# DOWNWARD MOVEMENT OF NITRATE AND PHOSPHORUS FROM HOG MANURES IN ANNUAL AND PERENNIAL CROPPING SYSTEMS

By

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# ABSTRACT

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Excess nitrate-N concentration (>10 mg $L^{-1}$) in drinking water can cause significant risk to human health. Also, at very low concentration (0.035-0.1 mg P $L^{-1}$), phosphorus is considered as a pollutant due to its effects of promoting algal growth and eutrophication of surface waters. This thesis' research was conducted at two different sites. The first study was conducted at Carman on a sandy loam soil with cropping system, perennial versus annual, as the main plot and manure nutrient management system, as the subplot to measure nitrate and phosphorus leaching from hog manures. The second field experiment, located northwest of the town of Carberry, Manitoba, was conducted on a loamy sand soil. A two year rotation was employed for the annual cropping systems with a randomized complete block design. Treatments included two rates of liquid hog manure, two rates of fertilizers corresponding to the amount of available nitrogen in the two rates of hog manure, a compost treatment and a control for a total of six treatments. The results from Carman site showed that while a substantial amount of nitrate-nitrogen was lost from the annual plots (40 to 60 kg $ha^{-1}$ in 2010 and 23 to 60 kg $ha^{-1}$ in 2011), a negligible amounts of nitrate was lost from the perennial (< 1 kg $ha^{-1}$). There was no evidence of significant downward movement of phosphorus below the top 15 cm soil layer in this study. However, repeated, annual application of manure at an N-based rate resulted in increased soil test P. In Carberry, total N leaching of fertilizer amended plots was greater than in plots that received manure. Based on the results, application of liquid hog manure at the rate of 2500 gallon $ac^{-1}$ was economically and environmentally more desirable and is recommended. We applied the multi-layer water balance model, VSMB, to the data that we generated in the field to gain an understanding of how well the model will simulate the loss of water that we measured from the lysimeters. The simulation study showed that the VSMB model grossly underestimated the amount of leached water, possibly due to an overestimation of evapotranspiration.

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# 1. REVIEW OF LITERATURE

## 1.1 INTRODUCTION

Leaching is a complex process, which often occurs on a time scale of decades or more periods. It is often ignored or simplified as the vertical movement of dissolved salts, compounds and nutrients past a given soil depth or the delivery of them to surface water through subsurface flow by means of interflow and tile drainage (Radcliffe and Cabrera 2007).

Through the process of leaching, a portion of plant nutrients is lost from the crop root zone, temporarily or permanently, which is economically undesirable. In addition, leached nutrients can reach ground waters and cause the contamination of these natural resources (Hillel 1998). Increasing population, especially during the past half century has increased demand for food and this has led to the use of increased levels agricultural inputs, including nitrogen fertilizer and manures. Mineralization of organic compounds can release a notable amount of nutrients but this amount cannot meet plant requirements, and applying fertilizer and manure is necessary to augment soil nutrient supply.

Nitrogen is a key essential element used in large quantity to produce greater crop yields. For its chemical properties, nitrate is readily leached from soils in agricultural and grassland areas since it has a negative charge and cannot be sorbed by negatively charged surfaces of soil colloids (Janzen et al. 2003). Depending on soil condition such as aeration, water content, pH and organic matter, some parts of soil nitrogen can be lost through leaching (Abril 2007). It has been reported that “nearly 66% of applied N in fertilizers is lost during irrigation events” in conventional agriculture (Beman et al. 2005).

Nitrate concentrations above 10 ppm in drinking water creates alarm for nitrate contamination and results in problems for young animals and human babies (Basso and Ritchie 2005). Public concerns about nitrate-N in drinking water arose from two serious medical conditions: methemoglobinaemia (blue-baby syndrome) in infants, and cancer in adults. Blue-baby syndrome occurs in infants with ages less than 1 year. In affected infants, stomach microbes convert nitrate to nitrite which then reacts with haemoglobin in the blood-stream to form methaemoglobin and decreases the capacity of blood haemoglobin to form oxyhaemoglobin with oxygen. Oxyhaemoglobin contains Fe (II) whereas methaemoglobin contains Fe (III) and this lessens the capacity of blood to carry oxygen (Basso and Ritchie 2005; Knobeloch et al. 2000).

Phosphorus is one of the most important macronutrients for crop production and is required for photosynthesis, respiration, seed production and root growth. Traditionally, most data on P movement in soil have focused on soil analysis of extractable P as a function of depth, which has led to the general assumption that no substantial vertical P movement or leaching loss occur because of the high P-fixation capacity of many mineral soils (Brye et al. 2002).

Significant P leaching can occur where certain combinations of land-use practices (i.e., overfertilization and/or excessive manure application), soil properties (i.e., sandy subsoil, high organic matter, and the presence of preferential flow paths), and climatic conditions (i.e., precipitation > evapotranspiration) exist. Recent field studies have indicated that soil solution concentrations and subsurface P leaching losses are larger than once thought. Repeated application of P via commercial fertilizers, organic wastes, or both may saturate the P adsorption capacity of those soils, thus altering the chemical equilibrium established by adsorption–desorption processes. This will lead to higher concentrations of P in solution and greater potential for P export by subsurface flow paths, which are hydrologically connected to both surface and ground water. Therefore, accurately quantifying nutrient leaching is extremely important (Brye et al. 2002). As an example of P leaching, Nelson et al. (2005) monitored P leaching in two different pastures with loamy soil which had received liquid hog manure for more than 20 years in Sampson County, NC. The results showed that the maximum soil solution P concentrations at 45 cm depth exceeded 18 mg $L^{-1}$ in both soils. They concluded that long term application of hog manure resulted in an increase of soil solution P concentration and considerable vertical P movement.

Phosphorus is considered a pollutant in surface water because of its potential to cause eutrophication. At very low concentrations (0.035-0.1 mg $L^{-1}$), P can cause excessive plant growth and algal blooms. When the plants and algae die, depletion of oxygen in water can cause odor and kill fish and other aquatic organisms (CCME, 2004). “In Manitoba, the total P concentration threshold in freshwater, above which eutrophication may be enhanced, is set as 0.025 mg $L^{-1}$ total P at the entry points of lakes, ponds and reservoirs and 0.05 mg $L^{-1}$ total P for streams and rivers” (Flaten et al. 2007).

Lake Winnipeg in the province of Manitoba is the third-largest freshwater lake in Canada. Nutrients loads and therefore eutrophication in Lake Winnipeg has increased in the last thirty years (Lake Winnipeg Stewardship Board, 2006). The major nutrient inputs, 54 % of total P and 30 % of total N come from the Red River (Lake Winnipeg Stewardship Board, 2006). In addition to the Red River, other tributaries from the United States, Saskatchewan, Alberta and Ontario contribute N and P loads to Lake Winnipeg. It is believed that Manitoba agriculture, including the hog industry in Manitoba, contributes 11% of N and 32% of P to the annual N and P loads coming from Manitoba into Lake Winnipeg (Flaten et al. 2007).

## 1.2 PROCESSES OF NUTRIENT LEACHING

Water is enriched during movement in the soil profile by dissolving different substances and ions including fertilizers, pesticides, salts and products of various chemical reactions (Lal and Shukla 2004).

Solutes can be broadly classified into two categories depending on chemical stability and reactivity: I) conservative solutes, which remain unchanged physically and chemically, and do not undergo any irreversible reaction. Examples of these are chloride ($Cl^-$) and bromide ($Br^-$). II) non-conservative solutes, which undergo irreversible reactions and change their physical or chemical state; this group is further divided to two groups of labile solutes and reactive solutes; nitrate is an example of labile solutes which is involved in mineralization, immobilization, or redox reactions (Lal and Shukla 2004).

Distribution of soil aggregates and pores with different sizes causes different velocity of solutes in larger and smaller pores (Hillel 1998). Basically, solute transport within a soil matrix occurs by three physical processes: advective, diffusive, and dispersive transport.

### 1.2.1 Advective Transport of Solutes

In large pores solutes migrate via water movement. This process is referred as "advection". In fact, advective transport of a solution inside a soil matrix, sometimes called the Darcian flow or mass flow, is passive movement with flowing soil water (Hillel 1998).

Advective transport ($J_m$) can be expressed as:

$$J_m = q_s C = -C\,(K\,\nabla H)$$

Where $J_m$ shows the flux density for convective or mass transport ($ML^{-2}T^{-1}$); $q_s$ is the volumetric fluid flux density with the same dimensions of velocity ($LT^{-1}$) and is defined as the volume of liquid flowing through a unit area per unit time; and C refers the mass of solute per unit volume of solution ($ML^{-3}$).

$q_s = -K\,\nabla H$ is Darcy's equation where K is the soil hydraulic conductivity and $\nabla H$ is the three-dimensional hydraulic gradient which is the change in pressure head (H) with distance.

### 1.2.2 Diffusion of Solutes

The random thermal motion of dissolved ions and molecules is responsible for diffusion. Contrary to advection, diffusion is an active process and results in decreasing concentration gradients and conducts the process towards homogeneity. Diffusion can be well defined by Fick's law. Fick's law for one-dimensional steady state transport follows as:

$$J_d = -Ds\,\partial C/\partial X$$

$$Ds = D_0\,\theta\,\xi$$

$J_d$: Solute flux density for diffusive transport of solute ($ML^{-2}T^{-1}$)

$\theta$: The volumetric moisture content of soil ($L^3L^{-3}$)

$D_s$: Diffusion coefficient in soils

$D_0$: Diffusion coefficient in pure water

$\xi$: soil tortuosity

$D_s$ is slightly less than $D_0$ for tortuous path of water in soil matrix

In smaller pores, diffusion is the main process of solute transport.

### 1.2.3 Hydrodynamic Dispersion

The velocity of water in pores of different characteristics such as shapes, sizes, and orientation varies. Based on Poiseuille's law assuming pores as capillary tubes total flow rate in each individual pore varies in proportion to the fourth power of the radius (R) of the pore. However, flow velocity in the tube decreases by increasing distance of each point from the center of the tube. These variations in water velocity account for dispersive transport of solute (Lal and Shukla, 2004). Dispersive transport can be described by an equation similar to diffusion as follows:

$J_h = -\theta\ D_h\ \partial C/\partial X$

$D_h$: mechanical dispersion coefficient assumed as a function of fluid velocity:

$D_h = \lambda V$

$\lambda$: dispersivity

Dispersive and diffusive transports are macroscopically similar and both tend to mix and eventually eliminate non-uniformity in solute. However, diffusion is an active process, relating to concentration gradients in spite of flow while dispersion is a passive process relating to fluid flow (Lal and Shukla 2004).

The three processes of solute movement can be combined into the advection-dispersion equation (ADE) usually written in the form of:

$$J = \nu\theta c - D_{sh}(\theta, \nu)\ \partial C/\partial X$$

"Here, J is the total mass of a solute transported across a unit cross-sectional area of soil per unit time, $D_{sh}$ is the lumped diffusion-dispersion coefficient (a function of volumetric wetness Θ and average pore-water velocity v), C is the solute concentration, and $\partial C/\partial X$ is the solute gradient" (Hillel 1998).

### 1.2.4 Preferential Flow Paths

Preferential flow is the transport pathway by which relatively large amounts of water flow through a small portion of the soil volume. Therefore, water can infiltrate downward much faster than what is predicted by homogeneous flow with plane wetting fronts (Selker et al. 1996).

Bypass flow is a type of preferential flow that refers to the entry of rain or irrigation water into macropores such as continuous vertical cracks or worm channels resulting in water loss for crops and redistribution of soil materials and plant nutrients. Factors such as clay mineralogy, clay content, fluctuation in soil moisture, adsorbed cations, rainfall distribution, natural vegetation and land use can determine the magnitude of cracking. Contrary to non-cracked soils, Darcy-Richards type flow theory is not applicable for cracked soils. Vertical flow of water into cracks occurs only when rainfall intensity outweighs the vertical infiltration rate into and through the soil aggregates (Kosmas et al. 1991). When cracks are vertically continuous, a notable amount of water may penetrate deep into the subsoil.

Fingering flow is the second type of preferential flow as a result of wetting front instability because of textural discontinuity. When fine-textured soils overlie coarse-textured soils, the wetting front during infiltration does not uniformly penetrate the coarse layer. At first, water cannot penetrate the coarse layer due to the low wetting front pressure head created in the fine-textured layer. When pressure head increases at the interface, water starts to enter the smallest pores in the coarse layer and then pressure head reaches a value where the flux through the coarse layer is equal to or greater than the flux through the top layer (Hillel 1998).

The third type of preferential flow is funneling which is one of the subdivisions of lateral flow. Funneling happens below the root zone in stratified soil and sediment profiles. Because of the low permeability of the underlying layer such as bed rock or a hardpan or even fine-textured soil, water moving vertically through a soil profile is partially inhibited at the interface and this compels water to accumulate above the restrictive layer and in turn flows laterally across it (Walter et al. 2000). Generally, lateral movement is important, especially in spatially variable soils and landscapes. Lateral movement can accelerate and increase nitrate leaching due to collection of water from different parts of the landscape to a point where hydraulic conductivity is large enough and then downward movement to ground water occurs.

Due to high P sorption capacity of most mineral soils, phosphorus can be lost from the soil if the amount applied exceeds the retention capacity of the soil and the rate of uptake by plant roots (Brye et al. 2002). However, most subsurface transport of phosphorus is in preferential pathways. Simard et al. (2000) reported that particulate P forms are the dominant fraction transported through tile drainage systems. It is hypothesized that these suspended particles may be transported rapidly through macropores of soil profile. Rapid flow through macropores decreases the contact time between the percolating water and the subsoil, and causes reduction of P adsorption. This suggests that preferential flow is an efficient transport mechanism for the leaching of phosphorus.

## 1.3 AGRICULTURE MANAGEMENT EFFECTS ON N AND P LEACHING

Common soil management activities in agriculture include tillage, cropping system, residue manipulation, irrigation system and application of fertilizers. Agriculture is known as a nonpoint source of N and P pollution in a watershed. In other words, agricultural lands are often considered as major sources of nutrients into local rivers and regional water bodies (Saso, 2009). Intensive agricultural practices usually impair water quality. However, negative effects can be reduced by means of environmental protection. For example, the balance between plant uptake and the rate of applied N and P is an important factor affecting the risk of leaching from arable areas.

### 1.3.1 Source of N and P

Aiming for greater yields, farmers apply different sources of N and P as fertilizers and organic amendments such as manures, slurries and composts. Organic and inorganic N fertilizers undergo transformations into plant available N forms. Although using these materials increase the quantity and quality of crops, excessive use of them, beyond plant requirements elevates the risk of nitrate leaching and ground water contamination. Under aerobic conditions, nearly all labile forms of nitrogen can be converted to nitrate and most agricultural crops absorb most of their N in the form of nitrate (Mangiafico 2005). However, there is no difference between nitrate derived from fertilizers and that from organic sources with respect to risk of leaching. The intensity of nitrate leaching induced by fertilizers or manures depends on the availability of N released by these amendments and the amount of excess of water.

#### 1.3.1.1 Performance of manure as source of nutrient

Manures have been successfully used in arable land all over the world for many years (Bakhsh et al. 2009). Manure is not only a good source of nutrients, manure is also a good source of organic matter, which can improve soil quality and have beneficial effects on agricultural productivity, especially for soils degraded by erosion (Larney et al. 2005). However, over time, extensive livestock operations in certain regions of the world have resulted in the production of very large quantities of animal waste, contributing to excess application of livestock manure to land, and consequently resulting in loss of P and $NO_3^-$-N from agricultural areas (Carpenter 1998).

Controlling potential nitrate leaching in manured soils is often much more difficult than in those receiving fertilizer. This could be due to the N availability in manures which depends on mineralization of organic N and soil moisture and temperature conditions (Power et al. 2001). Besides, variation in manure nutrient content is a challenging factor in estimating the amount of required manure for crop needs (Eghball et al. 2002). Therefore, it is difficult to control soil nitrate levels resulting in potential degradation of water quality at high manure application rates.

Among the animal manures, hog manure is an excellent source of nutrients for crops. Manitoba is Canada’s third largest hog producing province with a yearly production of about 2.5 million hogs (Nikiema et al. 2013). Without considering any losses, approximately 22,500 to 24,000 tonnes of N and 5,000 to 7,000 tonnes of P are produced by the hog industry every year in Manitoba. This is equivalent to 8-9% and 12-17% of the amount of applied N and P in synthetic fertilizers (Flaten et al. 2007). According to Manitoba Conservation (2006), approximately 120000 ha receive hog manure on an annual basis, corresponding to 2.5% of the crop land area in Manitoba.

### 1.3.1.2 N and P Availability of liquid manure

Generally, liquid hog manure with a large amount of available N in the form of inorganic ammonium-N is a cost effective source of N for plant uptake. However, under adverse environmental conditions, the readily available N from manure can be lost through volatilization, surface runoff, leaching or denitrification (Flaten et al. 2007).

Several studies have been conducted to determine the influence of hog manure on $NO_3^-$-N and P leaching (Daudén et al. 2004; Bergström and Kirchmann 2006; Bakhsh et al. 2005; Nikiema et al. 2013). For example, Bergström and Kirchmann (2006) showed that during three years of study on nitrate leaching with four rates (50, 100, 150 and 200 kg of N $ha^{-1}$) of liquid hog manure, the leached total N increased with increasing rate of manure application. These authors concluded that liquid hog manure may be more susceptible to leaching compared to synthetic N fertilizer on sandy soils due to greater crop use efficiency of added nitrogen and phosphorus from synthetic fertilizers than liquid manure.

Nikièma et al. (2013) measured N leaching from liquid hog manure at three rates of 64, 128 and 192 kg N $ha^{-1}$ on a sandy soil in Manitoba. They found that increasing manure application rate increased soil available N concentration. This resulted in enhanced N leaching as 4.7, 28.4 and 54.5 kg $ha^{-1}$ nitrate-N was lost in the low, medium and high manure-N treatments. Despite the fact that liquid hog manure increased grain yield and N removal by about 40%, the nitrogen use efficiency decreased at higher rates of manure N application.

In monogastric animals (e.g., pig and poultry) with limited ability to digest phytic acid, the use of phytase enzymes combined with reductions in the quantity of non-phytate phosphorus in diets, have successfully reduced manure total P concentrations (Ige et al. 2006). However, changes in animal diets can also change the chemical composition of manures and speciation of the manure P. These changes can affect the potential for P retention and loss from manured soils (Toor et al. 2005). Depending on the manure type, the inorganic P fraction in manures and consequently P leaching losses can vary. As an example, Kumaragamage et al. (2011) reported water soluble inorganic P was higher in liquid hog manure (3.42 kg $t^{-1}$) than solid cattle manure (0.8 kg $t^{-1}$) on a dry weight basis. Inorganic P concentration in $NaHCO_3$, NaOH and HCl extractable fractions were also higher in hog manure than solid cattle manure resulting in about five-fold higher total inorganic P in liquid hog manure than in solid cattle manure. As a result, the concentration of labile P in liquid hog manure was far greater than in solid cattle manure.

Although liquid hog manure has a lower content of P than solid hog manure, the smaller amount of organic carbon and carbon to phosphorus ratio (C:P) in liquid hog manure result in greater mineralization and release of P compared to solid manure. For instance, Tarkalson et al. (2009) conducted a column study on P leaching in a calcareous soil treated with inorganic fertilizer (MAP), solid dairy manures, and liquid dairy manures. Results showed that P mobility in soil was in the order liquid dairy manures >MAP > solid dairy manures. They concluded that the greater leaching of P in the liquid manure treatment compared with the solid manure treatment may be caused by a combination of factors including microbial activity, coating of P adsorption sites on clay particles by organic C compounds, and P-Ca and P-Al reactions.

#### 1.3.1.3 N and P availability of solid manure and compost

Only a few studies have reported the effect of solid hog manures on $NO_3^-$-N. However, solid or composted manure have large amounts of organic N that can be converted to available inorganic N by mineralization and nitrification processes over the years, depending on soil and weather factors (Flaten et al. 2007). Eghball (2000) reported 21% mineralized organic N for solid cattle feedlot manure and 11% mineralized organic N for composted manure during the first growing season.

Due to the high ratio of carbon to nitrogen (C:N) frequently found in solid manure, slow release of N can limit the availability of N for crops. For example, Qian and Schoenau (2002) showed a negative correlation between N mineralization and C:N ratio of solid manure. They reported a decrease of N availability for various solid manures at C:N ratios above 15:1.

Composting manure decreases and stabilizes the amount of nutrients such as N and P and thereby reduces the rate of release of nutrients into the soil (Gagnon et al. 2012). Helgason et al. (2007) reported small nitrogen uptake from composted cattle manure due to low N availability of organic N in the first year of application. In the compost, more than 70% of total P was in the inorganic form which showed high availability of P (Gagnon et al. 2012).

Maeda et al. (2003) studied the effects of different fertilizer applications through seven years on N leaching under the rotation of corn-cabbage in an Andisol in Japan. The treatments were composted hog manure, coated urea, ammonium nitrate and no fertilizer. Their results showed that the N concentration at the depth of 1 m increased significantly for plots that received ammonium nitrate and coated urea in the second year of application while this started to happen for composted hog manure after four years of application. Although compared with inorganic fertilizers, composted hog manure acts as a slow release N fertilizer when used excessively it can increase the risk of $NO_3^-$-N leaching.

Eghball et al. (2003) determined the leaching of different P fractions following beef cattle manure and composted manure application. Manure, composted manure, and fertilizers were applied to meet the N and P needs of corn (Zea mays L.). Following four years of manure and compost applications, leaching of plant-available P was observed to 30 cm below the soil surface. The differences in total and inorganic P among treatments were significant only in the top 15 cm of the soil profile. Greater concentrations of total, available, and inorganic P fractions were observed for the N-based manure and compost treatments as these management strategies received more P than P-based treatments. Inorganic P constituted more than 70% of beef cattle manure or composted feedlot manure. Only a small fraction of total P in beef cattle feedlot manure or composted manure was related to water soluble P (<13%).

#### 1.3.1.4 Comparison between P derived from manures and fertilizers

Synthetic phosphorus fertilizers are different in chemical properties in comparison with organic P amendments such as: manures and composts. Generally, P fertilizers have greater P concentration than organic by-products and, except for P rock, they are nearly 100% soluble and available in soil before retention by soil or immobilization by microorganisms. Therefore, they are potentially highly leachable, can easily transfer to subsurface flows, and their environmental risk is higher than for other sources of P (Sims and Sharpley 2005).

Manures contain both organic and inorganic P fractions with the dominance of inorganic P. Soluble inorganic P fractions of manures are mainly orthophosphates so they can be easily leached in water passing through the soil profile. Organic fractions may be mineralized via hydrolysis by a variety of phosphatase enzymes through biological processes. Therefore, P losses from manures are more dependent on seasonal changes (e.g., moisture and temperature) and organic P sources may have lower risk of P losses than fertilizers, especially in the short term. Following long term application of manures in soils, the risk of P leaching from soil is increased, particularly in the case of liquid hog manure (Sims and Sharpley 2005; Abul Kashem, et al. 2004).

Kumaragamage et al. (2011) studied P runoff and leaching losses in ten fertility treatments including solid cattle manure, liquid hog manure, and inorganic fertilizer (MAP) in clay loam and loamy sand soils by conducting a rainfall simulation runoff study and a column leaching study. The researchers reported that the P leaching for liquid hog manure was generally greater than that in solid cattle manure, but less than in MAP.

In contrast, McDowell et al. (2004) used Mehlich-3 to study the effect of different sources of P including fertilizer (triple superphosphate), dairy and poultry manure, and dairy and poultry compost on P leached from a silt loam soil. The results showed that more P was leached from dairy compost and poultry manure amended soils compared with mineral superphosphate and dairy manure. The authors suggested that the presence of small amounts of organic matter in dairy compost facilitated P loss by blocking P sorption sites whereas P applied as superphosphate was more susceptible to P sorption by the soil matrix, resulting in lower P leaching losses.

#### 1.3.1.5 Soil testing and rate of manure application

Although N and P soil tests do not account for transport pathway and proximity to water bodies, they can predict nutrient availability for crops and optimum yield, to help prevent soil nutrient buildup, as well as decrease the risk of nutrient loss through leaching and runoff. Annual soil testing is a useful tool for adjusting manure application rates (Olson et al. 2010). Historically, manure nutrient management in North America and Europe has been based on crop N requirements to meet yield potentials and to minimize $NO_3^-$-N leaching and potential for groundwater contamination (Miller et al. 2011). The imbalance between N and P requirements of the crop and the supply of these nutrients in manure lead to two different manure application strategies including N- and P-based managements (Olson et al. 2010).

Due to the greater N:P ratio of crop removal than the N:P ratio in manure, N-based manure management results in the accumulation of P (Eghball 2002). Therefore, many regions of Canada, including Manitoba, have applied manure to crop lands based on P-based management to decrease the risk of water contamination by runoff P. However, the amount of land required for P-based application is five to seven times that for N-based application (Olson et al. 2010). Multiyear P-based manure application is more cost effective in labor and equipment compared with annual P-based manure application with probably the same environmental benefit (Miller et al. 2011).

Most studies have been conducted on the effect of P-based application versus annual N-based applications on P and N in soil and runoff (Ferguson et al. 2005, Sharpley et al. 2009, Miller et al. 2011). For example Olson et al. (2010) used cattle manure and, found that the apparent N and P recovery was the greatest for treatments that received N and P fertilizers, intermediate for the P-based organic amendments, and smallest for the N-based manure and compost amendments. Their results showed that the P-based manure and compost amendments have relatively large apparent N recovery values due to the application of urea as complementary N fertilizer. Ferguson et al. (2005) in a study comparing N- and P-based application of composted and fresh beef feedlot manure on crop yield and soil N and P movement, reported that N-based manure and compost treatments had the highest yield, soil $NO_3^-$-N and P concentration.

At present, few studies have reported on $NO_3^-$-N and P leaching under N- and P-based manure application. For example, Eghball et al. (2003) reported less soil buildup of P fractions in the soil from P-based manure applications than N-based. They also found evidence of downward movement of P to a depth of 30 cm of soil under N-based manure application. Conversely, Toth et al. (2006) compared N- and P-based manure applications on N and P crop uptake, leaching below the root zone and soil P build up. These authors reported no differences in the amount of

N and P loss by leaching. However, an increase of 47% of soil test P after four years of N-based manure applications suggest high risk of P movement.

### 1.3.2 Water Management

Water management practices may have important impact on nitrogen availability for crops and N losses through volatilization, denitrification and leaching. Coarse textured soils and intensive production of shallow root crops can lead to considerable $NO_3^-$-N losses by leaching. For instance, in Ontario, 17% of the total agricultural land has a water surplus, with $NO_3^-$-N concentration that is greater than 14 mg $L^{-1}$ in the soil profile (Chambers et al., 2001). Surplus water can be estimated by the following relation:

$$\text{Surplus water} = (\text{Precipitation} + \text{irrigation}) - \text{Evapotranspiration} \pm \Delta \text{ soil water storage}$$

The surplus water can leach $NO_3^-$-N below the root zone and this can reach subsurface water bodies. Althaus et al. (2009) indicated that spatial and temporal variations in $NO_3^-$-N in the soil solution were related to the pattern of water movement induced by the applied irrigation method.

In addition to precipitation in dry land farming, or irrigation in irrigated farming systems, surplus water can contribute to soil moisture content and consequently microbial activities. Nitrification rate in the soil increases by increasing soil moisture and reaches a maximum near-field capacity moisture (-33 kPa in medium to heavy textured soils, and 0 to -10 kPa in light-textured or sandy soils; Sahrawat, 2008). Therefore, a high rate of nitrification results in greater risk of N leaching by precipitation or irrigation.

### 1.3.3 Cropping System

Perhaps after N addition and precipitation, cropping systems have a greater effect on the movement of $NO_3^-$-N below the root zone and its delivery to subsurface water sources than any other agricultural practices. Campbell et al. (2006) showed that fallow frequency was the main factor influencing N leaching in southwestern Saskatchewan due to the promotion of N mineralization and soil water content and consequently facilitating N leaching. Legume plants can fix atmospheric $N_2$ and need little or no N fertilizer to complete their growth cycle. However, nitrogen derived from legumes can be an important source of N inputs for watersheds. For example, in the Mississippi River Basin, biological nitrogen fixation is the second most important N input, after synthetic nitrogen fertilizer (Howarth 2008).

Due to their greater nutrient uptake and multiple hay harvests, perennial grasses are recommended for areas with heavy application of manure and risk of nutrient contamination (Read et al. 2008). Dinnes et al. (2002) reported that cover crops can reduce $NO_3^-$-N concentration in leachate by 20-80% compared to the control without cover crop. They also reported that the efficiency of perennial forage grasses for reducing leached $NO_3^-$-N were two to three times as much as that of legumes.

Entz et al. (2001) showed that alfalfa extracted significant amounts of $NO_3^-$-N to soil depths of 90, 180, 210 and 270 cm, respectively from one to four years of the stand, but annual crops extracted $NO_3^-$-N to depths of only 150 cm. However, one of the issues with alfalfa is the large amount of $NO_3^-$-N which can be mineralized and leached after the alfalfa is terminated. Based on this, they concluded that the optimum stand length of alfalfa for deep extraction of nitrate was less than 6 years. Campbell et al. (2006) reported that deep-rooted perennial grasses reduced nitrate leaching compared to shallow-rooted annual crops, such as flax, that use less N and consequently, leave more $NO_3^-$-N in the soil.

The extensive rooting system of perennial grasses could result in greater phosphorus uptake from the soil due to the increased surface area of the roots and the ability to explore more of the soil (Bundy et al. 2005). However, van Es et al. (2004) reported that P losses on perennial forage crops were more than those on annually cropped land due to higher numbers of continuous biopores. In a lysimeter study, Leinweber et al. (1999) determined that grassland on a sandy soil and winter grain crop rotation on a loamy sand soil had the smallest amounts of P loss by leaching. Therefore, the combination of soil texture and cropping system are important for leaching of P if the labile P is not consumed by crops.

### 1.3.4 Tillage

Tillage practices affect soil properties and losses of N through volatilization, denitrification and leaching (Malhi et al. 2001). The time of ploughing and the selection of crops in the following years are very important in utilizing increased N mineralization and thereby minimizing nitrate leaching. Considerable N mineralization occurs in the first and second year after ploughing. Djurhuus and Olsen (1997) studied the effect of the time of ploughing on nitrate leaching after a cut of grass-clover. Their results indicated that winter wheat did not have the potential for taking up the mineralized N in autumn after early autumn ploughing of grass-clover, resulting in high N leaching.

Comparing N losses from perennial cropping systems, Di and Cameron (2002) reported that nitrate leaching potential typically increases in the order of hayed grassland < grazed pastures < ploughed pasture. They suggested the hayed grassland has the least leaky system for $NO_3^-$-N due to the removal of N in harvested biomass. However, ploughing accelerated N mineralization rate in a short period of time causing a large amounts of N leaching in ploughed pastures, especially when ploughed in late summer and early fall.

Management strategies aimed at reducing the N loading to water bodies have stressed conservation tillage. Though conservation tillage is usually effective in reducing runoff from cropland, increases in drainage flux ultimately could increase the leaching of a large proportion of N fertilizer to groundwater (Abril et al. 2005). For example, a no-tillage system encourages the formation of continuous macro-pores and results in preferential flow of nitrate. Conversely, Mkhabela et al. (2008) reported greater $NO_3^-$-N in conventional tillage than no-tillage from plots fertilized with cattle manure. They concluded that the smaller amount of nitrate leaching under no-tillage may be due to greater denitrification rate and $NH_3$ losses through volatilization in the no-tillage plots.

Generally, the relationship between tillage system and risk of nitrate leaching also depends on the stage of maturity of the tillage system. For example, in the early stages of the transition from intensive to conservation tillage, N immobilization may be much greater than N mineralization, resulting in low risk of nitrate accumulation and leaching. However, in mature conservation tillage system, N immobilization and mineralization should reach equilibrium, where the risk of nitrate accumulation and leaching is greater than in the early stages of conversion.

In many regions, P loss mostly occurs through soil erosion and runoff, as such, the influence of tillage on P loss in runoff is more relevant. As an example, Gaynor and Findlay (1995) measured P loss from a corn field under conventional and conservation treatments in southwestern Ontario. They found that runoff P concentrations from conservation tillage plots were 2.2 times greater than from conventional tillage plots. However, conservation tillage that is an effective practice to decrease sediment and sediment-bound P export from croplands can be less effective in regions such as Manitoba where snowmelt plays a dominant role in the export of phosphorus from the field (Tiessen et al. 2010). So, in the Canadian Prairies conservation tillage is not an effective practice for reducing P loss by runoff.

Few studies have investigated the effect of tillage on phosphorus leaching. For example, Rubæk et al. (2006) examined P leaching from soil columns of a recently tilled soil following incorporation of liquid hog manure derived from centrifugation, addition of flocculants and anaerobic digestion of manure. All treatments had high particulate and dissolved P concentrations in leachate, probably due to the recent tillage and high soil P status.

## 1.4 METHODS OF MEASURING NITRATE LEACHING, ADVANTAGES AND DISADVANTAGES

Soil nitrogen is so dynamic that it is difficult to monitor nitrogen forms in soil with one method solely. Several methods of monitoring nitrate leaching have been developed, but three are more common: soil cores, ceramic suction cup and subsurface drainage lysimeters (Zotarelli et al. 2007). There are some advantages and drawbacks for each method.

The soil core method is simple, relatively cheap, widely used, and appropriate to most soils, but it is time consuming and destructive, and it only provides an image of N distribution at one point in time. In addition, soil coring is an indirect measure of inorganic N in the soil solution and is associated with errors caused by spatial variability in soil nitrate concentrations. Also, the appearance or disappearance of nitrate cannot be attributed to leaching alone. For example, when there is no sign of nitrate leaching, such as bulge of nitrate concentration within the soil profile, two completely different stories could have happened: leaching of nitrate has occurred and the bulge of nitrate has passed the specific sampled depth of soil (eg, 120 cm), or no nitrate leaching has been happened. Therefore, in order to quantify nitrate leaching, soil coring must be either combined with modeling approaches, or linked with water flow dynamics below the rhizosphere (Zotarelli et al. 2007).

Ceramic suction cup lysimeters are suitable methods for monitoring nitrogen leaching in unsaturated conditions. They can be easily set up and allow repeated measurements from the same location. However, collected leachate is from various directions rather than from one direction. The radial distance through which a water sample is drawn to the lysimeter varies as a function of soil water content, soil texture, and the amount of suction applied to the lysimeter. Finally, suction cup lysimeters only provide a measure of solute concentration without any consideration of water flow. Thus, without one of the components, concentration and flow, no determination of solute loading can be made. Also, in coarse sandy soils, it is frequently impossible to obtain adequate sample volumes because of low soil water availability and dry conditions, and this may cause large uncertainties in calculating N losses (Zotarelli et al. 2007).

Drainage lysimeters are a common instrument for monitoring N leaching dynamics. Below a specific soil depth, N load passing through the soil can be calculated because drainage lysimeters capture the entire leachate volume and N concentration. However, lysimeter installation may cause significant soil disturbance. In addition, drainage lysimeters must be placed deep enough if they are to represent overall field-conditions in the crop root zone (Zotarelli et al. 2007). The other disadvantage is that it collects water and solute only if the soil above the lysimeter is saturated, or it intercepts water traveling along a preferential flow path.

There are basically two categories of lysimeters: weighing and non–weighing. As the name suggests, weighing lysimeters use differences in lysimeter mass to account for changes in hydrologic cycling and crop growth (Mulla and Strock 2008).

The zero-tension pan lysimeter is one of the most common non-weighing lysimeters, with numerous geometric shapes and sizes used in the literature. Pan lysimeters generally consist of a pan placed under an experimental plot at a predetermined depth in the soil, usually just below the maximum rooting depth of the crop being studied (Mulla and Strock 2008).

Another type of lysimeter consists of a soil monolith isolated from the surrounding soil by encasing it in a material such as polyvinyl chloride (PVC). Soil monolith collection can involve extracting a core by drilling methods or by hydraulically forcing a rigid PVC pipe into the soil to a specified depth followed by excavation (Enns 2004; Nikiema et al. 2013). Leachate from a zero-tension lysimeter commonly drains into a reservoir and is periodically retrieved using a vacuum pump to collect leachate samples.

However, one of the disadvantages of lysimeters is the controlled boundary condition of sampling region. Moreover, physical soil properties such as soil texture and depth of each morphological horizon are varied between soil profiles of individual lysimeters. As well, "all the agronomic activities within the lysimeters such as tillage, sowing, and manure management are limited to hand operations" (Goss and Ehlers 2009). Therefore, lysimeters cannot provide a complete picture of nitrate leaching under natural field conditions with spatial variability (Olatuyi 2011).

## 1.5 SOIL WATER CONTENT MODELLING

Computer models are useful tools to estimate the risk of $NO_3^-$-N contamination of surface and ground waters resources. However, the models must be tested based on actual field data for each region where soil and climate conditions are different. Direct soil moisture measurements are not feasible in many agronomic applications. Thornthwaite and Penman (1948) introduced the concept of potential evapotranspiration, and this allowed soil moisture models to be developed in many countries to estimate water content and its distribution in the soil profile based on climatological data. The agrometeorological techniques such as biometeorological time scale, crop parameters, soil characteristics and day length were used in the first generation of soil moisture models (Baier and Roberrson 1996).

The development of models started with the simple moisture budget models which require few input parameters. However, these water budget models have to be validated for each geographic region with new soil, plant, and climatic conditions due to the empirical nature of several of its input variables such as root growth coefficients (Akinremi et al. 1996).

Over the years, process-oriented models which required more input variables with detailed measurements (Akinremi et al. 1996) and less frequent validation have been developed. Later, a number of different process-oriented models were developed to describe $NO_3^-$-N coupled with soil water movement in soils at different spatial and temporal scales. These mechanistic models can be divided into deterministic and stochastic models. In deterministic models, the output variable has a specific value at a given place which simplifies the complexity of the natural environment (Fetter 1999). The numerical estimates of these models offer a better understanding of the physical processes than stochastic models; however, the uncertainty of the final results limit their ability to predict the existing environmental $NO_3^-$-N water pollutions (Acutis et al. 2000).

Nevertheless, deterministic models are not able to overcome naturally existing spatial and temporal variability in field situations. Therefore, stochastic-mechanistic approaches have been developed. In stochastic models, the value of the output variables has statistical uncertainty that it is due to uncertainty in the values such as hydraulic conductivity and porosity. So, in these models, we have a range of possible outcomes. These models need many measurements of variables, which limits their extensive application.

As an example of process-oriented models, LEACHMN (Hutson and Wagenet 1992) a process-based, one dimensional model simulates water and solute movement, and related chemical and biological processes in the unsaturated zone (Sogbedji et al. 2006). Sogbedji et al. (2001) calibrated the LEACHMN model by adjusting the values of nitrification, denitrification and volatilization rate constants to optimize the fit between predicted and measured data. They reported that the N transformation rate constants obtained from the calibration efforts were similar to those used in other model simulation studies. Akinremi et al. (2005) modified the

LEACHMN model by incorporating the van Genuchten retentivity function. They concluded that the modified model was adequate for predicting nitrate leaching in subsequent studies.

Because $NO_3^-$-N readily moves with water, an accurate simulation of water movement in the soil profile can be a useful tool for estimating $NO_3^-$-N leaching and its effect on groundwater contamination. The use of a simple moisture budget model which requires few inputs is the primary step for investigating water movement and consequently $NO_3^-$-N leaching of manured soils under different cropping systems. An N-dynamic subroutine can then be incorporated into the Versatile Soil Moisture Budget (VSMB) to determine its reliability for nutrient loss from different locations on an annual basis.

The VSMB model (Baier and Robertson 1966) has been calibrated and validated for estimation of soil moisture of croplands on the Canadian prairies (Akinremi et al. 1996). In addition, as a multi-layer water balance model, VSMB was developed for monitoring of soil moisture in a large area, planning of farming operations, and mapping of agroecological resources areas. For example, Akinremi et al. (1997) used VSMB to estimate available soil moisture within the root zone on a regional scale in Alberta. They reported that simulated soil moisture values across the province of Alberta, particularly in the fall, were comparable to those obtained from field surveys. Hayashi et al. (2010) simulated the evaporation from grasslands using VSMB in Calgary, Alberta. The simulated evaporation was close to what was measured in the field. They reported that the model was highly sensitive to depth of soil profile and drying curve function. Ojo (2012) replaced the Priestley-Taylor equation with Penman Monteith equation to estimate evapotranspiration at 13 sites across Central and Western Manitoba during the 2009 and 2010 growing seasons. He reported that this replacement did not significantly improve the soil moisture simulation at most of the locations. Similar to Akinremi's et al. (1996) results, the model accurately estimated the soil moisture content of the first surface depths; however, the root mean square error increased at lower depths.

## 1.6 SPATIAL DISTRIBUTION OF NITRATE LEACHING AND SCALE ISSUES

The importance of variability of soil properties within a landscape can influence the available plant nutrients leading to variability in yield and water contamination. Soil spatial variability can be either visible or not visible to the eye and these patterns are caused by both natural soil forming processes and human modifications to the landscape through management history.

Spatial distribution of nitrate leaching is a consequence of interactions between soil physical, chemical and biological properties, physiographic factors, and management such as irrigation, tillage, fertilizing and cropping system. These interactions can influence hydrology including infiltration, run off and erosion leading to changes in N mineralization, nitrification and losses via volatilization and denitrification. As an example of the influence of soil texture on soil nitrogen mineralization, Cote et al. (2000) reported a negative correlation between nitrogen mineralization and clay content due to less mineralization of N by the microflora. van Es et al. (2006) compared nitrate losses from manure in loamy sand and clay loam soils and reported lower retention of $NO_3^-$-N in the loamy sand soils. The smaller water holding capacity and greater hydraulic conductivity in the coarse-textured loamy sand soils contributed to greater $NO_3^-$-N leaching from loamy sand soil than the clay loam. Physiographic factors can produce spatial variation in the vertical and lateral rate of water movement through the soil to the stream, as well as spatial variation in N balance (in different parts of the landscape) that influences nitrate leaching (Bruckler et al. 1997; Burt and Arkel, 1987).

Spatial distribution of the $NO_3^-$-N concentration is also time dependent. For example, interplays between climate factors (precipitation and temperature), plant (growth stage, root system development and uptake ability) and management (source, method, timing of fertilization and irrigation) determine temporal patterns of nitrate leaching (Nelson et al. 1995).

Knowing the spatial variation in the risk of nitrate leaching may result in an increase of yield as well as environmental protection. Technological advances have occurred to develop site-specific management in line with the spatial distribution of nutrients and, of course, nitrogen. Contrary to potassium and phosphorus that show comparatively stable spatial distribution patterns, nitrate as a mobile anion shows large fluctuations in spatial distribution patterns (Lehmann and Schroth 2003). Nevertheless, investigation of the spatial variation of $NO_3^-$-N is important due to the economic impact of leached nitrogen. Having enough information of nitrate spatial variation allows the derivation of the necessary number of samples for a sound estimate of the mean. Geostatistical methods are good tools for determining the spatial correlation and the range across which data are spatially correlated (Stenger et al. 2002).

Changing the scale of $NO_3^-$-N leaching from the plot, to the field, to the watershed can be affected by different factors. At the plot scale, leaching, denitrification, mineralization and volatilization, as well as N application rate and timing, crop rotations, precipitation, and runoff may be important factors. At the field scale, spatial variability in soil and landscape positions, as well as water management practices such as drainage depth and spacing intensely affect the $NO_3^-$-N leaching (Mulla and Strock 2008). For example, Olatuyi (2011) used $^{15}N$ to estimate $NO_3^-$-N leaching in a hummocky landscape near Brandon, Manitoba. The plots were established in three landscape positions including upper, middle and lower slope. His results showed that $^{15}N$ loss was in order of lower > upper > middle of slope. As a result, it is necessary to minimize fertilizer application at the lower slope position. However, due to the least leaching potential of the middle slope position, more N fertilizer can be applied to this region in order to have high yield. At the watershed scale, new factors begin to affect the $NO_3^-$-N leaching including ground water base flow, denitrification and plant uptake, in-stream processing of nitrate in wetlands and riparian buffer strips. In addition, the spatial variation in field management practices and precipitation across the watershed can influence the amount of $NO_3^-$-N transport. Therefore, the methods to predict $NO_3^-$-N loss should vary based on scale of investigation (Mulla and 2008).

There is a lack of knowledge of cropping system effects on the movement of $NO_3^-$-N and P below the root zone and its delivery to subsurface water sources. Knowledge of nutrient management effects on $NO_3^-$-N and P leaching loss form farming lands as well as an understanding of the differences in nutrient loss measurements in the field is necessary to minimize the amount of water pollution.

## 1.7 THESIS ORGANIZATION

The thesis includes five chapters. This general introduction (Chapter 1) introduces some background information of N and P leaching from hog manure and its environment effects.

In Chapter 2, we compared the amount of nitrate that is lost below the root zone from liquid and solid manure management systems, and the effect of cropping system and N-based and P-based application rates, on crop yield, nitrogen removal, and the loss of water and nitrate below the root zone. In addition, the use of field core lysimeters will be compared with traditional soil profile sampling in measuring nutrient loss from the soil at the Ian Morrison Research Station of the University of Manitoba at Carman Manitoba.

In Chapter 3, the effect of cropping systems and manure management techniques including liquid and solid manure management systems and N-based and P-based application rates on leakage of phosphorus to the environment and the changes in soil test P will be compared. The results of a 3-year study at the Ian Morrison Research Station of the University of Manitoba at Carman, Manitoba will be presented in this chapter.

Chapter 4 reports the effects of hog slurry at different rates on nitrate and phosphorus leaching behavior of a permeable sandy soil at Carberry and compares the results with those of synthetic fertilizers and compost. That chapter focuses on the relationship between the amounts of N applied, nitrate leached and crop removal of nitrogen to find out the sustainable rates of hog manure and fertilizer application to reduce nitrate and phosphorus leaching on this soil type.

Chapter 5 estimates the accuracy of simulated soil moisture content in two different cropping systems by VSMB and compares simulated drainage obtained from the model with real data obtaining from field lysimeters on two soil types at Carman and Carberry.

The final chapter (Chapter 6) is a summary of the findings of the data chapters, and it includes general conclusions and implications of the results and suggestions for further research.

