or absence of these forms are *denitrification* (reduces the $NO_3$ concentrations) and *nitrification* (increases the $NO_3$ concentrations). In addition, CEC reactions influence the migration of $NH_4$. These two forms of N are intricately linked in the subsurface via redox related reactions.

## 4.2. Phosphorous in Groundwater

Phosphorus (P) typically exists in organic, soluble and adsorbed forms. Organic P is a part of all living organisms, including microbial tissues and plant residue, and is the principal form of phosphorus in the manure of most animals. About two-thirds of the phosphorus in fresh manure is in the organic form. Soluble P is sometimes termed available inorganic phosphorus. It can include small amounts of organic phosphorus, as well as orthophosphate – the form used by plants. It also is the form subject to loss by dissolution in runoff and to a lesser extent, leaching. When fertilizer or manure is added to soil, the pool of soluble phosphorus in the soil increases. The adsorbed P is unavailable inorganic phosphorus.

When present in groundwaters, P does not biodegrade or change into benign forms such as nitrate. It only has one important oxidation state in groundwater and is removed from the aqueous phase by adsorption on geologic media or precipitation by minerals. These processes can account for a substantial amount of phosphorus removal from the aqueous phase of leachate water, greatly retarding the transport of P in groundwaters [Colman, 2005].

Because P is present as an amphoteric oxyanion in water, it may strongly bind with metal hydroxides of aluminum and iron at groundwater-solid surface sites. The adsorption of oxyanions on pure metal hydroxide surfaces is well known [Dzombak and Morel, 1990; Moldovan et al., 2003; Moldovan and Hendry, 2005]. Because P is an oxyanion, its adsorption changes with pH and cation concentrations. Little is known about the reactions involving the precipitation of P in minerals. Iron-phosphate minerals vivianite and strengite [Parkhurst et al., 2003] and aluminum-phosphate mineral variscite [Robertson, 2003] are typically supersaturated in wastewater plumes [Colman, 2005], suggesting that their formation is kinetically driven. Although P has only one oxidation state in groundwater, the common metals with which it forms minerals are controlled by redox conditions in the groundwater. As a result, the migration of P in groundwaters must be considered in light of redox controls.

Accounting for the factors controlling the migration of P and N in groundwaters requires an understanding of both the physical controls and geochemical controls on the element. Such controls require models that simulate groundwater migration and the transport and geochemical reactions concurrently.

## 4.3. Dissolved Organic Carbon

Dissolved organic carbon (DOC) is operationally defined as the organic carbon present in water that passes a 0.45 ug/L filter. Most aquifer groundwaters contain <0.7 mg DOC/L. The concentrations of DOC in groundwaters associated with glacial till of the Interior Plains of North America are typically greater than those of aquifers. For example, Wassenaar et al. [1990] report DOC concentrations in oxidized tills at a study site in southern Alberta in a range between 16 and 45 mg/L. Hendry and Wassenaar [2005] report DOC concentrations in oxidized tills at a study site in southern Saskatchewan in a range between 16 and 170 mg/L. The source of DOC in the natural systems originates from either soil organic matter derived from infiltrating recharge water and biogenic or abiotic decomposition of buried peat or plant matter. In contrast to natural systems, DOC concentrations in CFO liquids can be much greater than natural systems, up to several thousand mg/L.

DOC plays an important role in the geochemical and biochemical evolution of groundwater [Hendry and Wassenaar, 2005], including acting as a carbon source in controlling microbial respiration (redox). The large concentrations of potentially labile DOC facilitate the rapid consumption of $O_2$ in groundwaters. The consumption of $O_2$ by this process prevents the oxidation of $NH_4$ to $NO_3$ (nitrification). However, nitrification should occur under conditions of excess $O_2$, such as in granular unsaturated zones (above the water table) or in the absence of DOC.

## 4.4. References


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# 5. Status of Livestock Production Contamination of Groundwater in Alberta

## 5.1. Potential factors affecting contamination in Alberta

### 5.1.1. Geology

The Quaternary geology of the Albertan Grassland and Peace Lowland regions is predominantly characterized by a combination of till (~60%) and fine and coarse-grained glaciolacustrine clay (~40%) [Rodvang and Simpkins, 2001]. The hydraulic conductivity (K) of till deposits – a poorly sorted and stratified sand/silt/clay mixture – typically range between $10^{-11}$ to $10^{-5}$ m/s, while glaciolacustrine clays – a clay/silt/sand mixture – typically have K values between $10^{-11}$ and $10^{-8}$ m/s [Rodvang and Simpkins, 2001]. In the event of contaminant release from a livestock production site, the low K of the glaciolacustrine clay deposits suggests they will act as a relatively impermeable barrier to groundwater transport processes over the long-term.

However, the upper portions of the Quaternary deposits are generally weathered as a result of mineral and organic matter oxidation, to a maximum depth of 25 m [Rodvang and Simpkins, 2001; Hendry, 1988, 1983, 1982, 1981; Hendry et al., 1984a]. Weathered layers are characterized by fractures, which significantly increase the local K providing preferential flow paths for water transport to occur. In the event of contaminant release from a livestock production site situated over a weathered deposit, the potential for rapid transport of the contamination to groundwater at depth is increased.

### 5.1.2. Soil Type

Areas of intensive livestock production (Figure 5.1) are characterized by two main soil groups: chemozemics and luvisols. Chernozemic soils are further subdivided into four additional sub-groups distinguished by their relative organic content: i) Brown (3 to 4%), Dark Brown (4 to 6%), Black (6 to 10%) and Dark Gray (6 to 10%) [AAFRD, 2002a]. Luvisol type soils only incorporate organic matter under long-term agricultural activities [AAFRD, 2002a]. A summary of the soil group distribution within the province of Alberta is presented in Figure 5.1.

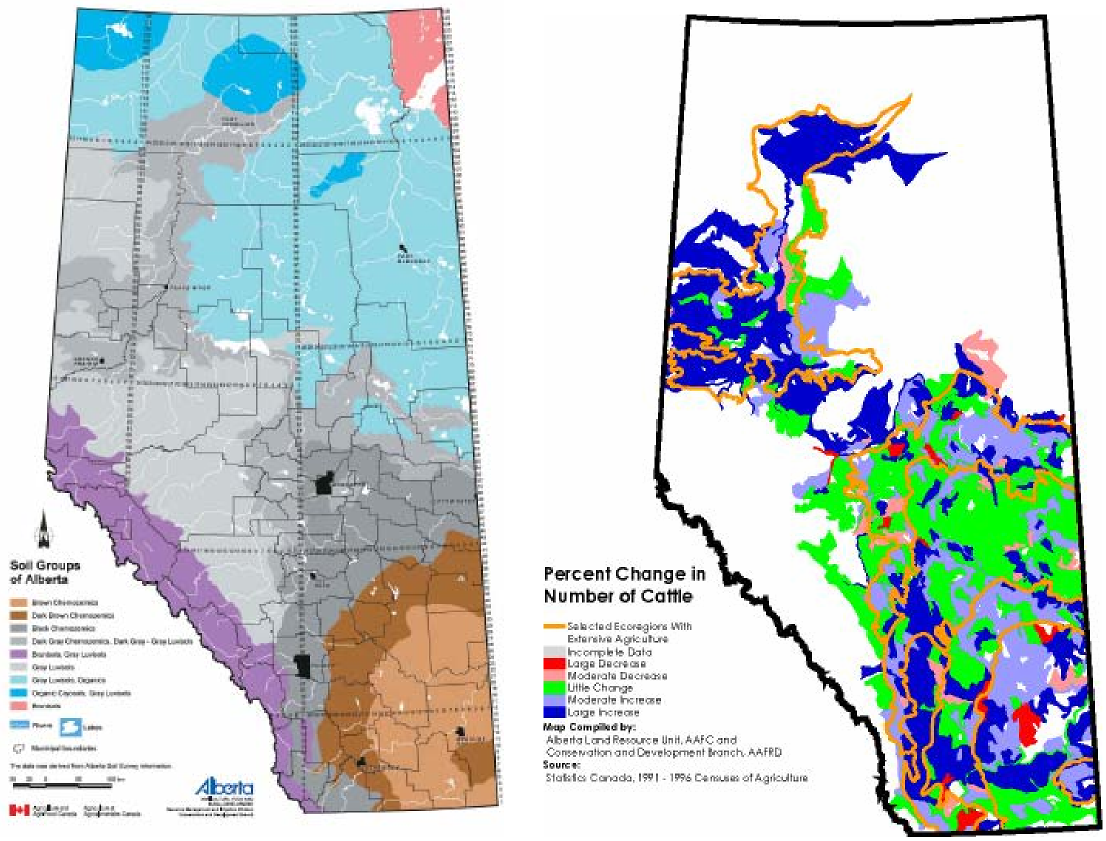


**Figure 5.1. Distribution of Alberta soil groups (left) and percent change in cattle numbers between 1996 and 2001 (right).**

### 5.1.3. Impact of releases on groundwater quality for human and livestock health

Approximately 440,000 Albertan's rely on groundwater as the drinking water source for themselves and for their livestock [AAFRB, 2002b]. A detailed study was conducted to quantify the extent of $NO_3$-N, total-P, pathogen, pesticides and heavy metal contamination occurring in the Alberta groundwater supply [AFFRB, 2002b]. However, no specific link was made between anthropogenic groundwater contamination and livestock manure storage.

Johnson et al. [2003] indicate that the Lethbridge area is noted to have a high cattle density, in addition to one of the highest incidence rates of gastroenteritis in Canada resulting from *E. coli* and *Salmonella* spp, commonly produced in cattle manure. However, their data suggest no correlation between infection rates and CFO sites, potentially as a result of the current best management practices being implemented in the province [Johnson et al., 2003].

### 5.1.4. Groundwater releases that have impacted surface waters

To the best of our knowledge, no cases of surface waters impacted by contaminated groundwater originating from livestock manure storage facilities in Alberta have been documented.

### 5.1.5. Groundwater releases that have impacted regional groundwater

To the best of our knowledge, no cases of regional groundwater contamination resulting from livestock manure storage facilities in Alberta have been documented.

## 5.2. Status of EMS (Earthen Manure Storage) Contamination

### 5.2.1. Source Chemistry

Very few examples are available in the scientific literature of swine manure wastewater concentrations from Alberta. Olson and Papworth [2006] characterized swine manure from two separate sites (Airdrie and Lethbridge); however, they expressed their results in terms of wet weight concentration and not as aqueous concentrations as typically used in wastewater studies (Table 5.1).

Similar to swine manure storage lagoon wastewater, studies involving characterization of cattle wastewater are generally lacking. In a study conducted on a 10 year old cattle feedlot near Vegreville, Kennedy et al. [1999] characterized the average concentrations associated with the wastewater held in a storage lagoon (Table 5.2).

***Table 5.1. Average swine and cattle manure storage lagoon wastewater concentrations from two Alberta sites.***

| **Measured Parameter** *(g/kg)* | **Olson and Papworth [2006]** *Airdrie* | *Lethbridge* |
|---|---|---|
| n | 1 | 1 |
| Total-P | 1.08 | 1.69 |
| Total-N | 5.13 | 3.97 |
| $NH_4$-N | 2.98 | 1.76 |
| $NO_3$-N (mg/kg) | 0.04 | 0.07 |
| Ca | - | - |
| Mg | - | - |
| Na | 0.58 | 0.23 |
| K | 1.25 | 1.07 |
| Cl | - | - |

n = number of sites included in study

***Table 5.2. Average concentration of cattle lagoon wastewater collected from a 10 year old cattle feedlot located in Vegreville, Alberta.***

| **Measured Parameter** *(mg/L)* | **Kennedy et al. [1999]** *Vegreville* |
|---|---|
| n | 1 |
| Total-N | 240 |
| $NH_4$-N | 176 |
| Total-P | 47.2 |
| K | 572 |
| Na | 351 |
| Ca | 130 |
| Cl | 616 |

n = number of sites included in study

A recent study indicated that levels of *Salmonella* spp. and *E. coli* in hog manure samples collected from 90 sites across Alberta were below detection limits [CAHIDF, 2005a]. The same study demonstrated the presence of a non-infective strain of *Cyrptosporidium* and an infective strain of *Giardia* in collected samples, both of which have been shown to degrade rapidly in lagoon settings.

### 5.2.2. Site Hydrogeology

In a detailed study of several lagoon sites within Alberta, CAHIDF [2005b] indicates that despite poor design and maintenance, little seepage was detected from facilities constructed without clay liners. The lack of significant contamination is attributed to the formation of a manure seal around the lagoon perimeter, resulting in a significantly decreased hydraulic conductivity [MacMillan, 2000]. However, some contamination was detected below studied sites, suggesting that preferential flow paths were temporarily occurring potentially as a result of breaks in the manure seals [MacMillan 2000]. In addition, MacMillan [2000] indicated that the hydrogeologic integrity of lagoon sites could potentially be compromised by the presence of tree roots and subsurface occurrences of increased conductivity deposits, such as sand lenses.

### 5.2.3. Operation Practices

Manure storage lagoons are predominantly employed by hog producers for manure storage in Alberta. These structures consist of natural clay deposits or emplaced clay material that is engineered to achieve low hydraulic conductivities. An operational code of practice was implemented in Alberta prior to 2000 that outlines the minimum requirements for siting and design criteria, including depth to water table, material K properties and flood control [AAFRD, 2001]. However, a detailed investigation of five storage sites indicates that the design of sites prior to 2000 varies widely across the province [MacMillan, 2000]. Historical practices rely on the observed self-sealing action of lagoons to prevent the seepage of manure to the underlying groundwater. However, as the AAFRD site investigation details, periodic seepage likely occurs from temporary breaks in the self-sealed, resulting in localized contamination of the underlying soil system [MacMillan, 2000].

### 5.2.4. Site Age

No data was reviewed that addressed the evolution of contamination plumes from sites with time. A site investigation study conducted by the AAFRD suggests that swine lagoons constructed before at least 1991 did not include any formal engineering design or rigorous siting criteria [MacMillan, 2000]. Soil and groundwater analyses indicated elevated concentrations of $NO_3$, $NH_4$ and Cl below these older sites [MacMillan, 2000].

### 5.2.5. Extent of known contamination in Alberta

Groundwater contamination studies conducted on Alberta EMS sites are limited. However, the studies that have been conducted suggest the extent of contamination is not significant [MacMillan, 2000]. However, the prevalence of EMS sites currently employed and that were constructed without any engineering design or siting criteria are a source of potentially significant groundwater contamination, which will be dependent on the integrity of their developed manure seals over the long-term [MacMillan, 2000].

## 5.3. Status of Cattle Feedlot Contamination

### 5.3.1. Source Chemistry

Representative average cattle feedlot manure concentrations from several studies are summarized in Table 5.3. Olson et al. [2003] provide the most detailed characterization of feedlot manure from three separate feedlots sampled over an eight year period in Lethbridge. Casson et al. [2006], in a study of manure amended fields, detail the concentrations associated with manure collected from two feedlots in southern Alberta. Olson and Papworth [2006] collected manure from two feedlots (located in Airdrie and Lethbridge). Finally, Miller et al. [2006a] characterized the composition of manure collected from a single feedlot in Lethbridge. The amounts of total-P and total-N vary greatly between the four cited investigations. The cause of this variability is often attributed to the variability in the operational practices of CAFO, which include feed type, animal density, bedding material and local climatic conditions. In western Canada, feedlot pens are normally cleaned in spring, after snow melt, which results in manure with a high water content (up to 75% wet weight) [Larney et al., 2006]. However, the source chemistry of solid manure cannot be used in the modeling of contaminant transport within the soil and groundwater underlying feedlots. Instead, the source chemistry of the mobile liquid fraction needs to be characterized.

***Table 5.3. Average cattle feedlot manure chemistry from several Alberta feedlots.***

| **Measured Parameter** | **Olson et al. [2003]** | **Casson et al. [2006]** | **Olson and Papworth [2006]** | | **Miller et al. [2006a]** |
|---|---|---|---|---|---|
| ***(g/kg)*** | ***Lethbridge*** | ***Lacombe*** | ***Airdrie*** | ***Lethbridge*** | ***Lethbridge*** |
| n | 3 | 1 | 1 | 1 | 1 |
| Total P | 2.70 | 4.7 | 1.60 | 3.08 | 3.0 |
| Total N | - | 24.0 | 5.48 | 9.20 | 15.0 |
| $NH_4$-N | 2.59 | 1.5 | 0.74 | 1.04 | 0.64 |
| $NO_3$-N (mg/kg) | 0.02 | 0.8 | 0.40 | 0.11 | 0.06 |
| Ca | 24.9 | - | - | - | - |
| Mg | 7.87 | - | - | - | - |
| Na | 4.26 | - | 0.73 | 1.64 | - |
| K | 20.1 | - | 5.82 | 6.94 | - |
| Cl | 4.38 | - | - | - | - |

n = number of feedlots included in each study.

The average concentrations of feedlot runoff from two separate studies are summarized in Table 5.4. Miller et al. [2004] diverted the runoff from a three year old Lethbridge feedlot and collected samples using an automated sampler over a two year period. The authors indicated that the runoff, on average, exceeded selected government water quality guidelines for total-P, total-N, $NH_4$-N, Na, Cl and TDS. Similarly, Kennedy et al. [1999] also diverted and collected runoff from a 10 year old Vegreville feedlot over a two year period using an automated sampler (Table 5.4). A distinction should be made between fresh feedlot runoff and runoff that is stored in lagoon containment structures for extended periods of time, as the source chemistry will differ between the two depending on the length of storage.

*Table 5.4. Average water chemistry of runoff from two Alberta cattle feedlots.*

| Measured Parameter | Miller et al. [2004] | Kennedy et al. [1999] |
|---|---|---|
| | *Lethbridge* | *Vegreville* |
| n | 1 | 1 |
| Total-P | 35.3 | 56.2 |
| Total-N | 85.7 | 240 |
| Total-C | 604 | - |
| TOC | 524 | - |
| Na | 246 | 340 |
| K | 515 | 510 |
| Ca | 148 | 81.2 |
| Cl | 604 | 668 |
| $SO_4$ | 217 | - |
| TDS | 2671 | - |
| EC (dS/m) | 4.2 | - |

### 5.3.2. Site Hydrogeology

The control of local hydrogeology on the release of contaminants from cattle feedlots has not been extensively studied in Alberta. Olson et al. [2005] investigated the hydrogeologic evolution of a newly constructed feedlot located in Lethbridge, Alberta, over the first four years of operation. The site was characterized by a shallow groundwater table, with an average depth between 1.23 and 2.50 m. A seasonal variation in groundwater depth was observed with maximum elevations occurring between May and July each year and water table fluctuations closely followed precipitation patterns. In addition, the authors observed the development of a distinct water mound directly below the feedlot pens within the first year of observation, suggesting an increased rate of recharge due to cattle urination and the lack of vegetation transpiration.

Kennedy et al. [1999] conducted a detailed investigation on a 10 year old cattle feedlot constructed on sandy-clay loam soil in Lethbridge. The authors observed the formation of a manure interface layer, formed through the mixing and compaction of manure and the underlying soil profile. Mielke and Mazurak [1976] suggest the compact organic layer that develops between the feedlot manure and soil layer horizons restricts the downward transport of feedlot contaminants (see Chapter 6.2.2 for a detailed summary). The authors conducted a series of infiltration tests and determined the infiltration rate of a newly constructed feedlot pen floor was initially between $7.4 \times 10^{-4}$ and $3.7 \times 10^{-4}$ cm/s and decreased to between $2.9 \times 10^{-6}$ and $2.3 \times 10^{-6}$ cm/s after 216 hours. Identical testing of a three year old pen floor resulted in no observable infiltration of water after 137 hours. Finally, a third test conducted on a pen floor with the upper manure layer removed indicated an infiltration rate between $1.8 \times 10^{-7}$ and $7.8 \times 10^{-8}$ cm/s after 192 hours, and no observable infiltration after 336 hours.

In Central Alberta, between 16 and 40% of precipitation deposited on feedlots during rainfall events will become runoff [Kennedy et al., 1999]. Olson et al. [2006] noted greater runoff originating from the pen floor, as compared to the bedding area, during rainfall events. The authors suggested that the stability of the manure within the bedding area was increased by the bedding material, while the pen floor becomes unstable during rainfall events. The current catch-basin design criteria (as of 2002) were demonstrated to adequately contain rainfall over the three year duration of a feedlot runoff study in Lethbridge, Alberta [Miller et al., 2004].

### 5.3.3. Site Contamination

Sommerfeldt et al. [1973] observed elevated $NO_3$-N and total-P concentrations in the upper 50 cm of soil cores sampled adjacent to three feedlots in Southern Alberta (Figure 5.2). Feedlot site 1 was constructed on silty-clay soil and was in operation for 10 years, feedlot 2 was constructed on loam-till soil and in operation for 40 years, and feedlot 3 was constructed on loam-till and in operation for over 60 years. However, concentrations were observed to fall below natural background levels below approximately 60 cm depth at all three feedlot sites. In addition, monitoring of the groundwater downgradient and adjacent to all three feedlots indicated $NO_3$-N concentrations between 0.2 and 2.7 mg/L (feedlot 1), 12.6 and 58.8 mg/L (feedlot 2), and from 1.9 and 140 mg/L directly beneath feedlot 3.

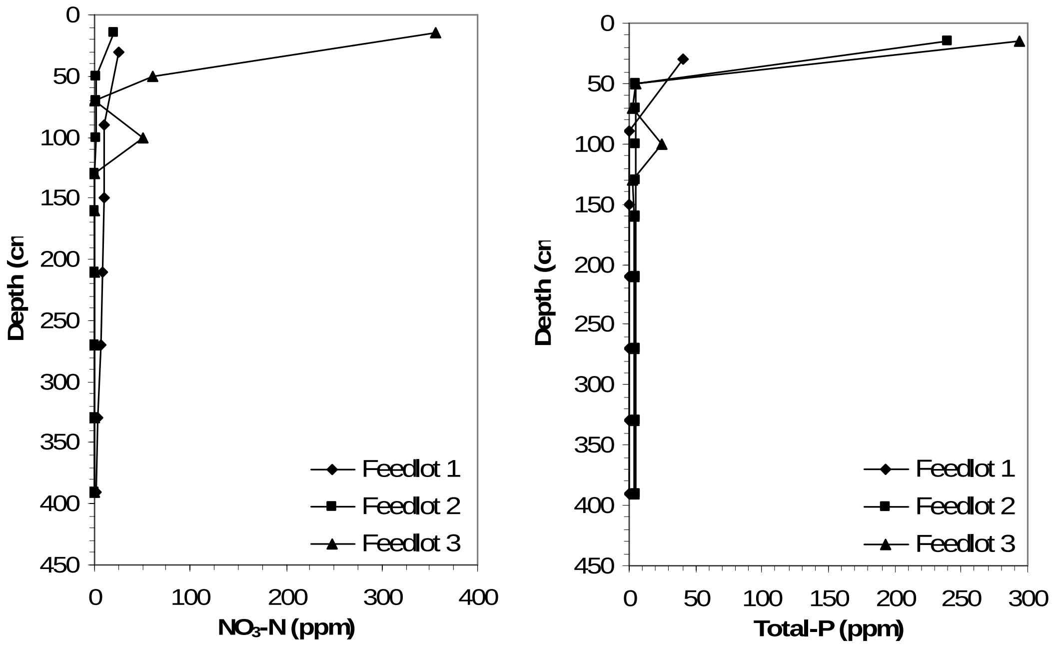


**Figure 5.2. $NO_3$-N and Total-P concentrations determined from soil cores sampled from adjacent to three feedlots in Southern Alberta. Feedlots constructed on a) silty-clay soil (feedlot 1); b) loam-till (feedlot 2) and c) loam-till (feedlot 3). (after Sommerfeldt et al. [1973]).**

Kennedy et al. [1999] sampled soil cores from three separate locations beneath a 10 year old cattle feedlot constructed on sandy-clay loam soil in Lethbridge. The three sampled sites were in use for five years (site 1), two years (site 2) and newly constructed (site 3). Elevated $NO_3$-N and Cl concentrations were observed in all three sites, suggesting that contaminants had infiltrated through the manure interface layer over time (Figure 5.3). The data suggest that $NO_3$-N migrated into the underlying soil over the two year lifespan of site 2 at a rate much greater then that observed at the five year old site (site 1). However, the authors do not address the potential reasons for this discrepancy. Comparing the Cl and $NO_3$-N migration fronts for site 2 suggests that the observed spike in $NO_3$-N concentrations at depth are not related, as Cl should move at a greater rate then $NO_3$-N.

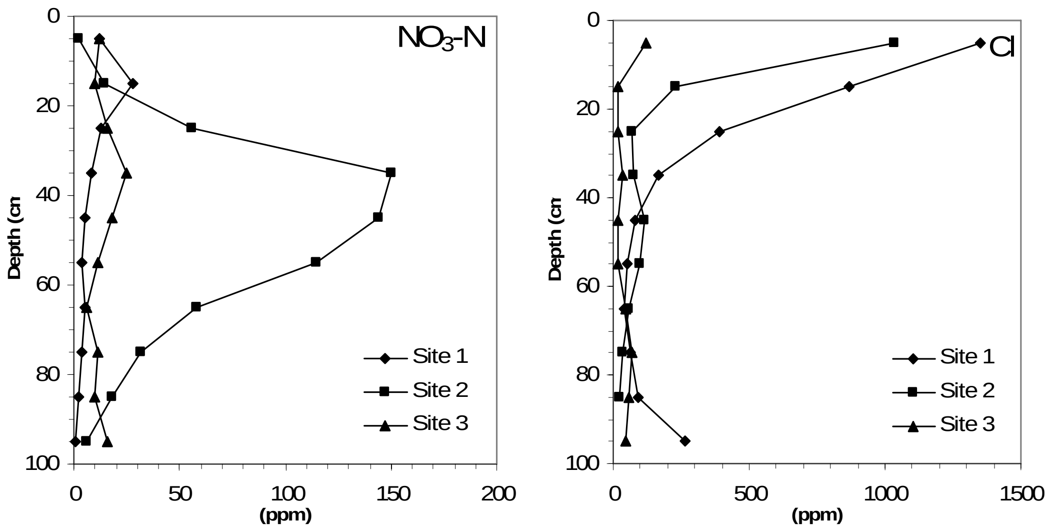


**Figure 5.3. $NO_3$-N and Cl concentrations of soil cores sampled from three cattle feedlot pens constructed on silty-sand soil. The sampled pens were active for five years (Site 1), two years (Site 2), and one week (Site 3), respectively (after Kennedy et al. [1999]).**

Olson et al. [2005], in a study investigating the contamination of soil and groundwater beneath a newly constructed feedlot characterized by sand-silt-clay overlying oxidized till, observed after three years a significant elevation in Cl concentrations to the studied depth of 1.4 m, and a significant increase in $NO_3$-N concentrations to 0.15 m depth and in $NH_4$-N concentrations to 0.30 m depth (Figure 5.4). These observations suggest $NH_4$-N migrates at a rate of approximately 5 cm/yr and its transport is retarded, relative to Cl, by over a factor of seven ($R > 7$). The authors also demonstrated a significant increase in Ca, Na and Mg concentrations between 0 and 50 cm depth in the soil profile, suggesting they are both associated with the migrating manure front and mobilized through exchange reactions with infiltrating $NH_4$-N [*c.f.*, Chang and Donahue, 2007].

