

Soil and Groundwater Quality under a Cattle Feedlot in Southern Alberta

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Environmental concerns related to manure often focus on land application and the negative effects on soil and water quality. However, there is also potential for contamination of soil and water directly from feedlot sites. The objective of this study was to determine if a newly constructed feedlot in southern Alberta would change soil and groundwater quality under the feedlot within the first four years of operation. A cattle feedlot was constructed at the Agriculture and Agri-Food Canada Research Centre in Lethbridge, Alberta, in 1995 and 1996. Sixteen groundwater wells were installed at the feedlot in 1996. Groundwater chemistry and microbiology were monitored from 1996 to 2000, which included a baseline period of 3 months at the start before the feedlot was stocked with cattle. Soil samples (0- to 0.15-m, 0.15- to 0.3-m, and every 0.3- to 1.5-m depth) were collected in 1996 and 1999. Mean water-table depth ranged from 1.23 to 2.50 m. Some soil chemical properties, PO₄-P, NO₃-N, NH₄-N and K, were only significantly affected in the top 0.15-m layer. Other soil properties, EC, SAR, SO₄-S, Mg, Ca and Na, increased significantly to a depth of 0.6 m. Chloride content increased significantly to a depth of 1.5 m. Groundwater analysis indicated that contaminants had leached to the water table. Chloride concentrations, *E. coli* counts and total coliform counts increased in the wells within the pen area, whereas there was little change in the wells outside the pen area.

Key words: feedlot, cattle, groundwater quality, soil, leaching, nutrients, bacteria

Introduction

Intensive livestock operations (feedlots) are a major component of the cattle industry in Alberta. Of the estimated 5.65 million cattle in Alberta in 1998 (Alberta Agriculture, Food and Rural Development 2004), about one million were fed in about 4500 feedlots (Dey 1998). Major environmental issues related to feedlots include the negative effects of disposal of large quantities of manure on surrounding farm land in areas where the density of feedlots is high. Improper manure disposal can adversely affect soil, groundwater and surface water quality (Ap Dewi et al. 1994). Though most of the concern has been with land application of manure and its effects on soil and surface water quality, there is also potential for contamination directly from feedlot sites, with particular concern of groundwater contamination.

Active cattle feedlot pens develop three layers: manure layer, interface layer and top of the soil profile, which has been physically and chemically altered (Mielke et al. 1974). The interlayer is a mixture of manure and soil and is formed by hoof action of cattle. The interlayer is believed to act as a barrier to air and water movement (Mielke et al. 1974; Mielke and Mazurak 1976). Also, the poorly aerated layers that form can potentially cause

denitrification resulting in minimal leaching and accumulation of nitrogen below feedlot pens (Saint-Fort et al. 1995). Researchers in Nebraska, United States, analyzed soil cores for nitrate (NO₃) and concluded that infiltration into the subsurface below feedlots was essentially zero (Lorimor et al. 1972; Mielke et al. 1974; Ellis et al. 1975; Mielke and Mazurak 1976). Miller (1971) sampled soil and groundwater beneath 80 feedlots in the Texas High Plain, and concluded that there was no regional subsurface NO₃ pollution from cattle feedlot runoff. If NO₃ is readily denitrified in feedlots, it is not surprising these researchers concluded there is little infiltration below feedlots. In contrast, Coote and Hore (1979) reported contamination of groundwater by NO₃, chloride (Cl) and sodium (Na) from a feedlot in eastern Canada. Elliott et al. (1976) found greater concentrations of calcium (Ca), magnesium (Mg) and manganese (Mn) in soil solution below a feedlot compared to nearby cropland. Maulé and Fonstad (1998, 2000) sampled groundwater near five 25- to 35-year-old cattle feedlots in central Saskatchewan and concluded that 50 to 67% of the samples had elevated concentrations of solutes due to the presence of manure. Maulé and Fonstad (2002) also concluded that NO₃ is not a reliable indicator of manure seepage due to biological transformations.

There has been very little work on hydrology of feedlots in Alberta with respect to nutrients and other

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pollutants. The only major recent study in this regard was carried out by Kennedy et al. (1999) on a feedlot in central Alberta. They studied surface runoff and infiltration, but did not examine the groundwater beneath the feedlot. Earlier work by Sommerfeldt et al. (1973) found that nitrate-nitrogen (NO₃-N) and phosphate-phosphorus (PO₄-P) concentrations in shallow groundwater were increased adjacent to feedlots, but values generally remained within safe limits.

Findings from the literature suggest that other contaminants, in addition to NO₃, need to be assessed to determine movement of water below feedlots, and studies that looked at only soil showed minimal or no leaching beneath feedlots, whereas studies that examined groundwater usually revealed some contamination. This suggests macropore flow as a possible mechanism for groundwater contamination with minimal effects in the overlying soil matrix. Few studies have examined soil and groundwater beneath feedlots.

The objective of this study was to determine if a newly constructed cattle feedlot in southern Alberta would change soil and shallow groundwater quality under the feedlot within the first four years of operation. In this study a variety of constituents were measured in the soil and groundwater outside of the feedlot and at various locations within the feedlot during four years.

Materials and Methods

Feedlot Site

The feedlot was located at the Agriculture and Agri-Food Canada Research Centre, Lethbridge, Alberta. Construction on the feedlot began in 1995, and the first phase (32 pens) was completed in late spring of 1996. The first phase included four rows of eight pens in each row. The rows of pens were orientated north-south (Fig. 1). Each pen was 14 by 19.5 m in size. Topsoil that was removed during land grading, was stockpiled on the south side of the feedlot. The overall site was sloped from south to north for drainage, with a 0.35% slope. A catch basin, to hold drainage water from the feedlot, was constructed on the north side of the feedlot. Runoff from the individual pens was directed to the catch basin via drainage alleys (Fig. 1). These alleys were also used to handle and move cattle. Two more rows of pens (16 pens) were located on the east side of the initial 32 pens during the construction of phase two in 1998. A few larger pens were also constructed along the west side of the initial 32 pens.

The original 32 pens had a capacity to hold about 480 head of cattle (*Bos taurus*). Each pen held a maximum of 15 animals for a stocking density of 18 m² per animal. Cattle were generally stocked in the pens from October-December until June-August, varying among pens and requirements of research trials. The pens were

bedded mainly with barley straw, but some pens were bedded with wood chips. Manure was removed from the pens during the summer season.

