

**NITROGEN AND PHOSPHORUS IN SOIL AND GROUNDWATER
FOLLOWING REPEATED NITROGEN-BASED SWINE SLURRY
APPLICATIONS TO A TAME GRASSLAND ON COARSE TEXTURED SOIL**

BY

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ABSTRACT

Coppi, Luca. Ph.D., University of Manitoba, September, 2012. Nitrogen and phosphorus in soil and groundwater following repeated nitrogen-based swine slurry applications to a tame grassland on coarse textured soil. Major Professor: Dr. Mario Tenuta.

Swine slurry is a source of nutrients to grasslands. However, accumulation of N or P can lead to their movement to groundwater. This thesis' research was conducted using a tame pasture fertilized over six years with swine slurry at N-requirement rates, on a gravelly soil in south-eastern Manitoba. Objectives were to determine N and P soil surplus and accumulation over time, soil profile P fractions and soil surface P saturation, and nitrate and dissolved-P in shallow groundwater for treatment combinations of forage utilization (Hay and Grazed), slurry application in spring (Single), split in fall and spring (Split) and no slurry (Control), and Grassed areas of grazed paddocks and Bare areas where cattle congregated around water troughs.

There were less surplus and accumulation of extractable-P in the soil surface for the Split compared to the Single treatment because of less P in fall- than in spring-applied manure. With Grazing, there were greater surpluses of N and P than with Haying, and surface accumulation of extractable-P over time was linear being 16.5 and 11.9 mg P kg⁻¹ year⁻¹ at 0-5 cm for Grazed and Hay treatments, respectively. Labile inorganic-P fractions (water- and bicarbonate-extractable) increased in the Bare and in the Single Grassed treatments. In the Bare treatment, P-sorption capacity at 0-5 cm decreased compared to the Grassed treatment. The surface soil of the Bare areas had large concentrations of water-soluble-P up to 165 mg P kg⁻¹ associated with an increase in P-saturation to 88 % of the sorption capacity, and nitrate was elevated in the soil profile. Concentrations of

nitrate-N and dissolved-P in groundwater were below the environmental thresholds of 10 mg nitrate-N L⁻¹ and 0.025 mg P L⁻¹, respectively, in both the Hay and the Grazed Grassed treatments. In contrast, the Control and Single Bare treatments had nitrate concentrations always above the threshold, and the Bare areas in the Single paddocks had P concentrations of environmental concern in 2009, averaging 0.7 mg total dissolved P L⁻¹. Nitrogen-based slurry applications did not cause leaching of N or P, but Bare areas in grazed pastures are at risk of N and P leaching.

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1. INTRODUCTION

Nitrogen (N) and phosphorus (P) are two macro-nutrients in agriculture. Nitrogen is used by plants to produce proteins, enzymes, nucleic acids and chlorophyll, which are all necessary for metabolism and basic functioning. Phosphorus is used by plants to produce energy (adenosine tri-phosphate or ATP), genetic material (nucleic acids) and cell membranes (phospholipids). Nitrogen is absorbed by plants in higher amounts than P, with dry matter concentration in the ranges of 2.5 - 4 % for N and 0.2 - 0.4 % for P (Brady and Weil, 2002) in plant leaves.

Nitrogen and P are given to crops with inorganic fertilizers as well as organic fertilizers such as manures. When N and P additions are greater than the amounts removed by harvest or grazing, they can accumulate in soil (Toth et al., 2006; Schomberg et al., 2009; Whalen and Chang, 2001; Uusitalo et al., 2007), with the consequential increase of risk of loss to the environment (Kinley et al. 2010; Seidel et al., 2007; Nelson et al., 2005; Sonzogni et al., 1980). A result is the increased N and P concentration in surface waters. In aquatic environments, these nutrients have similar important functions to the ones they hold in soils. However, when their concentration increases by just a slight margin compared to background levels, the population of autotrophic algae grows disproportionately compared to the other aquatic organisms. The decomposition of this enlarged autotrophic biomass causes increased dissolved oxygen consumption through increased respiration. The aquatic environments affected by these algal blooms become low in oxygen level. As a result, the population of fish and other organisms shrinks and biodiversity plummets (Carpenter, 1998). Additionally, waters affected by algal blooms

are by far less attractive for recreational purposes (Kleinman et al., 2011). Thus, both economic activities living off waters and the aquatic biome become greatly damaged by this process called eutrophication. Usually, increased N concentrations are the primary cause of eutrophication in marine waters. In contrast, P is more commonly the growth limiting factor of algae in freshwaters (Carpenter, 1998), even though the role of P in coastal eutrophication has advanced in areas such as the Gulf of Mexico and the Baltic Sea (Kleinman et al., 2011). Thus, in lakes and rivers, it is a slight increase in average P concentration that triggers eutrophication. Eutrophication of rivers and lakes is, unfortunately, very common in human managed regions. A renowned example being the impoverishment of the water ecosystems in the Great American Lakes, a process that started at the beginning of the 20th century (Beeton, 1965).

Lake Winnipeg is the 10th largest lake in the world, and is located in Manitoba, Canada. It has been found that eutrophication is occurring in Lake Winnipeg since at least thirty years ago (Lake Winnipeg Stewardship Board, 2006; Schindler, 2006). The eutrophication of Lake Winnipeg has been linked to increased nutrient load from its main tributaries: the rivers Winnipeg, Red, and Saskatchewan. The Red River contributes a large proportion of the nutrient input into Lake Winnipeg, being 54 % and 30 % of total P and total N annual loading, respectively (Lake Winnipeg Stewardship Board, 2006). This is in spite of the fact that the Red River contributes a relatively small proportion of water volume to the lake. The Red River basin drains the northern part of US states Minnesota and North Dakota, as well southern Manitoba and Eastern Saskatchewan. It is believed that a significant part of N and P inputs into the rivers is from urban centres. However, a larger proportion of nutrients comes from agricultural lands through surface runoff (Lake Winnipeg Stewardship Board, 2006), while leaching of fertilizer added nutrients can also

occur (Cicek, 2006). Agricultural sources of water pollution are called non-point sources because of their dispersed origin. Point sources of pollution, instead, are localized areas such as urban centres and livestock operations. Livestock operations, in particular, can contribute to nutrient loading both as point source pollution from confined operations, and as non-point sources when the stored manure is land applied, usually in areas close to the storages (Pitois et al., 2001).

The current thesis focuses on the effect of manure, obtained from a confined hog operation, applied onto a pasture on N and P concentration and forms in soil and groundwater. It is a small contribution that fits into the much bigger picture of understanding to what extent and how livestock operations contribute to surface and ground water pollution in Manitoba.

1.1 Nitrogen and Phosphorus in Swine Slurry

Livestock herding has gradually shifted from extensive to enclosed systems of pastures and barns over the last one hundred years (Chadwick and Chen, 2002). An example being swine (*Sus scrofa* L.) production. Herd sizes commonly range from 100 to more than 5000 swine slaughtered per year (Zheng et al., 2007), with the largest farms can reach an annual output of 1,800,000 pigs/year (McGlone and Pond, 2003). Manitoba hog population grew from 600,000 in 1976 to 3 million animals in 2006. At the same time, the average number of hogs per farm grew from 762 in 1976 to 3,818 in 2006 (Statistics Canada, 2006). The progressive concentration of hog production reduced the costs of transport of animals and feeds, as well as of storage of animal waste. However, because of this intensive production methodology, large amounts of nutrients are imported as feed into small areas (Carpenter, 1998). Because hogs have a low retention of

nutrients ingested with feeds, a large part of these are emitted in feces and urine that are stored in manure earthen storages (Cooperband and Ward Good, 2002). Every pig produces on average 161-178 kg feces and urine year⁻¹ (Muller, 1980), and for every sow and their progeny, 20 t of manure are excreted every year, or 1.8 t year⁻¹ per growing pig space (Whates and Whittermore, 2006). A common method of hog manure storage in Western Canada is the anaerobic earthen manure storage, where feces and urine with wash water is collected in earthen, open air pools. The mixture of feces and urine is a liquid phase with suspended solids, called slurry. The slurry typically has a moisture content of approximately 94 % (Chadwick and Chen, 2002). Manure composition changes during storage, as anaerobic conditions develop under the surface of the storage lagoons. These conditions lead to mineralization of organic N and losses of N through ammonia volatilization (Cabrera and Gordillo, 1995). The remaining N is approximately two-thirds in inorganic forms, mainly ammonium, that are solubilised in the liquid phase of the manure. The remaining one-third is organic N associated with the solid phase (Cabrera and Gordillo, 1995).

In contrast, P is not susceptible to losses during storage, and is mainly associated with the solid phase of the slurry as inorganic orthophosphate, calcium and aluminium phosphates of various solubility and organic P in phytic acid and esters (Ajiboye et al., 2008). Concentration of N and P in slurry is highly variable among swine operations, and in the same storage it can change over time. Specifically, P content is usually lower if pigs are fed with phytase, an enzyme that contributes to the breaking down of plant organic P during digestion (Simons et al., 1990). Phosphorus concentration is also lower in effluent from secondary and tertiary lagoon cells, because these are filled by the liquid phase of the slurry overflowing from the primary cell, while the P rich solid phase

accumulates in the primary cell. Water soluble fractions of P also tend to increase during storage (Maguire et al., 2006). Despite these efforts to reduce P concentration in tertiary lagoons, the N:P ratio in swine slurry stored anaerobically commonly ranges from 2:1 to 5:1 compared to ratios of 5:1 to 8:1 in plants (Toth et al., 2006; Sistani et al., 2008; Whalen and Chang, 2001).

1.2 Land Applications of Swine Slurry and Surplus Nitrogen and Phosphorus

Animal manures, and among them swine slurry, have always been used as a form of fertilizer in agriculture (Wang et al., 2004). However, the intensive and concentrated nature of today's livestock operations has contributed to a shift towards considering manures as an industrial by-product whose disposal is a nuisance. The large amounts of nutrients imported in swine feeding operations and retrieved in the slurry create a nutrient imbalance at the regional scale. It is often difficult to find sufficient land base nearby to apply this large amount of manure based on crop requirements, thus over application can occur (Carpenter, 2008). The nutrient imbalance at the regional scale is evident in that, in the long-term, more nutrients are imported with feeds that can be disposed of with land applications of manures, and transport of untreated or non composted manure outside the region is not economically viable (Kleinman et al., 2012).

The La Broquerie Rural Municipality in south-eastern Manitoba has one of the highest livestock concentrations in Canada at 129 animal units km⁻² (Flaten et al., 2003). Here, swine and cattle production systems are both widespread, as the soils are usually not suited for annual crop production (Rural Municipality of La Broquerie Information Bulletin, 1999). Thus, it is the common practice for swine slurry to be spread over pastures. In Western Canada and Manitoba, land applications of swine slurry to pastures

are commonly based on N requirement rates for the grass (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006). Applications at N requirement rates for grasslands are problematic for two reasons. Firstly, in hay pastures, the N needed to sustain the growth of the grass is exported out of the field with the hay. At the field scale, these exports maintain a balance between plant-available N additions and removals (Jarvis, 1993). However, since N requirement rates assume that just 25 % of organic N is available to plants in the year of application, a surplus of organic N is allowed to accumulate in the soil. The increase in organic-N is beneficial to soils, but it may lead to greater risk of losses following mineralization and nitrification. Secondly, if the slurry is applied to grazed pastures, removals of N are much lower than in hay pastures because of the low rate of N incorporation in grazing cattle biomass, with the greatest part of grass biomass ingested N being returned to the soil with urine and feces (Williams and Haynes, 1990; Zhang et al., 2006). Thus, in grazed pastures, a larger surplus of N than in hay pastures is to be expected from N requirement based slurry applications (Jarvis, 1993; Wilson et al., 2011). Grazed pastures have also the problem of redistribution of nutrients ingested with the grass and excreted by back grounding cattle. Livestock tend to concentrate and loaf for most of the day in the shade and around water troughs and mineral feeders (Williams and Haynes, 1990). Thus, in these areas large amounts of animal waste accumulate over time (Haynes and Williams, 1999; Sigua and Coleman, 2006), while the combination of trampling, excess nutrients and salinity prevents grass growth, leaving these areas bare and thus devoid of any nutrient removal (Cluzeau et al., 1992; Turner, 1998).

Concentration and accumulation of the inorganic N forms, nitrate and ammonium in soil, is expected to be low with N based applications (Read et al., 2008; Seidel et al.,

2007), except for, possibly, the uppermost soil layer and the bare areas of grazed pastures. Of more direct concern is the accumulation of surplus P. Accumulation of surplus P cannot be avoided with N based application rates, because the N:P ratio of the slurry is lower than that of most crops and grasses (Verloop et al., 2010; Whalen and Chang, 2001). As a consequence, with N based applications, more P is applied to soil than what is needed by the plants, and this P is not exported with grass or animal removals and accumulates (Toth et al., 2006; Schomberg et al., 2009). Accumulation of surplus P in the surface soil over time can often be modelled using a linear function (Duan et al., 2011). An agronomic index of practical importance is the ratio between surplus P and the rate of increase of P concentration in the surface soil as determined by chemical extractions (soil test P), as this value helps to understand how fast soil test P increases following additions. This ratio ranges 5 to 25 kg ha⁻¹ of surplus P per mg kg⁻¹ of increase of soil P, and it varies among soils depending on soil characteristics and on fertility status prior to additions, as well as on the chemicals used to extract P (Zhang et al., 1995; Singh et al., 2001) and on age of applied P, as P is usually stabilized with time.

To limit accumulation of surplus N and P in soils, it is preferable to match the times of application and utilization of nutrient in pastures (Read et al., 2008). Slurry should be added when the growth rate of the grass is higher, to maximize removals. Manures can be land applied in spring, corresponding to the phase of fast activity of plants and microorganisms that allow the breakdown and absorption of nutrients, in summer or in fall. However, in Western Canada, manures are commonly applied in fall, when time and labour are available for application. Split fall-spring applications of manure could be warranted where spring temperatures are low and fall growth can occur

before freeze-up. In this scenario, split applications could reduce environmental losses of surplus N by improving its utilization.

1.3 Nitrogen and Phosphorus Forms and Losses Following Slurry Applications in Coarse Soils and in Presence of a Shallow Water Table

Following addition with manure and removal by crop and grass harvest or by grazing and live weight animal gains, N in soil is prevalently associated with organic matter, especially if manure application is based on crop N requirements (Hooda et al., 2000). This organic bound N is not readily available to plants and not at immediate risk of losses to the environment. However, the organic N can be mineralized to ammonium when conditions are favourable, such as warm temperatures and high moisture.

Ammonium N is bound to the negatively charged soil surfaces and not likely to be mobilized (Hooda et al., 2000). However, this N form can face gaseous losses in the form of volatilized ammonia, and is the substrate of microbial nitrification in soil (Trehan and Wild, 1993). Nitrification of ammonium leads to formation of the highly mobile nitrate ion. Gaseous losses of nitrate as N_2 and NO_x following nitrification and denitrification can reduce the availability of N in soils (Jarvis, 1993; Tenuta et al., 2010). If nitrate accumulates following nitrification, and it is not readily absorbed by plants, leaching losses can occur (Hooda et al., 2000). These losses are more likely in coarse soils with high porosity and fast drainage (Di and Cameron, 2002). If a water table is near the soil surface where nitrate from nitrification is present, leaching can be fast and the concentration of nitrate in groundwater can become a concern for the environmental quality of surrounding surface waters and for the health of people if the groundwater is used for drinking. Concentrations of nitrate above $10 \text{ mg NO}_3^- \text{-N L}^{-1}$ have been linked to

the respiratory health illness in infants called methemoglobinemia or “baby blue syndrome” (USEPA, 1987). There is no clear understanding of what concentration of nitrate in waters eutrophication can occur, thus the drinking water threshold is commonly used also for environmental purposes. In Canada, environmental eutrophication thresholds for N were set at 2.9 mg $\text{NO}_3^- \text{-N L}^{-1}$ for freshwaters and 3.6 mg $\text{NO}_3^- \text{-N L}^{-1}$ for marine waters, as an interim guideline (Canadian Council of Ministers of the Environment, 2003), and recently at 0.39-0.98 mg total N L^{-1} for the Canadian prairies by Environment Canada (Chambers et al., 2012).

Unlike N, surplus P following manure applications does not go through redox reactions in soil, and it accumulates in the surface soil because of the high affinity of P to soil colloids and formation of sparingly soluble precipitates (Edwards and Withers, 1998). Inorganic P in soil is present as phosphate in fractions of varying lability, ranging from water soluble to recalcitrant (Brady and Weil, 2002). These fractions form pools of inorganic P that are in a dynamic equilibrium in soil, and P can be transferred from one fraction to the other depending on additions and removals of P and changes in soil pH and redox status. The equilibrium is a negative feedback, meaning that higher concentrations of labile P fractions in soil lead to increased transfer of P from this pool to the more recalcitrant pools, and vice-versa (Edwards and Withers, 1998). The predominance of the pools is also highly dependent on the pH and other chemical properties of soils. Alkaline and calcareous soils have a prevalence of Ca and Mg bound P in minerals of various degrees of solubility. Acidic soils have a prevalence of Al and Fe bound P in hydroxides (Brady and Weil, 2002). A characteristic of swine slurry additions to soils is the increase of the labile forms of P (water and extractable) compared to the more recalcitrant forms (cation bound and inert) (Hountin et al., 2000; Kashem et al., 2004), the reason being that

the greatest part of untreated slurry P is in a labile form (Ajiboye et al., 2004). Thus, swine slurry additions have the two-fold effect of increasing the soil P concentration as determined with chemical extractions for agronomic purposes, as well as the proportion of environmentally mobile (labile) forms. Additionally, the amount of labile P that can be retained by the soil decreases with increasing P concentration, as more sorption sites become bound to P. This process is called P saturation, and leads highly P saturated soils to become less and less able to retain P until a point where P in runoff and leachate increases to concentrations of environmental concerns (Dou et al., 2009). Even though P losses to water are more common as particulate P through surface runoff, in coarse, flat soils with fast internal drainage the orthophosphate in soil solution can move vertically (Leinweber et al., 1999; Nelson et al., 2005). Shallow groundwater can at this point become contaminated by high P concentrations. A shallow, oscillating water table can also contribute directly to P mobilisation from soil colloids into soil solution because of redox reactions that occur with waterlogging. These reactions could mobilize previously non labile P bound to Fe and organic matter (Ajmone-Marsan et al., 2006; Young and Ross, 2001). Very low concentrations of P in waters can cause eutrophication. In Manitoba, the environmental thresholds have been set at 0.025 and 0.05 mg total P L⁻¹ for rivers and lakes, respectively (Manitoba Conservation, 2001), but new thresholds of 0.10 mg total P L⁻¹ were implemented recently by Environment Canada for the surface waters of the Canadian Prairies (Chambers et al., 2012)

The concentrations of N and P in a shallow groundwater underneath a pasture or other agricultural land are not exclusively dependent on the nutrient loading from the soil, as the groundwater volume has a dilution effect on incoming ions. It has also to be mentioned that high concentrations of N and P in shallow groundwater do not necessarily

pose an environmental concern. The nutrient load from the groundwater to the surface water depends on the dynamic of lateral subsurface water flow as well on processes, such as denitrification and soil sorption, that can reduce the nutrient concentration before seepage into a surface-water body (Hooda et al., 2000; Nelson et al., 2005).

1.4 The La Broquerie Pasture and Swine Manure Management Study Site

The research reported in this dissertation was conducted in the La Broquerie Pasture and Swine Manure Management Study Site, a 40 ha tame pasture, gravelly sand site located in South-Eastern Manitoba in the Rural Municipality of La Broquerie. Several studies had been conducted at the site prior to this research to address agronomical and environmental issues of swine slurry additions to pasture, and they helped to shape the hypotheses of research. Specifically, Holley et al. (2004) found that there was no survival of fecal bacteria from the slurry in the soil. Wilson et al. (2010) showed the beneficial effects of slurry addition to a grazed pasture on forage quality and productivity and on pasture carrying capacity of beef cattle steers. Split applications (spring and fall) of slurry decreased gaseous losses of N_2O in hay fields and CH_4 enteric emissions from grazing steers relative to single application of slurry in spring (Tenuta et al., 2010 and Wilson et al., 2010). Tenuta et al. (2010) also reported the presence of a shallow water table at the site that rose to the surface in wet years, and of a moisture gradient that influenced N dynamics in soil, particularly N_2O emissions. Wilson et al. (2011) did not find a difference over three years for the effect of Single-spring and Split-spring and fall swine slurry applications on N and P removal in pasture forages. However, they reported that haying increased nutrient utilization efficiency and reduced N and P accumulation in soil compared to grazing, and that extractable-P (Olsen-P) concentration increased by 5 mg

kg⁻¹ in the surface 30 cm depth over the three years across treatments. They estimated that 18.9 kg surplus-P ha⁻¹ was required to increase extractable-P (0-30 cm) by 1 mg kg⁻¹, but did not study the relationship in the near-surface soil (i.e. 0-5 cm) that receives the slurry and deeper soil, nor the difference between application timing and forage utilization.

Based on the results of Wilson et al. (2011), the research presented in this dissertation addressed questions that were still unanswered from previous studies. Specifically, soil accumulation of surplus N and P from manure additions has been widely studied in the past. However, there is a lack of knowledge about the effect of additions of swine slurry to grasslands, since most studies have involved cattle manure and field crops (Toth et al., 2006; Damodar Reddy et al., 1998; Zheng et al., 2004). Of the experiments that have studied swine slurry and pastures, most were short-term studies covering two field seasons and involved warm temperature grasses such as bermudagrass (*Cynodon dactylon* L. Pers.) (Adeli et al., 2003; Read et al., 2008). It is also unknown whether surplus P from swine slurry additions accumulates linearly over time, as most studies modelled time linear accumulation of P following addition of fertilizers and other manures (Wang et al., 2010; Huang et al., 2011). Particularly lacking is an understanding of the differences between hay and grazed pastures in P accumulation following swine slurry additions at N requirement rates, as well as an understanding of the effect of manure application timing on P accumulation (Franzluebbers et al., 2002; Read et al., 2008).

Even fewer studies have analyzed the effects of swine slurry additions to grasslands on P fractions and P saturation in calcareous soils, which have largely different chemical sorption mechanisms for P compared to acidic and neutral soils. Most studies have involved other manures, crops or acidic soils (Welsh et al., 2009; Lehmann et al.,

2005; Brock et al., 2007; Griffin, 2003). Also severely lacking is an understanding of the effects of animal waste accumulation in bare areas of pastures on N and P quantities and fractions (Mathews et al., 1994; Sanjari et al., 2008). Specifically, it is unclear how additions by waste deposition in bare areas compare to the addition of swine slurry in the grassed areas of the pasture, and how they interact, given the effect of slurry addition in increasing grazing density (Wilson et al., 2010).

Finally, previous studies analyzing the effect of manure additions to grasslands on N and P leaching, have focused mainly on subsurface drainage and on leachate collected by suction samplers or lysimeters (Booltink, 1995; Legdard et al., 1999; Nelson et al., 2005; Magesan et al., 2006). There is a lack of knowledge of the effect of swine slurry additions on N and P concentrations in a shallow water table, and how the concentrations are affected by groundwater dilution. Knowledge of the differences in nutrient concentration in groundwater between grassed and bare areas of pasture is also lacking (Chardon et al., 2007), as well as an understanding of the effect of groundwater proximity to the soil surface on nutrient leaching and of waterlogging on P solubilisation in the field rather than through laboratory incubations (Shober and Sims, 2009; Villapando and Graetz, 2001).

1.5 Thesis Organization

The thesis is composed of five chapters. This general introduction (Chapter 1) introduces the reader to the background of the issues that the thesis intends to address. Chapters two to four are the data chapters, presented in a manuscript format. In Chapter 2, findings are reported regarding N and P accumulation in the soil profile and in the near-surface soil over five years following swine slurry additions at N requirement rates to

grassland on a coarse soil. In that chapter, comparisons in N and P accumulation in soil between hay and grazed pasture, as well as spring and split fall/spring slurry additions are also made. Chapter 3 reports on extractable N and P, P fractions and P saturation in the same grassland soil following three years of slurry addition. That chapter focuses exclusively on the more nutrient rich grazed pasture and compares N and P fractions in soils between grassed and bare areas. In Chapter 4, the concentration of N and P in the surface of the shallow water table is reported following four years of slurry additions, and during two additional years of slurry applications. In that chapter, both hay and grazed pastures are studied in two separate experiments, with the first experiment focusing on a relative water table depth gradient in the hay pasture, and the second on the comparison between grassed and bare areas of the grazed pasture. In both experiments, the effect of groundwater proximity to the soil surface on saturated zone water concentration of N and P is also studied. In Chapters 2 and 4, chloride in the soil profile and in groundwater, respectively, is used as a natural inert tracer of nitrate in the slurry. Comparison of nitrate and chloride concentrations helps to elucidate the effect of root and microbial immobilization that affect nitrate but not chloride (Booltink, 1995; Peters and Ratcliff, 1998). The final chapter (Chapter 5) 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.

1.6 References

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2. THE EFFECT OF NITROGEN BASED SWINE SLURRY APPLICATION AND OF FORAGE UTILIZATION ON NITROGEN AND PHOSPHORUS SOIL PROFILE CONTENTS AND SURFACE ACCUMULATION RATES

2.1 Abstract

Consecutive applications of swine slurry can lead to accumulation of nitrogen (N) and phosphorus (P) in the soil. This study was conducted over five years to determine if applications of slurry to meet N-demand of forage grass and the grazing or haying of the forage affected N and P accumulation in a coarse texture soil in south-eastern Manitoba. Grassed paddocks were subjected to factorial combinations of slurry application (Control-none, Single-spring, Split-half rate in spring and fall) and forage utilization (Grazed-grazing by steers, Hay-mechanical removal). For each treatment combination, surpluses of N and P (additions of N and P with slurry above the amount removed in hay or steer weight gain), and in residual standing forage in fall were determined. Soil profile concentration to 1.2 m was determined each fall for extractable inorganic N (extract) and P (Olsen), as well as chloride used as a tracer of slurry constituents. Nitrogen and P surpluses were greater with Grazed than Hay treatment because of lower removal as live-weight gain of steers. Surplus-P was greater with Single than Split treatment because of greater concentration of P in the slurry applied in spring. Average nitrate-N content to 1.2 m depth was only 5.5 kg ha⁻¹ greater in slurry treatments than in the unamended Control and slightly more for Grazed than Hay treatment. Just 1.9 % of surplus-N across all treatments was recovered as extractable inorganic N in the top 30 cm of soil in the year of application. Nitrogen and P did not accumulate in appreciable amounts below 30 cm though an increase in soil chloride content below this depth indicated potential for

movement of conservative constituents added with the slurry. Phosphorus accumulated most rapidly at 0-5 cm, and with the Grazed compared to Hay treatment, with rates of 16.5 and 11.9 mg kg⁻¹ year⁻¹, respectively. The accumulation rate of extractable-P was greater with Single than Split treatment. Overall, 3.56 and 10.61 kg ha⁻¹ year⁻¹ surplus-P increased extractable-P by 1 mg kg⁻¹ at 0-5 and 5-30 cm, respectively. Nitrogen-based swine slurry applications to grass forage appear to limit movement of the nutrient to lower soil depths but extractable-P accumulation in the near-surface soil over time will necessitate reducing amounts of surplus-P additions. Particularly, for grazed pastures, P-based application of slurry will be warranted sooner than for hay forage utilization. In this study, here seems to be no environmental benefit to split applying the slurry in spring and fall to reducing N loss, and the benefit in reducing build-up of extractable-P was due to differences in P concentration of the spring and fall slurry.

Abbreviations: ANOVA, analysis of variance; **Cl**, chloride; **DM**, dry matter; **EC**, electrical conductivity; **M**, molar; **N**, nitrogen; **P**, phosphorus; **S.E.**, standard error.

2.2 Introduction

Manures have been used since ancient times to fertilize soils because manures can be rich in nutrients (Wang et al., 2004). However, large confined livestock operations require importation of nutrients in feed, often imported from afar. These nutrients are recycled with manure applications to soil only short distances from operations (Ramirez et al., 2009; Schroder et al., 2010). A specific case of nutrient and manure management issues is the slurry produced by swine (*Sus scrofa* L.) operations. The rural municipality of La Broquerie in south-eastern Manitoba is one of the most intensive livestock

production areas in Canada, with a livestock density of 129 animal units km⁻² (Flaten et al., 2003). Swine manure is stored as a slurry in earthen manure storages that become anaerobic. The slurry has most of its nitrogen (N) in the ammonium and the phosphorus (P) in the inorganic form (Ramirez et al., 2009; Ajiboye et al., 2004; Sharpley and Moyer, 2000), though there might be variability among different farms and times of the year. Because of high transportation costs, manures are usually applied to land close to the storages, and with repeated applications can cause nutrients to build-up in the soil (He et al., 2009; Olson and Papworth, 2006; Nelson et al., 2005).

Repeated application of manure to agricultural soils results in trade-offs. On one hand, manure may build tilth and organic matter as well as soil CEC, on the other hand nitrate from manure can leach to the water table and ammonium and orthophosphate can accumulate in the top soil and be susceptible to run-off or leaching (Kinley et al., 2010; Seidel et al., 2007). Phosphorus as orthophosphate can move downward in soil when the concentration is above the retention capacity of soil (Dou et al., 2009) through a complex combination of pathways and transfer routes. Specifically, leaching losses of P are possible in coarse soils or soils with a shallow water table and prone to wetting and drying cycles that may increase P solubility (Nelson et al., 2005; Griffin et al., 2003; Sonzogni et al., 1980; Whalen and Chang, 2001).

Phosphorus and N are the main contributors to the eutrophication of water bodies with the former being of greater concern for fresh water and the latter for marine systems (Carpenter et al., 1998), even though the role of P in coastal water eutrophication have increased recently (Kleinman et al., 2011). High nitrate concentrations in groundwater (> 10 mg N L⁻¹) also pose concerns for the health of people, particularly infants, if that water is used for drinking (Spalding and Exner, 1993). Lake Winnipeg, the largest fresh water

body in Manitoba, is undergoing a P-limited eutrophication process comparable to that of Lake Erie in 1970 (Schindler et al., 2006). The Red River basin, despite being a minor hydrological contributor to the Lake, is the main source of nutrient loading. Within the basin, non-point sources such as crop production, animal husbandry and forestry contribute to nutrient loading in Manitoba, Saskatchewan and the United States (Cicek et al., 2006).

Rates of manure application to land are usually determined based on the N requirement of the crop grown. While N-based application has the advantage of providing the entire N needed by a crop, it often leads to a greater amount of P added than utilized by the crop (Verloop et al., 2010; Whalen and Chang, 2001; Sharpley et al., 2009). The N:P ratio of manures is commonly 2:1 to 5:1 while harvested parts of most crops have a ratio of 5:1 to 8:1 (Toth et al., 2006; Sistani et al., 2008; Whalen and Chang, 2001). Thus, N-based application rates of hog slurry tend to result in accumulation of P in soil (Toth et al., 2006; Shomberg et al., 2009). Many studies have reported that high concentrations of extractable-P in soil can lead to run-off and leaching losses of P, particularly when the sorption capacity of the soil is overcome (Griffin et al., 2003). Phosphorus based manure applications have the benefit of preventing P build-up, but they require additions of commercial fertilizers or changes to crop and pasture species mixes to ensure adequate N is provided by legumes, to make up for the less than optimal availability of N from the manure to the crop (Sharpley et al., 2009; Kleinman et al., 2007). Additionally, P-based manure applications require a larger land base to utilize manure since the application rates are lower than with the N-based approach.

Swine slurry in Western Canada is usually land applied once or twice a year, in spring, summer or fall. One-time application at high rates might lead to short-term

excesses if the nutrients are readily available, and the soil is bare or the crop is not growing. Partitioning the slurry application into two may improve N and P uptake by the plant and reduce losses. Read et al. (2008) reported the importance of relating the timing of swine slurry application to the growth stage and N-uptake of grass to avoid accumulation of N in soil. Particularly, split applications may have the benefit of providing N at lower rates per application and thus minimize potential for loss. Additionally, split applications (spring and fall) have decreased gaseous losses of N₂O in hay fields and CH₄ enteric emissions from grazing steers (Tenuta et al., 2010 and Wilson et al., 2010). However, fall applications may increase the risk of over-winter and spring-melt losses of N and P.

In the rural municipality of La Broquerie, 89% of the land is in agriculture capability classes 3 to 6, which have substantial limitations for the production of annual crops (Canada Land Inventory, 1969). This results in more than 40% of the area being used for grass forage production (Rural Municipality La Broquerie Information Bulletin 98-24, 1999). Both hog and cattle production systems are present in the area. The pasture land commonly receives nutrient additions from swine slurry, leading to increased carrying capacity of the pastures and coincident rises in live-weight gains per hectare and concentration of forage crude protein, ash, P, K, and Mg (Wilson et al., 2010). Both harvesting hay as forage and grazing grasslands by cattle are common practices in the area for providing feed and for removing nutrients from the soil after slurry application. Harvesting forages for hay is a means of exporting more nutrients from soil than grazing, because with grazing, most nutrients are recycled directly back into the forage system from animal excreta (Zhang et al., 2006, Wilson et al., 2011). Thus, three consecutive

years of grazing led to greater extractable N and P accumulation in pasture soils compared to hay (Wilson et al., 2011).

There is a lack of understanding of the effects of consecutive additions of swine slurry on amounts of surplus-N and P and relation to accumulation in grasslands. Wilson et al. (2011) did not find a difference over three years between Single-spring and Split-fall and spring swine slurry applications on N and P removal. They also reported that haying reduced N and P accumulation in soil compared to grazing by 8 kg N ha⁻¹ and 6 kg P ha⁻¹, respectively, and that extractable-P (Olsen-P) concentration increased by 5 mg kg⁻¹ in the surface 30 cm depth over the three years.. They estimated that 18.9 kg surplus-P ha⁻¹ was required to increase extractable-P (0-30 cm) by 1 mg kg⁻¹, but did not study the difference between the near-surface soil (i.e. 0-5 cm) that receives the slurry and the deeper soil, nor did they study the difference between the effects of application timing and forage utilization on P accumulation.

Thus, a study was conducted at the La Broquerie Pasture and Swine Manure Management Study Site to include an additional two years of consecutive slurry applications to the study of Wilson et al. (2011), to determine the effects of treatments of manure application timing (Single and Split) and forage utilization (Hay and Grazed) on (1) the presence of extractable-N and P in the soil profile (down to 120 cm), (2) the rate of N and P accumulation in near surface soil; and (3) the relation of surplus-N and P additions to the rate of near-surface soil accumulation.

2.3 Materials and Methods

2.3.1 Site Location and Description

The research site was the La Broquerie Pasture and Swine Manure Management Study Site located in the Rural Municipality of La Broquerie in South Eastern Manitoba, Canada. The study site was established in 2003 on 40 ha of tame pasture land that had not recently received fertilizer or manure additions. The pasture site was mainly quackgrass (*Elytrigia repens* L. Nevski) and Kentucky bluegrass (*Poa pratensis* L.) (Wilson et al., 2010). The soil is mapped as Berlo series (70%), a lacustrine loamy fine sand and imperfectly drained soil (FAO Gleyed Luvisol, Canadian system Gleyed Dark Gray Luvisol) and the remaining 30% of area to the Kergwenan series, an outwash loamy sand to gravel and also imperfectly drained soil (FAO Greyzem, Canadian system Gleyed Dark Gray Chernozem) (Hopkins, 1985). The agriculture capability class of soil is 3m (Canada Land Inventory, 1969), meaning suitable for pasture but unsuitable for annual crop production due to fast drainage and low water retention of the coarse soil. The slope at the site was gentle to level (< 2%).

Soil properties were determined in 2003 by collecting soil profile samples in 30 cm depth increments to 1.2 m, and bulk density in soil pits in 2007 with the excavation method. Based on the Canadian classification, soil texture of the site was gravelly loamy sand in the top 30 cm and very gravelly sand in the 30-120 cm depths. Percentage of stones averaged 37 % of total soil weight in the whole profile. Soil bulk density was determined in the field using the excavation method in two soil pits excavated to the water table (125 cm) in August 2007 in each of the west replicate paddocks of the Control Hayed and Single Hayed treatment. Bulk density was 1.68 Mg m⁻³ in the top 30 cm, and 1.98 Mg m⁻³ over the 30-120 cm depth. pH was 7.9 in the top 30 cm and increased with depth to 8.6 at 90-120 cm. Soil was weakly to moderately calcareous, with carbonate content increasing with depth being 20 g CaCO₃ equivalent kg⁻¹ at 0-30 and 79 at 90-120

cm. The cation exchange capacity was 15.1 and 17.0 mol_c kg⁻¹ at 0-30 and 90-120 cm depths, respectively. Generally, plant available-N and P were low when the study was initiated, with total soil N being 0.77 g kg⁻¹, extractable ammonium-N 2.0 mg kg⁻¹, nitrate-N 1.7 mg kg⁻¹, and sodium bicarbonate extractable-P (Olsen-P) 4.0 mg kg⁻¹ in the top 30 cm. Organic carbon concentration decreased with depth being 12.6 and 6.4 g kg⁻¹ for the 0-30 and 90-120 cm depths, respectively.

Drainage was imperfect due to an impermeable clay layer at a depth of 2 meters in the western half of the site which grades deeper (> 5 m) to the eastern edge of the study site (Appendix III.1). As a result, a shallow water table (< 2 m deep) is present year-long in the western half and periodically rose to surface during spring-melt and after successive days of heavy rains.

2.3.2 Experimental Design

The site was divided into 12 paddocks (Fig. 2.1). The experimental design was a generalized complete block with two factors, manure application and forage utilization treatments. Manure application treatments were no slurry (Control), slurry applied at crop N requirement rates in May (Single), and half rate applications of slurry in fall and spring (Split). Swine slurry was applied annually for five years starting in fall 2003 with the Split treatment. Forage harvest treatments were removal of baled dry forage (Hay) and grazing by yearling beef steers (*Bos primigenius Taurus* Bojanus) from June to August (Grazed). Combinations of the levels of the two factors were randomly assigned to a paddock in each of two blocks, with blocking being the depth of the clay layer (shallow in the west and deeper to the eastside). Grazed paddocks were 8 ha (Control) and 4 ha (Single and Split) whereas Hay paddocks were 1.2 ha each. Paddock size was adjusted

with the goal of maintaining a number of eight to ten animals per paddock and the standing forage at 1.0-1.5 Mg dry matter ha⁻¹ throughout the growing season. When the number of animals was in excess and standing forage was < 400 kg ha⁻¹, some of the animals were removed.

Hay paddocks were hayed once a year at the early head phenological stage of the grass, usually in late June (an additional cut in August was obtained in 2004). Steers were provided with fresh water and trace minerals *ad libitum* with Rancher's Choice 8:4:50 Interlake Beef Pasture Premix (Puratone Feeds, Niverville, MB), containing 8% calcium and 4% phosphorus.

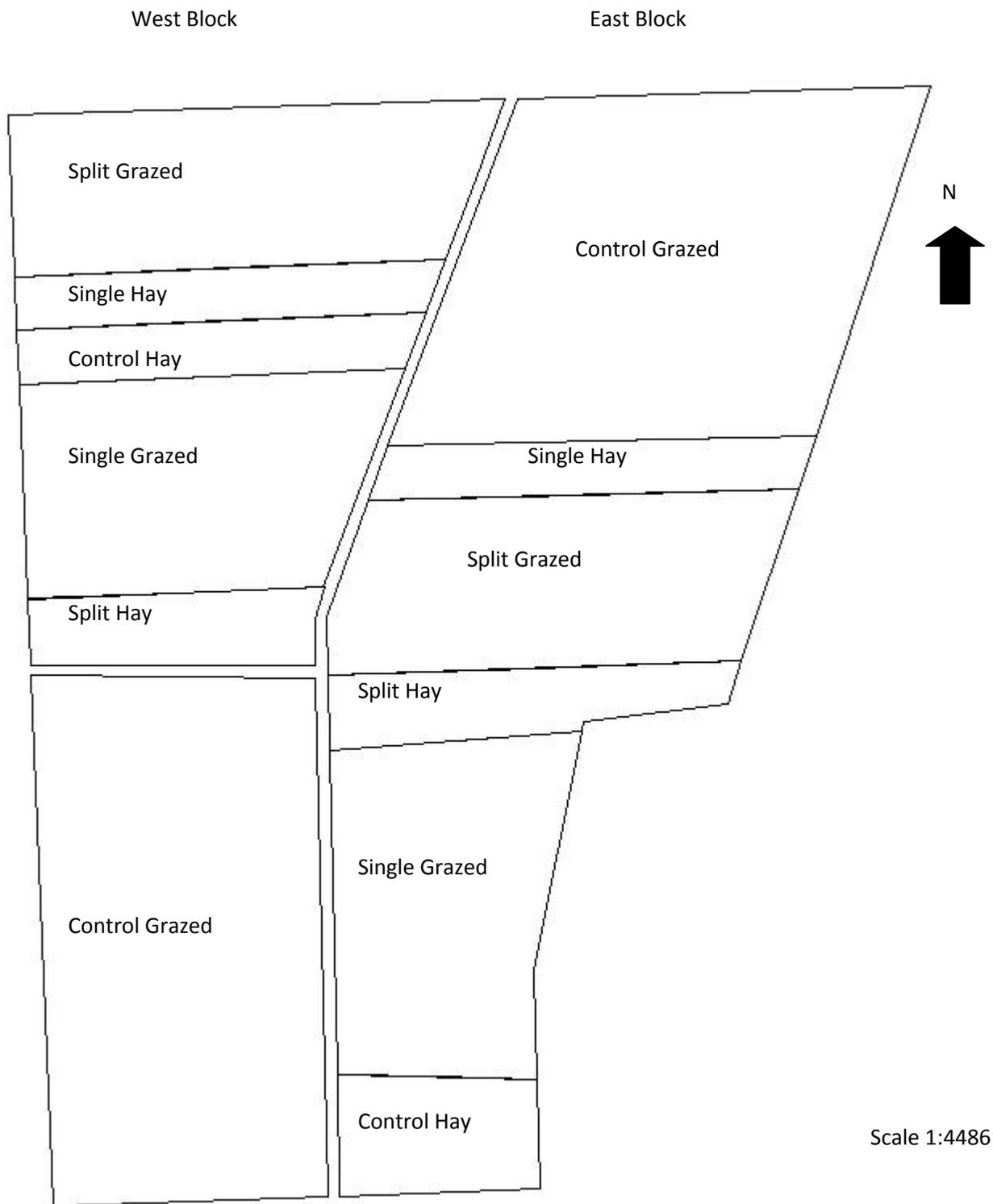


Figure 2.1 Layout of the La Broquerie Research Site (40 ha in total size) showing the subdivision into paddocks for slurry (Control-no manure, Single-slurry applied once at full rate in spring, and Split-slurry applied twice at half rate in spring and fall) and forage utilization (Hay and Grazed) treatment with two replicate blocks.

2.3.3 Slurry Applications

Swine slurry was obtained from the primary cell of an earthen manure storage located next to the field site that received swine waste from five feeder barns. The hogs were fed a diet with phytase, and the mean N:P ratio of the applied slurry was 3.5 and 6.0 for the Single and Split treatments, respectively, (Appendix I.2). Since the swine slurry was sourced from the primary cell of a three-cell earthen lagoon storage system, P addition rates to the soil were greater because of the higher solid content than that of slurry used by most producers in Manitoba. Swine slurry in the storage was agitated (slurry pump agitator GEA Houle Inc., Drummondville, QC) for at least three hours before application to land. The target application rate of swine slurry was according to recommendation (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006) of 123 kg plant available-N ha⁻¹ for grass forage. The Single treatment was applied the first week of May, while the Split treatment received a half rate (62 kg plant available-N ha⁻¹) the beginning of October and the following May (Table 2.1). Manure ammonium- and organic-N contents were estimated prior to each application with an Agros Nova Mk3 (Agros, Lidköping, Sweden) manure N meter and hydrometer, respectively. Plant available-N concentration of the slurry was calculated based upon the in-field determinations of ammonium- and organic-N concentration and the assumptions that 25% of the ammonium-N was volatilized as ammonia under cold humid weather conditions at the time of application, and 25% of the organic-N was readily available to plants in the first year of application (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006). The slurry was surface applied using drag noses in the fall 2003, using a single splash-plate from 2004 to 2006, and dribbled on the surface (24 nozzles across 5 m) for 2007 and 2008. The

application in spring took two to three days and in fall took one to two days to complete.

