

A REPORT ON CURRENT KNOWLEDGE OF KEY ENVIRONMENTAL ISSUES RELATED TO HOG PRODUCTION IN MANITOBA

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PREFACE

Background and scope

The geographical area that is the province of Manitoba stretches across five distinct terrestrial ecozones, each with a unique combination of land forms, soils, water features, vegetation and climate. Within each ecozone exists a number of ecodistricts characterized with distinctive relief, geography, landforms and soils, vegetation, water, fauna and land use. The majority Manitoba's agricultural activity, including pig production, lies within the Prairies and southern portions of the Boreal Plains ecozones. Agriculture activity exists to a lesser degree in the northern Boreal Plains and Boreal Shield ecozones, however, industry and public initiatives are likely to increase agriculture, including animal production in these regions. The challenge to Manitoba regulators is great as environmental regulation or policy requires the flexibility to address issues across this broad and geographically diverse landscape.

Trends over the past 20 years confirm that Manitoba's agriculture activity is diverse and remains very dynamic. Pig production has played a major role in this evolution and today leads the agriculture sector in farm cash receipts. However, unlike many other pork producing regions of the world, Canadian farm investment and management practices have two to four fold greater reliance on export markets. The challenge is to identify environmental protection policies that continue to offer Manitoba's agriculture sector, and the pig industry specifically, opportunities to adapt as market forces change and new technologies become available.

In November, 2006, Manitoba Conservation formally instructed The Manitoba Clean Environment Commission to undertake a review of the environmental sustainability of hog production in Manitoba. As a part of that process, researchers at the Faculty of Agricultural and Food Sciences, University of Manitoba were asked to provide an overview of current knowledge relevant to the key environmental issues associated with pig production in Manitoba; to identify mitigation strategies and public policy currently considered or in use in other jurisdictions; and to provide an assessment of these environmental protection measures on the basis of environmental and economic impact.

Over the course of two meetings with representatives of the Manitoba Clean Environment Commission, seven key environmental issues were identified and report format was confirmed. The seven issues to be addressed included nutrient management and water quality, water use, odour management and air quality, manure storage, manure processing, energy use, and pathogens and antibiotic resistance. The report format was to provide a description of each environmental issue, including an overview of the basic science principles, current scientific and technical knowledge and identification of mitigation strategies. Where a mitigation strategy is related to a review or description of public policy, public policy was defined to include education, research, government policy and regulation as well as best management practices. The intent of the report was not to provide evidence of all research done in any particular area. Source of information used to prepare the report was predominantly scientific information, but personal communication and technical information, such as adopted management practices, were included and referenced.

Report summary

Chapter 1 provides relevant statistics characterizing Manitoba's pig production industry. The chapter includes methodology for calculation of total annual pig manure production as well as total land base on which pig manure is applied, developed following consultation (summer 2007) with representatives from Manitoba Conservation, Manitoba Agriculture, Food and Rural Initiative as well as industry. Given that the manner in which pig manure is managed has a significant impact on both the economic competitiveness and the environmental sustainability of Manitoba's pork industry, an improved mechanism is required for estimation of manure produced, manure composition and land area on which manure application is practiced.

The importance and practice of efficient nitrogen and phosphorus use in pig production and cropland management are the focus of **Chapter 2**. High risk areas for nitrogen and phosphorus losses from Manitoba agricultural lands are identified and significant effort is directed towards our current understanding of sources and transport mechanisms for these nutrients. A review of the literature revealed that best management practices (BMPs) effective at reducing the risk of nutrient loss at the source are better understood than transport BMPs in our relatively cold, dry and flat environment and that source-oriented BMPs have greater potential for minimizing manure P loss to surface waters than transport and interception oriented BMPs.

Water is used in pig production systems as drinking water for the animals, for cleaning the barns, sanitizing equipment and, in liquid manure systems, for flushing or moving the manure to the storage site. A small proportion of water usage is also for domestic use which includes human usage associated with barn operation. **Chapter 3** provides an overview on water usage and wastage in hog operations and opportunities for conservation.

Air quality, the subject of **Chapter 4**, is one of the greatest concerns to the public when considering the siting of new or the expansion of existing hog operations in Manitoba. The odour problem in Manitoba is to some extent a perception issue, similar to other jurisdictions. None-the-less this chapter addresses the major scientific issues associated with odour including odour emission (how much odour is released), odour dispersion (how far it travels downwind), and the odour impact on health and property values. The chapter also addresses how feeding regiment, the type of confinement facility, type of manure management system, and the method of land application may influence other air quality parameters, including ammonia, greenhouse gases, total reduced sulphur compounds, hazardous air pollutants and particulate matter. Odour mitigation technologies and strategies are discussed on the basis of research and evaluation in Manitoba and other jurisdictions.

Chapter 5 addresses manure storage facilities and some of the regulatory requirements related to manure storage and handling. A large segment of Manitoba's hog industry has adopted liquid hog manure storage systems, and as such this is featured in the chapter. Lagoon design, operation, and engineering assumptions made in developing regulations related to liquid manure storage are discussed for four provinces of Canada (Manitoba, Alberta, Ontario, Quebec) and four states in the USA (North Carolina, Minnesota, Iowa, Illinois). The factors such as siting, total storage volume, depth, shape, detention time, and number of cells, embankment and excavation, inlet and outlet pipes design, effluent utilization, and water supply are compared for the different locations.

Manure processing strategies are combinations of techniques and technologies that achieve one or more of the following objectives related to the environment: nutrient management and the isolation of nitrogen and phosphorus, renewable energy production, odour control, and or greenhouse gas (GHG) reduction. **Chapter 6** focuses on manure processing and treatment technologies, which are currently utilized in various parts of the world, and hold significant promise as possible technical solutions for a sustainable hog industry in the Province of Manitoba.

Chapter 7 offers a detailed breakdown of the energy costs of applying liquid hog manure. Energy use is often described as the sum of direct (e.g. diesel fuel consumption) and indirect (e.g. energy for production of fertilizer or machinery) energy. AN energy balance sheet related to a Manitoba study suggests that the availability of large quantities of liquid manure from hog production facilities presents an opportunity for some farmers to dramatically reduce indirect energy consumption by replacing energy-intensive synthetic fertilizer with manure.

The importance of zoonotic pathogens in animal agriculture and its relationship with food and waterborne illnesses in humans as well as the contribution made by animal production practices to the development and persistence of antibiotic resistance in zoonotic pathogens are discussed **Chapter 8**. Evaluation of Manitoba's monitoring systems for food borne illness and water sources, and relevant environmental protection policy are assessed with respect to zoonotic pathogens and antibiotic resistance.

A wide range of environmental protection policy instruments exist. **Chapter 9** identifies relevant stakeholders and describes the range of policy instruments from voluntary measures, such as information provision or market incentives, through to the "command and control" or involuntary measures. Involuntary measures may include charges for an offence or outright banning of certain activities. Future environmental and economic sustainability for an industry is best ensured in a policy environment that encourages continual development of new strategies to reduce environmental risk and/or generate environmental benefit.

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CHAPTER 1 Statistics for Manitoba Pig Industry

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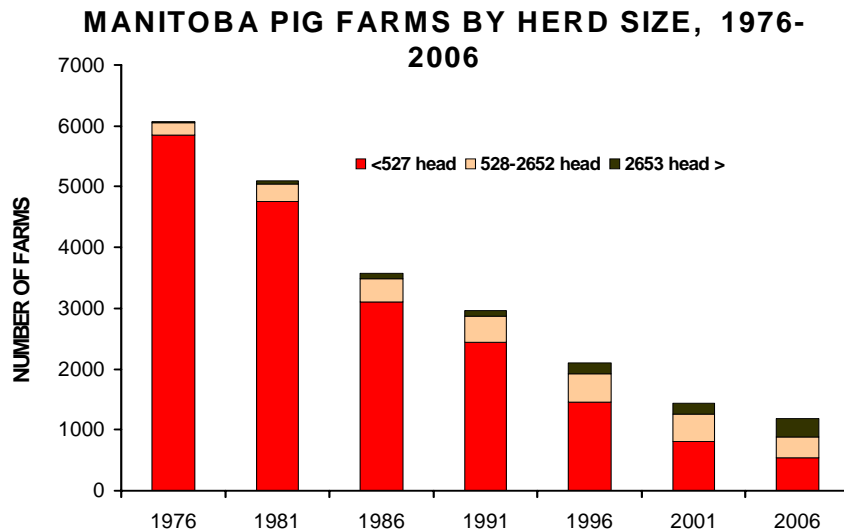
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1.1 Farm statistics

1.1.1 Current pig/livestock farm numbers

Statistics Canada's *2006 Census of Agriculture* shows that there were 1,188 Manitoba farms with pigs in May 2006, of which only 768 were classified by the North American Industrial Classification System (NAICS) as "hog and pig farms". Only 6.2% of all Manitoba farms had pigs in 2006, and of all Canadian farms with pigs, 10.3% were in Manitoba. According to the census five years earlier, there were 1,668 farms with pigs in 2001 (7.9% of all Manitoba farms) and 995 "hog and pig farms" using the NAICS (Statistics Canada, 2007).

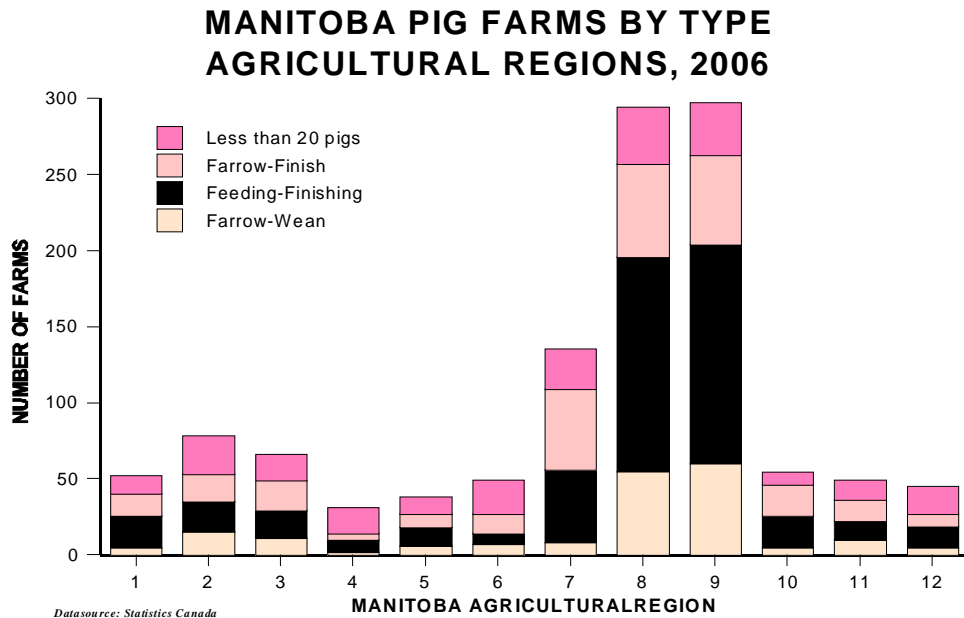
Of the 1,188 Manitoba farms with pigs in 2006, 240 or 20.2% had fewer than 20 pigs on the farm, 189 or 15.9% produced weanlings only, 293 or 24.7% were farrow-to-finish and 466 or 39.2% were feeder operations. The Manitoba swine industry has a much higher percentage of farms which produce only weanling pigs (those under 23 kg) than do other provinces.



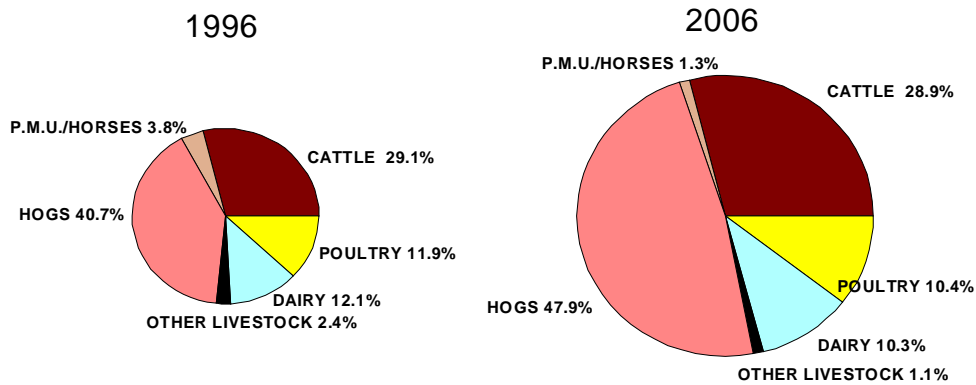
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The 2006 Census showed that about 29.6% of Manitoba's pig farms are located in the Eastern Region (9 and 10 in MB Pig by Type and Region chart), about 36.1% in the Central Region (7 and 8 in chart) about 16.5% in the Southwest Region (1, 2 and 3 in chart), about 9.9% in the Northwest Region (4, 5 and 6 in chart) and about 7.9% in the Interlake Region (11 and 12 in chart).

Further analysis using Statistics Canada's Census data showed that more than half of the pig farms in Manitoba in 2006 had no sows. These 629 units had an average of 1,505 pigs per farm and 32.3% of the total number of pigs in Manitoba. There were 559 Manitoba pig farms, or 47.1% of the total number of operations, with sows and, of these, 227 or 40.6% had fewer than 100 sows. These smallest farrowing units had only 1.6% of the total number of sows in Manitoba. On the other end of the scale, there were 111 pig production units (compared to 75 in 2001) with more than 1,000 sows per unit. These large units averaged about 2,396 sows per unit and, in total, comprised 72.4% of all sows and 40.0% of total pigs in the province. Another 70 units had 600-999 sows, averaging 744 sows per unit, with a total of 14.2% of all sows and 14.9% of total pigs in the province. There were 61 units in the 300-599 sow category, which had 7.5% of all sows and averaged 451 sows per unit. The 90 pig farms with 100-299 sows had 4.3% of all sows and 4.0% of all pigs.

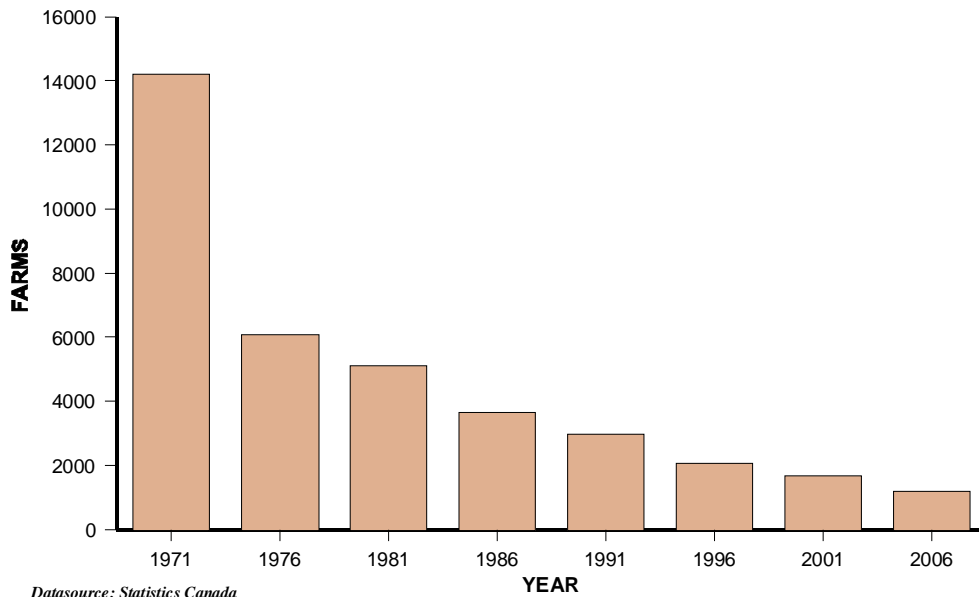


MANITOBA LIVESTOCK TYPE: PERCENT OF TOTAL PRODUCTION VALUE, 1996 and 2006



Datasource: MAFRI, author's estimate for 2006

NUMBER OF MANITOBA PIG FARMS 1971- 2006



Datasource: Statistics Canada

1.1.2 Change in farm numbers and farm size

According to Statistics Canada, the number of farms with pigs in Manitoba decreased from 14,200 in 1971 to 1,668 in 2001. By 2006, there were 1,188 pig farms in the province.

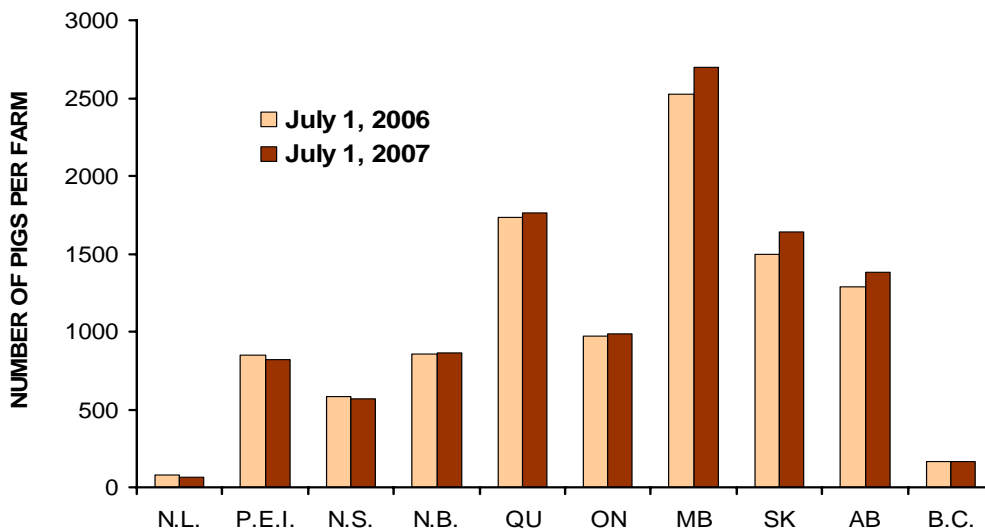
Manitoba has the largest average number of pigs per farm of any province. From 1,495 pigs per farm in 2001 the number rose to 2,695 pigs per farm in mid-2007, well above the next province, Quebec, which had 1,761 pigs per farm in mid-2007.

1.2 Animal Statistics

1.2.1 Animal Categories

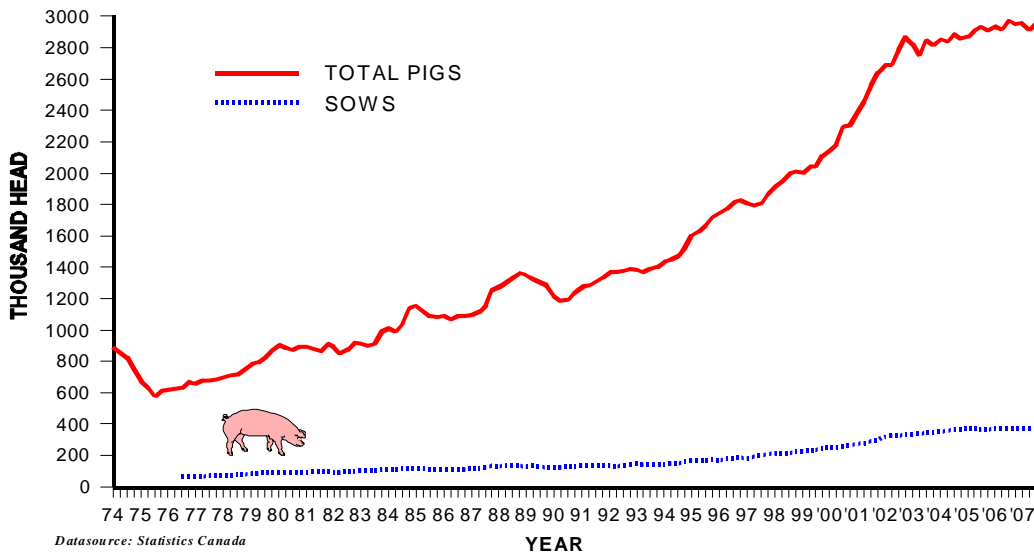
According to Statistics Canada data, only 31.2% of all Manitoba farms with more than 300 animal units (AU) were pig farms in 2006, while 881 Manitoba pig farms had 300 (AU) or less comprising 4.9% of all Manitoba farms in this category. Of all farms with sows, 208 or 37.2% had more than 300 AU in 2006.

AVERAGE NUMBER OF PIGS PER FARM, BY PROVINCE, 2006 - 2007



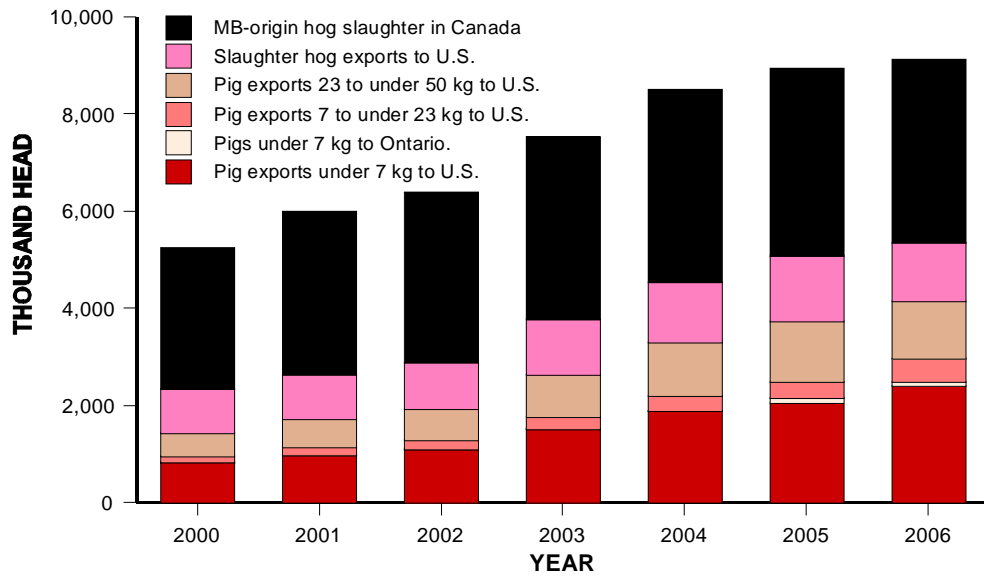
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PIGS ON MANITOBA FARMS BY QUARTER 1974- 2007



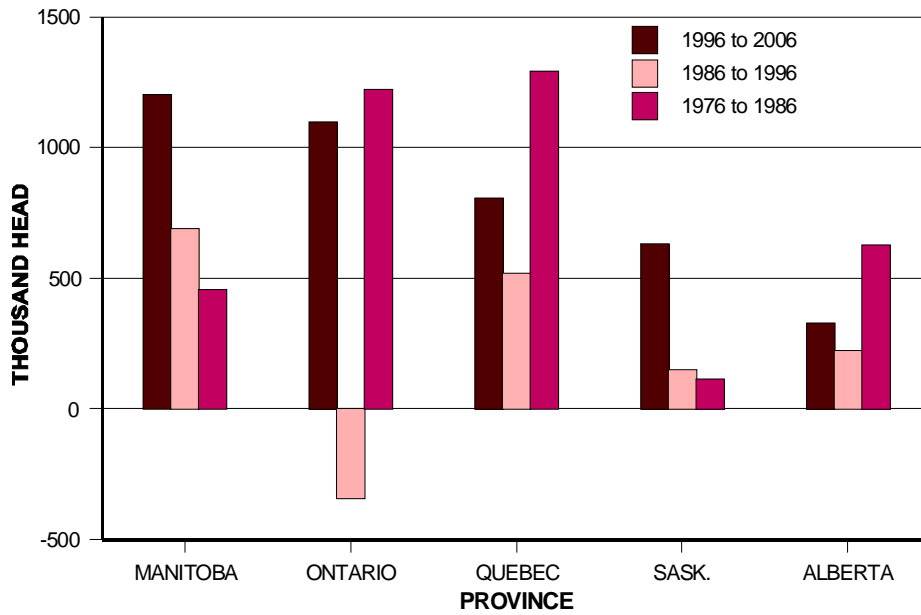
Datasource: Statistics Canada

MANITOBA PIG SALES BY TYPE 2000 - 2006



Datasource: Statistics Canada, AAFC, author's estimate

CHANGE IN TOTAL PIG HERDS BY PROVINCE TEN-YEAR INTERVALS, 1976 TO 2006



Datasource: Statistics Canada

1.2.2 Current pig numbers

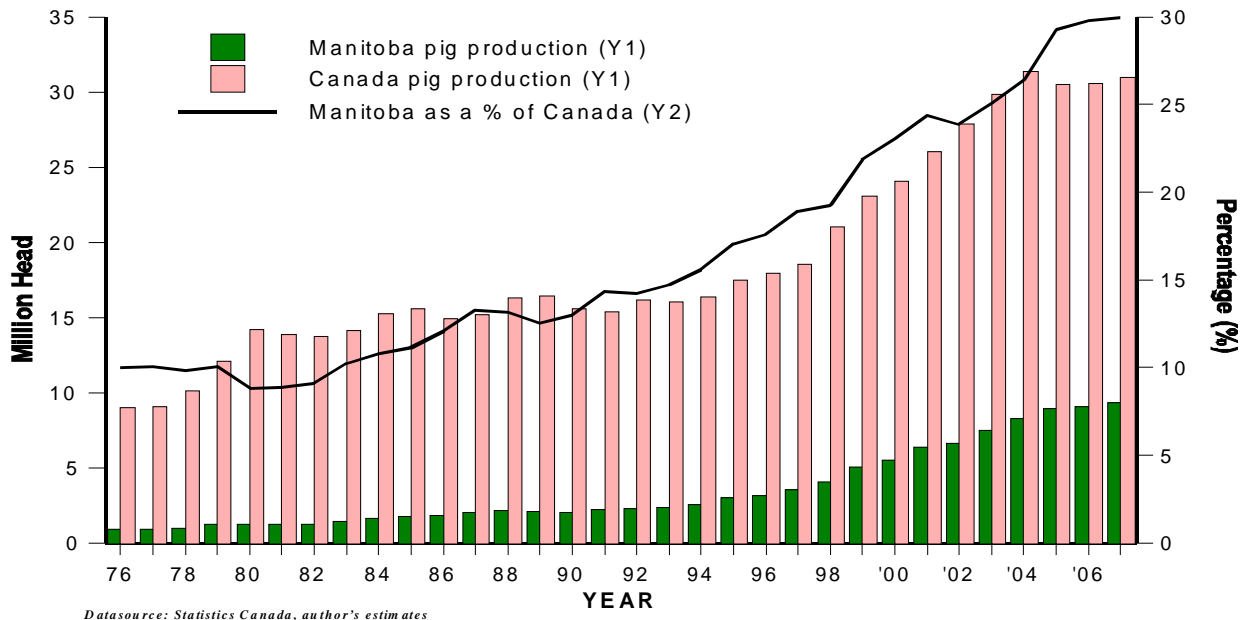
Interpretation of media and other reports often lead to an incorrect conclusion that Manitoba's total annual pig production (9.1 million pigs in 2006) reflects the number of animals on Manitoba farms at any given point in time. In fact, the total number of pigs on Manitoba farms at any given point in time would consist of the breeding herd of sows and boars, which are on farm all year, and fewer than 30% of the pigs destined for market annually. Sows produce approximately 2.2 litters annually. A large percentage of Manitoba's piglets are exported at 18 to 50 days of age. The remainder remain in Manitoba barns until they reach slaughter weight of 115 kg.

According to Statistic Canada estimates, there were a record 370,500 sows on Manitoba farms on July 1, 2007, an increase of 0.7% from a year earlier. The total number of pigs on farms at that time was 2,965,000 head, slightly below the July 1, 2006 record of 2,980,000 head.

January 1, 2007 statistics indicate that 38% (989,000 head) of the 2,586,000 pigs destined for market from Manitoba farms were either newborn or weanling pigs weighing less than 20 kg. More than half of these were targeted for export, most weighing less than 7 kg. The amount of feed consumed and manure produced by these animals during their time in Manitoba is small. Approximately one-third of the 799,000 of the pigs on farms in January weighing 20 to 60 kg were targeted for export with an average export weight of less than 50 kg. Total feed consumed and manure produced by this category of animal is significantly lower than for a slaughter pig. Finally, there were 789,000 head in the over 60 kg category, all of which were being fed to slaughter weight of about 115 kg.

1.2.3 Change in pig numbers

CANADA AND MANITOBA PIG PRODUCTION 1976 - 2007P



Manitoba is the largest pig-producing and pig-exporting province in Canada with close to 30% of national pig production in 2006. The quality of Manitoba pigs is among the best in Canada. Manitoba sows are also the most efficient producers of pork in Canada, producing an annual average of over 2.29 tonnes per sow in 2006 compared to the Canadian average of almost 1.72 tonnes of pork per sow.

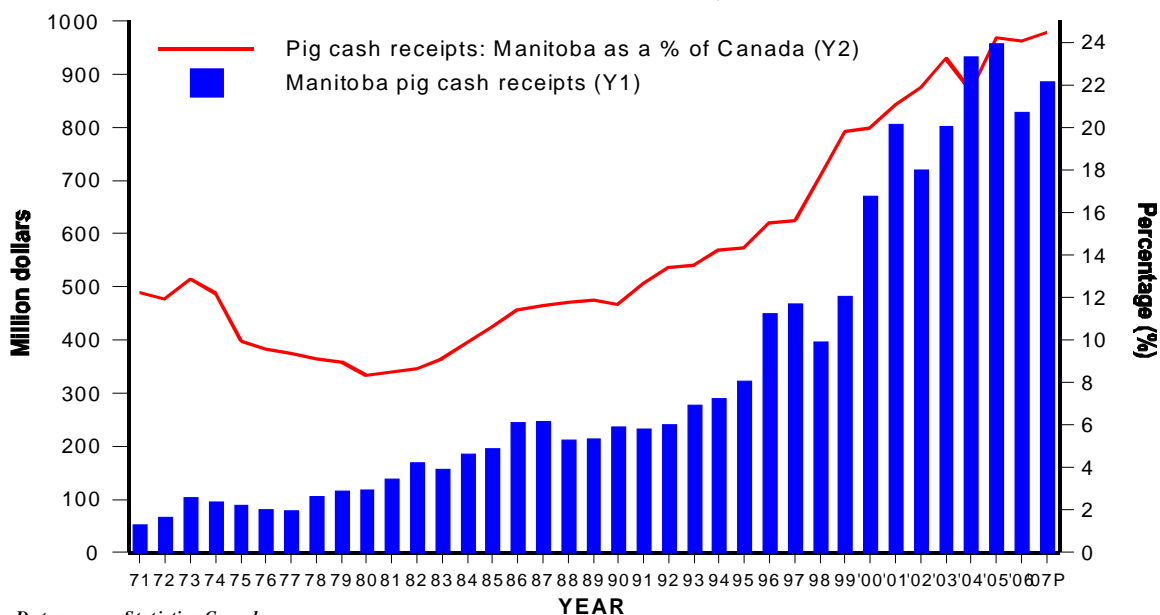
Manitoba's swine industry grew significantly from the mid 1990s to the early 2000s, but growth began to slow after 2003. The average annual rate of growth in production from 1995 to 2004 was 12.6% compared to an average increase of 4.6% for the previous decade. Annual production growth fell from 12.8% in 2003 to less than 2% in 2006.

Pig sales in the province totalled 9.1 million head in 2006, of which about 5.3 million head or 58 percent were sold out of the province, with 2.4 million or 26% of total sales as baby pigs under 7 kg. Total pig production in 2006 was up by 43% from the 6.35 million pigs produced five years earlier and ten times the pig production in 1976.

Pig producers and brokers exported over 1.2 million slaughter hogs and more than 4 million weanlings/feeder pigs directly to the U.S. Close to 0.1 million weanling/feeder pigs were sold to Ontario in 2006, well below the level ten years earlier, when many more pigs were shipped east and fewer to the U.S. Reduced demand for hogs by Manitoba slaughter plants led to 3.8 million Manitoba hogs being killed in federal and provincial plants in 2006, down by 2 percent from 2005.

The swine industry is the largest source of farm cash receipts of any single agricultural commodity sold in the province. Pig sales produced about 27% of Manitoba's farm cash receipts from the marketplace (excluding direct program payments) in 2006. Increased pig sales and slightly higher prices for slaughter hogs and iso-wean pigs are expected to raise cash receipts from pig sales by about 7% in 2007.

PIG CASH RECEIPTS: MANITOBA AND AS A % OF CANADA, 1971- 2007P



Datasource : Statistics Canada

1.3 Economic Impact of the Industry

The cash income from pigs in Manitoba was estimated for 2006 by Statistics Canada at \$834 million, the largest income from any single commodity in the province. In their study on the regional impacts MacMillan et al. (2004) estimated a dollar of output in pig production in the Pembina Valley can mean \$2.66 in output the rest of the supply chain in the province. Manitoba has a significant pork processing plant in Brandon, MB owned by Maple Leaf. At current single shift capacity it can slaughter 45,000 pigs per week. Grier and Kohl (2003) argue that they would need to move to double shifts to be competitive with U.S. plants.

The pig sector is also a huge user of feed. Kraft and Rude (2002) showed that nearly 2 million tonnes of feed grains go to pig feed each year, contributing to a feed grain deficit and the need for feed imports to supply Manitoba's livestock sector. Kraft and Rude showed an average 174,000 tonnes of corn were imported into Manitoba each year to supplement feed supplies. This is of great concern if the prices in the U.S. rise due to new ethanol demand. The price for corn in Chicago has stayed above \$3/bu. for more than six months now which is a 50% increase over the long term floor price of \$2. Increased grain demand is resulting in similar price increases for wheat and corn in Canada, and the increasing value of the Canadian dollar has reduced US importer demand for pigs and causing a decline in pig prices.

Pig manure can be given economic value. The crop industry, important to Manitoba's economy, is the main beneficiary of pig manure nutrients. Although no single commodity generates the income of pigs, the crop sector as a whole, normally generates over \$1.3 billion a year in cash income for farmers in Manitoba. The estimated annual manure production by Manitoba's pig industry is 346,000 tonnes, dry basis (calculations provided in next section of Chapter). The value of nitrogen and phosphorus supplied by this manure is in the range of \$30 to \$40 million and is often given away to crop producers.

The recent Census of Agriculture for 2006 identifies various types of manure management used in Manitoba. Seventy-three percent of pig farmers apply their pig manure to their own land. Despite its value in nutrients, less than 3% of pig farmers get any compensation for their manure from crop farmers. The value of the manure used by mixed farmers on their own farms is not necessarily 73% of the \$30 million since large pig farms with the lion share of manure are less likely to have large tracts of crop land.

The 2006 census had some good news. Seven hundred and seventy-five farms, or 65% of all pig farms, were incorporating manure into the soil which improves its availability to plants, lowers odour generation and prevents the loss of nitrogen through volatilization. This should be expected from those farmers who can make use of the nutrients for crop production, but might not be expected by producers simply wanting to dispose of the manure as cheaply as possible.

1.4 Manure production/storage and land application statistics

1.4.1 Current estimates

The methods outlined below provide estimates of the total phosphates or phosphorus contained in pig manure produced in Manitoba in 2006. The industry has embraced new feed formulations and feeding practices in recent years which are expected to reduce manure phosphorus concentration and in some cases reduce the amount of manure produced per pig. Examples would include the use of enzymes, in particular phytase

enzyme and phase feeding. Annual estimates for total phosphorus in pig manure ranged from a high of over 7,000 tonnes with no phytase enzyme inclusion in the diet to about 6,100 tonnes with 50% adoption of phytase enzyme addition in the diet to 5,000 tonnes with 90% adoption of phytase use. Exact statistics related to level of adoption of this practice are not available, but industry and government consultation suggest that the adoption rate is high.

Pigs in Manitoba excreted approximately 22,500-24,000 tonnes of nitrogen in the form of manure in 2006. However a substantial portion of that nitrogen was lost due to volatilization of ammonia during the handling and storage of that manure.

1.4.2 Confidence of current estimates

Calculation of the total annual manure production by Manitoba's hog population can be calculated on the basis of statistics of total animal numbers within each category on Manitoba farms at a given point in time and estimates of daily manure output by each category. The values are scaled up to estimate annual manure production. Alternatively, total manure production can be estimated on the basis of total animals marketed and a calculation of manure production associated with each animal marketed.

Source of pig inventory numbers:

The number of pigs on Manitoba farms on May 16, 2006 was derived from answers to a Statistics Canada 2006 Census question, which asked Manitoba farmers if there were any pigs on the farm and if so, how many were boars, sows and gilts for breeding, nursing and weaner pigs and grower/finishing pigs. This is the most comprehensive count of the Manitoba pig inventory. Accuracy of the data depends on the willingness of pig producers to enter correct numbers on the questionnaire and on subsequent auditing and correction by Statistics Canada.

Post-censal estimates of pig inventories are conducted on a quarterly basis by Statistics Canada using the Census inventory as a base, surveying a relatively small sample of Manitoba pig producers and using hog slaughter and other production data to determine the percentage change in the pig inventory over time. In the past, Statistics Canada's estimates of the Manitoba pig inventory were carefully checked and often corrected by Manitoba Agriculture and Food staff prior to release. The quarterly inventories are adjusted by Statistics Canada back five years once new Census results are available. The inter-censal revisions for 2001-2006 and post Census corrections for 2006-2007 were released on August 16, 2007 by Statistics Canada.

Total annual pig production in Manitoba is calculated by adding (sources below)

- i) the number of hogs of Manitoba origin slaughtered in Canadian federally- and provincially-inspected plants during the year to
- ii) the number of all Manitoba pigs exported to the United States to
- iii) the known number of weanling/feeder pigs shipped to other provinces for finishing to
- iv) the January-January change in the Manitoba pig inventory.
- v) Any pigs imported into Manitoba are deducted from this total.

Sources of this data are as follows:

1. Province-of-origin hog slaughter data is available from Canadian Food Inspection Agency/Agriculture and Agri-Food Canada (AAFC, 2007). (In recent

years, the improved accuracy of Manitoba Pork Council's slaughter levy data has allowed the latter to be used).

2. Manitoba pig export data is collected at the ports of exit by the United States Department of Commerce. The data is split into type and weight categories, such as breeding sows and boars, pigs <7kg (iso-wean pigs) (about 2.4 million pigs in 2006 or more than 25% of total pig production), pigs 7-23 kg, pigs 23-50 kg (almost 1.2 million in 2006 or 13% of total pig production) and pigs > 50 kg, most of which are slaughter hogs, cull sows and cull boars. Statistics Canada and AAFC compile and disseminate this data. It should be noted that about 45% of the pigs produced in Manitoba are exported as weanling or feeder pigs (AAFC,2007) .

3. The number of pigs shipped to other provinces for feeding is obtained by phoning the large weanling production companies in Manitoba. This number is relatively small compared to exports to the U.S. (fewer than 100,000 pigs in 2006.)

4. Statistics Canada's January 1 inventory estimates are used to calculate the change in the Manitoba pig inventory from one year to the next.

5. The number of pigs imported into Manitoba from the U.S. is very small and is obtained from Statistics Canada or AAFC. However, there are weanlings and slaughter hogs shipped from Saskatchewan through Manitoba to the United States that may be counted by the US Dept. of Commerce as Manitoba-origin if the exporter has a Manitoba address. This number is provided by Saskatchewan Pork Development Board and is deducted from total Manitoba pig exports.

Estimating pig manure production was more complicated than expected as every source has different daily manure production data for pig types and weights.

1. One source used was the "CONCENTRATED ANIMAL FEEDING OPERATION (CAFO) AMENDED FACT SHEET", National Pollutant Discharge Elimination System (NPDES) and State Waste Discharge General Permit, June 21, 2006. (see attached table adapted from the American Society of Agricultural Engineers (ASAE))

2. Another source was the Manitoba Agriculture, Food and Rural Initiatives (MAFRI) pre- and post phytase use manure nutrient tables 4a, 4b and 5 in <http://www.gov.mb.ca/agriculture/livestock/pork/swine/pdf/bah09s04.pdf>

3. The third source was from a paper by *Clarence Froese, DGH Engineering* <http://www.banffpork.ca/proc/2003pdf/17cFroese.pdf>

4. Also used were Prairie Agricultural Machinery Institute (PAMI) estimates of the percentage of P and N in stored liquid manure.

It is understood that the amount of phosphorus (organic and inorganic) in manure is highly variable and the averages used will only give a rough idea of total phosphorus content. The total amount of dry matter in the manure (Statistics Canada: 9% – this method was not used due to uncertainty of type of manure) and liquid manure (Froese/DGH: 5% and ASAE: given at 9%) were used in order to estimate both the phosphate and phosphorus content. MAFRI's estimates of nutrient content of manure, pre-and post-phytase use (50% each) were used to pro-rate phosphate production per pig type, but this data had to be applied to liquid manure estimates made by Froese/DGH because of the difficulty getting the average MAFRI manure production by farrow-finish sows/pigs to fit into the Statistics Canada inventory types.

The methods used for the N estimates were similar to those used for the P estimates. The calculation using PAMI's estimate of the organic N content of manure are much lower than the other two calculations (ASAE and MAFRI) using total N content of manure.

1.4.3 Mechanisms used to collect relevant data on animal numbers, manure production and land area receiving manure on land for the Province of Manitoba

Various ways of estimating total manure production in Manitoba were attempted using both Manitoba and U.S. reported averages of solid and liquid amounts of manure each pig type or weight group produces daily and applying these first to a breeding stock inventory on farms/total annual sales by weight of pig combination and then to Statistics Canada's 2006 Census pig inventory numbers by type of pig.

As manure produced per pig at each stage of development cannot be determined with any accuracy due to the difficulty of separating sow manure production from pre-weaned pig manure production, the breeding stock/sales by weight methods were abandoned in favour of the methods using inventory numbers by type of pig. The one fault found with the latter method was that some iso-wean pigs may be missed, but the amount of manure they produce is probably smaller than the variation in total manure production estimates.

The 2006 Census of Agriculture asked Manitoba farmers if manure was produced on the farm in 2005 and if that manure was applied to the farm operator's own land or sold or given to another operation. Farmers were also asked if manure was purchased or received and asked about the method of manure application and areas and types of farm land on which the manure was applied.

Not all of the 1,188 Manitoba pig producers answered these questions. Only 1,114 pig operations said manure was produced on the farm and of these, 877 applied the manure to their own land, 242 sold or gave manure to other farms and 34 pig farms bought or received manure. Of all this activity, 181,200 acres were used to apply pig manure. There is no way of separating out non-pig farms, which used only pig manure, but it is assumed that most of the liquid manure applied was from pigs. Of non-pig farms, the Census shows 124,600 additional acres on which liquid manure was applied. As there are farms which did not answer the question on manure application, it is possible there could be an unknown number of acres on which pig manure was used and not reported.

The total amount of phosphorus produced in pig manure in 2006 (5,000 tonnes estimated) was divided by the minimum area of farm land on which pig manure was known to be applied in 2005 (about 120,000 hectares) to estimate the rate of phosphorus application in 2006 (42 kg/ha).

1.5 References

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CHAPTER 2 Manure Nutrient Management and Water Quality

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2.1 Executive Summary

Manitoba's pigs require substantial quantities of nutrients; therefore feed comprises a large portion of a pig producer's cost of production. And, like other animals, pigs excrete the majority of nutrients that they are fed, creating a challenge to ensure that feed nutrients are used as efficiently as possible and that manure nutrients are not land-applied at rates that exceed crop removal or the land's capacity to retain those nutrients. Therefore, the manner in which pigs are fed and pig manure is managed has a significant impact on both the economic competitiveness and the environmental sustainability of Manitoba's pork industry.

Pig manure is an excellent source of crop nutrients that can be used to offset the use of synthetic fertilizers, even though the nutrients in manure are not as soluble as in synthetic fertilizers. The total gross amount of N and P in pig manure produced province-wide is approximately 22,500 to 24,000 tonnes of N and 5,000 to 7,000 tonnes of P (with no accounting for losses), figures that are equivalent to 8-9% and 12-17% percent of the amount of N and P synthetic fertilizers applied and 8-9% and 11-15% percent of N and P removed by Manitoba crops. Therefore, if pig production and manure application were evenly distributed throughout the province and properly offset by reductions in the use of synthetic fertilizers, excess accumulation of nutrients should not occur. However, this manure is applied to only 2.5% of Manitoba's agricultural land base, approximately 120,000 hectares, with a high concentration of that land in southeastern Manitoba.

The net rate of N application, after accounting for losses such as volatilization during manure handling, storage and application is probably well balanced with crop removal in most cases. However, the relatively low ratio of N:P in pig manure, compared to crop removal, along with significant losses of N from manure subsequent to excretion, result in excess application of P for pig manure that is applied to meet crop N requirements. As a result, the average rate of P application for pig manure in Manitoba appears to be 42-58 kg P per hectare, while the provincial average rate of P removal by crops is approximately 10 kg P per hectare. Applying pig manure P in balance with the average rates of crop P removal would increase land requirements to 500,000 to 700,000 hectares, equivalent to 10-15% of Manitoba's agricultural land base.

Nitrate leaching into groundwater and phosphorus loss to surface water are the main environmental risks associated with excess accumulation of N and P, respectively, regardless of whether the original source is livestock manure, synthetic fertilizer or plant residues. However, source factors, such as the accumulation of N or P in a field, are not the only issue that determines the risk of losing that nutrient to ground or surface water. Transport factors, such as climate, soil texture and topography must also be considered. Although the transport factors affecting nitrate leaching in Manitoba are reasonably well understood, the factors affecting the transport of phosphorus are not. Manitoba's landscapes are relatively flat, the soils are relatively alkaline (high pH) and the climate is

relatively cold and dry, compared to areas where most research on P loss has been conducted. Across landscapes typical of agricultural land in Manitoba, over 80% of runoff occurs during snowmelt, when soils are frozen and vegetation is dormant, creating a unique challenge for developing beneficial management practices that will intercept the predominantly dissolved forms of P between the field and the watercourse. In the interim, a strong focus on minimizing excessive loading of surplus P in the field, at the source, is the best strategy for private industry and public agencies to focus upon.

In order to be effective, beneficial management practices (BMPs) for minimizing the risk of N and P loss from pig manure must be targeted at the source and transport factors that are controlling those losses under Manitoba conditions. Public agencies and private industry must continue to invest in research that will develop and adapt BMPs to ensure that pig producers have the proper tools to respond to those policies. This research is especially important for some of the transport-oriented BMPs, such as cover crops and conservation tillage, that may decrease losses of some forms of nutrients (e.g., particulate P) but increase losses of other forms (e.g., dissolved P). Some of the BMPs that warrant further attention from public agencies, producers and research, alike, include:

- Source-oriented BMPs such as:
 - Managing feeding strategies to reduce N and P content in manure
 - Formulating diets to improve utilization of feed N and P by pigs
 - Managing manure N and P to balance application with crop removal

- Transport-oriented BMPs such as:
 - Reduce erosion of particulate N and P, where necessary
 - Reduce runoff of dissolved N and P, the dominant forms in runoff
 - Reduce direct, incidental losses (e.g., application into watercourses)
 - Reduce leaching of N and P into groundwater

Risk indicators and regulatory strategies, too, are most likely to succeed in predicting and reducing N and P losses if they are based on a scientifically sound understanding of the major source and transport factors controlling those losses. Manitoba's N-based manure management regulations incorporate both groups of factors by imposing progressively restrictive residual soil test nitrate limits on soils and landscapes where the risk of water movement below the root zone is greatest. Manitoba's newly introduced manure P regulations also include source factors (e.g., Olsen soil test P thresholds) and transport factors (e.g., special restrictions for the Red River Valley and areas adjacent to waterways). More sophisticated measures for source and transport risks affecting P loss (e.g., degree of P saturation soil tests, site-specific P indexes) have been used in other states or provinces and have the potential to improve our ability to predict and restrict P loss, but appropriate versions of these tools have not yet been fully validated for Manitoba's soils, crops, landscapes, climate and management practices, especially under field conditions during snowmelt.

Overall, Manitoba's approach to dealing with the environmental challenges of livestock manure nutrient management is reasonably sound, but worthy of continued refinement. Such refinement will only succeed if the Province works closely with other levels of government, researchers and livestock producers to ensure that the BMPs that livestock producers are expected to implement are technically sound, economically affordable and adequately encouraged by a coherent, comprehensive and supportive range of education, incentive and regulatory policies.

2.2 Introduction

Nitrogen and phosphorus are nutrients required for all forms of terrestrial and aquatic life. However, according to recent studies completed by the Provincial Government, there are increasing concentrations of nitrogen (N) and phosphorus (P) in the rivers draining major watersheds in Southern Manitoba (Jones and Armstrong 2001). Agriculture is one of many "non-point" contributors to the N and P loads in these watersheds. Manure N and P from pigs and other livestock can enrich soil N and P in areas of high density of confined livestock operations. Especially if concentrations of soil test N and P are allowed to rise to excessive levels, some of this N and P may reach important water bodies such as Lake Winnipeg. Existing information regarding the behaviour of N and P in agricultural production systems is an important tool for developing and encouraging appropriate agricultural management practices that will minimize the risk of N and P transfer to water bodies.

2.2.1 Benefits of nitrogen and phosphorus for crop and livestock production

As with all plants and animals, crops and livestock require nitrogen (N) and phosphorus (P) to live and grow. For livestock and poultry production, the required amounts and proportions of nutrients deemed essential for optimal production and performance are recommended by the National Research Council (Table 2.1).

Table 2.1 Daily phosphorus and nitrogen requirements of dairy cattle, beef cattle, poultry, and growing pigs.

	Phosphorus Requirement g/day	Nitrogen Requirement g/day
<i>Dairy Cattle</i>		
lactating dairy cow	66	459.2
bred heifer	29	258.4
<i>Beef Cattle</i>		
growing and finishing	13	70.7
<i>Poultry</i>		
broilers	0.57	5.2
<i>Pigs (by kg weight)</i>		
3-5	1.75	10.4
5-10	3.25	19.0
10-20	6.00	33.4
20-50	9.28	53.4
50-80	11.59	63.9
80-120	12.30	64.9

Source: NRC 2001; NRC 1996; NRC 1994; NRC 1998.

Basis for determination of N and P requirements is explained in note 2.1 in the appendix.

For crop production, the typical N and P requirements of crops vary with species and yield (Table 2.2). When these nutrients are exported to consumers, they must be replaced or recycled in order to maintain the sustainability of our food production system, especially with steadily increasing global demand for crop products. For example, over a five year period (2000-2004) crop removal of nitrogen and phosphorus in Manitoba averaged 279,552 tonnes N and 47,190 tonnes P (A. M. Johnston, personal communication, International Plant Nutrition Institute, Saskatoon, SK). In livestock production systems, a high proportion of the nutrients fed to the animals is excreted as manure. Therefore, application of livestock manure onto cropland provides an important means of recycling and replacing the nutrients exported by crop production systems.

Most crop N and P requirements are supplied with synthetic fertilizers

On typical farmland in Manitoba, nitrogen and phosphorus are often the most limiting nutrients for good crop growth and long term fertility. Therefore, N and P are the nutrients most commonly added as synthetic fertilizer onto Manitoba farmland. For example, in the 2006 crop production year, 292,904 metric tonnes of N and 41,414 metric tonnes of P were sold as synthetic fertilizers in Manitoba (Canadian Fertilizer Institute 2007). In Manitoba, fertilizer inputs of N and P have closely followed crop removal of N and P over the past 40 years (Figure 2.1). Interestingly, both N and P fertilizer use has declined in Manitoba in recent years, perhaps as a result of substantial increases in the cost of synthetic fertilizers. For example, N and P fertilizer sales for 2006 decreased by 12.3% and 25.7%, respectively from 2003 sales.

Table 2.2. Typical nitrogen and phosphorus requirements and removals for Manitoba crops.

Crop Species	Typical Yield	Average Nutrient Requirement for Production of Whole Crop			Average Nutrient Removal Rate in Harvested Portion		
		Nitrogen	Phosphorus	N:P	Nitrogen	Phosphorus	N:P
	tonnes/ha	kg/tonne			kg/tonne		
Spring Wheat	2.7	35	5.7	6.2	25	4.4	5.7
Winter Wheat	3.4	23	4.4	5.3	17	3.9	4.3
Barley	4.3	29	5.2	5.6	20	3.9	5.1
Barley (Silage)	10.1	n/a	n/a	n/a	17	2.6	6.5
Oats	3.8	32	5.2	6.1	18	3.5	5.2
Rye	3.5	30	6.5	4.6	19	3.5	5.4
Corn (Grain)	6.3	27	4.8	5.6	17	3.5	4.9
Corn (Silage)	11.2	n/a	n/a	n/a	16	2.6	6.1
Canola	2.0	64	13.0	4.9	39	9.1	4.3
Flax	1.5	52	6.5	8.0	38	5.2	7.3
Sunflowers	1.7	50	7.4	6.8	36	4.8	7.5
Alfalfa	11.2	n/a	n/a	n/a	29	3.0	9.5
Grass	6.7	n/a	n/a	n/a	17	2.2	7.8

Source: Farm Practices Guidelines for Pig Producers in Manitoba (2007).

Meeting crop N and P requirements with pig manure

The value of manure as a source of crop nutrients depends on the abundance and relative availability of N and P in manure. Livestock manure's value as a source of crop nutrients varies with livestock species (Table 2.3). With specific reference to pig manure, the value depends on the type of pig operation (i.e. based on pig growth stage) (Table 2.4), feed management (e.g. use of phytase to reduce P excretion) (Table 2.4) and manure storage and handling (Tables 2.5 and 2.6). The variability in manure nutrient content is more pronounced for P than for N. For example, there is over 500 times difference in the minimum and maximum P content of the pig manures sampled versus a 17 fold range in N content variability (Table 2.3). The relative amounts of N to P in manure of N and P in manure can also vary substantially, particularly when based on the availability of manure N for crop use.

Overall though, all livestock manures have a low concentration of nutrients compared to synthetic fertilizers. For example, the N concentration in typical liquid pig manure is 150 times more dilute than in urea fertilizer (46-0-0) and the P concentration is 230 times more dilute than monoammonium phosphate fertilizer (11-52-0). Due to this low nutrient content in manure, a much larger amount of manure must be applied to supply an equivalent amount of N and P supplied in synthetic fertilizer and the costs for transporting manure more than a few miles from the manure storage are very high.

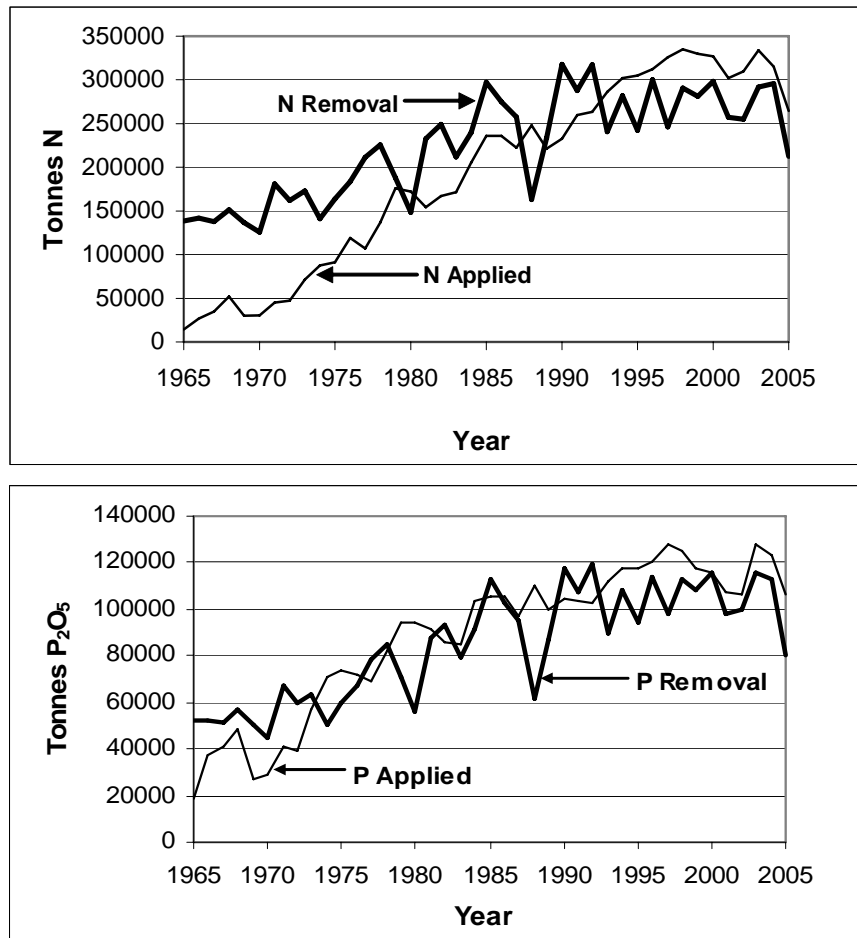


Figure 2.1. Fertilizer additions and crop removal of nitrogen and phosphate for agricultural land in Manitoba from 1965-2005 (Source: A. M. Johnston, personal communication, International Plant Nutrition Institute, Saskatoon, SK).

Table 2.3. Average nitrogen and phosphorus content for various types of livestock manures.

Operation Type	Dry Matter %	Total N mean (range)	Ammonium-N mean (range) kg/1000L	Available N	Total P mean (range) kg/1000L	Ratio Total N to Total P	Ratio Available N to Total P
Liquid Pig (n=133)	3.4	3.1 (0.4-6.8)	1.9 (0.2-5.2)	1.2	1.0 (0.0-5.1)	3.1	1.2
Liquid Poultry (n=35)	9.1	8.0 (3.0-14.2)	5.8 (0.1-10.5)	3.8	2.8 (0.6-5.1)	2.9	1.4
Liquid Dairy (n=252)	8.9	3.4 (0.7-7.6)	1.5 (0.0-7.2)	1.0	0.9 (0.1-8.5)	3.8	1.1
Solid Beef (n=45)	26.4	6.0 (1.4-20.2)	0.6 (0.0-2.7)	0.4	1.4 (0.3-6.4)	4.3	0.3

Note: P is expressed as elemental P, not P₂O₅ as in fertilizer analyses; Available N (calculated for average values only) = NH₄-N x 0.65 (35% average ammonia loss for incorporation within 3 days which equals loss if applied to standing crop) + 0.25 Organic N (25% of organic N); N:P calculated for average values only.

Source: Tri-Provincial Manure Application and Use Guidelines (2003)

Table 2.4. Manure nitrogen and phosphorus content for liquid pig manure from various types of operations.

Operation Type	Without Phytase Supplemented Diets						
	Dry Matter %	Total N mean (range)	Ammonium-N mean (range) kg/1000L	Available N	Total P mean (range) kg/1000L	Ratio Total N to Total P	Ratio Available N to Total P
Farrow (n=37)	3.0	1.7 (0.6-6.5)	1.2 (0.5-3.2)	0.7	0.7 (0.0-5.5)	2.6	1.1
Nursery (n=11)	3.1	2.7 (1.5-4.6)	2.0 (1.3-3.0)	1.5	1.2 (0.1-2.6)	2.3	1.3
Finisher (n=92)	3.7	3.4 (1.5-6.4)	2.6 (1.2-4.1)	1.9	1.0 (0.0-3.5)	3.4	1.9
Farrow to Finish (n=5)	2.1	2.8 (1.2-4.0)	2.4 (1.1-3.4)	1.7	1.0 (0.1-2.1)	2.7	1.6
Operation Type	With Phytase Supplemented Diets						
Farrow (n=132)	2.1	2.2 (0.4-6.0)	1.6 (0.4-2.7)	1.2	0.7 (0.0-4.1)	3.4	1.9
Nursery (n=58)	2.2	2.7 (1.1-5.6)	2.0 (0.6-3.3)	1.5	0.5 (0.0-2.0)	5.6	3.1
Finisher (n=181)	3.4	3.4 (0.4-6.7)	2.4 (0.4-4.7)	1.8	0.7 (0.0-2.8)	5.2	2.8

Note: P is expressed as elemental P, not P₂O₅ as in fertilizer analyses; Available N (calculated for average values only) = NH₄-N x 0.65 (35% average ammonia loss for incorporation within 3 days which equals loss if applied to standing crop) + 0.25 Organic N (25% of organic N); N:P calculated for average values only.

Source: Farm Practices Guidelines for Pig Producers in Manitoba (2007).

Table 2.5. Manure nitrogen and phosphorus content of various forms of solid pig manure from finishing operations.

	Dry Matter %	Total N Mean (range)	Ammonium-N Mean (range)	Available N	Total P Mean (range)	Ratio Total N to Total P	Ratio Available N to Total P
n=10	%	kg/tonne			kg/tonne		
Fresh	33.2	6.3 (4.8-8.9)	1.4 (1.0-1.8)	2.14	2.83 (2.2-4.3)	2.2	0.8
Stockpiled ¹	47.1	6.0 (3.7-12.9)	2.1 (1.1-3.1)	2.34	2.39 (1.5-3.8)	2.5	1.0
Composted: Fresh	66.4	6.4 (5.0-8.1)	0.5 (0.1-1.2)	1.8	4 (3.3-4.6)	1.6	0.5
Compost: Stockpiled	58.2	7.2 (5.8-8.8)	0.7 (0.2-1.0)	2.08	3.74 (2.8-4.3)	1.9	0.6

Note: P is expressed as elemental P, not P₂O₅ as in fertilizer analyses; Available N (calculated for average values only) = NH₄-N x 0.65 (35% average ammonia loss for incorporation within 3 days which equals loss if applied to standing crop) + 0.25 Organic N (25% of organic N); N:P calculated for average values only.

¹ Minimally disturbed for approximately 6 months.

Source: Farm Practices Guidelines for Pig Producers in Manitoba (2007).

Table 2.6. Manure nitrogen and phosphorus content for various pig manure storage systems.

	Dry Matter	Total N	Available N	Total P	N to P Ratio	Available N to P Ratio
	%	kg/1000L				
<i>Sample Depth</i>						
Top (n=62)	2.3	2.5	2.2	0.6	4.2	3.6
Middle (n=30)	3.7	2.7	2.2	0.9	3.2	2.6
Bottom (n=53)	4.6	3.2	2.5	1.3	2.5	1.9
<i>Type of Manure Storage</i>						
Open Earthen (n=114)	3.3	2.9	2.3	0.9	3.3	2.7
Open Earthen - Primary (n=9)	8.0	3.1	2.3	1.3	2.4	1.7
Open Earthen - Secondary (n=14)	1.7	2.0	1.6	0.8	2.6	2.1
Slurry (n=8)	2.9	2.8	2.6	1.1	2.5	2.3

Note: P is expressed as elemental P, not P₂O₅ as in fertilizer analyses; Available N (calculated for average values only) = NH₄-N x 0.65 (35% average ammonia loss for incorporation within 3 days which equals loss if applied to standing crop) + 0.25 Organic N (25% of organic N); N:P calculated for average values only.

Source: Fitzgerald and Racz (2001)

Numerous studies have demonstrated the capacity of pig manure to increase both yield and quality for annual crops and perennial forages (Beauchamp et al. 1997; Mooleki et al. 2002; Schoenau et al. 2005; Ominski et al. 2007; Sager and Przednowek 2005). Therefore, in spite of the challenge of high transportation costs for manure, recent increases in fertilizer prices have made the use of manure as a source of crop nutrients an increasingly cost effective option.

Land availability and utilization for applying manure

The amount of pig manure nutrients currently produced is far less than the province-wide crop nutrient demand and therefore could supply only a small portion of the annual requirements for total cropland production in Manitoba. The total gross amount of N and P in pig manure produced province-wide is approximately 22,500 to 24,000 tonnes of N and 5,000 to 7,000 tonnes of P (for more discussion of the range in estimates, please refer to the Introduction chapter in this report), figures that are only 8-9% and 12-17% percent of the amount of N and P synthetic fertilizers applied and 8-9% and 11-15% percent of N and P removed by Manitoba crops.

Pig manure is applied to approximately 120,000 ha of land in Manitoba or 2.5% of crop land (for more information on the methods, challenges and uncertainties of this estimate, please refer to the Introduction to this report). By comparison, synthetic fertilizer is applied to about 3.45 million hectares or 73% of Manitoba's 4.70 million ha of agricultural land. Based on the quantity of nutrients produced as pig manure and the land area used for liquid manure application in Manitoba, this means that, on a gross basis, pigs excrete the equivalent of approximately 190-200 kg N and 42-58 kg P per ha of manured land. However, these values for N and P production are not equivalent to actual rates of application, since they do not account for losses during the storage, handling and application of the manure, losses that are particularly substantial for N. For example, USDA scientists estimated that 75% of N excreted by U.S. pigs was lost prior to land application (Kellogg et al. 2000) in a recent study on manure nutrient balances. Although N losses from Canadian pig operations are probably less than those from U.S. operations, the real rates of N application in Manitoba are still much less than rates of excretion; otherwise, the frequency of excess nitrate concentrations in manured land would be much higher than currently observed in Manitoba

Conservation's audit program. Conversely, these figures indicate that P, which is not lost as easily as N during manure storage and handling, is being applied at rates that substantially exceed the average rate of P removed by crops in the province (10 kg P per ha as calculated by A. M. Johnston, International Plant Nutrition Institute, Saskatoon, SK). Even after accounting for above-average crop yields and P removal for crops that are grown on manured land (e.g., the 15 kg P per ha figure used in some of Manitoba Pork Council's reports), a substantial surplus of P is often applied, reflecting the fact that P-based manure application regulations were introduced only recently, in November 2006. Also, if pig manure P was applied at a rate to match average crop removal in Manitoba, approximately 500,000 to 700,000 ha of land would be required, representing 10-15% of the province's agricultural land base, figures that are much larger than 300,000 ha (742,000 acres) or 6% cited in Manitoba Conservation's report on the sustainability of Manitoba's pig industry (Manitoba Conservation 2006).

Other constituents in manure

In addition to N and P, manure applications to soil supply nutrients, salts, metals and minerals such as Na, K, Al, Mg, Ca, Mn, Fe, Cu, Zn, Cl, etc, which can lead to build up of some manure constituents (Fitzgerald and Racz 2001). However, the risk of excessive accumulations of these constituents is generally low if manure is applied at recommended rates, especially under the new P based manure application regulations. For example, following five to eight years of applying liquid pig manure addition at agronomic rates at various sites in Saskatchewan, Schoenau et al. (2005) reported no apparent increases in salinity (high salts accumulation) and sodicity (high sodium accumulation) and limited accumulation of trace metals.