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# 2. EFFECT OF CROPPING SYSTEM AND TYPE OF HOG MANURE ON NITRATE LEACHING IN A SANDY LOAM SOIL

## 2.1 ABSTRACT

In Manitoba, hog manure is widely applied as a fertilizer for crop production. Excess applications of livestock manure, however, can result in the loss of nitrate-nitrogen ($NO_3^-$-N) and degrade surface and groundwater. The main objectives of the study were to compare the effect of cropping system (i.e. perennial versus annual), N-based and P-based manure application rates on crop yield, nutrient removal and the loss of water and nitrate below the root zone. Another objective was to compare the use of field core lysimeters with traditional soil profile sampling in measuring $NO_3^-$-N loss from the soil. The 3-year study was performed at the Ian Morrison Research Station of the University of Manitoba at Carman, Manitoba. The experiment was a split-plot design with annual and perennial cropping systems as the main plots. Five nutrient management treatments were the subplots: nitrogen based liquid hog manure application (LH-N), phosphorus based liquid hog manure application (LH-P), nitrogen based solid hog manure application (SH-N), phosphorus based solid hog manure application (SH-P) and a control. The study was initiated in the spring of 2009 with the perennial cropping system consisting of a mixture of timothy and orchard grass. The annual crop was canola in 2009; barley in 2010 and canola in 2011. Soil samples, leachate and plant samples were taken and were analyzed for nitrogen. The major finding of this study was that while a substantial amounts of nitrate-nitrogen was lost from the annual plots (40 to 60 kg ha$^{-1}$ in 2010 and 23 to 60 kg ha$^{-1}$ in 2011), a negligible amounts of nitrate was lost from the perennial (< 1 kg ha$^{-1}$). These differences could not be attributed to differences in N uptake as the annual crops sometimes took up more nitrogen than the perennial. The lack of nitrate leaching from the perennial system was probably due to a combination of several factors including better synchrony between available nutrients and crop uptake. The results of this study also indicate that the current formula used to estimate N availability from solid manures overestimated N supply. The P-based application of solid hog manure followed by urea in subsequent years reduced the risk of nitrate leaching over the course of the rotation, likely due to immobilization of N with the addition of the straw in the manure. In conclusion, perennial cropping systems consisting of a mixture of grasses have the capacity

to receive and utilize significant amounts of nutrients without nutrient leakage to the adjacent environment. The inclusion of grasses in a crop rotation and their use to utilize excess nutrients are sustainable practices that will benefit the environment. In the annual cropping system of this study, nitrate leaching occurred in the control plots where no nutrients were applied suggesting that nitrate leaching can be reduced by nutrient management but not eliminated.

## 2.2 INTRODUCTION

Agricultural crops respond positively to manure applications and manure has been successfully used in arable land all over the world for many years (Bakhsh et al. 2009). In Canada and Manitoba, hog manure is applied to a significant portion of agricultural lands (Flaten et al. 2003). Hog manure provides nutrients and organic matter to the soil (Maule et al. 2006), so it can be an excellent resource for agriculture. Excess applications of livestock manure, however, can result in the loss of nitrate-nitrogen ($NO_3^-$-N) from agricultural land and consequent degradation of groundwater, streams and lakes (Allen et al. 2006).

The chemical characteristics of $NO_3^-$-N make it susceptible to leaching through the soil system and into water. The drinking water limit for nitrate is 10 mg $NO_3^-$-N $L^{-1}$. The primary health hazard from drinking water that is contaminated with $NO_3^-$-N occurs when nitrate is transformed into nitrite by bacteria that are present in the digestive system. The presence of nitrite oxidizes iron in the haemoglobin of red blood cells resulting in the formation of methemoglobin, which lacks the ability to carry sufficient oxygen to the individual body cells. This causes infants to develop a blue coloration and respiratory problems known as methemoglobinemia, sometimes referred to as the ‘‘blue baby syndrome’’. Also potential cancer risk from contaminated water and food by nitrate and nitrite has been reported (Basso and Ritchie 2005).

Many studies have indicated that there is an increased risk of $NO_3^-$-N leaching when soil receives large or repeated applications of manure. Basso and Ritchie (2005) found that manure applications resulted in higher amounts of $NO_3^-$-N leaching compared to compost and inorganic fertilizer. Bakhsh et al. (2005) reported in their long-term study at Nashua, Iowa, that liquid hog manure resulted in significantly greater $NO_3^-$-N losses and showed no difference in corn grain yields in comparison with synthetic fertilizer application under a continuous corn production system. Bergström and Kirchmann (2006) showed that during three years of study on nitrate leaching with four rates (50, 100, 150 and 200 kg of N $ha^{-1}$) of liquid hog manure, the leached total N increased with increasing rate of manure application. Also they concluded that liquid hog manure may be more susceptible to leaching compared to inorganic N fertilizer on sandy soils due to greater crop use efficiency of added nitrogen and phosphorus from synthetic fertilizers than liquid manure. Daudén et al. (2004) found that applying hog manure above N crop needs did not increase corn yield and led to a greater risk of nitrate loss to water bodies. In southern Alberta, Mayer et al. (2004) found $NO_3^-$-N reached groundwater under soils that received manure, but no significant $NO_3^-$-N reached groundwater under soils that received inorganic fertilizers (as stated by Olson et al. 2009).

Several studies have been conducted to determine the influence of crop type on $NO_3^-$-N leaching (Entz et al. 2000; Campbell et al. 2006; van Es et al. 2006; Bakhsh et al. 2009; Schroder et al. 2010). For example, perennial, deep-rooted perennial grasses were found to reduce nitrate leaching compared to annual crops. Even within annual cropping systems, there are differences in nitrate movement. Shallow-rooted annual crops such as flax use much less N (and water) than wheat and, consequently, they leave more $NO_3^-$-N and water in the soil (Campbell et al. 2006).

The frequency of manure application has also been found to have an effect on leaching losses. Bakhsh et al. (2009) observed that the $NO_3^-$-N concentrations and leaching losses increased by more than 50% when manure was applied every year to a corn-soybean system in comparison with manure application in the corn years only. Soybean yield increases, however, were less than 4% when manure was applied to the soybean crop.

Only one other study is being conducted in Manitoba that compares N and P-based application of hog manure to different cropping systems that is being conducted on a heavy clay soil (Fraser and Flaten 2014). Such a study is needed to understand the movement of nitrate from manure-amended soils. The first objective of this study was to determine the influence of cropping system (perennial versus annual); nutrient management system (N versus P-based); and the type of hog manure (liquid versus solid) on crop yield, N removal, and the loss of water and nitrate below the root zone. The second objective of this study was to compare the use of field core lysimeters to measure $NO_3^-$-N loss from the soil profile with the traditional soil sampling technique.

## 2.3 MATERIALS AND METHODS

### 2.3.1 Site Characteristics

The study was conducted at the Ian N. Morrison Research Farm, University of Manitoba field research station, Carman, Manitoba from 2009 until 2011. The site was located on the Hibsin soil series (Canada-Manitoba Soil Survey Report D60). Surface soils are coarse loamy underlain by clayey deposits. This soil is moderately well drained (Appendix 2.7.A.).

In the fall of 2006, the entire experimental area was seeded to a blend of 50% alfalfa, 34% timothy and 16% orchard grass which was maintained until the experiment began in the spring of 2009. Forty field core lysimeters were installed at the corner of each sub-plot in the summer of 2006 so that water movement and nutrient leaching could be measured directly. Each lysimeter included three main parts: the main column, the schedule 80 PVC pipe with an internal diameter of 54 cm and 106 cm in length, representing root zone extension of annual crops; a circular perforated plate and a collection bottom cap. To reduce the disturbance of soil during installation a custom made hydraulic press was used to push down the main column of the lysimeter to the desired depth. The main column was then pulled out of the soil and turned upside down. Geotextile fabric was placed on the soil to separate the soil from the perforated plate and collection basin. The perforated plates, collection caps and extraction pipes were then installed on the main columns. Details of the lysimeters' design and installation have been previously provided by Enns (2004) and Nikiema et al. (2013) for the Carberry experimental site.

In the spring of 2009, May 29, the alfalfa was killed on the perennial plots by spraying with the 0.34 L $ac^{-1}$ clopyralid (Lontrel) and 0.4 L $ac^{-1}$ of 2-methyl-4-chlorophenoxyacetic acid (MCPA) herbicide. The perennial plots were then left with about 68% timothy and 32% orchard grass. The alfalfa-timothy-orchard grass crop where the annual plots were to be established was killed with the 2.25 L $ac^{-1}$ of glyphosate (Roundup) herbicide and plowed under.

### 2.3.2 Experimental Design

A split-plot treatment structure was established in 2009 with cropping system (annual and perennial) as the main plot and manure/urea treatment as the sub-plot (10 m x 10 m) with 4 replications (Figure 2.1). A three year rotation was employed for the annual and perennial cropping systems. For the annual rotation, canola (Argentine, 1960 kg ha$^{-1}$) was grown in year 1 (2009), barley (Tradition, 4417 kg ha$^{-1}$) was grown in year 2 (2010) and canola (Liberty Link, 2356 kg ha$^{-1}$) was grown again in year 3 (2011). For the perennial rotation, timothy/orchard grass was maintained for all three years.

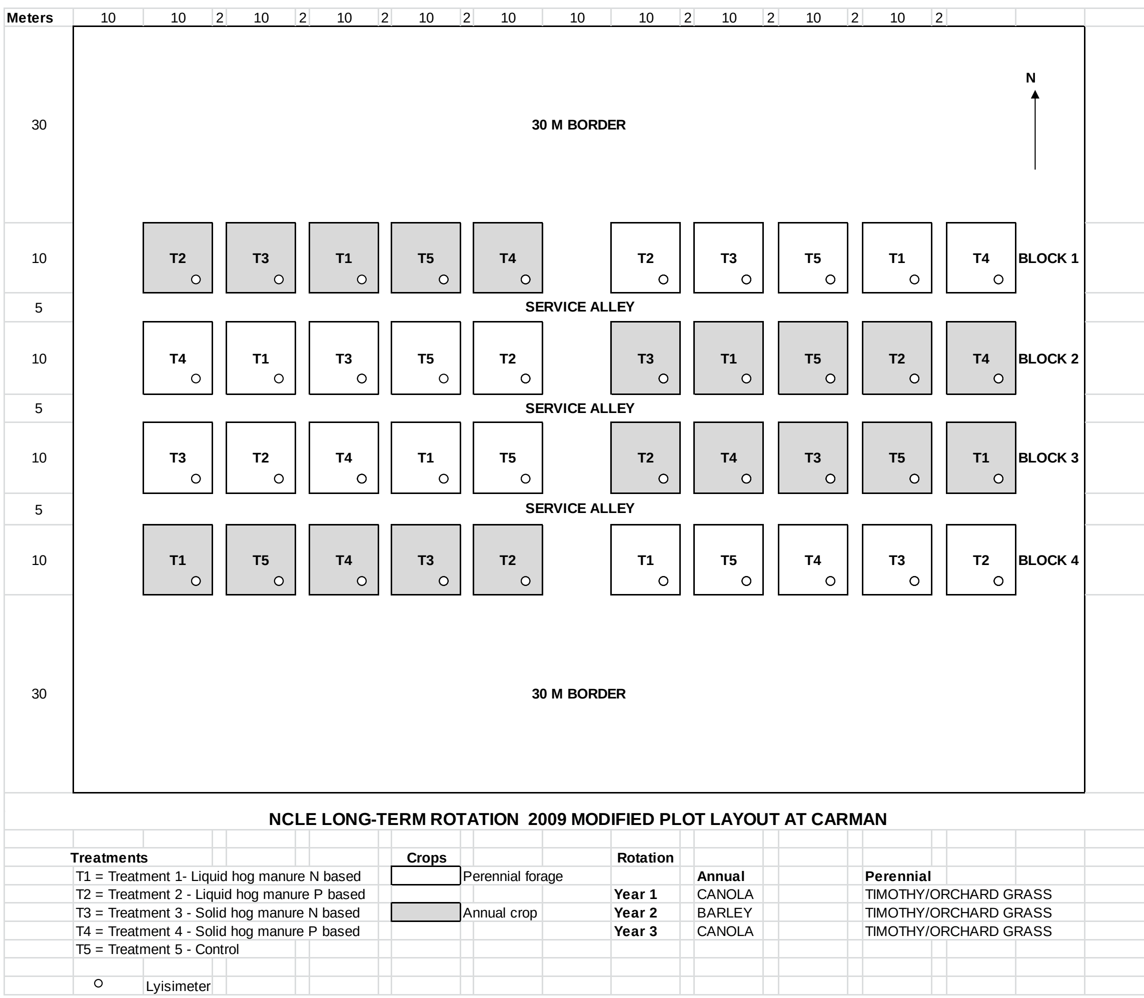


Fig. 2.1. Plot layout at the Ian Morrison Research Station, Carman, Manitoba during three years of study

The manure/urea treatments included:

- **N-based liquid hog manure** where liquid hog manure was applied in 2009, 2010 and 2011 at rates of application that targeted the N needs of the crops;
- **P-based liquid hog manure or urea** where liquid hog manure was applied in 2009 only, at a rate of application to meet the N requirements of the 2009 crop, and urea was applied to meet the N needs of the crops in 2010 and 2011;
- **N-based solid hog manure** where solid hog manure was applied in 2009, 2010 and 2011 at rates of application that targeted the N needs of the crops;
- **P-based solid hog manure or urea** where solid hog manure was applied in 2009 only, at a rate of application to meet the N requirements of the 2009 crop, and urea was applied to meet the N needs of the crops in 2010 and 2011; and
- **Control** where no manure or synthetic fertilizer was applied.

There was a buffer of 5 m between the replicates and a buffer of 2 m between the sub-plots.

The lysimeters received the same manure/urea treatments and were planted to the same crops as their surrounding plots.

### 2.3.3 Manure Application and Seeding Dates

On the 2nd and 3rd of June 2009 liquid hog manure was applied while on the 11th and 12th of June 2009 solid hog manure was applied to the designated plots manually. The annual plots were roto-tilled on June 11 with alfalfa and grass incorporated into the plot together with the liquid manure that had earlier been applied to these plots. After the application of solid hog manure the annual plots were rotor-tilled for the second time to incorporate the solid hog manure on June 15 just before seeding the plots to canola (Table 2.1). On 15th, 16th and 17th of June, 2010 and on 16th and 17th of June, 2011 hog manures (solid and liquid) and urea for P-based treatment were applied to the appropriate plots and incorporation and seeding were carried out on the same day (Table 2.1).

Table 2.1. Manure application, incorporation and seeding dates for the three years of this study

| | *Liquid Manure* | *Solid Manure* | *Urea* | *Incorporation* | *Seeding* |
|---|---|---|---|---|---|
| ***2009*** | June 2, 3 | June 11, 12 | n/a | June 11 liquid<br>June 15 solid | June 15 |
| ***2010*** | June 16 | June 17 | June 15 | June 17 | June 17 |
| ***2011*** | June 16 | June 17 | June 16 | June 17 | June 17 |

### 2.3.4 Manure and Urea Application Rates

**2009**

In 2009, manure application rates were based on the N requirements of the crop using residual soil nitrate analyses for the entire experimental area and target yields for canola and grass. The alfalfa that was chemically killed on the perennial plots and the alfalfa/grass mixture that was chemically killed and plowed down on the annual plots were credited with supplying 60.5 kg N $ha^{-1}$ (MARC 2008). Solid manure application rates were based on standard reference values (MAFRI 2009) for solid and liquid hog manure in the first year of the study. Actual manure N and P application rates were back-calculated from the analyses of representative solid manure samples collected from the Glenlea research station (Table 2.2) and liquid manure nutrient analyses conducted by Ag-Vise Laboratories and provided by Agra-Gold consulting (Table 2.2). The 2009 solid and liquid hog manure application rates are presented in Table 2.3a.

**2010**

In 2010, manure was applied to the N-based treatments only. The N requirements of the crops on the P-based treatments were supplied by urea. Application rates took into consideration residual soil nitrate analyses and target yields for wheat. However, due to the late seeding date, barley was planted instead of wheat. Actual manure nutrient analyses were used for the solid manure based on representative samples collected from the Glenlea research station in the spring prior to application. Liquid manure application rates were based on the Nova meter estimate of ammonium-N and standard reference values (MAFRI 2009) for organic N for liquid manure from a commercial hog barn in Manitoba. Actual liquid manure N application rates were back-calculated (Table 2.3b) using manure nutrient analyses results (Table 2.2) from samples collected at the time of application. The manure samples were collected in plastic buckets and were kept cool (4°C) until analyzed in the laboratory where the samples were mixed and subsamples were analyzed for moisture content, total P, total N and ammonium N.

**2011**

In 2011, manure was again applied to the N-based treatments only and the N requirements of the crops on the P-based treatments were supplied by urea (Table 2.3c). Application rates took into consideration residual soil nitrate analyses and target yields for canola. Manure application rates were based on actual manure nutrient analyses (Table 2.2) of representative samples collected from the Glenlea research station (solid) and a commercial hog barn (liquid) in the spring prior to application.

### 2.3.5 Nitrogen Availability

To estimate available N in manure the following formula was used (MAFRI, 2007):

$$\text{Total available N} = \text{Ammonium N} \times (100\% - \%\ \text{Volatilization loss}) + 25\%\ \text{Organic N}$$

Ammonia volatilization losses were estimated to be 25% of the ammonium N for the annual plots and 35% for the perennial.

Table 2.2. Manure analyses and N availability assuming 25% volatilization loss for annual crops.

| | ***Solid Manure*** | | | | | | ***Liquid Manure*** | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ***Year*** | Total P | Total N | $NH_4$ | Org N | Avail N | Moisture | Total P | Total N | $NH_4$ | Org N | Avail N | Moisture |
| | kg tonne$^{-1}$ | | | | | % | kg 1000 L$^{-1}$ | | | | | % |
| ***2009*** | 3.6 | 3.1 | 1.7 | 1.3 | 1.6 | 70 | 0.9 | 3.8 | 2.2 | 1.6 | 2.0 | 93 |
| ***2010*** | n/a | 5.3 | 1.4 | 3.9 | 2.1 | 70 | 2.2 | 5.0 | 3.0 | 2.0 | 2.8 | 84 |
| ***2011*** | 3.8 | 6.5 | 0.4 | 6.2 | 1.8 | 78 | 0.7 | 2.7 | 2.1 | 0.6 | 1.7 | 98 |

Table 2.3a. Residual soil nitrate levels, target N application rates and manure application rates to annual (canola) and perennial (grass) treatments in 2009.

| ***Rotation*** | ***Treatment*** | ***Target Yield*** | ***Residual Soil $NO_3$*** | ***Target N*** | ***Alfalfa N*** | ***Manure Appl*** | ***Avail Manure N Appl*** | ***Urea Appl*** | ***Urea N Appl*** | ***Manure $P_2O_5$ Appl*** |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | ***----------------kg ha$^{-1}$*** | ***----------------*** | | | ***-------------------------kg ha$^{-1}$*** | | | ***-------------------------*** |
| ***Annual Canola*** | Liquid-N | 1960 kg ha$^{-1}$ | 28 | 140 | 60.5 | 28758 L ha$^{-1}$ | 58 | 0 | 0 | 57 |
| | Liquid-P | 1960 kg ha$^{-1}$ | 28 | 140 | 60.5 | 28758 L ha$^{-1}$ | 58 | 0 | 0 | 57 |
| | Solid-N | 1960 kg ha$^{-1}$ | 28 | 140 | 60.5 | 22.4 tonne ha$^{-1}$ | 37 | 0 | 0 | 188 |
| | Solid-P | 1960 kg ha$^{-1}$ | 28 | 140 | 60.5 | 22.4 tonne ha$^{-1}$ | 37 | 0 | 0 | 188 |
| | Control | | | | 60.5 | 0 | 0 | 0 | 0 | 0 |
| ***Perennial Grass*** | Liquid-N | 6.7 tonne ha$^{-1}$ | 28 | 140 | 60.5 | 32352 L ha$^{-1}$ | 59 | 0 | 0 | 65 |
| | Liquid-P | 6.7 tonne ha$^{-1}$ | 28 | 140 | 60.5 | 32352 L ha$^{-1}$ | 59 | 0 | 0 | 65 |
| | Solid-N | 6.7 tonne ha$^{-1}$ | 28 | 140 | 60.5 | 24.6 tonne ha$^{-1}$ | 37 | 0 | 0 | 207 |
| | Solid-P | 6.7 tonne ha$^{-1}$ | 28 | 140 | 60.5 | 24.6 tonne ha$^{-1}$ | 37 | 0 | 0 | 207 |
| | Control | | | | 60.5 | 0 | 0 | 0 | 0 | 0 |

Table 2.3b. Residual soil nitrate levels, target N application rates and manure application rates to annual (barley) and perennial (grass) treatments in 2010.

| ***Rotation*** | ***Treatment*** | ***Target Yield*** | ***Residual Soil $NO_3$*** | ***Target N*** | ***Alfalfa N*** | ***Manure Appl*** | ***Avail Manure N Appl*** | ***Urea Appl*** | ***Urea N Appl*** | ***Manure $P_2O_5$ Appl*** |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | ----------------***kg ha$^{-1}$***--------------- | | | | ------------------------ ***kg ha$^{-1}$*** --------------------- | | | |
| ***Annual Barley*** | Liquid-N | n/a | 23 | 157 | 0 | 79309 L ha$^{-1}$ | 219 | 0 | 0 | 393 |
| | Liquid-P/Urea | n/a | 24 | 158 | 0 | 0 | 0 | 291 | 134 | 0 |
| | Solid-N | n/a | 28.7 | 151 | 0 | 60.5 tonne ha$^{-1}$ | 124 | 0 | 0 | n/a |
| | Solid-P/Urea | n/a | 23.5 | 157 | 0 | 0 | 0 | 291 | 134 | 0 |
| | Control | | | | 0 | 0 | 0 | 0 | 0 | 0 |
| ***Perennial Grass*** | Liquid-N | 6.7 tonne ha$^{-1}$ | 10.3 | 133 | 0 | 82342 L ha$^{-1}$ | 202 | 0 | 0 | 408 |
| | Liquid-P/Urea | 6.7 tonne ha$^{-1}$ | 11.2 | 135 | 0 | 0 | 0 | 269 | 123 | 0 |
| | Solid-N | 6.7 tonne ha$^{-1}$ | 41.6 | 120 | 0 | 40.7 tonne ha$^{-1}$ | 76 | 0 | 0 | n/a |
| | Solid-P/Urea | 6.7 tonne ha$^{-1}$ | 10.5 | 133 | 0 | 0 | 0 | 269 | 123 | 0 |
| | Control | | | | 0 | 0 | 0 | 0 | 0 | 0 |

Table 2.3c. Residual soil nitrate levels, target N application rates and manure application rates to annual (canola) and perennial (grass) treatments in 2011.

| ***Rotation*** | ***Treatment*** | ***Target Yield*** | ***Residual Soil $NO_3$*** | ***Target N*** | ***Alfalfa N*** | ***Manure Appl*** | ***Avail Manure N Appl*** | ***Urea Appl*** | ***Urea N Appl*** | ***Manure $P_2O_5$ Appl*** |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | ***---------------- kg ha$^{-1}$--------------*** | | | | ***-----------------------kg ha$^{-1}$----------------------*** | | | |
| ***Annual Canola*** | Liquid-N | 2356 kg ha$^{-1}$ | 68.2 | 145.6 | 0 | 31488 L ha$^{-1}$ | 54 | 0 | 0 | 40 |
| | Liquid-P/Urea | 2356 kg ha$^{-1}$ | 36.2 | 163.1 | 0 | 0 | 0 | 275 | 126 | 0 |
| | Solid-N | 2356 kg ha$^{-1}$ | 45.7 | 156.8 | 0 | 61 tonne ha$^{-1}$ | 111 | 0 | 0 | 532 |
| | Solid-P/Urea | 2356 kg ha$^{-1}$ | 45.2 | 157.5 | 0 | 0 | 0 | 244 | 112 | 0 |
| | Control | | | | 0 | 0 | 0 | 0 | 0 | 0 |
| ***Perennial Grass*** | Liquid-N | 6.7 tonne ha$^{-1}$ | 6.4 | 140.4 | 0 | 62998 L ha$^{-1}$ | 94 | 0 | 0 | 81 |
| | Liquid-P/Urea | 6.7 tonne ha$^{-1}$ | 3.2 | 140.4 | 0 | 0 | 0 | 297 | 136 | 0 |
| | Solid-N | 6.7 tonne ha$^{-1}$ | 3.8 | 140.4 | 0 | 74 tonne ha$^{-1}$ | 132 | 0 | 0 | 647 |
| | Solid-P/Urea | 6.7 tonne ha$^{-1}$ | 3.5 | 140.4 | 0 | 0 | 0 | 294 | 135 | 0 |
| | Control | | | | 0 | 0 | 0 | 0 | 0 | 0 |

### 2.3.6 Field and Laboratory Procedures

Soil samples were collected during the growing seasons of 2009, 2010 and 2011 at three times: spring, mid-season, and harvest. Spring soil sampling occurred prior to manure application. Sampling dates are provided in Appendix 2.7.B. Soil was sampled at six depth intervals, 0-15, 15-30, 30-45, 45-60, 60-90 and 90-120 cm, for spring and harvest using a Giddings soil coring machine and at five depths of 0-15, 15-30, 30-45, 45-60, 60-90 cm for midseason using a Dutch auger. Two sub samples were taken from each plot and then composited.

Gravimetric moisture content was determined on all samples after oven drying (105° C). Ammonium-N and nitrate-N were determined on air dried samples in 2009 and field moist samples in 2010 and 2011. Due to probable release of ammonium by dead microbial biomass in dried soil, for accurate ammonium analysis, field moist samples were used in 2010 and 2011. Following soil extraction with 2 M KCl, soil $NO_3$-N was determined by the automated cadmium reduction method while $NH_4$-N was measured by the automated phenate method (Clesceri et al., 1998) using a Technicon auto-analyzer II (Pulse Instrumentation Ltd, Saskatoon, SK). The remaining portion of each soil sample was dried and stored as an archived sample from this site.

Plant samples were collected in each year at mid-season and harvest (i.e. two cuts for forage with complete removal of above ground biomass in both cuts; seed harvest only for annual crops with returned crop residues to soil). Sampling dates are provided in the appendix 2.7.B. In each plot, biomass samples were taken in four randomly-selected areas using a 0.25 $m^2$ quadrat. In 2011, 2.0 $m^2$ were sampled to reduce variability in the biomass data. The plant material was put in cloth bags and hung in a drying room at room temperature (25°C) for 30 d after which the seed was threshed and the seed, straw and grass weights determined. The mid-season and harvest biomass were sub-sampled and finely ground with a mini-ball mill for total N using the wet oxidation technique of Akinremi et al (2003). Because of improbable high P concentration, the 2011 plant biomass samples were reanalyzed by Agvise Laboratories, Northwood, North Dakota in 2013 using a nitric acid/hydrogen peroxide digestion method followed by P determination using a Perkin Elmer 5400 ICP (Jones J. B. 2001).

Leachate was collected from the lysimeters three to five times per year depending on the amount of precipitation during the growing seasons. Sampling dates are provided in Appendix 2.7.B. The leachate was collected from the catch basin by a vacuum pump connected to a hose that ran through one of the extraction tubes. The second extraction tube was opened during leachate collection for equalization of pressure; otherwise, both tubes were covered with caps to prevent rain water from running down the tubes. The total volume of leachate from each lysimeter was recorded and the nitrate-N concentration determined. The procedure for measuring the concentration of nitrate in the leachate was the same as that outlined for the soil samples. Total flux of nitrate was determined by multiplying concentration of nitrate in the leachate by total water flow for each year.

### 2.3.7 Calculation of Nitrogen Use Efficiency

Nitrogen use efficiency was calculated by subtracting N removal for the control from N removal for each treatment in each block and dividing by the available N including sum of soil available N plus available N in manure or fertilizer applied in the year when crop response was measured. Read et al. (2008) calculated the nitrogen use efficiency based on applied total N. However, because treatments received different amounts of total N, in this study we considered available N as the sum of soil available N and added N to calculate nitrogen use efficiency.

### 2.3.8 Statistical Analyses

Analysis of variance (ANOVA) using PROC MIXED procedure (SAS Institute, 2008) was conducted on soil, leachate and biomass to determine significant cropping system, nutrient treatment effects and their interaction in each year. Assumption of normality distribution was checked using PROC UNIVARIATE. Since Shapiro-Wilk's normality test did not show normal distribution for leachate and soil measurements, the log transformed data was used to generate normal distribution of residuals and homogeneity of variance prior to statistical analysis. For total above-ground biomass and their nutrient removals as well as leachate, the statistical model included block (with four levels) as a random factor and treatments (five levels) and cropping systems (two levels) as fixed factors. For soil variables (i.e. nitrate, ammonium and moisture) the statistical model included block (with four levels) as a random factor and treatments (five levels), cropping systems (two levels) and depth (six levels) as fixed factors with depth treated as a repeated measurement. The spatial power [SP(POW)] covariance structure was used in the model for the repeated measures data in which the depth intervals were unequal. Due to variation in manure application by hand a predefined 0.1 significant level was considered (Olatuyi et al. 2012; Zvomuya et al. 2003). Treatment differences were accepted if P<0.1 using Tukey-Kramer method.

## 2.4 RESULTS AND DISCUSSION

### 2.4.1 Total Above-ground Biomass and Nitrogen Uptake

Because of common parameters for two cropping systems, the total above-ground biomass and nitrogen uptake were analyzed as a factorial experiment. In 2009, there was a significant effect of cropping system on biomass as the canola crop produced significantly greater biomass than the grass (Table 2.4). There was no significant effect of manure treatment (P >0.1) on biomass yield or N uptake[1], due in part to the high variability and also the similarity between the various treatments in the first year of the study. Although not statistically significant, the control treatments of both annual and perennial cropping systems produced numerically the smallest biomass yield, and N uptake.

In 2010 there was a significant crop effect, manure effect and crop x manure interaction for total above-ground biomass and nutrient uptakes (Table 2.4). Because of the interaction between crop and manure, the manure treatment effects were tested for each cropping system separately.

In 2011, there was a significant crop effect, manure effect and crop x manure interaction for total above-ground biomass (Table 2.4). Similar to 2009, greater above-ground biomass was produced in the annual plots than in the perennial plots (11450 versus 8612 kg ha$^{-1}$). Because of the statistically significant interaction between crop and manure, the manure treatment effects were tested separately for each cropping system. The above-ground biomass of canola was significantly greater for all manure treatments than the control. Consistent with the overall yield differences for canola and grass, N uptake by the above-ground biomass was greater for canola (172 kg ha$^{-1}$) than grass (148 kg ha$^{-1}$) in 2011 (Table 2.4). The N uptake by the above-ground biomass in treated plots was significantly greater than in the control.

[1] Removal: Nutrient removed in the harvested portion of the crop
Uptake : Total nutrient taken up by the crop (adapted from the Canadian Fertilizer Institute, 2001)

Table 2.4. Above ground plant biomass and nitrogen uptake of canola and grass at harvest

| **Group Means** | | | **Biomass (kg ha$^{-1}$)** | | | **Nitrogen uptake (kg N ha$^{-1}$)** | | |
|---|---|---|---|---|---|---|---|---|
| | | | ***2009*** | ***2010*** | ***2011*** | ***2009*** | ***2010*** | ***2011*** |
| *Crop×Manure* | | | | | | | | |
| Annual | Liquid-N | | 11732 | 9105 a | 11505 a | 141 | 173 a | 182 |
| | Liquid-P/Urea | | 11634 | 7680 ab | 12315 a | 150 | 147 ab | 195 |
| | Solid-N | | 11715 | 8610 ab | 12560 a | 152 | 155 ab | 182 |
| | Solid-P/Urea | | 11916 | 7400 ab | 13715 a | 139 | 153 ab | 200 |
| | Control | | 10397 | 5833 b | 7155 b | 123 | 101 b | 100 |
| Perennial | Liquid-N | | 8195 | 10865 a | 10476 a | 122 | 241 a | 192 |
| | Liquid-P/Urea | | 7983 | 9893 ab | 9164 ab | 133 | 188 ab | 169 |
| | Solid-N | | 7350 | 7838 ab | 7629 ab | 112 | 135 bc | 128 |
| | Solid-P/Urea | | 7293 | 10978 a | 9399 ab | 117 | 232 a | 184 |
| | Control | | 6848 | 7453 b | 6392 b | 104 | 101 c | 83 |
| *Crop* | | | | | | | | |
| Annual[z] | | | 11479 a | 7725 | 11450 | 141 | 146 | 172 a |
| Perennial | | | 7533 b | 9405 | 8612 | 118 | 179 | 147 b |
| | *Manure* | | | | | | | |
| | Liquid-N | | 9964 | 9985 | 10991 | 132 | 207 | 187 a |
| | Liquid-P/Urea | | 9808 | 8786 | 10739 | 141 | 168 | 182 ab |
| | Solid-N | | 9532 | 8223 | 10094 | 132 | 145 | 155 b |
| | Solid-P/Urea | | 9604 | 9188 | 11557 | 128 | 193 | 182 ab |
| | Control | | 8622 | 6642 | 6773 | 113 | 101 | 91 c |
| ***Model effect*** | | ***d.f.*** | ---------------------------------- | | | ***P value*** | -------------------------------- | |
| Crop | | 1 | 0.0138 | 0.0298 | 0.0029 | 0.2172 | 0.0087 | 0.0726 |
| Manure | | 4 | 0.3926 | 0.0017 | 0.0001 | 0.4043 | <0.0001 | <0.0001 |
| Crop×Manure | | 4 | 0.8833 | 0.0233 | 0.0436 | 0.7291 | 0.0072 | 0.1123 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.
[z] Canola: 2009 and 2011; Barley: 2010

#### 2.4.1.1 Canola oilseed and grass yield and nitrogen removal in the first year of study - 2009

All of the canola oilseed and grass yields were greater than the target yields of 1960 kg ha$^{-1}$ for canola and 6.7 tonne ha$^{-1}$ for grass, even on the control plot which was not fertilized (Tables 2.5 and 2.6). The high yield on the control plot may be due to residual nutrients provided by the previous crop of alfalfa and grass. Baseline soil samples taken in May 2009 indicate that the experimental area had high background STP with 65 kg ha$^{-1}$ Olsen extractable P (data not shown). As well, the plow-down of the alfalfa/grass forage may have supplied more N than was credited using the MARC software.

Table 2.5. Canola oilseed yield, nitrogen concentration and removal at harvest in 2009

| | | ***Seed (kg ha$^{-1}$)*** | ***N Conc. (%)*** | ***Nitrogen Removal (kg N ha$^{-1}$)*** |
|---|---|---|---|---|
| Liquid-N | | 2142 | 3.5 | 73.5 |
| Liquid-P | | 2561 | 3.5 | 90.3 |
| Solid-N | | 2682 | 3.4 | 90. 1 |
| Solid-P | | 2323 | 3.5 | 79.3 |
| Control | | 2107 | 3.4 | 71.9 |
| ***Model effect*** | ***d.f.*** | ------------------------------- ***P value***$^{z}$ ---------------------------- | | |
| Manure | 4 | 0.8319 | 0.9592 | 0.8195 |

$^{z}$ Probability value is significant at P <0.1

Table 2.6. Grass yield, nitrogen concentration and removal in 2009

| | | ***Total Yield (kg ha$^{-1}$)*** | ***N Conc. (%)*** | | ***Total N Removal*** |
|---|---|---|---|---|---|
| ***Treatment*** | | ***(1$^{st}$ cut)+(2$^{nd}$ cut)*** | ***1$^{st}$ cut*** | ***2$^{nd}$ cut*** | ***(kg N ha$^{-1}$)*** |
| Liquid-N | | 8195 a | 1.6 | 1.3 | 122.4 |
| Liquid-P | | 7982 ab | 1.8 | 1.4 | 133.6 |
| Solid-N | | 7350 ab | 1.7 | 1.3 | 112.6 |
| Solid-P | | 7292 ab | 1.9 | 1.3 | 117.2 |
| Control | | 6847 b | 1.7 | 1.3 | 104.2 |
| ***Model effect*** | ***d.f.*** | --------------------------------- ***P value***$^{z}$ -------------------------------- | | | |
| Manure | 4 | 0.0427 | 0.4419 | 0.1205 | 0.1034 |

$^{z}$ Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

Although there was no significant effect of manure on canola oilseed yield, the yields ranged from a low of 2107 kg ha$^{-1}$ in the control to a high of 2682 kg ha$^{-1}$ in the solid hog manure N-based treatment (Table 2.5). Achieving the high yield in the solid N-based manure treatment was surprising as solid manures often do not supply adequate available N. However, the high fertility of this site, coupled with the N from the alfalfa/grass plowed down, may have provided sufficient fertility to the canola crop and the solid manure may have provided micro-nutrient or non-fertility benefits such as moisture retention.

Nitrogen removal in the canola seed ranged from 33.6 to 35.3 kg tonne$^{-1}$ (calculated from Table 2.5) and falls within the range of what the Canadian Fertilizer Institute (CFI) reported for western Canada (Table 2.7).

Table 2.7. Nitrogen removal ranges for grass, canola and barley for Western Canada (adapted from the Canadian Fertilizer Institute, 2001).

| ***Crop*** | ***Yield (tonne ha$^{-1}$)*** | ***N Removal*** | |
|---|---|---|---|
| | | ***kg ha$^{-1}$*** | ***kg tonne$^{-1}$*** |
| Grass | 6.7 | 103-127 | 15-19 |
| Canola | 1.96 | 68-83 | 35-42 |
| Barley | 4.3 | 78-95 | 18-22 |

There were significant effects of manure application on total grass yield (Table 2.6). Total grass yields ranged from 6.8 tonne ha$^{-1}$ on the control plots to 8.2 tonne ha$^{-1}$ on the liquid N-based manure plots. The only statistical difference was between the control and the liquid N-based manure application rate (Table 2.6). Again, the control plots achieved the target yield without fertilization, supported by the high background fertility of the experimental site and/or from the previous crop. The grass removed between 14.9 to 16.7 kg N tonne$^{-1}$ (calculated from Table 2.6) which is at the low end of the range reported by the Canadian Fertilizer Institute (Table 2.7). However, the N concentration of 1.3-2 % (Table 2.6) converts to a crude protein of 8 - 12% which is reasonable for grass.

Although not statistically significant ($P<0.1034$) N removal tended to be the highest on the N and P-based liquid manure plots where yields tended to be higher. Nitrogen concentrations in the grass tended to be greater in the first cut of hay than the second (Table 2.6)

### 2.4.1.2 Barley grain and grass yield and nitrogen removal in the second year of study - 2010

In 2010 the barley grain yield (Table 2.8) from the annual application of liquid (4475 kg $ha^{-1}$) and solid (4357 kg $ha^{-1}$) manure at an N based rate was typical of target barley yields for Manitoba and were significantly higher than the control (2942 kg $ha^{-1}$). Yields on the P-based manure plots that received urea in 2010 were not as high (3320 and 3812 kg $ha^{-1}$) as the N-based manure plots and were not statistically different from the control.

Table 2.8. Grain yield, nitrogen concentration and removal of barley at harvest in 2010

| | | *Grain (kg $ha^{-1}$)* | *N Conc. (%)* | *Nitrogen Removal (kg N $ha^{-1}$)* |
|---|---|---|---|---|
| Liquid-N | | 4475 a | 2.3 | 97.3 a |
| Liquid-P/Urea | | 3320 bc | 2.3 | 74.6 ab |
| Solid-N | | 4357 ab | 2.1 | 92.2 a |
| Solid-P/Urea | | 3812 abc | 2.5 | 88.8 a |
| Control | | 2942 c | 2.0 | 60.6 b |
| ***Model effect*** | ***d.f.*** | ------------------------------- | ***P value*** | ------------------------------- |
| Manure | 4 | 0.0015 | 0.1392 | 0.003 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

In 2010, the target N applications were for 3450 kg $ha^{-1}$ wheat. This resulted in more N being applied than required for barley on the plots that received manure and urea (Table 2.3b). As well, the Nova Meter was used to determine the application rate for liquid manure in 2010. It measured the ammonium-N content of the manure and did not account for the organic N in the manure. The use of the Nova Meter resulted in further over-application of N for the liquid manure at the N-based rate. As a result of the over-application of N, lodging of the crop was noted at mid-season and especially at harvest on some of the plots. The yield from the annual application of solid manure at the N based rate was surprisingly large as it was greater than the P-based manure treatments (3320 and 3812 kg $ha^{-1}$for the liquid P and solid P, respectively) that had received urea. The barley grain removed between 20.6 and 23.3 kg N $tonne^{-1}$ (calculated from Table 2.8) which is in the range of what is reported by CFI (Table 2.7). All of the manure treatments, except the liquid P-based/urea treatment, resulted in significantly greater N removal than the control.

Overall, the grass yields in 2010 were better than in 2009 and were above the target yield of 6.7 tonne $ha^{-1}$, even on the unfertilized control. This may be due, in part, to higher precipitation in 2010 (data not shown). In 2010 the grass yields were the highest for the P-based solid manure (applied in 2009) that received urea in 2010 and the N-based liquid manure. These two treatments had biomass yields that were statistically greater than the control (Table 2.9). The N-based liquid manure oversupplied both N and P in 2010 and this may be responsible for the greater yield of this treatment (Table 2.3b). The P-based solid manure applied in 2009 supplied adequate P in 2010 (Table 2.3a) and the urea supplied the required N. The grass yields on the liquid P-based manure that received urea in 2010 and the solid manure applied annually at an N rate were numerically smaller than the other manure treatments, but not statistically different from any of the treatments (Table 2.9). The liquid P-based manure application that received urea in 2010 may not have had adequate P. Nitrogen availability from the solid manure applied annually at an N rate may have limited yield as the manure supplied only 76 kg N $ha^{-1}$ in 2010 (Table 2.3b).

The concentration of N in the grass was greater in 2010 (Table 2.9) than in 2009 (Table 2.6). An N concentration of 2.6% in the N-based liquid manure converts to 16.2 % crude protein, which is high for forage grasses (Popp and German, MAFRI). The N removals for the liquid manure at the N rate (22.3 kg $tonne^{-1}$) and the solid manure at the P rate plus urea (21.2 kg $tonne^{-1}$) were higher than expected using the Canadian Fertilizer Institute range for grass (Table 2.7). Only the control removed less N (13.6 kg $tonne^{-1}$) than estimated using the CFI values. Nitrogen removal from the N-based solid manure was not statistically different from the control, again indicating that there was less N available from the annual solid manure treatment. Previous research has shown that the formula which is used to estimate available N in manure (available N in manure = 75% of $NH_4$ + 25% of organic N) overestimates N availability for solid manures resulting in inadequate N fertilization which can result in decreased N removal and yields (W. Akinremi, personal communication, University of Manitoba, Winnipeg, MB).

Table 2.9. Grass yield, nitrogen concentration and removal in 2010

| ***Treatment*** | | ***Total Yield (kg ha$^{-1}$)*** ***(1$^{st}$ cut)+(2$^{nd}$ cut)*** | ***N Conc. (%)*** ***1$^{st}$ cut*** | ***2$^{nd}$ cut*** | ***Total N Removal (kg N ha$^{-1}$)*** |
|---|---|---|---|---|---|
| Liquid-N | | 10865 a | 2.6 a | 2.0 | 241.8 a |
| Liquid-P/Urea | | 9892 ab | 2.1 ab | 1.8 | 188.4 b |
| Solid-N | | 7837 b | 1.7 bc | 1.8 | 135.3 c |
| Solid-P/Urea | | 10977 a | 2.4 a | 2.1 | 232.8 ab |
| Control | | 7452 b | 1.3 c | 1.6 | 101.3 c |
| ***Model effect*** | ***d.f*** | ---------------------------------- ***P value*** | | | -------------------------------- |
| Manure | 4 | 0.0094 | 0<.0001 | 0.5093 | <0.0001 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

### 2.4.1.3 Canola oilseed and grass yield and nitrogen removal in the third year of study - 2011

Canola oilseed yields were extremely high in 2011 for all treatments except the control (Table 2.10). The control plot, which yielded significantly less than all of the manure treatments, was still above the target yield of 2356 kg ha$^{-1}$.

Table 2.10. Canola oilseed yield, nitrogen concentration and removal at harvest in 2011

| | | ***Seed (kg ha$^{-1}$)*** | ***N Conc. (%)*** | ***Nitrogen Removal (kg N ha$^{-1}$)*** |
|---|---|---|---|---|
| Liquid-N | | 3825 a | 3.7 a | 137.7 a |
| Liquid-P/Urea | | 4122 a | 3.6 a | 143.5 a |
| Solid-N | | 4070 a | 3.4 ab | 136.0 a |
| Solid-P/Urea | | 4440 a | 3.4 ab | 148.9 a |
| Control | | 2600 b | 3.2 b | 80.3 b |
| ***Model effect*** | ***d.f.*** | -------------------------------- ***P value*** | | ----------------------------- |
| Manure | 4 | 0.0024 | 0.0212 | 0.0043 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

The highest yield (4440 kg ha$^{-1}$) was on the P-based solid manure treatment (i.e. plots that received solid manure in 2009 at the P-based rate and then urea-N in 2010 and 2011). The high yields were consistent with visual observations of the crop at mid-season and at harvest. The plants at mid-season exceeded 150 cm in height and it was difficult to enter the plot due to stem density. At harvest, the crop stems were difficult to cut using hand sickles.

For the canola oilseed (Table 2.10), N removals on the manure treatments were all significantly greater than the control. Because of the very high canola yields, the N removals were also high (80.3 to 148.9 kg N $ha^{-1}$). The very high canola yields resulted in low concentrations of N in the oilseed (ranging from 32 kg $tonne^{-1}$ on the control to 37 kg $tonne^{-1}$ on the liquid manure N-based treatment).

Table 2.11. Grass yield, nitrogen concentration and removal at mid-season and harvest in 2011

| ***Treatment*** | | ***Total Yield (kg ha$^{-1}$)*** | ***N Conc. (%)*** | | ***Total N Removal (kg N ha$^{-1}$)*** |
|---|---|---|---|---|---|
| | | ***(1$^{st}$ cut)+(2$^{nd}$ cut)*** | ***1$^{st}$ cut*** | ***2$^{nd}$ cut*** | |
| Liquid-N | | 10476 a | 1.9 ab | 1.8 a | 192.6 a |
| Liquid-P/Urea | | 9163 a | 2.1 a | 1.6 b | 169.8 a |
| Solid-N | | 7628 b | 1.6 bc | 1.9 a | 128.3 b |
| Solid-P/Urea | | 9398 a | 1.9 ab | 1.6 b | 164.5 ab |
| Control | | 6392 b | 1.2 c | 1.5 b | 83.0 c |
| ***Model effect*** | ***d.f*** | ---------------------------------- ***P value*** ---------------------------------- | | | |
| Manure | 4 | <0.0001 | <0.0001 | 0.0002 | <0.0001 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

For the grass biomass, liquid N-based and P-based manure treatments provided sufficient available N for the highest yields and were statistically greater than the control. This is reflected in the N removals for these treatments (Table 2.11). While the N-based solid manure treatment produced one of the greatest yields in the annual plot (Table 2.10), it produced the smallest yield amongst the treated perennial plots. The reason for these differences in performance of the N-based solid manure was probably a lack of incorporation in the perennial plot unlike the regular incorporation in the annual plot following application. Regular tillage and incorporation of the solid manure ensured nitrogen mineralization and available nitrogen supply to the crop. Crude protein for the fertilized 2011 grass crop ranged from 9.8 to 13.0%. The N removals were in the range of what is reported by CFI (Table 2.7) for all treatments (16.8-18.5 kg $tonne^{-1}$) except the control (13.0 kg $tonne^{-1}$) which was lower than reported by CFI.

### 2.4.1.4 Nitrogen Use Efficiency

Nitrogen use efficiency estimates the amount of applied available N that is taken up by the crop. In this study, only above-ground biomass N was considered, therefore total uptake (including N in the roots) is underestimated.

In 2009, there was no significant effect of cropping system or manure treatments on N use efficiency and there was no interaction between crop and manure treatment (Table 2.12).

In 2010, there was a significant crop effect, manure effect and crop x manure interaction on N use efficiency. Because of the interactions in 2010, the manure effects were tested for each cropping system separately. However, nitrogen use efficiency generally was greater for the perennial cropping system in 2010 than the annual cropping system. This is consistent with other research that showed greater nitrogen utilization efficiency in perennial cropping systems than annual systems, thereby making perennials more effective in reducing nitrate movement through the soil profile (Entz et al. 2001). Again, in 2011, the perennial grass showed higher N use efficiency compared to the annual cropping system.