Actual applied plant available-N was determined by chemical analysis of manure samples by a commercial laboratory (Norwest Lab, Winnipeg, MB or Bodycote, Winnipeg, MB).

Table 2.1 Means of chemical properties and N, P and Cl⁻ concentration of spring and fall swine slurry applied at the La Broquerie site from fall 2003 to spring 2008.

Manure characteristics	-----Spring slurry†-----				-----Fall slurry‡-----			
	Mean	Max	Min	CV	Mean	Max	Min	CV
Dry matter, %	9	12	6	26	4	6	2	40
Electrical Conductivity, dS m ⁻¹	22.1	26.6	18.0	16	24.3	30.1	18.8	20
pH	7.0	7.3	6.8	3	7.2	7.6	7.0	3
NH ₄ ⁺ -N, g N L ⁻¹	3.7	4.1	3.3	7	3.3	3.8	2.6	14
NO ₃ ⁻ -N, mg N L ⁻¹	2.1	3.2	1.3	39	17.3	74.5	0.4	185
Organic-N, g N L ⁻¹	1.9	2.8	1.2	35	1.2	1.7	0.6	34
Total-N, g N L ⁻¹	5.7	6.5	5.1	9	4.6	5.6	3.2	20
Total Phosphorus, g P L ⁻¹	1.6	1.8	1.3	14	0.7	1.2	0.2	49
Total N: Total-P	3.5	4.2	3.0	15	8.5	19.2	4.9	71
Chloride§, g Cl L ⁻¹	1.8	1.9	1.7	7	2.0	2.4	1.7	19

N = 5

† Spring slurry was the only source of manure for the Single treatment and in the spring application of the Split treatment.

‡ Fall slurry and spring slurry were applied to the Split treatment.

§ Slurry analyses for chloride were not available from fall 2003 to spring 2005. The statistics shown for chloride were obtained from the values of three applications (2006-2008).

2.3.4 Weather Conditions

Daily average air temperature and precipitation as well as the climate normal for 1971-2000 were obtained from the Meteorological Service of Canada Long-term Climatic Data for Steinbach Airport (Steinbach, MB; Environment Canada) located 22 km from the site. The growing season (April-October) in 2005, 2006 and 2007 were warmer than the climate normal average by 0.4 to 1.0 °C, while 2004 and 2008 were cooler by 1.1 and 0.8 °C, respectively. All years except 2006 and 2008 were wetter than the normal for the April-October period (422 mm), with April to October cumulative rainfall of 611, 574, and 470 for 2004, 2005 and 2007, respectively. Very wet months during the study periods were May-June in 2005 and 2007, and August-October 2004. The 2008 growing season was slightly drier than average by 9 mm, while 2006 was a very dry year, with just 283 mm falling during the growing season (Table 2.2).

Table 2.2 Monthly mean air temperature and total rainfall for the 2004-2008 growing seasons (April-October) and long-term climate normal (1971-2000).

Month	-----2004-----		-----2005-----		-----2006-----		-----2007-----		-----2008-----		Long-term average	
	Mean Temp (°C)	Rainfall (mm)	Mean Temp (°C)	Rainfall (mm)								
April	3.8	27	7.4	21	8.9	12	4.0	7	3.2	21	4.1	21
May	7.8	137	10.4	110	11.9	23	12.3	122	8.7	45	11.9	59
June	14.4	90	17.6	232	17.4	50	17.9	109	15.2	98	16.6	95
July	18.2	85	19.9	68	21.3	42	20.6	60	18.0	73	19.1	80
August	14.2	137	17.1	42	18.5	26	16.9	51	18.4	67	18.1	69
September	15.1	89	14.4	19	12.8	94	12.3	25	12.5	60	12.1	60
October	6.1	46	7.1	82	3.9	36	6.2	97	6.0	49	5.4	39
Season	11.4	611	13.4	574	13.5	283	12.9	470	11.7	413	12.5	422

2.3.5 Soil, Forage and Animal Measurements

Composite soil samples were collected in the fall (late September before the fall manure application) of every year starting in 2003. Soil samples were taken with a hydraulic flight auger (Giddings Machine Company Inc., Windsor, CO) at 15 locations per paddock at 30 cm depth intervals from the soil surface to 120 cm. Samples at a depth were combined for each paddock. These samples were used to determine contents of extractable-N and -P in the grass main root zone (0-30 cm) and underneath the main root zone (30 to 120 cm). Fifteen surface samples per paddock were also taken by hand with a Dutch auger at 0-5 and 5-30 cm depths and composited. These samples were collected to determine the surface accumulation of extractable-N and -P.

Samples were stored at -20 °C before analysis and were processed by air-drying for three days at 30 °C then ground and passed through a 1.5 mm mesh screen using an in-house rolling mill. The stone fraction (> 1.5 mm) was weighed and discarded. Gravimetric moisture of dried samples was determined by drying at 105°C for 24h. Nitrate and ammonium concentration of soil was determined by extraction using 2 M KCl (5 g dry soil to 25 mL solution) with colorimetric analysis (Keeney and Nelson, 1982) using a Technicon II Autoanalyzer (Pulse Instruments, Saskatoon, SK). Extractable-phosphorus was determined using the Olsen bicarbonate method (Olsen and Sommers, 1982). The extracts were filtered through Whatman filter (no. 42) and analyzed colorimetrically (Murphy and Riley, 1962) using a UV/Visible spectrophotometer at a wavelength of 820 nm (Ultrospec 2100 pro, Biochrom Ltd., Cambridge, UK). Chloride was present in the manure and used to determine the potential for manure nitrate downward movement in soil as relatively small amounts of chloride are utilized by grass forage (Chang and Entz, 1996). Extraction for chloride was performed by shaking one

part soil to two parts deionized water for one hour, then centrifugation of the solution at $12,500 \times g$ for three minutes to separate the supernatant. A 20 mL aliquot of the supernatant was stored frozen until analysis by ion chromatography (ICS-1000, Dionex Canada Ltd., Oakville, ON).

Extractable-N and P content to 120 cm depth (kg ha^{-1}) was determined from each sampled 30 cm layer after accounting for stones and assuming stones had no nutrient retention capability using equation [1]:

$$\text{Extractable-N and -P} = nc_a \times (1-sf) \times bd \times 0.3 \text{ m} \times 10,000 \text{ m}^2 \quad [1]$$

where a is the soil depth increment (0-0.3, 0.3-0.6, 0.6-0.9 or 0.9-1.2 m), nc_a is the extractable-N and P concentration of a (kg kg^{-1} dry soil < 1.5 mm), sf is stone fraction (weight of stones >1.5 mm / (weight of stones + dry weight of soil < 1.5 mm)) and bd is bulk density (kg m^{-3}). Dry matter forage production and total-N and -P concentration of baled hay, standing forage during the grazing season and fall residual standing forage in Hay and Grazed paddocks, as well as steer live-weight gain on the paddocks were also determined by the methods described in Wilson et al. (2010 and 2011).

2.3.6 Nitrogen and Phosphorus Balance, Surplus and Accumulation

2.3.6.1 Balance and Surplus. A simple balance for N and P was performed by taking into account addition with manure, outputs (live-weight gain, hay export and standing fall residual forage), and yearly change in extractable-N and P content in the root zone. Input with manure was calculated from application rate, the total N and P concentration of manure analyzed by the commercial laboratories and estimated volatilization of ammonia-N with surface manure application. The exports in forage tissue were

determined from the hay and residual dry matter yields and their total-N and -P concentration. For export by steers, seasonal live-weight gain and an average estimated total nutrient concentration in animal tissue of 3.03 % N and 0.84 % P were used (Haecker, 1920 via Berg and Butterfield, 1976).

Average annual N and P surpluses ($\text{kg ha}^{-1} \text{ year}^{-1}$) for each paddock were calculated as the difference between cumulative inputs and outputs for study years 2003-2004 to 2007-2008, divided by the number of years:

$$\text{Surplus-N and -P} = (nc_m - nc_o - nc_{r2008}) / y \quad [2]$$

where nc_m is cumulative total N and P applied with slurry (kg ha^{-1}), nc_o is cumulative total N and P content removed with hay or animal biomass (kg ha^{-1}), nc_{r2008} is total N and P content in residual forage in fall 2008 (kg ha^{-1}) and y is the study period in years (5). The amount of total N applied considered was that after accounting for estimated volatilization of ammonia following surface application.

Nitrogen and P in residual forage/hay were considered an output only in fall 2008 (end of the trial). In the previous years, it was assumed that the nutrients stored in the residual forage and residual hay biomass in the fall before freezing were released to the subsequent crop. This nutrient balance doesn't consider other inputs and outputs not measured. The amount of P added to the Grazed paddocks with the mineral supplements was not quantified. However, this addition was likely very small compared to the amount of P added as swine slurry. Also, denitrification and nitrate leaching were not measured and volatilization losses were estimated but not measured.

2.3.6.2 Nutrient Accumulation. The rate of nutrient accumulation over six years (mg

extractable-N and -P kg^{-1} dry soil year^{-1}) was the slope of linear regression models of soil extractable-N and P concentration (mg kg^{-1} dry soil) against time (year). A regression was performed for each treatment combination at two depths (0-5 and 5-30 cm). The amount of surplus-P needed to increase the concentration of extractable-P by 1 mg kg^{-1} at 0-5 and 5-30 cm depth was determined using equation [3]:

$$\text{Surplus-P:Extractable-P} = \text{Surplus-P} / r_a \quad [3]$$

where a is soil depth increment (0-5 and 5-30 cm), Surplus-P ($\text{kg ha}^{-1} \text{ year}^{-1}$) is from equation [1], r_a is the slope of the regression model of extractable-P concentration over years (rate of P accumulation for the soil layer a , $\text{mg Olsen-P kg}^{-1}$ dry soil year^{-1})

2.3.7 Statistical Analysis

Statistical analyses were executed with SAS Software 9.2 (SAS Institute Inc., Cary, NC). Extractable-N and -P, and CI in the soil profile were analyzed using a 3-way factorial analysis of variance (ANOVA; proc Mixed), with manure treatment (Control, Single and Split), forage utilization (Grazed and Hay) and year (2004 to 2008) as main effects. The blocking factor (east/west replicates) was considered a random effect. Surplus-N and -P (equation [1]) were analyzed using a 2-way factorial ANOVA, with fixed effects being manure application (Single and Split) and forage utilization (Grazed and Hay), and block being random. In this analysis, the Control was excluded because this treatment had a nutrient deficit instead of a surplus. The rate of change in extractable-N and P was analyzed using linear regression (Proc REG) over six years, with two replicates for each treatment combination. A slope test was performed to compare the accumulation rate of extractable-P for the treatments. Controls were excluded from this

last analysis because their regression slopes were not different from zero ($P < 0.05$).

All significance levels unless otherwise specified are at $\alpha = 0.10$. When relevant, comparison of treatment means was done with the Tukey-Kramer test with adjustment for β error. For interaction effects, comparison of means was sliced by forage utilization treatment, in order to compare slurry treatments with their respective Control. When data did not meet assumptions of normality (Shapiro-Wilk statistic < 0.90), data were transformed for analysis with transformation given with the presentation of data. Means are reported for untransformed data. Possible heteroschedasticity of residuals was minimized by the Kenward-Roger method for calculation of denominator degrees of freedom in the F test and the Restricted Maximum Likelihood method as a model type (SAS Institute Inc., 2009). A power-type covariance matrix was used among sampling depths (Giesbrecht and Gumpertz, 2004).

2.4 Results and Discussion

2.4.1 Nitrogen and Phosphorus Application, Utilization and Surplus

Spring applied slurry had several properties that differed from that of the fall resulting in a greater amount of slurry applied to the Split treatment (Tables 2.1 and 2.3). The slurry in the fall was more diluted than in spring with dry matter being 4% and 9%, respectively. The greater solid content of the slurry in spring coincided with higher concentrations than the fall slurry for total N (5.7 and 4.6 g L⁻¹), NH₄⁺-N (3.7 and 3.3 g N L⁻¹), and organic N (1.9 and 1.2 g N L⁻¹). However, the most striking difference in characteristics of the slurry from spring and fall was their total P concentrations. Total P concentration was 1.6 in spring compared to 0.7 g L⁻¹ in fall, respectively. The N:P ratio of the fall manure was 8.5, which was greater than 3.5 in spring (Table 2.1).

The smaller concentrations of NH_4^+ and organic-N in the slurry in the fall resulted in greater application rates for the Split treatment to achieve the target of plant available-N addition (Table 2.3). The average application rate of slurry to the Split treatment was $5,000 \text{ L ha}^{-1}$ greater than to the Single treatment. The total and plant available-N added to the slurry treatments was similar, as application rates were adjusted to provide the same rates of available N for the treatments. Plant available-N applied was slightly more than the target addition of $123 \text{ kg N ha}^{-1} \text{ year}^{-1}$ by 7 and $11 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for the Single and Split treatments, respectively. The marginally higher than expected addition was because of underestimation of NH_4^+ concentration at application time using the Nova meter. Despite the greater amounts of manure applied to the Split treatment, the much lower N:P ratio for the manure in the fall resulted in $15 \text{ kg ha}^{-1} \text{ year}^{-1}$ more P added with the Single than Split treatment (Table 2.3).

The removal of N and P through above ground forage utilization by growth of steers and by haying increased with slurry treatment compared to the unamended Control (Table 2.3). Live-weight gain removal of N and P by steers increased with slurry treatment because of the previously observed increase in forage productivity resulting in a 2.6 fold greater density of animals on manured than control paddocks (Wilson et al. 2010). Live-weight gain removal of N was 3, 11, and $10 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for the Control, Single and Split treatments, respectively (Table 2.3). For P, the live-weight gain removal was smaller than for N, being 1, 3, and $3 \text{ kg P ha}^{-1} \text{ year}^{-1}$ for the Control, Single, and Split treatments, respectively. Slurry addition increased hay yield, being 3.4 fold greater compared to without addition (Table 2.3). This resulted in larger removal of N with hay for the manure treatments. Removal of N with hay was 10, 67, and $58 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for the Hay Control, Single, and Split treatments, respectively. Similarly, P removal as hay

was greater with slurry treatment and it was 1, 11, and 10 kg P ha⁻¹ year⁻¹, respectively. Both N and P removals were much greater in Hay than Grazed treatments (Table 2.3).

Nitrogen and P in residual fall standing forage also increased with slurry treatment (Table 2.3). This was a result of an increase in dry matter levels and concentration of crude protein and P in the forage (Wilson et al. 2010). Nitrogen in the residual fall standing forage for the Grazed treatment increased by 1.7 and 1.4 fold of Control levels whereas removal was even larger for the Hay treatment, being 2.4 fold more for Hay Single and Split treatments compared to the Hay Control. Phosphorus in residual fall standing forage among treatments had a similar pattern to N, with P amounts increasing with slurry application and Hay than Grazed forage utilization.

Table 2.3 Average \pm 1 standard error of the mean of annual nutrient inputs and outputs for the study period (2004-08). Treatments were combinations of manure applications (Control, Single and Split) and forage utilization (Grazed and Hay).

		-----Grazed-----			-----Hay-----		
		Control	Single	Split	Control	Single	Split
Inputs							
	Slurry Rate, '000 L ha ⁻¹	0	40 \pm 2	45 \pm 2	0	40 \pm 2	45 \pm 2
	Total N [†] , kg ha ⁻¹	0	189 \pm 14	187 \pm 7	0	189 \pm 14	187 \pm 7
Manure application	Available-N, kg ha ⁻¹	0	130 \pm 10	134 \pm 6	0	130 \pm 10	134 \pm 6
	Total P, kg P ha ⁻¹	0	63 \pm 3	48 \pm 3	0	63 \pm 3	48 \pm 3
	Total N:P [†]	NA	3.0 \pm 0.2	4.9 \pm 1.1	NA	3.0 \pm 0.2	4.9 \pm 1.1
	Soluble Cl, kg ha ⁻¹ §	0	63 \pm 7	75 \pm 7	0	63 \pm 7	75 \pm 7
Outputs							
LiveWeight Gain	Rate, kg ha ⁻¹	97 \pm 3	350 \pm 17	318 \pm 15	-	-	-
	Total N, kg ha ⁻¹	3 \pm 0.1	11 \pm 0.5	10 \pm 0.4	-	-	-
	Total P, kg ha ⁻¹	1 \pm 0.1	3 \pm 0.1	3 \pm 0.1	-	-	-
Hay	Yield, t DM ha ⁻¹	-	-	-	0.8 \pm 0.1	3.5 \pm 0.3	3.5 \pm 0.3
	Total N, kg ha ⁻¹	-	-	-	10 \pm 1	67 \pm 7	58 \pm 5
	Total P, kg ha ⁻¹	-	-	-	1 \pm 0	11 \pm 1	10 \pm 1
	DM, t ha ⁻¹	1.1 \pm 0.1	1.5 \pm 0.1	1.5 \pm 0.1	2.1 \pm 0.6	3.4 \pm 0.8	3.8 \pm 0.6
Residual forage/hay‡	Total N, kg ha ⁻¹	14 \pm 2	45 \pm 5	40 \pm 4	24 \pm 6	68 \pm 18	71 \pm 14
	Total P, kg ha ⁻¹	2 \pm 0	4 \pm 0	4 \pm 0	3 \pm 1	9 \pm 2	10 \pm 2

† Added-N applied is the summation of nitrate-N, organic-N and 75% of ammonia-N assuming that 25% of NH₃ is lost through volatilization at the time of application (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006).

‡ Residual forage /hay indicate the forage and hay left standing in the field at the end of the growing season. Residual forage was an output only the last year of the study (2008). In the previous years, residual nutrients were assumed to be returned to the soil with above ground biomass decomposition in the fall and subsequent spring.

§ Manure analyses for chloride were not available from fall 2003 to spring 2005 (n = 3).

The amount of plant available-N and P applied with slurry, removed with steers or hay, and remaining in residual fall forage was used to determine surplus amounts for the treatments. Surplus-N was similar in Single and Split application timing because rates of manure N application and forage N removal were similar for both timings (Table 2.4). Surplus-N was 76 % of added N averaged across treatments. In contrast, the surplus was 55.5 kg N ha⁻¹ year⁻¹ greater with Grazed than the Hay treatment (Table 2.4), and surplus-N was 62 % of added N for the Hay treatment and 90 % of added N for the Grazed treatment. Considerably less N removal with steers than with haying accounted for the difference in surplus for the two methods of forage utilization.

Manure ($P = 0.01$) and to a lesser degree, forage utilization ($P = 0.07$) treatment, significantly affected the amount of surplus-P (Table 2.4). The surplus was 15 kg P ha⁻¹ year⁻¹ more in the Single than Split treatment because P added with the slurry was greater in the former. As a result, the percentage of surplus P relative to applied P was similar between the manure application treatments, and it was 87 and 85 % for the Single and Split, respectively. The Grazed treatment resulted in an 8.5 kg P ha⁻¹ year⁻¹ larger surplus than with forage utilization by haying (Table 2.4) due to low rates of P removal by grazing steers. Surplus-P was 94 % of added P for the Grazed treatment and 79 % of added N for the Hay treatment.

Table 2.4 Mean values for annual N and P surplus of manure and forage utilization treatments for the study (2004-08). Shown are results of analysis of variance for main treatment and interactions effects.

	Surplus-N	Surplus-P
	-----kg ha ⁻¹ -----	
<i>Manure application</i>		
Single	143.0	55.0a
Split	142.6	40.8b
<i>Forage utilization</i>		
Grazed	169.8a	52.2a
Hay	115.8b	43.7b
ANOVA P > F		
Manure	ns	0.006
Utilization	0.003	0.01
Interaction	ns	ns

N=12. Significance at $\alpha = 0.10$. Means followed by different letters indicate significant difference at $\alpha = 0.10$ with the Tukey-Kramer comparison test. Control treatments were not included in the analysis because these treatments had a nutrient deficit instead of a surplus.

A surplus of N and P added to cropping systems can result in accumulation and movement of the nutrients to the environment. Surprisingly, few experiments have determined surplus-N and P additions with swine slurry to grass, while more have studied a variety of annual crops and a variety of management practices affecting additions and removals. Swine slurry applied to bermudagrass (*Cynodon dactylon* L. Pers.) in Mississippi resulted in 29 to 57 % surplus of total N applied (Adeli et al. 2003) compared to 62 and 91 % for the Hay and Grazed treatment in the current study. Furthermore, with bermudagrass in Mississippi, Read et al. (2008) found 37 and 40 % surplus of total N added with swine slurry at rates of 263 and 468 kg N ha⁻¹. The surplus-N in the current study was greater than for the studies with bermudagrass likely because generally only one cut was taken for hay each year and in the case of the Grazed treatment, low N removal rates.

Adeli et al. (2003) and Read et al. (2008) reported smaller N removals and larger surplus-N for swine slurry applied in fall than in spring to bermudagrass. They suggested that lesser productivity of the grass in the cool fall months reduced removal of N with mowing harvest. In contrast, we did not observe such a difference between Single and Split treatments, indicating that N applied in the fall and spring was utilized similarly by forage. The pasture in the current study was comprised of cool-season grasses that should be better suited to N uptake in fall than warm season bermudagrass.

One of the main factors affecting surplus-P in a cropping system receiving manure is whether applications are at rates of N or P requirement for the crop. Toth et al. (2006) reported surplus-P of 81 and 15 kg ha⁻¹ year⁻¹ for solid dairy manure applied to orchardgrass (*Dactylis glomerata* L.) at N and P requirement rates, respectively, over four years in Pennsylvania. The surplus-P corresponded to 79 and 44 % of applied P supplied

at N and P requirement rates, respectively. Adeli et al. (2003) reported surplus-P of 82 and 51 % of swine slurry applied at a higher (154 kg P ha⁻¹) and lower (37 kg P ha⁻¹) rate to bermudagrass. For the reasons discussed previously, Adeli et al. (2003) also reported more surplus-P with fall swine slurry application than with spring application. These values are comparable to 85 and 64 % of added P being surplus in the current study for Grazed and Hay treatments, respectively. For annual crops, studies reported very low surplus-P from cattle (Damodar-Reddy et al., 1998) and dairy manure (Zheng et al., 2004) applied at crop P requirement rates, being 0 and 12.5 kg ha⁻¹ year⁻¹, respectively.

2.4.2 Extractable Nitrogen and Phosphorus in the Soil Profile

2.4.2.1 Nitrogen. Extractable ammonium and nitrate were stratified within the soil profile, with higher contents for the surface (0-30 cm) than lower depths. Extractable ammonium was not affected by slurry and, generally not affected by forage utilization treatment (Appendix I.7). An exception was ammonium content being higher ($P = 0.04$) with Grazed (4.4 kg N ha⁻¹) than Hay (3.7 kg N ha⁻¹) treatment at 30-60 cm.

Generally, nitrate content was greater for the 0-30 cm than lower depths (Table 2.5). For this surface layer, the amount of nitrate increased with manure ($P = 0.03$) and for Grazed than Hay ($P = 0.04$) utilization of forage (Table 2.5). There was a weak year effect ($P = 0.07$) for nitrate in soil with 2004 having less nitrate than 2005. Similarly, there was a weak manure x forage utilization interaction ($P = 0.07$) caused by greater nitrate content with the Single than the Split treatment in the Grazed utilization type, while the Split had greater N content than the Single treatment in the Hay utilization type (Fig. 2.2a). Slurry treatment did not affect nitrate content below 30 cm (Table 2.5).

However, below 30 cm, there were consistent differences among depths over the years (P

< 0.02) with 2005 generally having more and 2008 less nitrate. Rainfall in June 2005 was abnormally high (232 mm) compared to other years and the long-term climate normal average for the last 30 years (95 mm; Table 2.1), but May 2005 had less rain than other years except 2006 (Table 2.1). There was a weak effect of forage utilization on nitrate in soil for only the 30 – 60 cm depth with nitrate in the Grazed treatment (4.5 kg N ha⁻¹) being slightly greater than in the Hay treatment (3.4 kg N ha⁻¹). A stronger manure x forage utilization effect ($P = 0.02$) on nitrate soil content was caused by nitrate in the Single treatment being greater with Grazed than Hay treatment (Fig. 2.2b).

Table 2.5 Mean values for nitrate N, extractable-P (Olsen-P), and Cl⁻ soil profile contents for manure and forage utilization treatments and year of the study. Shown are results of analysis of variance for main treatment and interactions effects.

	NO ₃ ⁻ -N				Extractable-P				Cl ⁻			
	Depth, cm											
	0-30	30-60	60-90	90-120	0-30	30-60	60-90	90-120	0-30	30-60	60-90	90-120
	kg ha ⁻¹											
<i>Manure</i>												
Control	7.5b	3.9	2.1	2.0	17.9b	4.5b	2.9b	3.7	12.1b	15.4b	18.9b	21.9b
Single	11.8a	4.0	1.9	2.3	83.6a	11.9a	8.3a	5.2	42.3a	33.1a	29.7a	35.7a
Split	11.8a	3.9	2.2	2.3	64.0a	10.3a	5.6a	4.4	47.8a	33.5a	29.8a	33.3a
<i>Forage utilization (above ground only)</i>												
Grazed	12.3a	4.5a	2.1	2.2	62.5a	10.3	5.5	5.0a	38.0	30.1	26.4	30.8
Hay	8.4b	3.4b	2.0	2.2	47.9b	7.6	5.7	3.9b	30.1	24.6	25.9	29.7
<i>Year</i>												
2004	6.3b	2.8bc	1.5abc	1.2b	22.5b	5.7ab	2.7b	1.3ab	16.6d	15.4b	18.7	23.4
2005	13.4a	9.1a	6.0a	6.8a	36.1b	4.5b	5.7b	0.6b	28.4c	22.3ab	22.6	24.8
2006	11.6ab	4.2ab	1.9ab	2.1b	55.4a	18.4a	11.0a	8.7a	29.0bc	38.3a	31.3	33.6
2007	8.1ab	2.6bc	0.9bc	0.9bc	73.5a	10.2ab	7.2a	7.9ab	53.0a	31.8a	27.3	36.0
2008	12.5ab	1.0c	0.0c	0.0c	88.3a	5.8ab	1.5b	3.7ab	43.5ab	28.9a	30.8	33.5
ANOVA $P > F$												
Manure	0.03	ns	ns	ns	<0.0001	0.05	0.01	ns	0.001	0.001	<0.0001	0.0001
Forage utilization	0.04	0.08	ns	ns	0.03	ns	ns	0.099	ns	ns	ns	ns
Year	0.07	0.01	0.02	0.002	<0.0001	0.09	0.001	0.06	0.002	0.04	ns	ns
Manure x Forage utilization	0.07	0.02	0.07	ns	ns	0.06	ns	0.07	ns	0.04	ns	ns
Manure x Year	ns	ns	ns	ns	0.002	ns	ns	0.08	0.04	0.03	ns	ns
Forage utilization x Year	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Manure x Forage x Year	ns	ns	ns	ns	ns	ns	0.02	ns	ns	ns	ns	ns

N=60. Analysis was conducted separately for each depth on log-transformed data. Significance at the 0.1 level. Means followed by different letters in the same column for each utilization type (interaction means) and main effect indicate significant difference at $\alpha = 0.1$ with the Tukey-Kramer comparison test.

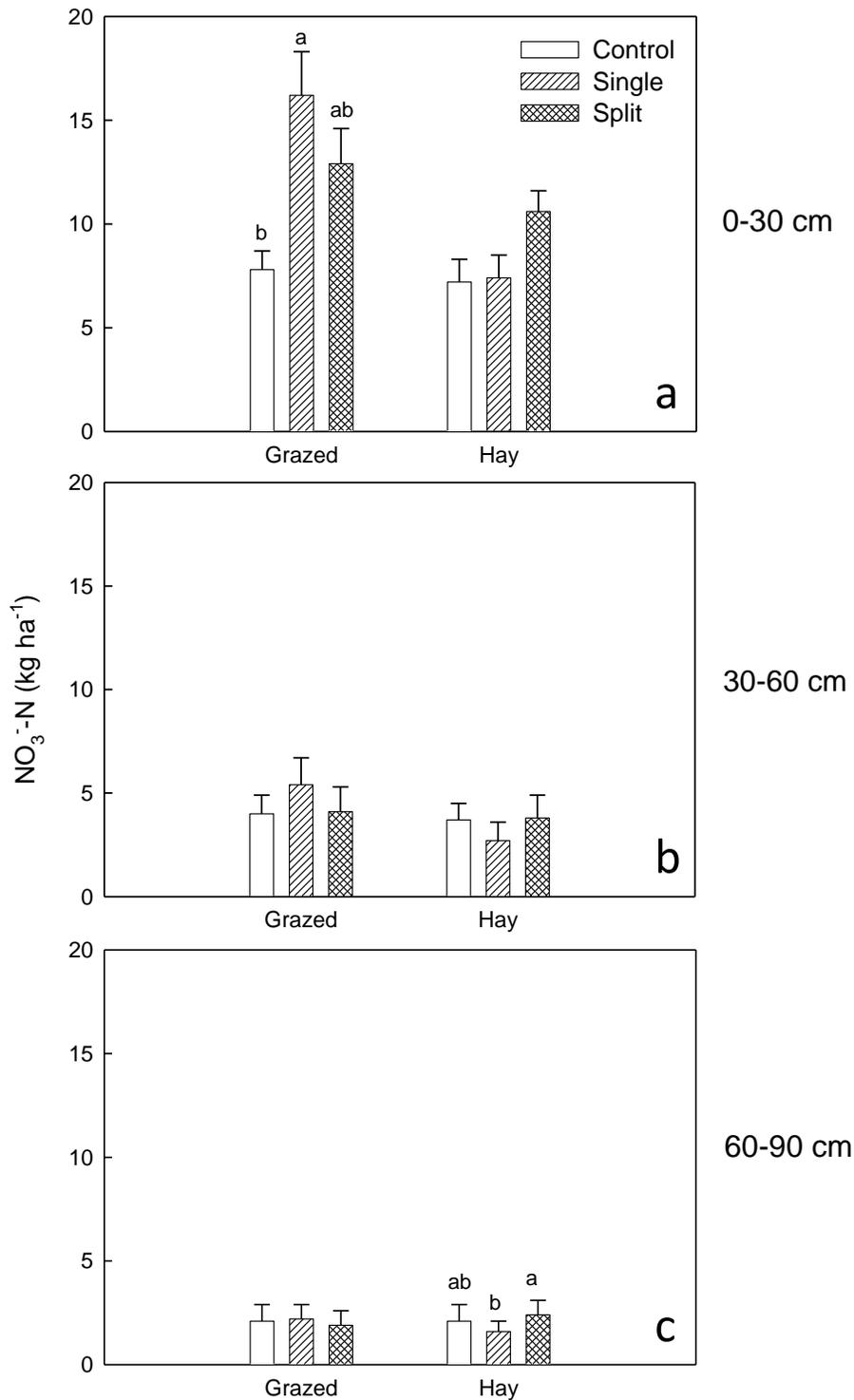


Figure 2.2 Mean nitrate-N content + 1 standard error of the mean for manure x forage treatment combinations in the 0-30 (a), 30-60 (b) and 90-120 (c) cm depths of soil. Means indicated by the same letters are not significantly different at $\alpha = 0.1$ with the Tukey-Kramer comparison test.

Despite an estimated surplus of 144 kg N ha⁻¹ across manure addition and forage utilization treatments due to the fact that the slurry was applied at plant available N, rather than total N, forage requirement rates, nitrate did not appear in appreciable amounts in the soil profile. This confirms that application rates set to plant available-N concentration of manure to meet N requirement by forage crops are environmentally viable. Nitrate content for the whole profile (0-120 cm) was 15.5 kg N ha⁻¹ for the Control and 20 kg N ha⁻¹ for Single and Split treatments across forage utilization types. Others have also reported that with swine and cattle manure application at N requirement rates, nitrate accumulation was limited in soil cropped to bermudagrass and a mixture of perennial ryegrass (*Lolium perenne* L.), meadow fescue (*Festuca pratensis* L.), timothy (*Phleum pratense* L.) and Kentucky bluegrass (Read et al., 2008, Seidel et al., 2007). On the contrary, when application rates of swine and cattle manure are in excess of crop N requirement rates to bermudagrass, alfalfa (*Medicago sativa* L.) and timothy, nitrate can accumulate in surface soil and below the root zone (Adeli and Varco, 2001; Olson and Papworth, 2006), particularly in sandy soil grasslands (perennial ryegrass) (Schroder et al., 2010).

Overall, just 1.9 % of surplus-N was recovered as extractable ammonium and nitrate in the top 30 cm of soil in the year of application. This percentage is much less than the 22-81 % of surplus-N recovered as extractable mineral N reported by others (Read et al, 2008; Whalen and DeBerardinis, 2007, King et al., 1985, Adeli et al., 2003).

The fate of the 98.1 % of surplus-N not recovered as extractable mineral N is uncertain. It is likely that most of the surplus N was in organic form, and ammonium and nitrate were made available through mineralization and nitrification slowly over time. Gaseous losses through nitrification and denitrification were not determined in the current

study. Whalen and DeBerardinis (2007) estimate < 2.1 % of added N from swine slurry to bermudagrass and tall fescue to be lost by denitrification. Tenuta et al. (2010) determined < 2 kg N₂O-N ha⁻¹ was lost with slurry treatment to the Hayed paddocks. They did not determine N₂ losses through denitrification or NO_x emissions by nitrification and denitrification. As well ammonia volatilization was estimated but not measured in this current study. The latter was assumed to be 25% of ammonium-N of the slurry added as a surface application in cool humid conditions (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006) when quantifying surplus-N. For this project site, Ominski et al. (2008) reported ammonia volatilization to be 35 kg N ha⁻¹ for spring applied slurry in 2006, which is 23 % of applied ammonium-N.

Leaching losses of N were also not determined in the current study. There were no large increases in extractable-N below the root zone (0-30 cm) but leaching still cannot be ruled out because sampling was done only once a year, every fall. McGechan et al. (2001) for a mixed grass site modelled leaching losses of 4-20% of applied swine slurry N. At our site, we speculate that the largest part of not-accounted-for surplus-N was incorporated into the root system of the grasses and assimilated into soil organic matter. In 2006, average root dry matter in the top 5 cm soil in the Control Hay was 8.8 g kg⁻¹ whereas it was greater (P < 0.05), being 12.8 g kg⁻¹, in the Single Hay treatment. Similarly, soil organic carbon at 0-5 cm also increased (P < 0.01) from 2.38 to 2.64 % in the Control and Single treatments, respectively.

A difference in extractable ammonium and nitrate between Single and Split treatments was not found. This was surprising because Read et al. (2008) and McGechan et al. (2001) reported larger unaccounted for surplus-N and leaching losses with fall than spring application of swine slurry to grass in Mississippi. In the current study, Single and

Split treatments had equal removal, surplus and extractable contents of N, and this result seems to indicate that N applied in fall and spring is utilized similarly by the grass.

2.4.2.2 Phosphorus. Similarly to extractable-N in soil, extractable-P was strongly, vertically stratified, with highest contents in the surface layer (0-30 cm). This depth of soil was the most responsive to the effect of manure, forage utilization and year on extractable-P content. Manure addition resulted in extractable-P increasing by 63.7 and 46.1 kg P ha⁻¹ year⁻¹ for the Single and Split treatments, respectively, above that of the Control (Table 2.5). Averaged across all treatments, extractable-P content in soil increased with each year of manure application. However, the lack of slurry application to the Control resulted in little change in extractable-P content versus a steady increase with each manure application to the Single and Split treatment (Fig. 2.3a). This resulted in a significant manure x year interaction ($P = 0.002$) for the 0-30 cm depth. Average extractable-P with the Grazed treatment was 14.6 kg P ha⁻¹ greater than with the Hay forage utilization treatment (Table 2.5).

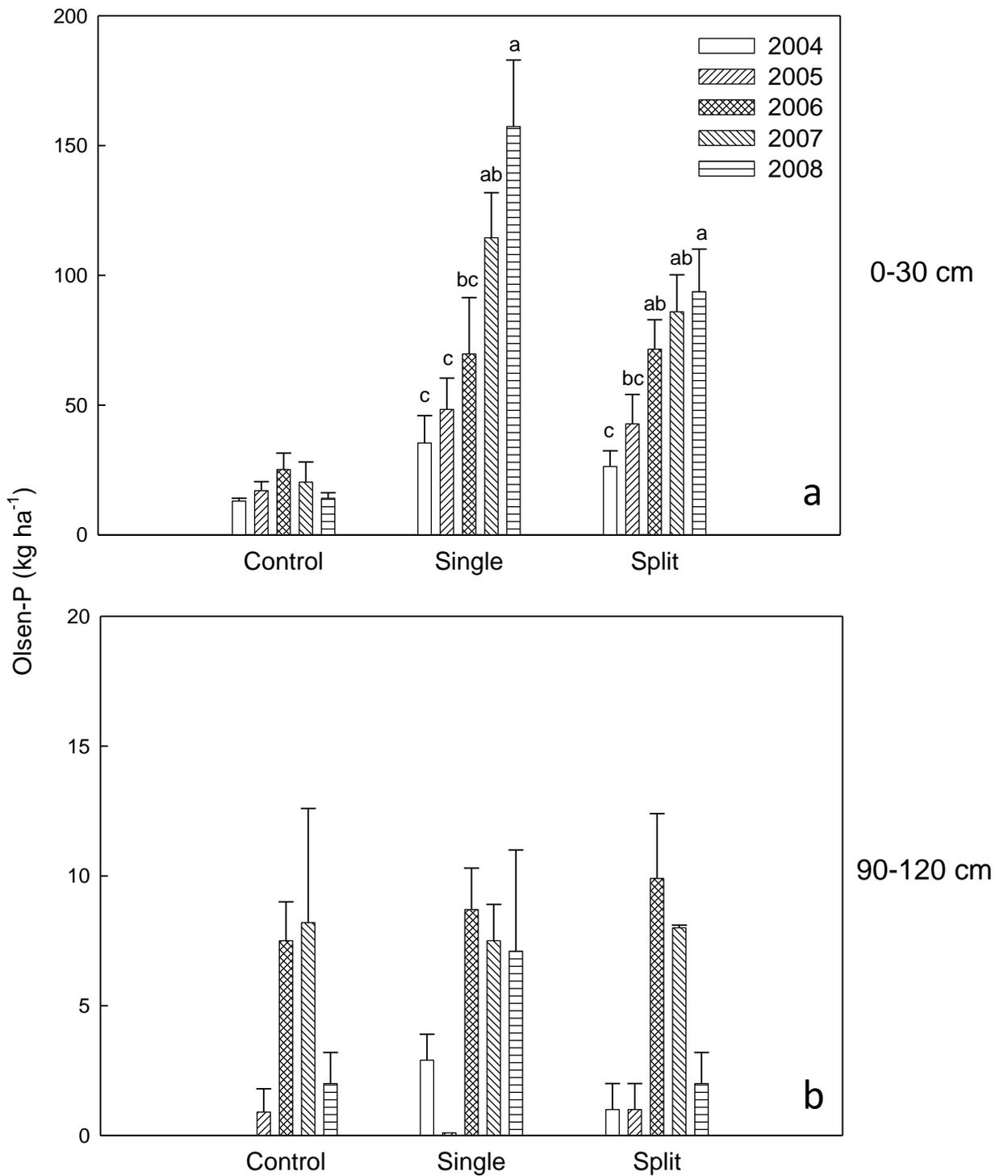


Figure 2.3 Mean extractable-P content + 1 standard error of the mean for manure x year treatment combinations in the 0-30 (a) and 90-120 (b) cm depths of soil. Means indicated by the same letter are not significantly different at $\alpha = 0.1$ with the Tukey-Kramer comparison test.

Slurry application also increased extractable-P content in the 30-60 and 60-90 cm depths (Table 2.5). However, these depths showed a less marked increase in extractable P as a result of slurry application than for the surface depth. Average extractable-P content across treatments was strongly (60-90 cm) and weakly (30-60 and 90-120 cm) affected by year (Table 2.5). Generally, 2006 had the highest P contents at these depths than other years. The 2006 growing season was very dry in spring and summer, but wetter than normal in September (Table 2.1). It is possible that water stress limited forage growth and P uptake, and rains before the sampling in fall of that year contributed to movement of P lower in the soil profile.

A weak manure x forage utilization interaction was observed for the 30-60 ($P = 0.06$) and 90-120 ($P = 0.07$) cm depths (Table 2.5). The interaction was due greater P content for Single than the Split treatment in the Grazed utilization type, while the Split had greater P content than the Single treatment in the Hay utilization type (Fig. 2.4a and b). A manure x forage x year interaction ($P = 0.02$) was observed for only the 60-90 cm depth. Here, extractable-P content was larger with Single than Split application under Grazed utilization, while with Hay, Single had more P than Split application in 2005 and 2006, but Split had more P than Single in 2007 and 2008 (data not presented). It should be noted that the differences were smaller than 7 kg P ha^{-1} except in 2005 when Single Hayed had an unusually high P content of $26.8 \text{ kg P ha}^{-1}$. Furthermore, there were no significant differences among means ($\alpha = 0.1$).

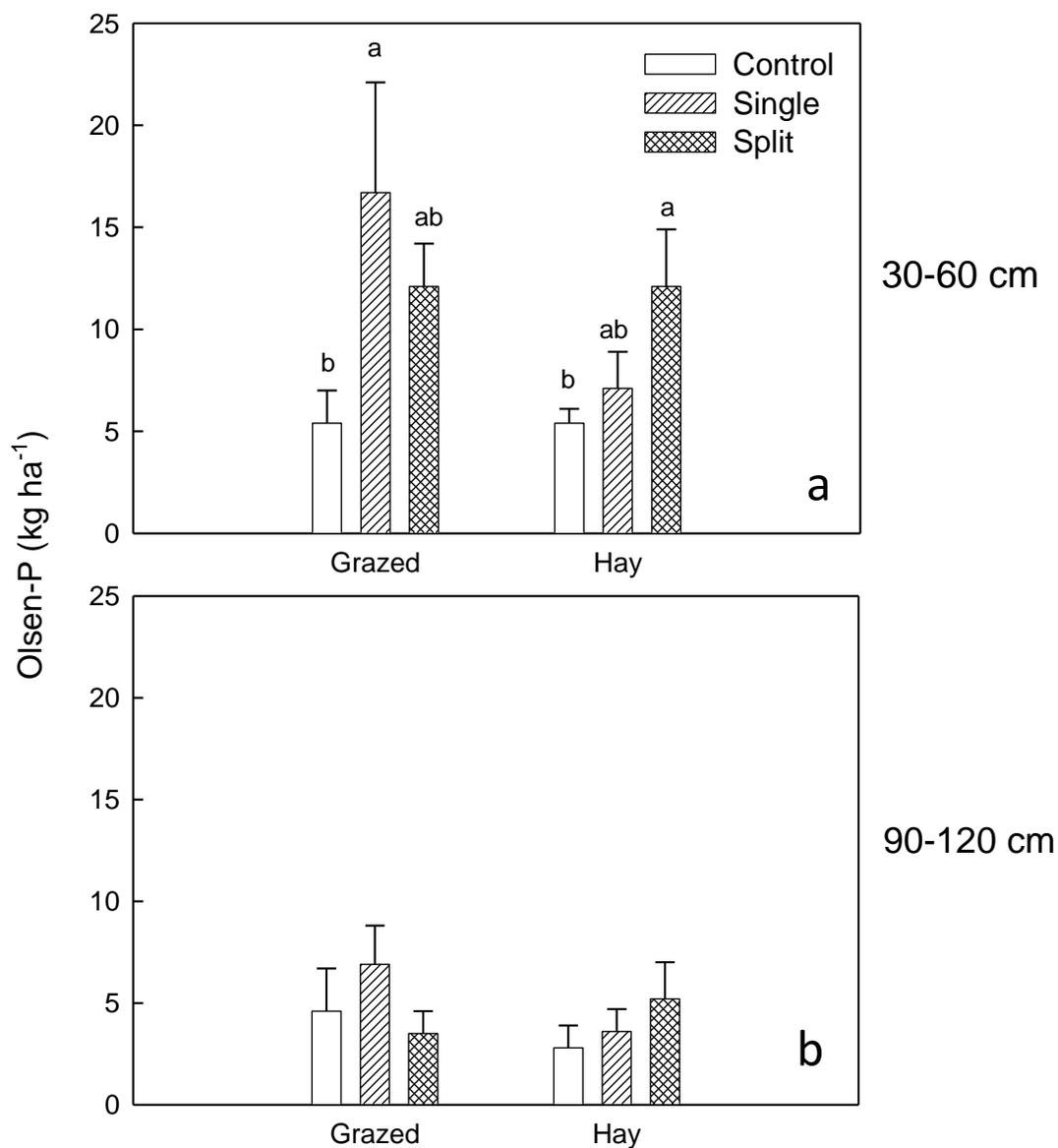


Figure 2.4 Mean extractable-P content + 1 standard error of the mean for manure x forage utilization treatment combinations in the 30-60 (a) and 90-120 (b) cm depths of soil. Means indicated by the same letter are not significantly different at $\alpha = 0.1$ with the Tukey-Kramer comparison test.