The soil type at the site was predominantly an Orthic Dark Brown Chernozem (CAESA-Soil Inventory Project Working Group 1998). The AGRASID (Agricultural Region of Alberta Soil Inventory Database) map unit is LEWN4/U11, with Lethbridge and Whitney as co-dominant soils, and Chokio and miscellaneous eroded as significant soils (CAESA-Soil Inventory Project Working Group 1998). The site consisted of lacustrine material (about 1 m thick) over an oxidized till deposit at Wells 1 to 4 and Wells 14 to 16 (Fig. 1). The top soil layer had been removed from the pen area where Wells 5 to 13 were located. The following soil parameter values are for the soil profile within the pen area after the topsoil was removed. Soil texture was relatively uniform, and predominately clay loam in the top 1.5 m (33% sand, 34% silt, 33% clay). Total N and total P concentrations decreased with soil depth. Total N ranged from 1.23 g kg⁻¹ (0 to 0.15 m) to 0.34 g kg⁻¹ (1.2 to 1.5 m), and total P ranged from 1.62 g kg⁻¹ (0.15 to 0.3 m) to 1.47 g kg⁻¹ (1.2 to 1.5 m). Total carbon (C) generally decreased with

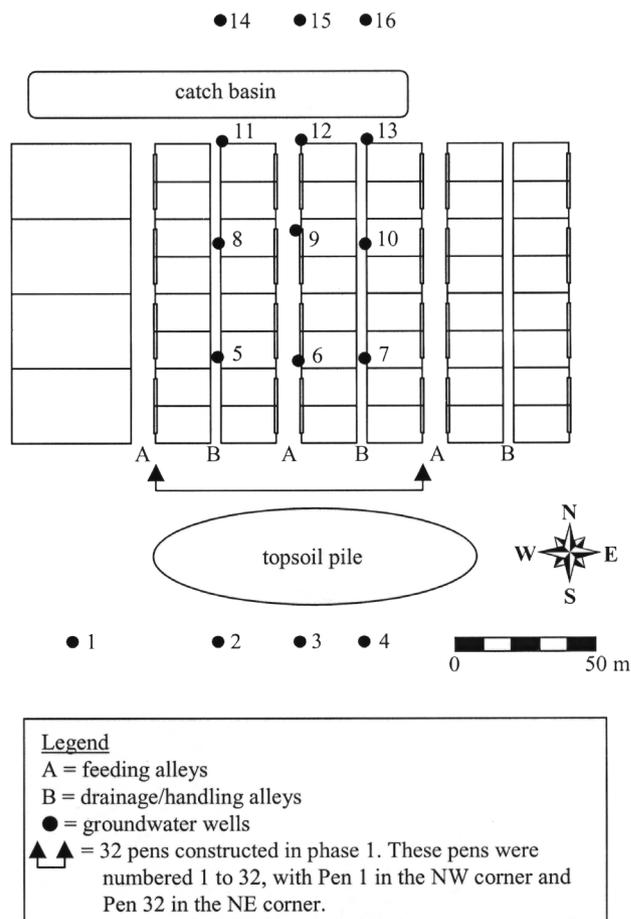


Fig. 1. Layout of the research feedlot at the Agriculture and Agri-Food Canada Research Centre at Lethbridge, Alberta.

depth from 20.8 g kg⁻¹ in the 0.15- to 0.3-m layer to 9.49 g kg⁻¹ in the 1.2- to 1.5-m layer. The proportion of total C as inorganic C ranged from 40 to 66%. The proportion of total C as inorganic C increased with depth.

Groundwater Well Installation

Groundwater wells were installed at 16 locations in and around the feedlot (Fig. 1). The wells were located so they were protected from normal feedlot activity. Wells were installed from March 20, 1996, to May 10, 1996. Slotted plastic (PVC; 50.8 mm diameter) and stainless steel (38.1 mm diameter) wells were installed to an average depth of about 5.87 m below ground level. Stainless steel pipe was used for Wells 1 to 4 and 14 to 16, and PVC pipe was used for the other wells. The wells protruded (i.e., the wellhead) about 0.24 m above the soil surface. Wells were slotted to within 1.5 m from the top end of the wellhead, and the slotted portion was covered with a filter sock. The sides of the wells were backfilled with sand to within about 1.1 m below the soil surface. The remaining distance to the soil surface was backfilled with bentonite clay. A protective steel casing (1 m long; 169 mm diameter) was installed around each well. The steel casing protruded about 0.26 m above the soil surface. The elevations (i.e., metres above sea level) of the top of each wellhead and the adjacent soil surface were measured.

Groundwater Sampling and Analyses

Wells were bailed several times during a 10-day period prior to the first groundwater sampling. Sampling protocol involved first measuring the water table. Groundwater samples were collected with a Waterra foot valve attached to high-density polyethylene tubing. The foot valve and tubing were first flushed with water from the well. A 250-mL sample bottle was then rinsed once with well water before collecting the sample volume. The Waterra sampler was replaced with a stainless steel bailer in May 1998. Collected samples were placed in coolers with ice packs and transported to the laboratory for analysis. All groundwater samples were filtered under vacuum through 0.45- μ m filter paper within a few hours of collection.

Groundwater was monitored from May 1996 to November 2000. During the first three months wells were sampled weekly for a total of 11 sample times. These samples served as the baseline period (May 22 to August 21, 1996) during which conditions were relatively dry and only a few animals occupied the feedlot. Water samples were analyzed for: pH; electrical conductivity (EC); Ca and Mg (atomic absorption); sodium (Na) and potassium (K) (flame photometry); sulphate-sulphur (SO₄-S) (turbidimetric method); Cl (thiocyanate colorimetric method); carbonate (CO₃) and bicarbonate (HCO₃) (acid titration); NO₃-N (hydrozine reduction

colorimetric method); Mn, iron (Fe), copper (Cu) and zinc (Zn) (atomic absorption) (Greenberg et al. 1992). Sodium adsorption ratio (SAR) was calculated using the Na, Mg and Ca data.

Groundwater sampling was continued on a monthly basis (weather permitting) after the baseline period until February 1998. These samples were only analyzed for NO₃-N and Cl. Electrical conductivity and pH were also measured on a few sample sets.

More extensive analysis was carried out from April 1998 to November 2000. The analytical suite included PO₄-P, ammonia-nitrogen (NH₃-N), pH, EC, Ca, Mg, Na, K, SAR, SO₄-S, Cl, CO₃, HCO₃, NO₃-N, Mn, Fe, Cu and Zn. Selected biological parameters were also measured according to the methods described by Miller et al. (2004), and these included *Escherichia coli* (*E. coli*), total coliform and total aerobic heterotrophs at 27°C. The microbiological results are presented as common log most probable number per 100 mL (log MPN 100 mL⁻¹) for the total coliform and *E. coli* data, and as common log colony-forming units per 100 mL (log CFU 100 mL⁻¹) for the heterotroph data. The detection limit for the log MPN 100 mL⁻¹ was 1.56. For calculating means, a value of half the detection limit was assigned to those samples that were below the detection limit. The antilog of 1.56 is 36.3. Half this value is 18.15, which has a common log of 1.26.

Soil Sampling

Baseline soil samples were collected in March 1996, during construction of the feedlot, from 16 of the 32 pens (Pens 1, 3, 5, 7, 10, 12, 14, 16, 17, 19, 21, 23, 26, 28, 30 and 32; Fig. 1) using a hydraulically powered core tube mounted on a truck. Three core samples were collected to a depth of 1.5 m from each pen. Composite samples were prepared from the three cores in increments of 0 to 0.15, 0.15 to 0.3, 0.3 to 0.6, 0.6 to 0.9, 0.9 to 1.2 and 1.2 to 1.5 m. Soil samples were air dried and ground (<2 mm). A second set of soil samples was collected from the same pens in August 1999. The pens were empty of cattle, and were in the process of being cleaned of manure at the time. Where the underlying soil was not exposed, the manure pack and compact organic layer were first removed with a shovel before soil cores were collected. Therefore, the 0- to 0.15-m soil increment was the soil layer immediately below the compact organic layer.

Soil samples were analyzed for pH, EC, Ca and Mg (atomic absorption), Na and K (flame photometry), Cl (thiocyanate colorimetric method), HCO₃ (acid titration) and SO₄-S (turbidimetric method) using the saturated-paste extraction method (Carter 1993); extractable NO₃-N and NH₄-N using the 2 M KCl extraction method (Carter 1993); extractable PO₄-P using the Modified Kelowna extraction method (Qian et al. 1994);

and extractable Mn, Fe, Cu and Zn using the diethylene triamine pentaacetic acid (DTPA) extraction method (Carter 1993).