In 2010 and 2011, N use efficiency was highest for the urea and the annual applications of liquid manure at the N-based rate in both cropping systems. Nitrogen use efficiency was always lowest for the annual application of solid manure applied at the N rate. Fifty percent or more of the total N in the liquid manure was in the readily available ammonium form (Table 2.2). Solid manure, on the other hand, typically contains most of the total N in the organic form. It takes time for mineralization of organic N to ammonium and, depending on the C:N ratio of the manure, immobilization of N may also occur. Therefore, it is likely that the method used to estimate available N from organic N in the solid manure overestimated the quantity of N that would be mineralized. Research in Manitoba has shown that the current method of estimating available N from solid manure overestimates available N (W. Akinremi, personal communication, University of Manitoba, Winnipeg, MB).

Table 2.12. Comparison of nitrogen use efficiency (%) in two cropping systems affected by five different treatments

| ***Group Means*** | | | ***2009*** | ***2010*** | ***2011*** |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid-N | | 14.9 | 34.5 a | 82.8 |
| | [z] Liquid-P/Urea | | 18.6 | 29.4 a | 28.8 |
| | Solid-N | | 23.9 | 35.5 a | 46.1 |
| | [z] Solid-P/Urea | | 15.4 | 33.1 a | 44.6 |
| Perennial | Liquid-N | | 20.0 | 65.8 ab | 89.2 |
| | [z] Liquid-P/Urea | | 21.7 | 64 ab | 55.5 |
| | Solid-N | | 12.3 | 28.9 b | 33.3 |
| | [z] Solid-P/Urea | | 9.9 | 98.5 a | 56.0 |
| *Crop* | | | | | |
| Annual | | | 18.2 | 33.1 | 50.6 |
| Perennial | | | 16.0 | 64.3 | 58.5 |
| | *Manure* | | | | |
| | Liquid-N | | 17.5 | 50.2 | 86.0 a |
| | Liquid-P/Urea | | 20.1 | 46.7 | 42.2 b |
| | Solid-N | | 18.1 | 32.2 | 39.7 b |
| | Solid-P/Urea | | 12.6 | 65.8 | 50.3 b |
| ***Model effect*** | | ***d.f.*** | -------------------------------------- | ***P value*** | ----------------------------------- |
| Crop | | 1 | 0.8239 | 0.0249 | 0.3793 |
| Manure | | 3 | 0.9047 | 0.0482 | 0.0036 |
| Crop×Manure | | 3 | 0.417 | 0.0141 | 0.1505 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

[z] The NUE is expressed for manure in 2009 and for urea in 2010 and 2011.

### 2.4.2 Soil Ammonium

**The distribution of soil ammonium-nitrogen during the 2009 growing season**

The soil ammonium results were most interesting in 2009 and are presented herein. In 2010 and 2011, soil ammonium was always below 5 mg $kg^{-1}$. The results for 2010 and 2011 are provided in Appendix 2.7.C.

The ammonium concentration in both cropping systems was unusually high in the spring of 2009 (before treatments were applied) and ranged from 10 to 20 mg $kg^{-1}$ in the top 15 cm layer (Figure 2.2a and b). This was more than double the amount of nitrate in the same soil layer in 2009 (compare Figures 2.2a and b with Figure 2.3a and b). The high ammonium level was mainly due to the presence of alfalfa and grass on the field for 3 years prior to onset of the experiment that was chemically killed prior to soil sampling. The cool spring weather probably reduced nitrification of ammonium to nitrate. It has been shown that the bacteria that nitrify ammonium are sensitive to cool soil temperature (Sahrawat 2008). However, soil samples in 2009 were dried prior to KCl extraction resulting in probable increase of ammonium level. Results from another study in soil ecology laboratories of the Department of Soil Science, University of Manitoba, has shown that measuring ammonium in dried soil is in error as it overestimates $NH_4^+$ probably due to release of ammonium by dead microbial biomass in dried soil (D. Flaten, personal communication, University of Manitoba, Winnipeg, MB).

By mid-season in 2009, the ammonium concentration in the entire soil profile approached the 3 to 4 mg $kg^{-1}$ range that was expected for this soil type. The decline in the concentration of ammonium from spring to mid-season could be due to plant removal, immobilization, or nitrification as a result of a warmer soil.

By harvest, the ammonium concentrations increased to the levels they were in the spring including the control. This is an indication of the mineralization potential of this soil. The high ammonium concentration in the control plots indicate that this was likely due to the chemical killing of the alfalfa on the perennial plots and the chemical killing and incorporation of the alfalfa and grass on the annual plots.

For all three sampling periods, the distribution of ammonium in the perennial cropping system was similar to that of the annual cropping system in 2009. This was not surprising as the two systems received similar treatment in the spring of 2009.

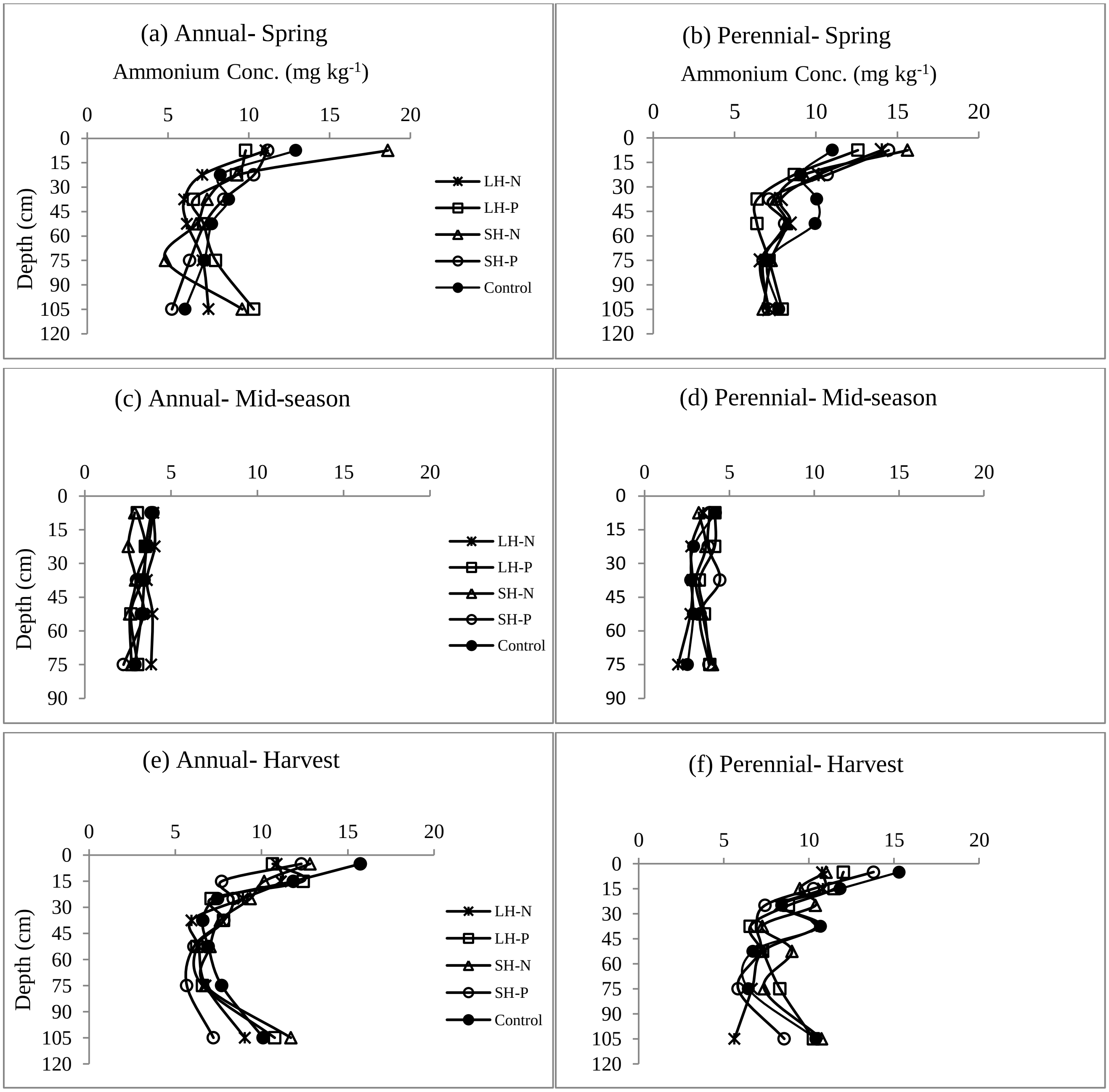


Fig. 2.2. Concentration of ammonium-N in the soil profile during the 2009 growing season

### 2.4.3 Soil Nitrate

#### 2.4.3.1 Soil nitrate distribution during the 2009 growing season

Soil samples were taken to a depth of 120 cm to determine if the manure treatments resulted in nitrate leaching and to compare nitrate leaching between the two cropping systems. Although soil sampling does not provide an accurate estimate of the quantity of nitrate leached because nitrate could have moved beyond the sampling depth (Olatuyi et al. 2012) accumulation of nitrate with depth is an indirect measure of leaching (Miller et al. 2011).

There was a significant crop effect on the average concentration of nitrate in the soil profile at mid-season and harvest (Table 2.13b). At both sampling periods, the annual cropping system resulted in greater nitrate concentrations than the perennial system. The annual plots may have had more nitrate than the perennial due to the alfalfa residues that were mineralized following incorporation. Thomsen and Christensen (1998) found that incorporation of ryegrass may result in nitrate leaching in the second year after it is ploughed under. The nitrate concentrations in manure treatments were not significantly different from those in the control. The nitrate concentration in the control plot of the annual and perennial systems were similar to those for the manured plots indicating that the killed alfalfa and grass provided sufficient N for the crops.

In spring 2009 prior to the application of manure, the concentration of nitrate-N in the top 15 cm of soil was between 3 to 5 mg $kg^{-1}$ in all plots (Figure 2.3a, b). By mid-season, following the application of manure, nitrate concentration in the top 15 cm increased to 24 to 30 mg $kg^{-1}$ in the annual plots and 16 to 24 mg $kg^{-1}$ in the perennial plots (Figure 6c, d). By harvest, due to crop uptake of N and leaching loss, the concentration of nitrate in the top 15 cm decreased to between 5 and 8 mg $kg^{-1}$ in the annual plots and less than 3 mg $kg^{-1}$ in the perennial plots (Figure 2.3e, f). For both cropping systems, by harvest, the concentrations of nitrate decreased with depth (Table 2.13b).

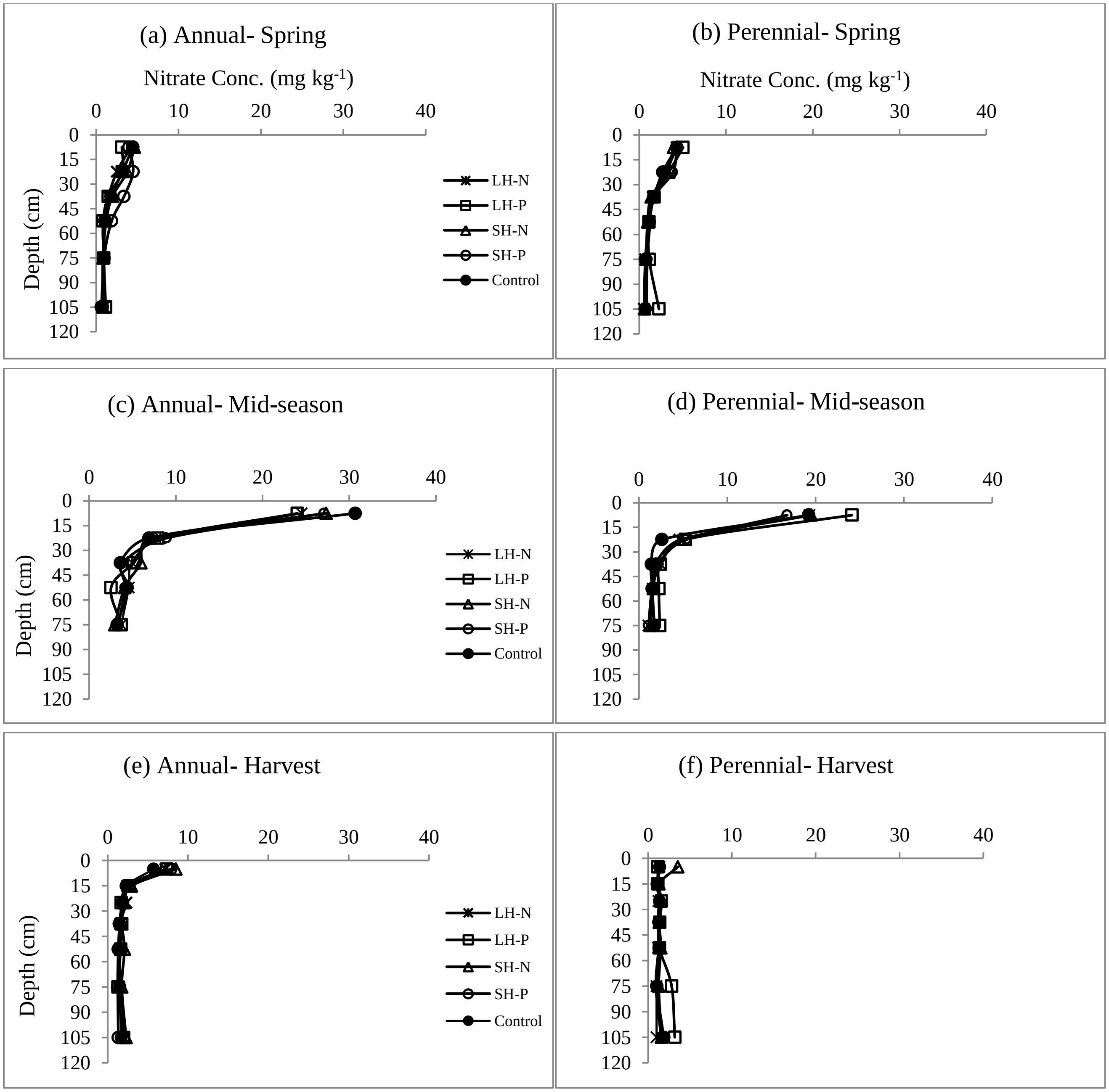


Fig. 2.3. Concentration of nitrate-N in the soil profile during the 2009 growing season

Table 2.13a. Effect of manure treatments and cropping system on the vertical distribution of ammonium (mg $kg^{-1}$) in 2009, 2010 and 2011.

| | | *2009* | | | *2010* | | | *2011* | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Spring | Mid-season | Harvest | Spring | Mid-season | Harvest | Spring | Mid-season | Harvest |
| ***Crop Means*** | | | | | | | | | | |
| Annual | | 8.07 | 3.47 | 8.48 | 0.93 | 0.78 | **0.54** | 0.94 | **0.89** | **0.73** |
| Perennial | | 8.60 | 3.44 | 8.86 | 1.03 | 0.80 | **0.69** | 0.96 | **1.29** | **0.61** |
| ***Treatment Means*** | | | | | | | | | | |
| Liquid-N | | 8.00 | 3.43 | 8.26 | 1.02 | 0.79 | 0.59 | 0.95 | 1.17 | 0.70 |
| Liquid-P/Urea | | 8.16 | 3.58 | 8.77 | 0.90 | 0.77 | 0.59 | 0.89 | 0.92 | 0.64 |
| Solid-N | | 8.65 | 3.22 | 9.15 | 1.03 | 0.77 | 0.58 | 1.09 | 0.94 | 0.63 |
| Solid-P/Urea | | 8.25 | 3.67 | 7.94 | 0.98 | 0.85 | 0.64 | 0.89 | 1.23 | 0.68 |
| Control | | 8.62 | 3.39 | 9.29 | 0.96 | 0.77 | 0.66 | 0.95 | 1.14 | 0.69 |
| ***Depth*** | | | | | | | | | | |
| 0-15 | | **12.9** | 3.84 | **11.52** | **1.14** | **1.53 a** | **0.88** | **1.51a** | **2.07** | **0.95** |
| 15-30 | | **9.14** | 3.51 | **9.51** | **1.01** | **0.90 b** | **0.59** | **0.95 ab** | **1.34** | **0.68** |
| 30-45 | | **7.53** | 3.40 | **7.63** | **0.94** | **0.64 c** | **0.60** | **0.89 b** | **1.05** | **0.68** |
| 45-60 | | **7.54** | 3.28 | **6.97** | **0.94** | **0.59 c** | **0.57** | **0.84 b** | **0.85** | **0.62** |
| 60-90 | | **6.78** | 3.29 | **6.76** | **0.86** | **0.60 c** | **0.56** | **0.82 b** | **0.80** | **0.59** |
| 90-120 | | **7.39** | --- | **9.25** | **0.85** | --- | **0.55** | **0.87 b** | **0.78** | **0.57** |
| ***Model effect*** | ***d.f.*** | | | | | ***P value*** | | | | |
| Crop | 1 | 0.75 | 0.89 | 0.73 | 0.38 | 0.85 | **<0.01** | 0.85 | **0.01** | **0.06** |
| Manure | 4 | 0.99 | 0.84 | 0.90 | 0.95 | 0.98 | 0.82 | 0.70 | 0.57 | 0.93 |
| Depth | 5 | **<0.01** | 0.10 | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **0.00** | **<0.01** | **<0.01** |
| Crop×Man | 4 | 0.99 | **0.08** | 0.99 | 0.86 | 0.19 | 0.85 | 0.28 | 0.95 | 0.66 |
| Crop×Dep | 5 | 0.83 | 0.92 | 0.94 | 0.10 | 0.49 | **0.07** | 0.79 | **0.05** | **0.05** |
| Man×Dep | 20 | **0.04** | 0.66 | **0.09** | **0.09** | 0.91 | 0.59 | 0.90 | 0.27 | 0.54 |
| Crop×Man×Dep | 20 | 0.18 | 0.23 | 0.15 | 0.90 | 0.68 | 0.71 | 0.98 | 0.74 | **0.08** |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

Table 2.13b. Effect of manure treatments and cropping system on the vertical distribution of nitrate (mg $kg^{-1}$) in 2009, 2010 and 2011.

| | | 2009 | | | 2010 | | | 2011 | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Spring | Mid-season | Harvest | Spring | Mid-season | Harvest | Spring | Mid-season | Harvest |
| ***Crop Means*** | | | | | | | | | | |
| Annual | | 1.68 | **6.59** | **2.19** | **3.67** | **3.14** | **4.10** | **4.28** | **1.95** | **1.08** |
| Perennial | | 1.62 | **3.35** | **1.38** | **0.96** | **1.00** | **0.42** | **1.18** | **1.40** | **0.46** |
| ***Treatment Means*** | | | | | | | | | | |
| Liquid-N | | 1.49 | 4.78 | 1.62 | 1.90 | **2.11** | **2.51** | 2.65 | 1.93 | 0.84 |
| Liquid-P/Urea | | 1.79 | 5.22 | 1.90 | 2.18 | **1.99** | **1.59** | 2.00 | 1.70 | 0.86 |
| Solid-N | | 1.56 | 4.68 | 2.03 | 1.70 | **1.37** | **1.82** | 2.85 | 1.63 | 0.72 |
| Solid-P/Urea | | 1.87 | 4.59 | 1.56 | 1.94 | **2.57** | **1.68** | 2.26 | 1.94 | 0.66 |
| Control | | 1.56 | 4.27 | 1.63 | 1.72 | **1.18** | **1.05** | 1.69 | 1.18 | 0.50 |
| ***Depth*** | | | | | | | | | | |
| 0-15 | | **4.21** | **22.88** | **2.58 a** | **3.12** | **6.09** | **2.59** | **11.44** | **5.65** | **2.14** |
| 15-30 | | **3.29** | **5.82** | **1.68 bc** | **1.67** | **2.71** | **1.50** | **4.53** | **3.10** | **1.08** |
| 30-45 | | **1.76** | **3.08** | **1.45 bc** | **1.85** | **1.34** | **1.59** | **2.09** | **1.48** | **0.70** |
| 45-60 | | **1.14** | **2.54** | **1.45 bc** | **1.98** | **0.98** | **2.07** | **1.52** | **1.06** | **0.47** |
| 60-90 | | **0.87** | **2.21** | **1.36 c** | **1.46** | **0.82** | **1.47** | **1.02** | **0.82** | **0.38** |
| 90-120 | | **0.84** | --- | **1.77 b** | **1.36** | --- | **1.18** | **0.79** | **0.92** | **0.45** |
| ***Model effect*** | ***d.f.*** | ***P value*** | | | | | | | | |
| Crop | 1 | 0.82 | **<0.01** | **0.00** | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **0.00** | **<0.01** |
| Manure | 4 | 0.86 | 0.79 | 0.66 | 0.50 | **<0.01** | **0.00** | 0.14 | **0.05** | 0.14 |
| Depth | 5 | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **<0.01** | **<0.01** |
| Crop×Man | 4 | 0.85 | 0.57 | 0.92 | 0.41 | **0.00** | **0.05** | 0.66 | 0.82 | 0.95 |
| Crop×Dep | 5 | 0.86 | **0.02** | **<0.01** | **0.00** | **0.00** | **0.00** | **0.00** | **0.00** | **0.00** |
| Man×Dep | 20 | **0.09** | **0.06** | 0.74 | 0.79 | **0.01** | 0.78 | 0.47 | **0.08** | **0.06** |
| Crop×Man×Dep | 20 | 0.86 | 0.27 | 0.15 | 0.91 | 0.63 | 0.76 | 0.33 | **0.00** | 0.87 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

### 2.4.3.2 Soil nitrate distribution during the 2010 growing season

In 2010, there was a significant crop effect on the average concentration of nitrate in the soil profile in all three sampling periods (Table 2.13b), where the annual cropping system had greater nitrate concentrations than the perennial system. There was also a significant manure treatment effect on the average concentration of nitrate in the soil profile at mid-season and harvest (Table 2.13b). When analyzed for each cropping system separately, there were no manure treatment differences in the perennial plots; however, there were manure treatment differences at mid-season and at harvest in the annual plots (data not shown). At mid-season, the P-based/Urea treatment and the N-based liquid manure treatment resulted in greater soil nitrate than the control (Table 2.13b). The N-based solid manure treatment had similar nitrate-N as the control. The P-based treatments received urea in 2010 to meet the N needs of plants, a readily available source of N compared to solid manure which needs time to mineralize and release nitrate afterward. Liquid manure also contained a significant quantity of readily available N as ammonium. By harvest, the N-based manure treatments had more nitrate throughout the profile than the control but were no different than the P-based/Urea treatment (Table 2.13b).

In 2010, the pattern of soil nitrate was different for the annual and perennial cropping systems (Figure 2.4). In the spring, the annual plots had higher soil nitrate in the top 15 cm than the perennial plots (Figure 2.4 a, b). The concentration of nitrate across the annual plots ranged from 6 to 8 mg kg$^{-1}$, similar to what was measured in the fall of 2009 (Figure 2.4a versus 2.3e). There was a bulge of nitrate-N around the 45 to 60 cm depth in the annual plots in the spring of 2010 with concentrations that were similar to those in the top 15 cm. This bulge is an indication of nitrate movement possibly due to excess residual nitrogen from the previous year. Late planting date of annual crop resulted in downward movement of nitrate in annual plots compare to significant growth in the perennial crop by the time the annual crop was sown. Unlike the annual plots, the nitrate concentrations in the perennial system was small (<4 mg kg$^{-1}$) in the spring with no apparent nitrate bulge (Figure 2.4b). Generally, on the Canadian Prairies most of nitrate leaching occurs in the spring before the plants can remove the nitrogen that is mineralized during fall and early spring after snowmelt (Enns 2004). As well, Sun et al. (2008) showed that accumulated residual soil nitrate can be readily leached during wet seasons, such as the spring in Manitoba. The differences in the concentration of soil nitrate between the perennial and annual cropping systems in this study can also be explained by the differences between long and short season crops and the density of their root systems. The perennial grass root system was dense and well established prior to onset of the experiment and would have been able to take up soil nitrate in early spring until fall. The annuals, on the other hand, were not seeded until mid-June, would have taken additional weeks to establish their root systems, had sparser root systems and were harvested earlier in the fall. As with results obtained by Entz et al. (2001) showing no downward movement of nitrate in the unfertilized system, there was no evidence of nitrate leaching in control plots under perennial cropping system but nitrate movement and leaching was observed in the control plots of annual cropping system particularly in the spring (Figure 2.4a). Campbell et al. (1993) concluded that insufficient fertility may lead to poor crop growth and consequently increase nitrate leaching. The control plot in this study yielded less barley than the fertilized plots.

The bulge of nitrate that was observed in the spring at about the 45 cm depth has disappeared by mid-season (early August, see soil sampling dates in Appendix 2.7.B). This could have been due to nitrate leaching or plant uptake of N. Mid-season biomass sampling occurred in late July (Appendix 2.7.B) and indicated that the above-ground barley plant biomass had removed from 74 kg N ha$^{-1}$ (control) to 150 kg N ha$^{-1}$ (liquid N-based) (data not shown).

By September (post-harvest), nitrate concentration in the top 15 cm ranged from 3 to 9 mg kg$^{-1}$ in the annual plots and about 1 mg kg$^{-1}$ in the perennial (Figure 2.4e and f). Nitrate concentrations in the annual plots decreased with depth but showed the reappearance of a nitrate bulge (Figure 2.4e) centered at 60 cm. Again this nitrate bulge was an indication of downward nitrate movement in the soil. Other studies also reported maximum concentration of nitrate at the 30 to 60 cm depth of soil profile after eight years of manure application (Miller et al. 2011 and Olson et al. 2009).

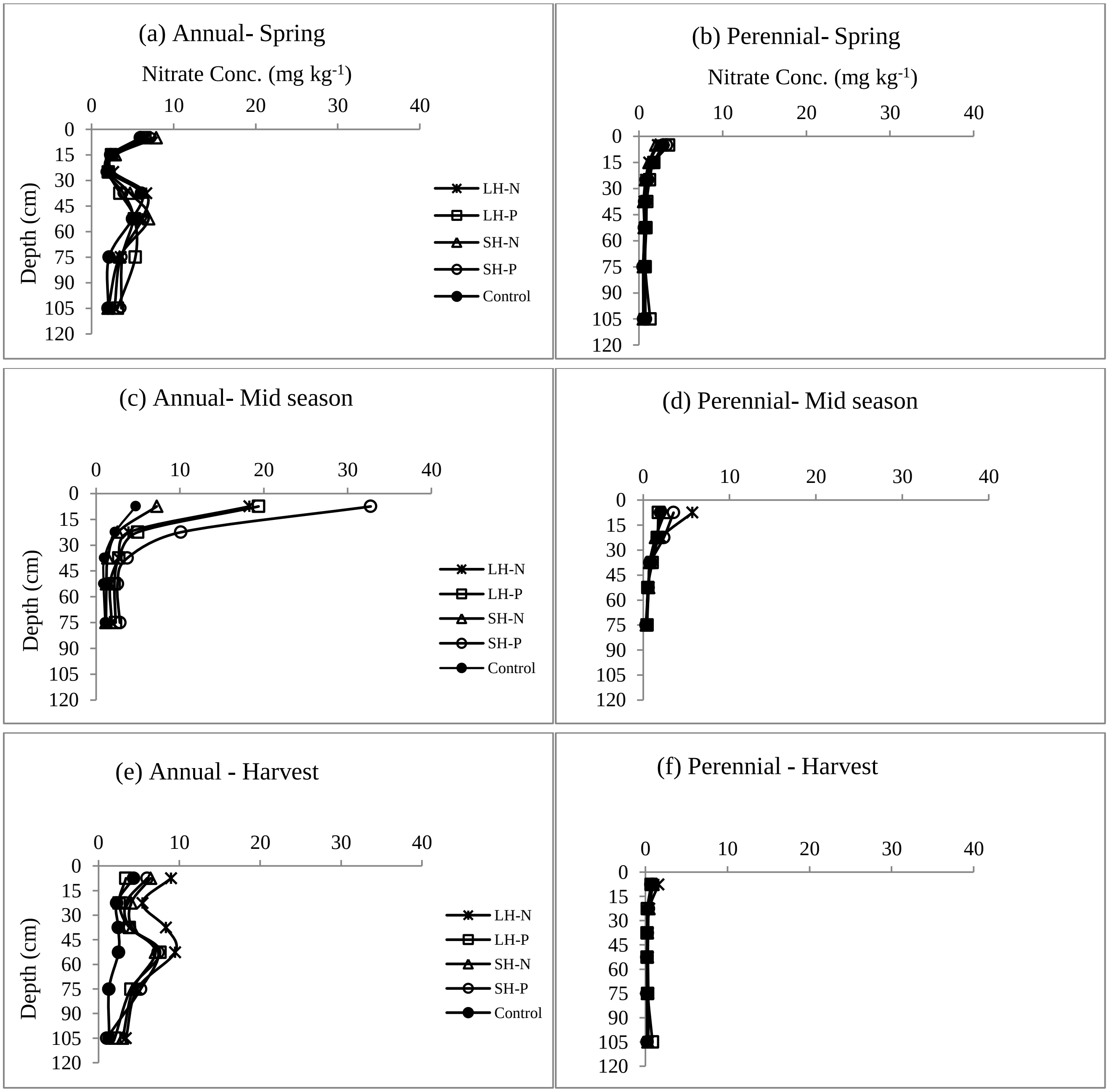


Fig. 2.4. Concentration of nitrate-N in the soil profile during the 2010 growing season

### 2.4.3.3 Soil nitrate distribution during the 2011 growing season

In 2011, there was a significant crop effect on the average concentration of nitrate in the soil profile in the spring, mid-season and harvest (Table 2.13b) where the annual cropping system resulted in greater nitrate concentrations than the perennial system.

In the spring of 2011, the concentration of nitrate within the top 15 cm depth was between 8 to 20 mg $kg^{-1}$ in the annual plots and 5 to 23 mg $kg^{-1}$ in the perennial plots (Figure 2.6a and b). For some treatments, this was more than double what was measured in the fall of 2010 and was an indication of the mineralization potential, especially where solid manure was applied. Mineralization of organic N and nitrification of $NH_4^+$ resulted in higher nitrate concentrations in the top 15 cm.

In the annual plots the bulge of nitrate that was observed at harvest in 2010 had disappeared by the spring of 2011 possibly due to movement of nitrate below our sampling depth by the high amount of precipitation in the wet fall (August and September) of 2010 ( Figure 2.5) and snow melt in the early spring of 2011 (Enns 2004).

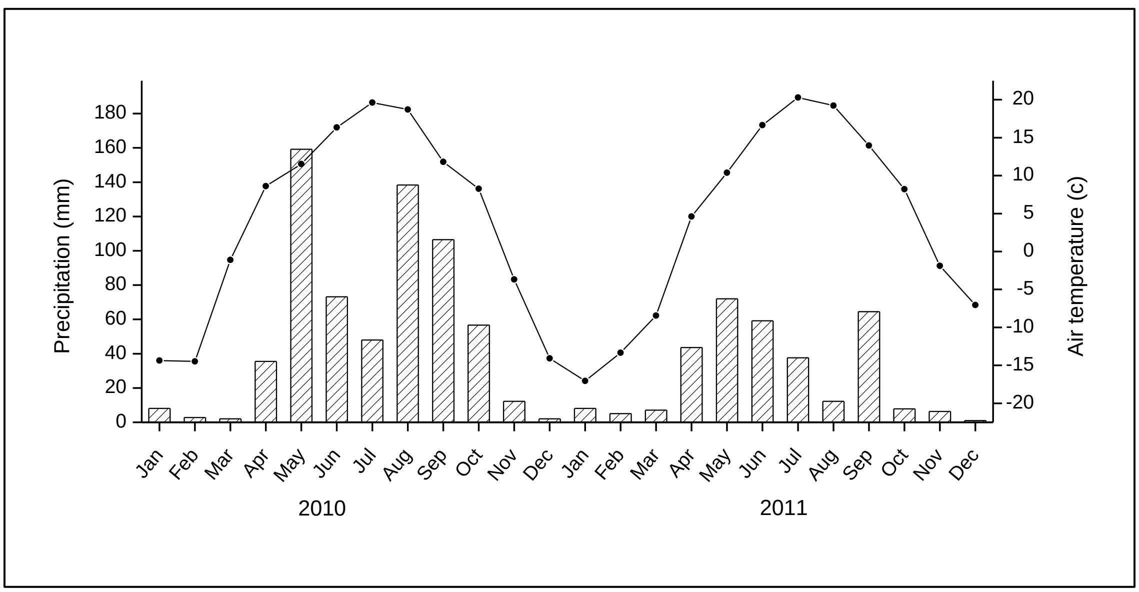


Fig. 2.5. Monthly precipitation (bars) and monthly mean air temperatures (line chart) during two years of study, based on Environment Canada's weather data (Carman station)

The amount of nitrate-N in the soil profile at mid-season was less than in the spring due to plant uptake and also some nitrate leaching (Figure 2.6). In the annual plots, the nitrate content of soil increased slightly at the 90-120 cm depth by mid-season, indicating that the decrease in soil nitrate from spring to mid-season was primarily a result of plant uptake but also that there was probably some downward movement of nitrate. Similar to what we observed during the two previous growing seasons, soil nitrate levels were lowest at harvest, likely due to plant uptake and reduced mineralization rate as a result of cooler soil temperature (Woodard et al. 2003). The annual cropping system (Figure 2.6e) resulted in greater soil nitrate concentrations than the perennial system at harvest (Figure 2.6f).

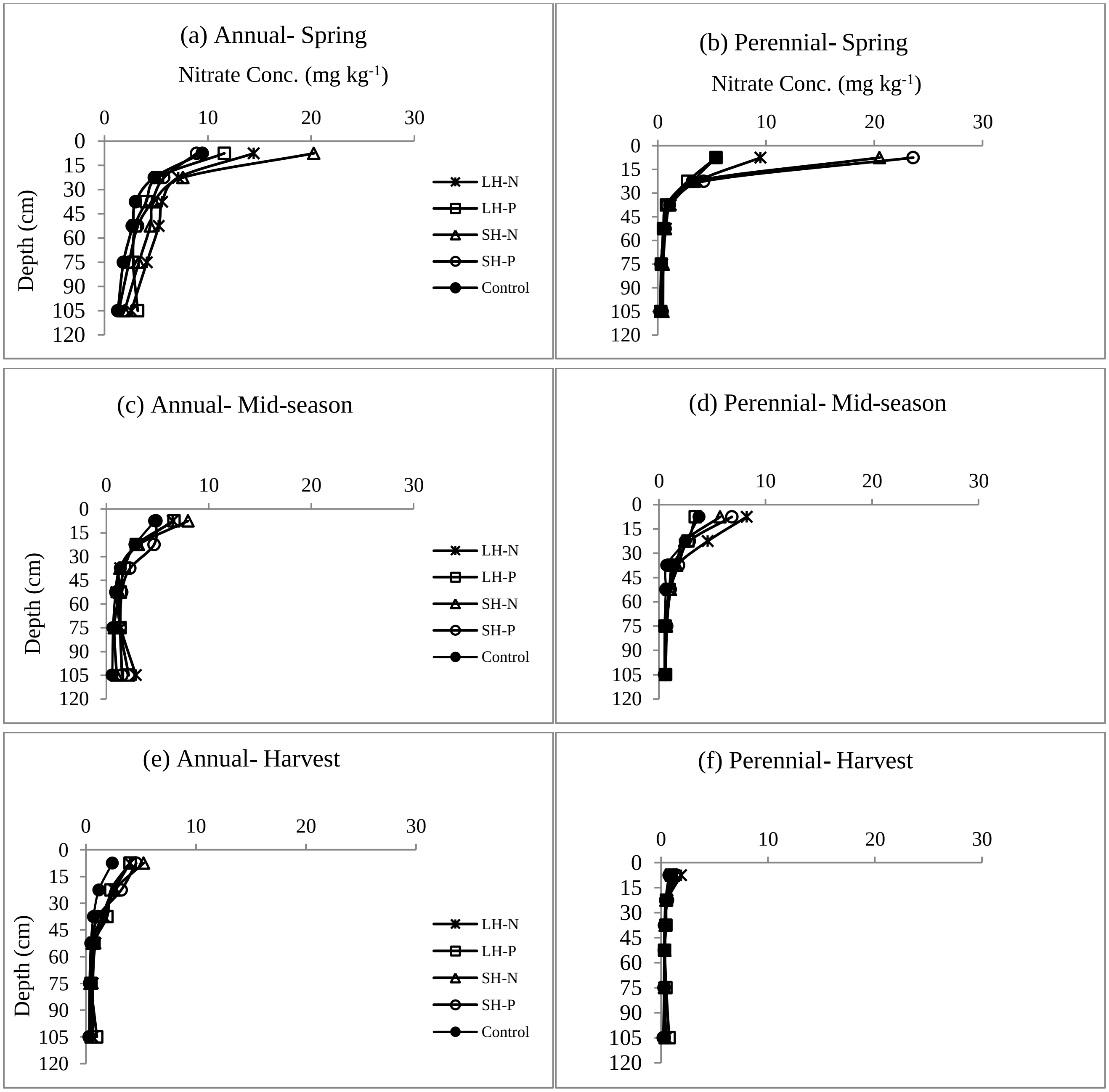


Fig. 2.6. Concentration of nitrate-N in the soil profile during the 2011 growing season

Overall, during three years of the study, greater concentrations of total nitrate were observed in the soil profile of the annual cropping system than the perennial system. This could not be explained by the crop N uptake data, because the canola crops in 2009 and 2011 removed more N in the aboveground biomass than the grass. Therefore, greater storage of N in the root system of the grasses as well as other soil or plant processes may be responsible for the decreased risk of nitrate leaching under the perennial system. Although organic matter and organic N were not measured, microbial activity and organic matter content may have been greater in the perennial plots than in annual plots due to greater root density of grass and reduced oxidation due to a lack of tillage. Higher immobilization of nitrate in the soil organic matter pool may have led to reduced nitrate leaching under perennials than annuals. A similar observation was made by Lipiec et al. (2011) who concluded that the smaller nitrate leaching in 35 year-old apple orchard soil than in conventionally tilled field was due to greater C:N ratio in soil and greater immobilization of nitrogen in the apple orchard. Huggins et al. (2001) reported lower concentration of nitrate in subsurface drains under alfalfa plots than under continuous corn and corn-soybean rotation (3 mg $L^{-1}$ vs 32 and 24 mg $L^{-1}$ respectively). They concluded that perennial grasses can cause significant reduction in nitrate losses in drainage water and considering these crops in rotation may improve nitrogen use efficiency and water quality.

### 2.4.4 The Distribution of Soil Water during the 2009, 2010 and 2011 Growing Season

Although we took soil samples in spring 2009, the spring 2009 soil moisture data were lost. At mid-season and at harvest in 2009, there were no significant differences in soil moisture between cropping systems and nutrient treatments down to a depth of 90 cm (Figure 2.7 and Table 2.14). A bulge of water at the 45 cm depth showed significant downward movement of water (Table 2.14). By harvest, volumetric water content in the top 90 cm depth was about 10% on both annual and perennial plots, an indication of the ability of both crops to use available water in the soil. At the 90-120 cm depth soil moisture increased.

In the spring of 2010 the soil moisture content increased compared to the values at harvest in 2009 due to snowmelt and precipitation during fall and early spring. A bulge of water in the spring and at harvest at the 45 cm depth is consistent with the bulge of nitrate observed in the spring and at harvest in 2010 (Figure 2.8 and Figure 2.7). This indicates that nitrate-N moved with the water within the soil profile.

Table 2.14. Treatments effect, cropping system and their interactions on the vertical distribution of soil moisture in 2009, 2010 and 2011

| ***Group Means*** | | ***2009*** | | | ***2010*** | | | ***2011*** | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Spring[z] | Mid-season | Harvest | Spring | Mid-season | Harvest | Spring | Mid-season | Harvest |
| ***Model effect*** | ***d.f.*** | ------------------------------------- ***P value*** ---------------------------------- | | | | | | | | |
| Crop | 1 | --- | 0.2959 | 0.9433 | **0.0057** | 0.9091 | **0.0161** | 0.5021 | 0.7092 | 0.2375 |
| Manure | 4 | --- | 0.900 | 0.6683 | 0.9861 | 0.8672 | 0.8909 | 0.1423 | 0.9262 | 0.3129 |
| Depth | 5 | --- | **0.0878** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** |
| Crop×Man | 4 | --- | 0.8129 | 0.7124 | 0.3530 | 0.9161 | 0.9978 | 0.9467 | 0.3943 | 0.1905 |
| Crop×Dep | 5 | --- | 0.4612 | 0.5362 | 0.1151 | 0.2760 | **0.0612** | 0.9923 | 0.2041 | 0.2266 |
| Man×Dep | 20 | --- | **0.0470** | 0.7825 | 0.6521 | 0.7759 | 0.9873 | **0.0756** | 0.3458 | **0.0338** |
| Crop×Man×Dep | 20 | --- | 0.1474 | 0.1796 | **0.0833** | 0.5889 | 0.4328 | 0.9882 | 0.2490 | **0.0843** |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.
[z] Missing soil moisture data

The amount of water in the annual plots was greater than that in the perennial plots in spring and fall of 2010 (Table 2.14) likely because perennials absorbed water during fall and early spring.

In 2011, the pattern of soil water distribution was similar to what we observed in 2010 (Figure 2.9 and Figure 2.8, respectively). Although less water was leached from the perennial plots than the annual plots (Table 2.16) which indicates greater uptake of water by perennials due to longer growing season, there were no differences between soil water content of perennial plots and annual plots for all three sampling times in 2011 (Table 2.14). At harvest 2011, both perennial and annual plots showed significant greater soil moisture content at the 90-120 cm depth compare to the other depths. Although, 3-way interaction was significant, it only represented a small proportion of the total variance compared to the depth and manure main effects.

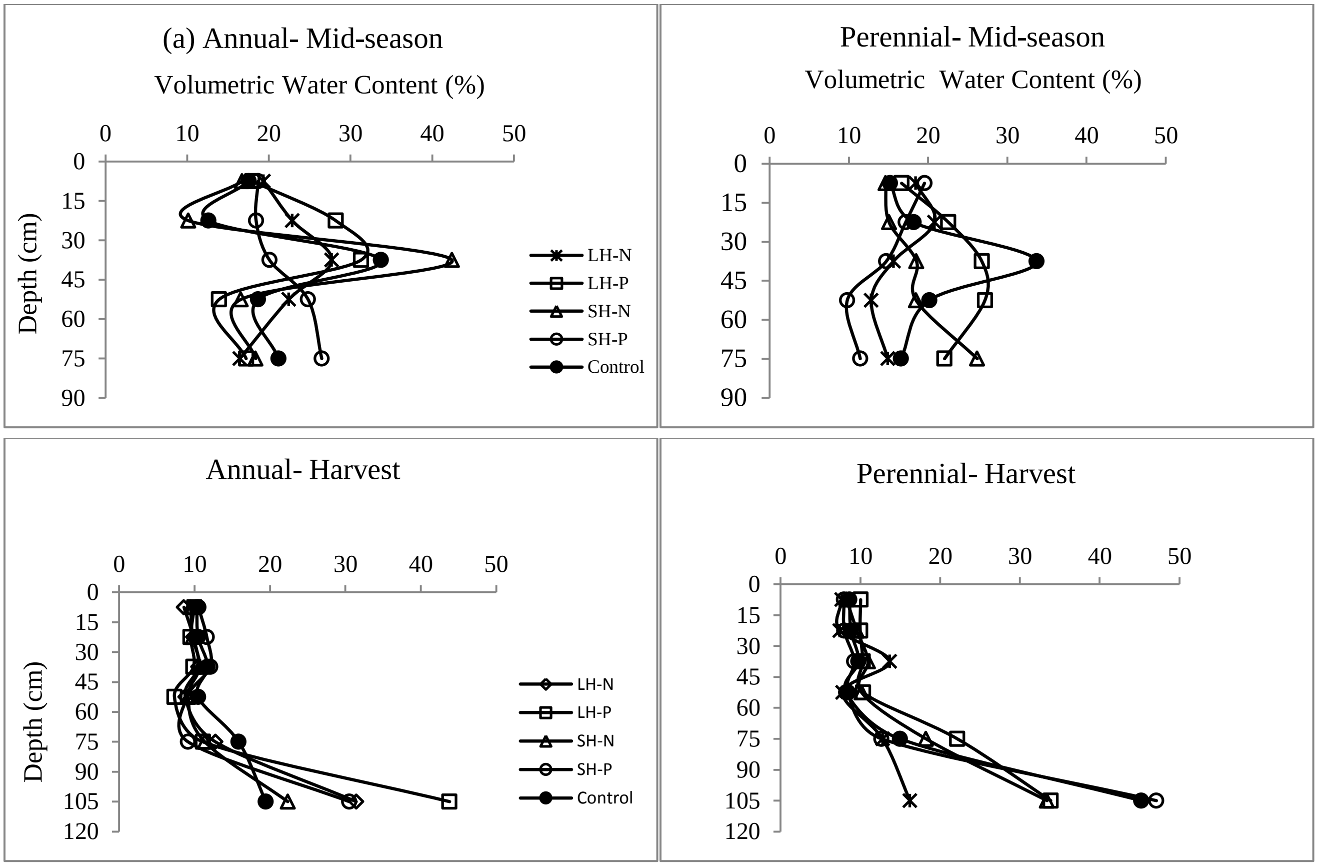


Fig. 2.7. Soil water distribution during the growing season of 2009.