Other studies have also reported the surface layer of soil to increase in extractable-P with surface application of swine slurry to bermudagrass (Adeli et al., 2006), swine slurry and solid cattle manure to irrigated alfalfa and rainfed timothy (Olson and Papworth, 2006) and poultry litter to pasture and a cotton (*Gossypium hirsutum* L.) - corn (*Zea mays* L.) rotation (He et al., 2009 and Schomberg et al., 2009).

Phosphorus can move in the soil profile in coarse textured soils and with continuous addition of a large amount of surplus-P. In the coastal plains of North Carolina, Nelson et al. (2005) found that 21 years of swine slurry applications in surpluses of 30-50 kg ha⁻¹ year⁻¹ above removals to a bermudagrass-triticale (*x Triticosecale* Wittmack ex A. Camus) pasture increased water soluble, Mehlich-3, and oxalate extractable-P as well as total P to 75 cm. In Alberta, Whalen and Chang (2001) found irrigated barley (*Hordeum vulgare* L.) receiving solid cattle manure applied for 16 years at N requirement rates increased Olsen-P concentrations below the crop root zone (60 – 150 cm), with available P concentrations up to 176 and 48 mg P kg⁻¹ at 30-60 and 120-150 cm, respectively. In their experiment, total P applied with manure far exceeded the crop's requirements on average by 382 and 171 kg P ha⁻¹ year⁻¹ for irrigated and non-irrigated barley. Idowu et al. (2008) found that very large additions of P (200 kg P ha⁻¹) as swine slurry and beef cattle manure resulted in leaching of P in repacked columns. Thus, large surplus-P additions can favour mobility of P in soil. In the present study, surplus-P additions were constrained to a relatively small rate of 43 kg P ha⁻¹ year⁻¹ compared to studies having observed P movement in soil. In regard to general farm practice in Manitoba, the swine slurry used in this current study was sourced from the primary cell of a three-cell earthen lagoon storage system, which resulted in greater P addition rates to

the soil because of the higher solid content than that of slurry used by most producers. Therefore, current rate recommendations for application of swine slurry to meet forage N demand seems to not increase P and also inorganic N presence below the surface rooting depth for at least five consecutive years of slurry application compared to the unamended Control.

2.4.2.3 Implications of Chloride in the Soil Profile. The presence of chloride in the swine slurry allowed the opportunity to determine the potential for movement of nitrate below the root zone as a result of manure application. Chloride is subjected to leaching in soil as it is soluble in water. It is usually considered inert and not affected by ion exchange on mineral surfaces (Peters and Ratcliff, 1998; White and Broadley, 2001), and is taken up by crop plants in modest amounts (Cameron et al., 1996) due to selective exclusion (Pasternak, 1987). McKenzie (2001) summarizes chloride aboveground uptake for grass cereal crops to range between 2 to 9 kg ha⁻¹ and for alfalfa hay to be 6 kg ha⁻¹. In Quebec, Pelletier et al. (2007) found that chloride uptake by timothy was proportional to the amount applied with fertilizer at rates of 48 to 144 kg Cl ha⁻¹. In their study, hay exported 45 % of applied chloride in the above ground biomass. Heckman (2006) explained that chloride accumulates in aboveground green biomass with about half being leached following senescence. Generally, chloride is a commonly used tracer added to soil to follow percolation of water and leaching losses of soluble mobile compounds (Allison and Hughes, 1983; Cameron et al., 1996; Chang and Entz, 1996; White and Broadley, 2001). Across the study years, 63 and 75 kg ha⁻¹ year⁻¹ of soluble chloride in the slurry was added to Single and Split treatments, respectively (Table 2.3). The soluble chloride concentration of slurry is indicative of the total chloride content of manures (Dr.

Geza Racz, pers. comm.). Therefore, not unexpectedly, slurry application increased extractable chloride content in each of the four soil depth intervals sampled the fall of each year (Table 2.5). The Single and Split treatments had similar chloride contents for the four depth intervals with substantial amounts at the 90-120 cm depth interval (Table 2.5). Forage utilization did not affect chloride contents in soil, but contents across treatments for the 0-30 and 30-60 cm depths were greater in 2006-2008 than in 2004.

The greater chloride content in the soil profile of manure treatments was most likely caused by additions with the slurry. However, it cannot be ruled out that upward water flux from the saturated zone of soil driven by evapotranspiration (Schmidhalter et al., 1994) increased in manure treatments as a result of more plant biomass production, and caused deposition of chloride with mass flux in the soil. By observation, the majority of the root mass of the grasses was concentrated in the top 5 cm of soil with roots obvious to 35 cm, but roots extending to 90 cm were rarely observed. The water table during the growing season fluctuated between the soil surface to 2 m below the surface (data not presented). The extent that the few deep roots of the cool-season grass at the site resulted in concentrating chloride in the soil profile is unknown. The coarse texture of the soil would not favour capillary rise of chloride from the water table because of the dominance of large diameter pores in this soil, and the deep roots would have to be in contact with the groundwater surface to obtain water (Ayars et al., 2006). In any case, deposition of chloride by mass movement in upward water flux driven by evapotranspiration was not determined.

The presence of chloride in the soil profile as a result of slurry application indicates the potential for constituents in manure to move through the profile. That inorganic N and P were not found in appreciable amounts below the soil surface depth

analyzed suggests that forage uptake of N and P sorption by soil colloids prevented downward movement of the nutrients. For N, this could be the case as aboveground N uptake for the Hay treatment receiving slurry was $76 \text{ kg N ha}^{-1} \text{ year}^{-1}$ or 58 % of applied available and 40 % of total applied N. Chang et al. (1991) found similar distribution of nitrate and chloride in the soil profile of non-irrigated and irrigated plots, after eleven years of application of cattle feedlot manure to barley at one to three times the recommended rates based on N availability to crops. However, Cameron et al. (1996) reported that placement of dairy slurry affected nitrate movement. In their study using lysimeters receiving slurry at 305 kg N ha^{-1} to perennial ryegrass and white clover (*Trifolium repens* L.), nitrate was present in the soil profile with injection but not surface application, despite chloride presence in the profile with both treatments. They speculated that root absorption had a major role in limiting nitrate movement in the surface applied treatment. In contrast, crop uptake of P in the slurry was low relative to N, being $12 \text{ kg P ha}^{-1} \text{ year}^{-1}$ (22 % of total applied P). Phosphorus is prone to numerous reactions forming insoluble precipitates and also sorption to surfaces of soil minerals (Edwards and Withers, 1998). Thus, the forage probably had a minor role in prevention of P downward movement in the soil profile.

2.4.3 Phosphorus Accumulation and Relation to Surplus Phosphorus

2.4.3.1 Phosphorus Accumulation. Manure application resulted in an increase in extractable-P at shallow soil depths through the duration of the study, whereas concentrations did not change for the Control treatment (Fig. 2.5 and Table 2.6). Linear regression analysis indicated that the rate of P accumulation with manure treatments was about two-fold greater in the top 5 cm compared to the 5-30 cm depth being 14.2 and 4.7

mg P kg⁻¹ year⁻¹, respectively. Clearly, most of the added P was retained in the uppermost soil layer (0-5 cm) that was in direct contact with the applied slurry.

The manure and forage utilization treatments affected the rate of extractable-P accumulation (Table 2.6). The Single treatment caused significantly faster accumulation of extractable-P than the Split treatment by 5.4 and 1.2 mg kg⁻¹ year⁻¹ in the 0-5 and 5-30 cm depths, respectively. Furthermore, accumulation was significantly quicker with Grazed than Hay utilization by 5.4 and 1.5 mg kg⁻¹ year⁻¹ at 0-5 and 5-30 cm depths, respectively.

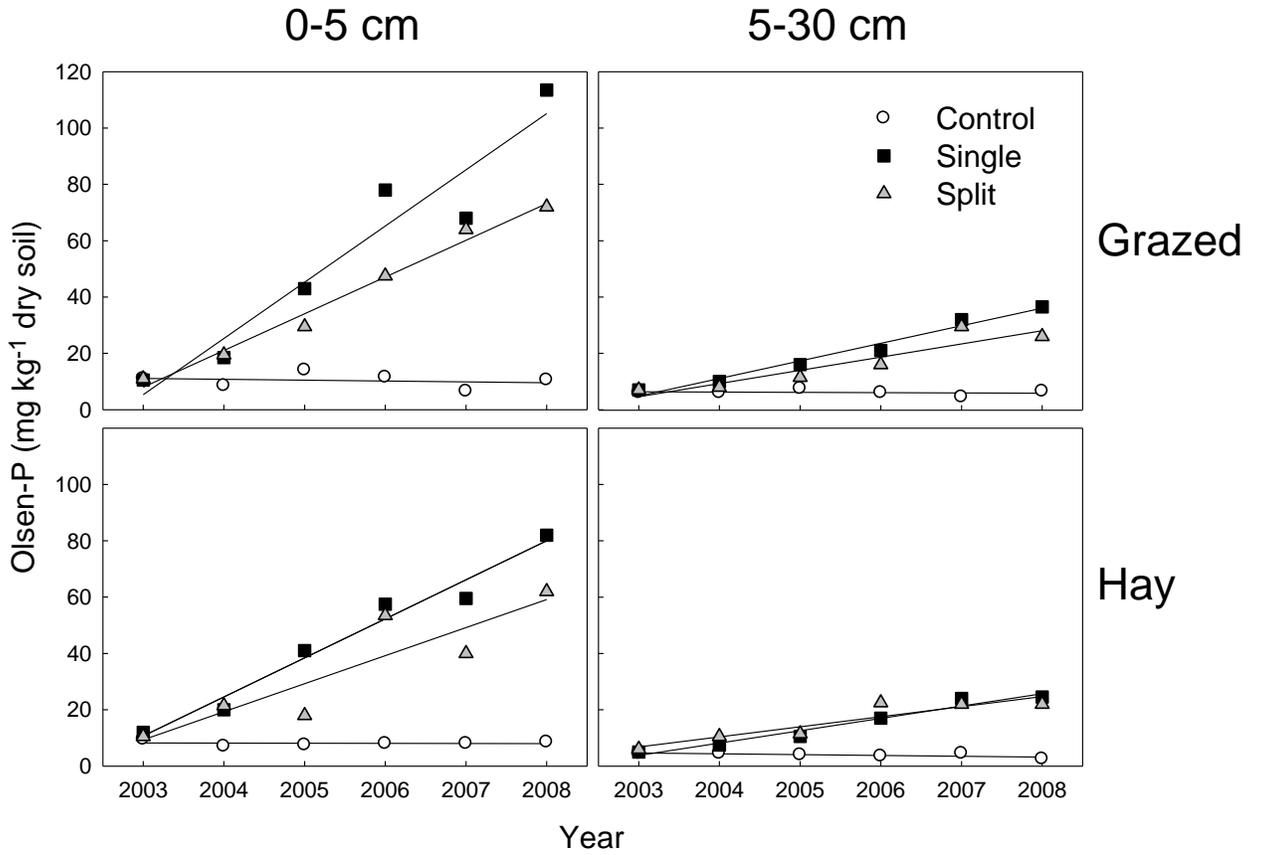


Figure 2.5 Extractable-P (Olsen-P) concentration in the surface soil from samples collected in fall of each year of the study. Treatments were combinations of manure (Control, Single and Split) and forage utilization (Grazed and Hay) treatments. Data points are averages of two replicate plots. Lines represent the regression equations given in Table 2.6

Table 2.6 Regression analysis of extractable-P over the study period and two-way analysis of variance of the slopes of the regression and of the ratio of surplus-P and slopes for the 0-5 and 5-30 cm depths of soil. Treatments were combinations of manure (Single and Split) and forage utilization (Grazed and Hay) treatments with two replicates.

Utilization	Manure	Extractable-P Increase						Surplus-P/Extractable-P	
		0-5 cm			5-30 cm			0-5 cm	5-30 cm
		mg kg ⁻¹ year ⁻¹	r ²	P > F	mg kg ⁻¹ year ⁻¹	r ²	P > F		
<i>Interaction Means</i>									
Grazed	Single	20.0	0.86	<0.001	6.2	0.81	<0.001	3.0	9.6
	Split	13.0	0.90	<0.001	4.7	0.68	0.001	3.7	9.6
Hay	Single	13.9	0.87	<0.001	4.4	0.67	0.001	3.8	11.6
	Split	10.0	0.72	0.001	3.6	0.64	0.003	3.7	11.6
<i>Main Effect Means</i>									
Grazed		16.5a	0.79	<0.001	5.5a	0.72	<0.001	3.4	9.6
Hay		11.9b	0.75	<0.001	4.0b	0.64	<0.001	3.8	11.6
	Single	16.9a	0.82	<0.001	5.3a	0.69	<0.001	3.4	10.6
	Split	11.5b	0.79	<0.001	4.1b	0.64	<0.001	3.7	10.6
Effect		ANOVA P > F							
Manure		0.03			0.05			ns	ns
Utilization		0.05			0.02			ns	ns
Utilization x Manure		ns			ns			ns	ns

For the regression analysis, n=2. Regression models for the Control treatments were found not significant at $\alpha = 0.10$ and were omitted. For the ANOVA analysis of the regression slopes, n=8. Significance at $\alpha = 0.10$. Means followed by different letters in a column for each main effect indicate a significant difference between levels of the effect at $\alpha = 0.10$ with the Tukey-Kramer comparison test.

Others have shown P to accumulate in the uppermost soil layer with surface application of manures. Franzluebbbers et al. (2002) found that four years of consecutive applications of broiler litter to bermudagrass at a surplus-P addition of 112 kg P ha⁻¹ resulted in accumulation rates being the most rapid closest to the soil surface.

Accumulation rates of Mehlich-1 P were 15, 6 and 5 mg kg⁻¹ year⁻¹ at 0-2, 2-4 and 4-6 cm, respectively. There was no change in extractable-P in the soil profile below 12 cm.

Accumulation rates of soil extractable-P appear to be linear over time when similar additions of manure P are consecutively applied. Numerous studies have modelled steady soil accumulation rates in extractable-P with time (year) from P additions from fertilizers (Wang et al., 2010; Selles et al., 1999; Regmi et al., 2002, Fan et al., 2005; Franzluebbbers et al., 2002; Hedtcke et al., 2010), poultry, dairy, cattle, barnyard, farmyard and horse manures (Huang et al., 2011; Sistani et al., 2008; Franzluebbbers et al., 2002; Regmi et al., 2002, Fan et al., 2005, Hedtcke et al., 2010, Duan et al., 2011) and swine slurry (Duan et al., 2011). Most of these studies sampled soil at 0-15 cm or deeper though Franzluebbbers et al. (2002) sampled in two cm depth intervals to 6 cm and still found accumulation rates to be linear.

Even considering differences in sampling depth or in P extraction methods, accumulation rates for extractable P were greater in the current experiment than in other field studies with surface application of manure (Huang et al., 2011; Selles et al., 1999; Fan et al., 2005; Hedtcke et al., 2010; Franzluebbbers et al., 2002; Regmi et al., 2002; Salmeron et al. 2010; Singh Antil and Singh, 2007; Lloveras et al., 2004) and with injection or incorporation of swine slurry (Qian et al., 2004; Hedtcke et al., 2010). These other studies used manure of a similar or lower N:P ratio compared to the one in the current experiment yet reported lower accumulation rates. The effect of surplus-P on

extractable-P needs to be taken into consideration to understand these differences. In contrast, other studies that used lower N:P ratio manures, such as cattle manure (Qian et al., 2004; Duan et al., 2011, Sistani et al., 2008), resulted in more rapid accumulation than for the current study.

As mentioned previously, the slurry applied in the current study was obtained from the primary cell of a three-cell earthen manure storage system of hogs fed a diet with phytase, and the mean N:P ratio of the applied slurry was 3.5 and 6.0 for the Single and Split treatments, respectively, (Appendix I.2). Because it was sourced from the primary cell of the storage where P accumulates, the swine slurry in the current experiment applied in spring had a lower N:P ratio than commonly used in Manitoba. Additionally, the N:P ratio of slurries in Manitoba has increased after the widespread adoption of phytase in feeds. Based on 92 samples of swine slurry used by farmers, mean N:P ratio for feeder barn manure was 3.4 prior to phytase introduction (Racz and Fitzgerald, 2001). After widespread phytase adoption, the average ratio based on 181 samples was 5.2 (MAFRI, 2007).

2.4.3.2 Relation of Surplus to Extractable Phosphorus. The ratio of surplus-P to rate of extractable-P increase (Equation [3]) was not affected by slurry or forage utilization treatments (Table 2.6). This indicates that P accumulation rates were simply a function of surplus-P associated with the treatments rather than caused by other factors. The greater concentration of P in the slurry applied in spring than in fall increased surplus-P and consequently, rate of extractable-P accumulation for the Single than Split treatment (Tables 2.3 and 2.4, Appendix I.2). Similarly, greater surplus-P for the Grazed treatment increased the rate of accumulation of extractable-P than with the Hay treatment.

Franzluebbers et al. (2002) also reported higher linear accumulation rates for high and low density grazed pastures compared to hay, from surplus-P of 120 and 104 kg ha⁻¹ for grazed and hay utilization treatments, respectively.

The amount of surplus-P required to increase extractable-P by 1 mg kg⁻¹ (Equation [3]) averaged across treatments was 3.56 and 10.61 kg ha⁻¹ year⁻¹ for the 0-5 and 5-30 cm depths, respectively. The response of extractable-P to surplus-P additions in the current study is greater than those reported by others for manure applications to coarse (Zhang et al., 1995; Shepherd and Withers, 1999; Richards et al., 1995), medium and fine texture soils (Singh et al., 2001; Toth et al., 2006; Zheng et al., 2004; Spratt and McIver, 1978). Stones were removed before P extraction and assumed to have negligible P sorption capability. Even if the stones were not excluded from the calculation, the ratio of surplus P to soil P would only increase from 3.15 to 4.27 kg ha⁻¹ year⁻¹ for the 0-5 cm and from 10.61 to 12.71 kg ha⁻¹ year⁻¹ for the 5-30 cm depth. An explanation for the greater rate of conversion of surplus P to Olsen P in this study compared to other studies is probably the relatively weak retention of P by Ca and Mg in calcareous soils, and consequently lower buffer capacity, compared to the stronger retention by Fe and Al that occurs in most other soils. .

2.5 Conclusions

Five years of swine slurry applications at N requirement rates to a coarse soil pasture did not increase extractable inorganic N and P below the main rooting zone to 1.2 m. Recovery of N applied in excess of utilization by the grass forage above ground tissue as extractable inorganic N was very low indicating either storage of N in soil roots, organic matter or loss to the environment. In contrast, a considerable portion of surplus-P

was recovered as extractable-P closest to the soil surface. However, extractable-P steadily increased with surplus-P additions and more so with Grazed than Hay forage utilization and Single than Split slurry treatment. The accumulation rates of extractable-P were increased by the lower P removal with Grazed and larger P addition rates with the Single treatments. Nitrogen-based swine slurry applications to grass forage seem appropriate to prevent movement of the N and P to groundwater but extractable-P accumulation in the near-surface soil over time will necessitate reducing amounts of surplus-P additions. Particularly, for grazed pastures, P-based application of slurry will likely be warranted sooner than for hay forage utilization. There seems to be no benefit to split applying slurry in spring and fall to reducing N loss and build-up of extractable-P in consideration of the extra management involved with two applications.

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**3. LABILE NITROGEN, PHOSPHORUS AND PHOSPHORUS FRACTIONS
IN A COARSE TEXTURED GRASSLAND SOIL GRAZED BY CATTLE
AND FERTILIZED WITH SWINE SLURRY**

3.1 Abstract

Repeated applications of swine slurry to pastures in excess of crop removals can increase concentration of labile nitrogen (N) and phosphorus (P) in soil, increasing risk of environmental losses. This study was conducted in South-Eastern Manitoba on a coarse textured soil to determine to what extent three years of slurry treatment to a grazed pasture affected N and P forms in soil. Four paddocks were subjected to factorial combinations of slurry application (Control-none, Single-spring application) and vegetation cover (Grassed-pasture forage, Bare-areas around waterers and salt supplements). Concentrations of extractable N and P in the soil profile were determined for six depths to 1.2 m. Soil P fractions were determined by sequential extraction for four depths to 0.6 m. Phosphorus sorption capacity and degree of phosphorus saturation were determined for the near surface soil (0-5 cm). Nitrate concentrations were greater throughout the soil profile in Bare compared to Grassed areas. In both Single and Bare treatments, significant increases in labile-P (water and bicarbonate extractable) compared to the Control occurred to 30 cm, and organic- and recalcitrant-P were unaffected. Phosphorus sorption capacity significantly decreased to 261 mg P kg⁻¹ in Bare from 367 mg P kg⁻¹ in Grassed areas. Degree of phosphorus saturation increased significantly in the Single Grassed and Bare treatments compared to the Control and Grassed treatments. Water soluble inorganic P at 0-5 cm increased significantly with degree of phosphorus saturation ($r^2 = 0.71$), and ranged from 22 to 165 mg H₂O-P_i kg⁻¹ in the Single Grassed and in the Bare treatments. The results

indicate that Bare areas are of concern for potential N and P mobility in the environment because of excreta depositions and lack of plant nutrient removal. Bare areas in the slurry amended pasture, which had better forage quality and higher stock densities because of slurry additions, had greater concentrations of labile N and P than Bare areas from Control paddocks. There is concern for movement of labile N and P to the water table in the Bare areas.

Abbreviations: ANOVA, analysis of variance; N, nitrogen; P, phosphorus; P_{150} , one point isotherm of 150 mg P L⁻¹; P_i , inorganic P; P_o , organic P; P_t , total P; S.E., standard error; S_{max} , sorption maximum.

3.2 Introduction

Pastures close to livestock production operations are often fertilized with animal manures. However, manure addition to pastures increases the risk of accumulation of nitrogen (N) and phosphorus (P) in soil (Cuttle, 2008; Hao et al., 2004) and the risk of N and P movement into ground and surface water. A consequence being nitrate (NO₃⁻) contamination of groundwater and P as the main contributor to eutrophication of rivers and lakes. It was found that 16 % of private wells exceeded 10 mg NO₃⁻N L⁻¹ in Manitoba (Manitoba Water Stewardship Report No. 2006-02, 2006). Moreover, Lake Winnipeg is the largest fresh water body in Manitoba and is undergoing eutrophication similar to that experienced by Lake Erie of the North American Great Lakes in the 1970s (Schindler et al., 2006).

Nitrogen and P can accumulate in grasslands when the application of the nutrients exceeds removals as hay, animal weight gain and losses as transfers to the environment

(Toth et al., 2006; Schomberg et al., 2009). Nutrient removals are usually very small in grazed pastures because most of the N and P ingested by livestock are returned as urine and feces (Williams and Haynes, 1990). This can result in greater accumulation of extractable P in soil with grazing compared to hay utilization of grassland (Coppi et al. – Chapter 2; Wilson et al., 2011).

Plant-available nutrient in soils are often determined through the use of salt solutions such as potassium chloride (inorganic N), sodium bicarbonate (Olsen-P) and mixtures with dilute acids (Bray-P, Mehlich-P).. These solutions are suitable to determine the accumulation of N in excess of plant requirements because of the eventual formation of the inorganic anion, nitrate (NO_3^-). However, P applied to soil as manures or fertilizers enters numerous fractions that are in varying degrees of equilibrium with the most readily extractable form of P in soil, orthophosphate dissolved in water (H_2PO_4^- and HPO_4^{2-}), that is water soluble and may be prone to loss in percolating or runoff water.

Swine (*Sus scrofa* L.) slurry contains 17-70% P in dissolved orthophosphate, dissolved organic, and weak base (Olsen), readily extractable P fractions, defined here as labile P (Ajiboye et al., 2004; Kumaragamage et al., 2009). Not surprisingly, swine slurry can increase soluble-P in soil (Hountin et al., 2000; Kashem et al., 2004) resulting in increased plant growth. However, a portion of the labile P added with manures becomes adsorbed to soil surfaces as extractable-P (Ajiboye et al., 2008) with the total amount of P possibly held in this way referred to as the sorption capacity of soil (Brock et al., 2007). A portion of P added with manure also forms minerals of Ca, Mg, Fe and Al of varying solubility (Ajiboye et al., 2008) or remains as or is assimilated by organisms to become organic P (Koopmans et al., 2007). There is a lack in understanding the effect of successive

swine slurry applications to grazed pastures on the amounts of soluble and weak base extractable P fractions in soil, as well as on the organic and more recalcitrant fractions of P.

When P accumulates above the sorption capacity of a soil it is increasingly present as water soluble P (Dou et al., 2009). A soil's ability to retain P is related to its degree of P saturation, defined as the ratio of extractable P to P sorption capacity of soil (Beauchemin and Simard, 1999). Phosphorus sorption capacity and degree of P saturation are highly correlated with loss of water soluble P with leaching or over-land flow (Leinweber et al. 1999). Particularly, the degree of P saturation often shows a two-phase relation to water soluble P concentration in which the relation is weak before a "change point" where concentrations dramatically increase linearly with the degree of P saturation (Kleinman et al., 2000; Nelson et al., 2005). In coarse textured soils, values for the degree of P saturation above a "change point" have been related to leaching of P (Nelson et al. 2005). In a field study in Manitoba, Coppi et al. (Chapter 2) showed that grazed pastures accumulated more Olsen extractable P with five successive applications of swine slurry compared to hay utilization of forage grass. That rapid increase in Olsen extractable-P in the near-surface layer lead to the question as to whether the soil's capacity to retain P and prevent it from being released to the soil solution was compromised.

Nitrogen and P can accumulate in pastures where livestock concentrate near water troughs, mineral feeders, and shelters (Williams and Haynes, 1990). The tendency to of animals to congregate in these areas results in a higher density of deposition of feces and urine than for other pasture areas (Haynes and Williams, 1999; Sigua and Coleman, 2006). These areas can often become bare of vegetation because of trampling, soil compaction and high nutrient and salt concentrations (Cluzeau et al., 1992; Turner, 1998). These areas tend to enlarge with each deployment of animals to pasture and nutrient concentrations in soil

rise because of further feces and urine deposition and lack of removal by forages (Dahlin et al., 2005). Whether the concentrations of N and P in these animal congregation areas cause environmental losses of the nutrients in pastures is not known. Moreover, very little is known about N and P accumulation in the soil profile, the degree of P saturation and the P fractions following depositions in these areas.

Thus, a study was conducted to determine the effect of swine slurry applications on P saturation and fractions and extractable-N in the soil profile of a coarse textured grassland soil. Paddocks of grazed grassland at the La Broquerie Pasture and Swine Manure Management Study Site in south-eastern Manitoba were selected to determine the effect of three consecutive years of swine slurry application to grass compared to bare areas impacted by animal congregation on: a) extractable-P and -N concentrations and soil chemical properties to 120 cm; b) P sorption capacity and degree of P saturation of near-surface soil and c) concentration of P fractions to 60 cm depth.

3.3 Materials and Methods

3.3.1 Site Location and Description

The research site was the La Broquerie Pasture and Swine Manure Management Study Site located in the Rural Municipality of La Broquerie in South Eastern Manitoba, Canada. Twelve paddocks were established in 2003 on 40 ha of tame pasture that had not recently received fertilizer or manure additions. A description of the site was given in section 2.3.1 (Coppi et al.).

Four paddocks were used in this study. The soil is mapped as Berlo series (70%), a lacustrine loamy fine sand and imperfectly drained soil (FAO Gleyed Luvisol, Canadian system Gleyed Dark Gray Luvisol) and the remaining 30% of area to the Kergwenan series,

an outwash loamy sand to gravel and also imperfectly drained soil (FAO Greyzem, Canadian system Glayed Dark Gray Chernozem) (Hopkins, 1985). Based on the Canadian classification, the stone-free fraction (< 1.5 mm in diameter) of the paddocks was loamy sand in the top 60 cm, and sand in the 60-120 cm depths (Table 3.1). When considering all particles, the soil was very gravelly to gravelly as % stones averaged 37 % of total soil weight (Table 3.1). Soil bulk density determined using the excavation method (section 2.3.1; Coppi et al.) averaged 1.68 Mg m^{-3} in the top 30 cm and 1.98 Mg m^{-3} for the 30-120 cm depth interval. All results for soil properties except bulk density are reported for the stone-free fraction of soil. Soil pH in 1:2 (soil:water) solutions was 7.9 in the top 30 cm and increased with depth to 8.6 at 90-120 cm. The soil was weakly to moderately calcareous, with carbonate content increasing with depth being 21 g CaCO_3 equivalent kg^{-1} at 0-30 to 82 at 90-120 cm. Cation exchange capacity also increased slightly with depth from 15.7 to 17.1 $\text{mol}_c \text{ kg}^{-1}$ at 0-30 and 90-120 cm depths, respectively. Generally, N and P concentrations in the surface soil were low prior to addition of hog slurry in 2004 with total soil N being 0.8 g kg^{-1} , extractable ammonium-N of 1.9 mg kg^{-1} , nitrate-N of 1.6 mg kg^{-1} , and sodium bicarbonate extractable-P (Olsen-P) of 5.5 mg kg^{-1} in the top 30 cm. Organic carbon concentration decreased with depth from 10.2 to 5.2 g kg^{-1} for the 0-30 and 90-120 cm depths, respectively (Table 3.1). An impermeable layer of clay was present at 2 m depth in the western half of the site grading deeper in the eastern half (> 5 m), and a shallow water table, particularly on the western half (< 2 m), rose to the surface occasionally at springmelt and after heavy rains (Tenuta et al., 2010).

Table 3.1 Mean and one standard error (S.E.) for the treatment paddocks (n = 4) of soil chemical and physical properties at the study site, determined in 2003 before the start of the trial.

Soil Depth		pH†	EC†	CEC	Organic carbon	CaCO ₃ eq	Total-N	NH ₄ ⁺ -N	NO ₃ ⁻ -N	Olsen -P	Cl ⁻	K ⁺	Sand	Silt	Clay	Texture‡	Stones	Bulk density§
cm			dS m ⁻¹	mol _c kg ⁻¹	----- g kg ⁻¹ -----	-----	-----	-----	-----	mg kg ⁻¹	-----		% weight of fine soil¶			Class	% weight	Mg m ⁻³
0-30	Mean	7.9	0.1	15.7	10.2	20.5	0.8	1.9	1.6	5.5	0.9	45.0	83.5	9.3	7.3	Gravelly Loamy Sand	29.6	1.68
	S.E.	0.1	0.0	1.7	0.7	6.5	0.1	0.4	0.3	1.7	0.2	4.1	1.9	1.5	0.5		3.8	0.11
30-60	Mean	8.3	0.1	15.2	7.8	47.8	0.5	1.0	1.3	1.3	2.0	22.0	83.0	14.0	3.0	Very Gravelly Loamy Sand	41.0	2.04
	S.E.	0.0	0.0	1.8	1.0	7.3	0.2	0.2	0.2	0.7	0.3	1.6	7.4	7.4	0.4		4.4	0.12
60-90	Mean	8.4	0.1	16.0	4.9	72.0	0.3	0.5	0.7	0.3	2.9	16.3	92.8	5.8	1.5	Very Gravelly Sand	39.4	1.94
	S.E.	0.0	0.0	1.5	1.9	9.1	0.2	0.1	0.0	0.3	0.4	1.1	0.3	0.3	0.3		3.2	0.09
90-120	Mean	8.6	0.1	17.1	5.2	82.3	0.2	0.3	0.6	0.3	3.3	14.0	92.8	5.5	1.8	Very Gravelly Sand	37.6	1.96
	S.E.	0.1	0.0	0.7	1.2	9.9	0.1	0.0	0.0	0.3	0.4	0.7	0.6	0.6	0.3		3.8	0.09

†1:2 soil:water ratio.

‡Textural classes according to Canadian classification.

§Bulk density was determined with the excavation method from four soil pits dug in summer 2007 in paddocks not included in the study. Values reported here are weighted averages of bulk density of two replicates of each pedogenetic horizon and four pits.

¶ Fine soil is soil that passed through a 1.5 mm mesh.

3.3.2 Experimental Design

The experimental design of the study was described in detail in section 2.3.2 (Coppi et al.), with only a brief description given here. The site was divided into 12 paddocks. For the current study, four paddocks were sampled in spring 2007. Two paddocks had three consecutive annual swine slurry applications that started in the spring of 2004 (Single) and two received no slurry (Control). The paddocks had tame forage grasses grazed by cattle (*Bos primigenius Taurus* Bojanus) from June to August starting in 2004. The Control paddocks were 8 ha and the Single paddocks 4 ha. The size of paddocks varied according to the carrying capacity of the pasture with the goal of maintaining eight to ten animals per paddock and the standing forage at 1.0-1.5 Mg dry matter ha⁻¹ throughout the growing season. When the standing forage was < 400 kg ha⁻¹, some of the animals were removed. Steers were provided with fresh water and mineral supplements *ad libitum* with Rancher's Choice 8:4:50 Interlake Beef Pasture Premix (Puratone Feeds, Niverville, MB) containing 8% calcium and 4 % phosphorus.

The experimental design was a generalized complete block and split plot with the main plot factor being manure treatment (Control and Single). The subplot factor was vegetation cover with treatments of forage grass area (Grassed) and a smaller area around water troughs and mineral feeders with no vegetation (Bare). The Bare areas did not receive slurry application. A Bare area in each paddock was caused by visitation and loafing of steers around a water trough and mineral supplement feeder. The blocking factor was depth to the clay layer, with one replicate on the western side of the study site with shallow depth to the clay layer and the other on the eastern side having a greater depth to the clay layer.

3.3.3 Slurry Application

A detailed description of the slurry application methodology was given in section 2.3.3 (Coppi et al.). Briefly here, swine slurry obtained from the primary cell of a three-cell earthen manure storage was surface applied by tanker with a single splash-plate in early May of each year starting in 2004. Applications averaged 44,000 L slurry ha⁻¹ year⁻¹ which provided 142 kg plant available-N ha⁻¹ year⁻¹ assuming 25% volatilization of ammonium-N and 25% of organic-N of the slurry available to the grass in the year of application (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006). Total P applied in the slurry averaged 65 kg P ha⁻¹ year⁻¹ for the period of this study (Wilson et al., 2011).

The average annual surplus of total N and P added with the slurry (additions minus removals with animal liveweight gain) for the Single treatment was 185 kg N ha⁻¹ and 58 kg P ha⁻¹ (Wilson et al., 2011). In contrast, the unamended Control treatment had an annual net deficit of 7 kg N ha⁻¹ and 2 kg P ha⁻¹.

3.3.4 Soil Sampling and Analysis

Soil samples were collected in late April to early May 2007. Ten and one sample locations in the Grassed and Bare area per paddock, respectively, were randomly chosen using arcGIS (ESRI, Redland, CA). Sample locations were at least 7 m from the perimeter of paddocks. If present in the sampling location, dung pats were removed from the soil surface prior to sampling. Each location was excavated and sampled by hand to avoid contamination of soil depths. First, a 1x1 m pit was dug and samples were collected using trowels from pit walls at 0-5, 5-15 and 15-30 cm depth intervals. Then, a sample at 30-60 cm was obtained from the centre of the pit floor using a 3.8 cm wide soil auger.

Finally, a hydraulic flight auger (Giddings Machine Company Inc., Windsor, CO) was inserted carefully into the previous auger hole to collect samples in the 60-90 and 90-120 cm increments. Samples were transported on ice from the field and stored at -20 °C. They were then air-dried for three days at 30 °C, and ground and passed through a 1.5 mm mesh screen using an in-house rolling mill. The stone fraction (> 1.5 mm) was weighed and discarded. Gravimetric moisture of dried samples was determined by oven drying at 105 °C for 24 hours. Electrical conductivity and pH were determined on supernatants of 1:2 (soil:water) mixtures using a CDM2e Conductivity Meter (Radiometer Medical ApS, Bronshoj, Denmark) and a Benchtop pH/ISE Meter (Orion Research Inc., Beverly, MA), respectively.

Olsen extractable P concentration of samples was determined by shaking soil for 30 minutes with 0.5 M NaHCO₃ at pH of 8.5 and a 1:20 soil:extractant ratio (Olsen and Sommers, 1982). The extracts were filtered through no.42 Whatman filters (2.5 µm pore size) and analyzed colorimetrically (Murphy and Riley, 1962) for orthophosphate using a UV/Visible spectrophotometer at a wavelength of 820 nm (Ultrospec 2100 pro, Biochrom Ltd., Cambridge, UK). Nitrate and ammonium concentration of soil was determined by shaking soil slurries of 5 g dry soil and 25 mL 0.5 M K₂SO₄ for 60 minutes and filtering supernatants (Whatman no. 42) with analysis done by colorimetry (Keeney and Nelson, 1982) on a Technicon II Autoanalyzer (Pulse Instruments, Saskatoon, SK).

3.3.5 Phosphorus Saturation

All samples of the 0-5 depth interval were analyzed for their Degree of P Saturation (Ige et al., 2005; Brock et al., 2007) according to equation [1]:

$$\% \text{ Degree of P saturation} = (\text{extractable-P} / \text{P sorption capacity}) * 100 \quad [1]$$

where P sorption capacity (mg kg^{-1} dry soil) was determined for each soil sample as given in equation [2]:

$$\text{Phosphorus sorption capacity} = \text{extractable-P} + S_{max} \quad [2]$$

where S_{max} is the Langmuir sorption maximum (mg kg^{-1} dry soil). We estimated S_{max} using the equation [3] (Ige et al., 2008):

$$S_{max} = 1.7 * P_{150} \quad [3]$$

where the one-point isotherm P_{150} (mg kg^{-1} dry soil) was determined using the method of Bache and Williams (1971) by equilibrating 2 g of dry screened soil (< 1.5 mm) with 20 mL of a 150 mg $\text{K}_2\text{HPO}_4\text{-P L}^{-1}$ and 0.01 M KCl solution for 24 hours on an end-to-end shaker at room temperature. The suspension was centrifuged at 7,000 x g for ten minutes and filtered through a 0.45 μm cellulosic membrane (Micronsep, GE Water and Process Technology, Oakville, ON). The concentration of P in solution was determined by colorimetry (Murphy and Riley, 1962). The difference between the original P content of the solution (150 mg P L^{-1}) and the P content of the solution after equilibration was the amount of P adsorbed to soil (P_{150}). The coefficient of 1.7 was fitted from a pedotransfer function developed by Ige et al. (2005), and modified by Ige et al. (2008), that correlates P_{150} with S_{max} Langmuir isotherm determinations of 40 soils from the surface layer of agricultural fields across Manitoba.

3.3.6 Phosphorus Sequential Extraction

Five locations for the Grassed and the one for the Bare area per paddock at depth intervals of 0-5, 5-15, 15-30 and 30-60 cm were subjected to a P sequential extraction procedure of Hedley et al. (1982) with modification by Kashem et al. (2004). Air dried soil samples (0.5 g) were placed into 50 mL screw-cap centrifuge tubes containing 30 mL of increasingly stronger extractant solutions and shaken for 16 hours at 120 excursions per minute using an end to end shaker. The sequence of extractants was deionized water, 0.5 M NaHCO₃ at pH 8.5, 0.1 M NaOH and finally, 1 M HCl. Each solution targets removal of different fractions of P in soil; deionized water removes those P forms readily dissolved in water, 0.5 M NaHCO₃ is thought to remove forms of P adsorbed on the surface of soil particles and readily mineralizable organic P, while the 0.1 M NaOH solution is thought to remove Al³⁺ and Fe²⁺ bound P, and P bound in more stable organic compounds. The 1 M HCl solution removes Ca²⁺ and Mg²⁺ bound phosphate (Tiessen and Moir, 1993). The soil residue remaining after this last extraction was then digested in 4.4 mL of a stronger acid, H₂O₂-H₂SO₄ (1:1.2) mixture in 50 mL Kjeldhal tubes at 350 °C for 3 hours (Akinremi et al., 2003) to obtain residual-P fractions. It must be noted that the sequential extraction of P is a chemical procedure that cannot define the exact minerals and forms of P in soil, but it is used to classify P pools by their lability in sequentially stronger extracts rather than by their specific chemical form (Tiessen and Moir, 1993).

Water, NaHCO₃ and NaOH extracts were centrifuged at 9000 *x g* for ten minutes, and then passed through a 0.45 µm cellulosic membrane. A 5 mL portion was analyzed immediately for inorganic P (P_i; Murphy and Riley, 1962). A 20 mL portion of all extracts was digested using the H₂O₂-H₂SO₄ mixture for determining total P concentration of digests (P_t) by colorimetry after neutralization with drop-wise additions of dilute NaOH solution.

Organic P (P_o , mg kg^{-1} dry soil) of the water, NaHCO_3 and NaOH extractable fractions was determined using equation [4]:

$$\text{Organic P } (P_o) = \text{total-P of digest } (P_t) - \text{P in extract } (P_i) \quad [4]$$

The organic P component of the HCl extraction is commonly considered to be very small as dilute acid is inefficient at extracting organic carbon in soil and most organic P should be extracted in previous steps (Tiessen and Moir, 1993).

Total soil P for each sample was estimated as the sum of contents of the $\text{H}_2\text{O}_2:\text{H}_2\text{SO}_4$ digests of extracts (P_t). Inefficiencies in recovering soil residues left on membranes during vacuum filtration and in transferring the soil sample from screw-cap to Kjeldhal tubes for the determination of residual-P are possible. Thus, a check of the recovery efficiency of P for the sequential extractions was obtained from determinations of total soil P from a “one-step” digestion of 0.2 g of dried soil in the 4.4 mL $\text{H}_2\text{O}_2\text{-H}_2\text{SO}_4$ mixture.

The P concentration of the digests of all the extracts was increased by 7.1% because of imperfect P recovery of the acid digestion method for determination of P_t in extracts (data not shown), since even a small volume loss of the digest during transfer from the Kjeldhal tubes to the screw-cap tubes before colorimetric analysis affect the concentration of P in the digests.