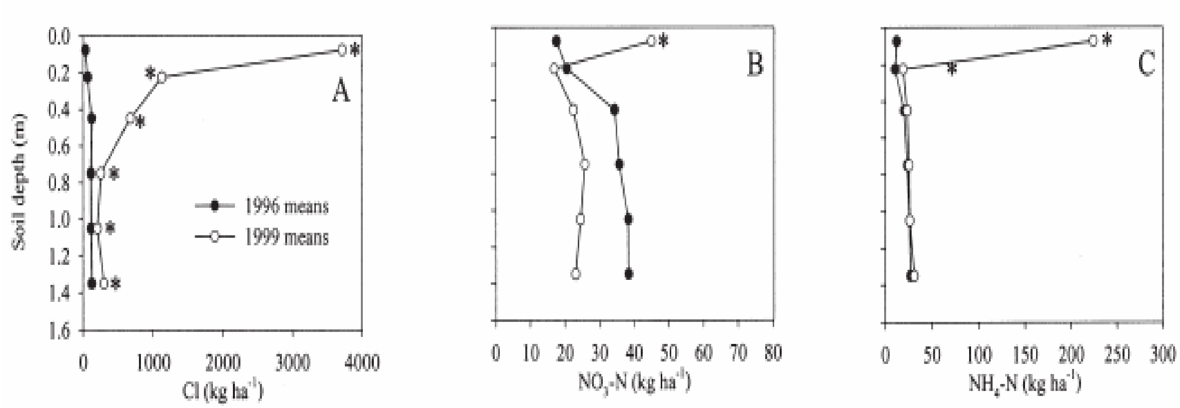


**Figure 5.4. Average paste extract concentrations in a soil core sampled from beneath a feedlot before (1996) and after (1999) construction. The feedlot was constructed on 1 m of (33:33:33) sand-silt-clay soil overlaying an oxidized till deposit (taken from Olson et al. [2005]). * denotes significant difference ($P < 0.05$) between the 1996 and 1999 mean concentrations for each soil layer.**

In addition to soil profile sampling, Olson et al. [2005] compared the groundwater before and three years after construction at an average depth of 5.8 m beneath the feedlot. Monitoring prior to feedlot construction indicated that several monitoring wells within the feedlot exceeded water quality guidelines for $NO_3$-N, Na and EC. Previous studies by both Hendry et al. [1984 b] and Rodvang et al. [1998] have shown that $NO_3$-N occurs naturally in groundwater located in oxidized till at many locations in southern Alberta, and that the concentration is dependent on the redox conditions and water flow regimes.

Greater but variable Cl concentrations were observed in monitoring wells within the feedlot after three years as compared to outside of the feedlot, indicating that the Cl originated from manure seepage from the feedlot [Olson et al., 2005]. The largest increases in Cl concentrations were observed in the feedlot drainage alleys, which are characterized by a gravel layer overlying compacted soil and which lack the compacted manure layer found over the rest of the feedlot surface. In addition, increases in K and $NH_4$-N concentrations were observed to closely follow increases in Cl concentrations [Olson et al., 2005]. Finally, although the authors observed elevated levels of $NH_4$-N, K and $PO_4$-P in monitoring wells, corresponding elevations in the soil profiles were not present suggesting the potential for preferential flow of contaminants through soil macropores. However, the presence of macropores was not confirmed within the parameters of the study.

A recent study indicates the use of woodchips for bedding material significantly increases the potential loading capacity of $NH_4$-N, $SO_4$, and total coliforms to feedlot runoff relative to straw [Miller et al., 2006b]. In addition, Miller et al. [2006b] suggest feedlot bedding areas are significant stores for total-N, Na, K, $SO_4$, Cl, and total coliforms relative to the remaining pen floor, which correlates well with the observation that the majority of manure is deposited within the bedding areas.

### 5.3.4. Site Age

No specific studies have been conducted that correlate groundwater contamination and CFO ages. However, examination of the results presented by Olson et al. [2005] demonstrates a progression of downward contaminant migration in successive years of feedlot operation. This contaminant evolution resulted in Cl migration to a depth of 1.5 m after five years of operation and a significant increase in selected contaminants in the shallow groundwater.

### 5.3.5. Extent of known contamination in Alberta

To the best of our knowledge, the three studies cited in this review [Sommerfeldt et al., 1973; Kennedy et al., 1999; Olson et al., 2005] represent the extent of known subsurface contamination within Alberta specifically as a result of cattle feedlots.

## 5.4. Status of Poultry Production Contamination

No documented case studies involving poultry production and potential groundwater contamination are available from Alberta at this time.

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# 6. STATUS OF CFOs IN NORTH AMERICA

## 6.1. Manure Storage Lagoons (Swine, Dairy, Cattle)

Studies of concentrated animal feed operations (CAFO) and the effluent originating from earthen manure storage lagoons (EMS) have, to date, focused on determining the spatial differences in nutrient and solids concentration vertically within the manure wastewater. Determining the manure wastewater chemistry is typically conducted after chemical digestion of the unfiltered samples. This methodology results from a prevailing interest in determining the nutrient content of the manure, which can vary considerably with the solids content. The average values obtained from wastewater source chemistry determinations may vary widely depending on factors such as animal age, animal diet, type of storage, and manure handling system [Rieck-Hinz et al., 1996; Campbell et al., 1997].

### 6.1.1 Source Concentrations

#### 6.1.1.1. Swine Manure

Studies attempting to characterize wastewater in swine EMS sites have reported significant differences in solids and nutrient content with depth and between individual sites. These differences are exemplified in a comparison of studies summarized in Table 6.1 [Campbell et al., 1997; Ham et al., 1999; Fonstad et al., 2000; Fonstad, 2004; Fernando et al., 2005].

The large variation in ammonium ($NH_4$-N) in the liquid manure may result from variations in solids content or animal diet. At the time of the Campbell et al. [1997] study, American swine were sold at a higher finished weight than their Canadian counterparts. Older animals tend be less efficient at feed conversion, which is reflected in higher nutrient levels in the manure. The lower wastewater $NH_4$-N values observed by Ham et al. [1999] (Table 6.1) are potentially an artifact of some American operations where the waste storage system is flushed, which would result in the dilution of the manure nutrients through the addition of water. The typical swine diet consists of corn and soybeans, which contain P sources not readily available to swine; therefore, inorganic sources of P are added to the feedstock. Consequently, high levels of Total-P are usually present in swine manure. However, the concentration present in the manure wastewater would be dependent upon the amount of added P and the amount available to the animals at each individual EMS site. DOC concentrations in swine manure wastewater also vary considerably between EMS sites, attributed to variations in diet, bedding material and the age of manure wastewaters [Levi-Minzi et al. 1986]. Similar to $NH_4$-N, the concentration of Cl in swine wastewater varies considerably between sites, potentially as a result of variation in diet, amount of urine mixed in with manure solids, and dilution of wastewater through precipitation.

***Table 6.1. Comparison of average nutrient concentrations in swine EMS wastewater from CFO sites in North America.***

| **Measured Parameter** | **Campbell et al. [1997]** | **Fonstad et al. [2000]** | **Fonstad [2004]** | **Ham et al. [1999]** | **Fernando et al. [2005]** |
|---|---|---|---|---|---|
| ***(mg/L)*** | ***PEI*** | ***Manitoba*** | ***Saskatchewan*** | ***Kansas*** | ***Kansas*** |
| n | 8 | 8 | 7 | 4 | 1 |
| $NH_4$-N | 3,530 | 1,874 | 3,879 | 673 | 475 |
| Total-P | - | 804 | 114 | 42.5 | 272 |
| DOC | - | - | 5,110 | - | 1,232 |
| Ca | 710 | 716 | 190 | 79.8 | 39 |
| Mg | 270 | 361 | 96.1 | 19.3 | 3 |
| K | 1,640 | 1,373 | 2,023 | 647 | 1,527 |
| Na | - | 519 | 734 | 270 | 392 |
| Cl | - | 959 | 1,351 | 276 | 878 |

n = number of EMS sites included in each study.

#### 6.1.1.2. Cattle and Dairy Manure

The source chemistry of cattle and dairy EMS wastewater differs from that of swine, and is generally characterized by lower concentrations of nitrogen (Total-N, $NH_4$-N and $NO_3$-N) and potassium (K, not presented) (Table 6.2).

***Table 6.2. $NH_4$-N variation between swine, dairy and cattle EMS sites.***

| **$NH_4$-N *(mg/L)*** | **Type** | **Location** | **Reference** |
|---|---|---|---|
| 3,530 | Swine | PEI | Campbell et al. [1997] |
| 702 | Swine | Kansas | Ham and DeSutter [1999] |
| 639 | Swine | Kansas | Ham and DeSutter [1999] |
| 711 | Swine | Kansas | Ham and DeSutter [1999] |
| 300 | Swine | North Carolina | Westerman et al. [1995] |
| 140 | Cattle | Kansas | Ham and DeSutter [1999] |
| 159 | Cattle | Texas | Sweeten et al. [1992] |
| 162 | Dairy | Texas | Sweeten et al. [1992] |
| 210 | Dairy | Tennessee | Sewell [1978] |

Several researchers have reported values for the chemistry of runoff water from cattle CFOs (Table 6.3). Substantial variation is evident amongst the studied cattle EMS sites, similar to that observed in the swine EMS source chemistry investigations. The variability in wastewater concentrations is most likely a result of variations in diet, evaporative losses as a result of local climate variations, or sample collection method.

Dairy runoff wastewaters are characterized by greater nutrient concentrations than cattle CFO site runoff wastewaters (Tables 6.3 and 6.4). However, considerable variation is also evident between EMS sites, as demonstrated in a comparison of studies by Mukhtar et al. [2004] and Ullman and Mukhtar [2007] (Table 6.4). The great variation between EMS sites can potentially be attributed to differences in animal diet, EMS system type (number of treatment cells, etc.), sampling method, and bedding material type [Mukhtar et al., 2004].

*Table 6.3. Comparison of average nutrient concentrations in cattle EMS wastewater from different CAFO sites across North America.*

| **Measured Parameter** *(mg/L)* | **Low [2006]** *Saskatchewan* | **DeSutter et al. [2000]** *Iowa* | **Ham et al. [1999]** *Kansas* | **Sweeten et al. [1992]** *Texas* | **Sweeten et al. [1992]** *Texas* |
|---|---|---|---|---|---|
| n | 10 | 5 | 5 | 1 | 1 |
| $NO_3$-N | 1.0 | 0.4 | 0.5 | - | - |
| $NH_4$-N | 59 | 72 | 98.3 | - | - |
| Na | 210 | 212 | 148 | 256 | 230 |
| Mg | 115 | 74 | 87.8 | 72 | 20 |
| Ca | 110 | 129 | 145 | 99 | 180 |
| Cl | 367 | 593 | 569 | 623 | 1,000 |
| K | 479 | 460 | 552 | 445 | 1,145 |
| Total-P | 16 | - | 47.5 | 43 | - |

n = number of EMS sites included in each study

*Table 6.4. Comparison of average nutrient concentrations in dairy EMS wastewater from different CAFO sites across North America.*

| **Measured Parameter** *(mg/L)* | **Mukhtar et al. [2004]** *Central Texas* | **Mukhtar et al. [2004]** *North Carolina* | **Ullman and Mukhtar [2007]** *Texas* |
|---|---|---|---|
| N | 12 | 15 | 29 |
| $NH_4$-N | 303 | - | 373 |
| Na | 357 | 372 | 425 |
| Mg | 400 | 600 | 0.04 |
| Ca | 1,800 | 1,200 | 0.19 |
| K | 1,379 | 2,000 | 1,727 |
| Total-P | 470 | 733 | 547 |
| EC (us cm-1) | 7,324 | 7,191 | 9,121 |

n = number of EMS sites included in each study

For both cattle and dairy operations, fresh manure and urine do not contain great quantities of $NH_4$-N, as N is predominantly present in undigested or partially digested proteins and as urea in urine [Braam et al., 1997; Webb, 2001]. Because $NH_4$-N formation from manure occurs relatively slowly [Misselbrook et al., 1998], it primarily occurs during storage and the amount produced will vary depending on the time of storage. Feedstock variability potentially accounts for the observed differences in the concentrations of Na, Ca, and Mg [Ullman and Mukhtar, 2007]. P is not typically added to cattle diets, as it is for swine, since it is acquired from natural dietary sources [Vasconcelos et al., 2006]; thus the Total-P concentration found in EMS wastewater can vary significantly between study sites.

### 6.1.2. Site Hydrogeology

Studies of sealing or clogging of soils were prevalent in the late 1940s and early 1950s when groundwater recharge by surface infiltration ponds was used to rejuvenate depleted surficial aquifers. Researchers speculated that EMS sites may be good candidates for infiltration rate reduction due to soil clogging. Field and laboratory research on clogging by manure was conducted by numerous researchers under differing environmental conditions. The clogging has been consistently attributed to the initial physical blockage soil pores by manure solids followed by the development of a microbial growth layer [Chang et al., 1974; Lo, 1977; DeTar, 1979; Rowsell et al., 1985; Barrington and Madramootoo, 1989; Fonstad and Maule, 1995; Fonstad, 1996; Maule et al., 2000].

Culley and Phillips [1982] suggest the soil type used in constructing these structures does not play a significant role in the clogging layer development. However, numerous studies suggest the soil type affects the time required for significant clogging to occur, following the order clay < till << sand/gravel, although a similar reduced infiltration rate is observed in all soil types after a sufficient time period [Lo, 1977; Rowsell et al., 1985; Fonstad, 1996; Cihan et al., 2006].

Laak [1970] and Roswell et al. [1985] observed that the clogged surface interface layer ranged in thickness from three to 15 mm. In addition, Maule and Fonstad [2000] reported that this interface ingressed into the soil matrix at a rate of 0.3 mm/month and demonstrated a hydraulic conductivity of approximately $5x10^{-11}$ m/s. Similarly, numerous studies focused on determining the hydraulic conductivity of EMS sites and report highly variable results, ranging between $10^{-7}$ to $10^{-10}$ m/s [Chang et al., 1974; Hills, 1976; Barrington et al., 1983; Phillips et al., 1983; Roswell et al., 1985; Barrington and Madramootoo, 1989]. However, several researchers suggest the observed reduction in near-surface hydraulic conductivity is not translated downward into the soil column [Barrington and Madramootoo, 1989; Fonstad, 1996; Maule et al., 2000]. In all cases, the clogged layer is relatively unstable and can be easily ruptured by cleaning, gas bubbles, and wet-dry cycles, and it should not be relied upon to prevent contamination of the underlying groundwater [Chang et al., 1974; Nordstedt and Baldwin, 1975; Ciravolo et al., 1979; Gangbazo et al., 1989; Withers et al., 1998; Fonstad, 2004].

Once an EMS site has experienced sufficient clogging to result in a corresponding decrease in the hydraulic conductivity, additional mechanisms have been identified that can increase the potential for the migration of contaminants into the underlying groundwater regime. The sidewalls of EMS sites potentially represent areas of increased secondary hydraulic conductivity, attributed to cracking caused by wetting and drying, holes from burrowing animals and worms, and channels due to decaying weed roots. Parker et al. [1999a] demonstrated increased side-wall seepage with lagoon age in a study of a 22 year old cattle feedlot manure storage lagoon. Although not specifically mentioned by these authors, sidewall seepage can also be attributed to the absence of a less conductive manure layer and differences in compaction relative to the EMS floor [Glanville et al., 2001]. In addition, the formation of biological and/or physiochemical induced macropores as identified by McCurdy and McSweeney [1993] can result in the rapid migration of contaminants from storage sites.

In addition to EMS sites, both above and in-ground concrete structures are commonly used for CAFO manure storage. However, concrete structures are susceptible to failure because of design flaws that can be partially attributed to higher than normal tank pressures created by large tank depths (4 to 5 m; Fleming et al. [1999]). Cracking of walls in in-ground concrete manure tanks has been observed and attributed to uneven settlement, poor structural design, or improper backfilling [Fleming et al., 1999].

Concrete structures are characterized by extremely low hydraulic conductivities ($10^{-13}$ m/s). However, leakage can occur between floor and wall joints and cracks that develop over the lifespan of the tanks [Barrington et al., 1991]. These authors suggest that sealing of joints and cracks should be considered for tanks exposed to high groundwater tables.

### 6.1.3. Site Contamination

Thirty to 40 years ago, studies showed that significant amounts of effluent could seep into groundwater from EMS constructed in granular soils. Consequently, many studies to date have been designed to determine the potential for contaminant transport from EMS by establishing the geochemical extent of vertical effluent plumes in several different soil types to determine a set of best management practices in EMS location, design, and management.

The rate of seepage from EMS structures into the subsurface environment is dependent upon the hydraulic conductivity of both the accumulated manure layer and the underlying geologic media. In addition, the hydraulic conductivity of the underlying media is dependent upon the media type, and for the purpose of this review will include sand and gravels and fractured oxidized glacial tills and clays. Phillips and Culley [1985] observed greater $NO_3$-N concentrations in leachate beneath lagoons constructed of coarser grain sediments in the order of clay < till << sand, attributed to the introduction of oxygen and the aerobic conditions required for nitrification of $NH_4$-N (Korom and Jeppson [1994]; Chapter 4). Also, the presence of a shallow permeable or confined aquifer will control the extent of groundwater contamination resulting from EMS seepage into the underlying geologic media. Finally, once the contamination moves beyond the EMS facility, cation exchange reactions with the underlying media will influence the migration of N contaminants (Chapter 4).

#### <u>6.1.3.1. Sands and Gravels</u>

Several studies have investigated seepage from EMS sites constructed on loam type soils, and report hydraulic conductivities ranging from 0.13 to 3.60 cm/d and seepage rates from 0.06 to 14.4 cm/d [Chang et al., 1974; DeTar, 1979; Culley and Phillips, 1982; Barrington and Madramootoo, 1989] (Table 6.5). Consequently, studies of swine and dairy lagoons in the coastal plain states (Delaware, Florida, North Carolina and Virginia), characterized by alluvial soils and shallow water tables, have identified elevated $NH_4$-N and $NO_3$-N concentrations at distances exceeding 30 m after eight years of operation [Ciravolo et al. 1979; Ritter and Chirnside, 1987; Westerman et al., 1995; Halloway et al., 1996].

***Table 6.5. Hydraulic conductivity and seepage rates observed from EMS sites constructed in sand/gravel dominated soils (after Parker et al. [1999a]).***

| **Study (*Animal Type*)** | **Soil Type (*Age*)** | **Hydraulic Conductivity *(cm/d)*** | **Seepage Rate *(cm/d)*** | **Location** |
|---|---|---|---|---|
| Chang et al. [1974] *(Dairy)* | Silica Sand (*64d*) | 3.60 | 14.4 | California |
| | Sandy Soil (*17d*) | 0.18 | 0.72 | |
| DeTar [1979] *(Dairy)* | Gravel-Sand (*15d*) | - | 0.24-0.48 | Pennsylvania |
| Culley and Phillips [1982] *(Beef)* | Sand (*10d*) | 0.15 | 0.06 | Ontario |
| Barrington and Madramootoo [1989] *(Swine)* | Sand (*69d*) | 0.13-0.18 | 0.26-0.36 | Quebec |

The Minnesota Pollution Control Agency [2001] investigated the extent of lateral migration of contaminants from three manure storage basins constructed without soil liners that were in operation for between 13 and more than 20 years, and from three manure storage basins constructed with liners that were in operation for between 6 and 12 years. In all cases the storage basins were constructed on coarse textured soils (coarse sand and coarse sand and gravel). They determined that the contaminant plumes emanating from the basins with no liners extended from >40 to >150 m downgradient whereas the plumes from the basins with soil liners extended from 90 to 130 m downgradient. Elevated concentrations of $NH_4$ (attributed to the anoxic conditions in the plume), phosphorus, organic N and organic carbon and Cl were measured in the plumes.

$NH_4$ and P concentrations were found to decrease along the length of the plumes. In two of the plumes emanating from soil lined basins, high concentrations of $NO_3$ were measured. The presence of $NO_3$ was attributed to the coarse texture of the soils and deep depths to the water table (>6 m). Interpretation of the data sets suggested that the chemical loadings and plume lengths were greater under unlined storage systems than under soil lined systems. The authors concluded that these facilities should not cause exceedances of surface water criteria for $NH_4$ and drinking water criteria for $NO_3$ when the distance from the source to a well or surface water body is >70 m for an earthen lined basin or >100 m for an unlined basin. They do caution that in cases where $NO_3$ is the dominant form of N, these distances may not be appropriate. They further concluded that these facilities should not adversely impact surface water quality for P when distances from the source to a surface water body are >30 m. In one case, however, excess P was observed in groundwaters 80 m from an unlined basin.

An extreme end member of contaminant transport in sandy-soils was observed during monitoring of a dairy lagoon constructed on highly permeable soil with an elevated water table in Japan. A detailed sampling regime observed elevated total-N concentrations at a distance of 75 m downgradient within four days, and at 15 m after 85 days [Kanazawa et al., 1999], suggesting an initial rapid migration of contaminants followed by a reduction in migration. These authors attributed the reduction to the formation of a manure seal in the EMS structure.

In a study of older swine EMS sites in North Carolina, Huffman [2004] observed a general trend of increased seepage and correspondingly higher $NO_3$-N concentrations in the underlying groundwater profile from swine EMS sites constructed in predominantly sandy soils. Groundwater samples were collected approximately 38 m downgradient from sites that ranged in age from six to 20 years, with elevated $NH_4$-N concentrations ranging from 0.3 to 473 mg/L. However, the author also indicated that a few sandy soil dominant sites showed little indication of $NH_4$-N seepage, which was attributed to natural variation of soil type within individual study sites. In addition, groundwater monitoring of several sites in Kansas, characterized by hydraulic parameters indicative of high seepage rates, revealed negligible $NO_3$-N contamination [Hobson, 1991].

Ammonium originates from the source manure wastewater and remains stable in the anaerobic conditions typically found beneath EMS sites [Fonstad and Maule, 1996]. Transport of $NH_4$-N can be retarded through exchange reactions when significant clay content is present in the materials underlying EMS sites (Chang and Donahue [2007]; Chapter 4). These exchange reactions typically result in elevated Ca and Mg pore-water concentrations in advance of the $NH_4$ plume, as preferential sorption of $NH_4$-N to exchange sites results in their release to the soil porewater [Ciravolo et al., 1979; Fonstad and Maule, 1996]. The transport of Cl in the underlying hydrogeologic profile is considered conservative, and significant porewater Cl concentrations typically delineate the leading edge of contaminant plumes from EMS sites [Fonstad, 2004].

#### 6.1.3.2. Oxidized Till and Clays

Oxidized tills are common throughout Alberta and the Interior Plains of North America, and are characterized by varying amounts of clay, silt and sand [Rodvang and Simpkins, 2001]. The groundwater velocity through these tills should be controlled, at least in part, by their textures. Seepage from EMS sites constructed on oxidized tills and clay-rich soils has been demonstrated to have a hydraulic conductivity ranging between 0.004 and 0.91 cm/d and a rate between 0.005 and 1.18 cm/d [Chang et al., 1974; DeTar, 1979; Culley and Phillips, 1982; Roswell et al., 1985] (Table 6.6). The variable nature of the measured rates is most likely dependent upon the relative clay content of the geologic media and the duration that each EMS was exposed to manure wastewater. The clay content of EMS sites will serve to retard the migration of contaminants, relative to conservative Cl, from the EMS site due to cation exchange reactions with the clay surface [Chang and Donahue, 2007]. Also, examination of the measured rates (Table 6.6) suggests they decrease with increasing exposure to EMS wastewater, corresponding to the development of a clogging layer with time.

***Table 6.6. Hydraulic conductivity and seepage rates observed from EMS sites constructed in clay-rich soils (after Parker et al. [1999a]).***

| **Study (*Animal Type*)** | **Soil Type (Age)** | **Hydraulic Conductivity (cm/d)** | **Seepage Rate (cm/d)** | **Location** |
|---|---|---|---|---|
| Chang et al., [1974] (*Dairy*) | Silty-Clay (*7d*) | 0.91 | 3.6 | California |
| DeTar [1979] (*Dairy*) | Clay (*7d*) | - | 0.22-1.18 | Pennsylvania |
| | Clay (*12d*) | - | 0.12 | |
| | Shale-Clay (*7d*) | - | 0.14-0.50 | |
| Culley and Phillips [1982] (*Beef*) | Clay (*10d*) | 0.13 | 0.04 | Ontario |
| Roswell et al. [1985] (*Beef*) | Clay (*30d*) | 0.0038-0.0059 | 0.035-0.0054 | Ontario |

Chang and Donahue [2007] observed retardation of swine manure derived $NH_4$-N transport through glacial clay cores, with a retardation factor of between 3 and 4. In addition, their results indicated a corresponding increase in aqueous Ca and Mg concentrations as a result of exchange reactions with $NH_4$-N. Huffman [2004] observed a weak correlation between decreased seepage and reciprocally lower $NH_4$-N concentrations below swine EMS sites constructed in clayey soils. However, significant seepage was observed in a number of EMS structures situated in clayey soils, which was attributed to the variations in soil type within individual study sites. Additionally, Fernando et al. [2005] suggest that $NH_4$ in swine manure experiences increased sorption and decreased de-sorption rates due to the presence of significant quantities of dissolved organic carbon (DOC). The mitigating effect of sorption was demonstrated in a study of lagoon liners by Reddi and Davalos [2000], where increased breakthrough times and decreased concentrations were observed with increasing liner thickness and constructed permeability (Fig. 6.1).

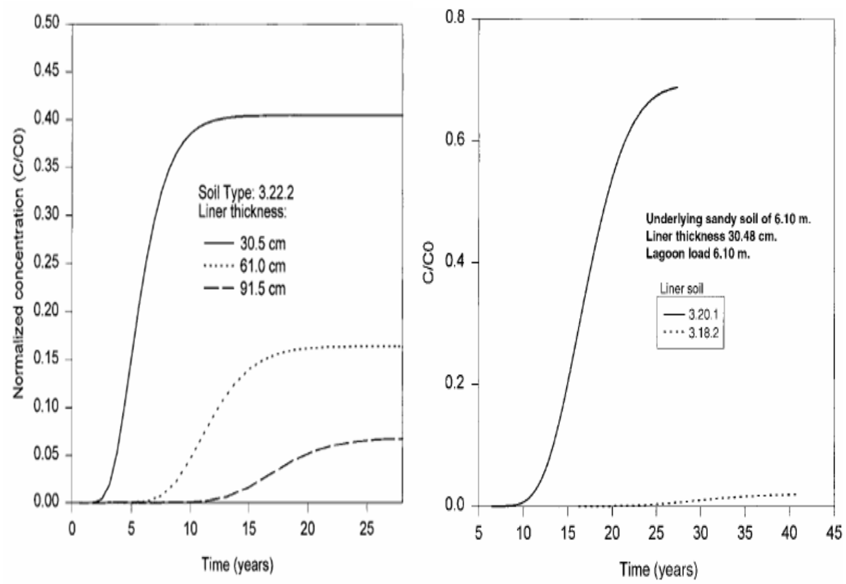


**Figure 6.1. $NH_4$-N breakthrough as a function of lagoon liner thickness and constructed permeability (after Reddi and Davalos [2000]).**

The retardation of $NH_4$-N migration, relative to Cl, is evident from a comparison of the $NH_4$-N profiles determined by Fonstad and Maule [1996] (Figures 6.2 & 6.3). The authors examined two swine EMS sites constructed on clay till and in operation for 10 years (Figure 6.2) and 17 years (Figure 6.3), respectively. Examination of the measured Cl and $NH_4$-N fronts in Figures 6.2 and 6.3 suggest the $NH_4$-N front is retarded with respect to the Cl front by a factor of five. In addition, the migration rate of $NH_4$-N is approximately 2.5 cm/yr and 6.0 cm/yr for the 10 and 17 year old sites, respectively.