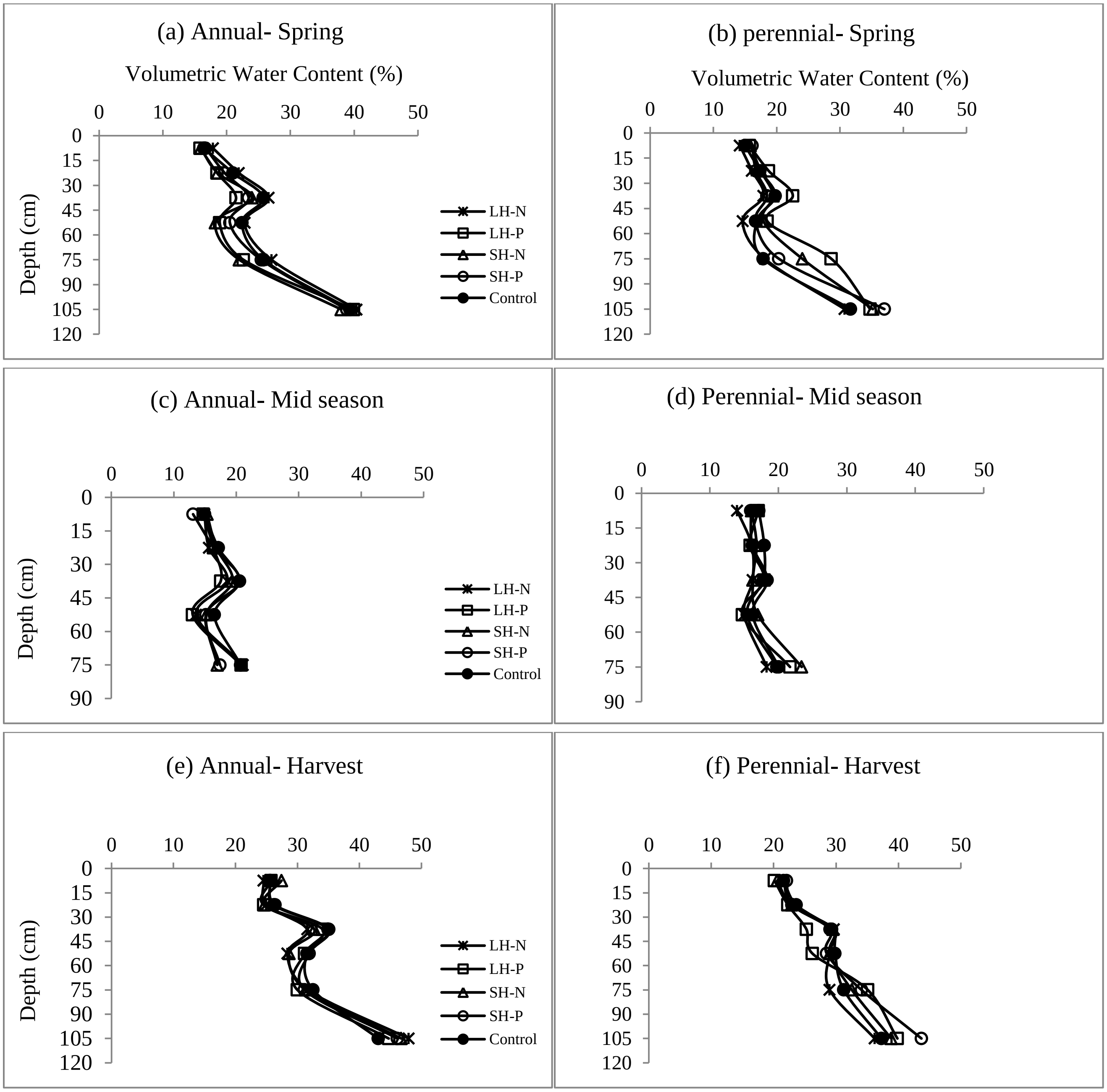


Fig. 2.8. Soil water distribution during the growing season of 2010.

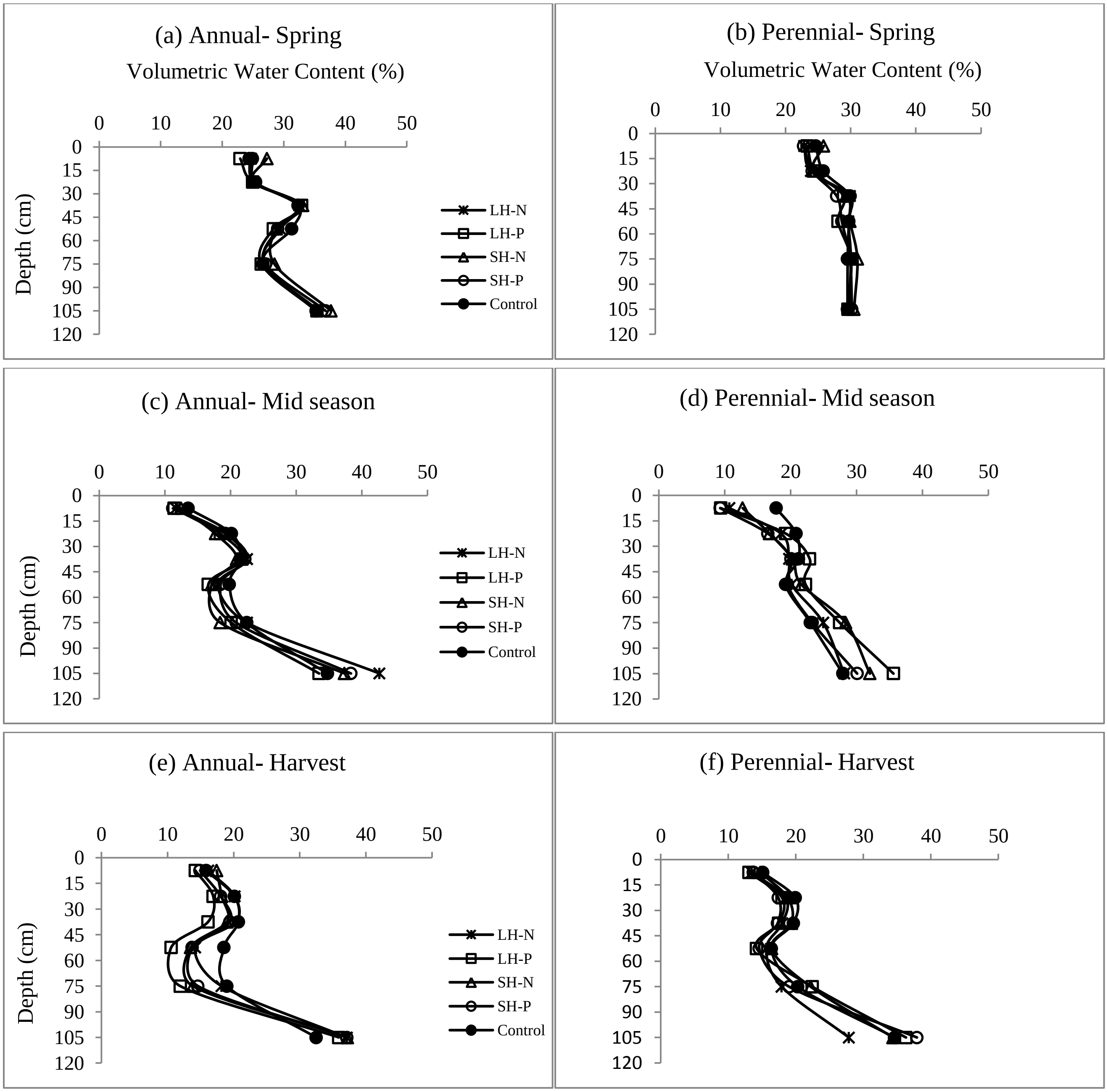


Fig. 2.9. Soil water distribution during the growing season of 2011.

### 2.4.5 Leachate

#### 2.4.5.1 Amount of water and nutrient leached below the root zone in 2009

Lysimeters were installed in each plot to measure nutrient leaching and to compare the use of field core lysimeters with traditional soil profile sampling. In order to make treatment comparisons or to compare the results from the soil samples with leachate from the lysimeters, conditions in the lysimeters must reflect conditions within the plots. In this study, the soil conditions within the lysimeters reflected the soil conditions in the plots, the lysimeters received the same nutrient treatments, and incorporation of nutrients and seeding was simulated by hand.

Although the amount of water that was lost below the root zone of the annual crop in 2009 appeared to be greater than the water lost below the perennial crop (Table 2.15), the difference was not statistically significant. Deeper root activity and higher water use by perennial crops usually decreases the water available for leaching (Entz et al. 2001). Greater transpiration by perennial crops also reduces the water available for leaching (Mueller et al. 2005 and Hatfield et al. 2001). In the 2009 growing season, there was no significant difference between the canola and grass crops in the amounts of nitrate that was lost below the root zone (Table 2.15). This is consistent with the results of the soil samples. The lack of treatment differences in nitrate and water loss may also have been due, in part, to the killing and plowing down of alfalfa and perennial grasses on all plots prior to treatment applications.

Table 2.15. Amounts of water, nitrate leached from annual and perennial plots in 2009

| ***Group Means*** | | ***d.f.*** | ***Water (cm)*** | ***FWMC (N) (mg $L^{-1}$)*** | ***Nitrate (kg N $ha^{-1}$)*** |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid-N | | 10.83 | 0.21 | 0.18 |
| | Liquid-P | | 6.73 | 0.14 | 0.09 |
| | Solid-N | | 8.90 | 0.17 | 0.13 |
| | Solid-P | | 11.58 | 0.27 | 0.27 |
| | Control | | 7.31 | 0.15 | 0.10 |
| Perennial | Liquid-N | | 5.23 | 0.12 | 0.03 |
| | Liquid-P | | 4.37 | 0.21 | 0.08 |
| | Solid-N | | 7.85 | 0.15 | 0.10 |
| | Solid-P | | 7.47 | 0.05 | 0.04 |
| | Control | | 9.12 | 0.13 | 0.11 |
| *Crop* | | | | | |
| Annual | | | 9.07 | 0.18 | 0.14 |
| Perennial | | | 6.81 | 0.12 | 0.06 |
| | *Manure* | | | | |
| | Liquid-N | | 8.03 | 0.16 | 0.08 |
| | Liquid-P | | 5.55 | 0.17 | 0.08 |
| | Solid-N | | 8.37 | 0.16 | 0.12 |
| | Solid-P | | 9.53 | 0.12 | 0.10 |
| | Control | | 8.22 | 0.14 | 0.11 |
| ***Model effect*** | | ***d.f.*** | -------------------------- | ***P value***$^{z}$ | -------------------------------- |
| Crop | | 1 | 0.3101 | 0.5216 | 0.3986 |
| Manure | | 4 | 0.3267 | 0.9967 | 0.9961 |
| Crop×Manure | | 4 | 0.3746 | 0.7499 | 0.6941 |

FWMC: Flow Weighted Mean Concentration
$^{z}$ Probability value is significant at P <0.1

### 2.4.5.2 Amounts of water and nutrient leached below the root zone in 2010

The amount of precipitation received during the growing season in 2010 was 596 mm, which was 158% of the 30 year normal growing season precipitation. The large amount of precipitation led to a leaching loss in the range of 18 to 33 cm in the perennial and 23 to 36 cm in the annual cropping system (Table 2.16), although there was no statistical difference between these amounts.

There was a significant difference between the annual and perennial cropping systems in the flow weighted mean concentration (FWMC) of N and the amount of nitrate lost (Table 2.16). The amount of nitrate lost below the root zone of the annual cropping systems ranged from 40 kg N $ha^{-1}$ in the control to 61 kg N $ha^{-1}$ in liquid manure P-based treatment, whereas for the perennial cropping system, the amount lost was less than 1 kg N $ha^{-1}$ in all treatments. The downward movement of nitrate in the soil profile of the annual plots was also apparent in the 2010 spring and harvest soil nitrate analyses (Figures 2.4a and 2.4e). Similar results have been reported by Nikiema et al. (2013) who reported 23, 48 and 51 kg N $ha^{-1}$of leached N from the low, intermediate and high rates of liquid manure treatment, respectively, using field core lysimeters under annual cropping system. Furthermore, Toth and Fox (1998) reported 81 and 55 kg N $ha^{-1}$ of $NO_3$-N leached from the root zone of continuous corn and 9 kg N $ha^{-1}$ of $NO_3$-N leached from the root zone of alfalfa during first and second year of their study.

The negligible loss of nitrate in the lysimeters of the perennial cropping system was consistent with the soil nitrate data and may be explained by the high yields, N concentration and N removal by the grasses in the main plot area (Table 2.9). Russelle et al. (2001) used perennial forages to take up excess nitrate from soil in order to improve water quality. The perennial grass in this study was a long season crop with well established, dense and deep root systems that could take up nitrogen from early spring into the fall.

The magnitude of the N losses from the barley plots may have been elevated due to a number of factors including being a shorter season crop, plowdown of alfalfa and grass that supplied more N than expected, over-application of N to the fertilized plots due to a change in crop, the combination of early spring leaching and late seeding date. The spring and summer of 2010 were wet and there was a delay in field work. The barley in this study was planted in mid-June and, in contrast to the perennial grasses, would have taken additional weeks to fully establish its root system and was harvested in September. In 2010, most of the leachate was collected in the spring prior to manure application and seeding. It is likely that delayed uptake of residual soil nitrate resulted in greater spring leaching of nitrate under the annual cropping system than the perennial. The large N loss from the barley control plot (40 kg N $ha^{-1}$) was surprising considering that no manure N or fertilizer had been applied. The N in this plot may have originated from the alfalfa/grass crop that was chemically killed and plowed down prior to the onset of the experiment. Although high (42.6 to 60.8 kg N $ha^{-1}$), the N leached from the manure treatments on the annual plots were not statistically different from the control. In 2010, barley was planted rather than wheat for early maturity. The target yield of barley (4417 kg $ha^{-1}$) required 162 kg N $ha^{-1}$. However, 179 kg N $ha^{-1}$ was applied based on calculations for 3450 kg $ha^{-1}$ of wheat which was originally planned.

The same results were presented by Olatuyi et al. (2012) using the dual tracer technique (Bromide and $^{15}N$-labelled fertilizer) for $NO_3$-N leaching. these authors demonstrated that the distribution of $NO_3$-N in the soil cannot be used solely to infer the relative loss of $NO_3$-N between treatments or to deduce $NO_3$-N leaching over time.

Table 2.16. Amounts of water, nitrate leached from annual and perennial plots in 2010

| ***Group Means*** | | ***Water (cm)*** | ***FWMC (N) (mg L$^{-1}$)*** | ***Nitrate (kg N ha$^{-1}$)*** |
|---|---|---|---|---|
| *Crop×Manure* | | | | |
| Annual | Liquid-N | 22.58 | 20.75 | 49.51 |
| | Liquid-P/Urea | 32.06 | 21.77 | 60.80 |
| | Solid-N | 29.06 | 17.94 | 47.01 |
| | Solid-P/Urea | 36.16 | 14.72 | 42.55 |
| | Control | 30.50 | 15.68 | 40.14 |
| Perennial | Liquid-N | 18.79 | 0.81 | 0.15 |
| | Liquid-P/Urea | 29.66 | 0.27 | 0.16 |
| | Solid-N | 18.16 | 0.05 | 0.04 |
| | Solid-P/Urea | 30.39 | 1.38 | 0.37 |
| | Control | 32.82 | 0.06 | 0.05 |
| *Crop* | | | | |
| Annual | | 30.07 | 18.17 a | 47.51 a |
| Perennial | | 25.97 | 0.51 b | 0.12 b |
| | *Manure* | | | |
| | Liquid-N | 20.68 | 10.78 | 2.72 |
| | Liquid-P/Urea | 30.87 | 11.02 | 3.12 |
| | Solid-N | 23.61 | 8.99 | 1.44 |
| | Solid-P/Urea | 33.28 | 8.05 | 4.00 |
| | Control | 31.66 | 7.87 | 1.45 |
| ***Model effect*** | ***d.f.*** | ---------------------------- | ***P value$^{z}$*** | ------------------------------ |
| Crop | 1 | 0.5430 | 0.0014 | <0.0001 |
| Manure | 4 | 0.4228 | 0.2462 | 0.8509 |
| Crop×Manure | 4 | 0.9217 | 0.1578 | 0.8760 |

FWMC: Flow Weighted Mean Concentration
$^{z}$ Probability value is significant at P <0.1

### 2.4.5.3 Amounts of water and nutrient leached below the root zone in 2011

Similar to results obtained in 2010, there was a statistically significant cropping system effect on the flow weighted mean concentration (FWMC) for nitrogen and the amount of nitrate that was leached below the root zone in 2011 (Table 2.17). When the cropping systems were analyzed separately, in the annual cropping system, liquid manure treatments (liquid-N and liquid-P/urea) showed greater FWMC than control. The amount of nitrate loss through leaching under the annuals was 80 to 250 times greater than perennials. Higher soil nitrate levels were also measured for the annual cropping system than the perennial. When the cropping systems were analyzed separately, there were no significant differences in the amount of nitrate lost among manure treatments, probably due to spatial variability. Similar to what we observed in 2010, the solid P-based/urea treatment in the annual cropping system produced the smallest nitrate leaching of 23 kg ha$^{-1}$ which was less than one-half of what was lost from the liquid P-based/urea treatment (50 kg $ha^{-1}$). Both P-based treatments received manure in 2009 and urea in 2010 and 2011. It is possible that the solid manure treatment immobilized more nitrogen than their liquid manure counterpart due to the high C: N ratio of the straw in the solid manure. Although not statistically different, nitrate loss from the annual N-based liquid manure treatment was numerically the greatest. Liquid hog manure has more of the total N in the ammonium form. As well, liquid manure has a lower C:N ratio and will immobilize less N than solid manure. Beckwith et al. (1998) showed a loss of 26.7% of slurry-N and 9% of broiler litter-N during four years of study. They reported the difference in N loss was as a result of diversity in C:N ratio of slurry (5:1) and broiler litter (10.6:1).

The amount of water leached from the soil profile showed no statistical difference between annuals and perennials (Table 2.17), although the amount lost from the perennial system tended to be numerically less than the annual system, similar to what was observed in the previous two years. This showed that the differences between nitrate losses in the perennial and the annual cropping systems were mostly related to the high concentration of nitrate in the soil profile of the annual system. Thomsen (2005) showed that the quantity of drained water had a minor effect compared to the concentration of nitrate in influencing nitrate leaching losses.

Table 2.17. Amounts of water, nitrate leached from annual and perennial plots in 2011

| ***Group Means*** | | ***Water (cm)*** | ***FWMC (N) (mg $L^{-1}$)*** | ***Nitrate (kg N $ha^{-1}$)*** |
|---|---|---|---|---|
| *Crop×Manure* | | | | |
| Annual | Liquid-N | 21.86 | 28.90 a | 59.98 |
| | Liquid-P/Urea | 32.31 | 18.88 ab | 50.52 |
| | Solid-N | 22.11 | 17.26 bc | 34.12 |
| | Solid-P/Urea | 35.71 | 9.83 bc | 22.72 |
| | Control | 31.33 | 8.66 c | 24.01 |
| Perennial | Liquid-N | 17.42 | 3.71 a | 0.55 |
| | Liquid-P/Urea | 27.96 | 0.26 a | 0.21 |
| | Solid-N | 23.63 | 0.51 a | 0.24 |
| | Solid-P/Urea | 27.18 | 0.16 a | 0.28 |
| | Control | 29.14 | 0.08 a | 0.16 |
| *Crop* | | | | |
| Annual | | 28.66 | 16.71 | 35.50 a |
| Perennial | | 25.07 | 0.95 | 0.26 b |
| | *Manure* | | | |
| | Liquid-N | 19.64 | 16.30 | 5.76 |
| | Liquid-P/Urea | 30.14 | 9.57 | 3.29 |
| | Solid-N | 22.87 | 8.89 | 2.87 |
| | Solid-P/Urea | 31.45 | 5.00 | 2.51 |
| | Control | 30.24 | 4.38 | 1.99 |
| ***Model effect*** | ***d.f.*** | ---------------------------- | ***P value***$^{z}$ | ----------------------------- |
| Crop | 1 | 0.5472 | <0.0001 | 0.0002 |
| Manure | 4 | 0.6280 | 0.0019 | 0.8107 |
| Crop×Manure | 4 | 0.9788 | 0.0157 | 0.9812 |

FWMC: Flow Weighted Mean Concentration
$^{z}$ Probability value is significant at P <0.1

As mentioned previously, the amount of aboveground biomass and nitrogen uptake by annuals was greater than perennials in 2011 (Table 2.4) so it was expected that the nitrate lost from annual plots would be less than perennials. However, this was not the case as the amount of nitrate lost was greater in annuals than perennials. In this study, the belowground biomass was not measured. Despite the greater aboveground biomass yield of canola than perennial grasses in 2009 and 2011, greater leached nitrate was measured from lysimeters from the annual cropping system in all three years of this study. It is possible that the belowground biomass of perennial grasses was greater than that of canola with greater uptake of nitrate in the roots of the perennial crop that was not taken into consideration in this study. There could be additional mechanisms, such as microbial biomass differences, that also contributed to the lack of nitrate leaching from the perennial system. This observation warrants further study.

### 2.4.5.4 Leached water and nitrate above the control treatment

Subtracting the amount of leached nitrate-N and water in the control plots from each treatment, provides an estimate of the effect of treatment on the amount of nitrate and water leaching (Table 2.18). Negative values for the amount of water indicate that less water was lost from the treatment plot than the control and is an indication that water uptake was greater than the control.

In all three years, the manure and fertilizer treated perennial plots consistently lost less water by leaching than the control perennial plot. This is consistent with higher yields and, by implication, greater water use by the grass crops that received manure and urea in 2010 and 2011. In 2010 and 2011, less water appeared to be lost by leaching from the N-based manure treatments (both solid and liquid) than the control treatments of the annual cropping system. Greater uptake of water often correlates with greater yield. For example, in 2010, N-based manure treatments were the only treatments that out yielded the control (Table 2.8). Campbell et al. (1993) demonstrated that the amount of water in the soil profile is inversely related to the fertility level and crop yield. Poor fertility and subsequent low yield is often associated with greater soil water content due to reduced root growth and leaf area.

Table 2.18. Leached nitrate and water above the control treatment during the three years of study

| | ***Annual plots*** | | | | | |
|---|---|---|---|---|---|---|
| | ***Amount of water (cm)*** | | | ***Amount of nitrate (kg N ha$^{-1}$)*** | | |
| ***Treatments*** | ***2009*** | ***2010*** | ***2011*** | ***2009*** | ***2010*** | ***2011*** |
| Liquid-N | 3.52 | -7.92 | -9.47 | 0.08 | 9.37 | 35.97 |
| Liquid-P/Urea | -0.58 | 1.56 | 0.98 | -0.01 | 20.66 | 26.51 |
| Solid-N | 1.59 | -1.44 | -9.22 | 0.03 | 6.87 | 10.11 |
| Solid-P/Urea | 4.27 | 5.66 | 4.38 | 0.17 | 2.41 | -1.29 |
| | ***Perennial plots*** | | | | | |
| | ***Amount of water (cm)*** | | | ***Amount of nitrate (kg N ha$^{-1}$)*** | | |
| ***Treatments*** | ***2009*** | ***2010*** | ***2011*** | ***2009*** | ***2010*** | ***2011*** |
| Liquid-N | -3.89 | -14.03 | -11.72 | -0.08 | 0.1 | 0.39 |
| Liquid-P/Urea | -4.75 | -3.16 | -1.18 | -0.03 | 0.11 | 0.05 |
| Solid-N | -1.27 | -14.66 | -5.51 | -0.01 | -0.01 | 0.08 |
| Solid-P/Urea | -1.65 | -2.43 | -1.96 | -0.08 | 0.32 | 0.12 |

Reduced loss of water did not necessarily reduce nitrate loss (Table 2.18). In 2009, nitrate leaching under the treated plots was no different than the control. Treatment differences may not be apparent due to the elimination of alfalfa and grass from the plots prior to the onset of the experiment and the subsequent mineralization of N. For the annual plots in 2010 and 2011, with the exception of the solid P-based manure/urea treatment in 2011, manure and urea applications always resulted in more nitrate leaching than the control. Nitrate leaching under the perennial plots, however, was no different than the control.

## 2.5 CONCLUSIONS

Forage grasses can provide a significant quantity of N when chemically killed and plowed under. In 2009, high background soil P and N from the alfalfa/grass forage masked treatment differences and supplied sufficient fertility to reach the target yields of 1960 kg $ha^{-1}$ for canola and 6.7 tonne $ha^{-1}$ for grass without fertilization. Nitrate levels in the control plot of the annual cropping system in 2009 were no different than the plots that received manure, supporting that the alfalfa residues mineralized N following incorporation. The alfalfa/grass forage crop appears to have supplied more N than was credited using MAFRI's MARC software.

The traditional formula for estimating available N in manure (available N in manure = 75% of $NH_4$ + 25% of organic N) over-estimates N availability from solid hog manure. The 2010 and 2011 grass yields, N removals and N use efficiency from the annual applications of solid hog manure at N-based rates suggest that solid hog manure does not mineralize enough N to meet the crop's N requirement in the first year following application.

Annual cropping systems presented a higher risk of nitrate leaching than the perennial cropping systems on the soils in this study. Both the soil samples and the lysimeters showed low levels of nitrate leaching in the perennial plots and higher levels of nitrate leaching in the annual plots. This was probably due to longer life span of perennials, early season uptake of nitrogen and having dense, deep root system compared to annual crops. However, late planting of the annual crops in this experiment may have exaggerated the impact of the difference in growth, nitrogen and water uptake for these two cropping systems, especially during early spring, when a large proportion of leaching occurs in Manitoba.

Although nitrate leaching may be reduced through nutrient management, this study suggests that it cannot be eliminated in annual cropping systems altogether as nitrate leaching occurred in the canola and barley control plots where no manure or fertilizer was applied. Solid manures with lots of bedding may reduce the risk of nitrate leaching. Application of solid manure at the P-based manure application rate followed by urea in subsequent years, reduced the risk of nitrate leaching over the course of the rotation, likely due to immobilization of N with the addition of the straw in the manure.

Both soil samples and lysimeters provide valuable information regarding nitrate leaching. Soil samples throughout the profile show the pattern of nitrate distribution up to the sampling depth. Unfortunately, nitrate that has moved below the sampling depth is not captured. Lysimeters capture all of the nitrate and water moving through the profile. Scaling up the concentration of nitrate in the lysimeters to the field scale, however, requires that the conditions in the lysimeters mirror the conditions in the field with respect to soil properties, crop management and crop yields.

## 2.6 ACKNOWLEDGMENTS

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## 2.7 APPENDICES

### Appendix 2.7.A: Hibsin Series (HIN)

The Hibsin series consists of moderately well drained Orthic Black Chernozem soils developed on a mantle (40 to 90 cm) of weakly to moderately calcareous, shallow, coarse loamy to sandy (LVFS VFS, FSL, FS, LFS), lacustrine deposits over moderately calcareous, uniform, deep, clayey (SiC, C), lacustrine deposits. These soils occur in upper positions of very gentle slopes on undulating landscapes and have moderately rapid over slow permeability, moderate surface runoff, and a low water table during the growing season. Hibsin soils are non-eroded, non-stony, and non-saline. They have medium available water holding capacity, high organic matter content, and medium natural fertility. Native vegetation includes aspen, oak, shrubs and prairie grasses. The majority of these soils are currently cultivated for crop production.

In a representative profile the solum is approximately 40 cm thick. The profile is characterized by a very dark gray to dark gray Ah horizon, 15 to 25 cm thick, a brown $B_m$ horizon, 20 to 30 cm thick, a $C_{ca}$ horizon, 8 to 13 cm thick and a $C_k$ horizon. The profile is usually developed entirely in the coarser material. Hibsin soils occur in close association with Rosebank and Layland soils. They are similar to Hochfeld soils by having an Orthic Black profile in coarse loamy deposits but differ by having a clayey substrate. Hibsin soils were previously mapped as associates of the Altona (light) Association in the Winnipeg-Morris (1953) soil report.

**Appendix 2.7.B**

Table. 2.7.B. Schedule of soil, biomass and leachate sampling during the 2009, 2010 and 2011

| Field Operation | ***2009*** | | ***2010*** | | ***2011*** | |
|---|---|---|---|---|---|---|
| ***Soil sampling*** | | | | | | |
| Spring | May 21 | | April 27 and May 11 | | June 14 | |
| Mid-season | August 18,19 | | August 4, 5 | | August 11 | |
| Harvest | September 30 and October 6 | | September 16, 17 | | September 26, 30 | |
| ***Biomass sampling*** | | | | | | |
| | Perennial | Annual | Perennial | Annual | Perennial | Annual |
| Mid-season | July 3 | August 18 | June 30 | July 29 | July 6 | August 11 |
| Harvest | September 28 | September 28 | September 14 | September14 | August 29 | September 20 |
| ***Leachate sampling*** | | | | | | |
| | June 25, August 7, September 28 and November 17 | | June 4, July 14, August 24, September 30 and November 2 | | May 16, June 9, July 6 and October 11 | |

**Appendix 2.7.C:**

### The distribution of soil ammonium-nitrogen during the 2010 growing season

Unlike nitrate, which showed a bulge that was centered at 45 cm depth in the spring of 2010 (Figure 2.4a), ammonium concentration was small in the spring of 2010 and was about 1 mg $kg^{-1}$ throughout the soil profile in the annual cropping system (Figure 2.7a). The amount of ammonium in the perennial cropping system was slightly greater than in the annual cropping system at the soil surface and similar to the annual system in the remainder of the soil profile (Figure 2.7b). In general, ammonium levels in the perennial system near the soil surface were significantly greater than in the annual system (Table 2.13). This is surprising considering that the nitrate levels in the annual system were much greater than in the perennial system. This may be an indication that the perennial system interfered with the nitrification process leading to the accumulation of ammonium. This speculation will also explain why there was less nitrogen leaching in the perennial than in the annual. This speculation warrants further study to confirm if this was the case.

### The distribution of soil ammonium-nitrogen during the 2011 growing season

The results obtained in 2011 were similar to that in 2010 with very small levels of ammonium in the spring, slightly increasing at mid-season following manure and fertilizer additions and diminishing at harvest due to crop uptake and nitrification (Figure 2.8). The greater concentration of ammonium that was observed in 2011 was also observed at mid-season in 2011. It is possible that the incorporation of manure in the annual plots played an important role in promoting nitrification resulting in less concentration of ammonium in annual plots. In general, ammonium did not show a trend of leaching in the soil (as compared to nitrate) possibly due to the high cation exchange capacity of soil and absence of limitation for rapid nitrification.

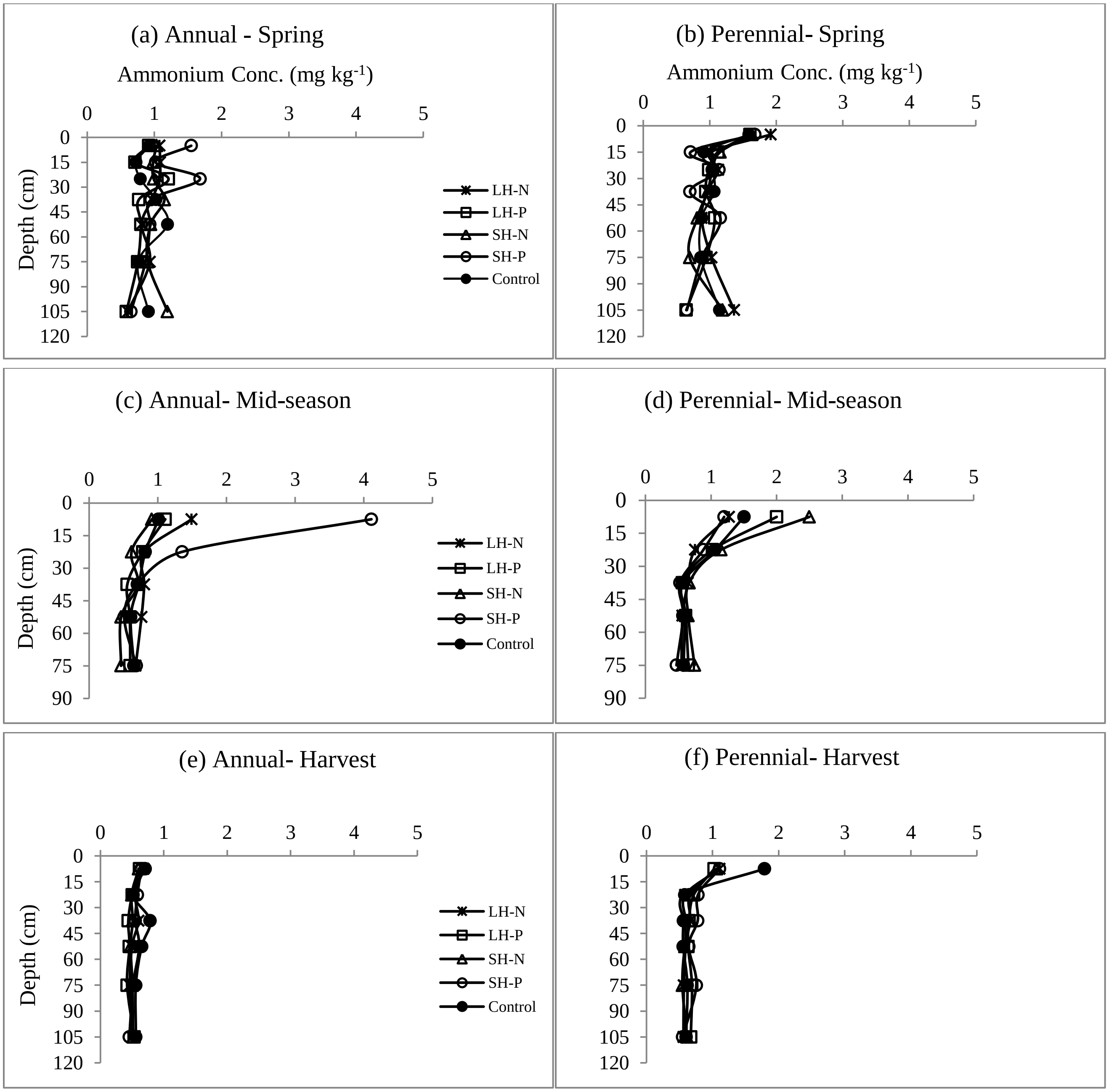


Fig. 2.7.C1. Concentration of ammonium-N in the soil profile during the 2010 growing season

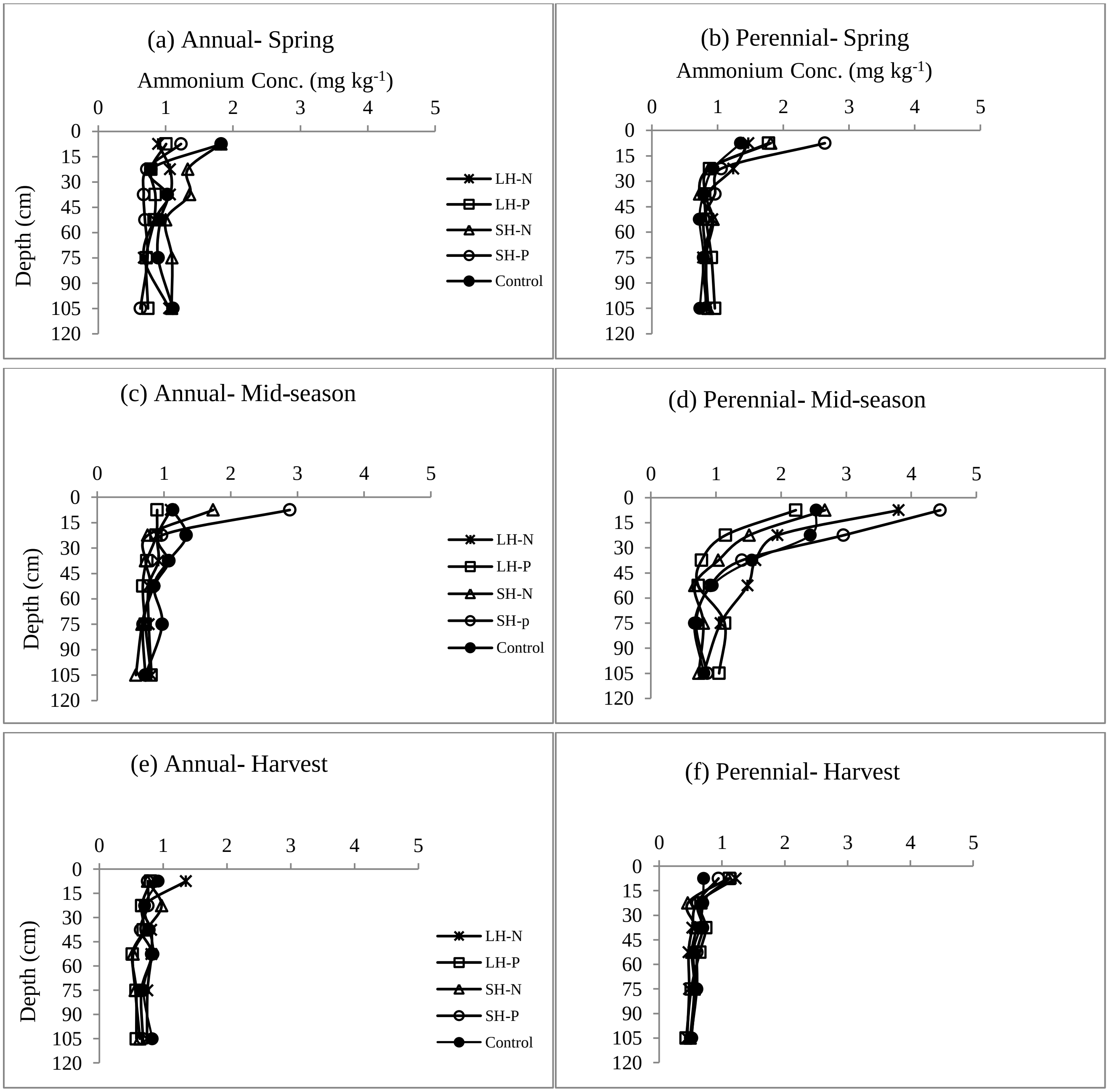


Fig. 2.7.C2. Concentration of ammonium-N in the soil profile during the 2011 growing season

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# 3. NITROGEN- OR PHOSPHORUS-BASED HOG MANURE APPLICATION RATES AFFECT SOIL TEST PHOSPHORUS AND RISK OF PHOSPHORUS LOSS

## 3.1 ABSTRACT

The leakage of nutrients from production systems into the environment is a challenge that is facing agriculture in general and the swine industry in particular in the province of Manitoba. This study was conducted to investigate the effect of cropping systems and manure management techniques on the leakage of phosphorus to ground water and the changes in soil test P (STP) from animal production. To accomplish these objectives, a 3-year study was performed at the Ian Morrison Research Station of the University of Manitoba at Carman, Manitoba. The experiment was a split-plot design with annual and perennial cropping systems as the main plots. Five nutrient management treatments were the subplots and were: nitrogen based liquid hog manure application, phosphorus based liquid hog manure application, nitrogen based solid hog manure application, phosphorus based solid hog manure application and a control. The study was initiated in the spring of 2009 with the perennial cropping system consisting of a mixture of timothy and orchard grass. The annual crop was canola in 2009; barley in 2010 and canola in 2011. Soil samples were taken three times during the growing season, in the spring, mid-season and at harvest. Leachate samples were collected from field core lysimeters after snowmelt and intermittently based on rainfall events. Plant samples were taken at mid-season and at harvest. Soil, plant and water samples were analyzed for phosphorus. The results of this study indicate that there was no evidence of significant downward movement of phosphorus below the top 15 cm depth. However, repeated, annual application of manure at an N-based rate resulted in increased soil test phosphorus (STP). By the end of the 3-year study, STP concentrations in the N-based solid and liquid manure management treatment were significantly greater (48 and 43 mg kg$^{-1}$ for the solid and the liquid manure, respectively) than the P-based solid and liquid manure treatments (26 and 17 mg kg$^{-1}$ for the solid and the liquid manure, respectively). Because increasing STP results in increasing P concentration in runoff, STP buildup should be managed through field rotation when N-based manure application rates are applied. In contrast to the N-based annual applications of manure, soil test P levels were not different from the control treatment by the end of the 3$^{rd}$ year of this study when manure was applied at a multi-year, P-based manure application rate (where manure was applied at the N rate in 2009 and

received urea 2010 and 2011). These results demonstrated that a multi-year, P-based manure application rate resulted in soils with low STP and low risk of P loss through runoff and leachate.

## 3.2 INTRODUCTION

In Canada and Manitoba, hog manure is widely applied as an amendment to agricultural lands (Flaten et al. 2003). Hog manure provides nutrients and organic matter to the soil (Maule et al. 2006), so it can be an excellent resource for agriculture. Excess applications of livestock manure, however, can result in the loss of phosphorus (P) from agricultural land and consequent degradation of groundwater, streams and lakes (Allen et al. 2006).

Phosphorus is considered a pollutant in surface water due to eutrophication. At very low concentrations (0.035-0.1 mg $L^{-1}$) P can cause excessive plant growth and algal blooms. When the plants and algae die, their decomposition can deplete oxygen in water and cause odors and fish kills (CCME 2004).

Large algal blooms have occurred in Lake Winnipeg as a result of P driven eutrophication (Flaten et al. 2003; Stainton et al. 2003). The main cause of this process appears to be the excess nutrient load to the Red River and other tributaries in the upstream regions (Alberta, Saskatchewan, Ontario and USA), from rural and urban areas alike. To address this issue, the Manitoba Government has proposed new regulations affecting mainly livestock production.

Generally, most research on P movement in soil has been focused on soil P concentration as a function of depth which has resulted in the general assumption that insignificant P leaching occurs because of high P-fixation capacity in many mineral soils (Sims and Sharpley 2005). Nevertheless, phosphorus can be lost from the soil if the amount applied exceeds the retention capacity of the soil and the rate of uptake by plant roots (Brye et al. 2002). As a result of repeated application of phosphorus, the chemical equilibrium of soil P adsorption capacity established by retention–release processes may change. This will lead to higher concentrations of P in solution, greater potential for P movement and possibly ground water pollution (Brye et al. 2002). Phosphorus leaching loss is a slow process and can continue for many years; however, it still can be an environmental threat, especially in sandy soils with low P sorption capacities that receive high P additions from livestock manure (Nelson et al. 2005). Therefore, assessment of P leaching losses is a key tool to prevent this gradual and quiet environmental danger.

Manure is typically applied to meet the N requirements of the crop. Repeated, annual applications of manure based on N requirements of the crop often result in over-application of manure P and a build-up of soil test P. As soil test P increases, the concentration of P in runoff also increases (Sharpley et al. 2009). Application of manure to meet the annual P requirements of the crop most often does not supply adequate N for optimum crop yields. Therefore, on high soil test P soils, it is recommended that manure be applied intermittently, based on multi-year P-based application. In the intervening years, synthetic N fertilizer is applied to meet the crops' needs (Miller et al. 2011).

Toth et al. (2006) studied the effects of annual N- vs. multi-year P-based manure applications on N and P uptake by three perennial crops (alfalfa, corn silage, and orchard grass), leaching below the root zone, and accumulation of P in soil. The results showed that application of manure at N-based and P-based rates (where ammonium sulfate fertilizer was used to meet the N requirement of crop), supplied adequate nutrients for crop growth, and had equal losses of nitrate and total P in leachate. However, the N-based manure resulted in greater accumulation of soil test P in the surface 5 cm of the soil. Surface soil P accumulation has implications for increased risk of P movement.

Kumaragamage et al. (2011) studied P runoff and leaching losses from different sources of solid cattle manure, liquid hog manure and mono ammonium phosphate (MAP) in sand and clay loam soils. The results of this study showed that the proportion of P in liquid hog manure that was susceptible to runoff and leaching losses was generally greater than that in solid cattle manure, but less than in MAP.

To our knowledge, only one other study is being conducted in Manitoba that compares N and P-based applications of hog manure to different cropping systems and it is being conducted on heavy clay soil (Fraser and Flaten 2014). Also, few studies have measured P leaching following manure application (Coppi 2012). These studies are needed to understand the movement of phosphorus from manure-amended soils. The objectives of this study were to determine the influence of cropping system (perennial versus annual); nutrient management system (N versus P-based); and the type of hog manure (liquid versus solid) on STP, crop P uptake, and the loss of water and phosphorus below the root zone.

## 3.3 MATERIALS AND METHODS

The site characteristics and experimental design of the study as well as manure and urea application rates were described in detail in Chapter 2.

### 3.3.1 Field and Laboratory Procedures

A detailed description of the field and laboratory procedures was given in section 2.3.6. Briefly here, soil samples were collected during the growing seasons of 2009, 2010 and 2011 at three times: spring, mid-season, and harvest. Sampling dates are provided in Appendix 3.6.A. Soil was sampled at six depth intervals of 0-15, 15-30, 30-45, 45-60, 60-90 and 90-120 cm for spring and harvest using the Giddings soil sampler and at five depths of 0-15, 15-30, 30-45, 45-60, 60-90 cm for mid-season using a Dutch auger. Two soil samples were taken from each plot and composited.

Olsen-P was determined on air dried samples in 2009 and field moist samples in 2010 and 2011. $NaHCO_3$- extractable P (Olsen-P) was measured following the methods of Olsen and Sommers (1982). Phosphorus in the extract was determined using the colorimetric method of Murphy and Riley (1962).

Plant samples were collected in each year at mid-season and harvest. Sampling dates are provided in the Appendix 3.6.A. In each plot, biomass samples were taken in four randomly-selected areas using a 0.25 $m^2$ quadrant. In 2011, 2.0 $m^2$ were sampled to reduce the variability in yield data. The plant material was put in cloth bags and hung in a drying room at room temperature (25°C) for 30 d after which the seed was threshed and the seed, straw and grass weights determined. The mid-season and harvest biomass were sub-sampled and finely ground with a mini-ball mill for total P using the wet oxidation technique of Akinremi et al (2003). In one year, 2011 only, plant biomass samples were reanalyzed by Agvise Laboratories, Northwood, North Dakota.

Leachate was collected from the lysimeters three to five times, depending on the amount of precipitation during the growing seasons. Sampling dates are provided in the Appendix 3.6.A. The leachate was collected from the catch basin by a vacuum pump connected to a hose that ran through one of the extraction tubes. The second extraction tube was opened during leachate collection to equalize pressure, otherwise, both tubes were covered with a cap to prevent rain water from running down the tubes. The total volume of leachate from each lysimeter was recorded and the phosphorus concentration determined. The procedure for measuring the concentration of phosphorus in the leachate was as outlined for the soil samples. Total flux of phosphorus was determined by multiplying concentration of P in collected leachate by total water flow for each year.

### 3.3.2 Statistical Analyses

Analysis of variance (ANOVA) using PROC MIXED procedure (SAS Institute, 2008) was conducted on soil, leachate and biomass to determine significant cropping system, nutrient treatment effects and their interaction in each year. Assumption of normality distribution was checked using PROC UNIVARIATE. Since Shapiro-Wilk's normality test did not show normal distribution for leachate and soil measurements, the log transformed data was used to generate normal distribution of residuals and homogeneity of variance prior to statistical analysis. For total above-ground biomass and their nutrient uptakes as well as leachate, the statistical model included block (with four levels) as a random factor and treatments (five levels) and cropping systems (two levels) as fixed factors. For soil phosphorus, the statistical model included block (with four levels) as a random factor and treatments (five levels), cropping systems (two levels) and depth (six levels) as fixed factors with depth treated as a repeated measurement. The spatial power [SP(POW)] covariance structure was used in the model for the repeated measures data in which the depth intervals were unequal. Due to variation in manure application by hand a predefined 0.1 significant level was considered (Olatuyi et al. 2012; Zvomuya et al. 2003). Treatment differences were accepted if $P<0.1$ using Tukey-Kramer method.

## 3.4 RESULTS AND DISCUSSIONS

### 3.4.1 Yields and Phosphorus Uptake/Removal

Phosphorus uptake of the two cropping systems were compared in a full factorial analysis (Table 3.1); however, further analysis of individual crops were presented for harvested yields (i.e. canola oil seed, barley grain and grass), and their phosphorus removals in 2009, 2010 and 2011(Tables 3.2, 3.4, 3.5, 3.6, 3.7, 3.8).

#### 3.4.1.1 Biomass and Phosphorus Uptake/Removal - 2009

In 2009, there was a significant effect of cropping system on biomass as the canola crop produced significantly greater biomass than the grass (Chapter 2, Table 2.4). There was no significant effect of manure treatment ($P > 0.1$) on biomass yield or phosphorus uptake[2], due to the similarity between the various treatments in the first year of the study. However, although not statistically significant ($P<0.1055$), the solid manure tended to result in the greatest P uptake (Table 3.1), particularly for the canola. As well, although not statistically significant, the control treatments of both annual and perennial cropping systems produced numerically the smallest biomass yield and P uptake.