3.3.7 Statistical Analysis

Statistical analysis was performed with the Statistical Analysis Software program (SAS 9.2, SAS Institute Inc., Cary, NC). A randomized complete block with split plot

design was used in a three-way ANOVA. The Mixed procedure was used with manure treatment (Control and Single) as the main plot effect and vegetation cover, Grassed and Bare, as the split plot effect. The third factor was Depth. Phosphorus sorption capacity and degree of phosphorus saturation of the 0-5 cm depth were analyzed using a two-way split plot ANOVA, with manure as the main plot and vegetation as the subplot effect. For all the datasets, a blocking factor for the east/west replicates was considered a random effect.

Unless otherwise specified, effects are significant at $P < 0.10$, with effects at $P > 0.05$ considered to be weak. When relevant, comparison of means was done at $\alpha = 0.1$ using the Tukey-Kramer mean comparison test with adjustment for β -type error. When required because of non-normality (Shapiro-Wilk statistic < 0.90), data were transformed prior to ANOVA with the transformation indicated in the tables where the results are presented. Possible heteroschedasticity of residuals was dealt with using the Kenward-Roger method for calculation of denominator degrees of freedom in the F test and the Restricted Maximum Likelihood as model type (SAS Institute Inc., 2009). A spatial power covariance structure was used among sampling depths (Giesbrecht and Gumpertz, 2004).

3.4 Results and Discussion

3.4.1 Olsen Extractable Phosphorus and Nitrogen in the Soil Profile

3.4.1.1 Extractable Phosphorus. Olsen extractable P varied with vegetation cover ($P < 0.0001$), soil depth ($P < 0.0001$) and manure treatment ($P = 0.001$; Table 3.2). Extractable P concentration across depths and manure treatments averaged 28.3 and 11.4 mg P kg⁻¹ for Bare and Grassed areas, respectively. Slurry application increased Olsen extractable P

to 30.8 mg P kg⁻¹ compared to 9.0 for the Control, across depths and vegetation covers (Table 3.2). However, a two-way interaction for manure x depth ($P = 0.02$; Fig. 3.1) indicated the effect of slurry application to be confined to the top 30 cm of the soil profile. The vertical stratification of extractable P was also more pronounced with the Single treatment where concentrations were 2.5, 3 and 2-fold greater than for the Control at the 0-5, 5-15, and 15-30 cm depths, respectively. The difference in concentration of extractable P for Bare and Grassed areas was greatest for the surface (0-5 cm), being 57.9 mg P kg⁻¹ compared to 1.5 mg P kg⁻¹ on average at 60-120 cm (Fig. 3.2a and 3.2d).

Table 3.2 Mean values of soil NaHCO₃ extractable P (Olsen-P), ammonium- and nitrate-N after three years of annual swine manure applications for manure and vegetation treatments and sampling depth. Shown are results of analysis of variance for main treatment and interaction effects.

Treatment	Olsen-P	NH ₄ ⁺ -N	NO ₃ ⁻ -N
	—mg kg ⁻¹ dry soil —		
<i>Manure</i>			
Control	9.0b	1.0	1.6
Single	30.8a	1.8	3.6
<i>Vegetation</i>			
Grassed	11.4b	1.9a	0.6b
Bare	28.3a	0.9b	4.6a
<i>Depth, cm</i>			
0-5	73.8a	4.9a	5.1a
5-15	25.8b	1.4b	4.0b
15-30	13.7c	0.9b	1.7bc
30-60	2.1d	0.5c	0.9c
60-90	2.1d	0.3c	1.7bc
90-120	1.8d	0.2c	2.2bc
ANOVA†		P > F	
Manure	0.001	ns	ns
Vegetation	<0.0001	0.001	0.001
Depth	<0.0001	<0.0001	0.001
Manure x Vegetation	ns	ns	ns
Manure x Depth	0.02	0.02	0.07
Vegetation x Depth	ns	<0.0001	ns
Manure x Vegetation x Depth	ns	0.07	0.06

†ANOVA was performed on log-transformed data: log_n (x + 1). Non transformed arithmetic means are shown in the table.

N = 264. Main effects are means of two experimental units (paddocks) each split in grass (ten subsamples) and bare earth (one subsample). Each subsample location was sampled in six depths intervals. Model was a randomized complete block split plot with covariance among depths. Effects are significant at P < 0.1. Separately for each effect, means followed by different letters are significantly different at α = 0.1 and Tukey-Kramer adjustment for beta error.

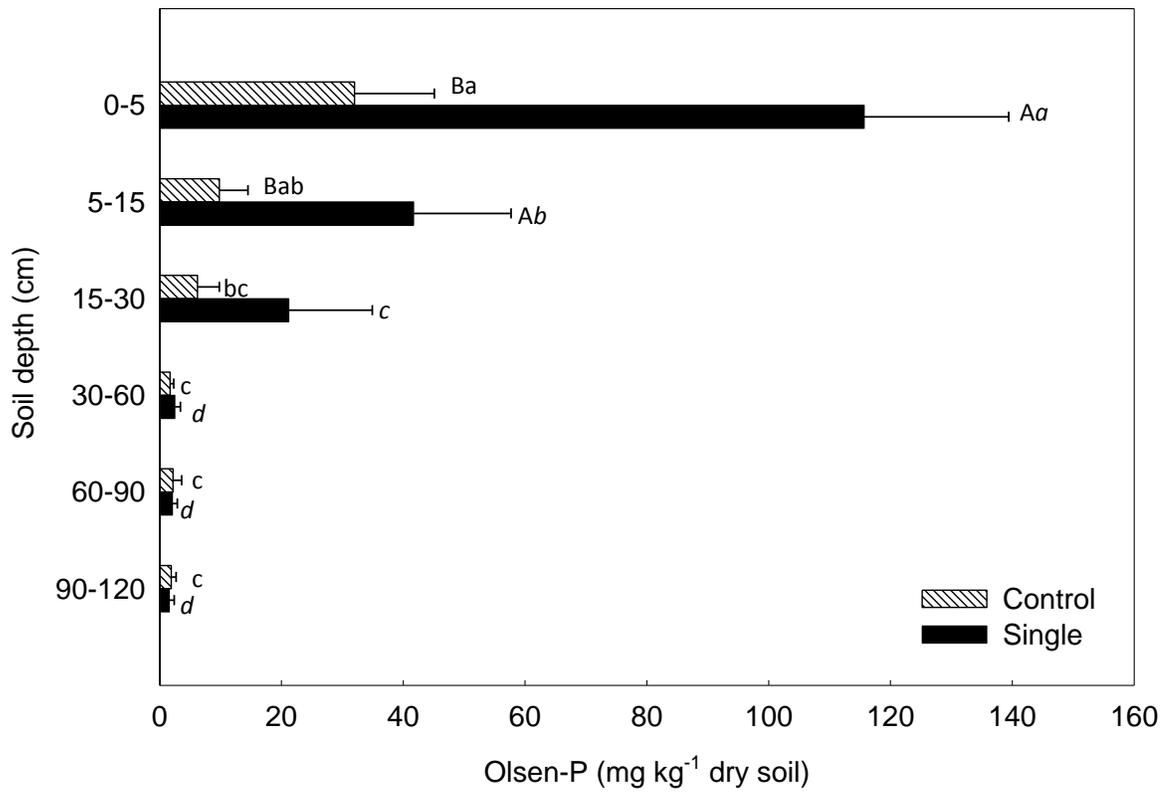


Figure 3.1 Mean concentration of Olsen extractable-P + one standard error of the mean for manure x soil depth treatment combinations (n = 4). Means followed by the same capital letter for each depth are not significantly different. Means followed by the same lower case letter for each manure treatment are not significantly different

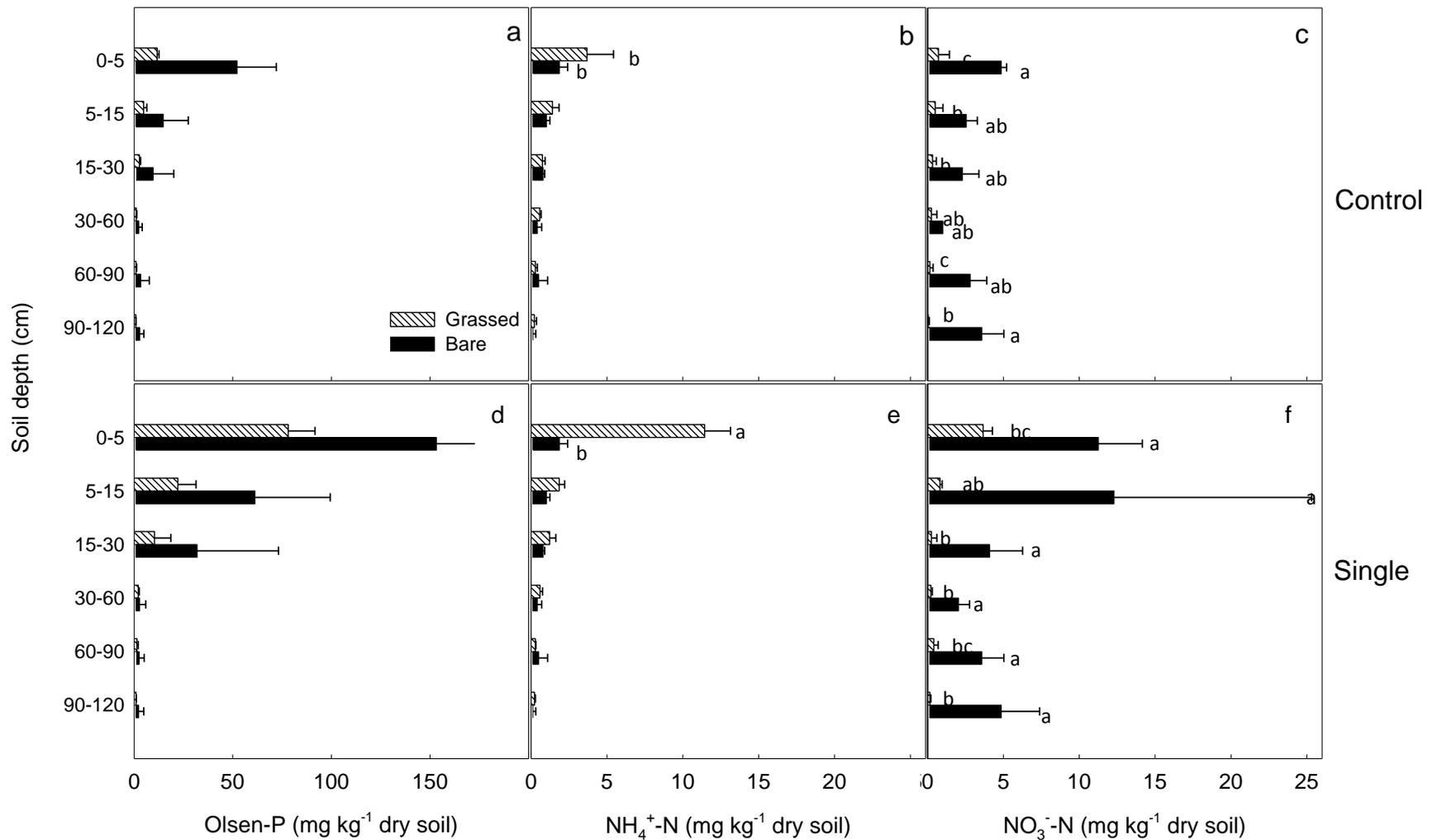


Figure 3.2 Mean concentration of (a) Olsen-P, (b) ammonium-N and (c) nitrate-N + 1 standard deviation for manure (Control and Single) x vegetation (Grassed and Bare) x soil depth treatment combinations (n = 2). Means followed by the same letter for each depth are not significantly different. For clarity, difference of means among depths is not shown.

The greater concentration of Olsen extractable P at all soil depths in the Bare compared to Grassed areas was likely a result of feces deposition around water troughs and mineral supplements (Cuttle, 2008; Williams and Haynes, 1990). Similarly, Mathews et al. (1994) found that extractable P concentration increased in cattle lounging areas compared to the rest of an unfertilized bermudagrass (*Cynodon dactylon* L.) pasture. Sanjari et al (2008) also reported an increase in extractable P concentration in the soil surface (0-10 cm) along a 500 m transect in a pasture of couch grass (*Agropyron repense* L.) toward stocking camps of sheep.

Not unexpectedly, extractable P was stratified under Grassed and Bare areas decreasing sharply over the 0-5, 5-15 and 15-30 cm depth, and reaching 5.9 fold lower concentrations at 60-120 cm. Coppi et al. (Ch. 2) showed stratification of Olsen extractable P for Grassed locations at the site but in this current study, stratification was also demonstrated for Bare areas. The stratification was not surprising as P is not highly mobile in soil (Holanda et al., 1998) and P addition in Bare areas was to the surface by way of feces deposition.

3.4.1.2 Ammonium and Nitrate. Extractable ammonium concentration varied with depth ($P < 0.0001$) and vegetation cover ($P = 0.001$) (Table 3.2). Ammonium concentration was greatest in the surface 5 cm of soil and for the Single Grassed treatment (11.5 mg $\text{NH}_4^+\text{-N}$ kg^{-1} , Fig. 3.2b and 3.2e) resulting in a weak manure x vegetation x depth interaction ($P = 0.07$).

Ammonium was stratified in the soil profile, with a concentration of 4.9 mg $\text{NH}_4\text{-N}$ kg^{-1} in the surface 5 cm decreasing to 1.4, 0.9 and 0.3 mg $\text{NH}_4\text{-N}$ kg^{-1} in the 5-15, 15-30 and 60-120 cm layers, respectively. Ammonium concentration was significantly, but slightly,

greater in the Grassed ($1.9 \text{ mg NH}_4^+\text{-N kg}^{-1}$) than Bare ($0.9 \text{ mg NH}_4^+\text{-N kg}^{-1}$) vegetation covers (Table 3.2). The difference in concentration between vegetation covers was most pronounced in the top 5 cm of soil averaging 7.6 (Grassed) and 2.3 (Bare) $\text{mg NH}_4^+\text{-N kg}^{-1}$. Further, stratification of extractable ammonium was more pronounced under Grassed than Bare vegetation (Fig. 3.2b and 3.2e). Together, these differences between vegetation covers with depth resulted in a very strong interaction effect for vegetation x depth ($P < 0.0001$).

An increase in extractable ammonium concentration with manure application was evident in the surface 5 cm of soil, averaging 7.1 and $2.8 \text{ mg NH}_4^+\text{-N kg}^{-1}$ in Single and Control treatments, respectively, across vegetation covers. The lack of increase in concentrations below this depth resulted in a manure x depth ($P = 0.02$) interaction effect.

Extractable nitrate concentration was affected by depth ($P = 0.001$), vegetation cover ($P = 0.001$) and a three way interaction between depth, vegetation and manure ($P = 0.06$) (Table 3.2 and Fig. 3.2c and 3.2f). Nitrate concentration varied with depth but not with as much stratification as for ammonium (Table 3.2). Further, unlike the situation for extractable ammonium, the concentration of nitrate was greater in the Bare ($4.6 \text{ mg NO}_3\text{-N kg}^{-1}$) than Grassed ($0.6 \text{ mg NO}_3\text{-N kg}^{-1}$) areas (Table 3.2). The concentration of nitrate tended to decrease from the surface to the 30-60 cm depth and then, below 60 cm, nitrate tended to increase for the Bare areas, especially with Single treatment, resulting in a weak manure x vegetation x depth interaction effect ($P = 0.06$; Fig. 3.2c and 3.2f).

Overall, the concentration of extractable ammonium was significantly but slightly greater in the surface 5 cm of soil of paddocks receiving slurry under Grassed than Bare vegetation. In contrast, nitrate concentration was greater under Bare than Grassed vegetation cover and was by far the prevalent of the two forms of extractable N in soil.

This was likely a result of the lack of plant uptake of inorganic N allowing accumulation of nitrate from nitrification in Bare areas. Similarly, Sanjari et al (2008) reported an increase in nitrate concentration in surface soil nearer a sheep stocking camp. The lack of stratification in the profile for nitrate concentrations is likely because, unlike ammonium, nitrate is mobile in soil water and thus prone to leaching (Eriksen, 2011).

3.4.1.3 Bare Area Nitrogen and Phosphorus. Bare areas in manured paddocks did not receive slurry. Thus, the source of N input to the Bare areas was deposition of feces and urine from steers. Phosphorus input to the Bare areas likely was feces and waste from the mineral supplement containing 4 % P provided *ad libitum* in the centre of a car tire placed 5 to 10 m from an above-ground tub waterer.

Williams and Haynes (1990) explain that most of the N and P ingested with forage by grazing livestock are not utilized (60-99%) and excreted with dung and urine. Donohoe (2011) in cows fed with low quality forage in Manitoba, found that 42 % of N was excreted in urine, and 58 % in the dung. She found that 30 % of N in urine was urea, and 66 % was other organic forms. An urination event can provide highly localized rates of 500 to 1000 kg N ha⁻¹, even though 20-60 % of urea can be lost through ammonia volatilization after hydrolysis to ammonium (Williams and Haynes, 1990; Di and Cameron, 2002). Unlike N, P in excreta is almost exclusively present in the dung at 99 %, just 1% in the urine (Donohoe, 2011). Phosphorus in the dung has a concentration of 2.6 g P kg⁻¹ of dry matter (Donohoe, 2011), with 75% in inorganic form (Williams and Haynes, 1990). In contrast, N is present in the dung for 90 % in organic form (Floate et al., 1970; Donohoe, 2011). Excretion of N and P with feces is very large. Donohoe (2011) calculated a release of 16.1 g total N and 3.5 g total P cow⁻¹ 8 hours⁻¹ for feces. Because

of the livestock camping behaviour around water troughs and other concentration areas, depositions can cover just 5% of the pasture area (Williams and Haynes, 1990). Nutrient addition from depositions in this study was likely considerable, and there was no plant and thus uptake of N and P through grazing in the Bare areas.

The Bare areas in the Single treatment likely received larger amount of N and P deposition from excreta compared to the Control. Particularly, because of higher productivity of the manured pasture, the stocking density of Single treatment ranged 1.2 to 5.6 animals ha⁻¹ compared to 0.7 to 2.4 for the Control (Wilson et al., 2010). Additionally, crude protein and P concentration of forage increased 90 and 70%, respectively, in Single compared to Control paddocks (Wilson et al., 2010). The amount of total N and P ingested by steers averaged 19.7 and 2.2 kg ha⁻¹ year⁻¹, respectively, with the Single treatment, but lower at 2.9 and 0.4 kg ha⁻¹ year⁻¹, respectively, for the Control treatment. Subsequently, greater total excretion of N and P likely resulted in not only greater concentrations of extractable forms of the nutrients but also Bare areas of larger size in the Single than Control treatment.

The distribution with depth of nitrate in Bare areas, particularly of the Single paddocks (Fig. 3.2c and 3.2f), may indicate downward movement of the ion and leaching to the shallow water table. High concentrations of extractable-P in the surface layers may also increase the risk of P leaching in this coarse soil when the water table rises to the surface and causes anaerobic conditions (Villapando and Graetz, 2001). Bare areas, thus, clearly represent hotspots of N and P in pasture systems, with annual extent of the Bare areas depending on management practices such as stocking density, that in turn depends on forage productivity and pasture area, as well as on rotational grazing practices and feed and watering systems.

3.4.2 Phosphorus Sequential Extraction: Phosphorus Fractions

Total soil P recovered by sequential extractions was similar to the “one-step” digestion being 3.4 % greater than the latter. The concentration of total soil P by sequential extraction in the Single treatment was generally similar to what reported for Central and Eastern USA for swine slurry treated pastures, and lower compared to dairy and poultry manure treated pastures and croplands (Sharpley et al., 2004). Swine slurry additions across depths and vegetation covers caused total soil P in the Single treatment to increase to 431 mg P kg⁻¹ compared to 341 mg P kg⁻¹ in the Control ($P = 0.03$; Table 3.3). Total soil P in the Bare areas across depths and manure treatments was 430 mg P kg⁻¹ compared to the lower concentration of 342 mg P kg⁻¹ under Grassed cover ($P = 0.02$). As expected, total soil P was stratified in the profile. It was on average 512 mg P kg⁻¹ in the surface 5 cm across manure and vegetation cover treatments and greater than concentrations at other depths ($P < 0.001$). A two-way interaction of manure x depth ($P = 0.04$; Fig. 3.3a) was apparent because slurry application increased total soil P only in the 0-5 cm increment resulting in 629 mg P kg⁻¹ with the Single treatment compared to 395 mg P kg⁻¹ for the Control.

Table 3.3 Mean values of the concentration (mg P kg⁻¹ dry soil) of P fractions for manure and vegetation treatments and sampling depth after three years of annual swine manure applications. Shown are results of analysis of variance for main and interactions effects.

Treatments	∑Total-P†	H ₂ O-Pi‡	NaHCO ₃ ⁻ Pi	NaHCO ₃ ⁻ Po	NaOH-Pi	NaOH-Po	HCl-P	Residual-P
	mg P kg ⁻¹ dry soil							
<i>Manure</i>								
Control	341.4b	8.4b	14.2b	1.7b	14.6b	28.3	163.6	111.4
Single	430.6a	34.2a	39.1a	6.8a	30.4a	36.5	180.2	101.6
<i>Vegetation</i>								
Grassed	342.4b	11.0b	15.6b	3.6	13.1	30.1	161.0	106.5
Bare	429.7a	31.6a	63.2a	4.9	32.0	34.8	182.8	106.5
<i>Depth</i>								
0-5	511.8a	62.1a	63.2a	10.1	45.4	58.5a	142.3b	129.2a
5-15	368.3bc	15.0b	25.7b	3.5	25.4	27.8bc	143.9b	125.9a
15-30	382.3b	6.5c	15.1c	2.2	15.6	31.2b	188.0a	121.8a
30-60	281.6c	1.6c	2.8d	1.2	3.8	12.2c	213.4a	49.1b
ANOVA	P > F							
Manure	0.03	0.06	<0.10	0.06	0.05	ns	ns	ns
Vegetation	0.02	0.08	<0.01	ns	ns	ns	ns	ns
Depth	<0.001	<0.001	<0.001	ns	ns	<0.001	<0.01	<0.001
Manure x Vegetation	ns	ns	ns	ns	ns	ns	ns	ns
Manure x Depth	0.04	0.03	ns	ns	ns	ns	0.07	ns
Vegetation x Depth	ns	0.09	ns	ns	ns	ns	ns	0.04
Manure x Vegetation x Depth	ns	ns	ns	ns	ns	ns	ns	ns

N = 96. ANOVA was performed on log-transformed data: log_n (P + 1). Non transformed arithmetic means are shown in the table. Manure main effects are means of two experimental units (paddocks) each split in grass (ten sampling locations) and bare earth (one sampling location). Each location was sampled in six depth intervals. Model was a randomized complete block split plot with covariance among depths. Effects are significant at P < 0.1. Separately for each effect, means followed by the same letters are not significantly different at α = 0.1 and Tukey-Kramer adjustment for beta error. Pi, Po and Pt refer to Inorganic (P analyzed directly in the extract), Organic (Po=Pt-Pi) and Total (P analyzed after digestion) P respectively. †Total Soil P = H₂O-Pt + NaHCO₃-Pt + NaOH-Pt + HCl-Pi + Residual-P. ‡H₂O-Po is not shown because it didn't have any significant effect. Overall mean across treatments for H₂O-Po is 3.5 mg P kg.

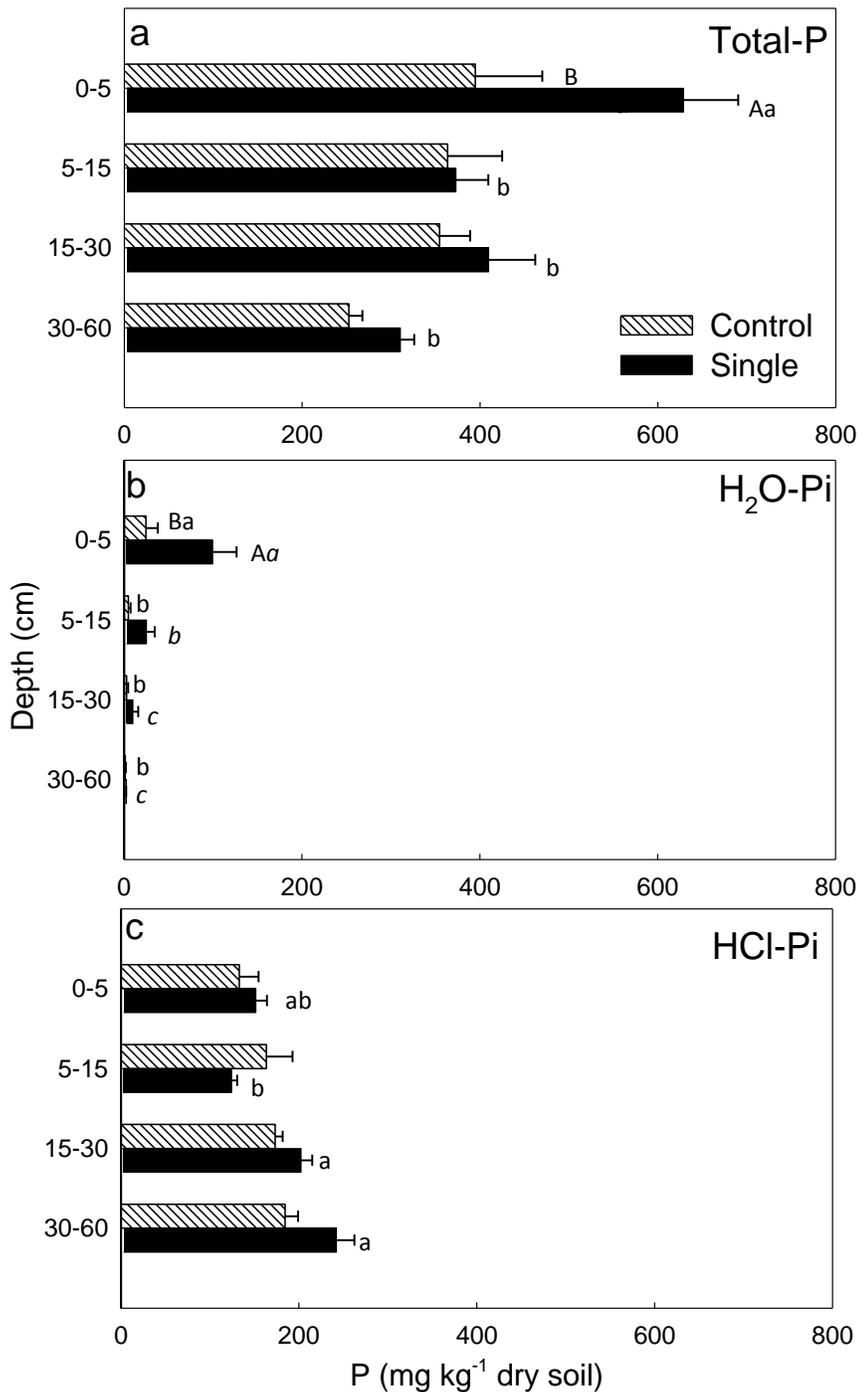


Figure 3.3 Mean (a) total soil- (b) water extractable- and (c) hydrochloric acid extractable-P concentration and + one standard error of the mean for manure x soil depth treatment combinations from sequential extraction analysis (n = 4). Means followed by the same capital letter for each depth are not significantly different. Means followed by the same lower case letter for each manure treatment are not significantly different

The concentration of water extractable P_i (H_2O-P_i) was greatest in Bare areas and in Grassed areas having received slurry, and it also decreased with depth (Table 3.3, Fig. 3.3b and Fig. 3.4a). Water extractable P_i as a proportion of total P seems to have increased with slurry addition and under Bare cover. The fraction of H_2O-P_i ranged from 8 % of total soil P in the surface 5 cm for Single and Bare treatments to 3 % for the Control and Grassed treatments. The proportion of H_2O-P_i decreased with depth from 12 % at 0-5 cm to 0.6 % at 30-60 cm. A two-way interaction of manure x depth ($P = 0.03$; Fig. 3.3b) highlights that additions of slurry increased concentrations of H_2O-P_i for only the surface (0-5 cm). At the surface across vegetation covers, the concentration was four times greater with the Single treatment than in the Control. A weak two-way vegetation x depth interaction ($P = 0.09$) was due to a greater increase in H_2O-P_i concentrations for Bare than Grassed vegetation in the surface 0-5 cm compared to lower depths (Fig. 3.4a). Compared to H_2O-P_i , the concentration of H_2O-P_o was smaller and not affected by treatments. The overall average was $3.5 \text{ mg } H_2O-P_o \text{ kg}^{-1}$ or 0.9 % of total soil P across all treatments.

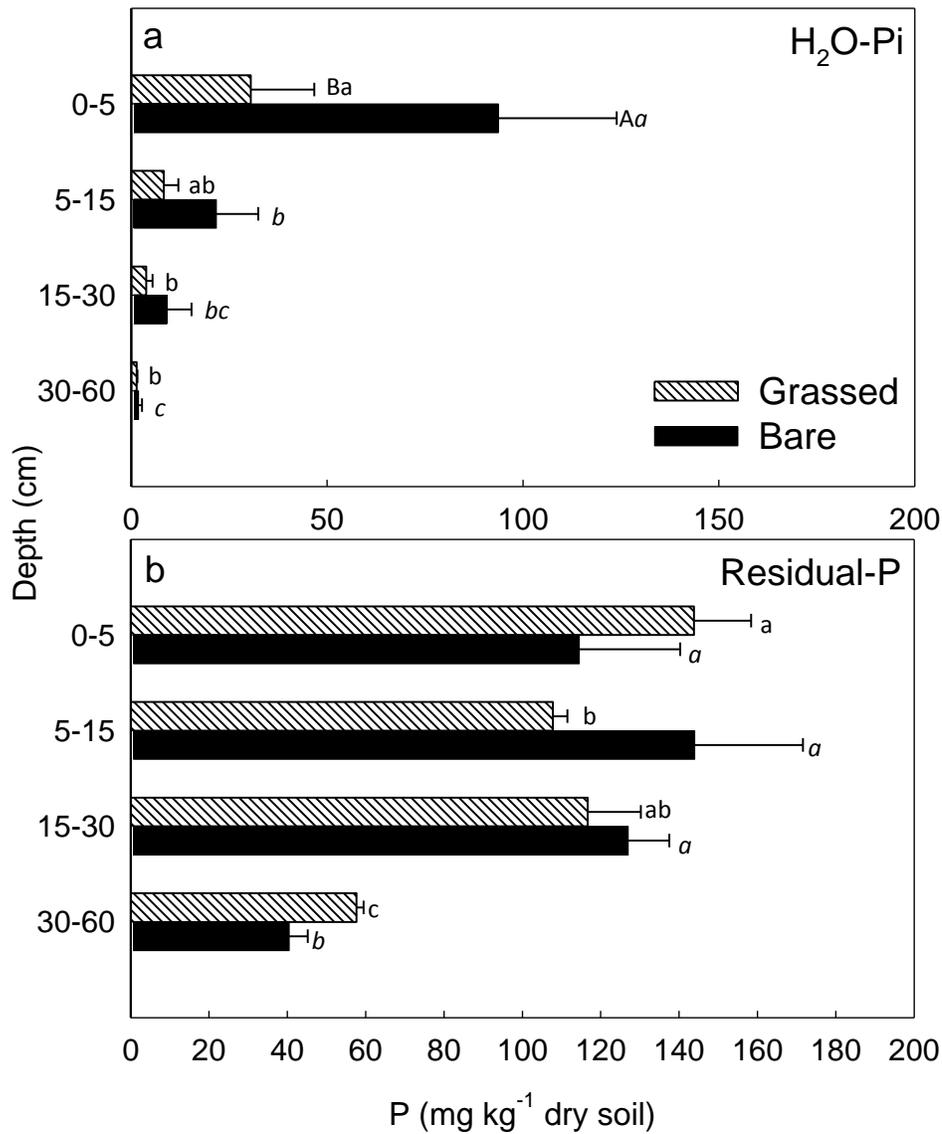


Figure 3.4 Mean (a) water extractable- and (b) residual-P concentration + one standard error of the mean for vegetation x soil depth treatment combinations from sequential extraction analysis (n = 4). Means followed by the same capital letter for each depth are not significantly different. Means followed by the same lower case letter for each vegetation treatment are not significantly different

Sodium bicarbonate extractable P ($\text{NaHCO}_3\text{-P}$) is thought to include the pool of soil P weakly adsorbed onto colloidal minerals and organic matter (Tiessen and Moir, 1993). The concentration of $\text{NaHCO}_3\text{-P}_i$ varied strongly with vegetation cover and depth ($P < 0.01$) and more weakly with slurry addition ($P < 0.1$; Table 3.3). It increased 3.1-fold under Bare than Grassed cover compared to a 1.7-fold increase in concentrations for the Single than Control treatment (Table 3.3). The concentration of $\text{NaHCO}_3\text{-P}_i$ decreased steadily with depth from 63 mg P kg^{-1} in the 0-5 to 3 mg P kg^{-1} in the 30-60 cm depth (Table 3.3). Bicarbonate extractable P_i represented 15 and 9 % of total soil P in the Bare and Single treatments respectively, and 5 and 4 % in the Grassed and Control treatments. It also decreased from 12 to 1 % from the surface to the lowest depth increment. Sodium bicarbonate extractable organic P ($\text{NaHCO}_3\text{-P}_o$) increased with slurry treatment, with an average concentration across depths and vegetation covers of 1.7 mg P kg^{-1} in the Control and 6.8 mg P kg^{-1} in the Single treatment. However, $\text{NaHCO}_3\text{-P}_o$ concentration ranged over just 0.4-2.0 % of total soil P for all treatments and depths combinations.

Sodium hydroxide extraction (NaOH-P) is assumed to release Al and Fe bound-P and other forms stable at acid pH (Iyamuremye et al., 1996). Its concentration decreased with depth, amounting for 20 % of total P at 0-5 cm and 6 % of total soil P in the 30-60 cm soil layer. Manure treatment was the only significant effect on NaOH-P_i ($P = 0.05$), with NaOH-P_i concentrations of 14.6 mg P kg^{-1} in the Control and 30.4 mg P kg^{-1} in the Single treatment, and 4.3 and 7.1 % of total soil P, respectively. This indicates that slurry application was related to increased accumulation of Al- and Fe-bound P fractions. The concentration of NaOH-P_o was generally greater than NaOH-P_i and differed by only depth ($P < 0.001$). It decreased from 59 to 12 mg P kg^{-1} in the 0-5 to the 30-60 cm increments, respectively (Table 3.3). A large part of NaOH-P_o may be composed of

orthophosphate mono and di-esters with the former being more stable than the later.

Orthophosphate mono and di-esters are organic-P compounds often found in the alkaline soluble fraction of manures and manured soils (Lehmann et al., 2005).

Hydrochloric acid extractable-P (HCl-P_i) is assumed to include alkaline stable soil-P forms, particularly inorganic P bound to Ca²⁺ and Mg²⁺ in minerals of varying degrees of stability (Iyamuremye et al., 1996; von Wandruszka, 2006). The concentration of HCl-P_i was lower at 0-5 (142.3 mg P kg⁻¹) and 5-15 (143.9 mg P kg⁻¹) cm compared to 188.0 and 213.4 mg P kg⁻¹ at 15-30 and 30-60 cm, respectively (depth effect, *P* = 0.002). Its proportion of total soil P increased with depth from 24 % at the surface (0-5 cm) to 78 % for the 30-60 cm increment of the Single treatment across vegetation covers. This was probably due to the increasing pH and CaCO₃ with depth (Table 3.1). As expected because the soil was alkaline, HCl-P_i was the dominant soil P fraction. A weak two-way interaction for manure x depth (*P* = 0.07; Fig. 3.3c) was present because the HCl-P_i concentration at 5-15 cm with slurry application was 124 mg P kg⁻¹ and significantly lower than 203 and 242 mg P kg⁻¹ at 15-30 and 30-60 cm, respectively. This result could have been related to the retention of P leached through macropores from the surface soil, to the effect of the organic matter protecting P from sorption in the surface soil, or to acidification of the 5-15 cm layer with slurry application (Appendix II.3) that may have caused Ca- and Mg-P forms to illuviate into the deeper, more alkaline layers.

The residual-P fraction includes all the recalcitrant forms of soil P not obtained using the previous extractants of soil. Residual-P was the second largest fraction after HCl-P_i. Its percentage of total soil P concentration was smallest at 14 % for the 30-60 cm Bare areas to greatest at 36 % with the 5-15 cm Grassed soil. Residual-P was greater in the top three depths ranging from 122 to 129 mg P kg⁻¹ but decreased abruptly at 30-60

cm to 49 mg P kg⁻¹ (depth effect, $P < 0.001$). Similarly, residual-P as a percentage of total soil P decreased from a range of 25 to 32 % over the top three depths to 17 % at 30-60 cm. A two-way interaction for vegetation x depth ($P = 0.04$; Fig. 3.4b) likely resulted because under Bare vegetation there was a redistribution of residual-P in the uppermost profile. Specifically, in the Grassed cover, residual-P concentration at 0-5 cm was 144 mg P kg⁻¹ and greater than the 108 mg P kg⁻¹ at 5-15 cm while in the Bare areas residual-P concentrations were inverted with depth, being 115 and 144 mg P kg⁻¹ at 0-5 and 5-15 cm, respectively, but not significantly different. This result may indicate that trampling of the surface soil caused a physical redistribution of soil P pools or a mixing of soil with depth.

3.4.3 Soil Phosphorus Saturation and Relation with Water Soluble Phosphorus

Manure treatment did not affect the total soil P sorption capacity at 0-5 cm. In contrast, many studies report increased (Brock et al., 2007; Sharpley et al., 2004; Sato et al., 2005, Weaver and Richie, 1994) or decreased P sorption capacity (Siddique and Robinson, 2003; Reddy et al., 1980; Holford et al., 1997; Laboski and Lamb, 2003; Iyamuremye et al., 1996; Lehmann et al., 2005) after manure additions. The lack of effect of manure treatment on P sorption capacity may be due to limited amount of manure applied; i.e., only three consecutive years of applications occurred with this study. Phosphorus sorption capacity however tended to decrease with Bare compared to Grassed areas, being 261 and 367 mg P kg⁻¹ ($P = 0.099$), respectively (Appendix II.6).

Nonetheless, there are indications that soil P was approaching 100 % saturation of the sorption capacity in the Bare and Single treatments. A degree of P saturation ‘change point’ has been observed in past studies in which H₂O-extractable P increases

dramatically above a threshold value of degree of P saturation (Nair et al., 2004). However, there is no evidence of change point behaviour in transport studies in Manitoba soils. The Control Bare, Single Grassed and Single Bare treatment combinations had a degree of P saturation at 0-5 cm higher than the change point value reported by others. The change point corresponded with a degree of P saturation of 30 % in Brock et al. (2007), 25 % in Kleinman et al., (2000), 20 % in Griffin et al. (2003) and Maguire and Sims (2002) and 10% in Hooda et al. (2000). In the current study, regression analysis was used to quantify the relation of degree of P saturation level to H₂O-P_i at 0-5 cm. The analysis revealed that H₂O-P_i was not dependent upon P sorption capacity (Fig. 3.5a), but was strongly dependent upon degree of P saturation ($r^2 = 0.71$, $P < 0.0001$; Fig. 3.5b). Hooda et al. (2000) also found H₂O-P_i increased with degree of P saturation but not with P sorption capacity. Water soluble P_i (mg kg⁻¹) could be predicted using the following linear equation [5]:

$$\text{H}_2\text{O-P}_i = 13.5 + 124 (\text{degree of P saturation}) \quad [5]$$

A change point for degree of P saturation above which H₂O-P_i increases greatly was not determined in the current study. H₂O-P_i steadily increased with degree of P saturation with bi-plot values for the Control Grassed areas grouping lower than for the Single Grassed areas (Fig 3.5b). Control Bare areas had bi-plot values positioned at the low range of the Single Grassed bi-plot and the Single Bare values being at the high end. A further regression analysis excluding the Control Grassed treatment that had values below an hypothetical change point of 10 % of DPS did not improve the statistical output ($r^2 = 0.60$, $P < 0.001$; data not presented). Thus, it appears that a change point in the relationship between DPS and H₂O-P_i was not present in this study.

Slurry addition as well as animal congregation around supplements and waterers resulted in an increase in soil P saturation and in an increase of H_2O-P_i . Probably as a consequence of more P in the soil solution at soil surface, extractable P in the 5-15 and 15-30 cm depths of the Single Grassed, Control Bare and Single Bare combinations increased compared to the Control Grassed (Fig. 3.1a). This suggests that some slurry and deposited P moved in the soil profile. Determining degree of P saturation at lower depths than conducted here and determinations of soluble P concentration in the water table would be required to understand whether this downward movement resulted in P leaching to the shallow water table. Nelson et al. (2005) reported very high P leaching losses below 45 cm of soil after 20 years of swine slurry additions to a sand soil pasture, where the degree of P saturation was greater than 45 %. Sims et al. (2002) also reported a split-line relationship between degree of P saturation measured with the Mehlich-3 method and dissolved reactive P concentration in column leachate, noticing very high P leachate concentrations of more than 10 mg P L^{-1} at degree of P saturation of 60 %.

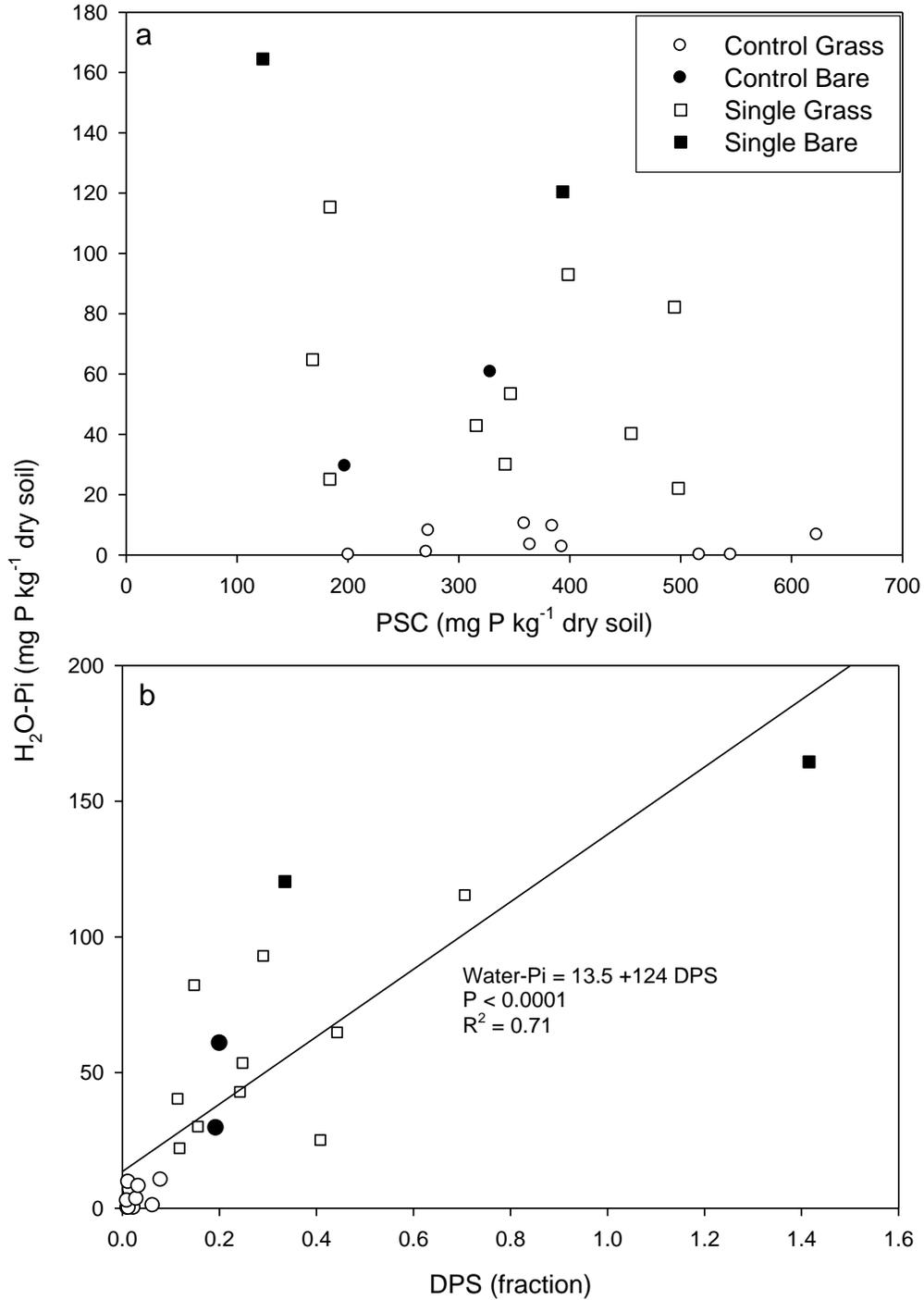


Figure 3.5 Scatter plots of (a) P Sorption Capacity and (b) Degree of P Saturation by inorganic water extractable P determined with sequential analysis, for manure (Control and Single) x vegetation (Grassed and Bare) treatment combinations at 0-5 cm soil depth. For (b), the regression statistics referring to the independent variable degree of P saturation are given. A data point is a sampling location (n = 24).