Non-nutritional benefits of manure

Manure may also indirectly benefit crop productivity by improving many soil properties associated with crop growth. Manure addition to soil may improve soil tilth by decreasing soil bulk density, increasing soil aeration or porosity, increasing soil organic matter and soil organic carbon and increasing water infiltration (Sommerfeldt and Chang 1985; Meek et al. 1982; Schoenau et al. 2005). These improvements may be attributed directly to manure addition or indirectly via enhanced plant root growth and uptake of nutrients. For example, Coppi and Tenuta (2007) measured substantial increases in root biomass of grasses following 3 years of liquid pig manure addition to grassland near La Broquerie, Manitoba while in Saskatchewan, Schoenau et al. (2005) reported increases in the light fraction of organic carbon (organic material associated with decomposing plant roots and residues) after 8 years of liquid pig manure.

2.2.2 Environmental risks of N and P loading from manured land to surface and ground water

Protecting surface and ground water quality is important for everyone in Manitoba. Rural and urban residents depend on wells and local surface water sources for drinking water, domestic use, municipal (community and local) use, and for use in industrial and livestock/agricultural operations. These waterways are the initial receiving bodies of nutrient input and are therefore more directly impacted by nutrient management, compared to lakes such as Lake Winnipeg, the end receiving bodies.

Water quality concerns and guidelines for N in groundwater

In groundwater, nitrate contamination is a specific concern when limits exceed 10 mg N/L, because high nitrate concentrations are linked with methemoglobinemia or blue baby syndrome in infants (Ward et al. 2005).

Water quality concerns and guidelines for N and P in surface water

Nutrient concentrations in many Manitoba rivers approach or exceed existing guidelines for water quality for both aquatic organisms and human consumption. Smaller, more static water bodies such as dugouts and reservoirs that may serve as drinking water for livestock or people, may be particularly sensitive to nutrient loading where mixing and water cycling is limited. Human health concerns should supersede esthetic water quality concerns when managing nutrient entry to waterways.

Nutrient enrichment of streams, rivers, lakes and other surface waters in Manitoba (e.g., Lake Winnipeg) has led to concern regarding natural and human-caused loading of nutrients to surface water. The primary nutrients of concern are phosphorus and nitrogen as these are the two main nutrients associated with algae growth and eutrophication (Bourne et al. 2002).

- **Phosphorus** is commonly the most limiting nutrient for algae growth in aquatic freshwater environments, particularly for large lakes. In smaller or shallow water bodies or rivers, water flow, nutrient cycling and light penetration are different than in large lakes, complicating the relationship between nutrient loading and eutrophication (Environment Canada 2004). However, based on CCME trophic categories for Canadian freshwater ecosystems, water bodies (includes lakes and streams) are classified as eutrophic if the total P content is within a range of 0.035 - 0.10 ppm (Table 2.7; CCME 2004). In Manitoba, the total P concentration threshold in freshwater, above which eutrophication may be enhanced, is set as 0.025 ppm total P for lakes, ponds, reservoirs and entry points and 0.05 ppm total P for streams and rivers (Table 2.7; Manitoba Conservation 2002).

However, numerous rivers in Manitoba have elevated phosphorus concentrations (Table 2.8), many naturally, and would be classified as eutrophic using these values. The differences in nutrient levels of water bodies can be related to regional location. For example, P concentrations along the Roseau River in Manitoba gradually increase from east to west as the river enters the fine-textured and fertile soils of the Red River Basin (Ralley 1998). Similarly, long-term water monitoring data shows as a whole, watersheds in the Eastern region of Manitoba contain higher total phosphorus concentrations and export greater total P quantities annually than watersheds located in the Western region of the province (Salvano and Flaten 2006).

- **Nitrogen** can induce eutrophication in freshwater environments at concentrations of 0.5 to 1.0 mg N/L (Pierzynski et al. 2005b), but this effect is also dependent on the N:P ratio in water. The United States Environmental Protection Agency established “eco-regional criteria” for total N at 0.12 to 2.18 mg/L for streams and rivers and 0.10 to 1.27 mg/L for lakes and reservoirs (Pierzynski et al. 2005b). The nitrogen content of many streams and rivers in Manitoba is within the upper range of these limits (Table 2.8). Canadian Water Quality Standards for the Protection of Aquatic Life (CCME updated 2006) also define thresholds for several forms of N in fresh water (Table 2.7). Ammonia, in particular, is highly toxic to aquatic life in low concentrations and receives the most attention of the N species found in surface water. In water, ammonia and ammonium (the ionized form) exist in equilibrium, a relationship that is pH and temperature dependent, favouring a shift to ammonia at high pH and temperature (Jones and Armstrong 2001).

Table 2.7. Maximum recommended nitrogen and phosphorus concentrations for freshwaters.

	CCME Guidelines for Aquatic Life	Manitoba Water Guidelines
Total P	0.035-0.10 mg /L	water bodies: 0.025 mg /L rivers and streams: 0.050 mg /L
Ammonia-N	pH and temperature dependent 0.015 mg N/L	pH and temperature dependent
Nitrate-N	2.9 mg N/L	10 mg N/L nitrate+nitrite N in drinking water
Nitrite-N	0.06 mg N/L	

Sources: Canadian Council of Ministers of the Environment (2004); Manitoba Conservation (2002).

Table 2.8. Nitrogen and phosphorus concentrations of select rivers in Manitoba.

Rivers and Tributaries	Mean Total Nitrogen mg/L	Mean Total Phosphorus mg/L
Assiniboine River	1.55	0.21
Assiniboine River Tributaries		
Qu'Appelle River	1.25	0.17
Little Saskatchewan	2.06	0.15
Souris River	2.03	0.29
Cypress River	1.61	0.26
Red River	2.08	0.26
Red River Tributaries		
LaSalle River	2.00	0.46
Roseau River	1.18	0.09
Rat River	1.35	0.15
Boyne River	1.52	0.15
Pembina River	1.49	0.28
Seine River	1.53	0.23
Selection of Other Streams Sampled		
Brokenhead River	1.10	0.06
Dauphin River	1.40	0.02
Nelson River	0.52	0.03
Ochre River	0.67	0.06
Swan River	1.28	0.18
Turtle River	0.73	0.06
Whitemud River	1.50	0.12
Winnipeg River	0.51	0.02
Minimum	0.41	0.02
Maximum	2.35	0.46
Mean	1.32	0.16
Median	1.30	0.15

Adapted from Manitoba Water Stewardship (2006).

2.2.3 An example of excessive nutrient loading: Lake Winnipeg

General overview of N and P sources

The total contribution of Manitoba sources to annual nitrogen and phosphorus loading to Lake Winnipeg represents 49% and 47% of the total N and P loading respectively, to the Lake (Table 2.9). The remaining N and P comes from the United States, Saskatchewan, Alberta and Ontario. The Red River supplies the greatest amounts of phosphorus and nitrogen to Lake Winnipeg: 54% of the annual phosphorus load and 30% of the annual nitrogen load (Lake Winnipeg Stewardship Board 2006). Bourne et al. (2002) estimate total N and total P loads to Lake Winnipeg from the Red River have increased by 13% and 10%, respectively over the past 3 decades.

Table 2.9. Estimated annual nitrogen and phosphorus loading to Lake Winnipeg from Manitoba sources (1994-2001).

	Total Nitrogen tonnes/year	% of Total Nitrogen	Total Phosphorus tonnes/year	% of Total Phosphorus
Manitoba Sources	47,100	49%	3700	47%
Manitoba Point Sources of Wastewater (City of Wpg, etc)	5,100	5	700	9
Manitoba Watershed Processes				
Natural background + other ¹	18,100	19	1,300	17
Current Agriculture	5100	5	1,200	15
Atmospheric Deposition	9,500	10	500	6
Internal Lake Processes (N fixation by blue-green algae)	9,300	10		

Adapted from Lake Winnipeg Stewardship Board (2006).

¹would also include contributions from sources such as forests, wildlife and septic fields.

Sources of agricultural N and P to Lake Winnipeg, including Manitoba's pig industry

Nutrients from crops, fertilizers and manures can be transferred from agricultural land to surface and ground waters. Manitoba agricultural production contributes an estimated annual loading of 5,100 tonnes of nitrogen per year and 1,200 tonnes of phosphorus per year to Lake Winnipeg, representing 11% and 32%, respectively, of the total inputs coming from Manitoba. In comparison, estimated natural background sources contribute 18,100 tonnes of nitrogen and 1,300 tonnes of phosphorus per year.

2.2.4 Manure production and use in Manitoba

Livestock Sources of Manure in Manitoba

Transfer of nutrients from livestock manures to surface and ground water receives considerable attention, particularly the contribution from the expanding pig industry in Manitoba. As of January 1, 2007, there were an estimated 3.0 million pigs and 1.4 million cattle (dairy and beef combined) in Manitoba (Statistics Canada 2007). The amount of manure produced from a pig operation will depend on the type of operation and the age of pigs as pigs of different size produce different amounts of manure. For example, according to Statistics Canada (2006) census data, on census day nursing/weaner pigs comprised about 40% of the total number of pigs on farms, compared to about 47% grower and finisher pigs. However, the small pigs produced less than 20% of the total pig manure and phosphorus due to much lower feed demand, while grower and finisher pigs generated approximately 51% of the total manure and 55% of the total P calculated.

Although the total amount of manure N and P excreted by cattle is much greater than for pigs, a high proportion of the manure N and P excreted by cattle is deposited in pasture grazing systems, where the nutrients from the local forage crop are recycled in place, with little risk of excessive accumulation of imported nutrients. Conversely, pigs are generally fed with imported feed in barns where all pig manure is mechanically collected and applied onto cropland. Therefore, pig manure must be managed carefully to ensure that nutrients are not applied at rates that exceed crop removal.

Manitoba's Pig Industry: by region and relative to other livestock types

Intensive livestock operations, as are common in Manitoba's pork industry, have the potential to concentrate large numbers of animals on a comparatively small land area. Therefore, access to enough suitable land for spreading manure without detrimental impact on water quality may be a concern in areas with a high density of intensive livestock production. The RMs of La Broquerie and Hanover, both located in southeastern Manitoba, have the highest livestock densities in the province, reporting 445,683 and 388,905 pigs respectively on census day in 2006 (Statistics Canada 2006) which represents 28% of the total number of pigs in the province on census day.

N based manure management regulations often lead to excessive P loading

For any type of agricultural operation to be sustainable over the long term, nutrient inputs should be balanced with nutrient removals. To minimize manure-sourced nutrient loading to Manitoba water ways, the province has implemented N based manure management regulations for many years. These regulations limit the rate of manure application to farm land based on the crop's agronomic requirements for N. However, the proportions of all nutrients in manure do not match the proportions required for optimal crop growth, and vary depending on animal and operation type and manure storage and handling. For example, the typical crop ratio of nitrogen to phosphorus is 6 to 1 while the typical nitrogen to phosphorus ratio of manures can vary between 2 and 4 to 1 (Tables 2.3 to 2.6). Applying manure based on *available* N, to account for the portion of nitrogen that may be unavailable for crop uptake due to losses or slow rates of release, further lowers the nitrogen to phosphorus ratio to less than 2 to 1 for many manures. Therefore, regular applications of manure based on crop nitrogen requirements will apply excess P and result in accumulations of P in soil, which in turn increases the risk of P loss to surface water.

P based manure management regulations for reducing the risk of P loss

Awareness of the detrimental effects of excess P entering surface waters and the divergence between N:P ratios in manure and crops has led many countries including the United States to adopt phosphorus-based manure management regulations (reviewed in Heathwaite et al. 2005 and Flaten et al. 2003). Similarly, Manitoba's concern about excess phosphorus loading of surface waters prompted the adoption of regulations for manure applications based on soil phosphorus concentrations, effective November 2006. However, Quebec and Ontario are the only other provinces in Canada with phosphorus based manure management regulations.

2.2.5 The Source-Transport-Receptor Model for describing risk of N and P loss to water

Manure nutrient management regulations vary from one jurisdiction to another for a variety of reasons, including differences in the aquatic and terrestrial ecosystems for the region. However, to avoid inadvertent harm to the environment or to livestock producers or both, all regulations should be developed on a sound science base that considers the risk of nutrient loading.

In a variation of the source-pathway-receptor model that is frequently used to assess the risk of any form of unwanted contamination, the risk of nitrogen and phosphorus transfer from field-applied pig manure or any other source of nutrients to ground water or surface water depends on two main groups of factors (Figure 2.2):

1) **Source Factors** – affect the quantity of N and P in the soil available for environmental losses. Source factors address the:

- net loading rate of N and P – the balance between N and P inputs and removals
- availability of N and P for plant uptake or loss to surface water as affected by:
 - form and amounts of N and P in manure
 - the rate at which manure is applied
 - placement of manure
 - timing of manure application
- capacity of the soil and vegetation to retain added N and P.

The potential for loss increases as N and P accumulate in soil and overwhelm the soil's ability to retain those nutrients. However, some risk for nutrient loss still exists at all concentrations of N and P because the system is not closed -- it is "leaky".

2) **Transport Factors** – determine the loss pathway and ease of N and P movement along that pathway. Transport factors for N and P contamination of surface and groundwater include:

- climate and weather factors - quantity of snow and duration of snowmelt period; frequency, duration, intensity and quantity of rainfall events
- landscape - slope, hydrology, surface condition, depth to water table, proximity to water bodies
- soil – texture, structure, stratification, water retention and infiltration capacity
- land and water management – soil and crop management, equipment and operations, drainage, vegetated areas

Transport factors connect the source to the receptor and determine if the potential for N or P loss at the source translates to actual nutrient loss to surface or ground water. Therefore, source and transport factors must coincide at the same location and at the same time in order for N and P to move from land to water.

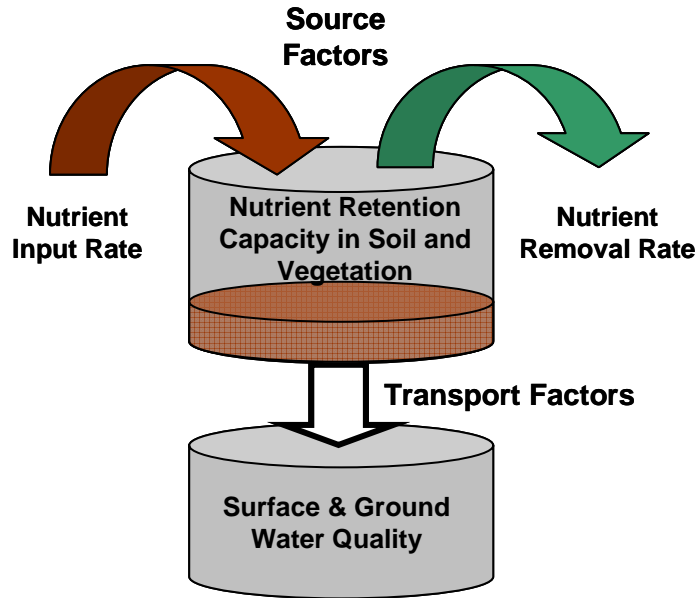


Figure 2.2. The risk of nutrient loss from land to water depends on the balance of nutrient inputs and removal, along with the capacity of soil and vegetation to retain nutrients and the intensity of transport factors (adapted from Flaten 2003).

Phosphorus and nitrogen in soil can be transported to surface water with overland or subsurface flow of water (Pierzynski et al. 2005b; Haygarth and Sharpley 2000). Phosphorus, and more commonly nitrogen, can also leach to groundwater over time. Atmospheric nitrogen or phosphorus potentially derived from manure or manured soil may also be deposited with snowfall (Maule and Elliott 2006b).

As mentioned previously, the combination of source and transport factors must occur at the same time in the same place for N or P to be lost from land to water. For example, there is little risk of nutrient transport to surface water from runoff-susceptible land that has low concentrations of nutrients (low source risk) and there is little risk of nutrient transfer from a nutrient rich soil that does not drain to surface water (low transport risk). It is the combination of susceptible source and transport factors existing together in the presence of favourable weather event (e.g., rainfall or snowmelt) where nutrient transfer to surface or ground water can occur. As such, nutrient loss is generally confined spatially and temporally (Sharpley et al. 2001a). The relative importance of the various source and transport factors is governed by regional land and climatic factors (Salvano and Flaten 2006; Sharpley et al. 2005). Therefore, an understanding of how those factors interact in the region is necessary for developing beneficial management practices that will control nutrient loss in the region (Kleinman et al. 2006). Conversely, without this understanding, the practices purported to be beneficial and encouraged by education, incentives and regulations are likely to be ineffective, inefficient and/or counter-productive.

2.3 NITROGEN MANAGEMENT AND WATER QUALITY

Nitrogen is the most limiting nutrient in agricultural crop production. Liquid pig manure is a good source of nitrogen and other essential nutrients. However, compared to synthetic fertilizer, the nutrient concentrations in manure are low and a much larger quantity of manure is required to supply N at agronomic rates. Once applied to the soil, pig manure N is largely available for crop uptake or potentially available for environmental loss. The processes and management practices that determine the risk of N loss can be classified in two main groups: source factors and transport factors.

2.3.1 Factors Affecting Risk of N Loss at the Source

Source factors can be divided into the subfactors of i) inputs, ii) removals and iii) soil and vegetation retention and release. Generally, when inputs of N exceed crop removal, nitrogen can accumulate in the soil.

2.3.2 Sources of N Inputs

In addition to pig manure, there are many sources of N inputs to agricultural land that can contribute to N losses to surface water. The contributions from these other sources and how they fit within the context of manure management is important to account for in developing manure management strategies to reduce N losses from agricultural land.

○ **Atmospheric Nitrogen Inputs**

Atmospheric ammonia volatilized from manures can be re-deposited directly into surface waters or onto land (Pierzynski et al. 2005b). For example, nitrous oxides released to the atmosphere from human activity and denitrification of nitrate can be converted to nitric acid and deposited as acid rain or dry deposition (Pierzynski et al. 2005b). As a result, direct atmospheric deposition of nitrogen contributes an estimated 9,500 tonnes of nitrogen per year to Lake Winnipeg, representing 20% of the total loading from Manitoba sources (Lake Winnipeg Stewardship Board 2006; note that this amount does not include the estimated 9,300 tonnes of nitrogen loaded into Lake Winnipeg from the atmosphere by N fixing blue-green algae). This amount is equivalent to 186% of the contribution from Manitoba point sources and 186% of the estimated contribution from Manitoba's agricultural industry.

Additionally, atmospheric loading with snow and onto snow-covered land can contribute significant quantities of N that may be indirectly responsible for some of the N loading into surface waters during snowmelt. For example, Maule and Elliott (2006b) measured nitrogen concentrations in snow over a six-year period near Elstow, Saskatchewan and reported concentrations of ammonia-N ranging from 0.240 to 1.45 mg N/L and nitrate+nitrite-N from 0.210 to 1.35 mg N/L. However, the mobility and environmental impact of this "snow N" is not clear as some of this N can be lost prior to snowmelt.

○ **Synthetic Nitrogen Fertilizer Inputs**

In Manitoba, synthetic nitrogen fertilizer is routinely applied to agricultural land in a variety of forms, predominantly as granular urea (46-0-0), pressurized anhydrous ammonia (82-0-0), and liquid nitrogen solutions such as urea ammonium nitrate solution (e.g. 28-0-0). Fertilizer sales for the 2006 crop production year totaled 552,756 tonnes of nitrogen fertilizer with similar amounts of each of these three forms sold. Use of N fertilizer has declined 12% in Manitoba since 2003 (Canadian Fertilizer Institute 2007).

- **Manure Nitrogen Inputs**

As stated earlier, the amount of manure nitrogen excreted by pigs in Manitoba is equivalent to only 8-9% of the synthetic N fertilizer applied to crops and 8-9% of the N removed by crops in Manitoba. After accounting for N losses during manure storage, handling and application, the net quantity of N applied as pig manure represents a relatively small portion of the N applied onto agricultural land in the province. Nonetheless, care must be taken to ensure manure is applied at appropriate rates and that the small amount of land used for manure application does not generate large losses of nutrients.

2.3.3 Agronomic and Environmental Performance of Pig Manure as a N Fertilizer

Availability of N in liquid pig manure versus synthetic N fertilizers

Liquid pig manure generally contains a high percentage of its N in the form of inorganic ammonium-N (Table 2.4) which is readily converted in soil to nitrate-N; therefore, pig manure has a large pool of N readily available for plant uptake or potential loss under adverse environmental conditions. The forms of N in *synthetic commercial fertilizer* can be ammoniacal-N, urea-N or nitrate-N and can be in granular or liquid form. Nitrogen in synthetic fertilizers is either readily available as nitrate or is rapidly converted (e.g. urea and ammoniacal-N) to nitrate-N. Regardless of its original source, nitrate-N, which is highly available in soil solution, can be removed from the soil by plant uptake or by environmental loss via surface runoff, leaching or denitrification. The main risk of loss for ammoniacal N from fertilizer or manure is ammonia volatilization.

Availability of N in solid manure

Very little research has been conducted with solid pig manure in the Prairie Provinces. However, solid or composted pig manure (Table 2.5) or other solid manure such as cattle manure (Table 2.3) contains a large pool of organic N that converts to available inorganic N over a period of years, gradually releasing N to the soil for plant uptake or potential loss, the rate of conversion depending on the size and stability of the organic pool. As an example of the gradual release of mineralized N from solid manures and composts, Eghball (2000) found 21% of solid cattle feedlot manure and 11% of composted manure was mineralized during the first growing season following late fall application. In a separate study, first year availability of solid cattle feedlot manure N and compost N were 40% and 15%, respectively, while availability in the 2nd year diminished to 18% and 8%, respectively (Eghball and Power 1999).

The reason for the slow release of N from solid manure is its high ratio of carbon to nitrogen (C:N); the microorganisms have to decompose the carbon first, before they will release the nutrients for plant uptake. For example, Qian and Schoenau (2002) found that for various solid forms of manure, manures with a C:N ratio above 15:1 decreased short-term N availability (indicating net immobilization) while manures with a C:N of about 13-15:1 showed limited release of available N.

Composting manure decreases the amount of carbon and nitrogen, where the remaining N fraction is more stable organic N (Eghball et al. 1997) which further reduces the available N content. Helgason et al. (2007) found nitrogen uptake from composted cattle manure was directly proportional to manure inorganic N content and that only a small proportion of the organic manure N would have been available in the year of application. Therefore, to fully supply adequate N to a crop, higher rates of manure are required with solid or composted

manures, and therefore greater amounts of manure P, because of this lower and more gradual N availability relative to liquid manures.

As a result of their slow release of N, manures or composts with a large proportion of N in the organic form, such as solid manures, may not supply adequate N to the crop early in the season and therefore may not meet crop N requirements as well as liquid manures. Also, continued release of organic N from high organic N manures over the growing season may result in increased crop uptake of N late in the growing season which may be either beneficial (e.g. wheat - protein boost, or corn - a long season crop) or detrimental (e.g. malt barley – decreased quality), depending on the crop.

Effect of placement on manure N benefit

Subsurface placement of manure or synthetic N fertilizer greatly improves the efficiency of the application by minimizing the amount of N lost as ammonia to the atmosphere. Developments in application technology for liquid manures have greatly improved the efficiency of manure placement to increase plant availability of manure N and decrease N loss in runoff and ammonia volatilization (Bittman et al. 2005; van Vliet et al. 2006). Liquid manure can now be applied in much more narrow bands with closer spacing than previously, either on the surface or subsurface using direct or passive (e.g. over soil aeration slots) injection, suitable for use in both annual crops and established forages. Reducing N loss increases the proportion of *available* N in manure to reduce the total amount of manure required to meet crop needs, in turn reducing the amount of P added.

Effect of application timing on manure N benefit

The most efficient timing of synthetic or manure fertilizer applications coincide with periods of active crop removal of nutrients and the lowest risk of environmental nutrient loss. Therefore, from a nitrogen efficiency standpoint, fall application is less efficient than spring application as a percentage of fall-applied N is either immobilized by the soil microorganisms or lost via denitrification, volatilization or leaching prior to active crop uptake the following spring. Manure applied in the winter is an even less agronomically and environmentally efficient practice as substantial N loss may also be lost with snowmelt runoff, in addition to volatilization and denitrification losses.

Timing manure applications to coincide with crop demands may be more difficult with solid than liquid manure. Relative to liquid manure, solid manure has a larger portion of N as organic N that must first be converted to a plant available form of N, which may temporarily delay the supply of utilizable N to crops. Therefore, application well in advance of the crop may be agronomically desirable and environmentally acceptable for solid manures.

Effect of application rate on manure N benefit

Ideally rates of synthetic or manure fertilizers based on N should parallel crop N requirements to meet yield potentials yet avoid accumulation of N in soil at levels susceptible to runoff or leaching losses. For example, Burns et al. (1990) and King et al. (1990) observed that at excessive rates of manure N application, the relative efficiency of plant N uptake diminishes and high levels of nitrogen accumulate and move downward in the soil.

Variability in manure N content

As mentioned previously, the variability of N content in manure is less pronounced than for P content. For example, across all liquid pig manure operations, total N concentrations varied from 4 to 6.7 kg/1000L while P varied from 0.0 to 5.5 kg/1000L (Table 2.4). Whether

manure is liquid, solid, fresh or composted also affects this variability. Total N content of solid finishing pig manure or composted solid manure is higher than for liquid pig manure due to the lower water content (Tables 2.4 and 2.5). This variability in N content and form makes managing manure as a fertilizer challenging, particularly when trying to match manure nitrogen to phosphorus ratios to crop requirements.

2.3.4 Removal of Nitrogen

There are three main nitrogen removal streams from the agricultural system, excluding inadvertent runoff, leaching or volatilization losses:

1. **Crop Removal** – Exporting grains, oilseeds, hay, silage and straw removes N from the system. Crop rotations that maximize crop removal of N from the soil include deep rooted crops in rotations where soil nitrate has accumulated at depth. The greater proportion of the total plant that is exported, the greater the removal of N from the system. For example, a high N-demand crop like silage corn may be grown in fields with excess soil nitrogen, as a typical yield of silage corn can remove up to 170 kg N/ha.

Another important factor to consider with crop removal of manure nutrients is the increased crop yield when manure is applied. For example, in a study conducted at La Broquerie, Manitoba, forage crops (mainly grass) fertilized with pig manure removed four to five times more nitrogen than unmanured forage crops (Ominski et al. 2007). The authors report the nitrogen content of harvested hay represented approximately 33% of the N that was applied according to the crop's N requirements.

2. **Removal by Grazing Livestock** – Exporting cattle removes the percentage of the total N in the system that is associated with cattle biomass. As with the amount of P removed with cattle, the amount of N removed by cattle therefore depends on factors related to cattle weight gain such as i) the quality and use-ability of P in manured pasture grasses, ii) the growth stage of the cattle during grazing and iii) the overall health of the animals.

Fertilizing pastureland with manure improves forage productivity (Sager and Przednowek 2005) and quality relative to un-fertilized pastures (Ominski et al. 2007), which in turn should increase the sustainable grazing density to increase the proportion of applied manure nitrogen being exported with live cattle. For example, in a study conducted at La Broquerie, Manitoba, cattle grazed on manured pasture gained more weight and removed about three times the nitrogen on a per hectare per year basis as cattle grazed on an unfertilized pasture. In this project, Ominski et al. (2007) estimated that approximately 31% of the manure N applied was exported as body weight gain.

3. **Manure Export Off the Farm** - When there is insufficient land to apply an operation's total manure production, the manure can be exported from the farm, either in a raw or treated form. Although this type of export is rarely required in Manitoba for applying manure according to crop N requirements, the implementation of the new P-based manure management regulations may necessitate more export of manure offsite to avoid a build up of soil P (see Phosphorus Section).

2.3.5 Retention and Release of Nitrogen by Vegetation and Soil

2.3.5.1 N Retention and Release by Vegetation

Actively growing crops accumulate and retain N in vegetative matter and in the grain or seed. However, once plants are no longer growing, N can be released from dead and decomposing plants. Although N in runoff is predominantly released from the soil, vegetation may also release N during runoff events. Residues retained on the crop surface, as is common with reduced tillage management, have been shown to release small amounts of ammonium and nitrate nitrogen during rainfall events (Schreiber and McDowell 1985). This released N may be intercepted by the soil, depending on the volume of runoff water.

However, thawing of frozen vegetative residues can release large amounts of N (Miller et al. 1994), which may then be transported with snowmelt runoff to surface waters. Under simulated rainfall, release of N from frozen/thawed cover crops, measured as nitrate and ammonium N, represented between 5 and 10% of the total biomass N (Miller et al. 1994). Laboratory studies in Manitoba showed that the amount of dissolved N released from frozen and thawed canola residues collected from zero-till fields and conventional till fields was 3 to 7 times higher than from the top 1 cm of soil collected from the same fields (Flaten et al. 2005). In the same study, frozen and thawed riparian vegetation released approximately half as much dissolved N as canola residues from a zero-tilled field. Dissolved organic N was the predominant form of N extracted with water in these studies. Therefore, surface residues, perennial or fall-seeded crops may be additional sources of N in snowmelt runoff.

2.3.5.2 N Retention and Release by Soil

Agricultural soils require regular additions of N as synthetic fertilizer, animal manures or as green manures (N-fixing crops) to remain productive, as growing crops require N in amounts greater than most soils can naturally supply in order to produce optimal yields. Once applied to the soil, nitrogen in manure undergoes chemical and biological transformations that either increase or decrease N availability for plant uptake or environmental loss, or that may even remove N from the soil.

Forms of N in soil

Nitrogen exists in the soil as organic or inorganic N. Organic N is associated with microbial or plant biomass or soil organic matter in soil. Inorganic N, the plant available pool of N, comprises only a very small percentage of total N in soil at less than 5%. Ammonium, nitrite and nitrate are the inorganic forms of N in soil. Ammonium and nitrite are readily converted to nitrate, the predominant inorganic form in cultivated and/or annually cropped soils and the most mobile form of N. Nitrate is a negatively charged ion that is not readily retained in soil, is soluble and therefore is present in soil solution where it is highly mobile and moves with soil water. Ammonium-N, present in smaller amounts, is positively charged and can be retained at soil or soil organic matter surfaces where it can be held tightly (slowly available) or held in an exchangeable (available) form where it can easily be released for plant uptake, converted to nitrate, or potentially be lost with surface erosion runoff. Ammonium that is “held” is not readily susceptible to leaching or volatilization loss.

Transformations of N in Soil

Inorganic and organic forms of N in soil are subject to various chemical and biological processes that determine the availability of N for environmental loss or plant uptake.

Mineralization and Immobilization of N by Soil Organisms

Organic N must first undergo *mineralization* to become plant available, a process whereby microorganisms convert organic N to ammonium. Mineralization is a complex and variable process which occurs during manure storage and following application to soil. *Immobilization* is the reverse process where inorganic N is converted into organic N associated with live microbial tissues, by-products and decay products. This conversion is only temporary as N is released back to the soil once the microorganisms die.

Whether microbial activity results in net immobilization or net mineralization depends primarily on the ratio of carbon to nitrogen in manure and in soil. The C:N threshold for determining whether soil microorganisms will immobilize N or mineralize N is generally regarded as 20:1 (Killham 1994) but may vary from 15:1 to 25:1. Therefore, manures with high C:N ratios may be slow in releasing their N following application due to temporary immobilization by soil microorganisms. This temporary unavailability can reduce nutrient availability to the crop if immobilization coincides with the period of active crop uptake. Conversely, manures with a low C:N ratio, such as liquid pig manure, release their N quickly. Consequently, Schoenau et al. (2006) reported that liquid pig manure produced high crop yields at lower rates of application than for solid cattle manure.

Volatilization of Ammonia

Ammonium N in manure can be lost to the atmosphere as ammonia gas via a process called *volatilization*. Ammonia losses from liquid pig manure, where N is predominantly present as ammonium-N, occurs when manure is exposed to air during handling and storage and following surface application to agricultural land, particularly when weather conditions promote surface drying. Therefore, ammonia losses can be minimized by using covers on manure storage structures and by injecting liquid manure or incorporating directly after surface application to minimize manure exposure to the air. Ammonia loss reduces the efficiency of manure use as it requires higher rates of manure addition, and therefore higher P addition, in order to meet crop N requirements.

Conversion of Ammonium to Nitrate (Nitrification)

Once applied to soil, ammonium (NH_4^+) in pig manure is converted to nitrite (NO_2^-) and then to nitrate (NO_3^-) during a process called *nitrification* where nitrifying bacteria sequentially oxidize ammonium to nitrate:



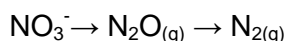
The conversion of ammonium to nitrate is relatively rapid in well aerated, alkaline soils such as those typical of agricultural land in Manitoba, but may be slower at low soil temperatures (Killham 1994). In Manitoba experiments with pig manure, Akinremi (2005) observed nitrate to generally increase near the soil surface with time, showing mineralization and nitrification to be occurring. Injecting or incorporating manure into the soil increases manure contact with the soil environment which thereby increases the rate of microbial nitrification, compared to applying manure to the surface where contact is restricted to the soil surface and significant quantities of ammonium N may be lost to the atmosphere prior to nitrification.

Nitrate-N is the dominant form utilized by plants as it is highly mobile and more readily abundant in soil than ammonium, the other plant available N form. As nitrate N is highly mobile, it is also susceptible to downward movement with excess water moving into the soil profile. Therefore, the rate of liquid pig manure applied and the timing of the application should match the N requirements of the actively growing crop to avoid excess amounts of

residual nitrate in soil, especially at times (e.g., early spring) or in places (e.g., depressional areas) where excess water may also occur.

Denitrification of Nitrates in Wet Soils

Under wet conditions, the oxygen content in soils may become low or absent (anaerobic conditions) and nitrate (NO_3^-) may be used by bacteria as a substitute for oxygen during respiration. This process is called *denitrification* and, if completed, results in conversion of nitrate to dinitrogen (N_2) gas, the gas that comprises almost 80% of the atmosphere.



During denitrification nitrate is converted to nitrous oxide (N_2O) a gas that can also be lost to the atmosphere. Nitrous oxide is a strong greenhouse gas with a global warming potential approximately 300 times that of CO_2 . Denitrification is enhanced when soils containing high amounts of carbon and nitrate, such as following manure addition, are subject to high soil water content where anaerobic areas can occur. Denitrification losses of fall applied manure can be great from poorly drained soils such as fine textured clay soils subject to low oxygen conditions. These losses can occur late in the fall and early in the spring when soils are more likely to be saturated with water as microbial activity proceeds even at low temperatures near freezing.

2.3.6 Factors Affecting Manure, Soil and Vegetative N Transport to Surface and Ground Water

Transport of manure N from a source to a surface water body requires a connecting landscape pathway and surplus water as a vehicle to mobilize and deliver the N. Transport of manure N to groundwater also requires surplus water, but in combination with an in-soil water flow pathway to carry nitrates to the groundwater or along a lateral subsurface pathway.

Forms of Nitrogen Transported

Nitrogen is transported to water in two forms:

Particulate N – water erosion transports organic or inorganic nitrogen that is associated with soil and/or soil organic matter. Erosional loss of sediment with runoff water is not a dominant loss pathway in Manitoba as only a small proportion of agricultural land in Manitoba has enough slope and precipitation to carry large amounts of soil particles to surface water. However, in some areas, snowmelt and rainfall runoff can induce erosion of soil and organic matter particles containing N whereby particles are separated from the bulk soil and delivered to surface water.

Dissolved N – snowmelt and rainfall runoff can also transport dissolved organic or inorganic N from vegetative residues, soil organic matter, livestock manure or synthetic fertilizers. However, due to its high solubility and ease of movement through the soil, nitrate is the predominant form of N transported. Nitrate may also move downward with excess infiltration of water or be transported laterally with subsurface water movement and may ultimately be discharged to surface water.

For manure's contribution to this loading, the proportions of the various forms of N present in manure (e.g., the proportion of N in water soluble vs. insoluble forms) as well as the physical characteristics of the manure influence the form and amount of N available for transport with

water to surface or groundwater. Soil and landscape characteristics as well as climate and weather also govern the transport and transformations of N. For example, in depressional areas of a field where the soils may be periodically saturated, nitrate accumulation may be limited due to intense denitrification losses of N or due to limited production of N from mineralization of organic N and nitrification of ammonium N. Conversely, manure applied to moist, well drained soils may result in higher soil nitrate concentrations due to higher rates of mineralization of organic N and nitrification of ammonium N.

Transport Processes and Pathways

There are three common transport processes and pathways for N movement with water to surface water or groundwater (Figure 2.3):

1. **Leaching of N** – Transport of nitrates with excess soil water to groundwater is the main pathway of concern for N loss to water. Once applied to the soil, ammonium and organic N in manure are converted to nitrate which is readily available for use by crops. If excess nitrate is present in the soil in the spring prior to crop uptake or in the fall after crop removal, this nitrate pool may be moved downward in the soil if there is net downward movement of water in the soil. Nitrates may also be transported with water moving laterally below the soil surface which can subsequently be transported to surface water. However, the risk of substantial quantities of N leaching in semi-arid climates such as Manitoba is less than in humid regions where excess moisture movement below the root zone is much more common (Heathwaite et al. 2000).
2. **Surface runoff and erosion of N** – Nitrogen from agricultural land can be transported with snowmelt or rainfall to surface water. Under Manitoba conditions, surface runoff of the dissolved fraction of N is more likely as water erosion runoff of N associated with soil erosion is not a common transport pathway for our agricultural landscapes. However, loss of N with surface runoff is generally a minor concern for freshwater quality.
3. **Direct Incidental Transfer of N** – Nitrogen in manure or fertilizer which is surface applied in or close to waterways can be washed directly into surface water when rainfall occurs shortly after application. Manure placed on top of frozen soils or snow in the winter can also be directly transported to surface water with spring snowmelt runoff. This manure N has not interacted with the soil and can be detrimental to fresh water quality, particularly with high concentrations of ammoniacal N, which is highly toxic to fish.

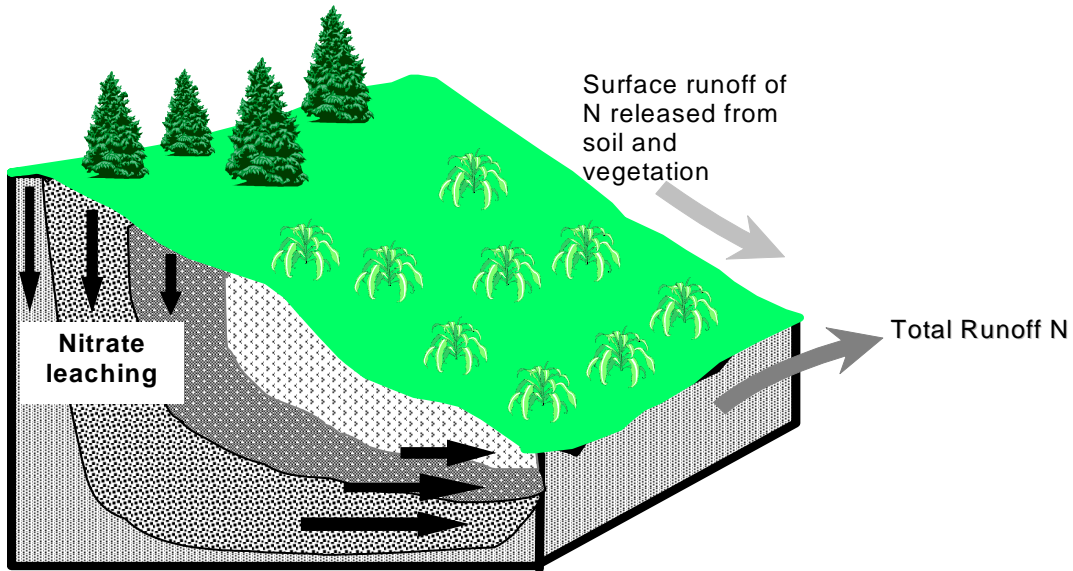


Figure 2.3. Nitrogen can be transported to groundwater and surface water along surface and subsurface pathways (adapted from Sharpley 2007).

Factors that Control Transport of Nitrogen

For leaching of soil nitrates, when there is nitrate present in the soil, the movement of water in and over the soil largely governs the extent of total N loss from agricultural land (Heathwaite et al. 2000). Movement of surplus soil nitrates downward in the soil profile occurs only when excess nitrates are present and the amount of water infiltrating exceeds the capacity of the soil to retain the water, causing net downward movement of water. The intensity of leaching is governed by climate and weather conditions (e.g. volume of water reaching the land per unit time) and soil (e.g. rate and extent of water inflow), landscape (e.g. rate, extent and direction of water movement into and within the soil) and land and water management as they influence the hydrology of the field. Transport of N with surface runoff is also governed by these four transport factors (Table 2.10). For a more detailed description of transport factors for nitrate leaching to groundwater and surface runoff of N, please see note 2.2 in the appendices.

Nitrate Leaching

Nitrate leaching is generally regarded as the most significant concern for N loss to water. For nitrate leaching to occur, there must be a combination of a i) pool of soil nitrate (source) and ii) net downward movement of water into the soil (transport) at the same time and place. Generally, the magnitude of N loss with leaching increases as the concentration of nitrate in the soil solution and the volume of water moving through the soil profile increase (Pierzynski et al. 2005b). Soil nitrates are transported with soil water through soil pores or along soil cracks or old earthworm and plant root channels. Once nitrates move below plant rooting depth, they cannot be recovered from the soil. This pool of nitrates leached to depth can be transported to underlying groundwater with future leaching events, or the water table may rise at times of high water infiltration, to intercept the nitrates. Nitrate contamination of groundwater is a greater risk from soils with shallow water tables versus soils with water tables located at depth due to the shorter distance to interception (Maule and Elliott 2006a).

The risk for nitrate loss from high nitrate soil is greater for coarse textured soils or cracking clay soils where a high volume of water, such as occurs during snowmelt or times of high rainfall, can move rapidly through the soil profile and where the groundwater is located near the soil surface. Nitrate-N can also be drained to surface waters upon entering tile drainage or natural subsurface lateral flow paths with subsequent discharge to surface water.

Table 2.10. Conditions where transport factors increase N movement to groundwater or surface water.

Factor	Conditions for Increased Nitrate Movement with Soil Water to Groundwater
climate and weather	<ul style="list-style-type: none"> - rainfall: high amount or duration - snowfall: high snow quantity - result in net downward movement of water within the soil
landscape	<ul style="list-style-type: none"> - shallow depth to groundwater or seasonally fluctuating water table depth - recharge areas of landscape - flow gradient to recharge areas - for subsurface flow: impermeable subsoil layer along which soil water moves to surface discharge site
soil	<ul style="list-style-type: none"> - increased inflow of water and preferential flow: high infiltration capacity - high connectivity of pore space to depth, lower capacity for water retention (e.g. leaching potential sand (coarse texture) >> clay (fine texture))
land and water management	<ul style="list-style-type: none"> - increased inflow of water and preferential flow: high infiltration capacity - artificial tile drainage, old root and/or earthworm channels (e.g. under minimum tillage, established forage or perennial grasses) - high nitrates: repeated manure or fertilizer applications beyond crop removal rate - reduced or no active water uptake: manure or fertilizer application to low productivity area, summerfallow or bare field
Factor	Conditions for Increased N Movement with Water to Surface Water
climate and weather	<ul style="list-style-type: none"> - rainfall: high amount, duration or intensity - snowmelt: high snow quantity, rapid snowmelt, frozen soil
landscape	<ul style="list-style-type: none"> - steep slope susceptible to erosion (particulate N) - water ponding areas: extended contact of water with surface soil (for dissolved N) with connectivity to surface water - connectivity of field water to surface water along a flow gradient - reduced capacity for water infiltration: groundwater discharge site
soil	<ul style="list-style-type: none"> - slow or restricted water infiltration: lateral flow of runoff water - fine texture, surface crusting, frozen soil, or thin layer of thawed soil over frozen soil - high water content or saturated soil: lateral flow of runoff water
land and water management	<ul style="list-style-type: none"> - slow water infiltration - lateral flow of runoff water: intensive tillage, soil compaction by equipment or livestock - high lateral flow of runoff water: artificial in-field surface drainage - high release of N: intensive tillage, annual crops, bare soil (summerfallow or intensive fall tillage) - incidental flow of manure: manure surface-applied into/next to a ditch or water way - high N at or near soil surface: repeated manure (or fertilizer) applications beyond crop removal rate, surface applications to fields under reduced tillage, perennial crops

Nitrate losses depend strongly on the hydrological conditions in the field. Soil water moving laterally that discharges at the surface is capable of transporting high amounts of nitrates from within the soil to surface water, as nitrates are picked up along the way. For example, in laboratory rainfall simulations, Zheng et al. (2004) recovered 11-16% of the total fertilizer N applied where the net water movement was out of the soil due to “artesian seepage” compared to only about 0.01% of total fertilizer N being lost with rainfall runoff over a free draining soil. Under natural conditions in Manitoba, lateral movement of water within the landscape has been observed (Whetter et al. 2007) but the total flux of nitrate movement is not large by comparison to more humid areas.

Nitrate from mineralized organic matter, synthetic N fertilizer and pig manure behaves similarly

The potential for leaching of manure N is no greater, and in many cases is less than similar rates of synthetic nitrogen fertilizer (Burton et al. 1994; Stumborg et al. 2007). For example, immediately following application of liquid pig manure and synthetic N fertilizer to sandy soils at Carberry, Manitoba, Akinremi (2005) measured higher nitrate content near soil surface from fertilizer than from pig manure. The reason for manure's slower rate of nitrate release is that manure N is predominantly in the form of ammonium and organic matter, which must be nitrified and mineralized, respectively, to generate nitrate. In the same study, trends in downward nitrate migration in soil by the end of a three year period were similar for both manured and synthetically fertilized soils, with similar nitrate levels below 60 cm, but above this depth, soil nitrates were lower in the manured plots than in the urea-fertilized plots. Nitrate leaching can also occur in the absence of manure or fertilizer addition from mineralization of background organic nitrogen (Akinremi 2005), as over 90% of soil N is organic N (Stevenson 1982).

Risk of nitrate leaching is minimal with proper management of manure or fertilizer

On cropland, nitrate accumulation in the soil is limited under proper manure or fertilizer management, and may even be *reduced* compared to other management strategies. For example, in long term experiments in Saskatchewan, nitrogen fertilization resulted in less nitrate leaching than with unfertilized crops because of improved use of water by the crop and less downward movement of water, below the root zone (Campbell et al. 1984). In other Saskatchewan experiments, Stumborg et al. (2007) found that nitrate did not accumulate in the soil when pig manure was applied at agronomic rates, even following multiple years of annual applications. The authors also observed that plots fertilized with urea contained higher soil nitrate than plots where pig manure was applied at similar or even greater rates. They attributed this difference to improved crop growth from manured plots due to the additional nutrients present in the manure, but not added with urea fertilizer, that may have been limiting to crop growth. On coarse textured soils where pig manure was applied based on N requirements to grazed or hayed forage consecutively for three years, nitrate did not accumulate in soil relative to an unfertilized pasture (Ominski et al. 2007). Higher productivity of forages on manured land likely increased water and nutrient uptake to limit the amount of residual soil nitrate as well limit downward movement of water.

In some cases where pig manure has been applied at a rate several times higher than annual requirements, nitrate loading or downward migration has been very limited (Schoenau et al. 2006); however, this research was conducted in central Saskatchewan, where the climate is drier than in parts of Manitoba. Therefore, in general, repeated manure applications at excessive rates or at rates that do not account for residual soil N results in nitrate loading and increases the risk of downward movement of nitrates with excess soil water (Stumborg et al. 2007; Schoenau et al. 2006). As such, manure applications to soils

with limitations to crop yield (e.g., drought stress) should be managed to account for any potential accumulation of nitrate in soil that may be subsequently lost by leaching (Stumborg et al. 2007). For example, during a wet year following two drier years, Akinremi (2005) found nitrate that had accumulated at depth over that period migrated downward below the root zone prior to crop establishing deeper roots.

Risk of nitrate leaching is greatest in fall and early spring

As nitrate N is highly soluble and mobile in water, the extent of water movement in soil largely determines the extent of nitrate leaching. Relatively little nitrate is leached on cropped land during the growing season in our climate because evapotranspiration losses limit water accumulation in the soil. The greatest potential for leaching is in the early spring and fall (Akinremi 2005) where snowmelt or seasonal precipitation, in the absence of crop uptake and removal of water and nitrates from the soil, results in net downward movement of water in soil. Akinremi (2005) detected the highest volume of leachate water and highest amount of nitrates collected in lysimeters following spring snowmelt and in the fall at times where high amounts of water moved through the soil profile.

In cold regions such as Manitoba, infiltration, and therefore nitrate leaching is limited during the winter when the soil is frozen. However, the potential for nitrate leaching is often the greatest immediately after the snow melts, when runoff water accumulates in depressional areas and the soil begins to thaw. As a result, water table levels often rise significantly during this period (Burton and Ryan 2000; Maule and Elliott 2006a).

Runoff and Erosion of N to Surface Water

Nitrogen can be transported off agricultural fields with rainfall and snowmelt runoff. However, transport of N to surface water along this pathway represents only a very small proportion of the total N loss to surface water and is not considered a dominant pathway for N transport (Kleinman et al. 2006). In comparison, the quantity of N deposited with *precipitation* generally exceeds the amount of N lost with runoff from agricultural lands (Nicholaichuk and Read 1978; Burwell et al. 1975). For example, as mentioned previously, annual N loading to Lake Winnipeg by atmospheric deposition of N (9500 tonnes/year) far exceeds the annual contribution from agriculture (5100 tonnes/year) in Manitoba (Table 2.9).

Dissolved inorganic and organic N can be lost with snowmelt and rainfall surface runoff, while sediment-bound organic N is the main form in erosion runoff (Pierzynski et al. 2005b). For example, the majority of N from inorganic fertilizer or pig manure measured in snowmelt and post-snowmelt runoff by Gangbazo et al. (1997) was as nitrate (43 kg N/ha), followed by total Kjeldahl N (8.0 kg N/ha) and ammonium (1.8 kg N/ha). In agreement, Kleinman et al. (2006) found nitrate accounted for 65% and 85% of the total N measured in rainfall runoff from two soils. In Manitoba using constructed plots designed for high rates of erosion (slope 9%), Hargrave and Shakyewich (1997) found the sediment fraction of N accounted for over 99% of the total N recovered from rainfall-induced erosion runoff.

The relative proportion of N forms in runoff is reflective of the relative amounts of the various forms of N in soil in combination with site hydrology. Runoff from a soil that has recently received manure has a greater proportion of N as ammonium and organic N and a smaller proportion as nitrate (Sharpley 1997), reflective of the amounts and forms of N in manure. In support of this, Gangbazo et al. (1995) observed that while ammonium and total Kjeldahl N load concentrations in snowmelt and drainage water collection increased linearly with increasing application rates of manure, nitrate loads in leachate were not influenced by manure N rate which they attributed to nitrate coming from soil rather than from the manure.

Although fall-applying manure is commonly practiced for a variety of practical reasons, a portion of fall-applied manure N is lost prior to crop uptake in the spring. A portion of fall applied manure N is volatilized, particularly if surface applied, and denitrified during the period from application to crop uptake. The reduced efficiency of fall manure applications is well known and to compensate, higher amounts are routinely added so that sufficient manure N is available for the spring crop. Also, fall applied manure can contribute to snowmelt runoff losses as observed by Maule and Elliott (2006a) in Saskatchewan where they measured increased ammonia, but not nitrate in snowmelt runoff waters, from manure injected the previous fall. However, in the second snowmelt season, higher nitrate concentrations in snowmelt runoff were measured relative to the control, two years after manure was applied.

Another problem with fall applied manure is that if it is applied late, near the time of soil freezing, injected manure may not be properly stabilized by the soil and therefore may be very susceptible to runoff loss in the spring (Maule and Elliott 2006a). These losses of N represent an environmental cost, and as manure is a fertilizer resource, this practice also represents an economic cost. Therefore, timing of manure application should ideally be close to the time of crop N demand to limit the amount of time manure nutrients are in the soil prior to crop uptake.

Direct, Incidental Transfer of N

Winter surface applications of manure are at high risk for direct transport of dissolved inorganic and organic N with snowmelt runoff to surface water, particularly in cold regions like Manitoba where snowmelt runoff constitutes the largest single runoff event. The potential for significant nutrient losses from manured fields in Manitoba was demonstrated by Schulte et al. in 1979 where they reported the total N measured in surface runoff from plots receiving winter manure applications was equivalent to 12.0% of the N applied in manure. In recent studies conducted in Saskatchewan, Maule and Elliott (2006b) observed high losses of ammonium N (5.9 kg N/ha) in snowmelt runoff from a field where manure was applied onto the snow during the winter. Nitrate concentrations were not affected as the majority of manure N would have been in ammoniacal form, with minimal nitrate content. Moving manure into the soil by incorporation or injection is highly effective for reducing ammonia losses of N and loss of N with runoff. For example, in Alberta, incorporation of cattle manure using various tillage methods reduced total N loads in surface runoff by 26-95% compared to surface application, conserving more N in the soil for use by the crop (Little et al. 2005). Therefore, incorporating or injecting manure and avoiding fall or winter application onto frozen or snow covered ground can greatly reduce incidental transfer of manure N.

2.4 PHOSPHORUS MANAGEMENT AND WATER QUALITY

As mentioned earlier, due to the low ratio of N:P in manure vs. crops, when manure is applied based on crop N needs, excess P will accumulate in soil and increase the risk of P loss to water. Therefore, the importance of managing manure on the basis of P, in addition to N, has recently been recognized in Manitoba, just as it has in many other areas of the world. As with N, the processes and management practices that determine the risk of P loss can be classified in two main groups: source factors and transport factors.

2.4.1 Factors Affecting the Risk of P Loss at the Source

Source factors can be divided into the subfactors of i) inputs, ii) removals and iii) soil and vegetation retention and release. In general, the risk for P loss to surface water increases when P inputs exceed removals and P builds up in the soil. Although the risk of P loss from various soils will vary with the soil's capacity to retain P, no soil has an infinite capacity to retain P. Therefore, P source inputs (e.g., pig manure) and exports (e.g., crop products) must be balanced over the long term.

2.4.2 Sources of P Inputs

Pig manure is one of many possible sources of P inputs to land. An overall understanding of these various sources and the role of manure management within this context is important in order to develop a realistic approach for reducing P losses from agricultural land.

o Atmospheric Phosphorus Inputs

Atmospheric P deposition can be both wet (with precipitation) and dry and can be a significant source of P directly entering lakes (Schindler et al. 1976). For example, in a study of twelve lakes in the Riding Mountain Park area, Beck (1985 *in* Flaten et al. 2003) estimated an atmospheric P deposition loading rate of 0.41 kg/ha/yr. As a result, atmospheric deposition of phosphorus directly contributes an estimated 500 tonnes of phosphorus per year to Lake Winnipeg, representing 14% of the total loading from Manitoba sources (Lake Winnipeg Stewardship Board 2006). This amount is equivalent to 71% of the contribution from Manitoba point sources and 42% of the estimated contribution from agriculture.

Furthermore, atmospheric loading with snow and onto snow-covered land also contributes significant quantities of P that may be indirectly responsible for some of the P loading into surface waters during snowmelt. For example, very fine particulate phosphorus in the atmosphere has been detected in snow deposited in Riding Mountain National Park (Thompson et al. 2000, *in* Flaten et al. 2003) where total P measured in snowpack averaged 1.28 mg/L. Maule and Elliott (2006b) reported a range in total P (0.077-0.72 mg/L) and orthophosphate P (0.016-0.402 mg/L) concentrations in snow over a six year period in Saskatchewan. However, the mobility and environmental impact of this "snow P" is not clear.

o Synthetic Phosphorus Fertilizer Inputs

In Manitoba, phosphorus is most commonly added to soil as synthetic fertilizer, predominantly as monoammonium phosphate (e.g., 12-51-0 or 11-52-0). For the 2006 crop year, monoammonium phosphate comprised 93% of the 191,445 metric tonnes of phosphate fertilizer sold in Manitoba (Canadian Fertilizer Institute 2007). Use of phosphorus fertilizer in Manitoba appears to be declining, as P fertilizer sales have dropped 26% since 2003 (Canadian Fertilizer Institute 2007). Phosphorus fertilizer, in its original form, is highly water soluble, but reacts quickly with soil and is commonly applied underneath the soil surface, in or near the seed row of annual crops at modest, agronomic rates that are usually close to the rates of crop removal. Therefore, synthetic fertilizer P is generally not as susceptible to environmental loss as surface or higher rate applications of other P sources.

o Manure Phosphorus Inputs

Use of manure as a fertilizer source for cropland in Manitoba is markedly lower than synthetic fertilizer use, as mentioned previously. Injecting manure allows for subsurface placement and greater placement control to increase plant availability but reduce the risk of

environmental loss relative to surface applications. Injection of liquid manure is increasing in popularity as the area of land receiving injected manure has increased more than twelve fold compared to 1995 census values. The amount of pig manure produced in Manitoba in 2006 contained approximately 5,000-7,000 tonnes of phosphorus, equivalent to only 11-15% of the average total P removed by crops and 12-17% of the total fertilizer P applied in Manitoba in 2006. Nevertheless, as stated earlier for manure N management, manure P should be applied at appropriate rates, even if the acreage of manured land is small. Otherwise, there is a risk of large losses of P from that small area.

2.4.3 Agronomic and Environmental Performance of Pig Manure as a P Fertilizer

Availability of pig manure versus synthetic P sources for plant uptake and environmental loss

Phosphorus in synthetic fertilizer is inorganic P that is highly soluble and almost entirely available for plant uptake or environmental loss when initially applied to soil. In contrast, the availability of manure P is highly variable, depending on the amount and forms of P in the manure. Research has shown the initial availability of P in pig manure can range from 10-50% (Tri-Provincial Manure Application and Use Guidelines 2003). Sharpley and Moyer (2000) found the available fraction of inorganic P in pig manure comprised 31% of total P while 23-33% of the total P in pig manure samples collected in Manitoba was in this form Ajiboye et al. (2004). In comparison, the initial availability of P from synthetic fertilizer P (e.g., monoammonium phosphate) is regarded as 100% even though this availability declines rapidly after the fertilizer P reacts with soil.

Effects of placement on manure P benefit

The agronomic availability of P in synthetic and manure fertilizers also depends on nutrient placement. The ideal placement maximizes nutrient availability near plant roots for early crop uptake yet minimizes nutrient availability at the soil surface for transport loss to the atmosphere or water. From an agronomic perspective, manure P is generally regarded as 50% less efficient than synthetic fertilizer P, primarily due to the impracticality of banding manure P in the seedrow, which is a standard, highly efficient practice for synthetic P fertilizers. From an environmental standpoint, the risk of P loss to surface water increases as: injection or banding < surface application + incorporation < surface applications without incorporation.

Effect of application timing on manure P benefit

The most efficient timing of synthetic or manure fertilizer applications coincide with periods of active crop removal of nutrients and the lowest risk of nutrient loss. Therefore, from a nitrogen efficiency standpoint, fall application is less efficient than spring application, for example. However, the relative agronomic efficiency of phosphorus utilization from fall vs. spring or summer applications of manure is not known. What is known is that the majority of runoff and P loss in the Prairies occurs during snowmelt, rather than rainfall runoff events (Sheppard et al. 2006; Green and Turner 2002; Glozier et al. 2006) as snowmelt typically accounts for over 80% of the total runoff (Nicholaichuk 1967; Glozier et al. 2006). Therefore, the risk of P loss from surface applications of manure in winter or late fall, without incorporation may increase the environmental losses of late fall or winter-applied manure relative to manure applications in the spring.

Effect of application rate on manure P benefit

Generally, increasing the rate of manure or fertilizer P application increases the agronomic and environmental availability of P in soil (Kashem et al. 2004ab), indicating the soil has a

limited capacity for P retention that diminishes as the retention capacity becomes saturated. As a result, high rates of P application on soils with large existing supplies of P are much more likely to result in significant losses of P, compared to low rates of P application on soils that are low in P.

Variability of P (and N) in pig manure

As mentioned previously, there is a high degree of variability in the phosphorus content of pig manure depending on animal growth stage, feed inputs, and manure storage and handling (Tables 2.4, 2.5 and 2.6). The range in variability of phosphorus content can even be high for the same type of operation. For example, the range in phosphorus content of liquid pig manure for finisher operations is 0 to 3.52 ppm. This “natural” range in phosphorus content is further widened by recent developments in pig feeding practices to reduce P excretion in waste. For example, looking at Table 2.4, use of phytase in feed formulations has lowered phosphorus content of manure from finisher operations by an average of 0.35 ppm compared with finisher operations that do not use phytase, although the variability within phytase-fed finisher operations was still high, ranging from 0 to 2.83 ppm.

Imbalance of N:P in pig manure versus crop requirements - P build up with N based applications

Livestock manure's inherent imbalance of N and P relative to crop requirements combined with the greater range of variability in P than N content, makes managing manure applications to meet crop needs very challenging for all livestock producers, including pig producers. Therefore, when pig manure is applied based on crop N needs, there is an imbalance which can result in excess supply of P. For example, in La Broquerie, Manitoba, researchers observed an average increase in soil phosphorus from 11 to 59 ppm in the first 5 cm of soil after 3 years of applying liquid pig manure on an N basis to grazed and hayed fields on coarse textured soil (Ominski et al. 2007).

In addition to P content variability, the ratio of N:P in pig manure also varies based on type of operation (e.g. finisher operation) and operation practices (e.g. use of phytase in feedstuffs) as well as manure handling and storage, much like the P content of manure varies. For example, since P is associated with the solid fraction of "liquid" pig manure, even when manure in a storage unit is vigorously mixed, P content is highest at the bottom of the storage unit (Fitzgerald and Racz 2001). Therefore, P content varies during storage pumpout to a greater degree than ammonium content which is associated with the liquid fraction of manure and is more uniformly distributed in the storage unit (Dick 2003; Fitzgerald and Racz 2001). This high degree of variability and nutrient imbalance make managing manure as a fertilizer very challenging, particularly when trying to match manure nitrogen to phosphorus ratios to crop requirements.

It should be noted however, that some of the liquid manure from some pig operations can have very high N:P ratios. During long term studies in Saskatchewan, N:P ratios of pig manure collected from earthen storages ranged from about 10 to 40:1 (Schoenau et al. 2005), compared to less than 3:1 for representative Manitoba pig manure samples (Table 2.4). The proportion of total N as ammonium N was similar to that measured in the Manitoba samples, typically between 70 and 80%. Some of this apparent variation in N:P ratio is likely due to the degree of agitation in the manure storages prior to sampling or the completeness of the pumpout, since the ammonium N concentrations are fairly uniform in a storage, but P-rich solids settle to the bottom (Fitzgerald and Racz). Regardless of the reasons for such

wide N:P ratios in these studies, applications of liquid pig manure with a high N:P ratio results in limited P accumulation in the soil (Qian et al. 2004 and Schoenau et al. 2005).

Although little research has been conducted with solid or composted pig manure, solid manures or composts have a high proportion of N in stable, organic forms. Therefore, solid manures must be applied at higher rates than manures with a high proportion of N in readily available forms such as in liquid pig manure to meet crop N demands (Schoenau et al. 2006). The stable nature of N in solid manure also causes the ratio of available N to total P to be very low in solid manure (Tables 2.3 and 2.4). As a result, application of solid manure at a rate to fully supply a crop's N needs usually results in a substantial over-application of P (Qian et al. 2004), even worse than for liquid pig manure.

2.4.4 Removal of Phosphorus

There are three main phosphorus removal streams from the agricultural system, excluding inadvertent runoff or leaching losses:

1. **Crop Removal** – Exporting grains, oilseeds, hay, silage and straw removes P from the system. Rotations can be managed to maximize crop removal of P from soil, as certain crops remove greater amounts of P and have greater yields than others. For example, the harvested portion of an alfalfa crop removes approximately 34 kg P/ha based on a forage yield of 11.2 tonnes/ha whereas a canola crop removes only about 18 kg P/ha at a grain yield of 2 tonnes/ha. Although the same canola crop requires a total of 26 kg P/ha, a portion of P is used for roots and un-harvested above-ground plant material, which remains in the field after harvest.

Crop removal of P, however, may be increased dramatically as a result of manure application. For example, in a study conducted at La Broquerie, Manitoba, forage crops (mainly grass) fertilized with pig manure removed over 5 times the amount of phosphorus than an unmanured forage crop (Ominski et al. 2007). The authors report the phosphorus content of the harvested hay represented approximately 22% of the P that was added when the manure was applied to meet the crop's N requirements.

2. **Removal by Grazing Livestock** - Exporting cattle grazed on manured land removes a percentage of total P as P in bone and tissue matter (Shewmaker 1996). As with the amount of N removed with cattle, the amount of P removed by cattle therefore depends on factors related to cattle weight gain such as i) the quality and use-ability of P in manured pasture grasses, ii) the growth stage of the cattle during grazing and iv) the overall health of the animals. In many cases, given the improvement in forage productivity and quality that results from application of manure, rates of P removal during grazing are higher on manured than on non-manured forages. For example, in a study conducted at La Broquerie, Manitoba, on a per hectare basis, cattle grazing on manured pasture gained three times the weight and removed about three times the phosphorus as cattle grazed on an unfertilized pasture (Ominski et al. 2007). However, the amount of P removed in grazing systems is still much less than the P applied at rates that are sufficient to supply the forage crop's need for N. In the La Broquerie project, the authors estimated that only 5% of the manure P applied to the pastures was exported as body weight gain (see note 2.3 in the appendix for calculation of P removal in cattle).

3. **Manure Export Off the Farm** – When there is insufficient land to apply an operation's total manure production, the manure can be exported from the farm, either in a raw or treated form. Manure can be exported for disposal or as a commodity as i) raw

product (high volume, low P content, high transportation cost), ii) compost product (lower volume, weight and N:P ratio), and iii) high P solid product from solid-liquid separation (low weight, high P content, lower transportation cost).

2.4.5 Retention and Release of Phosphorus by Vegetation and Soil

2.4.5.1 P Retention and Release by Vegetation

Soil is not the only possible source of P released during runoff events. When plants are actively growing, they retain their tissue P quite well. However, repeated freezing and thawing of plant material breaks up plant cells and releases P from plant tissues which can be subsequently transported with snowmelt runoff to surface waters. Surface residues or perennial cover may therefore also contribute significant amounts of dissolved P to snowmelt runoff, particularly with reduced tillage or perennial cropping. Under simulated rainfall, Bechmann et al. (2005) measured approximately ten times more total P loss in runoff from catch-cropped soils after freeze/thaw cycles than from bare manured or bare unmanured soils. Similarly, laboratory studies in Manitoba showed that the amount of soluble P released from frozen and thawed canola and oat residues collected from zero-till fields and conventional till fields was much higher than from the top 1 cm of soil collected from the same fields. Riparian vegetation also released a significant amount of soluble P (Flaten et al. 2005).