Additional soil samples were collected in 1996 and 1999 to determine soil bulk densities. Cores were taken to a depth of 1.5 m in six incremental depths (0 to 0.15, 0.15 to 0.3, 0.3 to 0.6, 0.6 to 0.9, 0.9 to 1.2 and 1.2 to 1.5 m). A 0.05-m section was removed from the mid-section of each incremental layer and oven dried (105°C) to determine soil bulk density.

Climatic Data

Climatic data were obtained from a meteorological station operated by the Agriculture and Agri-Food Canada Research Centre. The station was located about 0.8 km north of the feedlot (49°42'N, 122°46'W, 921 m elevation).

Data Processing and Statistical Analysis

Data processing was carried out using SAS (SAS Institute Inc. 2000). The 1996 and 1999 soil data were treated as paired samples. The null hypothesis that means between years were the same was tested using the UNIVARIATE procedure in SAS. Since the data were not normally distributed, the nonparametric statistic, Wilcoxon Signed Rank Test, was used to test for differences between the two years in each soil layer.

Results and Discussion

Climatic Conditions

Precipitation during the baseline period (May 22, 1996, to August 21, 1996) was 53% below the May-to-August 30-year normal (Fig. 2). Total precipitation was 10% below the 30-year normal in 1996, and total precipitation was normal in 1997. In contrast, precipitation during 1998 was 32% above the 30-year normal, with March and June being particularly high. Precipitation was below the 30-year normal by 6% in 1999 and by 25% in 2000. The months May to August, October, and November were particularly dry in 2000. The 30-year normal mean monthly temperature ranged from -7.8°C for January to 18.2°C for July at this site (Environment Canada 2004). Daily average temperatures for each month from 1996 to 1998 were similar to the 30-year normal. Temperature in January to March 1999 (-0.4°C) was above the 30-year normal (-4.1°C). Temperature in November to December 1999 (3.4°C) was also above the 30-year normal (-3.6°C). Temperature during the rest of 1999 was comparable to the 30-year normal. Temperature in 2000 was above the 30-year normal throughout most of the year. The daily average temperature was 6.2°C in 2000, compared to the 30-year normal of 5.8°C.

Pen Floor Development

Four layers were visible where manure was still present at the time of soil sampling in August 1999. The top layer was the fresh manure pack. Located under the manure pack was a black, compact organic layer, which was on top of the soil. The soil was visibly darker in colour within the top 0.15 m, representing the third layer. This colouring of the soil was most likely the result of organic material from the manure pack and the compact organic layer moving into the soil or mixing with the top soil layer. Below this layer was the visually unaltered soil material. However, the soil was chemically affected by the manure pack to depths greater than 0.15 m, as evident from the following results comparing data from the two sampling dates. Mielke et al. (1974) described three distinctive layers above the unaltered soil, referring to the second layer as the interface layer (i.e., compact organic layer) of mixed organic material and mineral soil that forms under the surface manure layer.

Soil Bulk Density

Mean soil bulk density values ranged from 1.70 to 1.87 Mg m⁻³ for the six soil layers (0 to 0.15, 0.15 to 0.3, 0.3 to 0.6, 0.6 to 0.9, 0.9 to 1.2 and 1.2 to 1.5 m) in 1996. The 0- to 0.15-m layer had a mean bulk density of 1.73 Mg m⁻³. The bulk density of this layer was significantly increased to 2.08 Mg m⁻³ in 1999, showing that soil compaction had occurred from pen activity through cattle movement and pen cleaning. Soil bulk density values were similar for the 0.15- to 0.3-m layer between the two sampling dates. Mielke et al. (1974) reported that soils in feedlots are generally compacted to a depth of about 0.15 m.

Soil Chemistry

Chloride content was significantly higher throughout the soil profile in 1999 compared to 1996 (Fig. 3A). The

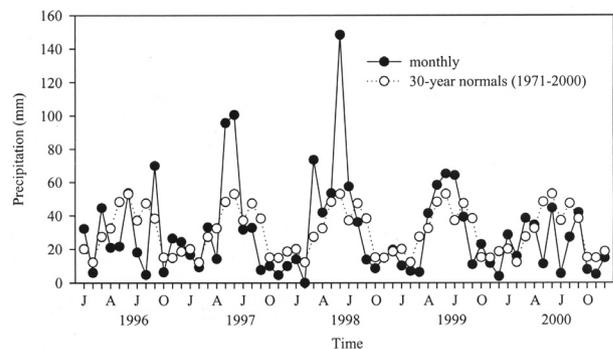


Fig. 2. Monthly precipitation during the study period (1996 to 2000) compared to the 30-year normal values (Environment Canada 2004).