3.4.4 Implications of Slurry Application and Animal Congregation in Pastures on Phosphorus Lability.

3.4.4.1 Implications of Slurry Application to Grass on Phosphorus Lability. The increase in extractable P and total P in the surface soil of the manured pasture compared to the Control, would not represent an environmental concern, since runoff is unlikely in this coarse texture soil and flat terrain and from the results here, there is no evidence of substantial downward P movement. However, slurry addition also had a subtle effect in shifting the balance of soil P pools towards labile fractions. Similar findings of manure P addition selectively increasing labile P were reported by Leinweber et al. (1999), Lehmann et al. (2005) and Zheng et al. (2004). Welsh et al. (2009) concluded that P release or inclusion from and into recalcitrant pools is slow. Thus, only the labile-P pools are affected, at least initially, by additions and plant uptake of P, and this is in agreement with the results here. It is unlikely that slurry addition increased the lability of previously resident soil inorganic P (Oberson et al., 1993) because the initial concentrations of P in paddock soils were very low.

Another effect of swine addition was the lack of increase in concentration of organic P fractions. Some organic fractions of soil P can be very mobile, specifically because they are not as easily adsorbed onto soil colloids. Many studies have reported limited or no effects of swine slurry applications on organic P fractions (Oberson et al., 1993; Sharpley et al., 2004). Manure applications likely increased phosphatase activity as well as microbial biomass and microbial activity (Oberson et al. 1993; Lalande et al. 2000); thus, increasing the turnover of organic P. Other hypotheses are that the methods used in this study may not be adequate to detect P_o changes, as P_o may have been hydrolyzed by key extracts. Also, timing of sampling may have missed periods of greater

P_o, prior to mineralization. Alternatively, the low application rates may have been insufficient to build the P_o pool (Kleinman, pers. comm.).

Sequential P extraction of slurry obtained from the same storage used in this study (Kumaragamage et al., 2009) showed that labile-P (H₂O-P plus NaHCO₃-P) at 78% of total P was the vast majority of total P. This is in agreement with other studies that reported swine slurry and other liquid manures to have the greatest proportion of water extractable P (9.2 g kg⁻¹) among livestock manures (Kleinman et al., 2005). It seems labile-P in the slurry contributed to the increase in soil labile-P with slurry treatment, as swine slurry commonly behaves as source of soluble and labile P to soils (Elliott et al., 2006)

According to Kumaragamage et al. (2009), 16 and 48 % of H₂O-P and NaHCO₃-P, respectively, in swine manure were organic. In the current study, organic soil-P fractions did not increase with slurry addition indicating these fractions were likely mineralized and likely contributed to the labile inorganic-P. Further understanding of the conversion of organic P to labile P is important, especially considering interest in removal of inorganic P in slurry such as struvite formation.

3.4.4.2 Interpretation of Saturation and Sorption Capacity Effects on Phosphorus Lability with Continue Slurry Application to Grass. Slurry addition to the pasture did not modify the P sorption capacity of the surface soil. However, continued additions of slurry for more than three years could lead to either increase by Ca additions at high rates, or more commonly decrease of P sorption capacity. Water soluble P was strongly dependent on degree of P saturation, and concentrations ranged from 0 to 10 mg P kg⁻¹ for Control Grassed and 22 to 115 mg P kg⁻¹ for Single Grassed. Thus already after three

years of swine slurry application, an increase in amounts of soluble P was evident for the surface soil. The risk related to these concentrations would mainly be of leaching at this site, due to its flat terrain and coarse subsoil texture. The pathway of loss could be macropore flow, as the movement of surface P downward through matrix flow is unlikely since plant roots would be able to intercept P in the soil solution, and the P that moved down in the profile would be absorbed on very low P saturated subsoil. However, with continued slurry additions, as it can be common, it is possible that the degree of P saturation of the deeper layers would increase to levels of environmental risk.

3.4.4.3 Bare Areas as Hotspots of Labile Nitrogen and Phosphorus in Pasture

Systems. The results here indicate that cattle congregation around water and supplements can quickly result in hotspots for risk of N and P loss to the environment. Nitrate concentrations in Bare areas were distributed through the monitored soil depth with substantial concentrations at 60-120 cm, indicating vertical movement of this nutrient likely occurred.

Bare areas were also hotspots of Olsen-extractable and H_2O -P, but unlike NO_3^- , only in the top 30 cm (Fig. 3.1a). Compared to Grassed areas, Bare areas had increased total soil P and labile, inorganic P fractions, while there was little effect on recalcitrant and organic-P. Donohoe reported 99 % of total P excreted to be with feces and only 1 % with urine. She also found 75 % of P in open beef-cow feces was inorganic, with 65 % of total P as labile fractions (H_2O -P plus $NaHCO_3$ -P). The effect of animal deposition on P fractions in Bare areas was, not surprisingly, similar to that of slurry to grassed pasture.

Unlike grassed areas having received slurry, Bare areas had decreased P sorption capacity at 0-5 cm after just three years. The decrease in P sorption capacity in Bare areas

could be attributed perhaps to that competition from anions such as nitrate and displacement of calcium by cations prevented sorption of orthophosphate (Whalen and Chang, 2001; von Wandruszka, 2006). The degree of P saturation of Bare areas at 0-5 cm also increased to values similar or greater than Single Grassed, indicating high P saturation and soluble P concentration. It is also interesting to note that Bare areas in the Single paddocks, more densely grazed and where the grass biomass was greater and with higher % of N and P compared to the Bare areas in the Control, had greater extractable nutrients build-up, labile-P, and degree of P saturation and water soluble-P at 0-5 cm. The absence of grass in these Bare areas means remediation of these locations relies upon preventing animal access and promotion of grass reestablishment. In a practical sense, moving water and supplement supplies in pastures each year is recommended to prevent establishment of bare areas, and if a bare area begins to form, absence of traffic in subsequent years can promote removal of N and P by plants (Dahlin et al., 2005). Furthermore, the practice of placing mineral supplements in car tires laid on the soil surface should be replaced with tubs allowing for trapping and periodic removal offsite of drained water.

3.5 Conclusions

Three years of swine slurry applications to a coarse soil pasture at rates appropriate for forage N requirements did not increase extractable nitrogen but caused an increase in extractable P in the surface 30 cm and of total P in the surface 5 cm of soil. Labile inorganic fractions of P (water and sodium bicarbonate extractable) were the soil pools primarily increased by slurry addition. As a result, P saturation and concentration of water soluble P were increased for the surface 0-5 cm soil. Soil organic P fractions were

unaffected by slurry addition. Overall, three years of slurry application did not pose a significant risk of N and P loss to the environment, but the presence of a water table at times reaching the soil surface requires further attention if P is mobilized.

Bare areas at congregation sites of grazing livestock around waterers and salt supplement locations were hotspots of nitrate build-up in the soil profile. As well, these areas were hotspots of labile inorganic P in the surface 30 cm soil, compared to the grassed pasture. Here, P saturation at 0-5 cm was particularly high, and the soil sorption capacity was lowered. Thus, areas without vegetation in the pasture where nutrients from animal excreta build-up are areas of particular risk of both P and N losses to the environment, especially in the more densely grazed and nutrient rich manured pastures. Management of these areas is of primary importance and should involve periodic movement of the waterers in the pasture during the grazing season to limit waste deposition thus allowing grass reestablishment and nutrient removal.

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4. NITROGEN AND PHOSPHORUS CONCENTRATIONS IN THE SURFACE OF SHALLOW GROUNDWATER IN A COARSE TEXTURED PASTURE SOIL RECEIVING SWINE SLURRY

4.1 Abstract

Applications of swine slurry to coarse texture soils can increase the concentrations of labile nitrogen (N) and phosphorus (P) leading to leaching of the nutrients to groundwater. A study was conducted in South-Eastern Manitoba on a gravelly sandy soil with the objective to determine whether five years of slurry addition to a tame pasture resulted in N and P transfer to the surface of shallow groundwater at the site. Swine slurry was applied from 2004 to 2008 at grass N requirement rates, and in 2009 at double the rate. In the first experiment, two paddocks harvested by haying and receiving two slurry treatments (Control-none, Single-spring application) were monitored for the concentrations of the nutrients along a gradient of groundwater table depth from to the soil surface over four periods in 2007 to 2009. In the second experiment, paddocks having received the slurry treatments and grazed by steers were monitored for the nutrients in the surface of groundwater in grassed and bare areas around water troughs and mineral supplements over the four periods. Concentrations of nitrate, dissolved reactive P, total dissolved P, and chloride in groundwater were determined at 1.5 to 2 m using monitoring wells and also at 0.3, 0.6, 0.9 and 1.2 m from soil surface using multi-level samplers, when groundwater height permitted. In the hay paddocks, nitrate and dissolved phosphorus concentrations in the monitoring wells were low ($< 0.5 \text{ mg NO}_3^- \text{-N L}^{-1}$ and $< 0.015 \text{ mg P L}^{-1}$). Under the wetter than normal conditions in 2009, analysis of water from the multi-level samplers showed that high concentrations of P occurred at 30 cm of

groundwater depth when the water table rose to soil surface and to a lesser extent at 120 cm of groundwater depth in the Single treatment before and after the double rate application of slurry. Chloride concentration in groundwater increased in the Single compared to the Control treatment, suggesting that potential nitrate leaching was prevented, likely by plant uptake of N. In grazed paddocks, nitrate in monitoring wells increased with the Single and Bare treatments compared to Control and Grassed treatments, respectively. Concentrations of nitrate in the wells were generally greater than 10 mg NO₃⁻-N L⁻¹ in the Bare areas regardless of the slurry treatment. Dissolved P was always lower than 0.03 mg P L⁻¹, except in the Single Bare areas under wet conditions that occurred in 2009, both before and after the double rate application of slurry. In 2009, reactive and total dissolved P in the wells averaged 0.64 and 0.70 mg P L⁻¹, in the Single Bare before and after slurry application, respectively. The results show that Bare areas in grazed paddocks had significant N and P groundwater concentration whereas grassed areas did not lead to accumulation of the nutrients in the groundwater surface. A further understanding as to what extent leaching from Bare areas, which are relatively small compared to total pasture area, affects nutrient loading to groundwater is needed.

Abbreviations: ANOVA, analysis of variance; HDPE, high density poly-ethylene; N, nitrogen; P, phosphorus; PVC, polyvinyl chloride; PTFE, polytetrafluoroethylene; S.E., standard error.

4.2 Introduction

Swine (*Sus scrofa* L.) slurry additions to grasslands can improve the productivity, nutrient content and carrying capacity of pasture for backgrounding beef cattle (Wilson et

al., 2010). At the same time, application of slurry to pastures recycles the manure nutrients in an economically viable way for hog producers and users of the forage. However, over the long term, swine slurry and other manures should be applied at rates that match the removal of nutrients, especially N and P (Carpenter et al., 1998). If surpluses of N and P are allowed to accumulate, their concentrations in the surface soil build-up over time (Coppi et al., Chapter 2; Toth et al., 2006; Schomberg et al., 2009), increasing the risk of surface and groundwater contamination.

Two thirds of swine slurry N is in the ammonium, plant available form, the rest being in more recalcitrant, organic N form (Cabrera and Gordillo, 1995). Manure applications are based upon plant available N only, thus a considerable surplus of potentially mineralizable organic N accumulates in the surface soil (Coppi et al, Chapter 2). Moreover, the N:P ratio in manures (including slurry) is usually larger than the N:P ratio of plant biomass (Toth et al., 2006; Whalen and Chang, 2001). Thus, applications based on N requirements lead to surpluses of P in soil (Coppi et al., Chapter 2; Sistani et al., 2008; Toth et al., 2006; Whalen and Chang, 2001). This is especially true in grazed pastures where the efficiency of nutrient removal by cattle is low (Wilson et al., 2011), but it also occurs to a lesser extent in hay pastures (Coppi et al., Chapter 2; Franzluebbers et al., 2002).

Accumulation of excess, labile N in soil can lead to the downward movement of nitrate in wet climates and especially on coarse texture soils (Hooda et al., 2000; Di and Cameron, 2002). Accumulation beyond the soil's ability to retain P (sorption capacity) can result in increased concentrations of water soluble P. The water soluble P is at risk for leaching in coarse texture soils (Dou et al., 2009; Coppi et al., Chapter 3; Hooda et al., 2000; Leinweber et al. 1999; Nelson et al. 2005). Also, swine slurry additions tend to

increase the concentration of the labile (water and sodium bicarbonate extractable) more than the recalcitrant P fractions (Coppi et al., Chapter 3; Hountin et al., 2000; Kashem et al., 2004).

In grazed pastures, livestock tend to congregate and loaf in areas around water troughs and mineral feeders. This leads to the accumulation of nutrient rich beef cattle manure in small areas, loss of grass cover and absence of nutrient and water removals. Build-up and movement of labile forms of N and P in the soil profile are thus more likely in these areas (Coppi et al., Chapter 3; Williams and Haynes, 1990; Haynes and Williams, 1999; Sigua and Coleman., 2006). Moreover, in manured pastures, nutrient concentration and biomass production of the grass are higher (Wilson et al., 2010). This leads to an increased carrying capacity of the pasture and a greater number of grazing animals per unit area. As a result, bare areas of manured pastures are likely to receive larger additions of N and P from beef cattle manure than bare areas of non manured pastures (Coppi et al., Chapter 3).

When nutrients are added to soil as fertilizers and manure, a water table close to the surface increases the risk of groundwater contamination during recharge from precipitation or through solubilisation of soil adsorbed ions as the water table rises. This mechanism is particularly important for P, since it has been shown that some of its recalcitrant fractions, such as P bound to Al and Fe oxides and organic P, can be mobilised under low redox conditions that develop in water-saturated soils (Ajmone-Marsan et al., 2006; Young and Ross, 2001). Since shallow groundwater is usually hydrologically connected to surface water in rivers or drains, N and P that leach to the groundwater eventually reach the surface waters depending on base and storm flow

dynamics, flow velocity and on the sorption and denitrification processes that occur in the saturated zone (Kleinman et al., 2007; Hooda et al., 2000; Nelson et al., 2005).

It was found that 16 % of private wells exceeded the drinking water threshold of $10 \text{ mg NO}_3^- \text{ N L}^{-1}$ in Manitoba (Manitoba Water Stewardship Report No. 2006-02, 2006). High concentration of P in surface water is the main cause of algal blooms or eutrophication, a process that leads to depletion of oxygen in water and loss of biodiversity of aquatic ecosystems (Carpenter et al., 1998). In the province of Manitoba in central Canada, Lake Winnipeg, one of the largest lakes in the world, is undergoing eutrophication (Schindler et al., 2006). The deterioration of the water quality of Lake Winnipeg has prompted the legislative authority to issue new environmental regulations. Among them are limitations to the hog industry and to manure application to land (Manitoba Conservation, 2008 and 2011), since manure and livestock operation are considered among the main sources of nutrients to the watershed.

There is a lack of understanding on N and P concentration in shallow groundwater after slurry application and/or in pastures. Most studies involving N leaching have focused on losses to subsurface drainage (Booltink, 1995) and leachate from suction samplers (Ledgard et al., 1999; Eriksen et al., 1999; Steele et al., 1984; Toth et al., 2006) rather than concentration in groundwater. Studies about N losses to groundwater were mainly conducted with slurry amended field crops (Mantovi et al., 2006; Stites and Kraft, 2001). Similarly, the focus for P losses in grasslands has mainly been on overland run-off (Turtola and Kemppainen, 1998; Turtola and Yli-Halla, 1999; Haygarth et al., 1998) and on concentration of P in lysimeters' leachate (Chardon et al., 2007; Turner and Haygarth, 2000; McDowell and Sharpley, 2001; Tarkalson and Leytem, 2009), in soil solution extracted by suction cups (Toth et al., 2006; Nelson et al., 2005;) and in subsurface

drainage (Haygarth et al., 1998; Turtola and Kemppainen, 1998; McGechan et al., 2005; Magesan et al., 2006). The dynamics of N and P leaching to groundwater are different from drainage and soil solution concentrations, because of the dilution effect of the groundwater volume on the nutrient rich percolate reaching the water table. Even less is known in field studies about the effect of an oscillating water table on P solubilisation, as most research has been conducted with lysimeters or in the laboratory (Villapando and Graetz, 2001; Shober and Sims, 2009; Young and Ross, 2001).

The rural municipality of La Broquerie in the Red River basin in south-eastern Manitoba is one of most livestock dense regions in Canada, where both the hog and the cattle-herding industry thrive (Flaten et al., 2003). In the La Broquerie Pasture and Swine Manure Management Study Site, an experiment showed the beneficial effects of slurry addition to a grazed pasture on forage quality and productivity and on pasture carrying capacity (Wilson et al., 2010). It was also found that hay harvest strategy had better nutrient utilization efficiency than grazing (Wilson et al., 2011). Following these results, a study determining the effect of consecutive additions of swine slurry at grass N requirement rates showed that surface soil accumulation of extractable P in both hay and grazed pasture occurred, with a faster rate of accumulation in the grazed pasture (Chapter 2, Coppi et al.). Another experiment at the site showed that the swine slurry addition mainly affected the labile soil P pools, thus increasing the risk of environmental losses. The same experiment also revealed that bare areas around water troughs and mineral feeders in grazed pastures had accumulations of nitrate and P, and the surface soil was quickly being saturated with P, leading to increased concentrations of water soluble P (Chapter 3, Coppi et al.). These results raised concerns about the possibility of N and P reaching the shallow groundwater, especially at times when the groundwater rises to the

soil surface. Additionally, another experiment revealed the presence of a shallow water table at the site that rose to the surface in wet years, and of a moisture gradient that influenced N dynamics in soil, particularly N₂O emissions (Tenuta et al., 2010).

Thus, the La Broquerie Pasture and Swine Manure Management Study Site was chosen to perform this study with the objective of determining whether yearly swine slurry additions to a gravelly, sand soil and with an underlying shallow water table cause: (1) increased concentrations of N and P compared to a Control in the groundwater of a hay pasture along a gradient of water table proximity to the soil surface; (2) increased concentrations of N and P compared to Controls in the groundwater of a grazed pasture and in bare areas of livestock congregation compared to the grassed pasture; and (3) increased concentration of dissolved P in shallow depths of groundwater close to the soil surface.

4.3 Materials and Methods

4.3.1 Site Location and Description

The research site was the La Broquerie Pasture and Swine Manure Management Study Site located in the Rural Municipality of La Broquerie in South Eastern Manitoba, Canada. Twelve paddocks were established in 2003 on 40 ha of tame pasture that had not received recent additions of fertilizer or manure. A description of the site's soil characteristics was given in section 2.3.1 (Coppi et al.).

The soil at the site is mapped as Berlo series (70%), a lacustrine loamy fine sand and imperfectly drained soil (FAO Gleyed Luvisol, Canadian system Gleyed Dark Gray Luvisol) and the remaining 30% of area as Kergwenan series, an outwash loamy sand to gravel and also imperfectly drained soil (FAO Greyzem, Canadian system Gleyed Dark

Gray Chernozem) (Hopkins, 1985). Soil texture is coarse (loamy sand to sand) and gravelly to very gravelly, with stone content ranging from 31 to 41 % of the total soil weight. The bulk density determined with the excavation method averaged 1.68 Mg m^{-3} in the top 30 cm and 1.98 Mg m^{-3} for the 30-120 cm depth interval. All results for soil properties except bulk density are reported for the stone-free fraction of soil. pH in 1:2 (soil:water) solutions was 7.9 in the top 30 cm and increased with depth to 8.6 at 90-120 cm. The soil was weakly to moderately calcareous, with carbonate content increasing with depth being $21 \text{ g CaCO}_3 \text{ equivalent kg}^{-1}$ at 0-30 and 81 at 90-120 cm. Cation exchange capacity also increased slightly with depth, being 16.5 and $17.1 \text{ mol}_c \text{ kg}^{-1}$ at 0-30 and 90-120 cm depths, respectively. Concentrations of extractable nitrogen and potassium and of sodium bicarbonate extractable-P (Olsen-P) were very low according to fertility status of perennial grasslands (Brummer et al., 2011) prior to addition of swine slurry in 2004 (Table 4.1). Initial organic carbon concentration was also low and decreased with depth being 12.1 and 5.6 g kg^{-1} for the 0-30 and 90-120 cm depths, respectively (Table 4.1).

Table 4.1 Mean and one standard error (S.E) of soil chemical and physical properties in the paddocks included in this study (n = 6), determined in 2003 before the start of the trial.

Soil Depth		pH†	EC	CEC	Organic carbon	CaCO ₃ eq	Total-N	NH ₄ ⁺ -N	NO ₃ ⁻ -N	Olsen-P	Cl ⁻	K ⁺	Sand	Silt	Clay	Stones	Texture‡	Bulk density§
cm			dS m ⁻¹	mol _c kg ⁻¹	----- g kg ⁻¹ -----	-----	-----	-----	-----	mg kg ⁻¹ -----	-----		% weight of fines¶			% weight		Mg m ⁻³
0-30	Mean	7.9	0.1	16.5	12.1	21.0	0.9	1.9	1.7	4.3	1.0	46.0	82.3	9.8	7.8	33.1	Gravelly Loamy Sand	1.68
	S.E.	0.1	0.0	1.2	1.3	4.2	0.1	0.3	0.2	1.3	0.2	3.0	1.4	1.0	0.5	3.3		0.11
30-60	Mean	8.3	0.1	16.5	8.2	52.0	0.4	0.9	1.3	0.8	2.2	20.8	85.0	11.7	3.3	44.5	Very Gravelly Loamy Sand	2.04
	S.E.	0.0	0.0	1.4	1.0	5.4	0.1	0.1	0.1	0.5	0.2	1.3	4.8	4.9	0.3	3.6		0.12
60-90	Mean	8.4	0.1	16.3	6.3	74.3	0.2	0.4	0.7	0.2	3.2	14.8	92.8	5.5	1.7	41.3	Very Gravelly Sand	1.94
	S.E.	0.1	0.0	1.0	1.5	5.9	0.1	0.1	0.0	0.2	0.3	1.1	0.3	0.3	0.2	2.4		0.09
90-120	Mean	8.6	0.1	17.2	5.6	81.0	0.2	0.3	0.5	0.2	3.5	13.7	92.8	5.3	1.8	39.3	Very Gravelly Sand	1.96
	S.E.	0.1	0.0	0.5	0.8	6.4	0.0	0.0	0.1	0.2	0.3	0.6	0.4	0.4	0.2	2.6		0.09

†1:1 soil:water ratio.

‡Textural classes according to Canadian Soil classification.

§Bulk density was determined with the excavation method from 4 soil pits dug in summer 2007. Values reported here are weighted averages of bulk density of 2 replicates of each pedogenetic horizon in the 4 pits.

¶ Fine soil is soil that passed through a 1.5 mm mesh.

Soil at the site has a low water retention capacity, with volumetric moisture content ranging between 0.20 and 0.35 cm³ cm⁻³ at 0 kPa (saturation) and 0.10 cm³ cm⁻³ at 15 kPa (lowest measured value). Drainage is usually fast in the unsaturated zone but is sometimes imperfect due to an impermeable clay layer at a depth of 1.75 - 2 m in the western half of the site grading to deeper depths (> 5 m) towards the eastern side. Thus a shallow water table (< 2 m deep) rises to the soil surface during spring-melt and after successive heavy rains (Tenuta et al. 2010). The slope at the site is gentle to level (< 2%), and the groundwater flows north-west horizontally towards a drainage ditch that flows into the Seine River, a tributary of the Red River.

4.3.2 Experimental Design

The experimental design of the site was detailed in Wilson et al. (2011), with only a brief description given here. The site is divided into 12 paddocks of factorial combinations of slurry application timing (Control, Single spring and Split fall-spring) and forage harvest type (Hay and Grazed) with two replicates per treatment combination. Six paddocks were used for two experiments in this study (Fig. 4.1) and sampled for shallow groundwater from fall 2007 to summer 2009, with swine slurry application occurring yearly from 2004 to 2009.

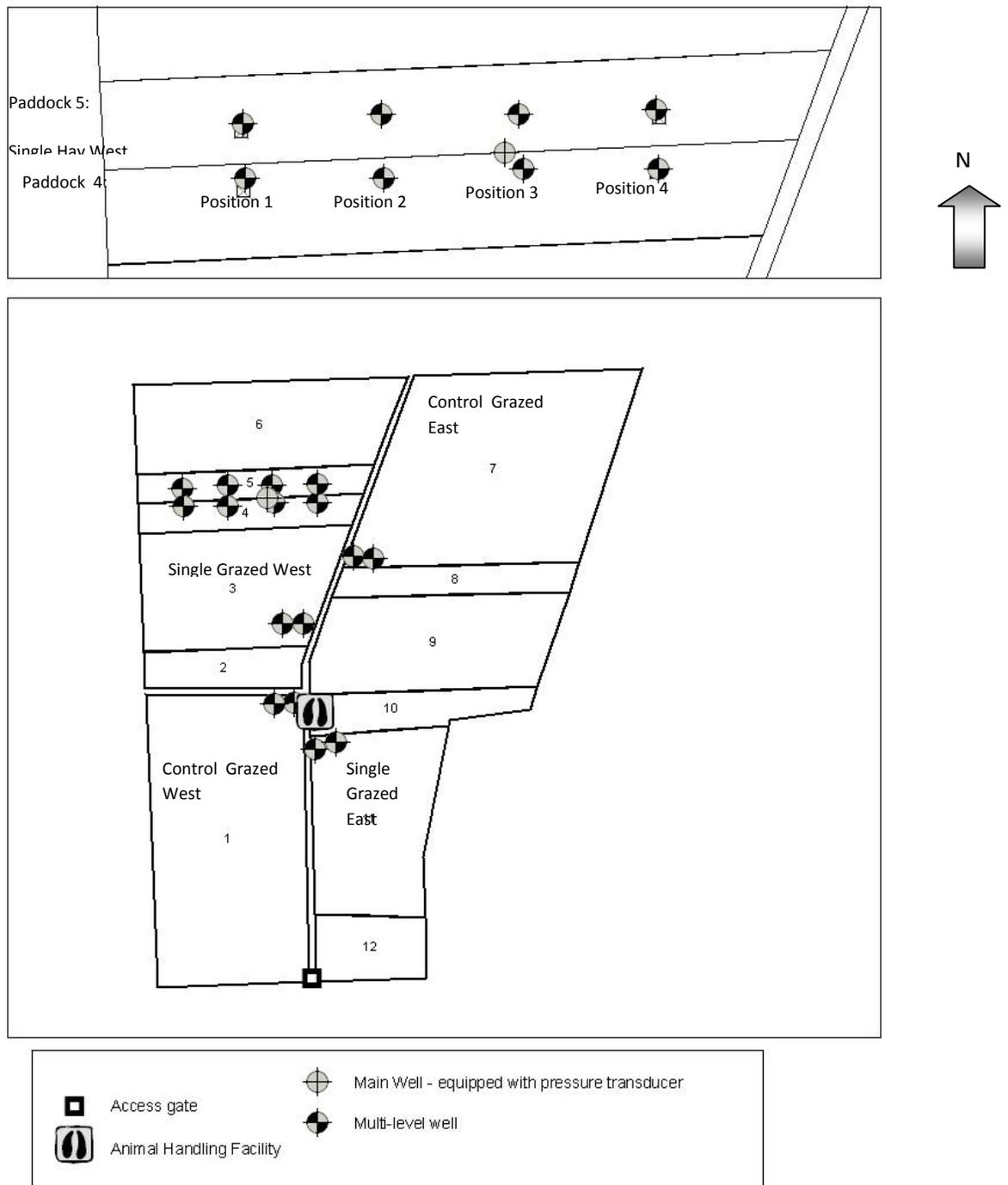


Figure 4.1 Layout of the La Broquerie Research Site (40 ha in total size) showing the paddocks used in this study for slurry (Control-no manure and Single-slurry applied in spring) and forage utilization (Hay and Grazed) treatment with two replicate blocks (west and east). Shown are also the positions of the wells.

4.3.2.1 Experiment 1: Two hay paddocks of 1.2 ha each (Fig. 4.1), located in the west replicate were used to determine the effect of slurry addition and relative water table elevation on N and P concentration in groundwater. The experimental design was a three-way factorial with factors being manure treatment (Control and Single), position (1 to 4) and sampling period (1 to 4). One paddock received six consecutive annual swine slurry applications that started in the spring of 2004 (Single) and the other no slurry (Control). The paddocks were mechanically harvested for hay once per year at the early head phenological stage of the grass, usually in late June (an additional cut in August was done in 2004). The paddocks had a sandy clay layer 1.75 to 2 m below the soil surface that deepened from the west side (1.75 m) to the east side (2 m). This feature- caused a water table depth gradient from west (shallower) to east (deeper). Thus, four monitoring wells were installed in each paddock every 60.6 m along the west-east direction. The north-south distance of the wells between paddocks for the same position was 30 m (Fig. 4.1, inset). Water table depth over the 2007-09 growing seasons averaged from west to east positions, 0.68 (Position 1), 0.82 (Position 2), 0.94 (Position 3) and 1.02 m (Position 4) in the Control paddock. Water table depth in the Single treatment averaged 0.72, 0.88, 0.88 and 0.92 m in the Positions 1 to 4, respectively. There were four monitoring periods. Period 1 was from Oct 29 2007 to April 29 2008 with 5 sampling dates. Period 2 was from May 13 2008 to October 23 2008 with 12 sampling dates. Period 3 was from April 8 2009 to May 26 2009 and Period 4 from June 2 2009 to August 26 2009, with 8 and 7 sampling dates, respectively. Periods 1 and 3 coincided with spring-melt of snow to slurry application. Periods 2 and 4 coincided with the growing season from slurry application to end of the growing season.

4.3.2.2 Experiment 2: The second experiment included four grazed paddocks (Fig. 4.1), previously described in Wilson et al. (2010), to determine the effect of slurry application and grazing congregation areas on N and P concentration in groundwater. Experimental design was a generalized complete block three-way factorial, with factors being manure treatment (Control and Single), vegetation cover (Grassed and Bare), and sampling period (1 to 4, same as for study one). Two paddocks had six consecutive annual swine slurry applications that started in the spring of 2004 (Single) and two received no slurry (Control). The paddocks had forage grasses grazed by cattle (*Bos primigenius Taurus* Bojanus) from June to August starting in 2004. The Control paddocks were 8 ha and the Single paddocks 4 ha in size. Additionally, steers were provided with fresh water and mineral supplements *ad libitum* with Rancher's Choice 8:4:50 Interlake Beef Pasture Premix (Puratone Feeds, Niverville, MB) containing 8% calcium and 4 % phosphorus. The paddocks were mainly covered by tame forage vegetation, but also contained smaller bare areas. A bare area in each paddock was caused by visitation and loafing of steers around a water trough and mineral supplement feeder. The water troughs and mineral feeders in these areas were kept in place from 2003 to 2008 relocated elsewhere in the paddocks in the summer of 2008. The bare areas did not receive slurry application after 2004. The bare areas' size averaged 20 m² in the Control paddocks and 78 m² in the Single paddocks. Two shallow groundwater monitoring wells were installed in each paddock: one in the Bare area and the other 25 m away in the grassed area of the pasture (Grassed). The blocking factor was west –east replicate.

4.3.3 Slurry Applications

A detailed description of the slurry application methodology was given in section 2.3.3 (Coppi et al.). Briefly, swine slurry obtained from the primary cell of a three-stage earthen manure storage was surface applied in early May of each year using a single splash-plate from 2004 to 2006, and dribbled on the surface from 2007 to 2009. Applications averaged 44,000 L slurry ha⁻¹ year⁻¹ (2004-2006) before water sampling started and were 31, 36 and 67 thousand L slurry ha⁻¹ in 2007, 2008 and 2009, respectively. In the 2004-2006 period, applications provided 142 kg plant available-N ha⁻¹ year⁻¹ assuming 25% volatilization of ammonium-N and 25% of organic-N of the slurry available to the grass in the year of application (The Prairie Provinces' Committee on Livestock Development and Manure Management, 2006). In 2007, 2008 and 2009, applications provided 105, 119 and 224 kg plant available-N ha⁻¹, respectively. Total P applied in the slurry averaged 65 kg P ha⁻¹ year⁻¹ in 2004-2006 and was 56, 66 and 107 kg total P ha⁻¹ in 2007, 2008 and 2009, respectively. Chloride application rate averaged 75 kg Cl⁻ ha⁻¹ year⁻¹ in 2004-06, and was 52, 62 and 83 kg Cl⁻ ha⁻¹ in 2007, 2008 and 2009, respectively.

4.3.4 Well Design and Installation

The monitoring wells were installed in the summer of 2007. The wells were hollow PVC pipes with an internal diameter of 2.54 cm. The pipes were 2.17 to 2.70 m in length and perforated by 1.6 mm diameter holes from the 1.8 m length to the bottom of the tube. The pipes were sealed at the bottom with a cap. A removable cap at the top allowed access to the pipes. A 30 cm long portion of the pipes was left aboveground. Thus, the well screen started at 150 cm below the soil surface, thus above the clay horizon.

Each well was also equipped with four Teflon (PTFE) multi-level sampling tubes of 0.476 cm internal diameter to sample at depths within shallow groundwater. These tubes were cut to 1.3, 1.6, 1.9 and 2.2 m lengths. Their bottom ends were wrapped with a 50 µm mesh Nylon screen to exclude coarse particles during groundwater sampling. These samplers were attached to the external wall of the well with several zip ties. A 1 m portion of the samplers was left aboveground. Thus, the sampling depths of the multi-level samplers were 30, 60, 90 and 120 cm below the soil surface, in addition to the 150 cm sampling depth of the well pipe.

The wells were installed using a flight auger of 7.62 cm of diameter mounted on a drill rig (Mobile Drill Intl, Indianapolis). The wells were positioned into a hole dug by the driller and granite gravel (diameter > 1.6 mm) was packed into the annulus. Bentonite pellets were placed in the depth intervals 136.5-133.5, 106.5-103.5, 76.5-73.5, 46.5-43.5 cm to provide physical separation for the groundwater at the five sampling depths. Bentonite powder was poured in the upper 15 cm of the annulus to seal the well at the soil surface. A PVC case of 15 cm diameter, with length of 30 cm above and 15 cm below ground, capped with a 30 cm long PVC tube of 30 cm diameter protected the well from weather and wild animals. Wells in the Grazed paddocks were further protected from steers using a car tire and a metal cage.

After installation and once before start of first sampling in 2007, the wells were developed by bailing with a 1.9 cm diameter HDPE inertial pump system (Waterra Pumps Ltd., Mississauga, ON) and by pumping at 1700 mL min⁻¹ for an hour with a peristaltic pump (Portable Masterflex Sampling Pump Model # 7570-10, Cole-Palmer Canada, Montreal, QC).

4.3.5 Groundwater Sampling and Analysis

The wells and multi-level samplers were sampled from 2007 to 2009, approximately every week during spring-melt and every two weeks or after rains in summer, at the dates previously reported in section 4.3.2.1. During each sampling, first the water depth was measured with a water level meter (Water Level Meter Model # 101, Solinst Canada Ltd., Georgetown, ON), then 1.5 L of water were pumped out of the well and discarded, and finally the well was sampled with the peristaltic pump using Teflon tubes. These samples provided for a measure of nutrient concentration in the shallow groundwater in the range between 1.5 m below surface (start of the well screen) to the clay layer. Depending on the measured water depth in the well, the multi-level samplers were then sampled with the pump. These samples gave a measure of nutrient concentration at the discrete groundwater depths of 30, 60, 90 and 120 cm. All samples were 0.5 L. The samples were placed in HDPE bottles and stored at 4 °C for 24 hours.

After the 24 hours of storage, two 50 mL subsamples of each sample were then vacuum-filtered through a 0.45 µm pore acetate membrane (Micronsep, GE Water and Process Technologies). One subsample was analyzed for nitrate-N and chloride concentrations using an ion chromatograph (ICS-1000, Dionex Canada Ltd., Oakville, ON), and for dissolved reactive phosphorus (DRP) by colorimetry (Murphy and Riley, 1962) using a UV/Visible spectrophotometer (Ultrospec 2100 pro, Biochrom Ltd., Cambridge, UK) with a 10 cm path length cell that provided a detection limit of 0.0025 mg P L⁻¹. The second subsample was digested with a 30 % sulphuric acid solution and potassium persulfate at high pressure (USEPA, 1983; APHA, 1992) and total dissolved P determined by colorimetry as above with a detection limit of 0.003 mg P L⁻¹ (Greenberg et al., 1992). Determination of total dissolved P was done on water samples from the

wells and for only three dates for the samples obtained from the multi-level samplers (May 14 and 26 2009 in Period 3 and June 29 2009 in Period 4), when water table was shallowest and allowed collection of samples from all depths. Dissolved reactive P is considered to be mainly inorganic orthophosphate P (Turner and Haygarth, 2000), thus the difference between total dissolved and dissolved reactive P is often read as a measure of organic P in water (Haygarth et al., 1998).

4.3.6 Weather and Continuous Water Table Height Monitoring

Weather conditions including daily maximum, minimum and average air temperature, and daily rainfall were monitored on-site starting April 2006 using a Campbell Scientific weather station equipped with a datalogger (CR1000, Campbell Scientific Canada Corp., Edmonton, AB). Daily water table depths from fall 2007 to fall 2009 were determined using a pressure transducer with datalogger (Aquistar PT2X with Aqua4Plus Control Software, INW, Kirkland, WA) installed in a monitoring well in the Control Position 3 treatment of Experiment 1.

4.3.7 Statistical Analysis

Statistical analysis was performed with the Statistical Analysis Software program (SAS 9.2, SAS Institute Inc., Cary, NC). For Experiment 1, separate analyses were done for samples collected from wells and multi-level samplers. Nutrient concentrations in samples were analyzed with Proc Mixed as a three-way ANOVA using a nested factorial design with factors position (1 to 4), manure treatment (Control and Single) nested in the position factor, and sampling period (1 to 4). Nutrient concentrations in samples from multi-level samplers were analyzed using a four-way ANOVA, having the same factors

as the previous model and in addition depth of sampling (30, 60, 90, 120, 150 cm). Only dates when the water table depth was 120 cm or above were included in the analysis. Since samples were not available from all the depths at every date, this factorial model was incomplete. Sampling date was considered a replicate in the models used for analysis of data from wells and multi-level samplers.

A randomized complete block split-plot, with separate analyses for samples collected from wells and multi-level samplers, was used in Experiment 2. Nutrient concentration in samples from the wells was analyzed with using a Proc Mixed three-way ANOVA model with the main plot factor being manure treatment (Control and Single) and the split-plot factor being vegetation cover (Grassed and Bare). The third factor was sampling period (1 to 4). Nutrient concentration in samples from the multi-level samplers was analyzed using incomplete four-way ANOVA with the same factors as the previous model and addition of depth (30, 60, 90, 120, 150 cm). Sampling date was a replication and a blocking factor for the east and west paddock replications was considered a random effect.

Unless otherwise specified, effects were considered significant at $P < 0.10$. When relevant, comparison of means was done at $\alpha = 0.1$ using the Tukey-Kramer mean comparison test with adjustment for β -type error. When required because of non-normality (Shapiro-Wilk statistic < 0.85), data were transformed prior to ANOVA with the transformation indicated in the tables where the results are presented. Arithmetic means are reported for non transformed data. Possible heteroschedasticity of residuals was dealt with using the Kenward-Roger method for calculation of denominator degrees of freedom in the F test and the Restricted Maximum Likelihood as model type (SAS

Institute Inc., 2009). A spatial power covariance structure was used among sampling depths (Giesbrecht and Gumpertz, 2004).

4.4 Results and Discussion

4.4.1 Weather and Depth to Water Table

The growing season (April-October) of 2007 was similar to that of the long term average climate (12.5 °C), while the 2008 and 2009 growing seasons were cooler than average by 0.8 and 1.1 °C, respectively. The growing season in 2008 was similar to the long term cumulative precipitation for April to October (422 mm). In contrast, the growing seasons of 2007 and 2009 were wetter than average by 48 and 113 mm, respectively. Particularly, 2009 was a very wet year, with precipitation in May, June and August being 103, 134 and 113 mm compared to the long term averages for those months of 59, 95 and 69 mm, respectively. As a result of the unusual large amount of summer rains, the water table in the Position 3 Control Hay treatment rose to the soil surface six times in 2009. Specifically, it did so at the end of the sampling Period 3 at spring melt, and during most of Period 4 in summer. In contrast, water table depth in the same location was always deeper than 75 cm in the 2008 and 2007 growing seasons.

4.4.2 Nitrogen and Phosphorus in the Shallow Groundwater for Hay Forage-Experiment 1

4.4.2.1 Nitrate. Nitrate concentration in the shallow groundwater of the hay paddocks (Experiment 1) across all four positions at 1.5 m depth averaged less than 0.5 mg NO₃⁻-N L⁻¹ for both Control and Single treatments on any date (Fig. 4.2a). Thus, nitrate accumulation in groundwater generally was likely not a concern with harvest of grass by

hayage. This supports our previous results at the site showing that soil nitrate content at 30-120 cm averaged $7.6 \text{ kg NO}_3^- \text{-N ha}^{-1}$ and was not increased by slurry addition compared to the Control (Chapter 2, Coppi et al.). The greater than N requirement slurry application in 2009 did not cause a large increase of nitrate in the groundwater (Fig. 4.2a). The slight increase in concentration from spring to summer 2009 (Periods 3 and 4) in both Control and Single treatments, was likely a result of a rising water table with the weather conditions (Fig. 4.2e), shortening the leaching path for surface N. Di and Cameron (2002) reported that cut grasslands are of low concern for nitrate leaching with fertilization rates up to $400 \text{ kg N ha}^{-1} \text{ year}^{-1}$, especially if applications are made in spring when temperatures and mineralization rates are low. Concentrations of nitrate in the current study are also low compared to the average of $18 \text{ NO}_3^- \text{-N L}^{-1}$ found by Vadas et al. (2007) in shallow groundwater at 1.5 m under a corn (*Zea mays* L.) – wheat (*Triticum aestivum* L.) – soybean (*Glycine max* L. Merr.) rotation in a silty soil in Maryland receiving poultry slurry and liquid ammonia.