However, *soil* may still play a very important role as an interceptor for *vegetative P* during snowmelt, in contrast to the traditional view that *vegetation* has a role in intercepting *soil P*. Maintaining low soil test P concentrations may be an important means of ensuring that the soil can readily retain additional P that may be lost from the vegetation.

2.4.5.2 P Retention and Release by Soil

The P status of agricultural soils in Manitoba is generally low to medium, meaning that most soils are likely to respond to P addition; only a few areas have high soil P concentrations because of historical land management (Johnston and Roberts 2001; Johnston 2006). Therefore, P is commonly added to agricultural soils to meet crop P requirements. However, phosphorus can accumulate in soil with repeated fertilizer or manure P additions in excess of crop removal rates to levels that are susceptible to runoff loss or even to leaching, which may occur under long-term manure use at N-based rates. Certain agricultural activities, such as high input vegetable production, intensive livestock operations with high manure production on a small land base (e.g. poultry, dairy or pig operations, cattle feedlots), or allowing grazing cattle to congregate for long periods in certain areas (e.g., near a permanent water source), for example, can result in localized areas of high soil P that may be subsequently mobilized and transported to water bodies.

Forms of P in Soil

Phosphorus exists in the soil as organic or inorganic P and is either attached to soil particles (bound) or in soil solution. Bound P, can be associated with soil surfaces (adsorbed) or with other compounds in the soil to form new P compounds such as Ca and Mg phosphates (precipitates). This fraction may be susceptible to transport loss as erosion of particulate P. Dissolved P is present as soluble complexes and as free phosphate ions in soil solution. This fraction may be susceptible to transport loss as runoff or leached dissolved P. The relative amounts of inorganic and organic P in a manured soil will depend on the proportions contained in the added manure (Heathwaite 1997).

Transformations of P in Soil

Phosphorus in soil is subject to numerous continuous chemical and biological processes that transform organic and inorganic P and affect P retention and release (Figure 2.4). There are three paired transformation processes: immobilization and mineralization, precipitation and dissolution, adsorption and desorption. Understanding the effects of these processes on the availability of P for transport out of the soil system is an important step in determining the risk of P loss and the appropriate management practices required to minimize that risk.

Immobilization and Mineralization of Phosphorus in Soil

Immobilization and mineralization are the microbiological transformations between inorganic P and organic P. Immobilization is the process where inorganic P is converted by microorganisms into organic P (e.g., as microbial tissue P) and mineralization is the reverse process (e.g., where soil organic matter P is converted to soluble inorganic P). Both processes occur continuously and simultaneously in soil (see note 2.4 in the appendices for more detail on immobilization and mineralization processes in soil).

Precipitation/Dissolution and Adsorption/Desorption Reactions of Phosphorus in Soil

Phosphorus precipitation involves the formation of new solid compounds from soluble phosphorus and soil constituents; dissolution is the reverse process; both processes are highly pH dependent. At low pH or acid soils, P predominantly forms precipitates with two major cations (positively charged ions), iron (Fe) and aluminum (Al). However, in neutral to calcareous soils, such as the prevalent soils in Manitoba, P precipitates primarily with calcium (Ca) and magnesium (Mg) to form Ca and Mg phosphates (Havlin et al. 1999). Thus, soils with abundant supplies of available calcium, magnesium, aluminum or iron (e.g., clay soils) have a higher capacity to precipitate P than soils with low supplies of these elements.

Phosphorus adsorption involves the attachment of P to an *existing* surface (which is unlike absorption where something enters into another substance, for example P uptake into a plant root). Phosphate anions (negatively charged ions), such as H_2PO_4^- and HPO_4^{2-} (orthophosphate) form bonds of varying strength with the surface to which they are attached (Pierzynski et al. 2005a). Adsorption reactions occur on a variety of soil surfaces, including clay minerals, carbonates or with aluminum and iron oxides/hydrous oxides surface coatings or particles (Morgan 1997; Pierzynski et al. 2005a). Therefore adsorption of phosphorus increases as reactive surface area increases, such as in clay soils.

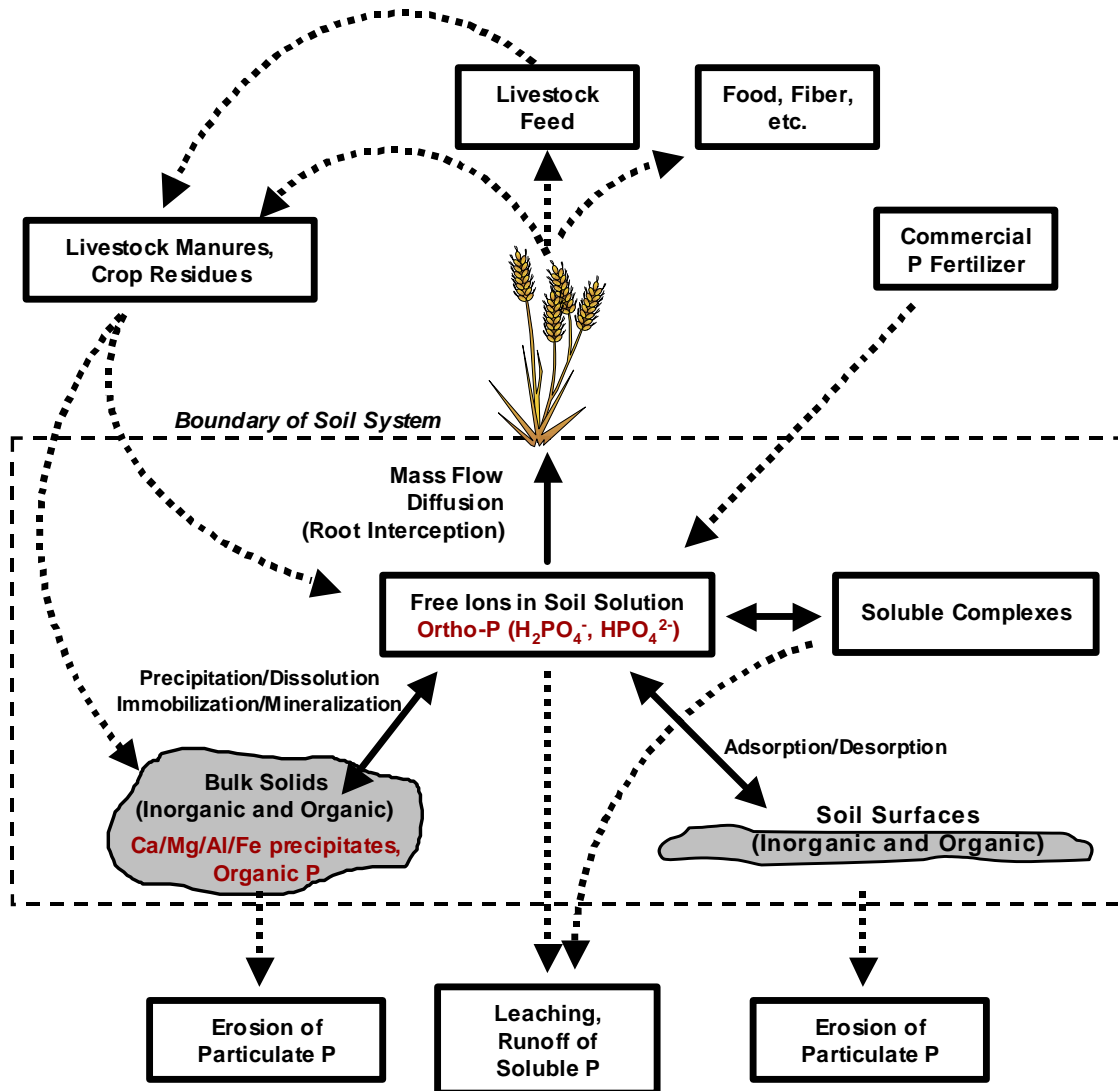


Figure 2.4. Transformations and losses of phosphorus in soil (Adapted from Flaten 2003).

To soil scientists, the process of phosphorus precipitation and adsorption in soil represent two distinctly separate processes for retaining phosphorus. However, from a practical standpoint, both of these processes occur together when manure is applied to soil. The retention reactions of precipitation and adsorption reactions remove phosphorus from soil solution (decreasing dissolved P but increasing particulate P) while the release reactions, dissolution and desorption, release bound P to the soil solution (decreasing particulate P but increasing dissolved P). For both sets of processes, the proportion of applied P that is retained by the soil decreases as the amount of P in soil increases, increasing the risk of P loss in high P soils (Sims 2004).

2.4.5.3 Soil Properties and Phosphorus Retention

There is a great range in capacity for Manitoba soils to retain P (Akinremi 2007a, Ige et al. 2005, Kumaragamage et al. 2007); this capacity is related to several key soil characteristics

(see note 2.5 in the appendices for more information on P retention in soil). The most important soil properties governing P retention in Manitoba soils appear to be the abundance of Ca and Mg cations (Ige et al. 2005b and 2007). Soil texture, organic C, soil pH, cation exchange capacity and carbonates were also shown to be related to the P retention capacity, but primarily because they influence the abundance and availability of Ca rather than for being important for P retention in-and-of themselves (Ige et al. 2007). Ige et al. (2007) also concluded that Al is important for P retention for the neutral to calcareous Manitoba soils they investigated, although the role of Al was much less important than the role of Ca and Mg.

Soil Texture, Organic Matter, Cation Exchange Capacity and Cation Content

Soils with high clay and organic matter content have a high surface area and a high amount of negative charges on those surfaces. Therefore, these soils have a high capacity to hold cations (positively charged ions) such as Fe, Al, Ca and Mg on their surfaces (i.e. these soils have a high *cation exchange capacity*) and a greater capacity to retain P via both precipitation and adsorption reactions with these cations. These cations can bond with P to remove it from soil solution. Generally, for a neutral to alkaline soil, P retention increases with Ca and Mg content while for an acidic soil, P retention increases with Al and Fe content.

Soil Carbonates

Many of Manitoba's soils have developed from material that is rich in calcium and magnesium carbonates. This characteristic has been used on some occasions to regard Manitoba soils as having extremely high capacity to retain P. However, 65% of Manitoba soils do not have significant concentrations of carbonates at the surface and the role of carbonates in retaining phosphorus is not clear. For example, some researchers have claimed that carbonates play an important role in P retention in soil (James et al. 1996; Torbert et al. 2002), while others have shown that carbonates are relatively unimportant (Weir and Soper 1963; Soper and El Bagouri 1964; Leclerc et al. 2001; Mnkeni and MacKenzie 1985; Akinremi et al. 2004; Ige et al. 2005). For additional information on the role of soil carbonates on P retention in soils, see note 2.6 in the appendices.

Part of the reason for this contradictory evidence is that the presence of carbonates in soil is also associated with other soil characteristics that may be fundamentally more important indicators of P retention. For example, carbonates are found only in high pH soils, whereas the solubility of P precipitates is highest at neutral or slightly acid pHs. In addition, soils with high pHs and high concentrations of calcium carbonate are also likely to have high concentrations of exchangeable Ca that can react more readily than carbonates with added P. As a result, Ige et al. (2007) found carbonate content in Manitoba soils was related to P retention due to its relation to Ca content, which was a much better indicator of P retention capacity.

Flooding

Areas of agricultural land in Manitoba, for example the fine textured, nearly level soils of the Red River Valley and the depression areas of the Western Manitoba parklands, are subject to periodic flooding, particularly in the spring during snowmelt and during periods of high rainfall. Flooding reduces oxygen availability which can increase release of retained P to soil solution (Pierzynski et al. 2005a). Soils with high amounts of Fe oxides have great potential to hold P, but also may release large amounts of P to solution during periods of flooding (Racz 2006). In addition, flooding traps carbon dioxide in soil, increasing the solubility of calcium phosphates and releasing P from soils with high amounts of Ca (Lindsay

1979). Therefore, prolonged periods of flooding can release P into soil solution especially in soils with a substantial accumulation of manure P (Ajmone-Marsan et al 2006).

Solubility of manure phosphorus relative to other sources

While transformations of fertilizer P in soil have been well studied (Hedley and McLaughlin 2005), reactions of manure P in soil are more complex and less understood. Mineralization and immobilization occur during manure storage and following application to the soil, so the relative proportion of organic and inorganic P is continually changing, the extent of which depends on environment conditions and initial organic P composition of manure. Recent research with Manitoba soils demonstrated that the availability of manure or fertilizer P added to soil depends on both the P form and soil type to which it is applied (Kashem et al. 2004b). Comparing synthetic fertilizer to manure sources of P following soil application, fertilizer P is highly water soluble and therefore initially more available for plant uptake or loss compared to manure (Shigaki et al. 2006). However, the availability of fertilizer P in soil rapidly decreases, more rapidly than for manure P, depending on the manure (Kashem et al. 2004a). Therefore, although manure P is not as soluble as fertilizer P, it may remain available for a longer period, with the degree of availability varying substantially with manure and soil type (Kashem et al. 2004ab). For information on the stability of pig manure P compared to cattle manure P, see note 2.7 in the appendix.

Impact of manure additions on soil capacity to retain and release phosphorus

To complicate matters further, addition of manure to soil can alter P retention mechanisms by changing the soil solution chemistry and physical environment. Long term manure addition to soil can increase soil organic matter, pH and abundance of cations such as Ca, that can change the chemical and physical properties of the soil and thereby alter the P retention capacity of the soil (Whalen and Chang 2002; Simard et al. 2001; Sharpley et al. 2004). Increases in soil organic matter content may interfere with surface retention sites or enhance movement of P as suspended organic P (Leytem and Westermann 2003; Whalen and Chang 2002; Laboski and Lamb 2003). Cations added with manure can react directly with phosphorus or may displace Ca, Mg, Al and Fe from the soil exchange to subsequently react with P to form precipitates. For example, increases in calcium content may alter soil chemistry such that Ca-P interactions control P availability more than Al- and Fe-P (Sharpley et al. 2004).

2.4.5 Factors Affecting Manure, Soil and Vegetative P Transport to Surface and Ground Water

Transport of manure P to surface water requires a source of available P, connectivity of the field with a waterway, a suitable landscape pathway and water to mobilize and deliver soil P to surface water. The greatest potential of P loss from manure and inorganic fertilizers occurs where agricultural lands drain into nearby waterways (natural or artificial) or are located over a shallow water table that connects with surface water. However, if soil P at the source is not elevated, there is limited risk of P transfer from the soil to water.

Forms of Phosphorus Transported

Phosphorus is transported to surface waters in two forms:

1. **Particulate Phosphorus** – water erosion transports phosphorus that is chemically bound with soil particles and/or organic matter and is operationally defined as not being able to pass through a 0.45 µm filter (Haygarth and Sharpley 2000). Although particulate P is the dominant form of P loss from cultivated land in the U.S., this form of P loss accounts for only a small proportion of P losses from cultivated land in Manitoba, even along the escarpment

where the risk of soil erosion is relatively high compared to other areas of the Province (Glozier et al. 2006).

Particulate P is P that is i) adsorbed to existing soil surfaces or other suitable surfaces within the soil such as Fe and Al oxides, calcium carbonate, organic matter, etc, ii) precipitated or bound with soil or organic matter to form new particles, or iii) organic P that has not been mineralized or chemically transformed. Water erosion generated by rainfall or snowmelt detaches particulate P from the bulk soil and transports the particles to surface water where the P can be slowly released to become “bio”available to algae and other aquatic life (Sims 2004).

2. **Dissolved Phosphorus** – runoff can also carry soluble phosphorus that has been released to soil solution during mineralization and desorption and dissolution reactions. This type of P is operationally defined as being able to pass through a 0.45 µm filter (Haygarth and Sharpley 2000) and accounts for the majority of P loss from land in Manitoba (Glozier et al. 2006). For example, over a 12 year study near Miami, MB, 77% and 84% of P loss from a conventional tilled and a zero tilled field, respectively was in the form of dissolved P (J. Elliott, personal communication, National Water Research Institute, Saskatoon, SK). Similarly, high proportions of dissolved P have been reported in Alberta (Little et al. 2007)

Soluble dissolved P can exist as i) *inorganic P* from P fertilizer and manure, mineralized organic P from manure or soil organic matter, and ii) *organic P* from manure and soil organic matter; both forms are largely bioavailable upon water entry. Very small particles of P (< 0.45 µm) are also included in the dissolved P fraction. For example, the dissolved fraction of P transported with snowmelt runoff may also contain a fraction of very small particulate P derived from mixing with soil surface for a long period, or from atmospheric P in snowfall.

The proportions of the various forms of P present in the manure as well as physical characteristics of the manure can affect the risk of P loss during rainfall or snowmelt (Miller et al. 2006; Sharpley and Moyer 2000). However, climate and landscape play a very large role in determining the balance between dissolved and particulate P in runoff. For example, the movement of snowmelt water over the landscape is gradual and the water is in contact with the soil for prolonged periods, which facilitates release of dissolved P from the soil. Conversely, the energy associated with an intense rainfall event on a sloped landscape will facilitate erosion of particulate P, particularly if the soil is fine-textured (e.g. clay or clay loam), which requires a small amount of energy for detachment and transport (Haygarth and Sharpley 2000; Sims 2004). The interaction between soil P and runoff water is greatest at the surface and decreases rapidly with depth (Ahuja et al. 1981). Therefore, placing manure below the soil surface by injection or incorporation greatly reduces the amount of manure at the soil surface in contact with runoff water. As a result, the potential for P loss with runoff water from subsurface placed manure is much less than from surface-applied manure.

Transport Processes and Pathways

There are four common transport processes and pathways for P movement with water to surface waters (Figure 2.5).

1. **Erosion of Particulate P** - Erosion of soil by water is an important pathway for steeply sloped land in humid climates (e.g., much of the U.S. and Europe) but is a less important loss pathway in Manitoba where the majority of agricultural land is nearly level and the climate is relatively cold and dry. Transport distances for this form of P are relatively

short and losses of this form of P can be reduced with erosion control processes. For more detailed information on erosion of particulate P, see note 2.8 in the appendices.

2. **Runoff of Dissolved P** - Most of the P lost from agricultural land to surface water in the Prairies is dissolved P. This form of P can be transported for long distances and is very difficult to intercept once it enters a water pathway.

3. **Direct Incidental Transfer of P** - Phosphorus in manure or fertilizer which is deposited in or close to waterways directly by animals or mechanical broadcasting can be washed into surface water when rainfall occurs shortly after application, or if manure is placed on top of frozen soils in the winter and is transported with spring snowmelt runoff. This can be particulate and dissolved P, but does not contain P that has interacted with soil.

4. **Migration of Leached P into Groundwater and then to Surface Water** - Groundwater contaminated with leached P (dissolved P or suspended particulate P) can connect with surface waters or resurface at discharge areas. Transport of groundwater containing leached P is a secondary pathway for surface contamination, but can be significant under certain management such as long term manure addition in excess of crop removal and/or artificial drainage, especially in coarse-textured (sandy) soils. Leaching of P to groundwater and subsequent transport to surface water poses a long-term risk to P contamination of surface water, as the high reactivity of P in soil usually restricts P mobility in the short term. However, little is known of this P loss pathway in Manitoba. For more detailed information on P leaching to groundwater, see note 2.9 in the appendices.

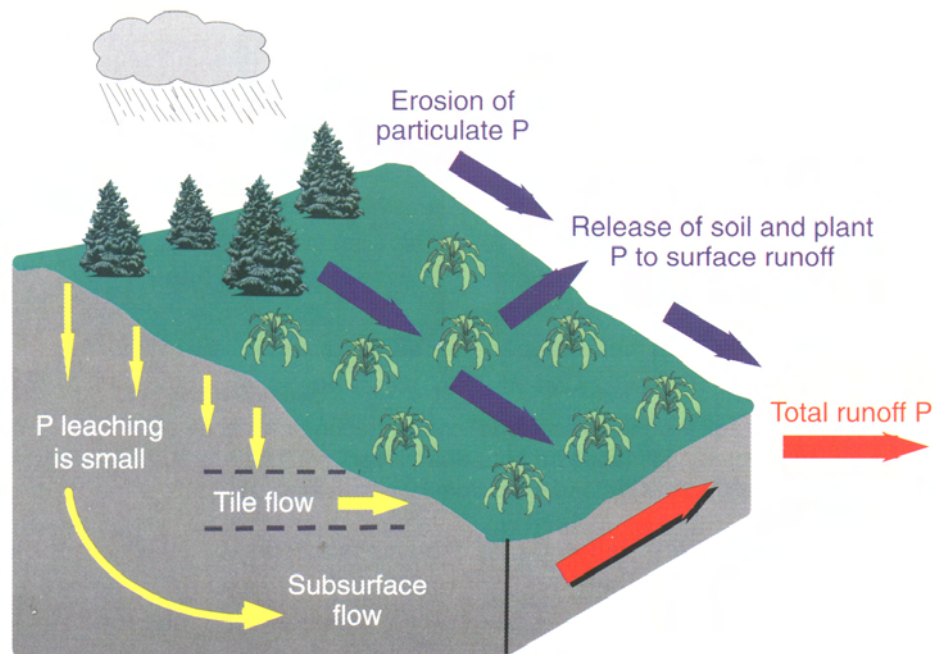


Figure 2.5. Phosphorus can be transported to surface water along surface or subsurface pathways. (Adapted from Sharpley et al. 2003a).

Factors that Control Transport of Phosphorus

Transport factors are those which control P separation from the bulk soil (mobilization) and facilitate movement of mobilized P to surface water (delivery). Transport factors can be separated into four sub factors of climate and weather, landscape, soil, and land and water management factors that contribute to sustained contact of water with soil, and transport with water to surface water via overland or subsurface flow (Flaten 2003; Sims 2004) (Table 2.11). The dominant pathway of P loss can vary spatially by region, farm or field scale depending on soil, landscape and climate factors, as transport factors can be highly interrelated, and can even vary with time, based on weather factors (Sharpley et al. 2001a). Most of these factors have a common influence on the risk of P loss, whether the source is pig manure or any other agricultural or natural source of nutrients. For more detailed information regarding the role of P transport factors on surface runoff losses, please refer to note 2.10 in the appendices.

Table 2.11. Conditions where transport factors increase P movement to water.

Factor	Conditions for Increased P Loss from Soil to Water
climate and weather	<ul style="list-style-type: none">- rainfall: high amount, duration or intensity- snowmelt: high snow quantity, rapid snowmelt, frozen soil
landscape	<ul style="list-style-type: none">- steep slope susceptible to erosion (particulate P)- water ponding areas: extended contact of water with surface soil (for dissolved P) with connectivity to surface water- shallow depth to ground water table, or seasonally fluctuating water table depth- short distance, connectivity of field water to surface water along a flow gradient
soil	<ul style="list-style-type: none">- slow or restricted water infiltration: lateral flow of runoff water - fine texture, surface crusting, frozen soil, or thin layer of thawed soil over frozen soil- high water content or saturated soil: lateral flow of runoff water- high preferential flow (P leaching): high soil cracking (e.g. dry clay soil)- high matrix flow (P leaching): high water content of coarse textured soil
land and water management	<ul style="list-style-type: none">- slow water infiltration - lateral flow of runoff water: intensive tillage, soil compaction by equipment or livestock- high preferential flow (P leaching): artificial tile drainage, old root and/or earthworm channels (e.g. under reduced tillage, established forages or perennial grasses)- high lateral flow of runoff water: artificial in-field surface drainage- high release of particulate P: intensive tillage, annual crops, bare soil (summerfallow or intensive fall tillage)- incidental flow of manure: manure surface-applied into/next to a ditch or water way- high P at or near soil surface: repeated manure or fertilizer applications beyond crop removal rate, surface applications to fields under reduced tillage, perennial crops

Unique Challenges Associated with P Loss during Snowmelt in the Prairies

In most of North America and Europe, where the climate is relatively warm and humid, rainfall is the main source of runoff. However, in the Prairies, 80-90% of runoff typically occurs during spring snowmelt (Nicholaichuk 1967; Glozier et al. 2006). As a result, the vast majority of runoff and P loss in this region also occurs during snowmelt, rather than rainfall runoff events (Sheppard et al. 2006; Green and Turner 2002; Glozier et al. 2006) (Figure 2.6). Other Canadian studies have also reported that the greatest amount of P loss to surface water occurs with snowmelt runoff (Goulet et al. 2006; Jamieson et al. 2003; Gangbazo et al. 1997).

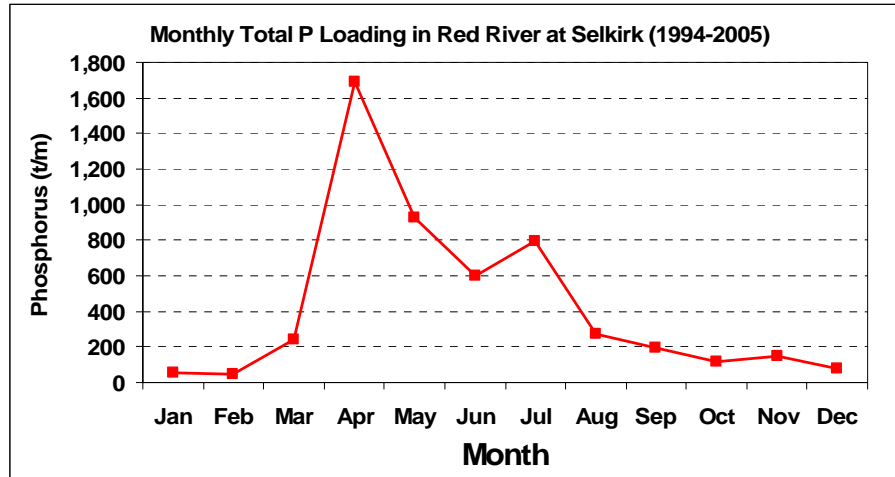


Figure 2.6. Monthly total phosphorus loading in the Red River at Selkirk (1994-2005) (Lake Winnipeg Stewardship Board 2006)

Most snowmelt P in the Prairies is in the dissolved form

In Manitoba, the nature of our cold, dry climate combined with our relatively level landscapes makes dissolved P the predominant form of P lost from agricultural fields during snowmelt. In addition, snowmelt runoff extends over a longer time period than rainfall runoff and the long duration of the soil-water contact encourages solubilization reactions at this interface. Soluble P is also released during freezing and thawing of dead plant tissues (Bechmann et al. 2005; Flaten et al. 2005). As a result of these factors, 79% of the runoff flow and 83% of P loss in the South Tobacco Creek watershed of south-central Manitoba occurred during snowmelt (Glozier et al. 2006) with the vast majority of the P lost in dissolved forms, especially at the edges of fields. Alberta researchers have also observed that snowmelt runoff is the dominant source of dissolved P (Feng et al. 2004). However, this may not be the case in Eastern Canada, where Jamieson et al. (2003) found the majority of P loss from snowmelt runoff in Quebec was associated with sediment loss.

P loss during snowmelt is more complex than during rainfall

The water flow pattern for snowmelt runoff is much slower and more complex than for rainfall runoff (Figure 2.7). During snowmelt, conditions at the soil surface determine the degree of P loss since most of the soil is frozen and water infiltration is restricted to the soil surface or a narrow thawed layer. Therefore, in addition to the complex hydrological character of snowmelt runoff, losses of P will vary with:

- Sources of available P at the soil surface:
 - P released from plant residues and perennial crops

- P originally present in snow
- P released from soil (background, from manure, or from fertilizer P)
- Depth of thawed layer of soil
- Duration and degree of contact between snowmelt and soil surface/thawed layer

These pools of P can enter into melt water along the drainage pathway for transport to surface water. It is largely land management factors that determine the soil surface conditions and the amount of soil and vegetative P available for interaction with snowmelt. Once the dissolved P begins to move with snowmelt water, it is very difficult to intercept due to most of the soil being frozen and a lack of plant growth and uptake. Therefore, reducing the quantity of soil P located at or near the soil surface via input and crop management is the most effective means for reducing the risk of dissolved P entering surface waters with snowmelt runoff.

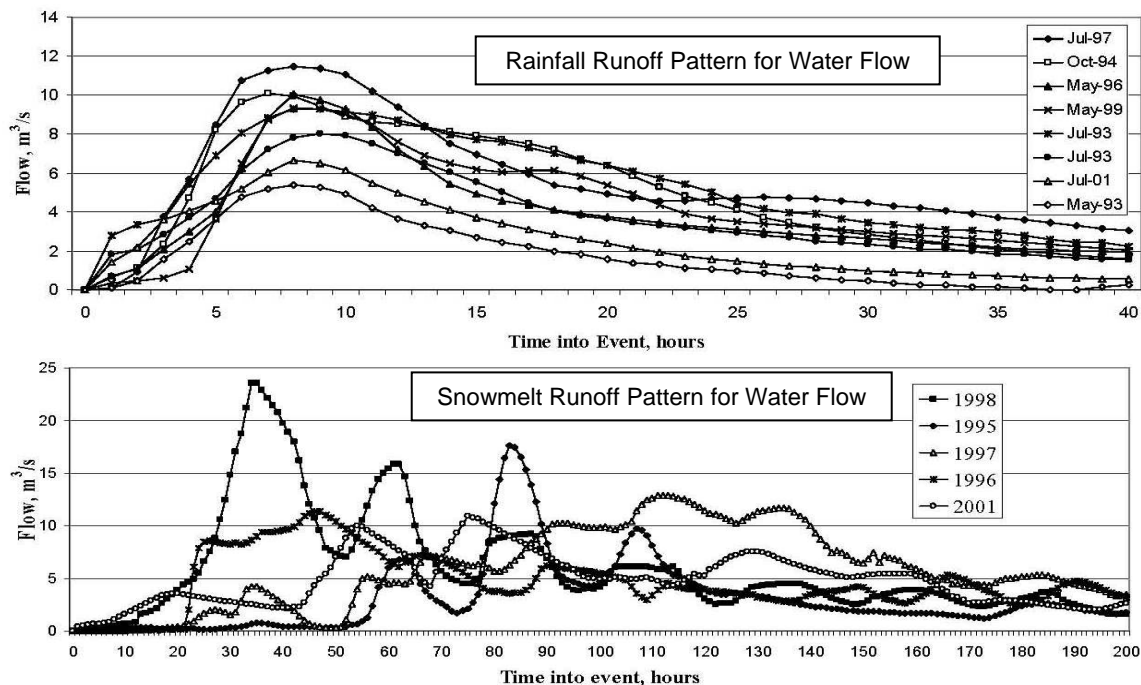


Figure 2.7. The water flow pattern for snowmelt runoff is much slower and more complex than for rainfall runoff (Glozier et al. 2006).

Winter applications of manure or fertilizer are at high risk of loss

The predominance of snowmelt runoff in Manitoba also creates a substantial risk of direct, incidental loss of manure P that is applied in winter. The potential for significant nutrient losses from manured fields with snowmelt runoff has been long recognized in this province (Schulte et al. 1979). Direct application of manure onto frozen or snow covered soil results in high runoff losses of particulate P (Srinivasan et al. 2006) and dissolved reactive P losses in runoff water with snowmelt (Gentry et al. 2007). In a Manitoba study, Green (1996) measured higher P from a field where manure was surface applied to frozen soil relative to fields that did not receive manure. American scientists have found similar problems with winter applications of manure. For example, Klausner et al. (1976) found that solid dairy manure application during thawing periods resulted in significant phosphorus movement to surface water bodies. Young and Mutchler (1976) found under conditions in Minnesota that up to 16% of orthophosphate was lost during spring runoff when manure was applied to

frozen soil. In the same study, losses of less than 4% of phosphorus were observed when the manures were incorporated into the soil in the fall following application. Incorporating or injecting manure markedly reduces all fractions of P in runoff relative to surface applications (Daverede et al. 2004; Tarkalson and Mikkelsen 2004). The authors found runoff P load and concentration from soils where manure was incorporated were not significantly different than from unamended soils. Therefore, incidental transfer of manure P can be greatly reduced by incorporating or injecting manure and by avoiding fall or winter application onto frozen or snow covered ground.

Late fall applied manure may also be at risk if not injected

For a variety of reasons, including accessibility to land without an actively growing crop, a large proportion of livestock manure is applied in the fall. Although this timing allows for injection and incorporation of the manure, it may increase the concentration of P at the soil surface and elevates the risk of nutrient loss during the subsequent spring snowmelt period, unless the manure is completely buried. For example, in the South Tobacco Creek region of Manitoba, Green and Turner (1999) found elevated concentrations of total phosphorus in spring runoff water from a field to which manure had been applied and incorporated during the previous fall relative to fields not receiving manure. In Minnesota, Gessel et al. (2004) found losses of dissolved P in runoff during the spring thaw period increased with increasing rates of liquid pig manure fall-applied and incorporated.

2.5 BENEFICIAL MANAGEMENT PRACTICES FOR REDUCING THE RISK OF N AND P LOSS TO SURFACE WATER OR GROUND WATER

Minimizing phosphorus and nitrogen loss from soil to surface or ground water is an essential component of any environmentally sustainable agricultural system, including pig farms. Nutrient management Beneficial Management Practices (BMPs) represent a suite of management practices specifically directed towards reaching this objective: reducing the risk of detrimental N and P entry to surface or ground water with consideration for economic feasibility of implementation. However, to be effective, the BMPs employed by farmers and encouraged by government incentives or regulations must be adapted to deal with the processes that control N and P loss in the local region. Therefore, nutrient management BMPs can be sorted into the two main groups of factors responsible for N and P loss, source factors and transport factors, as follows:

1. **Source Oriented** - management practices that impact agricultural N and P inputs, removals and net accumulation or loading in soil (e.g., reducing the amount of N and P being added to the soil and therefore the amount susceptible to loss), and
2. **Transport Oriented** - management practices that impact detachment, mobilization and delivery of N and P from the agricultural source to surface or ground water (e.g., restricting the transport of N and P and therefore the amount reaching surface or ground water and/or by recapturing N and P before entry to surface water).

To be effective and efficient, the suite of BMPs selected should address managing loss from identified **critical source-areas** which are the areas most susceptible to N and P loss; areas that have a high risk from both **source factors** and **transport factors** (e.g., available soil N or P content and direct connectivity to a susceptible water body along surface or subsurface pathways).

BMPs can be likened to tools in a box; a farmer can select the combination of “tools” (BMPs) most suited to the needs and capabilities of his/her operation based on prevailing economic, source and transport factors relevant to the unique set of conditions for that given operation. Conversely, there is no single BMP (e.g., anaerobic digestion or manure treatment) that will eliminate all the nutrient risks in every situation.

Nutrient management experts in Manitoba, such as the Manitoba Phosphorus Expert Committee (2006) have compiled an inventory of BMPs for reducing the risk of manure nutrient loss to surface water. However, most of those BMPs have not been fully evaluated for effectiveness under Manitoba conditions. Therefore, the following recommendations for BMPs must be regarded as preliminary in nature.

2.5.1 Source-Oriented Beneficial Management Practices

Managing for N and P at the source is critical as the risk of environmental loss is mainly based on *availability* of N and P for transport, especially in the Prairies, where soil erosion is not a major direct cause of nutrient loss to surface water. Producers prefer N-based manure applications because N is the most limiting nutrient to crop growth and this practice allows for higher rates of manure application than P-based rates. However, since the N:P ratio of livestock manures, including pig manure, is lower than the N:P ratio for crop nutrient removal, application of manure on an N-basis inevitably leads to a build up of soil test P and, in Manitoba at least, the need to comply with P-based regulations.

Source-oriented beneficial management practices should reduce the risk of N and P loss to surface and ground water by managing the:

- i) amount of N and P in manure (nutrient inputs)
- ii) amount of manure N and P applied to soil (nutrient inputs) and
- iii) the amount of N and P being removed from the soil by crops (nutrient removals) to prevent oversupply of manure N and P to the soil (nutrient balance).

The goal is to manage for balance of inputs and removals over the long term, with the flexibility to allow short term accumulation of soil N and P to concentrations which do not pose an unacceptable risk to water quality. The difference between inputs and removals is an indicator of the trend in N and P balance management, which manifests as accumulation of N and P in soil and/or loss to the environment.

The concept of balancing nutrient inputs with removals is encouraged by Provincial regulations. Effective November 2006, new and expanding pig operations in Manitoba are required to submit a manure management plan demonstrating the capacity of the operation to sustainably comply with regulations for soil test N and P, either by having access to suitable land base or by treating and/or exporting manure to reduce nutrient accumulation on the farm. To aid in maintaining a sustainable N and P budget, existing pig operations should maintain records of yearly manure operations and cropping history.

In addition, existing livestock operations of 300 or more animal units (for definitions of animal units for various classes of pigs, see Note 2.11 in the appendices) are required to submit an annual manure management plan to Manitoba Conservation. However, all operations (regardless of size) are required to comply with existing soil test N limits; existing operations will be required to conform to the new soil test P thresholds over a period ranging from 2008

to 2020. Pig producers can use a computer software program such as MARC 2005, the Manure Application Rate Calculator for Manitoba and Saskatchewan (2005) to help with the task of formulating an N-based manure management plan. This computer program contains a farm and a land application component, accommodates liquid or solid manure, and has elements of crop type, soil and manure nutrient concentrations, animal units, manure storage system, manure and crop management history, and manure application rate and timing. However, the program has not yet been updated to fully reflect the issue of managing for P balance.

BMPs for reducing the risk of pig manure P loss at the source can be grouped into two broad categories, one for each end of the animal: diet management practices and manure management practices (Table 2.12).

Table 2.12. Source-oriented BMPs for reducing loading of pig manure N and P to agricultural soils.

Source-Oriented BMP Objective	Recommended Action
Manage feeding strategy to minimize N and P content in manure	<ul style="list-style-type: none"> - decrease feed wastage - formulate diets to increase N and P use-efficiency in pigs
Manage manure as a nutrient resource to balance N and P additions with removals at the farm and field scales	<ul style="list-style-type: none"> - store or treat (chemically or physically) manure to optimize N:P ratio for crop use and removal - analyze manure and apply according to soil test recommendations; - optimize crop N and P utilization and yield via manure handling, timing, method and placement - select land or access additional land to optimize manure N and P additions and removals - manage crop rotations, maximize amount of P in exports of harvested crop material and grazing livestock

Adapted from Manitoba Phosphorus Expert Committee (2006).

2.5.1.1 Manage feeding strategy to reduce N and P content in manure

Minimize feed wastage

Feed wastage contributes a substantial portion of N and P in manure. Practices such as the use of pelleted rather than mash feed or use of wet/dry feeders can reduce wastage by up to 60% for pigs in the 20-80 kg weight range. This avoids unnecessary excess N and P entering the waste stream.

2.5.1.2 Formulate diets to improve utilization of feed N and P by pigs

Nitrogen and phosphorus inclusion in pig diets is commonly based on nutritional requirements developed by the National Research Council (1998). The amount of N and P in manure depends on the amount of N and P in feed, the amounts ingested and the N and P use-efficiency in the pig. An estimated 50-80% of P and 45-60% of N in pig diets is excreted as urine and feces (NRC 1998).

There are several strategies to decrease the amount of N and P excreted from pigs (Table 2.13), including:

- Match N and P supply with N and P requirements based on pig growth stage.
 - Employ feeding strategies such as (multi) phase feeding or split sex feeding as feed requirements differ at various growth stages and for different sexes. These practices can reduce N and P in pig manure by 5-10%.
- Increase the utilizable proportion of N and P in feed ingredients and reduce the addition of non-necessary ingredients to reduce N and P in pig manure and/or the total quantity of pig manure produced.
- Feed processing: inclusion of processed feeds which are more digestible such as finely ground peas to reduce total fecal excretion from pigs (Nyachoti et al. 2002).
- Diet formulation:

To reduce N in pig manure:

- Reduce addition of non-necessary ingredients - feed low protein diets supplemented with essential amino acids, as research has shown that reducing crude protein by 1 percentage unit can reduce N excretion by 8-10% (Nyachoti et al. 2003). A 2 percentage point reduction plus 0.15% supplemental lysine (amino acid) can reduce N excretion by 20-30% (Cromwell 1996) (for further information on reducing N in pig manure, see Note 2.12 in the appendices).
- Improve the use-ability of N in feedstuffs - improving N use-ability by 1% can reduce N excretion per kg of pork by 1.4% (Nyachoti et al. 2003). For example, research has shown N excretion from pigs fed a diet including finely ground peas versus raw peas was reduced on average 21% (Nyachoti et al. 2002).

To reduce P in pig manure:

P in manure can be reduced by improved feed formulations comprised of P ingredients and/or additives that increase the bioavailable portion of P (see Note 2.13 in the appendices for a comparison of the bioavailability of various feedstuffs). To improve use-ability in feedstuffs, increase the inclusion of:

- phytase enzyme – added to feedstuffs to increase the digestibility of phytate P, an extremely stable form of P in feed grains (for more information regarding phytate and phytase, please see note 2.14 in the appendices). Farrow, nursery and finisher operations in Manitoba that include phytase have lower total P than corresponding operations that do not utilize phytase (Table 2.4).
- higher phytase-containing feedstuffs – examples include seed coats of wheat, barley and triticale grains; hulls of soybean seeds and plants with bioengineered phytase.
- lower phytate P feedstuffs – examples include low phytate corn, barley and soybean varieties. Feeding low phytate barley can reduce manure P concentrations by approximately one third (Leytem et al. 2004b).
- higher bioavailable feedstuffs P due to pretreatment by fermentation or other processing – examples include distillers grain and high-moisture corn. However, heating during processing destroys natural and added phytase activity of feed ingredients (Orschak 2005; Walz and Pallauf 2002).
- Eliminate the practice of insurance feeding or adding P to pig diets beyond National Research Council (NRC 1998) requirements. Exceeding NRC

requirements for P in pig feed by as little as 0.2 percentage points results in a 70% greater P excretion than feeding at requirements (Cromwell 2005).

Table 2.13. Diet management to reduce N and P in pig manure.

Strategies	Reduction in manure N content (%)	Reduction in manure P content (%)
Phase feeding	5-10%	5-10%
Split sex feeding	5-8%	NA
Precision diet formulation	10-15%	10-15%
Lower crude protein plus amino acid supplementation	20-40%	-
Selecting digestible feeds	5%	5%
Lower inorganic P supplementation plus phytase use	-	20-30%
Supplementing selected enzymes	2-5%	5%
Combination low N + essential limiting amino acids plus low P + microbial phytase	30%	35-44%

Sources: Oryschak (2005); Walz and Pallauf (2002).

Special extension or educational programs may be required to improve the adoption of these practices, especially by small pig operations. For example, according to the 2001 Statistics Canada agriculture census, large pig operations are more likely than smaller operations to utilize feeding strategies that will improve manure management.

Potential long-term strategies to improve use-efficiency of feed P by pigs:

- Re-evaluate National Research Council P requirements for pigs to account for the relative bioavailability of P in barley-based diets commonly used in Manitoba (Flaten et al. 2003). Recent research in Manitoba shows supplying P at concentrations slightly below National Research Council requirements (0.1 percentage unit reduction of non-phytate P) does not negatively impact animal quality (Akinremi et al. 2007b).
- Reduce the minimum P requirements in commercial feed formulations where phytase is added as a supplement. The Canadian Food Inspection Agency regulations currently require commercial feeds for pigs to contain more P than is required when phytase enzymes are added to diets to aid utilization of phytate P from feed grains.
- Adapt Canadian regulations limiting use of high available P (HAP) feedstuffs (e.g. HAP barley) (Nyachoti 2007). A new variety of feed barley has been developed by the University of Saskatchewan through conventional plant breeding techniques to have a higher proportion of P in a non-phytate form than traditional barley varieties. This new variety has the same P content, overall, as traditional varieties, but its higher availability of P would reduce or eliminate the need for supplementing the diet with mineral P. However, this variety is currently regarded by the Canadian Food Inspection Agency as a potentially harmful "novel feed" which will severely restrict its commercial viability.

- Genetically select or develop pigs for higher utilization of P. Conventional genetic selection may have limited potential in this area. However, future genetic selection may include use of transgenic pigs that have high concentrations of phytase in their saliva to improve digestibility of phytic acid. Research at Guelph, Ontario showed up to a 75% reduction in P excretion from transgenic pigs capable of producing salivary phytase (Golovan et al. 2001).

2.5.1.3 Manage manure as a nutrient resource to balance N and P additions with crop removal

The rate, timing, method and placement of pig manure applications can be managed to improve the efficiency of manure applications to more closely meet crop N and P requirements and thereby limit accumulation of N and P in soil at concentrations susceptible to loss.

Handle, store and treat manure to increase nutrient content or modify the N:P ratio

- Reduce the amount of manure generated. Reduce in-barn water use to lower the volume of manure produced and to concentrate nutrients; this will not decrease available N:P ratio, but lowers transport and application costs.
- Minimize N loss from manure during storage and handling. Use covers on storage areas and reduce manure exposure to air during handling to reduce loss of N as ammonia. This increases the available N:P ratio of manure.
- The relative amounts of N and P varies with type of storage system and at different depths within a storage system (Table 2.6). Nitrogen to P ratio is lowest at the bottom of the storage tank where P content is highest due to bottom settling of solids. Storage tanks with thorough mixing homogenize manure for more uniform nutrient content during pumpout (Dick 2003).
- Compost solid manure to:
 - reduce manure mass and increase nutrient concentrations. When there is insufficient land to apply manure, composting of solid manure can be an alternative management as it reduces manure weight and volume so that the manure is more economical to transport to land that may benefit from organic matter and stable nutrient inputs (Larney et al. 2006).
 - stabilize N and P. Composting converts the ammonium-N fraction to organic N. Although some N is lost during the composting process, the remaining N is stabilized and more resistant than fresh manure to volatilization, leaching and denitrification losses during and after land application (Qu et al. 1999; Miller et al. 2006). However, the increased stability of the N also lowers the ratio of available N to total P; therefore, application of compost to meet crop N requirements results in a larger surplus of P than for manure that is not composted.
 - The compost pile must be carefully located and managed to reduce the risk of nitrate leaching to groundwater.
- Physical separation of solid and liquid manure phases to increase the N:P ratio of the liquid and increase the P concentration of the solids. Most of the P content in liquid

pig manure is in the solid fraction. Therefore, after separation, the liquid phase has a higher N:P ratio than in raw manure and is well suited to application on fields with elevated soil test P, low P retention capacity or low crop removal of P. The solid phase has a high concentration of P, allowing manure P to be economically transported to areas further away from the manure storage, for example onto fields with low soil test P, high P retention capacity or high crop removal of P. The solid phase may even be dried to reduce export costs further.

- Chemical treatment to increase N:P ratio of liquid manure effluent. For example, aluminum, iron or manganese can be added to precipitate P, zeolites to adsorb P or polymers to flocculate P. The net effect of these chemical treatments is to settle the P as solids and increase the N:P ratio of the liquid portion of the manure to more closely match crop removal rates.
- Biological treatment to increase N:P ratio of manure effluent. Aerobic or anaerobic decomposition reactions can be used for biological immobilization of P and generation of bioenergy. The effectiveness of these treatments has not been thoroughly evaluated in Manitoba and may be limited due to the long cold winters and short summers.

Physical, chemical or biological methods of reducing P are costly and, particularly for small operators, would be less attractive and less feasible than applying untreated manure as a fertilizer (Salvano et al. 2006; for more information on the economic impact of manure treatment on the viability of pig farms, please refer to note 2.15 in the appendices). However, these BMPs may become necessary for operations that run out of suitable nearby land for manure application or other less costly alternatives.

Analyze manure and apply according to soil test recommendations

Liquid pig manure is a valuable source of N and P; however, the concentration and availability of N and P in the manure is variable from one source to another (Table 2.4). Testing before and during the pumpout with a portable ammonia meter and hydrometer provides a reasonable estimate of the nutrient content in liquid manure, so that the rates of application can be adjusted accordingly. In addition, manure samples should be collected during the pumpout for analysis by a professional laboratory; keeping ongoing records of manure's nutrient content in individual storages helps pig producers to develop a reliable estimate of the typical nutrient content for future management decisions.

The availability of N and P from solid pig manure is not well understood. Solid manures from straw bedding systems, for example, have a very high ratio of carbon to nitrogen in the manure. This high carbon content will slow down the microbial decomposition of the manure and the subsequent release of nutrients, compared to liquid pig manure. Such delays in availability of nutrients decrease the risk of excessive nutrient accumulation in the short term, but may increase the risk of excessive accumulation in the long term due to our lack of control over the rate of nutrient release from these organic sources.

Pig manure can be applied to meet crop N requirements for low soil test P fields, but should be based on crop removal of P for higher soil test P fields. Oversupplying manure beyond agronomic rates of N and P will result in build up of N and P in soil, particularly with repeated applications, increasing the risk of nutrient loss by leaching or runoff (Stumborg et al. 2007; King et al. 1990). Applying pig manure at high N rates also decreases crop uptake efficiency

(Mooleki et al. 2002; Burns et al. 1990), which results in inefficient use of valuable, and otherwise expensive, crop nutrients.

- Determine manure application rate based on annual soil P (0-15 cm) and N (0-60 cm) tests, manure N and P analyses prior to application, and realistic target yield(s) based on land productivity or agricultural capability (see Note 2.16 in the appendices). Due to the high variability of manure N and P content, particularly for P, estimates based on “typical” nutrient content or regional databases may over or underestimate N and/or P availability.
- Manage manure for uniformity of N and P and minimal ammonia volatilization before and during pumpout and during application via agitation and/or chopping up of larger manure chunks in an enclosed system. Vigorous agitation is also important to minimize residual manure solids of high P content remaining in storage tank which would reduce N:P ratio of subsequent manure entering storage tank (Dick 2003).
- Maintain adequate N fertility to ensure water is used efficiently. Adding N to soil will reduce nitrate leaching by improving the crop’s ability to utilize water efficiently (Campbell et al. 1984).
- Manure rates based on crop P removal that result in undersupply of N should be supplemented with N fertilizers to meet crop N needs. The suitability of this practice will also depend on the economic cost of application versus value of increased yield. Apply other nutrients (e.g., sulphur) if recommended by soil test and not supplied by manure, to ensure proper crop nutrition to optimize yield and therefore N and P removal.
- Applying liquid manure in amounts that exceed or overwhelm the soil’s infiltration capacity can lead to manure remaining on the soil surface for a longer time where it is susceptible to runoff and volatilization loss.

Apply manure at the appropriate time to minimize losses:

Application of manure or synthetic fertilizer should ideally be timed to supply N and P at the time of crop growth to optimize the efficiency of the application and minimize the risk of environmental loss of nutrients. However, the 2001 Statistics Canada Agriculture Census showed 66% of farms in Manitoba with livestock applied manure in the fall which represents 54% of the manure produced on these farms (Beaulieu 2004). Applications are also common during the spring (21%) and summer (22%). Only 3% of the manure being produced in Manitoba was reported to have been winter-applied in (11% of livestock farms).

Generally, the risk of manure N and P loss is greatest when high manure N and P concentrations at or near the soil surface coincide with overland water movement, as can occur with rainfall shortly after manure application or when manure is applied onto frozen or snow covered fields in the late fall and winter (Sharpley et al. 2001a; Maule and Elliott 2006b). In order to minimize losses of N and P, the following practices for timing of manure application are recommended:

- Avoid manure applications at times when there is a high probability for high amounts of rainfall. Rainfall runoff losses of P (dissolved and bioavailable) and N (total and

ammonium-N) are reduced with increased time between application and rainfall (Sharpley 1997).

- Avoid applying manure late in the fall or in the winter when the ground is frozen to limit the risk of manure nutrient loss with snowmelt, when there is an elevated risk for loss based on the local physical and climatic conditions and no crop for active water and P uptake.
- Minimize applying manure in the fall to improve the efficiency of manure-N availability and reduce the total amount of manure required to meet crop N requirements. Nitrogen in fall-applied manure is subject to denitrification or leaching losses prior to crop growth in the spring, requiring higher rates of manure to compensate for this reduced availability. However, this practice may be the only practical option for operations that need to empty manure storages for accommodating manure produced over the winter months or where early spring applications may result in compaction and soil structure problems on annual cropland (e.g., heavy clay soils).

Use application methods that maximize nutrient utilization and minimize losses:

Volatilization losses of liquid manure can be high when the manure is applied onto the surface, without tillage to incorporate the manure into the soils (Gordon et al. 2001), particularly when warm, windy weather promotes soil drying (Rochette et al. 2001) (see Note 2.17 in the appendices for volatilization losses under various management practices). Therefore, when manure is being applied to meet crop N requirements, surface application rates have to be increased to account for this loss of N. Surface-applied manure is also susceptible to P and N losses with rainfall or snowmelt runoff and N loss with ammonia volatilization. Therefore, wherever possible, subsurface placement of manure by injection or incorporation should be used to minimize N and P loss (Sharpley et al. 1999; Kleinman et al. 2002; Little et al. 2005) and increase crop recovery of nutrients (Schoenau et al. 2005; Mooleki et al. 2002). Since subsurface placement of manure minimizes the amount of manure-N lost as ammonia (Bittman et al. 2005), this increases the amount of available N which in turn reduces the total amount of manure that needs to be applied to meet yield targets and therefore decreases the amount of P being applied.

Additional options such as aerating the soil, in combination with applying liquid manure reduces runoff losses of total N, total P, soluble dissolved P and ammonium-N (van Vliet et al. 2006). Also, using a drag hose system to supply a manure applicator reduces the risk of runoff losses compared to a tank delivery system because of reduced compaction of the soil (Tri-Provincial Manure Application and Use Guidelines 2003).

Select land to optimize nutrient removal or export off the farm, if necessary

- If the farm's land base is sufficient, prioritize fields to apply highest rates of manure on most productive land, based on N requirements, if possible.
- Rotate lands if manure application is based on N. Repeated annual applications of manure based on N have a high potential to cause P build-up. Rotating fields for manure application using an N-based rate in one year and only commercial N fertilizer in subsequent years can slow or halt the long-term build-up of soil test P.

- Access additional neighbouring land to obtain a large enough land base to accommodate all of the manure produced, based on soil P concentrations. The increased land required for the transition from N-based to P-based manure application is substantial and much greater than expected according to simple differences between the N:P ratio in manure and the N:P ratio in crops. Although N-based rates are determined by the *available* N content in the manure and total crop requirements for target yields, P-based rates are determined by *total* P content in manure and crop removal for long term average yields (for more information about the reasons for these differences, please refer to note 2.18 in the appendices).
- If manure production exceeds the availability of suitable land for application, manure may need to be transported off the farm, e.g.:
 - transport raw manure to neighbouring farm lands or nearby rented land;
 - export treated/solid manure containing concentrated P which is more economical to transport larger distances than liquid manure;
 - export value-added manure by-product.

However, liquid and solid pig manure is a bulky product with a very low nutrient content, therefore, the costs of transporting untreated liquid or solid manure long distances are substantial while treatment costs to reduce manure P content or manure bulk are also costly (Salvano et al. 2006).

Select crops to maximize nutrient removal

- Removal of N and P from field and farm is maximized when both seed and residue of annual crops are harvested and exported. For forage or perennial crops, harvesting and exporting hay removes more P than grazing where P is retained in vegetative matter or re-deposited with cattle manure.
- Select crops suited to:
 - high P removal for higher P soil
 - high N removal for soils with elevated nitrates
 - removal of nitrates leached below rooting zone of annual crops by deep rooted perennial forages such as alfalfa.
- If high concentrations of soil test P become a concern, include crops in rotations with high rates of P removal to maximize P removal.
- Apply high P manure (e.g. solid phase from solid-liquid separation) to supply crop P requirements for a few years such as to forage crops at seeding to meet forage P demand over few years.
- Use manures with high concentrations of organic N (e.g., solid pig manure), as slow release fertilizers and use with crops that benefit from late season availability of N such as corn and perennial forage.
- Include cover crops or fall seeded crops as "catch" crops to take up excess inorganic reserves of N and P during the fall and minimize early spring losses during snowmelt

2.5.2 Transport-Oriented Beneficial Management Practices

Transport-oriented BMPs are designed to reduce the mobilization and delivery of N and P from the field, farm or watershed to surface water or ground water (Table 2.14). Transport factors work in concert with source factors and therefore source and transport BMPs should be used together. Transport BMPs are particularly important when repeated applications of high rates of manure result in high concentrations of soil N and P that may be highly susceptible to loss.

Table 2.14. Transport-oriented BMPs for reducing loading of pig manure N and P to agricultural soils.

Transport-Oriented BMP Objective	Recommended Action
Reduce erosion of particulate N and P	<ul style="list-style-type: none"> - establish actively growing crops early and include fall seeded crops - maintain permanent vegetative cover - retain surface residues - use cover crops or mulches - establish vegetated barrier strips or conservation buffers - use controlled surface or subsurface soil drainage - restore or construct wetlands - reduce tillage frequency - eliminate summerfallow or bare fields from rotations
Reduce runoff of dissolved N and P	<ul style="list-style-type: none"> - avoid accumulation of P in high risk areas such as: <ul style="list-style-type: none"> o soils prone to flooding o areas of fields subject to overland or channelized flow - perhaps wetlands, vegetative buffer strips and periodic tillage of soils with high surface P and low erosion risk, but these are not well documented for snowmelt runoff
Reduce direct, incidental losses of N & P	<ul style="list-style-type: none"> - inject or incorporate manure to reduce runoff exposure - avoid manure application on frozen soils or snow, or into ditches or waterways
Reduce leaching of N and P	<ul style="list-style-type: none"> - apply manure at conservative rates, especially in areas with sandy texture and preferential flow channels - avoid summerfallow - plant "catch" crops to utilize excess water and nutrients

Adapted from Manitoba Phosphorus Expert Committee (2006)

Beneficial management practices suitable for reducing manure N loss to surface or ground water may not be suitable for reducing P loss to surface water and vice versa as sources and dominant loss pathways for N and P can be impacted differently by management practices (Sharpley et al. 2001a; Heathwaite et al. 2000) (Table 2.15). In addition, BMPs suitable for reducing the risk posed by one form of nutrient may be unsuited for reducing the risk posed by another form of the same nutrient. For example reduced tillage or use of artificial drainage reduces total P and total N losses via surface runoff but increases dissolved P and nitrate leaching losses. However, as new P-based manure management regulations focus on restricting rates of manure P application, P-based BMPs, in general, may be more relevant than N-based BMPs because excess manure N application is less likely to occur.

Table 2.15. Typical impacts of select transport-oriented beneficial manure management practices on N and P loss from soil¹.

Transport BMPs	Impact on loss of P		Impact on loss of N	
	Particulate P	Dissolved P	Total N	Nitrate-N
Reduced or no-till	decrease	increase	decrease	increase
Cover crops and residues	decrease	increase	decrease	decrease
Vegetated buffer strips ²	mixed results in Manitoba, with slight decrease overall			
Soil drainage (tile or ditch)	decrease	increase	decrease	increase
Critical Source Area Treatment	decrease		decrease	

¹Adapted from Sharpley et al. (2006); most of these BMPs have not been validated in Manitoba.

² Sheppard et al. (2006)

In addition, transport-oriented BMPs that are generally effective at reducing or intercepting rainfall runoff and erosion losses during the growing season may not be effective, during spring snowmelt, the dominant runoff process in the Prairies, because:

- the snow, itself, contains P and N (Maule and Elliott 2006b),
- the soil is frozen and water infiltration is minimal so surface soil N and P is mobilized and carried with snowmelt runoff,
- there is no active plant uptake of water or nutrients as plants are dormant or dead at this time,
- phosphorus is released from thawing plant material retained at the soil surface (Bechmann et al. 2005; Flaten et al. 2005),
- plant material retained at surface can increase snow trapping and reduce evaporation losses, which can, in turn:
 - increase the total amount of N and P deposited with the snow that is released during spring snowmelt,
 - increase the volume of water that runs off the fields, as well as
 - increase the duration of flooding in the spring which increases the duration of soil-water contact to release more soil N and P to snowmelt water.

Therefore an essential step for reducing nutrient loading to Manitoba surface and groundwater is to understand the impact of a variety of management practices on N and P transport with snowmelt at the field, farm and watershed scale.

2.5.2.1 Reduce Erosion of Particulate N and P:

The majority of BMPs intended to reduce N and P transport have been developed for *rainfall-induced erosion runoff of particulate N and P* to surface water (e.g., Sharpley 2006). Some of these BMPs increase water uptake to limit downward movement of nitrates and soluble or suspended P with water. Others can reduce the quantity of runoff water and the erodibility of the soil and thereby limit the amount of N and P reaching surface waters by:

- reducing rainfall impact energy,
- impeding water movement over the soil,
- enhancing water infiltration, or
- increasing water uptake.

Some BMPs that manage water movement and may be suitable for controlling the limited amounts of N and P lost by erosion in Manitoba include:

- establish actively growing crops early and include fall seeded crops
- maintain permanent vegetative cover
- retain surface residues
- use cover crops or mulches
- establish vegetated barrier strips or conservation buffers
- use surface or subsurface soil drainage
- restore or construct wetlands
- reduce tillage frequency
- eliminate summerfallow or bare fields from rotations

However, because the predominant nutrient loss pathway for most areas of Manitoba is not erosion (Salvano and Flaten 2006), use of BMPs specific to loss of N and P with erosion runoff may have little benefit or, in some cases, may actually increase losses of dissolved P or nitrate N. For example, Sharpley and Smith (1994) observed conversion from conventional till to no till resulted in a decrease in total P loss (as particulate P) but a simultaneous increase in dissolved P in the initial years following conversion to no till. If such an increase in losses of dissolved P occurred in an area where the decrease in losses of particulate P was not large, the net effect would be negative.

These observations do not mean that erosion control practices should not be implemented for the benefit of reducing erosion and sedimentation. However, these practices are likely to have very limited benefits for reducing nutrient losses in most areas of Manitoba. For more information on these water management and erosion control BMPs, please refer to note 2.19 in the appendices.

2.5.2.2 Reduce Runoff of Dissolved N and P:

Wetlands can intercept runoff of N or P (overland or subsurface) or nitrates in groundwater if the wetland is a natural discharge area for groundwater from agricultural lands. Nitrates from runoff water or groundwater discharge can subsequently be removed via denitrification as wetlands adjacent to or within agricultural lands provide a carbon rich anaerobic environment conducive to denitrification (Pierzynski et al. 2005b). Normally, most of the denitrification proceeds completely to dinitrogen, the harmless form of N gas that forms nearly 80% of the atmosphere. However, incomplete denitrification may generate nitrous oxide, a potent greenhouse gas which can also contribute to ozone depletion.

Vegetated buffer strips have been shown to intercept soluble N as nitrate, ammonium or dissolved organic N in runoff in some studies but not in others, and have even been shown to be a source of N in other studies (Bedard-Haughn 2004; Dosskey 2001; Dillaha et al. 1989). Successfully intercepted N can be taken up by the vegetation, infiltrate into the soil, or be denitrified or immobilized. Bedard-Haughn et al. (2004) measured decreased nitrate (28-42%), ammonium (34-48%) and to a lesser extent dissolved organic nitrogen (9-21%) loads from buffer strips of two different widths. Decreases were highest initially following fertilizer application reduction but gradually began to contribute N to the runoff water, mainly as nitrate and dissolved organic N.

Once dissolved P is mobilized, control is difficult, because physically slowing or filtering the runoff water has little impact on nutrient retention, especially during snowmelt. However, as

mentioned in the discussion of P sources, the risk of P loss as dissolved P increases with the accumulation of P in the soil, as the soil's retention sites become saturated. Therefore, the best way to manage for dissolved P loss is to identify critical source areas at high risk for runoff of dissolved P and manage at the source level, reducing P inputs and maximizing P removals. Although little is known about high risk areas for P transport under Manitoba conditions, those areas where P source management may be especially important include:

- areas subject to periodic flooding due to restricted infiltration, poor drainage or saturation, especially at critical times such as spring snowmelt which induces release of soil P to water, and
- areas in an overland flow or channelized pathway connected to a surface or ground water body, a ditch or low area of a field (surface water) or recharge area over a sensitive aquifer (ground water)

There is very little quantitative information regarding the effectiveness of any other BMPs that will reduce transport of dissolved P during snowmelt, when the majority of runoff occurs in Manitoba. However, in areas where soil test phosphorus is very high at the soil surface and which are not susceptible to erosion, periodic tillage might be useful for burying high P surface soil to depth of tillage. Intercepted dissolved P may also be removed from water by wetland vegetation or P sorption by sediments. However, this practice is not generally regarded as particularly effective for this form of P. Furthermore, as dissolved P loading progresses or increases, there is a reduced capacity of wetland sediments to retain additional dissolved P (White et al. 2000).

2.5.2.3 Reduce Direct, Incidental Losses of N and P:

Incidental transfer is the direct addition of N and P with manure to an adjacent waterway. Most pigs in Manitoba are raised in barns, under confinement conditions, therefore livestock access to surface water is not a major issue for pig producers. However, pig manure that is mechanically broadcast too close to ditches or other watercourses can easily wash into the surface water with rain or snowmelt runoff. Similarly, manure surface-applied in the winter remains on the frozen surface and is easily moved with snowmelt runoff to surface waters during spring thaw. Therefore, regulated and recommended setback distances from ditches and other waterways should be respected; winter manure application should be avoided; and manure should be incorporated or injected as much as possible.

2.5.2.4 Reduce Leaching of N and P:

Nitrate leaching

Nitrate leaching is a high risk for soils with shallow depth to ground water and sandy/coarse texture or preferential flow pathways. The risk of nitrate leaching is further increased if poor growing conditions limit crop N uptake during the growing season and/or increase downward movement of nitrate at any time of year. Therefore:

- for high risk soils in particular, extra careful attention to source management is critical. For example, manure N should be applied at conservative rates, certainly no greater than soil test recommendations. In addition to testing soil annually for residual nitrate-N, producers should also account for accumulated reserves of manure organic N that may be released, especially from solid manure. Also,

following P-based rates for manure application will generally result in lower rates of manure N application, reducing the risk of excess nitrate accumulation and leaching.

- eliminate summerfallow from rotations, as water and nutrients may percolate below the root zone depth in the absence of crops to uptake water
- plant "catch" crops such as perennial forage or fall planted cereals to utilize water and nitrate that would otherwise leach below the root zone of spring annual crops.

Phosphorus leaching

High P soils with a low P retention capacity, such as sandy or coarse textured soils, or with preferential flow pathways such as cracking clay soils, or fields with earthworm or plant root channels or tile drains, may pose a long term risk for loading shallow groundwater with excessive P. This groundwater often flows into surface water bodies, where the discharge of groundwater P causes eutrophication. Avoiding excess accumulation of P in areas prone to P leaching greatly reduces this risk.