[2] Removal: Nutrient removed in the harvested portion of the crop
Uptake : Total nutrient taken up by the crop (adapted from the Canadian Fertilizer Institute, 2001)

Table 3.1. Phosphorus uptake of annual and perennial cropping systems at harvest in 2009, 2010 and 2011

| ***Group Means*** | | | ***P uptake - 2009*** | ***P uptake - 2010*** | ***P uptake - 2011*** |
|---|---|---|---|---|---|
| | | | ---------------- | ***kg ha$^{-1}$*** | ---------------- |
| *Crop× Manure* | | | | | |
| Annual | Liquid-N | | 20.7 | 31.9a | 28.6 ab |
| | Liquid-P | | 21.6 | 21.1 b | 21.6 bc |
| | Solid-N | | 27.4 | 30.8 a | 33.4 a |
| | Solid-P | | 24.8 | 26.6 ab | 28.5 ab |
| | Control | | 19.1 | 18.2 b | 15.9 c |
| Perennial | Liquid-N | | 19.2 | 40.8 a | 26.2 a |
| | Liquid-P | | 18.3 | 30.7 bc | 20.4 ab |
| | Solid-N | | 18.2 | 30.6 bc | 22.0 ab |
| | Solid-P | | 18.7 | 36.1 ab | 22.5 ab |
| | Control | | 15.4 | 24.2 c | 16.7 b |
| *Crop* | | | | | |
| Annual | | | 22.7 | 25.7 | 25.4 |
| Perennial | | | 17.9 | 32.5 | 21.5 |
| | *Manure* | | | | |
| | Liquid-N | | 19.9 | 36.3 | 27.4 |
| | Liquid-P | | 19.9 | 25.9 | 21.0 |
| | Solid-N | | 22.8 | 30.7 | 27.7 |
| | Solid-P | | 21.8 | 31.4 | 25.5 |
| | Control | | 17.3 | 21.2 | 16.3 |
| ***Model effect*** | | ***d.f.*** | ---------------- | ***P value*** | ---------------- |
| Crop | | 1 | 0.1271 | 0.007 | 0.0007 |
| Manure | | 4 | 0.1055 | <0.0001 | <0.0001 |
| Crop×Manure | | 4 | 0.2343 | 0.0402 | 0.0082 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

As described in Chapter 2, all of the canola oilseed yields were greater than the target yields of 1960 kg ha$^{-1}$ for canola, even on the control plots which were not fertilized (Table 3.2).

There was no significant effect of manure application on canola oilseed yield, as yields ranged from a low of 2107 kg ha$^{-1}$ in the control to a high of 2682 kg ha$^{-1}$ in the solid hog manure N-based treatment (Table 3.2). Phosphorus removals in the canola seed ranged from 6.4 to 7.2 kg tonne$^{-1}$ (calculated from Table 3.2) and were below the range of what is reported by CFI for western Canada (Table 3.3).

Table 3.2. Canola oilseed yield, phosphorus concentration and removal at harvest in 2009

| | | Seed (kg ha$^{-1}$) | P Conc. (%) | Phosphorus Removal (kg P ha$^{-1}$) |
|---|---|---|---|---|
| Liquid-N | | 2142 | 0.66 ab | 13.8 |
| Liquid-P | | 2561 | 0.64 b | 16.4 |
| Solid-N | | 2682 | 0.71 ab | 19.1 |
| Solid-P | | 2323 | 0.72 a | 16.8 |
| Control | | 2107 | 0.65 ab | 13.8 |
| ***Model effect*** | ***d.f.*** | | ---------------------------- ***P value*** [z] ---------------------------- | |
| Manure | 4 | 0.8319 | 0.0308 | 0.6173 |

[z] Probability value is significant at $P < 0.1$

Table 3.3. Phosphorus removal ranges for grass, canola and barley for Western Canada (adapted from the Canadian Fertilizer Institute, 2001).

| ***Crop*** | ***Yield*** | ***P Removal*** | |
|---|---|---|---|
| | ***tonne ha$^{-1}$*** | ***kg ha$^{-1}$*** | ***kg tonne$^{-1}$*** |
| Grass | 6.7 | 13-16 | 2-2.4 |
| Canola | 1.96 | 16-20 | 8-10 |
| Barley | 4.3 | 14-18 | 3.5-4.3 |

There were significant effects of manure application on total grass yield and P removal (Table 3.4). A more detailed discussion of the effect of manure application on total grass yield was presented in Chapter 2. Phosphorus removal was significantly higher on the N-based liquid manure plots and the P-based solid manure plots than the control. Phosphorus removals for the control plots and the liquid manure treatments (Table 3.4) were in the range of what was reported by CFI for western Canada (Table 3.3); whereas P removals for the solid manure treatments were slightly above what was reported by CFI (2.5 and 2.6 kg tonne$^{-1}$ for N based solid manure and P based solid manure, respectively). Phosphorus concentrations in the grass were much higher for the second cut than for the first cut, even for the control (Table 3.4).

Table 3.4. Grass yield, phosphorus concentration and removal in 2009

| ***Treatment*** | | ***Total Yield (kg ha$^{-1}$)*** ***(1$^{st}$ cut)+(2$^{nd}$ cut)*** | ***P Conc. (%)*** ***1$^{st}$ cut*** | ***2$^{nd}$ cut*** | ***Total P Removal (kg P ha$^{-1}$)*** |
|---|---|---|---|---|---|
| Liquid-N | | 8195 a | 0.15 | 0.30 | 19.2 a |
| Liquid-P | | 7982 ab | 0.15 | 0.29 | 18.3 ab |
| Solid-N | | 7350 ab | 0.15 | 0.32 | 18.2 ab |
| Solid-P | | 7292 ab | 0.16 | 0.32 | 18.7 a |
| Control | | 6847 b | 0.15 | 0.29 | 15.4 b |
| ***Model effect*** | ***d.f.*** | ------------------------------------ | ***P value*** | | --------------------------------- |
| Manure | 4 | 0.0427 | 0.9117 | 0.4238 | 0.0571 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

### 3.4.1.2 Biomass and Phosphorus Uptake/Removal - 2010

In 2010 there was a significant crop effect, manure effect and crop x manure interaction for total biomass (Chapter 2, Table 2.4) and P uptake (Table 3.1). Because of the interaction between crop and manure, the manure treatment effects were tested for each cropping system separately.

In 2010, barley grain yields from the annual application of liquid and solid manure at an N based rate were significantly higher than the control. However, yields on the P-based manure plots were not statistically different from the control (Table 3.5).

The barley grain removed 4.3 to 4.9 kg P tonne$^{-1}$ (calculated from Table 3.5) which is slightly above the range published by the Canadian Fertilizer Institute (Table 3.3). All manure treatments, except the liquid P-based/urea treatment, resulted in greater P removal than the control. Regardless of the forms of manure, the greatest P removal was in the N-based manure treatments. This may reflect the effect of cumulative P addition from two years of N-based manure application.

Table 3.5. Grain yield, phosphorus concentration and removal of barley at harvest in 2010

| | | ***Grain (kg ha$^{-1}$)*** | ***P Conc. (%)*** | ***Phosphorus Removal (kg P ha$^{-1}$)*** |
|---|---|---|---|---|
| Liquid-N | | 4475 a | 0.48 a | 21.6 a |
| Liquid-P/Urea | | 3320 bc | 0.44 c | 14.4bc |
| Solid-N | | 4357 ab | 0.48 ab | 20.8a |
| Solid-P/Urea | | 3812 abc | 0.49 a | 18.8ab |
| Control | | 2942 c | 0.45 bc | 13.3c |
| ***Model effect*** | ***d.f.*** | ---------------------------------- ***P value*** ---------------------------------- | | |
| Manure | 4 | 0.0015 | 0.0005 | 0.0003 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

As discussed in Chapter 2, the grass yields in 2010 were greater than in 2009 and were above the target yield of 6.7 tonne ha$^{-1}$ even on the unfertilized control. This may be due, in part, to higher precipitation in 2010. In 2010 the grass yields were the highest for the P-based solid manure (applied in 2009) that received urea in 2010 and the N-based liquid manure. These two treatments had biomass yields that were statistically greater than the control (Table 3.6). The grass yields on the liquid P-based manure that received urea in 2010 were numerically smaller than the other manure treatments, but not statistically different from any of the treatments. Nitrogen availability from the solid manure applied annually at an N rate may have limited yield as the manure supplied only 76 kg N ha$^{-1}$ in 2010 (Table 2.3b).

The concentration of P in the grass was higher in 2010 (Table 3.6) than 2009 (Table 3.4) particularly for the first cut measurements. The high concentration of P in the grass combined with the high yields for the manure treatments resulted in a very high P removal on a per hectare basis.

Table 3.6. Grass yield, phosphorus concentration and removal in 2010

| | | ***Total Yield (kg ha$^{-1}$)*** | ***P Conc. (%)*** | | ***Total P Removal (kg P ha$^{-1}$)*** |
|---|---|---|---|---|---|
| ***Treatment*** | | ***(1$^{st}$ cut)+(2$^{nd}$ cut)*** | ***1$^{st}$ cut*** | ***2$^{nd}$ cut*** | |
| Liquid-N | | 10865 a | 0.36 | 0.39 a | 40.8 a |
| Liquid-P/Urea | | 9892 ab | 0.31 | 0.32 b | 30.7 bc |
| Solid-N | | 7837 b | 0.36 | 0.41 a | 30.6 bc |
| Solid-P/Urea | | 10977 a | 0.32 | 0.33 b | 36.1 ab |
| Control | | 7452 b | 0.31 | 0.35 b | 24.2c |
| ***Model effect*** | ***d.f.*** | ---------------------------------- ***P value*** ---------------------------------- | | | |
| Manure | 4 | 0.0094 | 0.8036 | <0.0001 | 0.0002 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

The annual solid and liquid manure applications at the N-based rate resulted in statistically greater P concentrations in the grass than the P-based/urea applications and the control treatments. In 2010 the liquid manure application rate over-applied P (393 kg $P_2O_5$ $ha^{-1}$, Table 2.3b). The $P_2O_5$ application from the N-based solid manure application was not available.

### 3.4.1.3 Biomass and Phosphorus Uptake/Removal - 2011

In 2011, there was a significant crop effect, manure effect and crop x manure interaction for total biomass (Chapter 2, Table 2.4) and P uptake (Table 3.1). The biomass yield is added to the table as a matter of course and described in Chapter 2 of this thesis.

The concentration of P in the canola seed was slightly lower in 2011 (Table 3.7) than in 2009 (Table 3.2). Phosphorus removals per unit of crop ranged from 4.6 to 6.9 kg $tonne^{-1}$ (calculated from Table 3.7) and are below what is reported by CFI (Table 3.3). The lower concentration of P in the grain was due to a dilution effect brought about by the relatively high canola yield in 2011. Similar results regarding nutrient removal and nutrient dilution were reported by Lieffering et al. (2004). These authors concluded that an increase of biomass and grain production due to increased $CO_2$ led to a decrease of grain N concentrations. Because of the very high canola yields, however, the P removal per hectare was greater than what is reported by CFI (Table 3.3) for the manure treatments (19.2 to 28.1 kg P $ha^{-1}$). The canola grown on the control plots removed less P than the treated plots except the liquid P/urea treatment.

Table 3.7. Canola oilseed yield, phosphorus concentration and removal at harvest in 2011

| | | ***Seed (kg ha$^{-1}$)*** | ***P Conc. (%)*** | ***Phosphorus Removal (kg P ha$^{-1}$)*** |
|---|---|---|---|---|
| Liquid-N | | 3825 a | 0.58 | 24.4 ab |
| Liquid-P/Urea | | 4122 a | 0.41 | 19.2 bc |
| Solid-N | | 4070 a | 0.63 | 28.1 a |
| Solid-P/Urea | | 4440 a | 0.52 | 25.1 ab |
| Control | | 2600 b | 0.50 | 14.2 c |
| ***Model effect*** | ***d.f.*** | ------------------------------------ ***P value*** ---------------------------------- | | |
| Manure | 4 | 0.0024 | 0.5543 | 0.0004 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

Table 3.8. Grass yield, phosphorus concentrations and removal in 2011

| ***Treatment*** | | ***Total Yield (kg ha$^{-1}$)*** | ***P Conc. (%)*** | | ***Total P Removal (kg P ha$^{-1}$)*** |
|---|---|---|---|---|---|
| | | ***(1$^{st}$ cut)+(2$^{nd}$ cut)*** | ***1$^{st}$ cut*** | ***2$^{nd}$ cut*** | |
| Liquid-N | | 10476 a | 0.25 ab | 0.25 bc | 26.2a |
| Liquid-P/Urea | | 9163 a | 0.21 b | 0.24 c | 20.4bc |
| Solid-N | | 7628 b | 0.26 a | 0.32 a | 22.0b |
| Solid-P/Urea | | 9398 a | 0.23 ab | 0.25 bc | 22.5ab |
| Control | | 6392 b | 0.23 ab | 0.30 ab | 16.7b |
| ***Model effect*** | ***d.f.*** | ------------------------------------ ***P value*** ---------------------------------- | | | |
| Manure | 4 | <0.0001 | 0.0869 | 0.0021 | 0.0003 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test.

Grass yields (Table 3.8) from the control and the annual application of solid manure at the N-rate were significantly lower than the other manure treatments. Liquid manure at the annual N-based rate and urea provided sufficient available N for the highest yields.

There was a significant treatment effect on P concentration and removal. Phosphorus concentrations in the grass were lower in 2011 than in 2009 and 2010. Phosphorus removals per unit of crop ranged from 2.2 to 2.9 kg P tonne$^{-1}$ grass (calculated from Table 3.8) and were at the high end of the range reported by the CFI. It is possible that several months of wet soil (near saturation) following periods of freezing and thawing from fall 2010 to spring 2011 increased soluble phosphorus and plant uptake in 2011. Soil test P in the control plots (Table 3.11) was in the agronomic high range in the spring and harvest of 2011, but decreased to the low to very low range at mid-season. Due to the effect of yield, P removal was greatest for the annual liquid manure application at the N rate which was significantly greater than the control (Table 3.8).

### 3.4.2 Soil Phosphorus

#### 3.4.2.1 Treatment effect of soil test phosphorus during the 2009 growing season

Soil sampling and analyses indicated that the accumulation of P was in the upper layer of soil (0-15 cm) and there was no evidence of P movement beyond this layer (data not shown). Miller et al. (2011) found maximum soil test P within the 0 to 30 cm depth and no treatment differences on soil P concentration below 30 cm for different manure treatments after nine years of manure application. Since the Hibsin series consists of soils developed on moderately calcareous lacustrine deposits (Canada-Manitoba Soil Survey Report D60), it seems the high P-fixation capacity of the Carman's subsoil due to the high content of calcium resulted in the reduction of P concentration in the percolating water and reduced downward movement of P.

In comparison, Eghball (2003) reported accumulation of P at the 30 to 60 cm of the soil profile in a sandy loam after 20 years of manure application. Therefore, with long term application of manure, P may finally be subjected to leaching which often occurs on a time scale of decades or more (Radcliffe and Cabrera 2007). Since most of the agronomic and environmental recommendations in Manitoba use residual phosphorus level within the top 15 cm, the soil phosphorus data that was collected at the 0-15 cm depth is the primary focus of the discussion herein.

In 2009 soil test P behaved similarly for both cropping systems (i.e. no crop effect or crop×manure interaction, Table 3.9). There was also no interaction between the crop effect and the manure treatment. There was a significant effect of the manure treatment on soil test P at mid-season and at harvest. The significant manure treatment differences are based on pooled data for the annual and perennial cropping systems.

In 2009 soil test P in the control plots (Table 3.9) were agronomically high (>15 mg $kg^{-1}$ Olsen P) to very high (>20 mg $kg^{-1}$ Olsen P) according to the Manitoba Soil Fertility Guide (MAFRI 2007), indicating that the background P fertility of the site was excellent even without the addition of manure. Solid manure application resulted in the greatest residual (harvest) soil test P (Table 3.9) with only the solid P-based treatment being significantly higher than the control and the liquid P-based treatment. This was likely due to the greater quantity of P that was in the added solid manure: 188 and 207 kg $P_2O_5$ $ha^{-1}$ was applied on the annual and perennial cropping systems, respectively (Table 2.3a). The liquid manure provided much less P than the solid manure at 57 and 65 kg $P_2O_5$ $ha^{-1}$ for the annual and perennial cropping systems, respectively (Table 2.3a), and soil test P from these plots was not statistically different from the control.

Table 3.9. Phosphorus concentration (mg $kg^{-1}$ Olsen P) within the first 15 cm of soil in 2009

| Group means | | | Spring | Mid-season | Harvest |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid- N | | 25.0 | 25.5 | 27.8 |
| | Liquid- P | | 18.2 | 25.0 | 30.0 |
| | Solid- N | | 32.9 | 35.7 | 40.6 |
| | Solid- P | | 25.3 | 30.9 | 46.6 |
| | Control | | 26.1 | 18.5 | 24.9 |
| Perennial | Liquid- N | | 31.0 | 17.8 | 30.6 |
| | Liquid- P | | 25.9 | 19.0 | 23.7 |
| | Solid- N | | 23.1 | 29.4 | 27.3 |
| | Solid- P | | 29.0 | 29.2 | 38.9 |
| | Control | | 21.4 | 21.8 | 22.5 |
| *Crop* | | | | | |
| Annual | | | 25.5 | 27.1 | 34.0 |
| Perennial | | | 26.1 | 23.4 | 28.6 |
| | *Manure* | | | | |
| | Liquid- N | | 28.9 | 21.7 ab | 29.2 ab |
| | Liquid- P | | 22.0 | 22.0 ab | 26.9 b |
| | Solid- N | | 28.0 | 32.5 a | 33.9 ab |
| | Solid- P | | 27.1 | 30.0 ab | 42.8 a |
| | Control | | 23. 8 | 20.2 b | 23.7 b |
| ***Model effect*** | | ***d.f.*** | ------------------------------------ | ***P value*** | ---------------------------------- |
| Crop | | 1 | 0.8256 | 0.4714 | 0.1234 |
| Manure | | 4 | 0.5047 | 0.0196 | 0.0115 |
| Crop×Manure | | 4 | 0.2029 | 0.6835 | 0.6593 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

### 3.4.2.2 Treatment effect of soil test phosphorus during the 2010 growing season

In 2010, there was a significant interaction between the cropping system and manure treatment in spring and at harvest. For this reason, the effect of the manure treatment on spring and harvest soil test P was analyzed for the annual and perennial cropping systems separately.

In 2010 soil test P in the control plots (Table 3.10) were lower than in 2009 but still in the medium (> 10 mg $kg^{-1}$ Olsen P) to high range except for the mid-season perennial which was in the low (< 10 mg $kg^{-1}$ Olsen P) range according to the Manitoba Soil Fertility Guide (MAFRI, 2007). Although not statistically significant, soil test phosphorus was numerically greater in the annual plots than in the perennial plots (Table 3.10). Higher crop P uptakes for the perennial system in 2010 (Table 3.1) may explain, in part, the lower soil test P levels in the perennial system than in the annual crop system. Braman, (2012) reported higher capacity of perennials for building phosphorus in the microbial biomass than annuals due to favorable moisture conditions in perennial rotations.

For the annual cropping system, the N-based liquid and N-based solid manure applications resulted in higher STP levels at harvest than the control. In 2010, the liquid manure was very thick and had a high concentration of P (Table 2.2) resulting in over 390 kg $P_2O_5$ $ha^{-1}$ being applied (Table 2.3b). From this, very high STP levels from the N-based liquid manure could be expected. The actual amount of P applied with the 2010 N-based solid manure application was not calculated; however, the high STP for this treatment was apparent in the spring 2010 soil test which was taken prior to the 2010 manure application. The STP pattern for the annual cropping system (Appendix 3.6.B) was similar to the pattern for barley grain yields (Table 3.5).

For the perennial cropping system, the N-based liquid manure application rate resulted in the highest STP at harvest. This treatment was significantly greater than the liquid P-based manure treatment and the control; however, it was not significantly different from the N- and P-based solid manure treatments. Again, the STP pattern for the perennial cropping system (Appendix 3.6.B) was similar to the pattern for the grass yields (Table 3.6).

The liquid P-based manure showed a decrease in soil test P at harvest in 2010 in both the annual and perennial plots, indicating that the 2009 manure application rates supplied less P than required for multiple crop years. As well, retention of the manure P by the soil may have decreased phosphorus availability for plant uptake (Kashem et al. 2004).

Table 3.10. Phosphorus concentration (mg $kg^{-1}$ Olsen P) within the first 15 cm of soil in 2010

| Group Means | | d.f. | Spring | Mid-season | Harvest |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid-N | | 22.1 b | 39.1 | 48.8 ab |
| | Liquid-P/urea | | 30.3 ab | 16.2 | 13.8 c |
| | Solid-N | | 54.4 a | 41.3 | 54.8 a |
| | Solid-P/urea | | 28.9 ab | 30.1 | 29.0 bc |
| | Control | | 16.6 b | 14.6 | 15.4 c |
| Perennial | Liquid-N | | 23.6 ab | 47.6 | 38.8 a |
| | Liquid-P/urea | | 17.6 b | 11.2 | 7.7 b |
| | Solid-N | | 29.3 ab | 32.6 | 19.1 ab |
| | Solid-P/urea | | 43.8 a | 25.2 | 25.4 ab |
| | Control | | 17.5 b | 9.7 | 11.1 b |
| *Crop* | | | | | |
| Annual | | | 30.5 | 28.2 | 32.3 |
| Perennial | | | 26.3 | 25.3 | 20.4 |
| | *Manure* | | | | |
| | Liquid-N | | 22.8 | 43.3 a | 43.8 |
| | Liquid-P/urea | | 24.0 | 13.7 b | 10.8 |
| | Solid-N | | 41.8 | 36.9 a | 36.9 |
| | Solid-P/urea | | 36.3 | 27.7 ab | 27.2 |
| | Control | | 17.1 | 12.1 b | 13.2 |
| ***Model effect*** | | ***d.f.*** | ---------------------------------- | ***P value*** | --------------------------------- |
| Crop | | 1 | 0.2779 | 0.5828 | 0.0013 |
| Manure | | 4 | 0.0011 | 0.0002 | <0.0001 |
| Crop×Manure | | 4 | 0.0222 | 0.7312 | 0.0247 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

### 3.4.2.3 Treatment effect of soil test phosphorus during the 2011 growing season

In 2011 manure treatments affected soil test P for all three sampling periods and the effect of manure treatments was consistent for both cropping systems (i.e. no crop × manure interaction). Soil test P for perennial system was lower than for the annual system at mid-season only (Table 3.11).

In 2011 soil test P in the control plots (Table 3.11) were in the medium (> 10 mg $kg^{-1}$ Olsen P) to high range in the spring and fall but were in the low to very low (< 5 mg $kg^{-1}$ Olsen P) range at mid-season, according to the Manitoba Soil Fertility Guide (MAFRI, 2007). The repeated, annual N-based manure application rate resulted in the greatest STP at harvest in 2011. The multi-year, P-based treatment resulted in significantly smaller STP levels than the N-based rate at harvest. The STP in the P-based treatment was not significantly different from the control. Comparison of Olsen-extractable P in the control plots at three different sampling times showed the temporal change of soil phosphorus. The relative seasonal variation in the control plots and also in the other treatments was greater in the 2011 growing season than that in 2010. The reason for less fluctuation in phosphorus concentration in 2010 can be related to high moisture content of soil and increasing of soil phosphorus during the wet season.

Table 3.11. Phosphorus concentration (mg $kg^{-1}$ Olsen P) within the first 15 cm of soil in 2011

| ***Group Means*** | | | **Spring** | **Mid-season**[z] | **Harvest** |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| annual | Liquid-N | | 48.7 | 12.7 | 42.2 |
| | Liquid-P/Urea | | 19.3 | 5.5 | 20.9 |
| | Solid-N | | 69.8 | 25.4 | 55.2 |
| | Solid-P/Urea | | 30.4 | 7.8 | 26.4 |
| | Control | | 15.5 | 5.5 | 18.8 |
| | | | | | |
| Perennial | Liquid-N | | 54.1 | 11.5 | 43.6 |
| | Liquid-P/Urea | | 13.5 | 3.2 | 13.6 |
| | Solid-N | | 52.8 | 19.4 | 41.2 |
| | Solid-P/Urea | | 43.1 | 7.5 | 24.7 |
| | Control | | 16.4 | 4.2 | 18.1 |
| | | | | | |
| *Crop* | | | | | |
| Annual | | | 36.7 | 9.5 a | 32.7 |
| Perennial | | | 36.0 | 7.4 b | 28.3 |
| | *Manure* | | | | |
| | Liquid-N | | 51.4 ab | 12.1 b | 42.9 a |
| | Liquid-P/Urea | | 16.4 c | 4.2 d | 17.2 b |
| | Solid-N | | 61.3 a | 22.2 a | 48.2 a |
| | Solid-P/Urea | | 36.7 b | 7.6 bc | 25.6 b |
| | Control | | 16.0 c | 4.2 cd | 18.47 b |
| ***Model effect*** | | ***d.f.*** | ------------------------------------ | ***P value*** | ---------------------------------- |
| Crop | | 1 | 0.8637 | 0.0673 | 0.3904 |
| Manure | | 4 | <0.0001 | <0.0001 | <0.0001 |
| Crop×Manure | | 4 | 0.2735 | 0.7528 | 0.3486 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.
[z] Log transformed data

### 3.4.3 Leachate

#### 3.4.3.1 Amount of water and phosphorus leached below the root zone in 2009

In this study, the lysimeters received the same nutrient treatments as the surrounding plot, and the incorporation of nutrients and seeding were carried out manually. Although the amount of water that was lost below the root zone of the annual crop in 2009 was numerically greater than the water lost below the perennial crop (Table 3.12), the differences were not statistically significant. The absence of a significant crop effect was surprising. Deeper rooting depth and greater water use by perennial crops often decreases the water available for leaching (Entz et al. 2001; Mueller et al. 2005; Hatfield et al. 2001). In the 2009 growing season, there was no significant difference between the canola and grass crops in the amounts of phosphorus that was lost below the root zone (Table 3.12). This is consistent with the soil phosphorus data (Table 3.9).

Table 3.12. Amounts of water and phosphorus leached from annual and perennial plots in 2009

| ***Group Means*** | | ***d.f.*** | ***Water (cm)*** | ***FWMC (P) (mg $L^{-1}$)*** | ***Phosphorus (kg P $ha^{-1}$)*** |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid-N | | 10.8 | 0.002 | 0.002 |
| | Liquid-P | | 6.7 | 0.004 | 0.003 |
| | Solid-N | | 8.9 | 0.010 | 0.008 |
| | Solid-P | | 11.6 | 0.015 | 0.015 |
| | Control | | 7.3 | 0.020 | 0.014 |
| Perennial | Liquid-N | | 5.2 | 0.004 | 0.002 |
| | Liquid-P | | 4.4 | 0.008 | 0.003 |
| | Solid-N | | 7.8 | 0.003 | 0.002 |
| | Solid-P | | 7.5 | 0.005 | 0.004 |
| | Control | | 9.1 | 0.005 | 0.004 |
| *Crop* | | | | | |
| Annual | | | 9.1 | 0.008 | 0.006 |
| Perennial | | | 6.8 | 0.005 | 0.003 |
| | *Manure* | | | | |
| | Liquid-N | | 8.0 | 0.003 | 0.002 |
| | Liquid-P | | 5.5 | 0.006 | 0.003 |
| | Solid-N | | 8.4 | 0.005 | 0.004 |
| | Solid-P | | 9.5 | 0.009 | 0.007 |
| | Control | | 8.2 | 0.010 | 0.008 |
| ***Model effect*** | | ***d.f.*** | ------------------------------------ | ***P value***[z] | -------------------------------- |
| Crop | | 1 | 0.3101 | 0.3086 | 0.2540 |
| Manure | | 4 | 0.3267 | 0.1283 | 0.1262 |
| Crop×Manure | | 4 | 0.3746 | 0.1092 | 0.5733 |

FWMC: Flow Weighted Mean Concentration
[z] Probability value is significant at P <0.1

#### 3.4.3.2 Amounts of water and phosphorus leached below the root zone in 2010

The amount of precipitation received during the growing season in 2010 was 596 mm, which was 158% of the 30 year normal growing season precipitation. The large amount of precipitation led to a leaching loss in the range of 18 to 33 cm in the perennial and 23 to 36 cm in the annual cropping system (Table 3.13), although there was no statistical difference between these amounts.

Phosphorus leaching in 2010 was negligible with no significant effects of cropping system, manure or crop x manure interaction.

Table 3.13. Amounts of water and phosphorus leached from annual and perennial plots in 2010

| ***Group Means*** | | | ***Water (cm)*** | ***FWMC (P) (mg $L^{-1}$)*** | ***Phosphorus (kg P $ha^{-1}$)*** |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid-N | | 22.6 | 0.007 | 0.008 |
| | Liquid-P/Urea | | 32.1 | 0.020 | 0.095 |
| | Solid-N | | 29.1 | 0.004 | 0.015 |
| | Solid-P/Urea | | 36.2 | 0.022 | 0.053 |
| | Control | | 30.5 | 0.012 | 0.043 |
| Perennial | Liquid-N | | 18.8 | 0.015 | 0.045 |
| | Liquid-P/Urea | | 29.7 | 0.012 | 0.075 |
| | Solid-N | | 18.2 | 0.011 | 0.020 |
| | Solid-P/Urea | | 30.4 | 0.007 | 0.021 |
| | Control | | 32.8 | 0.006 | 0.019 |
| *Crop* | | | | | |
| Annual | | | 30.1 | 0.013 | 0.043 |
| Perennial | | | 25.9 | 0.011 | 0.036 |
| | *Manure* | | | | |
| | Liquid-N | | 20.7 | 0.011 | 0.026 |
| | Liquid- P/Urea | | 30.9 | 0.017 | 0.085 |
| | Solid-N | | 23.6 | 0.008 | 0.017 |
| | Solid-P/Urea | | 33.3 | 0.015 | 0.037 |
| | Control | | 31.7 | 0.009 | 0.031 |
| ***Model effect*** | | ***d.f.*** | ----------------------------------- | ***P value***$^{z}$ | --------------------------------- |
| Crop | | 1 | 0.5430 | 0.3977 | 0.5663 |
| Manure | | 4 | 0.4228 | 0.8240 | 0.6426 |
| Crop×Manure | | 4 | 0.9217 | 0.1882 | 0.4132 |

$^{z}$ Probability value is significant at P <0.1

### 3.4.3.3 Amounts of water and phosphorus leached below the root zone in 2011

The amount of water leached from the soil profile showed no statistical difference between annual and perennial in 2011 (Table 3.14), although the amount lost from the perennial system tended to be less than the annual system, similar to what was observed in the previous two years.

Table 3.14. Amounts of water and phosphorus leached from annual and perennial plots in 2011

| *Group Means* | | | *Water (cm)* | *FWMC (P) (mg $L^{-1}$)* | *Phosphorus (kg $ha^{-1}$)* |
|---|---|---|---|---|---|
| *Crop×Manure* | | | | | |
| Annual | Liquid-N | | 21.9 | 0.022 | 0.05 |
| | Liquid-P/Urea | | 32.3 | 0.022 | 0.05 |
| | Solid-N | | 22.1 | 0.012 | 0.03 |
| | Solid-P/Urea | | 35.7 | 0.020 | 0.06 |
| | Control | | 31.3 | 0.027 | 0.07 |
| Perennial | Liquid-N | | 17.4 | 0.027 | 0.04 |
| | Liquid-P/Urea | | 27.9 | 0.024 | 0.10 |
| | Solid-N | | 23.6 | 0.012 | 0.02 |
| | Solid-P/Urea | | 27.2 | 0.014 | 0.04 |
| | Control | | 29.1 | 0.014 | 0.03 |
| *Crop* | | | | | |
| Annual | | | 28.7 | 0.021 | 0.050 |
| Perennial | | | 25.1 | 0.017 | 0.039 |
| | *Manure* | | | | |
| | Liquid-N | | 19.6 | 0.024 a | 0.043 |
| | Liquid-P/Urea | | 30.1 | 0.023 a | 0.072 |
| | Solid-N | | 22.9 | 0.012 b | 0.024 |
| | Solid-P/Urea | | 31.4 | 0.017 ab | 0.051 |
| | Control | | 30.2 | 0.019 ab | 0.044 |
| ***Model effect*** | | ***d.f.*** | | ***P value***[z] | |
| Crop | | 1 | 0.5472 | 0.6341 | 0.5828 |
| Manure | | 4 | 0.6280 | 0.0753 | 0.3462 |
| Crop×Manure | | 4 | 0.9788 | 0.4532 | 0.7098 |

[z] Probability value is significant at P <0.1

In 2011, there was a significant effect of manure on the flow weighted mean concentration (FWMC) of P (P<0.1). The FWMC was greater for the liquid manure treatments than the solid N-based manure treatment, but was not significantly different from the control. The FWMC for P in 2011 was about 10 times greater than 2009 and 2010. As previously mentioned, the amount of precipitation received in 2010 was above normal and it is possible that several months of wet soil (near saturation) following periods of freezing and thawing from fall 2010 to spring 2011 increased soluble phosphorus and P leaching in spring 2011 when most of leachate was collected. Very low concentrations of P (0.035-0.1

mg P $L^{-1}$) are enough to cause eutrophication and algae growth in some surface waters (CCME 2004; Lipiec et al. 2011). However, P concentrations in leachate did not exceed this threshold during three years of study.

## 3.5 CONCLUSIONS

In this study, accumulation of P following manure application was restricted to the upper layer of soil (0-15 cm) and there was no evidence of STP increasing below this layer in this study. Solid manure application in 2009 resulted in the greatest residual soil test P in 2009. This was because 188 and 207 kg $P_2O_5$ $ha^{-1}$ were applied with the solid manure on the annual and perennial cropping systems, respectively. By the 3rd year of the study, however, STP levels in the solid P-based application plots had returned to pre-2009 levels and were not significantly different from the control. The liquid P-based manure applications also did not result in increased STP by the 3rd year of the study, although much less P was applied with the liquid manure than the solid. Regarding P leaching risk and water contamination hazard, our results showed that phosphorus concentrations in leachate did not exceed the threshold of 0.035-0.1 mg P $L^{-1}$. Therefore, the short term of P leaching and water contamination is low at this site. However, repeated, annual applications of both forms of manure at an N-based rate resulted in increased STP compared to the control by the 3rd year of the study. Therefore this practice is likely to increase the risk of P leaching over the long term period. Also, because the risk of P loss in surface runoff increases as STP increases, STP accumulation should be minimized by rotating fields when N-based manure application rates are applied.

## 3.6 APPENDICES

### Appendix 3.6.A

Table. 3.6.A. Schedule of soil, biomass and leachate sampling during the 2009, 2010 and 2011

| Field Operation | ***2009*** | | ***2010*** | | ***2011*** | |
|---|---|---|---|---|---|---|
| ***Soil sampling*** | | | | | | |
| Spring | May 21 | | April 27 and May 11 | | June 14 | |
| Mid-season | August 18,19 | | August 4, 5 | | August 11 | |
| Harvest | September 30 and October 6 | | September 16, 17 | | September 26, 30 | |
| ***Biomass sampling*** | | | | | | |
| | Perennial | Annual | Perennial | Annual | Perennial | Annual |
| Mid-season | July 3 | August 18 | June 30 | July 29 | July 6 | August 11 |
| Harvest | September 28 | September 28 | September 14 | September14 | August 29 | September 20 |
| ***Leachate sampling*** | | | | | | |
| | June 25, August 7, September 28 and November 17 | | June 4, July 14, August 24, September 30 and November 2 | | May 16, June 9, July 6 and October 11 | |

**Appendix 3.6.B**

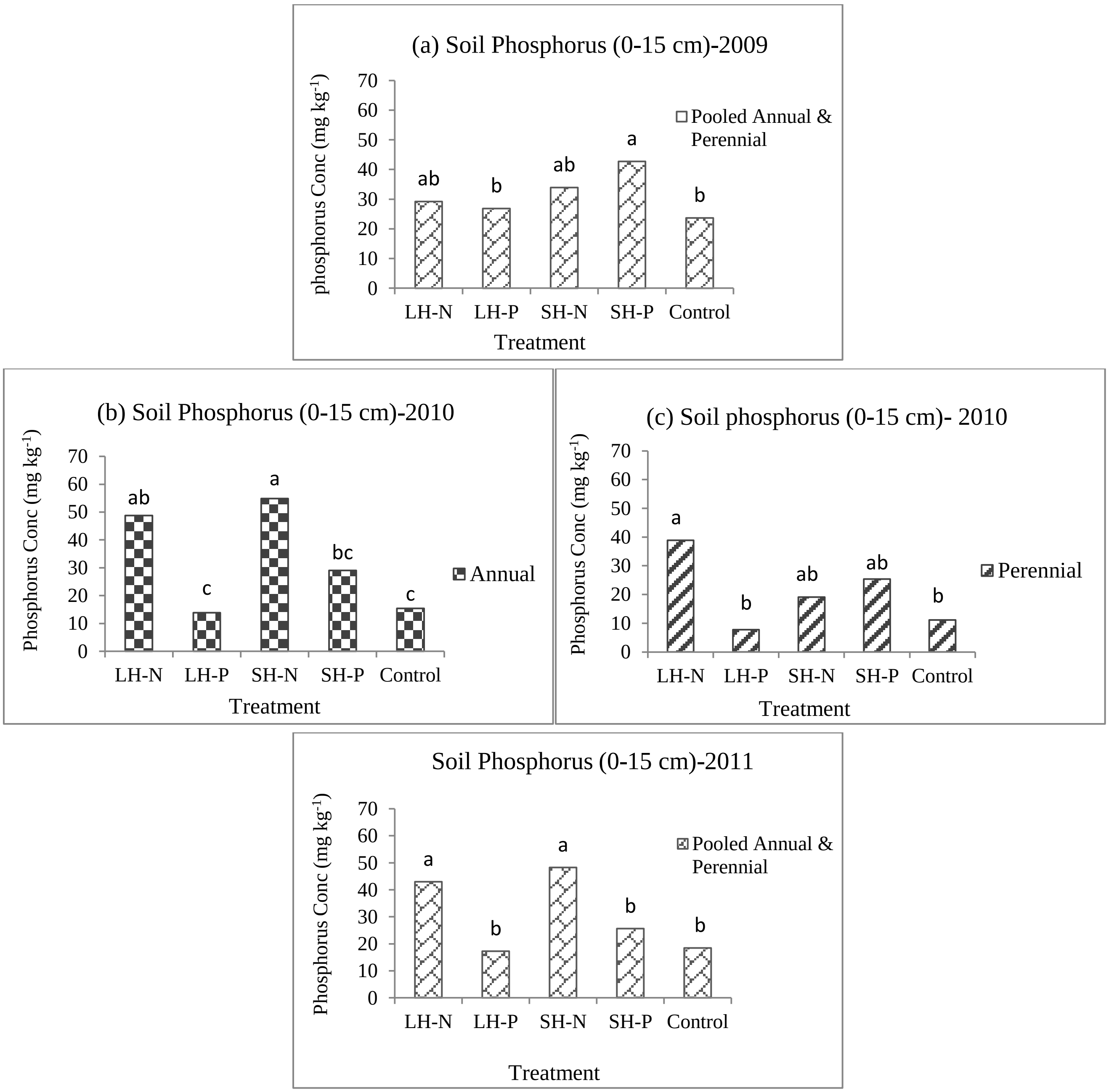


Fig. 3.6.B. Effect of manure treatment and cropping system interaction on soil phosphorus at harvest during three years of study

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# 4. NITRATE AND PHOSPHORUS LEACHING IN A LOAMY SAND SOIL RECEIVING TWO RATES OF LIQUID HOG MANURE AND FERTILIZER

## 4.1 ABSTRACT

To monitor the effect of liquid hog manure and commercial fertilizer on $NO_3^-$-N and P movement from the root zone, a field experiment was conducted at Carberry in a Loamy sand soil. The field experiment was a randomized complete block design with six treatments and four replicates. Treatments included two rates of liquid hog manure (LH-2500 and LH-5000 gallons per acre), two rates of fertilizers (F-2500 and F-5000) corresponding to the estimates for available nitrogen in manure, a compost (com-2500) treatment supplemented with urea to match the estimates for available nitrogen in 2500 gallons per acre of liquid hog manure, and a control. Leachate was collected from lysimeters (one per 10 m by 10 m plot) at intervals dependent on observed precipitation during the 2010 and 2011 growing seasons. The total volume of leachate from each lysimeter was recorded and the $NO_3^-$-N and P concentrations were measured. Available nitrogen was measured three times during spring, mid-season and harvest at soil depths of 0-15, 15-30, 30-60, 60-90, and 90-120 cm and analyzed for inorganic N and Olsen-P concentrations. Results showed in 2010, 79% (112 kg $ha^{-1}$), 55% (40 kg $ha^{-1}$) and 27 % (19.5 kg $ha^{-1}$) of applied available N was lost from F-5000, com-2500 and F-2500, respectively. The 2011 results showed that F-5000 (80%, 63.6 kg $ha^{-1}$) and F-2500 (79%, 31.5 kg $ha^{-1}$) had greater N loss than LH-5000 that lost 40% (32 kg $ha^{-1}$) and LH-2500 that lost 9%.(3.5 kg $ha^{-1}$). Treatments that received fertilizer lost more than one-half of the added N by leaching, with a possible negative effect on the environment. There were no significant differences between rates or sources of nitrogen on grain yield in the two years of study. Phosphorus leaching was negligible and the maximum loss was from the higher rate of liquid hog manure (58 and 83 g P $ha^{-1}$ in 2010 and 2011, respectively). The results of this study showed that caution should be exercised in applying nutrients to this pervious sand.

## 4.2 INTRODUCTION

Excess nitrate (>10 mg N $L^{-1}$) in leachate from farmlands, regardless of the source of nitrate (fertilizers or organic amendments), leads to degradation of water quality. Serious health problems occur when the bacteria of the digestive system transform the nitrate of drinking water to nitrite. The nitrite can form methemoglobin by oxidizing the haemoglobin of red blood cells. Therefore, body cells get insufficient oxygen resulting in respiratory problems (Basso and Ritchie 2005). At very low concentrations of phosphorus (0.035-0.1 mg P $l^{-1}$) in water, eutrophication occurs resulting in algal blooms and the depletion of oxygen in water which cause odor, fish and other aquatic organisms kills (CCME 2004; Lipiec et al. 2011).

In Canada, hog manure is widely applied as an amendment to agricultural lands (Flaten et al. 2003). Manitoba is the third largest hog-producing province in Canada with production of 2.5 million hogs per year. About 2.5% of the crop land area in Manitoba receives hog manure which is an excellent resource for agriculture (Nikièma et al. 2013). However, repeated applications of livestock manure to land can result in accumulation of P and N in the soil and the loss of both nutrients from agricultural areas (Allen et al. 2006; Eghball 2003).

There is no difference between $NO_3$-N derived from fertilizers and from organic sources with respect to risk of leaching. The intensity of nitrate leaching induced by fertilizers or manures depends on the availability of N released by these amendments (Hallberg and Keeny 1993). However, nitrogen in manures is generally less plant available than the N in synthetic fertilizers. This could be due to losses by ammonia volatilization and/or the lack of synchrony between uptake and supply of N from manure (Schröder et al. 2010).

A number of studies have compared the effect of liquid manure rates and synthetic fertilizers on yield, N uptake and groundwater quality. Mooleki et al. (2002) reported that low to medium liquid hog manure application rates (100-200 kg N $ha^{-1}$) on two different soil types in east-central Saskatchewan led to greater wheat grain yield and less $NO_3^-$-N leaching than repeated large application rates. Olson et al. (2009) conducted an eight year field experiment in southern Alberta, Canada, to determine the effects of different rates of manure on $NO_3^-$-N accumulation in two irrigated soil types and $NO_3$-N leaching to shallow groundwater. The results showed that the greatest manure rate significantly increased $NO_3^-$-N concentrations in groundwater at the coarser-textured site. Bergström and Kirchmann (2006) showed that during three years of study on nitrogen leaching at four different rates (50, 100, 150 and 200 kg total N ha$^{-1}$) of liquid hog manure applications, leached nitrogen increased with increasing manure application rate. These authors concluded that liquid hog manure may have been more susceptible to leaching compared to synthetic N fertilizer on sandy soils due to the greater crop efficiency of added nitrogen and phosphorus from synthetic fertilizers than manure. Basso and Ritchie (2005) compared the impact of synthetic fertilizer, compost and manure on crop yield and $NO_3^-$-N leaching based on 120 kg N ha$^{-1}$ annual application. They reported that the greatest amount of $NO_3^-$-N leaching was from manure treatments (55 and 59 kg $NO_3$-N ha$^{-1}$ in a six-year rotation of maize-alfalfa and alfalfa-maize respectively) indicating that the environmental impact of manure is large. Bakhsh et al. (2005) reported in their long-term study at Nashua, Iowa, that liquid swine manure resulted in significantly greater $NO_3^-$-N losses and showed no difference in corn grain yields in comparison with urea ammonium nitrate fertilizer application under a continuous corn production system. In contrast, in a study conducted by Salazar et al. 2012 on a volcanic soil at Southern Chile, they reported greater nitrogen uptake for urea than for liquid manure. However, there was no significant difference in N leaching loss from urea and liquid dairy manure plots in two years of study receiving 400 kg N ha$^{-1}$ yr$^{-1}$.

Traditionally, there is a general assumption that no substantial vertical P movement or leaching losses due to the high P-fixation capacity in many mineral soils. However, P can move downward in soil when the repeated application of phosphorus exceeds the retention capacity of the soil and the rate of uptake by plant roots (Brye et al. 2002). Nelson et al. (2005) monitored P leaching at two different grazed pasture loamy soils which had received hog manure for more than 20 years. They reported maximum P concentrations at the 45 cm depth of soil exceeding 18 mg L$^{-1}$ in both pasture soils. High soil P concentration, increased of degree of phosphorus saturation up to more than 90%, and consequently substantial downward movement of P were the results of long term application of hog manure.

Kumaragamage et al. (2011) studied P runoff and leaching losses from different sources of solid cattle manure, liquid hog manure and mono ammonium phosphate (MAP) in sand and clay loam soils. The results of this study revealed that the proportion of P in liquid hog manure that was susceptible to runoff and leaching losses was generally greater than that in solid cattle manure, but less than in MAP.

The Assiniboine Delta Aquifer (ADA) is a large unconfined aquifer located in southwestern Manitoba covering a land area of about 4,000 $km^2$ including the city of Carberry. Many of the fields in this region are used for growing potatoes, with sprinkler irrigation, resulting in a high risk of groundwater nitrate contamination. It is estimated that the average $NO_3^-$-N concentration of the recharge water under agricultural fields utilizing irrigation is 18 mg N $L^{-1}$ with 50% of samples exceeding the Canadian drinking water standard (Burton and Ryan, 2000). Therefore, more research to quantify nitrate and phosphorus leaching following land application of manure and compost on the sandy soils overlying the ADA is needed.

The objectives of this study were to assess the effects of hog slurry applied at different rates on nitrate and phosphorus leaching in a permeable sandy soil using field core lysimeters and to compare the results with those of synthetic fertilizers and compost; as well as to determine the relationship between the amounts of N applied, nitrate leached and crop uptake of nitrogen and to find out the sustainable rates of hog manure and fertilizer application to reduce nitrate and phosphorus leaching.