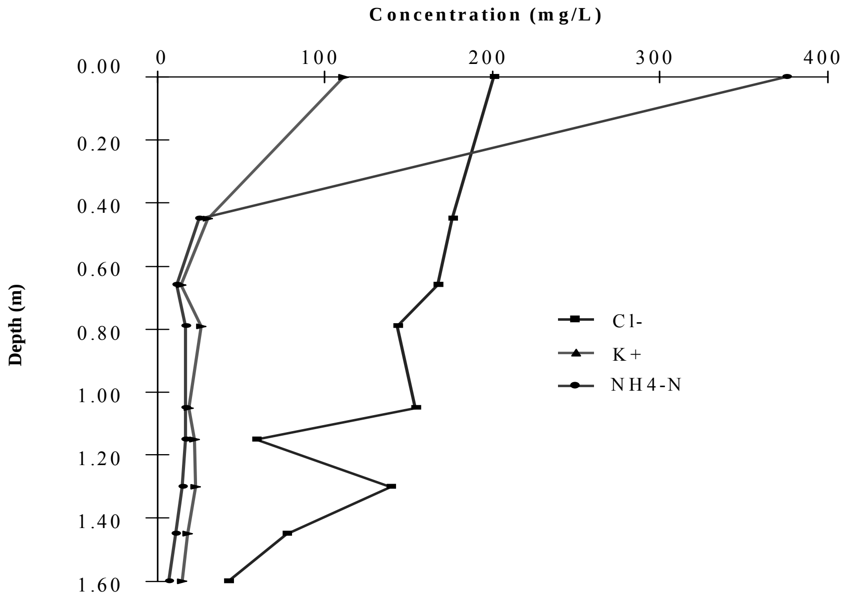


**Figure 6.2. Porewater concentrations determined by paste extraction of soil cores taken from beneath a 10 year old Saskatchewan EMS site constructed in clay till (after Fonstad and Maule [1996]).**

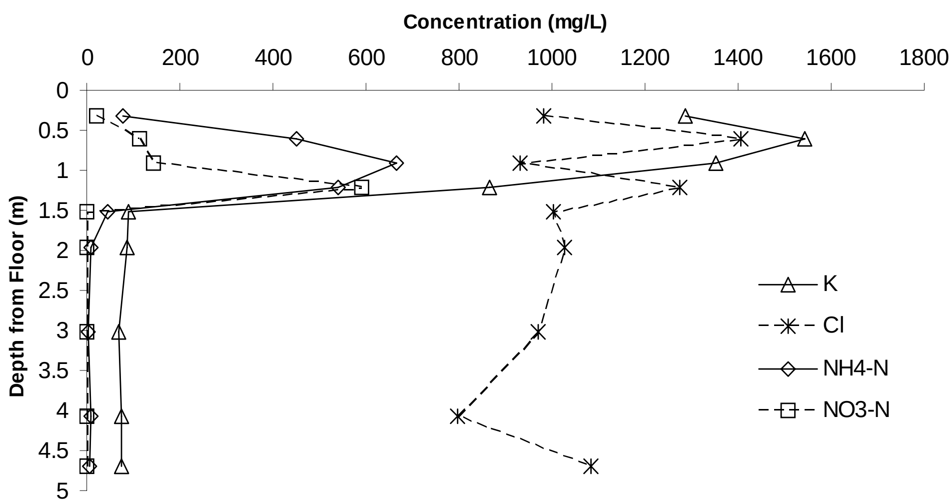


**Figure 6.3. Porewater concentrations determined by paste extraction of soil cores taken from beneath a 17 year old Saskatchewan EMS site constructed in clay till (after Fonstad and Maule [1996]).**

Miller et al. [1976] investigated the seepage of swine wastewater from a two year old Ontario EMS site constructed on clay till soil (Figure 6.4) and another two year old Ontario EMS site constructed on lacustrine clay (Figure 6.5). The authors measured $NH_4$-N, $NO_3$-N and total-P concentrations from soil cores taken below each EMS site and observed no elevated $NO_3$-N or total-P throughout the sampled profiles. However, elevated $NH_4$-N concentrations were measured to an approximate depth of 0.25 m and 0.20 m in the till and clay sites, respectively (Figure 6.4 & 6.5), suggesting that the migration of $NH_4$-N, given an identical duration, will occur to a greater depth in clay till soil than in a clay-rich soil. This is supported by the estimated $NH_4$-N migration rates of 7.0 cm/yr for the clay-till and 6.0 cm/yr for the lacustrine clay sites. The authors did not provide any Cl concentration data, and therefore an estimate of the retardation factor of $NH_4$-N cannot be extrapolated from the study.

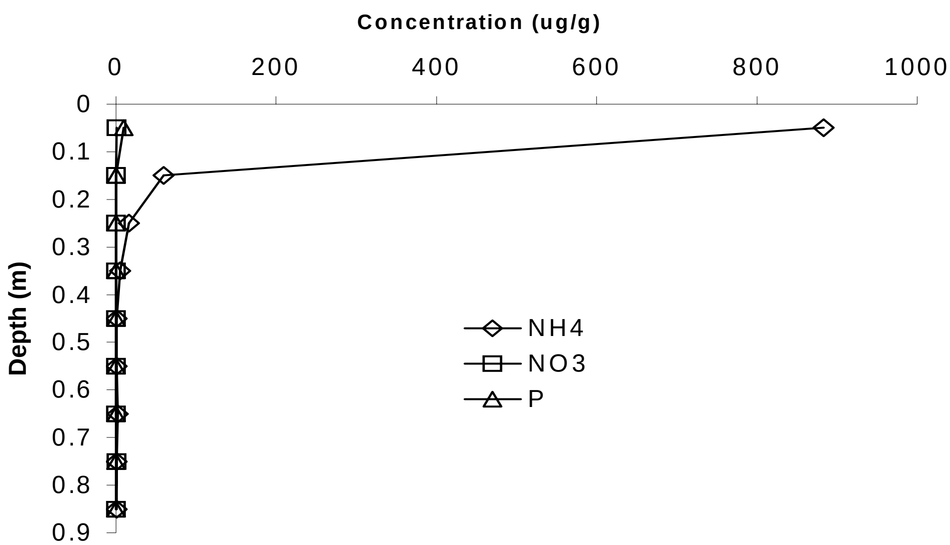


**Figure 6.4. Concentration profiles from soil samples taken from beneath a two year old Ontario EMS site constructed in calcareous clay till (after Miller et al. [1976]). The migration rate of $NH_4$-N is approximately 7.0 cm/yr.**

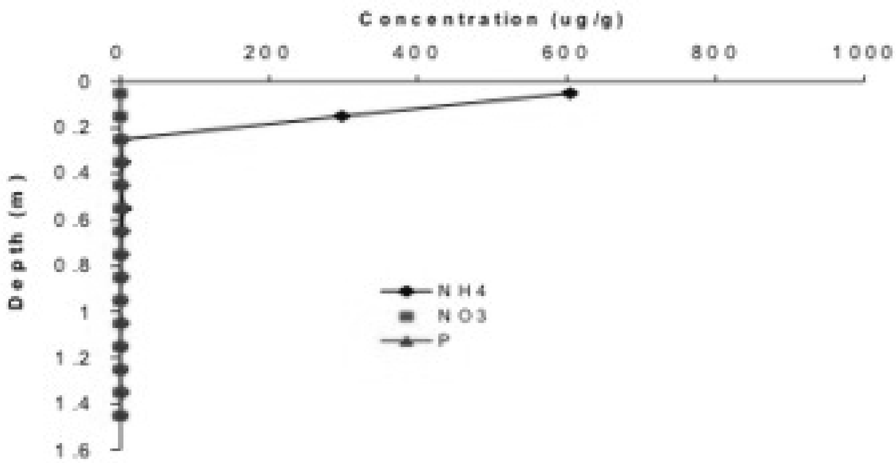


**Figure 6.5. Concentration profiles from soil samples taken from beneath a two year old Ontario EMS site constructed in lacustrine clay (after Miller et al. [1976]). The migration rate of $NH_4$-N is approximately 6.0 cm/yr.**

In a similar study, Fonstad and Maule [1996] reported soil paste extract concentrations from two Saskatchewan swine EMS sites constructed on clay-rich till and in operation for 10 and three years, respectively (Figure 6.2 & 6.6). The migration of Cl and $NH_4$-N from the 10 year old site was observed to occur to depths of >1.6 m and approximately 0.40 m, respectively (Figure 6.2). Therefore, the migration rate of $NH_4$-N is roughly 2.5 cm/yr and is retarded by at least a factor of four. Conversely, the migration rate of $NH_4$-N from the three year old site is approximately 12 cm/yr and is only retarded by approximately a factor of 1.6 (Figure 6.6).

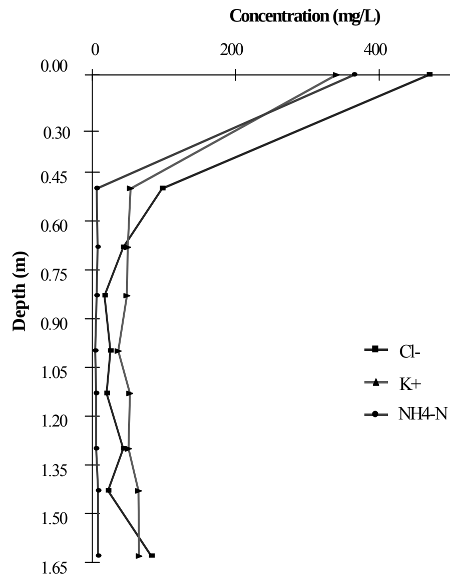


**Figure 6.6. Ion concentrations from saturated paste extraction from soil core sampled beneath a three year old swine EMS site in Saskatchewan constructed in clay till (after Fonstad and Maule [1996]). The migration rate and retardation factor of $NH_4$-N are 12 and 1.6, respectively.**

The seepage of EMS wastewater into sandy clay till can be illustrated from the results presented by Fonstad and Maule [1996] and Miller et al. [1976] (Fig. 6.7 & 6.8). Fonstad and Maule [1996] determined the porewater concentrations of Cl, K and $NH_4$-N from a soil core sampled from beneath a Saskatchewan swine EMS site constructed in sandy clay till and in operation for seven years (Figure 6.7). The authors observed elevated Cl concentrations throughout the entire 1.75 m core interval and elevated $NH_4$-N concentrations to approximately 0.75 m depth. In this case, the retardation factor for the $NH_4$-N, with respect to the Cl, was about 3 and the $NH_4$-N migration rate was approximately 4.5 cm/yr. In the study conducted by Miller et al. [1976] elevated $NH_4$-N concentrations and background $NO_3$-N and total-P concentrations were observed throughout the entire 1.5 m sample depth (Figure 6.8). The distribution between $NH_4$-N and $NO_3$-N concentrations in Figure 6.8 is typical of an anaerobic environment.

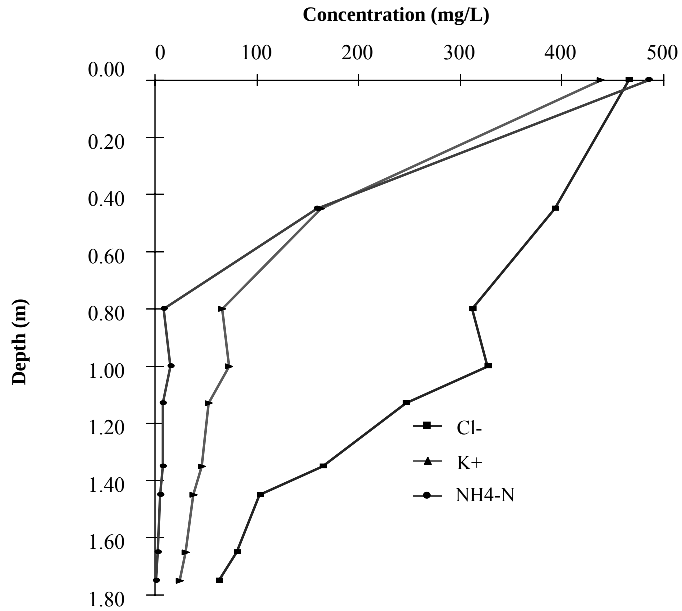


**Figure 6.7. Porewater concentrations determined by paste extraction of soil cores taken from beneath a seven year old Saskatchewan EMS site constructed in sandy clay till (after Fonstad and Maule [1996]). For $NH_4$-N, the retardation factor is approximately 3 and the migration rate is about 4.5 cm/yr.**

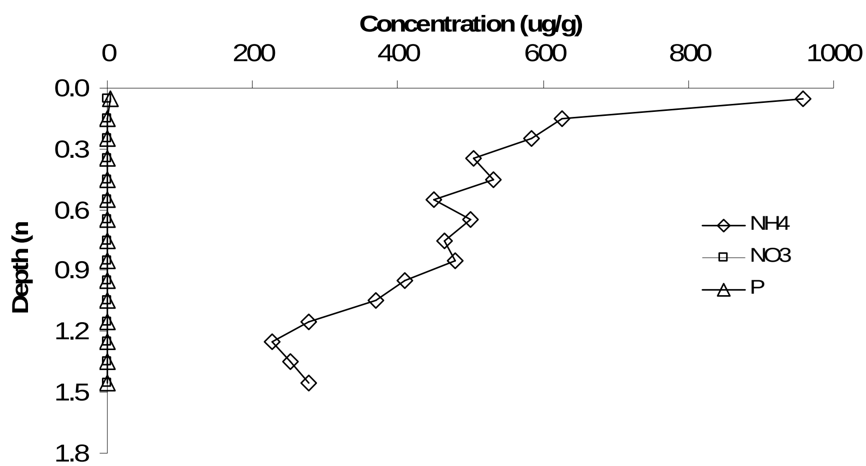


**Figure 6.8. Concentration profiles from soil samples taken from beneath a 10 year old Ontario EMS site constructed in sandy till (after Miller et al. [1976]). The retardation factor of $NH_4$-N is approximately three.**

### 6.1.4. Site Age

Miller et al. [1976] observed an increasing trend in both the maximum $NH_4$-N concentration and infiltration depth with EMS site age (between two and ten years) in the underlying soil profiles, while no significant $NO_3$-N concentrations were observed. A similar trend was noted at a separate Ontario field site by Betcher et al. [1996]. Glanville et al. [2001] observed a slight inverse correlation between seepage rates and storage structure age. Fonstad et al. [2000] concluded that the anaerobic and reducing conditions prevalent beneath active EMS sites reduced significantly reduced the threat of $NO_3$-N migration. However, the oxidation of $NH_4$-N enriched soils beneath abandoned EMS sites has been demonstrated to be of concern [Miller et al., 1976; Culley and Phillips, 1989; Fonstad and Maule, 1996].

### 6.1.5. Pathogens

There was a lack of data related to pathogen contamination of groundwater associated with EMS sites. Most data pertained to the storage of manure and surface water contamination. A recent analysis of stored swine manure, conducted by the USDA, suggests that the dominant bacteria present in stored swine manure are anaerobic members of the *Eubacteria*, *Lactobacillus* and *Streptococcus* groups [USDA, 2000]. Himathongkham et al. [2000] observed the survival of *E. coli* and *Salmonella* in cattle manure to be directly related to a decrease in temperature and suggest that *E. coli* was observed to persist at low levels in stored manure solids and wastewaters. The authors further suggest that manure should be stored for 105 days at 4°C, compared to 45 days at 37°C. In addition, although the observed *E. coli* did not proliferate to significant levels, the authors caution that a significant potential for re-cultivation of the bacteria once outside of the lagoon environment. *E. coli* was observed to persist in swine manure holding tanks from a Quebec study, at 13-16°C, for approximately 30 days [Cote et al., 2006]. Studies by Cote et al. [2006] and Ajariyakhajorn et al. [1997] indicate *Salmonella* persistence in swine manure for durations of 88 days (at 13-16°C) and 56 days (at 4°C), respectively. Johnson et al. [2003] present evidence suggesting a link between *E. coli* contamination of surface water and high livestock densities in Alberta.

### 6.1.6. Pharmaceuticals

Approximately 88% of US swine producers use antibiotics in therapeutic and prophylactic capacities and an estimated >75% are excreted through urine and manure [Elmund et al., 1971]. Tetracycline was detected in EMS sites from eight undisclosed US swine facilities between 11 and 540 ug/L [Campagnolo et al., 2002]. In addition, these authors quantified significant concentrations of tetracycline in groundwater samples collected from an undisclosed distance from the same EMS sites. Mackie et al. [2006] observed variable, detectable concentrations of tetracycline and its breakdown products in groundwater and manure samples collected from a distance up to 30 m downgradient from a seven year old Illinois swine facility constructed on silt loam soil (average of 0.5 ug/L) [Mackie et al., 2006]. However, the detection of antibiotics was variable, which the authors attributed to the potential non-reversible sorption onto the underlying soil and organic matter.

A per animal estrogen excretion rate of 3-6 mg/d for dairy cattle was recently estimated from a Tennessee study. This rate equates to a release rate that is an order of magnitude greater than human waste facilities, when averaged across the total US dairy cattle population [Raman et al., 2004]. Estrogen is a concern because low concentrations (ng/L) can adversely affect the reproductive biology of aquatic vertebrates (fish, turtles, frogs, etc.) by disrupting the normal function of their endocrine systems [Hanselman et al., 2003]. There was a lack of data related to estrogen contamination of groundwater associated with EMS sites.

### 6.1.7. Summary

A critical component needed to assess the impact of EMS facilities on groundwater contamination is the source chemistry. Studies of EMS wastewaters have typically focused on the concentration differences within the vertical profile of an EMS facility. The chemical characterization of manure wastewater is usually conducted on unfiltered samples.

Swine EMS wastewater is characterized by very high concentrations of $NH_4$-N, which can reach upwards of 4,000 mg/L. However, the relative amount of $NO_3$-N is typically low, as a result of the anaerobic conditions prevalent in swine EMS structures. In addition, elevated levels of K and Cl, associated with the urine portion of the wastewater, can reach average concentrations 2,000 and 1,400 mg/L, respectively. In addition, total-P concentrations have been observed approaching 800 mg/L and DOC values between 1,000 and 5,000 mg/L appear common, according to the cited literature.

In contrast to swine, cattle and dairy EMS wastewaters are characterized by much lower $NH_4$-N, $NO_3$-N, total-P, K and Cl concentrations than those observed in swine EMS wastewaters. These components have been observed, in the cited studies, at maximum concentrations of approximately 100, 1.0, 50, 1000, and 1000 mg/L, respectively. We were unable to find any studies that quantified the amount of DOC present in cattle EMS wastewater.

Considerable variability is observed in the source chemistry of swine, cattle and dairy manure wastewaters in the literature. The cause of this variability, both between studied sites and within individual sites, is often attributed to the variability in the operational practices of CAFOs. These practices include, but are not necessarily limited to: feed type, animal density, EMS system employed, and bedding material. Further, the variability in climate amongst studied sites should be considered when determining source chemistry values because both evaporation and precipitation events can alter concentrations even within a single study site [Conn et al., 2007]. In addition, the variability could be an artifact of the sampling method [Fonstad, 2004], as most studies citied in this section did not clearly indicate how wastewater samples were collected.

Historically, the formation of a clogging layer at the base of the EMS has been relied upon to minimize the long-term seepage of manure wastewater into the soil and groundwater underlying EMS sites. Numerous studies indicate these layers, while effective at preventing seepage once formed, cannot be depended upon to prevent seepage of EMS wastewater. In addition, several studies demonstrate that seepage of manure wastewater from EMS sites is controlled both by the material used in structure construction and the underlying geologic media, and therefore both should be considered in the design and construction of EMS sites.

Hydraulic conductivities and seepage rates vary considerable between study sites. Sites situated on sandy-soils, on average, experience greater seepage rates than those situated in oxidized tills and clays, and thus provide the greatest potential for extensive groundwater contamination.

The cited data suggest that EMS structures should not be located on sites characterized by sand/gravel media. In situations where these types of geologic settings cannot be avoided, the construction of an engineered liner, consisting of clay-rich soils, is warranted to retard the movement of contaminants into the underlying soil and groundwater.

The transport of Cl in the subsurface is generally considered to be conservative, undergoing limited reactions in the subsurface. An elevated Cl concentration typically indicates the leading edge of contamination plumes emanating from EMS sites.

In contrast to Cl, the migration of $NH_4$-N undergoes retardation. Because the retardation is controlled by exchange reactions through the replacement of Ca and Mg on exchange sites, retardation of $NH_4$-N within the soil underlying EMS sites is related to the clay content.

According to the data cited in this Chapter, contaminant plumes emanating from EMS sites are characterized by elevated concentrations of $NH_4$-N, total-P, DOC, Ca, Mg, K and Cl, in keeping with the aqueous chemistry of the EMS.

The majority of studies cited in this review focus on delineation of vertical contamination plumes originating from manure storage lagoons, through both soil core and groundwater sample analyses. The horizontal extent of these plumes is not well characterized.

In the case of sandy (permeable) media, the retardation of $NH_4$-N with respect to Cl appears limited. However, the lack of available data on the migration of $NH_4$-N and Cl in the literature precludes an estimation of the retardation factor and migration rates associated with $NH_4$-N. However, given the available contaminant seepage rates determined from a number of studies (between 0.2 and 3. m/year) and limited definition of contaminant plumes in permeable media, an EMS contaminant plume could over 100 years migrate between 20 and 250 m from the EMS.

The retardation of $NH_4$-N with respect to Cl is greater in glacial tills and clays than for sandy media. Available data suggest the retardation can be between a factor of three and seven with rates of migration of $NH_4$-N in the subsurface of between 2 and 7 cm/year. These data suggest that over 100 years of use, the $NH_4$ plume could migrate between 2 and 10 m from the EMS facility.

Because $NH_4$ is the stable N species in anoxic environments, $NO_3$ concentrations in EMS facilities and in the underlying contaminant plume will be low and should remain low as long as anaerobic conditions are maintained. However, discharge of $NH_4$-rich contaminated groundwater into wells or surface waters will result in nitrification of the $NH_4$.

Although the occurrence of $NO_3$-N has been observed in contaminant plumes beneath active EMS sites in North America, associated with the nitrification of $NH_4$-N within the manure wastewater, the prevailing hydrogeologic conditions in Alberta suggest that development of elevated $NO_3$-N concentrations associated with EMS seepage is unlikely.

The nitrification of $NH_4$-N enriched soils (on the exchange sites) beneath abandoned EMS sites has been demonstrated. This observation suggests a potentially large reservoir of oxidizable $NH_4$-N that may enter the groundwater regime at a later date (e.g., after site closure).

Transport of elevated total-P concentrations, associated with manure wastewater, below EMS sites constructed in sand/gravel, till and clay dominant geologic material was not clearly addressed in the available literature.

Evidence suggests that seepage rates from EMS sites decrease with time due to the development of an impermeable manure barrier. The significant variation in observed contamination from EMS seepage within each of the studied soils suggests accurately predicting the potential for contamination from individual EMS sites from the soil texture/grain size alone may be difficult.

The cited data suggest that storage of EMS wastewater is required to reduce the amount of bacteria to acceptable levels. This storage time appears to vary between 50 and 100 days at an average temperature of 4°C and decreases as the storage temperature is increased. Although the requisite data regarding groundwater contamination is lacking, the cited data suggest that bacteria could potentially persist in the groundwater because of its characteristically low temperatures.

The persistence of pharmaceuticals in EMS wastewater and in the surrounding soil and groundwater profiles is not well understood in the scientific literature. Results of the limited number of studies cited suggest that pharmaceuticals rapidly degrade in the subsurface environment. However, the resulting breakdown products have been shown to persist in the same environments and there long-term effects on human and animal health are not well understood at this time.

## 6.2. Cattle Feedlots

### 6.2.1. Source Chemistry

Cattle feedlot manure is traditionally cleaned from pens in the spring and either stockpiled or applied directly to agricultural fields. In addition, bedding material can be incorporated into the manure, especially in colder climates where its use is common practice [Larney et al., 2006]. Whereas EMS wastewater includes runoff collected from cattle feedlots, the manure stored on cattle feedlots will be characterized by lower water contents; its nutrient concentrations are, therefore, typically measured in terms of weight (g/kg). To fully understand the implications of manure storage of cattle feedlots on soil and groundwater quality, the effects of feedlot runoff (also discussed previously) not captured by constructed containment systems must also be considered.

#### 6.2.1.1. Feedlot Solid Manure

Studies characterizing feedlot manure have reported considerable differences in the nutrient content between individual sites. These differences are exemplified in a comparison of studies summarized in Table 6.7 [Gilbertson et al., 1975; Eghball et al., 2000; Olson and Papworth, 2006]. Eghball and Power [1994] indicate that total-P is primarily contained in the feces (96%). Conversely, the majority of N and K are contained in the urine of feedlot cattle, at approximately 58 and 73%, respectively.

*Table 6.7. Average concentrations of feedlot manure from studies conducted in North America.*

| **Parameter** (ppm) | **Gilbertson et al. [1975]** Nebraska | **Olson and Papworth [2006]** Alberta | **Eghball et al. [2000]** Nebraska |
|---|---|---|---|
| N | 1 | 1 | 1 |
| Total-P | 960 | 3,083 | 3,410 |
| $NH_4$-N | 1,390 | 1,035 | 3,006 |
| $NO_3$-N | 1.4 | 112 | 47.0 |
| Na | 1,180 | 1,948 | - |
| K | 4,080 | 8,536 | - |
| Ca | 1,900 | - | - |
| Mg | 1,230 | - | - |

n = number of sites included in study.

### 6.2.1.2. Feedlot Runoff

In addition to the solid manure, manure contaminants are mobilized during runoff events from feedlot surfaces. These events are primarily instigated by rainfall events, and to a lesser extent through snowmelt [White, 2006]. Few studies have attempted to quantify the major contaminants associated with feedlot runoff events [Clark et al., 1975; Coote and Hore, 1979; Edwards et al., 1986] (Table 6.8).

The great variation associated with the volume of runoff produced by precipitation events is evident from the results of several studies, which are summarized in Table 6.9.

***Table 6.8. Average concentrations of feedlot runoff from studies conducted in North America.***

| Runoff Parameter | Clark et al. [1975] | Coote and Hore [1979] | | Edwards et al. [1986] |
|---|---|---|---|---|
| (mg/L) | Texas | Ontario | Ontario | Ohio |
| No. Cattle | 20,000 | 600 | 150 | - |
| Area ($m^2$) | 220,000 | 2450 | 1646 | - |
| Total-N | 1,083 | 772 | 335 | - |
| $NO_3$-N | - | 0.97 | 0.53 | 0.7 |
| $NH_4$-N | - | 264 | 86 | 209 |
| Total P | 205 | 133 | 102 | 118 |
| K | 1,320 | - | - | 701 |
| Ca | 449 | - | - | - |
| Na | 588 | - | - | - |
| Mg | 199 | - | - | - |
| Cl | 1,729 | - | - | - |

***Table 6.9. Percentage of rainfall associated with individual runoff events from feedlots in Canada, USA and Australia (after White [2006]).***

| Author | % Rainfall Associated with Runoff Events | Location |
|---|---|---|
| Coote and Hore [1979] | 19 – 25 | Ontario |
| Gilbertson et al. [1981] | 36 – 86 | USA |
| Lott [1995] | 22 – 50 | Australia |
| Parker et al. [1999b] | 38 | Nebraska |
| Kennedy et al. [1999] | 16 – 40 | Alberta |
| Miller et al. [2003] | 19 | Alberta |

### 6.2.2. Site Hydrogeology

Several studies have demonstrated that distinct layers develop on the floor of cattle feedlots over time. Although these layers vary in depth within and between individual feedlots, they consist of three defined layers: i) an upper loose manure layer; ii) an interfacial compacted manure layer; and iii) a bottom compacted mixture of manure and soil [Mielke et al., 1974; Mielke and Mazurak, 1976; Norstadt and Duke, 1985; McCullough et al., 2001]. Mielke et al. [1971] determined that the compacted interface layer forms as a result of compaction and plugging of the soil pore-space. The soil pore-space is plugged through the dual action of compaction by hoof action and particle dispersion induced by high Na and K urine concentrations [Mielke and Mazurak, 1976]. In addition, Mielke and Mazurak [1976] observed that the downward seepage of manure contaminants become restricted by the formation of the interface layer. However, several studies indicate that the compacted interface does not completely restrict the movement of contaminants below feedlots [Mielke et al., 1974; Elliott et al., 1972; Norstadt and Duke, 1982; Maule and Fonstad, 2002; White, 2006]. Therefore, the interface layer should not be considered as a sufficient barrier to long-term contamination of the soil and groundwater profiles underlying cattle feedlots.

The hydraulic conductivity of feedlot floors has been characterized using a number of methods including ring infiltrometers, tension infiltrometers and laboratory-based falling and constant head permeameters. Ring infiltrometer tests measure field saturated conductivities and involve ponding water within a metal ring and measuring the rate needed to maintain a constant head on the soil surface. Tension infiltrometer tests are representative of the soil matrix conductivity as they keep a slightly negative pressure on the water as it seeps into the soil. Finally, constant or falling heads are applied to undisturbed soil cores and Darcy's law is applied to determine the saturated hydraulic conductivity in constant and falling head permeameter tests. Each method is characterized by particular difficulties in determining field hydraulic conductivities and the method employed should be selected based on the individual study site [White, 2006].

Detailed overviews of feedlot hydraulic conductivities and seepage rates within North America are limited within the scientific literature, with few studies conducted within the past 20 years (Table 6.10). A study conducted by Mielke et al. [1974] on an active Nebraska feedlot observed no measurable infiltration during a 20 day ring infiltrometer test. Mielke and Mazurak [1976] quantified seepage rates and hydraulic conductivities from several soil cores taken from an active Nebraska feedlot. Kennedy et al. [1999] observed a decreasing seepage rate with time during ring infiltrometer tests on a recently active Alberta feedlot. Laboratory analyses were conducted on soil cores taken from a newly constructed Texas feedlot to determine the change in hydraulic conductivity after eight months of active use [McCullough et al., 2001]. The authors observed a significant decrease in the measured hydraulic conductivity, between 5 and 23 times the original values, over the study period, which was attributed to the formation of a manure interface layer. The downward seepage rate through several Saskatchewan feedlots was determined by Maule and Fonstad [2002] through a series of calculations using the observed migration of contaminants into the underlying soil profile. In addition, the authors suggest that contaminants of concern need to be monitored in order to delineate the extent of migration below feedlots, which include $NH_4$-N, $NO_3$-N, Cl, Ca and Mg. Finally, Miller et al. [2003] investigated the difference between the hydraulic conductivity of an Alberta feedlot with and without the uppermost manure layer.