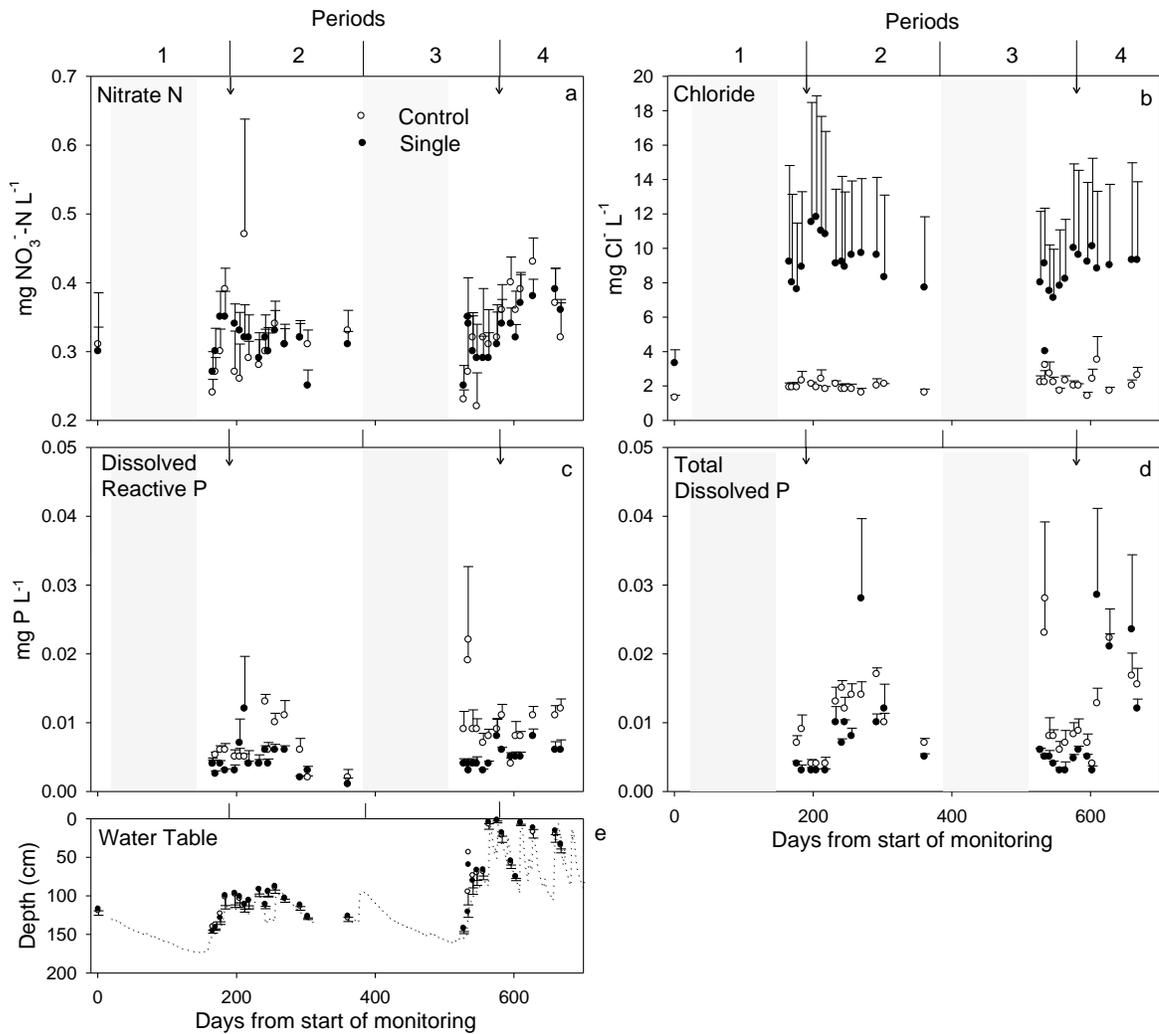


Figure 4.2 Results for (a) nitrate, (b) chloride, (c) dissolved reactive P and (d) total dissolved P concentrations in surface groundwater at 1.5 m depth and (e) depth to water table during Fall 2007 to Summer 2009 for the hay paddocks in Experiment 1. The dotted line in (e) shows continuous water depth measurement with a pressure transducer mid-way along a gradation of four locations of depth of water table to the soil surface. Shaded areas are winter periods. Arrows indicate time of slurry application. Day of start of monitoring (1) was October 29 2007; last sampling day (668) was August 26 2009. Shown are mean values of four wells per treatment plus one standard error of the mean.

Nitrate in groundwater was generally unaffected by manure application, but despite being in low concentrations, it did appear to be affected by the gradient in water table height ($P < 0.0001$; Table 4.2). Position 3 with an average water table depth of 0.94 and 0.88 m in the Control and Single treatments, respectively, had the highest average concentration, at $0.37 \text{ NO}_3^- \text{-N L}^{-1}$. Position 1 averaged 0.68 and 0.72 m in the Control and Single treatments, respectively, and had the lowest concentration of $0.27 \text{ NO}_3^- \text{-N L}^{-1}$. Positions 2 and 4 had concentrations of $0.32 \text{ NO}_3^- \text{-N L}^{-1}$ averaged across the two positions (Table 4.2). Nitrate in groundwater varied with sampling period ($P < 0.0001$). Nitrate concentration was the greatest during Period 4 in summer 2009 by just $0.07 \text{ NO}_3^- \text{-N L}^{-1}$ compared to the other periods. Moreover, the weak three-way interaction manure x position x period ($P = 0.09$; Fig. 4.3) confirms that concentrations were low in all the treatment combinations, and that in Period 4 nitrate was only slightly increased, in both the Control and Single treatments, likely a result of the wet conditions (Fig. 4.2e). This result confirms that the larger than N requirements swine addition of 2009 did not affect significantly nitrate in groundwater.

Overall, it is clear that nitrate accumulation in groundwater did not occur with swine additions to the hay field, independently of water table height gradient and application rate. Nitrogen in surplus of above ground grass removal was $116 \text{ kg N ha}^{-1} \text{ year}^{-1}$ in the hay treatment during the 2004-08 period (Coppi et al, Chapter 2). Even though a large part of this surplus N was likely organic, mineralization and nitrification would have provided a supply of nitrate over time in the soil (Magesan et al., 1996). These results seem to suggest that the roots of the perennial grass were able to intercept most of the nitrate thereby preventing leaching (Chapter 2, Coppi et al.), especially in the western side of the paddocks where the water table was shallower (Positions 1 and 2) and

the vegetation greater and more dense, compared to the eastern side where the groundwater was deeper (Positions 3 and 4) and the grass productivity and density were lower (data not shown).

Table 4.2 Mean concentration of nitrate-N, chloride, dissolved reactive P and total dissolved P in surface groundwater at 1.5 m depth in hay paddocks (Experiment 1) and associated analysis of variance results for the 2007 to 2009 years.

	Nitrate-N	Dissolved Reactive P	Total Dissolved P	Chloride
	mg L ⁻¹			
<i>Manure treatment</i>				
Control	0.32	0.008a	0.010a	2.0b
Single	0.32	0.004b	0.008b	8.6a
<i>Position</i>				
1 (west)	0.27c	0.008a	0.011	4.1b
2	0.33b	0.005ab	0.008	2.3d
3	0.37a	0.006ab	0.010	3.7c
4 (east)	0.31b	0.005b	0.007	11.2a
<i>Period</i>				
1 (spring '08)	0.31b	0.004b	0.006bc	4.5b
2 (summer '08)	0.31b	0.005b	0.009b	5.8a
3 (spring '09)	0.29b	0.007a	0.007c	5.2a
4 (summer '09)	0.37a	0.007a	0.013a	5.8a
ANOVA		<i>P > F</i>		
Manure	ns	<0.0001	0.0002	<0.0001
Position	<0.0001	0.02	0.10	<0.0001
Period	<0.0001	<0.0001	<0.0001	0.01
Manure x Position	0.001	0.03	0.08	<0.0001
Manure x Period	ns	<0.10	ns	ns
Position x Period	0.01	ns	ns	0.03
Manure x Position x Period	0.09	ns	ns	<0.0001

N for manure means ranges 109 to 125, N for position means ranges 54 to 63 and N for period means ranges 16 to 96. ns is not significant. Separation of means for effects with $P < 0.10$ was performed with $\alpha = 0.1$ and Tukey-Kramer adjustment for beta error. Means in columns for an effect followed by the same letters are not significantly different. ANOVA was done using log transformed data. Non transformed arithmetic means are shown

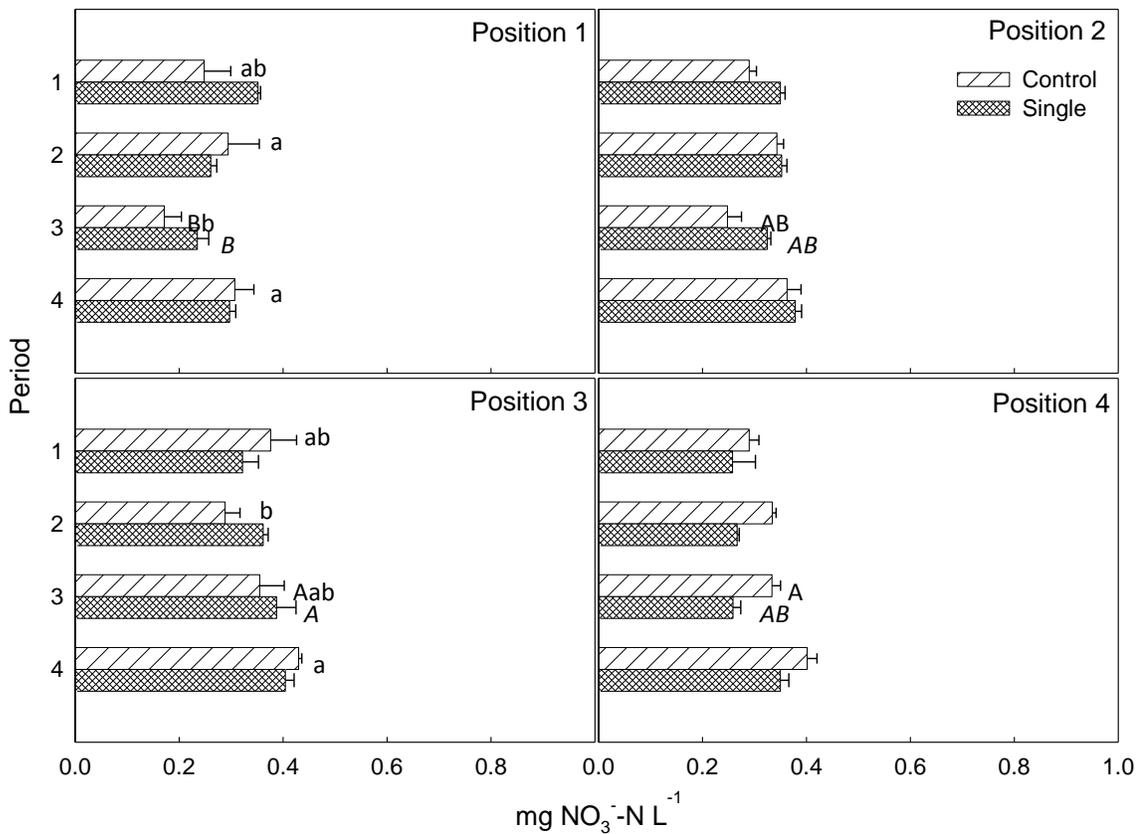


Figure 4.3 Three-way interaction means (manure by position by period) plus one standard error of nitrate concentration in surface groundwater of hay paddocks in Experiment 1. Means followed by the same capital letter for each period across positions are not significantly different. Means followed by the same lower case letter for each manure treatment across periods are not significantly different.

4.4.2.2 Dissolved Phosphorus. Across all sampling dates for both Control and Single treatments, groundwater concentrations of dissolved reactive and total dissolved P were almost always lower than the interim water quality standards in Manitoba ($0.025 \text{ mg P L}^{-1}$ for lake and ponds and 0.05 mg P L^{-1} in all other water streams; Manitoba Conservation, 2001). On three occasions average concentrations of total dissolved P did elevate between 0.025 and $0.030 \text{ mg P L}^{-1}$, one time in the Single and two times in the Control treatment (Fig. 4.2c and 4.2d). It is important to note that Provincial water quality thresholds are for total P, while in the current study only dissolved reactive and total dissolved P was determined, since particulate P content was likely unnaturally increased by the turbidity generated while sampling the groundwater. Many studies report that leaching of P mainly involves dissolved-P while particulate P is predominant in surface runoff (Hooda et al., 2000; Turner and Haygarth, 2000; Haygarth et al., 1998), with the exception of sediment movement in macropore flow in some soils (Kleinman, pers. comm.). Concentrations of total dissolved P and, less visibly, of dissolved reactive P increased after manure application in summer 2008 (Period 2) and even more in summer 2009 (Period 4). These increases seemed to happen when the water table was rising (Fig. 4.2e) as a result of the spring snow melt and of precipitation in late spring and summer.

Dissolved reactive P was generally greater in the Control than the Single treatment ($P < 0.0001$), with low concentrations in both treatments (Table 4.2). This result clearly shows that groundwater inorganic and reactive P was not increased by slurry application. Surpluses of were $44 \text{ kg ha}^{-1} \text{ year}^{-1}$ during 2004-2008. Olsen extractable soil P of 80 and 20 mg P kg^{-1} at 0-5 and 5-30 cm in the fall of 2008 (Coppi et al., Chapter 2) shows that P mainly built up in the surface and this soil layer did not seem involved in downward movement. Similarly, Turtola and Kemppainen (1998) reported little concern for P

accumulation in drainage water under a ley pasture of timothy (*Phleum pratense* L.) and meadow fescue (*Festuca pratensis* Huds.) for a sand soil receiving cattle slurry in surplus of hay removals equal to $18 \text{ kg P ha}^{-1} \text{ year}^{-1}$. Brock et al. (2007) also reported that only 1 % of liquid dairy manure P applied onto intact cores from previously manured coarse loamy soils under an alfalfa (*Medicago sativa* L.) – corn rotation was recovered in the leachate.

In contrast to nitrate trends, dissolved reactive P was the greatest at the west end of the paddocks with shallower water table (Position 1) and the lowest at the east end with deeper water table (Position 4; $P = 0.02$), but again concentrations of P were very small ($< 0.008 \text{ mg P L}^{-1}$). Spring and summer of the very wet 2009 (Periods 3 and 4) had the highest concentrations of dissolved reactive P ($P < 0.0001$; Table 4.2). An interaction of manure x position ($P = 0.03$) was due to that in the Single treatment, dissolved reactive P was not different among positions, while in the Control, the higher concentrations were in the Positions 1 and 3 (Fig. 4.4a). The weak manure x period interaction ($P = 0.099$) was caused by more dissolved reactive P in the Control than Single treatment only in Periods 3 and 4 in 2009 in contrast to the previous periods (Fig. 4.4b).

Treatment means and significance of effects for total dissolved P had a similar pattern to those for dissolved reactive P, with slightly higher P concentration in the Control than Single treatment and in Period 4 than in Period 1. The only difference being that the concentration of total dissolved P in Period 3 was the lowest among sampling periods, while the concentration of dissolved reactive P in Period 3 was the highest (Table 4.2). A weak manure x position interaction ($P = 0.08$) was caused by the Control having the highest total dissolved P across periods in the Positions 1 and 3, while for the Single treatment, P concentrations did not vary among positions and were lower than Control at

Position 1 (Fig. 4.4c). While groundwater nitrate concentration seemed the highest in the eastern part of the paddocks where water table was on average deeper, concentration of dissolved P seemed the highest in the wetter western part with shallow water table. Here, dissolved P movement could have been favored by the wetting-drying cycles and reducing conditions, but, similar to nitrate, concentrations were extremely small. 33 % of total dissolved P was unreactive or organic, but because of the low concentrations, this percentage amounts to an overall mean of just 0.003 mg unreactive-P L⁻¹.

It must be noted that P movement to groundwater is highly dependent on timing relative to storm occurrence (Vadas et al., 2007), thus peaks of P concentration in groundwater may have been missed if storm flow events were not sampled due to the discrete nature of groundwater sampling in this study. Even though care was used to sample groundwater in the field the day after every rainfall event that occurred in the growing seasons of 2008 and 2009, the possibility of having missed storm flow events immediately after a rain cannot be ruled out.

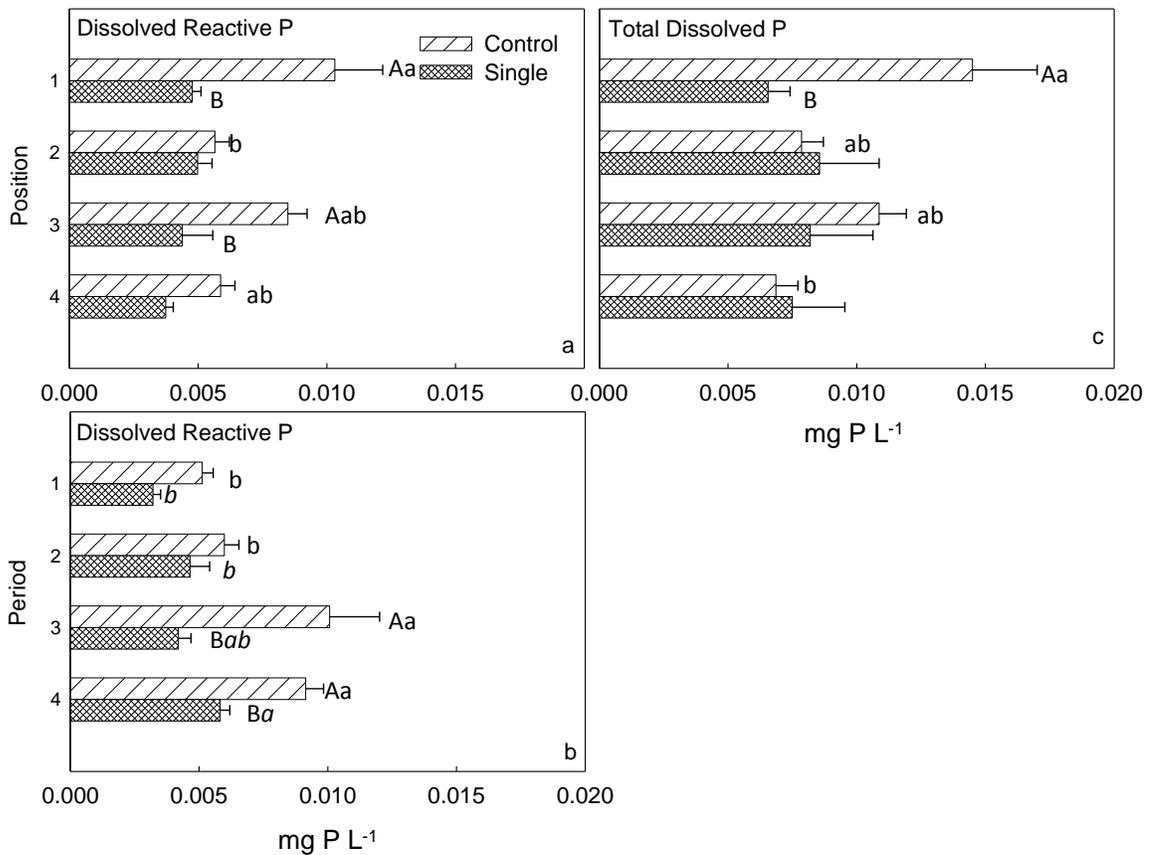


Figure 4.4 Two-way interaction means plus one standard error for hay paddocks in Experiment 1 for manure by position for dissolved reactive P (a) and total dissolved P (c) and for manure by period for dissolved reactive P (b) concentrations in surface groundwater. Means followed by a different capital letter for each position and period across manure treatments are significantly different. Means followed by the same lower case letter for each manure treatment across positions and periods are not significantly different.

Concentrations of dissolved P in groundwater in our study were very low and close to the detection limit for the analytical methods used. Concentrations of water extractable P measured monthly in the vadose zone in 2008 with soil extractions and with a detection limit of $0.067 \text{ mg P kg}^{-1}$ averaged from undetectable to 1 mg P kg^{-1} at 0-30 cm in the Single treatment across positions. They were lower than 0.5 mg P kg^{-1} in the Control at 0-30 cm and undetectable at 30 to 120 cm in both Single and Control treatments (data not shown). It is possible that if P leaching occurred on occasions, concentrations of soil solution P were diluted greatly in groundwater. The concentrations of P in the shallow groundwater measured in this study are one order of magnitude lower than those found by Heathwaite and Dils (2000) at 1.5 m of depth of a shallow groundwater under fertilized ryegrass (*Lolium perenne* L.) grassland in the UK. Specifically, they reported average concentrations, over a two year period, of 0.07 mg P L^{-1} for both dissolved reactive and total dissolved P. That experiment was carried out in an area with P-rich parent material, thus background levels may have been high. Vadas et al. (2007) also reported greater dissolved reactive P concentrations averaging 0.03 mg P L^{-1} at 1.5 of a shallow groundwater under annual crops receiving poultry slurry and soil Melich-3 P concentrations exceeding 350 mg P kg^{-1} . In the current study nitrate concentrations in groundwater were also one to two orders of magnitude lower than most reported by others for arable crops and grasslands (Mantovi et al., 2006; Booltink, 1995; Scanlon et al., 2005), implying that overall the nutrient management of the hay pasture was excellent and the groundwater remained oligotrophic.

4.4.2.3 Chloride. Chloride is a natural tracer widely used for estimating groundwater recharge and nitrate leaching (Scanlon et al., 2005; Dauden and Quilez, 2004; Booltink,

1995; Stites and Kraft, 2001). Chloride present in the slurry was used as a tracer for potential movement of nitrate because of its relative inertness to biological reactions and colloidal adsorption in the soil (Peters and Ratcliff, 1998; White and Broadley, 2001; Cameron et al., 1996).

Chloride concentration in the groundwater of the hay paddocks across positions was relatively uniform during the two monitored years, with spikes occurring after slurry application in summer of both 2008 and 2009 in Periods 2 and 4 (Fig. 4.2b). The standard error of the Single treatment means across positions was unusually high.

The concentration of chloride was greater in the Single than Control treatment, in the Position 4 compared to the others, and in Periods 2, 3 and 4 in summer 2008 and spring and summer 2009 compared to Period 1 in spring 2008 (Table 4.2). A partial chloride mass balance can be calculated from 0-120 cm soil contents in Coppi et al. (Chapter 2), measured as input with slurry minus change in storage from one year to the previous. After subtracting the chloride in the Control from the Single treatment balances, the yearly average surplus chloride in the 0-120 cm soil profile of the Single paddock was $70 \text{ kg Cl}^- \text{ ha}^{-1}$. Assuming low chloride plant absorption (Cameron et al., 1996; McKenzie, 2001), it is likely that a considerable part of this amount was lost to the groundwater. This result suggests there was a potential for nitrate groundwater accumulation at the site, but that did not occur, as previously observed, and it was likely prevented by root absorption. Considerable amounts of N in surplus of above ground crop requirements, especially if provided by swine slurry, can be immobilized into root biomass and organic matter (Jarvis, 1993; Dauden and Quilez, 2004). Roots of the perennial grass were very thick until 35 cm of depth but they could extend as far down as 85 cm on a visual

characterization through soil pits dug in summer 2007. Additionally, root biomass as well as organic carbon was increased by slurry application, (Coppi et al., Chapter 2).

The strong three-way interaction manure x position x period ($P < 0.0001$) clearly shows that chloride in groundwater was increased in the Single treatment compared to the Control in most of the treatment combinations, confirming what we previously reported. However, compared to the other monitoring wells, much higher concentrations were measured constantly across periods in the Position 4 of the Single treatment (Fig. 4.5).

The reason for this difference may be that the well was positioned downstream a horizontal flow line of the groundwater which moves south-west to north-east towards a drain located east of the site (Graham Phipps, personal communication). Thus, chloride may have been increased in the area monitored by this well because of conveyance of horizontal fluxes and from capillary rise.

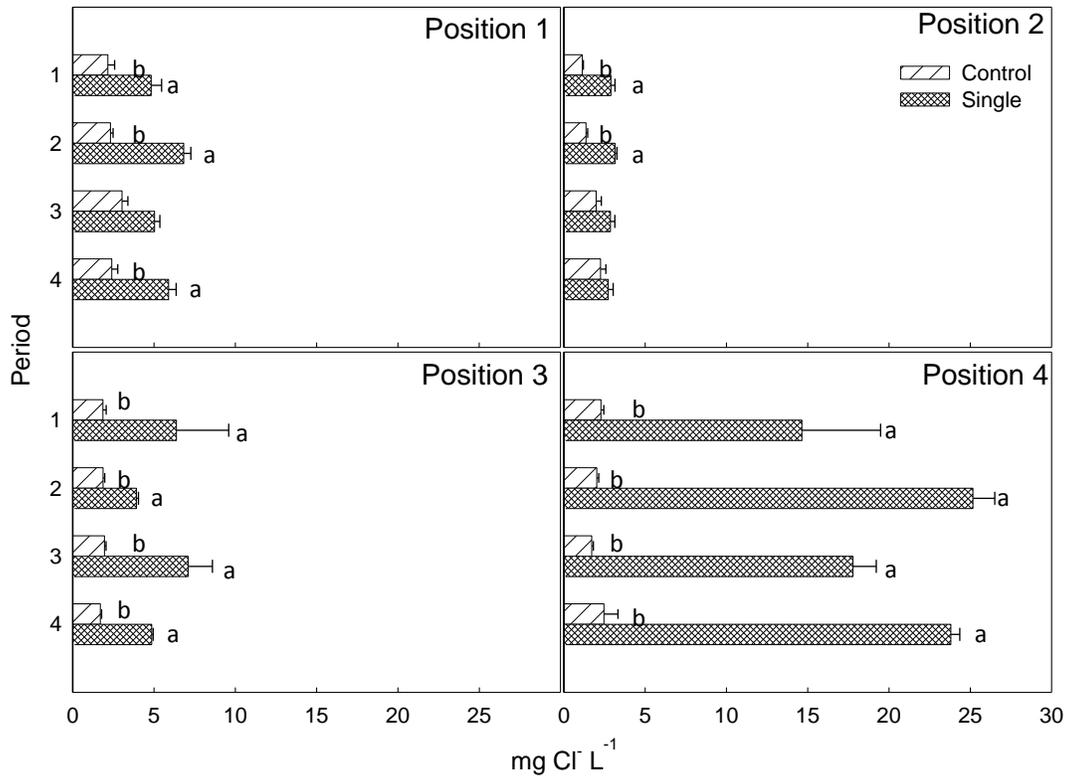


Figure 4.5 Three-way interaction means (manure by depth by period) plus one standard error of chloride concentration in surface groundwater of hay paddocks in Experiment 1. For reasons of clarity, only differences between Control and Single manure treatments per period by position combinations are reported. Means followed by different letters are significantly different.

4.4.2.4 Phosphorus Distribution within Shallow Groundwater. In 2008 in Periods 1 and 2, dissolved reactive P concentrations within shallow ground water depths of 30, 60, 90 and 120 cm from the soil surface across all the four positions were very low and similar to those measured at 1.5 m (data not presented). That year's growing season was 9 mm drier than normal and the saturated zone did not come to within 88 cm of the soil surface when averaged across positions (Fig. 4.2e). But in 2009, both before and after slurry addition in Periods 3 and 4, respectively, high concentrations of P were found across positions at the 30 and 120 cm depths of the Single treatment compared to the Control ($P < 0.0001$; Fig. 4.6). Similarly, Vadas et al. (2007) reported the highest concentrations of dissolved reactive P in shallow groundwater at 0-30 cm, but they also reported a concentration gradient from the soil surface to 1.5 m that was not observed in the current study.

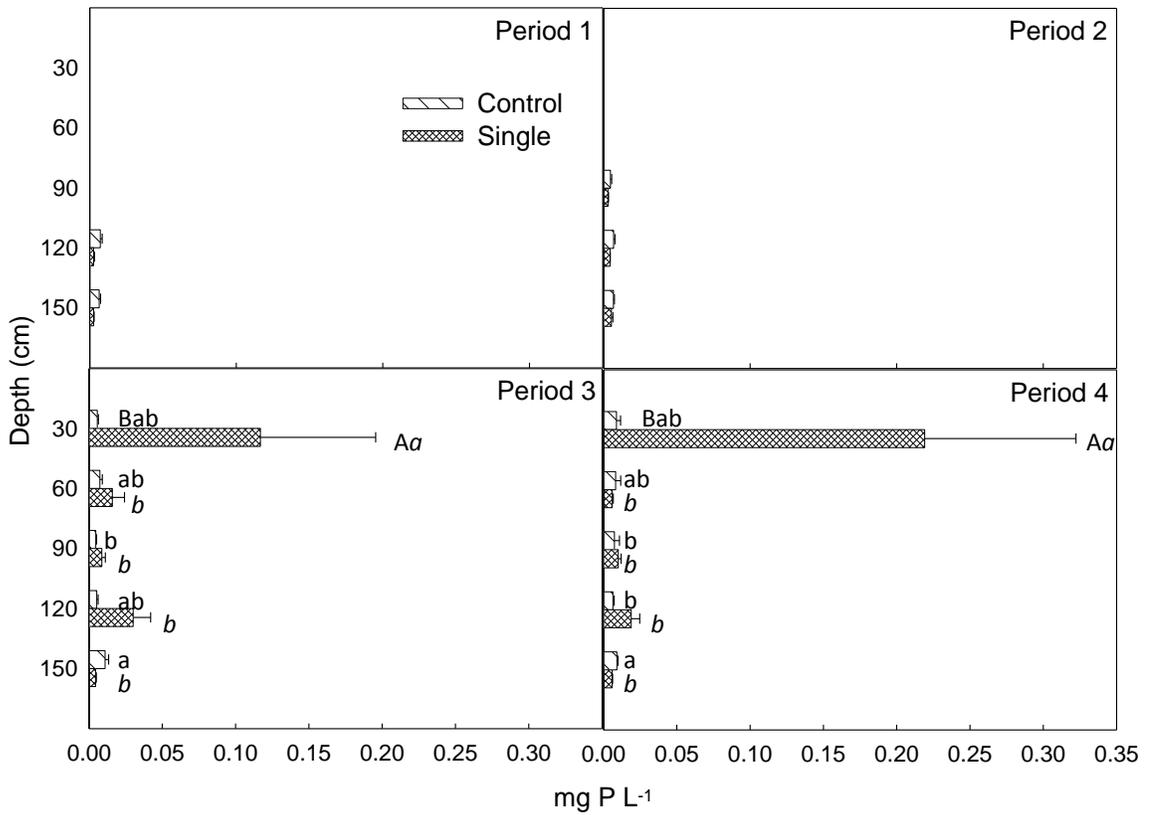


Figure 4.6 Three-way interaction means (manure by period by sampling depth) plus one standard error of dissolved reactive P concentration in shallow groundwater of hay paddocks in Experiment 1. Means of manure treatments followed by a different capital letter for each period by depth combination are significantly different. Means of depths followed by the same lower case letter for each manure treatment by period combination are not significantly different.

Phosphorus previously shown to accumulate in the soil surface (Chapter 2, Coppi et al.) may have been mobilized under reducing conditions caused by saturation (Young and Ross, 2001; Ajmone-Marsan et al., 2006), thus increasing P in soil water at 30 cm. However, if this process occurred, it did not cause accumulation of P in the deeper groundwater, as evident by the lower than Control P concentrations in the Single treatment at 1.5 m (Table 4.2). Any P that may have dissolved at 30 cm was arguably re-adsorbed on the soil matrix of deeper layers (Young and Ross, 2001; Shober and Sims, 2009; Villapando and Graetz, 2001). However, while most studies report that under anoxic conditions P sorption mainly occurs on amorphous, mixed Fe (II)/ Fe(III) oxides, the soil in the current study was very alkaline, so alternative sorption processes active in reducing conditions must have been present. The higher concentrations of dissolved P at 120 cm than at shallower depths could suggest that P moving down was accumulating and being re-adsorbed in this layer or extracted by roots above that layer. It must be noted that redox potential was not measured in the current study, thus we can only speculate that the saturated conditions lead to reducing conditions.

Determination of total dissolved P in the shallow water was done only in 2009 for three sample dates. The results have high variability but are similar to those for dissolved reactive P, with higher P concentration at 30 and 120 cm of the Single treatment compared to the other treatment combinations. Concentrations of dissolved reactive P were also only < 20 % lower than those of total dissolved P, confirming that little of the P in groundwater was organic or non-reactive (Fig. 4.7). The exception was very high total dissolved P concentrations found in the Control treatment at 30 cm at one location for one sampling date of Period 4 in summer 2009.

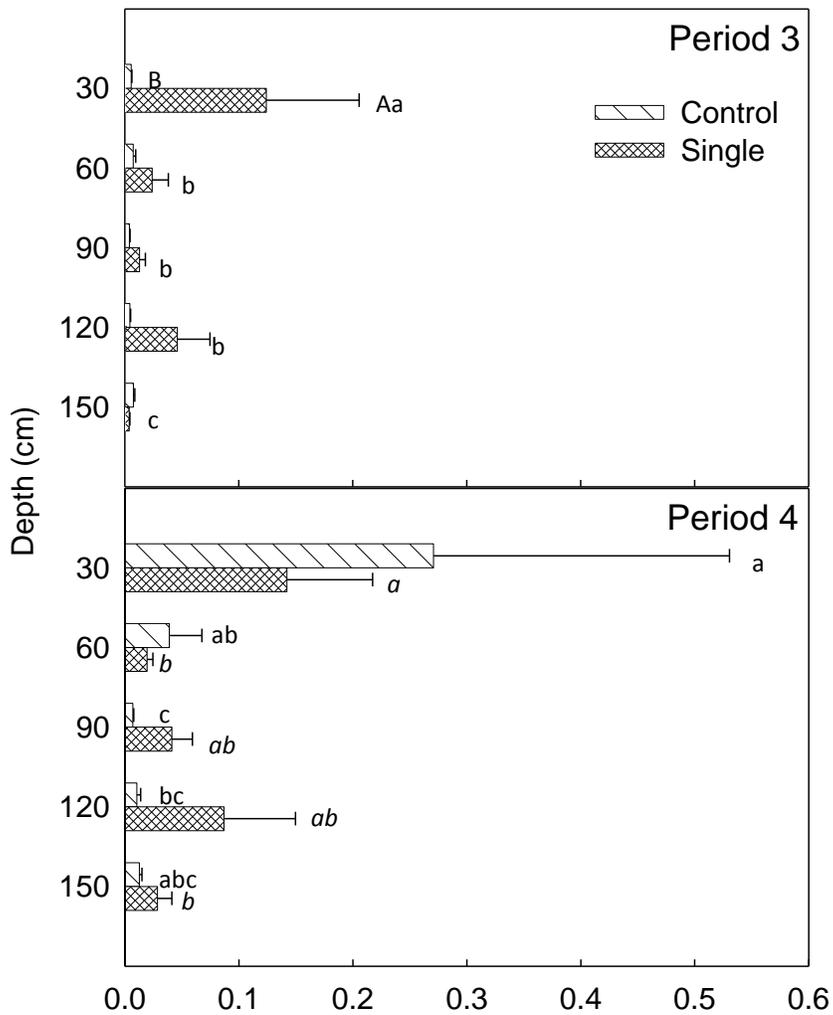


Figure 4.7 Three-way interaction means (manure by period by sampling depth) plus one standard error of total dissolved P concentration distribution in surface groundwater of hay paddocks in Experiment 1. Means of depths followed by the same for each manure treatment by period combination are not significantly different.

4.4.3 Nitrogen and Phosphorus in the Shallow Groundwater for Grazed Forage - Experiment 2.

4.4.3.1 Nitrate. Water table depth in the grazed paddocks in 2008 ranged from 1.5 to 2 m (Fig. 4.8e). Thus, well samples were obtained at depths ranging from 1.5 to 2 m rather than at 1.5 m in that year. Trends of concentrations over time show that nitrate at 1.5 - 2 m of groundwater depth across sampling dates was consistently different among the manure x vegetation treatment combinations. Specifically, Single Bare had concentrations between 10 and 90 mg NO₃⁻-N L⁻¹ and the Control Bare between 3 and 30 mg NO₃⁻-N L⁻¹ (Fig. 4.8a). The Grassed vegetation cover had lower concentrations of nitrate than the Bare treatment: between 1 and 7 mg NO₃⁻-N L⁻¹ in the Single Grassed, and less than 1 mg NO₃⁻-N L⁻¹ in the Control Grassed combinations.

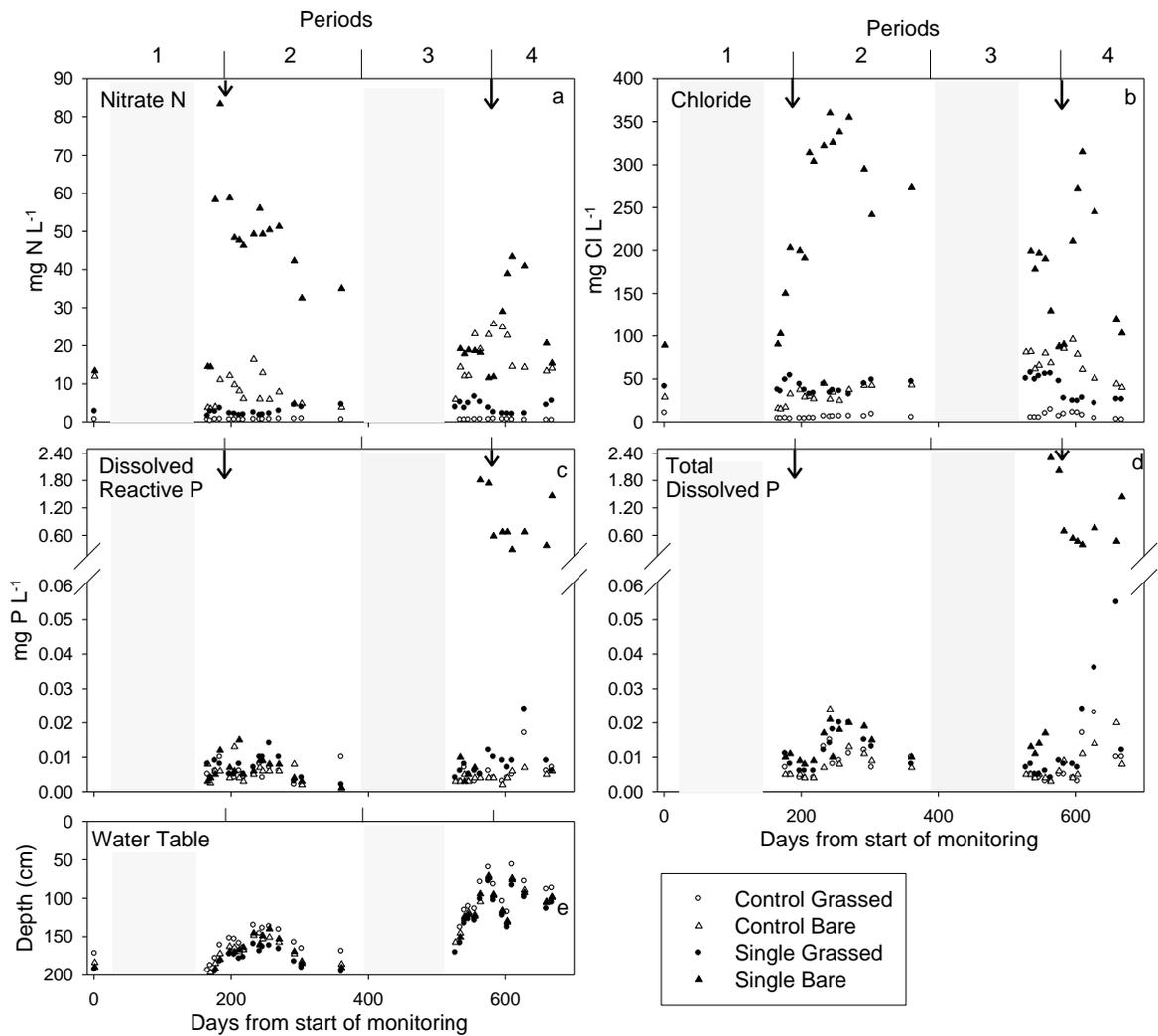


Figure 4.8 Results for (a) nitrate, (b) chloride, (c) dissolved reactive P and (d) total dissolved P concentrations in surface groundwater and (e) depth to water table during Fall 2007 to Summer 2009 for the grazed paddocks in Experiment 2. Shaded areas are winter periods. Arrows indicate time of slurry application. Day of start of monitoring (1) was October 29 2007; last sampling day (668) was August 26 2009. Shown are mean values of two wells per treatment combination.

Table 4.3 Mean concentration of nitrate-N, chloride, dissolved reactive P and total dissolved P in surface groundwater at 1.5 – 2 m of grazed paddocks (Experiment 2) and associated analysis of variance results.

	Nitrate-N	Dissolved Reactive P	Total Dissolved P	Chloride
	mg L ⁻¹			
<i>Manure treatment</i>				
Control	6.5b	0.005b	0.007b	27.4b
Single	17.7a	0.164a	0.185a	112.2a
<i>Vegetation</i>				
<i>n</i>				
Grassed	1.9b	0.007b	0.010b	23.0b
Bare	22.4a	0.162a	0.182a	116.6a
<i>Period</i>				
1	11.7	0.006bc	0.007b	49.3b
2	14.4	0.006c	0.011b	83.3ab
3	9.8	0.152b	0.186b	74.4a
4	12.6	0.174a	0.180a	72.3ab
ANOVA		P > F		
Manure	0.001	0.03	0.02	0.03
Vegetation	0.0002	0.04	<0.0001	<0.0001
Period	ns	<0.0001	0.005	0.06
Manure x Vegetation	ns	0.02	<0.0001	0.03
Manure x Period	ns	<0.0001	<0.0001	0.07
Vegetation x Period	ns	<0.0001	<0.0001	ns
Manure x Vegetation x Period	ns	<0.0001	<0.0001	ns

Number of replicates (N) for manure means ranges 104 to 129, N for vegetation means ranges 105 to 128 and N for period means ranges 16 to 91. ns is not significant. Separation of means for effects with $P < 0.10$ was performed with $\alpha = 0.1$ and Tukey-Kramer adjustment for beta error. Means in columns for an effect followed by the same letter are not significantly different. ANOVA was done using log transformed data.

Mean nitrate concentrations across periods and vegetations were 17.7 and 6.5 mg NO_3^- -N L^{-1} for the Single and Control treatments, respectively ($P = 0.001$). Means across periods and manure treatments were 22.4 and 1.9 mg NO_3^- -N L^{-1} for the Bare and Grassed vegetation covers, respectively ($P = 0.0002$; Table 4.3). Thus, average nitrate concentration across periods was below the drinking water threshold of 10 mg NO_3^- -N L^{-1} for the Control and Grassed treatments, and above the threshold for the Bare and Single treatments. Absence of significance for interactions indicates that the effects of slurry addition and animal waste deposition were additive and not multiplicative. More specifically, the Grassed vegetation, either with or without slurry application mitigated concerns for nitrate accumulation in groundwater. Di and Cameron (2002) and similarly Magesan et al. (1996) reported that even though grazed pastures are at higher risk of leaching than harvested ones because of lower removals, as also shown by the current study, they are less likely to experience nitrate leaching than arable lands. In contrast, Rotz et al. (2009) modeled increased leaching loads in a clay loam Pennsylvanian soil following conversion from crop to grazed pasture system, but the groundwater concentration of nitrate was predicted to be unchanged because of increased percolation and thus dilution under perennial pasture.