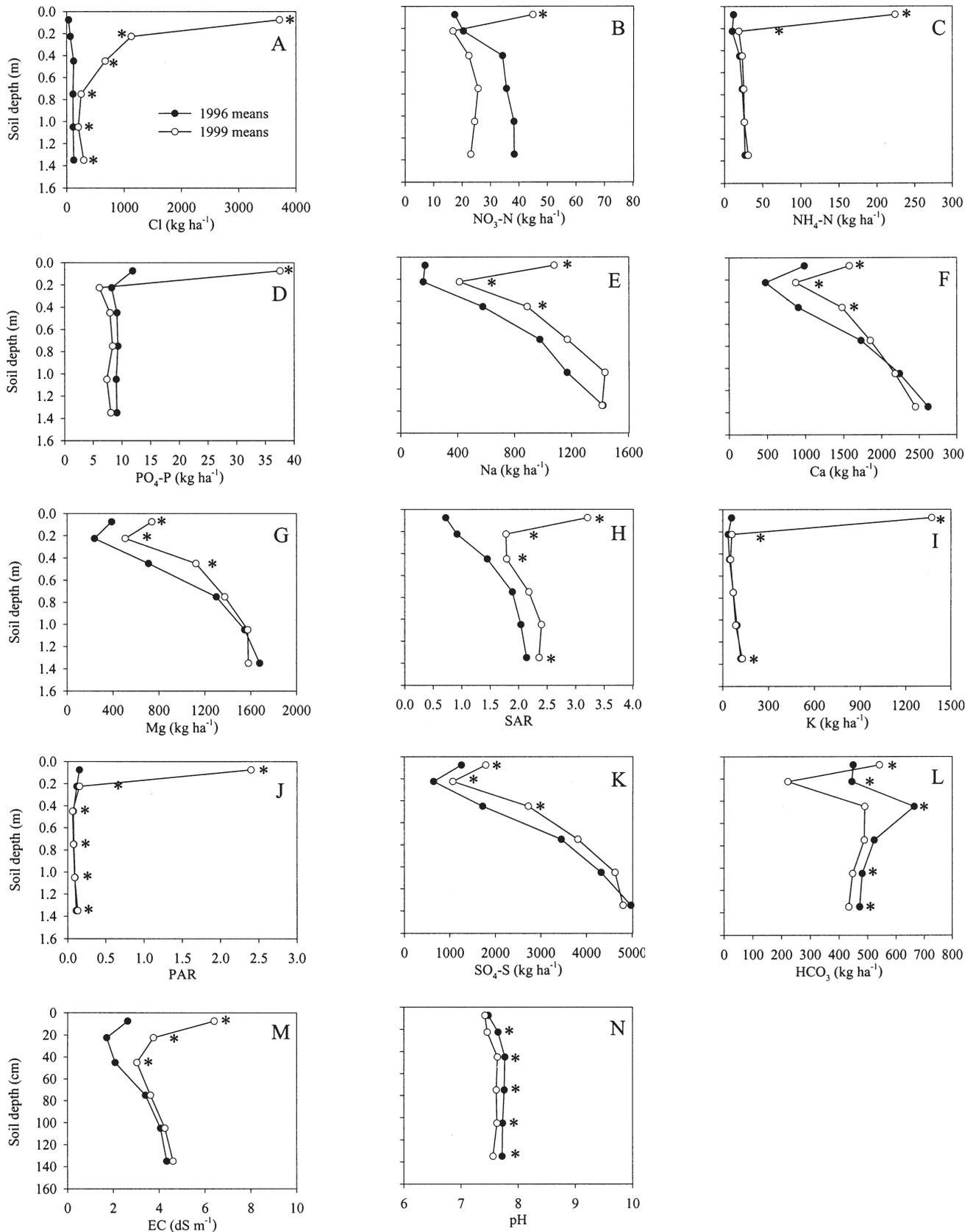


Fig. 3. Mean soil chemistry when the feedlot was constructed (1996) and after three years of operation (1999) for (A) extractable Cl, (B) extractable NO₃-N, (C) extractable NH₄-N, (D) extractable PO₄-P, (E) extractable Na, (F) extractable Ca, (G) extractable Mg, (H) SAR, (I) extractable K, (J) PAR, (K) extractable SO₄-S, (L) extractable HCO₃, (M) EC and (N) pH. Asterisks indicate significant differences (P < 0.05) between means (n = 16) for each soil layer.

largest increase was more than 100-fold in the 0- to 0.15-m layer. The smallest increase was about twofold in the 0.9- to 1.2-m layer. Chloride readily leaches downward with the movement of water and it does not interact chemically or biologically with soil. Since the only source of Cl would have been from the manure, Cl serves as a good indicator of leaching. These results show that Cl moved below the manure pack and compact organic layer and throughout the soil profile within the three-year period. Therefore, the compact organic layer was not completely impermeable to the movement of contaminants from the manure pack into the underlying soil. Kennedy et al. (1999) collected 1-m deep soil cores from five-year-old, two-year-old and newly constructed pens at a feedlot in central Alberta. They also observed significantly elevated Cl content in the 0- to 0.2-m layer in five-year-old and two-year-old pens compared to newly constructed pens. They concluded that some water movement occurred from the pen surface into the soil profile.

Extractable $\text{NO}_3\text{-N}$ increased from 17 kg ha^{-1} in 1996 to 45 kg ha^{-1} in the 0- to 0.15-m soil layer (Fig. 3B). However, $\text{NO}_3\text{-N}$ content in lower depths was less in 1999, though there were no significant differences. The soil profile (0 to 1.5 m) actually contained less $\text{NO}_3\text{-N}$ in 1999 (157 kg ha^{-1}) than in 1996 (184 kg ha^{-1}), representing a 15% decrease during the three-year period. Data for the 0- to 0.15-m layer clearly shows that some $\text{NO}_3\text{-N}$ moved into the soil. Nitrate leaching may have occurred after pen cleaning and prior to the reforming of the compact organic layer where it had been removed. Soil $\text{NO}_3\text{-N}$ concentrations may have been reduced by denitrification.

Kennedy et al. (1999) found no significant difference in the $\text{NO}_3\text{-N}$ gradient under five-year-old and two-year-old pens compared to the newly constructed pens. They proposed that any NO_3 that leached into the soil was lost through denitrification under anaerobic conditions. Saint-Fort et al. (1995) also concluded that denitrification can occur in feedlots. Elliott and McCalla (1972) measured the composition of soil air beneath a feedlot near Lincoln, Nebraska, and found high concentrations of methane (8 to 27.5% average) and carbon dioxide (12.5 to 23% average). They concluded that the feedlot soil profile had a low redox potential, which facilitated methane production and provided favourable conditions for denitrification. They reported that previous work at the same feedlot showed that $\text{NO}_3\text{-N}$ concentrations in the shallow groundwater (0.93 to 1.52 m below the soil surface) were generally below 10 mg L^{-1} (Mielke et al. 1970), and suggested that the reduced conditions prevented NO_3 from reaching the water table. In a field study carried out in Nebraska by Ellis et al. (1975), soil core samples were collected beneath 15 feedlots ranging in age (new to 50 years) and soil texture (clay to coarse sand). They found that continuously

stocked feedlots did not have problems with $\text{NO}_3\text{-N}$ accumulation, whereas soil under abandoned feedlots contained large amounts of $\text{NO}_3\text{-N}$. Maulé and Fonstad (2002) suggested that $\text{NO}_3\text{-N}$ is not always a reliable indicator of solute leaching from manure.

Like Cl, $\text{NO}_3\text{-N}$ is also very soluble and can be readily leached. However, $\text{NO}_3\text{-N}$ did not increase in the soil profile, except in the 0- to 0.15-m layer (Fig. 3B). Some workers have suggested that changes in Cl to $\text{NO}_3\text{-N}$ ratio can be an indicator of denitrification (Kimble et al. 1972; Saint-Fort et al. 1995). However, we caution that the ratio alone may be misleading and the actual distribution of the two anions needs to be examined (Fig. 3A,B). The amount of Cl and $\text{NO}_3\text{-N}$ added to the pen surface also needs to be considered. The Cl concentration in the manure pack was, on average, 300 times greater than the $\text{NO}_3\text{-N}$ concentration (Miller et al. 2003). Even without denitrification, Cl content in the soil would increase more than $\text{NO}_3\text{-N}$ simply because there was more Cl than $\text{NO}_3\text{-N}$ added relative to the levels measured in 1996.

Ammonium-nitrogen was relatively uniform throughout the soil profile in 1996, with a slight increase below the 0.15- to 0.3-m layer (Fig. 3C). There were no changes in 1999, except in the 0- to 0.3-m layer. Ammonium-N was increased by a factor of 19 in the 0- to 0.15-m layer and by a factor of two in the 0.15- to 0.3-m layer. Elliott et al. (1972) also reported that $\text{NH}_4\text{-N}$ was high in soil water at the 0.15-m depth, but declined markedly at greater depths. The movement of $\text{NH}_4\text{-N}$ can be restricted by interactions (sorption) with the soil matrix and microbial reduction (Saint-Fort et al. 1995).

Phosphate-phosphorus followed a similar pattern to $\text{NH}_4\text{-N}$, with a threefold increase in the 0- to 0.15-m layer after three years (Fig. 3D). There were no differences between 1996 and 1999 below 0.15 m. Campbell and Racz (1975) measured P content below a 13-year-old feedlot and found higher P content in the 1.2-m depth compared to adjacent cropland. They concluded that movement of P occurred in organic and inorganic forms. One possible reason why we only observed elevated $\text{PO}_4\text{-P}$ content in the 0- to 0.15-m layer is our site was only three years old in 1999.

Extractable Na, Ca and Mg increased significantly in the upper soil profile (Fig. 3E-G). The largest increase occurred in the 0- to 0.15-m layer, particularly for Na (Fig. 3E). Sodium tended to increase more than Ca and Mg, and this resulted in a significantly higher SAR in the 0- to 0.15-m and 0.15- to 0.3-m layers (Fig. 3H). The increase in SAR may have contributed to soil structure degradation, which may have partly increased soil bulk density in the 0- to 0.15-m layer.