## 4.3 MATERIAL AND METHODS

### 4.3.1 Site Description and Experimental Design

A field experiment located northwest of the town of Carberry, Manitoba, was conducted on a loamy sand soil overlying the Assiniboine Delta Aquifer. The site consists of Orthic Black Chernozem soils of Fairland series, which developed on lacustrine deposits. These soils have a medium texture with the upper 75-90 cm soil classified as loamy sand and the underlying layer has a texture of sandy loam to loam. The sand content is about 78% in the upper layer and the percent sand gradually decreases with depth. The mean annual temperature is 2.1$^{o}$ C while mean annual precipitation is 472 mm of which 351 mm is received as rainfall and the maximum of rainfall is received during the months of May to July. Figure 4.1 shows monthly precipitation and mean temperatures during two years of study (2010 and 2011).

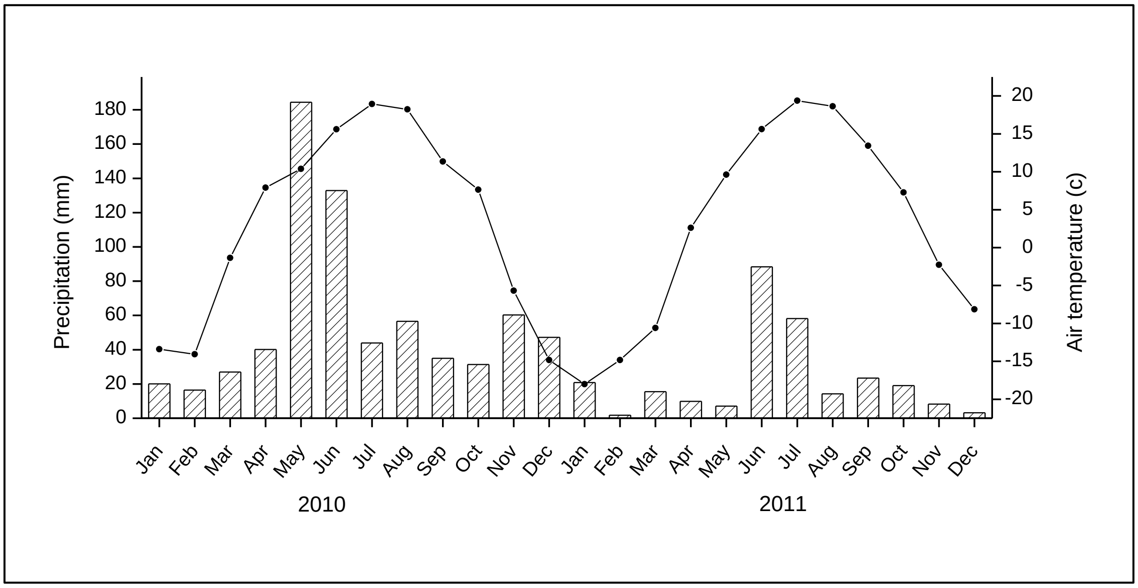


Fig. 4.1. Monthly precipitation (bars) and monthly mean air temperatures (line chart) during two years of study, based on MCDC station's weather data near Carberry

The experimental field is a long term field study that was initiated in 2002 to monitor the effect of liquid hog manure and commercial fertilizer on nutrient movement (Enns 2004; Nikièma et al. 2013). Treatments included two rates of liquid hog manure (2500 and 5000 gallons per acre), two rates of fertilizers corresponding to the amount of available nitrogen in the two rates of hog manure, a composted beef manure treatment, and a control for a total of six treatments. The compost was added primarily to provide P, while urea was used as a supplementary source of nitrogen in the compost treatment such that total available nitrogen was the same as that provided by the 2500 gallons per acre of liquid hog manure.

The experimental design was a randomized complete block design with four replications for a total of 24 treatment plots. Treatment plots were 100 $m^2$ (10 m × 10 m) in size and one field core lysimeter was installed along the centerline for each plot, 2 m from the southern edge to directly measure water, nitrate and phosphorus moving below the rooting zone (Enns 2004).

Each lysimeter included three main parts: the main column, the schedule 80 PVC pipe with an internal diameter of 54 cm and 106 cm in length, representing root zone extension of annual crops; a circular perforated plate and a collection cap. To reduce soil disturbance a drop hammer with a collapsible tower was used to drive down the main column to the desired depth. The main column was then pulled out of the soil and a geotextile material, a perforated plate, a collection cap and two extraction pipes were installed on the main column. Leachate was collected from the catch basin by a vacuum pump connected to a hose that ran down one of the extraction pipes to the collection cap, while the second extraction pipe was kept opened for equalization of pressure (Nikièma et al. 2013). Treatments that were applied to the surrounding plot were also applied to the lysimeter in the plot.

### 4.3.2 Manure Analysis and Application

Representative samples of liquid hog manure were collected from a hog barn in Manitoba and the composted beef cattle manure from Agriculture and Agri-Food Canada (AAFC) composting facility in Brandon, Manitoba. Manure and compost samples were collected in plastic buckets and were kept in a fridge (4° C) until analyzed in laboratory. Samples were thoroughly mixed; subsamples were analyzed for moisture content, total P and total N using the wet oxidation technique of Akinremi et al. (2003). To determine available N in manure and compost, ammonium-N was determined by extraction with a 1:1 extract (20 mL liquid manure : 4 M KCl), shaking the mixture for about 30 minutes, centrifuging and filtering using Whatman No. 40 filter paper. Ammonium was measured in prepared solution within 24 hours using Technicon auto-analyzer II (Pulse Instrumentation Ltd, Saskatoon, SK).

To estimate available N in manure the following formula was used (MAFRI, 2007):

Available N = Ammonium N × (100%- %Volatilization loss) + 25% Organic N

Ammonia volatilization was assumed to be 25% of the ammonium N. Liquid hog manure, fertilizer and compost were applied manually to the designated plots on the 6th of July 2010 and 23rd of June 2011. Since the N:P ratio of composted hog manure was lower than that of liquid manure, urea was applied simultaneously to plots that received compost. In 2010, manure was not analyzed prior to field application due to delay of field work and late seeding. However, manure and compost analyses from previous years were used for rate of manure, compost and fertilizer application. The treatments were applied based on manure available N during the two years of study (Table 4.1). After the manure was applied, all plots were roto-tilled the same day and planted to barley (Hordeum vulgare) and hard red spring wheat (Triticum aestivum L.) at a seeding rate of 100 kg $ha^{-1}$ in 2010 and 2011. The same crops were planted inside the lysimeters after adding equivalent rate of manure, compost or urea to the lysimeters.

Table 4.1. Manure and compost analysis and rate of application

| Treatment | Amount of manure | *$N_t$ | $NH_4$-N | Moisture | Applied Total P | Applied Available N |
|---|---|---|---|---|---|---|
| **2010** | | kg tonne$^{-1}$ | | (%) | --------------- kg ha$^{-1}$ | --------------- |
| LH-2500 | 2500 gal ac$^{-1}$ | 2.4 | 2.3 | 98.8 | 1.4 | 65 |
| LH-5000 | 5000 gal ac$^{-1}$ | 2.4 | 2.3 | 98.8 | 2.8 | 130 |
| Com-2500 | | 6.9 | 0.013 | 51.5 | 22.2 | 73 |
| (10100 kg ha$^{-1}$ Com+121.7 urea kg ha$^{-1}$) | | | | | | (17.6 com+55.4 urea) |
| F-2500 (114 kg ha$^{-1}$ urea+173 kg ha$^{-1}$ MAP) | | ----- | | ----- | 39.2 | 71.5 |
| F-5000 (228 kg ha$^{-1}$ urea+346 kg ha$^{-1}$ MAP) | | ----- | | ----- | 78.4 | 142 |
| **2011** | | | | | | |
| LH-2500 | | 1.6 | 1.3 | 99 | 3 | 39.8 |
| LH-5000 | | 1.6 | 1.3 | 99 | 6 | 79.6 |
| Com-2500 | | 5.7 | 0.013 | 38 | 18.6 | 40 |
| (10100 kg ha$^{-1}$ Com+55.2 urea kg ha$^{-1}$) | | | | | | (14.6 com+25.4 urea) |
| F-2500 (only urea) | | ----- | | ----- | 0 | 40 |
| F-5000 (only urea) | | ----- | | ----- | 0 | 80 |

***$N_t$:** Total Nitrogen

- In 2010 com-2500 treatment was mistakenly applied to plots that were meant for F-2500

### 4.3.3 Field and Laboratory Procedures

Soil samples were collected three times during the growing seasons of 2010 and 2011, in the spring, mid-season, and at harvest. Samples were taken from six depths of 0-15, 15-30, 30-45, 45-60, 60-90 and 90-120 cm using the Giddings soil sampler (spring and harvest) and Dutch auger (mid-season). Soil samples were taken from two holes per plot. Gravimetric moisture content was determined after oven drying (105° C) for all three sampling times.

The field moist soil samples were analyzed for ammonium-N, nitrate-N and $NaHCO_3$-extractable P (Bicarb-P). The Bicrab-P was measured following the methods of Olsen and Sommers (1982). Phosphorus in the extract was determined using the colorimetric method of Murphy and Riley (1962). Following soil extraction with 2 M KCl, soil $NO_3^-$-N was determined by the automated cadmium reduction method while $NH_4^+$-N was measured by the automated phenate method (Clesceri et al., 1998). The remaining portion of the soil samples were dried and stored as archived samples from this site.

Two plant samples were collected in each year, one at mid-season and the other at harvest (Table 4.2). In each plot, biomass samples were taken in four randomly-selected areas using a 0.25 m$^2$

quadrat. In 2011, 2.0 $m^2$ were sampled to reduce variability in the biomass data. Final harvest within the lysimeters and the plots was taken at the same time. The plant material was put in cloth bags and hung in a drying room at room temperature (25°C) for 30 d, after which the seed was threshed and the seed and straw weights determined. The mid-season and harvest biomass were sub-sampled and finely ground with a mini-ball mill for total N and P using the wet oxidation technique of Akinremi et al. (2003).

Leachate was collected from the lysimeters approximately monthly depending on the amount of precipitation during the growing season (Table 4.2). The total volume of leachate from each lysimeter was recorded and the nitrate-N and phosphorus concentration determined. The procedure for measuring the concentration of nitrate and phosphorus in the leachate was as outlined for the soil samples. The total flux of nitrate and phosphorus was determined by multiplying the concentration of nitrate in the leachate by the total water flow for each year.

Table 4.2. Schedule of soil, biomass and leachate sampling during the 2010 and 2011

| Field Operation | 2010 | 2011 |
|---|---|---|
| ***Soil sampling*** | | |
| Spring | June 23 | May 5 |
| Mid-season | August 11 | August 3 |
| Harvest | September 21 | September 14 |
| ***Biomass sampling*** | | |
| Mid-season | August 11 | August 3 |
| Harvest | September 21 | September 14 |
| ***Leachate sampling*** | | |
| | June 9, July 16, August 26, September 28 | May 5, June 27 and 29 and October 13 |

### 4.3.4 Statistical Analyses

All soil, leachate and plant variables were analyzed as a randomized complete block design in SAS software (Littell et al. 1998; SAS Institute, 2008). For soil variables, the statistical model included block (with four levels) as a random factor and treatments (six levels) and depth (six levels) as fixed factors. Analysis of variance (ANOVA) using PROC MIXED was conducted on soil nitrate, ammonium, phosphorus and moisture to determine significant treatment effects, depth and their interaction for each year. The spatial power [SP(POW)] covariance structure was used in the model for the repeated measures data in which the depth intervals were unequal. Treatment means were compared using Tukey-Kramer test at probability level of $P \leq 0.1$. Due to variation in manure application by hand, a predefined 0.1 significant level was considered (Olatuyi et al. 2012; Zvomuya et al. 2003). Since Shapiro-Wilk's normality test did not show a normal distribution for residuals of leachate and soil analysis, the log transformed data were used to get a normal distribution of residuals and homogeneity of variance and the back transformed data was presented on the tables and graphs. For plant and leachate variables, the statistical model included block (with four levels) as a random factor and treatments (six levels) as fixed factors. Effects of the treatment on crop yield, N and P content and uptake as well as leachate nitrate and phosphorus levels were assessed using PROC MIXED procedure in SAS software. Treatment differences were accepted if $P<0.1$ using Tukey-Kramer method.

## 4.4 RESULTS AND DISCUSSION

### 4.4.1 Biomass, Nitrogen and Phosphorus Uptake at Mid-season

The effect of treatment on the biomass yield, N concentration, N uptake[3], P concentration and P uptake of barley and wheat was significant (Table 4.3). In 2010, all amended treatments, except com-2500, produced greater biomass than the control. The F-5000 had the greatest biomass and N concentration and N uptake among the treatments. Due to greater application of P in spring, 78.4 kg ha$^{-1}$ for the high rate of fertilizer, phosphorus concentration was greater in this fertilizer treatment than any other treatments. However, there was no significant difference on P uptake between fertilizer and liquid manure treatments. The compost treatment applied a high rate of P and the availability of the compost P was expected to be 60% of that for inorganic P fertilizer in the year of application (Gagnon et al. 2012). However, the depression in biomass yield and P uptake in com-2500 treatment relative to the other treatments at mid-season suggested that both N and P release from compost was much slower than that in synthetic fertilizer (Chiyoka 2011).

In 2011, the biomass yields for the high rate of manure and fertilizer were similar to those of their respective low rates. All of amended treatments outyielded the control. The LH-2500, com-

[3] Uptake : Total nutrient taken up by the crop (adapted from the Canadian Fertilizer Institute, 2001)

2500 and control had similar N concentration while LH-5000, F-2500 and F-5000 had greater N concentration than that of the control (Table 4.3). As a result of high spatial variability, there were no significant differences in N uptake among the amended treatments. However, all treatments had significantly greater N uptake than the control. Similar to N concentration, LH-2500, com-2500 and the control had similar P concentration while P concentration of LH-5000, F-5000 and F2500 was significantly greater than that of the control. Although com-2500 received the greatest amount of P in 2011 (18.6 kg $ha^{-1}$), LH-5000 showed the greatest P uptake. A smaller N mineralization and slower rate of N release in compost than in hog manure may be responsible for the reduced availability of P in plots treated with compost.

Several studies have reported that for the same P-based application rate, the amount of soil labile P generally increased in the order inorganic P fertilizers > liquid manures > solid or composted manures (Kashem et al. 2004). However, in this study, compost was applied on the bases of its N content and, as a result; substantial amounts of excess P were applied. Therefore, in spring and mid-season 2011, STP values for the compost treatment (Figure 4.7) were extremely high.

Table 4.3. Mid-season above ground plant biomass and nutrient uptake of barley in 2010 and wheat in 2011

| ***Treatment*** | ***Biomass (kg ha$^{-1}$)*** | | ***N Conc. (%)*** | | ***N uptake (kg ha$^{-1}$)*** | | ***P Conc. (%)*** | | ***P uptake (kg ha$^{-1}$)*** | |
|---|---|---|---|---|---|---|---|---|---|---|
| | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** |
| LH-2500 | 1945 a | 1850 ab | 2.9 c | 2.9 c | 68.1 ab | 51.2 a | 0.40 b | 0.34 c | 7.8 ab | 6.2 ab |
| LH-5000 | 2125 a | 1981 a | 3.9 b | 3.3 ab | 76.2 ab | 62.4 a | 0.41 b | 0.37ab | 8.7 ab | 7.2 a |
| Com-2500 | 1640 ab | 1627 ab | 3.87 b | 3.2 bc | 62.0 b | 49.3 a | 0.40 b | 0.35 bc | 6.6 bc | 5.7 b |
| F-5000 | 2212 a | 1761 ab | 4.8 a | 3.5 a | 89.0 a | 58.5 a | 0.55 a | 0.38 ab | 10.3 a | 6.6 ab |
| F-2500 | 1980 a | 1522 b | 4.0 ab | 3.4 ab | 76.1 ab | 49.2 a | 0.44 ab | 0.39 a | 8.7 ab | 5.8 b |
| Control | 967 b | 1085 c | 2.7 c | 3.0 c | 31.3 c | 30.9 b | 0.44 b | 0.33 c | 3.7 c | 3.6 c |
| | | | ------------------------------------ | | *P > F* | | ------------------------------------ | | | |
| ***Trt effect*** | 0.0023 | 0.0001 | <.0001 | 0.0001 | 0.0002 | 0.0002 | 0.0043 | 0.0004 | 0.0024 | <0.0001 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

### 4.4.2 Biomass, Nitrogen and Phosphorus Uptake at Harvest

In 2010, there was a significant treatment effect for biomass, N and P uptake and NUE. The F-5000 treatment was the only treatment that produced significantly greater biomass than the control (Table 4.4). This same treatment (F-5000) had significantly greater N uptake than compost and control treatments. This was due to the large amount of N (142 kg ha$^{-1}$) that was added to this treatment in the spring. However, there was no significant difference in biomass yield, N and P uptake between F-5000 and its corresponding manure treatment, LH-5000.

Fertilizer treatments had significantly greater P uptake than the control; however, there were no significant differences in the P uptake between manure and fertilizer treated plots (Table 4.4).

Table 4.4. Above ground plant biomass and nutrient uptake at harvest by barley in 2010 and wheat in 2011.

| ***Treatment*** | ***Biomass*** | | ***N uptake*** | | ***P uptake*** | | ***NUE*** | |
|---|---|---|---|---|---|---|---|---|
| | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** |
| | ---------------------------- kg ha$^{-1}$ | | | ------------------------------- | | | ---------- *%* --------- | |
| LH-2500 | 4642 ab | 2960 | 94.2 ab | 43.6 ab | 15.4 ab | 5.4 | 55.54 ab | 15.6 |
| LH-5000 | 5012 ab | 3455 | 100.8 ab | 56.4 a | 16.8 ab | 6.8 | 30.7 ab | 22.1 |
| Com-2500 | 4630 ab | 3410 | 88.7 b | 51.7 ab | 14.7 ab | 6.5 | 21.87 b | 18.8 |
| F-5000 | 5690 a | 3057 | 130.2 a | 51.6 ab | 20.4 a | 6.0 | 48.4 ab | 19.5 |
| F-2500 | 5115 ab | 3130 | 101.6 ab | 51.1 ab | 18.7 a | 6.2 | 56.8 a | 38.0 |
| Control | 3267 b | 2557 | 61.1 b | 36.0 b | 10.1 b | 4.8 | ----- | ----- |
| | | ---------------------------------- *P > F* | | | ------------------------------------ | | | |
| ***Trt effect*** | 0.0709 | 0.2470 | 0.0029 | 0.0245 | 0.0176 | 0.1711 | 0.0285 | 0.1082 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test

In 2011, there was no significant treatment effect on biomass, P uptake and NUE. However, considering the trend, the control plots produced the smallest biomass (2557 kg ha$^{-1}$) and LH-5000 produced the largest biomass yield (3455 kg ha$^{-1}$). Overall, the biomass yields in 2010 were greater than in 2011 (Table 4.4) due to greater precipitation in 2010 (Figure 4.1).

There was a significant treatment effect on nitrogen uptake in 2011 due to the significantly greater N uptake from LH-5000 compared to the control. This may be due to a combination of the ammonium nitrogen and the other crop essential nutrients in the manure compared to fertilized plots. Our results in 2011 support the findings of Nikièma et al. (2013) who reported greater yields in manured plots compared to fertilizer amended plots.

Manure and compost treatments received 6 and 18 kg $ha^{-1}$ phosphorus, respectively; however, treatment effect was not statistically significant for P uptake. This might have resulted from the residual effect of MAP applied in 2010.

Nitrogen use efficiency was calculated by subtracting N uptake for the control from N uptake for each treatment in each block and dividing by the amount of total supplemental N applied (Read et al. 2008). The compost treatment had the smallest NUE while the F-2500 had the greatest NUE in 2010. As such, the NUE in F-2500 was significantly greater than from compost-2500 in 2010. The trend in NUE was in the order of F-2500 > LH-2500 > F-5000 > LH-5000. In 2011, treatment effect on NUE was not statistically significant ($P<0.1082$, Table 4.4). These results support the hypothesis that F-5000 and LH-5000 supplied more N than the crop required; therefore, the crop had little or no response to the additional N applied over and above their respective F-2500 and LH-2500.

### 4.4.3 Grain Yield

#### 4.4.3.1 Barley grain yield in 2010

There were significant effects of treatment on grain yield and N and P removal (Table 4.5). The maximum yield was 1.4 tonne $ha^{-1}$ from LH-5000 which was smaller than the average barley yield for the rural Municipality of North Cypress in southwestern Manitoba (4.5 tonne $ha^{-1}$) in 2010 (Manitoba Agricultural Services Corporation). Because of above normal precipitation, soil moisture was not a limiting factor for grain yield in 2010. It is likely that the late seeding of barley, July 6, contributed to the low yields. June 20 is considered the last seeding date for barley for full crop insurance coverage by Manitoba Agricultural Services Corporation. Late planting significantly reduced grain yield of spring wheat, oat and barley during 4-year field study at Pullman, Washington (Ciha 1983). Juskiw and Helm (2003) investigated the response of five cultivars of barley to seeding date at Lacombe, central Alberta. The authors reported 54 to 76% reduction of grain yields compare to the mean site yield due to late seeding date (mid-June).

These data gathered in Alberta are similar to those gathered by the Manitoba Agricultural Services Corporation (MASC) for crops grown in Manitoba.

Table 4.5. Grain yield of barley and nutrient uptake at harvest (2010)

| | | ***Grain (kg ha$^{-1}$)*** | ***N Conc. (%)*** | ***N Removal (kg ha$^{-1}$)*** | ***P Conc. (%)*** | ***P Removal (kg ha$^{-1}$)*** |
|---|---|---|---|---|---|---|
| LH-2500 | | 1292 a | 1.8 | 23.3 a | 0.39 c | 5.1 ab |
| LH-5000 | | 1385 a | 2.0 | 26.7 a | 0.44 ab | 6.1 a |
| Com-2500 | | 887 ab | 1.7 | 17.9 ab | 0.41 bc | 3.6 ab |
| F-5000 | | 1112 ab | 2.2 | 22.5 a | 0.45 a | 5.0 ab |
| F-2500 | | 1072 ab | 1.9 | 19.1 ab | 0.42 abc | 4.5 ab |
| Control | | 570 b | 1.8 | 9.9 b | 0.40 c | 2.3 b |
| ***Model effect*** | ***d.f.*** | ---------------------------------- ***P value*** --------------------------------- | | | | |
| Treatment | 5 | 0.0235 | 0.917 | 0.0093 | 0.0015 | 0.0195 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test

Plots that received manure produced the greatest grain yield regardless of rate. Grain yields from the manure plots were significantly greater than those from the control plots (Table 4.5). Although grain yield from fertilizer and compost treatments were greater than that of control, there was no statistically significant difference between these treatments and the control.

The two rates of manure and high rate of fertilizer showed the greatest N removal which was significantly greater than that of the control. There were no statistically significant differences in the N removal between Com-2500 and F-2500; however, these two treatments had approximately twice the amount of grain nitrogen in the control. Thomsen (2005) reported the greatest N removal from plots that received liquid manure compared to plots amended with calcium ammonium nitrate. He concluded that synthetic fertilizer was more efficient than liquid manure at the beginning of growing season and later in the grain filling time, liquid manure N became available resulting in greater N content in the grain. Nikièma et al. (2013) concluded that other than N and P, liquid hog manure can provide other plant nutrients such as Ca, K, Mg and S, resulting in greater yield in plots that received liquid hog manure than in control plots.

Although fertilizer plots received MAP as a source of phosphorus (39 and 78 kg ha$^{-1}$ for F-2500 and F-5000 respectively), LH-5000 had the greatest phosphorus removal (6.1 kg ha$^{-1}$) that was significantly greater than for the control plot (2.3 kg ha$^{-1}$).

#### 4.4.3.2 Wheat grain yield in 2011

In 2011, the maximum yield was 0.9 tonne ha$^{-1}$ from F-2500 and com-2500 treatment (Table 4.6) which was about one-third of the average wheat yield for the rural Municipality of North Cypress in southwestern Manitoba (3 tonne ha$^{-1}$) in 2011. The total amount of precipitation in 2011 (270 mm) was about one-half of 10-yr mean precipitation (470 mm) and could explain the low yield in 2011.

Nikièma et al. (2013) reported wheat grain yield that ranged between 1.1 tonne ha$^{-1}$ from control plots to 1.3 tonne ha$^{-1}$ from high manure N rate (192 kg N ha$^{-1}$) in 2002 and 2003 from the same study site. In contrast, grain yield in 2004 ranged from 2.5 (control plots) to 3.9 tonne ha$^{-1}$ (high manure N rate). They concluded that the increased precipitation in 2004 (16% more than the long-term average) resulted in the greater wheat yield compared to the two previous years.

Table 4.6. Grain yield of wheat and nutrient uptake at harvest (2011)

| | | Grain (kg ha$^{-1}$) | N Conc. (%) | N Removal (kg ha$^{-1}$) | P Conc. (%) | P Removal (kg ha$^{-1}$) |
|---|---|---|---|---|---|---|
| LH-2500 | | 780 | 2.9 b | 22.2 | 0.4 | 3.2 |
| LH-5000 | | 881 | 3.0 ab | 26.3 | 0.4 | 3.9 |
| Com-2500 | | 911 | 2.8 b | 25.8 | 0.4 | 3.8 |
| F-5000 | | 801 | 3.1 a | 24.9 | 0.4 | 3.6 |
| F-2500 | | 925 | 2.9 b | 26.7 | 0.4 | 3.9 |
| Control | | 750 | 2.6 c | 19.8 | 0.4 | 3.2 |
| ***Model effect*** | ***d.f.*** | ----------------------------------- | ***P value*** | ---------------------------------- | | |
| Treatment | 5 | 0.4979 | <0.0001 | 0.2042 | 0.1048 | 0.4571 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test

There was no statistically significant effect of treatment on grain yield and N and P uptake perhaps due to high variability (P>0.1, Table 4.6). Evanylo et al. 2008 reported that the application of compost with fertilizer increased the effectiveness of fertilizer resulting in increased crop productivity. In 2010, com-2500 treatment was mistakenly applied to plots that were meant for F-2500. It is likely that the relatively large grain yield in F-2500 is a result of the residual effect of compost from the previous year.

### 4.4.4 Straw Yield and Nutrient Removal in 2010 and 2011

In 2010, similar to grain yield, barley straw yield, N and P removal were significantly affected by treatment (Table 4.7). Straw yield was the greatest at the high fertilizer N rate. Application of fertilizer at both rates showed significant greater straw N removal than the control. However, there were no significant differences in N removal between the two rates of LHM, the lower rate of fertilizer or the compost. Unlike in the grain where LH-5000 treatment had the greatest P removal, F-5000 had the highest concentration of P in the straw. The N concentration of straw at harvest was almost the same as that of the grain, an indication that the crop did not have enough time to transfer nitrogen from stem to seed at harvest.

In 2011, similar to the grain, there were no significant treatment effects on straw yield. Nitrogen uptake was significantly different among treatments. Although there was no significant source or rate effect on N uptake, the high rate of manure and fertilizers (LH-5000 and F-5000) showed the greatest N uptake. Only the plot that received the high rate of manure (LH-5000) had a significant greater P removal in the straw than the control.

Table 4.7. Straw yield and nutrient removal at harvest 2010 and 2011

| | | ***Straw (kg ha$^{-1}$)*** | | ***N Removal (kg ha$^{-1}$)*** | | ***P Removal (kg ha$^{-1}$)*** | |
|---|---|---|---|---|---|---|---|
| | | *2010* | *2011* | *2010* | *2011* | *2010* | *2011* |
| LH-2500 | | 3396 ab | 2180 | 72.4 bc | 20.9 ab | 14.1 ab | 2.2 ab |
| LH-5000 | | 3628 ab | 2574 | 74.1 bc | 30.1 a | 10.7 ab | 2.9 a |
| Com-2500 | | 3743 ab | 2499 | 70.8 bc | 25.8 ab | 11.1 ab | 2.7 ab |
| F-5000 | | 4578 a | 2256 | 107.7 a | 26.6 a | 15.4 a | 2.5 ab |
| F-2500 | | 4043 ab | 2205 | 82.5 ab | 24.4 ab | 14.2 ab | 2.3 ab |
| Control | | 2698 b | 1808 | 51.2 c | 16.1 b | 7.8 b | 1.7 b |
| ***Model effect*** | ***d.f.*** | ------------------------------------ ***P value*** ---------------------------------- | | | | | |
| Treatment | 5 | 0.0373 | 0.1773 | 0.0022 | 0.0198 | 0.0922 | 0.0598 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test

### 4.4.5 Nitrate and Phosphorus Leaching

The amount of nitrogen that was lost below the root zone of barley in 2010 ranged from 21.4 to 133.5 kg ha$^{-1}$ and was greater than that in 2011 which ranged from 18.8 to 82.4 kg ha$^{-1}$ (Table 4.8). This could be related to greater N application in 2010 than in 2011 (Table 4.1) coupled with greater amount of precipitation in 2010. Thomsen (2005) concluded that the concentration of nitrate had a greater influence on nitrate leaching than the volume of leachate. Overall, the F-5000 had the greatest loss of nitrate-nitrogen. This suggests that, from an environmental point of view, the F-5000 treatment may not be sustainable on this sandy soil. In 2010, 79% (112 kg ha$^{-1}$), 55% (40 kg ha$^{-1}$) and 27 % (19.5 kg ha$^{-1}$) of applied available N was lost from F-5000, com-2500 and F-2500, respectively. A similar result was obtained in 2011 where F-5000 lost 80% (63.6 kg ha$^{-1}$), F-2500 lost 79% (31.5 kg ha$^{-1}$) compared to LH-5000 that lost 40% (32 kg ha$^{-1}$) and LH-2500 that lost 9%.(3.5 kg ha$^{-1}$). In other words, treatments which received fertilizer lost more than half of their plant available N through leaching. This loss of fertilizer may adversely affect the adjacent environment and may account for the lack of a significant increase in grain yield during the two years of study. Payet et al. (2009) reported that smaller, or equivalent, nitrate leaching occurred in fields that received LHM at sufficient rates than fields which received mineral fertilizers.

Table 4.8. Total amount of water, nitrate and phosphorus leached in 2010 and 2011

| ***Treatment*** | ***Water (cm)*** | | ***FWMC (N)*** | | ***FWMC (P)*** | | ***Nitrogen*** | | ***Phosphorus*** | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | ----------------(*mg L$^{-1}$*)-------------- | | | | ---------------(*kg ha$^{-1}$*)-------------- | | | |
| | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** | ***2010*** | ***2011*** |
| LH-2500 | 19.5 | 19.9 | 19.1 | 12.3 ab | 0.021 | 0.023 | 37.3 | 22.3 | 0.036 | 0.045 |
| LH-5000 | 18.5 | 21.5 | 16.4 | 25.1 ab | 0.032 | 0.039 | 29.6 | 50.9 | 0.058 | 0.083 |
| Com-2500 | 18.2 | 14.6 | 33.9 | 18.5 ab | 0.024 | 0.006 | 61.5 | 34.7 | 0.043 | 0.012 |
| F-5000 | 17.0 | 20.7 | 81.1 | 40.7 a | 0.013 | 0.004 | 133.5 | 82.4 | 0.022 | 0.009 |
| F-2500 | 18.1 | 25.0 | 22.8 | 22.0 ab | 0.006 | 0.016 | 40.8 | 50.3 | 0.012 | 0.040 |
| Control | 17.9 | 23.1 | 12.1 | 10.9 b | 0.018 | 0.014 | 21.4 | 18.76 | 0.032 | 0.034 |
| | ----------------------------------- *P > F*----------------------------------- | | | | | | | | | |
| ***Trt effect*** | 0.9482 | 0.4707 | 0.1795 | 0.0867 | 0.8571 | 0.4346 | 0.2575 | 0.1759 | 0.8334 | 0.4346 |

Means with the same letter within the column are not significantly different at P <0.1 according to Tukey-Kramer test

FWMC: Flow Weighted Mean Concentration

Generally, the late planting in this experiment resulted in lack of soil water use in the early part of the growing season and yield reduction. These two factors probably have exaggerated the amount of nitrate leaching loss. The magnitude of $NO_3^-$-N loss would probably be less if we had planted at the recommended time and achieved normal crop yield.

The total nitrate leached from the control plots was 21 and 19 kg $ha^{-1}$ in 2010 and 2011, respectively, suggesting that although nitrate leaching can be reduced by withholding supplemental N, it may be unavoidable particularly in sandy soils.

The amount of precipitation received during the growing season in 2010 was 36 cm, which was 142% of the 30 year normal growing season precipitation. The large amounts of precipitation led to a leaching loss in the range of 17 to 19.5 cm (Table 4.8). In 2011, the amount of precipitation received during the growing season was 75% of the 30 year normal growing season precipitation. Although the amount of precipitation in 2010 was greater than that of 2011, loss of water in all treatments, except com-2500, was greater in 2011 than in 2010. Most of the leaching that was measured in 2011 was due to the precipitation in fall 2010 and snowfall till April of 2011. It is likely that the wet year of 2010 predisposed the soil to leaching in spring 2011. There were no statistical differences in the amounts of water lost from various treatments in the two years of study as a result of spatial variability of leached water. A similar result was obtained by Bergstrom and Kirchmann (2006) who reported no significant difference in drainage volume between fertilizer and slurry treatments.

Although there were no treatment effects on the flow weighted mean concentration (FWMC) of N in 2010 its trend was in the order of F-5000 > com-2500 > F-2500 > LH-2500 > LH-5000 > control. Also, the FWMC of N in all treatments was greater than 10 mg $L^{-1}$ which is the drinking water threshold for health risk to humans. When the residual soil nitrogen is not taken into account in obtaining the appropriate rate of manure application, accumulation of excess nitrate in soil occurs, leading to nitrate leaching, especially in a sandy soil with high hydraulic conductivity (Olson et al. 2009).

In 2011, F-5000 showed the greatest FWMC of N and was significantly greater than the control. There were no statistical differences in the FWMC in manured compared to fertilized plots. Nevertheless, F-5000 and F-2500 had numerically greater FWMC of nitrate (40.7 and 22.0 respectively) compared to LH-5000 and LH-2500 (25.1 and 12.3 respectively). This shows that when applied at equivalent rates, nitrogen from fertilizer is more environmentally available than nitrogen from LHM. Bittman et al. (2005) found that fertilizer plots had more N leaching loss than manured plots after six years of liquid dairy manure and fertilizer application. They suggested that bacteria in manured soil may immobilize nitrate resulting in less mineral N for leaching compared to fertilized soil. Qian and Schoenau (2000) reported that 20–30% of the organic N in the liquid hog manure became available during the year of application, while LHM as a source of nitrogen had 60–70% N availability as urea. In contrast, Bergstrom and Kirchmann (2006) reported that nitrate leaching from different rates of liquid hog manure was significantly greater than that from fertilizer.

Although the amount of added phosphorus was different for individual treatments in both years of the study, there was no significant treatment effect on FWMC of P. In 2010, maximum P leaching occurred in the LH-5000 treatment (58 g P $ha^{-1}$) followed by com-2500 (43 g P $ha^{-1}$) and LH-2500 (36 g P $ha^{-1}$). Only 12 and 22 g P $ha^{-1}$ was lost through leaching from F-2500 and F-5000 respectively, while 39 kg P $ha^{-1}$ was applied to F-2500 and 78 kg P $ha^{-1}$ to F-5000. In 2011, LH-5000 and LH-2500 had the greatest amount of leached phosphorus, 83 and 45 g P $ha^{-1}$ respectively. Although not statistically significant the results obtained in both years showed that more P was lost from liquid hog manure than from fertilizer, a trend that was the opposite that of nitrogen. Ajiboye et al. (2003) reported greater concentration of P in soil solution after hog manure application compared to inorganic P fertilizer (MAP) application. Campbell and Racz (1975) suggested that the greater movement of P under manure amended soil was partly due to the movement of organic P and the interference of organic matter in manure with the sorption of inorganic P by soil particles resulting in an increase of the plant available P and leaching loss.
Our results support the findings of Bergstrom and Kirchmann (2006); Sørensen and Rubæk (2011) who reported 23-148 and 40-165 g P $ha^{-1}$ $year^{-1}$ respectively from a sandy soil after application of liquid hog manure. They indicated that adsorption of phosphorus by the soil and

the absence of preferential flow through soil cracks and macropores resulted in small phosphorus leaching.

#### 4.4.5.1 Treatment effect on seasonal variation of nitrate leaching in 2010 and 2011

Results of monthly leachate collection in 2010 (Figure 4.2) showed that in spite of spatial variability resulting in no significant treatment effect, in June before treatment application, F-5000 had the greatest amount of leached nitrate-N (76.7 kg $ha^{-1}$) followed by com-2500 (30 kg $ha^{-1}$) (Figure 4.2). In July, after treatment application and seeding, F-5000 also had the greatest leached nitrate (50 kg $ha^{-1}$) that was significantly greater than nitrate leaching from the control (4.7 kg $ha^{-1}$). Thomsen, 2005 also reported an increase of nitrate loss with an increase in nitrogen application rate. Total precipitation between the first two leachate collections was 12.2 cm, about one-half of which was leached though the soil matrix. This amount of water can easily transport the added nitrogen out of the root zone without the need for time to mineralize the organic N. In August 2010, there were no significant effects of treatment on nitrate leaching. However, F-5000 and LH-5000 had the greatest nitrate leaching among all treatments.

In May 2011, there were no significant effects of treatment on nitrate leaching (Figure 4.2). The amount of nitrate lost from com-2500 treatment was 19.8 kg $ha^{-1}$ which was the greatest among all treatments. It is presumably due to residual nitrogen of the added urea in 2010 and movement of nitrate after snow melt in spring. Moreover, the availability of compost-N depends on the mineralization rate of the organic-N. This can reduce the risk of nitrate leaching during the growing season but there is the risk of N mineralization and nitrification after crop harvest that leads to N leaching (Gingerich, 1999).

However, in June 2011, 48 kg $ha^{-1}$ N was lost from F-5000, which was twice of that from LH-5000. In October 2011, nitrate leaching was significantly ($P<0.1$) affected by treatment effects. LH-5000, F-5000 and F-2500 lost about 7 kg $ha^{-1}$ nitrogen which was significantly greater than the losses from the control and LH-2500 (Figure 4.2). The reason why there was no statistical difference in the amount of nitrate-N that was lost from LH-5000 and F-5000 could be due to the mineralization of organic N in manure during the warm and wet growing season, and also because a larger amount of nitrate-N was lost from fertilizer treatment at the beginning of growing season than from manure. The same trend was observed in August 2010 although treatment effect was not statistically significant. These results show that the risk of nitrate leaching is smaller with manure than with synthetic fertilizer when applied at the same estimated rate of available nitrogen. Total rainfall from June to October, the time period between the second and third leachate collections, was 10 cm and about one-third (29%) of this percolated below the root zone. Crop water uptake during the growing season caused less leached water and consequently less N leaching than in spring time.

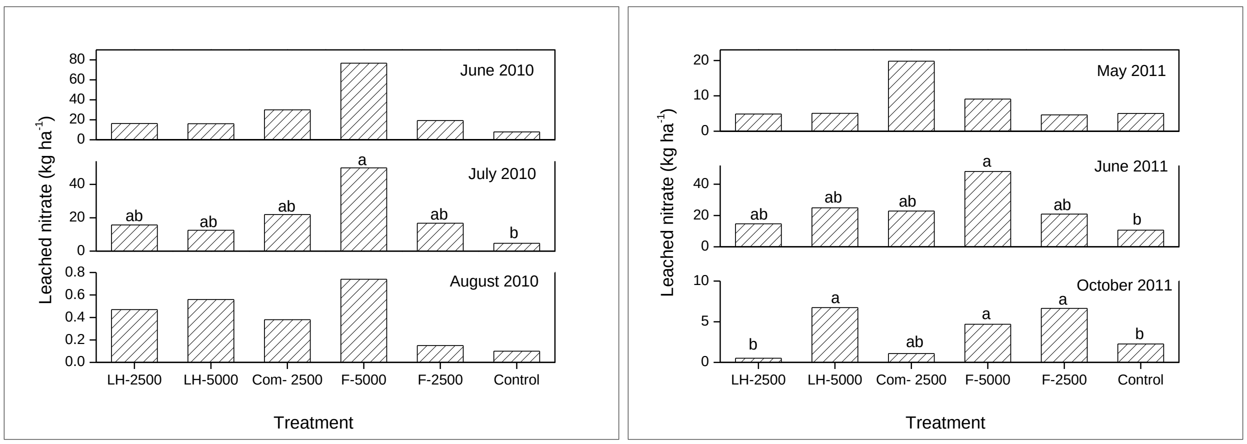


Fig. 4.2. Amount of $NO_3^-$ that was leached at different collection times during the 2010 and 2011 growing seasons

Means with the same letter are not significantly different at P <0.1 according to Tukey-Kramer test.
LH-2500: Hog manure at 2500 gal ac$^{-1}$
LH-5000: Hog manure at 5000 gal ac$^{-1}$
Com-2500: Compost application at N rate similar to LH-2500
F-5000: Fertilizer application at N rate similar to LH-5000
F-2500: Fertilizer application at N rate similar to LH-2500
Control: Seeded-No treatment

#### 4.4.5.2 Relationship between leachate volume and the amount of leached nitrate

In further data analysis, leachate volume was plotted against the amount of nitrate leached to determine the influence of drainage volume on nitrate leaching for each rate and nitrogen source. Also, this analysis helps account for spatial variability between plots with respect to infiltration and hydraulic conductivity. For fertilizer plots, the relationship between drainage volume and the amount of nitrate leached increased with increasing rate of N and this may be due to the surplus N in the soil profile that was available for leaching (Figure 4.3). The slope of the regression between nitrate leached and leachate volume showed the mean concentration of nitrate. The slope for the F-5000 treatment was 0.5 which indicates that the amount of nitrogen leached (kg $ha^{-1}$) was approximately one-half of the amount of drainage volume (eg. 100 mm of drainage produced 50 kg $ha^{-1}$ of leached nitrate-N). However, for F-2500, the relationship between nitrate leached and leachate volume was weak and the slope of the regression was similar to that of the control (Figure 4.3). In addition, the significantly greater slope of F-5000 than that of F-2500 (P=0.006) represents a greater loss of $NO_3^-$-N from F-5000 than from F-2500. The same trend was observed for manure treated plots. There was also significant difference in the slopes of F-5000 and LH-5000 (P= 0.07). The slope for LH-5000 was 0.25 which is one-half of the slope of F-5000. It can, thus, be concluded that with the same amount of drainage, F-5000 produced twice the amount of leached nitrate-N compared to LH-5000.

When the three treatments that received N based on the 2500 gallon $ac^{-1}$ of liquid hog manure are compared, the slope for F-2500 and LH-2500 was the same (Figure 4.4). This suggests that the risk of nitrate leaching from both treatments is the same. However, com-2500 had significant greater slope (P=0.02 and P=0.04) than F-2500 and LH-2500, perhaps, as a result of adding urea to the compost treatment.

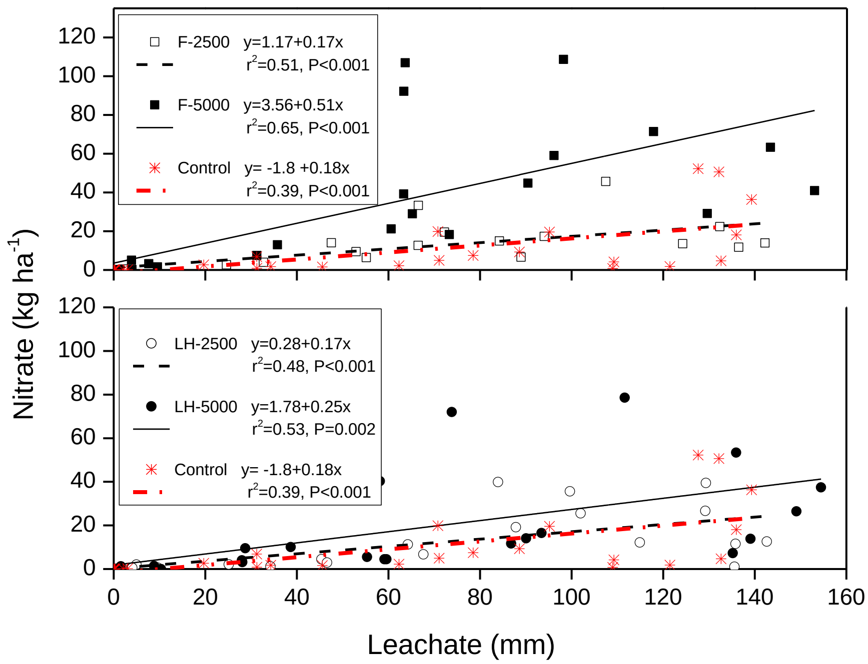


Fig. 4.3. Relationship between leachate volume and nitrate leached for different rate of applied fertilizers and liquid manures in 2010 and 2011.

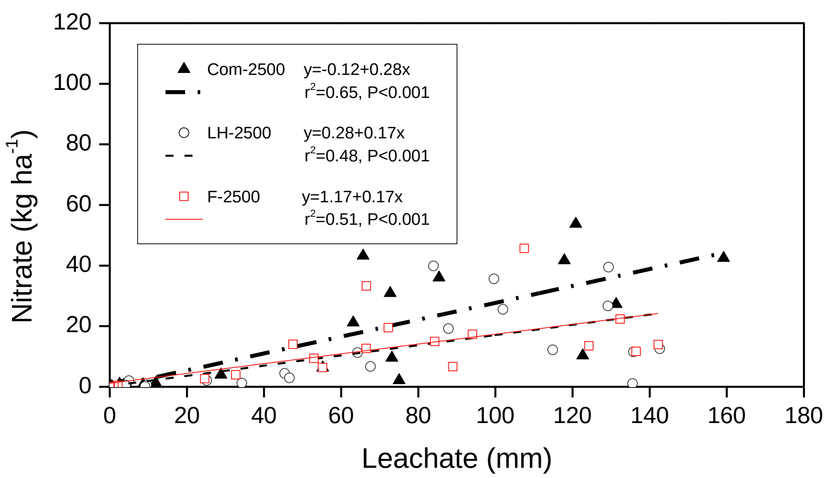


Fig. 4.4. Relationship between leachate volume and nitrate leached for low N rate treatments in 2010 and 2011.

### 4.4.6 Distribution of Nitrate within the Soil Profile

In the spring of 2010, prior to manure addition, the concentration of nitrate-N in the top 15 cm was between 4 to 6 mg $kg^{-1}$ in all plots (Figure 4.5a). Nitrate distribution within the soil may not provide us with an accurate measure of nitrate loss since it could have moved beyond the sampling depth (Olatuyi et al. 2012). However, accumulation of nitrate with depth is an indirect evidence of leaching (Miller et al. 2011).

After the application of manure, nitrate concentration in the surface soil layer increased, ranging from 5 to 24 mg $kg^{-1}$ and the trend in nitrate concentration in the top 15 cm depth was in the order of F-5000 > com-2500 > F-2500 > LH-5000 ≥ LH-2500 > control. The fertilizer and compost treatments that received urea, a readily available source of nitrogen, had greater nitrate concentration in the top 15 cm depth compared to manure (Figure 4.5b). The trend shown above parallels the available nitrogen in the various treatments. Sayem (2014) have shown that LHM is about 50% the equivalence of nitrogen fertilizer when applied on the same available nitrogen basis. These authors measured negligible amounts of mineralized nitrogen from LHM in field and laboratory studies.