2.5.3 BMP Summary

BMPs that are effective at reducing the risk of nutrient loss at the source are better understood than transport BMPs in our relatively cold, dry and flat environment. This, as well as the observation that most nutrient losses to Prairie surface and ground waters are in the dissolved, rather than particulate form mean that source-oriented BMPs have greater potential for minimizing manure P loss to surface waters than transport and interception oriented BMPs. Transport-oriented BMPs that reduce transfer of rainfall runoff P once vegetation is actively growing are well documented for other areas of the world; however, transport-oriented BMPs that are effective during peak loss periods at snowmelt, when the soil is frozen and the vegetation is dormant, have not been validated.

BMPs that are proven, practical, relatively easy to implement and cost effective (e.g. timing and placement of manure application), will be more readily adopted by producers than more cost-prohibitive measures such as manure treatment. BMP adoption may also be encouraged via local demonstration, producer-directed cost/benefit and how-to information and education sessions or fact sheets, and provision of incentives rather than regulations, with allowances for operation scale differences.

2.5.4 Indicators for Estimating Risk of N and P Loss and Encouraging Adoption of BMPs

Various indicators or criteria have been developed to estimate the risk of N or P loss from agricultural land. Once validated for local conditions, these tools can be used voluntarily by farmers and can also be used by public agencies to develop extension programs, incentive programs and regulations that will encourage farmers to adopt the appropriate beneficial management practices that will reduce those risks, accordingly.

2.5.4.1 Nitrogen Risk Indicators

Nitrogen risk indicators are designed to assess the risk of nitrate loss from agricultural land leading to nitrate contamination of water, in determining the environmental sustainability of certain agricultural practices, such as supplying cropland with nutrients by applying pig manure onto agricultural fields. As with P risk indicators, N risk indicators should account for both source and transport factors and should adequately address critical source areas. Critical source areas are those areas of agricultural land where source and transport factors

coincide to create risk for nitrate loss to water, whether surface or ground water. The key source factor is excess soil nitrate content. Transport factors are related to excess water: landscape position (e.g. low-lying positions that collect water from surrounding landscape and/or are groundwater recharge areas), climate (e.g. excess water for nitrate transport: spring snowmelt or high rainfall), seasonal variation in water availability (where precipitation exceeds evapotranspiration). In more humid areas, the presence of nitrate in soil is often a sufficient indicator for the risk for N loss, as water supply is not limiting for N transport (Heathwaite et al. 2000). However, in the semi-arid, sub-humid environment of agricultural Manitoba, excess water is often limiting for downward nitrate movement in the soil. Therefore, the potential for nitrate leaching is more spatially and temporally confined for Manitoba's climate than in a more humid environment.

Residual Soil Test Nitrate Content

Residual soil nitrate-N content is usually measured on a soil sampled to a depth of 24" (60 cm) in the fall, after crop removal (Manitoba Soil Fertility Guide) and provides an accurate historical perspective for determining if fertilizer or manure N was applied in excess of crop removal. For the semi-arid, sub-humid environment of agro Manitoba, fall residual soil nitrate is also a well-documented predictor of nitrate that will probably be available to the subsequent crop, but which may also be at risk of environmental loss (leaching, denitrification) before crop uptake the following spring.

However, soil sampling to this depth does not always provide an accurate indicator of the risk of nitrate leaching. For example, the accumulation of nitrate will not result in leaching unless excess water is also present. Also, the absence of nitrate in 0-24" depth is not a 100% reliable indicator that nitrate leaching hasn't already occurred. Under conditions highly conducive to nitrate leaching such as high rainfall or substantial infiltration of snowmelt water between the time of nitrate accumulation and crop establishment on a coarse textured soil or a soil with preferential flow paths, nitrates may leach below 24" and therefore would not be detected with standard soil sampling procedures. Therefore, periodic deep sampling to 3-5 ft or 100-150 cm is recommended, generally on a 3 to 5 year basis (Tri Provincial Manure Application and Use Guidelines 2003). This "due-diligence" practice is particularly important for fields with a history of manure applications or high application rates.

National Agri-Environmental Indicators for Risk of N Loss

Recently a number of agri-environmental indicators were applied to data available for agricultural land across Canada to assess the risk of contamination of water with nitrate from agricultural land on a provincial and national scale (Drury et al. 2007; De Jong et al. 2007). The results, scaled up from a regional basis, based on large spatial units (soil landscape polygons of scale 1:1million), and coarse-scale climatic data (based on ecodistricts), have not been validated and therefore should be interpreted with caution.

RSN- Residual Soil Nitrogen – N Source

Residual soil N (RSN), the amount of inorganic N (ammonium- and nitrate-N) at the end of the growing season, has been used as an indicator for risk of N loss from agricultural land in Canada (Drury et al. 2007). RSN is soil N in excess of crop removal (due to over-application or poor crop utilization due to adverse growing conditions) and is the difference between all N inputs (N source: chemical or manure fertilizer N, atmospheric sources: biological N

fixation, atmospheric deposition) and all N outputs (crop removal, denitrification, volatilization), assuming mineralization and immobilization are balanced (Drury et al. 2007).

The risk of RSN leaching is much greater in humid areas where excess water from high rainfall more readily leaches nitrate to depth than in drier regions such as agro-Manitoba. For example, throughout the Northern Great Plains region of North America, fall nitrate concentrations are routinely used as the basis for soil testing and fertilizer recommendations throughout this region. Also, several historical cultural practices in this region, such as summerfallow and green manuring, are generally reliable means of increasing the supply of nitrate nitrogen for use by subsequent crops. Therefore, the risk of excess accumulation of water, for transporting the nitrates, must also be considered.

IROWC-N-Indicator of Risk of Water Contamination by Nitrogen – N Source + N Transport

IROWC-N is intended to 'link' the RSN indicator with soil and climatic conditions during the non-growing season to assess the risk of N movement out of the agricultural system to water (De Jong et al. 2007), accounting for both source and transport factors. Risk is based on both the amount of N lost and the concentration of nitrate in water leaving the agricultural system, with the thinking that higher water volume losses would dilute high N losses (i.e. would dilute nitrate concentrations) and would therefore have less of an impact environmentally (De Jong et al. 2007), such as in regions with higher annual precipitation. However, this model uses cumulative precipitation and does not account for the effect of individual events which actually have a greater impact on nitrate loss, particularly for Manitoba conditions where the risk of nitrate leaching is largely confined to the early spring and late fall when snowmelt or large rainfall occurs in the absence of active crop growth.

Nitrate Leaching Risk Indicators in Manitoba

Although Manitoba has not formally developed a set of risk indicators for nitrate leaching, the regulatory approach adopted by Manitoba Conservation reflects the fundamental necessity of rating both source and transport risk factors. For manured land, progressively restrictive limits on residual nitrate concentrations are applied on the basis of soil survey/land inventory information about the soil's productivity and moisture status. As a result, no manure or only very low rates of manure can be applied on extremely sandy soils, for example, where excess moisture could easily percolate below the root zone of a crop. Manitoba Water Stewardship has recently proposed expansion of this approach to regulate the application of all sources of nutrients, including synthetic fertilizers and municipal biosolids. There is, however, debate within the province about the accuracy and scale of the soil survey information being used for delineating the "water quality management zones" and the degree to which the criteria used for determining the agricultural productivity classes and subclasses adequately reflect the environmental risk of N loss.

2.5.4.2 Phosphorus Risk Indicators

Many countries and U.S. states have developed soil test P thresholds and site-specific phosphorus indexes to assess the risk of P loss to surface water and identify remedial strategies for reducing that risk. However, developing or applying an existing risk indicator for estimating the risk of P loss to surface water in Manitoba requires careful consideration of many source and transport factors as well as the indicator's ultimate purpose. For

example, an indicator developed for acidic soils may not be suitable for use in Manitoba where soils are predominantly neutral to calcareous (alkaline), where calcium may be more important for P retention and release more so than aluminum, which is important under acidic conditions (Simard et al. 2001).

A risk indicator should adequately predict the risk of P loss under the dominant loss mechanism and pathway for the agricultural area(s) of concern, which is regionally sensitive, with no negative unintended consequences (Sims et al. 2000). Therefore, an ideal risk prediction tool should account for both source and transport factors. However, the importance of this requirement is debatable, especially in the Prairies where water erosion and its associated landscape factors do not appear to be major factors determining P loss. For example, Feng et al. (2004), after evaluating environmental risk of P runoff loss under manure application for Alberta soils using the Soil P Export Model, concluded that "a single soil P limit for all soils and climates will not work". However, more recent, watershed-scale research in Alberta showed that soil test P values alone accounted for nearly 90% of the variability in P loss (Little et al. 2007). Similarly, various measures of soil test P such as modified Kelowna, Mehlich III, Bray and Olsen soil tests have been well correlated with P concentration in agricultural runoff in many other field and laboratory studies (Pote et al. 1996; Fang et al. 2002; Sharpley et al. 2001b; Guidry et al. 2006) and have been used in the development of P thresholds or P Indexes in many regions to mitigate loss of soil P to surface water (reviewed in Maguire et al. 2005; Heathwaite et al. 2005).

Once an appropriate risk indicator is developed or selected, it must be validated under field conditions representative of Manitoba climate, landscape, soil and management conditions. This essential task will be expensive; calibration and field validation of an environmental risk indicator is more complex and time consuming than for agronomic tests (Sims et al. 2000) and may present a high degree of regional sensitivity.

2.5.4.2.1 Soil Test Phosphorus

Increases in runoff loss of P and water-extractable P are directly related to increases in soil test P (STP) for both acid and neutral to alkaline/calcareous soils (Sims et al. 2002; He et al. 2006; Fang et al. 2002; Casson et al. 2006; Sawka et al. 2007ab; Little et al. 2007). The relationships for Alberta and Manitoba soils in particular are strong and reasonably linear (Figures 2.8 and 2.9) illustrating that:

- conventional agronomic soil tests, including the Olsen test commonly used in Manitoba, are a reasonably reliable indicator of the risk of P loss in the Prairies
- P losses increase steadily with increases in soil test P. When these soil tests are used to measure the amount of available P in soil, the soil's capacity to retain P does not appear to be fixed or absolute; these soils are "leaky" vessels that consistently "leak" more P with rising concentrations of soil test P (for more information on soils and soil tests that exhibit sharp changes or "break points" in P release with changes in soil test P, please refer to note 2.20 in the appendices)
- the selection of realistic, regionally-specific, ecologically sound targets for the P concentrations in runoff water is a critical step in the process of selecting a soil test P threshold that will be regarded as "excessive" from a water quality perspective

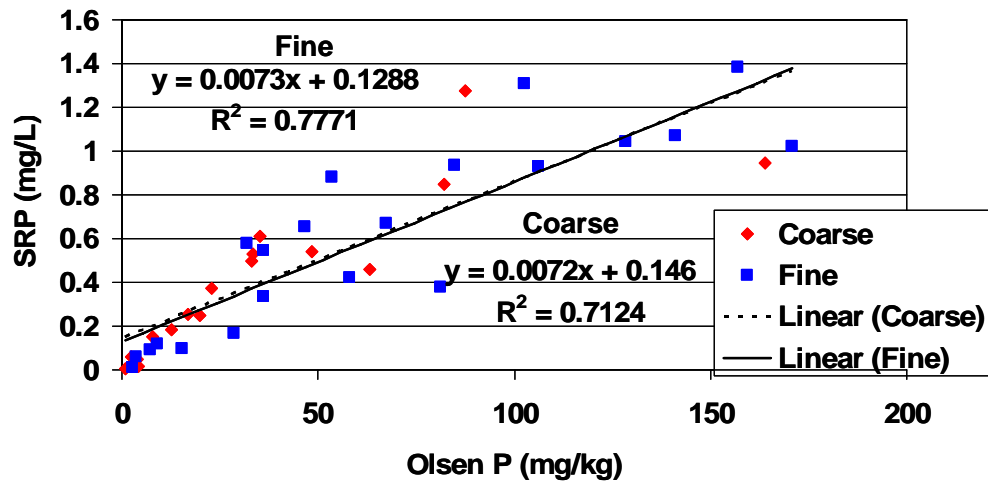


Figure 2.8. Olsen soil test P and soluble reactive P concentrations in simulated runoff from coarse-textured (sandy soils) and fine-textured (loamy or clay) soils in Manitoba (Sawka et al. 2007ab).

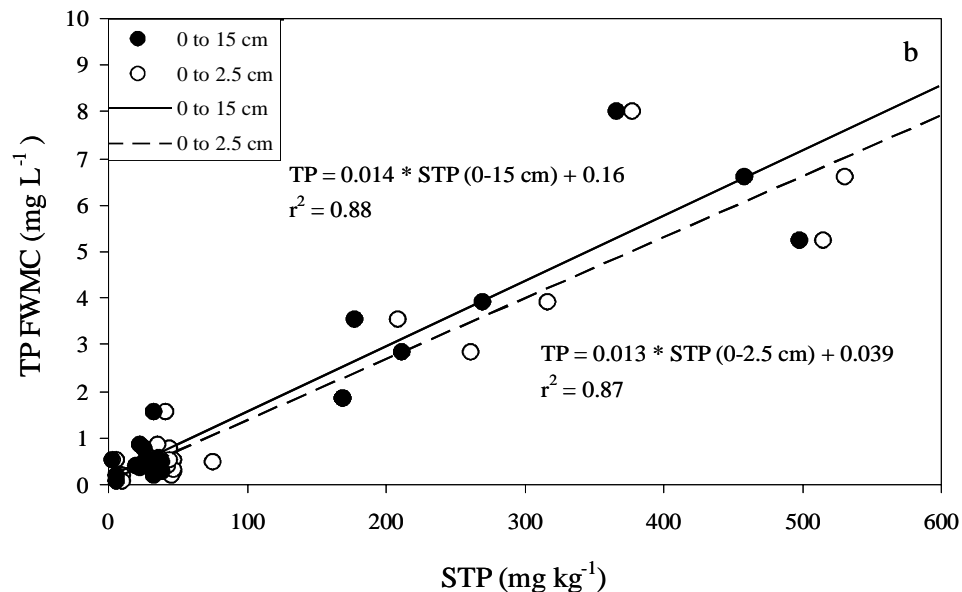


Figure 2.9. Modified Kelowna soil test P and total P concentrations in runoff (flow weighted means) from small Alberta watersheds (Little et al. 2007).

Sampling methods for soil testing

The appropriate soil sampling depths for environmental soil testing for P may be shallower than for agronomic tests (e.g., 0-5 cm vs. 0-15 or 0-30 cm). Use of the deeper samples may underestimate rainfall or snowmelt runoff loss where only the upper layer of soil interacts with runoff water (Sims et al. 2000). However, due to the high correlation between soil test values for 0-5 cm and 0-15 cm samples in most cultivated soils, this limitation may not be significant for most situations in the Prairies. For example, Little et al. (2007) showed that 0-2.5, 0-5, and 0-15 cm samples were equally effective in accounting for runoff P concentrations from 8 small watersheds in Alberta.

Extraction methods

There are two general groups of phosphorus soil tests: agronomic and environmental. Extraction methods for determining agronomic soil test P include Olsen (commonly used in Manitoba, in neutral to alkaline soils), Modified Kelowna (frequently used in Alberta and Saskatchewan in a wide variety of soils), Mehlich 3 (used in the U.S. and Quebec, in acid soils), and Bray (used in the U.S., in acid soils). Extraction methods for determining environmental soil test P include these methods, plus water, calcium chloride and various resin or oxide sink-based methods.

Each of these methods extracts different amounts of "available P." However, for soils that are neither strongly acid nor alkaline, there are reasonably strong correlations between the values for many of these soil test P methods. The equations for these relationships enable approximate conversions for values from one soil test method to another (Ige et al. 2006). These relationships, though, are approximate and may vary with nutrient management history. For example, Kumaragamage et al. (2007) found that at higher levels of STP, the relationships between various soil test P methods and water-extractable P were not as strong as at lower STP levels, and were different for manured and non-manured Manitoba soils.

Determining appropriate environmental thresholds for soil test P

To determine the level of soil test P that should be regarded as environmentally excessive, researchers will investigate the relationship between these various methods of measuring soil test P and the concentration of P in runoff. Researchers then use these relationships to develop environmental soil test P thresholds for different regions. Environmental soil test P thresholds vary with region (Maguire et al. 2005) and soil characteristics (Torbert et al. 2002) because the relationship with P in runoff or water-extractable P and some soil test P values can vary widely between soils and with soil management (Sharpley et al. 1996).

In practice, the basis for developing environmental thresholds varies widely from one study to another and so do the threshold values that are recommended. For example, based on literature values for Mehlich 3 thresholds (Sharpley et al. 2001, 2003a) and laboratory-based conversion equations for 214 Manitoba soil samples, Kumaragamage (2007) suggested an upper range of environmental P thresholds at 88-118 ppm for Olsen P and 15-20 ppm for water-extractable P. Conversely, for calcareous soils in southern Minnesota, Fang et al. (2002) recommended 40-70 ppm Olsen P as a critical range for the soils used in their study. Fang's recommended thresholds were based on the target of 450-600 µg biologically available P or total P per L in runoff, but other targets might be used in other regions.

Therefore, various regions select different critical threshold values to define environmentally excessive levels of soil test P (Table 2.16).

Table 2.16. Agronomic and environmental thresholds for soil test P values and P management recommendations for various states in the U.S.

State	Soil Test P Threshold Values (ppm)		Soil Test P Method	Management recommendations to protect water quality
	Agronomic	Environmental		
Arkansas	50	150	Mehlich-3	<i>At or above 150 ppm soil P: Apply no more P, provide buffers next to streams, overseed pastures with legumes to aid P removal, and provide constant soil cover to minimize erosion.</i>
Delaware	25	50	Mehlich-3	<i>Above 50 ppm soil P: Apply no more P until soil P is significantly reduced.</i>
Idaho	12	50 100	Olsen	<i>Sandy soils, above 50 ppm soil P: Silt loam soils, above 100 ppm soil P: Apply no more P until soil P is significantly reduced.</i>
Ohio	40	150	Bray-1	<i>Above 150 ppm soil P: Reduce erosion and reduce or eliminate P additions.</i>
Oklahoma	30	130	Mehlich-3	<i>30-130 ppm soil P: 50% P rate on >8% slopes 130-200 lb/acre soil P: 50% P rate and reduce surface runoff and erosion Above 200 lb/acre soil P: Amount of P applied should not exceed that taken up by the crop and removed as harvested produce</i>
Michigan	40	75	Bray-1	<i>Below 75 ppm soil P: P application not to exceed crop removal. Above 75 ppm soil P: Apply no P from any source.</i>
Texas	44	200	Texas A&M	<i>Above 200 ppm soil P: Amount of P applied should not exceed that taken up by the crop and removed as harvested produce.</i>
Wisconsin	20	75	Bray-1	<i>Below 75 ppm soil P: Rotate to P-demanding crops and reduce P additions. Above 75 ppm soil P: Discontinue P applications.</i>

Source: Sharpley et al. (2003a)

2.5.4.2.2 Degree of P Saturation

The degree of P saturation (DPS) is another means of assessing the risk of P loss. DPS is meant to account for both the *capacity* of the soil to retain P and the *degree* to which that capacity is *occupied* or *saturated* (Akinremi et al. 2007a). Therefore, this measurement may be more useful as an environmental risk indicator than an agronomic-based indicator, providing the equation for predicting DPS is a reliable indicator of P loss risk for the region in question (Figure 2.10). For example, at the same soil test P value, a soil with a low retention capacity (b) will have a higher degree of P saturation than a soil with a higher retention capacity (a).

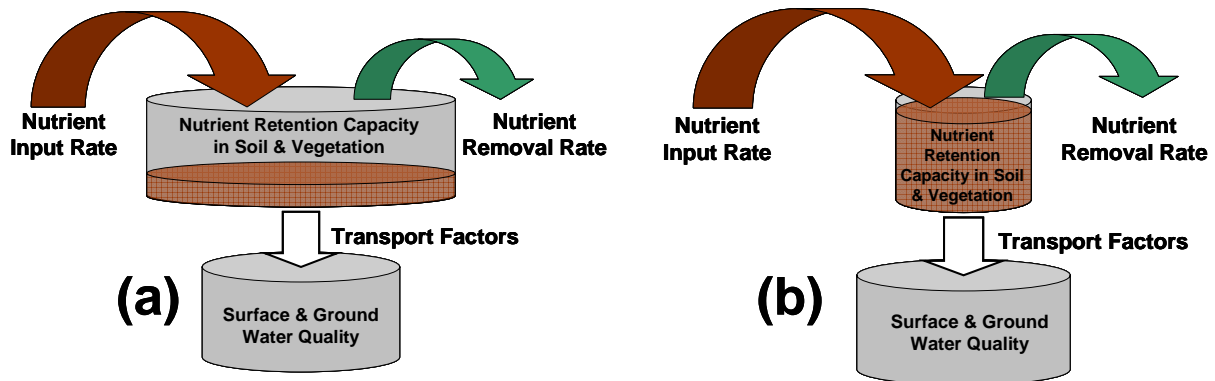


Figure 2.10. The risk of losing applied P is low if the soil has a high retention capacity for P and a low degree of saturation (a); however, the risk of P transfer is high if the soil has a small retention capacity for P and a high degree of saturation (b) (adapted from Flaten et al. 2003).

However, analyses required for actual determination of the P saturation of a soil are not usually routine laboratory procedures for agronomic soil testing (Kleinman and Sharpley 2002). Therefore, researchers have looked at ways of estimating the actual degree of P saturation using more commonly measured variables such as Mehlich III extractable Al, Fe, Ca and/or Mg (Beauchemin and Simard 1999; Sims et al. 2002; Ige et al. 2005a; Pellerin et al. 2006a and Pellerin et al. 2006b). Various equations developed to estimate the degree of phosphorus saturation have been well correlated with actual P saturation determination in the U.S. (Kleinman and Sharpley 2002). Researchers at the University of Manitoba have developed preliminary DPS equations based on Manitoba soil samples (Ige et al. 2005a and Akinremi et al. 2007b). However, recent validation studies with simulated rainfall have shown that further research on these DPS methods is required before they can be used in Manitoba (Sawka et al. 2007ab). For more information on this research, see note 2.21 in the appendices.

2.5.4.2.3 Phosphorus Index

The site-specific phosphorus index is the most widely used tool for assessing the risk of P loss (Sharpley et al. 2003b; Heathwaite et al. 2005). These indices incorporate key source and transport factors with the capacity to identify “critical source areas” or areas of combined high source and transport risk for P loss.

However, phosphorus indexes developed for nearby jurisdictions do not appear suitable for use in Manitoba. Salvano and Flaten (2006) evaluated the suitability of three phosphorus index risk indicators (Minnesota, Canadian IROWC-P, preliminary P risk indicator for Manitoba) for assessing the susceptibility of P loss to surface water within 14 watersheds in Eastern and Western Manitoba (correlation of P risk indicators with water quality). None of the indicators were significantly correlated with water quality parameters. The indexes underestimated the importance of STP and overestimated the importance of soil erosion for the watersheds examined. Part of the reason for the poor performance of these indexes is that they have been designed to assess transport factors based on rainfall-induced erosion of particulate phosphorus, which is not the main loss pathway from agricultural soils in Manitoba.

A Phosphorus Index for Manitoba therefore would need to be developed using the most relevant source and transport indicators of P loss under Manitoba conditions, where rainfall induced erosion is not a major factor. All of these factors require evaluation under snowmelt dominated runoff conditions, with the soils, landscapes, climates, and agricultural practices that are typical for this province.

P source coefficients, where differential weighting is assigned to different sources (e.g., different manures, biosolids, synthetic fertilizers, or vegetative residues) most susceptible to loss based on runoff loss potential, must also be developed for P sources that are typical for the Prairie region. P source coefficients are the weighting factors assigned to P sources based on their relative potential for P loss with runoff (Leytem et al. 2004a). P source coefficients may be standardized or weighted relative to mineral fertilizer, which is considered to represent the highest potential for dissolved P runoff loss or based on the availability of a particular fraction of P (Elliott et al. 2006). However, Simard et al. (2001) observed P Index studies over-estimated P loss from manure P relative to inorganic fertilizer P due to an excessively high weighting assigned to manure than fertilizer P. Therefore a suitable P index needs to appropriately assign weighting of P source based on comparative P losses from fertilizer and manure P as many P indexes assign higher weighting to manure than fertilizer P. As crop residues are also an important source of dissolved P, including this loss mechanism in a Manitoba-based phosphorus index may improve P loss predictability.

More research is also required to quantify the transport mechanisms for P loss from the various landscape regions of Manitoba such as the Red River Valley with its fine textured soils subject to periodic inundation, areas in Western Manitoba with “pothole” topography where water can be retained, or areas such as the Upper Assiniboine Delta, Interlake and Southeastern Manitoba with permeable coarse textured soils over shallow important aquifers (Flaten 2003).

Phosphorus Loss Risk Indicators in Manitoba

As is the case for nitrate leaching risk indicators, no phosphorus loss risk indicators have been formally adopted in Manitoba. However, Manitoba Conservation’s new manure P regulations, similar to its well-established N regulations, reflect a combination of source and transport factors. Source factors, most notably balancing P application with removal, are addressed through soil test thresholds based on 0-15 cm samples analyzed for Olsen extractable P. Subsequent to the introduction of this regulation in November 2006, this sampling depth and this extraction method have been supported by Prairie-based research discussed earlier in this section. There is, however, concern about allowing livestock producers to build up soil test P to the 120 ppm threshold before mandating a balance between further application of P and P removal by crops. This threshold is in the upper range of soil test thresholds permitted by other jurisdictions; however, there is little evidence from local watershed studies that could be used to justify lowering the 120 ppm threshold at this time.

From a transport perspective, the new manure P regulations are designed to minimize the direct, incidental transfer of manure P into waterways by restricting manure applications nearby the waterways and in areas such as the Red River Valley, areas that are regularly inundated by overland flooding during spring snowmelt. It's important to note that these restrictions were not intended to increase the capacity for intercepting P between the field and the waterway, because the effectiveness of practices developed elsewhere (e.g., vegetative buffers and conservation tillage) for this purpose have not yet proven to be very effective during snowmelt runoff in the Prairies.

Although Manitoba's risk assessment strategy for its new manure P regulations is relatively simplistic, it represents a progressive first step towards reducing excessive P loading. And, given the technical challenges already mentioned in the sections on degree of P saturation and the P index, considerable research will be required before the Province has the science base to implement more sophisticated combinations of source and transport risk assessment tools for its P regulations.

2.6 Conclusions

Pig manure is a relatively small source of nutrients being used on Manitoba's agricultural land, compared to synthetic fertilizers, for example. However, from a technical perspective, pig manure is still a significant source of N and P that must be managed properly to avoid overloading of nutrients that could occur due to excessive rates of manure application on too small a land base. In addition, from a social perspective, Manitoba's pig industry should not lag behind the multitude of small sources, rural and urban, who must reduce their contribution to surface and ground water quality problems. Everyone must do their share.

The Province's longstanding N-based manure application regulations appear to be reasonably sound and have been respected by the majority of Manitoba's pig industry. Therefore, the risk of excessive leaching of nitrate from pig manure appears to be small. However, compliance with the new P-based regulations will present substantial challenges, especially in southeastern Manitoba, where livestock densities are already high and the availability of additional land that is suitable for manure application is very restricted. It's very important to note that the challenges in these regions are not unique to the pig industry; beef, dairy and poultry farmers in these areas will also face substantial challenges with manure P management.

Manitoba's P-based manure management regulations have been in place for less than one year. From the research perspective, there is ongoing controversy about whether the 120 ppm P threshold for balancing P application and removal is too high for maintaining water quality in the Province. However, in the absence of local data on soil test P relationships with runoff, this threshold was developed as a first step. Additional land and water research will be required to develop a scientifically sound basis for adjusting that threshold.

Given the newness of the P regulations, Manitoba's livestock industry is only beginning to adapt to the challenge of balancing P application with removal. However, in order to accelerate this adaptation and to minimize further increases of runoff P losses from manured land, the Government of Manitoba (all relevant departments) and the province's livestock industry should work together to develop and implement a comprehensive and coherent approach for dealing with manure P. A wide range of beneficial management practices (BMPs) will be required to achieve this goal, because every livestock producer will face different opportunities and constraints for their operations and in their communities. Effort and money are also required to develop management practices that will be more technically effective and economically affordable, combined with additional investment in a suite of education, incentive and regulatory programs that will encourage the adoption of BMPs that are proven to be effective in Manitoba's soil, climate and landscape conditions. To be fully effective, Provincial policies must be a well-balanced blend of "carrots" and "sticks" and not excessively reliant on regulations, alone.

Lastly, improving nutrient management within Manitoba's livestock industry is not simply a challenge; it's also an opportunity. At the same time as livestock producers are concerned about the restricted rates of manure application in the new regulations, grain and oilseed producers are concerned about the substantial increases in the cost of synthetic fertilizers. With proper planning and management, it should be feasible to develop ways of improving the economic and environmental sustainability of both types of farms while also improving the welfare of all Manitobans and the ecosystem that we share.

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2.8 Technical Notes

2.1 Basis for determination of N and P requirements

Dairy cattle N and P requirements: N was calculated from the sum of ruminally undegraded feed protein (g/day) plus rumen-degradable protein (g/day) / 6.25. For lactating dairy cow, P and N requirements are estimates generated from a computer model with parameters of a sample diet for a holstein at 680 kg weight, 90 days in milk, with a milk production of 25 kg, with milk having a 3.5% fat content, and a true milk protein of 3.0%. For bred heifer, N and P requirements are based on a large breed heifer 240 days pregnant (mature weight = 650 kg), weighing 550 kg at an average daily gain of 0.8 kg/day.

Beef cattle N and P requirements: N was calculated from the metabolizable protein required for gain (g/day) / 6.25. N and P requirements for growing and finishing cattle are based on a growing and finishing Angus at 300 kg (finished weight = 533 kg) at an average daily gain rate of 1.5 kg/day.

Poultry N and P requirements: N was calculated from crude protein content (as a percentage of dry matter intake) / 6.25. N and P requirements are based on a male broiler at 6 weeks receiving 1141 g feed over a one week period.

Growing pigs N and P requirements: N was calculated from crude protein content (as a percentage of dry matter intake) / 6.25. The factor for converting crude protein to N content based on 100 g protein being equivalent to 16 g nitrogen (NRC 1998).

2.2. Factors that control nitrate leaching to groundwater and surface runoff of N

Climate and weather factors

In Manitoba's climate, there is limited risk of net downward movement of water *during the growing season* as plants continually remove water from the soil, which is lost to the atmosphere during evapotranspiration. However, there is modest risk of nitrate leaching during the spring and fall when rainfall is not being used by crops and immediately after snowmelt, when most runoff into depressional areas occurs. In years where fall and winter precipitation are low, N transport with runoff water is reduced due to the reduced quantity of melt water (Maule and Elliott 2006ab). For surface runoff losses of N, providing there is a pool of N available, surface losses will also be higher at these times which correspond to greater potential for overland flow of water. Therefore, the potential for surface runoff losses of N is also highest during snowmelt runoff as this process typically accounts for at least 80% of runoff in the Prairies (Nicholaichuk 1967; Glozier et al. 2006).

Landscape factors

Nitrate supplied with synthetic N fertilizer or manure in excess of crop uptake can move below the crop rooting depth, and in upper slope positions may remain at depth while at lower positions nitrate may be leached or denitrified and therefore may not accumulate (Whetter et al. 2007). The risk of nitrate leaching to groundwater is higher at groundwater recharge locations in the landscape, which are typically depressional areas where snowmelt or rainfall water from surrounding land may drain into, thereby increasing the amount of water and nitrate in that area. During snowmelt, in depressional areas the water table may reach the surface and can therefore pick up accumulated soil nitrates in these shallower depths, to be transported downward as the water table lowers again (Maule and Elliott 2006a). Nitrate can also be transported with lateral subsurface movement of water along an

impermeable layer in the soil and, if this flows to a discharge area of the landscape, leached nitrate may end up in surface water. However, in low lying areas of the landscape where the soil is periodically saturated due to poor draining or where “exfiltration” of water occurs (net upward movement of soil water, not related to crop removal), the risk of nitrate leaching is low and nitrates may be removed by denitrification.

Soil factors

Soil texture and structure that enhance infiltration of water into the soil as well as continued net draining of water within the soil will enhance the risk of nitrate leaching. Typically soils with a high degree of permeability and hence high risk for nitrate leaching are coarse textured (sands) or have preferential flow pathways (vertical channels) that occur either naturally (e.g. high cracking clay soils) or are the result of land management (e.g. artificial drainage, root and earthworm channels in reduced tillage or perennial forage land). In terms of surface runoff losses of N, soils with a low degree of permeability or low infiltration capacity would increase the risk of N runoff, but would decrease the risk of nitrate leaching.

Land and water management factors

Generally, land and water management that enhance the rate and amount of water infiltration into the soil to depth will also increase nitrate leaching, providing sufficient nitrate is present at times of water inflow. Land and water management practices that enhance the ability of soil to allow water infiltration include minimum tillage or established forage, and the use of tile drainage. However, some of these practices, such as growing perennial forage, also improve water utilization and reduce the risk of excess water accumulation in the soil. Practices that eliminate plant utilization of water and nitrate, such as summerfallow or bare land create the highest risk of nitrate leaching. Use of irrigation increases the amount of water being applied to the field, and if applied in excess or if followed by substantial rainfall can transport nitrates downward. However, in a climate such as Manitoba's, where irrigation is applied on a supplemental basis, well-managed irrigation can keep the crop healthy during intermittent dry periods and, by maintaining the crop's ability to use water and nutrients, may reduce the risk of leaching during subsequent wet periods. For surface runoff losses of N, land and water management practices that restrict water infiltration and increase surface flow of N with water to surface water, such as in-field artificial drainage or application of manure or synthetic fertilizer into or next to a drainage ditch would increase the risk of N runoff; these practices are also likely to increase the risk of nitrate leaching.

2.3 Removal of P by grazing cattle

The vast majority of P injected by cattle grazing on pasture is excreted as feces and urine. The net removal of P from the pastures is simply equal to the amount of P found in the weight gained by the animals between the time that they arrive at the pasture and the time that they are moved elsewhere. As a result, P removal by grazing animals can be estimated by the P content in live cattle, which Lynch and Caffrey (1997) estimated 7-9 kg P for each 1000 kg. However, this value has also been measured at 7 kg P removed per 1000 kg of gain in experiments where "stocker cattle" were destructively sampled and analyzed in a U.S. grazing systems (Gibson et al. 2002).

2.4. Immobilization and Mineralization of Phosphorus in Soil

In soil, organic (immobilized) P is associated with particulate P while the inorganic (mineralized) P represents both particulate and soluble/dissolved P fractions. Net immobilization results in low P availability/high P retention while net mobilization results in increased availability of inorganic P for subsequent reactions, plant uptake or loss from the soil. These transformations are controlled by the amount of organic C relative to organic P

in soil organic matter (net immobilization at C:P > 300:1, [P]) and activity of phosphatase enzymes in soil (high activity leads to net mineralization) (Condrón et al. 2005). Research has shown organic P mineralization to coincide with organic C decomposition, but overall, the interactions and transformations of organic P in soil are complex and little understood (Condrón et al. 2005).

2.5. Soil Properties and Phosphorus Retention

The capacity of a soil to retain additional P depends on the portion of the retention capacity that is already occupied, or the **degree of phosphorus saturation** of the soil. Soils with a higher P retention capacity and a lower degree of P saturation can hold more P than soils which have a higher degree of P saturation (Kleinman et al. 2000). As P builds up in the soil, the soil has an increasingly reduced capacity to hold additional P, thereby increasing the risk for P loss. Many soils exhibit a “change” or “break point” in soil test P above which the increase in P release from soil per increase in soil test P is much higher than below the change point (Casson et al. 2006; McDowell et al. 2001). Several soil factors establish the P retention capacity of the soil (Leclerc et al. 2001) and at what level of soil test P the change point occurs. P inputs to the soil such as manure P determine the degree to which that capacity is filled or *saturated*.

2.6. Soil carbonates and P retention

The role of calcium carbonate for P retention has been briefly summarized in Flaten (2003), pointing out the somewhat contradictory importance of calcium carbonate as a sink for P in soils with high calcium carbonate content. In further support of the less dominant role of calcium carbonate as a sink for P, in a more recent report on the P retention capacity of Manitoba soils, calcium carbonate was not nearly as important as other soil factors (Akinremi et al. 2004). For example, as published in a subsequent paper, the best relationships between the adsorption parameters, P₁₅₀ and S_{max}, and the soil properties were obtained with the sum of Mehlich-3 extractable Ca and Mg ($R^2 = 0.66$) and the sum of exchangeable Ca and Mg ($R^2 = 0.64$). Mehlich-3-Ca and -Mg each explained 56% of the variation, while clay content explained 40% of the variation in the P retention capacity of these soils (Ige et al. 2005b). In contrast, the R^2 for calcium carbonate and these retention parameters was only 0.15; therefore % calcium carbonate was dropped from further equations for predicting P retention capacity.

2.7. Stability of liquid pig manure compared to solid cattle manure

In laboratory studies, Kashem et al. (2004) measured a shift from water extractable P to Olsen P, the P fractions considered available to plant uptake or environmental loss, for pig manure but these fractions decreased, presumably due to net immobilization for cattle manure over a 16 week period. From these results, a greater proportion of the total manure P added would be available in the year of application for pig manure than for cattle manure. In support of this, Stumborg et al. 2005 (in Schoenau and Davis 2006) found residual (slowly available) inorganic P (15 x higher) and organic P (3 x higher) fractions were much higher for cattle than for pig manure in the top 15 cm of soil following 8 annual applications of manure at agronomic rates of N. Adding manure to soil increases the amount of organic matter in the soil (SOM), and if applied to agricultural land with low SOM, inorganic P may be immobilized for a period after application, particularly if the manure contains high amounts of stable organic P such as phytic acid.

2.8. Phosphorus Transport Processes - Dissolved vs. Particulate P

Total phosphorus in rainfall runoff is comprised of both dissolved and particulate P fractions (Kleinman et al. 2006). Kleinman et al. (2006) reported dissolved reactive P comprised an average of 71% of the total P in rainfall runoff from two positions on sloped land (6% and 30% slope) with varying initial soil P content. The proportion of dissolved P in total runoff P is directly related to the water-soluble content of the manure source (Kleinman et al. 2002; Shigaki et al. 2006; Vadas et al. 2007). Vadas et al. (2007) observed that the highest amount of dissolved P loss from surface applied manure occurs in the first runoff event. Addition of manure can increase the proportion of total P as dissolved P (Miller et al. 2006).

In rainfall runoff simulations, the largest release of P to runoff water is during the initial period of water flow (Sharpley 1997; Sharpley and Moyer 2000) when available soil P levels are highest. Researchers have also observed peak rainfall runoff to occur with fall rains after crop harvest (Goulet et al. 2006; Hargrave and Shakyewich 1997) when there is no active crop uptake of water and nutrients and no crop cover to intercept rainfall impact. In regions where the majority of P loss occurs during snowmelt where a large volume of water is in contact with the soil surface, the contribution of rainfall events to total annual P loss is generally low due to the much lower water volume of individual rainfall events, even though the P concentration in rainfall runoff may be higher than in snowmelt runoff (Jamieson et al. 2003).

Rainfall-Induced Erosion of Particulate Phosphorus - This loss pathway has been well studied in the United States where it is the dominant P loss pathway (Sharpley et al. 1999). In localized areas of Manitoba where agricultural lands are susceptible to erosion, loss of particulate P in rainfall runoff may be an important loss pathway. Rainfall runoff studies conducted in the escarpment region of Manitoba with inorganic P fertilizer application under conventional tillage on sloped (9% slope) erosion plots provide a good example (Hargrave and Shakyewich 1997). The authors reported total runoff P was predominantly particulate P, and was directly related to the amount of soil loss during the rainfall events. However, rainfall-induced erosion of particulate P is generally a less important P loss pathway in Manitoba. As evidence that erosion loss of P is not an important P loss pathway in Manitoba, Salvano and Flaten (2006) found erosion parameters in P Indexes developed for other regions were poorly correlated with Manitoba data.

There are three components to erosion of **particulate** phosphorus:

1. **Soil structure, texture and degree of aggregation, which determine the susceptibility for detachment.** Soils that support a lower weight fraction (e.g. high clay content, high soil organic matter content) and smaller aggregate size have a greater potential for detachment (Haygarth and Sharpley 2000). Clay and soil organic matter fractions support high surface area and high diversity of fractions for P bonding (Gburek et al. 2005) and are more readily mobilized and transported with water-runoff energy due to smaller particle size. Long-term cattle manure applications resulted in higher proportion of erosion susceptible microaggregates enriched with N, P and C relative to unmanured soil (Whalen and Chang 2002).
2. **Nature of the rainfall event, which determines the amount of energy available to detach, mobilize and deliver soil particles to surface water.** Rainfalls of greater intensity and duration have greater energy to detach more and larger

particles from the bulk soil for transport. Hargrave and Shakyewich (1997) found high runoff erosion P loss was most strongly related to high rainfall intensity.

- 3. Landscape slope and interference due to management (e.g. residue, barriers), which determine flow momentum and impediment to delivering mobilized particulate P to surface water.** Landscapes with a greater degree of slope and less physical interference to overland water flow will enable greater flow energy (Sims 2004). Runoff water with higher flow energy can transport particulate phosphorus for larger distances with less settling of the larger size fractions.

Therefore, in areas susceptible to runoff erosion, particulate P losses may be reduced with management practices that reduce the impact and flow energy of rainfall such as conservation tillage, residue retention, riparian zones and buffer strips to name a few (Sharpley et al. 2006).

2.9. Migration of Leached P into Groundwater and then to Surface Water

Movement of water through soil occurs as preferential flow and matrix flow and can reach surface waters with subsurface lateral flow. Leaching losses of P can become an important loss pathway when soils contain high amounts of P and are at a high level of P saturation, such under long term manure application (Nelson et al. 2005), particularly if the soil supports preferential flow such as high cracking clay soils, fields with established root channels or fields with tile drainage (Sims et al. 1998). Matrix flow of P may even occur for high P coarse textured soils that contain a high water content to facilitate flow (Sims et al. 1998) or a shallow water table. Generally P movement with soil water to ground water is greater with preferential flow than with matrix flow because of the reduced contact time for P in soil water to be precipitated or sorbed from solution.

Long term manure additions can overwhelm the P retention capacity of the soil, thereby increasing the availability of P in solution for downward transport, particularly with shallow water tables (Eghball et al. 1996) or with irrigation (Whalen and Chang 2001). Also, long term applications of manure can enhance leaching of P during periods of flood-induced reducing conditions under conducive hydrological conditions (Ajmone-Marsan et al. 2006). Phosphorus that leaches below the rooting zone cannot be removed by crop uptake and may be susceptible to deeper leaching to groundwater or lateral transport to surface water (Schoenau 2006; Nelson et al. 2005).

However, in Manitoba studies on coarse textured soils showed minimal to no downward movement of P following repeated applications of liquid pig manure (Ominski et al. 2007; Akinremi 2005). Nitrogen-based pig manure applications over a 3 year period to established pastures on coarse textured soils in Eastern Manitoba showed P accumulation in the top 15 cm but minimal downward movement (Ominski et al. 2007). Similar results were obtained following a 3 year study at Carberry, Manitoba where phosphorus accumulated at increasing rates, based on manure rate, in the top 10 cm of soil but did not move downward below 20 cm on a fine sandy loam (Akinremi 2005).

Artificial drainage reduces overland flow of water and redirects water through the drainage channels which can reduce runoff P losses but may enhance P loss via subsurface lateral transport (Goulet et al. 2006; Sims et al. 1998). Installation and configuration of artificial drainage may also alter soil P retention and release mechanisms. In one study looking at fertilizer P leaching, controlled drainage that decreased water loss by 27% compared to free drainage, but that artificially decreased the depth of water table, led to increased P (96%

dissolved P) in leachate. This large increase, relative to another drainage site that did not alter the depth to water table, was thought to be related to increased release of P from soil under anoxic conditions (Sanchez Valero et al. 2007).

2.10. Factors that control surface runoff of phosphorus

Climate and weather factors

Snowmelt or rainfall water *mobilize* and *deliver* soil P to surface water. More soil P can be mobilized and transported with snowmelt or rainfall runoff when a large quantity of water is in contact with a large soil surface area for a prolonged period such as during ponding of snowmelt or rain, and when there is sufficient energy for runoff water to reach surface water. In Manitoba, the majority of P movement to water occurs with snowmelt. Appreciable amounts of P from fall-applied manure may be transported with fall rains.

This seasonal variability as well as yearly variability can be illustrated by temporal flow patterns of the Red river which represent land that is predominantly used for agriculture. Peak periods of P loading to these rivers correspond with snowmelt while later, smaller P loading events correspond with high rainfall amounts in the spring (Figure A1).

In Manitoba, there is a relatively high risk of snowmelt runoff and rainfall runoff based on index maps developed for the Prairie Provinces (Eilers and Buckley 2002).

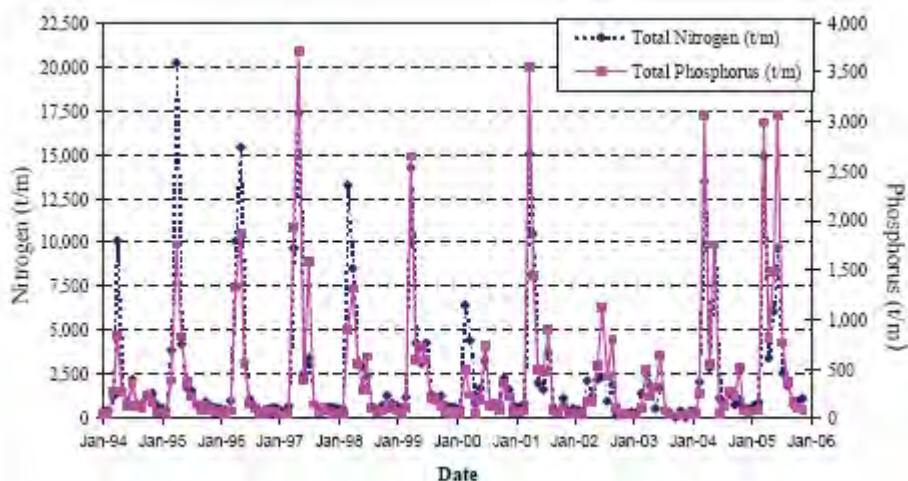


Figure A1. Monthly total phosphorus and total nitrogen loading rates (tonnes/month) from the Red River at Selkirk (1994-2005) (Adapted from Lake Winnipeg Stewardship Board 2006).

Landscape factors

There is a higher potential for P loss to surface water from manured land that is close to surface waterways or that supports a shallow water table with high connectivity to surface water. In addition to determining the pathway for P loss to surface water, the hydrology of a landscape or watershed also determines:

- the rate and volume of drain water moving over a soil surface area,
- the duration and degree of water contact with soil and organic material at the soil surface, and

- whether overland flow is due to poor infiltration or due to saturation of the soil (Sims 2004). Infiltration of unsaturated soils can be impeded due to poor soil structure, high clay content, or due to high rainfall intensity. Saturation flow excess runoff occurs on soils with a high water content or high water table.

As snow melts in the spring on nearly level landscapes with localized small variations such as commonly present in Manitoba, overland flow of meltwater covers the land with a shallow depth of floodwater, maximizing contact with the surface layer of soil until the water either slowly infiltrates or gradually drains to surface water along natural drain pathways (Sheppard et al. 2006). As a result of a variety of processes, higher soil P has been measured along natural drainage flow channels compared to background field P concentrations in Manitoba soils (Sheppard et al. 2006).

Melt water may remain for a longer period in small localized depressions where infiltration is restricted such as at a discharge site, or high clay or saturated soil if the soil is fine textured soil saturated soil. Prolonged soil-water contact in these low lying, saturated areas can lead to increased release of soil P and increased concentrations of dissolved P in runoff water (Feng et al. 2004), especially for soils with a long term history of manure application (Ajmone-Marsan et al. 2006). Temporary ponding of snowmelt or rainfall waters allows for settling out of mobilized particulate P, but over time enhances P desorption to increase soluble P concentration in the runoff waters (Ginting et al. 2000).

Soil factors

The soil surface is in direct contact with rainfall or snowmelt water. Water-soil contact results in release of bound or adsorbed P from soil clays, oxides and organic matter (dissolved P) and/or physical separation of P bound with fine, lightweight particles such as silts, clays, fine sands and organic matter from the bulk soil (particulate P). Released soil P is carried with water along flow pathways to surface water. Although most research has shown P loss in runoff to be directly related to soil P content for a given type of soils, the numerical relationships between P concentrations in soil and in runoff water often vary greatly, depending on soil type and texture (Pote et al. 1999; Sharpley et al. 2001a).

Soil texture and structure, as related to the rate and capacity of water infiltration, are important for rainfall runoff and leaching processes but not for snowmelt runoff when the soil is frozen and prevents infiltration (Feng et al. 2004), aside from the thin thawed surface layer. Zhao et al. (2002) determined soil texture was not an important factor governing snowmelt infiltration into frozen soils in Saskatchewan.

Where the predominant form and loss pathway is erosion runoff of particulate P, tillage practice/cropping treatment (land management) and intrinsic soil erodibility are key factors for determining loss potential, moreso than manure addition (Michaud and Laverdière 2004). Michaud and Laverdière (2004) in southwestern Québec determined soil type accounted for 70% of the variability in particulate P bioavailability, specifically related to clay content and total phosphorus concentration.

Land and water management factors

Land and water management factors that accelerate water infiltration or enhance overland or subsurface lateral flow can transport high amounts of phosphorus with water over a relatively short time period. Examples of management that promote **overland flow** of P with water include summerfallow or intensive tillage, soil compaction by animals or equipment and artificial surface drainage. Examples of management that promote rapid **infiltration** of

P with water include zero-tillage, conservation tillage, perennial forages for hay or pasture where preferential vertical flow channels can form, and subsurface tile drainage.

The impact of various management practices varies substantially from one area to another, even within the same province. For example, in a recent Quebec study, Goulet et al. (2006) found subsurface drainage from tile drains accounted for an average of 98% of the water and 95% of the P movement from study plots, with particulate P comprising an average 84% of the total P recovered from the drainage system. Conversely, in another Quebec study, although subsurface flow accounted for 51.7% of the total runoff volume, less than 40% of the annual P lost was collected from subsurface drainage waters (Jamieson et al. 2003).

In addition, the overall influence of management practices may be masked by landscape and climate factors. For example, Glozier et al. (2006) found nutrient loading during runoff in Manitoba's South Tobacco Creek watershed was more strongly related to the type and nature of the runoff event than land use and fertilizer management within the watershed. Furthermore, management practices, such as land use, crop selection and nutrient application rates, are highly influenced by the nature of the land and climate; so, the "nature" and "nurture" of the land are not truly independent of each other.

2.11 Definitions of animal units for different classes of pigs in Manitoba

Pig Operation Classes	Animal Units Produced by One Pig	Number of Pigs for One Animal Unit
Sows, farrow to finish (110-115 kg)	1.25	0.8
Sows, farrow to weanling (up to 5 kg)	0.25	4.0
Sows, farrow to nursery (23 kg)	0.31	3.2
Weanlings (5-23 kg)	0.03	30
Growers/finishers (23-113 kg)	0.14	7.0
Boars (artificial insemination operations)	0.20	5.0

Source: Farm Practices Guidelines for Pig Producers in Manitoba (2007).

2.12. Formulating diets to reduce N in pig manure

The minimum requirements for key amino acids in pig diets often leads to excessive amounts of non-essential or non-limiting amino acids in feed when N is supplied as crude protein. Excess dietary N is excreted in urine and feces. Formulating pig diets to reduce overall protein use and increase the supply of essential and limiting amino acids such as lysine, methionine, threonine and tryptophan and using feed ingredients that are highly digestible has been shown to reduce N excretion while having no negative effect on pig performance (Walz and Pallauf 2002).

2.13 Bioavailability¹ of phosphorus in select pig feed ingredients

Feedstuff	P Bioavailability %	Feedstuff	P Bioavailability %
Cereal grains		High protein meals-Plant origin	
Corn	14	Sunflower meal	3
Oats	22	Canola meal	16
Barley	30	Soybean meal, dehulled	23
Triticale	46	Soybean meal, low-phytic acid	50
Wheat	40	High protein meals-Animal origin	
Corn, high moisture	53	Meat and bone meal	90
Corn, low-phytic acid	75	Dried skim milk	91
Grain by-products		Blood meal	92
Corn gluten meal	15	Fish meal	93
Wheat bran	29	Dried whey	100
Brewers grains	33		
Cooked cereal fines	40	Inorganic Phosphates	
Wheat middlings	41	Monocalcium phosphate	100
Corn gluten feed	59	Dicalcium phosphate	100
Distillers grains plus solubles	76		
Phytase Addition		Miscellaneous	
Corn-soy diet without phytase	15	Soybean hulls	78
Corn-soy diet with phytase	45	Alfalfa meal	100

¹Bioavailability is expressed relative to the availability of P in monosodium- or mono-calcium phosphate, considered to be 100% bioavailable.

Adapted from Cromwell (2005).

2.14. Phytate and phytase

The majority of P in cereal grain or oilseed meals is organic P as phytate or phytic acid. Phytate is largely indigestible by monogastrics such as pigs, due to the limited presence of phytase in the intestinal tract, the enzyme which breaks down phytic acid (Cromwell 2005). Mineral P supplements are commonly added to pig diets to compensate for this low P availability in feed grains. Feeding above guideline requirements (insurance feeding) and overall low use-ability of P in feed grains lead to high P excretion. Adding phytase to pig feedstuffs improves P digestibility and retention, which in turn reduces the amount of P being excreted and decreases the need for supplementation with mineral P (Akinremi et al. 2007b).

2.15. Economic impact of manure treatment on viability of pig farms

Salvano et al. (2006) demonstrated that Manitoba's new manure P-based regulations will require some livestock operations typical of Manitoba to access a much greater land base in order to apply an amount of manure equivalent to the amount currently applied based on N rates, depending on current source management practices and operation factors. Salvano et al. (2006) conducted multiple scenarios to assess the potential economic impact of the new P-based manure management regulations (as various nutrient management strategies based on the regulations and land accessibility) on different types and sizes of pig operations. When insufficient land was available for manure application, cost per pig increased substantially, as manure needed to be treated (increased cost of \$1.08 to \$3.50/marketed pig) or transported off site (average increased cost of \$1.11/marketed pig). These costs represent a very significant proportion of long-term average returns for pig operations, estimated at \$2.08 per weanling pig and \$6.62 per finishing pig. The scenarios

showed greatest negative impact for farrowing operations while finishing operations had lower management costs when fed phytase-enriched feedstuffs, which decreases P in manure and therefore increases amount of manure loading rate compared to finishers not fed phytase. Thus, adoption of appropriate source-based BMPs to reduce manure P content, such as addition of phytase to pig feed, could reduce the need for additional land or manure treatment in some pig livestock operations.

2.16. Suitability of agricultural land for agricultural production

Agriculture capability classes for mineral soil in agro-Manitoba.

Agriculture Capability Class	Percentage of Agro-Manitoba
1	1.6
2	26.5
3	23.3
4	16.1
5	15.1
6	8.0
7	1.9

Adapted from Manitoba Conservation (2006)

2.17. Ammonia volatilization losses under various management situations

Estimated volatilization losses following manure application.

Method of Application	Cool Wet % loss	Cool Dry % loss	Warm Wet % loss	Warm Dry % loss	Average % loss
Injected	0	0	0	0	0
Incorporated within 1 day	10	15	25	50	25
Incorporated within 2 day	13	19	31	57	30
Incorporated within 3 day	15	22	38	65	35
Incorporated within 4 day	17	26	44	72	40
Incorporated within 5 day	20	30	50	80	45
Not incorporated	40	50	75	90	64
Irrigated	Above+10%	Above+10%	Above+10%	Above+10%	Above+10%
Applied to Standing Crop	25	25	40	50	35

Source: MARC 2005.

2.18. Increased land area required to adapt from N-based to P-based manure rates

The increase in land area required to adapt from an N-based manure application rate to a P-based rate is often greater than most people anticipate for the following reasons (Salvano et al. 2004).

- P-based rates are determined by **concentration of total P** in the manure, not just the portion of P that is immediately available to the crop. However, N-based rates of manure application are based on **available N** in the manure and are estimated at 25% of the manure's organic N and anywhere from 40 to 100% of the ammonium-N content, depending on manure placement and weather and soil conditions at the time of application. Therefore, when manure is applied on an N-basis, the rate must be increased to account for these potential losses or un-availabilities for the current crop year, effectively increasing the total amount of P added to soil.
- P-based rates are determined by **long term average rates of removal by the crop**. However, N-based rates are based on **target** yields which are often significantly higher than long term average yields. Therefore the actual N requirements for a crop in a typical year are often overestimated, effectively increasing the total amount of P added to soil versus applying based on average yields. Any over-applications of manure N should be reflected in subsequent soil tests and result in decreased applications of N in the subsequent year, but the short term effect is to exaggerate the P surplus even further.
- P-based rates are determined by the **rate of nutrient removal for the crop's harvested portion, only**. However, N Manure is applied based on the crop's **total N uptake** requirements, including N lost, immobilized into soil organic matter or used for vegetative growth. Crop removal of nutrients is much lower because only a portion of the plant matter is harvested and removed from the field.

2.19. Water management and erosion control BMPs

Some BMPs that manage water movement and may be suitable for controlling the limited amounts of N and P lost by erosion in Manitoba include:

- **establish actively growing crops early and include fall seeded crops** in rotations to extend the season of crop growth for water and nutrient uptake to include periods of post-harvest fall rains and early spring rains, and to physically intercept rainfall and impede overland flow. Inclusion of a fall seeded crop with high N demand reduces downward nitrate leaching related to early spring growth and therefore earlier removal of nitrates than spring seeded crops (Campbell et al. 1984). Include crops with high water use or high tolerance for wet conditions in areas subject to periodic inundation.
- **maintain permanent vegetative cover**, particularly on lands susceptible to erosion to protect the soil from erosion by reducing rainfall impact and runoff energy. On steeply sloped plots along the escarpment area of Manitoba, Hargrave and Shaykewich (1997) found N and P runoff losses from alfalfa plots were negligible, attributable to the absence of soil loss compared with wheat, corn or fallow plots over a 3 year period.
- **retain surface residues** to protect soil from erosion by reducing rainfall impact and runoff energy.

- **use cover crops or mulches** to increase water and nutrient uptake and to protect soil from erosion by reducing rainfall impact and runoff energy. However, although cover crops reduce loss of nutrients with erosion runoff, they do not reduce dissolved P losses (Sharpley and Smith 1991), the dominant form of P loss in the Prairies. Furthermore, large quantities of N and P can be released from vegetative residues when thawed after freezing (Flaten et al. 2005). For example, Miller et al. (1994) found release of N from frozen/thawed cover crops, measured as nitrate and ammonium N, represented between 5 and 10% of the total biomass N while dissolved P release accounted for 20-30% of the total biomass P. These authors found the potential leaching losses of ammonium ($1\text{-}3\text{ mg nitrate-N L}^{-1}$) and nitrate ($4\text{-}11\text{ mg ammonium-N L}^{-1}$) from the cover crops were in the same range as ammonium-N and nitrate-N in runoff from manured (non-incorporated) fields measured in an earlier study (Spires and Miller 1978). The dissolved P concentration in runoff measured by Spires and Miller (1978) of 0.70 mg L^{-1} from manured fields were many times lower than the potential P concentration in runoff from the cover crops ($1\text{-}16\text{ mg L}^{-1}$) examined by Miller et al. (1994). As with other vegetative residues, mulches and composts can reduce runoff volume and erosion losses, but may increase soluble and total nutrient losses in some situations, depending on compost nutrient composition and some physical and chemical properties (Faucette et al. 2004).
- **establish vegetated barrier strips or conservation buffers** along waterbodies or in flow channels to intercept and trap mobilized N and P associated with sediment (Dosskey 2001; Dillaha et al. 1987; Uusi-Kämppe et al. 2000) and to increase water and nutrient uptake. However, over time, as nutrients accumulate, the trapping efficiency of the VBSs may decrease, thereby becoming a source of nutrients in runoff water passing through the VBS (Dillaha et al. 1989). While this practice has been effective for reducing particulate P loss in areas where erosion loss is a major loss pathway, the capacity of vegetated strips to reduce total P loss from landscapes in Manitoba is not substantial (Sheppard et al. 2006). Overall, there is a general lack of understanding of optimal buffer strip design and management and long-term effectiveness, although it is known that P trapped in the vegetation can accumulate with time beyond the retention capacity of the soil (Lovell and Sullivan 2006; Dorioz et al. 2006). As this vegetation can be a source of N following freezing and thawing (Flaten et al. 2005), the effectiveness of these strips for reducing N loss, much like for reducing P loss, may be limited and has not been adequately evaluated under Manitoba conditions. Also, during the major period of runoff loss (spring snowmelt) the vegetation is dormant and therefore not actively removing nutrients. Uusi-Kämppe et al. (2000) recommended periodic harvesting of vegetation in buffer strips to reduce losses of dissolved reactive P.
- **use controlled surface or subsurface soil drainage** to manage water removal, reduce erosion and remove sediment.
- **restore or construct wetlands** as wetlands can retain N and P (overland or subsurface) from agricultural lands, but the capacity is limited and variable (White et al. 2000; Kovacic et al. 2000; Mitsch et al. 2000). For example, a portion of particulate P can settle over time to the bottom of the wetland basin. However, if over the long term high levels of particulate P accumulate in the wetland, excess sediment P may need to be removed from the wetland by other measures

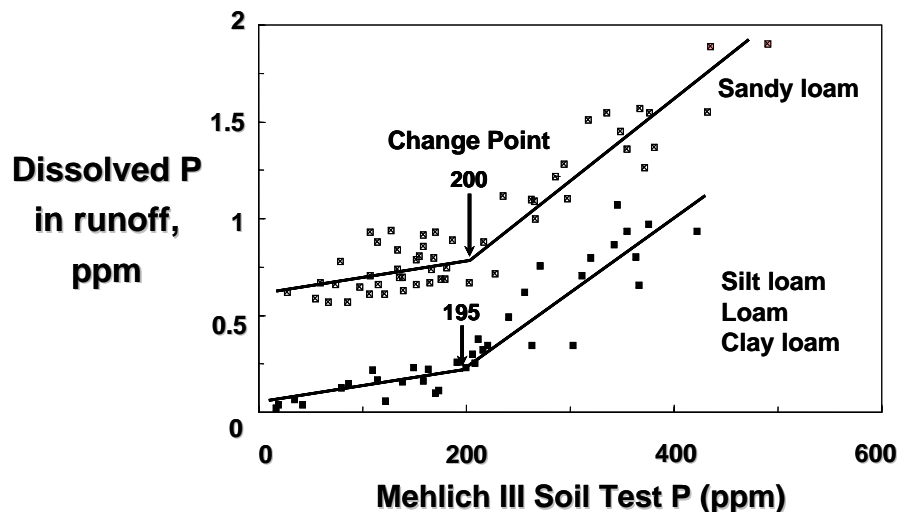
(Pierzynski et al. 2005) to maintain the effectiveness of the wetland as a net P sink.

- **reduce tillage frequency** to increase water infiltration and limit generation of loose soil (smaller sized aggregates) susceptible to erosion loss. While reduced tillage may reduce erosion loss of particulate P relative to conventional tillage, the higher abundance of surface crop residues in reduced tillage fields may result in greater release of dissolved P (Sharpley and Smith 1994), particularly with spring snowmelt.
- **eliminate summerfallow or bare fields from rotations** as water and nutrients may accumulate in the absence of crops to utilize water and bare tilled soil is more susceptible to erosion losses.

As mentioned previously, retaining vegetated areas can reduce P and N loss during the growing season but can also contribute to N, and particularly P loss in a variety of ways, particularly during spring snowmelt (Schellinger and Clausen 1992; Uusi-Kämpä et al. 2000). Therefore, more field research is needed on the effectiveness of setbacks and vegetative barriers before these can be regarded as effective for reducing nutrient loss under Prairie conditions.

2.20 Breakpoint or change point behaviour for soil test P and runoff P

No soil has an infinite capacity to hold P. Therefore, from a theoretical perspective, when the loading of phosphorus fully saturates the soil's retention capacity, the concentration of P in runoff from that soil should increase sharply, at the "breakpoint" or "change point" concentration of P in the soil (see figure below, adapted from Sharpley et al. 2001a). Using the "bucket" analogy from Figure 2.2, when the soil's capacity to retain P is full, the "bucket" starts to overflow and release large quantities of excess P.