***Table 6.10. Average seepage rates and hydraulic conductivity values determined for selected feedlots.***

| **Author** | **Location** | **Seepage Rate** (cm/d) | **Hydraulic Conductivity** (cm/s) |
|---|---|---|---|
| Mielke et al. [1974] | Nebraska | 0 | - |
| Mielke and Mazurak [1976] | Nebraska | 0.12-0.38 | $6.4 \times 10^{-6}$ - $1.1 \times 10^{-6}$ |
| Kennedy et al. [1999] | Alberta | $4.5 \times 10^{-4}$ – $4.4 \times 10^{-7}$ | $1.8 \times 10^{-7}$ – $7.8 \times 10^{-8}$ |
| McCullough et al. [2001] | Texas | - | $1.8 \times 10^{-5}$ – $5.3 \times 10^{-7}$ |
| Maule and Fonstad [2002] | Saskatchewan | $1.4 \times 10^{-3}$ – $5.5 \times 10^{-4}$ | $1.6 \times 10^{-8}$ - $6.3 \times 10^{-9}$ |
| Miller et al. [2003] | Alberta | - | $6.1 \times 10^{-7}$ – $5.1 \times 10^{-7}$ |

### 6.2.3. Site Contamination

In contrast to EMS site investigations, the relatively small number of studies conducted on cattle feedlots preclude an in-depth examination of the effect of soil type on soil and groundwater contamination resulting from contaminant seepage from stored manure. The most important factor affecting groundwater vulnerability may be the depth to the static water table because research shows the probability of finding contamination caused by feedlots decreases rapidly as the depth to groundwater increases [Maule and Fonstad, 2000]. The rate of seepage from feedlots into the surrounding environment is dependent upon the hydraulic conductivity of both the compacted manure interface layer and the underlying geologic media.

The majority of studies conducted on cattle feedlots typically focus on the movement of nitrogen ($NO_3$-N and $NH_4$-N), with minor attention to Cl, K and P. Nitrate has been detected at elevated concentrations directly beneath cattle and dairy feedlots [Elliott et al., 1972; Partridge and Racz, 1972; Sommerfeldt et al., 1973; Mielke et al., 1974; Coote and Hore, 1979]. However, similar to conditions prevalent beneath EMS sites, denitrification processes beneath cattle feedlots significantly mitigate the transport of $NO_3$-N within the underlying soil column [Stewart et al., 1967; Partridge and Racz, 1972, 1973]. This is attributable to the prevalent anaerobic conditions that persist beneath a majority of feedlot operations [Elliot et al., 1973; Mielke et al., 1974; Schuman and McCalla, 1975, Nordstadt and Duke, 1982].

#### 6.2.3.1. Sand Gravel

Gilbertson et al. [1971] observed elevated total-N, $NO_3$-N and total-P concentrations in soil cores sampled from beneath a Nebraska feedlot in operation for one year with an animal density of 18.6 $m^2$/head (Fig. 6.9, Gilbertson et al. [1971]). The soil type was not noted by the authors; however, the large migration rate of $NO_3$-N (50 cm/yr) suggests it was a very coarse grain soil. A study conducted on a 10 year old cattle feedlot in Florida, constructed on a sandy loam soil, noted an increase in Ca, K, Na and total-P in the upper 30 cm layer with time, and with depth over time (Fig. 1.10, Dantzman et al. [1983]). Although the authors did not quantify the concentration of $NH_4$-N or $NO_3$-N, an average rate of migration can be calculated from the observed contaminant species, which is approximately between 2 and 4 cm/yr

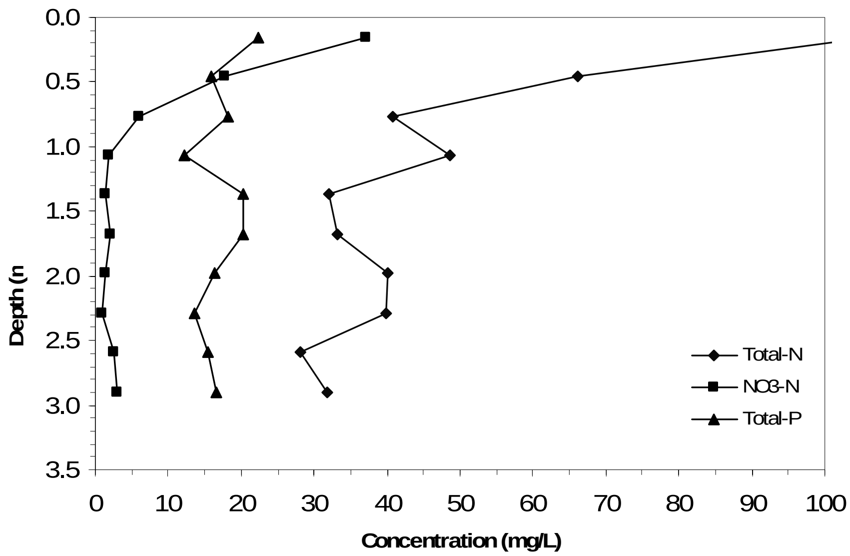


**Figure 6.9. Relative average paste extract concentrations in a soil core sampled from a Nebraska feedlot with an animal density of 18.6 $m^2$/head after one year of operation. Total-N concentrations are expressed (x 10) mg/L (adapted from Gilbertson et al. [1971]). The migration rate is approximately 50 cm/yr for $NO_3$-N, suggesting a very coarse grain soil type.**

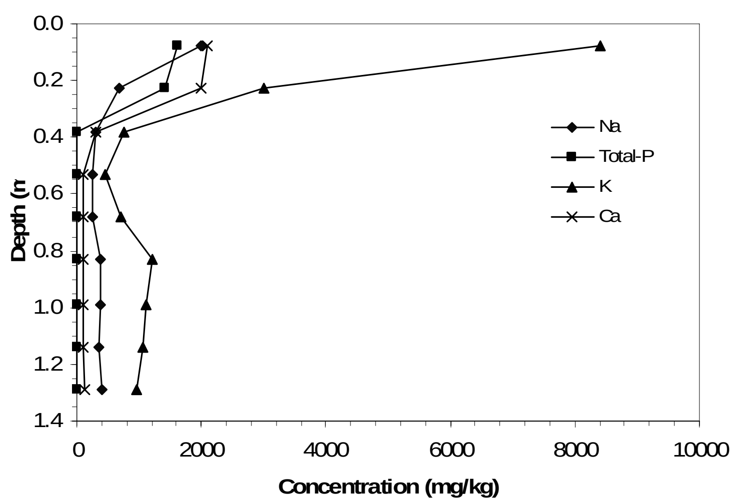


**Figure 6.10. Average paste extract concentrations in a soil core sampled from a 10 year old Florida feedlot constructed on a sandy loam soil with an animal density of 22.4 $m^2$/head (adapted from Dantzman et al. [1983]).**

In a study of two 30 year old Saskatchewan feedlots constructed on coarse textured sandy soil (Site 2) and sandy-till (Site 3), Maule and Fonstad [2002] observed elevated $NO_3$-N and $NH_4$-N concentrations to depths of about 1.5 m of the sampled depth profile (Fig. 6.11). Significant Cl concentrations were observed to depths between 3 and 4 times those measured for $NH_4$-N suggesting a retardation factor between 3 and 4 for $NH_4$-N migration. In addition, the migration rates of $NH_4$-N are approximately 1.0 cm/year and 2.5 cm/year for the sandy soil (Site 2) and sandy-till (Site 3), respectively. The presence of greatly elevated $NO_3$-N and lower $NH_4$-N concentrations in the sandy soil (Site 2) suggests the presence of elevated oxygen levels with depth, which is probably a result of the coarse nature of the soil. Finally, the amount of contamination observed by these authors greatly exceeds that indicated in the other studies cited in this section (Fig. 6.10 to 6.12), which were characterized by finer grained soils. Therefore, similar to the observations made regarding EMS studies, the extent of contamination in coarse grain soils such as those detailed by Maule and Fonstad [2002] is much greater than that observed in finer grained soil types [Gilbertson et al., 1971; Elliott et al., 1972; Dantzman et al., 1983].

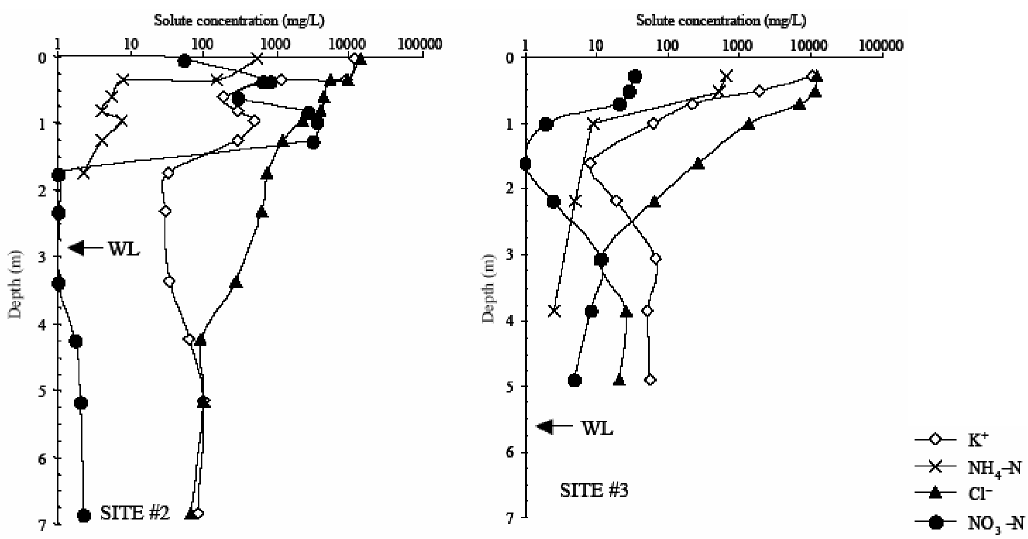


**Figure 6.11. Average paste extract concentrations in a soil core sampled from two 30 year old Saskatchewan feedlots constructed on coarse textured sandy soil (Site 2) and sandy till soil (Site 3) (from Maule and Fonstad [2002]). The migration rates are 1.0 cm/year (Site 2) and 2.5 cm/year (Site 3) and the retardation factors are >5.0 (Site 2) and >5.0 (Site 3).**

Campbell and Racz [1975], investigating a 13 year old feedlot constructed on sandy soil, observed minimal P contamination (to a studied depth of 3.0 m) of underlying soil and groundwater, which was attributed to the rapid mineralization of organic-P under anaerobic conditions. Similarly, Gilbertson et al. [1971] did not observe any significant increase in Total-P concentrations, above background levels, for soil cores taken from beneath a Nebraska feedlot constructed on coarse grained soil (Fig. 6.9).

The Minnesota Pollution Control Agency [2001] investigated the extent of lateral migration of contaminants from four open feedlots with no liquid manure storage (2 beef, one diary, and one hog) that were in operation for more than 20 years on coarse textured soils (sand and gravel, coarse sand, loamy sand). They determined that the contaminant plume emanating from the feedlots extended from >30 m to >130 m. Elevated concentrations of $NH_4$ (attributed to the anoxic conditions in the plume), phosphorus, organic N and organic carbon were measured in the plumes. In two of the plumes, high concentrations of $NO_3$ were measured. The presence of $NO_3$ was attributed to the coarse texture of the soils and deep depths to the water table.

PFRA [2002] investigated the spatial extent of an $NO_3$ plume in a discontinuous, thin, and often silty sand in contact with fractured till and bedrock shale located south of Regina, SK. The $NO_3$ plume extends 60 and 90 m downgradient from livestock pens. The authors suggest that the $NO_3$ is derived from the oxidation of $NH_4$ in the manure and, based on $NO_3$/Cl ratios, that some denitrification occurs in the plume.

#### **6.2.3.2. Oxidized Till and Clay**

Elliott and McCalla [1972] demonstrated anaerobic conditions by measuring the percent by volume concentration of $O_2$, $CO_2$ and $CH_4$ gases in the sandy-silt loam soil beneath a 15 year old Nebraska feedlot (Fig 6.12), where the presence of significant $CH_4$ concentrations suggested an anoxic environment. The dominant N species observed by Elliott et al. [1972] in soil cores taken from beneath the same 15 year old Nebraska feedlot, with an animal density of 37 $m^2$/head, was $NH_4$-N (Fig. 6.13). The observed $NH_4$-N concentration profile translates to a migration rate of 2.0 cm/yr; however, a retardation rate cannot be determined as the authors did not present Cl concentrations.

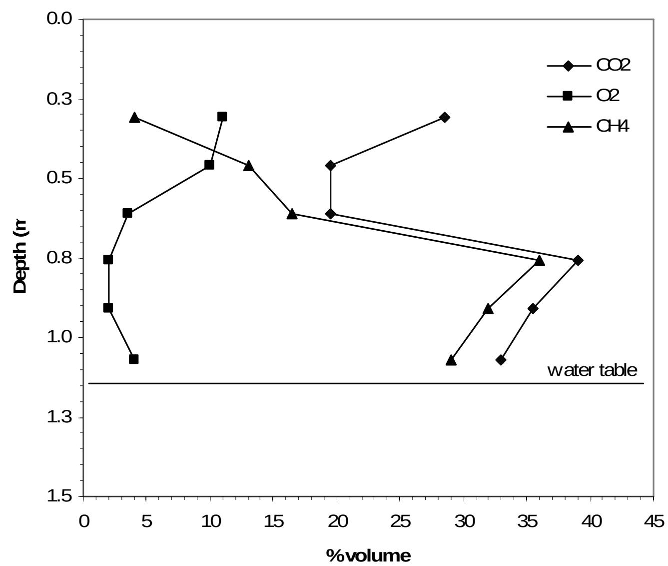


**Figure 6.12. Average gaseous concentration of soil pore-space beneath a 15 year old Nebraska feedlot constructed sandy-silt loam soil with an animal density of 37 m$^2$/head (adapted from Elliott and McCalla [1972]).**

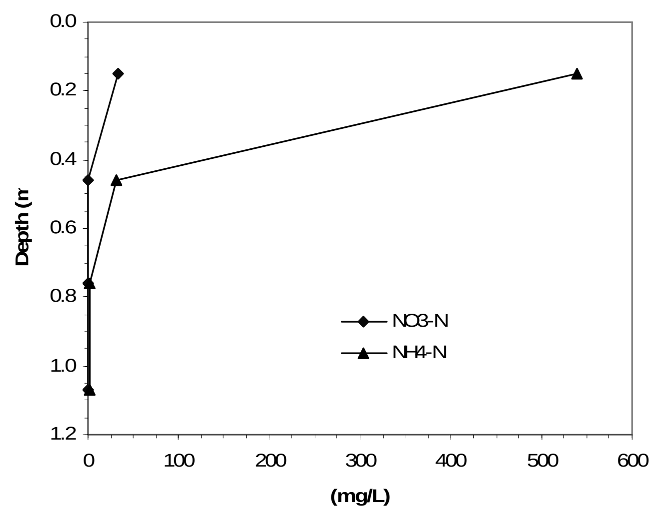


**Figure 6.13. Average yearly concentration of nitrogen species beneath a 15 year old Nebraska feedlot constructed on sandy-silt loam soil with an animal density of 37 m$^2$/head (adapted from Elliot et al. [1972]). The migration rate of $NH_4$-N is about 2.0 cm/year.**

### 6.2.4. Site Age

Zhu et al. [2004] conducted a comparison study of four active Minnesota dairy feedlots, all constructed on loam soil, in operation for 20, 40, 60 and 100 years. Soil cores were sampled to a maximum depth of 1.53 m from all four lots and the $NH_4$-N and $NO_3$-N concentrations are summarized in Table 6.11. Overall, the authors observed an increase in the concentration and migration depth of $NH_4$-N and $NO_3$-N with an increase in feedlot age. While the migration rates of $NH_4$-N and $NO_3$-N cannot be calculated from the observed data, the depth of migration, even in the 100 year old site, is relatively small. In addition, the presence of elevated $NO_3$-N concentrations suggests nitrification processes are occurring in the underlying soil profile.

Eigenberg and Nienaber [2003] observed elevated $NO_3$-N concentrations in soils beneath feedlot manure composting piles constructed on clay loam soil, operated for three years and abandoned for an additional four years. Measured $NH_4$-N concentrations were at background levels throughout the sampled profile. However, peak Cl concentrations had migrated to a depth of approximately 1.2 m and corresponding peak $NO_3$-N concentrations (104 mg/kg) were observed between 1.0 and 1.2 m. Although not considered by these authors, the corresponding low $NH_4$-N concentrations suggest that the usual reduced environment observed beneath manure storage facilities had become oxidized, resulting in the nitrification of $NH_4$-N to $NO_3$-N.

***Table 6.11. Nitrate and ammonium nitrogen distribution in the soil profile of four dairy cattle feedlots in southern Minnesota (after Zhu et al. [2004]).***

| Depth | 20 yrs (mg kg$^{-1}$) | | 40 yrs (mg kg$^{-1}$) | | 60 yrs (mg kg$^{-1}$) | | 100 yrs (mg kg$^{-1}$) | |
|---|---|---|---|---|---|---|---|---|
| (cm) | $NO_3$-N | $NH_4$-N | $NO_3$-N | $NH_4$-N | $NO_3$-N | $NH_4$-N | $NO_3$-N | $NH_4$-N |
| 30 | 31.8 | 125 | 108.1 | 152 | 153.2 | 739 | 195.7 | 1036 |
| 61 | 17.5 | 119 | 40.2 | 73 | 133.6 | 913 | 255.5 | 1674 |
| 92 | 16.4 | 93 | 42.6 | 70 | 88.7 | 499 | 307.2 | 2469 |
| 122 | 13.8 | 70 | 31.9 | 41 | 65.3 | 366 | 271.2 | 1962 |
| 153 | 14.9 | 73 | 23.5 | 44 | 53.1 | 185 | 72.1 | 1363 |

### 6.2.5. Pathogens

The available data on pathogens associated with cattle feedlot operations focus strictly on surface contamination. We found no data on the potential contamination and persistence of pathogens of groundwater associated with cattle feedlots. The most commonly studied pathogen originating from cattle feedlots is *E. coli*, specifically *E. coli* 0157:H7 [APHIS, 1999a; Galland et al., 2001; Burkholder et al., 2007]. Cattle infected with *E. coli* show no signs of disease, and shedding of the species through the manure is sporadic and difficult to detect. In a comprehensive study involving 11 of the top producing beef cattle states in the US, 73 feedlots were selected and 25 manure samples were taken from three pens on each site over the course of a year [APHIS, 1999a]. Approximately 11% of the total samples tested positive for *E. coli*, with the highest rate observed in September and lowest in February; no geographic trend was observed in the data. In addition, all 73 feedlots had at least one positive test, suggesting that *E. coli* is widely distributed amongst cattle feedlots in the US. Similarly, in a study of four feedlots in Kansas, of 24,184 samples collected only 45 were positive for *E. coli* (0.2%), with 44 from manure samples and one from a water trough [Galland et al., 2001].

The US Animal and Plant Health Inspection Service [APHIS, 1999b] conducted a study on *Salmonella* in conjunction with the previously described *E. coli* study, using the same collected manure samples. A total of 6.3% of samples tested positive for *Salmonella*, and a higher incidence rate was observed in samples collected from the southern states relative to northern states. In addition, the majority of *Salmonella* strains identified in the study were not associated with those known to cause human illness.

*Cryptosporidium parvum* (*C. parvum*) is a waterborne pathogen that has been identified within cattle manure at rates between 0 and 10% (e.g., Villacorta et al. [1991]; Atwill et al. [1998, 2003]; Huetink et al. [2001]) or higher [Scott et al., 1995; Grazyck et al., 2000], with most studies focused on dairy feedlot cattle [Atwill et al., 2006]. Atwill et al. [2006] investigated the prevalence of *C. parvum* by collecting 5274 manure samples from 22 feedlots in seven states and found detectable levels in only nine samples (0.17%), which is similar to the results of Hoar et al. [1999]. Hoar et al. [1999] noted that the prevalence of *C. parvum* in manure collected from feedlot floors (0.18%) was an order of magnitude less than samples collected directly from the cattle rectum (1.1%), suggesting *C. parvum* is susceptible to environmental stresses that reduce the overall environmental loading of the parasite.

Although the incidence rate of *E. coli* in cattle feedlots and other CAFO types has been demonstrated to be extremely low, human outbreaks of *E. coli* O157:H7 and *Salmonella* have originated from a variety of animals, including cattle [Tuttle et al., 1999; Michel et al., 2006]. Of particular concern is the persistence of pathogens released into the environment surrounding feedlots; *E. coli* O157:H7 can be recovered from environmental water for up to 12 weeks [Porter et al., 1997]. The prevalence of *E. coli* in Canada is wide-spread and a significant body of research has been developed on its source and occurrence within the surface environment of Alberta feedlots and surface water bodies (e.g., Van Donkersgoed et al. [2001]; Hyland et al. [2003]; Larney et al. [2003]; Johnson et al. [2003]; Gannon et al. [2004]; Stanford et al. [2005]; Byrne et al. [2006]). Therefore, as a cautionary step, the prevalence, transportation and persistence of these pathogens within cattle feedlots need to be considered.

### 6.2.6. Pharmaceuticals

The available data on pharmaceutical contamination associated with cattle feedlots focus on surface contamination. We found no data on the potential contamination and persistence of pharmaceuticals of groundwater associated with cattle feedlots. Tetracycline, an antibiotic, is used in cattle feedlots as the treatment of sick animals and as a preventative measure against the transmission of respiratory infection via cattle moving from one herd to another [Peak et al., 2007]. In a study of the persistence of tetracycline and its breakdown products in feedlot runoff, Peak et al. [2007] observed an increased incidence in the runoff from feedlots with higher usage rate of the antibiotic and during the autumn months. In addition, the authors suggest that without capture of feedlot runoff in storage lagoons a substantial release of tetracycline and associated resistant genes would occur to associated waters. Conversely, a study of the antimicrobial drug tylosin observed a relatively rapid breakdown in cattle manure (6.2 day half-life), which suggests it does not persist in the environment [Teeter and Meyerhoff, 2003]. These observations are also supported by the results of Gavalchin and Katz [1994] and

Kolpin et al. [2002], who investigated the degradation of tylosin in soils and surface waters, respectively.

Antibiotics beyond tetracycline and tylosin are employed in typical feedlot practices within North America (e.g., amoxicillin, ampicillin, bacitracin, dihydrosteptomycin, neomycin, tilmicosin, sulfadimethoxine; Committee on Drug Use in Food Animals, [1999]). No in-depth investigations into the persistence of antibiotics in feedlot runoff were identified in this review.

### 6.2.7. Summary

Although the number of studies conducted on cattle feedlots is limited, they comprise a large amount of the scientific literature on the source chemistry, hydrogeology and subsurface contamination association with feedlots. As a result, a number of Alberta studies have been cited in this Chapter, in addition to being included in the previous Chapter.

Considerable variability is observed in the source chemistry of manure and runoff wastewater from cattle feedlots. The variation in feedlot cattle manure chemistry can be attributed to variations in cattle type, diet and feedlot management practices and climatic variables.

Similar to cattle EMS wastewater, feedlot runoff concentrations can exhibit variations in concentration, which are attributed to fluctuations in the length and intensity of rainfall events. These events control the dilution experienced by the feedlot manure.

Examination of the scientific literature shows that studies of feedlot cattle manure have focused, for the most part, on its suitability as a nutrient source for application to agricultural fields. Consequently, studies of feedlot manure are typically limited to quantifying the available nutrient content (total-N and total-P).

To address the potential for seepage and transport of manure-derived contaminants, the source terms (leachable fractions) of associated contaminants (e.g., $NH_4$-N, $NO_3$-N,) must be quantified to obtain an understanding of the migration of contaminants in groundwaters associated with feedlots.

Similar to the clogging layers that form in EMS structures, the formation of the interface layer in feedlots has been relied upon to prevent the long-term seepage of manure wastewater into the underlying soil and groundwater. Numerous studies have shown that this layer once formed can be effective at reducing seepage. It cannot, however, prevent seepage into the underlying soil and groundwater.

The majority of studies cited in this review focus on delineation of vertical contamination plumes originating from cattle feedlots, through both soil core and groundwater sample analyses. The horizontal extent of these plumes is not well characterized.

Although no specific studies have been conducted on the subject, data suggest that the migration of $NH_4$-N from feedlots is likely controlled by exchange reactions, through the replacement of Ca and Mg on exchange sites (as was the case for EMS facilities). As a result, the retardation of $NH_4$-N within the soil underlying feedlots should be directly related to the clay content of the soil.

In the cases studied, the migration rate of $NH_4$ is estimated to be between 1 and 3 cm/yr for finer grained soils and as high as 50 cm/yr in coarser grained soils. These data suggest that over 100 years of CFO operation, the $NH_4$-N plume could migrate between 1 and 50 m from feedlot facilities.

Similar to EMS studies, the concentrations of $NO_3$-N, $NH_4$-N, total-P, K, Na, Ca, Cl and DOC (as previously suggested) are required to properly delineate the extent of contamination below feedlots and to assess the migration (and retardation) of effluent seepage.

The cited studies suggest that the accumulation of N, predominantly as $NH_4$-N due to anaerobic conditions, may be a concern in the long-term operation of cattle feedlots. The introduction of oxygen into the underlying soil can occur through either feedlot closure or removal of the manure interface layer, and may result in the production and mobilization of $NO_3$-N in the groundwater environment. Nitrification of this N should be avoided.

Although the cited data suggest that the incidence rate of *E. coli* and other pathogens in the surface environments of cattle feedlots is low, studies have demonstrated the ability of *E. coli* and other pathogens to persist in the environment for extended periods of time. Furthermore, pathogen persistence has been demonstrated to increase with decreasing temperature, which suggests pathogens could potentially persist in groundwater environments for extended periods of time. Currently, however, there are no data available on the contamination of groundwater by pathogens originating from cattle feedlots.

The persistence of pharmaceuticals in feedlot wastewater and in the surrounding soil and groundwater profiles is not known; it is an emerging field of study in the scientific community.

## 6.3. Poultry Litter/Manure

### 6.3.1. Source Chemistry

#### 6.3.1.1. Litter and Manure

Poultry litter is a mixture of bedding material, typically sawdust, wood shavings, wheat straw, peanut hulls, or rice hulls, and poultry manure [Edwards and Daniel, 1992]. A detailed characterization of poultry litter and manure is required to fully understand the source chemistry involved with the production and storage of poultry waste.

The source chemistry of both poultry litter and manure are summarized in Table 6.12. The values reported by Edwards and Daniel [1992] for both litter and manure concentrations represent an average value complied from an exhaustive literature review of over 20 investigations from the past 40 years. Kelley et al. [1996] collected litter from two poultry barns in Georgia that used wood shavings as bedding material. Faucette et al. [2004] collected aged poultry litter from a Georgia poultry farm, and Tasistro et al. [2004] collected 30 samples from different areas in Georgia poultry barn. Additionally, the concentrations determined by Kpomblekou et al. [2002] and Kpomblekou [2006] include the average concentrations determined from 25 samples incorporating a wide variety of bedding materials used in poultry production.

A comparison with cattle and swine manure (Table 6.13) demonstrates greater concentrations of total-N and total-P are associated with poultry litter [Hooda et al., 2000]. In addition, an investigation by Nahm [2003a] indicates that between 60 and 70% of the total-N is present in the organic form. Phosphorus is also found in higher concentrations, relative to swine and cattle manure, mainly in the inorganic form (60-90% of total-P) [Nahm, 2003b].