The wells in the Bare vegetation in both the Control and especially the Single treatment had concentrations of nitrate in groundwater consistently higher than 10 mg NO_3^- -N L^{-1} (Fig. 4.8a). These results for N groundwater accumulation in Bare areas are consistent with elevated nitrate in soil to 120 m observed at the site by Coppi et al. (Chapter 3). Movement of nitrate may have been promoted by deposition of feces- and urine-N (Ledgard et al., 1999) and subsequent mineralization and nitrification (Di and Cameron, 2002). Steele et al. (1984) and Rotz et al. (2009) suggested that nitrate leaching

in a grazed pasture was mainly occurring from urine affected areas, as N loading from a single cow urine patch is equivalent to approximately 1000 kg N ha⁻¹ (Di and Cameron, 2002). Further in Bare areas, plant uptake of N did not occur to provide a sink for soil nitrate and also transpiration losses of water would have been eliminated, potentially promoting leaching. I conclude that nitrate groundwater accumulation is a concern in Bare areas of livestock loafing especially in manured pastures where the stocking density is possibly higher than in non manured pastures.

Dissolved Phosphorus. Dissolved reactive P in the groundwater of the grazed paddocks was generally lower than 0.025 mg P L⁻¹ throughout the study. However, exceptions were very high concentrations up to 1.8 mg P L⁻¹ within the Single Bare treatment on two dates in spring 2009 (Period 3) and seven dates in summer 2009 (Period 4) when the water table rose from 151 to a 73 cm depth for the soil surface following spring-melt and heavy rains (Fig. 4.8c and 4.8e).

The Single treatment had higher average concentration of dissolved reactive P than the Control treatment across vegetation types and periods. The Bare had higher average dissolved reactive P concentration than Grassed vegetation across manure treatments and periods, and Periods 3 and 4 in 2009 had higher concentrations than Periods 1 and 2 in 2008 (Table 4.3). However, a very strong three-way interaction ($P < 0.0001$) occurred because of increased P concentrations in Period 4 compared to the previous periods, but only for Bare areas of manured pastures. Specifically, dissolved reactive P concentrations were lower than 0.01 mg P L⁻¹ and not different among the Control Grassed, Control Bare and Single Grassed treatment combinations in Periods 1, 2 and 3 (Fig. 4.9a and 4.9b). In contrast, concentrations of dissolved reactive P increased

slightly to 0.10 mg P L^{-1} in the Single Grassed combination in Period 4, likely a result of rains in the summer of 2009 combined with the double amount of P added with slurry in that year. Overall, there was no indication of significant P accumulation in groundwater in the manured or unmanured, grassed pasture. Surpluses were $52 \text{ kg P ha}^{-1} \text{ year}^{-1}$ in 2004-08 and extractable P was 120 and 30 mg P kg^{-1} at 0-5 and 5-30 cm in the fall of 2008 (Coppi et al., Chapter 2). Thus, similarly to Experiment 1, surface accumulated P was not of concern for P groundwater accumulation in the grassed area of grazed pasture. Dissolved P concentrations in this area are much lower than the 1.3 to 0.03 mg P L^{-1} range reported by Sigua et al. (2010) in a shallow groundwater under a fertilized bahiagrass (*Paspalum notatum* Flugge) pasture grazed by cows, even though they measured total P rather than dissolved P. It seems that, overall, P management in the grazed pasture in our study was sustainable in the short term for water quality.

In the Single Bare treatment combination, concentrations across sampling dates increased greatly from $0.006 \text{ mg P L}^{-1}$ in both Periods 1 and 2 to 0.60 and 0.67 mg P L^{-1} in Periods 3 and 4, respectively. These increases were likely related to the wet conditions of 2009 (Fig. 4.9a and Fig. 4.9b). The water troughs and mineral feeders were moved to other sites in the pasture in 2008. Thus, the Bare areas did not receive additional animal depositions in 2009. However, the surface soil of the Single Bare treatment was already saturated with P in 2006 (Coppi et al., Chapter 3).

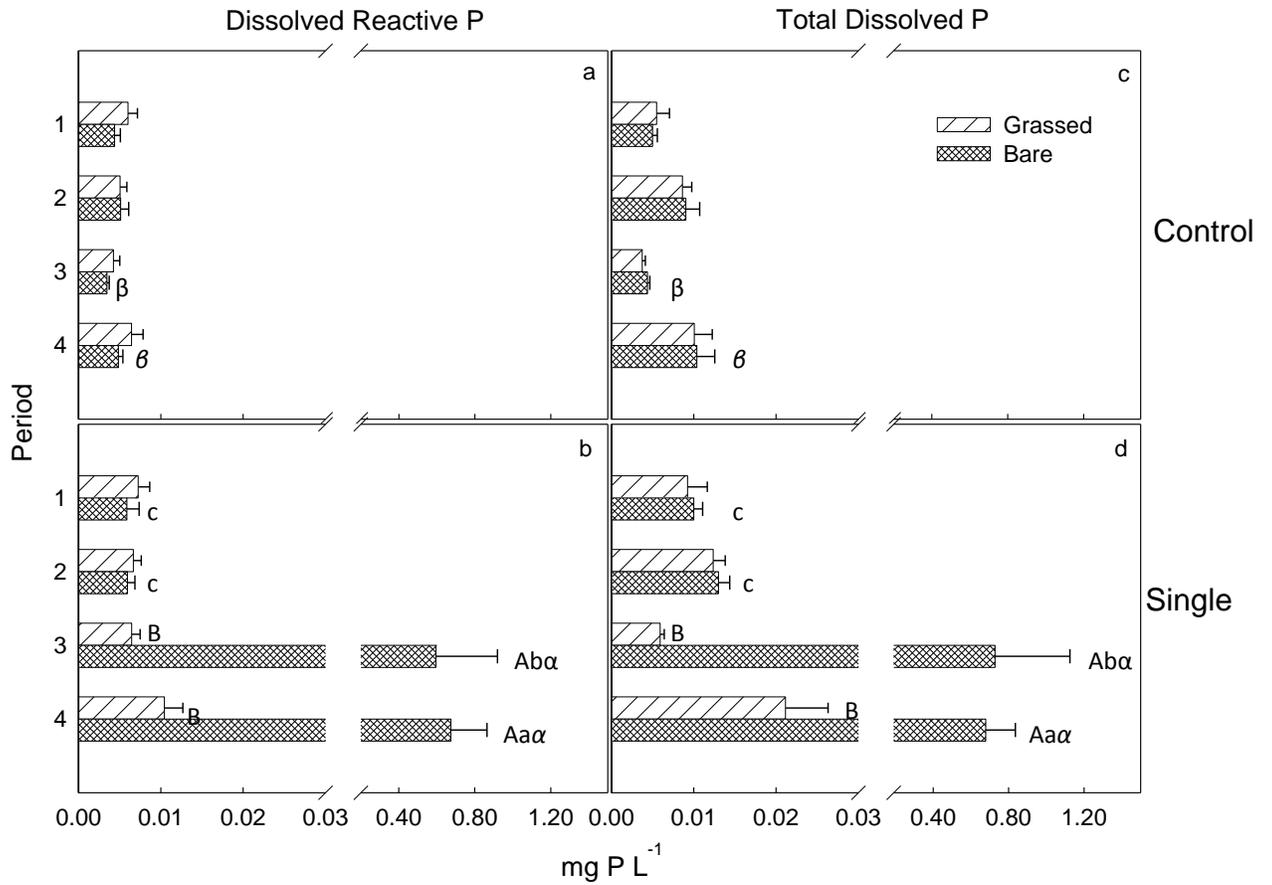


Fig. 4.9. Three-way interaction means (manure by vegetation by period) plus one standard error of dissolved reactive P (a and b) and total dissolved P (c and d) concentrations in the surface groundwater of grazed paddocks in Experiment 2. Means of vegetation treatments per manure treatment by period combinations followed by different capital letters are significantly different. Means of manure treatments per vegetation by period combination followed by different Greek letters are significantly different. Means of periods per manure by vegetation combination followed by the same lower case letter are not significantly different.

The trends over time of dissolved total P in the groundwater of the grazed paddocks were very similar to those of reactive P, with the exception that it seems that the wet conditions of 2009 caused, on occasions, groundwater accumulation of total dissolved P in the Grassed manure pasture. However, these high total dissolved P concentrations only occurred on two sampling dates, and afterwards, in the last sampling date of the experiment, P concentration was back to previous, low levels (Fig. 4.8d and Fig. 4.8e).

The treatment means of total dissolved P (Table 4.3) and the three-way interaction (Fig. 4.9c and 4.9d) had the same trends of those of dissolved reactive P. On average, total dissolved P in groundwater was 12 % greater than dissolved reactive P across all treatments. The relatively small increase indicates that a minor fraction of total dissolved P was organic and/or non reactive. It is evident for this site that reactive and inorganic P fractions were the most mobile and at risk of loss to the environment. This result agrees with the findings of Coppi et al. (Chapter 3), who at the same site reported that organic P fractions in soil were little affected by slurry application and comprised a small portion of labile P. Similarly, deposition of animal feces affected mainly the labile inorganic P fractions rather than the organic and recalcitrant ones. This is different from the observation of Chardon et al. (2007), but in agreement with several studies reporting the vast predominance of dissolved reactive P over unreactive and particulate P in leachate and surface groundwater under clayey to sand grasslands (Gerritse and Schofield, 1989; Turner and Haygarth, 2000; Haygarth et al., 1998).

I postulate that the increased P concentrations in the groundwater of the Single Bare treatment in 2009 are linked to decreased P sorption capacity and high P saturation in the surface soil of these areas already reported in previous years. It is possible that

macropore flow of P rich leachate from the surface soil occurred in the bare areas (Kleinman et al., 2003), considering the gravelly nature of the soil. Particularly, high P saturation coincided with high concentrations of inorganic water soluble P in the surface soil in 2006 (Coppi et al., Chapter 3). Additionally, water extractable soil P measured during the growing season of 2008 ranged from to 26 mg P kg⁻¹ at 0-30 cm in the Single Bare treatment, while concentrations were lower than 3.5 mg P kg⁻¹ at 30-120 cm (data not shown). Macropore flow of P from the saturated surface soil in the Bare areas could by-pass the soil matrix and lead to P leaching to groundwater independently of the saturation status of the deeper soil horizons (Brock et al., 2007). In contrast, soil profile distribution of P is not necessarily an indication of P leaching in soils, as reported by Kleinman et al. (2003). Thus, the high concentration of P at 0-30 cm could already be a signal of risk of P leaching in the coarse soil where macropore flow was likely present. Chardon et al. (2007) found total dissolved P concentrations up to 25 mg P L⁻¹ in leachate from bare sand lysimeters under fresh cow dung, while concentrations were much lower to maximums of 4 and 2 mg P L⁻¹ in grassed lysimeters with and without dung, respectively. Interestingly, in our study, P accumulation in groundwater did not occur in the Bare areas of the Control paddocks, where a lower density of steers would have provided lower amounts of P deposition from feces to these areas. Thus, stocking density and weather conditions seem to be important controls of P losses in congregation areas of pastures.

Results indicate that leaching of P occurred in the Bare areas of the manure grazed paddocks in the wet season of 2009. Water samples obtained from a nearby drainage ditch had relatively high P concentrations both upstream and downstream of the probable location of shallow groundwater flow obtained from flow maps (data not presented).

Kleinman et al. (2010) reported that the largest losses to drainage ditches from heavily manured fields in the Delmarva Peninsula in Maryland, occurred from manure storage locations, indicating risk of P losses from point sources of P accumulation within the field, particularly with storm flow after intense rainfall. In this study, the contribution of the Bare areas to the overall P load to surface water in the ditch was not determined, but was likely small because their position was too far from the ditch to indicate potential direct involvement in P losses through run off or leaching during storm flow after rains.

4.4.3.3 Chloride. Chloride in the groundwater of the grazed paddocks was generally greater in the Single than Control treatment and in Bare than Grassed treatments (Table 4.3). The manure x vegetation interaction ($P = 0.03$) occurred because, across periods, the Control Grassed combination had the lowest chloride concentration of $6 \text{ mg Cl}^- \text{ L}^{-1}$. The Single Grassed and Control Bare combinations were intermediate with 40 and $49 \text{ mg Cl}^- \text{ L}^{-1}$, respectively, and the Single Bare had very high concentration of $184 \text{ mg Cl}^- \text{ L}^{-1}$ (Fig. 4.10a). The trends of chloride concentration over time for these four treatment combinations (Fig. 4.8b) were generally similar to those for nitrate (Fig. 4.8a). However, while nitrate was consistently greater in the Control Bare than in the Single Grassed, chloride concentration was similar in these two treatment combinations. These results seem to confirm the role of the perennial grass in absorbing nitrate and preventing groundwater N accumulation from slurry addition in the Grassed pasture, the potential for which is demonstrated by the occurrence of chloride accumulation. On the contrary, without grass cover nitrate accumulated in groundwater under Bare areas. Thus nitrate concentration in groundwater followed similar trends to those of chloride in the Bare areas.

The manure x period weak interaction ($P = 0.07$) was caused by chloride concentration in the groundwater of the Control treatment across vegetations increasing in Periods 3 and 4 in the wet year 2009 compared to previous periods, while there was not a clear increase for the Single treatment (Fig. 4.10b).

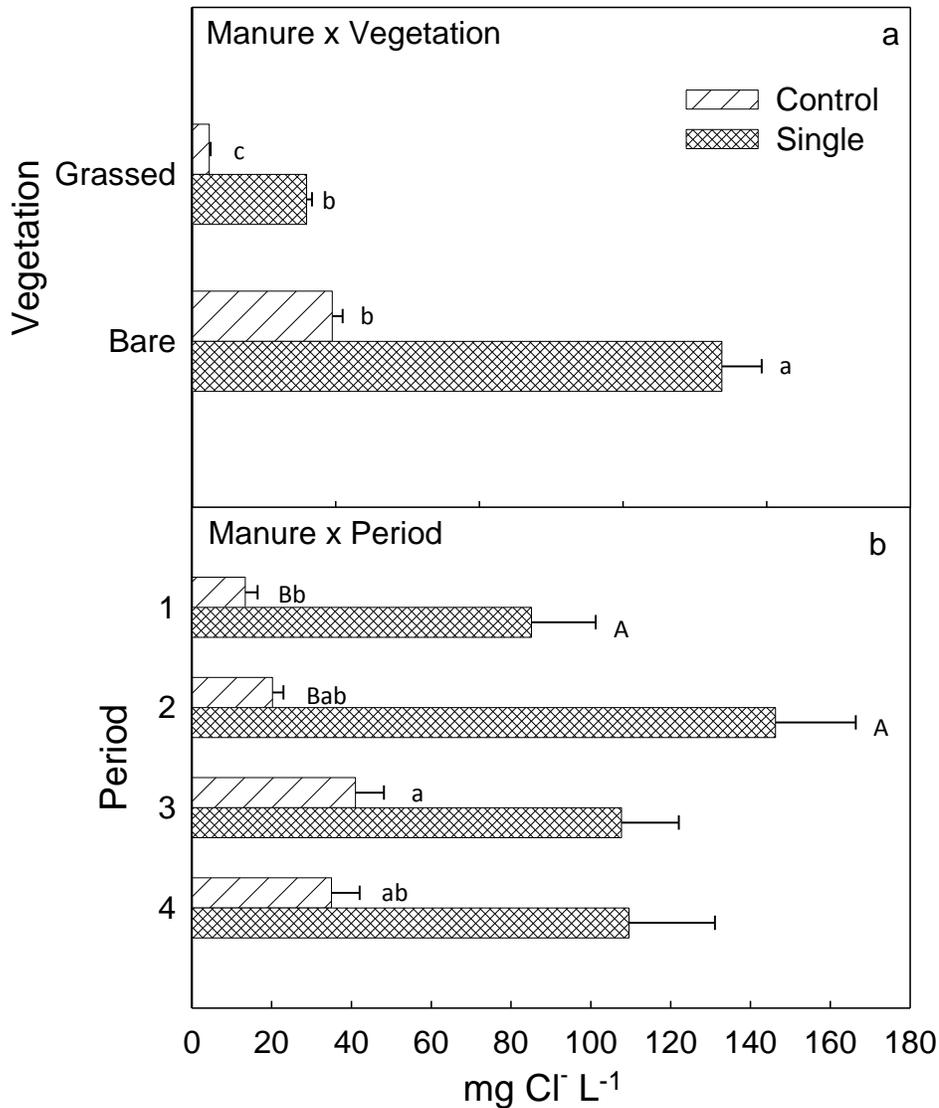


Figure 4.10 Two-way interaction means plus one standard error of chloride concentration in surface groundwater of grazed paddocks in Experiment 2 for (a) manure by vegetation and (b) for manure by period. Means followed by the same letter are not significantly different

4.4.3.4. Phosphorus Distribution Within Shallow Groundwater. The water table in the grazed paddocks was deeper than 60 cm throughout the monitored periods (Fig. 4.3e). Thus, it was not possible to sample the groundwater at the shallowest depths of 30 and 60 cm with the multi-level samplers, but only at 90 and 120 cm of depth, in addition to the 150-200 cm depth sampled in the well pipe.

Generally, concentration of dissolved reactive P tended to decrease with groundwater depth for all the treatments (data not presented). The manure x depth ($P = 0.003$; Fig. 4.11a) and the vegetation x depth ($P = 0.005$; Fig. 4.11b) interactions were caused by dissolved reactive P being more stratified in groundwater of the Single and Bare treatments that had received P additions from slurry or animal depositions, compared to the Control and Grassed treatments, respectively. Total dissolved P was affected by depth similarly to dissolved reactive P (data not shown).

Higher dissolved P concentrations with shallower groundwater depths may be due to P released to the soil solution under reducing conditions (Ajmone-Marsan et al., 2006) in the more P rich surface layers. It is commonly believed that mineralization of organic P and dissolution of iron phosphates following reduction of Fe (III) to Fe (II) are the main processes that cause P release under waterlogging (Scalenghe et al., 2002; Smith et al., 1998; Willet, 1989). However, Ajmone-Marsan et al. (2006) reported that recalcitrant fractions of P such as P bound to Ca and extractable by HCl, as well as residual P, decreased in soil samples incubated under reductive conditions, while soluble, labile and NaOH extractable P increased. Coppi et al. (Chapter 3) found that HCl extractable P was the predominant fraction of soil P at the site, especially in the deeper soil at 30-60 cm. Thus, it is possible that, as the water table rose, P bound to calcium was released. As we did not measure the redox potential of the soil under saturation and the changes in soil P

fractions after saturation, no conclusive remarks on the mechanisms of P release to the rising water table can be drawn. It is also possible that the increased P concentrations at shallower water table depths were because of the shorter leaching path of P from the surface soil, or simply due to gradient driven desorption, along with less dilution from the regional groundwater system.

It seems that dilution of P in groundwater or adsorption of dissolved P in deeper, saturated soil layers occurred as its concentration was eight times lower at 1.5-2 m than at 90 cm. A similar, though weaker, stratification of dissolved P with groundwater depth was reported by Vadas et al. (2007).

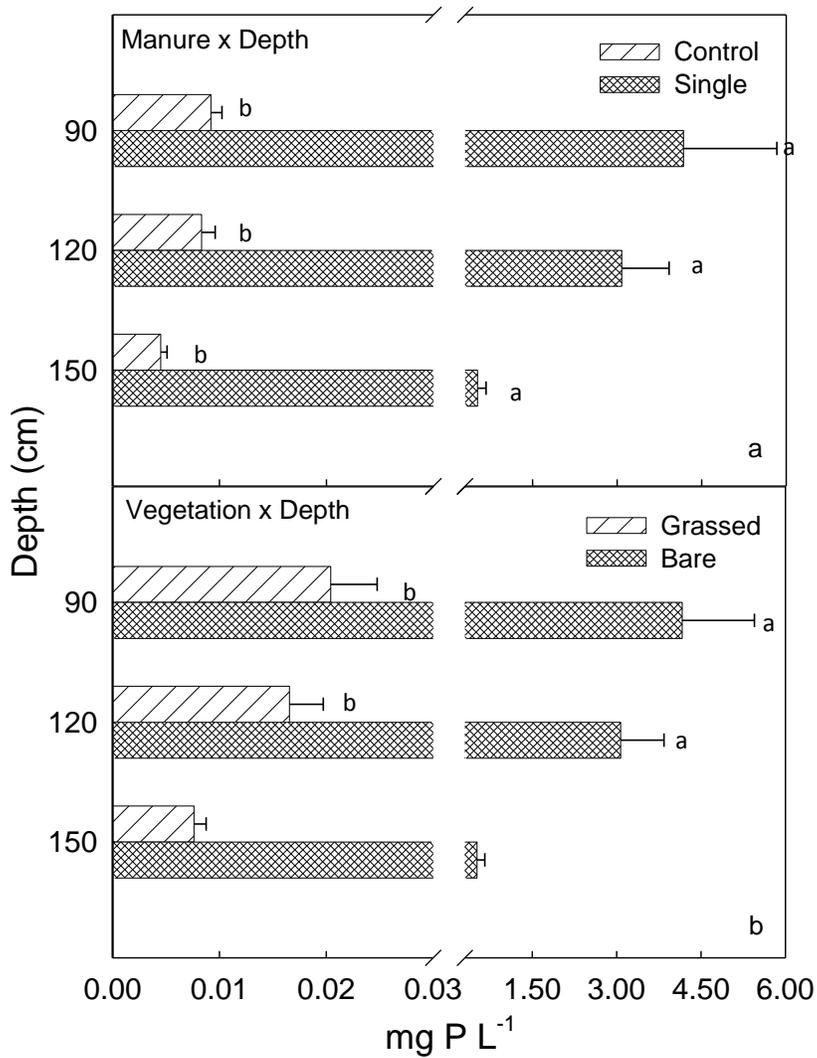


Figure 4.11 Two-way interaction means plus one standard error of dissolved reactive P concentration distribution in surface groundwater of grazed paddocks in Experiment 2 for (a) manure by sampling depth and (b) for vegetation by sampling depth. Means followed by different letters for each depth are significantly different.

4.5 Conclusions

Six years (2004-09) of swine slurry applications to a hay pasture on gravelly sand soil did not cause significant accumulation of nitrate and dissolved P in the shallow groundwater, either in a dry (2008) or wet (2009) year, and either during spring-melt or in the summer growing season. Thus, nitrate and dissolved P leaching from soil to groundwater was not evident, while higher chloride concentration in the groundwater of the manured treated pasture indicates that nitrate leaching was likely prevented by root absorption and P leaching was prevented by soil sorption. The water table rose to the soil surface in spring and summer of 2009. In that year, dissolved P concentrations were high in the shallow groundwater at 30 cm compared to the deeper groundwater.

Following the same slurry treatments of the hay pasture, concentrations of nitrate and dissolved P in shallow groundwater were not of environmental concern in grassed areas a grazed pasture on gravelly sand soil, even though nitrate levels were occasionally close to drinking water thresholds. In contrast, nitrate leaching above environmental concern levels occurred in the Bare areas where livestock congregate around water troughs and mineral feeders. Leaching of large amounts of dissolved P seemed to occur in the groundwater of the Bare areas in the more nutrient rich and intensely grazed manured paddocks. Leaching of P in these areas seemed triggered by large amounts of precipitations and the water table rising to 75 cm from the surface in 2009. In both hay and grazed pastures, most of the dissolved P in groundwater was reactive.

In pastures, especially when fertilized with manures, areas bare of grass where deposition of urine and feces from loafing cattle are allowed to build-up, are of concern for nitrate and dissolved P leaching to the groundwater in coarse soils. It is unknown how much these relatively small areas can contribute to the pasture's overall nutrient loading

into groundwater, as well as what is the effect of denitrification and sorption in deep, calcareous and anaerobic saturated soil and of lateral base and storm flow, on the nitrate and P that reach the groundwater. In any case, prevention measures should involve moving the water troughs and mineral feeders during the grazing season in order to limit accumulation of animal waste and the consequent loss of grass cover.

4.6 Acknowledgements

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5. SYNTHESIS

This dissertation tested the hypothesis that repeated swine (*Sus scrofa* L.) slurry additions based on N requirements to a coarse soil pasture were not a source of concern for nitrogen (N) and phosphorus (P) accumulation in the surface soil, in the soil profile and in the shallow groundwater. Specifically, the results of the research of Wilson et al. (2011) at the La Broquerie Pasture and Swine Manure Management Study Site led to conclude that haying utilization of the grass and application of the slurry split in fall and spring would cause lower accumulation of the nutrients than grazing utilization and single spring application. An unanswered question remained about nutrient accumulation and movement in areas of animal congregation around water troughs and mineral feeders in grazed pastures, as these areas were not included in previous research but were hotspots of animal depositions that could lead to nutrient loss to groundwater and surface water. The objective of the research presented in this dissertation was thus to determine N and P surplus and accumulation in the surface soil, contents and concentrations of extractable N and P in the soil profile to 1.2 m and concentration of nitrate and dissolved P in shallow groundwater at 30 cm incremental depths to 1.5 m. The dissertation also studied the effects of slurry application to the grazed pasture on P fractions in the soil profile to 0.6 m and on P sorption and saturation at 0-5 cm.

As reported in Chapter 2, application of the slurry based on the grass yearly requirements for plant available-N caused N and P surpluses that were in the medium-low range of values reported in literature for grasslands, and also lower than the surpluses that often are reported for manure applications in Western Canada, especially in Alberta (Unterschlutz and Jeffrey, 2001). The surplus N was also presumably in the organic, not

readily available form, thus made available through mineralization only slowly over the years. For this reason, N based slurry additions were not expected to be an environmental concern for N in the hay pasture, as they already represent best management practices for grasslands and swine slurry disposal. Our results confirmed this; there was not extractable N (nitrate- and ammonium-N) accumulation of environment concern either in the surface soil or in the soil profile to 1.2 m in the hay pasture. Considering that just 1.9 % of surplus N was recovered as extractable in the surface 30 cm, and that chloride contents increased in the soil profile with slurry addition compared to the non-manured Control, it is most likely that the surplus N remained in or was converted to organic forms, and as it was being mineralized, was quickly immobilized into the roots and organic biomass. This is suggested by the increased root dry matter and soil organic C in the manured paddocks compared to the Control, as determined in an ancillary experiment.

Calculated surplus N was slightly greater in the grazed than hay treatment, as expected considering the lower removals with animal consumption of the grass compared to hay harvest (Williams and Haynes, 1990; Zhang et al., 2006). However, even in the grazed treatment, measured accumulation of extractable N in soil did not indicate an environmental concern. In Chapter 3, we found the concentration of ammonium-N to be slightly above common values in agricultural soils (Marx et al., 1999) at $12 \text{ mg NH}_4^+ \text{-N kg}^{-1}$ after three years of slurry additions to the grazed pasture, but this value was for the grass sod (0-5 cm) and concentrations in the deeper layers were lower than $3 \text{ mg NH}_4^+ \text{-N kg}^{-1}$. Those concentrations of ammonium-N in the grass sod are not of concern even in the coarse soil, since ammonium is slowly mobile and nitrate from nitrification was likely to be absorbed quickly by the roots or immobilized by microorganisms active in the surface soil. Indeed, nitrate-N concentrations reported in Chapter 3 for the grazed pasture were

lower than $5 \text{ mg NO}_3\text{-N kg}^{-1}$ at all depths, and these concentrations are low (Sullivan and Cogger, 2003; Marx et al., 1999). However, nitrate is a very mobile ion in soil, and a once a year soil testing was deemed not sufficient for ruling out the possibility of nitrate leaching, especially considering that such an overwhelmingly high percentage of surplus N was not accounted for in extractable forms. Thus, the shallow groundwater was tested for nitrate. The results in Chapter 4 did not report nitrate concentrations in groundwater higher than the drinking standard of $10 \text{ mg NO}_3\text{-N L}^{-1}$ (USEPA, 1987) in either the hay or grazed grassed pasture following swine addition, even though concentrations in the grazed pasture were one order of magnitude greater than in the hay pasture. Low concentrations were also found in the last sampling season with very humid conditions caused by higher than normal rainfall, and with water table reaching the soil surface in the hay paddocks and as high as 50 cm in the grazed paddocks.

Of greater environmental concern were the bare areas around water troughs and mineral supplements in the grazed pasture, as reported in Chapter 3. Here, ammonium-N concentrations were lower than in the grassed areas. However, nitrate accumulated following nitrification because of the lack of absorption by plants. Nitrate-N was increased compared to the grassed area of the pasture to 1.2 m, even though concentrations did not exceed $13 \text{ mg NO}_3\text{-N kg}^{-1}$, their distribution in the soil profile was a concern. Another finding was that nitrate was greater in the bare areas of the manured than of the non-manured pasture. These areas did not receive swine slurry additions after 2004 as they were skipped during application. Instead, nutrient accumulation was an effect of the redistribution of nutrients caused by the cattle grazing the pasture and then congregating and spending most of the time in these areas. Because the largest part of nutrients ingested by grazing are re-emitted with urine and feces (Williams and Haynes,

1990), nutrients were redistributed from the grassed area of the pasture to these small spots. As grass in the slurry amended pasture had higher concentration of nutrients and its productivity was increased, it could sustain a higher stocking density (Wilson et al., 2010). Thus, the amount of nutrients deposited through waste by the livestock was greater in the bare areas of the manured pasture. The risk of nitrate loss was confirmed by the findings of Chapter 4. Nitrate-N concentration in groundwater exceeded $10 \text{ mg NO}_3\text{-N L}^{-1}$ in the bare areas, and it was greater in the bare areas of the manured than of the non-manured pasture.

Finally, for N, the findings in Chapter 2 show that there was no difference in calculated surplus and measured accumulation in soil following single spring or split fall-spring application, similarly to what reported for N use efficiency by Wilson et al. (2011) in the same site.

Swine slurry additions for five years at N requirement rates for the grass caused calculated surpluses of P that were intermediate compared to values reported in literature and for Western Canada, as reported in Chapter 2. The surplus of total P was expected following N based applications, since the N:P ratio of the slurry was lower than the ratio of the grass. The slurry was obtained from hog operations using phytase in feed to enhance P absorption by the hogs. However, the N:P ratio of the slurry was still lower than the grass ratio since the slurry was obtained from the P rich primary cell of the storage lagoon. Following manure addition, extractable P accumulated in the soil profile to 1.2 m compared to the non-manured Control, with contents lower than 10 kg P ha^{-1} . Part of this P was likely derived by cross-contamination, as sampling was done using a flight rather than a core auger because of the stoniness of the soil. Accumulation of extractable P occurred in the surface 30 cm of soil, with the highest accumulation rate for

the 0-5 cm layer. Differently than for N, calculated surplus and measured accumulation of P were greater in the grazed than in the hay pasture because of lower removals in the former, and with single application compared to split because the spring slurry had a consistently higher P concentration than the fall slurry. This difference was due to that fall slurry was diluted by the use of water for washing the barns during summer, while spring manure had higher solid content following the winter freeze. The ratio of surplus to accumulation rate for P was slightly lower than values reported by others, indicating that accumulation in the tested soil was rather fast, likely because of its alkaline pH, dominated by Ca and Mg retention of P rather than by Al and Fe. The effect of grazing and single application treatments were thus tested further in Chapter 3 following three years of swine additions. In this study, with a more accurate sampling methodology that avoided sample contamination, it was confirmed that P concentration was not affected by swine addition deeper than 30 cm (main rooting zone). It was also found that swine addition increased the soil's labile-P fractions (water and bicarbonate extractable) more than the recalcitrant fractions (sodium hydroxide and hydrochloric acid extractable, and residual), indicating a potential shift in the equilibrium of soil P pools towards environmental mobility. However, because P seemed to accumulate only in the surface soil, we did not expect P concentration in groundwater to be increased by slurry addition by matrix flow, and because of the very coarse texture of the soil and shallow rooting, we did not expect macropore flow to be prevalent. This hypothesis was confirmed in Chapter 4, with dissolved P concentrations in groundwater being lower than the environmental thresholds in both grass covered hay and grazed pastures with either dry or wet soil conditions. There were indications that shallow groundwater could mobilize sorbed P in the surface (0-30cm) soil; however, this process was occasional and did not seem to affect

the concentration of P in the deeper groundwater. Another finding was that organic P in soil was largely unaffected by swine additions, and organic P in groundwater was a very minor component of total dissolved P.

Similarly to the results for N, a special case needs to be made for the bare areas in the grazed pasture. Findings in Chapter 3 report that extractable P accumulated in greater concentration in the bare areas than in the grassed pasture, even though seemingly only in the surface 30 cm of soil. The deposition of animal waste in these areas had a similar effect on P fractions than the application of slurry to the grassed pasture, with the labile fractions being increased more than the recalcitrant ones, and organic P being largely unaffected. However, the P sorption capacity, determined in the surface 5 cm of soil, was decreased in the bare areas compared to the grassed pasture, indicating that the accumulation of P with lack of removals was affecting the capacity of the soil to retain additional P. As a result, the surface soil of the bare areas was saturated in regard to P, with the degree of P saturation values in the bare areas of the manure pasture being considerably higher than the values at which, in literature, it is reported that P downward movement can occur. Furthermore, concentration of water soluble P was increasing proportionally to the increase of P saturation levels in the bare areas. These results suggested there was a risk of P movement to the groundwater in these areas. The hypothesis was indeed proven in Chapter 4, with very high dissolved P concentrations reported in the groundwater in the bare areas of the manure pasture during a wet season characterized by greater than average precipitation and by the water table rising to 75 cm from the soil surface on two occasions.

5.1 Implications and Suggestions for Best Management Practices

The research presented in this dissertation suggests that N based swine slurry applications are an acceptable management practice for grasslands in the short and medium term. The environmental soundness of the application was tested in a very coarse soil with the shallow water table. Even under such favourable conditions for subsurface movement of N and P, the nutrient accumulation in the soil profile was not environmentally significant over the study period compared to the non-manured Control, and groundwater concentrations were under the environmental thresholds. Spring melt and a wet season with greater than normal rainfall and surface level water table also did not result in concentrations above the thresholds.

Phosphorus surplus and accumulation were greater with single spring than split fall-spring application, but the difference was due to the higher P concentration of spring than fall manure. Thus, there was substantially no difference in nutrient soil accumulation between the two methods, as N surplus and accumulation and the effect of surplus P on P accumulation were not different. Considering the higher operational costs, a split fall-spring application is not recommended. However, improved nutrient utilization could result with split spring-summer applications for the hay pasture, with the summer application occurring after the first hay cut. Further research in this direction is suggested.

Of the two harvest strategies that were compared in this research, haying, as expected, provided greater nutrient removals than grazing, because of the low rate of nutrient incorporation into animal biomass (Zhang et al., 2006; Wilson et al., 2011). Haying allowed for lower N and P accumulation in both the surface and deeper soil, and lower nitrate concentrations in groundwater, while P concentrations were equally low for both harvest strategies. Harvesting the grass as hay allows the export of a considerable

part of the manure added nutrients out of the field. However, since the hay is likely to be consumed by overwintering cattle nearby, the equalization of additions and removals at the field scale does not solve the outstanding issue of nutrient imbalance at the regional scale, as more nutrients are imported with swine feeds than exported through animal production. The solution of regional nutrient imbalances would have to address the cost of transportation of manures and hay. Treatments of manure such as soil-liquid separation and pelletization could increase the range of economically viable transportation of manure, and these methods should be studied for implementation in Manitoba.

The results of the research also indicate that P surface accumulation is not preventable with N based manure applications, as long as the N:P ratio of the manure is lower than that of the crop. Accumulation of P was linear, meaning that further years of slurry addition are likely to cause over saturation of soil sorption capacity for P. How fast the oversaturation occurs depends on the sorption capacity of the soil, as well as on the surplus P added. In Manitoba, a soil test P threshold has been set at 60 mg P kg⁻¹ for the surface 0-15 cm of soil, above which manure applications are to be no greater than two times the P requirements of the crop (Nutrient Management Regulation, 2008). The shift would require coupling slurry with N fertilizer additions to meet the N requirements, thus increasing operational costs for farmers. As slurry applications affect mainly labile, inorganic P fractions, we suggest that soil test P is adequate for determination of P concentration in soils with similar properties at the study site, especially considering that the Olsen-P bicarbonate testing method common in Manitoba extracts P approximately corresponding to the labile P fractions in soil. Even though soil test P has been found to correlate well with soil P saturation (Kleinman et al., 1999), a fixed threshold of soil test P could be replaced by the determination of the soil P saturation, as soils vary greatly in

their ability to bind P and thus the same value of a soil test indicates different saturation conditions in different soils. Furthermore, soil P saturation status controls the concentration of the water-soluble P, which is the most mobile form of P in leaching prone soils. However, the degree of P saturation is not a well established measurement of soil P status in Manitoba (Kumaragamage et al., 2011). Even more appropriate, if more laborious, would be the application of a P Index system to take into account sources and transport factors of P (Sharpley et al., 2011). This solution would allow for the characterization of areas that are hydrologically connected to surface waters through either surface or subsurface transfer, as these areas should be the focus of the most critical management.

The areas of bare soil around water troughs and mineral supplement in the grazed pasture were the main environmental concern identified in this research. Grazing cattle contributed to a redistribution of nutrients from the main grassed area of the pasture to these relatively small spots solely through waste deposition. After just three years of grazing and slurry additions, nitrate and extractable P accumulated in the surface 30 cm of soil in greater concentration than in the pasture, P sorption capacity was decreased at 0-5 cm and nitrate also accumulated in the soil profile to 1.2 m. Because of the 2.6 fold greater grazing density in the manure amended than in the non amended pasture, the bare areas in the manure amended grazed pasture were particularly nutrient rich, and at 0-5 cm the degree of P saturation was increased above an environmental concern level, triggering the release of P to soil solution. Similarly, to slurry addition, waste deposition also increased the proportion of labile over recalcitrant P, further increasing the environmental risks associated with these areas. Nitrate concentration in the groundwater below bare areas was above the drinking water threshold both in the dry and wet years, and both in

the spring and in summer during the grazing season. Differently, from N, dissolved P in groundwater was above environmental thresholds only in the bare areas of the slurry amended pasture, and only in the wet year. These results suggest that bare areas of cattle congregation in grazed pastures could result in nitrate leaching with or without slurry addition. However, it has to be considered that the soil in our study was very coarse and prone to vertical movement of nitrate, and the bare areas of the non-manured Control paddocks were rather small. Thus, their contribution to the overall nutrient load into groundwater from the pasture was likely small. Bare areas of the more densely grazed manured pasture not only had nitrate concentrations in groundwater of one order of magnitude greater than the bare areas of the Control, but were also considerably larger. Thus, we conclude that bare areas of grazed pastures, which may develop around water troughs, mineral feeders, shaded areas and other areas where cattle congregate, are a minor concern for nitrate leaching if the pasture is unmanured, while become a considerable environmental risk if the pasture is manured and has an increased grazing density according to its greater productivity. These areas can also become hotspots of P leaching in coarse soils in wet seasons. Moreover, the bare areas remain without a grass cover and at risk of nutrient losses outside of the grazing season. Best management practices should be introduced for grazed pastures to limit the grazing density of manured pastures, especially in wet season and in coarse textured soil pastures or to avoid the cattle from congregating in the same spots. The solution for water troughs and mineral feeders could be quite simple by moving them during the grazing season at regular intervals of time. While this practice increases the costs of grazing operations because of the need of extra water lines and workforce, it would prevent nutrient losses especially in coarse and manure amended soils.

5.2 Recommendations for Further Research

Following the research presented in this dissertation, the study of nutrient dynamics at the La Broquerie Pasture and Swine Manure Management Study Site has proceeded with focus on application of slurry obtained from the tertiary cell of the lagoon. The objective is to use the more diluted slurry with a higher N:P ratio compared to plants to create an annual deficit of P in soil and to determine the effectiveness of soil P removal with hay harvest and the role of the so called “legacy P” from past additions in the low P load pasture system (Kleinman et al., 2011). Another study in the grazed pasture has tested the effectiveness of periodical movement of the water troughs and mineral supplements during the grazing season in limiting the formation and size of bare areas of nutrient accumulation.

In the current research, determination of P saturation was done only for the surface 5 cm of soil. Since saturation of the deeper layers may indicate decreased capacity of the soil to immobilize P flowing through the soil matrix, and increased risk of P desorption during recharge events (Brock et al., 2007), determination of P saturation in the deeper soil, especially of the P rich bare areas in grazed pastures, would be warranted.

In the current research, we did not quantify the leaching volume, and the N and P concentration in groundwater was likely diluted by the surrounding water mass. This is especially true for the bare areas, as the likely increased percolation in these areas with no vegetation would mitigate the groundwater concentration of large nutrient loads (Roatz et al., 2009). This approach allowed for a direct estimation of the risk of groundwater contamination with excess N and P. However, it will be important in the future to quantify the contribution of bare areas to the overall nutrient load in grazed pastures.

Their contribution depends on their size as well as on the nutrient accumulation occurring, on the stocking density of the pasture and on the water and solute movement dynamics. Actual measurements of leachate volumes through subsurface drainage or lysimeters, or the use of modelling tools to estimate percolation and recharge would be required to quantify nutrient loads. For this purpose, a large dataset of vadose zone soil water extractions, soil moisture with tensiometers and TDR probes, soil temperature with thermocouples and soil vegetation cover has been collected in 2007 to 2009, as well as water depth from two shallow piezometers to monitor vertical pressure head gradients.

Finally, the groundwater flow net of the La Broquerie site showed that the groundwater was flowing horizontally into a drainage ditch. Further studies determining the hydrological connectivity between the field site and the catchment, as well as the water quality of the surface waters would be warranted in order to determine how fast and to what extent groundwater contamination results in nutrient enrichment of surface waters (Kleinman et al., 2007). In this sense, samples from a system of previously existing background and site-wide monitoring wells and from three locations in the drainage ditch have been collected in 2003 to 2009, and the ditch storm flow has been monitored with staff gauges. Additionally, nutrient transport through the saturated zone often involves denitrification processes that could result in gaseous losses of nitrate, especially in unconfined, shallow aquifers and under vegetated buffer strips (Osborne and Kovacic, 1993), as well as deep sorption of P (Villapando and Graetz, 2001). These processes can reduce dissolved N and P concentration in groundwater before flow into surface waters. Moreover, surface waters in Manitoba are often naturally rich in nutrients and sediments, and it is not clear what levels of groundwater concentration would result in a sensible increase in surface water concentrations.

5.3 References

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APPENDICES

APPENDIX I. Additional Tables and Figures for Chapter 2

Appendix I.1 Mean and ± 1 standard error (S.E., N = 12) of the mean of soil chemical and physical properties at the study site determined in 2003 prior to start of treatments.

Soil Depth	Parameter	pH [†]	EC dS m ⁻¹	CEC mol _c kg ⁻¹	Organic carbon g kg ⁻¹	CaCO ₃ g kg ⁻¹	Total- N	NH ₄ ⁺ - N	NO ₃ ⁻ - N	Olsen- P mg kg ⁻¹	Cl ⁻	K ⁺	Sand	Silt	Clay	Texture‡	Stones % weight	Bulk density§ Mg m ⁻³
cm													% weight of fine soil¶			Class		
0-30	Mean	7.9	0.13	15.5	12.6	19.6	0.77	2.0	1.7	4.4	1.0	44.4	82	10	8	Gravelly Loamy Sand	31	1.68
	S.E.	0.0	0.01	0.8	0.9	2.5	0.08	0.2	0.2	0.8	0.1	2.0	1	1	0		2	0.11
30-60	Mean	8.3	0.08	15.1	9.3	47.4	0.32	0.9	1.2	1.9	2.0	21.9	87	10	3	Very Gravelly Sand	41	2.04
	S.E.	0.0	0.00	0.9	0.8	4.0	0.06	0.2	0.1	1.0	0.2	1.0	2	2	0		2	0.12
60-90	Mean	8.4	0.07	15.9	6.9	73.6	0.17	0.4	0.7	0.5	3.1	14.7	93	6	2	Very Gravelly Sand	39	1.94
	S.E.	0.0	0.00	0.6	0.8	4.0	0.05	0.1	0.0	0.0	0.2	1.0	0	0	0		2	0.09
90-120	Mean	8.6	0.06	17.0	6.4	79.3	0.13	0.3	0.5	0.1	3.5	13.3	93	5	2	Very Gravelly Sand	36	1.96
	S.E.	0.0	0.00	0.3	0.6	3.0	0.03	0.1	0.0	0.0	0.2	0.0	0	0	0		2	0.09

†1:1 soil:water ratio.