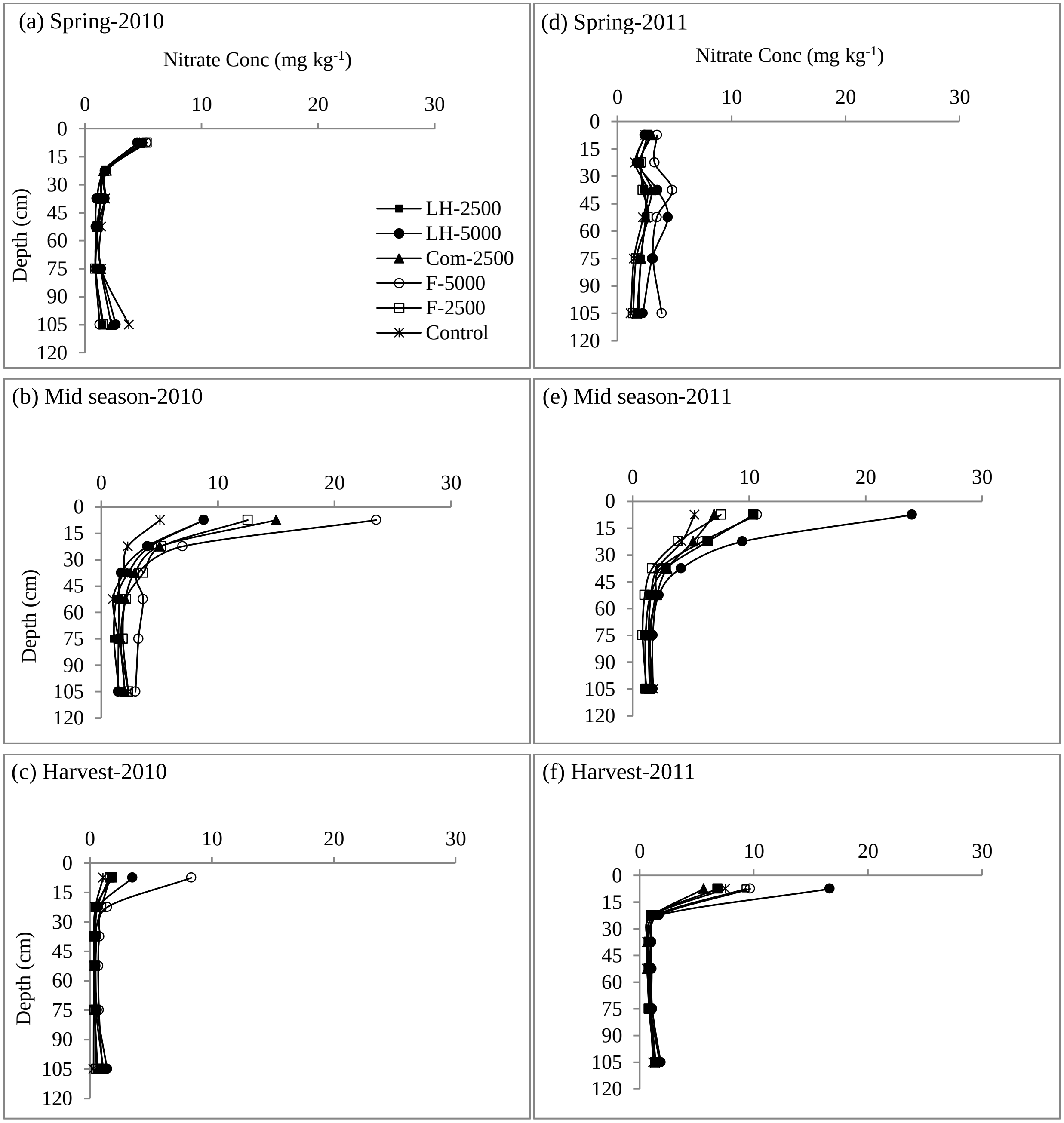


Fig. 4.5. Distribution of nitrate within the soil profile in 2010 and 2011

By September (harvest), there was a significant treatment effect, depth and treatment x depth interaction for nitrate movement (Table 4.9). Nitrate concentration at the soil surface (0-15cm) had declined compared to what was measured at mid-season partly due to crop uptake and partly as a result of the downward movement of nitrate as indicated by the lysimeter data (Figure 4.2). Generally, high N rate of manure and fertilizer showed significantly greater concentrations of N within the first depths of the soil profile than other treatments (Figure 4.5c and Table 4.9a).

In the spring of 2011, there was a bulge of nitrate-N with concentrations that were similar to those in the top 15 cm in the fall of 2010. This bulge of nitrate concentration that was centered around the 45 cm depth is an indication of nitrate movement due to excess of nitrate from the previous year (Figure 4.5d). Sun et al. (2008) showed that accumulated residual soil nitrate can be readily leached during wet seasons, and dry and wet cycles resulted in large nitrate losses. Generally, on the Canadian Prairies most of the nitrate leaching occurs in the spring after snowmelt and before plants can take up the mineralized nitrate (Enns 2004). The F-5000 and the LH-5000 treatments showed the greatest nitrate concentration at the surface and throughout the soil profile which was significantly greater than that of the control (Table 4.9b). This trend is somewhat parallel to the total amount of leachate nitrate in these treatments in 2011 (Figure 4.5d and Table 4.8). Unlike the results obtained by Entz et al. (2001) showing no downward movement of nitrate in the unfertilized system, the results of soil sampling in our study showed evidence of nitrate movement and leaching in the unfertilized plots. This supports the results that we obtained from the lysimeters where 21.4 and 18.7 kg $ha^{-1}$ of nitrogen were lost from the control plot in 2010 and 2011, respectively (Table 4.8).

Table 4.9a. Treatment effects on the vertical distribution of nitrate, ammonium and soil moisture in 2010

| | | ***Spring*** | | | ***Mid-season*** | | | ***Harvest*** | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $NO_3$ | $NH_4$ | $H_2O$ | $NO_3$ | $NH_4$ | $H_2O$ | $NO_3$ | $NH_4$ | $H_2O$ |
| ***Model effect*** | ***d.f.*** | | | | ***---------------------- P value --------------------*** | | | | | |
| Treatment | 5 | 0.9542 | 0.981 | 0.998 | 0.1314 | 0.1189 | 0.999 | **<.0001** | 0.4113 | 0.9447 |
| Depth | 5 | **0.0006** | 0.4457 | **<.0001** | **<.0001** | **0.0017** | **<.0001** | **<.0001** | **0.0504** | **<.0001** |
| Treatment×Depth | 25 | 0.8167 | 0.9119 | 0.9317 | 0.739 | 0.5253 | 0.989 | **0.0023** | 0.7016 | 0.9901 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

Table 4.9b. Treatment effects on the vertical distribution of nitrate, ammonium and soil moisture in 2011

| | | ***Spring*** | | | ***Mid-season*** | | | ***Harvest*** | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $NO_3$ | $NH_4$ | $H_2O$ | $NO_3$ | $NH_4$ | $H_2O$ | $NO_3$ | $NH_4$ | $H_2O$ |
| ***Model effect*** | ***d.f.*** | | | | ***---------------------- P value ---------------------*** | | | | | |
| Treatment | 5 | **0.099** | 0.167 | 0.9999 | 0.5268 | 0.6058 | 0.9717 | 0.3627 | 0.8733 | 0.9977 |
| Depth | 5 | 0.3261 | **<.0001** | **<.0001** | **<.0001** | 0.5259 | **<.0001** | **<.0001** | **0.0014** | **<.0001** |
| Treatment×Depth | 25 | 0.6063 | 0.7085 | 0.9726 | 0.8272 | **0.0206** | 0.8771 | 0.6858 | 0.8973 | 0.9522 |

Means with the same letter within the column are not significantly different at P< 0.1 according to Tukey-Kramer test.

Campbell et al. (1993) concluded that insufficient nutrient content of soil may lead to poor crop growth and consequently increase of nitrate leaching. This may be the case for the control plots.

At mid-season, due to high variability, the treatment effect was not statistically significant ($P>0.1$). However, LH-5000 and F-5000 had the greatest nitrate concentration in the top 15 cm depth (Figure 4.5e). By harvest, nitrate content of soil increased slightly at the depth of 120 cm showing downward movement of nitrate. Our results support the findings of Woodard et al. (2003) that soil nitrate at harvest was the smallest due to plant uptake and reduced mineralization rate as a result of cooler soil temperature. The LH-5000 and fertilizer plots showed the greatest concentration of nitrate in the first 15 cm of soil. These results are in line with leachate data collected on October 2011 after final soil sampling (Figure 4.5f).

### 4.4.7 Distribution of Ammonium within the Soil Profile

In the spring of 2010, the soil ammonium contents for all treatments were similar and ranged from 1 to 1.5 mg $kg^{-1}$ (Figure 4.6a). A similar result was reported by Eghball et al. (2004) who showed no treatment effects on ammonium concentration in all three residual years after three continuous years of manure and compost application. Trindade et al. (2008) reported that, in contrast to soil nitrate, ammonium showed no treatment effects after three years of application of cattle slurry and synthetic fertilizer at different rates. In our study, after application of manure and fertilizer, the amount of ammonium increased with F-5000 showing the greatest concentration of ammonium (Figure 4.6b). By harvest, the ammonium concentrations decreased to the levels they were in the spring which is an indication of the nitrification and crop uptake. A bulge that was centered at 45 cm depth (Figure 4.6c), is presumably as a result of a decrease of ammonium at the soil surface and its downward movement in a sandy soil.

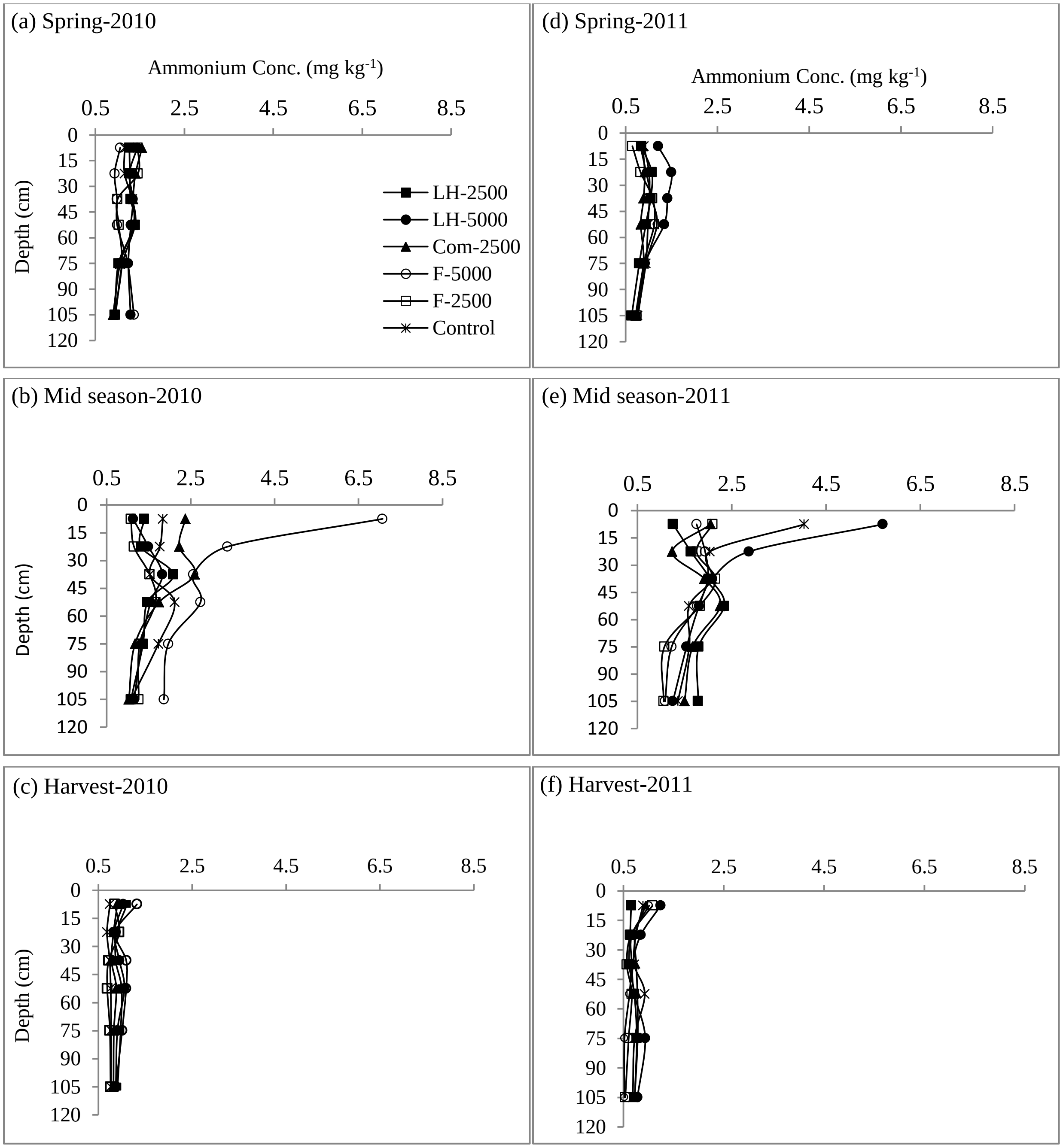


Fig. 4.6. Distribution of ammonium within the soil profile in 2010 and 2011

The results obtained in 2011 was similar to that in 2010 with very small amounts of ammonium in the spring, slightly increasing at mid-season following manure and fertilizer additions, and diminishing at harvest due to crop uptake and nitrification. Although there was no significant treatment effect, LH-5000 showed the greatest ammonium concentration within the soil profile (Figure 4.6d). Sørensen and Rubæk (2011) reported less than 0.3 kg $ha^{-1}$ of ammonium leaching in one year after application of hog slurry in two different loamy sand and sandy loam soils which was negligible compared to nitrate leaching.

### 4.4.8 Distribution of Phosphorus within the Soil Profile

Compared to nitrate which showed significant downward movement, accumulation of phosphorus was evident only in the upper layer of soil (0-15 cm) (Figure 4.7). Miller et al. (2011) found maximum soil test phosphorus in the 0 to 30 cm depth and no treatment differences in soil phosphorus concentration below 30 cm for different manure treatments after nine years of application of manures. In comparison, Eghball (2003) reported accumulation of phosphorus in the 30 to 60 cm of the soil profile in a sandy loam after 20 years of application of manure. As such, in long term application of manure, phosphorus may be subjected to leaching which often occurs on a time scale of decades or more (Radcliffe and Cabrera 2007). Since most of the agronomic and environmental recommendations considered the first 15 cm of the soil profile, soil phosphorus data of 0-15 cm depth was reported in this study separately.

In spring 2010, soil test P in the control and treatment plots (Figure 4.7a) were agronomically in the low (< 10 mg $kg^{-1}$ Olsen P) range according to the Manitoba Soil Fertility Guide (MAFRI, 2007), except LH-5000 which had high STP (20 mg $kg^{-1}$ Olsen P) indicating that the background P fertility of the treatment was high, even without the addition of manure. This could be as a result of high rate of manure application (5000 gallons per acre) and eight years of phosphorus accumulation (Figure 4.7a).

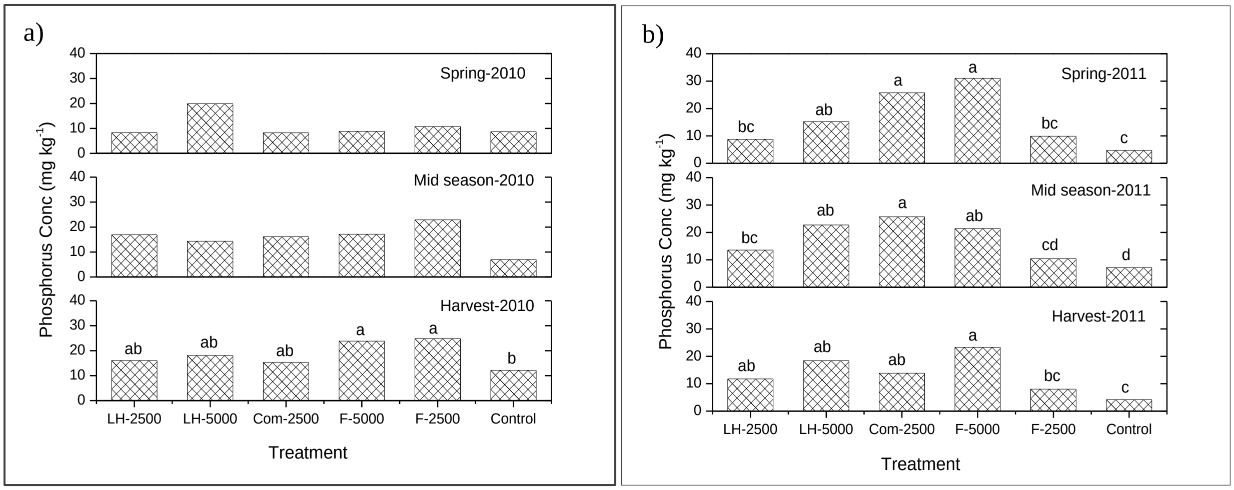


Fig. 4.7. Soil test P during three times of soil sampling in 2010 and 2011

Means with the same letter are not significantly different at P <0.1 according to Tukey-Kramer test.
LH-2500: Hog manure at 2500 gal ac$^{-1}$
LH-5000: Hog manure at 5000 gal ac$^{-1}$
Com-2500: Compost application at N rate similar to LH-2500
F-5000: Fertilizer application at N rate similar to LH-5000
F-2500: Fertilizer application at N rate similar to LH-2500
Control: Seeded-No treatment

At harvest in 2010, the highest concentration of soil test phosphorus was observed in the fertilizer treatments (F-5000 and F-2500), both of which were significantly greater (P <0.1) than the control plots. Kashem et al. (2004) showed that at the same P-based application rate, the labile soil P was in the order of inorganic P fertilizers > liquid manures > solid or composted manures.

In the spring of 2011, soil test P in the control plots was in the very low (< 5 mg kg$^{-1}$ Olsen P) range according to the Manitoba Soil Fertility Guide (MAFRI 2007). However, before application of manure, F-5000 and com-2500 showed the greatest concentrations of phosphorus in the very high range (>20 mg kg$^{-1}$ Olsen P) which were significantly greater than the control (Figure 4.7b). The high concentration of phosphorus at the 15 cm depth of F-5000 was due to the residual effect of applied MAP (78 kg ha$^{-1}$ P) in 2010. The N:P ratio of compost was lower than the N:P ratio of the harvested portion of the crops, resulting in the accumulation of phosphorus (Evanylo et al. 2008). Sharpley and Moyer (2000) reported that the application of compost based on crops N requirements may raise the risk of P transport in runoff.

The increase of soil phosphorus content of com-2500 and F-5000 from harvest 2010 to spring 2011 was presumably due to mineralization of organic matter or cell lyses of microorganisms after spring snowmelt and release of phosphorus. However, STP of other treatments including LH-2500, LH-5000, F-2500 and control decreased. Generally, this decrease and increase of STP could be related to variability of Olson P with environmental conditions such as soil temperature and moisture.

By mid-season in 2011, com-2500, LH-5000 and F-5000 had STP values that were significantly greater than for the control. It is likely that the residual effect of MAP from 2010 in fertilized treatments resulted in no significant difference between these treatments and their corresponding manured ones.

At harvest, STP decreased in all treatments partly due to plant uptake. However, F-5000 had the greatest STP and was significantly greater than the control. Regardless of the type of amendments, STP in the high rates of fertilizer and manure (F-5000 or LH-5000) was rated high to very high, according to the Manitoba Soil Fertility Guide (MAFRI, 2007). There were no statistically significant differences between F-2500, LH-2500 and com-2500 (Figure 4.7).

### 4.4.9 Water Distribution in Soil Profile

In the spring (2010), due to snowmelt and spring rainfall, volumetric soil water content in the first 15 cm of soil profile was about 20% and increased with depth (Figure 4.8a). There were no discernible differences in soil moisture between treatments down to a depth of 120 cm at different sampling times (Table 4.9a). Soil water in the top 60 cm depth was about 10% by volume at harvest, an indication of water uptake by the crop. However, at depth, soil water content increased up to 30% by volume, showing the movement of water through the soil profile during the growing season.

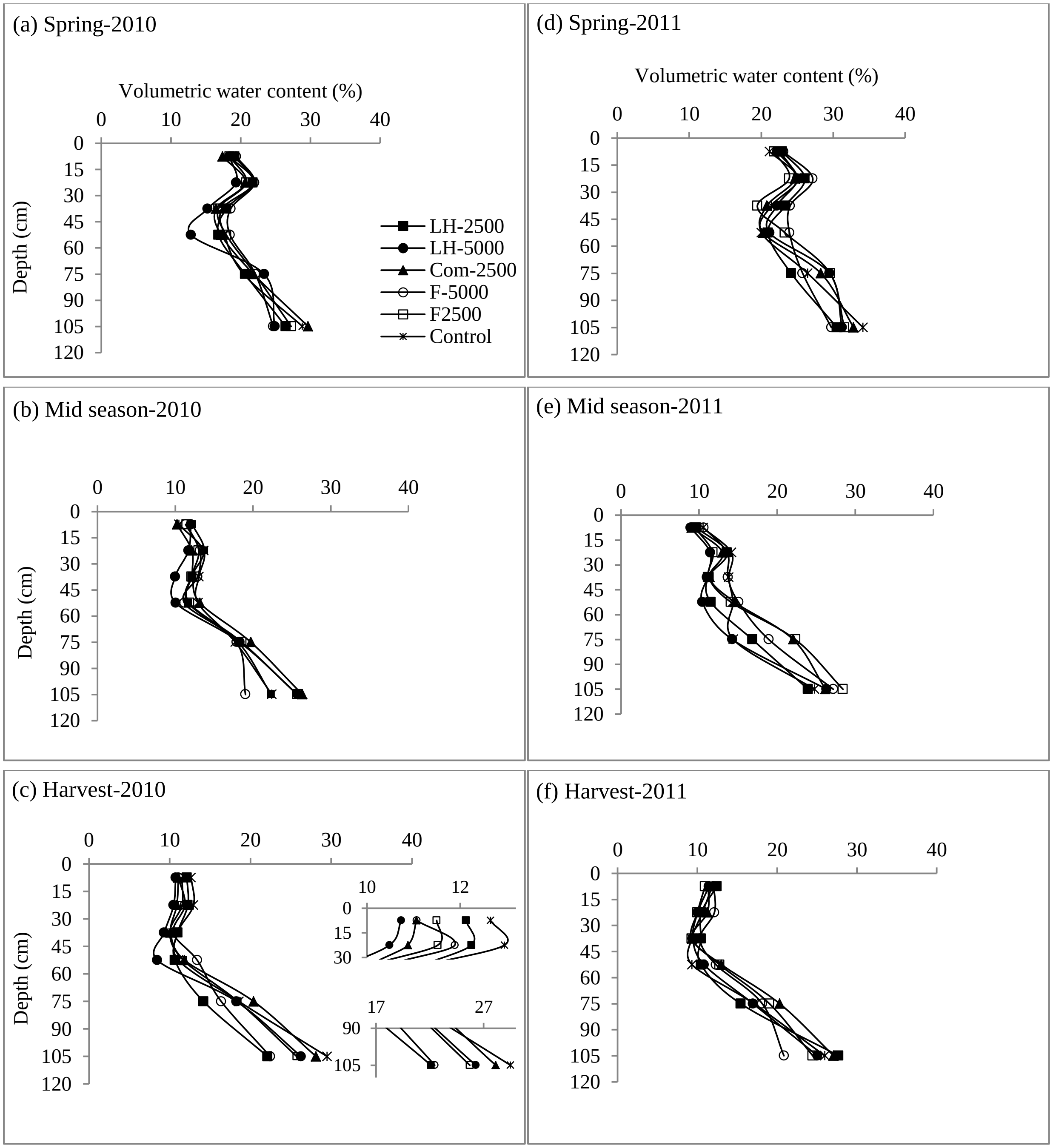


Fig. 4.8. Soil water distribution during the growing season of 2010 and 2011

Compared to the harvest of 2010, soil moisture content in the spring of 2011 increased due to snowmelt and precipitation during fall and early spring (Figure 4.8d). The pattern of soil water distribution in 2011 was similar to what we described above for 2010. Although there was no significant treatment effect on soil moisture content at harvest, the greater uptake of water and subsequently greater yield in the manure and fertilizer treatments resulted in smaller soil water than the control (Figure 4.8c). Campbell et al. (1993) demonstrated that the amount of water in the soil profile was inversely related to fertility level and crop yield. Poor fertility and subsequent low yield is often associated with greater soil water content due to reduced root growth and leaf area. The greater biomass yield in the treated plots resulted in greater water uptake and smaller soil water content than the control treatment.

## 4.5 CONCLUSIONS

Results from this study showed the application of high rates of fertilizer and hog manure would increase the risk of nitrate leaching on the sandy soils over the ADA. The concentration of N in the leachate and total N leached from fertilizer amended plots were greater than in manured plots. More than one-half of the plant available N of this treatment was lost through leaching, representing a substantial agronomic, environmental and economic loss. Therefore, application of nitrogen equivalent to that of F-5000 is not recommended on this pervious sand.

There were no differences between liquid hog manure and their corresponding fertilizer treatments for biomass and N uptake. Lack of difference between LH-2500 and LH-5000 for grain yield suggested that application of liquid hog manure at rate 2500 kg $ha^{-1}$ was economically and environmentally more desirable and recommended for the permeable sandy soil of the Carberry site.

Phosphorus leaching was negligible and the maximum loss of phosphorus was from the higher rate of liquid hog manure and compost. The previous eight years of manure application without considering soil test P as well as the low N:P of liquid hog manure and compost resulted in accumulation of phosphorus in the soil with the potential to increase the concentration of P in leachate.

In regions with a high risk of nutrient contamination, spring soil N and P soil tests may provide better estimates of available nutrients on soils that have good internal drainage such as coarse loamy sand soil.

As well, spring soil nitrate sampling may also provide a better estimate of nitrate availability in soils that have a history of receiving solid manure due to mineralization of the organic N in the manure

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# 5. ESTIMATE OF SOIL WATER CONTENT AND AMOUNT OF LEACHED WATER USING THE VSMB MODEL


## 5.1 ABSTRACT

The chemical characteristics of $NO_3^-$-N make it susceptible to leaching through the soil system and into water. Therefore, accurate simulation of soil moisture content and quantitative assessment of leached water can provide valuable information about the potential distribution and movement of $NO_3^-$-N through the soil profile and $NO_3^-$-N loss. The Versatile Soil Moisture Budget (VSMB) has been widely used on the Canadian prairies to simulate soil moisture content and evapotranspiration of crop and grass lands. However, the VSMB has not been tested for simulation of drainage water below the root zone. The objectives of this study were to evaluate the accuracy of simulated soil moisture content of two different cropping systems and compare model simulated drainage with field data obtained from field core lysimeters on two soil types. In this study, two modified versions of the VSMB model reported by Akinremi et al. (1996) and Hayashi et al. (2010) were used. Results showed that Akinremi's version had better agreement between simulated and observed soil moisture distribution than Hayashi's version. Both versions of VSMB underestimated soil moisture content in the 45-120 cm depth especially in the second year of simulation. The VSMB grossly underestimated the amount of leached water, possibly due to an overestimation of evapotranspiration. For example, the amount of leached water that was simulated during the two years at Carberry was 6.6 cm compared to 39.0 cm that was measured using field core lysimeters. As well, at Carman, the amount of simulated leachate was 7.8 and 6.7 cm for annual and perennial cropping system respectively while 58.7 and 51.0 cm of leachate on average were collected from field lysimeters. In general, our results show that a good simulation of soil water content may not translate into good simulation of other components of the hydrologic cycle. Both versions of VSMB will need to be modified before they can be used to estimate $NO_3^-$-N leaching.

## 5.2 INTRODUCTION

Accurate measurement of the distribution and amount of $NO_3^-$-N in the soil profile is needed for precise fertilizer recommendations and for reducing the risk of $NO_3^-$-N contamination of ground water. This information can be obtained from either direct measurements or from model simulations (Beckie et al. 1995). Estimation of nutrient transport by models is more cost effective and less labor and time consuming compared to direct field measurements especially for assessing long term risk. Therefore, simulation models with varying degrees of complexity are used to simulate $NO_3^-$-N leaching from croplands (Olatuyi 2011). Soil $NO_3^-$-N is a mobile ion that can move readily with water. Hence, evaluation of water flow models can provide valuable information on the distribution and movement of $NO_3^-$-N through the soil profile (Beckie et al. 1994; Akinremi et al. 2005).

The Versatile Soil Moisture Budget (VSMB) was developed to simulate vertical, one-dimensional soil moisture fluxes and it has been used extensively under several conditions on the Canadian prairies (Baier and Robertson 1996; Akinremi et al. 1996; Hayashi et al. 2010). As a simple moisture budget model, VSMB requires few input parameters. However, this model needs to be validated for each new soil, crop, and climatic condition (Akinremi and Mc. Ginn 1996).

Akinremi et al. (1996) modified the VSMB to estimate soil moisture for drought monitoring and crop yield estimation at Swift Current, Saskatchewan and Lethbridge, Alberta, Canada. Four major components of the original VSMB (Baier and Robertson, 1966) were modified including potential evapotranspiration, redistribution of soil moisture between soil layers, rainfall runoff and soil surface temperature. They replaced the Baier-Robertson equation with the Priestly and Taylor equation to estimate potential evapotranspiration (PE). In this approach solar radiation created a better estimate of PE than temperature and latitude which were used in Baier-Robertson equation. The cascade algorithm from the Ceres model was used for the redistribution of soil moisture by unsaturated flow. The value of 80 was assumed for Soil Conservation Service (SCS) curve number and was included in input data in order to calculate the rainfall runoff. Finally, the temperature algorithm of the EPIC model was used to calculate the soil surface temperature for determining the freeze-thaw cycle and estimating snowmelt runoff.

Hayashi et al. (2010) simulated soil moisture content and evaporation under grassland using field data near Calgary, Alberta. They revised the cascade algorithm using the hydraulic diffusivity that was based on measured soil water storage parameters, coefficient of van Genuchten's equation and saturated hydraulic conductivity of studied field. The grass extraction coefficient was modified in order to get better simulate evapotranspiration. They reported that the best evaporation estimate was obtained with a deeper soil profile than 1.2 m. In the semi-arid weather condition, the model was not very sensitive to soil water storage parameters including soil moisture content at field capacity, saturation, permanent wilting point, and available water holding capacity.

In Manitoba, Ojo (2012) conducted a study to modify and validate the VSMB (Akinremi et al. (1996) version) for estimating volumetric soil water content at 13 different locations across Central and Western Manitoba. The author included physiological days as a crop growth simulation function for canola growth and a three-phase growth stage approach for grasslands adapted from Hayashi et al. (2010). Due to an increase in the number of weather stations with record of additional weather parameters such as wind speed, humidity and solar radiation, Ojo replaced the Priestley-Taylor equation with the Penman-Monteith equation which needed more weather parameters for estimating evapotranspiration. Ojo reported that the modified VSMB did not significantly improve the soil moisture simulation at most of the locations. Similar to the results of Akinremi et al. (1996), the model estimated the soil moisture content of the surface layer with greater accuracy; however, the root mean square error increased at lower depths.

The VSMB has not been tested for simulation of drainage water below the root zone from field studies. Because accurate simulation of one part of the hydrologic cycle can affect the accuracy of the other parts, the objectives of this study were to evaluate the accuracy of simulated soil moisture content of two cropping systems by VSMB and compare simulated drainage with real biophysical data obtained from field lysimeters on two soil types.

## 5.3 MATERIALS AND METHODS

### 5.3.1 VSMB MODEL PARAMETERS

VSMB is a one-dimensional water balance model that divides the soil column into a user-defined number of layers up to a maximum of six. (typically 0.3 m in thickness) and calculates soil water balance daily. Rainfall and snowmelt, after subtracting any surface runoff, is added to the top layer, which is subsequently distributed to the next layer by gravity and soil hydraulic diffusivity. Evapotranspiration is extracted from individual layers according to the meteorological forcing, soil moisture condition, and plant growth stage. The model uses weather and soil/plant information as input data (see Appendix 5.7.A and 5.7.B). In this study, two modified versions of the VSMB model reported by Akinremi et al. (1996) and Hayashi et al. (2010) were used.

#### 5.3.1.1 Plant Growth

Both versions of the VSMB simulate crop growth using the bio-meteorological time (BMT) approach of Robertson (1968). While Akinremi's version simulates cereal crop growth (wheat and barley), Hayashi's version can be set up either for a grass or a cereal crop. The BMT crop growth subroutine for cereals has five stages including planting, emergence, jointing, heading, soft dough and maturity. Saiyed et al. (2009) reported that wheat phenological development was better estimated using the BMT compared to growing degree days (GDD) after testing the accuracy of BMT for six hard spring wheat varieties in western Canada. The VSMB determines the seeding date for spring wheat based on daily mean air temperature, precipitation, and soil moisture condition; and the harvest date based on daily minimum air temperature. If the meteorological conditions for seeding have not been met by June 15, then the VSMB sets the seeding date to June 15 (Hayashi et al. 2012). In this study, the Akinremi version was run using real seeding dates for the annual cropping system at Carman and for the Hayashi version, seeding date was set to the 15$^{th}$ of June which was closer to actual seeding date than the maximum seeding date (June 1$^{st}$) calculating by model. At Carberry, the real seeding date could not be used due to the late seeding date in the first year of study. Using actual or a simulated seeding date led to incomplete crop growth staging within a year as the model could not reach stage 5 which is the maturity stage in a realistic time period. Therefore, seeding date was set between June 13 and 15 at Carberry.

In the case of the perennial cropping system, since the Akinremi et al. (1996) version of VSMB did not have a plant phenology model for grass, the seeding date for grass was set to April 15 which is close to the start of the grass regrowth period in Manitoba. However, the Hayashi et al. (2010) modified version used a three-stage model for grass growth including dormant, growing and full cover. In response to heat unit accumulation, the growth stage goes from dormant (stage 1) to growing (stage 2) when the daily mean temperature exceeds 1˚C for five consecutive days, and from growing (stage 2) to full cover (stage 3) when cumulative growing degree day (GDD) reaches 240. The growth stage goes back to dormant when the maximum sunshine duration is below 10.5 hour which was on October 16 at Hayashi's site located near Calgary, Alberta (Hayashi et al., 2010).

#### 5.3.1.2 Drainage

Three volumetric soil moisture parameters are necessary as input data including field capacity ($\theta_{FC}$), actual moisture content of each layer, saturated water content ($\theta_{SAT}$) and permanent wilting point ($\theta_{PWP}$). The model uses the initial soil moisture content of the first day of the time period being simulated. Then, moisture content changes based on precipitation, evapotranspiration, drainage and runoff from snow and rain. The model can estimate water drainage in two ways, one is by gravitational drainage between soil layers and the other is the diffusive exchange of soil water between layers.

In terms of gravitational drainage between soil layers, the excess water above field capacity from the top layer is drained into the lower layer and no further drainage occurs until the water content exceeds field capacity again. Akinremi et al. (1996) allows the lowest soil layer to drain to 75% of field capacity as a means of obtaining drainage water from the lowest layer.

The available water holding capacity (mm) of the i-th soil layer which is the amount of soil moisture content after drainage is defined as:

$$AWHC_i = \Delta z_i \, (\theta_{FC} - \theta_{PWP}) \times 10^3 \qquad (1)$$

where $\theta_{FC}$ (mm) is the soil moisture at field capacity, $\theta_{PWP}$ (mm) is the soil moisture at permanent wilting point and $\Delta z_i$ (m) is the thickness of the soil layer.

Diffusive exchange of soil water between adjacent layers calculated by the Ceres algorithm which is adapted from Ritchie and Otter, 1985 (Akinremi et al. 1996). Hayashi et al. (2010) presented the equation as:

$$qi = -10^3 D_h (\theta_A)\, \partial\theta_A/\partial z = -10^3 D_h (\theta_{Ai} + \theta_{Ai+1}/2)(\theta_{Ai+1} - \theta_{Ai}/z_{i+1} - z_i) \quad (2)$$

where $D_h$ ($m^2\ d^{-1}$) is hydraulic diffusivity, $\theta_{Ai}$ ( $= \theta_i - \theta_{PWPi}$) is the available water content of the i-th layer, and $z_i$ (m) is the depth to the center of the i-th layer. Since z increases downward, a positive value of qi indicates downward flow. The $D_h$ function is defined by:

$$D_h = min[8.8\times10^{-5} exp(35.4\ \theta_A), 0.01] \quad (3)$$

where *min[,]* indicates the minimum of the two values.

### 5.3.1.3 Evapotranspiration

The VSMB computes actual evapotranspiration (*AET*) from individual soil layers using empirical factors dependent on soil moisture and plant condition:

$$AET = \sum_{i=1}^{n}(Ep \times Ri \times DC \times (PAW\ /\ AWHC]\ ) \quad (4)$$

where *Ep* is the potential evaporation calculated by Priestley and Taylor (1972) equation, *Ri* is the root extraction coefficient depend on crop type and it's growth stage, *DC* is the drying curve and *PAW/AWHC* is the plant available water to available water holding capacity ratio (Hayashi et al. 2010).

Because plant roots seek moisture sources at lower depths when the moisture in shallower layers becomes depleted, Baier et al. (1979) modified the *Ri* (Eq. 5). The *Ri* is calculated sequentially as

$$R_1 = r_1$$

$$Ri = ri + ri\sum_{j=1}^{i-1} Rj(1 - (PAW\ /\ AWHC]\ ) \qquad \text{for } i \geq 2 \quad (5)$$

where *ri* are dimensionless constants that are specific for soil depth and plant growth stage (Hayashi et al. 2010).

Due to the sensitivity of model to the root extraction coefficient, the exact root coefficient modified by Akinremi et al. (1996) using the five-layer approach was used for the annual cropping system (Table 5.1). For Hayashi's version of the model, grass coefficients from the four-layer approach (Hayashi et al. 2010) were retained, but adjusted for differences in soil depth after calibration with different sets of coefficient.

Table 5.1. Root extraction coefficients

| | Depth (cm) | | | | |
|---|---|---|---|---|---|
| Annual (Wheat) | 0-15 | 15-30 | 30-60 | 60-90 | 90-120 |
| Planting-emergence | 0.40 | 0.10 | 0.05 | 0.10 | 0.01 |
| Emergence-jointing | 0.40 | 0.15 | 0.10 | 0.02 | 0.01 |
| Jointing-heading | 0.55 | 0.20 | 0.20 | 0.10 | 0.05 |
| Heading-soft dough | 0.55 | 0.25 | 0.25 | 0.15 | 0.05 |
| Soft dough-ripening | 0.55 | 0.25 | 0.25 | 0.15 | 0.05 |
| | | | | | |
| Perennial (Grass) | | | | | |
| Dormant | 0.165 | 0.0865 | 0.033 | 0.0112 | 0.006 |
| Growing | 0.507 | 0.239 | 0.075 | 0.030 | 0.008 |
| Full cover | 0.0507 | 0.275 | 0.120 | 0.080 | 0.015 |

The drying curve (*DC*) function given in the VSMB is calculated as:

$$DC(x)=(x/C_r)^{Cm\ Cn\ Ch}+x(C_m/C_r)\ (1-x/C_r)^{Cn} \qquad (6)$$

where $x$ is the ratio of plant available water to the available water holding capacity of the soil. *DC* is equal to zero at permanent wilting point and increases by increasing of *PAW/AWHC* toward one, $C_m$,$C_n$,$C_h$ and $C_r$ are dimensionless fitting parameters. Hayashi (2010) showed that the drying curve function and root coefficient have a strong influence on the actual evapotranspiration from each soil layer and consequently the change in soil moisture content of each layer. In Hayashi's version of VSMB, the applied parameters were presented in table 5.2.

Table 5.2. Parameter values for drying curve used in Hayashi et al. (2010) version

| | $C_m$ | $C_n$ | $C_h$ | $C_r$ |
|---|---|---|---|---|
| Wheat | 0.27 | 0.90 | 0.30 | 1.58 |
| Fallow | 0.66 | 0.95 | 0.30 | 1.47 |
| | | | | |
| Grass | 0.11 | 0.95 | 0.05 | 1.00 |
| Dummy | 0.70 | 0.80 | 1.00 | 1.00 |

In Akinremi's version of VSMB these values are given as 1, 1, 1 and 0.7 respectively for cropped field and 1, 1, 1 and 0.5 respectively for fallow fields. This drying curve function denotes curves *E* for cropped soils and D for fallow which was adapted by Akinremi et al (1996) from Baier et al (1979). Curve *E* assumes no reduction in *AE:ETo* ratio over the range of available soil moisture from 100 – 50% and from 100 - 70% for curve *D*. Beyond these points, the *AE* decreases sharply with decreasing available soil moisture content (Ojo. 2012).

#### 5.3.1.4 Run off

The VSMB estimates runoff using two different approaches, one is for snowmelt runoff and the other is for rainfall runoff. The model assumes that precipitation comes as rain if the mean air temperature ($T_m = (T_{max} + T_{min})/2$) is greater than 2 °C and snow otherwise. Usually, thirty percent of snowfall is assumed to be lost by sublimation and the rest accumulates on the ground. Snowmelt is calculated based on the method of McKay (1964) by using the temperature-index method with coefficients that vary depending on the latitude and the day of year (Hayashi et al. 2012). However, complicated freeze-thaw processes can affect the release of melt water from the snowpack.

In the original VSMB, the daily maximum temperature of -6.7 was defined for the freeze-thaw process. However, Akinremi et al. (1996) adopted the temperature algorithm from the EPIC model (Williams et al. 1990) to calculate soil surface temperature ($T_s$, *°C*) by

$$T_s = I_s T_{m-1} + (1 - I_s)[T_m + (T_{max} - T_{min})/4 + T_{m-1} + T_{m-2}]/3 \qquad (7)$$

where $T_{m-1}$ and $T_{m-2}$ (°C) are daily mean air temperature of one and two days, respectively, before the current day (Eq. 7); and $I_s$ is a dimensionless variable which represents the effects of thermal insulation by the snowpack having snow water equivalent (*SWE*) of *A* (mm);

$$I_s = A / [A + exp(2.303 - 0.2179A)] \qquad (8)$$

Calculating runoff in frozen soils, Akinremi et al. (1996) found that the ratio of soil moisture content to field capacity content was approximately close to Hobb-Krogman's overwinter storage constant which was not necessary to be specified. Therefore, daily surface runoff ($R_{off}$, mm) was defined by:

$$R_{off} = W_{in} \times (\theta / \theta_{FC}) \qquad (9)$$

where $W_{in}$ (mm) is the daily amount of water input, which includes snowmelt and rainfall, $\theta$ is the volumetric water content of the first soil layer, and $\theta_{FC}$ is the volumetric water content at field capacity (Akinremi et al. 1996, Hayashi et al. 2012). The remaining portion of $W_{in}$ is added to the top layer as infiltration.

For unfrozen soil ($Ts$ > 0 °C), the Curve Number method (*NRCS 2004*) is used to calculate $R_{off}$ from $W_{in}$ and varies based on the soil type and antecedent moisture condition. Akinremi et al. (1996) modified the standard curve number (*CN*) method to compute the dimensionless *CN*. The two dimensionless parameters, $C_d$ and $C_w$, reflecting the moisture condition of all soil layers were calculated as:

$$C_d = \sum_{i=1}^{n}[wf_i \times (\theta_i - \theta_{WPi})/(\theta_{FCi} - \theta_{WPi})] \qquad (10)$$

$$C_w = \sum_{i=1}^{n}(wf_i \times \theta_i/\theta_{FCi}) \qquad (11)$$

where $wf_i$ is a depth weighting factor the i-th soil layer, with i increasing from 1 at the top and n at the bottom, $\theta$ is total volumetric moisture content and $\theta_{WP}$ and $\theta_{FC}$ are the volumetric water content at wilting point and field capacity, respectively. The weighting factor is specified by an exponential function, which takes the maximum value at the soil surface and becomes negligible (< 0.01) at a depth of 0.5 m. The *CN* is given by:

$$CN = CN_1 + cd\,(CN_2 - CN_1) \qquad cd < 1 \qquad (12)$$

$$CN = CN_2 + cw\,(CN_3 - CN_2) \qquad cd \geq 1 \qquad (13)$$

where $CN_2$ is the "master" curve number dependent on soil type, land cover, and agricultural practices; and $CN_1$ and $CN_3$ are calculated from $CN_2$ by

$$CN_1 = CN_2 - 20 \times (100 - CN_2) / [100 - CN_2 + exp\{2.533 - 0.063(100 - CN_2)\}] \qquad (14)$$

$$CN_3 = CN_2\, exp[0.006729(100 - CN_2)] \qquad (15)$$

The value of $CN_2$ = 80 was tested for the soils of Swift Current, Saskatchewan and Lethbridge, Alberta (Akinremi et al. 1996). From the calculated value of $CN$, $R_{off}$ (mm) is given by: $S = 254 (100 – CN) / CN$

$$R_{off} = (W_{in} – 0.2S)^2 / (W_{in} + 0.8S) \qquad \text{if } W_{in} > 0.2S \qquad (16)$$

$$= 0 \qquad \text{if } W_{in} \leq 0.2S \qquad (17)$$

where S (mm) is a retention parameter representing the effects of soil and plant canopy.

### 5.4.1 Site Characteristics and Field Measurements

Field data were collected at two sites in Manitoba with loamy and loamy sand soils. The first study was conducted at the University of Manitoba Field Research station, Carman, Manitoba. The site was located on the Hibsin soil series with moderately well drainage and coarse loamy underlain by clayey deposits soil (Canada-Manitoba Soil Survey Report D60).

A split-plot treatment structure was established with cropping system (annual and perennial) as the main plot and manure/urea treatment as the sub-plot (10 m x 10 m) with 4 replications. In this study, two years of the rotation data were used for the annual and perennial cropping systems. For the annual rotation, barley was grown in 2010 and canola was grown in 2011. For the perennial rotation, timothy/orchard grass was maintained for both years.

Forty field core lysimeters were installed at the corner of each plot so that water movement and nutrient leaching could be measured. Each lysimeter included three main parts: the main column, the schedule 80 PVC pipe with an internal diameter of 54 cm and 106 cm in length, representing root zone extension of annual crops; a circular perforated plate and a collection bottom cap. To reduce the disturbance of soil during installation a custom made hydraulic press was used to push down the main column of the lysimeter to the desired depth. The main column was then pulled out of the soil and turned upside down. Geotextile was placed on the soil to separate the soil from the perforated plate and collection basin. The perforated plates, collection caps and extraction pipes were then installed on the main columns.

The second field experiment, located northwest of the town of Carberry, Manitoba, was conducted on a loamy sand soil overlying the Assiniboine Delta Aquifer. The site consists of Orthic Black Chernozem soils of Fairland series, which developed on lacustrine deposits. These soils have a medium texture with the upper 75-90 cm soil classified as loamy sand and the underlying layer has a texture of sandy loam to loam. The sand content is about 78% in the upper layer and the percent sand gradually decreases with depth.