***Table 6.12. Average concentrations of poultry litter and manure from poultry production sites across North America. Results from Edwards and Daniel [1992] incorporate average results from several studies.***

| **Parameter** | **Edwards and Daniel [1992]** | | **Kelley et al. [1996]** | **Faucette et al. [2004]** | **Tasistro et al. [2004]** | **Kpomblekou et al. [2002] / Kpomblekou [2006]** |
|---|---|---|---|---|---|---|
| **(g/kg)** | **Litter** | **Manure** | **Litter** | **Litter** | **Litter** | **Litter** |
| Total-C | 376 | 289 | - | - | 342 | - |
| Total-N | 40.8 | 46.0 | - | - | 37.4 | 411 |
| $NH_4$-N | 2.60 | 14.4 | - | 0.04 | 6.40 | 3.03 |
| $NO_3$-N | 0.20 | 0.40 | - | 4.88 | - | - |
| Total-P | 14.3 | 20.7 | 14.1 | 35.0 | 16.2 | - |
| K | 20.7 | 20.9 | 21.6 | 15.0 | 26.7 | 25.5 |
| Ca | 14.0 | 38.9 | 17.8 | 29.8 | 20.1 | 26.6 |
| Mg | 3.10 | 4.70 | 4.48 | 3.49 | 4.69 | 6.30 |
| Na | 3.30 | 4.20 | - | 4.66 | 7.03 | 6.9 |
| Cl | 12.7 | 24.5 | - | - | - | - |
| Mn | 0.27 | 0.30 | 0.30 | - | 0.60 | 0.39 |
| Zn | 0.19 | 0.35 | 0.32 | 0.26 | 0.30 | 0.40 |
| Cu | 0.06 | 0.05 | 0.32 | - | 0.49 | 0.45 |
| Fe | 0.84 | 0.32 | 0.90 | - | 1.77 | 2.07 |
| Al | - | - | 0.70 | 2.35 | 2.27 | 2.20 |

***Table 6.13. Comparison of total-N and P concentrations in swine, cattle and poultry manure demonstrating the greater concentrations associated with poultry litter (after Hooda et al. [2000]).***

| Animal Type | Dry Matter (%) | Total-N | Total-P |
|---|---|---|---|
| *Solids (kg/t)* | | | |
| Cattle | 25 | 6 | 3.1 |
| Swine | 25 | 6 | 2.6 |
| Poultry | 60 | 29 | 9.6 |
| *Slurries (kg/m)* | | | |
| Cattle | 6 | 3 | 0.5 |
| Swine | 6 | 5 | 1.3 |

Several studies have characterized the trace metal concentrations associated with poultry litter (Table 6.12) since they are routinely included as additives in poultry feedstock and are readily soluble once excreted [Kpomblekou et al., 2002; Jackson et al., 2003]. A comparison of these concentrations shows the poultry manure, on average, contains more N, P, Cl, Ca, Na, Cu and Zn than poultry litter. In addition, poultry manure contains a greater water content, as it is not mixed with bedding material [Edwards and Daniel, 1992].

The composition of poultry litter and manure vary considerably. The variability is attributed to a variety of factors including: the number of flocks grown on the same litter; type of bedding material used; poultry age and type; animal density; feedstock type; climatic conditions; and nutrient losses during storage [Edwards and Daniel, 1992].

#### 6.3.1.2. Litter Stockpiles

A recent study documented the elevated total-N and total-P concentrations associated with runoff water collected from covered and uncovered poultry manure stockpiles (Table 6.14, Felton et al. [2007]). Felton et al. [2007] did not observe any significant differences in total-N or total-P concentrations between covered or uncovered, stored wet or stored dry, poultry manure stockpiles.

*Table 6.14. Average characteristic concentrations of runoff wastewater poultry litter stockpiles (after Felton et al. [2007]).*

| Parameter | Dry | | Wet | |
|---|---|---|---|---|
| (mg/L) | Covered | Uncovered | Covered | Uncovered |
| Total-N | 10.8 | 5.37 | 0.75 | 21.2 |
| $NO_3$-N | 2.94 | 2.75 | 9.80 | 9.53 |
| Total-P | 9.42 | 13.0 | 0.28 | 5.45 |
| $PO_4$-P | 1.28 | 0.79 | 0.21 | 10.8 |

### 6.3.2. Site Hydrogeology

The majority of available poultry litter investigations focus on the effects of land application on soil and groundwater contamination (e.g., Jackson et al. [2006]; Mitchell and Tu [2006]; Pengthamkeerati et al. [2006]; Pirani et al. [2006]; Sadeghi et al. [2006]; Jalai and Khanboluki [2007]; Moore and Edwards [2007]), which is beyond the scope of this study. In contrast, a detailed search of the literature for the hydrogeologic characteristics of poultry manure and litter storage did not reveal any documented case studies.

### 6.3.3. Site Contamination

As was the case for studies involving site hydrogeology, most site contamination studies address contamination of soil and groundwater associated with the spreading of poultry litter on agricultural land as a nutrient source. A limited database of groundwater and soil contamination is available on the on-site stockpiling of litter prior to field spreading and soil contamination beneath poultry barn floors, which are presented in the following sections.

#### 6.3.3.1. Groundwater Contamination

A limited number of studies investigated the direct contamination of groundwater from poultry litter production and storage. A detailed study involving four poultry production sites in Florida, located on sandy-clay soils, indicated that poultry manure spreading was responsible for elevated concentrations of $NO_3$-N, K and Cl in groundwater monitoring wells located at each site [Hatzell, 1995]. However, slightly elevated $NO_3$-N concentrations, to a maximum of 4.0 mg/L, observed in two monitoring wells on Site 4 correlated well with measured organic-N, which is present in elevated levels in poultry litter stockpiles (Table 6.15).

***Table 6.15. Maximum concentrations measured in groundwater monitoring wells adjacent to poultry litter stockpiles constructed on sandy-clay soils in Florida (after Hatzell [1995]).***

| Measured Parameter (mg/L) | Maximum Concentration Well 4-1 | Maximum Concentration Well 4-2 |
|---|---|---|
| $NO_3$-N | 4.0 | 2.6 |
| $NH_4$-N | 0.06 | 0.01 |
| Total-P | 0.08 | 0.04 |
| Organic-N | 0.75 | 0.20 |
| Organic-C | 1.9 | 1.6 |
| Cl | 5.0 | 4.2 |
| K | 1.3 | 0.7 |

Ritter et al. [1994] monitored groundwater surrounding both covered and uncovered litter stockpiles for three years. Measured $NH_4$-N reached a maximum concentration of 1.15 mg/L and $NO_3$-N was observed to increase over time in downgradient groundwater monitoring wells.

The contamination of the phreatic sand and gravel Abbotsford-Sumas aquifer southern British Columbia and northern Washington State by nitrate is attributed to the leaching of N from the long-term land application of poultry manure and the uncontrolled storage of poultry stockpiles [Wassenaar et al., 2006]. Rapid leaching of the applied and stored N is attributed to intense fall rains. The concentrations of Cu and Zn (which are used as additives in poultry feedstock) in all groundwater samples collected from this aquifer were not elevated (M.J. Hendry, unpublished data, 2005).

### 6.3.3.2. Soil Contamination

Lomax et al. [1997] conducted a detailed study of 30 Delaware poultry barns to compare the $NO_3$-N, $NH_4$-N and organic-N concentrations in the soil with depth below the barn floors. Three floor soil types were investigated, including 19 loose soil floors with an average operation age of 21 years, nine hard soil floors with an average age of 13 years, and two concrete floors with an average age of 30 years. Of the three floor types, concrete floors were demonstrated to substantially reduce the amount of nitrogen, in all forms, entering the underlying soil profile (Figures 6.14-6.16). Given the average age of each floor type and the average $NH_4$-N concentrations, the infiltration rates of $NO_3$-N are approximately 3, 5 and 0 cm/yr for the loose soil, hard soil and concrete floors, respectively. In addition, the authors observed 97 to 99% less nitrogen in soil samples taken from outside of the studied poultry barns, suggesting that contamination from poultry farms is limited to the area directly beneath the earthen barn floors.

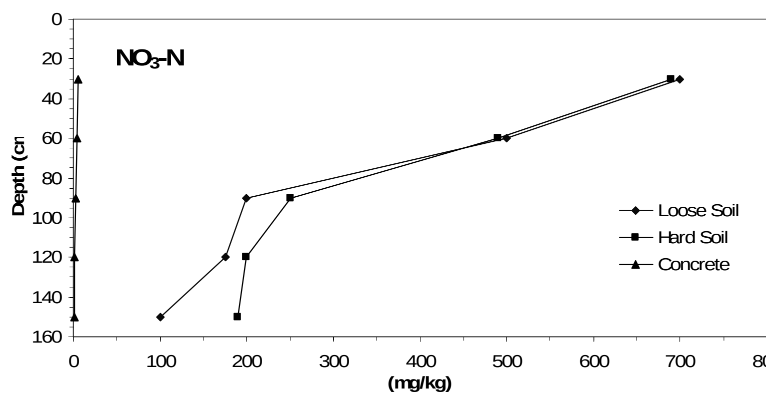


**Figure 6.14. Average $NO_3$-N concentrations observed beneath Delaware poultry barns constructed with loose soil, hard soil, and concrete floors (after Lomax et al. [1997]).**

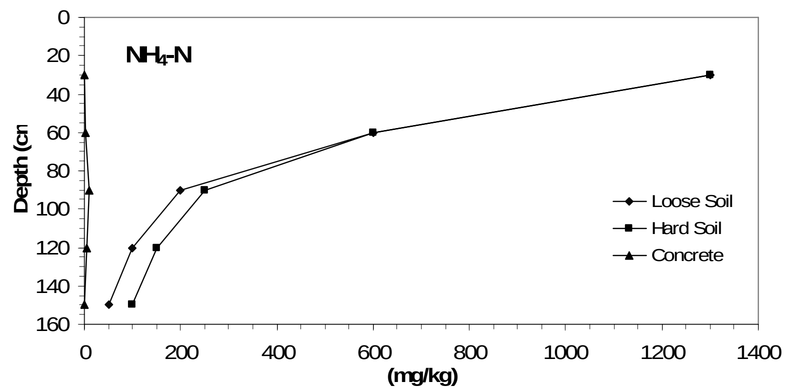


**Figure 6.15. Average $NH_4$-N concentrations observed beneath Delaware poultry barns constructed with loose soil, hard soil, and concrete floors (after Lomax et al. [1997]).**

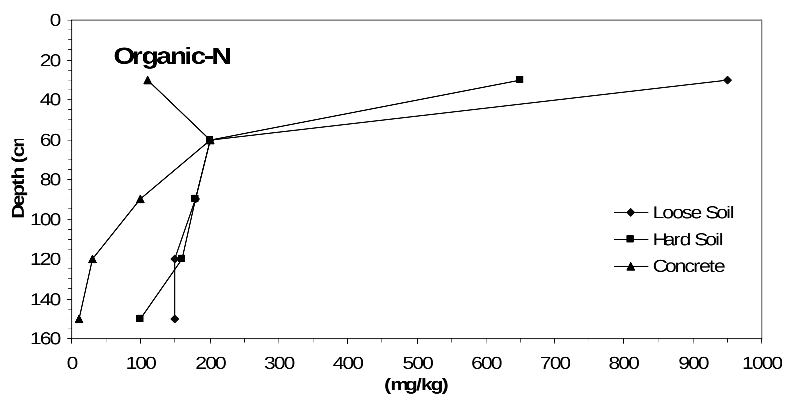


**Figure 6.16. Average organic-N concentrations observed beneath Delaware poultry barns constructed with loose soil, hard soil and concrete floors (after Lomax et al. [1997]).**

A British study on 20 year old poultry litter stockpiles constructed on unsaturated chalk noted elevated concentrations of $NO_3$-N, $NH_4$-N, Cl and organic-C to a depth of 5 m, with $NH_4$-N only observed in the upper 2 m depth [Gooddy, 2002], suggesting it is retarded by a factor between two and three. However, the authors noted that below approximately 2 m depth the soil was characterized by oxidizing conditions, which corresponded with the observed decrease in $NH_4$-N concentrations and significant increase in $NO_3$-N concentrations.

Significant loading of $NH_4$-N (approximately 1000 mg/kg) was measured directly beneath a six year old manure stockpile, constructed on a coarse grained, cultivated, agricultural field in British Columbia, to a depth of 3.7 m, while maximum elevated total-P concentrations of 730 mg/kg were observed within the initial 0.60 m depth [Zebarth et al., 1999]. Elevated $NO_3$-N concentrations, to a maximum of approximately 30 mg/kg were also observed directly beneath the stockpile; however, comparison to values determined prior to construction of the stockpile suggests they were not significantly different than background concentrations (Figure 6.17).

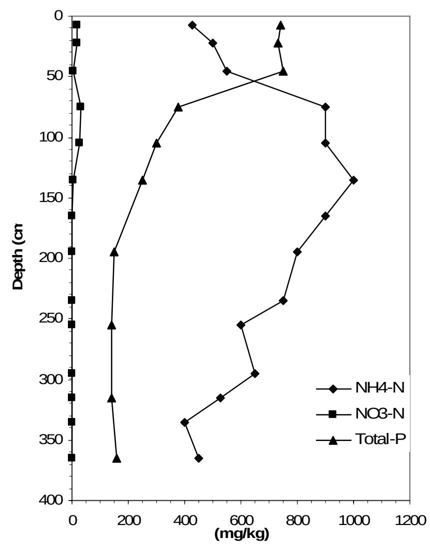


**Figure 6.17. Average soil core concentrations sampled from beneath a poultry manure stockpile constructed on a coarse grained, cultivated, agricultural field (after Zebarth et al. [1999]).**

In a parallel study, soil sampling conducted after removal of covered and uncovered stockpiles demonstrated a continued leaching of contamination through the profile, characterized by a significant increase in $NO_3$-N concentrations (Zebarth et al. [1999]; Table 6.16). In addition, more nitrogen was mobilized into the underlying soil profile from uncovered manure stockpiles, which was attributed to increased seepage associated with heavy rainfall events that are prevalent in the British Columbia study area [Zebarth et al., 1999].

***Table 6.16. Average $NH_4$-N and $NO_3$-N porewater concentrations beneath covered and uncovered poultry manure storage stockpiles constructed on coarse-textured cultivated soil in British Columbia (after Zebarth et al. [1999]).***

| Depth | Covered Stockpile | | Uncovered Stockpile | |
|---|---|---|---|---|
| (cm) | $NH_4$-N | $NO_3$-N | $NH_4$-N | $NO_3$-N |
| *Before pile formation (kg/ha)* | | | | |
| 0-15 | 8 | 33 | 9 | 54 |
| 15-30 | 8 | 27 | 7 | 31 |
| 30-210 | 116 | 129 | 125 | 210 |
| *After pile removal (kg/ha)* | | | | |
| 0-15 | 371 | 237 | 2123 | 8 |
| 15-30 | 54 | 130 | 33 | 60 |
| 30-210 | 97 | 244 | 248 | 324 |
| *After soil left bare (kg/ha)* | | | | |
| 0-15 | 54 | 326 | 552 | 252 |
| 15-30 | 9 | 147 | 257 | 176 |
| 30-210 | 21 | 312 | 578 | 772 |

### 6.3.4. Site Age

We were unable to find any documented case studies involving poultry production effects of storage site age.

### 6.3.5. Pathogens

Gooddy [2002] observed no detectable fecal bacteria in the contaminated soil beneath a long-term (20 years) turkey litter stockpile constructed on chalk in Britain. Testing of both fresh and composted poultry litter, by Hartel et al. [2000], resulted in predominantly non-detectable counts of fecal coliforms. In addition, litter samples spiked with fecal coliforms demonstrated reductions in the pathogen to below detectable limits within eight days [Hartel et al., 2000]. Investigations by Himathongkham et al. [2000] indicated that increased survival of fecal coliforms, *E. coli,* and *Salmonella* were directly related to decreasing poultry litter temperature. Additionally, Jones [1986] suggests that *Salmonella* can survive from 3 days to 36 months in the litter and 5 to 598 days in soil depending on the individual environmental conditions.

A detailed Polish examination of poultry litter observed quantifiable populations of *E. coli*, *Klebsiella sp.*, *Shigella sp.*, *Salmonella OC*, *Pseudomonas sp.*, *Pasteurella sp.,* and *Staphyllococcus* [Latala et al., 1999]. A study of 86 poultry litter samples, collected from production facilities throughout Georgia, indicated quantifiable but insignificant counts of pathogenic bacteria in 47 of the samples, with *Staphyllococcus* being the dominant species [Martin et al., 1998]. Terzich et al. [2000] conducted a detailed examination of poultry litter from 12 of the top poultry producing states in the US, collecting samples from five locations within each barn from at least 10 poultry farms in each state. The results indicated that, on average, *Staphyllococcus* was the predominant bacteria found amongst the sampled locations and that prevalence of pathogens increased with increasing litter pH.

### 6.3.6. Pharmaceuticals

The impact of poultry waste storage on the environmental loading of antibiotics into the surrounding soil and groundwater is not established in the scientific literature. Similar to the study of groundwater contamination from poultry litter, the majority of studies, to date, have focused on the effects of litter application to agricultural fields as a fertilizer source (e.g., Nichols et al. [1997, 1998]; Hemmings and Hartel [2006].

An estimated 75% of antimicrobial agents administered are potentially subsequently excreted by poultry [Addison, 1984]. Kumar et al. [2005] provide a detailed summary of antibiotic usage in agricultural production of animals, which is summarized in Table 6.17. Campagnolo et al. [2002] observed detectable levels of poultry administrable antibiotics in 67% of ground and surface water samples collected near undisclosed US poultry facilities. However, the authors suggest that field application of poultry litter was responsible for their occurrence and not the poultry barns or litter stockpiles.

***Table 6.17. Concentration of antibiotics in poultry litter samples from Virginia, United States (after Kumar et al. [2005]).***

| Antibiotic | Level | |
|---|---|---|
| | Range | Average |
| Oxytetracycline (mg $kg^{-1}$) | 5.5-29.1 | 10.9 |
| Chlortetracycline (mg $kg^{-1}$) | 0.8-26.3 | 12.5 |
| Penicillin (units $g^{-1}$) | 0.0-25.0 | 12.5 |
| Zn bacitracin (mg $kg^{-1}$) | 0.8-36.0 | 7.2 |
| Amprolium (mg $kg^{-1}$) | 0.0-77.0 | 27.3 |
| Nicarbazine (mg $kg^{-1}$) | 35.1-152.1 | 81.2 |

### 6.3.7. Summary

The average concentrations of both poultry litter and manure are well characterized in the scientific literature, including $NH_4$-N, $NO_3$-N, total-P as well as a large number of trace elements.

Similar to EMS wastewater and cattle feedlot manure, poultry manure concentrations vary widely between and within sites. In addition, the concentrations of individual constituents are dependent on a number of management (feedstock, litter type, etc.) and livestock physiological conditions.

Limited data are available on the concentrations of wastewater associated with runoff from poultry litter and manure.

Poultry litter and manure are characterized by higher total-P and total-N concentrations than swine and cattle manure. In addition, studies suggest that poultry litter is also characterized by a higher proportion of organic-N.

The studies cited in this review focus on delineation of vertical contamination plumes originating from poultry litter storage, through both soil core and groundwater sample analyses. The horizontal extent of these plumes is not well characterized to date.

The single study cited in this review suggests that although litter stockpiles are demonstrated to increase groundwater $NH_4$-N and $NO_3$-N concentrations, the relative concentration was well below the MCL of 10 mg/L ($NO_3$-N < 4.0 mg/L).

Investigations focused on contaminant loading of underlying soils suggest that nitrogen and phosphorus are mobilized from poultry manure and litter stockpiles, which implies the potential for contamination of groundwater.

The review of the literature suggests the presence of pathogens in poultry litter is lower in occurrence relative to that observed in swine and cattle wastes. Although quantifiable populations of several bacteria and fecal coliform have been observed in poultry litter and manure, most studies suggest they occur in numbers below levels of concern for human health. To date, we found no studies that address pathogen contamination of groundwater associated with poultry production.

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# 7. HYDROGEOLOGIC CONTROLS ON THE MIGRATION OF CONTAMINANTS

## 7.1. Introduction

The literature review highlights that some chemical constituents in effluents migrate in the groundwater in a relatively unattenuated manner (e.g., Cl is transported at the same rate as the groundwater velocity), while other contaminants (e.g., $NH_4$-N or K) may be attenuated, moving at velocities much lower than those of the groundwater.

In the case of sandy (permeable) media, the retardation of $NH_4$-N with respect to Cl appears limited. The lack of available data on the migration of $NH_4$-N and Cl in the literature precludes an estimation of the retardation factor and migration rates associated with $NH_4$-N. However, given the contaminant seepage rates cited in a number of studies (between 0.2 and 3. m/year), a contaminant plume from a CFO facility such as an EMS pond could migrate between 20 and 250 m over a 100 year time period.

The retardation of $NH_4$-N with respect to Cl is greater in glacial tills and clays than in sandy media. Available data suggest the retardation can be between a factor of three and seven with rates of migration of $NH_4$-N in the subsurface of between 2 and 7 cm/year. These data suggest that over 100 years of use, the $NH_4$ plume could migrate between 2 and 10 m.

Understanding the processes controlling the transport rate (mass flux and velocity) of the various chemical constituents of effluent from CFO operations is important as these will influence the citing of CFOs, the design of containment systems, or the development of monitoring strategies (location, frequency).

This section illustrates the nature of plume development adjacent to CFO operations using a few examples which encompass a range of hydrogeologic settings encountered in the Prairie provinces. The modeling presented in this chapter demonstrates the potential use of these types of simulations; however, the modeling would need to be refined and possibly expanded to deal with site specific cases. The parameters used in the model (recharge rates, K values, etc.) are based on values reported in the literature as discussed in more detail by Geonet [2000].

The Geonet [2000] study was undertaken for a Tri-provincial committee (Manitoba, Saskatchewan, and Alberta Agriculture) and used transport modeling to help identify characteristics of 'geologically secure' sites for the siting of Earthen Manure Storage (EMS) facilities. 'Geologically secure' sites were considered to be preferred sites for EMS facilities, which would not require detailed site investigation, engineered control structures, or intensive monitoring.

## 7.2. Rationale for Model Development

The evaluation of the impact of a contaminant release from a point source such as a CFO involves the characterization of three basic facets; source, pathway and receptor. The description of the source of contamination (e.g., N speciation) in the case of CFOs is discussed in Chapters 5 and 6 of this report. The particular chemical constituent of concern may be reactive (e.g., $NH_4$) or non-reactive (e.g., Cl, DOC) and consequently may be attenuated or unattenuated relative to the groundwater flow.

The potential release pathways from a CFO are a series of overlapping transport pathways that include the following:

1. Surface runoff and overland flow. This provides a potential release pathway to surface water bodies or local depressions that act as recharge areas for unconfined or confined groundwater flow systems.
2. Infiltration from CFOs or from runoff derived waters from the CFO into a surficial unconfined groundwater flow system. The receptor in this case may be nearby water wells or surface water bodies (e.g., streams or lake) into which this unconfined flow system discharges. The unconfined flow system may also recharge deeper groundwater flow systems.
3. Vertical deep percolation from the surface, or from the unconfined aquifer through a confining layer (i.e., aquitard) to a confined aquifer. The final receptor may be a water supply well within the confined aquifer, a surface water body into which the confined aquifer discharges, or the aquifer itself.

We assumed for the purposes of this discussion that uncontrolled runoff from a CFO does not occur and that the source for potential contamination is from direct infiltration from the CFO. This effluent is assumed to be unrestricted (no liners or engineered barriers) and that any vadose zone present below the CFO is relatively shallow and does not significantly increase the transport time. Steady state groundwater flow systems are postulated to develop based on average annual recharge rates. Recharge is assumed to be uniformly distributed across the surface of the CFO and the adjacent land surface.

The effluent from the CFO is then assumed to be transported along two potential pathways: along a horizontally flowing unconfined aquifer comprised of either coarse textured soil (e.g., sand) or weathered, highly oxidized, high hydraulic conductivity (K) glacial sediments; or vertically through an oxidized glacial till to an underlying aquifer. This conceptualization of potential contaminant migration pathways from CFOs is similar to that developed in the report by Geonet [2000] (Figure 7.1).

The unconfined flow system may be geologically similar to the clay aquitard; however it will likely have a higher K as a result of freeze/thaw and desiccation processes. A contrast of more than one or two orders of magnitude between the K of these layers will tend to produce a rectilinear flow system in which effluent from the CFO moves laterally within the unconfined aquifer or vertically downward through the clay aquitard towards the underlying aquifer.

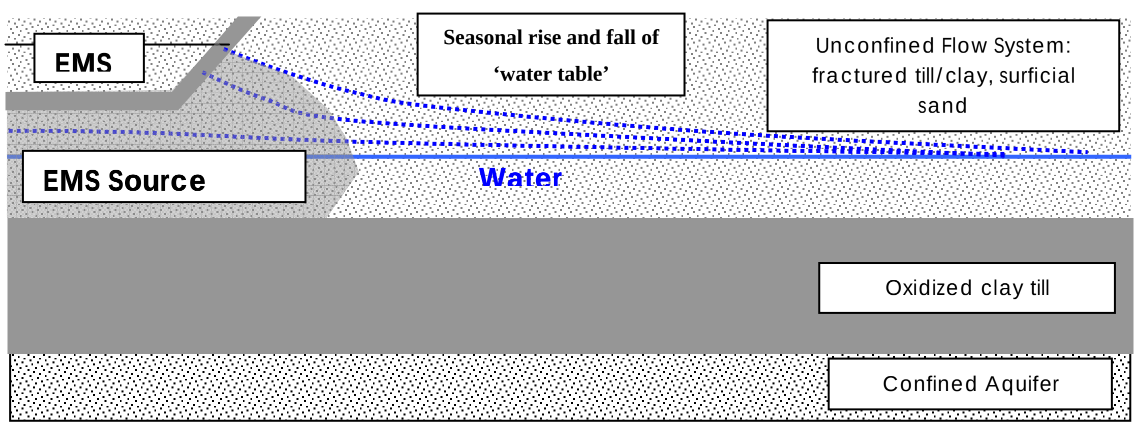



**Figure 7.1. Mechanisms of Solute Transport from an Earthen Manure Storage Facility [GEONET, 2000]**

Infiltration and seepage from the CFO through any vadose zone (zone above the water table) and into the unconfined flow system will define the extent (geometry) and concentration of the ‘source’ for subsequent transport through the confining layer to the confined aquifer. The ‘source’ can be characterized as the ‘shadow’ of the CFO that develops as a result of infiltration and lateral flow with this surficial layer. The geometry and concentration of solutes within this ‘shadow’ will vary depending on the nature of the unconfined material (thickness, soil-water characteristic curve, unsaturated K, depth to the water table, operation of the EMS, seasonal water table fluctuations and recharge rates, etc.).

The hydrogeologic systems associated with each of these transport pathways are unique. Unconfined flow systems are often relatively thin zones of lateral flow that have their upper surface potentially exposed to near atmospheric conditions into which fresh, oxygenated water continues to recharge the upper portion of the unconfined aquifer. Flow through the aquitard into an underlying aquifer is generally vertical through the confining layer followed by lateral flow within the aquifer. These deep flow systems are typically oxygen depleted and the width of the plume within the confining layer is relatively large, reflecting the areal extent of the footprint of the CFO. Consequently, no opportunity exists for oxic conditions to develop and no dilution from recharging waters occurs until the onset of lateral flow in the aquifer itself.

The following sections illustrate features of the transport of a conservative (Cl and DOC) and a reactive ($NH_4$) solute from a CFO footprint for a range of typical flow systems using numerical flow modeling. The results of this model exercise are not meant to represent all, or any specific hydrogeologic setting, but rather to highlight several issues associated with operation monitoring and closure.

## 7.3. Background Study – Solute Transport from Earthen Manure Storages [GEONET, 2000]

The modeling conducted by GEONET [2000] was performed to assist regulators in ‘visualizing’ the constraints required to produce a ‘geologically secure’ hydrogeologic setting for an EMS. The single pathway of concern was vertical transport through a clay aquitard to a lower confined aquifer and the contaminant of concern was considered to be conservative (no geochemical or biochemical reactions influence the concentrations of the contaminants). One dimensional numerical modeling was used to simulate a range of possible transport rates and loadings to the underlying aquifer for a range of typical hydrogeologic conditions including both fracture dominated and equivalent porous media (EPM) transport.

The flow chart shown in Figure 7.2 was used to highlight a sequence of key issues and material parameters to be considered when evaluating whether a potential site for a CFO was geologically secure. The first decision point in this chart is to establish that the hydrogeologic setting is sufficiently understood to ensure that any transport through the aquitard will satisfy the conditions for Equivalent Porous Media (EPM) transport, even if the aquitard is fractured. Siting of a CFO on a hydrogeologic system that demonstrated fracture dominated transport behaviour was assumed to not be prudent.