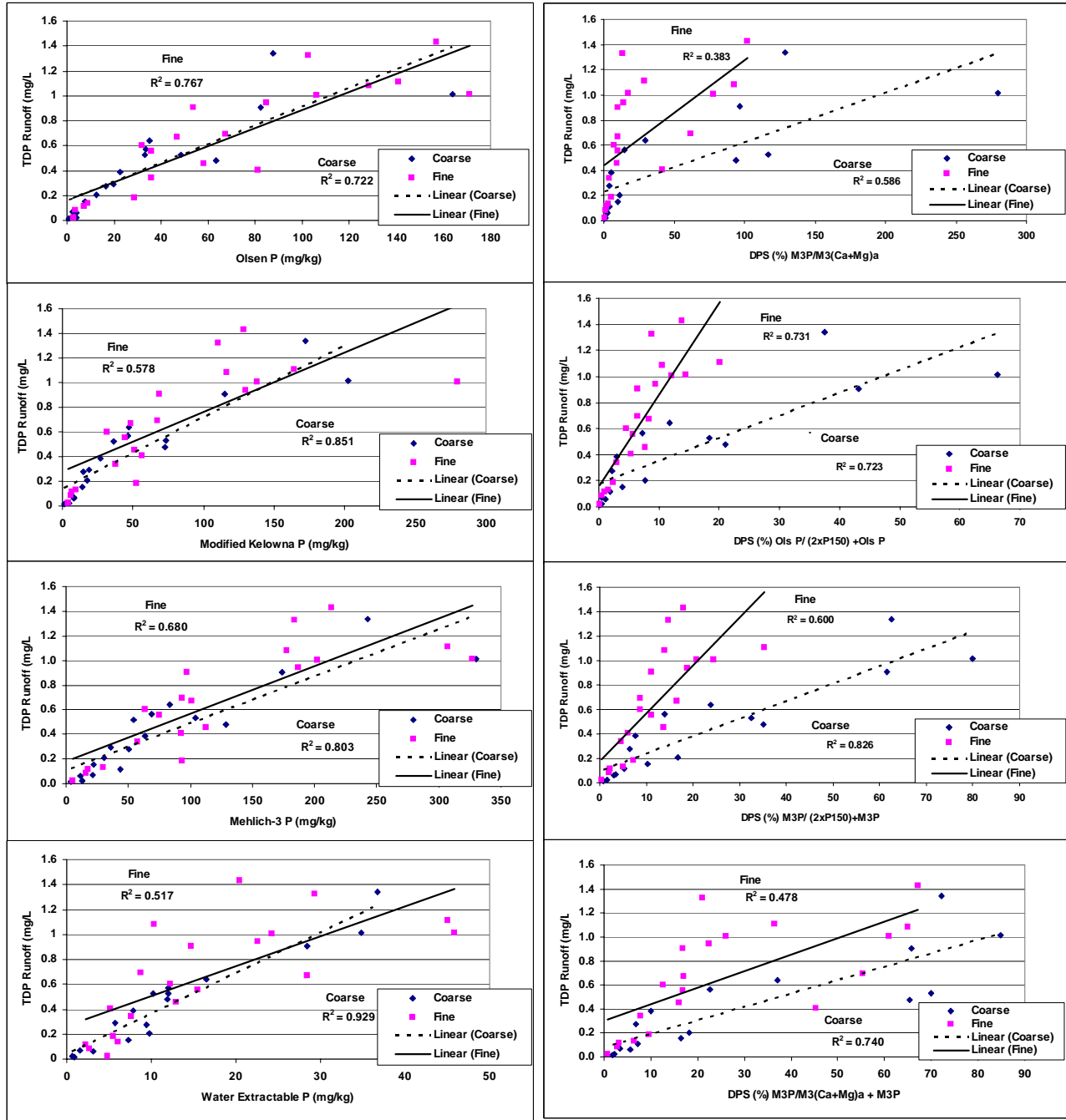


However, two important precautions should be considered when attempting to use change points for specific purposes, such as establishing regulatory thresholds:

- 1) change points are not always observed in P research, because the soil's capacity to retain P may not be as "leakproof" as a "bucket" ... in other words, sometimes the bucket is large and leaky and does not produce an obvious "overflow" at a certain concentration of P. For example, Figures 2.8 and 2.9 in the main text do not show any sign of "change point" behaviour for the relationship between soil test P and runoff P in Manitoba and Alberta soils, respectively.
- 2) change points may not occur at concentrations of P in runoff water that are suitable aquatic environmental thresholds. For example, some soils may not show change points until P concentrations in runoff are well above eutrophication thresholds while others may show change points at runoff P concentrations that are well below levels that present a major environmental concern.

2.21 Methods for estimating degree of P saturation for Manitoba soils

Methods for estimating the degree of P saturation (DPS) are well-developed for acid soils, but are not well developed for neutral to calcareous soils such as most of the soils in Manitoba. In laboratory studies, Akinremi et al. (2007a) tested DPS equations developed for Manitoba soils which incorporate the soil parameters most strongly correlated to P loss from Manitoba soils (Ige et al. 2005a). Using highly variable manured and non manured soil samples, they found the P sorption capacity to be best predicted for a DPS equation using Mehlich III-extractable Ca and Mg and oxalate-extractable Al, accounting for existing STP. Akinremi et al. (2007a) presented an overall critical P saturation limit of 20% (threshold limit) and an average Olsen-extractable P concentration of 64 mg kg⁻¹ for Manitoba soils based on breakpoint degree of P saturation above which loss of water-extractable P from soil becomes very high. However, these preliminary DPS equations developed for Manitoba soils have not been validated under field conditions and show need for further refinement according to recent simulated runoff studies. For example, in subsequent laboratory runoff studies at the University of Manitoba, runoff losses of dissolved P showed a stronger correlation between runoff P and STP (Olsen > Mehlich 3 > Kelowna > water) than with the estimated DPS, using the equations developed by Ige et al. (2005) and Akinremi et al. (2007b), when conducted with 40 Manitoba soils representing a range of soil textures (see figure below, from Sawka et al. 2007b). In addition, distinct textural differences were observed for the relationships of the various DPS equations with runoff dissolved P but were minimal for various STP method relationships with runoff dissolved P (Sawka 2007ab). Therefore, the Olsen soil test (the test currently used by Manitoba Conservation for the manure P management thresholds) was the strongest predictor of Total Dissolved P (TDP) losses across a broad range of soils, especially fine-textured soils where the risk of runoff and P loss is greatest. Several DPS methods show promise for predicting P losses within textural groups in Manitoba soils, but need further refinement for use with neutral to alkaline soils across a range of soil textures.



Linear regression for soils grouped by texture for Olsen P, Water, Modified Kelowna and Mehlich-3 P as well as Mehlich-3 P/M3(Ca+Mg) α , Olsen P/ (2xP150) + Ols P, Mehlich-3 P/(2xP150) +M3P, and Mehlich-3 P/M3(Ca+Mg) α + M3P with TDP for the first 30 minutes of runoff over a common STP range of 0-200 mg kg⁻¹ Olsen P (Sawka et al. 2007b).

CHAPTER 3 Water Use by Manitoba Pig Operations

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3.1 Executive Summary

In modern swine operations water is mainly used for drinking by the animals, cleaning the barns, enforcing dunging habit, sanitizing equipment and in liquid manure systems, for flushing or moving the manure to the storage site. An accurate account of water use in hog operations is critical for determining the requirements for wells and/or reservoirs serving the barn, the size of the manure storage system and the land base required for effluent disposal. From this review, it is clear that little information is available on the partitioning of total water usage into various components (i.e. drinking, washing, waste movement, etc.) within a modern hog operation in Manitoba. Moreover, available estimates are too variable to be relied upon. Various technologies for conserving water in hog production have been developed but the extent to which these are used in Manitoba is unclear. The lack of information on water usage and wastage would hamper efforts to focus and prioritize water conservation practices in Manitoba hog operations. Nonetheless, it is critical that policies developed regarding water usage and conservation in modern hog operations be based on accurate data on actual water usage and wastages. This chapter provides an overview on water usage and wastage in hog operations and opportunities for conservation. Also, it highlights information gaps which need to be addressed and offers recommendations to ensure sustainability of water use in modern hog operations in Manitoba.

3.2 Introduction

In swine operations water is used for drinking by the animals, for cleaning the barns, sanitizing equipment and in liquid manure systems, for flushing or moving the manure to the storage site. A small proportion of water usage is also for domestic use which includes human usage associated with barn operation.

3.3 Biological functions of water

Water is the nutrient that is required in the largest quantity by swine, and fulfills a number of physiological functions necessary for life. It is a major structural element giving form to the body through cell turgidity. Water is an important transport medium for hormones and other chemical messengers, nutrients, waste material, as well as ingested food in the gastrointestinal tract. Water has a major function in regulating body temperature, acts as a lubricant for skeletal joints and is a component of many basic chemical reactions. Without water, death can occur within a matter of days “water is life”. Pigs obtain water from three sources: water contained in the feed, metabolic water and water consumed by drinking. Pigs can successfully tolerate a wide range of water quality and will adapt to water which contains compounds that impart a negative effect on performance. Above all else, pigs need sufficient quantities of water to maintain optimal production levels.

3.4 Water usage by Manitoba hog operations

An accurate account of water use in hog operations is critical for determining the requirements for wells and/or reservoirs serving the barn, the size of the manure storage system and the land base required for effluent disposal. However, there is a dearth of information on the partitioning of total water usage into various components within an operation. Most published data on water usage deal with the amount used for animal drinking only and not the amounts used for washing, cooling, and other functions within a fully operational, modern production unit. This serves to invalidate existing figures and leads to unnecessary speculation about actual total water use, particularly for large operations being scrutinized by the public (Froese, 2003). Importantly, this lack of information hampers efforts to focus and prioritize water conservation practices. An attempt to estimate water usage by Manitoba hog operations is a daunting task given the lack of sufficient information. In this context, the following sections give background information on drinking water requirements by various classes of swine and some indication of water usage for other functions in an operation. All together this information forms the basis for estimating total water usage by Manitoba hog operations.

3.4.1 Water requirements by various classes of swine

Although water is an important nutrient in pork production, there is little or no well conducted scientific studies to establish its requirements for different classes of pigs. In part, this is because water consumption can vary considerably depending on physiological and environmental conditions including: age and production stage of the animals, temperature, humidity and activity level, as well as the water and nutrient content of the feed. Thus, available estimates of water requirements for pigs are quite variable (Table 1). The following sub-sections outline the water usage by various classes of swine.

Suckling pigs

It is a common assumption that suckling pigs do not drink water and can completely satisfy their water requirements by drinking milk because milk contains 80% water. However, as reviewed by Thacker (2001) and Nyachoti (2004), water consumption by suckling pigs is critical for optimal performance and therefore it is important to ensure that they have access to good quality drinking water. Water consumption at this stage is closely related to milk consumption, effective environmental conditions in the creep area, and creep feed intake. High water intake may encourage creep feed consumption, but this may negatively impact on milk intake levels (Mroz et al., 1995; Thacker, 2001). In general, suckling pigs should be provided water for *ad libitum* intake. This is particularly critical if milk intake is limited, in which case water intake may help prevent dehydration and increase survival rate of piglets with low milk intake (Fraser et al., 1988).

Weanling pigs

The importance of water intake as a factor affecting the performance of weaned pigs cannot be overstated (Nyachoti et al., 2005). During the first few days following weaning, water intake is reduced as piglets learn to seek and drink water (Nyachoti, 2004). This is undesirable as it might compromise the process of digestion and absorption thus leading to increased incidences of diarrhea (Stockill, 1990). However, because feed intake is also low soon after weaning, piglets tend to increase water intake so as to achieve gut fill and thus the feeling of being satisfied (McLeese et al., 1992). Weaned pigs should be encouraged to drink because this is an important factor determining feed intake levels (Brooks et al., 1984).

Growing-finishing pigs

Water consumption by growing-finishing pigs generally has a positive relationship with feed intake and body weight (Thacker, 2001). Water intake in growing finishing pigs is essential for lean muscle growth as lean meat is 72% water (Kober, 1993). The amount of water consumed per day by a growing finishing pig will depend largely on the feeding program. If pigs are allowed *ad libitum* feed intake their water consumption will be around 2.5 kg per kg of feed whereas pigs with restricted feed intake may consume up to 3.7 kg of water per kg of feed (NRC, 1998). The variation in water intake is most likely due to consuming water to feel satiated, although, it may also reflect increased water usage due to boredom. The water should therefore be available *ad libitum* for pigs given *ad libitum* access to feed. The pigs receiving restricted feed intake should also have access to water *ad libitum* as their welfare may be impaired with restricted water intake.

Gestating Sows

In addition to satisfying physiological needs, water consumption by gestating sows is also influenced by behavioral characteristics. Because gestating sows are limit fed, they may consume additional water in order to feel satiated. Also, individually housed gestating sows may experience some degree of boredom, which they often try to offset by excessive drinking. Despite these problems, it is recommended to provide gestating sows with water for *ad libitum* intake as this may play an important role in fulfilling their welfare requirements (Mroz et al., 1995).

Lactating Sows

Daily water consumption by lactating sows provided with free access to drinking water varies widely among individual sows. Consequently, it has been recommended that lactating sows should be allowed between 15 and 20 liters per day of drinking water, depending on size and milk production levels (NRC, 1998; Fraser et al., 1990). Adequate water intake is required for optimal milk production, which in turn impacts on litter performance. In addition to milk production levels, lactating sow water intake is influenced by dietary factors such as presentation (pellets, mash), protein levels and salt content (e.g. Seynaeve et al., 1996), as well as environmental temperature. From a practical pork production stand point, excessive water consumption by sows is an important concern as it relates to the amount of urine production and the associated environmental challenges.

Water consumption estimates from several sources for different classes of pigs are shown in Table 3.1.

Table 3.1 Daily water consumption (litres/head) of different classes of pigs¹

Production Stage	Manitoba ¹	Prairie Swine Centre ²	North Carolina ³	The Netherlands ⁴	NRC ⁵
Breeding/Gestation	15.7	15.0	26.0	10.0	15.8
Farrowing	37.4	20.0	32.0	N/A	26
Nursery	3.4	3.0	3.0	1.4	1.0
Grow/Finish ⁶	7.7	7.0	17.0	4.6	6.4

¹Small (2001)

²Pork Production Reference Guide (2000).

³Swine News (1999).

⁴The Dutch water consumption, research institute for pig husbandry (1999).

⁵Source: Nyachoti (2004).

⁶*ad libitum* feeding.

3.4.2 Other water uses in swine operations

Of the total water usage in modern swine operations 10-15% has been estimated to be used for cooling the animals and 5-10% for washing purposes. However, it is not clear whether these estimates are truly representative of the majority of swine operations. They do not appear to fully account for water used to move manure from the barns to the storage site.

Cooling Pigs

Water is used for evaporative cooling of lactating sows and growing/finishing pigs during warm weather and for reinforcement of dunging habits in partially-slatted grow/finish pens. Evaporative cooling nozzles for lactating sows deliver about 2.3 litres of water per hour (Small, 2001), but may only account for 0.3 liters/sow/day (Froese, 2003). Cooling of growing/finishing pigs is achieved by placement of a separate water line over the slatted area of a row of pens. The spray nozzles on this line are designed to deliver 0.90 litres/minute/nozzle which has been estimated to account for between 8.1 and 37.1 liters/sow/day (Froese, 2003). Both systems are temperature activated with an adjustable activation point, commonly set at 25°C for lactating sows and 20°C for growing/finishing pigs. Whereas the sow dripper system operates fully on or off, the grow/finish sprinkler system can be programmed to operate intermittently with an adjustable cycle frequency and length, to reinforce proper dunging habits. This likely contributes to the large range of water usage for grower/finisher sprinklers since in some operations they have become a management tool in addition to their use for evaporative cooling.

Washing Purposes

Washing water includes water used for all aspects of washing pens, floors, sow crates and feeders. According to Dr. Phil Wilson of the VIDO Swine Technical Group, a considerable amount of water is used to pre-soak or soak pens prior to pressure washing. Most commonly in Manitoba production systems, with totally or partially slatted floors, animal areas are completely washed down and disinfected between groups of animals. However, similar to estimates of water used for cooling, little information is available on water requirements for washing. The VIDO Swine Technical Group (1998) conducted a survey of Western Canada swine farms and reported the following water usages for washing (Table 3.2).

Table 3.2 Washing water usage in swine operations in Western Canada

Area	Average	Range
Farrowing	152 L/crate/wash *	85-318
Nursery	12 L/pig place/wash	6-26
Finishing	80 L/pig place/wash	21-246

Source: VIDO Swine Technical Group (1998).

* can also be expressed as 18.7 L/pig sold

Extrapolated to an annual, whole-herd basis, these averages would result in usages of 1.0 L/sow/day for farrowing, 0.2 L/pig/day for nursery, and 0.66 L/pig/day for growing/finishing (Small, 2001). In other terms, considering the 3 production stages (farrowing, nursery, finishing) use an average of 110 L of water for washing for every pig sold, two thirds of this (~80 L) is used in the grow-finish area and less than 10% is used in the nursery (VIDO, 1998)

An important observation from these estimates is the extreme variability, which makes it difficult to determine accurately the amount of water used for washing purposes in commercial hog production. Nonetheless, it makes it clear that wash water can have a big impact on the total amount of slurry produced in a farm. Thus, there is a need for further research to more accurately determine the amount of water that is used for this purpose in commercial hog production.

Domestic Use

Domestic water usage is the water used for human consumption, laundry, showering and hand washing, and other cleaning activities related to the barn office area. Domestic consumption can account for approximately one percent of the total water usage depending on the size of the operation (Small, 2001).

Manure Management

Water is used to efficiently move waste in livestock facilities (MAFRI, 2007). However, there is lack of quantitative data on the amount of water used to move waste in swine operations. Most current swine operations use a liquid manure handling system in which manure and waste water collected from the barn are subsequently moved into an earthen manure storage facility outside the barn. Various systems are used to move the slurry from the barn into the outside storage facilities, including the use of flush tanks or simply pulling a plug and the manure flowing by gravity to the earthen storage. Both systems require large amounts of water (about 800 to 1,600 gallons every six hours for the flush tank system) to facilitate the flow of the slurry. The flush tank system is described in detail in U.S. Pat. No. 4,913,095, "Flushing System for Hog Houses", issued to Morrow et al. Apr. 3, 1990.

Manure production rate is an important consideration in establishing swine operations. As alluded to earlier, a greater percentage of manure in conventional slurry systems is water from urine, spillages, leaks, washing and water used specifically to move the waste into the lagoon. Arguably, therefore, the waste production rate should provide an indication of the amount of water used to move waste. Wastage production rates are generally listed in provincial farm practices guidelines or codes for hog producers. Accordingly, they are used in key calculations when planning and siting operations by both producers and regulatory bodies. The waste production rates used for Manitoba along with those for Saskatchewan and Prince Edward Island are shown in Table 3.3. Clearly, there are considerable differences in these estimates, especially between values for Manitoba and those for Saskatchewan and PEI. These may point to differences in the methodologies used to generate these values. None-the-less, efforts to ascertain the 'true' estimate total as well as for waste water production in swine operations will have a distinct benefit for water and manure management strategies in swine production and in setting regulatory guidelines.

Table 3.3 Values for total daily waste production by stage of production and comparison to provincial guidelines.

Production Stage	MB ¹	SK ²	PEI ³
Gestation (L/sow/day)	7.6	15.9	15.9
Farrowing (L/sow/day)	14.2	21.8	21.8
Nursery (L/pig/day)	1.1	1.6	2.4
Grow/Finish (L/pig/day)	5.4	8.5	9.3

¹Farm practices guidelines for hog producers in Manitoba (2007).

²Manual for: Developing a manure and dead animal management plan, Saskatchewan Agriculture and Food (1997).

³Guidelines for manure management for Prince Edward Island (1999).

3.4.3 Estimate of water usage in Manitoba hog operations

Generally, there is very little information on total water usage by swine operations in Manitoba. Unfortunately, most of the published figures relate only to water used for drinking and the amount wasted but not the amount of water used for other functions. Small (2001), did a survey of 9 hog operations in an effort to apportion total water usage by function and by production stage within modern swine operations in Manitoba. Nine hog operations equally representing small (<500 sows), medium (501 to 1000 sows), and large (> 1000 sows) size operations were selected as survey participants. All production stages (gestation, farrowing, nursery, and grow/finish) within each operation were monitored for water usage for animal drinking, animal cooling, and washing. In addition, domestic usage was monitored as a separate variable. Alberta Agriculture (2000) has also published values for total water usage for different types of swine operations, and a comparison of those values to those obtained by the Small (2001) survey are presented in Table 3.4.

Table 3.4 Total daily water usage (litres/head) by type of operation in Manitoba and Alberta¹.

Operation type	Manitoba ²	Alberta ³	Average
Farrow – Finish	89.5	91	90.25
Farrow – 20 kg	31.6	30	30.8
Farrow – wean	21.1	25	23.05
Nursery	3.8	2	2.9
Grow/Finish	11.7	7	9.35

¹Total of water required for drinking, cooling, washing and domestic use.

²Source: Small (2001).

³Source: Alberta Agriculture (2000).

Extrapolated to pig inventories in Manitoba as of April 1, 2007, total daily water usage in Manitoba swine operations can amount to ~30 million litres (Table 3.5). However, it should be noted that the amount of water used for moving waste is obviously not included in these calculations, thus indicating that much more water than this can be used routinely in pig production. Examination of tables 3.1 to 3.4 reveals considerable variation in estimates of water requirements and use. While these estimates provide some values to work with, the need remains for more detailed studies to clearly define water usage in Manitoba hog operations, and ways to effectively conserve this resource.

Table 3.5 Total water usage (litres/day) by pig inventories in Manitoba as of April 1, 2007

Pig type	Number of pigs ¹	Water usage ²
Breeding stock	372,700	9,895,185
Nursery	1,110,800	4,221,040
Grow-Finish	1,436,500	16,807,050
Total	2,920,000	30,923,275

¹Statistics Canada (2007)

²Product of average total daily water usage (Table 3.4) and number of pigs. The amounts used by gestating and lactating sows were averaged (from Table 3.1) for this calculation.

3.5 Sources of water for hog operations

Adequate supplies of good quality water are critical for intensive, modern pig production. Currently, the majority of pigs on the Prairies receive ground water, but a growing number are being raised in areas where good quality ground water is unavailable, and surface (dugout) water must be used (Nyachoti et al., 2005). Although pigs can tolerate a wide range of water quality, there is no information indicating use of recycled water for drinking purposes. Recycled water from lagoons is used to some degree to move waste in lagoon systems. There is lack of information on the proportions of the various sources (i.e. wells water, pipeline water, dugouts) of water used in Manitoba hog operations.

3.6 Conservation actions and options

3.6.1 Water wastage in swine operations

Identifying areas of water wastage in an operation is essential in order to evaluate and implement measures for conservation. On an industry-wide basis this is another area in which good numbers are not readily available. Some potential causes of water wastage from drinkers, as reviewed by Thacker (1998), include inappropriate flow rates, inappropriate drinker height, poor drinker design, failure to use drinker wings, poor location of drinker, improper drinker angle, inadequate maintenance and leaking water pipe. Water wastage has been documented to exceed 40% of the water provided by a nipple drinker. Excess water spillage through leaking bowls and poorly maintained nipple drinkers leads to additional manure volume and to handling costs.

Another source of wastage is that used by the pig beyond its physiological needs. Many factors, such as the environment, diet constituents, feed allowance, boredom, age, season of the year and the stage of the breeding cycle can influence the water needs of the pig. Pigs also drink or play with the waterers to alleviate feelings of hunger and/or boredom. Such feelings can increase the water usage (intake plus wastage) several fold over requirements. Water consumed above requirement is eliminated as urine and thereby also increases the volume of slurry produced.

Typical swine manure contains about 93% water. Estimates from the VIDO Swine Technical Group shows that a 5000 head finishing barn would need an additional 5.5 million litres of annual storage capacity just for the wasted drinking water.

3.6.2 Opportunities for conserving water in swine operations

It will be important to design future production systems to reduce water input into the waste stream. This will serve to lower the total farm water requirements and reduce requirement for removal of water in order to achieve efficient concentration of nutrients. The following subsections outline approaches which would be useful in conserving water in swine operations.

Diet Composition

Various dietary factors including dry matter content and concentrations of salt, crude protein and fibre levels are known to influence water intake in pigs. A study by Greary et al. (1996) suggests that when weaner pigs are fed diets with low dry matter content their water consumption increases dramatically. Water intake of piglets fed a low dry matter diet was almost 1000 ml/pig/day more than for those piglets consuming a high dry matter diet. Similarly, McLeese et al. (1992) reported that during the first week post-weaning, when dry matter intake is very low, piglets tend to have high water intake. Furthermore, Yang et al. (1984) reported excessive water drinking in growing-finishing pigs consuming less than 30 g of dry matter per kg body weight. Results of these studies can be explained by the fact that pigs consume excessive water to feel satiated when feed (dry matter) intake is low.

The form in which feed is offered may also influence the amount of water consumed by pigs. Although not consistently observed in all three trials performed by Laitat et al. (1999), there was some suggestion that pigs fed mash diets have a higher need for water than those fed pelleted diets.

As well, the crude protein and mineral content in swine diets have major impacts on water intake levels (Thalin and Brumm, 1991; Mroz et al., 1995; NRC, 1998). When pigs are fed diets with a protein concentration that exceeds their requirements for maintenance and growth or production purposes, the excess protein is broken down and excreted as urea (NRC, 1998). This process exerts an additional need for water to help in the excretion of the excess nitrogen, which explains why pigs consuming high protein diets have high water intake levels (NRC, 1998; Figure 1). Lower protein diets, formulated to maintain the essential amino acids and therefore pig performance, may lower nitrogen excretion by 40% (Smith and Crabtree, 2005). As well, the accompanying decrease in water intake reportedly can decrease overall slurry volume by approximately 30% (Smith and Crabtree, 2005).

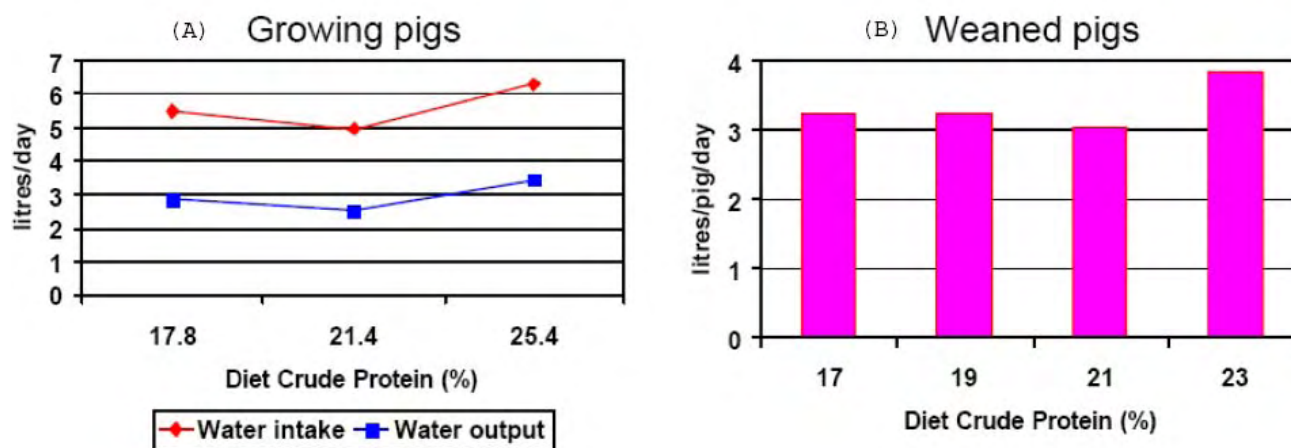


Figure 3.1 Effect of protein level on water intake in growing (panel A) and weaned (panel B) pigs (Adopted from Nyachoti, 2004).

From the data summarized by Mroz et al. (1995) and the study by Seynaeve et al. (1996), it is evident that high salt (NaCl) intake also results in increased water consumption in all classes of pigs and that this is associated with increased urine output. Again, this is undesirable as it adds to the challenges associated with manure handling and disposal in the swine industry. Overall, poor diet formulation can increase water usage unnecessarily, as well as impact on manure nutrient levels, and thus care should be taken to provide appropriately formulated rations, and in particular, to avoid excessive dietary salt and protein levels.

While not directly related to water intake, manure phosphorus levels, also of environmental concern, may also be decreased by 25 to 50% with addition of phytase enzyme to the rations (Maguire et al., 2005).

Drinker Design Technologies

The type of pig drinker will affect water wastage. Gill and Barber (1990) compared four types of drinkers and observed a 40% difference in water usage, with nipple drinkers wasting more water

than bite drinkers. These authors proposed that mounting a metal flange (wing) on both sides of the drinker so that a sideways approach cannot be made was useful in reducing water wastage. In a study conducted later, a 55% reduction in wastage was recorded using drinkers with wings (Gadd, 1992). There are no on-farm studies reported for Manitoba comparing the effectiveness of drinkers in controlling water wastage in hog barns. However, at the 2007 Banff Pork Seminar, it was reported that on an Alberta farm, ball-bite nipples reduced the amount of water used for drinking purposes by growing-finishing pigs by up to 46% compared with the standard nipple drinkers (Green Matters, 2006). As well, utilization of a nipple drinker inside the feeder, as incorporated into wet/dry feeder systems are common in some areas and are considered to decrease water wastage (Froese and Yaceniuk, 1990). These observations clearly show that technologies do exist to effectively manage water usage in pork production. However, it is not known how many farms are using these technologies in Manitoba.

The position of the drinker also affects waste. Positioning a drinker 10 to 15 cm above the pigs' backline wastes the least amount of water by getting the pigs' head up (Gadd, 1988a). If a drinker is set too low, the pig turns sideways to drink and up to 60% of water flows out the other side of the mouth (Gadd, 1988b). Drinkers installed with height adjustment flexibility as the pigs grow could contribute greatly to reduced water wastage.

The flow rate is also important. An excessive flow rate of 900 ml/min compared with a more conventional rate of 300ml/min with pigs 30 to 60 kg produces an extra 78 litres of slurry per pig over 40 days (Gadd, 1988a). Equally, leaking or poorly maintained drinkers will increase water usage. Overall, new technologies, especially design of drinkers, and appropriate management have potential to conserve water and lower the volume of manure that must be stored.

Animal Management

Factors such as boredom, season of the year and the stage of the breeding cycle can influence the water needs of the pig and therefore potential wastage. For instance, group housing gestating sows rather than confining them in individual crates may reduce boredom which has been associated with excessive intake of water. Additionally, such factors as stocking densities in hot weather will affect the environmental temperature felt by the pigs and influence their need for water for cooling purposes.

Recovery of Wastewater and Recycling

Wastewater can be reclaimed for reuse purposes when conventional treatment is combined with advanced treatment technologies. However, according to a report compiled by Premium Standard Farms, there are three major reasons why water reclamation for reuse purposes is not widely practiced; 1) perception associated with direct or indirect reuse for human consumption; 2) the cost of advanced treatment necessary for wastewater reclamation, and 3) health concerns with the consumption of reclaimed water. Animal health concerns may be the most difficult challenge to overcome. If all of the wastewater is reused, the concentrations of dissolved ions, or total dissolved solids (TDS) will increase in concentration in the recycle loop over time. If allowed to build to levels higher than can be tolerated by the livestock, adverse health effects could occur.

Premium Standard Farms has conducted several wastewater reclamation demonstration projects using a conceptual flow diagram for a water reuse system as shown in Figure 3.2.

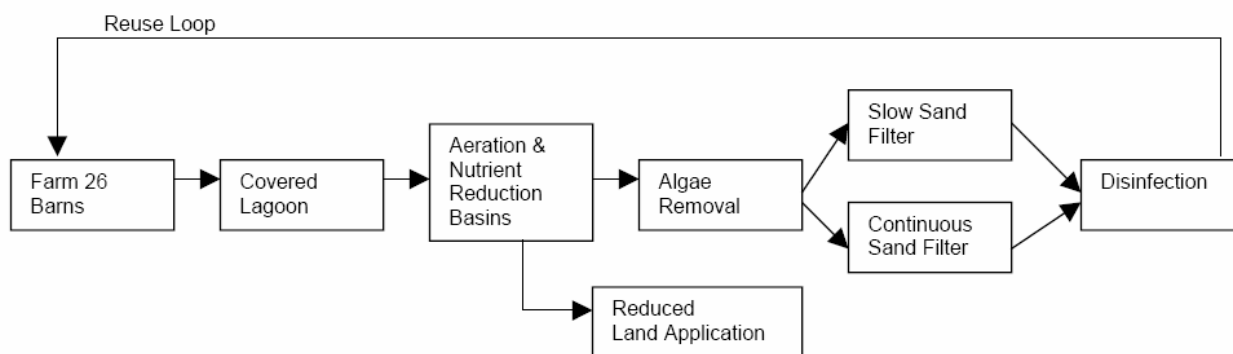


Figure 2 – Water reuse alternatives

Using this model several preliminary studies have been conducted under experimental conditions by Premium Standard Farms to determine the feasibility of recovering water from animal waste, removing nutrients, coliform/pathogens, and reusing the recovered water for drinking by pigs. In those studies there was no evidence of any performance reduction or other adverse animal responses to the inclusion of a significant portion of the drinking water as recycled water from animal waste (Bull et al., 2005).

Other studies conducted or ongoing at the University of North Carolina have evaluated similar and other technologies for recycling waste water. These include collecting water from an aerobic digestion pond, removing solids and then filtering through sand or membrane filtration followed by chlorination. While such methods show definite promise, a number of challenges need to be overcome and the cost effectiveness has yet to be confirmed. In Manitoba, limited efforts towards recycling of waste water have been made, and mostly directed towards recycling the water for washing purposes rather than for animal consumption. Similarly, in Ontario, commercialization of a waste water recycling technology for washing purposes is being attempted (Nutrient Management technologies Ltd website), but again, it is directed towards very large hog operations and the economic feasibility is questionable.

Moving Waste with Minimal or no Water Addition

A strategy to minimize water use in hog operations is to reduce the amount of water needed to move the manure from the barn to storage facilities. To achieve this goal, various technologies have been tested but it is not clear whether these are currently used, other than experimentally, in any system within Manitoba or anywhere else in Canada. An example of these technologies is a system demonstrated by research at North Carolina University. The basic principle underlying this technology is that urine is separated from feces and that the feces is swept away from the barn using conveyor belts underneath the slatted floors. Thus, the amount of water required to move the waste is drastically reduced and because of the drier nature of the manure, nitrogen loss through volatilization is reduced and the cost of transporting the manure is also reduced. There may be other examples of these technologies but as indicated their use in the Manitoba swine industry is unclear.

Solid Manure Animal Housing Systems

The largest majority of swine operations in Manitoba, as in the rest of North America, are based on slurry manure systems. Solid manure systems are used primarily by smaller operations and/or those looking for alternative systems. The greatest advantage of solid manure systems from a water usage standpoint would be the elimination of the water requirement for flushing or moving the manure. There may be some decrease in wash water, but in any facility

the need to thoroughly wash and disinfect between batches of pigs would still be required. There are a number of solid manure system concepts available, most of which have the pigs directly on bedding, which is seen to require more labor and more space per pig to maintain properly. One option studied in the United States evaluated a two-story hog building whereby the animal excrement falls through the slotted floor where it is collected onto a dry bulking agent (e.g. sawdust, straw, shredded newspaper), stored and treated before removal (Mescher et al. 1999). This unique design, which incorporates traditional confinement production practices including slotted flooring, incorporates manure composting within the facility and was considered most appropriate for areas with limited local land base for manure application. While the concept has many positive attributes, it has not seen wide adoption by the swine industry. There are likely several reasons for this; economic viability being one deterrent not fully assessed. In general, adopting solid manure technologies for most current production systems would be cost prohibitive. Although there may be a place for such technologies, with the right available resources, for new construction there is minimal research to support the viability and management strategies necessary for such enterprises to be economically viable.

A Case Study for Potential of Water Conservation in Manitoba Hog Operations

Froese (2003) reports a study that identified a number of areas where significant water wastage, and hence excess manure production, was occurring. These mainly focused on management practices for cooling and watering (e.g. drinker design) in the grower/finisher and gestating sow herds. Based on these observations, a cumulative reduction of 50% of current usage was identified as potentially achievable. However, the values used to obtain these estimates do not appear to be consistent with other estimates of water usage

3.7 Relevant water use regulations in other jurisdictions

In general, water regulatory programs in Canada, United States and European Union consist of both abatement and enforcement programs. The abatement component involves working cooperatively with system owners/operators to prevent and/or solve drinking water supply or quality problems; the enforcement component involves taking appropriate action when violations of specific requirements occur.

3.7.1 Canada

Responsibility for the protection and regulation of water use is shared among all the three levels of governments. Provincial and Territorial governments have put in place legislative mechanisms to protect the quality and safety of their water resources. To assist livestock producers in assuring that their operations do not compromise water safety, various government agencies and other stakeholders in different provinces publish guidelines outlining the various acts and regulations governing water use issues. In Manitoba, such acts and regulations are included in the recently revised *Farm Practices Guidelines for Pig Producers in Manitoba*. Table 3.6 lists acts which address water regulations at the provincial level.

However, these documents deal primarily with regulating the impact of animal manure on the water resources rather than on the quantity, quality or source of the water used by pork production facilities.

Table 3.6 Acts addressing water in Canada

Province	Regulation/Act
Manitoba	<ul style="list-style-type: none"> ➤ Water right act ➤ Water protection act
Saskatchewan	<ul style="list-style-type: none"> ➤ The water regulations, 2002 ➤ Environmental management and protection act
Alberta	<ul style="list-style-type: none"> ➤ Water act
Ontario	<ul style="list-style-type: none"> ➤ Environment protection act ➤ Nutrient management act ➤ Clean water act ➤ Ontario water resources act
Nova Scotia	<ul style="list-style-type: none"> ➤ Clean water act ➤ Water resources protection act
Prince Edward Island	<ul style="list-style-type: none"> ➤ Environment protection act ➤ Natural areas protection act
New Brunswick	<ul style="list-style-type: none"> ➤ Clean water act
Quebec	<ul style="list-style-type: none"> ➤ -
British Columbia	<ul style="list-style-type: none"> ➤ Water protection act ➤ Water act ➤ Environmental management act

3.7.2 The United States

The regulatory structures for water use in the US are organized much in the same manner as in Canada. Federal laws under the Clean Water Act are administered by the Environmental Protection Agency (EPA), which require each state to reduce the impact of various activities, including animal agriculture, on water quality and the environment. One document prepared by the EPA includes various aspects of animal facility impact on the environment, including manure treatment technologies, but does not detail issues related to actual water sources, quality or usage by animal operations (EPA website: Proposed Rule Development Document for Concentrated Animal Feeding Operations (CAFOs)).

Furthermore, individual states have put in place legislative and regulatory controls to protect water quality and safety and to govern water use. Some states with large numbers of hog producing units like North Carolina have instituted incentive programs under the *Environmental Quality Incentive Program* to encourage hog operations to carry out eligible conservation activities in order to facilitate a net savings in ground water resources by reducing the volume of water not consumed by the animal. It does not seem like such a program exist in Manitoba or anywhere else in Canada. In Michigan, as is the case in Manitoba, hog operations are required to report the amount they use if the total amount withdrawn on a daily basis exceeds 100,000 gallons or 70 gallons per minute. The Chapter 3 Technical Notes section outlines an example of how swine operations in Michigan account and report water usage.

3.7.3 European Union

Within the EU, issues related to the environment and water resources are the responsibility of the European Environment Agency (EEA). This agency coordinates efforts of Member States in such areas as protection of the quality and quantity of Community waters and assuring sustainable water use. A framework for Community action in the field of water policy was established in the Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000.

3.8 Conclusions and Recommendations

Available data on water usage in modern hog operations is sparse and highly variable. In addition to the water required by the pigs, an additional 5 – 30 % or more can be used for cooling, facility washing, domestic use and flushing of manure. For Manitoba, there is virtually no information on the proportion of water derived from the various available sources or on the number of swine farms that may be using recycled water to meet some of the operation's needs. There are a number of technologies, both commercially available and under development, as well as management practices that can reduce the total fresh water usage. These include: use of waterers that minimize wastage; correct diet formulation for the production stage of the animals; use of housing designs and management strategies that reduce the need for routine use of sprinkler systems for cooling and for dunging areas; use of management strategies that decrease water usage as a form of "recreation" for pigs; and incorporation of manure handling systems and water recycling technologies, as they become economically accessible. However, without having accurate values for actual water usage, as well as water source and quality, any changes in management strategies or adoption of technologies aimed at reducing water usage will be difficult to quantify and therefore to rationalize to pork producers. In addition, since estimates of manure production and therefore landbase requirements for a swine barn include estimates of water usage that becomes part of the manure handling system, it is imperative that accurate values of on-farm water use are available.

- 1) There is need to obtain accurate data on water source, quality and usage in different production systems (e.g. liquid vs. solid manure handling systems).
- 2) Pork producers should be encouraged to monitor water quality as well as usage. Water quality will impact on actual water consumption by the pigs. Knowledge of water usage, preferably for each production phase, will contribute to identifying the success of strategies adopted to reduce total water. If water monitoring is done at the production phase level and at the appropriate frequency, it will also help to detect potential problems associated with decreased water consumption and/or with faulty equipment.
- 3) Water wastage in hog barns needs to be accurately quantified. Current estimates are too variable to be relied upon. Related to this, economical methods for reducing water used for non-animal purposes are needed . This highlights the need to support technology development and knowledge transfer to the pork sector.
- 4) Swine producers should be encouraged to use technologies that reduce water wastage without compromising animal performance. There is considerable information available for those technologies that have been shown to work but it is unclear as to how widely these are used in Manitoba. Existing information on such technologies should be compiled and made readily available to producers.
- 5) Models are needed for accurate estimation of manure production by hog operations and therefore the amount of water, nutrients, etc. to be disposed of/retained and the landbase required.
- 6) There is a need for research into and promotion of animal management strategies and animal housing designs that minimize water usage and wastage and therefore reduce manure volume.
- 7) Research into and adoption of waste water recycling for existing operations that may not be able to significantly reduce water use because of their current facility design should be supported
- 8) Pig rations should be formulated to meet the specific needs of pigs at each stage of production, thereby minimizing excess nutrients and mineral intake which will influence water intake as well as manure output. In parallel, research into defining specific animal

requirements in consideration of their genotype, production phase, environment and health status should continue to be supported.

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3.10 Technical Notes

Estimating Water Usage on Michigan Swine Farms

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http://www.kbs.msu.edu/mgsp/Farmstead_Systems/WaterUse for Swine Operations.doc

Estimating Water Used by Swine Farms in Michigan

Swine farms use well water for watering animals, cleaning facilities, animal cooling and in some instances for moving manure from the barn to the storage structure. Most pigs are raised in an all-in/all-out environments where one group of pigs, at the same stage of production, is moved into a location and stays there until that group is ready to move to the next location or on to slaughter. Between groups the facility is thoroughly cleaned by pre-soaking and/or pressure washing. In the summer, during periods of extreme heat, pigs may be cooled by using drippers which emit small drops of water periodically on the animals back, or by misters giving off a small mist of water intermittently to cool the room. Some farms use well water to flush manure from the barn to the manure storage structure, but this practice is not very widespread in Michigan and therefore that water was not considered in these calculations.

Table 1 provides the estimated daily water consumption by pigs of various sizes. The range in daily water consumption within each stage of production is dependant on temperature and water conservation practices on the farm. For this example the average of the range will be used to estimate daily water use.

Table 1: Water requirements of pigs¹

Animal type	Gal/head/day	Animal type	Gal/head/day
Sow and litter	2.5 - 7	Finishing pig (100 – 250 [#] BW)	3 - 5
Nursery pig (up to 60 [#] BW)	.7	Gestating sow	3 - 6
Growing pig (60 – 100 [#] BW)	2 - 3	Boar	8
		Replacement Gilt	3

¹Swine Care Handbook

Michigan's average daily and annual water consumption for pigs at various stages of production is provided in Table 2. The Hog & Pig inventory information is from the *2002-2003 Michigan Agriculture Statistics* (NASS). Hog and pig inventories fluctuate from Quarter to Quarter, therefore the 2002 April 1, June 1, September 1, and December 1 inventories were averaged to report the 2002 numbers. The gallon per head per day is the average of the figures provided in Table 1.

Table 2: Direct water use - Drinking

Stage	Michigan hog and pig inventory ²	Gal/hd/day	Gallons daily use	Gallons annual use
Pigs under 60 [#]	305,000	0.7	213,500	77,927,500
60 – 119 [#]	202,000	2.5	505,000	184,325,000
120 – 179 [#]	156,000	4	624,000	227,760,000
Over 180 [#]	143,000	4	572,000	208,780,000
Gilts	6,500	3	19,500	7,117,500
Boars	3,000	8	24,000	8,760,000
Sow and Litter	12,800	5	64,000	23,360,000
Gestating Sow	94,200	4	376,800	137,532,000
Total Annual Water for Animal Drinking			2,206,650	875,562,000

²NASS

It was more difficult to estimate the indirect water use on swine farms. There is no reported information on which farms use which practice and the amount of water consumed by each practice. Table 3 Indirect Water Use – Cleaning and Table 4 Indirect Water Use - Cooling were developed using estimates from individuals working in the field. It was estimated that about two thirds of the pigs reared in Michigan reside on farms that regularly clean the farms facilities, the remainder of the pigs may reside in pasture or bedded situations where cleaning facilities with water is impractical. Therefore the number of head or litters was multiplied by 67% in determining total water use.

Table 3: Indirect water use - Cleaning

Operation	Approx. run time	Gal/hr.	% of pigs	Head or litters	Total water used
Wash Farrowing	20 hours / 100 litters	180	67%	184,000	4,438,080
Wash Nursery	4 hrs. / 1000 hd	180	67%	1,736,000	837,446
Wash Finish	15 hrs. / 1000 hd	180	67%	1,915,000	3,464,235
Pre soak Farr	1.25 hours / 100 litters	600	67%	184,000	924,600
Pre soak Nursery	.5 hours / 1000 hd	240	67%	1,736,000	139,574
Pre soak Finish	2 hrs per 1,000 hd	600	67%	184,000	147,936
Total water for cleaning					9,951,872

Table 4: Indirect water use - Cooling

Operation	Approx. run time	Days > 80° F	Gal/hr/ animal	% of Pigs	Head or litters	Total water used
Cool Farrowing	8 hrs/sow/day	70	2	67%	12,800	9,605,120
Cool Breeding	8 hrs/sow/day	70	2	4%	94,200	4,220,160
Cool Finishing	8 hrs/1,000 hd/day	70	2	10%	501,000	56,112
Total water for cooling						13,881,392

In Michigan larger swine farms may have up 2,500 sows at one location, or up to 4,000 finishing animals at one location. Michigan has farms that control more animals but the 2,500 sows and 4,000 finishing animals threshold represents the upper ranges of animals at one location being

provided water from one water system. Using the figures in Tables 2, 3, and 4 one may estimate that a location with 2,500 sows will use 4.84 million gallons of water annually (13,262 gallons per day), and a 4,000 head finishing location would use 5.13 million gallons of water annually (14,055 gallons per day).

Because of the swine industry structure, where large farms contract with other farms for growing pigs, it is estimated that there are no swine farms in Michigan that individually consume more than 100,000 gallons of water per day.

In Michigan, the estimated annual water used by swine farms is 899.39 million gallons. The water used for cooling is at low rates, on hot days when buildings are being well ventilated, therefore all of the cooling water should be considered as evaporative (consumptive). Very little of the water used for cleaning evaporates, therefore all of the cleaning water should be considered as non-consumptive.

Michigan does not have a large hog processor in the state therefore most of the market hogs produced in the state are shipped out of state for processing. There is a large cull sow processor in the state and most cull sows stay in Michigan for processing.

Market hogs are approximately 50% water (Tri-State Swine Nutrition Guide). The 2000 PigChamp Benchmarking publication reports that in year 2000 Michigan's sow herd had a 47% replacement rate (sows that are sold and replaced with younger gilts) (PigChamp). Michigan Agriculture Statistical Services (NASS) shows that in 2002 Michigan marketed 2.03 million head of hogs with a total weight of 522.9 million pounds. Using the PigChamp culling rate and the 2002 sow inventory, one may calculate that in 2002 there were 50,000 sows culled weighing approximately 17.60 million pounds and containing 1.1 million gallons water. Subtracting the cull sow sales from the total 2002 hogs sales indicates that Michigan's hog producers sold 1.98 million market hogs weighing approximately 505.3 million pounds 2002 (31.58 million gallons water). Table 5 provides the total consumptive/non-consumptive water use in Michigan.

Table 5: Consumptive water use

Water use	Consumptive³	Non-consumptive³	Totals³
Raising Market Hogs	31.58	667.21	698.79
Maintaining Breeding Herd		176.77	176.77
Cooling Facilities	13.88		13.88
Cleaning Facilities		9.95	9.95
Totals	45.46	853.93	899.39

³ Million Gallons

References

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CHAPTER 4 Odour Management and Air Quality

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4.1 Executive Summary

Numerous odourous compounds are generated from anaerobic decomposition of manure in hog operations. In concentrations at or above their chemical toxicity thresholds, these compounds can directly cause adverse health effects. The concentrations of odourous gases in communities downwind from hog operations are in general considerably lower than their toxicity thresholds, but they can induce odour sensations, which may trigger health symptoms by a variety of physiologic mechanisms. There is a need to identify potential health concerns associated with odour from livestock operations and to develop suitable acceptability criteria for community level exposure to odour from livestock operations.

Measuring odour presents challenges to researchers and regulatory agencies. There is a lack of accurate standardized measurement technologies for odour. There are no consistent standards/guidelines for odour measurement in Canada. Standards and procedures should be established in Manitoba for measuring odour emissions from barns and manure storage. The standards and procedures should also cover the assessment of odour downwind from livestock operations.

Odour emission data for hog operations are seriously lacking in Manitoba, as well as in North America. Limited Manitoba data indicated that odour emissions from hog operations in Manitoba are within the typical ranges reported in other jurisdictions. It appeared that that odour plumes traveled further than the separation distances for siting hog facilities recommended in the Farm Practices Guidelines for Pig Producers in Manitoba (FPGPPM). However, the measured odour annoyance-free distances were well within the range of the FPGPPM, and within those estimated by several setback models commonly used in North America.

Existing air quality regulations in Canada and other countries mainly address problems associated with industrial pollutants and few regulations are aimed specifically at odour-induced annoyance. With the current measurement technologies, it is not realistic to use concentration-limit based regulations for odour. A common practice used in many jurisdictions is to require the minimum separation (setback) distances between hog farms and the neighbouring communities. The method of estimating separation distances in the current Manitoba guideline (FPGPPM) does not have any flexibility to

account for many factors that influence odour emission and dispersion. A dispersion theory-based guideline, integrated with odour impact models, should be established in Manitoba.

Odour mitigation technologies may be based on diet manipulation, manure treatment, capture/treatment of emitted gases, or dispersion enhancement. Several technologies are considered to be good candidates for Manitoba conditions, including: (1) manure storage covers, both straw and synthetic (plastic); (2) biofilters for treating barn exhaust air; (3) shelterbelts for enhancing dispersion and creating a visual barrier to livestock facilities; and (4) dietary manipulation as an integrated approach to odour, as well as nutrient problems. Whenever possible, adopting the best management practices should be considered first in addressing odour problems before adopting technologies.

4.2 Understanding Odour

4.2.1 Odour Formation and Sources

“Pigs stink” seems to be common knowledge to most people. The fact of the matter is that a clean pig has about the same amount of body odour as a clean human being (NCARS, 1995). It is mostly hog manure along with feed that contributes to odour release from hog operations (Schaeffer, 1977). Odour compounds are produced if manure is decomposed by microorganisms under prevailing anaerobic conditions. O’Neil and Philips (1992) summarized 168 odour compounds identified in livestock odours by various researchers. Eaton (1996) listed 170 different compounds identified in hog manure odour. Schiffman et al. (2001) identified 331 VOCs (volatile organic compound) and fixed gases from swine facilities in North Carolina. Some common odour compounds identified in hog odour include ammonia, hydrogen sulfide, volatile fatty acids, p-cresol, indole, skatole, and diacetyl (Priest et al., 1994).

The four main sources of odour release from hog operations are: (1) building exhaust, (2) manure storage, (3) land application, and (4) mortality disposal. A shift to injection-spreading of manure seems to result in more odour complaints traceable to animal production facilities and manure storage than to the land application of manure (Jacobson et al., 1998). It should be cautioned, however, that odour emission during land application may be extremely high if inappropriate equipment and procedures are used. Little scientific research was found on emissions from mortality disposal sites. If managed properly, mortality disposal sites are not usually considered odour sources.

4.2.2 Odour Attributes

Odour is the sensations and perceptions that occur when a mixture of odourous compounds (odourants) stimulate receptors in the nasal cavity. In other words, odour is different from the actual odourants (chemical compounds) alone. Odour is a complex psychophysical variable, not a simple physical or chemical variable. Researchers agree that the identification of any number of individual odourants, such as hydrogen sulfide and ammonia, within an odour and measurement of their concentrations are not sufficient to describe the strength (concentration or intensity) or quality (character) of that odour (Jones et al., 1994). These odourants may be useful as indicators in odour monitoring, but do not give the complete picture of odour perception in a complex environment. Odour may be described by the following set of parameters: (a) concentration (dilutions-to-threshold value); (b) intensity, or perceived strength of the odour; (c) perceived hedonic tone (perceived offensiveness or pleasantness); (d) odour

character or odour quality; and (e) odour persistence or 'hang time' of odour in the ambient air (Jones, 1992).

Identifying the presence of odourants in hog odour is not enough to understand the characteristics of the odour since these odourous compounds are interactive and may smell differently than the pure compounds when mixed together. The combination of odourous compounds may result in five possible results as addition, reduction, independence, synergism, and averaging (Hill and Barth, 1976, cited from Feddes et al., 1999). Research on mixtures of odourants of known odour intensity proved that it is not possible to predict the odour intensity of a mixture of even two components (Rosen et al., 1962, cited from Zhu et al. 1999). Efforts have been made to correlate the odour intensity and the concentrations of some major odour compounds in hog odour. Barth and Poldowski (1974) identified the odourous components in stored dairy manure and found that the volatile organic acids correlated best with odour intensity. A study conducted by Spoelstra (1977) found that indole and skatole could not be indicators of hog odour because the concentrations of these compounds might decline during storage. He also reported that ammonia and hydrogen sulfide were not suitable indicators for hog odour (Spoelstra, 1980). Williams (1984) found that BOD (Biological Oxygen Demand) can be applied as an indicator in odour from both aerobic and post treatment manure storage. Pain et al. (1990, cited from Hobbs et al., 1995) reported a correlation between odour concentration and ammonia concentration in air, but the relationship was not constant for all farm odours and odour was still detectable at zero ammonia concentration. Many researchers have found that odour from hog operations cannot be well represented by any single or even a small group of compounds (Hobbs et al. 1999). At present, there is no consistency in the literature regarding the correlation between specific odourous gas emission and the odour sensation.

Human reactions to odour are associated with the **F**requency, **I**ntensity or concentration, **D**uration, and **O**ffensiveness of the odour (**FIDO**) (NCMAWM, 2001). The odour frequency is a measure of the number of perceivable odour events that occur in a given length of time. The odour intensity or concentration is a measure of the strength of a perceived odour. The odour duration is a measure of the length of time an uninterrupted odour event occurs. The odour offensiveness is a measure of the unpleasantness/pleasantness of a perceived odour.

4.2.3 Odour Measurement

4.2.3.1 Measuring odour concentration

Odour concentration is the most commonly used parameter for signifying the strength of a livestock odour (McGinley et al., 2000a). Although analytical techniques have been used to identify individual odourants in odour, there is no established correlation between the concentrations of these individual odourants or specific groups of odourants and the human response to odour (Jones et al., 1994). Thus, non-analytical techniques employing the human olfactory sense are commonly used to measure the strength of odours (NCMAWM, 2001; Kephart and Mikesell, 2000).

The dynamic-dilution olfactometer is considered to be the industry standard for measuring odour concentration (see Note: Olfactometer). The odour concentration measured by olfactometers is commonly expressed as "odour units" (OU) (mostly in North America) or "odour units per cubic meter" (OU/m³) (in Europe). The European standard (CEN, 2003) defines an European reference odour mass (EROM), which is equivalent to 123 µg n-butanol evaporated into 1 m³ of neutral gas (air). This leads to a definition of the

European odour unit, which is the amount of odourant(s) that, when evaporated into 1 m³ of gas air at standard conditions, elicits a physiological response from a panel (detection threshold) equivalent to that by one EROM. Therefore, the odour concentration is expressed as OU/m³.

Measuring odour is a complex process, and therefore standards must be followed to ensure the accuracy and consistence. A number of odour measurement standards exist, including:

- Australian/New Zealand standard (AS/NZS 4323.3:2001)
- Dutch standard (NVN 2820)
- European Union (EU) standard (NF EN13725)
- French standard (AFNOR X-43-101)
- German standard (VDI 3881)
- US standard (ASTM E679-91 and ASTM E544-99)

These standards specify criteria, procedures and protocols to be followed when odour concentration is measured. They also specify the types of materials that may be used in equipment for odour measurement and sampling. Among these standards, the EU and ASTM standards are most commonly followed for measurements of livestock odours in North America.

It should be noted that dynamic-dilution olfactometers may not be suitable for measuring downwind odours from hog operations. Because of constantly changing atmospheric conditions, obtaining downwind odour samples that are representative to what is actually experienced by the receptors in real life situations becomes impractical. For example, when a 10-L Tedlar bags are used for collecting samples for olfactometer measurement, it takes up to several minutes to collect a sample (fill the bag). The collected sample, therefore, reflects the odour strength “averaged” over the sampling period. However, the instantaneous bursts of high odour strength may be of more concern than the average odour strength. Zhang et al. (2003) reported that there was little correlation between the odour concentration of bagged samples measured with olfactometers and the odour intensity assessed by human sniffers (trained odour assessors) in the field. In other words, the bagged samples could not capture instantaneous high odour levels in the field.

A potentially more satisfactory method of evaluating odour directly in the field is quantifying the instantaneous odour intensity by using human sniffers (see Note: Odour Intensity Measurement by Human Assessors). There is a German guideline which describes specific procedures of determining field odour plumes by human sniffers (VDI, 1993). Hartung and Jungbluth (1997) followed the German guideline to measure the odour plumes from dairy and cattle barns. Sniffers ranked odour intensity in the field based on a 6-point intensity scale suggested by German VDI Guideline 3882 (VDI, 1992). Zhu et al. (2000a) used human sniffers to conduct on-site odour intensity measurement. The sniffers were trained to rank odour intensity on a scale of zero to five (0: no odour; 1: very faint; 2: faint; 3: distinctly noticeable; 4: strong; 5: very strong odour). Area residents who received limited training were used by Guo et al. (2001) in monitoring odour occurrences in a livestock production area. They used a relatively simple intensity scale of 0 to 3 (0: no odour; 1: faint odour; 2: moderate to strong odour, and 3: very strong odour). Zhang et al. (2001b) used trained human sniffers (Nasal

Rangers™) to measure odour on four hog farms. Nasal Rangers™ ranked odour based on an eight-point n-butanol intensity scale. Their results showed the promise of using human sniffers for downwind odour evaluation.

4.2.3.2 Measuring odour emission

Odour release from hog operations is quantified by the amount of odour (expressed as OU) being emitted per second (OU/s), which is commonly known as the odour emission rate. For easy comparisons, the emission rate is also often described in terms of odour units emitted per unit area (OU/s-m²) or per animal unit (OU/s-AU). To measure odour emission from buildings, air samples are taken from exhaust fans and analyzed for odour concentrations with olfactometers. The building ventilation rates are also measured (see Note: Ventilation Rate). The odour emission rate is then calculated as the product of odour concentration and ventilation rate.

Wind tunnels (or flux hoods) are commonly used for sampling odour emission from manure surfaces. A wind tunnel is a portable open-ended, open-bottomed enclosure placed over an odour emitting surface. Filtered ambient air is blown through the tunnel to pick and mix with odour emitted from the surface. The mixture is sampled at the downstream end of the tunnel for determining the odour concentration. The odour emission rate is then determined as the measured odour concentration multiplied by the airflow rate through the wind tunnel. Smith and Watts (1994) evaluated the performance of two wind tunnels of different sizes in measuring odour emission from feedlots and reported that there was strong dependence of the measured emission rate on the air velocity inside the tunnel. They recommended that the air velocity should be specified whenever emission rates measured by wind tunnels are reported. Schmidt et al. (1999) also investigated the effect of air velocity in the wind tunnel on the measurement of odour emission from manure storage. They found that measured emission rate increased exponentially with the tunnel wind speed. This means that odour emission rates from manure storage reported in the literature are “nominal rates”, which are the emission rates at certain wind speeds used in wind tunnels but may not reflect the emission rates in the field where the wind speed changes instantaneously.

4.2.3.3 Odour Dispersion and Transportation in the Atmosphere

Odour disperses in the atmosphere after it is released from hog facilities. The impact of odour on the surrounding areas is very much dependent on how far odour travels and how much odour disperses in the atmosphere. Odour transportation and dispersion are influenced by several variables interacting collectively, including the wind direction and speed, atmospheric stability, and topography. The worst condition for odour dispersion (impact) is when the atmosphere is stable – little solar radiation and low wind speed (see Note: Dispersion).

4.3 Odour Effect on Human Health

The effect of odour on humans may be *physiological or psychological* in nature. The physiological effect is caused by odourants in sufficiently high concentrations – generally at or above their chemical toxicity thresholds, whereas the psychological effect is due to the presence of unpleasant odours from the point of view of the basic emotions involved when odourant concentrations are generally much lower than their toxicity levels. The concentrations of odorous gases in communities downwind from hog operations are in general considerably lower than their toxicity threshold values. Agricultural odours are

rarely associated with chemical toxicity. However, the psychological effect of odour may lead to physiological symptoms. Noxious environmental odours may trigger health effects by a variety of physiologic mechanisms, including exacerbation of underlying medical conditions, innate odour aversions, aversive conditioning phenomena, stress-induced illness, and possible pheromonal reactions. Schiffman (1995) investigated the health effect of hog odour on neighbouring communities and showed that people living near intensive hog operations in North Carolina experienced significantly more anger, confusion, tension, depression, fatigue, and less vigor than control populations not living near intensive hog operations (see Note: Schiffman). Other researchers (e.g., Ackerman, 1992; Lohr, 1996) have demonstrated that people can have strong emotional and physiological responses to odours based in part on their previous experiences, social background, and expectations.

Schiffman et al. (2000) posited three paradigms by which odours could produce health symptoms. In the first paradigm, the adverse effects are caused by exposure to odourants at levels above the toxicological threshold for the effect. In this case, irritation, rather than odour, is the cause of the symptoms, and odour simply serves as an exposure marker. The threshold for irritancy is between 3 - 10 times higher than the concentration at which odour is first detected. There are extensive evidences indicating that high concentrations of volatile odour compounds can produce irritation in both the upper respiratory tract (nose and larynx) and lower respiratory tract (trachea, bronchi, and deep lung sites). Schiffman et al. (2000) summarized the physiological symptoms caused by sensor irritation: (1) changes in respiratory rate, (2) reduced respiratory volume, (3) increased duration of expiration, (4) alterations in spontaneous body movements, (5) contraction of the larynx and bronchi, (6) increased epinephrine secretion, (7) increased nasal secretion, (8) increased nasal airflow resistance, (9) increased bronchial tone, (10) decreased pulmonary ventilation, (11) bradycardia, (12) peripheral vasoconstriction, (13) increased blood pressure, (14) closure of the glottis, (15) sneezing, (16) closure of the nares, (17) decreased pulmonary blood flow, (18) decreased renal blood flow and clearance, and (19) lacrimation or tearing. Volatile chemical irritants can also cause local redness, edema, pruritis or pain, and eventually altered functions. Excessive irritation in the lower and upper airways may lead to tissue damage and, eventually, scarring. Airway irritation is also associated with non-respiratory tract health complaints such as headache and lassitude.

The second paradigm described by Schiffman et al. (2000) concerns cases where the concentration of the odourant exceeds its odour threshold but not its toxicological threshold. This typically occurs with exposure to certain odourant classes such as sulfur-containing compounds and organic amines with odour thresholds that are 3 - 4 orders of magnitude below the levels that cause classical toxicological or irritant symptoms. Schiffman et al. (2000) found that shallow and irregular breathing patterns were induced by exposure to unpleasant odours (hog odours, rotten fish, sulfides) while deeper stable breathing patterns were the characteristic of exposure to pleasant odours (chocolate chip cookies, orange cake). Odours perceived to be unpleasant can impair mood and increase reactivity to startling stimuli. Negative mood, stress, and environmental worry can potentially lead to a number of physiological and biochemical changes with subsequent health consequences. These include elevations in blood pressure, immune impairment, increased levels of peripheral catecholamines, increased glucocorticoids, increased secretion of adrenocorticotrophic hormone (ACTH) from the pituitary,

decreased gastric motility, increased scalp muscle tension in patients with muscle tension headaches, and even hippocampal damage.

In the third paradigm, the odourant is part of a mixture containing a co-pollutant that is actually responsible for the reported health symptom (Schiffman et al. 2000). Emissions from confined animal operations may contain co-pollutants such as particulates. Particulates associated with fecal waste are also known to carry bacteria. Thus, it is likely that some of the health complaints ascribed to odour may, in fact, be caused by particulate matter (liquid or solid) suspended in air or by a synergistic effect between odourants and particulates. For example, the adverse health effect of ammonia and particulates in combination was greater than the additive effect of ammonia and particulates by a factor of 1.5 to 2.0. Both fine and coarse particles in an odorous plume enter the nasal cavity and can induce nasal irritation.

4.3.1 Odour as Irritant

As discussed in the first paradigm of Schiffman et al. (2000), odours compounds from hog operations may cause negative physiological responses in human body if present in sufficiently high concentrations. The typical odour compounds in hog odour are ammonia, hydrogen sulfide, and VOCs (hundreds of VOC's have been identified in emission from livestock operations). Ammonia in air is an irritant and causes burning of the eyes, nose, throat and lungs. At levels greater than 100 ppm it can cause serious injury to such tissues (ODHS, 2000). Hydrogen sulfide is colorless and has a foul rotten egg odour. The hydrogen sulfide concentration at 300 ppm is considered by the ACGIH (American Conference of Governmental Industrial Hygienists) as immediately dangerous to life and health (OSAH, 2007).

Schiffman (1998) discussed the effects of livestock odours on the health and well-being of residents in the neighborhood of hog-rearing operations. The main findings are given under a series of 12 headings, some of which are quoted here:

- What health symptoms do persons exposed to odour complain about? – eye, nose and throat irritation as well as headache and drowsiness caused by stimulation of free nerve endings in the nose and throat by sub-threshold levels of volatile organic compounds (VOC's) that interacted additively or hyperadditively.
- Can odour from livestock operations cause rhinitis, asthma, bronchitis, or other immunological irregularities? Broadly, yes. Dust and ammonia are reported to cause problems for workers in barns. Residences near large operations are associated with increased respiratory inflammation. Increased pollution levels are linked to rhinitis, chronic nasal irritation and asthma. Formaldehyde, which is a component of these odours, is linked to asthma-like symptoms.
- Are any of the compounds in hog odours toxic and can these compounds get into the human body? VOC's can enter the body through the lungs, GI tract and skin and can affect metabolic and other physiological processes. They are generally fat soluble and can pass through the lining of the lungs to the bloodstream and be transported to the fat storage and organ sites where they can be deposited. The levels in livestock odour are low, so are probably not toxic. The issue of potential bioaccumulation is not addressed.

Schiffman et al. (2000) reported a variety of health complaints putatively associated with odours emanating from manure, including eye, nose, and throat irritation, headache,

nausea, diarrhea, hoarseness, sore throat, cough, chest tightness, nasal congestion, palpitations, shortness of breath, stress, drowsiness, and alterations of mood.

It should be noted that odour concentrations at downwind locations are typically very low, although they may exceed olfactory threshold values and create nuisance conditions (Sweeten et al., 2000). Odorous compounds generally have not been considered physiologically toxic at concentrations found downwind of livestock feeding facilities. Mackie et al. (1998) and Tamminga (1992) noted that the lowest toxic values (LTV) of frequently cited odorous gases from confinement buildings ranged from 5 to 20,000 times higher than cited odour threshold values for the compounds.