At the Carberry site, the experimental design was a randomized complete block design with four replications for a total of 24 treatment plots. Treatment plots were 100 $m^2$ (10 m × 10 m) in size and one field core lysimeter was installed 2 m inside of the southern boundary of each plot, approximately in the middle of the plot to directly measure water, nitrate and phosphorus moving past the rooting zone (Enns 2004; Nikiema et al. 2013). A two year rotation was employed for the annual cropping systems. Treatments included two rates of liquid hog manure, two rates of fertilizers corresponding to the amount of available nitrogen in the two rates of hog manure, a composted beef manure treatment and a control, for a total of six treatments.

Soil samples were collected during the growing seasons of 2010 and 2011 at three times: spring, mid-season, and harvest. Soil was sampled at six depth intervals of 0-15, 15-30, 30-45, 45-60, 60-90 and 90-120 cm for spring and harvest using the Giddings soil sampler and at five depths for midseason using a Dutch auger. Two soil samples were taken from each plot and composited.

Gravimetric moisture content was determined on all samples after oven drying (105° C). Volumetric moisture content was derived from the product of the gravimetric water content and bulk density. The measurement of soil bulk density was carried out at both sites in the 2009 and 2010. The mean value for each depth was determined and utilized in the volumetric soil moisture calculations. Soil bulk density was calculated as the total mass of dry soil ($M_s$) divided by the total volume ($V_t$) it occupies.

$$pb = M_s/V_t \qquad (18)$$

Leachate was collected from the lysimeters three to five times per year, depending on the amount of precipitation during the growing seasons. The leachate was collected from the catch basin by a vacuum pump connected to a hose that ran through one of the extraction tubes. The second extraction tube was opened during leachate collection for equalization of pressure; otherwise, both tubes were covered with a cap to prevent rain water from running down the tubes. The total volume of leachate from each lysimeter was recorded

### 5.4.2 Statistical Analyses

All soil and leachate variables of Carman and Carberry were analyzed as a split plot design and randomized complete block design using PROC MIXED procedure in SAS software (SAS Institute, 2008). Assumption of normality distribution was checked using PROC UNIVARIATE. Since Shapiro-Wilk's normality test did not show normal distribution of residuals, the log transformed data was used to generate normal distribution of residuals and homogeneity of variance prior to statistical analysis. The statistical model for Carman site included block (with four levels) as a random factor and treatments (five levels), cropping systems (two levels) and depth (six levels) as fixed factors with depth treated as a repeated measurement. The statistical model included block (with four levels) as a random factor and treatments (six levels) and depth (six levels) as fixed factors. The spatial power [SP(POW)] covariance structure was used in the model for the repeated measures data in which the depth intervals were unequal. Due to variation in manure application by hand a predefined 0.1 significant level was considered (Olatuyi et al. 2012; Zvomuya et al. 2003). Treatment differences were accepted if P<0.1 using Tukey-Kramer method.

To quantify the performance of the model simulation of soil moisture content, the root mean square error (*RMSE*) given by:

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(xsim - xobs)^2} \qquad (19)$$

where "*sim*" is the simulated soil moisture by VSMB model and "*obs*" is the observed soil moisture content in field, and n is the number of data point.

The mean bias error (*MBE*) indicates overestimation and underestimation of simulated soil moisture content compared with the measured moisture content as represented by positive and negative values of *MBE*, respectively.

$$MBE = 1/n \sum_{i=1}^{n}(xsim - xobs) \qquad (20)$$

These statistics were also used to compare results between the two version of the model and modified VSMB.

### 5.4.3 Model Evaluation and Modification

The two versions of the VSMB as reported by Akinremi et al. (1996) and Hayashi et al. (2010) were used as the base models. The soil moisture data during the growing season of 2010 were evaluated separately in the two cropping systems, annual and perennial, at Carman, and barley and wheat at Carberry. The model was initialized with the soil moisture in the spring of 2010 for each site and simulation was completed at harvest. In this study, because the study areas were flat (≤1% slope) and had permeable loamy and sandy soils in Carman and Carberry respectively, both rainfall and snowmelt runoff was considered to be zero and it was assumed that all precipitation infiltrated the soil. For the annual cropping system, the adjustment for evapotranspiration (k-adjustment) was restricted to stage 3 of crop phenology (jointing to heading stage). This assumption and model modifications were undertaken after the model predicted significantly smaller amounts of simulated leached water than was measured from the lysimeters. Because Hayashi's version did not simulate the soil moisture content properly at first and there were two cuts of the perennial grass at Carman, based on the actual dates of the first cut (first week of July), the source code was modified to restart the growth stages following the first cut. It was assumed that it took approximately three weeks for the grass to re-establish full cover stage again.

## 5.5 RESULTS AND DISCUSSION

### 5.5.1 Soil Moisture Distribution

The measured soil moisture content at Carberry and those simulated by the two models are shown in Figure 5.1 for the two years of field study. The results showed that the models were able to capture the trend in soil moisture.

#### 5.5.1.1 Carberry

Although similar input data and crop phenology were used for both models, the results showed that the Akinremi's version had better agreement between simulated and observed soil moisture distribution than Hayashi's version. This may be due to the differences in their drying curves. Both models underestimated the soil moisture in the 45-120 cm depth of soil especially in the second year of simulation (Figure 5.1). Akinremi et al. (1996) reported the underestimation of simulated soil moisture in lower depths by the original VSMB and this was attributed to the lack of a mechanism for redistributing soil water between the different soil layers when soil water content was below field capacity. Nevertheless, the modified model still underestimated soil moisture content of deeper layers (Akinremi et al. 1996). Ojo (2012) concluded that the drying curve coefficients in the VSMB could possibly dry the soil at lower depths faster than observed. Therefore, it is necessary to re-evaluate the drying curve at depths. In addition, the exclusion of water table depth in the model might also have an impact on the underestimation of moisture content at depth by the model, especially in wet years such as 2010.

The removal of rainfall and snowmelt runoff from model simulation as well as restricting the adjustment coefficient for evapotranspiration (k-adjustment) to stage 3 of crop phenology (jointing to heading stage), improved the performance of the Akinremi version of VSMB. Nevertheless, the model underestimated the moisture content in the 45-120 cm depths specifically in 2011 (Figure 5.1). Hayashi et al. (2010) reported that the drying curve function is an important factor which can affect the simulation of evapotranspiration. Therefore, the differences between the Akinremi and Hayashi versions could be due to differences in their drying curves. Hayashi et al. (2010) replaced $a_d$ = 8.8×10$^{-5}$ m$^2$ d$^{-1}$ with $a_d$ = 1×10$^{-5}$ m$^2$ d$^{-1}$ in hydraulic diffusivity ($D_h$, m$^2$ d$^{-1}$) formula based of measured field data. Moreover, the authors modified the model by using a new radiation algorithm. These might be responsible for the differences in the results obtained from the two versions of the model.

Table 5.3. Comparison of observed and simulated soil moisture content by two versions of VSMB

| ***Carberry*** | ***Akinremi*** | ***Hayashi*** | ***Akinremi_Modified*** [z] |
|---|---|---|---|
| | | ***($m^3/m^3$)*** | |
| MBE (n=30) | -0.02 | -0.02 | -0.01 |
| RMSE (n=30) | 0.04 | 0.05 | 0.02 |
| ***Carman-Annual*** | | | |
| MBE (n=29) | -0.04 | -0.01 | -0.01 |
| RMSE (n=29) | 0.06 | 0.04 | 0.04 |
| ***Carman-Perennial*** | | | |
| MBE (n=29) | | -0.02 | -0.02 |
| RMSE (n=29) | | 0.05 | 0.05 |

[z] Assuming no rainfall or snowmelt runoff; corrected for increased evapotranspiration to crop growth stage 3 only

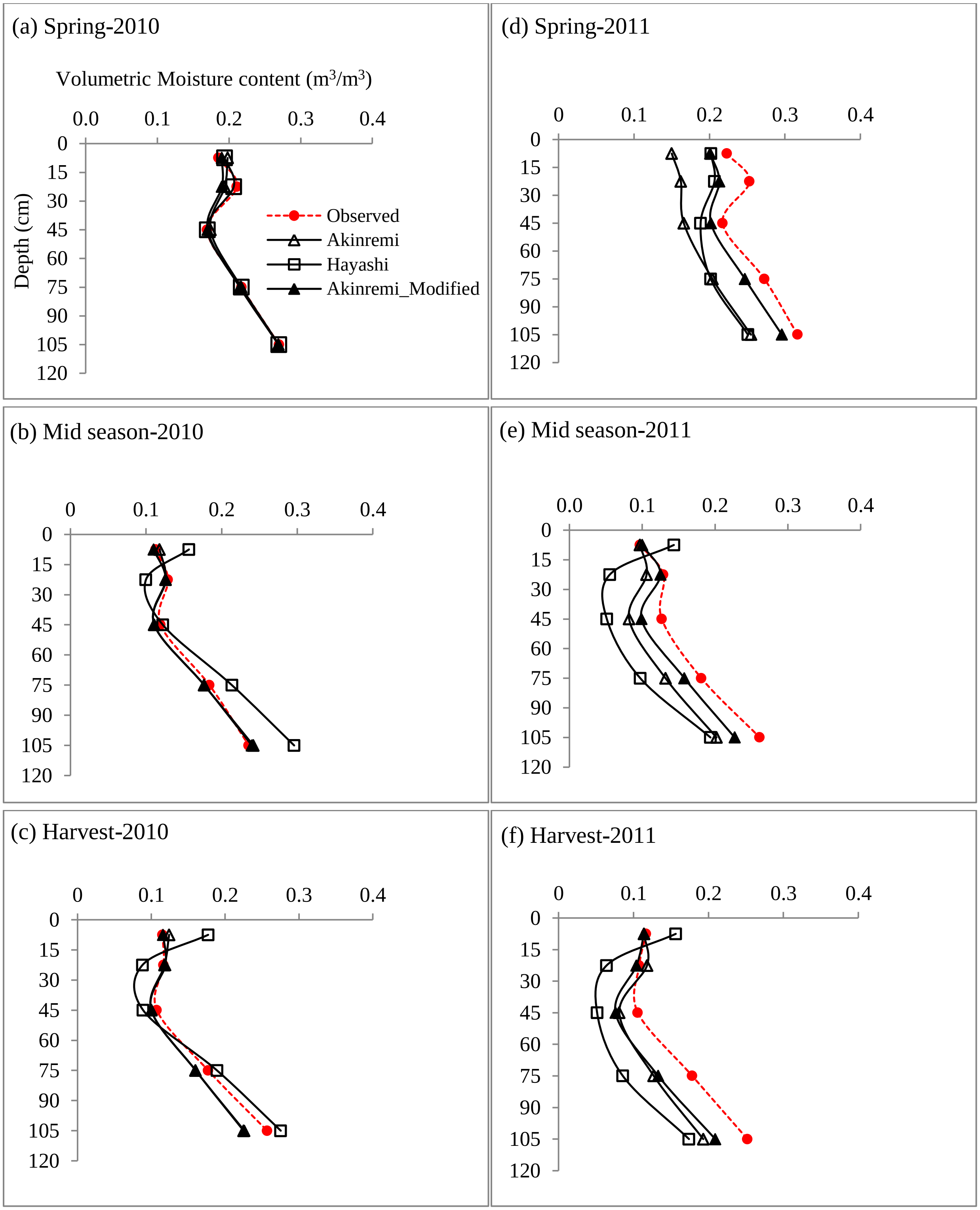


Fig. 5.1. Changes in soil moisture content at Carberry as simulated by the Akinremi, Hayashi and Akinremi_Modified versions of VSMB model.

### 5.5.1.2 Carman-Annual

Similar to model simulation at Carberry, the growth stage of wheat and input data were the same for both versions of the model. Unlike the results obtained at Carberry, the Hayashi version of the model captured the trend in soil moisture distribution better than the Akinremi version at Carman, especially in the first year of simulation (Figure 5.2). Because of differences in soil parameters including texture affecting the drying curve coefficients and hydraulic conductivity and diffusivity, the relative performance of the two models appeared to be site-specific. The RMSE between the simulated and the observed data was smaller for Hayashi's version (Table 5.3). However, assuming the absence of rainfall and snowmelt runoff as well as limiting the k-adjustment to stage 3, improved the performance of Akinremi's version in 2011, similar to the improvement observed at Carberry. The model accuracy decreased with depth, similar to the results at Carberry and results from previous studies (Akinremi et al. 1996; Ojo 2012). Akinremi et al. (1996) concluded that the poor simulation of soil moisture at the depth might be due to insufficient ability of model to transfer water to lower soil layers during the growing season. On the contrary, Mapfumo et al. (2003) suggested that "the conservative approach to drainage (the need to reach field capacity before drainage) may be partly responsible for overestimation of soil water at deep layers".

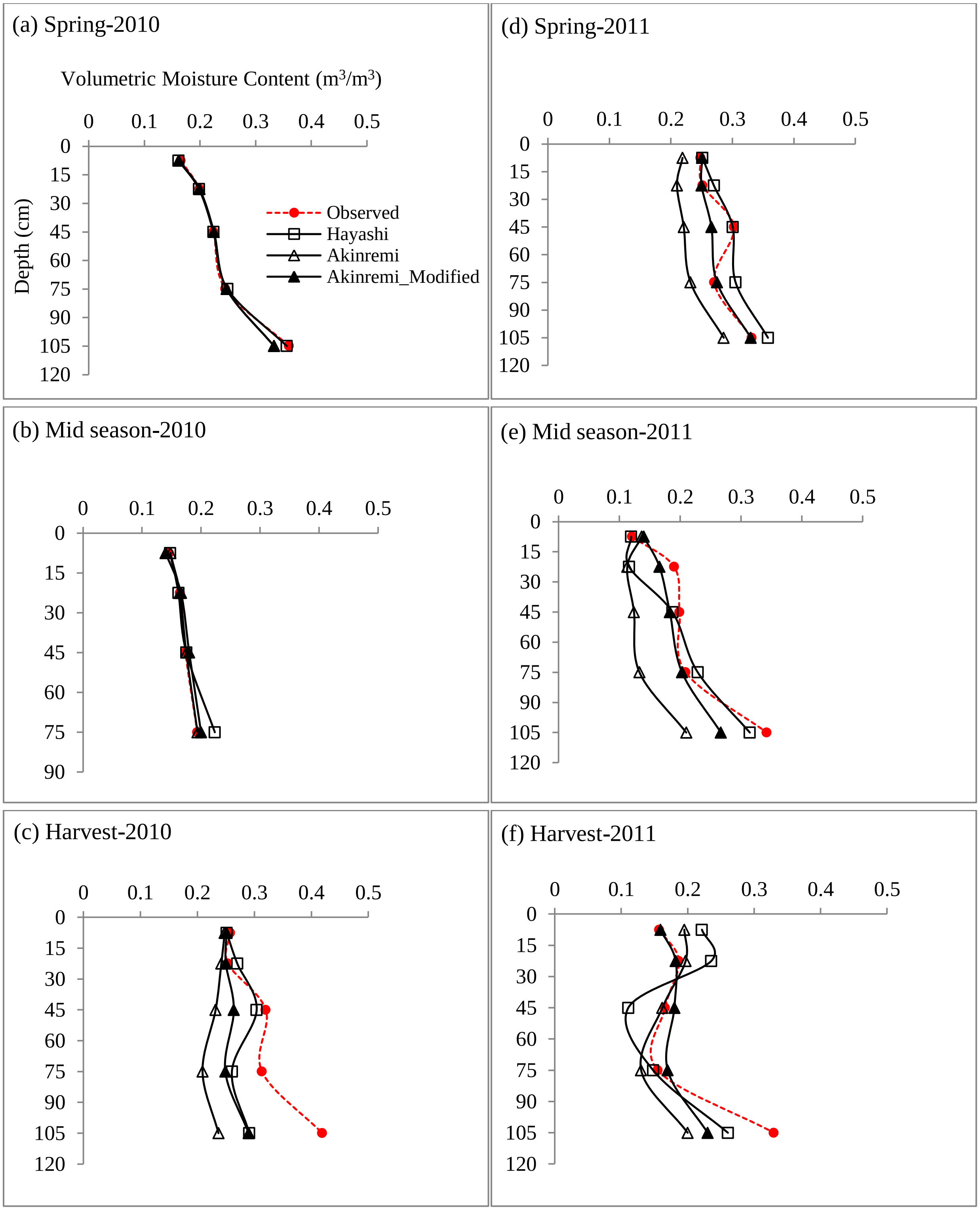


Fig. 5.2. Changes in soil moisture content at Carman's annual plots as measured and as simulated by the Akinremi, Hayashi and Akinremi_Modified versions of VSMB model.

#### 5.5.1.3 Carman-Perennial

The Akinremi version of VSMB was originally formulated to simulate soil water changes under a cereal crop (wheat or barley). Nevertheless, it was used to simulate changes in soil moisture under perennial grasses at Carman using the crop growth stages of wheat. Similar to the model simulation for the annual crop at Carberry and at Carman, the Akinremi version that assumed no rainfall and snowmelt runoff was used to simulate soil water changes under perennial grasses at Carman. Since perennial grasses are known to have greater evapotranspiration than annual crops (Bradshaw et al. 2007) the adjustment for evapotranspiration was kept the same as in the original Akinremi version. The simulated soil moisture followed a trend that was similar to that observed for both Akinremi_Modified and Hayashi versions. Both models were equally accurate for predicting volumetric moisture content (Figure 5.3, Table 5.3). To improve the simulation of soil moisture content, the Hayashi version was modified to take into account the mid-season harvest of the grasses and to simulate no rainfall or snowmelt runoff. The changes to the model did not result in improvement in the soil moisture output of the model.

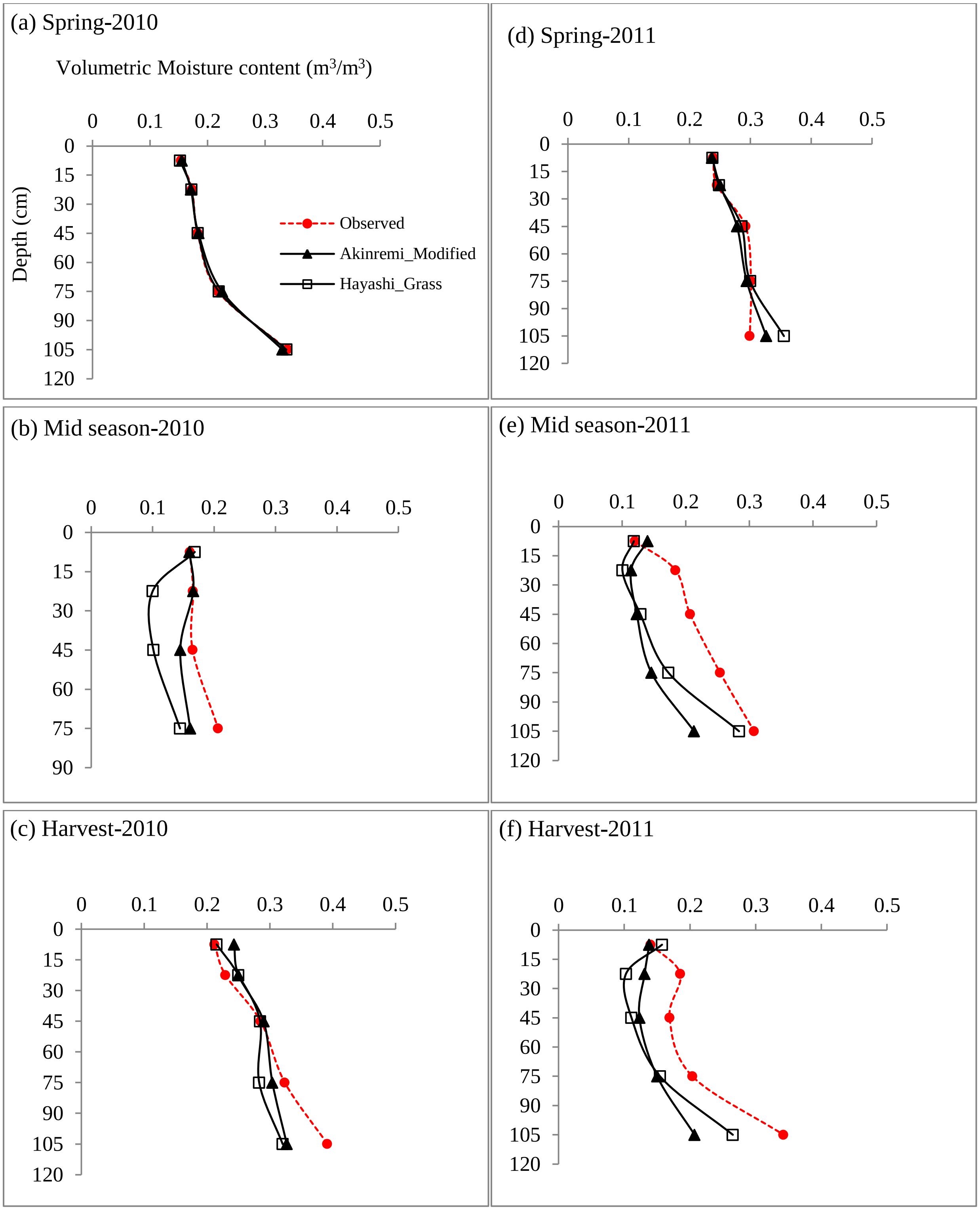


Fig. 5.3. Changes in soil moisture content at Carman's perennial plots as observed and as simulated by the Hayashi and Akinremi_Modified versions of VSMB model.

### 5.5.2 Drainage Water Simulation

The VSMB has been tested for soil moisture content and evaporation at different locations in Canada (Akinremi et al. 1996, Hayashi et al. 2010, Ojo 2012). However, this is the first case where the model has been tested for simulation of drainage water below the root zone. When both versions of the model were used to simulate drainage below the root zone in the two years of our field study, the results showed that neither the Akinremi, Hayashi, nor the Akinremi_ Modified versions, accurately predicted the amount of leached water collected from lysimeters at Carberry accurately. During two years of model simulation, the Akinremi and Hayashi versions predicted 2.4 and 1.7 cm of leached water, respectively, compared to 33-43 cm in the observed values. After the modification of Akinremi version, the amount of leached water simulated during the two years increased to 6.6 cm which was still smaller than the amount that we collected from the lysimeters (Table 5.4). Therefore, the VSMB grossly underestimated the amount of leached water, possibly due to the underestimation of gravitational drainage and mostly overestimation of evapotranspiration by the model. Conversely, Ojo (2012) and Hayashi et al. (2010) reported that the model substantially underestimated evapotranspiration. These conflicting results suggest that the evapotranspiration subroutine of the model needs to be tested and calibrated using real data.

Similar to the results of Carberry, the model did not accurately estimate the amount of leached water in annual and perennial cropping systems at Carman, which was 59 cm over the two years. For two continuous years of model simulation, Akinremi's versions simulated 3 cm of leached water for the annual cropping system. After modification, the amount of simulated leachate increased to 7.8 cm which was still smaller than the measured leachate (Table 5.5). The Hayashi version estimated 2.5 cm of leached water which showed less agreement between simulated and field collected leachate than the Akinremi version.

Table 5.4. Actual and simulated amounts of leached water (cm) from Carberry plots in 2010 and 2011

| ***Treatment*** | | ***2010*** | ***2011*** | ***Total*** |
|---|---|---|---|---|
| LH-2500 | | 19.5 | 19.9 | 40 |
| LH-5000 | | 18.5 | 21.5 | 40 |
| Com-2500 | | 18.2 | 14.6 | 33 |
| F-5000 | | 17.0 | 20.7 | 37 |
| F-2500 | | 18.1 | 25.0 | 43 |
| Control | | 17.9 | 23.1 | 41 |
| ***Model effect*** | ***d.f.*** | ---------------- ***P value***$^{z}$ | ---------------- | |
| Trt | 5 | 0.948 | 0.471 | |
| Simulated leached water (sum of 2-year) | | | | |
| ***Akinremi*** | | | | 2.4 |
| ***Akinremi_Modified*** | | | | 6.6 |
| ***Hayashi*** | | | | 1.7 |

$^{z}$ Probability value is significant at P <0.1
LH-2500: Hog manure at 2500 gal ac$^{-1}$
LH-5000: Hog manure at 5000 gal ac$^{-1}$
Com-2500: Compost application at N rate similar to LH-2500
F-5000: Fertilizer application at N rate similar to LH-5000
F-2500: Fertilizer application at N rate similar to LH-2500
Control: Seeded-No treatment

For perennial plots an overall average of 51 cm of leachate was measured over the two years. However, the Hayashi version with three grass growth stages did not simulate any leachate before and after modification. The modified Akinremi version with five growth stages of wheat estimated 6.7 cm of leachate for two continuous years of model simulation. It seems that the specified crop coefficients for grasses in the Hayashi's version overestimated evapotranspiration resulting in reduced leachate. In general, our results show that a good simulation of soil water content may not translate into good simulation of other components of the hydrologic cycle.

Table 5.5. Actual and simulated amounts of leached water (cm) from annual and perennial plots of Carman in 2011 and 2010

| | | ***2010*** | | ***2011*** | |
|---|---|---|---|---|---|
| ***Treatment*** | | Annual | Perennial | Annual | Perennial |
| Liquid-N | | 22.58 | 18.79 | 21.86 | 17.42 |
| Liquid- P/Urea | | 32.06 | 29.66 | 32.31 | 27.96 |
| Solid-N | | 29.06 | 18.16 | 22.11 | 23.63 |
| Solid-P/Urea | | 36.16 | 30.39 | 35.71 | 27.18 |
| Control | | 30.50 | 32.82 | 31.33 | 29.14 |
| ***Model effect*** | ***d.f.*** | ---------------- ***P value***[z] ---------------- | | | |
| Crop | 1 | 0.543 | | 0.547 | |
| Manure | 4 | 0.423 | | 0.628 | |
| Crop×Manure | 4 | 0.922 | | 0.979 | |

| | ***Akinremi*** | ***Akinremi_Modified*** | ***Hayashi*** |
|---|---|---|---|
| | Simulated leached water (sum of 2-year) | | |
| Annual | 3 | 7.8 | 2.5 |
| Perennial | ----- | 6.7 | 0 |

[z] Probability value is significant at P <0.1

## 5.6 CONCLUSIONS

Accurate simulation of soil moisture content and quantitative assessment of leached water can provide valuable information about the potential distribution, movement and loss of $NO_3^-$-N through the soil profile. However, simulated values should be close to the observed field data. This study showed both versions of VSMB underestimated soil moisture content in the 45-120 cm depth, especially in the second year of simulation. Based on our results, the amount of leached water that was simulated during the two years at Carberry was 6.6 cm compared to 39.0 cm of leachate that was measured in the field. As well, at Carman, the amount of simulated leachate was 7.8 and 6.7 cm for annual and perennial cropping system, respectively, while 59 and 51 cm of leachate, on average, were collected from field lysimeters. Although Ojo (2012) reported that the Priestley-Taylor equation in the Akinremi version of the VSMB underestimated the amount of water loss to the atmosphere when compared to the Penman-Montieth equation, in this study the model grossly underestimated the amount of leached water. This is related to the underestimation of soil water content at depth, most likely due to an overestimation of evapotranspiration. Moreover, the model did not include leaching occurring through macropore flow. In the future, to improve the model's drainage simulation, it is recommended that the evapotranspiration subroutine of the model to be tested independently and calibrated using measured evapotranspiration data from lysimeters or from micrometeorological techniques.

## 5.7 APPENDICES

### Input and Output Files

Most of the published papers on VSMB presented the simulated results by model and compared these to actual data. Akinremi and McGinn (1996) reported the lack of clear guidance for using soil moisture models as one of the challenges for model application. Therefore, a brief explanation of the source code and input files may improve our understanding and future application of the two versions of the model.

#### Appendix 5.7.A: Akinremi's Version Inputs

The model uses two input files: 1) a weather data file which includes year, month, day of month, Julian day, daily maximum and minimum temperature ($C^o$) and precipitation (mm) and 2) soil/plant information file which includes Field capacity ($\theta_{FC}$), Actual moisture content of each layer (mm), Saturated water content ($\theta_{SAT}$) and Permanent Wilting Point ($\theta_{PWP}$) that are arranged according to the formats of VSMB source code and also root coefficients at different growth stages and for different soil layers. The input table also contains the total number of crop development stages, crop stage for K adjustment start, starting year of run (yyyy), last year of the run, number of crop stages per year, seeding day (Julian) and latitude of the site (degrees) are included in soil/plant information file. The main output from the model is the simulated volumetric soil water content at the various depths specified by the user. Other outputs include soil temperature, precipitation, reference and actual evaporation, rainfall runoff, snow melt runoff and drainage.

#### Appendix 5.7.B: Hayashi's Version Inputs

In Hayashi's version input data divided to five individual files including cropwater.txt, data.txt, moist.txt, paras.txt and Mckay.txt files.

**cropwater.txt**: including crop water extraction coefficients ($k$) for wheat and grass separately, Drying curve index parameters for wheat and fallow and also grass and dummy, *KNTROL* is the zones using first Z-table assuming 7 and *NEW* is the crop stage for k-adjustment start which in original code is considered 3.

**data.txt:** number of time series data points and number of soil layers were entered in first line and parameters including day, month, year, Tmax, Tmin($C^o$), precipitation (mm), radiation ($W/m^2$), relative humidity(%), elevation (m) were entered for each individual day.

**moist.txt:** including soil parameters like field capacity **(***FC*), permanent wilting point (*PWP*), saturation (*SAT*) and plant available water (*PAW*), based on mm for each soil layer, *VALUE OF IFRNT* is the number of zones wetted during the first day of rain which is considered 3.

The *PAW* was calculated by subtracting *PWP* from measured volumetric water content. For instance, having the volumetric water content of 0.4 for a soil thickness of 100 mm and wilting point of 0.1, give the initial water content of (0.4-0.1)x100 = 30 mm. Then, to convert the available water content (*AWC*, mm) which is obtained from output file to total water content ($cm^3/cm^3$), it is needed to add the *PWP* to *AWC* and then divide by the thickness of the soil layer, i.e. total water content in the soil layer = (*AWC* + *PWP* (mm))/(thickness of layer (mm)).

**paras.txt:** this file includes various parameters LINE 1: wheat or grass option (>0 for wheat and <0 for grass), snow blow coefficient, soil temperature 2 days before, soil temperature 1 day before, radiation data type (1=short wave incoming radiation ,2=net radiation ,3=no radiation data), land cover type (1=alfalfa,2=wheat, 3=grass). In case of no radiation data, a number like -9999 can be considered to indicate no data in column 7 where radiation has to be specified.
LINE 2: albedo snow (0.64), albedo no snow (0.19), snow blow cof.1 (0.7), snow blow cof.2 (0.7), date blow change (701) and number of crop stage (5 for wheat and 3 for grass)
LINE 3: latitude
LINE 4: bottom depth of each soil layer (cm)
LINE 5: curve number 2 ($CN_2$), AWHC (available water holding capacity, mm)

**Mckay.txt:** Snowmelt is calculated using the temperature-index method with coefficients that vary depending on the latitude and the day of year (Hayashi et al. 2012). McKay.txt file is supposed to be 'generally' valid and it was based on McKay's(1964) paper.

The same as Akinremi's version after activation of vsmb.exe the output files are produced. The outputs are time series simulation results of the various hydrological components. The top header text on the output files shows the name of the variables.

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# 6. GENERAL SYNTHESIS

The annual investment of Manitoba farmers in nitrogen and phosphorus fertilizer is hundreds of millions of dollars. Some of the class 4 and 5 soils may have the capability of cropping with minimal loss of nutrients. On farms with both crops and livestock operations, or farms that are in the vicinity of a manure source, an integrated use of manure as a source of plant nutrients on such classes of soils will be of benefit through the reduction in cost of commercial fertilizers, while removing the negative impact to adjacent environments. The leakage of nutrients from production systems into the environment is a challenge that is facing the hog industry in particular within the province of Manitoba (W. Akinremi, personal communication, University of Manitoba, Winnipeg, MB).

There are two methods of managing manure in the province of Manitoba: the liquid manure management system; and the solid manure system that uses straw bedding and hence produces solid manure composed of feces, urine and bedding. While many studies have been carried out on nutrient release and nitrate loss from the liquid hog manure, there are very few studies on the solid hog manure. Also, the nutrient cycling from these two systems of manure management had not been compared in the field. The quantity of nitrate and phosphorus that is lost through downward movement of water in the soil following the application of these manures was unknown (W. Akinremi, personal communication, University of Manitoba, Winnipeg, MB).

Large algal blooms have occurred in Lake Winnipeg as a result of phosphorus driven eutrophication. The main cause of this process appears to be the excess nutrient load from the Red River and other tributaries from agricultural and urban lands. However, the Manitoba Government is trying to enforce stricter measures to reverse this process, with new regulations affecting mainly livestock production (Flaten et al. 2003).

For many years, manures have been applied to the soil based on nitrogen requirements of the crop. In recent years, because of accumulation of soil P and increased risk of P loss from the soil, nutrient management has changed to crop P removal (Toth et al. 2006). Based on the new nutrient regulation in the province, it is mandatory to apply manure based on the crop P requirements, especially, on soils with high soil test P. As such, it is important to compare nutrient loss from these two methods of manure application to the soil: one that is based on N requirements of the crop, and the other that is based on P removal by the crop.

The traditional method of studying nutrient movement within the soil is to sample the soil and analyze the nutrients at different depths. This method may not provide us with an accurate quantity of nutrient loss as the nutrient whose movement is to be studied could have moved beyond the sampling depth. Moreover, soil sampling techniques gives only the concentration of nutrients in solution and cannot be used alone to calculate a nutrient load (Meissner et al. 2014). Nevertheless, soil samples throughout the profile show the pattern of nitrate distribution up to the sampling depth.

"The ability to quantify soil water flow is a prerequisite for the accurate prediction of solute transfer within the unsaturated zone. Only lysimeters enable the quantity of water and solutes that percolate through a soil profile to be determined directly" (Meissner et al. 2014). Therefore, lysimeters are much more reliable in the quantitative assessment of nutrient leaching loss than soil sampling techniques (Goss and Ehlers 2009). In addition to discussing the advantages of using lysimeter, they have potential limitations in terms of financial support and installation effort and accuracy. There is a challenge not only for the accuracy and similarity between the data from the lysimeters (e.g., yield and nutrient leaching) and real agricultural lands, but also for understanding and interpreting of the results obtained. For example, because of the exclusion of lateral movement of water through the soil monolith, vertical movement of water may increase resulting in exaggeration of water and nutrient leaching. Depending on the water table depth, capillary suction causes the water in the field to rise in dry seasons, which does not happen inside the lysimeter (Hertel and Unold 2014). Further, lysimeters are small experimental pots with limitations that can differentiate them from actual agricultural lands. Lieffering et al. (2004) reported in the actual farming lands, the crop had a larger potential rooting volume and this would have led to a greater biomass yield compared to experimental pots.

After five site-years of lysimeter study, the results showed that there was a significant linear relationship between yield within the lysimeters and crop yield from the surrounding plots (Appendix 6.1.A and 6.1.B). Moreover, the leachate results showed with similar volumes of leached water, there was significantly greater nitrate leaching in annual cropping system than in perennials, which is similar to soil nitrate concentration in the annual and perennial plots from the surrounding plots. Therefore, it is reasonable to assess nutrient leaching loss using the lysimeter techniques with undisturbed soil monoliths that mirror the conditions in the field with respect to soil properties, crop management and crop yields. Our study combined the traditional soil profile sampling technique with the use of field core lysimeters to directly capture water and nutrient flowing past the root zone. A complete and accurate picture of the quantity of water, nitrate nitrogen and phosphorus moving past the root zone were obtained using these two techniques.

This dissertation tested the hypothesis that cropping systems and manure management techniques are useful tools that Manitoba hog producers can use to minimize the loss of both phosphorus and nitrate to the environment from animal production. Our results in Chapter 2 confirmed that using a perennial cropping system consisting of a mixture of grasses at the Carman site is a good management option for decreasing nitrate leaching either in the soil profile, or in leached water. Despite the greater aboveground biomass yield of canola than perennial grasses in 2009 and 2011, greater leached nitrate was measured from lysimeters on the annual cropping system in all three years of this study. The differences in nitrate leaching between the perennial and annual cropping systems in this study is best explained by the differences between long and short season crops and the density of their root systems. The long season crop begins to use water and nutrient very early in the growing season and very late into the fall period. During these periods the annual crop has not been established or has long been harvested. It is therefore expected that the perennial crop will use more water and take up more nutrients than the annual crop. This expectation was supported in part by the reduced amount of leached water that was measured on the plots that were seeded to grasses. Perhaps as indicated in studies elsewhere including a winter cover crop would reduce excess soil moisture and nitrate leaching from an annual cropping system.

We speculate that the belowground biomass of perennial grasses was probably greater than those of canola with greater storage of N in the roots of the perennial crop that was not taken into consideration in this study. Therefore, it has to be considered that the measurement of total biomass (below- plus above-ground biomass) as well as total N and P taken up by the plant is necessary. The N-based solid manure treatment produced one of the greatest yields in the annual plot, but this was not the case in the perennial plot resulting often in a significant cropping system by manure treatment interaction. Regular incorporation of the solid manure in the annual cropping system ensured nitrogen mineralization and available nitrogen supply to the crop. The solid hog manure P-based system lost the smallest amount of nitrate which was comparable to the loss from the control treatment. The annual addition of urea nitrogen to the straw based manure applied in 2009 might have resulted in nitrogen immobilization and hence reduced nitrate leaching. Solid hog manure P-based treatment may thus be more environmentally sustainable compared to the other nutrient management treatments examined in this study, however, the agronomic and economic sustainability of this practice may not be practical because of lower productivity.

The results of the research also showed that the P-based treatments resulted in significantly smaller STP levels than the N-based rates at the end of three years. The STP concentrations in the P-based treatment were not significantly different from the control. There was no evidence of significant phosphorus movement beyond the top 15 cm layer. Because increasing STP results in increase of P concentration in runoff, STP buildup should be managed through field rotation, where N-based manure application rates are applied intermittently. However, consecutive years of N-based manure application may result in the movement of P depending on the amount of P that is applied and the degree of soil P saturation. Determination of soil P saturation status is a useful tool for assessing the capacity of the soil to immobilize P. Therefore, this approach may help us to manage N and P application and minimize farmers' concerns about over-application of manure especially at the Carberry site where plots received liquid hog manures without consideration of plant N or P requirements.

Since fertilizer recommendations are based on target yields which are often 10-20% greater than long term average yield, the excess of nitrate above crop requirements can be moved through the soil profile by water. Therefore, nutrient application based on soil tests needs to be refined to reduce nitrate leaching. However, since nitrate leaching occurred even in the control plot where no nutrient was applied, our results suggest that nitrate leaching can be reduced by nutrient management but not eliminated.

Nitrogen application rates are based on soil tests for residual nitrate-N in the top 60 cm usually taken in the fall after harvest. This study showed that nitrate at the 45 to 60 cm depth in the fall of 2010 was no longer there in the spring of 2011 prior to seeding; and in the spring of 2011 considerably more soil nitrate was in the top 15 cm than was measured the previous fall, particularly where manure was applied. Therefore, spring soil nitrate tests may be superior in quantifying the amount of nitrate N that is available to the next crop in some circumstances.

The intensity of $NO_3^-$-N leaching induced by fertilizers or manures depends on the availability of N released by these amendments (Hallberg and Keeny 1993). However, nitrogen in manures is generally less plant available than the N in synthetic fertilizers (Ige et al. 2015). This could be due to losses by ammonia volatilization and the lack of synchrony between uptake and supply of N by mineralization of organic N (Schröder et al. 2010). At Carberry, the concentration of N in leachate and total N leached from fertilizer amended plots were greater than those amended with liquid hog manure (Chapter 4). More than one-half of the plant available N in the fertilizer received treatments was lost through leaching. Therefore, the application of high rates of N is not recommended on sandy soils similar to the one used in this study. There was substantially no difference between liquid hog manures and their corresponding fertilizer treatments for biomass and N uptake. The lack of differences between LH-2500 and LH-5000 for grain yield indicated that LH-5000 supplied more than more than required N to crop and consequently increased nitrate leaching hazard. However, Nikiema et al. (2013) concluded that the production capacity of the soil and the ability of crops to use the nutrient were limited by water supply. The authors reported that in 2002 and 2003, grain yield was not different among liquid hog manure treatments due to less soil moisture content resulting in the poor crop performance in all treatment plots. Based on the results, application of liquid hog manure at rate 2500 gal $ac^{-1}$ is economically and environmentally more desirable and recommended for the sandy soil of the Carberry site.

The field data on water movement in Chapters 2, 3 and 4 were used to simulate the movement of water in the soil profile using the VSMB and to compare model simulated drainage with field data obtained from field core lysimeters on two soil types (Chapter 5). Because $NO_3^-$-N readily moves with water, an accurate simulation of water movement in the soil profile can be a useful tool for estimating $NO_3^-$-N leaching and its effect on groundwater contamination. The use of a simple moisture budget model which requires few inputs is the primary step for investigating water movement and consequently $NO_3^-$-N leaching of manured soils under different cropping systems. Two modified versions of the VSMB model reported by Akinremi et al. (1996) and Hayashi et al. (2010) were used. The finding from this research showed the Akinremi's version had better agreement between simulated and observed soil moisture distribution than Hayashi's version. Both versions of VSMB underestimated soil moisture content in the 45-120 cm depth especially in the second year of simulation. The VSMB grossly underestimated the amount of leached water, possibly due to an overestimation of evapotranspiration. In general, our results indicated that a good simulation of soil water content may not translate into good simulation of other components of the hydrologic cycle. Both versions of VSMB will need to be modified before they can be used to estimate $NO_3^-$-N leaching. Some of the coefficients which are used in VSMB model are empirical including the drying curve parameters. Therefore, it is highly recommended to use the site specific data in the future.

Finally, the variability of soil properties within a landscape can influence the available plant nutrients leading to variability in yield and water contamination. Knowing the spatial variation in risk of nitrate leaching may result in an increase of yield as well as environmental protection. Technological advances have occurred to develop site-specific management in line with spatial distribution of nutrient and, of course, nitrogen (Lehmann, J. and Schroth, G. 2003). Having enough information of physical and chemical soil properties such as texture, structure and pH as well as pedologic characteristics in field can be useful in better understanding $NO_3^-$-N leaching and predicting soil water and $NO_3^-$-N movement through lysimeters.

For future environmental research and monitoring of real field situations, use of modern-day lysimeters with their sophisticated measuring, sampling, controlling and regulating instrumentation would be the best option. Combined with additionally measured meteorological data, lysimeter study is an essential tool for development of water balance models (Meissner et al. 2014).

Our current studies focused on management of cropping systems and nutrients; however, we did not focus on water management and drainage systems. Surface drainage is one of the important strategies to speed up water movement off from the lands and reduce subsurface nutrient losses, especially after spring snowmelt.  Therefore, it will be important in the future to assess the contribution of water management through surface drainage.

## 6.1 APPENDICES

### Appendix 6.1.A

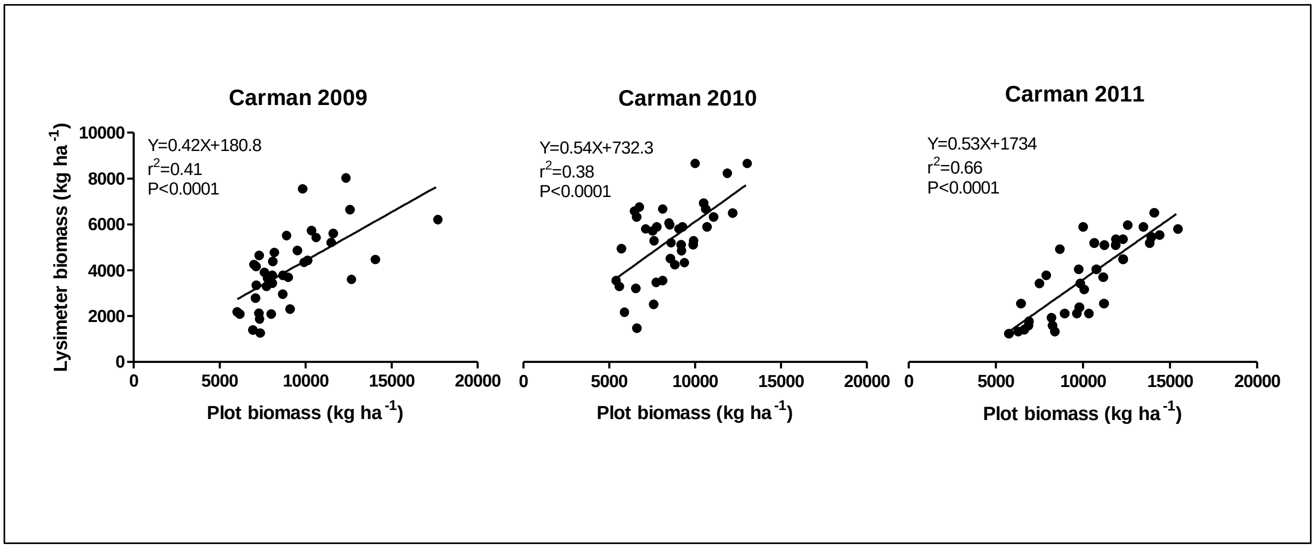


Fig. 6.1.A. Relationship between lysimeter biomass and plot biomass for 3-year study at Carman

### Appendix 6.1.B

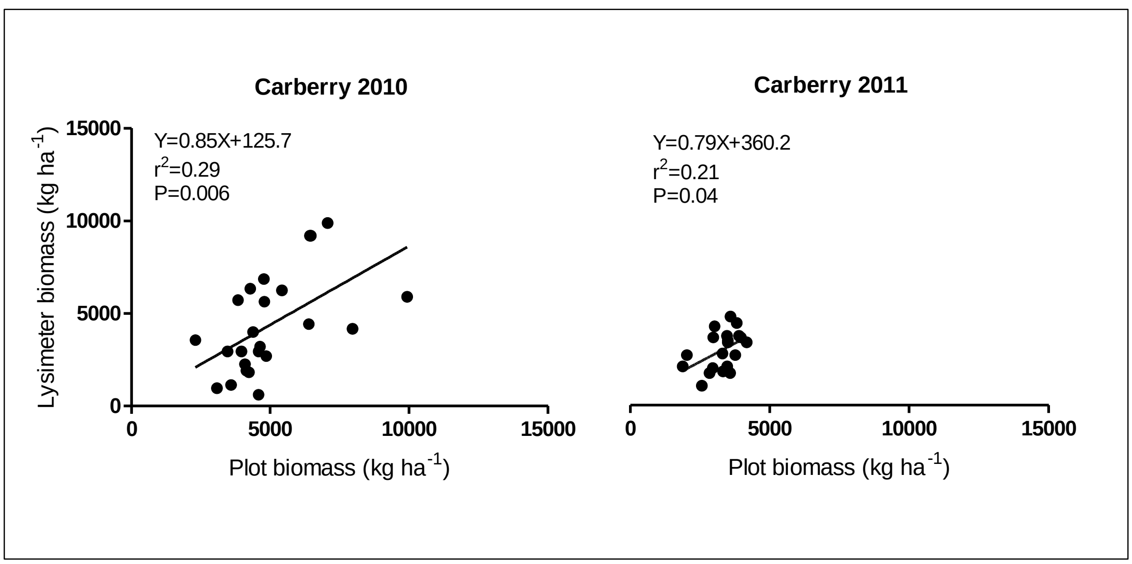


Fig. 6.1.B. Relationship between lysimeter biomass and plot biomass for 2-year study at Carberry

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