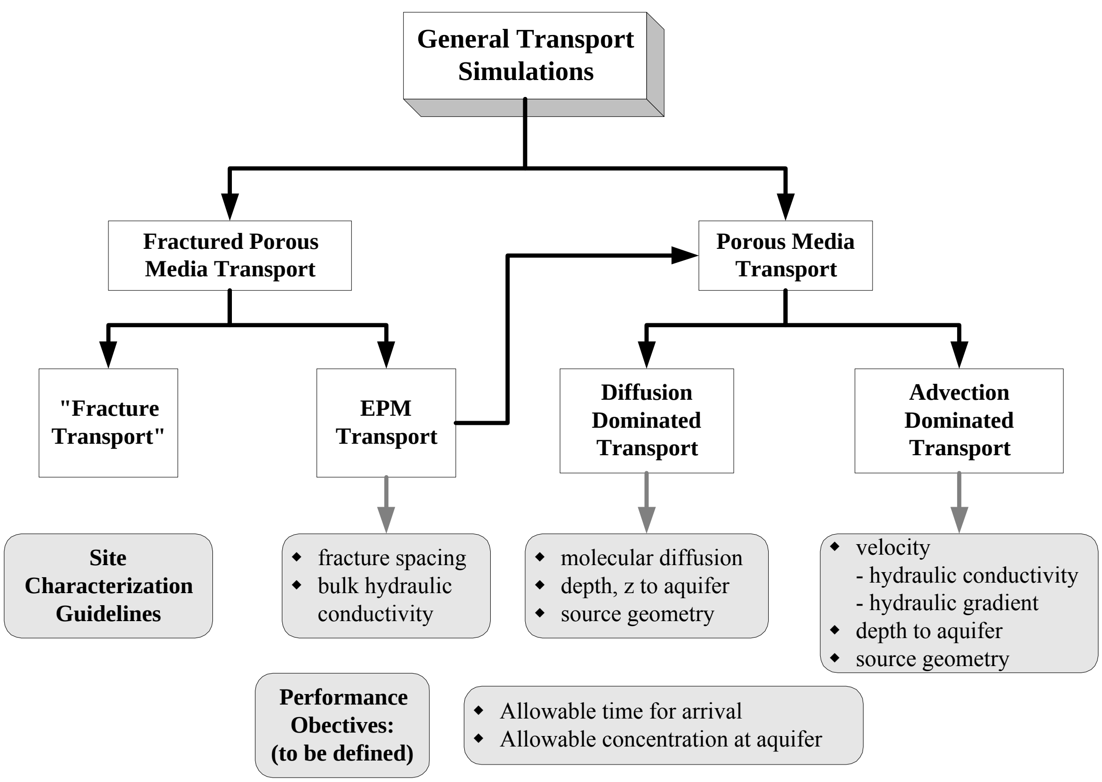



**Figure 7.2. Design Flow Chart [GEONET, 2000]**

The criteria used to define EPM conditions were those proposed by van der Kamp [1992]. This requires that data are available (measured or assumed) on fracture spacing, along with measured K, porosity and coefficient of diffusion values so that a potential range of EPM conditions can be established for various groundwater flow rates and transport distances.

An example of this evaluation is illustrated in Figure 7.3. In this graph, assumed or measured fracture spacing is plotted against hydraulic conductivity with fracture aperture estimated from the equations proposed by Snow [1968]. The estimated advection transport rate through an EPM is estimated from an assumed vertical hydraulic gradient of 1 and a porosity of 0.3. The rate of diffusion into the matrix between the fractures is based on an assumed coefficient of diffusion of 0.01 $m^2/y$.

The EPM calculation is simple. The time ($t_a$) for advective transport over a selected distance, L, is estimated from the equation:

$$t_a = L/v \qquad [7.1]$$

where v is the linear velocity (Darcy flux divided by porosity). The time to equalize the concentration within the matrix by diffusion ($t_d$) is estimated from Equation 2:

$$t_d = (S/2)^2/D \qquad [7.2]$$

where S is the fracture spacing and D is the coefficient of diffusion.

EPM conditions occur when $t_a > t_d$, that is, when the time to equalize the concentration in the matrix from diffusion from the fracture is less than the time to advectively transport the contaminant over the length, L, assuming all of the pore space is available for advective transport.

Similar charts can be constructed for any specific combination of parameters. The chart illustrates that EPM conditions are likely to exist, for example, for a transport distance of 10 m if the fracture spacing is less than 1 m and the bulk K is less than $1x10^{-9}$ m/s.

The second decision illustrated in Figure 7.2 was an evaluation of the relative importance of advection to diffusion in controlling ‘first arrival’ and loading to the lower aquifer.

Without a detailed hydrogeologic study, preliminary evaluations of time of arrival and contaminant loadings are often made solely based on advection and using parameters such as K, hydraulic gradient, porosity, and aquitard thickness. However, in many cases of low K and relatively thin aquitards, the arrival times and contaminant loading may be by diffusion. The report illustrated how simple calculations and charts can be used to estimate the times of first arrival for a specified concentration by considering these transport mechanisms.

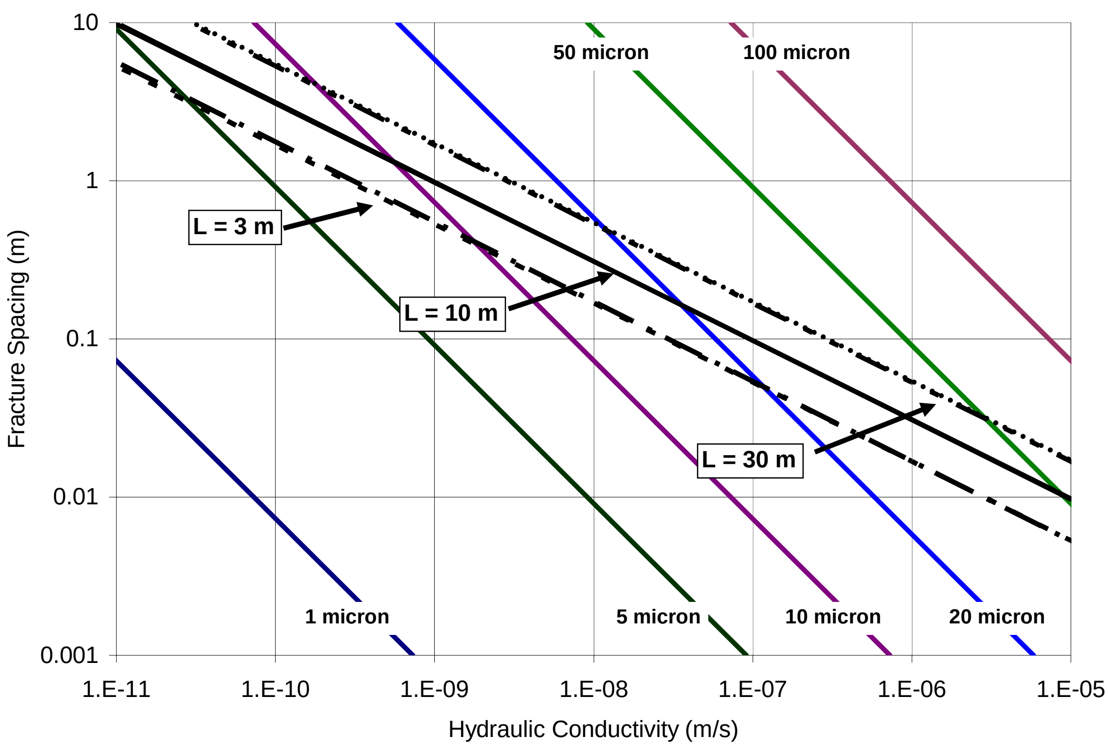



**Figure 7.3. EPM conditions related to bulk hydraulic conductivity, fracture spacing and fracture aperture. (Case: n=0.3, i = 1.0, D*= 0.01 m2/y) [GEONET, 2000]. Inclined lines marked as 3 m, 10 m, and 30 m separate EPM (lower left) and non-EPM (upper right) conditions for these assumed transport distances.**

## 7.4. Modeling Methodology

The illustrative transport models in this section were developed using the SEEP/W and CTAN/W software marketed by Geo-Slope International (www.geoslope.com). These are two-dimensional saturated/unsaturated groundwater flow and contaminant transport models. Capturing all the hydrogeologic settings of interest in just a few illustrative models is impossible. The hydrogeologic settings chosen in this modeling exercise attempted to capture a range of ‘typical’ conditions associated with CFO operation in glaciated terrain of Alberta.

The release of CFO effluent into two unconfined flow systems was simulated. The systems had a surface layer of relatively high K soil comprised of either coarse grained soil (e.g., sand) or fine grained soil (e.g., fractured oxidized glacial till). The systems were assumed to be underlain by an unoxidized fine-grained glacial till. A third case of vertical transport through a lower K oxidized till into a lower confined aquifer was also simulated. The three illustrative models are summarized in Table 7.1 below. All the settings were assumed to exhibit EPM transport. The range of values used in this illustrative modeling was similar to those in the Geonet [2000] modeling, where the justification of these typical values relative to the available literature is provided in detail.

The annual recharge rates of 50 mm/y for Case 1 (sand) and 10 mm/y for Case 2 (surficial oxidized till) are in the order of 10 to 2% of annual precipitation. This is a reasonable estimate of annual recharge in Alberta, although this range could easily be extended to 20 to 30% in some areas depending on surface soil texture. The ratio of 10:1 in recharge rates for Case 1 and 2 were simply chosen to ensure that the flow systems had a similar geometry given an assumed difference in hydraulic conductivity of one order of magnitude. The lateral underflow below the CFO within the unconfined aquifer (Case 1b and 2b) and lateral flow through the underlying aquifer in Case 3 was based on forcing an equivalent volume of recharge through the aquifer as would recharge through an equivalent surface area to that of the CFO, upstream of the CFO.

The recharge rates of 50 mm/year and 10 mm/year over the footprint of the CFO (50 m wide) produce a total annual water flux of 2.5 $m^3$/y and 0.5 $m^3$/y, respectively. Distributed over an assumed saturated thickness of unconfined aquifer of 5 m, this produces a Darcy flux of 0.5 m/y to 0.1 m/y. The sand is assumed to have a porosity of 0.3 and the oxidized till a porosity of 0.4, and consequently the linear groundwater velocity would be approximately 1.66 m/y in the sand to 0.25 m/y in the oxidized till. These velocities will vary with location as additional recharging waters are carried through the aquifer. In Case 3, 0.5 $m^3$/y is carried through an assumed confined aquifer that is 2 m thick. This results in a Darcy flux of .25 m/y and a linear velocity of approximately 0.75 m/y in the aquifer.

***Table 7.1. Simulation Cases***

| **Case** | **1** | **2** | **3** |
|---|---|---|---|
| **Soil type** | **Sand** | **Oxidized Till** | **Oxidized Till** |
| Hydrogeologic Setting | **Case 1a)** 5 m thick unconfined aquifer. No underflow. All flow in aquifer originates from CFO or downstream ground surface | **Case 2a)** 5 m thick unconfined aquifer. No underflow. All flow in aquifer originates from CFO or downstream ground surface | **Case 3a)** Recharge from CFO to horizontally flowing confined aquifer through thin (2 m) oxidized till |
| | **Case 1b)** As in 1a) plus flow under CFO from recharge upgradient of CFO | **Case 2b)** As in 1a) plus flow under CFO from recharge upgradient of CFO | **Case 3b)** Recharge from CFO to horizontally flowing confined aquifer through thick (7 m) oxidized till |
| Linear Velocity and direction (Hydraulic Conductivity, m/s, porosity) | **Case 1a)** ~ 1.6 m/y horizontally (1e-05, 0.3) (varies with position due to surface recharge) | **Case 2a)** ~ .25 m/y – horizontally (1e-06, 0.4) (varies with position due to surface recharge) | ~ .75 m/y in aquifer at upstream edge of CFO |
| | **Case 1b)** ~ 3.2 m/y horizontally at downstream edge of CFO (varies with position due to surface recharge) (1e-05, 0.3) | **Case 2b)** ~ .5 m/y horizontally at downstream edge of CFO (varies with position due to surface recharge (1e-06, 0.4) | |
| Transport Parameters<br>> Dispersivity ($\alpha_L$, $\alpha_T$)<br>> Diffusion Coefficient<br>> Retardation | <br>> 1 m, 0.1 m<br>> 0.015 $m^2$/y<br>> 1 | <br>> 1 m, 0.1 m<br>> 0.015 $m^2$/y<br>> 2 | <br>> 1 m, 0.1 m<br>> 0.015 $m^2$/y<br>> 2 |
| Boundary Conditions : | | | |
| Surface recharge (q) | 50 mm/y | 10 mm/y | 10 mm/y |
| CFO Transport boundary | C=1 | C=1 | C=1 |

The use of a relative (C/Co) concentration boundary condition (C=1) to represent the CFO is somewhat conservative in that it represents the CFO as an ‘infinite source’ of contamination. This enhances the diffuse mass flux from the CFO to the underlying soil at early times and under low groundwater recharge rates. A more physically realistic boundary may be a mass flux boundary in which recharging waters (q) carry a concentration of contaminant (C) and produce a mass flux boundary equivalent to q*C.

It is important to note that the lower boundary of the unconfined aquifer is assumed to be a zero water and mass flux boundary. The zero water flux assumption is not unreasonable if a lower confining layer of very low K is present, although some 'leakage' through this lower boundary would be likely. It is not, however, a good boundary condition in terms of mass flux since even in the absence of water 'leakage' through the confining layer, diffusion into this confining layer is likely. This lower confining layer would then act as a potential sink for contaminants diffusing out of the unconfined aquifer. This would result in some attenuation of the plume during CFO operation, but would also store contaminants which could be released post-closure. Further modeling could be developed to explore this effect in more detail in the future.

## 7.5. Presentation of Simulation Results and Discussion

### 7.5.1. Lateral Flow in Unconfined Sand Aquifer

The first case is illustrated in Figure 7.4a). The sand aquifer is assumed to be thin (~ 5m saturated thickness) with a shallow water table. The water table location is similar for both Case 1a and 1b; however the hydraulic head distributions (Figure 7.4 b and c) are slightly different due to the additional water flux added to the left side of the unconfined aquifer in Case 1b.

Figure 7.5 shows the plume developed after the first 50 years of operation for both Case 1a and 1b. The front edge of the plume (C/Co of 0.1) has arrived 150 m downstream but the plume is strongly diluted by recharging freshwater downgradient of the CFO (Case 1b). The presence of underflow beneath the CFO of freshwater has produced two somewhat contrary effects. The C/Co = 0.1 front has been transported farther downstream but the plume is more diluted in the vicinity of the CFO due to the fresh water flowing beneath the CFO from upstream. The dilution provided by these freshwater influxes will vary from site to site depending on the specific conditions present at each site.

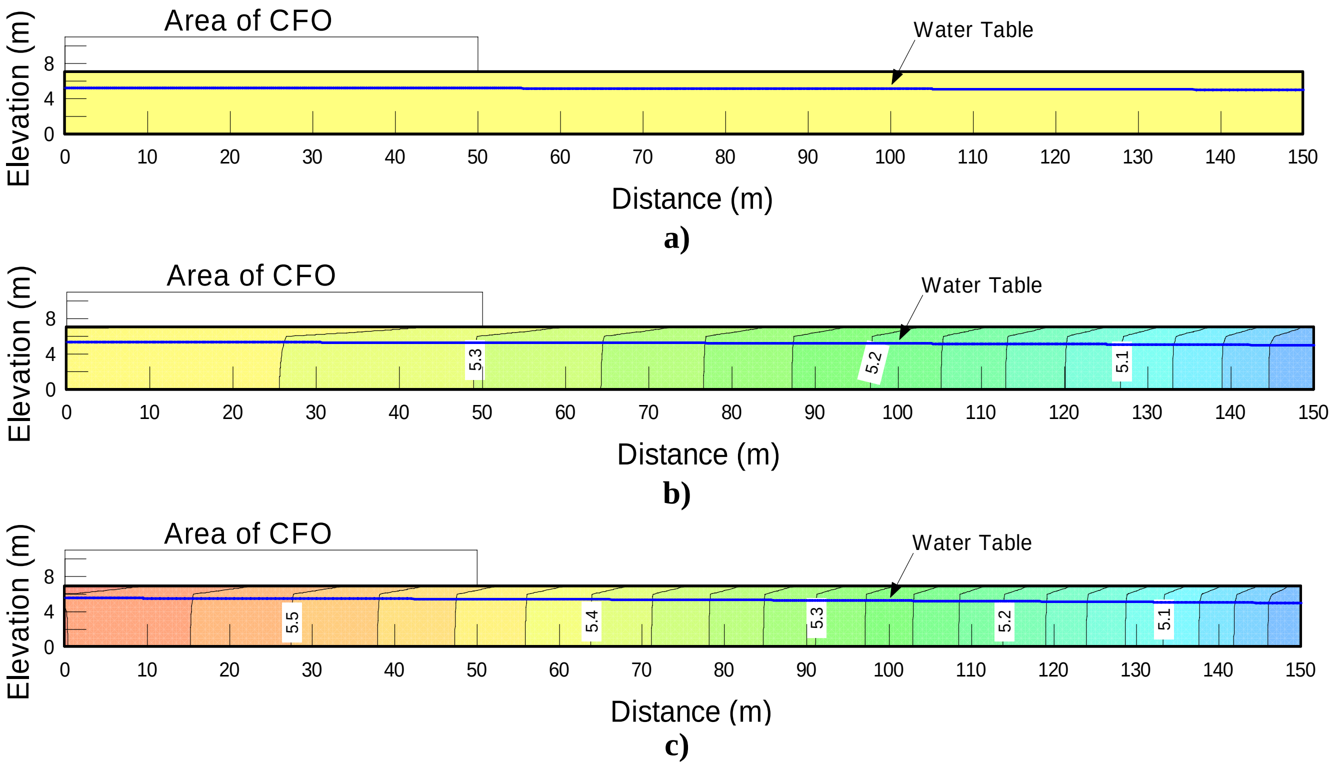



**Figure 7.4. a) Model geometry for flow and transport in unconfined sand aquifer;**
**b) Hydraulic head distribution for Case 1a;**
**c) Hydraulic head distribution for Case 1b.**

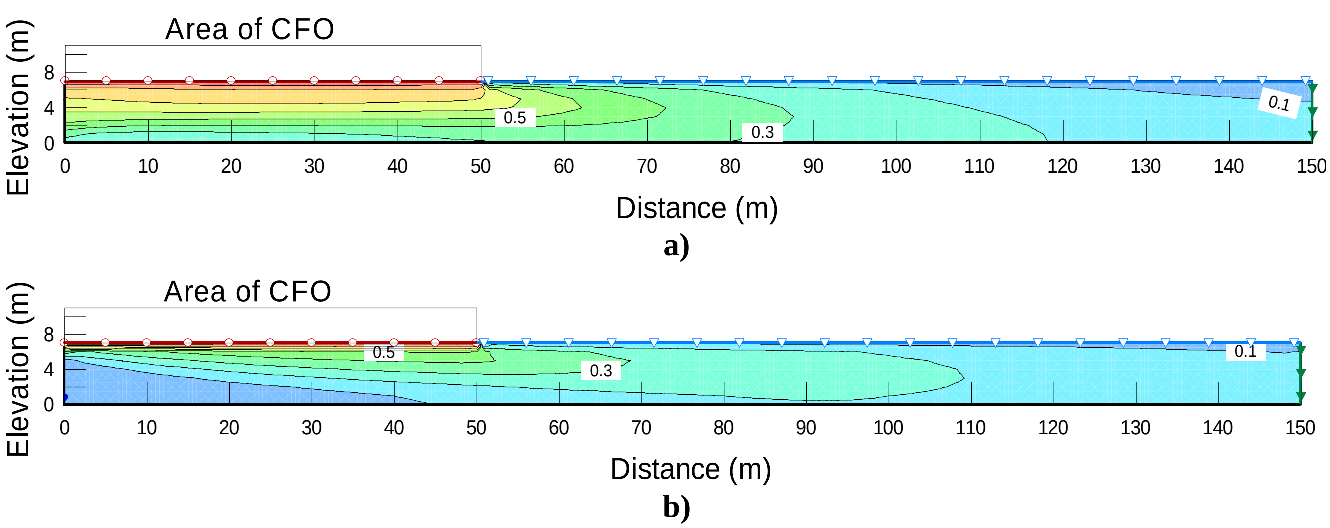



**Figure 7.5. Concentration Contours (C/Co increments of 0.1) in sand aquifer after 50 years of operation; a) Case 1a; b) Case 1b.**

It is evident that some stratification of the plume within the aquifer does occur, even in the presence of homogeneous soil. This stratification is limited in both cases by the high dispersive mixing assumed in the model which enhances mixing across the entire aquifer. This may not be the case if lower values of dispersivity or more heterogeneity (e.g. sand or gravel lenses) were used in the model. A longitudinal dispersivity value of 1.0 m was used in the model along with a transverse dispersivity of 0.1 m. These were the minimum values thought to be required to control numerical dispersion. The contaminant plume will mix over a greater depth of the aquifer with increasing time and distance. If the TDS of the effluent source is high, the plume may even sink under the influence of fluid density differences.

Monitoring at the base of the aquifer, immediately below the EMS, in Case 1b would suggest little transport away from the CFO. In fact, the mass flux from the CFO in Case 1b is slightly higher due to higher diffusive gradients near the source.

This simulation highlights how the concentrations within the sand aquifer are strongly controlled by mixing between upgradient waters, recharging waters, and the concentration of the CFO effluent. The distribution of contaminants can also be complicated by stratification of the plume.

### 7.5.2. Lateral Flow in Unconfined Oxidized Till

The simulation of flow and transport in a surficial, fractured, oxidized till was based on developing a linear velocity of approximately 0.25 m/y at the downstream edge of the CFO in Case 2a. The hydraulic conductivity assumed for the oxidized till is somewhat unrealistic (high) but was used so that the model could focus primarily on the role of adsorption and a lower linear velocity on plume evolution with a similar flow system geometry as in Case 1. Figure 7.6 shows the geometry of the model and the hydraulic head distribution for Case 2a and 2b. Figure 7.7 shows the extent of the plume after 100 years of CFO operation for both Case 2a and 2b, and with and without species retardation.

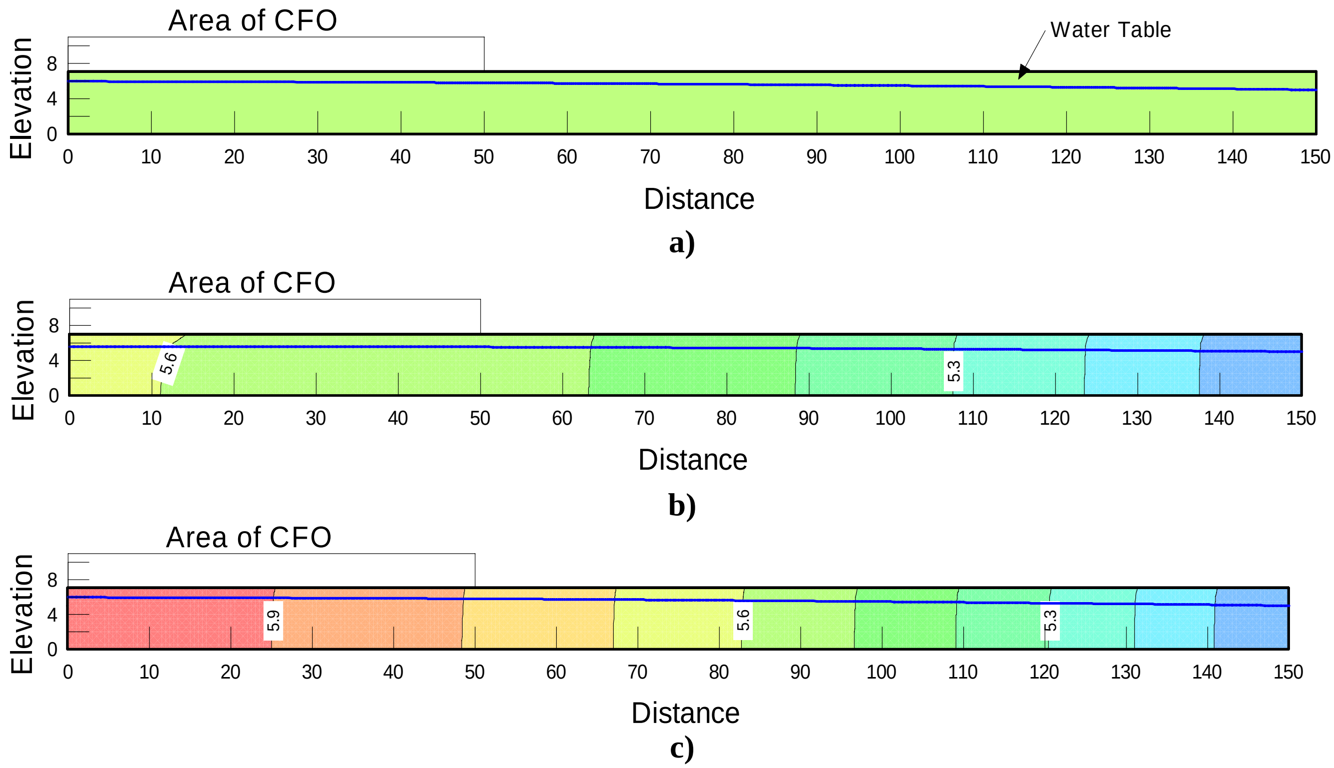
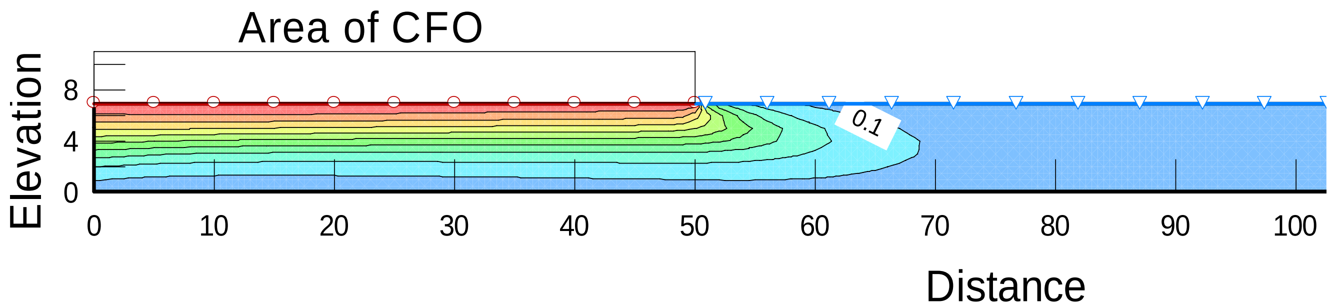
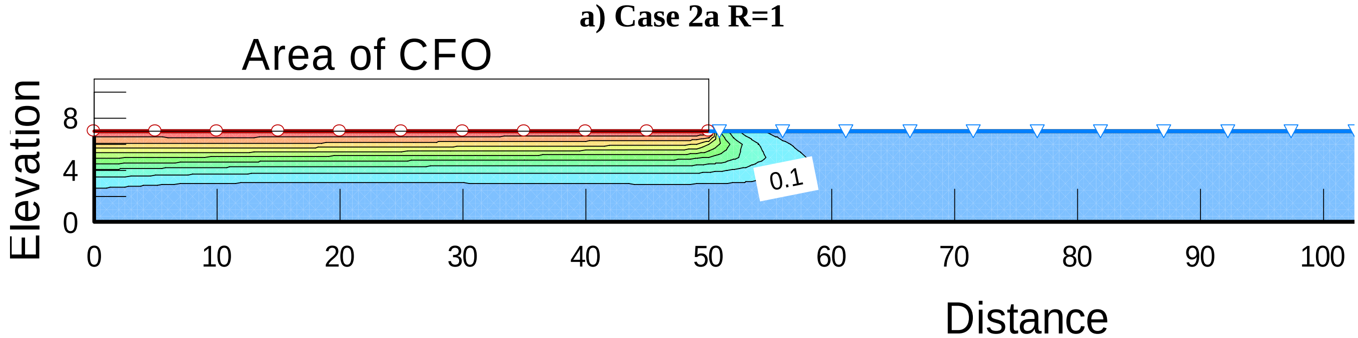