4.3.2 Odour as Nuisance

In the traditional context of environmental health, odour is considered to be a nuisance (not a pollutant) (Chrostowski et al., 2003). However, the definition of "health" has been expanded by the World Health Organization (WHO) as "...a state of complete physical, mental, and social well-being and not merely the absence of disease or infirmity." According to WHO, "environmental health comprises those aspects of human health, including quality of life, that are determined by physical, biological, social and psychosocial factors in the environment. It also refers to the theory and practice of assessing, correcting, controlling and preventing those factors in the environment that can potentially affect adversely the health of present and future generations." (USDHHS 1998). Thus, a symptom that diminishes physical, mental, or social well-being would be a "health effect".

Sensory information on odour is first processed through the area of the brain responsible for emotion and memory before it is processed for identity and decision-making. Therefore, odour nuisances can bring back memories and associations, even before the particular stimulus is identified. The effects of odour and the memories and emotions produced by odour have been studied from different view-points and all feed into the same conclusion, that odour and emotion are connected and humans respond to emotions in a manner which is physiological. Therefore, exposure to odour may lead to health (physiological) complaints besides psychological ones.

Caralini (1994) found that when exposed to odour, some people were annoyed, and of these people, only some reported general health complaints. Exposure to odour itself may not directly cause health complaints. But, annoyance is the intervening variable between odour exposure and general health complaints. A possible explanation for the relation between annoyance to odour and general health complaints might be found in the personality and attitudes of the exposed individual.

Dalton (2002) noted that numerous factors, including exposure history, expectations, personality, beliefs, social factors, and bias, can influence an individual's perception of odour, irritation, or health effects. Dalton (2002) also noted that anxiety over the consequences of exposure can worsen the perception of odour and/or irritation. Dalton (1996) reported that subjects rated an odour as more intense when they were told it was hazardous compared to those who were told that the same odour was a natural product. In another study, Dalton et al. (1997) found that people given positive information about odours to which they were exposed had less perceived irritation compared to people given negative information.

Thu et. al. (1997) collected mental and physical health information through personal interviews from a random sample of 18 residents living within two miles of a 4,000 sow operation. The data were compared to those collected from a demographically comparable sample of 18 rural residents living in an area with minimal livestock production. The results of the comparison indicated that neighbors of the large-scale hog operation reported experiencing significantly higher rates than the controls in four clusters of symptoms that are known to represent toxic or inflammatory effects on the respiratory tract. These clusters of symptoms have been well documented among workers in hog facilities. The specific symptoms reported included cough, increased sputum production, shortness of breath, chest tightness, wheezing, nausea, dizziness, headaches, runny nose, scratchy throat, burning eyes, muscle aches and pains, skin rash, fever. However, among the control group, symptoms of skin rash, muscle aches, and fever were more frequently reported. Additionally, there was no difference in the frequency of reported symptoms and distance from the hog facility, contrary to what one might expect. This study found that residents within two miles of the hog operation did not suffer higher rates of psychological health problems such as depression or anxiety when compared to controls. Thu et. al. (1997) also stated that all responders felt the owner of the farm was creating social and class divisions within that community.

Warner et al. (1990) assessed the impact of a 50,000 animal hog-growing facility as an odour source and potential health problems. The health problems were selected based on neighbors' reported complaints such as breathing difficulties, burning sensations in the nose and throat, nausea and vomiting, and headaches. They concluded that these responses contained complaints of symptoms attributable to the hog facility.

Chrostowski et al. (2003) found that in the mind of the public, the persistence of an odour, even after controls, implies the persistence of a health threat and the public perception problem remains unresolved although the environmental health problem may be resolved.

4.3.3 Other Airborne Emissions

Animal feeding operations may emit ammonia (NH_3), nitrous oxide (N_2O), hydrogen sulfide (H_2S), carbon dioxide (CO_2), methane (CH_4), total reduced sulfur (TRS) compounds, volatile organic compounds (VOC), hazardous air pollutants (HAP), and particulate matter (EPA, 2001). The emitted substances and the quantity of emissions can vary substantially depending on the design and operation of each facility. Factors that influence emissions include the feeding regiment, the type of confinement facility, the type of manure management system (storage, handling, and stabilization), and the method of land application of manure.

4.3.3.1 Hydrogen sulfide (H_2S)

Hydrogen sulfide is a colorless, flammable gas that smells like rotten eggs at low concentrations. Because H_2S has a specific gravity heavier than air, it stays close to the ground and can accumulate in enclosed, poorly ventilated, and low-lying areas. The odour detection threshold for H_2S ranges from 0.5 ppb to 30 ppb for 83% of the population, while the irritation threshold ranges from 2.5 to 20 ppm (MDH, 2007). Thus, the odour threshold for H_2S (as well as other sulfur-containing compounds) is 3-4 orders of magnitude (that is 10³ and 10⁴ times) below the level that causes irritant symptoms. The scientific literature on H_2S suggests that health symptoms can occur with chronic exposure to H_2S concentrations far below the levels at which acute irritation or toxicity occurs. Although H_2S has been identified as one of the main odours gas in hog

operations, little information is available in the literature on its effect on the communities nearby hog operations. Investigations on the impact of H₂S from paper mills, refineries, geothermal sources and meat packing plants indicate that exposure over a period of time to low levels (below the irritant threshold) of H₂S or other reduced sulfur compounds can cause health effects. The health effects were found for an average daily exposure to 10-11 ppb H₂S. These effects included eye, respiratory or neuropsychological symptoms. Acute exposure to H₂S at levels in the low ppm range (1 to 7 ppm) can also induce health symptoms including headache, increased airway resistance, coughing, throat irritation and eye pain. At 30 ppm, H₂S becomes neurotoxic and induces nasal lesions in olfactory mucosa. At 200 to 1000 ppm, brief exposure to H₂S can be fatal. Levels of H₂S inside livestock buildings (e.g., 1 to 2 ppm) tend to be above those that have been reported in other settings to elicit health symptoms with chronic (and in some cases acute) exposure.

Hydrogen sulfide is a concern to health within hog facilities, particularly during manure handling. Donham (1995) found that during agitation of liquid manure, the concentration of H₂S in the breathing zone of workers could climb from 5 ppm to lethal levels over 500 ppm within seconds. Ni et al. (2000; 2002) reported that the mean H₂S concentrations in hog finishing facilities in Indiana were between 65 and 536 ppb. Zhu et al. (2000b) studied the daily variations in H₂S emissions from various mechanically and naturally ventilated hog housing systems and found that the concentrations ranged from 200 and 3,400 ppb.

Ambient H₂S downwind from hog facilities can exceed 50 ppb (NCMAWM 2001). The Minnesota Pollution Control Agency randomly checked the property line H₂S concentrations on 138 farms throughout Minnesota (MPCA, 1999). Twenty-four of the 138 farms had at least one H₂S measurement that exceeded the Minnesota 30-ppb regulatory limit. Manure storage was found to cause a greater impact on ambient H₂S than barns. Bicudo et al. (2002a) investigated the ambient H₂S concentrations near hog barns and manure storage. The average H₂S concentrations measured about 15 meters from deep-pitted wean-to-finish barns were between 4 and 6 ppb. However, H₂S concentrations as high as 450 ppb were recorded about 5 meters downwind of a deep-pitted finishing barn.

4.3.3.2 Ammonia (NH₃)

Ammonia is a colorless gas at ambient temperature and pressure. It is a water-soluble irritant. As such ammonia can be rapidly absorbed in the upper airways, thereby damaging the upper airway epithelia. Ammonia is released from the natural decomposition of organic material, including manure as well as dead animals and plants. At concentrations above 0.7 ppm, ammonia has a pungent, sharp, repellent and acrid odour. The eye irritation threshold (irritation just barely noticeable) for ammonia is 4 ppm (NCMAWM, 2001). Decrements in baseline PFT tests (pulmonary function tests) have been reported in workers exposed to NH₃ at concentrations of 7 ppm in tandem with other aerial contaminants. At higher concentrations, a certain amount may bypass the upper airways, causing lower lung inflammation and pulmonary edema. Ammonia may also adhere to respirable particulates (< 5 μ m in aerodynamic diameter) that can reach alveoli and further adversely affect respiratory function. In addition to respiratory effects, ammonia can cause dermal and ocular irritation. Merchant et al. (2002) provided a comprehensive literature review of clinical, experimental, and epidemiological observations concerning human health effects of ammonia. For instance, exposure to 50-150 ppm ammonia can lead to severe cough and mucous production; whereas

exposure to >150 ppm ammonia can cause scarring of the upper and lower airways (Close et al., 1980; Leduc et al. 1992), though these concentrations are rarely reached in practice.

Most studies on ammonia associated with livestock production have been focused on ammonia concentrations inside and emissions from the facilities, and very limited information could be found in the literature on outdoor ammonia levels in the vicinity of confinement hog facilities. Ammonia concentrations up to 200 ppm have been found in some animal (e.g., poultry) confinement facilities, but typical levels are much lower (5 to 70 ppm). Reynolds et al. (1997) measured outdoor concentrations of ammonia near four types of swine production facilities (large, medium and small confinement and small conventional) and one control farm with no livestock. At a distance of 60 meters outside of facilities, the mean ammonia concentrations were: 0.251 ppm for the large confinement; 0.086 for the medium confinement; 0.214 ppm for the small confinement; 0.139 ppm for the small conventional; and less than 0.004 ppm for the control farm. These concentrations of ammonia measured outside the production facilities were below current occupational health standards, but they cautioned that it is possible that ammonia could be a physical irritant in combination with other exposures.

4.3.3.3 Volatile organic compounds (VOCs)

In an analysis of VOCs emitted from hog facilities in North Carolina utilizing gas chromatography and mass spectrometry (GC/MS), over 300 compounds were identified (NCMAWM, 2001). Many more compounds were present, but the GC peaks were too small to identify them. The compounds identified by GC/MS were diverse and included many acids, alcohols, aldehydes, amides, amines, aromatics, esters, ethers, fixed gases, halogenated hydrocarbons, hydrocarbons, ketones, nitriles, other nitrogen-containing compounds, phenols, sulfur-containing compounds, steroids and other compounds. Acids, phenolic compounds and aldehydes were present in the highest concentrations. The magnitude of total VOCs associated with animal feeding operations and/or waste management systems varies widely from as low as 0.60 mg/m³ in a recently cleaned hog facility to 108 mg/m³ from the headspace of a chamber containing slurries produced by weaner pigs. The effect of a large number of VOCs in aggregate is cumulative. Exposure to low concentrations of hundreds of compounds simultaneously can produce high levels of odour and irritation downwind of the facility. Introduction of irritant compounds into the upper and/or lower respiratory tract has been found to produce many systemic responses including altered respiration.

4.3.3.4 Particulate Matter Including Bioaerosols

Epidemiological evidences predominantly from urban settings indicate that exposure to increased levels of particulates is associated with increased mortality risk, especially among the elderly and individuals with preexisting cardiopulmonary diseases, such as chronic obstructive pulmonary disease, pneumonia and chronic heart disease (NCMAWM, 2001). Epidemiological studies also suggest that particulate exposure can increase the risk of respiratory and cardiovascular morbidity such as increased hospital admissions or emergency room visits for asthma or other respiratory problems, increased incidence of respiratory symptoms or alterations in pulmonary function. This effect can begin to occur when the ambient PM₁₀ (particles <10 µm in size, see Note: Particulate Matter) reaches a level of 0.03 to 0.15 mg/m³, according to the Committee of the Environmental and Occupational Health Assembly of the American Thoracic Society (NCMAWM, 2001). Researchers at University of Iowa (K.J. Donham and his colleagues)

suggested exposure limits for swine confinement workers as follows: 2.4 mg/m³ total dust and 0.23 mg/m³ respirable dust (cited from Predicala et al., 2001).

The sources and components of dust associated with hog operations are diverse and might include a range of microorganisms and their cell wall components, dried dung and urine, skin flakes, grain mites, spores, pollens, feed and bedding particles (Pedersen et al., 2000). Most studies on dust associated with livestock production have been focused on the air quality inside the buildings and little information could be found in the literature on dust levels in the vicinity of the confinement hog facilities. Typically the total dust levels inside hog confinement buildings are less than 5 mg/m³, but levels can reach up to 52 mg/m³ in some cases, with respirable dust comprising 5 to 50% of the total dust (NCMAWM, 2001). Pedersen et al. (2000) summarized typical dust levels in pig buildings measured in England, The Netherlands, Germany, and Denmark. The respirable dust concentrations ranged from 0.18 to 0.26 mg/m³. Predicala et al. (2001) reported that the total and respirable dust concentrations in mechanically ventilated hog buildings were 2.13 and 0.11 mg/m³, respectively. Reynolds et al. (1997) reported that dust was detected at a distance of 60 meters outside of four swine production facilities, but concentrations were generally below limits of accurate detection.

4.4 Assessment of Odour Problem in Manitoba

Odour is one of the greatest concerns to the public when considering the siting of new or the expansion of existing hog operations in Manitoba. The odour problem in Manitoba is to some extent a perception issue, similar to other jurisdictions. The major scientific issues associated with odour problem include odour emission (how much odour is released), odour dispersion (how far it travels downwind), and the odour impact on health and property values. Several studies were conducted in Manitoba to address odour emission and dispersion, but little information is available on odour impacts (See Chapter 9 for studies on the effect of livestock operations on the property value in the neighbouring areas).

4.4.1 Odour Emissions

Zhang et al. (2001a) measured odour emissions on ten (10) hog farms in Manitoba in 1999 and 2000. Five of the ten farms were farrow-to-finish operations (size ranging from 130 to 800 sows), two nursery operations (5,000 and 10,000 hogs), two farrow-nursery operations (2,500 and 3,000 sows), and one grow/finish operation (4,000 hogs). Seven of the ten operations included in this study were less than 5 years old, and the other three were 10, 35 and 40 years old, respectively. They reported that the farm-average odour levels from barn exhaust ranged from 131 to 1842 OU/m³ on ten farms. No apparent correlations were found between the odour level and the general farm characteristics, such as the age and type of operation, ventilation system, and manure handling system. Farm-average odour emission rates ranged from 12 to 39 OUs⁻¹m⁻². Odour levels and emission rates measured in Zhang et al's (2001a) study were similar to those reported for Minnesota livestock facilities by researchers from University of Minnesota (Jacobson et al., 1999a)

Zhang et al. (2007) reported odour emissions measured on two 3,000-sow farrowing farms in southern Manitoba, one with open earthen manure storage (EMS) and another with negative air pressure (NAP) covered EMS. The total odour emission was 303,1120 (open EMS) and 174,522 OU/s (NAP) for the two farms, respectively. Of the total emission, 129,276 OU/s was from buildings and 173,853 OU/s from EMS on the farm

with open EMS. For the farm with NAP EMS, 170,707 OU/s was from buildings and only 3.815 OU/s from NAP EMS.

4.4.2 Downwind Odour Levels

Guo et al. (2006) and Zhou et al. (2005) reported downwind odour levels (intensity) from the two 3000-sow farrowing facilities described by Zhang et al. (2007). Fifteen human odour sniffers were selected and trained to use an 8-point ASTM Odour Intensity Reference Scale to quantify the field odour intensity (see Note: Odour Intensity Measurement by Human Assessors). The variation of odour intensity with the downwind distance is summarized in Figure 1. For the two farms studied, odour traveled 5.3 and 4.5 km, respectively. However, the odour intensity was less than level 3 (little annoying), which is considered to be the acceptable odour level in most jurisdictions, at a distance of 667 m for the farm with NAP EMS and 926 m for the farm with open EMS.

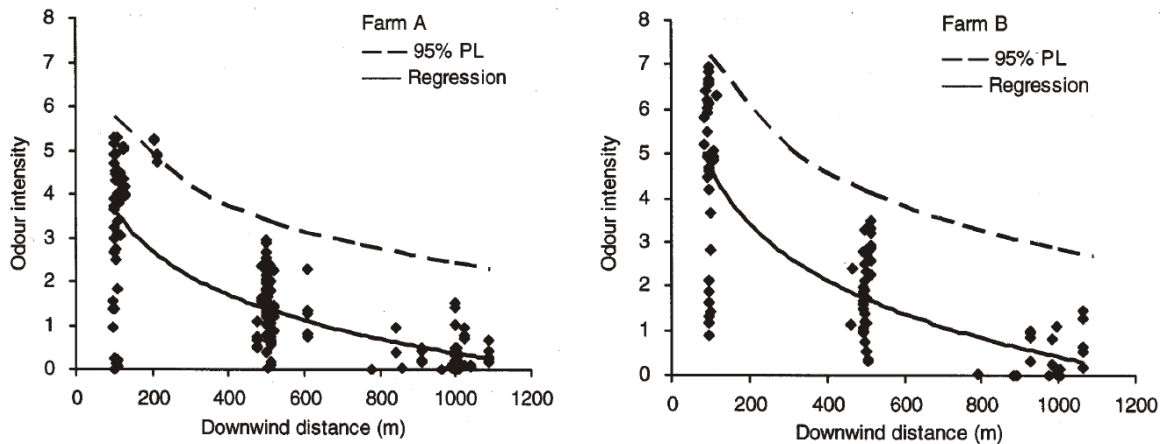


Figure 1. Variation of measured odour intensity with downwind distance for two 3000-sow farrowing operations in southern Manitoba (Guo et al., 2006).

Adam (1999) conducted a study to assess odour plumes downwind from six hog operations in southern Manitoba in locations ranging from sparsely- wooded flat terrain to heavily-treed rolling land. Two-person odour panels evaluated the strength of the odour plumes on a scale of 1 to 10 under varying weather conditions. The longest odour plumes reported ranged from 1.3 to 6.1 km (Table 1). It should be noted that the odour plume boundaries shown in Table 1 were defined by the odour level between 0 and 1 in a 0-10 scale, with zero (0) being nondetectable and one (1) being a barely detectable odour (Adam 1999). It was not clear what the odour annoyance limit was in the 0-10 scale used by Adam (1999). Detectable odour is not necessarily annoying. For example, the annoyance limit is commonly accepted as level 3 in the ASTM 0-8 n-butanol odour intensity scale, and level 2 in the 0-5 scale (Guo et al., 2000 and 2006). In other words, odour was detected, but might not be necessarily annoying within the plumes summarized in Table 4.1. Guo et al. (2006) used two measures, namely the odour-free distance (odour is not detectable) and the odour annoyance-free distance (odour is detectable but not annoying), to define the odour impact distance. It may not be realistic in most cases to maintain the odour-free distance and it is reasonable to use the annoyance-free distance as a criterion in regulating odour.

Table 4.1 Length (in kilometers) of the longest odour detection plumes found in six hog operations in southern Manitoba (Adam, 1999)

Farm	Animal Unit	Type of Manure Handling System	Longest Odour Plume, km	MB Setback*, km
Zhoda	596	Earthen storage with surface spreading	2.5	2.0 (1.3)
CMS	560	Vertical concrete tank with surface spreading	5.6	2.0 (1.3)
Dufrost	560	Earthen storage with surface spreading	4.1	2.0 (1.3)
Royal Pork	115.5	Two-stage earthen storage with surface spreading	1.3	1.2 (0.8)
Treesbank	230.5	Two-stage, covered concrete pit with overflow liquid earthen storage and surface spreading	2.0	1.6 (1.1)
Green Acres	812.5	Earthen storage with periodic surface spreading of solids	6.1	2.4 (1.6)

* Farm Practices Guidelines for Pig Producers in Manitoba. The number is the minimum separation distance from the designated residential or recreational area to earthen manure storage, or to buildings (number in “()”).

A common practice for reducing the impact of hog odour on the neighbouring communities is to maintain appropriate separation (setback) distance between hog farms and the neighbouring communities. Comparing Adam’s (1999) results with the Farm Practices Guidelines for Pig Producers in Manitoba, it can be seen that odour plumes traveled further than the setback (separation) distances recommended for siting hog facilities (Table 1). In other words, following the Guideline would not completely eliminate odour. This is further confirmed by the results reported by Guo et al. (2006), in which the odour-free distances were 5.3 and 4.5 km for the two 3000-sow farrowing farms, respectively, whereas the Guideline minimum separation distances from the designated residential or recreational area to earthen manure storage and to buildings are 2.4 and 1.6 km, respectively. However, The odour annoyance-free distances reported by Guo et al. (2006) were 667 and 926 m for the two farms, respectively, which were well within the minimum separation distances of the Guideline.

Guo et al.. (2006) also compared the measured odour-free and odour annoyance-free distances for the two 3000-sow farrowing facilities with several setback models commonly used in North America, including Minnesota, Purdue (developed at Purdue University), Alberta, and Ontario (Table 4.2). The measured odour-free distances were greater than the maximum distances determined by those setback models, but the measured odour annoyance-free distances were well within the ranges of the model predictions.

Table 4.2 Comparison of the measured odour-free and odour annoyance-free distances (m) with setback models for two 3000-sow farrowing operations in southern Manitoba (Guo et al., 2006) (Farm A – with NAP EMS, Farm B – with open EMS).

Farm	Minnesota OFFSET						Purdue		Alberta		Ontario		Measured	
	W1*	W2	W3	W4	W5	W6	Max	Min	Max	Min	Max	Min	O.A.F.+	O.F.**
	99.95%	99.1%	98.6%	97.5%	95.0%	90.8%								
Farm A	5061	3185	2042	1638	1173	894	2200 (2126)*	989 (1063)*	1873	702	1114	557	667	5284
Farm B	5244	3305	2120	1705	1222	933	2940 (2841)*	1321 (1420)*	2345	879	1114	557	926	4528

* W1 – W6 are six different weather conditions (see Note: Minnesota OFFSET), and the percentages represent the odour annoyance free frequencies

* Mean of 16 directions;

+O.A.F. = odour-annoyance free distance;

**O.F. = odour-free distance

4.5 Current Limits, Thresholds and basis for Odour Levels

Laws and regulations aimed at limiting the occurrence of nuisances have long been in force in many countries. However, regulations aimed specifically at odour-induced annoyance are relatively new. The first odour regulations in Europe appeared in the 1970's, which required the minimum setback distances for livestock operations (van Harreveld, 1991; Mahin, 2001). In recent years many jurisdictions have proposed and, in some cases, implemented policies and regulations specifically aimed at odours from commercial activities, both agricultural and industrial. In general terms, there are three basic approaches to regulating odours: 1) qualitative regulatory frameworks, which define the environmental quality in general terms, such as the absence of nuisance; odours not detrimental to the amenity; no justified complaints, 2) quantitative regulatory frameworks, which define the ambient air quality criteria, such as odour concentrations; frequencies of exceedence of concentration limits; ambient concentration limits of specific odourous compounds (e.g., hydrogen sulfide); and frequencies of odour detections, and 3) standard operational requirements for specific activities, such as setback distances for livestock operations and requirements for standard abatement techniques.

4.5.1 Air Quality Regulations in Canada and Manitoba

At the federal level, air quality standards, guidelines, objective, and criteria are covered under the Ambient Air Quality Criteria and the Canada-Wide Standards. These criteria and standards address primarily industrial pollution problems. The Manitoba Ambient Air Quality Criteria (MAAQC) addresses odour as a Guideline. The maximum acceptable odour level in the MAAQC is 2.0 OU (odour unit) for the Residential Zone, and 7.0 OU for the Industrial Zone, with the Maximum Desirable Level Concentration <1.0 OU (Manitoba Conservation, 2007). This range of odour concentration limits (1 – 7 OU) is in line with regulations/standards in many other jurisdictions. However, a close attention should be paid to the footnote in the MAAQC: *“..... It is intended that the odour unit limits be used only for evaluating potential impacts on a community during the environmental impact assessment of new or modified developments.”*

In regulating industrial pollutants, some time-average concentration limits are defined (e.g., the 1-h average maximum acceptable H₂S level of 15 µg/m³ in MAAQC), and a violation is cited if the measured concentration exceeds the limit. As discussed in Section 1.3, the downwind odour concentrations can not be accurately and reliably measured with the current technologies. This makes it almost unrealistic to regulate odours in the way industrial pollutants are regulated, i.e., setting the concentration limits. The footnote in MAAQC indicates that the odour limits (1 - 7 OU) are intended for *evaluating potential impacts*, normally through dispersion modeling (see Note: Dispersion Models). Dispersion models can predict odour concentrations as low as 1 OU or even lower. But there is no instrument capable of measuring downwind odour concentrations of 1 or 7 OU. The selection of 7 OU as the odour limit is probably related to an odour measurement instrument – Scentometer, developed in late 50's (see discussion on Scentometer measurement in Section 4.5.5.4).

Besides setting the concentration limits for downwind odours for assessing potential impacts through dispersion modelling, it is a common practice in many jurisdictions to require the minimum separation (setback) distances between hog farms and the neighbouring communities.

4.5.2 Setback/Separation Distances

The methods for estimating setback distances are either empirical (experience-based) or dispersion theory-based. Some European countries and some states and provinces in North America have developed setback guidelines during the last two decades (Schauberger and Piringer 1997a and 1997b; Klarenbeek and van Harreveld, 1995; OMAFRA, 1995; Lim et al., 2000; Jacobson et al., 2005; Guo et al., 2005). In Canada, an empirical guideline, entitled the MDS Guidelines, was developed in Ontario in the 1970s and has been incorporated into land use policy (OMAFRA, 1995; MacMillan and Fraser, 2003).

4.5.2.1 Ontario

Ontario's setback regulation was first established in 1976 - *Agricultural Code of Practice*. Most recently, the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA) issued a new set of Implementation Guidelines for MDS (Publication 707) and related software. The new MDS was effective as of January 1, 2007. The models determine the setback distance according to the animal species, animal numbers, and manure handling systems. These guidelines were generated with the help of some science-based information and a lot of personal experience with determining setbacks from livestock operations in the province (MacMillan and Fraser 2003). See Note: Ontario MDS for details of the Ontario setback model.

4.5.2.2 Alberta

The regulations and standards accompanying amendments to the *Agricultural Operation Practices Act* (AOPA) became effective January 1, 2002 (AOPA, 2002). The province-wide regulations and standards govern new and expanding Confined Feeding Operations (CFOs). They give the province responsibility for siting, monitoring and enforcing new and expanding CFOs through the Natural Resources Conservation Board (NRCB). Further amendments based on a targeted review of the legislation were made in 2004 and came into effect on October 1, 2006. The amendments clarify and enhance the province's ability to deal with nuisance, such as odour, noise, dust, smoke or other disturbances resulting from an agricultural operation.

AOPA sets out the Minimum Distance Separation (MDS) required between CFOs and neighbouring residences to reduce nuisance. The minimum separation distance is determined from three major variables: the odour production coefficient; odour objective coefficient; and the dispersion factor. Odour production is measured by Livestock Siting Units (LSU), which are tabulated in the Alberta Standards and Administration Regulation. The value of odour objective coefficient is dependent on what the land is zoned for. The dispersion factor considers the effect of topographical features and meteorological conditions. See Note: Alberta MDS for details of the Alberta setback model.

4.5.2.3 Manitoba

Farm Practices Guidelines for Pig Producers in Manitoba recommends a simple schedule of setback distances for different size hog operations, as measured in livestock (animal) units (AU) (Table 4.3). The schedule is not as comprehensive as those in other jurisdictions, such as Ontario and Alberta, and does not consider many factors that affect odour dispersion and impact. Table 4.4 compares the Manitoba guideline with other two Canadian setback guidelines for two 3000-sow farrowing farms in southern Manitoba (Guo et al., 2006). It can be seen that the distances estimated by the Manitoba guideline is slightly more conservative than the other two.

Table 4.3 Recommended Criteria for Siting Livestock Operations (Farm Practices Guidelines for Pig Producers in Manitoba, MAFRI 2007)

Animal Units ¹ (A.U.)	Maximum Number of Residences ² Within 1.6 km	Minimum Distance ³ (m)			
		From Single Residence		From Designated ⁴ Residential or Recreational Area	
		To Earthen Storage	To Buildings ⁵	To Earthen Storage	To Buildings
10 - 100	18	200	100	800	530
101 - 200	16	300	150	1200	800
201-300	15	400	200	1600	1070
301 - 400	14	450	225	1800	1200
401 - 800	12	500	250	2000	1330
801 -1600	10	600	300	2400	1600
1601 - 3200	8	700	350	2800	1870
3201 - 6400	6	800	400	3200	2130
6401 - 12800	4	900	450	3600	2400
12801 and greater	2	1000	500	4000	2670

¹ Refer to Table 14 for number of animals.

² Number of residences within 1.6 km (one mile) of the center of the facility applies only to new facilities. Expansions of existing facilities and the proponent's residence are excluded.

³ These separation distances apply to new and expanding operations; see Appendix C for imperial units.

⁴ Officially designated areas in a development plan or basic planning statement.

⁵ The distance to buildings includes barns and non-earthen manure storages such as above or below grade structures which may be covered or uncovered.

Table 4.4 Comparison of setback distances among three guidelines (in metres)

Farm*	Odour emission, OU/s	Alberta		Ontario		Manitoba	
		Max	Min	Max	Min	Earthen storage	Buildings
Farm A	174,522	1873	702	1114	557	2400	1600
Farm B	303,112	2345	879	1114	557	2400	1600

*Both Farms A and B are 3000-sow farrowing farms in southern Manitoba. Farm A had a negative air pressure (NAP) covered EMS, and Farm B open EMS.

4.5.2.4 The United States

There are no odour standards at the federal level in the United States. Regulations and management of odour-related annoyance is the responsibility of the state or county. Based on information from two technical papers (Redwine and Lacey, 2000; Sweeten, 1997), Sheffield (2005) presented a summary of odour standards/thresholds in various jurisdictions of the United States (Table 4.5).

For the residential zone, the odour limits range from 0 to 8 D/T and 7 D/T seems to be used by most jurisdictions. The unit D/T refers to "Dilution to Threshold", which is similar to OU (odour unit). It should be noted that the D/T values used in the standards are supposed to be measured with Scentometer or Scentometer-like device (see Note: Scentometer) and may be significantly different from the odour concentrations measured by the modern laboratory olfactometers. Newby and McGinley (2003) reported that a laboratory olfactometer measured a D/T of 110 for the odorous air samples collected whereas a field Scentometer measured 7 D/T. The D/T is related to odour level as follows (Huey, 1960): 2 – Noticeable odour; 7 – Objectionable; 15 – Nuisance; and 31 – Nauseating.

Table 4.5 Odour standards/thresholds in various jurisdictions of the United States (Sheffield, 2005)

States/Organizations	Residential	Commercial	Industrial
ATSDR - Agency for Toxic Substances and Disease Registry	7		
Cedar Rapids, Iowa	4	8	20
Chattanooga, Tennessee	0	4	4
Colorado	7	7	15
Dallas, Texas	2	1	1
District of Columbia	1	1	1
Illinois	8	8	15
Kentucky	7	7	24
Missouri	7	7	7
Nevada	8	8	8
North Dakota	2	2	2
Omaha, Nebraska	4	8	20
Polk County, Iowa	7	7	7
Wyoming	7	7	7

Many states use experience-based setback guidelines to maintain appropriate separation distances between livestock operations and the neighboring residences and communities. The two setback models that have received much attention are Purdue model and Minnesota OFFSET. The Purdue model was developed by researchers at Purdue University (Lim et al., 2000) and it is an empirical model based on the baseline odour emission data and the studies of existing setback guidelines, specifically the Austrian model (Schauberger and Piringer, 1997b) and the Williams and Thompson model (Williams and Thompson, 1986). The Purdue model estimates setback distances from over a dozen of parameters, accounting for the size of operation, building design, manure handling, weather conditions, and topography. See Note: Purdue Model for details.

The Minnesota OFFSET model was developed by researchers at University of Minnesota (Jacobson et al., 2005). The OFFSET, short for “Odour from Feedlots Setback Estimation Tool” is a design and planning tool for predicting the potential downwind odours from livestock facilities. The OFFSET model predicts the level and frequency of odour that can be expected at certain downwind (setback) distances from a livestock operation. Odour annoyance-free frequencies are based on the average weather data for a given odour detection threshold. Odour acceptability measured adjacent to each odour source were correlated to investigate whether a quantitative recommendation of allowable odour level can be determined. See Note: Minnesota OFFSET for details.

4.5.2.5 Europe

Austria

The Austrian guideline is based on a rough estimate of the odour source by considering the following parameters: number of farm animals, their use and the way they are kept, the geometry of air outlet, the vertical velocity of exhaust air, manure handling inside the building, the type of manure storage, and the feeding method (Schauberger et al., 1997; Schauburger and Piringer, 1997a and 1997b). Based on the above parameters, an odour number is calculated, and the separation distance is then estimated by a power function from the odour. The calculated separation distance is further modified to account for dispersion effects.

Belgium

Currently a policy review is under way to establish a concerted policy on odours in Flanders, the Northern part of Belgium. The Flemish Environmental Policy Plan 2002-2006 contains an initiative to define odour exposure standards for 16 sectors of economic activity (pig farms, slaughterhouses, paint application, wastewater treatment plants, and textile plants). The development of policy relies on field panels that determine the maximum distance at which the odour can be detected. This distance and the weather conditions during the field test are then used as input to a Gaussian dispersion model to estimate the emission of the source in ‘sniffing units’. The concept of ‘sniffing unit’ is similar to “odour unit”, but measured in the field rather than in the laboratory.

Germany

In Germany, separate guidelines exist for pig farms (VDI 3471, 1986), cattle (VDI 3473, 1994), and poultry (VDI 3472, 1986). The estimate of separation distances are carried out in three steps. In the first step, the odour source of a livestock farm is assessed by the number of livestock units (live mass of animals normalised by 500 kg). In the second step, manure handling, the ventilation system, the type of feed, and the topography of the site are evaluated by assigning scores to each category. The scores range from 100 for low odour emission to 25 for high odour emission. Depending on the score range, a different power function is used to calculate the separation distance from the number of livestock unit.

Switzerland

In the Swiss guideline (Richner and Schmidlin, 1995), a pollution factor is assessed by the number of animals and a weighting factor is determined based on the annoying potential of the type of animal. The product of these two factors gives the odour load. The standard separation distance is then calculated from the odour load by a logarithmic function and modified by nine factors covering the shape of the site, the sea level, the manure handling system, the type of manure produced, cleanliness of the farm, nutrition, the ventilation system and measures to abate odour release.

The Netherlands

The separation distance is calculated as a function of pig places (number of pigs the animal house is built for) for fattening pigs (Ministrie van Landbouw, 1991). For other species a conversion factor is defined in relation to fattening pigs.

Denmark

In Denmark, an exposure criterion is used, which states that the ground level odour concentration should not exceed 5 to 10 OU/m³, depending on the location (residential or non-residential), at a 99-percentile, with an averaging time of 1 minute.

4.5.2.6 Other Countries

Australia

In Australia, the states have the responsibility for setting air quality policies for odour. The different states have traditionally taken very individual approaches. Recently there appears to be a trend towards convergence. A main development supporting the shift from traditionally qualitative odour regulations to quantitative regulations is the development of an Australian standard for odour measurement, jointly with New Zealand. Most Australian states are expected to adopt this standard, with the exception of Victoria, which so far indicates that it will continue to use its own olfactometry method.

New Zealand

New Zealand's Resource Management Act 1991 imposes a duty upon industries to avoid causing "objectionable" or "offensive" odours to such an extent that they are likely to have adverse environmental effects. Since 1995, New Zealand has established a guideline for managing odour to make this general legal requirement operational: the Odour Management under the Resource Management Act (1995). Most regional authorities also have guidelines in more general common law terms: no objectionable odour at or beyond the property (site) boundary.

4.6 Odour Control Practices/Technologies in Manitoba and in other Jurisdictions

Approaches to odour control may be summarized in the following categories: ration/diet modification, manure treatment, capture/treatment of emitted gases, and enhanced dispersion (NCMAWM, 2001). Each of these mitigation approaches may include various specific technologies. This section discusses some technologies used or tested in Manitoba and other jurisdictions.

4.6.1 Manitoba

4.6.1.1 Negative Air Pressure Cover for Earthen Manure Storage

Earthen manure storage (EMS) is a main source of odour from hog operations in Manitoba. Covering the manure storage is an effective way of minimizing odour emission. Synthetic plastic covers have historically been relatively expensive. In order to ensure the cover is robust enough to withstand wind forces, these systems became too heavy and expensive to be attractive to the livestock industry. Recently, this problem has been overcome by the development of a technology that utilizes negative air pressure (NAP) to anchor lightweight plastic covers (Small and Danesh, 1999). Based on a capital cost of \$6.00 to \$8.00 per square meter, the cover is economically attractive, since the cover has a service life of up to ten years. Zhang et al. (2007) compared odour emissions between an open EMS and NAP covered EMS and reported that average emission rate from the NAP EMS was negligible (1%) in comparison with the open EMS (0.3 vs. 20.3 OU s⁻¹ m⁻²). Additional benefits include isolation of precipitation and the retention of manure nitrogen, factors that offset the cost of the cover.

4.6.1.2 Straw Cover for Earthen Manure Storage

Straw covers may provide a cost-effective and farmer-friendly solution to odour problems associated with EMS, however they have not been adopted on a widespread basis by the hog industry. The straw cover is easy to apply, but will sink over time, and thus reapplication is required. Additional agitation and straw chopping are required for pump-out if straw covers are used. Straw covers also increase the volume of manure that must be transported.

The Prairie Agricultural Machinery Institute (PAMI) conducted a series of projects to develop and assess straw covers for manure storage and showed that good quality barley straw floated well enough to form a cover layer on the liquid hog manure surface (PAMI, 1993). For effective odour control, a layer of straw 150-mm (6-inch) thick is required and reapplication of straw is also necessary. Straw may be applied by an applicator that shreds the straw to small pieces (150 to 200 mm or 6 to 8 in.) and throws the shredded pieces onto the manure surface.

Cicek (2005) compared odour emissions between an open EMS and a straw-covered EMS in two 3,000-sow farrowing operations in southern Manitoba. They observed that odour emission was on average 37.8% lower with the implementation of the straw cover. Clanton et al. (2001) tested the effectiveness of chopped oat straw in odour reduction and measured 47%, 69%, and 76% reduction for the 10-, 20-, and 30-cm (3.9, 7.9, and 11.8-in.) straw thicknesses, respectively.

In terms of costs, a year of 150 mm barley straw at \$1.00/square bale would result in \$0.23/m² per application (Cetac-West, 2002).

4.6.1.3 Biofilters

Biofilters have been tested in both the laboratory and field conditions by many researchers for livestock odour control (Sun et al., 2000, Chang et al., 2004, Hartung et al., 2001, Nicolai and Janni, 2001, Classen et al., 2000). Much of the research and development work in North America was conducted in Minnesota on low-cost, open-bed biofilters in hog operations (Nicolai and Janni, 1997). Odour reduction efficiencies between 75% and 90% were achieved. Janni et al. (1998) tested compost and brush chip biofilters over a ten-month period and achieved odour removal efficiency of 91% and 87%, respectively, hydrogen sulfide removal efficiency of 97 and 96%, and ammonia removal efficiency of 74% and 82%.

The use of biofilters in Manitoba is still at the experimental stage. Mann et al. (2002) tested an open-bed biofilter in a hog operation in southern Manitoba and showed that the biofilters operated reasonably well even in the winter conditions of Manitoba, maintaining temperatures of about +16°C when ambient temperatures varied between +9.2 and -34.2°C. The odour reduction ranged from 56% to 94%. Moreover, open-bed biofilters usually release enough heat to melt any snow that falls on them, providing moisture.

The cost information for Manitoba conditions is not available. Jacobson et al. (1998) estimated the construction, operating, and maintenance costs for an on-ground, open-bed, compost biofilter for a hog production unit, amortized over three years, to be between \$0.50 and \$0.80 per finished market pig for Minnesota conditions. "Costs to install a biofilter include the cost of the materials - fans, media, ductwork, and plenum - and labor to construct, Typically the cost for new construction on mechanically ventilated buildings will be between \$150 and \$250 per 1000 cfm. Annual operation/maintenance of the biofilter is estimated to be \$5 - \$10 per 1000 cfm. This cost includes the increase in electrical costs to push the air through the biofilter and the cost of replacing the media after 5 years. Both capital costs and operation and maintenance costs are quite variable." (Janni, 2000)

4.6.1.4 Shelterbelts

Many hog producers in Manitoba plant shelterbelts around their facilities. Although few quantitative studies have conducted to assess the effectiveness of shelterbelts on odour reduction, shelterbelts have been recommended for odour control in other jurisdictions. After reviewing the literature, Tyndall and Colletti (2000) concluded that shelterbelts have the potential to be an effective and inexpensive odour control device particularly when used in combination with other control methods for added effectiveness. The USDA Natural Resources Conservation Services (NRCS, 2004) published an information sheet that summarized six ways that shelterbelts can reduce the effects of livestock odour and improve visual perception of production buildings:

- Dilution and dispersion of gas concentrations of odour by a mixing effect created by windbreaks.
- Deposition of odourous dusts and other aerosols (like snow fencing) to the windward and leeward sides of windbreaks.
- Collection and storage (sinks) within tree wood of the chemical constituents of odour pollution.
- Physical interception of dust and aerosols odour particles on leaves, needles and branches.

- Containment of odour by placing windbreaks fore and/or aft of the odour source.
- Aesthetic appearance:
 - ◆ Trees create a visual barrier to livestock barns
 - ◆ Trees can make cropped fields and pastures more pleasing to look at
 - ◆ Trees represent an 'environmental statement' to neighbors that the producer is making every effort to resolve odour problems in as many ways as possible.

In a study conducted to measure odour plumes on hog farms in southern Manitoba, Adam (1999) observed that trees (bush) did not appear to significantly diminish odour travel distances. But he did suggest that trees (bush) generally deflect the wind, and therefore may be useful to cause dilution through mixing over shorter distances through strategic placement.

4.6.1.5 Solid Separation

Solid separation technologies have attracted much attention lately in Manitoba, primarily as a means of removing phosphorus from hog manure. Fresh hog manure contains high levels of suspended solids that are predominantly organic matter. It is the anaerobic decomposition of organic matter that produces odour compounds. Removing solids from the manure means reducing organic matter in manure storage, and therefore the process should reduce odour emission from manure storage. However, the quantitative effects of solid removal on odour reduction have been seldom reported. Some researchers (e.g., Heber et al. 1999) believed that solid separation for liquid manure was somewhat effective in reducing odour, but others observed that the effect was insignificant (Zhu et al., 2001). The discrepancies are probably attributed to the effect of particle sizes of manure solids. A study conducted by Zhang and Westerman (1997) showed that most of odour-generating compounds are contained in fine solid particles. Also, in a study related to the particle sizes distribution in hog manure, Jett et al. (1974) reported that approximately 83% of the crude protein and 93% of fat in the original manure samples were contained in the material which passed the 0.25 mm sieve. The results of a study conducted by Hill and Tollner (1980) showed that 40% of VS (volatile solids), 47% of COD (chemical oxygen demand), and 70% TKN (total Kjeldahl nitrogen) passed the 0.105 mm sieve. All of the above mentioned parameters are odour related. Research on the relationship between the solid particle size in manure and major odorous compounds conducted by Zhu et al. (2001) showed that most odorous compounds (VFA and BOD) were contained in manure particles smaller than 0.075 mm. For odour reduction, removal of fine particles (smaller than 0.075 mm) seemed necessary.

Most solid separation technologies that are currently used are not efficient in removing fine particles. Therefore, instantaneous odour reduction cannot be expected from the solid-liquid separation. For example, very little or no odour reduction can be achieved with mechanical screens (MWPS-18, 2002). The significance of solid-liquid separation process is, however, that it is a necessary pre-treatment for some other treatment techniques, such as drying and composting. Removing solids lowers the organic strength in the liquid portion, and this in turn would reduce the energy requirement for the further treatment of the liquid portion of manure.

4.6.1.6 Manure Injection

Odour from land application of manure used to be the main cause of odour-related complaints by the public. A shift to injection-spreading of manure seems to result in less odour complaints traceable to the land application than to animal production facilities and manure storage units (Jacobson et al., 1998). Zhang et al. (2001a) reported that injection of manure into soil caused little odour emission from soil. The emission rate measured from the soil with no manure applied was almost the same as that from the manured soil (3.6 vs. 4.0 OUs⁻¹m⁻²). Air samples collected downwind at the ends (or sides) of the fields on which manure was being applied showed very low odour levels (60 OU on average). Chen et al. (2001) compared four different techniques for applying liquid manure on grassland to examine the impact of each technique on the environment and agronomic responses for Manitoba conditions. These techniques were manure injection, sub-canopy banding using a sleighfoot, incorporation using an aerator, and surface banding using a dribble bar. They found that among the four techniques, the injection resulted in lower ammonia and odour concentrations on the land surface immediately after manure application.

From a crop productivity standpoint, mixing manure nutrients with soil through injection or incorporation often results in greater yields and reduced nutrient losses in runoff and volatilization to the environment (Sawyer et al., 1991; Schmitt et al., 1995; Warnemuende et al., 1999).

For grassland and zero-tilled fields, conventional injection equipment may create too much soil disturbance. Low disturbance, shallow injection equipment has been developed for these conditions (PAMI, 1999).

4.6.1.7 Dietary Manipulation

Zhang et al. (2002) conducted a thorough review on the dietary manipulation for odour reduction. Here are excerpts from their report.

“The main compounds associated with swine odours are products of degradation of excess dietary protein intake. Furthermore, it has been estimated that about 65 to 70% of dietary nitrogen intake by growing pigs is excreted (Lenis and Jongbloed, 1999). This means that considerable reduction in nitrogen excretion can be achieved by carefully matching dietary nitrogen supply with requirements (Sutton et al., 1996, 1999; Jongbloed and Lenis, 1998). Sulfur excretion can also be reduced by carefully managing intake levels, either through the diet (sulfur amino acid levels) or controlling sulfate levels in drinking water. However, it should be noted that little research has been done to determine how such modifications can influence odour emission from swine barns and manure storage facilities. It, therefore, is essential to establish the effect of such modifications on odour emission from swine manure.”

“Dietary manipulations have been shown to potentially reduce odour generation in hog operations. Most of the research to date in the area of dietary manipulations for hog odour control has been done elsewhere, primarily in the Netherlands and the Midwestern region of the U.S. Limited Canadian or Manitoba research is available. Fortunately, however, the majority of the research can be translated to the Canadian situation either directly or with further refinement. However, differences in primary cereal crops used (corn vs. barley or wheat) and the associated differences in chemical composition (NSP content and types) warrants further consideration”

“The bulk of studies examining the impact of diet on manure odour have focused on the impact that specific dietary regimens have on ammonia emissions from hog. Several dietary approaches have been taken, including (1) the use of low protein-amino acid supplemented diets (see above), (2) the increase of dietary non-starch polysaccharide levels to shift circulating urea towards microbial protein synthesis, (3) the use of binders or ion exchange compounds to sequester ammonia, and (4) dietary acidifiers to reduce ammonia volatilization.”

Although dietary manipulation is studied and used for reducing odour and ammonia emissions, little information is available in the literature in terms of quantifying odour reduction in commercial scale production facilities.

4.6.2 Technologies Used Elsewhere

4.6.2.1 Manure storage covers

Covering manure storage is one of the most effective and feasible ways of controlling odour. Three types of cover technologies have been used: solid covers (concrete, steel, and wood), impermeable covers (plastic), and biocovers (straw, straw and geotextile combination, etc). Comparisons of various types of covers are summarized in Table 4.6.

Solid covers can almost completely eliminate odour emission during storage, but are very expensive. Impermeable floating plastic covers may provide more than 99% odour reduction (Heber et al., 1999), and are commercially available. A biocover is usually composed of organic material (e.g., wheat straw, barley straw, chopped cornstalks, sawdust, wood shavings, rice hulls, etc.) that is blown onto the surface of the storage in a layer about 250 mm thick (Bicudo, 1999). Straw may be mixed with vegetable oil to keep the cover afloat longer (Barrington, 1997; Schmidt, 1997). The lifespan of a biocover can also be extended by combining it with geotextile fabric or floatation devices, such as the polystyrene pellets.

Biocovers provide an effective way of reducing odour emission from open manure storage units (Clanton et al., 2001; Mannebeck, 1985; Miner and Pan, 1995). Researchers at the University of Minnesota assessed the effectiveness of straw and geotextile in odour reduction and reported that the reductions in odour, ammonia and hydrogen sulfide emissions were 75%, 65% and 90%, respectively (Bicudo et al., 2002b). Biocovers may also act as biofilters and reduce the concentration of odours and gases in air which diffuse through the cover. Straw can be wetted with manure to make it more biologically active and promote biofiltration (Zhang et al., 1999).

Mannebeck (1985) estimated the useful lives of various covers: straw - six months; plastic pellets - two years; mats and tarpaulins -10 years; and lightweight roofs - 15 years. Jacobson et al. (1997) indicated that a straw cover can float for several months, depending on rainfall, depth of manure, surface area of the storage unit, winds, manure characteristics, etc. Biocovers are relatively inexpensive, as the raw materials are readily available. Heber et al. (1999) reported that 8” to 12” chopped straw (barley, wheat, oats or brome) could provide 50% to 80% odour reduction at a cost of about \$0.01- \$0.02 per square foot. This cost estimate may vary depending on the local price of straw. Jacobson et al. (1999b) estimated the cost of a 12” straw cover as being \$0.08 per square foot, not including the application cost. Zhang et al. (1999) estimated the cost of a straw cover to be between \$0.10 to and \$0.50/m². Bicudo (1999) reported the cost of a

straw cover on an open manure storage unit to be between \$1.07 and \$1.61 per m² (assuming 6 months of useful operation).

Table 4.6 Comparison of covers for manure storage (Cetac-West, 2002)

Type	Material	Strength	Weakness	Cost
Solid Emission Covers	Steel, concrete or wood	Almost complete odour control Long lasting	Expensive	Concrete cover: \$50,000
Flexible Emission Covers	A tarp and anchoring devices Dome style covers also require a low pressure blower	Excellent odour control – 95% Long lasting – 10 to 15 years Easy to remove	Expensive Some maintenance required	\$10,000 plus \$200/year in operating costs
Biocovers (Straw)	Quality barley straw works the best Blown over storage using a forage harvester Peat moss can be added to improve nutrient intake in the field	Inexpensive Effective odour control	Straw tends to sink requiring multiple applications Straw may interfere with pumping equipment	150 mm barley straw @ \$1.00/square bale = \$0.23/m ² per application
Supported Biocovers	Polystyrene pellets may be applied prior to the barley Oil can be added to the straw as well	Effective odour control Floatation measures may reduce straw applications Polystyrene pellets can be collected and reused	Recovery of floatation devices can be difficult More expensive alternative	25 mm polystyrene + 125 mm barley straw = \$2.45/m ²

4.6.2.2 Anaerobic digestion

Anaerobic digestion reduces odour emission by converting odorous intermediate products of anaerobic decomposition into odourless end products of carbon dioxide and methane. Because anaerobic digestion occurs in enclosed digesters, therefore intermediate odorous compounds produced during digestion are kept in the digesters. See Chapter 6 for details on anaerobic digestion.

4.6.2.3 Manure additives

“Many chemical and biological products are being marketed as manure additive for odour control. Research reported in the literature is mostly focused on the evaluation of manure additives, with little on the mechanism of action or the development of products. Researchers have reported conflicting results on the effectiveness of using manure additives. This is partially due to the lack of universally accepted protocols for evaluating manure additives. Although some existing manure additive products have been shown to be effective in odour reduction under laboratory conditions, they may not perform well in actual production facilities. Many other factors, such as the building, ventilation, manure handling, feeding, and the overall management practice, may conceal the effect of manure additives in actual production facilities.” (Zhang et al., 2002)

4.6.3 Other Technologies

- ◆ **Straw bedding.** Solid manure produces less odour. Various types of straw bedded systems are used in Europe. Hoop structures with deep straw bedding are used in US and Canada.
- ◆ **Composting.** The end product from composting has a non-offensive soil-like smell. However, the composting process itself may generate odour if the anaerobic condition occurs.
- ◆ **Surface aeration.** Maintaining the aerobic condition in the manure storage would eliminate odour production. The operating costs are very high.
- ◆ **Oil sprinkling.** Dust particles are carriers of odour compounds. Oil sprinkling reduces dust as well as odour inside the barn, thus reducing the odour emission from the barn.
- ◆ **Ozonation.** Increasing the ozone level in the barn to oxidize odour compounds, thus reducing the odour emission from the barn. Ozone is a regulated air pollutant. However, the concentrations used for ozonation are much less than the allowable limits (US EPA: 0.98 and 0.12 ppm for 1- and 8-hour averaging times).
- ◆ **Urine and feces separation.** Ammonia is formed mainly because of the contact between urea in urine and urease (an enzyme that catalyzes the hydrolysis of urea into carbon dioxide and ammonia) contained in the feces. Therefore, separating urine and feces would reduce ammonia emission.

4.7 RECOMENDATIONS

4.7.1 Mitigation Strategies

4.7.1.1 Adopting the best management practices (BMP)

Odour emissions from hog operations reported in the literature varied tremendously among similar facilities (e.g., Zhang et al., 2001a). This indicates that the management

practices in the day to day operation of a facility may have significant effect on odour emission from the facility.

Good housekeeping is a key in reducing odour emission from hog buildings. Keep floors clean and dry; keep manure and feed dry (<40% moisture); keep animals clean; prevent water leakage and feed spillage; frequent flush/wash/scrape; maintain adequate environment (temperature); minimize dust; and keep manure pits recharged properly (2-3 in of water in shallow pits).

A properly maintained barn environment is also a key in odour control. Adequate temperature and relative humidity should be maintained in the barns to prevent “dirty pens”. Furthermore, the barn should be adequately ventilated to remove airborne dust, gases, and bioaerosols from the building.

The in-barn manure handling system plays an important role in odour management. An effective manure handling system should promote quick separation of manure from animals to minimize odour generation. Properly designed slatted floor systems provide an effective way of separating manure from animals with minimum efforts.

Timing the land application of manure is a key to avoid conflicts with neighbours. Apply manure when neighbours are less likely to be outdoors and at the unstable atmospheric conditions (high solar radiation and wind speed) for better dispersion (dilution).

4.7.1.2 Maintaining adequate separation distances

Maintaining adequate separation distances between the hog operation and the neighbouring residences/communities provides the first line of defense to odour problems.

4.7.1.3 Adopting technologies

The technology adoption should not be required unless the BMP and separation distance approaches fail in addressing the odour problems. The following technologies are considered to be effective in odour reduction and economically feasible.

- ◆ **Manure storage covers.** Both straw and synthetic (plastic) covers are effective in reducing odour from earthen manure storage.
- ◆ **Biofilters.** Open-bed biofilters are effective in reducing odour and other gas (ammonia and hydrogen sulfide) emissions from hog barns, and economically feasible.
- ◆ **Dietary manipulation.** Dietary manipulation may provide an integrated approach to odour problems, i.e., reducing odour and other “waste” elements (N and P) while improving the feeding efficiency.
- ◆ **Shelterbelts.** Although the effectiveness of shelterbelts in odour control is difficult to be quantified, theoretically shelterbelts may improve odour dispersion, reduce odour travel distance, and trap odour-carrying dust particles. More importantly, shelterbelts create a visual barrier to livestock facilities, thus alleviating perceived odour problems.

4.7.2 Policies, Regulations, Standards, etc.

4.7.2.1 Odour credits

Technologies exist for mitigating odour problems. For example, the combination of an impermeable cover on manure storage, a biofilter to treat barn exhaust air, and injection of manure would eliminate most of the odour problems in a hog operation. But the challenge is the affordability of using these technologies. More producers will adopt technologies if there are financial incentives for them to do so. An odour credit system might be worth considering. A similar economic incentive system – Transferable Discharge Permit System, is discussed in Chapter 9. Also reported in Chapter 9, consumers are willing to pay for ozone reduction in Montreal and for pork products that originated from farms with shelterbelts for odour mitigation. This indicates that cost of abating odour may be shared by consumers.

4.7.2.2 Separation distance guideline

The current Manitoba separation distance guideline is simple to use, but does not have any flexibility to account for many factors that influence odour emission and dispersion, such as the use of technologies, the topographic conditions, etc. A dispersion theory-based guideline, integrated with odour impact models, should be established.

4.7.2.3 Odour measurement and assessment standards and procedures

Standards and procedures should be developed or adopted from other jurisdictions (such as the European Union) for measuring odour emissions from barns and manure storage. The standards should also cover the methods and procedures for assessing the odour impact downwind from livestock operations.

4.7.2.4 Educational programs

Educational programs should be developed to provide technical assistance to producers to address odour and other air quality issues in livestock operations. For example, developing Fact Sheets to assist producers in selecting odour abatement technologies.

4.8 INFORMATION GAPS

- There are insufficient data on potential health effects associated with odour downwind from hog operations for establishing suitable acceptability criteria for community level exposure to odour.
- Data on emission rates and characteristics of odour from hog operations are lacking. For example, very limited information is available on the emission rates as a function of diurnal, seasonal and climatic variations as well as design and management practices.
- Data on downwind distribution of odour and other air contaminants (dust and VOCs) are lacking.
- There is insufficient information on economic analysis of odour mitigation technologies for Manitoba conditions.

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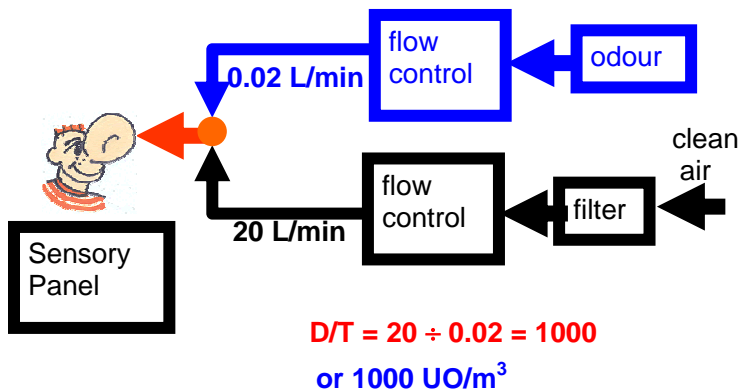
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4.10 TECHNICAL NOTES

Note: Olfactometer

Olfactometry is a psychophysical technique that utilizes the human sense of smell, i.e., the olfactory senses, to determine odour concentrations. An olfactometer is a dilution apparatus which mixes odorous air in specific ratios with clean air for the presentation to a panel of assessors. The diluted odour is presented to the panelists by continuously injecting a stream of odorous air at a known flow rate and pressure into a stream of the filter air (diluent), also flowing at a predetermined rate (Watts, 2000). The ratio of the volumetric flow rate of diluent to that of the odorous air at which odour can be detected is the dilution to detection threshold (D/T) of the odorous air. The D/T is used as a unit of odour concentration, OU, or OU/m^3 .



In operating an olfactometer, trained odour assessors sniff the diluted odour sample as it is discharged from one of three sniffing ports and must select one of the three different from the other two. Each assessor declares to the operator (panel leader) if the selection was a "guess", "detection" or "recognition" by pressing the corresponding button on the machine. The assessor then sniffs the next set of three sniffing ports, one of which also contains the diluted odor sample. However, this next set presents the odour at a higher concentration. The assessor continues to additional sets of three sniffing ports until he/she correctly detects the odour at two consecutive dilution levels. Generally, the first sample (mixture) presented to an odour panel is diluted with a very large volume of clean air. At this stage of the analysis, the diluted odour should be undetectable by the human nose.

It is typical to screen and select panelists based on their sensitivity and consistency. N-butanol is often used as a reference odour in the screening process. According to the European Union standards (CEN, 2003), a person is qualified as a panelist if his/her sensitivity to n-butanol is between 20 and 80 ppb.

Note: Odour Intensity Measurement by Human Assessors

Odour intensity is a measure of the human response to an undiluted odour. A common way of measuring odour intensity is comparing the intensity of an odour to the intensities of different but known concentrations of a reference odourant. ASTM (1999) recommends that successive concentrations of the reference odourant are greater than the preceding levels by a step factor of two. Odour intensity is obtained when a match is found between the intensity of the odour and the intensity of one of the concentrations of the reference odourant. N-butanol is commonly used as the reference odorant. Table below show a standard 8-point referencing n-butanol solutions based on the ASTM standards (ASTM, 1999). The three- and five-point scales are also commonly used in measuring livestock odour.

Table A1. Eight-point odour intensity referencing scale

Intensity level	n-butanol in water (ppm)	Annoyance scale
0	0	no odour
1	120	not annoying
2	240	a little annoying
3	480	a little annoying
4	960	annoying
5	1940	annoying
6	3880	very annoying
7	7750	very annoying
8	15500	extremely annoying

Various methods have been used to train sniffers to use the n-butanol odour intensity scales. St. Croix Sensory Inc. (Stillwater, Minnesota) train and “certify” Nasal Rangers for odour sniffing. At University of Manitoba, a training process was composed of a series of 6 individual sessions. The focus of the training sessions is to have the sniffers “memorize” the referencing scale which they would be using in the field. First of all, each sniffer is provided with a set of 8 n-butanol samples in 45 mL glass bottles with Teflon coated lids, and they sniff the samples from #1 to # 8 for several times. In between each sample, they wear carbon filtered masks for 10 – 20 seconds to “rinse” their noses. Then each sniffer is given 3 to 6 coded samples of known intensity (but unknown to the sniffer). They evaluate one sample at a time, assign a scale (1 to 8) to this sample, and record the scale on a ballot. Those who correctly rate the sample are asked to check with the standard solution bottle of n-butanol and sniff the sample again to reinforce the rating. Those who incorrectly rate the sample will sniff both the standard and the coded sample to “feel” the difference.

Once the sniffer are trained, they are ready for field odour sniffing. The sniffers “calibrate” their noses using the standard reference n-butanol samples before leaving for the field. To prevent nose from being “saturated”, the sniffers wear the carbon filtered masks. They only remove the masks briefly



to sniff odour. For every sniffing, the sniffer record the odour intensity (1 – 8) and odour description on a field data recording sheet

Note: Scentometer

A Scentometer measures the odour concentration of ambient air directly in the field (NCMAWM, 2001). Odorous ambient air and filtered (odourless) ambient air are drawn into the Scentometer and mixed each time the odour assessor sniffs from a sampling port inserted in his/her nostrils. The dilution ratio of the mixture is determined by the size of openings the odorous air flows through as it enters the Scentometer. The original Scentometer produced four (4) dilutions and the modified Scentometer produced six (6) dilutions, or Dilution to Threshold (D/T): 2, 7, 15, 31, 170, 350 (McGinley et al., 2000b). Scentometers have limited levels of dilution and usually one sniffer operates the instrument. They are, therefore, less accurate than dynamic-dilution olfactometers. It is also difficult for the operator to avoid breathing odorous air before the Scentometer is used. Therefore, the operator may experience odour fatigue when using the scentometer in the field.

Note: Particular Matter

The behavior of dust (or particular matter, PM) in the atmosphere and the effect on human health are dependent on the particle size, measured by the aerodynamic diameter in μm . Airborne particles range in size from 0.001 to 500 μm . Particles larger than 20 μm are airborne for relatively short time. Particles smaller than 10 μm are respirable, i.e., they can penetrate beyond terminal bronchioles to gas exchange region of the lungs. The term “total dust” refers to particles up to 20 μm in aerodynamic diameter. The term “PM₁₀” is used for particulate matter with an aerodynamic diameter of up to 10 μm , and PM_{2.5} for an aerodynamic diameter of up to 2.5 μm . The concentration limits of PM in the US EPA National Ambient Air Quality Standards (NAAQS) are: PM₁₀: 150 $\mu\text{g}/\text{m}^3$ (24 h average); PM_{10-2.5}: 70 $\mu\text{g}/\text{m}^3$ (24 h average); and PM_{2.5}: 15 $\mu\text{g}/\text{m}^3$ (annual average) and 35 $\mu\text{g}/\text{m}^3$ (24 h average). The proposed CWS (Canada-Wide Standards) for PM_{2.5} is 30 $\mu\text{g}/\text{m}^3$ (24 h average), and is to be achieved by the year 2010.

Note: Ventilation Rate

The odour emission rate from a hog building is calculated as the product of the odour concentration on the exhaust air and the ventilation rate. An important issue associated with measuring odour emission rates from hog buildings is quantifying building ventilation rates. Ni et al. (1999a and b) used a calibrated rotating impeller to measure ventilation rates in mechanically ventilated buildings. Naturally ventilated buildings or buildings with a combination of mechanical and natural ventilation require a more complicated method for calculating ventilation rates. One method is to measure temperatures and CO₂ concentrations for the heat balance and CO₂ balance, respectively (Heber et al., 2001). The use of an artificial tracer gas such as SF₆ is considered to be much preferred over the heat balance or CO₂ method (Phillips et al., 2001).

Note: Dispersion

Dispersion of air pollutants in the atmosphere has been widely researched and used for predicting the concentration of pollutants downwind from industrial sources. The Gaussian plume theory is the most widely air dispersion (diffusion) theory and forms the basis of several commercially available atmospheric dispersion models. The general form of the Gaussian dispersion equation is given as follows (e.g., de Nevers, 2000):

$$c = \frac{Q}{2\pi u \sigma_y \sigma_z} \exp \left[- \left(\frac{y^2}{2\sigma_y^2} + \frac{(z-H)^2}{2\sigma_z^2} \right) \right] \quad (A1)$$

where: C = downwind pollutant concentration, such odour concentration
 Q = emission rate
 y and z = coordinates (y – cross wind; z – elevation)
 u = horizontal wind speed
 H = effective emission (stack) height
 σ_y and σ_z = dispersion coefficients

According to the Gaussian equation (A1), the pollutant concentration in the atmosphere follows a normal distribution in both the crosswind and vertical directions, with standard deviations equal to the dispersion coefficients σ_y and σ_z , respectively. For a given emission rate Q and release height H, the distribution of pollutant (odour) concentration downwind from a source is dependent on the two dispersion coefficients and the wind speed. Various empirical equations and charts have been developed to determine the dispersion coefficients, as functions of the atmospheric stability and downwind distance. For example, the US EPA Industrial Source Complex Short Term (ISCST) dispersion model uses the Pasquill-Gifford dispersion parameters:

$$\begin{aligned} \sigma_y &= 0.84678x \tan(a - b \ln x) \\ \sigma_z &= cx^d \end{aligned} \quad (A2)$$

Where x is the downwind distance, and constants a, b, c and d are found from charts for various atmospheric stability conditions. The atmospheric stability is commonly described by the Pasquill (1961) classification scheme (Table A2), which defines a series of stability classes based on wind speed, insolation, and cloudiness. Class F represents the most stable conditions and results in the lowest values of dispersion coefficients. Therefore, the highest downwind concentration would occur under Class F. Jacobson et al. (2000) found that most odour events occurred under stable weather stability classes E and F at low wind speeds with odour detection up to 4.0 km (2.5 miles) from the livestock odour source.