Potassium increased more than 23 times in the 0- to 0.15-m layer (Fig. 3I). There was also a slight increase in the 0.15- to 0.3-m layer. Potassium content remained unchanged below the 0.3-m depth. Unlike many of the

other nutrients, accumulation of K was essentially restricted to the 0- to 0.15-m soil layer. Potassium adsorption ratio (PAR) also increased by over 15-fold in the top soil layer (Fig. 3J). Schuman and McCalla (1975) measured high exchangeable K in the surface of a feedlot soil profile. Potassium was the dominant cation in this layer and they suggested the high K content caused deterioration of soil physical properties.

Sulphate-sulphur followed a pattern similar to Na. Sulphate-sulphur increased throughout most of the soil profile, with significant increases in the top 0.6 m (Fig. 3K). Except for the 0- to 0.15-m soil layer, HCO₃ decreased in the soil profile (Fig. 3L).

At the start of the study, EC decreased from the 0- to 0.15-m layer to the 0.15- to 0.3-m layer, and then increased with depth below 0.3 m (Fig. 3M). Increases with depth of Na, Ca, Mg and SO₄-S reflect the natural higher salt content with depth in the subsoil. Electrical conductivity was significantly greater in the 0- to 0.6-m layer after three years of feedlot operation (Fig. 3M). The 0- to 0.15-m layer increased from a weakly saline condition (2.6 dS m⁻¹) in 1996 to a strongly saline condition (6.4 dS m⁻¹) in 1999.

Soil pH was generally similar throughout the soil profile in 1996 with a mean range of 7.5 to 7.8 (Fig. 3N). Soil pH changed very little after three years of feedlot operation; however, there was a slight but significant decrease in pH below 0.15 m.

Soil chemistry data showed that constituents from the manure pack leached through the compact organic layer and into the underlying soil. The change in accumulation and distribution in the soil profile varied among the measured parameters and could be divided into five groups: (1) Cl; (2) NO₃-N, HCO₃; (3) NH₄-N, K, PO₄-P; (4) Mg, Ca, Na, SO₄-S, EC; and (5) pH. The groups reflect how the various constituents interacted with the subsoil. In Group 1, Cl accumulated in the top soil layer and readily leached throughout the soil profile, whereas the constituents in Group 2 accumulated in the top soil layer and decreased below the 0.15-m depth. The constituents in Group 3 remained essentially in the top soil layer and the constituents in Group 4 leached only part way down the profile. In Group 5, pH decreased throughout the soil profile.

Groundwater Elevation

Water-table elevations had stabilized in all wells by early June 1996, within one month of installation. The average water-table depth below the soil surface for the site during the 4.5-year period ranged from 1.23 m (June 12, 1997) to 2.50 m (February 17, 1998). The minimum depth measured in a single well was 0.14 m for Well 13 on June 12, 1997. The maximum water-table depth in a single well was 4.77 m for Well 12 on May 8, 1996. A maximum fluctuation of 4.08 m was observed in Well 12. The least

fluctuation of 1.50 m occurred in Well 6. There was a clear annual fluctuation of water-table elevation, with the lowest elevation early in the year, and the maximum elevation in mid-May to mid-July (Fig. 4). Fluctuation of the water-table elevation closely matched the daily precipitation pattern, suggesting a direct relationship between precipitation and groundwater recharge.

After the water-table levels had stabilized by early June 1996, it was apparent the water table sloped downward from south to north (Fig. 5A). Most of the slope occurred in the south half of the study area, and the north half was relatively flat. By July 4, 1996, the water table was generally flat. A water-table mound began to develop by November 14, 1996, and became more distinctive throughout 1997. The water-table mound was located directly below the feedlot pens, reflecting increased recharge to groundwater below the pen area (Fig. 5B). The water-table mound persisted from 1998 to the last monitoring date in November 2000. It is probable that increased recharge in the pen area resulted from cattle urination and the lack of transpiration through plants. The response of the water table under the pen area indicates the compact organic layer was not effective in reducing the overall recharge rate below the pens.

Groundwater Chemistry

Baseline period. Groundwater chemistry, particularly NO₃-N, Cl, Na and SAR, varied widely among wells during the baseline period (May 22, 1996, to August 21, 1996) (Table 1). For example, Well 1 consistently had low NO₃-N concentrations, with a range of 0.0 to 1.5 mg L⁻¹, whereas Well 2 had higher concentrations, with a range of 152 to 185 mg L⁻¹. Nitrate-nitrogen was present at no more than 5 mg L⁻¹ in wells with more than 1 mg L⁻¹ Fe and 1 mg L⁻¹ Mn in the groundwater, consistent with the fact that NO₃ is not stable in an iron-reducing environment (Korom 1992). Nitrate is completely denitrified below the redoxcline in fine-textured and coarse-textured sediments (Robertson et al. 1996; Rodvang and Simpkins 2001). The redoxcline is the boundary between oxidized and reduced conditions (Postma et al. 1991), and can be identified by changes in

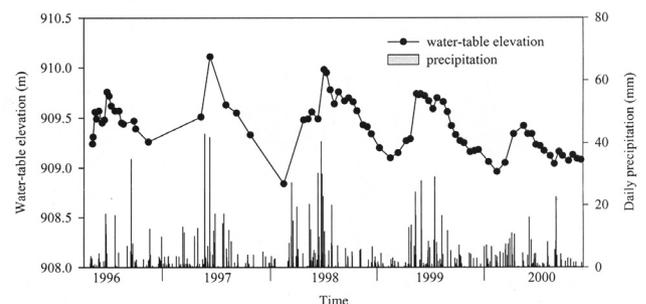


Fig. 4. Mean site water-table elevation and daily precipitation.

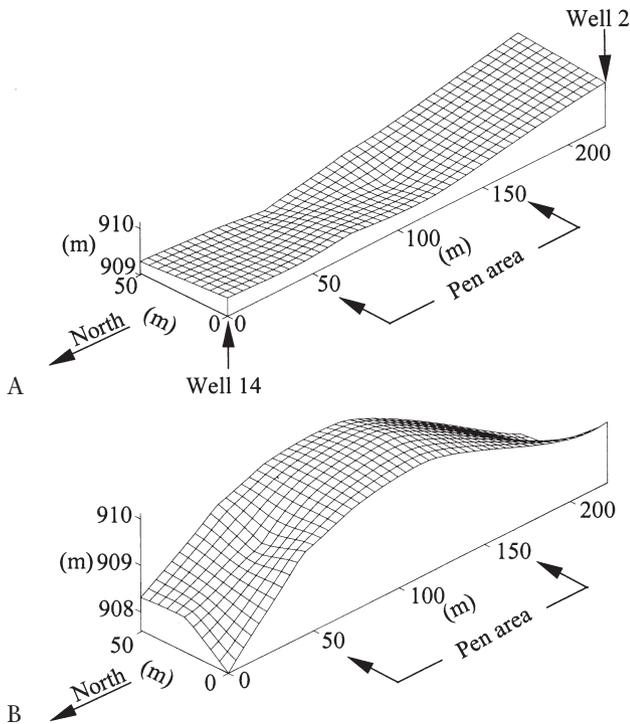


Fig. 5. Water-table elevation on (A) June 10, 1996, and on (B) November 27, 2000.

sediment colour and/or groundwater chemistry (Rodvang and Simpkins 2001). The redoxcline was not noted during well installation; however, low $\text{NO}_3\text{-N}$ (less than 5 mg L^{-1}) and elevated reduced Fe and Mn (greater than 1 mg L^{-1}) in Wells 1, 10, 12 and 14 suggest this groundwater was reduced. Hendry et al. (1984) and Rodvang et

al. (1998) showed that $\text{NO}_3\text{-N}$ occurs naturally in groundwater in oxidized till at many locations in southern Alberta, and that wide variations in concentration are related to redox conditions and groundwater flow. Sodium, $\text{SO}_4\text{-S}$ and Mg also occur naturally at high concentrations in oxidized till in southern Alberta (Rodvang et al. 1998; Rodvang and Simpkins 2001).