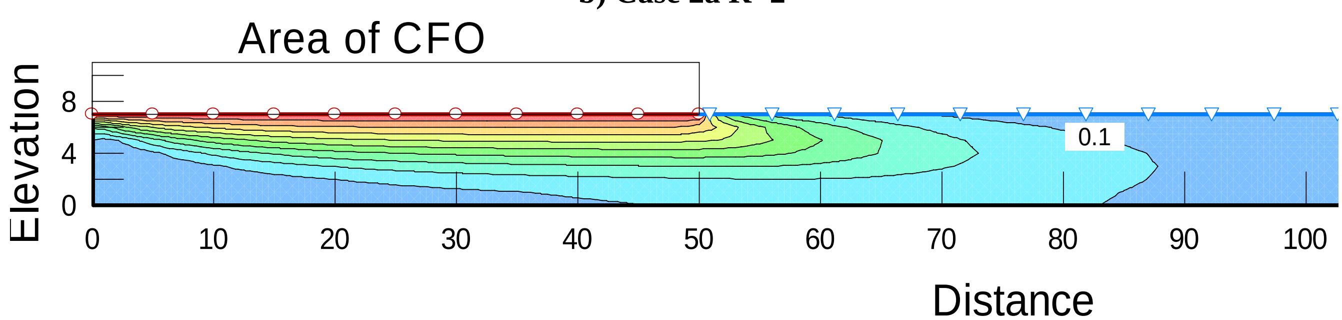

**Figure 7.6. a) Model geometry for flow and transport in unconfined oxidized till; b) Hydraulic head distribution for Case 2a; c) Hydraulic head distribution for Case 2b.**



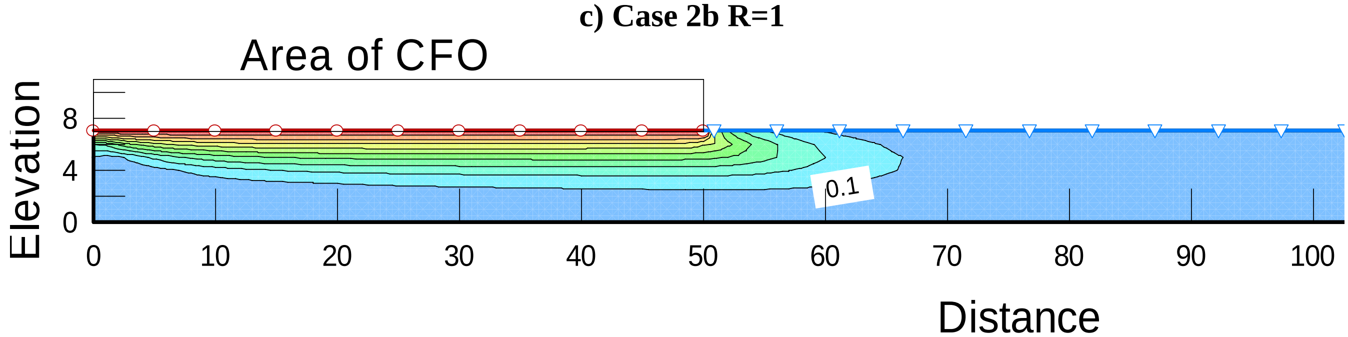

**a) Case 2a R=1**




**b) Case 2a R=2**




**c) Case 2b R=1**




**d) Case 2b R=2**

**Figure 7.7. Concentration contours in oxidized till after 100 years of operation: a) Case 1a (R=1, no attenuation); b) Case 1a (R=2); c) Case 2b (R =1, no attenuation); d) Case 2b (R=2).**

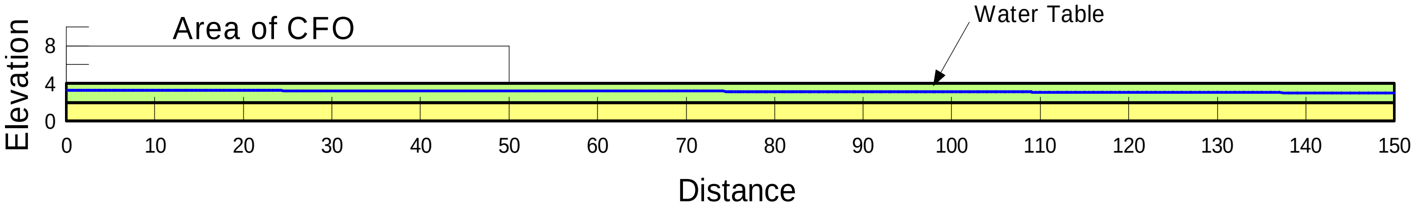
The results shown in Figure 7.7 illustrate that the front edge of the plume (C/Co=0.1) has only advanced 20 to 35 m downstream in approximately 100 years for the unattenuated cases. The shape of the plume and the effect of upstream underflow are similar in the unattenuated plumes (Figure 7.7 a and c) as for the unconfined sand aquifer, albeit at much smaller rates due to the lower assumed recharge rates and commensurate lower groundwater velocities. The dilution of the plume near surface due to fresh recharge downstream of the CFO and at the base of the CFO due to fresh groundwater flowing beneath the CFO are similar to Case 1.

Figures 7.7 b) and d) also highlight the significant impact on plume development due to even a rather nominal retardation factor of 2. In these cases, the plume advance is restricted to only 8 to 15 m downstream. In the case of underflow, the lower part of the unconfined flow system actually remains ‘clean’. Obviously, monitoring in this case at the bottom of the unconfined aquifer in the footprint of the plume would not necessarily be the best diagnostic to determine leakage losses.

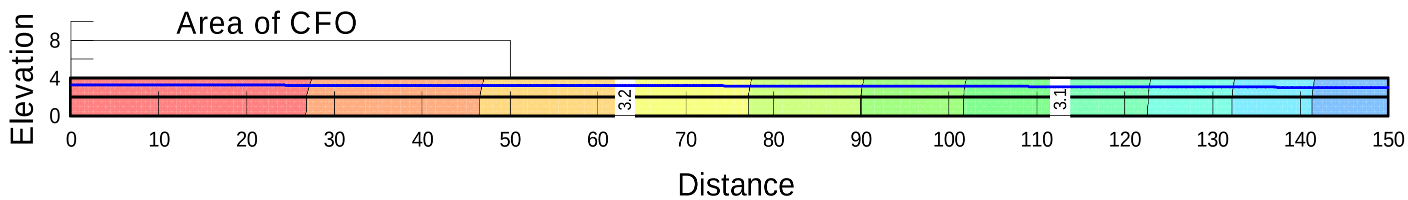
## 7.6. Vertical Flow in Oxidized Till to an underlying Aquifer

Two cases of vertical recharge across an oxidized till were simulated: the first where the aquitard is thin (2 m), and the second where the aquitard is thick (7 m). In both cases, a water table at a depth of 1 to 2 m below surface develops within the oxidized till. The geometry and hydraulic head distribution through these flow systems are shown below in Figure 7.8. In both cases, flow through the underlying aquifer was applied to the left hand side of the model.

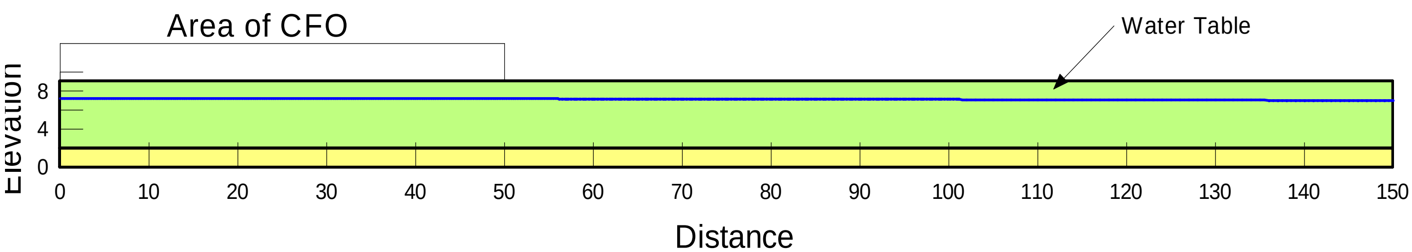
The plume for Case 3a is shown in Figure 7.9 at various elapsed times of 50, 100 and 200 years. In the first 50 years, the plume behaves very similarly to that for the full depth oxidized till shown in Case 2. The plume advances vertically, very slowly, with little lateral spreading as all recharging water moves directly downwards into the lower aquifer.



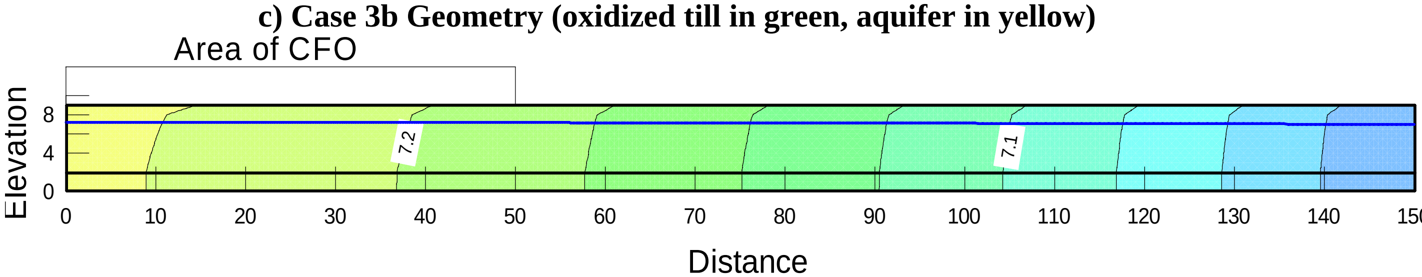

**a) Case 3a Geometry (oxidized till in green, aquifer in yellow)**




**b) Case 3a Hydraulic head contours (0.01 m increments)**

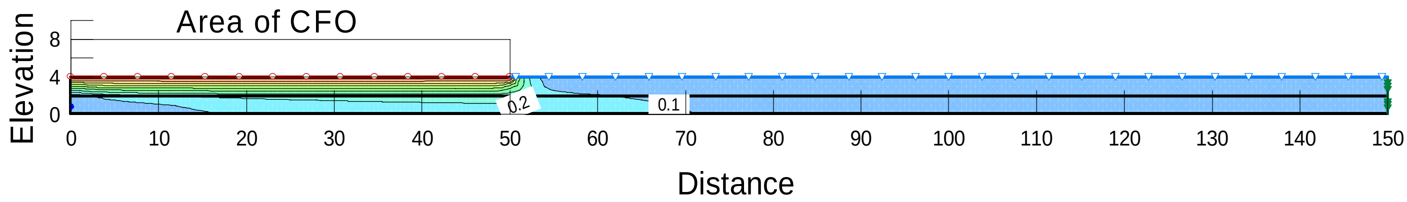



**c) Case 3b Geometry (oxidized till in green, aquifer in yellow)**

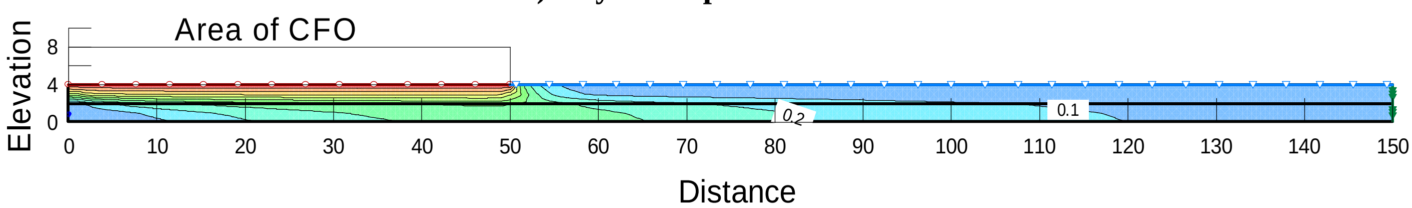



**d) Case 3b Hydraulic head contours (.01 m increments)**

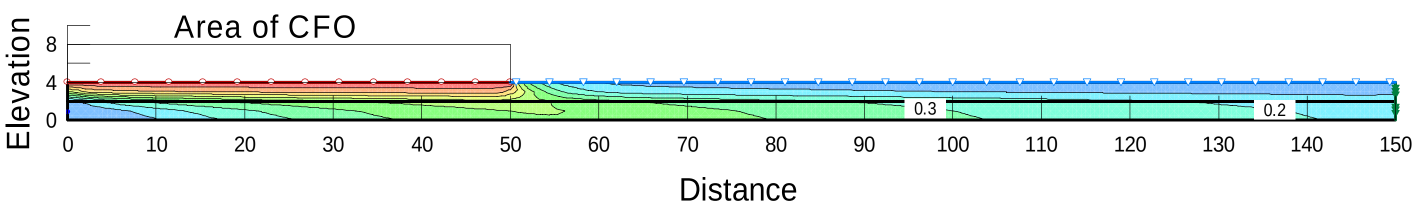
**Figure 7.8. Model geometry for vertical flow and transport through an oxidized till to an underlying confined aquifer: a) Geometry of thin till case (Case 3a); b) Hydraulic head distribution for thin till case (Case 3a); c) Geometry of thicker till case (Case 3b); d) Hydraulic head distribution for thick till case (Case 3b).**




**a) 50 year elapsed time**




**b) 100 year elapsed time**




**c) 200 years elapsed time**

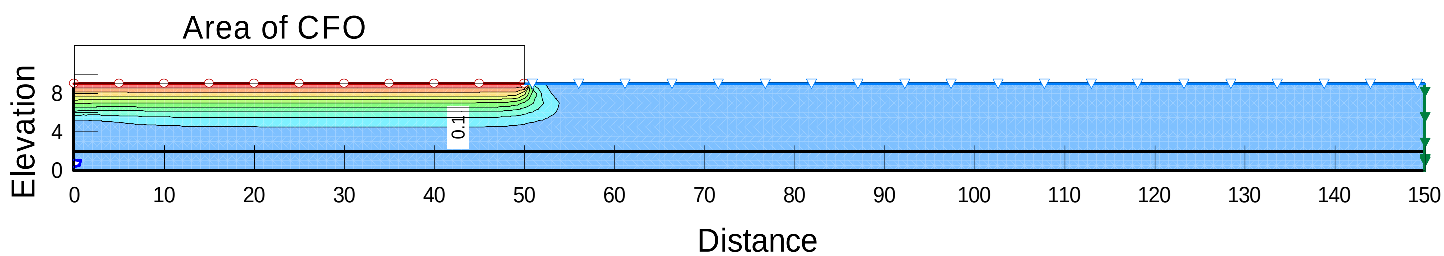
**Figure 7.9. Plume development below the thin till (Case 3a) at elapsed times of: a) 50 y; b) 100 y; and c) 200 y (Contour increment is C/Co=0.1).**

The plume geometry changes dramatically at elapsed times greater than 50 years. The slowly vertically advancing plume breaks through into the lower aquifer and is rapidly transported downstream within the aquifer moving nearly 70 m downstream (C/Co=0.1) by an elapsed time of 100 years. It is important to note that attenuation (e.g., R>1) is effective in slowing the advance of the contaminant front through the till; however once breakthrough into the lower aquifer occurs and concentrations within the till become constant, it has no further effect on mass flux. The mass released to the lower aquifer will be at the same rate regardless of the retardation value used in the simulation.

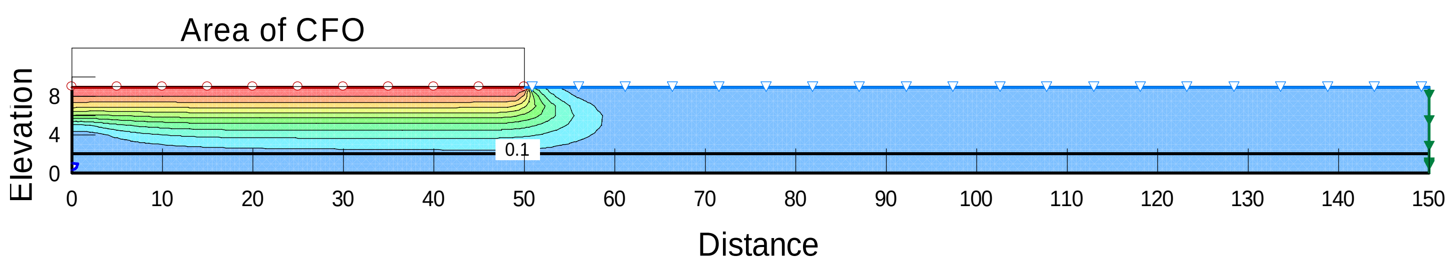
The specific time to breakthrough into the lower aquifer will vary depending on site specific conditions. The speed of transport and dilution and dispersion of the plume within the lower aquifer will also vary with site specific conditions. In fact, the actual pathway within the aquifer downstream of the CFO might be quite tortuous and narrow depending on the level of natural heterogeneity in texture within the aquifer and the seasonal variations in aquifer velocities [van der Kamp et al., 1994].

It is also less likely that the EPM conditions will occur over these short travel distances. In reference to Figure 7.3, it is evident that fracture spacings of less than 10 cm would be required with a bulk hydraulic conductivity of $1x10^{-8}$ m/s to ensure EPM conditions.

The plumes that development in the case of the thicker oxidized till sequence over the confined aquifer (Case 3b) are shown for elapsed times of 100 and 200 years in Figure 7.10. In this case the plume was contained within the oxidized till itself through to the elapsed time of 200 years, and consequently was identical to Case 2a described previously.




**a) Case 3b – Plume development after 100 years**




**b) Case 3b – Plume development after 200 years**

**Figure 7.10. Plume development below the thin till (Case 3b) at elapsed times of: a) 100 y; and b) 200 y (Contour increment is C/Co=0.1).**

## 7.7. Implications of Simulations

Simulations show that rapid plume advance may occur in an unconfined aquifer when relatively high groundwater flow velocities arise due to high hydraulic conductivity and recharge conditions. The actual geometry of the plume is controlled by the heterogeneity within the aquifer and the distribution of recharging waters both upstream and downstream of the plume.

In more stagnant unconfined aquifers (e.g., oxidized till case), lower hydraulic conductivity and recharge rates produce a plume that has very low advective velocities and in which diffusion/dispersion control the slow spread of the plume downgradient. In these cases, the contaminated soil is limited in depth and extent relative to the CFO footprint.

Case 3 provides an interesting transition case from the oxidized till sequence (Case 2) to the high lateral flow sequence. The presence of hydrogeologic complexity, such as the presence of a sand aquifer at depth below an oxidized till layer, can rapidly alter the nature of plume evolution. Initially the plume develops slowly, migrating vertically through the till below the CFO footprint, and at some later time the plume breaks through into a relatively rapid moving front within the underlying aquifer.

In this case, slow advance of the plume through the oxidized till may suggest little potential for transport; however, once the plume has advanced through the thin till, very rapid transport at significant concentrations can occur within the thin confined aquifer. The final concentrations within this aquifer will be dependent on the 'footprint' of the CFO, the thickness of the oxidized till, and the flow volumes and velocities within the lower aquifer. In addition, as the till thins, the potential for non-EPM conditions and rapid preferential flow along fractures will increase.

The footprint for the CFO in the simulations was assumed to be 50 m wide. In some cases, actual footprints may be 3 to 4 times greater than simulated. If the footprint was increased in the horizontal aquifer condition, it would increase the mass loading to the aquifer through advection/diffusion from the CFO. The location of the leading edge of the plume would likely not be affected greatly (e.g. advective front), however, the concentrations across the aquifer would more uniform with depth (i.e. less plume layering as observed in the simulations presented). In this case, any 'advantage' from upgradient freshwater dilution and stratification of the plume would reduced. In the case of simulations involving a deep aquifer, the impact on the groundwater may be felt more acutely. Any dilution effect due to mixing of from freshwater further upgradient causing dilution and stratification of plume would be reduced, resulting in greater concentrations of contaminants migrating into the aquifer and rapid advection of contaminants in the aquifer. These comments could be verified with additional simulations.

## 7.8 References


GEONET Consulting Ltd., 2000, 'Solute Transport from Earthen Manure Storages', prepared for a Tri-Province Committee on Site Characterization - Alberta Agriculture, Food and Rural Development, Manitoba Agriculture and Food, Saskatchewan Agriculture and Food, March 28, 2000.

Snow, D.T., 1968, Rock fracture spacings, openings and porosities: Proc. Amer. Soc. Civil Engineers, Vol.94, (SM 1) pp. 73–79.

van der Kamp, G., 1992, Evaluating the effects of fractures on solute transport through fractured clayey aquitards: 1992 Conference of International Assoc. of Hydogeologists.

van der Kamp, G., Luba, L.D., Cherry, J.A., and Maathuis, H., 1994, Field study of a long and very narrow contaminant plume: Ground Water 32(6): 1008-1016.

# 8. Summary of Literature Reviews and Modeling

An extensive literature review was conducted to assess information relating to the impact that seepage from CFOs may have on surface and groundwater resources. This review was augmented with numerical modeling of the fate and transport of contaminants in hypothetical groundwater regimes that were considered illustrative of several hydrogeologic settings encountered in Alberta.

Observations from the literature review and the modeling are synthesized below.

1. Soil and groundwater contamination can occur from CFOs.

2. Hydrogeologic conditions that are sensitive to contamination include sites characterized by coarse grained soils (sands and gravels), shallow unconfined aquifers, or thin natural clay barriers overlying laterally extensive confined aquifers. This last condition could also be seen to include coarse grained soil sites with a constructed engineered clay barrier or liner. Conditions that are more hydrogeologically stable are characterized by thick deposits of fine grained soils with high clay contents, deep and/or confined aquifers, and well designed and engineered waste storage sites.

3. In Alberta, the prevalence of relatively thick, clay till aquitards (fine grained soils) over much of the landscape and the lack of extensive shallow, aquifer systems suggest that 'hydrogeologically stable' sites should be common.

4. Few studies have delineated the lateral extent of contaminate plumes from CFO operations. More studies have delineated the vertical migration of plumes originating from CFOs.

5. The dominant contaminant in soil and soil pore-water regimes associated with CFOs is $NH_4$. Contaminant plumes emanating from CFOs are characterized by elevated concentrations of $NH_4$-N, Ca, Mg, K and Cl, in keeping with the aqueous chemistry of both EMS and feedlots. In some cases, $NO_3$-N contamination has been documented, often associated with permeable (sands and gravels) deposits with great depths to the water table.

6. In contrast to EMS, the accumulated knowledge on cattle feedlot contamination in North America is much less extensive and much of the data are from sites in Alberta.

7. . The transport of Cl derived from the CFOs in the soil and groundwater is generally considered to be conservative, undergoing limited geochemical reactions in the subsurface. An elevated Cl concentration typically delineates the leading edge of the contaminant plume.

8. In contrast to Cl, the migration of $NH_4$ undergoes retardation. Because the retardation is controlled by exchange reactions through the replacement of Ca and Mg on exchange sites, retardation of $NH_4$ within the soils is related to the clay content of the media.

9. In the case of permeable (sands and gravels) media, the retardation of $NH_4$ with respect to Cl appears limited. The lack of available data on the migration of $NH_4$ and Cl in the literature precludes an estimation of the retardation factor and migration rates associated with $NH_4$. However, given the contaminant seepage rates determined from a number of studies and field-based plume studies, contaminant plumes could migrate between 20 and 250 m from the CFO over 100 years. Numerical modeling of generic cases support this observation.

10. The retardation of $NH_4$ with respect to Cl is greater in glacial tills and clays than in sandy media. Data suggest the retardation can be between a factor of 3 and 7 with rates of migration of $NH_4$ in the subsurface of between 2 and 7 cm/year. These data suggest that, over 100 years of use, $NH_4$ plumes could migrate between 2 and 10 m from the CFO. Numerical modeling of generic cases support this observation

11. Numerical modeling of the illustrative case whereby a thin glacial till overlies a laterally extensive confined aquifer suggests that contaminant migration could be limited to a few centimeters per year in the short term, but increasing rapidly to several meters once the plume breaks through into the aquifer.

12. A review of the limited literature for Alberta supports the lack of contaminant migration in fine grained soils with the extent of soil contamination localized even after 60 years of CFO operation (on a loam till).

13. Because $NH_4$ is the stable N species in anoxic environments, $NO_3$ concentrations in EMS facilities and in any associated contaminant plume will be low and should remain low as long as anaerobic conditions are maintained. However, discharge of $NH_4$-contaminated groundwater into wells or surface waters will result in nitrification.

14. Although the occurrence of $NO_3$ has been observed in contaminant plumes beneath active EMS sites in North America (associated with the nitrification of $NH_4$ within the manure wastewater) the prevailing hydrogeologic conditions in Alberta suggest that the development of elevated $NO_3$ concentrations associated with EMS seepage is low.

15. Nitrification of $NH_4$ enriched soils ($NH_4$ on the exchange sites) beneath abandoned EMS sites has been demonstrated. This observation suggests a potentially large reservoir of oxidizable $NH_4$ that may enter the groundwater regime at a later date (e.g., after site closure).

16. . Literature suggests that total-P associated with manure wastewater is attenuated quickly in the geologic media below CFOs.

17. Evidence suggests that seepage rates from EMS sites decrease with time if soil clogging or the presence of a permanent manure seal on the soil surface develops over time. Evidence indicates that the formation of natural manure seals in both EMS and feedlot operations as the sole means of controlling effluent seepage should be avoided, since these seals (clogging layer in EMS sites and manure interface layer in feedlots) have been demonstrated to be unreliable and easily compromised.

18. The significant variation in observed contamination from EMS seepage suggests that accurately predicting the potential for contamination from individual EMS sites based on the soil texture/grain size alone may be difficult.

19. Data on the aqueous source chemistry of solid swine and cattle manure, EMS wastewaters, and cattle feedlot runoff waters are generally lacking in Alberta with respect to the rest of North America. Specifically, data are lacking on those constituents needed to delineate the extent of soil and groundwater occurring from CFOs. Results are often not comparable because of differences in sampling methodologies

20. Data on baseline groundwater chemistry in the vicinity of CFOs are lacking.

21. Extended storage of EMS wastewater is required to reduce the amount of bacteria to acceptable levels. This storage time appears to vary between 50 and 100 days at an average temperature of 4°C and decreases as the storage temperature is increased. Data on groundwater contamination by bacteria are lacking. However, available data suggest that bacteria could potentially persist in the groundwater because of its characteristically low temperatures.

22. Pathogen and pharmaceutical contamination of groundwaters are not well understood. Pathogen and pharmaceutical contamination requires additional investigation for all types of CFOs (swine, cattle and poultry).

23. The North American literature demonstrates that the N content of poultry manure/litter is greater than that of swine or cattle manure and manure wastewaters. In addition, poultry manure/litter is characterized by elevated heavy metal content.

24. Investigations into various aspects of groundwater contamination from poultry CFOs (source chemistry to site hydrogeology to site contamination studies) are limited in North America and none were noted for Alberta.

# 9. Areas for Further Study

Based on the material presented in previous chapters, additional research, specific to conditions in Alberta, is suggested. Some of the key areas that require research are presented below.

1. Literature reviews show that within Alberta and North America in general, the aqueous source chemistry of CFO manure and manure wastewater (swine, cattle and poultry) is not well defined. Although the literature indicates that source chemistry varies widely, determining the concentration range of manure and manure wastewaters of Alberta CFOs would have merit. Such an undertaking should ensure that a uniform and geochemically rigorous collection and analysis method is followed. Such a study should include both filtered and unfiltered sample quantification for typical geochemical parameters in addition to trace metals, total coliforms, and *E. coli*. This approach will allow a synthesis between older data sets, which the literature indicates have generally employed collection techniques without filtering of samples, and data from future studies, which incorporate the sampling protocol recommended in this report.

2. The literature reviews demonstrate that the N content of poultry manure/litter is greater than that of swine or cattle manure and manure wastewaters; however, an understanding of the impact of the storage of this waste on the surrounding environment is lacking within Alberta. To understand the extent of contamination associated with these types of CFOs, a study to quantify the methods employed by poultry producers with regards to manure/litter wastes would be beneficial.

3. The Alberta literature review demonstrates that few studies have examined baseline monitoring with subsequent delineation of the extent of contamination at a later time period during the operation of a CFO. To confirm the value of baseline monitoring, a series of pilot projects focused on each of the various CFO types (swine, cattle, and poultry) could be implemented. These sites could be situated on representative hydrogeologic conditions in Alberta. The delineation of the extent of contamination requires and understanding of the effects of fracturing

on contaminant migration and retardation of $NH_4$-N. The controls exerted by fracturing in glacial tills on contaminant transport from CFOs should be investigated to determine if the EPM approach used to characterize transport in Chapter 7 (and commonly used in hydrogeology) is valid. The rates and quantity of sorption (and desorption) of $NH_4$-N that can occur on fine textured soils also requires investigation; these sorption/desorption processes have implications for both the attenuation of advancing contaminant plumes and for the long-term fate of existing contamination follow CFO closure.