Table A2. Pasquill atmospheric stability classes.

Wind (at 10m)	Insolation			Night	
	Strong	Moderate	Slight	Thinly > 4/8	< 3/8 cloud low
<2	A	A-B	B	-	-
2-3	A-B	B	C	E	F
3-5	B	B-C	D	D	E
5-6	C	C-D	D	D	D
>6	C	D	D	D	D

A: strongly unstable
 C: slightly unstable
 F: moderately stable

B: moderately unstable
 E: slightly stable
 D: neutral (overcast)

Note: Schiffman

Schiffman et al. (1995) examined the effects of long-term exposure to odours by interviewing and testing a group of 44 people living near hog operations and 44 control subjects. They used the Profile of Mood States Questionnaire to measure the impact of the hog odours on mood. The responses are grouped into six factors: tension/anxiety, depression/dejection, anger/hostility, vigor/activity, fatigue/inertia, and confusion/bewilderment, as well as a score for total mood disturbance (TMD). They found a significant difference ($p < 0.0001$) between the control group and the resident group near hog operations. They showed higher scores for tension, depression, anger, fatigue, and confusion, and lower scores for vigor than the control group. The total mood disturbance was also greater for the group that lived near the hog operations. They discussed the possible causes of altered mood as being:

- the unpleasantness of the sensory quality of the odour – fecal odours are considered disgusting in most cultures (including North America) and the odours from the barns and lagoons of these operations are definitely fecal in nature.
- the intermittent nature of the stimulus. The odour is intermittent, depending on wind and climate conditions and so is out of the control of the persons living in its vicinity.
- learned aversions to the odour
- potential neural stimulation of immune responses via direct neural connections between odour centers in the brain and the immune system. They discuss the possibility that the brain structures involved in smell (limbic forebrain, hypothalamus, and other odour projection areas of the brain) can directly affect the individual's immune status.
- direct physical effects from molecules in the plume, including nasal and respiratory irritation
- possible chemosensory disorders
- unpleasant thoughts associated with the odour – which include fear of loss of property values, environmental concerns (manure load on fields and water table contamination), loss of enjoyment of life and property and concerns over the effect on health.

Note: Dispersion Models

Odour dispersion models are computerized mathematical tools used to predict the occurrence (frequency and duration) and concentrations of odours at any distance (location) from livestock odour sources. The commonly used dispersion models are either based on the Gaussian plume or Lagrangian particle trajectory theory. Gaussian plume models can be further subdivided into the steady-state models and the puff models. Gaussian plume dispersion models are relatively easy to use, and have simple and a small number of input requirements. These features make them widely used and well understood in terms of their applications and limitations. The applicability of Gaussian plume dispersion models to ground level odour emission from agricultural sources has been studied by some researchers (Carney and Dodd, 1989; Zhu et al. 2000a; Guo et al., 2003). Other researches have shown improved odour dispersion modeling by using Lagrangian particle models (Godfrey and Scire, 2000; Ormerod, 2001). Lagrangian particle models are generally capable of simulating more complex atmospheric dispersion by accounting for horizontally and vertically variable meteorological fields. A widely used Lagrangian particle model is FLEXPART (Stolh, 2002). A Lagrangian particle model - AUSTAL2000 has been developed to replace Gaussian plume models as the regulatory dispersion model in Germany (VDI, 2000).

In North America, commonly used dispersion models are ISCST3 (US EPA) (or its replacement AERMOD) (steady-state Gaussian) and INPUFF-2 (Gaussian puff). **ISCST3** (Industrial Source Complex Short Term) model is a steady-state Gaussian plume dispersion model developed by the US EPA. It can be used to assess pollutant concentrations from wide variety of sources associated with industrial complex. This model is specially designed to support the US EPA regulatory modeling programs, and is widely used in North America.

INPUFF-2 (INtegrated PUFF) model is based on the Gaussian puff theory and was originally developed by the US EPA, but is now marketed by a commercial company (Bee-Line Software Co., Asheville, NC). INPUFF-2 predicts atmospheric dispersion of pollutants released over a short time period, or dispersion of a "puff" of the pollutant released into the atmosphere. The model can simulate the dispersion of airborne pollutants from semi-instantaneous or continuous point sources, and it can also handle multiple point sources and multiple receptors at the same time. Researchers suggested that puff models might provide better predictions of odour concentrations downwind from livestock operations (McPhail, 1991; Gassman, 1992). INPUFF-2 allows for different time intervals in dispersion simulations as determined by the user rather than the hourly interval as required by other air dispersion models (Petersen and Lavdas, 1986).

Dispersion models require inputs of odour emission rates from various sources on livestock operations and historical weather data.

Note: Ontario MDS

Ontario MDS- II has separate procedures for buildings and for manure storage units. The building separation base distance is defined as the product of the following four factors (OMAFRA, 1995):

$$F (m) = \text{Factor A} \times \text{Factor B} \times \text{Factor C} \times \text{Factor D} \quad (\text{A3})$$

where: Factor A = tabulated value as function of type of animal (range of values from 0.65 for broiler chickens to 1.1 for adult mink). The value is 1.0 for swine barns.

Factor B = tabulated value as function of number of livestock units (LU), LU (range from 107 for 5 LU to 1,455 for 10,000 LU). For swine, five sows or boars, 20 nursery pigs, or 4 feeder hogs make up 1 LU.

Factor C = tabulated value as function of % increase in animal numbers (range from 0.7 for 0 to 50% increase to 1.14 for 700% increase or new facility); and

Factor D = tabulated value as function of type of manure system (solid = 0.7 and liquid = 0.8).

Then this base separation distance will be adjusted by a neighboring land use factor, i.e. Factor E. Factor E is 1 for nearest residence and areas zoned for agriculturally related commercial use, or 2 for areas zoned for residential, commercial or urban areas. The final distance from the barns required will be:

$$\text{Distance (m)} = F \times \text{Factor E} \quad (\text{A4})$$

The manure storage separation distance is a tabulated value that is a function of the base building distance F and the type of manure storage system (covered, open solid and runoff, open liquid tank and runoff, and earthen liquid and runoff). Manure storage separation distances in MSD-II vary from a minimum of 40 m to a maximum of 550 m. After this distance is obtained, Factor E is then used to adjust this base distance to required setback distance from the manure storages according to the neighboring land use.

Note: Alberta MDS

The Alberta MDS model is a modified version of the Ontario MDS- II. The minimum separation distance (MDS) is determined from the Odour Production (OP), Odour Objective (OB), Dispersion Factor (DF), and Expansion Factor (EF) (1 for new facilities) as follows (AOPA, 2002):

$$\text{MDS (m)} = \text{OP}^{0.365} \times \text{OB} \times \text{DF} \times \text{EF} \quad (\text{A5})$$

Odour production (OP) is measured by Livestock Siting Units (LSUs), which takes into account the number of animals or the size of the operation (not animal units), the nuisance value of the livestock species (Factor A), the technology of production system or the manure system (Factor B), and the amount of manure produced by the animals (MU Reciprocal).

$$\text{OP} = \text{LSU} = (\text{Factor A}) \times (\text{Factor B}) \times (\text{MU Reciprocal}) \times (\text{No. of Animals}) \quad (\text{A6})$$

Values of Factor A, Factor B and MU are tabulated for different livestock categories and types in the Alberta Standards and Administration Regulation (AOPA, 2002). Factor A measures the nuisance of the livestock species and addresses the fact that some species produce more odours than others. Factor B accounts for the type of manure system used in the operation and the effect that it has on reducing the odour nuisance level. As new technologies for reducing odour production evolve, this value can change. However, this information must be approved and proven to the NRCB, otherwise the values listed in the Standards are to be applied. MU Reciprocal measures the amount of manure produced by the animal and is expressed as 1/AU (Animal Unit: 454 kg or 1000 lb of live weight).

Odour objective (OB) describes the sensitivity or assumed tolerance level of neighbouring land uses and is based on the four zoning Categories. They are:

- Category 1: Land zoned for agricultural purposes such as farmsteads, acreage residences, etc.
- Category 2: Land zoned for non-agricultural purposes such as country residential, rural commercial businesses, etc.
- Category 3: Land zoned as large scale country residential, high use recreational or commercial purposes as well as from the urban fringe boundary or land zoned as rural hamlet, village or town which has an urban fringe.
- Category 4: Land zoned as rural hamlet, village or town without an urban fringe.

The Odour Objective is determined as follows:

$$OB = (\text{Category Base Distance}) \times (\text{Constant } 1.2) \times (\text{Initial Siting Factor } 1.14)$$

The OB values are determined to be 41.04, 54.72, 68.40, and 109.44 m for the four Categories, respectively.

Dispersion factor (DF) allows for a variance to the MDS based on unique climatic and topographic influences at the site that are proven to change the dispersion of odour. The conditions listed below must be proven to demonstrate a reduction and must be approved by the NRCB. If there is no evidence of a reduction, a value of 1.0 will automatically be applied.

Note: Purdue model

1.1.1. The equation for estimating setback distances has the form of (Lim et al., 2000):

$$SD = 6.19 \times F \times L \times T \times V \times (A_E \times E + A_S \times S)^{0.5} \quad (A7)$$

- where:
- SD = setback distance (m)
 - F = wind frequency factor, 0.75 to 1.00
 - L = land use factor, 0.5 to 1.0
 - T = topography factor, 0.80 to 1.00
 - V = orientation and shape factor, 1.00 to 1.15
 - E = building odour emission, E = N × P × B (OU/s)

N = number of pigs
 P = odour emission factor (OU/s-pig)
 B = building design and management factor, $B = M - D$
 M = manure removal frequency, 0.50 to 1.00
 D = manure dilution factor, 0.00 to 0.20
 S = odour emission from outdoor storage, $S = C \times G$ (OU/s)
 C = odour emission factor for outside liquid manure storage (50 OU/s-AU)
 G = animal units (AU) (500 kg of pig mass)
 A_E = odour abatement factor for buildings, 0.30 to 1.00 (no odour abatement measure)
 A_S = odour abatement factor for outside liquid manure storage, 0.30 to 1.00 (no odour abatement measure).

1.1.2.

1.1.3. Note: Minnesota OFFSET

1.1.4. Odour emission is quantified by odour emission numbers for livestock production facilities, and emission reduction by means of various odour control technologies are also accounted for in the model. The total odour emission factor is calculated as (Jacobson et al., 2005):

$$E = \sum_{i=1}^n E_i = \sum_{i=1}^n (E_{ei} \times A_i \times f_{ci}) \quad (A8)$$

where: E = total odour emission factor from an animal production site, dimensionless,
 E_i = odour emission factor from source i, dimensionless,
 E_{ei} = odour emission number per unit area from source i,
 A_i = area of source i (m^2),
 f_{ci} = odour control factor for source i, ranging from 0.1 to 0.6 for different odour control technologies such as biofilters, various basin covers, and oil sprinkling; $f_{ci} = 1$ if no odour control technology is used,
 n = total number of odour sources, and
 i = running index.

The odour emission number E_e for a source may be obtained from tables for various livestock operations and manure storage systems (Jacobson et al. 2005). The tabulated odour emission numbers were based on the measurements from over 200 sources on 80 farms in Minnesota between 1997 and 2001. However, these values may not be valid for other geographic areas (Jacobson et al., 2005). Alternatively, the odour emission factor E_i may be determined from the actual measured odour emission rate as follows:

$$E_i = K \times Q_{od} \quad (A9)$$

where: K = scaling factor, and
 Q_{od} = odour emission rate (OU/s- m^2).

Based on the dispersion simulations, the suggested value for scaling factor K is 35 for building emission and 10 for manure storage (Zhu et al. 2000a; Guo et al. 2001).

The total odour emission E determined by equation 7 is then used in the

dispersion model INPUFF-2 to predict downwind odour. Dispersion simulations were conducted for six typical weather conditions W1 to W6 that disadvantage odour dilution, resulting in high odour concentrations at the ground level. Based on the dispersion simulations of odour concentration downwind from the sources, setback distances were determined for W1 to W6 and a correlation between the separation distance and the total odour emission was established:

$$SD = aE^b \quad (A10)$$

where: SD = separation distance (m), and
a, b = weather influence factor constants for W1 to W6.

The occurrence frequencies of W1 to W6 were derived from the historical weather data in Minnesota. The odour annoyance free frequency (91% to 99%) in the OFFSET model is defined as the percentage of time when the odour intensity is below the selected annoyance level, i.e. 1 minus the occurrence frequency of the W1 to W6. For the 0-5 odour intensity scale (0 – no odour; 5 – very strong odour, ASTM, 1999), the selected odour annoyance free level is 2 (faint odour).

CHAPTER 5 MANURE STORAGE

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5.1 Executive Summary

In Canada, the laws relating to the design, operation, and management of manure storages are under the jurisdiction of the provinces. This report evaluates the regulatory requirements set out in the major pork producing areas in Canada and the USA. Alberta, Manitoba, Ontario, and Quebec were selected in Canada and Illinois, Minnesota, Iowa, and North Carolina were chosen in the US for detailed comparison. Animal waste handling is an important part of sustainable development of animal agriculture. A number of strategies can be used for animal waste handling which can be broadly classified into either open-lot or confinement facilities. Beef cattle operations use open lot method relying mainly on periodic solid removal. The hog industry has adopted the confinement method where waste is primarily handled as a liquid waste (Lorimor et al., 2006).

Animal manure storage facilities have been extensively used for animal waste treatment as well as storage due to the low cost and easy operation. However, there are serious environmental concerns associated with waste storages such as gaseous emissions and groundwater pollution. Anaerobic lagoon features are intended to minimize environmental risks and enhance biological treatment of wastes. This report compared the lagoon design, operation, and the assumptions made during this process in four provinces of Canada (Manitoba, Alberta, Ontario, Quebec) and four states in the USA (North Carolina, Minnesota, Iowa, Illinois). The factors such as siting, total storage volume, depth, shape, detention time, and number of cells, embankment and excavation, inlet and outlet pipes design, effluent utilization, and water supply were compared for the different locations.

The set backs for manure storages vary for the different jurisdictions. Manitoba has the largest set back of 100 m from water courses, sinkholes, wells, springs, and site boundaries. Only, Ontario has set back limits to drainage tiles mainly because of the prevalence of subsurface drains there. Manitoba has the most stringent requirement for the storages to be located a 0.6 m above the 100-year flood plain level. There is provision for the director to make site specific decisions on this aspect. Alberta and Quebec require the storage to be located above a 25-year and 20-year flood plain, respectively. The services of a Professional Engineer are required for site characterization in Ontario, only. In Manitoba, the director makes the decision based on site features information provided by the applicant. All Canadian jurisdictions require the services of a Professional Engineer for the design of the manure storage. In addition, Ontario requires the Professional Engineer to evaluate the manure management protocol associated with the storage. The storage capacity requirements vary between the different locations. While Ontario and Quebec require 240 days and 200 days of

storage capacity, Alberta requires 9 months storage capacity or a shorter period if an alternate manure management plan is submitted. Manitoba has banned the spreading of manure to unseeded land before August 15 if the land is not going to be seeded before the spring of the following year. To prevent manure being spread on frozen soils, manure spreading is banned prior to April 10 in Manitoba. The storage capacity is determined by the duration of this ban.

The seepage beneath the bottom of manure storages has been a contentious issue. All earthen manure storages result in some seepage and the maximum rate has been set by regulation. Although, research indicates that the seepage rate is primarily controlled by the sealing as result of the of the accumulation of sludge on the bottom, Manitoba Regulations have stipulated a minimum one meter thick compacted clay layer having hydraulic conductivity no greater than 10^{-7} cm/s. This requirement is similar to that of Alberta. If an aquifer exists beneath the manure storage, the Manitoba requirement is to have a five meter overburden having hydraulic conductivity no greater than 10^{-7} cm/s. Alberta requires 10 meters of soils having hydraulic conductivity no greater than 10^{-6} cm/s which is much less stringent than Manitoba. These requirements are more stringent than that exists in the USA. The formation of a low-permeability sludge accumulation layer at the bottom will greatly reduce the seepage rate than assumed by the design standards which do not take this layer into account. However, the disturbance of this seal-layer during agitation can temporarily destroy its sealing capability (Sri Ranjan et al, 2005). Therefore, the scouring of this layer during agitation should be avoided by the use of protective measures such as concrete agitation pads which are required by the Manitoba regulations. While, deeper manure storages promote anaerobic treatment they also increase the hydraulic gradient across the bottom, leading to an increase in seepage. The hydraulic conductivity limit of 10^{-7} cm/s was based on a 2.8 m of standing water. Recommended depths vary from 1.5 m to 6.0 m in the literature. Manitoba regulation does not specify any shape requirements for the manure storage. The use of multi-cell manure storages indirectly provides the advantage of rectangular shaped lagoons. The detention time has not been specified in the Manitoba regulations because they are primarily storage facilities rather than treatment facilities. In contrast to this, the municipal lagoons are concrete lined storages and not expected to have any seepage.

Manitoba and Ontario regulations recommend a maximum hydraulic conductivity of 10^{-7} cm/s for the berms surrounding the manure storages with a freeboard of 0.3 and 0.15 m, respectively. While, Manitoba does not have a secondary containment requirement, Alberta, Ontario, and Quebec have some requirement if the liquid level within the manure storage is above grade and the facility is located near a water body or water course. Quebec requires secondary containment if the manure storage is located within 5 km upstream of a surface water source supplying drinking water. Only, Alberta requires a protective liner for the berm. Manitoba regulations stipulate a compacted clay liner meeting standards similar to the one specified for the bottom of the storage. Manitoba requires a concrete pad where the agitation equipment is used to protect the bottom and sides of the manure storage.

None of the Canadian provinces allows the intentional release of manure to the environment. In comparison, the regulations in the USA permit spillage resulting from catastrophic storm events exceeding 25year/24hour storm. Among all the eight jurisdictions evaluated in this study, only Illinois has set out the requirement for a certified livestock manager to manage any farm having 300 or greater animal units. This

certification is only valid for three years and requires periodic renewal on completion of upgrade training.

All Canadian jurisdictions require a permit for the construction, modification, or expansion of manure storages. Only, Manitoba and Ontario require a manure management plan as part of the permit application. The Ontario regulation further requires a Professional Engineer to evaluate the manure management plan.

5.2 Lagoon Design Factors

The liquid waste stabilization pond, one of the most used wastewater treatment techniques (Abbas et al., 2006), is an economical and efficient system for wastewater treatment (Agunwamba, 1992). The American Society of Agricultural and Biological Engineers (ASABE) has developed standards titled “Design of Anaerobic Lagoons for Animal Waste Management” which has been used in the design of manure storages in the USA (ASABE 2004). According to these standards, manure storages have been defined as “anaerobic lagoons” if there is anaerobic treatment expected to take place in the lagoon. Such lagoons require the retention of a minimum treatment volume, except during maintenance, leading to partial anaerobic treatment of the diluted waste. When there is no intended treatment requirement, the manure is stored in “waste storage ponds” which are used for storing the manure for an extended period at the end of which they are emptied. In Manitoba, the predominant system in use is the multi-cell manure storage with partial anaerobic treatment happening while the manure is stored during the winter. The manure is first pumped into a primary cell where the solids are allowed to settle. The supernatant liquid is allowed to overflow into a secondary cell where further solids separation takes place under gravity. Some systems may have a third cell receiving the liquid overflowing from the secondary cell. In addition to the major role played by biological treatment processes, chemical and physical factors are also important in manure degradation (Hamilton et al., 2006). Hence, treatment efficiency will be proportional to the extent to which the lagoon promotes these processes (Figure 5.1).

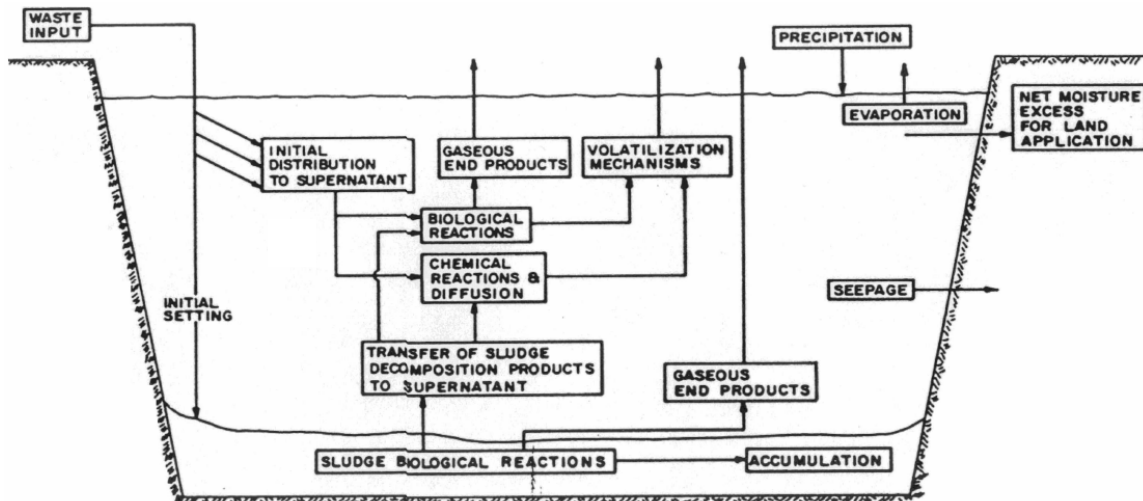


Figure 5.1: Schematic representation of processes occurring in an anaerobic lagoon (Source: Miner et al. 2000)

Lagoon design aims to enhance biological, physical, and chemical processes. However, criteria other than treatment efficiency are also taken into account (e.g. odour generation). The major design factors involved in a successful operation of lagoons are siting, total lagoon volume, depth, shape, detention time, number of cells, embankment and excavation, inlet and outlet pipes design, effluent utilization, and water supply (ASABE 2004). These factors vary widely due to climatic and site conditions, as well as operational goals.

There are many factors taken into consideration in the design of lagoons. The siting of the lagoon to mitigate environmental impact has been specified in the regulations in all of the jurisdictions evaluated in this study. As part of this requirement, site characterization standards and design features to be used in the lagoons have also been specified. The following sections will compare these requirements.

5.2.1 Selection of location to mitigate environmental impact

The primary consideration in the siting of manure storage is the odour concerns, potential for contamination of the groundwater resources, and operational considerations such as the need for spreading etc. Locating a lagoon is one of the most important factors because it influences several other design criteria. Special consideration is given to odour dispersion and groundwater pollution (ASABE 2004).

5.2.1.1 Odour Issues

Any manure storage/treatment facility, regardless of how well it is designed, will generate odour, which is an aesthetic problem that may require public intervention and regulatory agency involvement. Chen et al. (2003) have identified inadequate sizing and mismanagement as the main cause of odour problems in lagoons. The setback distance between the odour source and neighboring residences and public areas is a function of several factors. The proper setback distance can be determined by the use of verified computer models (ASABE 2007). To minimize the odour concerns, the ASABE standards prescribe a minimum separation of 400 m from any occupied dwellings and at least 760 m from any populated areas (ASABE 2004). Alberta has a minimum distance of 150 m to site boundaries where this distance is not covered by other odour separation regulations. The topographic features such as ravines should not be present to promote the transport of odour during atmospheric inversions. The prevailing direction of wind in relation to populated areas and the presence of trees to cause the natural mixing of the air stream should also be considered. Both Illinois and North Carolina have detailed setback requirements from private homes, schools etc. Areas located in relatively higher elevation and with good wind incidence should be preferred as they disperse lagoon odours better. Van Kleeck and Bulley (1985), applying a questionnaire directed to residents living between 50 and 1200 meters away from livestock farms, found that nuisance potential decreased with distance and that odour was more pronounced in the downwind direction than in the upwind direction. They also found that residents who could actually see the farm tended to report nuisance more than those who could not see the farm from their residence. In order to minimize the visual influence, features designed to hide the lagoon such as trees or natural barriers can be employed. These features also present the additional advantage of lifting and mixing the air (ASABE 2004).

Other methods for odour control in lagoons are aeration and covering. Such systems require application of different technologies in association with anaerobic lagoon itself. These two options are further discussed in a different chapter of this report.

5.2.1.2 Groundwater Contamination Potential

Seepage from animal-waste storages can have an effect on groundwater quality because it can contaminate subsurface water with nutrients and pathogens (Ham and DeSutter, 1999; Parker et al., 1999a). Due to this hazard, earthen manure storages should adopt design features to minimize seepage. Compacted soil liners between 0.3 meters and 0.46 meters thick are designed to avoid large seepage rates (Ham, 2002), where fine-grained soils permit lower seepage rates than coarse-grained soils (Parker et al. 1999a). In addition, organic sludge that accumulates on the bottom is expected to decrease the permeability of lagoons (Ham and DeSutter 1999), further reducing seepage losses. In fact, seepage seems to be controlled by sludge sealing rather than liner properties. Tyner et al. (2006), studying the effect of dairy waste on the flux through a silt loam liner with large macropores, found that flux was mainly controlled by the hydraulic properties of the seal layer and not related to liner thickness and weakly related to its saturated hydraulic properties. The authors observed that macropores in the liner were not water-filled and the soil within the liner remained unsaturated suggesting that the liner was not the main factor limiting the seepage rate. Barrington and Jutras (1985), studying sealing mechanisms of manure, found similar results. They stress that sealing process is primarily physical, in which suspended solids are entrapped within macro- and micro- pores. The maximum sealing efficiency is of the order of 10^{-7} cm/s (0.1 L/m²-day), which means that some seepage will always take place. Under such conditions, the soil beneath the lagoon should present a high filtration capacity (i.e. high cation retention capacity). Soils that lack this filtration capacity should have collection drains to collect the seepage and means to pump it back into the reservoir.

Despite generalization sought in terms of soil properties and seepage rate, there is no consensus about universal parameters for siting anaerobic lagoons. Given this scenario, some authors have proposed site-specific criteria for lagoon design when dealing with seepage (Parker et al. 1999a; Ham and DeSutter 2000). Ham and DeSutter (2000) propose a two-step framework for site-specific design of anaerobic lagoons. As a first step, geological assessment and soil analyses of the vadose-zone of the proposed site are performed. This investigation would reveal the occurrence of undesirable geological formation (e.g. karst, abandoned well) and important soil characteristics (e.g. texture, background level of nutrients, cation exchange capacity). The second step comprises selecting one or more criteria for lagoon design (e.g. ammonium nitrogen, fecal bacteria), estimating these criteria in the lagoon effluent, and calculating its maximum allowable seepage. According to Ham and DeSutter (2000), plastic-lined lagoons or alternative storage structures should be considered if the maximum allowable seepage (S_{max}) is less than 0.2 mm d^{-1} . According to this approach, S_{max} is a function of both the effluent composition and site background characteristics.

The proximity to other features is also a consideration in siting the manure storage. The presence of drinking water wells, watercourses, drainage tiles, and the potential for flooding of the site should also be taken into account. To prevent contamination of drinking water, a minimum clearance of 100 m is required in Manitoba between a watercourse and the edge of the manure storage. This is comparable to the requirements in Quebec. Iowa¹ requires a 305 m (1000 ft) distance while North

¹ Section 02T.1304(b)(4) Subchapter 2T Waste Not Discharged To Surface Waters Title 15A North Carolina Administrative Code <http://h2o.enr.state.nc.us/admin/rules/documents/2Tbook.pdf>

Carolina² only has a minimum distance of 100 ft. Ontario and Alberta require only 50 m and 30 m, respectively.

The minimum distance to a well, sinkhole, or spring is 100 m in Manitoba and Alberta, 30.5 m in Illinois³, 150 m to 300 m in Iowa⁴. In Ontario, this limit is only 15 m if the well is deeper than 15 m and has a watertight casing of 6 m and 30 m for any other wells. This limit increases to 100 m if the well is to be used for drinking water. North Carolina has set the minimum distance to a drinking water source to be 500 ft⁵. In Quebec, the minimum distance is 300 m if it supplies drinking water or waterworks and 75 m for non-potable wells.

Ontario has a minimum distance of 15 m similar to Illinois⁶ and Iowa⁷ from any drainage tiles. Iowa has 500 ft from surface intake of agricultural drainage well⁸ and 1000 ft from wellhead, cistern of an agricultural drainage well, or known sinkhole⁹. Only Iowa has set

² Section 02T.1304(b)(4) Subchapter 2T Waste Not Discharged To Surface Waters Title 15A North Carolina Administrative Code <http://h2o.enr.state.nc.us/admin/rules/documents/2Tbook.pdf>

³ Section 506.304(a)(8) General Design and Construction Standards, Title 35: Environmental Protection, Subtitle E: Agriculture Related Pollution, Chapter I: Pollution Control Board, Part 506 Livestock Waste Regulations, Joint Committee on Administrative Rules, Administrative Code <http://www.ilga.gov/commission/jcar/admincode/035/035005060C03040R.html> Online [Accessed June 11, 2007]

⁴ Iowa Code Chapter 459.310(1)(b) Animal Agriculture Compliance Act [http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=\[field folio-destination-name:'sec_459_310'\]&xhitlist_md=target-id=0-0-0-68257](http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=[field folio-destination-name:'sec_459_310']&xhitlist_md=target-id=0-0-0-68257)

⁵ Section 106-803(a)(4) Article 67 Swine Farms, Chapter 106 Agriculture, North Carolina General Statutes, http://www.ncleg.net/EnactedLegislation/Statutes/PDF/ByArticle/Chapter_106/Article_67.pdf

⁶ Section 506.304(a)(7) General Design and Construction Standards, Title 35: Environmental Protection, Subtitle E: Agriculture Related Pollution, Chapter I: Pollution Control Board, Part 506 Livestock Waste Regulations, Joint Committee on Administrative Rules, Administrative Code <http://www.ilga.gov/commission/jcar/admincode/035/035005060C03040R.html> Online [Accessed June 11, 2007]

⁷ Section 65.15(1)(b) Animal Feeding Operations Iowa Administrative Code Environmental Protection Commission[567] <http://www.legis.state.ia.us/Rules/Current/iac/567iac/56765/56765.pdf>

⁸ Iowa Code Chapter 459.310(1)(a) Animal Agriculture Compliance Act [http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=\[field folio-destination-name:'sec_459_310'\]&xhitlist_md=target-id=0-0-0-68257](http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=[field folio-destination-name:'sec_459_310']&xhitlist_md=target-id=0-0-0-68257)

⁹ Iowa Code Chapter 459.310(1)(a) Animal Agriculture Compliance Act [http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=\[field folio-destination-name:'sec_459_310'\]&xhitlist_md=target-id=0-0-0-68257](http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=[field folio-destination-name:'sec_459_310']&xhitlist_md=target-id=0-0-0-68257)

a minimum distance of 750 m from any wetlands¹⁰ while North Carolina prohibits construction of lagoons in wetlands¹¹.

The flood plain limit is less stringent in Alberta and Quebec with siting to be outside the 1 in 25-year and 1 in 20-year flood level, respectively. Both Manitoba and Ontario have a more stringent siting requirement outside the 1 in 100-year flood level. This requirement is similar to that required by North Carolina¹² and Illinois¹³. Manitoba has a further restriction of 0.6 m above the 1 in 100-year flood level. Table 5.1 compares the set back distances and floodplain limits.

¹⁰ Iowa Code Chapter 459.310(1)(b) Animal Agriculture Compliance Act
[http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=\[field folio-destination-name:'sec_459_310'\]&xhitlist_md=target-id=0-0-0-68257](http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xml&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=[field folio-destination-name:'sec_459_310']&xhitlist_md=target-id=0-0-0-68257)

¹¹ Section 3.3 Eighth Senate Bill (SB) 1217 Interagency Group Guidance Document, Online [Accessed June 5, 2007] http://www.enr.state.nc.us/DSWC/images/GuidanceDoc_8.pdf

¹² Section 106-803(a2) Article 67 Swine Farms, Chapter 106 Agriculture, North Carolina General Statutes, http://www.ncleg.net/EnactedLegislation/Statutes/PDF/ByArticle/Chapter_106/Article_67.pdf

¹³ Section 510 ILCS 77/13(b)(1) Livestock Management Facilities Act, Chapter 510 Animals, Illinois Compiled Statutes.
<http://www.ilga.gov/legislation/ilcs/ilcs3.asp?ActID=1720&ChapAct=510%26nbsp%3BILCS%26nbsp%3B77%2F&ChapterID=41&ChapterName=ANIMALS&ActName=Livestock+Management+Facilities+Act%2E> Online [Accessed June 8, 2007]

Table 5.1 Comparison of manure storage regulations governing location and siting. Note cells with “---“ designate no specific information in the regulation for that jurisdiction.

Regulations	Alberta	Manitoba	Ontario	Quebec
Minimum Distance from watercourse	30 m ¹⁴	100 m ¹⁵	50 m ¹⁶	100 m (river or lake) ^{17 18} 75 m (other watercourse) ¹⁹
Minimum distance from sinkhole, well or spring	100 m ^{20, 21}	100 m ²²	15 m (well of Minimum depth 15 m and 6 m of water tight casing) ²³ 30 m (from any other well) ²⁴	300 m (if supplies drinking water or waterworks) ²⁵ 75 m (non-drinking water / waterworks) ²⁶
Minimum distance	100 m ²⁷	100 m ²⁸	100 m ²⁹	300 m ³⁰

¹⁴ Section 7(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹⁵ Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹⁶ Section 63(3) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(3))

¹⁷ Section 11(a) Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

¹⁸ Appendix A Section 2. Canadian Code of Practice for Environmentally Sound Hog Production pg. 26

¹⁹ Section 11(a) Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

²⁰ Alberta Regulation 243/2003 Nuisance and General Sanitation Regulation Public Health Act http://www.qp.gov.ab.ca/documents/Regs/2003_243.cfm?frm_isbn=0779752767

²¹ Section 7(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

²² Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

²³ Section 63(1)(a) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(1))

²⁴ Section 63(1)(c) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(1))

²⁵ Section 11(a) Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

²⁶ Section 11(c) Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Regulations	Alberta	Manitoba	Ontario	Quebec
from drinking water source				
Minimum distance from site boundaries	---	100 m ³¹	---	---
Minimum distance from residential neighbours	150 m (otherwise setback distance determined by odour requirements) ³²	---	---	---
Drainage Tile Requirements	---	---	(a) determine location of all field drainage tiles or piped municipal drains within 15 metres of the perimeter of the facility; (b) remove all drainage tiles within 15 metres of the perimeter of the facility; and (c) redirect the flow of the field drainage system or piped municipal drain away from the facility ³³	---
Flood plain/level requirements	The 1 in 25-year maximum flood level at a	Not within 1 in 100-year flood plain unless	Not within 1 in 100-year flood lines (unless permit	Not within 1 in 20-year flood plain ³⁷

²⁷ Section 7(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

²⁸ Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

²⁹ Section 63(1)(b) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(1))

³⁰ Section 11(a) Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

³¹ Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

³² Section 7(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

³³ Section 63(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(2))

Regulations	Alberta	Manitoba	Ontario	Quebec
	<p>manure storage facility or at a manure collection area must be not less than one metre below any part of the facility where run-on can come into contact with the stored manure. ³⁴</p>	<p>a) Facility has flood protection for a flood water level at least 0.6 m higher than the 100-year flood water level the department anticipates at the location when the person proposes to construct, modify or expand the facility; or</p> <p>b) the director is otherwise satisfied that the facility will have satisfactory flood protection. ³⁵</p>	<p>issued under section 28 of the <i>Conservation Authorities Act</i> ³⁶</p>	<p>A manure storage site that is intended for an existing livestock operation and that is established in a flood plain must be equipped with watertight side walls that are as high as the highest flooding level, or must be surrounded by a watertight dyke of that height. ³⁸</p>

³⁴ Section 8 Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

³⁵ Section 5(1) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. E125

³⁶ Section 63(4) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(4\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(4))

³⁷ Section 11(d) Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

³⁸ Section 27 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

5.2.2 Site characterization requirements prior to construction of the manure storage

The site has to be characterized prior to the location of the manure storage. Only, Ontario requires that the site be characterized by a professional engineer or geoscientist. Iowa requires site characterization by a qualified person³⁹ using a minimum of four soil core samples obtained from the proposed storage site. Iowa also prohibits locations that exhibit Karst features such as sinkholes, or solution channeling generally occurring in areas underlain by limestone or dolomite⁴⁰. Illinois has a set back of 400 ft from any areas having Karst features⁴¹. Ontario requires subsurface site characterization to a depth 1.5 m below the bottom of the lined storage or 2.5 m below the bottom of earthen manure storage. In Manitoba, a 1-m clay layer is required below the bottom of the earthen manure storage. In addition, an examination of the potential to pollute surface and groundwater resources need to be carried out by the operator in a manner acceptable to the Director at the request on the Director of Manitoba Conservation. Table 5.2 compares the site characterization requirements.

³⁹ Section 65.15(6) Animal Feeding Operations Iowa Administrative Code Environmental Protection Commission [567] <http://www.legis.state.ia.us/Rules/Current/iac/567iac/56765/56765.pdf>

⁴⁰ Section 65.15(8) Animal Feeding Operations Iowa Administrative Code Environmental Protection Commission [567] <http://www.legis.state.ia.us/Rules/Current/iac/567iac/56765/56765.pdf>

⁴¹ Section 510 ILCS 77/13(b)(2) Livestock Management Facilities Act, Chapter 510 Animals, Illinois Compiled Statutes.

<http://www.ilga.gov/legislation/ilcs/ilcs3.asp?ActID=1720&ChapAct=510%26nbsp%3BILCS%26nbsp%3B77%2F&ChapterID=41&ChapterName=ANIMALS&ActName=Livestock+Management+Facilities+Act%2E> Online [Accessed June 8, 2007]

Table 5.2 Comparison of site characterization requirements. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Site Characterization	Alberta	Manitoba	Ontario	Quebec
Requires professional engineer / geoscientist	---	---	Yes ⁴²	---
Subsurface characterization requirements	---	Before deciding whether to register the manure storage facility, the director may require the operator to conduct an examination of the facility's integrity and proximity to surface water, surface watercourses, wells, springs, sinkholes, groundwater or other environmentally sensitive areas in a manner that is acceptable to the director and to submit a report to the director stating the results, if the director believes that the facility's integrity is causing or would likely cause	Stage one hydrogeologic or geotechnical investigation that identifies the soil types and the presence of any aquifer or bedrock, all to a depth of at least, (a) 1.5 metres below the lowest elevation of the excavation required for a structure made of concrete, steel or other materials that a professional engineer determines will provide equivalent protection; or (b) 2.5 metres below the lowest elevation of the excavation required for a structure made of earth. ⁴⁴	---

⁴² Section 64 Nutrient management Act Ontario Regulation 267/03, s. 64 (1) http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#64.

⁴³ Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁴⁴ Section 65(1) Nutrient management Act Ontario Regulation 267/03, s. 65 (1) [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(1\);](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(1);) O. Reg. 511/05, s. 36 (1)

Site Characterization	Alberta	Manitoba	Ontario	Quebec
		(a) pollution of surface water, groundwater or soil; or (b) livestock manure to escape from the boundary of the agricultural operation. ⁴³		
Subsurface Investigation Requirements	---	---	See Appendix A: Subsurface Investigation Requirements	---

5.2.3 Design and construction standards

Manitoba, Alberta, Ontario, and Quebec require that the owner/operator proposing a manure storage facility obtain the services of a Professional Engineer for the design of the manure storage structure with some variations in how it is implemented. Alberta does not require an engineer's approval for all structures. Ontario requires the engineer to look at the manure management protocol associated with the structure in addition to the design of the structure, inspection during and after construction to ensure compliance with the submitted plan.

The storage capacity requirements vary between the provinces. Alberta requires 9 months or fewer consecutive months of storage or approves an alternate plan submitted by the owner or operator. Ontario requires 240 days of storage and Quebec requires a minimum period of 200 consecutive days of storage. Illinois has a 150-day storage requirement with a 2-foot freeboard⁴⁵. Iowa requires storage capacity for up to 14 months with a freeboard⁴⁶. Manitoba does not specify the number of days of storage. However, there is a ban on spreading manure between a late fall date and a spring date during which the storage structure should be capable of holding the manure. Therefore, the number of days of storage is governed by this restriction. This no-spreading restriction is to prevent spreading on snow covered/frozen ground thereby minimizing the potential to contaminate surface water resources. Ontario has set a maximum storage depth of 3 meters and a maximum storage volume of 2500 m³ for earthen manure storages. There is no dimension limits set by Alberta, Manitoba, and Quebec.

Alberta and Manitoba require the material at the bottom of the manure storages to be at least 1 m thick composed of soil with a hydraulic conductivity of 1×10^{-7} centimetres per second. The basis of the hydraulic conductivity specification (1×10^{-7} centimetres per second) allows for a seepage rate of 2.5 cm per year, under a head of 2.8 m of standing liquid level, through a one-meter thick bottom. In most situations, this seepage rate is considered to have negligible impact on the nearby water resources provided the specified set backs are maintained.

The minimum separation between the bottom of the storage and the aquifer below has been specified to be no less than 1 m in Alberta and Ontario and 5 m in Manitoba. This separation is minimum of 4 ft and a recommended separation of 10 ft in Iowa⁴⁷. Manitoba has specified a hydraulic conductivity of 1×10^{-7} cm per second or less for the overburden. Illinois has requires the storage and transport surfaces constructed of concrete and intended to come into contact with livestock waste to be constructed or

⁴⁵ Section 510 ILCS 77/13(a)(1)(C) Livestock Management Facilities Act, Chapter 510 Animals, Illinois Compiled Statutes.

<http://www.ilga.gov/legislation/ilcs/ilcs3.asp?ActID=1720&ChapAct=510%26nbsp%3BILCS%26nbsp%3B77%2F&ChapterID=41&ChapterName=ANIMALS&ActName=Livestock+Management+Facilities+Act%2E> Online [Accessed June 8, 2007]

⁴⁶ Section 510 ILCS 77/13(a)(1)(C) Livestock Management Facilities Act, Chapter 510 Animals, Illinois Compiled Statutes.

<http://www.ilga.gov/legislation/ilcs/ilcs3.asp?ActID=1720&ChapAct=510%26nbsp%3BILCS%26nbsp%3B77%2F&ChapterID=41&ChapterName=ANIMALS&ActName=Livestock+Management+Facilities+Act%2E> Online [Accessed June 8, 2007]

⁴⁷ Section 65.15(9) Animal Feeding Operations Iowa Administrative Code Environmental Protection Commission[567] <http://www.legis.state.ia.us/Rules/Current/iac/567iac/56765/56765.pdf>

installed to achieve a hydraulic conductivity equal to or less than 1×10^{-6} cm/s⁴⁸. This limit is 10 times more than allowed in Manitoba. Ontario has specified at least 15 percent clay content for the soil below unlined storages and 10 percent clay content for the soil below concrete lined storages.

The material requirements for earthen manure storages differ between the provinces. Alberta requires the material to provide equal or greater protection than that given by 10 m of soil with a hydraulic conductivity of not more than 1×10^{-6} cm per second. In Iowa, the maximum hydraulic conductivity is set at 2×10^{-6} cm per second⁴⁹.

The manure storages can be constructed with concrete liners, plastic liners, or compacted earthen liners. The concrete can be un-reinforced or reinforced. The following tables compare the requirements for these different types in Alberta, Manitoba, Ontario, and Quebec.

⁴⁸ Section 506.304(a)(2) General Design and Construction Standards, Title 35: Environmental Protection, Subtitle E: Agriculture Related Pollution, Chapter I: Pollution Control Board, Part 506 Livestock Waste Regulations, Joint Committee on Administrative Rules, Administrative Code <http://www.ilga.gov/commission/jcar/admincode/035/035005060C03040R.html> Online [Accessed June 11, 2007]

⁴⁹ Section 65.15(11) Animal Feeding Operations Iowa Administrative Code Environmental Protection Commission[567] <http://www.legis.state.ia.us/Rules/Current/iac/567iac/56765/56765.pdf>

Table 5.3 Comparison of professional intervention and sizing requirements prior to construction. Note cells with “---“ designate no specific information in the regulation for that jurisdiction.

Professional need	Alberta	Manitoba	Ontario	Quebec
Requires a Professional Engineer	<p>Yes.</p> <p>If required by an approval officer or the Board, an owner or operator of a confined feeding operation or a manure storage facility must have a professional engineer</p> <p>(i) design and develop the plans for the system,</p> <p>(ii) stamp and sign the plans, and</p> <p>(iii) certify that the design and the plan meet the requirements of clauses (a) to (d).</p> <p>(a) the system must limit the amount of surface water and run-on and runoff flowing through and from the operation or facility;</p> <p>(d) the system must not be located on a fish</p>	<p>Yes.</p> <p>A professional engineer, contractor or other person who performs work for which a permit has been issued under this section shall</p> <p>(a) comply with any terms and conditions contained in the permit; and</p> <p>(b) ensure that the work complies with the siting and construction requirements set out in Schedule A.⁵¹</p> <p>Yes, must have a sealed professional engineer's certificate certifying that</p> <p>(i) the work of any contractor or other person performing work for which the permit is required conforms to the</p>	<p>Yes.</p> <p>(a) a professional engineer designs the construction or expansion, including any associated monitoring systems, having regard to the requirements of this Regulation and signs the Engineer's Commitment Certificate contained in the Nutrient Management Protocol, by which the engineer undertakes to have regard to those requirements and to inspect the construction or expansion upon completion;</p> <p>(b) the facility is designed to minimize leakage, minimize corrosion and to be structurally safe and</p>	<p>Yes.</p> <p>A project notice for erection work or increasing storage capacity must be served to the Minister of Sustainable Development, Environment and Parks at least 30 days before it is carried out.</p> <p>The project notice must be signed by the operator and by an engineer who is a member of the Ordre des ingénieurs du Québec and who will supervise the work. The engineer's signature certifies that the proposed work complies with this Regulation⁵⁴</p>

⁵⁰ Section 6(2) Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁵¹ Section 6(6.1) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁵² Section 6(7) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁵³ Section 71(1) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#71.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#71.(1))

⁵⁴ Section 40 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

Professional need	Alberta	Manitoba	Ontario	Quebec
	bearing water body as determined from maps described in the <i>Code of Practice for Watercourse Crossings</i> adopted in section 3(2) of the <i>Water (Ministerial) Regulation</i> (AR 205/98); ⁵⁰	siting and construction requirements set out in Schedule A and the permit, and (ii) the completed construction, modification or expansion of the manure storage facility conforms to the siting and construction requirements and the permit ⁵²	sound; (c) the construction or expansion complies with this Part; (d) the construction or expansion takes place under the supervision of a professional engineer; and (e) a professional engineer inspects the construction or expansion upon completion and confirms that it is in accordance with the design. O. Reg. 267/03, s. 71 (1); O. Reg. 511/05, s. 38 (1, 2). ⁵³	
Storage capacity	Sufficient to store all the manure produced by the operation over a period of at least 9 consecutive months or fewer consecutive months if an approval officer or the Board approves a manure handling plan submitted by the owner or operator ⁵⁵	Ensure that the manure storage facility, alone or in combination with other manure storage facilities located on the property of the agricultural operation, is of sufficient capacity to store all of the livestock manure produced or used in the	Capable of containing at least all of the nutrients generated or received in the course of the operation during a period of 240 days ⁵⁷	Collect all manure and liquid manure produced by livestock operation and all contaminated water for a minimum period of 200 consecutive days ⁵⁸ accumulate, without overflow, for the entire period where the livestock waste may not

⁵⁵ Section 10(1) Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

⁵⁶ Section 4(a) Livestock Manure and Mortalities Management Regulation Manitoba Regulation 42/2004 The Environment Act

Professional need	Alberta	Manitoba	Ontario	Quebec
		<p>agricultural operation until such time as the livestock manure can either be applied as fertilizer or otherwise removed from the manure storage facility;⁵⁶</p> <p>(a) the storage volume of the completed earthen storage facility shall meet the requirements of clause 4(a) of this regulation;</p>		<p>be spread, the livestock waste produced in the raising facilities as well as all other waste that may be accumulated in those facilities⁵⁹</p> <p>The storages of livestock producers shall have the capacity to receive and to accumulate, without overflow, in addition to the substances mentioned in section 59, the livestock waste produced in their livestock buildings for no fewer than 250 consecutive days⁶⁰</p>
Dimensions	---	---	Earthen based facility dimensions calculated in accordance with the Nutrient Management Protocol ⁶¹	---

⁵⁷ Section 69(1) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#69.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#69.(1))

⁵⁸ Section 30 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

⁵⁹ Section 10 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

⁶⁰ Section 38

⁶¹ 79(a) Nutrient Management Act Ontario Regulation 267/03 http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#79.

⁶² Section 65(3) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(3))

Professional need	Alberta	Manitoba	Ontario	Quebec
			Unlined earthen facility: maximum storage depth of 3.0 metres and a maximum storage volume of 2,500 cubic metres ⁶²	

Table 5.4 Comparison of material requirements for a concrete/steel facility. Note cells with “---“ designate no specific information in the regulation for that jurisdiction.

Material (concrete/steel)	Alberta	Manitoba	Ontario	Quebec
Material Requirements	The liner of a manure storage facility and of a manure collection area, if constructed of compacted soil or constructed of concrete, steel or other synthetic or manufactured materials, must provide equal or greater protection than that provided by compacted soil 1 m in depth with a hydraulic conductivity of not more than 1×10^{-7} centimetres per second for a liquid manure storage facility ⁶³	Dike and floor protection shall be constructed of concrete, or another material approved in writing by an environment officer ⁶⁴	Ensure that the concrete used in the facility is appropriate for the environmental conditions encountered on site to maintain the durability and corrosion resistance of the concrete and to protect the reinforcing materials, if any, in the concrete. O. Reg. 267/03, s. 72 (1); O. Reg. 447/03, s. 31. ⁶⁵	The ground on which a farm building is constructed or laid out must be protected from any contact with the livestock waste produced by means of a watertight floor. ⁶⁶
Thickness	---	---	(2) The permanent nutrient storage facility must be constructed with a minimum thickness of 125 millimetres of concrete on the floor of	---

⁶³ Section 9(6)(a) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁶⁴ Schedule A Section 2(h) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁶⁵ Section 72(1) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#72.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#72.(1))

⁶⁶ Section 8 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

Material (concrete/steel)	Alberta	Manitoba	Ontario	Quebec
			the structure unless a professional engineer specifies otherwise. O. Reg. 267/03, s.72 (2). ⁶⁷	

⁶⁷ Section 72(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#72.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#72.(2))

Table 5.5 Comparison of minimum separation to aquifer for unlined and unreinforced concrete floor. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Separation for unlined, unreinforced concrete	Alberta	Manitoba	Ontario	Quebec
Minimum separation between bottom of facility and aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource ⁶⁸	If an aquifer exists and less than 5 m of overburden having an expected hydraulic conductivity of 1×10^{-7} cm per second or less will separate the bottom of the facility from the top of the uppermost underlying aquifer or fractured rock, the proponent may be required to: (a) to implement a groundwater monitoring plan or install a groundwater monitoring system acceptable to the director; (b) to install a plastic or compacted clay liner and implement a groundwater monitoring plan or install a groundwater monitoring system acceptable to the	Unlined concrete or steel storage facilities with unreinforced concrete floors must have, between the bottom of the storage facility and the uppermost identified bedrock layer or aquifer, a minimum of 1.0 metres of hydraulically secure soil or a minimum of 1.0 metres of soil comprised of a clay content of at least 15 per cent. ⁷⁰	---

⁶⁸ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁶⁹ Schedule A Sections 3(1) and 3(2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁷⁰ Section 65(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(2))

	Alberta	Manitoba	Ontario	Quebec
Separation for unlined, unreinforced concrete		director; or (c) to use an alternative method of construction or manure storage acceptable to the director. ⁶⁹		
Minimum separation between bottom of facility and water table	If a protective layer is used, the bottom of the manure storage facility or manure collection area must be not less than 1 m above the water table of the site at the time of construction ⁷¹	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ⁷²

⁷¹ Section 9(3) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁷² Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Table 5.6 Comparison of minimum separation to aquifer for lined and unreinforced concrete floor. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Separation for lined, unreinforced concrete	Alberta	Manitoba	Ontario	Quebec
Minimum separation between bottom of liner and aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource. ⁷³	---	Lined concrete or steel storage facilities with unreinforced concrete floors must have a minimum of 1.0 metres of native undisturbed material or compacted granular material between the bottom of the storage facility and the uppermost identified bedrock layer or aquifer ⁷⁴	---
Minimum separation between bottom of liner and water table	The bottom of a liner of a manure storage facility and of a manure collection area must be not less than 1 m above the water table of the site at the time of construction. ⁷⁵	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ⁷⁶

⁷³ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁷⁴ Section 65(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(2))

⁷⁵ Section 9(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁷⁶ Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Table 5.7 Comparison of minimum separation to aquifer for unlined and reinforced concrete floor. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Separation for unlined, reinforced concrete	Alberta	Manitoba	Ontario	Quebec
Minimum separation between bottom of facility and aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource ⁷⁷	If an aquifer exists and less than 5 m of overburden having an expected hydraulic conductivity of 1×10^{-7} cm per second or less will separate the bottom of the facility from the top of the uppermost underlying aquifer or fractured rock, the proponent may be required to: (a) to implement a groundwater monitoring plan or install a groundwater monitoring system acceptable to the director; (b) to install a plastic or compacted clay liner and implement a groundwater monitoring plan or install a groundwater monitoring system acceptable to the director; or	Unlined concrete or steel storage facilities with reinforced concrete floors must have, between the bottom of the storage facility and the uppermost identified bedrock layer or aquifer, a minimum of 0.5 metres of hydraulically secure soil or 1.0 metres of soil comprised of a clay content of at least 10 per cent.	---

⁷⁷ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁷⁸ Schedule A Sections 3(1) and 3(2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

	Alberta	Manitoba	Ontario	Quebec
Separation for unlined, reinforced concrete		(c) to use an alternative method of construction or manure storage acceptable to the director. ⁷⁸		
Minimum separation between bottom of facility and water table	If a protective layer is used, the bottom of the manure storage facility or manure collection area must be not less than 1 m above the water table of the site at the time of construction ⁷⁹	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ⁸⁰

⁷⁹ Section 9(3) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁸⁰ Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Table 5.8 Comparison of minimum separation to aquifer for lined and reinforced concrete floor. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Separation for lined, reinforced concrete	Alberta	Manitoba	Ontario	Quebec
Layer between bottom of facility & aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource. ⁸¹	---	Lined concrete or steel storage facilities with reinforced concrete floors must have a minimum of 0.5 metres of native undisturbed material or compacted granular material between the bottom of the storage facility and the uppermost-identified bedrock layer or aquifer.	---
Minimum separation between bottom of facility and water table	The bottom of a liner of a manure storage facility and of a manure collection area must be not less than 1 m above the water table of the site at the time of construction. ⁸²	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ⁸³

⁸¹ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁸² Section 9(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁸³ Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Table 5.9 Comparison of material requirements for earthen manure storages. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

	Alberta	Manitoba	Ontario	Quebec
Material requirements	The protective layer of a manure storage facility and of a manure collection area must provide equal or greater protection than that provided by naturally occurring materials 10 m in depth with a hydraulic conductivity of not more than 1×10^{-6} centimetres per second for a liquid manure storage facility ⁸⁴	(a) topsoil shall be stripped from the area where any dike is to be constructed before excavation and compaction; (b) all excavated material shall be placed in 0.15 m lifts and then compacted to a minimum thickness of 1 m; (c) if a plastic or compacted clay liner will not be installed, the sides and bottom of the earthen storage facility shall be (i) disked to a minimum depth of 20 cm, and (ii) compacted with a fully ballasted sheepsfoot packer, or other compaction equipment approved by	Materials not excavated from site must be tested for appropriate hydraulic conductivity by professional engineer ⁸⁶	---

⁸⁴ Section 9(5)(a) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁸⁵ Schedule A Section 2 Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁸⁶ Section 75(1) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#75.\(1\) O. Reg. 447/03](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#75.(1) O. Reg. 447/03), s. 32

		<p>the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698 at a moisture content between 0.9 and 1.2 optimum and a maximum hydraulic conductivity no more than 1×10^{-7} cm per second;</p> <p>(d.1) if a plastic or compacted clay liner is installed, it shall be placed over a stable floor and dike that are to be compacted with a fully ballasted sheepsfoot packer, or other compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698;⁸⁵</p>		
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Table 10 Comparison of minimum separation to aquifer for unlined earthen manure storages. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Separation for unlined, earthen manure storage	Alberta	Manitoba	Ontario	Quebec
Layer between bottom of facility & aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource ⁸⁷	If an aquifer exists and less than 5 m of overburden having an expected hydraulic conductivity of 1×10^{-7} cm per second or less will separate the bottom of the facility from the top of the uppermost underlying aquifer or fractured rock, the proponent may be required to: (a) to implement a groundwater monitoring plan or install a groundwater monitoring system acceptable to the director; (b) to install a plastic or compacted clay liner and implement a groundwater monitoring plan or install a groundwater monitoring system acceptable to the director; or	Unlined storage facilities made of earth used to store agricultural source materials, other than manure and materials produced by intermediate generators, must have, between the bottom of the storage facility and the uppermost identified bedrock layer or aquifer, a minimum of 1.0 metres of hydraulically secure soil or a minimum of 1.0 metres of soil comprised of a clay content of at least 15 per cent ⁸⁹	---

⁸⁷ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁸⁸ Schedule A Sections 3(1) and 3(2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁸⁹ Section 65(2)(5) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(2))

Separation for unlined, earthen manure storage	Alberta	Manitoba	Ontario	Quebec
		(c) to use an alternative method of construction or manure storage acceptable to the director. ⁸⁸		
Minimum separation between bottom of facility and water table	If a protective layer is used, the bottom of the manure storage facility or manure collection area must be not less than 1 m above the water table of the site at the time of construction ⁹⁰	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ⁹¹
Construction	The protective layer of a manure storage facility and of a manure collection area must provide equal or greater protection than that provided by naturally occurring materials 10 m in depth with a hydraulic conductivity of not more than 1×10^{-6} centimetres per second	(d) if a plastic or compacted clay liner will not be installed, the sides and bottom of the earthen storage facility shall be (i) disced to a minimum depth of 20 cm, and (ii) compacted with a fully ballasted sheepsfoot packer, or other	Additional Requirements: (a) the facility has a maximum storage depth of 3.0 metres and a maximum storage volume of 2,500 cubic metres; (b) the facility has at least 2.0 metres of hydraulically secure soil between the bottom and	---

⁹⁰ Section 9(3) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁹¹ Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Separation for unlined, earthen manure storage	Alberta	Manitoba	Ontario	Quebec
	for a liquid manure storage facility ⁹²	compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698 at a moisture content between 0.9 and 1.2 optimum and a maximum hydraulic conductivity no more than 1×10^{-7} cm/s ⁹³	sides of the facility and the uppermost identified bedrock layer or aquifer; (c) the soil materials that form the interior surface of the facility are disked to a depth of at least 150 millimetres and recompact to meet a hydraulic conductivity of no more than 1×10^{-8} metres per second; (d) any soil anomalies that are discovered during construction, such as coarse material lenses, large rocks or soil fractures are excavated and filled with a clay based material to a depth of one metre to the satisfaction of the professional engineer; ⁹⁴ (b) the liner on the inside wall of the facility is constructed using at least six layers of a	

⁹² Section 9(5)(a) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

⁹³ Section 2(d) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

⁹⁴ Section 65(3) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(3))

⁹⁵ Section 75(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#75.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#75.(2))

Separation for unlined, earthen manure storage	Alberta	Manitoba	Ontario	Quebec
			thickness of no more than 150 millimetres; (c) the liner on the bottom of the facility is constructed using at least four layers of a thickness of no more than 150 millimetres; (d) the interface surface of layers is disked or scarified before placement of subsequent layers of material; ⁹⁵	

Table 5.11 Comparison of compacted earthen liners. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Separation for compacted earthen liner	Alberta	Manitoba	Ontario	Quebec
Layer between bottom of liner and aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource. ⁹⁶	---	Lined storage facilities made of earth must have a minimum of 2.0 metres of hydraulically secure soil between the bottom and sides of the lined storage facility and the uppermost identified bedrock layer or aquifer. ⁹⁷	---
Minimum separation between bottom of liner and water table	The bottom of a liner of a manure storage facility and of a manure collection area must be not less than 1 m above the water table of the site at the time of construction. ⁹⁸	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ⁹⁹
Construction	The liner of a manure storage facility and of a manure collection area, if constructed of compacted soil or	(d.1) if a plastic or compacted clay liner is installed, it shall be placed over a stable floor and dike that are to	(a) the minimum thickness of the completed liner is at least 0.9 metres on the sloping inside walls and	---

⁹⁶ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁹⁷ Section 65(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(2))

⁹⁸ Section 9(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

⁹⁹ Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Separation for compacted earthen liner	Alberta	Manitoba	Ontario	Quebec
	constructed of concrete, steel or other synthetic or manufactured materials, must provide equal or greater protection than that provided by compacted soil 1 m in depth with a hydraulic conductivity of not more than 1×10^{-7} centimetres per second for a liquid manure storage facility, ¹⁰⁰	be compacted with a fully ballasted sheep'sfoot packer, or other compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698; (d.2) if a compacted clay liner is installed, excavated material shall be placed to a minimum thickness of 1 m in successive 15 cm thick lifts compacted with a fully ballasted sheepsfoot packer, or other compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698 at a moisture	0.6 metres on the bottom of the facility; (b) the liner on the inside wall of the facility is constructed using at least six layers of a thickness of no more than 150 millimetres; (c) the liner on the bottom of the facility is constructed using at least four layers of a thickness of no more than 150 millimetres; (d) the interface surface of layers is disked or scarified before placement of subsequent layers of material; and (e) each of the layers has been compacted to at least 95 per cent of modified Proctor maximum dry density as determined for the soil at a specified optimum water content. O. Reg. 267/03, s. 75 (2). ¹⁰²	

¹⁰⁰ Section 9(6)(a) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

¹⁰¹ Schedule A Section 2(d.2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹⁰² Section 75(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#75.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#75.(2))

Separation for compacted earthen liner	Alberta	Manitoba	Ontario	Quebec
		content between 0.9 and 1.2 optimum, and a maximum hydraulic conductivity no more than 1×10^{-7} cm per second ¹⁰¹		

Table 5.12 Comparison of synthetic liners. Note cells with “---“ designate no specific information in the regulation for that jurisdiction.

Separation for synthetic liner	Alberta	Manitoba	Ontario	Quebec
Layer between bottom of facility and aquifer / bedrock	The bottom of a liner or the base of a protective layer of a manure storage facility or of a manure collection area must be not less than 1 m above the top of the groundwater resource. ¹⁰³	---	Lined storage facilities made of earth must have a minimum of 2.0 metres of hydraulically secure soil between the bottom and sides of the lined storage facility and the uppermost identified bedrock layer or aquifer. ¹⁰⁴	---
Minimum separation between bottom of facility and water table	The bottom of a liner of a manure storage facility and of a manure collection area must be not less than 1 m above the water table of the site at the time of construction. ¹⁰⁵	---	---	Above highest level of water table or lowered by gravitation (N/A to existing non-compliant sites) ¹⁰⁶

¹⁰³ Section 9(4) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

¹⁰⁴ Section 65(2)(6) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(2))

¹⁰⁵ Section 9(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

¹⁰⁶ Section 26 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act
<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Separation for synthetic liner	Alberta	Manitoba	Ontario	Quebec
Construction	The liner of a manure storage facility and of a manure collection area, if constructed of compacted soil or constructed of concrete, steel or other synthetic or manufactured materials, must provide equal or greater protection than that provided by compacted soil 1 m in depth with a hydraulic conductivity of not more than 1×10^{-7} centimetres per second for a liquid manure storage facility ¹⁰⁷	if a plastic or compacted clay liner is installed, it shall be placed over a stable floor and dike that are to be compacted with a fully ballasted sheepsfoot packer, or other compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698; ¹⁰⁸	Must be anchored or bonded to the facility, subgrade, or berms made of earth / structure according to good engineering practices or manufacturer's specification ¹⁰⁹	---
Inspection	---	---	The qualified professional or other person responsible for supervising the construction of the facility shall, (a) inspect the synthetic liner before the	---

¹⁰⁷ Section 9(6)(a) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹⁰⁸ Schedule A Section 2(d.1) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹⁰⁹ Section 74(1) and 74(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#74.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#74.(1))

¹¹⁰ Section 74(3) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#74.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#74.(3))

¹¹¹ Section 74(4) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#74.\(4\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#74.(4))

Separation for synthetic liner	Alberta	Manitoba	Ontario	Quebec
			<p>filling of the construction or the covering of the liner to ensure that there are no damage or perforations within the liner; and</p> <p>(b) ensure that any damage or perforations discovered during the inspection are repaired according to the engineer's instructions. 110</p> <p>(c) inspect any repairs made to the liner to ensure that the integrity of the liner is maintained 111</p>	

5.2.4 Total Lagoon Volume

The total volume of an anaerobic lagoon is the summation of several component volumes. Generally, these volumes are grouped into three main categories (Figure 5.2), namely sludge volume, treatment volume, and effluent storage (Mayes and George 1981; Hamilton et al. 2006). This subsection describes the rationale for the volume calculation process.

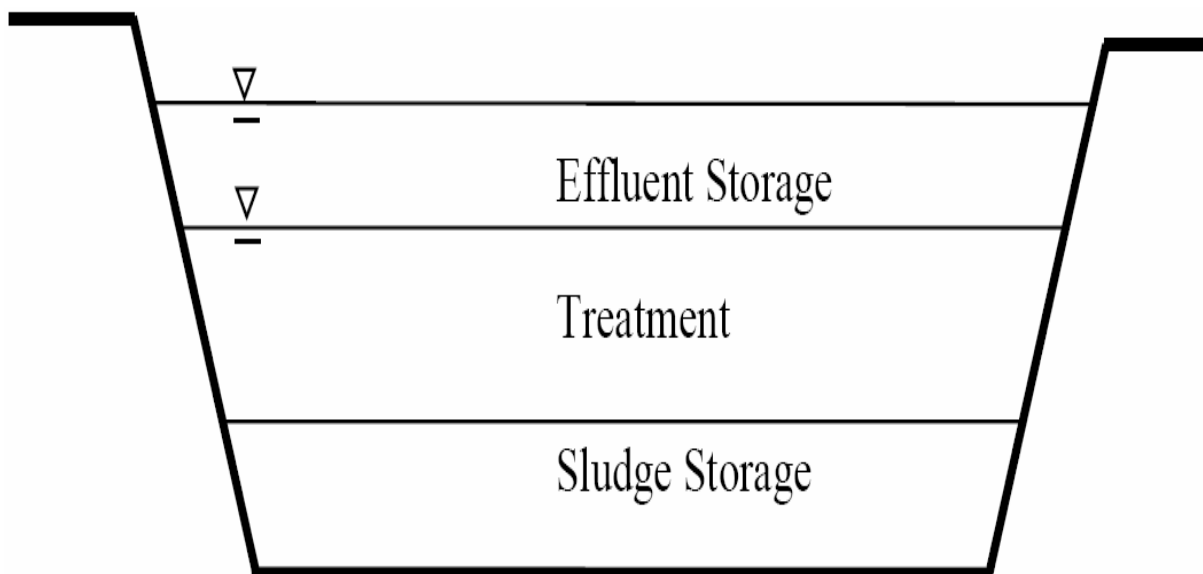


Figure 5.2: Lagoon volumes schematic (Source: Hamilton et al. 2006)

5.2.4.1 Sludge Volume

The allowance provided for accumulating the fixed and non-volatile solids that settles to the bottom of the lagoon is defined as the sludge volume (ASABE 2004). Definition of this volume is based on the amount of total solids (TS) that settles and accumulates as sludge. Some works used as reference for lagoon designing (ASABE 2004; Hamilton et al. 2006) refer to the sludge accumulation rates presented by Barth (1985a) (Table 5.13).