Most measured parameters did not show a consistent spatial pattern during the baseline period, except Cl and $\text{NO}_3\text{-N}$ concentrations, which tended to decrease from south to north within the study area. There were no obvious temporal trends during the three-month period; however, there were slight trends with time of decreased Na concentration and SAR, and a slight decrease in HCO_3 concentration. A consistent increase in Mn concentration was observed, with a mean value of 0.52 mg L^{-1} on May 22, 1996, increasing to 1.91 mg L^{-1} by August 22, 1996. Iron also increased with time, but remained consistently low (0.45 to 0.81 mg L^{-1}) during the first half of the three-month period and then increased (1.47 to 1.96 mg L^{-1}) after the July 4, 1996, sampling date.

Many of the parameters measured during the baseline period exceeded water quality guidelines, and included $\text{NO}_3\text{-N}$, Na, $\text{SO}_4\text{-S}$, SAR, Mn, Fe and EC (Table 1).

Post-baseline period. Data for each parameter were grouped according to three locations at the site (Fig. 1): the south wells ($n = 4$), the pen wells ($n = 9$) and the north wells ($n = 3$). If the feedlot affected groundwater chemistry, then any effect would most likely appear in some or all of the nine wells within the pen area before potentially appearing in the seven wells outside of the pen area.

TABLE 1. Descriptive statistics of groundwater chemical parameters measured during the baseline phase (May 22, 1996, to August 21, 1996)

Parameter	Units	Mean	Min.	Max.	Standard deviation	CV	No. of samples	Guideline limits
$\text{NO}_3\text{-N}$	mg L^{-1}	41.9	0	185	46	109	176	10^a
Cl	mg L^{-1}	97.9	7.1	291	79	80	176	250^a
Ca	mg L^{-1}	429	289	515	28	6.5	176	1000^b
Mg	mg L^{-1}	535	300	930	164	31	176	
Na	mg L^{-1}	1080	166	1791	381	35	176	200^a
K	mg L^{-1}	15.4	7.4	22.7	3.4	22	176	
$\text{SO}_4\text{-S}$	mg L^{-1}	1599	826	2411	362	23	176	$500^{a,c}$
HCO_3	mg L^{-1}	707	336	1159	145	21	176	
CO_3	mg L^{-1}	0.2	0	12.3	1.4	688	176	
SAR		8.2	1.4	12.7	2.8	34	176	4^d
Mn	mg L^{-1}	1.1	0	9	2.2	189	160	0.05^a
Fe	mg L^{-1}	1.1	0	14.6	2.9	256	160	0.3^e
Cu	mg L^{-1}	0.01	0	0	0.01	87	160	1.0^a
Zn	mg L^{-1}	0.01	0	0.21	0.02	180	160	5.0^e
pH		8.0	7.2	8.3	0.2	2.6	176	$6.5\text{--}8.5^a$
EC	dS m^{-1}	7.6	4.3	10.8	1.5	20	176	1.0^d

^aCanadian water quality guidelines (CCME 1987; FPCEOH 1996) for drinking water.

^bCanadian water quality guidelines (CCME 1987) for livestock watering.

^cAs sulphate (CCME 1987).

^dCanadian water quality guidelines (CCME 1987) for irrigation.

^eCanadian water quality guidelines (FPCEOH 1996) for aesthetic objective.

Chloride concentration in groundwater increased with time in the pen wells (Fig. 6A). There was essentially no change in the south and north wells. This shows that the north wells were not influenced by the catch basin. An annual cycle of Cl concentration was observed in the pen wells. Periods of maximum mean concentrations occurred in June 1997, June to July 1998, August 1999 and October 2000, coinciding with annual precipitation events (Fig. 6H). The difference between the pen wells and the seven wells outside of the pen area is evidence that Cl was leached to groundwater from the manure pack in the feedlot. This supports the soil data, which showed that Cl had leached through the soil profile. Because Cl is water soluble and does not interact with the soil matrix, either chemically or biologically, it serves as a good tracer for water movement through the soil profile.

Of the parameters that were measured, Cl showed the strongest evidence that water from the feedlot had moved through the soil and into shallow groundwater. Even though there was a general increase in groundwater Cl concentration in the pen wells, there was a wide range of variability among these wells. Some wells were not affected, whereas large increases in Cl concentration were observed in other wells. The nine pen wells were located adjacent to the pens and not in the pens (Fig. 1). Three wells were located in the feeding alley, and the other six wells were located in the two handling/drainage alleys. Most of the increases in Cl concentration occurred in the wells in the drainage alleys (Table 2). The three wells in the feeding alley either had no increase (Wells 6 and 9) or a modest increase (Well 12) in Cl concentration. The drainage alleys consisted of a gravel layer overlaying compacted subsoil, and they lacked the compact organic layer, which was observed in the pens. The compact organic layer that forms below the manure pack is believed to restrict the downward movement of water and solutes (Mielke and Mazurak 1976). The individual pens were sloped towards the drainage alleys and the movement of animals to and from the pens occurred in these alleys. As a result, the drainage alleys were often wet. In contrast, the feeding alley was generally dry and any surface water drained into the catch basin. Perhaps the drainage alleys allowed more water infiltration and leaching than other parts of the feedlot.

The site mean $\text{NO}_3\text{-N}$ concentration ($n = 16$) was highest during the three-month baseline period. Subsequent measurements showed mean $\text{NO}_3\text{-N}$ concentrations were lower, but consistently remained about 29 mg L^{-1} . Mean $\text{NO}_3\text{-N}$ concentration was generally highest in the south wells and lowest in the north wells (Fig. 6B). For the whole site ($n = 16$) there was a slight, but steady increase in $\text{NO}_3\text{-N}$ concentration from October 1998 to May 1999, followed by a decrease for about two months, after which $\text{NO}_3\text{-N}$ concentration slowly increased after July 1999 until the end of the study. The increase from October 1998 to May 1999 was mainly attributed to the