‡Textural classes according to Canadian Textural classification.

§Bulk density was determined with the excavation method from four soil pits dug in summer 2007. Values reported here are weighted averages of bulk density of two replicates of each pedogenetic horizon in the four pits.

¶ Fine soil is soil that passed through a 1.5 mm mesh.

Appendix I.2 Mean application rates of N, P and Cl⁻ in swine slurry applied at the site over the study period.

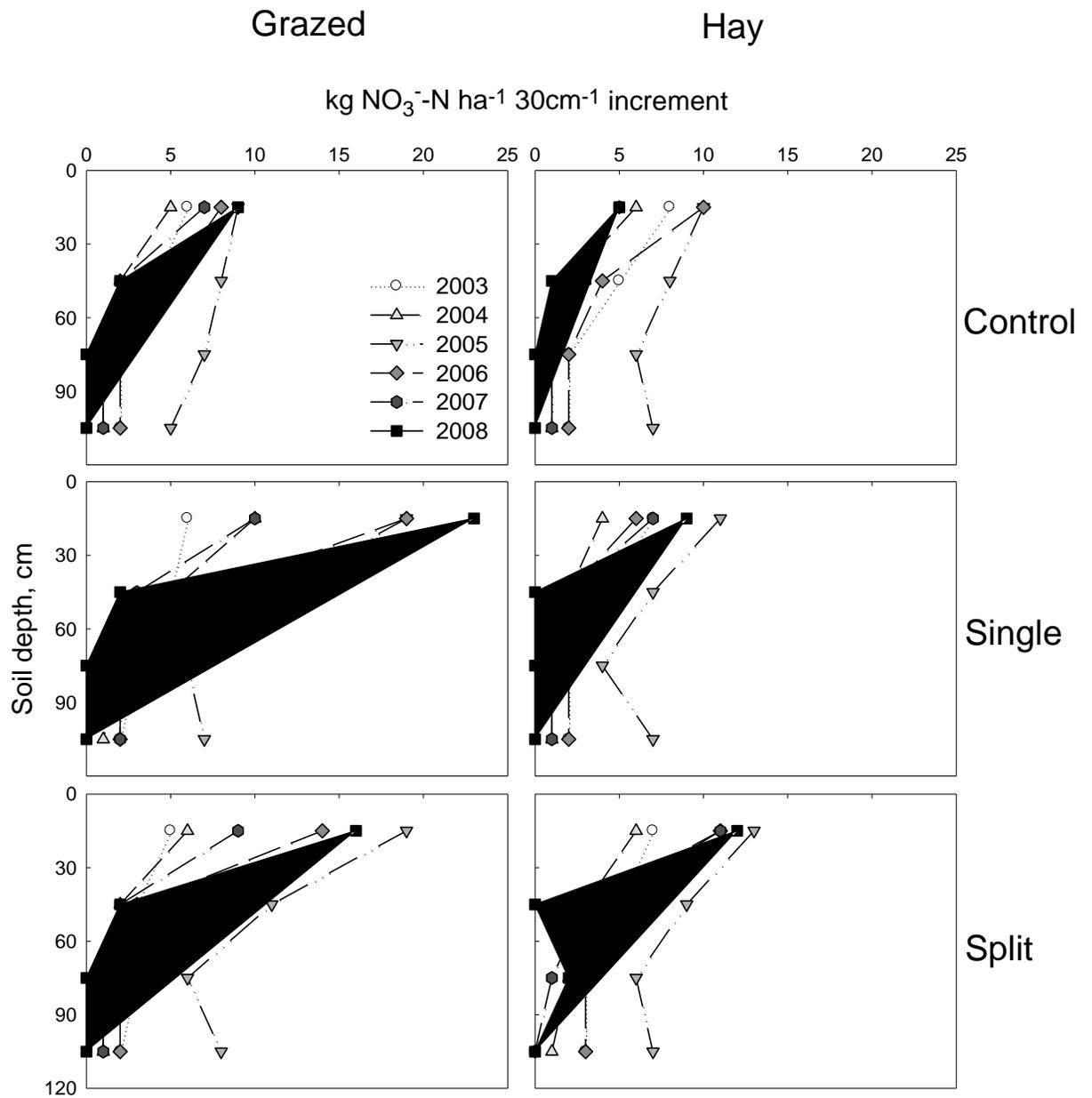
Application rates	-----Single slurry application†-----				-----Split slurry application‡-----			
	Mean	Max	Min	CV	Mean	Max	Min	CV
Manure application rate, '000 L ha ⁻¹	40	53	31	21	45	55	38	18
NO ₃ ⁻ -N, kg N ha ⁻¹	0	0	0	40	0	2	0	167
NH ₃ -N, kg N ha ⁻¹	147	178	128	13	155	169	139	8
Organic-N, kg N ha ⁻¹	78	121	37	47	69	90	51	22
Available-N, kg N ha ⁻¹	130	164	105	17	134	149	117	10
Total Nitrogen, kg N ha ⁻¹	226	299	165	23	224	259	190	12
Total Phosphorus, kg P ha ⁻¹	63	70	56	10	48	53	39	12
Total N:Total P	3.5	4.2	3.0	15	6.0	11.5	4.2	52
Chloride, kg Cl ha ⁻¹ ††	63	75	52	18	75	88	63	17

n = 5

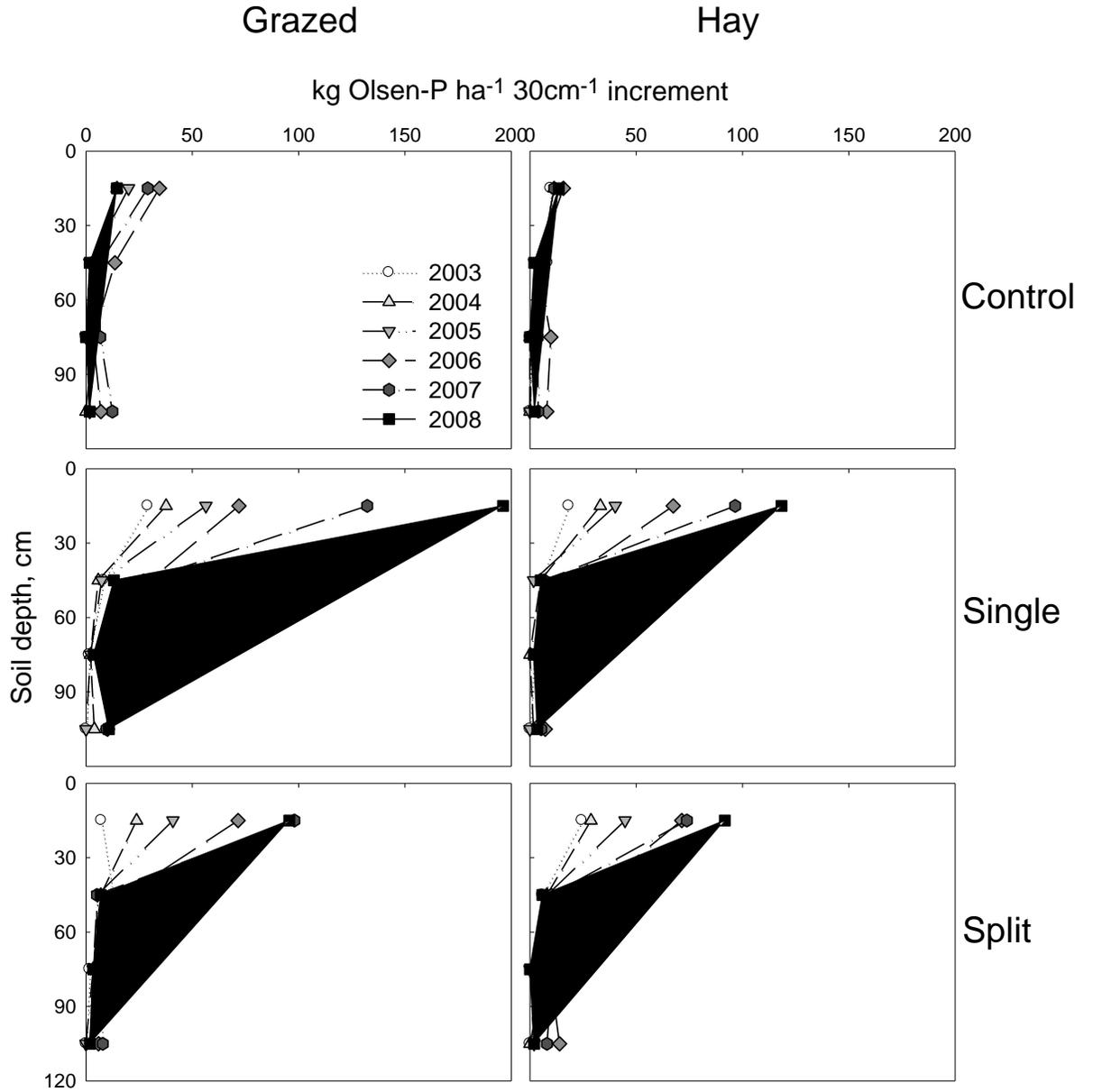
† Single application was manure applied once a year at full rate in spring 2004 to 2008

‡ Split application was manure applied twice a year at half rate in fall and spring from fall 2003 to spring 2008

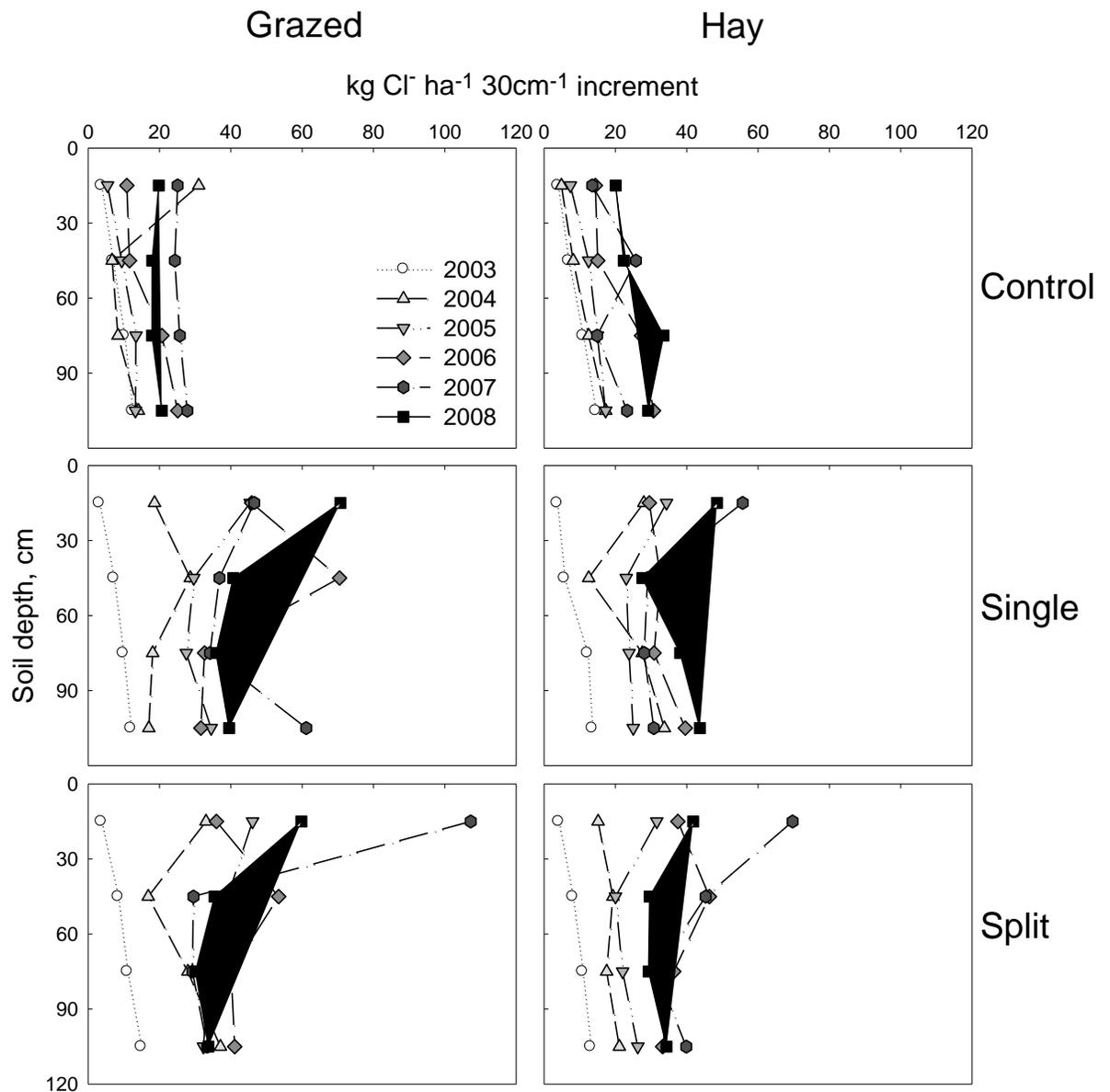
§ Manure analyses for chloride were not available from fall 2003 to spring 2005 (n = 3).



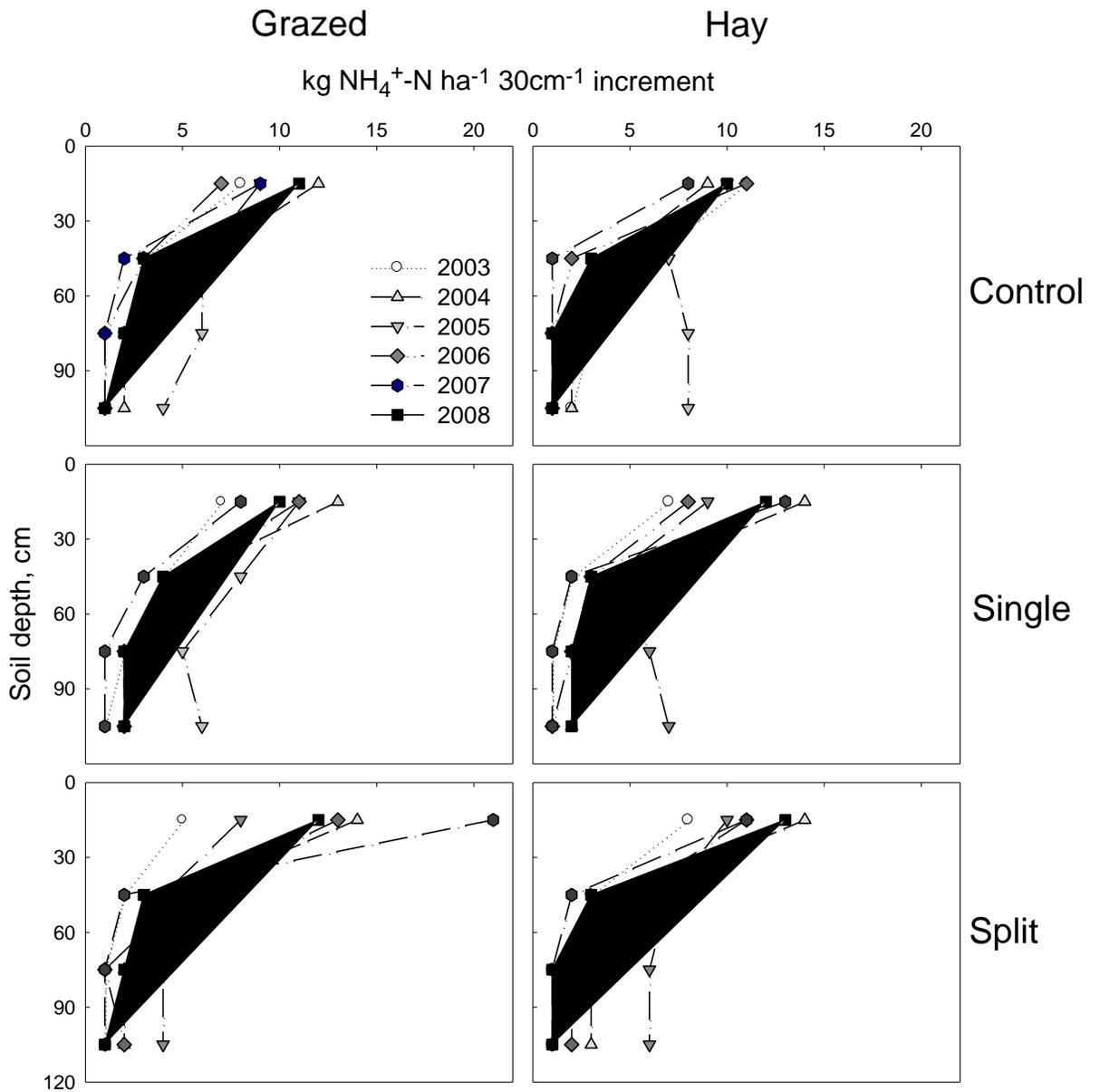
Appendix I.3 Fall extractable-NO₃⁻-N in the soil profile of treatments of combinations of manure (Control, Single and Split) and forage utilization (Grazed and Hay) treatments. Data points are means of two replicate plots.



Appendix I.4 Fall extractable-P (Olsen-P) in the soil profile of treatments of combinations of manure (Control, Single and Split) and forage utilization (Grazed and Hay) treatments. Data points are means of two replicate plots.



Appendix I.5 Fall extractable Cl⁻ in the soil profile of treatments of combinations of manure (Control, Single and Split) and forage utilization (Grazed and Hay) treatments. Data points are averages of two replicate plots.



Appendix I.6 Fall extractable-NH₄⁺-N in the soil profile of treatments of combinations of manure (Control, Single and Split) and forage utilization (Grazed and Hay) treatments. Data points are means of two replicate plots

Appendix I.7 Three-way analysis of variance of fall extractable ammonium-N and extractable-N (sum of ammonium-N and nitrate-N) content in 30 cm depth intervals to 120 cm. Treatments were combinations of manure (Control, Single and Split), forage utilization type (Grazed and Hay), and year (2004 to 2008).

Main Effect Means	NH ₄ ⁺ -N				Extractable-N			
	Depth, cm							
	0-30	30-60	60-90	90-120	0-30	30-60	60-90	90-120
	kg ha ⁻¹							
<i>Manure</i>								
Control	9.8	3.7	2.5	2.3	17.3a	7.6	4.6	4.3
Single	10.8	4.2	2.4	2.4	22.6ab	8.2	4.4	4.7
Split	12.8	4.2	2.3	2.3	24.6b	8.1	4.5	4.6
<i>Utilization</i>								
Grazed	11.3	4.4a	2.2	2.2	23.6a	8.9a	4.3	4.4
Hay	10.9	3.7b	2.6	2.5	19.4b	7.0b	4.7	4.7
<i>Year</i>								
2004	12.8	5.0a	2.4b	2.1b	19.0	7.8b	3.9b	3.3b
2005	9.6	6.0a	5.6a	5.9a	23.0	15.2a	11.5a	12.7a
2006	10.1	3.8ab	1.5bc	1.5bc	21.7	8.0b	3.4bc	3.6b
2007	11.8	2.0b	1.0c	0.8c	19.9	4.5c	2.0c	1.7c
2008	11.3	3.3ab	1.5bc	1.5bc	23.8	4.3c	1.8c	1.5c
Effect	ANOVA P > F							
Manure	ns	ns	ns	ns	0.07	ns	ns	ns
Utilization	ns	0.04	ns	ns	0.1	0.002	ns	ns
Year	ns	0.02	0.006	0.002	ns	<0.001	0.002	0.001
Manure x Utilization	ns	0.06	ns	ns	ns	0.002	ns	ns
Manure x Year	ns	ns	ns	0.1	ns	ns	ns	ns
Utilization x Year	ns	ns	ns	ns	ns	ns	ns	ns
Manure x Utilization x Year	0.08	0.06	ns	ns	ns	ns	ns	ns

N=60 Significance at $\alpha = 0.10$. Means followed by the same letter in the same column for each harvest management type (interaction means) and main effect indicate not significant difference at $\alpha = 0.10$ with Tukey-Kramer adjustment for beta error. Analysis was performed on log-transformed data.

APPENDIX II. Additional Tables and Figures for Chapter 3

Appendix II.1 Mean, maximum and minimum of chemical properties and nutrient application rates of swine slurry applied at the site annually for three years (from 2004 to 2006) in one spring application (Single).

Manure characteristics	—Spring manure—		
	Mean	Max	Min
Dry matter, %	9	12	6
pH	6.9	7.1	6.8
EC, dS m ⁻¹	21.9	24.2	19.7
Application rates	—Single treatment†—		
	Mean	Max	Min
Manure application rate, '000 L ha ⁻¹	44	53	39
NO ₃ -N, kg N ha ⁻¹	0	0	0
NH ₃ -N, kg N ha ⁻¹	158	178	146
Organic-N, kg N ha ⁻¹	94	121	51
Available-N, kg N ha ⁻¹	142	164	126
Total Nitrogen, kg N ha ⁻¹	253	299	202
Total Phosphorus, kg P ha ⁻¹	65	70	58
Calcium, kg Ca ha ⁻¹	53	65	34
Potassium, kg K ha ⁻¹	93	113	83
Magnesium, kg Mg ha ⁻¹	60	66	48
Sodium, kg Na ha ⁻¹	30	40	24
Sulphur‡, kg S ha ⁻¹	18	18	18
Chloride†, kg Cl ha ⁻¹	75	75	75

N = 3

† Manure analyses for chloride were not available in 2004 and 2005 (N = 1)

‡ Manure analyses for sulphur were not available in 2006 (N = 2)

Appendix II.2 Mean, standard deviation (stddev), standard error of the mean (stderr) and number of samples (N) of main and three-way interaction effects of total soil P obtained from one step digestion compared to obtained from sum of P fractions from sequential extraction. Absolute difference and difference as percentage of total P from one step digestion are also shown.

Manure	Vegetation	Depth cm	Total P from one step digestion			N	Total P as sum of P pools from sequential extraction		Difference	Difference %
			Mean ----- mg kg ⁻¹ dry soil -----	Stddev	Stderr		----- mg kg ⁻¹ dry soil -----			
Control			340.8	-	-	48	341.4	0.7	0.2	
Single			436.9	-	-	48	430.6	-6.4	-1.5	
	Grassed		336.3	-	-	80	342.4	6.0	1.8	
	Bare		441.4	-	-	16	429.7	-11.7	-2.7	
		0-5	507.2	-	-	24	511.8	4.6	0.9	
		5-15	322.3	-	-	24	368.3	46.0	14.3	
		15-30	424.4	-	-	24	382.3	-42.2	-9.9	
		30-60	301.4	-	-	24	281.6	-19.8	-6.6	
Control	Grassed	0-5	321.6	156.0	49.3	10	349.7	28.1	8.7	
		5-15	246.2	71.7	22.7	10	282.0	35.8	14.6	
		15-30	339.0	127.6	40.3	10	304.7	-34.3	-10.1	
		30-60	280.8	71.1	22.5	10	277.1	-3.7	-1.3	
Control	Bare	0-5	437.0	182.8	-	2	439.8	2.8	0.6	
		5-15	379.4	47.5	-	2	445.2	65.8	17.4	
		15-30	450.2	20.2	-	2	404.7	-45.5	-10.1	
		30-60	271.8	32.0	-	2	228.3	-43.5	-16.0	
Single	Grassed	0-5	543.5	207.6	65.6	10	543.2	-0.3	-0.1	
		5-15	273.3	84.3	26.7	10	311.7	38.3	14.0	
		15-30	361.2	161.2	51.0	10	382.7	21.6	6.0	
		30-60	325.0	66.1	20.9	10	287.8	-37.2	-11.4	
Single	Bare	0-5	726.6	35.2	-	2	714.6	-12.0	-1.7	
		5-15	390.3	117.4	-	2	434.4	44.1	11.3	
		15-30	547.3	314.4	-	2	437.0	-110.4	-20.2	
		30-60	328.1	35.9	-	2	333.2	5.1	1.5	
Overall Difference									3.2	

Appendix II.3 Mean values of soil pH after three years of annual swine manure applications for manure and vegetation treatments and sampling depth. Shown are results of analysis of variance for main treatment and interaction effects.

	Depth, cm					
	0-5	5-15	15-30	30-60	60-90	90-120
<i>Depth</i>	7.5e	7.8de	8.2cd	8.6bc	8.9ab	9.0a
<i>Manure x Depth</i>						
Control	7.7c	8.1bc	8.3b	8.6ab	8.8a	9.1a
Single	7.3c	7.5c	8.0b	8.6a	8.9a	9.0a
<i>Vegetation x Depth</i>						
Grassed	7.2Be	7.8de	8.1cd	8.5bc	9.0ab	9.1a
Bare	7.8Ab	7.8b	8.3ab	8.7a	8.8a	9.0a
ANOVA	P > F					
Manure	ns					
Vegetation	ns					
Depth	<0.001					
Manure x Vegetation	ns					
Manure x Depth	0.06					
Vegetation x Depth	<0.001					
Manure x Vegetation x Depth	ns					

N = 264. Main effects are means of two experimental units (paddocks) each split in grass (ten subsamples) and bare earth (one subsample). Each subsample location was sampled in six depths intervals. Model was a randomized complete block split plot with covariance among depths. Effects are significant at $P < 0.1$. Separately for each significant effect, means followed by a different capital letters in a column are significantly different. Means followed by the same lower case letters in a row are not significantly different at $\alpha = 0.1$ and Tukey-Kramer adjustment for beta error.

Appendix II.4 Mean values of soil electrical conductivity (EC) after three years of annual swine manure applications for manure and vegetation treatments and sampling depth. Shown are results of analysis of variance for main treatment and interaction effects.

	Across depth	Depth, cm					
		0-5	5-15	15-30	30-60	60-90	90-120
----- dS m ⁻¹ -----							
<i>Manure</i>							
Control	0.12B						
Single	0.16A						
<i>Vegetation</i>							
Grassed	0.11B						
Bare	0.17A						
<i>Depth</i>		0.23a	0.16b	0.13bc	0.10c	0.11c	0.11c
<i>Manure x Vegetation x Depth</i>							
Control Grassed		0.13C	0.10B	0.10	0.09	0.09	0.08
Control Bare		0.17BC	0.15B	0.15	0.09	0.14	0.14
Single Grassed		0.21Ba	0.10Bb	0.11b	0.10b	0.09b	0.08b
Single Bare		0.39Aa	0.30Aab	0.18c	0.10c	0.13c	0.15c
ANOVA				P > F			
Manure							0.001
Vegetation							0.04
Depth							<0.001
Manure x Vegetation							0.01
Manure x Depth							0.0003
Vegetation x Depth							0.01
Manure x Vegetation x Depth							0.001

N = 264. Main effects are means of two experimental units (paddocks) each split in grass (ten subsamples) and bare earth (one subsample). Each subsample location was sampled in six depths intervals. Model was a randomized complete block split plot with covariance among depths. Effects are significant at $P < 0.1$. Separately for each significant effect, means followed by different capital letters in a column are significantly different. Means followed by the same lower case letters in a row are not significantly different at $\alpha = 0.1$ and Tukey-Kramer adjustment for beta error.

Appendix II.5 Mean values of soil K₂SO₄ extractable dissolved organic carbon (EDOC) after three years of annual swine manure applications for manure and vegetation treatments and sampling depth. Shown are results of analysis of variance for main treatment and interaction effects.

	Depth, cm					
	0-5	5-15	15-30	30-60	60-90	90-120
	mg C kg ⁻¹ dry soil					
<i>Depth</i>	213.3a	146.4b	159.9ab	89.2c	49.5d	38.6d
<i>Vegetation x Depth</i>						
Grassed	250.1a	139.3b	130.3b	86.1c	41.2d	32.5d
Bare	176.5ab	153.5ab	189.5a	92.3bc	57.8c	44.8c
ANOVA	P > F					
Manure	ns					
Vegetation	ns					
Depth	<0.001					
Manure x Vegetation	ns					
Manure x Depth	ns					
Vegetation x Depth	0.04					
Manure x Vegetation x Depth	ns					

N = 264. Main effects are means of two experimental units (paddocks) each split in grass (ten subsamples) and bare earth (one subsample). Each subsample location was sampled in six depths intervals. Model was a randomized complete block split plot with covariance among depths. Effects are significant at P < 0.1. Means followed by the same letter in a row are not significantly different at $\alpha = 0.1$ and Tukey-Kramer adjustment for beta error.

The concentration of extractable dissolved organic carbon in the extracts was determined by UV-photochemical oxidation (Technicon Industrial Systems, 1978; Goulden and Brooksbank, 1975, Anal. Chem. 47:1943-1946) and colorimetry using a Technicon II Autoanalyzer (Pulse Instruments, Saskatoon, SK).

Appendix II.6 Means values of Phosphorus Sorption Capacity (PSC) and Degree of Phosphorus Saturation (DPS) for surface soil (0-5 cm) after three years of annual swine manure applications for manure and vegetation treatments. Shown are results of analysis of variance for main treatment and interaction effects.

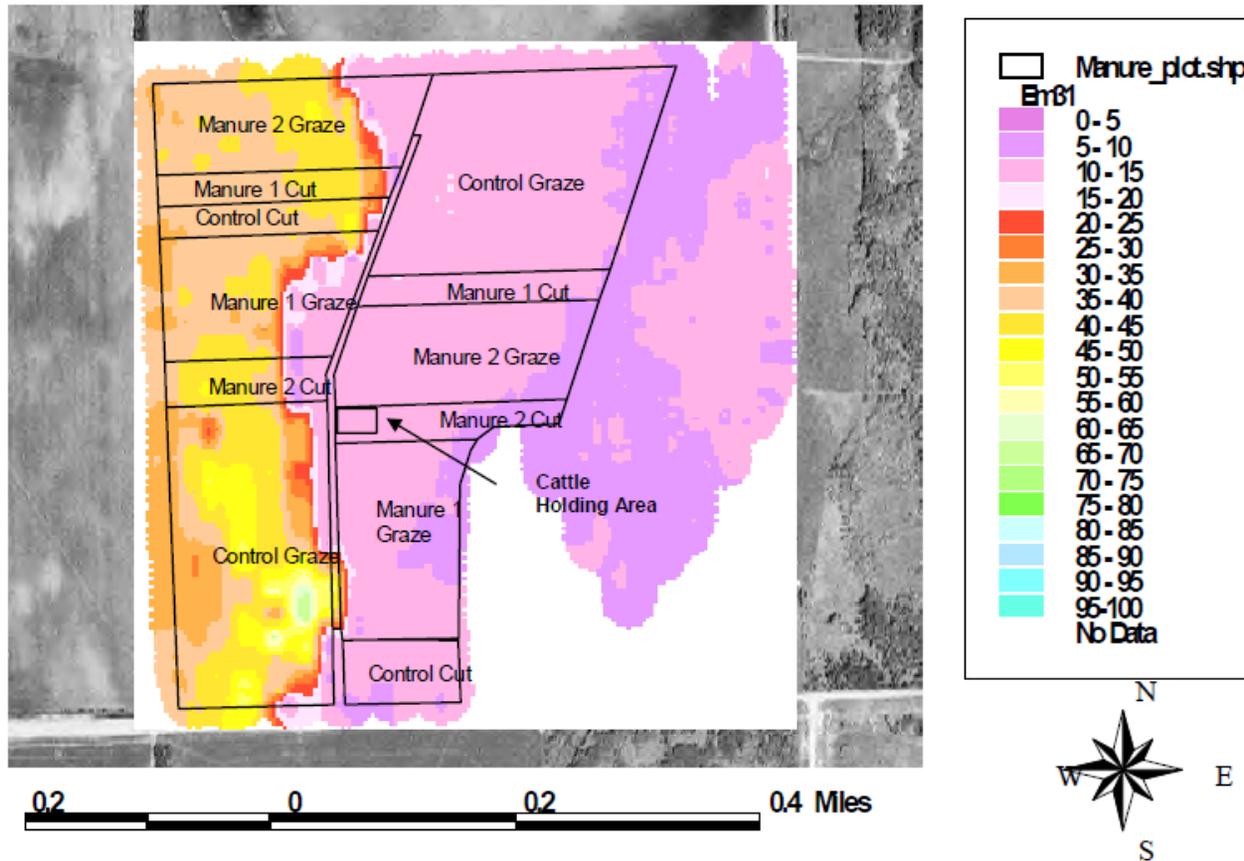
		PSC† mg kg ⁻¹	DPS‡ %
<i>Manure</i>			
	Control	328.0	11.6
	Single	299.3	56.9
<i>Vegetation</i>			
	Grassed	366.6a	14.8
	Bare	260.7b	53.7
<i>Manure x Vegetation</i>			
	Control		
	Grassed	393.0	3.5
	Bare	263.0	19.8
	Single		
	Grassed	340.2	26.2
	Bare	258.4	87.5
ANOVA		P > F	
<i>Manure</i>		ns	ns
<i>Vegetation</i>		0.09	ns
<i>Manure x Vegetation</i>		ns	ns

N = 44. Main effects are means of two experimental units (paddocks) each split in grassed (ten subsamples) and bare (one subsample). Model was a randomized complete block split plot. Effects are significant at $P < 0.1$. Separately for each significant effect, means followed by a different letter in a column are significantly different $\alpha = 0.1$ and Tukey-Kramer adjustment for beta error.

†PSC was calculated as $S_{\max} + \text{Extractable-P}$ where S_{\max} is the Sorption Maxima from the Langmuir isotherm estimated with Ige et al. (2008) using: $S_{\max} = 1.7 * P_{150}$, where P_{150} is sorbed P from a one point isotherm of 150 mg P L⁻¹.

‡DPS was calculated for each sample as $\text{Extractable-P} / \text{PSC} * 100$ (mg P kg⁻¹ / mg P kg⁻¹).

Appendix III. Additional Tables and Figures for Chapter 4



Appendix III.1 Results of an electromagnetic survey at the site conducted in 2003. The western section of the site shows to be underlain by clay (readings 60-20) and the eastern section by coarse material (readings 0-20).

Appendix III.2 Overall and sampling period means of water table depth measured with a water level meter in shallow wells in the hay paddocks. Factors are Control and Single manure application treatment and four positions in a moisture gradient West (wetter) –East (driest) in each paddock.

Paddock	Manure Treatment	Location	Position	Average Water Depth				
				Overall (Fall 2007- Fall 2009)	Period 1 (Fall 07- Spring 08)	Period 2 (Spring 08- Spring 09)	Period 3 (Springmelt 09- Spring 09)	Period 4 (Spring 09- Fall 09)
----- cm from soil surface -----								
4	Control	1	1	68	120	89	37	36
4	Control	2	2	82	127	101	65	41
4	Control	3	3	94	136	115	74	50
4	Control	4	4	102	146	123	83	59
5	Single	5	1	72	112	92	46	43
5	Single	6	2	88	130	108	74	52
5	Single	7	3	88	128	110	76	49
5	Single	8	4	92	134	113	78	51

Sampling seasons started and/or ended at spring manure application of each year.

†Period 1 spanned from October 29th 2007 to April 29th 2008 and included 6-9 sampling dates.

‡Period 2 spanned from May 13th to October 23rd 2008 and included 12 sampling dates.

¶Period 3 spanned from April 8th to May 26th 2009 and included 9 sampling dates.

§Period 4 spanned from June 2nd to August 26th 2009 and included 9-11 sampling dates.

Appendix III.3 Mean (n = 3) chemical properties and nutrient application rates of swine slurry applied at the site from 2004 to 2006 and yearly values for the 2007-2009 water sampling period.

	2004-2006	2007	2008	2009
Dry matter, %	9	7	10	13
pH	6.9	7.3	6.9	5.8
EC, dS m ⁻¹	21.9	26.6	18.0	9.0
Manure applicaton rate, '000 L ha ⁻¹	44	31	36	67
NO ₃ -N, kg N ha ⁻¹	0	0	0	0
NH ₃ -N, kg N ha ⁻¹	158	128	135	238
Organic-N, kg N ha ⁻¹	94	37	71	184
Available-N, kg N ha ⁻¹	142	105	119	224
Total Nitrogen, kg N ha ⁻¹	253	165	207	423
Phosphorus, kg P ha ⁻¹	65	56	66	107
Calcium, kg Ca ²⁺ ha ⁻¹	53	33	46	82
Potassium, kg K ⁺ ha ⁻¹	93	62	65	138
Magnesium, kg Mg ²⁺ ha ⁻¹	60	44	50	84
Sodium, kg Na ⁺ ha ⁻¹	30	17	23	39
Sulphur‡, kg S ha ⁻¹	18	13	34	49
Chloride†, kg Cl ⁻ ha ⁻¹	75	52	62	83

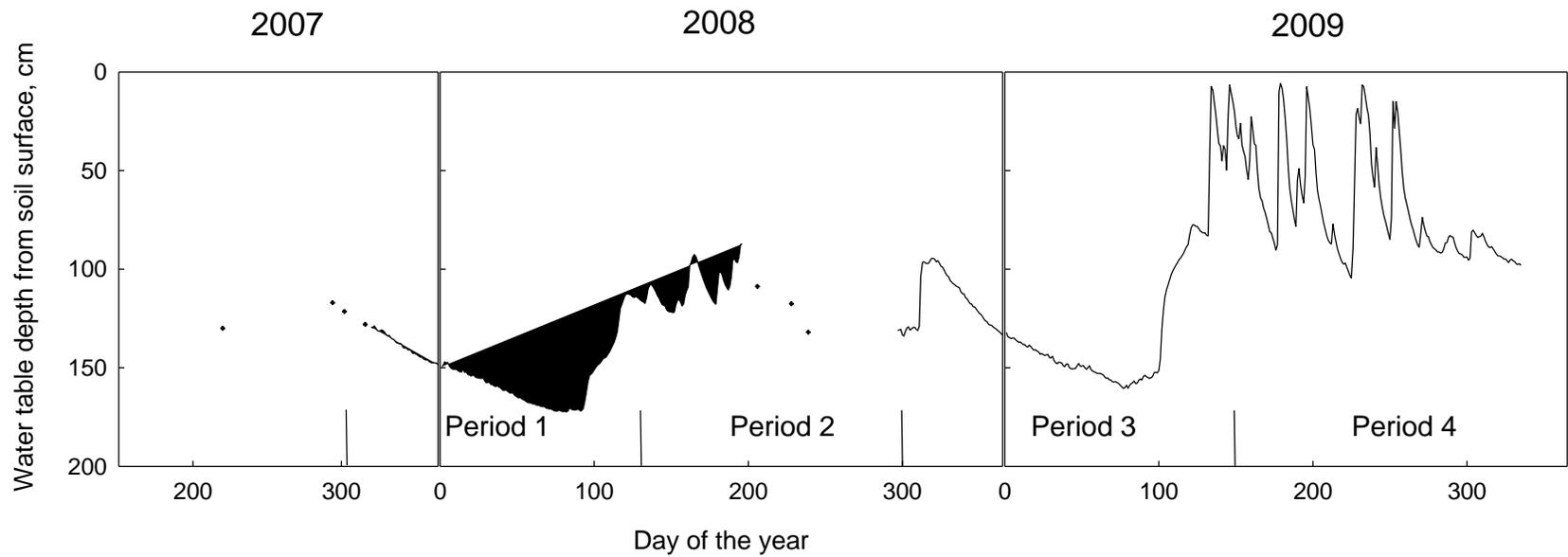
† Manure analyses for chloride were not available in 2004 and 2005 (n = 1)

‡ Manure analyses for sulphur were not available in 2006 (n = 2).

Appendix III.4 Monthly mean air temperature and total rainfall at the study site for the 2007-2009 growing seasons (April-October) and long-term climate normal (1971-2000).

Month	-----2007-----		-----2008-----		-----2009-----		Long-term normal†	
	Mean Temp	Rainfall	Mean Temp	Rainfall	Mean Temp	Rainfall	Mean Temp	Rainfall
	°C	mm	°C	mm	°C	mm	°C	mm
April	4.0	7	3.2	21	3.5	23	4.1	21
May	12.3	122	8.7	45	8.2	103	11.9	59
June	17.9	109	15.2	98	14.9	134	16.6	95
July	20.6	60	18.0	73	16.3	71	19.1	80
August	16.9	51	18.4	67	16.7	113	18.1	69
September	12.3	25	12.5	60	16.6	63	12.1	60
October	6.2	97	6.0	49	3.3	29	5.4	39
Season	12.9	470	11.7	413	11.4	535	12.5	422

† The climate normal for 1971-2000 was obtained from the Meteorological Service of Canada Long-term Climatic Data of the Steinbach Airport (Steinbach, MB; Environment Canada), distant 22 km from the site. .



Appendix III.5 Depth to water table (cm) from the soil surface measured in 2007, 2008 and 2009 in Position 3 Control of the hay pasture with a pressure transducer (solid line) and with a water level meter (dots). Elevation of the monitoring wells is 303.52 m a.s.l.

Appendix III.6 Mean concentration of dissolved reactive P and total dissolved P within surface groundwater at five depths in hay paddocks (Experiment 1) and associated analysis of variance results for the 2007 to 2009 years.

	Dissolved Reactive P	Total Dissolved P
	mg L ⁻¹	
<i>Manure treatment</i>		
Control	0.007	0.037b
Single	0.026	0.053a
<i>Position</i>		
1 (west)	0.016b	0.109a
2	0.008c	0.016ab
3	0.015a	0.038ab
4 (east)	0.034c	0.017b
<i>Period</i>		
1 (spring '08)	0.005b	not sampled
2 (summer '08)	0.006b	not sampled
3 (spring '09)	0.016ab	0.024b
4 (summer '09)	0.024a	0.068a
<i>Depth, cm</i>		
30	0.105a	0.136a
60	0.009bc	0.023b
90	0.007c	0.016c
120	0.011bc	0.037bc
150	0.007b	0.013bc
<hr/>		
ANOVA	<i>P > F</i>	
Manure	ns	0.004
Position	0.01	0.05
Period	0.008	<0.0001
Depth	<0.0001	<0.0001
Manure x Position	ns	ns
Manure x Period	0.03	ns
Position x Period	0.001	ns
Manure x Depth	<0.0001	<0.0001
Position x Depth	<0.0001	<0.0001
Period x Depth	0.03	ns
Manure x Position x Period	0.04	0.01
Manure x Position x Depth	<0.0001	<0.0001
Manure x Period x Depth	<0.0001	0.002
Position x Period x Depth	<0.0001	0.02
Manure x Position x Period x Depth	<0.0001	0.002

Appendix III.7 Mean concentration of dissolved reactive P and total dissolved P within surface groundwater at five depths in grazed paddocks (Experiment 2) and associated analysis of variance results for the 2007 to 2009 years.

	Dissolved Reactive P	Total Dissolved P
	mg L ⁻¹	
<i>Manure treatment</i>		
Control	0.007b	0.017b
Single	2.593a	3.187a
<i>Vegetation</i>		
Grassed	0.015b	0.008b
Bare	2.586a	3.196a
<i>Period</i>		
3 (spring '09)	1.272	1.787a
4 (summer '09)	1.329	1.417b
<i>Depth, cm</i>		
90	2.094	2.445a
120	1.545	2.036a
150	0.263	0.326b
ANOVA	<i>P > F</i>	
Manure	0.01	<0.0001
Vegetation	0.03	<0.0001
Period	ns	0.02
Depth	ns	0.02
Manure x Vegetation	0.01	<0.0001
Manure x Period	ns	0.04
Vegetation x Period	ns	0.05
Manure x Depth	0.003	ns
Vegetation x Depth	0.005	ns
Period x Depth	ns	ns
Manure x Vegetation x Period	ns	ns
Manure x Vegetation x Depth	ns	ns
Manure x Period x Depth	ns	ns
Vegetation x Period x Depth	ns	ns
Manure x Vegetation x Period x Depth	ns	ns