4. A poorly understood area of the scientific literature is the contamination of soil and groundwater by pharmaceuticals, including antibiotics and growth hormones. Given the breadth of research currently being conducted and the growing concern of the public with respect to the potential effects of these compounds in the environment, it would be prudent to investigate these potential contaminants. This could be addressed through the implementation of a pilot project that systematically identifies the major pharmaceuticals employed at CFOs in Alberta, quantifies the average concentrations within manure and manure wastewaters, and examines their persistence both in the soil and groundwater environments typical of CFOs.

5. A key area requiring study that came to light during preparation of this report, but is beyond its scope, is the decommissioning of CFOs. Release and transformation of $NH_4$-N from the exchange sites can potentially occur upon decommissioning (resulting from changes in the hydrogeologic and geochemical regimes proximal to the CFO). Under hydrogeochemical conditions resulting from the decommissioning of CFOs, the $NH_4$-N on the exchange sites could be released and could enter the groundwater regime. As a result, the future impact(s) of CFO decommissioning on the release of exchanged $NH_4$-N into the groundwater should be addressed.

# 10. MONITORING THE PERFORMANCE OF LIQUID MANURE STORAGE FACILITIES DEEMED TO BE A RISK TO THE ENVIRONMENT

Two provisions of the existing Regulations [Alberta Standards and Administration Regulation, 306/2006] relate to groundwater monitoring at liquid manure storage facilities. First, under s. 7 of the Alberta Standards and Administration Regulation (306/2006), a manure storage facility must be constructed more than 100 m from a spring, 100 m from a water well, and 30 m from a common body of water unless otherwise authorized by the NRCB based on site conditions. Second, under s. 18, if the NRCB determines that there is a risk to the environment, the approval officer may require the owner or operator of a liquid manure storage facility to install and maintain a leakage detection system for the liquid manure storage facility. This monitoring system will consist of at least one monitoring well upgradient of the facility and at least two monitoring wells downgradient from the facility. The wells must be monitored on a schedule determined by the NRCB.

Based on literature reviews and the illustrative groundwater modeling conducted for this report, several observations and comments can be made with respect to the existing AOPA and implementing regulations as they relate to groundwater monitoring of liquid manure storage facilities once the NRCB determines that there is a risk to the environment from the facility. This Chapter summarizes these observations and comments. It is assumed that the NRCB has an accepted definition of risk and, as such, we have not attempted to redefine it.

## 10.1. Monitoring versus Alternative Regulations

The majority of jurisdictions in the United States manage the risk of contamination to surface and groundwater from manure by controlling surface runoff through construction design standards and siting criteria, and through the development of site-specific nutrient management plans, rather than by groundwater monitoring. This approach relates in large part to the existing framework of the National Pollutant Discharge Elimination System under the Federal Water Pollution Control Act, which authorizes the EPA to regulate point source discharges to surface water [33 U.S.C. 1250, et. seq.], although underlying practical and political reasons for the format may also exist. The federal government and the delegated states created regulatory programs consistent with this framework by focusing on impacts to surface water. As noted above, the EPA stated in its rulemaking that regulation of point source discharges would have the added benefit of protecting groundwater.

Several provincial jurisdictions (e.g., Saskatchewan) also place the primary emphasis on site selection, design and waste management practices. Some reasons that have been expressed for this are:

- Minimizing the risk of contamination reduces the liability of both the producer and province, reducing the overall work load of regulatory agencies and providing the greatest benefit to environmental protection;
- Detection of contamination can be expensive and difficult. Improperly designed monitoring systems may give false positives or miss detection;
- If monitoring is taking place at a site in a higher risk location and if there is confidence in the monitoring system data, then a consensus must be reached as to the chemistry of the groundwater that would define contamination. That is to say, a rise in chloride concentration may not trigger concern but a rise in nitrogen at some concentration may; and,
- If a site is shown to be experiencing contaminant migration, the regulatory agency's ability to enforce is difficult and time consuming. Further, the operator of the site may choose to abandon the site leaving the responsibility with the province.

In some of the delegated states that chose not to implement groundwater monitoring (such as North Carolina), the agencies believe that groundwater monitoring data do not support groundwater monitoring in addition to the surface water and design controls called for by the point source program. In other states, such as New Mexico and California, groundwater data collected at existing facilities were believed to support the need for groundwater monitoring in addition to design criteria and nutrient management. For those states that require groundwater monitoring, the programs fall into two categories: regulatory programs that specifically call for a monitoring well program (i.e., "one up and two down"), and those that simply prohibit discharges to groundwater and use agency discretion to impose monitoring on a site-by-site basis. Other provincial jurisdictions reviewed are equally divided. Ontario, Quebec, and British Columbia do not currently require groundwater monitoring for CFOs.

## 10.2. Regulations versus Guidance

In some jurisdictions such as Alberta and New Mexico, the details of the groundwater monitoring program are set out in technical guidance. In some states, such as Nebraska and Oklahoma, the monitoring program is incorporated into enforceable regulations. An assessment of the two approaches is outside the scope of this report; however, regardless of the option chosen for groundwater protection, different regulatory mechanisms are available for imposing those options.

## 10.3. Recommendations relating to Groundwater Monitoring Protocols

Based on the literature review, groundwater modeling, and other information reviewed for this project, the following comments are provided to enhance the existing protocols for groundwater monitoring of liquid manure storage facilities in cases where the NRCB deemed that there is a risk to the environment.

### 10.3.1. Expand list of baseline parameters

Most jurisdictions reviewed do not use a baseline monitoring program, although New Mexico requires one round of monitoring for $NO_3$-N, total Kjeldahl nitrogen (organic nitrogen plus ammonia; TKN), Cl and TDS prior to discharge. The Alberta approach incorporating baseline monitoring is valuable because a complete suite of baseline parameters are important to establish benchmark groundwater quality. The existing frequency of baseline monitoring appears appropriate.

However, baseline data should include all information that would allow the operator to assess the effects of the CFO on the groundwater. $NH_4$ is the dominant groundwater contaminant from CFOs, and therefore should be added to the list of baseline parameters. Although TKN is required in the list of baseline parameters, it does not address the concentrations of $NH_4$ explicitly as it is the sum of organic N, $NH_3$ and $NH_4$. For clarity, we also recommended that the baseline parameter $HCO_3$ be replaced by alkalinity.

If these parameters are added to the list of baseline parameters (for a minimal additional cost), the baseline data would comprise a complete suite of major ion analyses. This suite of chemical analyses could be used to assess the geochemical controls on the changes in the groundwaters chemistry in the future. Further, because the analytical laboratories are providing analyses of all potentially major dissolved ions, they should be requested to provide an ion balance on the water sample (for no additional cost). This balance can provide a cursory assessment of the completeness/accuracy of these analyses.

If a contaminant is observed in a monitoring well at some later date, knowing if it was present in the baseline data would be important. As such, consideration should be given to including total coliforms and *E. coli* to the list of baseline parameters. However, sampling for these parameters requires care in the installation of monitoring wells and/or significant purging prior to sampling, which may be difficult to achieve. The installation of monitoring wells and purging of the wells is discussed below.

At this time we believe that insufficient data are available to support the addition of metals or pharmaceuticals to the list of baseline parameters.

### 10.3.2. Expand list of indicator parameters

Existing NRCB indicator analyses compare well with many jurisdictions, and in some cases exceed the number of parameters monitored under other programs. For example, Nebraska requires only monitoring of $NO_3$, Cl and $NH_3$ while Alberta also includes K, TKN, P and EC in their list of indicator parameters. Other jurisdictions include the parameters $NH_3$, TDS, Na, and fecal coliforms.

To effectively assess the full range of risks posed by liquid manure storage, determining $NH_4$ concentrations is considered important. Further, as DOC is present in these facilities (and thus is an indicator parameter) as well as being a control on the speciation of N, it should also be added to the list of indicators.

Under specific conditions, consideration should be given to the inclusion of additional indicator parameters. To make the monitoring cost effective, the analyses of these indicator parameters could be triggered by an exceedance of specific indicator parameter(s). These additional indicators could include fecal coliforms and *E. coli*.

As an alternative, some researchers have noted that an initial measurement of EC with subsequent annual EC comparison sampling would also be appropriate. If EC varies more than 10%, then a complete indicator analysis would be required.

### 10.3.3. Number and general location of monitoring wells

Like Alberta, most jurisdictions require one upgradient and two downgradient monitoring wells. This approach has been demonstrated to provide effective monitoring in other regulatory programs and under a diverse range of hydrogeologic conditions.

The selection of the number and location of monitoring wells must be based on an understanding of the existing geology/hydrogeology and the groundwater flow system that will develop during operation.

We believe these monitoring wells, if properly located, should be adequate. Ideally, however, a greater number of monitoring points could be used at early times to ensure that the system is understood, with fewer monitoring points at later times to act as checks. In this case, the number of wells installed and monitored would be related to the level of understanding required of the system. If the hydrogeologic system is well defined or simple, then fewer monitoring points would be required. In addition, the nature of the hydrogeologic system might suggest that only 1 or 2 wells are required to provide verification of overall flow system.

In hydrogeologic settings that are demonstrably slow moving, monitoring wells (with frequent sampling, e.g., annual) may not be as effective as direct sampling methods (such as coring and extraction of pore waters for analyses) on a less frequent basis (e.g., decadal).

### 10.3.4. Specific locations of monitoring wells

No specific guidance is provided in the NRCB guidance document with respect to the location of monitoring wells because the hydrogeology of each site is unique; well locations cannot be boiler plated. Field data and modeling suggest, however, that many of the monitoring wells in North America are likely either too far from the source or at the wrong depth in relation to the facility to provide meaningful data on the extent of groundwater contamination.

Some states, such as New Mexico and Oklahoma, provide guidance for the location of wells. The extent of contaminants (e.g., $NH_4$) as demonstrated in the literature and the groundwater modeling suggest at least one monitoring well should be located as close to the CFO as possible. These well(s) would provide an "early warning" of the migration of any contaminants. In the case of sites directly underlain by unoxidized till, in which the literature shows $NH_4$ can be retarded with respect to Cl, monitoring in close proximity to the CFO may be the only way to assess the migration of contaminants from the CFO.

### 10.3.5. Construction of monitoring wells

Proper and consistent construction of monitoring wells is critical because the near surface hydrogeologic materials in Alberta are typically complex, commonly consisting of clay-rich fractured glacial tills. Poor installations in these media may result in false positives because of the elevated concentrations of potential contaminants in CFO sources areas. Oklahoma and Nebraska impose specific criteria for monitoring well installation. Factors such as length of well screen and sand or gravel pack, and the use of a bentonite seal are included [OAR 35:17-3-11(e)(6)(G) and Title 178 NAR Ch 12].

The NRCB does not include guidance for the construction of monitoring wells in its regulations or technical guidance relating to CFOs, but requires that the construction of the wells be overseen by a qualified professional. Alberta Agriculture implements the Water Act Ch W-3, and the Water (Ministerial) Regulations AR 205/98 [Alberta, 1998]. The Water Act, as implemented by Alberta Agriculture, provides the following definition of a water well:

*(kkk) "water well" means an opening in the ground, whether drilled or altered from its natural state, that is used for*

*1) the production of groundwater for any purpose,*

*2) obtaining data on groundwater, or*

*3) recharging an underground formation from which groundwater can be recovered, and includes any related equipment, buildings, structures and appurtenances.*

This definition appears to be broad enough to include groundwater monitoring wells and if this is the interpretation, the Water (Ministerial) Regulations include well construction standards for “water wells” that could then be applied to groundwater monitoring wells [s. 66(4); Alberta, 1998]. Consideration should be given to providing more specific direction with regard to the construction of monitoring wells in the NRCB guidance. Existing guidance could be referenced.

### 10.3.6. Decommissioning monitoring wells

The NRCB guidance and regulations are silent on the requirements for decommissioning and abandonment of monitoring wells. The Water (Ministerial) Regulations set out requirements for reclamation of an abandoned water well, which includes specific requirements for removal or reduction of casing, liner and riser pipe, and filling the full length of the well [s. 66(4); Alberta, 1998]. Consideration should be given to including directions on proper well decommissioning in the NRCB guidance. Existing guidance could be referenced.

### 10.3.7. Groundwater sampling protocols

Sampling methodology is important to ensure that groundwater samples collected and analyzed are representative of formation waters. As such, the sampling protocol must be rigorous and can be complex. For example, the redox sensitive nature of $NH_4$ requires that the well water samples be collected in such a manner as to reduce aeration and that the sample be filtered. In addition, the number of well volumes purged prior to sampling must be clearly defined. This can be problematic in the case of some tills, which do not yield water rapidly. In addition, the use of filtering may have an impact on the concentrations of many constituents. In some handbooks, TKN (organic nitrogen plus ammonia) is determined on unfiltered samples while the concentrations of nitrogen occurring in the oxidized state ($NO_2$ and $NO_3$) and as ammonia are determined separately using the filtered sample. Further preservation of some samples (including acidification) is necessary while others should not be preserved.

The NRCB guidance states that a "suitably qualified professional will be responsible for the sampling protocol, and analysis and assessment of the chemical data. Sampling must be conducted by a properly trained individual, under the guidance of a professional" [NRCB, 2004]. Some states, such as Nebraska, require the submission and approval of a sampling and analysis plan prior to any sampling event [Title 130 NAR Ch. 13, s. 005]. Reference to acceptable sampling protocols could be included in NRCB guidance.

### 10.3.8. Frequency of sampling

With optimum placement of a monitoring well (see above), the NRCB guidance requiring biannual baseline sampling and indicator sampling intervals appears adequate. For comparison, New Mexico guidance describes quarterly sampling for indicator parameters.

Based on the limited extent of plume development observed in the literature and in the model simulations, one could argue that no modifications to the sampling intervals would be valid. However, a greater intensity of sampling and monitoring at early times could be valuable to verify assumptions on the hydrogeologic system and its evolution during operation, followed by less frequent sampling for validation. As noted above, a baseline EC measurement with annual comparisons may also be used as a trigger for increased sampling for indicator parameters.

### 10.3.9. Water level measurements

Water levels in monitoring wells can provide information on direction of groundwater and contaminant migration on the site. TG 2004-01 [NRCB, 2004] indicates that "Leakages from EMS facilities are detected by monitoring changes in groundwater levels and chemistry over time"; however, the collection of water level data is not included in that guidance. To allow comparisons of water level data across the site, all water levels should be reported with reference to an on site datum. Water level measurements should be taken immediately prior to water sampling and this direction could be included in NRCB guidance.

## 10.4. References


Alberta Standards and Administration Regulation 267/2001 consolidated up to 306/2006.

Alberta Water Act Ch W-3. http://www.qp.gov.ab.ca/Documents/acts/W03.CFM

Alberta Water (Ministerial) Regulations AR 205/98. http://www.qp.gov.ab.ca/Documents/REGS/1998_205.CFM

Federal Water Pollution Control Act, 33 United States Code Sections 1251-1376 (1987).

Natural Resources Conservation Board (NRCB) of Alberta, 2004, Technical Guideline TG 2004-01: Leak Detection and CFOs: http://www.nrcb.gov.ab.ca/downloads/documentloader.aspx?id=3634

Nebraska – N.A.R. Title 130 Chap 1 et. seq. and Title 178 NAR Chap 12 et. seq. http://www.deq.state.ne.us/

Oklahoma – Oklahoma Concentrated Animal Feeding Operations Act, OK Rev. Stat. Title 2, Chapter 1 Article 9.200 et. seq. and O.A.R. 35:17-3-11 et. seq. http://www.oar.state.ok.us/oar/codedoc02.nsf/frmMain?OpenFrameSet&Frame=Main&Src=_75tnm2shfcdnm8pb4dthj0chedppmcbq8dtmmak31ctijujrgcln50ob7ckj42tbkdt374obdcli00_

# 11. Monitoring the Performance of Liquid and Solid Manure Storage and Collection Facilities Constructed to the Standards

The Scope of Work for this project included the task to:

> *Recommend economically and technically viable groundwater monitoring protocols that could be used in Alberta to monitor the performance of liquid and solid manure storage and collection facilities constructed to the construction and performance standards specified in the Agricultural Operation Practices Act Standards and Administration Regulation.*

The purpose of this Chapter is to provide the information to respond to this task.

Under the existing regulations [Alberta Standards and Administration Regulation, 267/2001], a manure storage facility or manure collection area must have either a protective layer or a liner between the facility or area and the uppermost groundwater resource below the site. The bottom of the liner of a manure storage facility and of a manure collection area must be not less than 1 m above the water table of the site at the time of construction or, if a protective layer is used, the bottom of the manure storage facility or manure collection area must be not less than 1 m above the water table at the time of construction. Lastly, the bottom of a liner or the base of the protective layer must not be less than 1 m above the top of the groundwater resource.

The protective layer of a manure storage facility and of a manure collection area must provide equal or greater protection than provided by naturally occurring materials:

a) 10 m in depth with a hydraulic conductivity (K) of less than $1x10^{-8}$ m/s for a liquid manure storage facility,

b) 5 m in depth with a K of less than $1x10^{-8}$ m/s for a catch basin, or

c) 2 m in depth with a K of less than $1x10^{-8}$ m/s for a solid manure storage facility or solid manure collection facility.

If a liner (compacted soil or constructed of concrete, steel, or other synthetic or manufactured media) is used at a manure storage facility or of a manure collection area, it must provide equal or greater protection than that provided by compacted soil:

a) 1 m in depth with a K of less than $1x10^{-9}$ m/s for a liquid manure storage facility,

b) 1 m in depth with a K of less than $5x10^{-9}$ m/s for a catch basin, or

c) 0.5 m in depth with a K of less than $5x10^{-9}$ m/s for a solid manure storage facility or solid manure collection facility.

Further, an approvals officer may issue or amend an approval, registration or authorization for a manure storage facility or manure collection area if it has a liner or a protection system than uses biological methods, monitoring or performance standards that provide equal or greater protection than that provided above. In addition, a manure storage facility or manure collection area must be constructed to have positive drainage to prevent the collection of water.

Data from the literature suggest that all protective layers or liners can leak and contaminate local groundwaters. These systems include concrete lined storage systems [e.g., Minnesota Pollution Control Agency, 2001]. In the case of earthen barriers, including compacted clay barriers, there are several key performance considerations. In the short term, these barriers can achieve the specified K upon construction. However, over the medium and long terms the K can increase with time because of several factors including desiccation, shrink/swell, freeze/thaw, root penetration, thermal stresses, differential settlement, or chemical incompatibility. Secondary features, such as fractures and voids can significantly increase the K of the barriers. Such fractures are also present in natural till and clay deposits, as noted in the literature review sections of this report. Recognizing that engineered containment systems have only been in existence for a few tens of years and that their containment properties can change with time, the longevity of these systems requires study.

The most conclusive information on the performance of liquid and solid manure storage and collection facilities constructed to the construction and performance standards will come from monitoring data from the hydrogeologic environment adjacent to the protective layer or liner (i.e., the low K layer). As a result, long term groundwater monitoring at manure storage facility or manure collection area should be initiated.

The design of this leak detection monitoring system could be modeled after that presented in s. 18 of the Regulations. Under Section 18 of those Regulations, the approval officer may require the owner or operator of a liquid manure storage facility to install and maintain a leakage detection system for a liquid manure storage facility. This system will consist of at least one monitoring well upgradient of the facility and at least two monitoring wells downgradient from the facility.

As is the case for the liquid manure storage facilities (Chapter 10), the selection of the number and location of monitoring wells must be based on an understanding of the existing geology/hydrogeology and the groundwater flow system that will develop during operation. Ideally, a greater number of monitoring points could be used at early times to ensure that the hydrogeologic system is understood, with fewer monitoring points at later times to act as checks.

In addition to detecting the leaks in the hydrogeologic environment adjacent to the protective layer or liner, this monitoring data should be augmented with an appraisal of the migration of contaminants through the protective layer or liner. This would best be done via chemical analyses on core samples collected from core holes. The use of monitoring wells should not be considered an option for this purpose because wells yield an integrated water sample over the intake zone of the well and, as a result, will not yield the needed level of detail on the development of the contaminant profile with depth through the low K layer. Operationally, there are also benefits to not having too many monitoring wells located immediately adjacent to or in the facility. To provide detail on the time-depth profiles of potential contaminants, core samples should be collected from new core holes located adjacent to core holes drilled at an earlier time.

Profiles of contaminants through the protective layer or liner can be constructed by performing chemical analyses on paste extracts from the core samples. Alternately, and of more value to assessing contaminant migration, samples of porewater could be removed from the core samples for analyses. Successful porewater removal techniques from core samples of fine textured media have included squeezing or centrifugation.

If the media in the protective layer or liner is unsaturated (i.e., vadose zone), suction lysimeters could be used to collect porewater samples. However, lysimeters require routine maintenance.

As was the case for the observations and comments provided with respect to the exiting AOPA and implementing regulations as they relate to groundwater monitoring of liquid manure storage facilities (Chapter 10), no specific guidance can be provided with respect to the location of monitoring wells or core holes because the hydrogeology of each site is unique. Further comments on construction and decommissioning of these monitoring wells, groundwater sampling protocols, frequency of sampling, and water level measurements equally apply here.

## 11.1. References

Alberta Standards and Administration Regulation 267/2001 consolidated up to 306/2006.

Minnesota Pollution Control Agency, 2001, Effects of liquid manure storage systems on ground water quality. Minnesota Monitoring and Assessment Program, Environmental Outcomes Division, April 2001, 120 pp.

## 12. ASSESSING THE IMPACT OF SEEPAGE ON SURFACE AND GROUNDWATER RESOURCES

The scope of work for this project included the task to:

> *"…generate the information required to assess the impact that seepage from manure storage and collection facilities are having on surface and groundwater resources."*

Although available data suggest the fine grained nature of much of the soils in Alberta may limit the extent of contamination in many cases, the answer to this question cannot be obtained from available data. The purpose of this Chapter is, therefore, to outline a study approach to respond to this task. Specifically, a strategy is suggested to quantify the impacts of contaminants on receptors. It is important to stress that, as is the case for most hydrogeologic studies, each study site is hydrogeologically unique. As a result, only a conceptual approach can be presented here. The approach employed at a specific site must consider specific subsurface conditions and will need to be tailored to suit the unique hydrogeology of the site.

The conceptual study approach is as follows:

1. Identify a suite of potential long-term, high risk and typical (of various hydrogeologic settings) sites for consideration;
2. Using available data and a set of selection criteria, select a smaller number of representative sites for further investigation;
3. Develop site specific investigation plans for these representative sites;
4. Instrument, sample, and analyze data collected from each site; and
5. Determine large-scale implications of CFO seepage on surface and groundwater resources.

Some aspects of the conceptual study design may be based on CSA [2000], ASTM [2002], or SAFRR [2004] documents. In the case of CSA [2000], materials of use may include the Planning and Site Investigation (Chapter 6) and Conducting a Site Investigation (Chapter 7).

In contrast to general site assessments (as presented in the CSA Phase II document), quantifying the impacts of contaminants on receptors requires a research approach to define specific processes controlling the migration of contaminants in the context of Alberta's subsurface. These processes, identified within this report, include the rates of sorption and desorption of $NH_4$ onto the fine textured geomedia and the masses of $NH_4$ that can exchange onto these fine textured geomedia and thus retard the migration of the $NH_4$, as well as the effects of changes in redox conditions on the mobility of N species. Further, as part of this study, some of the limitations with respect to the existing knowledge base could also be addressed. These limitations include: defining the source chemistry of CFOs, and collecting and assessing baseline groundwater data, the extent of contamination of groundwater by pharmaceuticals, including antibiotics and growth hormones, and the controls exerted by fracturing on transport in glacial tills (i.e., the utility of the EPM approach). Once these processes and limitations are understood, the impacts that CFO contaminants are having (and will have in the future) on surface and groundwater resources can be determined.

Notably, most of the knowledge gaps identified in Chapter 9 are also identified as processes or limitations that should be addressed if an assessment of the impact that seepage from manure storage and collection facilities is having on surface and groundwater resources proceeds.

## 12.1. Selection of Potential Study Sites

From the literature reviews and the hydrogeologic modeling presented in previous chapters, some hydrogeologic environments are clearly more susceptible to CFO contamination than others. The susceptible sites include those constructed in permeable near surface aquifers and on thin fractured tills overlying aquifers. In contrast, less susceptible sites include CFOs installed on thick tills comprised of clay rich soils with high cation exchange capacities. Assessment of the processes controlling the migration of contaminants should be undertaken in both of these end member hydrogeologic environments. We believe that a key source of data for the selection of potential sites is the NRCB database on existing sites.

## 12.2. Selection of Study Sites

Criteria for selection of the types and numbers of sites for further investigation should be developed at this stage. Selection should be based on practical issues (e.g., site location and access), technical issues (e.g., complexity of hydrogeologic system), environmental and operational factors (e.g., chronology of CFO operation), while ensuring that sufficient information is available on geology, regional hydrology, and hydrogeology. Some specific information required should include: type and size (footprint) of CFO, agro-intensity of CFO, age of CFO, presence of any liners, review of any existing monitoring, knowledge of contaminants and their extent, geology, depth of bedrock, depth to groundwater, extent of any aquifers, type and texture of soils in the area (agronomic soils classification), proximity to surface waters and water supply wells, regional topography, and adjacent land use practices (e.g., manure spreading).

The information compiled at this stage can be used to: (1) determine if the site meets the requirements established for further investigation, and, if it does, (2) develop a conceptual model of the site hydrogeology and form the basis for developing a site investigation plan.

## 12.3. Develop Site Specific Investigation Plans

Information to be collected at this stage of the investigation include the plans for each site, topographical, soils and geological surveys, identification of depth of base of exploration, identification of the lateral extent of exploration, identification of numbers and depths of boreholes and their locations, depth of groundwater instrumentation (piezometers and water table wells), and whether piezometers can be nested or completed in individual boreholes. In addition, preliminary decisions will be required on the type of drill rig required, the type(s) and numbers of solids and groundwater samples to be collected, analytical methods to apply to the water and solids samples, and construction technique(s) for groundwater instrumentation. Further, sampling should include the source term chemistry for the CFO. At this stage of the study a cost estimate can be generated for each site.

## 12.4. Instrument, Sample, and Analyze

Based on the data collected from the field program, initial evaluations will be made of the groundwater flow characteristics (including residence times), aqueous geochemistry (including potential contaminants), types and extent of contaminant plume(s), as well as solids chemistry (including exchangeable cations on clays and hydrogeotechnical parameters).

The frequency of water sampling will be dictated by the specific nature of the flow system. If groundwater flow rates are low, water samples for chemical analyses can be collected less frequently than under more rapid flow conditions.

After adequate water level and pore water chemistry data sets have been collected from the instrumentation and interpreted in concert with the solids data, the decision as to whether enough data has been collected to characterize the plume(s) should be made. In most cases, characterizing the distribution of the plume(s) may require additional instrumentation and monitoring. At this stage in the study the potential receptors and the processes controlling the fate and transport of the contaminants should be clearly defined.

## 12.5. Determine Large-scale Implications

By developing an understanding of the processes that control the fate and transport of contaminants at the study sites, a comprehensive assessment of the migration of contaminants from CFOs under a range in hydrogeologic regimes common to Alberta will be possible. The implications of these findings will allow the impact(s) of seepage from manure storage and collection facilities on surface and groundwater resources (and could have in the future) in Alberta to be determined.

Importantly, this assessment program will require a lengthy study period, likely requiring a time commitment in excess of 5 years. It will also require the active involvement of a number of specialists, including hydrogeologists, soil chemists, hydrogeochemists, agrologists, and agricultural engineers.

## 12.6. References

ASTM, 2002, Standard Guide for Environmental Site Assessment: Phase II Environmental Site Assessment Process, ASTM Standard E1903-97.

CSA, 2000, Phase II Environmental Site Assessment, CSA Standard Z769, CSA International, Toronto, March 2000.

SAFRR (Saskatchewan Agriculture, Food and Rural Revitalization), 2004, Site Characterization Manual for the development of Intensive Livestock Operations and Earthen Manure Storage, May 2004.