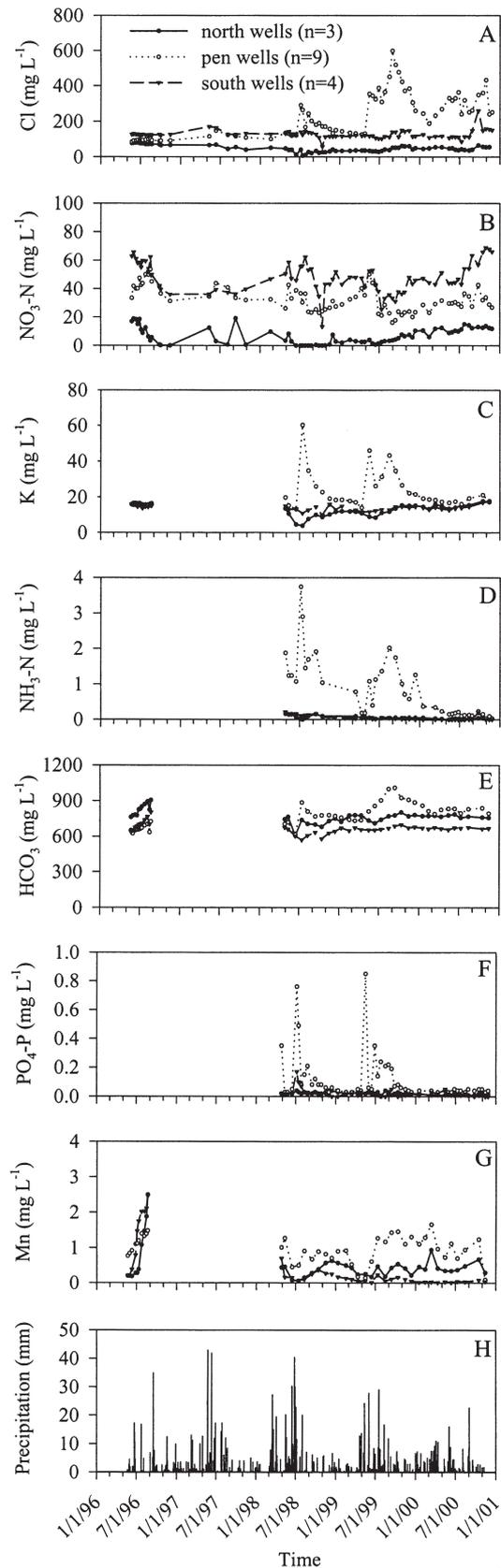


Fig. 6. Mean content of (A) Cl, (B) $\text{NO}_3\text{-N}$, (C) K, (D) $\text{NH}_3\text{-N}$, (E) HCO_3 , (F) $\text{PO}_4\text{-P}$ and (G) Mn in groundwater from the south wells, north wells and pen wells compared to (H) daily precipitation.

TABLE 2. Mean chloride content in groundwater from wells within the feedlot pen area

Well number	Mean (standard error) ^a for May 1996 to June 1998 (mg L ⁻¹)	Mean (standard error) ^b for July 1998 to November 2000 (mg L ⁻¹)
Drainage alley wells		
Well 5	159 (13)	511 (28)
Well 7	183 (8)	236 (20)
Well 8	71 (5)	534 (43)
Well 10	26 (1)	345 (44)
Well 11	54 (3)	134 (16)
Well 13	152 (20)	390 (26)
Feeding alley wells		
Well 6	231 (9)	234 (7)
Well 9	38 (1)	31 (1)
Well 12	28 (1)	126 (15)

^an = 24.^bn = 49.

wells within the pen area, whereas an increase was observed for all three locations after July 1999 (Fig. 6B).

As previously stated, the NO₃-N concentration was well above the Canadian drinking water quality guideline of 10 mg L⁻¹ at the time the feedlot was constructed (Table 1). Prior to construction of the feedlot, the pen area and south (i.e., south wells) were in crop/forage production and may have received applications of fertilizer and irrigation water. The area may have also been used to graze livestock. The area where the north wells were located was not used for any particular purpose. The higher NO₃-N in the groundwater under the pens and in the south wells, particularly in Wells 2, 6 and 8, may be the result of previous land-use activities or natural sources. The presence of the feedlot generally did not affect NO₃-N concentration in the groundwater. Some researchers have concluded that feedlots do not contribute significant amounts of NO₃, if any, to groundwater (Elliott et al. 1972; Lorimor et al. 1972; Schuman and McCalla 1975) due to reduced infiltration and denitrification. However, Elliott et al. (1972) collected soil water samples and Schuman and McCalla (1975) collected soil samples, and neither actually sampled groundwater beneath feedlots. Other researchers have reported groundwater contamination by NO₃ from feedlots (Coote and Hore 1979). The clear effects on Cl, combined with the lack of effects on NO₃-N, suggests denitrification may have prevented an increase in NO₃-N concentration in the groundwater.

Potassium concentrations were similar between the north and south wells, with a slight increase with time (Fig. 6C). Ammonia-N concentrations were very low in the north and south wells, with very little change with time (Fig. 6D). Potassium and NH₃-N concentrations increased in the pen wells from April 1998 and remained high until early 2000. These increases occurred mainly in Well 5, and to a lesser extent in Wells 7, 8, 10, 12 and 13. The increase in concentration for these two cations

corresponded closely with the Cl changes in 1998 and 1999. After March 2000, K and NH₃-N concentrations in pen wells were only slightly higher than the concentrations in the north and south wells (Fig. 6C,D). An increase in NH₃-N concentration, particularly in conjunction with an increase in Cl concentration, provides further evidence of NO₃ reduction.

Phosphate-phosphorus was not measured during the 1996 baseline period. Analysis of groundwater from April 1998 to November 2000 showed no major fluctuations in the north and south wells (Fig. 6F). Phosphate-phosphorus concentration in these wells was usually less than the suggested aquatic life water quality guideline used in Alberta of 0.05 mg L⁻¹ for total P (Alberta Environment 1999). There were several instances in 1998 and 1999 when the PO₄-P concentration increased in pen wells. These increases occurred in Wells 5, 7 and 13. The largest concentration measured was 7.2 mg L⁻¹ in Well 5 on May 17, 1999.

Soil analysis showed that NH₄-N, K and PO₄-P accumulated in the 0- to 0.15-m soil layer, with no accumulation below 0.15 m, suggesting no leaching of these three constituents. However, the groundwater data clearly show increased levels of NH₄-N, K and PO₄-P concentrations in the pen wells. The discrepancy between the soil and groundwater data may be explained by possible macropore flow, where constituents can be moved by water through macropores to shallow groundwater with minimal interaction with the soil matrix.

Bicarbonate concentration increased with time during the three-month baseline sampling phase, and this increase was consistent among the three well locations (Fig. 6E). After HCO₃ analysis was resumed in 1998, increased HCO₃ concentrations were not observed. On average, the pen wells contained the highest HCO₃ concentration, and the south wells contained the lowest concentrations after April 1998. The higher levels of HCO₃ concentration in the pen well may be due to denitrification.

Manganese concentration was generally greater in the pen wells than in the south and north wells in 1999 and 2000 (Fig. 6G). The increase in manganese was attributed to Well 5.

The feedlot pens did not have any noticeable influence on groundwater EC, Ca, Mg, SO₄-S, Na, Fe, Cu and Zn concentrations (data not shown).

Groundwater Microbiology. *Escherichia coli*, total coliforms and aerobic heterotrophs were detected in the groundwater from 1998 to 2000. Values for *E. coli* and total coliforms varied annually (Fig. 7A,B); and during the winter months, values were below the detection limit of 1.56 log MPN 100 mL⁻¹. Substantial increases in *E. coli* and total coliforms values occurred from July to October, 1998, and from April to October, 1999. Increases in 2000 were not as large, with a modest increase in April of *E. coli* and total coliforms followed by larger increases from August to November, peaking at 3.39 log MPN 100 mL⁻¹ for *E. coli* and 3.87 log MPN 100 mL⁻¹ for total coliforms. Aerobic heterotroph populations remained relatively stable throughout the period, ranging from 5.54 to 7.86 log CFU 100 mL⁻¹ (Fig. 7C). However, the higher values corresponded to the peaks observed for *E. coli* and total coliforms.

Figure 7 clearly shows that essentially all of the increases in the *E. coli* and total coliform populations occurred within the pen area. The pen area had little influence on aerobic heterotrophs (Fig. 7C). The short-term increase that was observed in April 2000 for *E. coli* was not within the pen area, but rather in the south wells, and this can be mainly attributed to Well 3 and to a lesser extent Well 4. Of the nine wells within the pen area, only Well 9 did not show the same increase as the other wells. Well 11 also contained lower populations, but not to the same extent as Well 9. The mean values of *E. coli* and total coliforms in the other seven wells (Wells 5–8, 10, 12–13) were 11- to 13-fold greater than in Well 9, and eightfold greater than in Well 11. Aerobic heterotrophs were, on average, fivefold and twofold greater in the seven wells compared to Wells 9 and 11, respectively.

The annual fluctuations of *E. coli* and total coliforms in the nine pen wells closely followed the annual precipitation patterns (Fig. 7). The amount of precipitation in 1998 was 32% above the long-term average, with June being particularly wet. In 1999, precipitation was about 6% below the long-term average, and the higher amounts of precipitation were spread over a longer time compared to 1998. Lower microbial populations in the groundwater in 2000 reflect the precipitation in 2000, which was 25% below the long-term average. Cho et al. (2000) reported that summer precipitation events were a major factor accelerating groundwater contamination with bacteria from livestock waste disposal.

Livestock manure is a source of microorganisms, including pathogens, and these microorganisms can

move vertically through soil (Mawdsley et al. 1995; Betcher et al. 1996). Betcher et al. (1996) reviewed the findings of a 1991 farm drinking water well survey in Ontario reported by Rudolph and Goss (1993). Twenty to forty-four percent of the wells tested for bacteria exceeded the recommended maximum concentrations. The major point sources for bacteria on farms were septic tanks, manure storage facilities and feedlots. Coliform bacteria contamination of groundwater decreased as the separation distance between wells and feedlots increased. Other work has shown that bacteria can readily move downward through soil. Fleming and Bradshaw (1991) applied liquid hog manure to undisturbed soil cores (0.15-m diameter by 0.6-m long) and found that a little more than 2% of the bacteria load in the manure was in the effluent that drained from the cores. The amount of bacteria in the effluent from the soil cores that received manure was 60 to 99 times greater than observed from the control. Stoddard et al. (1998) reported a significant increase of fecal bacteria in leachate

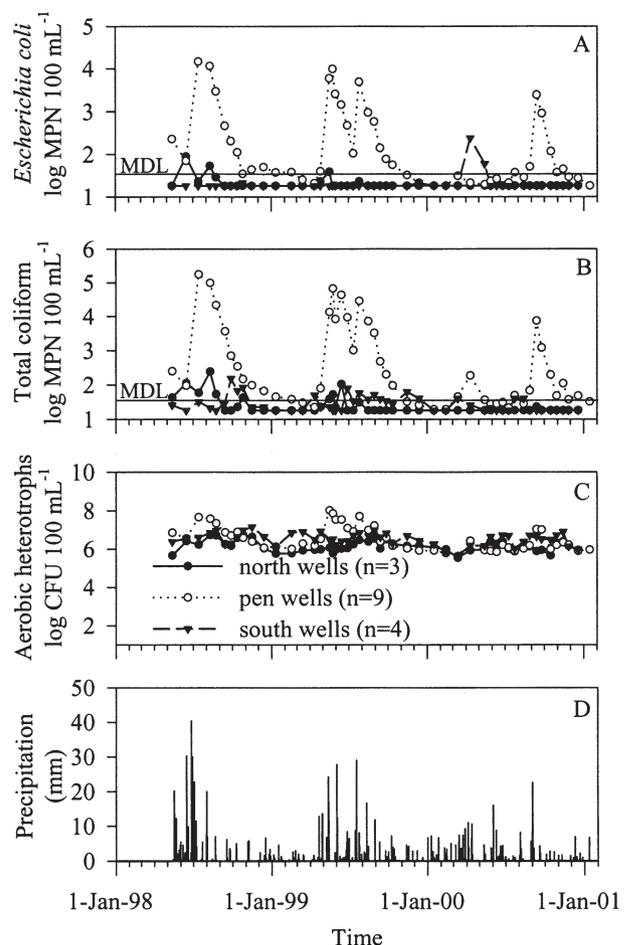


Fig. 7. Mean content of (A) *Escherichia coli*, (B) total coliforms and (C) aerobic heterotrophs in groundwater from the south wells, north wells and pen wells compared to (D) daily precipitation. Minimum detection limit (MDL) was 1.56 log MPN 100 mL⁻¹ as shown by the horizontal line.

at the 0.9-m depth after dairy manure was applied to soil, and this persisted for 60 days. Fecal bacteria moved beyond the root zone when sufficient rainfall occurred. They pointed out that groundwater contamination depends on soil structure and water flow more than on fecal bacteria survival at the soil surface. McMurry et al. (1998) showed that fecal coliforms moved with preferential water flow through excavated soil blocks that were treated with poultry manure. Krapac et al. (2002) found groundwater contaminated with bacteria from swine manure pits, and they questioned the common assumption that soil effectively filters bacteria.

Clearly, from the literature and our study, microorganisms can move from a manure source downward through soil. The groundwater Cl data provide evidence that the drainage alleys may be an important area where water can infiltrate and move downwards. The six wells located in the drainage alleys contained more Cl than the three wells in the feeding alley. Soil data from beneath the pens showed that Cl leached to the 1.5-m depth; however, the distinction between the drainage-alley wells and the feeding-alley wells was not as clear for the microbiology data.

Conclusions

A four-year monitoring study found that shallow groundwater was adversely affected by a new cattle feedlot and that soil chemistry beneath the feedlot pens was changed. The elevation of the water table changed under the feedlot. Water-table levels averaged 1.23 to 2.50 m below the soil surface. The water table began to mound below the feedlot pens beginning in late 1997, and the mound persisted through to the end of the study. It is probable that increased recharge in the pen area resulted from cattle urination and the lack of transpiration through plants. The response of the water table under the pen area indicates the compact organic layer was not effective in reducing the overall recharge rate below the pens. The top soil layer (0 to 0.15 m) became more compacted from the action of cattle hoofs under wet conditions.

Some soil chemical properties increased in the top layer, namely extractable PO₄-P, extractable NO₃-N, extractable NH₄-N and extractable K. These increases may simply be the result of mechanical mixing by cattle. Other soil properties increased significantly to greater depths (down to 0.6 m), such as EC, SO₄-S, extractable Mg, extractable Ca, extractable Na and SAR. Chloride content increased significantly through the entire soil profile (0 to 1.5 m), providing evidence that leaching occurred under the feedlot pens. There was evidence of potential macropore flow, where some constituents, such as NH₄-N, K and PO₄-P, did not accumulate in the subsoil layer but increased in the shallow groundwater beneath the pens. The compact organic layer that forms under the manure pack is generally thought to prevent the

downward movement of water and other material. However, we observed that the compact layer was sometimes removed from certain pens during annual pen cleaning.

Groundwater analysis also indicated that contaminants had leached to the water table, particularly as indicated by increases in Cl concentrations, *E. coli* counts and total coliform counts in the wells within the pen area. Potassium, PO₄-P, NH₃-N and Mn concentrations also increased beneath the pens. Drainage alleys, which did not have a compact organic layer, may have contributed to leaching of contaminants through the soil profile.

Our results show that shallow groundwater under a feedlot is vulnerable to contamination. Construction criteria, maintenance of the compact organic layer within pens, and management of non-pen areas, such as alleyways, may be critical factors to prevent or minimize groundwater contamination from feedlots.

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