Table 5.13: Sludge accumulation rates

Type of livestock	Sludge accumulation rates (m ³ /kg TS added)
Poultry – layer	0.00184
Poultry – pullet	0.00284
Swine	0.00303
Dairy	0.00455

Source: Barth 1985a

Accumulation rates presented in Table 13 are based on the amount of total solids present in the manure. ASABE (2005) and Barth (1985b) present reference values for manure components as excreted by different types of livestock. Data presented by the former source are calculated from diet numbers and literature values, and are expressed in kilogram per day per animal (kg/day – animal). The latter source, in turn, presents an approach called digestibility approximation of manure production (DAMP) to calculate manure composition. According to this approach, the characteristics of digestibility of feed components and their respective digestibility when fed to different groups of livestock are used to determine manure

characteristics. The total solids loads are expressed in kilogram per head per day (kg/hd/day). In addition, Barth (1985a) presents a different set of data that accounts for feed waste in the manure, which may have a significant impact on TS loads. For example, 5% feed waste mixed with manure increases the TS content by about 40%.

Regardless of the source of data used for TS loads calculation, the procedure to determine the sludge volume consists of multiplying the TS loads by the sludge accumulation rate. The result is multiplied by the number of animals in the facility and then by the number of days of accumulation. The result is the volume of sludge accumulated in the operational timeframe defined (i.e. days of accumulation). This procedure is mathematically expressed by

$$SV = TS \cdot SAR \cdot NA \cdot DA \quad (1)$$

where

SV = sludge volume (m³)

TS = total solids produced per animal per day (kg/head/day)

SAR = sludge accumulation rate (m³/ kg TS)

NA = number of animals (head)

DA = days of accumulation (day)

The operational timeframe is an important parameter to be defined as it determines how often the sludge would have to be removed from the lagoon. Sludge should be removed when its volume exceeds the calculated capacity. According to Mukhtar et al. (2004), the effectiveness of biodegradation is compromised when sludge build-up encroaches on the other volumes (e.g. treatment volume) comprising the lagoon. Thus, periodic sludge removal is necessary to restore volumes to their respective calculated values and, consequently, to preserve the effectiveness of the treatment processes. The use of the multi-cell lagoon will minimize the problem of reduced treatment volumes because the treatment can happen in the secondary cell where the sludge accumulation is much reduced.

5.2.4.2 Treatment Volume

Even though lagoon designers have tended to use number, size, and animal species to calculate treatment volume, traditional calculations are based on the weight or mass of volatile solids discharged into the lagoon on a daily basis (Miner et al 2000). The first approach is site-specific because it assumes the other factors involved in biodegradation (i.e. biological activity, temperature) to be defined a priori. In fact, this methodology relies on tabulated values that were calculated using the second approach. Therefore, the traditional method is a more general procedure that comprises organic material loads, specificities of different microorganism, and climatologic characteristics. The following discussion is based on the second approach.

Anaerobic digestion of animal manure is a complex process that involves serial and parallel microbial reactions (Masse and Droste, 2000). Ultimately, the treatment volume of an anaerobic lagoon depends upon these reactions that decompose the volatile solids added to the lagoon. According to Hamilton et al. (2006), minimum treatment volume is found by dividing the mass of volatile solids (VS) added each day by a volumetric loading factor. Such a loading factor is calculated from biological activity rates of several communities of microorganisms involved in the biodegradation of manure.

The three main groups involved in organic waste digestion are hydrolytic bacteria, fermenting and acid-forming bacteria, and methanogenic and sulfate-reducing bacteria (Hamilton et al. 2006). However, some authors describe anaerobic digestion as a two-stage process involving only the two last groups (Barth 1985a; Adams Jr. et al. 1999). The first stage is controlled by acid-producing bacteria, in which dissolved organic waste is decomposed into organic acids. In the second stage, these acids are further decomposed into methane, carbon dioxide, and water by methanogenic bacteria. The treatment process is directly linked to these two communities and proportional to their capacity to break down waste components.

The ability of microbes to decompose substrates relies on the rate of biological activity, which depends on temperature (Adams Jr. et al., 1999). In the lagoon context, the treatment process is restricted by methane formers because this group has limited activity below 15°C, while acid formers remain active up to about 4°C (Barth 1985a). Based on these limitations, Barth (1985a), proposes an approach for calculating treatment volume based on the Van't Hoff-Arrhenius relationship

$$KT = K_{20} \theta^{(T-20)} \quad (2)$$

where,

KT = rate of biological activity,

K_{20} = rate of biological activity at 20°C (assumed to be 1.0),

θ = constant for anaerobic animal waste decomposition (assumed to be 1.047),

T = temperature (°C).

In this approach, the average lagoon temperature for every single day is used in Equation 2. The annual biological activity rate (KTA) is then calculated by adding the values of KT for each day of the year. It is important to note that KT is assumed to be zero on days when the temperature is below 5°C. For simplification, the KTA can be divided by 365 to represent the daily biological activity (KTA_{daily}) that can be used for treatment volume definition.

The next step consists of determining the mass of volatile solids (VS) added to the storage each day. This value is calculated by multiplying the VS loads from the sources cited in the previous section (e.g. ASABE, 2005) by the number of animals in the facility. Finally, the operational timeframe must be defined. This procedure can be expressed as

$$TV = \frac{(VS \cdot NA \cdot DA)}{KTA_{\text{daily}}} \quad (3)$$

where,

TV = treatment volume (m^3),

VS = volatile solids produced per animal per day (kg/head/day),

NA = number of animals (head),

DA = days of accumulation (days),

KTA_{daily} = KTA divided by 365 (dimensionless).

Barth (1985a) shows a map of KTA_{daily} values for the continental United States (Figure 5.3). The effect of temperature is illustrated in this map by lower values in greater latitudes. That

means that lagoons in greater latitudes require larger volumes than those in lower latitudes to treat the same amount of VS. This fact is supported by White (1977), who states that anaerobiosis increases in warmer climates, allowing larger loading rates. White (1977) has reported that lagoons treat loadings of volatile solids from 6 to 166g/m³ per day. The values presented by Barth (1985a) for swine, poultry and dairy are within this range.

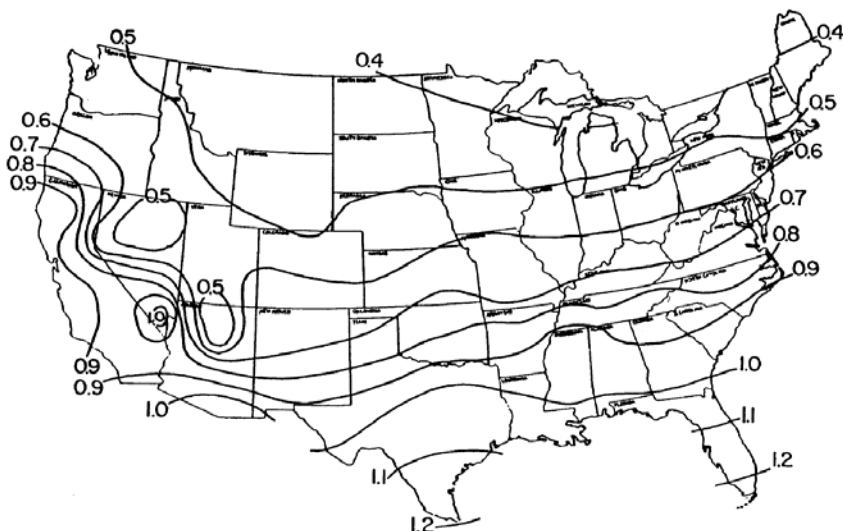


Figure 5.3: Values of KTA_{daily} for the continental United States (Source: Barth 1985a).

5.2.4.3 Effluent Storage

Storage volume should allocate space for the volume of manure loaded between drawdown events, the volume of any runoff that enters the lagoon between drawdown, and the volume corresponding to rainfall minus evaporation for the same period (Moore et al. 1980; Mayes and George 1981). In Manitoba, this period could last over seven months.

ASABE (2004) presents a simple mathematical procedure for calculating the total waste volume generated in the facility. This procedure is adapted here and shown by equation 4, which includes manure and other solid waste such as bedding and spilled feed. This procedure also recommends providing some allowance for other wastewater volumes of contaminated water according to operational aspects. This volume is determined by observation according to different wastewater sources in the livestock facility.

$$LWV = (VM \cdot NA \cdot DA) + (OS \cdot DA) \quad (4)$$

where,

LWV = livestock waste volume (m³),

VM = volume of manure produced per animal per day (m³/head/day),

NA = number of animals (head),

DA = days of accumulation (days),

OS = other solids (m³/day).

The second component of storage volume is designed for runoff. Miner et al. (2000) advises that including an appropriate volume to hold runoff is very important and that many disastrous

lagoon failures have been caused by inadequate runoff storage capacity. In fact, incorporating runoff in the storage volume may represent the difference between failure and success of waste containment. However, in Manitoba, the manure storages have berms that prevent runoff from entering the manure storages.

The last component of storage volume is the one corresponding to rainfall minus evaporation. According to Mayes and George (1981) this component is referred to as “net rain” and has been calculated by some authors using annual lake evaporation minus precipitation in combination with some other factors. Others make use of data collected from evaporation pans (Overcash et al. 1983; ASCE 2005). However, Parker et al. (1999b), comparing evaporation rates from feedyard pond effluent and clean water in Texas, observed that the feedyard effluent evaporated around 10% more than clean water. According to the authors, this is due to differences in physico-chemical characteristics of both liquids. These findings agree with Mayes and George (1981), who state that experience with lagoons has demonstrated that liquid levels in these lagoons varied more than design data indicated they would. Presumably, this difference between designed and actual volume is a decreased risk of overflow because the lagoon is losing more water than expected. Nonetheless, errors in estimation of evaporation rates may cause risks such as erroneous seepage rate predictions (Parker et al. 1999b), operational depth below the minimum level (ASABE 2004), and loss of anaerobic conditions (Adams Jr. et al. 1999).

5.2.5 Depth

Even though depth is an important criterion, some sources do not include it in lagoon design. Barth (1985a) and ASCE (2005) describe lagoon planning in terms of volume only. Other sources, however, include depth as a distinctive criterion for design.

Data shown suggest that depths between 2 and 6 meters are preferred, and that lagoon should be above the maximum water table level. The necessity of a freeboard of about 0.3 meter has also been stressed (Overcash et al. 1983; ASABE 2004). All these features are intended to keep proper operational conditions and avoid environmental risks.

The operational aspect related to depth is treatment efficiency. According to Hamilton et al. (2006), depth provides suitable sludge storage and favors the development of layered biological communities. The anaerobic conditions also increase with depth and products of anaerobic metabolism appear in the deeper layers (Hobson and Robertson 1977). These conditions are essential for biological treatment; therefore, they are fundamental for satisfactory lagoon performance. Recommendations of some authors are summarized in Table 5.14.

Table 5.14 Summary of recommended depth for anaerobic lagoons

Type of waste	Recommended operational depth (m)	Observations	Source
Dairy	Maximum 1.5; minimum 0.6	Assumes evaporation and seepage equal to rainfall	Crowe and Phillips (1972)
Undefined	As deep as groundwater conditions allow	Usually between 3 m and 9 m	Booram and Smith (1974)
Undefined	Maximum 6.0; minimum 1.5	Lowest operational level above the highest level of the water table	White (1977)
Undefined	About 3.0	---	Hobson and Robertson (1977)
Undefined	Maximum 4.5; minimum 2.4	Lagoons 6 m deep can be used but have high cost	Overcash et al. (1983)
Undefined	As deep as groundwater conditions allow	In practice, maximum 4.5 m and minimum 1.5 m	Loehr (1984)
Undefined	Maximum 6.0 or more; minimum 3.0	Lagoon bottom above maximum groundwater level	Miner et al. (2000)
Undefined	Maximum 6.0; minimum 2.0	Treatment volume at least 0.3m above water table	ASABE (2004)

Adequate depth is also important for environmental protection. ASABE (2004) suggests the minimum operational depth, which corresponds to treatment volume, to be 0.3 meters above maximum water table level. This caution is intended to avoid groundwater contamination by lagoon effluent. Another concern is the maximum depth, which is observed to avoid overtopping. If water level encroaches on the freeboard, there is a risk of overtopping the dike. In such cases, the dike is prone to erosion, which can cause its collapse and massive spilling of lagoon contents (Miner et al. 2000).

5.2.6 Shape

Shape is another significant factor when designing anaerobic lagoons because geometry affects other important characteristics of the pond such as resistance to flow and detention time (Agunwamba 1992). A well-shaped lagoon also avoids problems with solids accumulation (Miner et al. 2000).

The shape is determined by the need for a surface area as small as possible. Small surface area promotes anaerobic conditions and decreases the total area needed (Loehr, 1984). However, minimizing area without encroaching lagoon volume implies an increasing lagoon depth. Thus, surface area is ultimately restricted by total volume and depth, and lagoon can

only be reduced in terms of area to the extent to which these two factors are not critically affected.

Rectangular lagoons are more common due to the positive influence of the rectangular shape being better for internal distribution. Shindala and Murphy (1969), studying the influence of shape on mixing and load of sewage lagoons in Texas, report that rectangular lagoons presented a more uniform load distribution.

Some works have also discussed about length-to-width ratios that maximizes treatment efficiency. Tools such as numerical models have been used to this end. For example, Abbas et al. (2006) used computational fluid dynamic modeling (CFD) to verify efficiency of biochemical oxygen demand (BOD) removal in waste stabilization ponds. Their simulations showed that BOD removal efficiency increased with length-to-width ratio for no baffle cases. The BOD removal for a 300mg/l influent was 21.6% and 18.9% for length-to-width ratios of 4 and 3, respectively. These ratios (i.e. 4 and 3) are the maximum length-to-width recommended for anaerobic lagoons (Miner et al. 2000; ASABE 2004).

5.2.7 Detention Time

Lagoons, to be effective, need a minimum residence time for waste treatment. Residence time of about 90 days seems to be necessary in most climates for reasonable waste breakdown (Hobson and Robertson 1977). White (1977) presents narrower period from 30 to 60 days. ASABE (2004), however, suggests a minimum time of 50 days.

Detention time is not simple to determine. It is a function of a more complex process called hydraulic transport, which controls residence time itself and dispersion of waste (Nameche and Vasel 1998). A classical approach to assess hydraulic conditions in anaerobic lagoons is to perform experiments using dye tracer measurements. In such experiments, the system is sampled at different times after the injection of a dye tracer. Torres et al. (1997), performing a dye tracer experiment in a deep waste stabilization pond in Spain, found that the residence time (t) was below the spatial time (τ = total volume of the tank divided by total flow rate). This result implies that waste is leaving the pond before the theoretically designed time. In this case, biological treatment of waste may be insufficient. Unfortunately, such conditions may be prevalent in many anaerobic lagoons. In fact, very few tracer experiments have been performed in full-scale waste stabilization ponds (Nameche and Vasel 1998), suggesting that the detention time in most lagoons may not be known. In such instances, the risk of untreated waste being pumped out of the lagoon exists.

Numerical models have been coupled with dye tracer experiments in an effort to determine lagoon hydraulic conditions and their respective treatment efficiencies. These studies show that, in a general way, lagoons are described as an “intermediate” hydraulic model between plug-flow and completely mixed systems (Nameche and Vasel 1998), even though completely-mixed flow model compare well with experimental data (Ferrara and Harleman 1981). This knowledge about hydraulic behavior is essential for lagoon design as it can help engineers to design lagoons better and correct possible faults in terms of hydraulic behavior, thus enhancing treatment.

Another concern related to the detention time is flow rates. Residence time may be decreased if the volume of water heading to the lagoon is increased (Hobson and Robertson 1977). Therefore, detention time is not a function of design parameters only, but depends on operational aspects as well.

5.2.8 Number of Cells

Animal waste treatment can also be designed to occur in more than one cell (i.e. lagoon). According to Lorimor et al. (2006), treatment efficiency and management flexibility are enhanced by using two- or three-stage cells in series. Figure 5.4 depicts a system with a two-cell configuration.

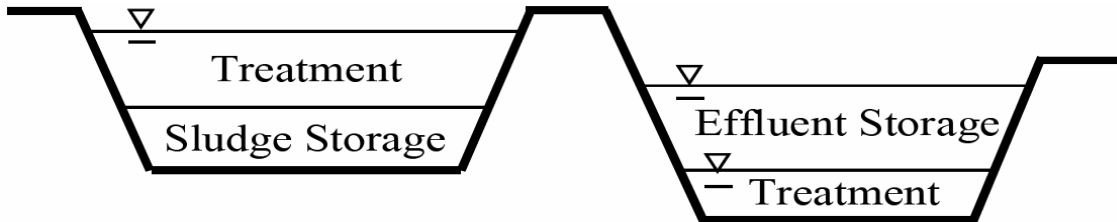


Figure 5.4 Two-cell lagoon configuration (Source: Source: Hamilton et al. 2006)

Hamilton et al. (2006) suggests a three-cell system in which the first is an anaerobic cell, followed by a facultative (naturally aerated) cell, and having an aerobic cell as the third stage. According to Miner et al. (2000), lagoon volumes should be designed as if for a single cell system, but the storage volume and an additional 25% treatment volume should be placed in the second cell. The third cell, also called tertiary lagoon or maturation pond, is used for removal of pathogenic organisms (Maynard et al. 1999). According to Maynard et al (1999), this system is an effective and low-cost method for removing pathogens from wastewater.

5.2.9 Embankment and Excavation

Despite being listed among the criteria for lagoon design, literature on embankment and excavation are, in most cases, sparse in the literature. Authors limit comments to appropriate slope and embankment protection. Slopes of 3:1 or less are generally recommended (Hobson and Robertson 1977; Overcash et al. 1983; Miner et al. 2000), while slopes steeper than 2:1 are discouraged (ASABE 2004). Such slopes are said to be sufficiently flat to avoid erosion and to withstand water pressures. If steeper slopes are required, erosion-control measures should be used.

There are several features for erosion control that may be used in earthen structures. A popular method is "blanketing" the dikes with different materials. The most common ones are rock riprap and concrete (US Department of Interior 1974). Geotextiles have been largely used in the past few years. According to Sarsby (2007), geotextiles reduce runoff, retains soil particles, and protect against sun, rain and wind.

Erosion can also be caused by factors other than environmental conditions. Routine operations can damage embankments of lagoons. Richard and Hinrichs (2002), in their study about management and maintenance of earthen manure structures in Iowa, point out that 27% of the lagoon studied showed erosion of compacted clay liners caused by agitation or manure inflow, 24% showed evidence of animal burrows in berms, and 6% had tree growth in berms. According to Richard and Hinrichs (2002), these conditions associated with normal operations of lagoons contribute to chronic form of environmental impacts. In Manitoba, concrete lined agitation pads are required in all new earthen manure storages to prevent erosion of the compacted clay bottom during agitation and pump out.

Lagoons are generally constructed mainly or entirely below ground level requiring excavation (Hobson and Robertson 1977). Embankment top elevation should be increased by 5% during construction to allow for settlement, while top width of 2.5 m should be observed (ASABE 2004). No comments about methods to be used for embankment construction were found in the literature examined. Nonetheless, a tractor-compacted embankment as described by US Department of Interior (1974) seems to be adequate because it improves engineering properties such as strength and consolidation, thus enhancing impermeability.

Table 5.15 compares berm construction standards in the different provinces.

Table 5.15 Comparison of above ground berms. Note cells with “---“ designate no specific information in the regulation for that jurisdiction.

Above ground berms	Alberta	Manitoba	Ontario	Quebec
Slope of inside walls	3:1 Minimum horizontal to vertical ratio of inside wall side slope ¹¹²	---	Consistent with the requirements of the liner design and pump out equipment and, unless a professional engineer specifies otherwise, slope is no steeper than 50% ¹¹³	---
Slope of outside walls	4:1 Minimum horizontal to vertical ratio of outside wall side slope ¹¹⁴	---	no steeper than 33% ¹¹⁵	---
Thickness	---	1m ¹¹⁶	---	---
Max. hydraulic conductivity of berm	---	1 x 10 ⁻⁷ cm/s ¹¹⁷	1 x 10 ⁻⁹ m/s ¹¹⁸	---
Minimum compaction density of berm	---	95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698 ¹¹⁹	95% modified Proctor density ¹²⁰	---
Excavation	---	(b) topsoil shall be	Topsoil is stripped to the	---

¹¹² Section 14(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹¹³ Section 79(c) Nutrient Management Act Ontario Regulation 267/03 http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#79.

¹¹⁴ Section 14(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹¹⁵ Section 79(d) Nutrient Management Act Ontario Regulation 267/03 http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#79.

¹¹⁶ Schedule A Section 2(d.2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹¹⁷ Schedule A Section 2(d.2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹¹⁸ Section 65(3)(f) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(3))

¹¹⁹ Schedule A Section 2(d.1) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹²⁰ Section 65(3) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(3))

Above ground berms	Alberta	Manitoba	Ontario	Quebec
		stripped from the area where any dike is to be constructed before excavation and compaction; (c) all excavated material shall be placed in 0.15 m lifts and then compacted; (e) excavation and compaction shall be completed during temperature conditions that are above freezing;	subsoil layer from the area where any berm is to be constructed and stockpiled for use in the outside slopes of the facility; ¹²¹	
Minimum freeboard	0.5 m ¹²²	0.3 m ¹²³	0.15 m (uncovered storage) ¹²⁴ 0.3 m (uncovered earthen facility) ¹²⁵	---
Secondary containment required	Yes, if possible for manure to be discharged	---	If the liquid level is partially or wholly above	if highest level of liquid is above grade and facility

¹²¹ Section 65(3)(e) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(3\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(3))

¹²² Section 11(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹²³ Section 16.2(2) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹²⁴ Section 5.2.3 Nutrient Management Protocol Ontario <http://www.omafra.gov.on.ca/english/nm/regs/nmpro/nmpro05j05.htm#523>

¹²⁵ 79(b) Nutrient Management Act Ontario Regulation 267/03 http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#79.

Above ground berms	Alberta	Manitoba	Ontario	Quebec
	into common body of water ¹²⁶		level of soil, then (a) the load factor, α_L , as defined in clause 4.1.3.1. (1) (c) of Part 4 of the Building Code made under the <i>Building Code Act, 1992</i> for liquid loads is 1.5 or another value that a professional engineer is satisfied should be used; (b) a professional engineer specifies that the storage and landscape features around the facility are adequate to ensure that a secondary containment system is not required; or (c) the above grade portion of the facility has a secondary containment system with a capacity equivalent to 110 per cent of the above ground portion of the facility. O. Reg. 267/03, s. 76; O. Reg.	is located less than 5 km upstream from a surface water source supplying a waterworks system, it shall be surrounded by an earth backfill forming a reservoir with a minimum capacity equal to 1.5 times the capacity of that part of the storage site above grade. This requirement does not apply where a liquid manure storage site is a natural earth reservoir ¹²⁸

¹²⁶ Section 11(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹²⁷ Section 76 Nutrient Management Act Ontario Regulation 267/03 ://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#76

¹²⁸ Section 32 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act

<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Above ground berms	Alberta	Manitoba	Ontario	Quebec
Protective layer/liner required	Yes ¹²⁹	No ¹³⁰	447/03, s. 33. ¹²⁷ No ¹³¹	---
General Liner Requirements	---	(d) if a plastic or compacted clay liner will not be installed, the sides and bottom of the earthen storage facility shall be (i) disced to a minimum depth of 20 cm, and (ii) compacted with a fully ballasted sheepsfoot packer, or other compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698 at a moisture content between 0.9 and 1.2 optimum and a maximum hydraulic conductivity no more	Liner must be continuous under the floor and footings of the facility and must extend up the wall to a level equal with the top of the ground surface, unless the qualified professional supervising the construction of the facility specifies otherwise ¹³³	---

¹²⁹ Section 9(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

¹³⁰ Schedule A Section 2(d) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹³¹ Section 65(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#65.(2))

¹³² Schedule A Section 2(d) and 2(d.1) Manitoba Regulation 52/2004 Livestock Manure and Mortalities Regulation The Environment Act C.C.S.M. c. E125

¹³³ Section 73(2) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#73.\(2\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#73.(2))

Above ground berms	Alberta	Manitoba	Ontario	Quebec
		<p>than 1 x 10⁻⁷ cm per second; (d.1) if a plastic or compacted clay liner is installed, it shall be placed over a stable floor and dike that are to be compacted with a fully ballasted sheepsfoot packer, or other compaction equipment approved by the director, to at least 95% of maximum Standard Proctor dry density, determined by testing in accordance with ASTM Standard D698;¹³²</p>		
Ventilation	---	---	<p>If covered or otherwise allows gases to accumulate or intensify, a ventilation (natural or powered) system must be installed to eliminate corrosive, noxious or explosive gases¹³⁴</p>	---

¹³⁴ Section 78 Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#78.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#78.(1))

Table 5.16 Comparison of dike and floor requirements. Note cells with “---“ designate no specific information in the regulation for that jurisdiction.

Dyke and floor	Alberta	Manitoba	Ontario	Quebec
Required material	---	(h) dike and floor protection shall be constructed of concrete, or another material approved in writing by an environment officer, at the following locations:	---	The ground on which a farm building is constructed or laid out must be protected from any contact with the livestock waste produced by means of a watertight floor. ¹³⁵
Required areas	---	(i) at the access ramp where agitation equipment or a pump is rolled into the earthen storage facility in connection with agitation or pumping operations, (ii) at the point of discharge of the inlet pipe, where livestock manure is pumped into the earthen storage facility, (iii) in the case of earthen storage facilities with two or more cells, at the overflow channel; (i) all dikes of the earthen storage facility	---	---

¹³⁵ Section 8 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

Dyke and floor	Alberta	Manitoba	Ontario	Quebec
		shall be seeded to grass within one year of construction.		
Location of inlet	Bottom ¼ of lagoon ¹³⁶	---	---	---
Sealing of liner, piping & other extrusions required	Yes ¹³⁷	---	test holes that are excavated in the course of the site characterization and that are not required for any further purpose after the site characterization are plugged and sealed to provide a level of hydraulic conductivity that is the same or less than the hydraulic conductivity of the surrounding undisturbed soil ¹³⁸	---
Drain		---	---	entire outer perimeter, at and below floor-level, is equipped with a drain not connected to the site and whose outlet is joined to an inspection cover accessible at all

¹³⁶ Section 16 Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹³⁷ Section 17 Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act
http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

¹³⁸ Section 68 Nutrient Management Act Ontario Regulation 267/03 http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#68.

Dyke and floor	Alberta	Manitoba	Ontario	Quebec
				<p>times for the taking of samples ¹³⁹</p> <p>The drain must not be connected to the storage and its outlet must be linked to a manhole with a minimum inside diameter of 40 cm accessible for sample taking. A permanent marker must indicate the drain outlet's location. The drain must be functional at all times and evacuate water by gravity or pumping. ¹⁴⁰</p> <p>No overflow or sump drain that allows manure liquid or contaminated water to run off directly or indirectly into the environment ¹⁴¹</p>

¹³⁹ Section 28 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

¹⁴⁰ Section 12 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

¹⁴¹ Section 31 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

5.2.10 Inlet and Outlet Design

Inlet and outlet devices are designed to control the flux of influent and effluent in the lagoon. These features, if correctly planned, facilitate the operation and improve the efficiency of treatment.

Properly installed inlets should prevent plugging, accumulating solids along the bank, and damages caused by inflow (Overcash et al. 1983). Inlets can be either above water or submerged, but this aspect is not critical. Important characteristics of inlets are size and gradient. ASABE (2004) suggests inlets of 150 and 200 mm for nondairy and dairy waste, respectively. Miner et al. (2000), however, recommend that inlet lines should be based on anticipated flow rate, but recommend a minimum size of 100 mm pipe to avoid frequent plugging. Pipe inclination at the inlet of between 4% and 15% can be used, whereas 7% to 8% should be preferred (ASABE 2004). Positioning of discharge is also important because it influences solids distribution. If internal distribution is a concern, slopping floor may help to improve it (Hobson and Robertson 1977).

Outlets, contrasting with inlets, are simpler to plan. These components work as a spillway to avoid overtopping of dikes in case of overflow (Overcash et al. 1983). In case of a very unusual rainfall (i.e. 25year/24 hours storm), the outlet should discharge the excess of liquid and maintain adequate freeboard (ASABE 2004). Outlets should be positioned as far as possible from inlets to avoid short-circuiting (Hamilton et al. 2006).

5.3 Operation and maintenance

None of the Canadian provinces allows the release of liquid manure to the environment. Additional safeguards to prevent unintentional release have also been recommended in the form of berms and dykes. This requirement is more stringent than in the US. The US Federal laws allow intentional point release of stored liquid manure into a water body if a permit is obtained. However, unintentional releases to a water body resulting from a catastrophic storm event exceeding 25year/24hour storm do not require a permit. The release of liquid from an anaerobic lagoon for spreading onto agricultural field does not require a permit.

Among the four states, only Illinois seems to have a regulatory requirement for a certified person to manage the facility. A livestock waste handling facility serving 300 or greater animal units is required to operate only under the supervision of a certified livestock manager. The certification program includes the following:

- (1) A general working knowledge of best management practices;
- (2) A general working knowledge of livestock waste handling practices and procedures;
- (3) A general working knowledge of livestock management operations and related safety issues;
- (4) An awareness and understanding of the responsibility of the owner or operator for all employees who may be involved with waste handling.

The certification issued is valid for 3 years and thereafter it is subject to renewal. A renewal is valid for a further 3-year period, only. The Department may require anyone who is certified in less than 3 years to re-certify, for just cause including but not limited to repeated complaints, where investigations reveal the need to improve management practices.

Illinois requires the owner or operator of a livestock waste handling facility serving 1,000 or greater animal units to become a certified livestock manager. This could be done by attending a training session conducted by the Department of Agriculture, Cooperative Extension Service,

or any agriculture association, which has been approved by or is in cooperation with the Department and successfully completing a written competency examination. The certificate has an expiry date requiring periodic updating training to keep current. The owner or operator of a livestock waste handling facility operating in violation of these provisions is issued a warning letter for the first violation and is required to have a certified manager for the livestock waste handling facility within 30 working days. Failure to comply with the warning letter within the 30-day period will result in an administrative penalty of up to \$1,000 by the Department. Continued failure to comply, the Department may issue an operational “cease and desist” order until compliance is attained¹⁴². Most states do not seem to have a similar requirement on an ongoing basis.

Table 5.17 compares the operational requirements in Canada. No jurisdiction is permitted to release the stored manure to the environment. Additional berms/dykes are required for the containment of any spills in Canada.

A comparison of the regulatory requirements related to permits, design advisors, regulating body and penalties for Alberta, Manitoba, Ontario, and Quebec is presented in Table 18.

¹⁴² Section 510 ILCS 77/30 Livestock Management Facilities Act, Chapter 510 Animals, Illinois Compiled Statutes.
<http://www.ilga.gov/legislation/ilcs/ilcs3.asp?ActID=1720&ChapAct=510%26nbsp%3BILCS%26nbsp%3B77%2F&ChapterID=41&ChapterName=ANIMALS&ActName=Livestock+Management+Facilities+Act%2E> Online
[Accessed June 11, 2007]

Table 5.17 Comparison of operational requirements. Note cells with “---” designate no specific information in the regulation for that jurisdiction.

Operational aspects	Alberta	Manitoba	Ontario	Quebec
Discharge to environment	Not allowed, the owner or operator of an open liquid manure storage facility must provide a system of secondary containment of the liquid manure if there is a reasonable possibility that liquid manure can be discharged into a common body of water. ¹⁴³	An operator who stores livestock manure in a manure storage facility shall (b) design and construct the manure storage facility, or ensure that it is designed and constructed, so as to prevent the escape of any livestock manure that may cause pollution of surface water, groundwater or soil; ¹⁴⁴ No person shall handle, use or dispose of livestock manure, or store livestock manure in an agricultural operation, in such a manner that it is discharged or otherwise released into surface water, a surface watercourse or	Not allowed. No person shall construct or expand a permanent nutrient storage facility used on a farm unit in the course of the operation if the facility permits liquid prescribed materials to enter a tile drainage system ¹⁴⁶	Not allowed. No one may deposit or discharge or allow the deposit or discharge of manure, manure liquid or contaminated water into the environment except where the said deposit or discharge is carried out in accordance with this Regulation ¹⁴⁷

¹⁴³ Section 11(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act

http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=0779750411

¹⁴⁴ Section 11(2) Manitoba Livestock Manure and Mortalities Management Regulation 219/2006 The Environment Act C.C.S.M. c. E125

¹⁴⁵ Section 11(1) Manitoba Livestock Manure and Mortalities Management Regulation 219/2006 The Environment Act C.C.S.M. c. E125

¹⁴⁶ Section 63(6) Ontario Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.\(6\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#63.(6))

¹⁴⁷ Section 17 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act

<http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

Operational aspects	Alberta	Manitoba	Ontario	Quebec
		groundwater. ¹⁴⁵		
Frequency of waste removal	---	---	---	Once / year ¹⁴⁸ Must be cleaned out to provide adequate storage capacity while ground is frozen or snow-covered ¹⁴⁹
Fly control required	Yes ¹⁵⁰	---	---	---
Dust control required	As required by approval officer ¹⁵¹	---	---	---

¹⁴⁸ Section 15 Agricultural Operations Regulation , R.Q. c. Q-2, r.11.1, Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

¹⁴⁹ Section 36 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

¹⁵⁰ Section 20(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹⁵¹ Section 20(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

Table 5.18 Comparison of regulatory requirements. Note that cells with the “---” designate no specific information in the regulation for that jurisdiction.

Regulatory aspects	Alberta	Manitoba	Ontario	Quebec
Permit Required	<p>Yes.</p> <p>No person shall commence construction, expansion or modification of a manure storage facility or manure collection area for which an authorization is required pursuant to the regulations or commence construction, expansion or modification of a manure storage facility for manure that is in a predominantly liquid state or manure to which water has been added unless</p> <p>(a) the person holds an authorization that authorizes the construction, expansion or modification, or</p> <p>(b) the person holds an approval or registration that authorizes the</p>	<p>Yes.</p> <p>No person shall construct, modify or expand a manure storage facility except under the authority of a permit issued by the director under this section. ¹⁵³</p>	<p>Yes.</p> <p>Nutrient management strategy for an agricultural or non-agricultural operation requires the approval of a Director if, the operation is an agricultural operation and a person who owns or controls the land on which the operation is carried out submits an application for a building permit under the <i>Building Code Act, 1992</i> in respect of any building or structure that is used to house farm animals or to store manure and that is located or to be located on the land ¹⁵⁴</p>	<p>Yes.</p> <p>Certificate of Authorization from the Minister of Environment and Wildlife required for any installation or modification ¹⁵⁵</p>

Regulatory aspects	Alberta	Manitoba	Ontario	Quebec
	construction, expansion or modification. ¹⁵²			
Manure Management Plan Required		Yes. No person shall store, handle or dispose of livestock manure, or apply livestock manure to land, except in accordance with a manure management plan registered with the director in accordance with subsection (4). ¹⁵⁶	Yes. No person shall construct a building or structure on a farm unit on which the operation is carried out, where the building or structure is used to house farm animals or store nutrients, unless the nutrient management strategy has been prepared and, if applicable, approved in accordance with this Regulation. ¹⁵⁷	---

Table 5.19 compares the monitoring requirements in Canada. Only, Iowa requires monitoring for unformed manure storage structure¹⁵⁸. Alberta and Manitoba require the installation of monitoring wells. Ontario requires the monitoring wells to be designed by a Professional Engineer.

¹⁵² Section 14(1) Alberta Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Acts/A07.cfm?frm_isbn=9780779723478

¹⁵³ Section 16.1 (1) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. E125

¹⁵⁴ Section 27(1)(a) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#27.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#27.(1))

¹⁵⁵ Section 1 Prevention of Water Pollution in Livestock Operations, Regulation Respecting the, Quebec Environment Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.18/20070307/whole.html>

¹⁵⁶ Section 13(1) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. 125

¹⁵⁷ Section 11.1 Nutrient Management Act Ontario Regulation 267/03 http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#11.1

¹⁵⁸ Iowa Code Chapter 459.311(3) Animal Agriculture Compliance Act

[http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xsl&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=\[field folio-destination-name:'sec_459_311'\]&xhitlist_md=target-id=0-0-0-68229](http://nxtsearch.legis.state.ia.us/NXT/gateway.dll?f=xhitlist&xhitlist_x=Advanced&xhitlist_vpc=first&xhitlist_xsl=querylink.xsl&xhitlist_sel=title;path;content-type;home-title;item-bookmark&xhitlist_d={2007code}&xhitlist_q=[field folio-destination-name:'sec_459_311']&xhitlist_md=target-id=0-0-0-68229)

Table 5.19 Comparison monitoring requirements. Note that cells with the “---” designate no specific information in the regulation for that jurisdiction.

Monitoring aspects	Alberta	Manitoba	Ontario	Quebec
Installation of monitoring wells	As required by approval officer, install at least 1 well up gradient and 2 well down gradient of facility ¹⁵⁹	The director may require an operator to install monitoring wells in relation to a manure storage facility. The operator shall install and maintain the monitoring wells in accordance with the director's requirements or in a manner that is satisfactory to the director. ¹⁶⁰	Must be designed by a professional engineer ¹⁶¹	---
Monitoring schedule	As required by approval officer, must monitor wells at regular intervals ¹⁶²	In addition to requiring an operator to comply with subsections (1) to (3), the director may require an operator to implement a monitoring and reporting program if the director believes that the storage, handling	---	---

¹⁵⁹ Section 18(1) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

¹⁶⁰ Section 6.1(1) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. E125

¹⁶¹ Section 71(1)(a) Nutrient Management Act Ontario Regulation 267/03 [http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#71.\(1\)](http://www.e-laws.gov.on.ca/DBLaws/Regs/English/030267_e.htm#71.(1))

¹⁶² Section 18(2) Alberta Regulation 267/2001 Standards and Administration Regulation Agricultural Operation Practices Act http://www.qp.gov.ab.ca/documents/Regs/2001_267.cfm?frm_isbn=9780779722358

Monitoring aspects	Alberta	Manitoba	Ontario	Quebec
		and management of livestock manure in the agricultural operation is causing or would likely cause pollution of surface water, groundwater or soil. ¹⁶³		
Submission of water analysis reports		<p>The operator shall submit water analysis reports of water samples from the monitoring wells collected and analyzed in accordance with the sampling, analysis and reporting protocol for monitoring wells approved by the director under subsection (4) ¹⁶⁴</p> <p>the operator of an agricultural operation with 300 animal units or more shall submit an annual water analysis report of water from the operation's livestock drinking water source collected and analyzed in</p>	---	---

¹⁶³ Section 6.1(5) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. E125

¹⁶⁴ Section 6.1(2) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. E125

¹⁶⁵ Section 6.1(3) Manitoba Livestock Manure and Mortalities Management Regulation 52/2004 The Environment Act C.C.S.M. c. E125

Monitoring aspects	Alberta	Manitoba	Ontario	Quebec
		<p>accordance with the sampling, analysis and reporting protocol for water sources approved by the director under subsection (4) ¹⁶⁵</p>		

Table 5.20 Comparison of regulatory structure and penalties. Note that cells with the “---” designate no specific information in the regulation for that jurisdiction.

Penalties	Alberta	Manitoba	Ontario	Quebec
Regulating Body	Natural Resources Conservation Board (NRCB)	Manitoba's Environmental Livestock Program (Manitoba Conservation), Farm Practices Protection Board, Manitoba Conservation	Ministry of Agriculture, Food and Rural Affairs and the Ministry of the Environment	Minister of the Environment and Wildlife / Ministère de l'Environnement et de la Faune (MEF) ¹⁶⁶ Minister of Sustainable Development, Environment and Parks ¹⁶⁷
Penalty for Non-Compliance	May issue enforcement order against producer if producer is: -Creating a risk to the environment. -Causing an inappropriate disturbance. -Contravening the Act or regulation. ¹⁶⁸	---	---	A fine of \$2 000 to \$20 000 for a first offence and of \$5 000 to \$50 000 for any subsequent offence, for a natural person or a fine of \$2 000 to \$150 000 for a first offence and of \$5 000 to \$500 000 for any subsequent offence, for a legal person ¹⁶⁹

¹⁶⁶ Prairie Swine Centre. Legislation and Regulation for Swine Facilities

[http://www.prairieswine.com/porkinsight/environmental_issues/pdf/Legislation%20and%20Regulation%20for%20Swine%20Facilities%20\(EIRC\).PDF](http://www.prairieswine.com/porkinsight/environmental_issues/pdf/Legislation%20and%20Regulation%20for%20Swine%20Facilities%20(EIRC).PDF)

¹⁶⁷ Section 40 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

¹⁶⁸ Beneficial Management Practices: Environmental Manual for Hog Producers in Alberta

¹⁶⁹ Section 44 Agricultural Operations Regulation R.Q. c. Q-2, r.11.1 Quebec Environmental Quality Act <http://www.canlii.org/qc/laws/regu/q-2r.11.1/20070307/whole.html>

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CHAPTER 6 Assessment of Potential Hog Manure Processing Strategies in Manitoba

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6.1 Introduction

This section provides information regarding potential hog manure processing strategies, their use in other jurisdictions and the feasibility of implementing these strategies in the Manitoba context. General groups of manure processing unit operations discussed in this document include:

- Solid-liquid separation
- Anaerobic digestion
- Composting

These unit operations can be implemented individually or combined to create a multifaceted manure processing strategy. The following chart illustrates potential configurations of these unit operations along with key inputs and outputs:

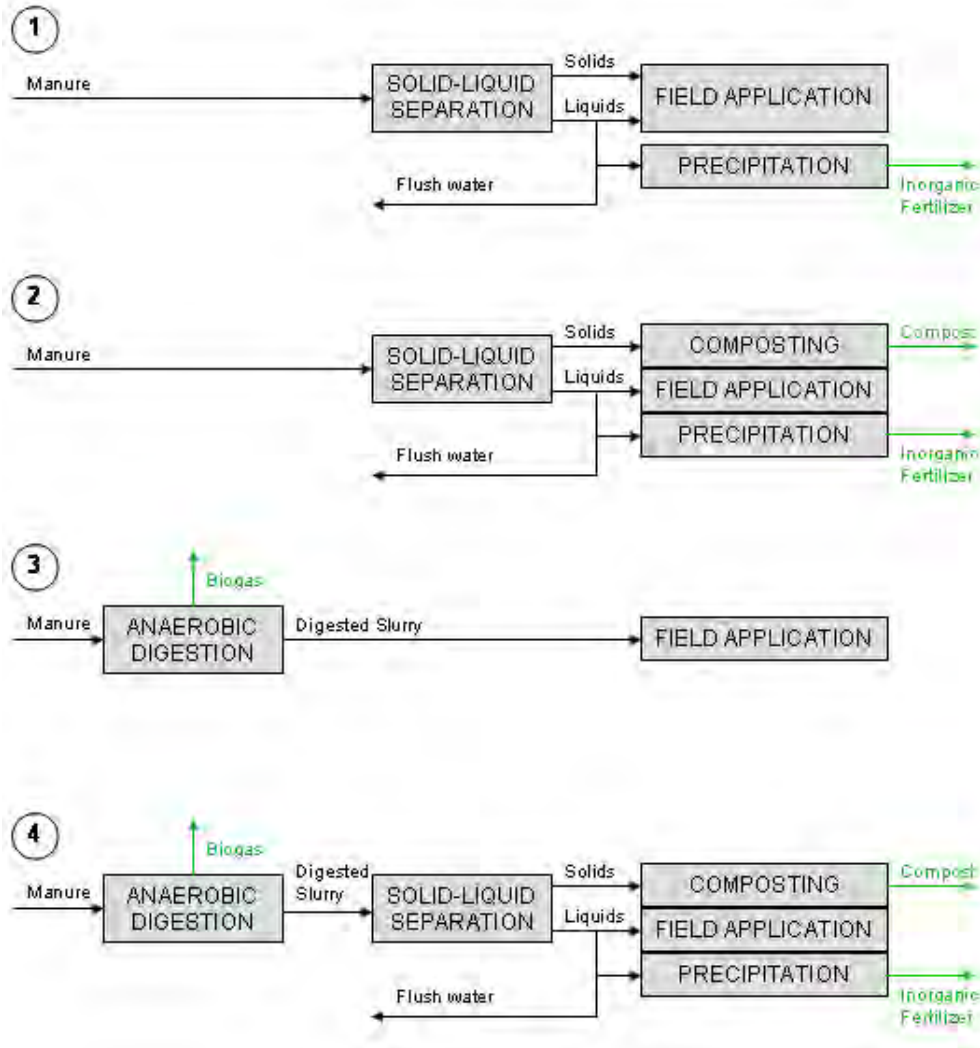


Figure 6.1: Manure processing configurations

The aforementioned manure processing strategies are combinations of techniques and technologies that achieve one or more of the following objectives related to the environment:

- nutrient management and the isolation of nitrogen and phosphorus
- renewable energy production
- odour control
- greenhouse gas (GHG) reduction

Since there are many differences between individual farms it must be recognized that there is no single best solution for manure processing and that a thorough investigation of available options and an understanding and respect for individual situations is necessary in order to decide which options should be implemented.

Each processing technology will be discussed in terms of technical principles, benefits, nutrient fates, practical application in a Manitoba context, and economic implications. These

technologies appear to be more widely used outside Manitoba, therefore, in order to understand existing uses other jurisdictions around the world will be explored. Until very recently, manure storage and manure land application as crop fertilizer, were considered “treatment” in Manitoba with the primary incentives being odour reduction and improving public perception of the hog industry. This is apparent from the following quote by Manitoba Agriculture, Food and Rural Initiatives (MAFRI) in 2001:

“For the majority of hog producers in Manitoba, most odour problems can be solved by a combination of good management and the separation of hog farms from residential areas. Manure produced can be easily disposed of on the available crop land. As long as energy and feed prices are relatively low, the most cost-effective treatment system is storage of the manure, followed by spreading the manure on cropland. In the future, however, there may be circumstances where another method of treatment is desirable.” (Manitoba Agriculture, Food and Rural Initiatives 2001).

However, new drivers and incentives for manure treatment have since emerged. These include reducing nutrient loading to surface and ground water bodies (primarily driven by the Lake Winnipeg Water Stewardship Board), renewed interest in green and renewable energy sources, a potential market for carbon credits arising from greenhouse gas mitigation, and higher inorganic fertilizer costs. While odour abatement and public relations continue to be strong motivations, a new regulatory framework, which is anticipated to follow the temporary moratorium on the expansion of hog facilities in the Province of Manitoba, will spurt the implementation of advanced manure treatment technologies. This section of the CEC report will focus on manure processing and treatment technologies, which are currently utilized in various parts of the world, and hold significant promise as possible technical solutions for a sustainable hog industry in the Province of Manitoba.

6.2 Solid-liquid separation

6.2.1 Process Description

Solid-liquid separation (separation) is a physical process which separates incoming slurry (solid-liquid mixture) into its two general components of solid particles and the liquid phase (generally water). The physical process can be augmented with chemical amendments in order to increase separation efficiency. Separation is a common process employed in a wide variety of industries, including water and wastewater treatment.

The two product streams generated from separation are the separated solids and separated liquids. Separated solids can be either field applied as a nutrient rich soil conditioner or composted. Separated liquids can be directly field applied, stored, recycled for flush water or processed further if required.

6.2.2 Benefits

Separation can be used to achieve the following objectives:

- Production of nutrient rich organic solids for field application
- Improved nitrogen to phosphorus (N:P) ratio in liquids for plant uptake after irrigation
- Volume reduction of solids for field application
- Odour reduction
- Facilitation of further processing such as composting and biogas generation
- Improved handling of liquid fraction
- Potential to recycle of liquid fraction for flush water in the facility
- Use of separated solids as animal feed supplementation

Separated solids can be used on farm land within proximity to hog operations or can be exported if the immediate land base is insufficient.

Since it has been shown that hog operations that generate surplus nutrients relative to crop requirements/land-base can lead to environmental problems, it is desirable to transport a portion of the nutrient rich solids to areas with nutrient deficits. Since hog manure slurry has a relatively low concentration of solids and nutrients, the cost of transporting the slurry itself is high. Effective separation will concentrate nitrogen and phosphorous in the solid fraction. With separation there will be a reduction in the volume of material required for transport leading to lower costs. If the liquid portion of the separated manure is land applied the nutrient loadings to the soil will be reduced per volume applied.

According to Zhang and Westerman (1997), separation reduces odour generation rates in liquid manure storage and treatment units. Less nutrients and organic carbon in the liquid phase means less food for odour producing bacteria, and since much bacteria is bound to solids, there will be fewer bacteria in the liquid phase after separation. Bicudo (2001) indicates that research conducted in the UK displayed that separation extended storage time of the liquids by one-third before offensive odours returned.

Separation of the solids from dilute hog manure will allow the application of composting. Typical hog manure is too dilute to be composted directly without significant addition of bulking agents such as straw, woodchips, etc. If composting is desirable, then some form of solid separation is necessary (Paul 2005).

Separator effluent (liquids) exhibit less potential to plug transfer pipes due to lower solids and reduced particle sizes, subsequently, less energy will be required to pump the material. Increased flowability of liquids facilitates more reliable operation of irrigation systems where liquids are pumped long distances (Ford and Fleming 2002).

With further processing, separated liquids can be recycled as flush water or grey water for other on farm uses (Ford and Fleming 2002). This effectively reduces freshwater consumption in the facility, which could be of great value in some of the more arid regions.

6.2.3 Technology Options

There are many commercial separation technologies available. Primary classes of separation technology include:

- Gravity settling
- Screening
- Centrifugation
- Filtration/pressing

Each technology presents a unique set of benefits and drawbacks that must be carefully researched and validated. Ultimately, the ideal separation technology should balance performance goals with capital cost, operating cost and ease of use.

6.2.3.1 Gravity Settling

Solids will naturally settle out of any manure storage structure due to gravity. Settling is a batch process which requires a certain amount of time to achieve the required level of separation and

nutrient reduction. Depending on manure composition and the desired degree of separation, this may take hours, days or weeks. A settling tank or basin is a basic physical installation such as a lined pit or large atmospheric tank. Settling tanks or basins require less capital over more complex separation technologies such as centrifugation or filtration. Settling rates are limited in terms of time and basin capacity.

Chemical amendments can be added to manure slurry to increase the speed and efficiency of settling. Chemical amendments would be added to the settling tank or basin under vigorous agitation then once the amendment is mixed thoroughly, agitation ceases and settling begins. Typical chemicals used to augment settling include (i) chlorides and sulfates of calcium, iron, and aluminum; (ii) lime; and (iii) cationic polymers (Zhang and Westerman 1997; Taylor 2001; Toth et al. 2001).

6.2.3.2 Screening

Screening is a separation technique where a screen of a specific pore/mesh size is used to allow solid particles smaller than the pore openings to pass through and particles larger than the pore openings to be blocked, accumulating on the surface of the screen. There are many different ways to configure a screening device. Common configurations are stationary inclined screens, vibrating screens, and rotary screens (Ford and Fleming 2002; Zhang and Westerman 1997; Shutt et al. 1975; Hegg et al. 1981).

Separation of solids and nutrients using screening technology is dependent on solid particle size distribution and the extent to which nutrients are associated with solid particles. Screening alone may not be a sufficient technique for solid-liquid separation of hog manure because of the relatively small solid particle size associated with hog manure (Buckley and Gross). Screening equipment is subject to fouling if screens are not cleaned consistently. This will result in highly variable and unreliable separation performance (Ford and Fleming 2002). Brushing to remove adhering organic matter is necessary since pressure washing alone may be insufficient.

6.2.3.3 Centrifugation

Centrifugation devices use centrifugal force to increase the settling velocity of suspended particles. Common centrifugation devices include the decanter centrifuge and the hydrocyclone (Zhang and Westerman 1997; Shutt et al. 1975). Similar to gravity settling, chemical amendments can be added to augment performance of centrifugation.

6.2.3.4 Pressing

Presses use mechanical pressure to achieve separation of manure. Presses tend to be used following other separation devices to remove additional liquid from the separated solids. In this sense, pressing can be viewed as a supplementary dewatering technique as opposed to a primary separation technique (Ford and Fleming 2002). Pressing technologies include the roller press, belt press, screw press and the filter press.

6.2.4 Selecting the Appropriate Technology (Performance)

Performance and feasibility of solid-liquid separation can be judged based on the following criteria:

- Manure characteristics (particle size distribution, initial total solids concentration)
- Total solids reduction
- Nutrient removal
- Odour reduction
- Capital and operating costs
- Labor requirements

- Final solids management options

A standard measure of the performance of solid-liquid separation is separation efficiency (in terms of solids or nutrient removal). Separation efficiency is commonly expressed in terms of % removal. To calculate % removal, the constituent of interest must be measured prior to and following separation.

$$\% \text{ removal} = \frac{\text{influent concentration (\%)} - \text{effluent concentration}}{\text{influent concentration (\%)}} \times 100$$

Initial manure solids concentration will provide preliminary selection criteria as shown in Table 6.1:

Table 6.1: Appropriate technology based on manure total solids

Raw manure initial TS level	Works Best With...
< 5.0%	Gravity settling ^a , Screening ^{b,c}
> 5.0%	Centrifuge ^{b,d} , Press ^b

^aPieters et al. (1999); ^bZhang and Westerman 1997; ^cBicudo 2001; ^dSheffield et al.

Table 6.2 summarizes separation efficiency data for four of the most common on-farm separation technologies. The table is based on data collected from numerous sources in which separation technologies were evaluated with hog manure. Test conditions reported in the literature are highly variable with respect to manure dilution/total solids concentration, influent flow rate, operating parameters, and duration of testing.

Table 6.2: Separator performance data

Parameter	Particle Size Range (Raw Manure)	Technology	Separation Efficiency (% removal)
Total Solids (TS)	40% < 0.45 μm 25% 0.45 – 10 μm 15% 10 – 50 μm 5% 50 – 250 μm 5 % 250 – 1000 μm 10% > 1000 μm ^b	Settling	40 ^d 55 ^e 64 ^f
		Settling (w/ chemical amendment)	65-70 ^e
		Screening	3-31 ^a
		Centrifugation	15-61 ^a
Phosphorous (TP)	20% < 0.45 μm 50% 0.45 – 10 μm 30% > 10 μm ^b	Settling	38 ^g 40 ^d
		Settling (w/ chemical amendment)	75 ^{d,g} 70-90 ^e
		Screening	2-12 ^a
		Centrifugation	43-68 ^a
Nitrogen (TN)	85% < 0.45 μm ^h	Settling	20 ^e
		Settling (w/ chemical amendment)	30-35 ^e
		Screening	3-6 ^a
		Centrifugation	3.4-32 ^a

^a Shutt et al. 1975; Hegg et al. 1981; Piccinini and Cortellini 1987; ^b L. Masse et al. 2005;

^d Taylor and Wood 2001; ^e Kruger et al. 1995; ^f Mukhtar et al. 1999;

^g Worley and Das 2000; ^hBicudo 2001

Manure composition is highly variable between test cases. Reported performance values may not represent actual performance under all conditions. Performance criteria must be identified early, performance goals must be set and performance must be verified at lab and pilot-scale in order to gain confidence with a particular technology and accurately assess capital and operational costs. Due to variability in test conditions it is important to recognize that the performance data serves as a starting point for further evaluation rather than a definitive statement about the performance of generic separator techniques.

Separation is best performed on fresh manure. Separation efficiency tends to decrease if manure is stored prior to separation due to decomposition. Zhu et al. (2000) suggest that in order to maximize separation efficiency manure should be separated within five days of storage. Francis Pouliot from the Centre de developpement du porc du Quebec Inc. (CDPQ) stressed that separation efficiencies were greater with fresh manure (Pouliot 2007).

Settling with chemical amendments and centrifugation display the highest phosphorous reduction potential. Chemical treatment has been shown to effectively remove a portion of dissolved phosphorous, increasing separation efficiency beyond the limits of a purely physical process.

6.2.5 Nutrient Fates

Phosphorus and nitrogen are differentially distributed in hog manure, either bound to the solids or dissolved in the liquid portion. Table 6.3 lists the forms in which these nutrients are found in hog manure. It has been suggested that most of the carbon compounds, proteins, and nutrient elements are contained in fine solid particles (Zhang and Westerman 1997; Sheffield et al.).

Table 6.3 Nutrient forms in hog manure

Phosphorous	Nitrogen
<ul style="list-style-type: none"> • Organic phosphorous • Inorganic phosphorous 	<ul style="list-style-type: none"> • Organic nitrogen • Ammonium nitrogen (aqueous phase) • Ammonia nitrogen (gas phase) • Nitrite • Nitrate

Phosphorous is generally more easily removed relative to nitrogen by separation since it is relatively insoluble and primarily associated with the solid fraction. Nitrogen in swine manure is heavily present in the form of ammonia, which is very soluble and is therefore not removed by solid separation (Atlantic Swine et al.). The organic fraction of the total nitrogen content in hog manure accounts for almost all nitrogen concentrated in the solids after centrifugation. The organic fraction represents approximately 15% of total nitrogen and total nitrogen in solids increases with increased manure solids content (Bicudo 2001). Thus, physical separation techniques will only transfer the organic nitrogen to the solids and not the majority of nitrogen which is dissolved ammonium (Møller et al. 2000). It has also been reported that only 13% of total phosphorous in hog manure is soluble, therefore a high amount of phosphorous compounds are concentrated in the solids (Bicudo 2001). It has been reported that over 95% of organic nitrogen and 50% of phosphorous is linked to particles between 0.45 and 10 μm (Masse et al. 2005). Therefore, a solid-liquid separation scheme must be capable of removing particles in this range in order to realize significant reductions in nitrogen and phosphorous.

There has been a general misconception that all separation technologies are capable of concentrating phosphorus in the separated solids. However, due to the association of P with particles of very small sizes, in many of the studies reviewed, separation could transfer less than 30% of the TP into the solids fraction for swine manure (Ford and Fleming 2002). Masse et al. (2005) suggested that except for a decanter centrifuge, most commercial solid-liquid separators would not be efficient for nutrient separation.

Along with the overall removal of nitrogen and phosphorus from the liquid stream by solid separation, it is also important to consider the new nitrogen to phosphorus ratio (N:P) of the effluent, which will generally be used for land application purposes. According to Dr. Bittman at AAFC, crops need approximately 10:1 of N:P, but hog manure, as excreted, has a N:P ratio ranging from approximately 4:1 to 5:1 (Bittman 2007). Assuming an incoming manure N:P of 5:1, solid liquid separation technologies may yield a new ratio in the liquid stream as shown in Table 6.4.

Table 6.4 N:P ratios in the liquid stream following various solid separation processes

Technology	New N:P of effluent
Settling	6.67:1
Settling (w/ chemical amendment)	13:1
Screening	5.34:1
Centrifugation	10.65:1

This indicates that solid separation produces a liquid fertilizer with an improved N:P ratio for plant growth. This could have significant benefits for farmers as well as the environment, preventing excessive P loading to agricultural fields and reducing mobilization to surface water.

6.2.6 Solid-Liquid Separation in Practice

Separation is a common process applied throughout a wide variety of diverse industries. The knowledge base for solid-liquid separation in the context of hog manure is not as extensive as conventional applications such as wastewater treatment. There remain significant knowledge gaps and misconceptions about the role of separation in the livestock industry. Much can be learned by examining existing operations utilizing separation.

Jamieson et al. (2001) suggest that the limited use of separation technologies in the North American hog industry is due to cost and that separation technologies have been more readily accepted in Europe and Asia due to shrinking or inadequate land base. However, the continual intensification of livestock operations and heightened environmental awareness in North America, are strong drivers for the application of solid separation technologies in the US and Canada.

Solid separation is uncommon in Manitoba, with few examples of ongoing use. In contrast, there are a number of operations in Quebec, which use some form of separation. Cited reasons for the advanced development in Quebec are incentives for manure treatment including (Pouliot 2007):

- Stricter environmental regulations
- Prime-Vert program (provincial subsidy)
 - 70% of costs (max \$200,000)
- Provincial income tax credit
 - 30% of costs (max \$200,000)
- Écoconditionnalité

- Linking agricultural assistance payments to respect of environmental standards

In a presentation at the recent Manure Management Conference in Winnipeg Francis Pouliot reported that 13 different marketers of separation technologies have installed 43 different separation units of various sizes (Pouliot 2007). The primary examples cited include:

- Vibrating sieve (1 installation)
- Centrifuge
 - Alfa Laval technology
 - Commercial scale use, 4500 head finishing operation, installed February 2007
- IRDA Mobile centrifuge
- Maximizer sieve
 - Demonstration installation only
- Lisox™ system
 - Developed by Corporation HET (Horizon Environment Technologies)
- Sequencia system
 - 3 step process
 - Separation by roller press
 - Chemical separation w/ dissolved air and polymers
 - Liquid fraction treated by an aerobic biological process

Based on conversations with Marc Trudelle from Manitoba Conservation, Quebec borrowed knowledge extensively from Europe from a regulatory perspective and regarding appropriate separation technology. For instance, there are approximately 150 centrifuges in use at hog operations in France. Delegates from Quebec spent one month in France a number of years ago touring operations and sharing knowledge (Trudelle 2007).

6.2.7 Feasibility in Manitoba Context

With current emphasis on phosphorous removal and odour reduction, as well as spiraling fuel costs affecting transport and land application economics, solid separation of hog manure has garnered renewed interest in Manitoba. This has not yet resulted in large number of installations across the Province. Brandon based Home Farms Technologies Inc. produced a multi-stage solid-liquid separator that is currently in use at Green Acres Colony, in Wawanesa, Manitoba. This represents the only full-scale manure separation system that could be confirmed in Manitoba. In this process, manure is transferred from the barn to an 18,000 gallon holding tank where it is agitated to keep solids suspended. It is then sent through a screen to remove large solids and a flocculation chamber for additional solids removal (Hofer, 2007).

This system is capable of removing 62-70% of total solids based on information obtained in conversation with Katherine Buckley from AAFC Brandon, who administered the research and monitoring associated with the Home Farms system at Green Acres (Buckley 2007). According to conversation with John Hofer, who manages the system at the colony, it has been operational six days per week for the last four years. Green Acres stockpiles the separated solids and with the addition of dead livestock, achieves a crude form of composting. A more sophisticated composting process involving a rotating-drum type system is being planned at the colony, which would use separated manure solids, residential organic waste, and livestock mortality as mixed feedstock (Hofer, 2007).

Based on personal communications with Dr. Shokry Rashwan, Project Engineer at, the company is currently exploring suitable separation technology options to be used in their

Manitoba hog facilities. Potential phosphorous regulations and the rising cost of handling manure using existing practices were stated as the primary drivers for implementation of separation. Puratone is actively investigating three options including a proprietary concept that is based on filtration principles, centrifugation, and settling using chemical amendments. They have yet to decide which route they will take and a decision on how to proceed with solid separation is expected by the end of 2007 (Rashwan 2007).

Puratone in conjunction with Manitoba Conservation have been evaluating centrifugation using the Research and Development Institute for the Agri-Environment (IRDA - Institut de recherche et de développement en agroenvironnement) mobile centrifuge from Quebec (Trudelle, 2007). The mobile centrifuge consists of an Asserva-300 decanter centrifuge mounted inside a trailer. The mobile centrifuge is removing up to 70% of the phosphorous and the solids are approximately 35% dry matter. The hog producer Hytek and to a lesser extent Puratone, appear interested in exploring mobile separation systems in an attempt to minimize overall cost (Pohlman, 2007). However, mobile separators may present significant challenges associated with logistics (scheduling, maintenance) and bio-security (Pouliot 2007; Trudelle 2007).



Figure 6.2: IRDA Mobile Centrifuge stationed in Manitoba (Trudelle 2007)

6.2.8 Economics

In general, separation economics are based on (a) the amount of manure to be processed; (b) the solids removal efficiency; and (c) the potential for solids use (value recovery). Table 6.4 shows a comparative cost analysis for two different sizes of hog operations and three different solid separation technologies. It does not, however, include the costs associated with solid and liquid storage infrastructure.

There are a number of reports that identify economic data based on individual cases. According to Møller et al. (2000) to treat 4000 tonnes of manure slurry it will cost £0.44 (\$0.95 CAD) per tonne using a screen separator and £2.21 (\$4.77 CAD) per tonne using a decanter centrifuge. Bicudo (2001) suggests that the total cost of a screw-press separator on a 3,600 head, finishing farm would be approximately \$0.44 per finished hog (\$0.35 for ownership of the equipment and \$0.09 for operational costs). On the other hand, Pouliot (2007) suggest that the total annual costs of solid separation can vary between \$9-19 per m³ of manure treated, which represents approximately \$6-12 per pig produced.

Table 6.4: Costs of various solid separation techniques as adapted from three separate case studies (E.A. Systems Pty Limited 2002)

Common specifications	Parameter	Units	200 sow	2000 sow
	Total effluent	ML/year	9	85
	Effluent solids content	% TS	3.1	3.3
	Solids	t/year	270	2800
Technology				
Trafficable Settling Basin (Gravity Settling)	Solids removal	%	50	50
	Capital cost	\$	8000	30,000
	Operating cost	\$/year	14,900	52,300
	*Total Cost	\$	26,565	95,165
	Total Cost/m ³	\$	2.95	1.12
	Total cost/tonne solids	\$	98.38	33.98
Flamingo Vibrating Screen	Solids removal	%	20	20
	Capital cost	\$	31,000	68,500
	Operating cost	\$/year	3156	7551
	*Total Cost	\$	34,932	77,908
	Total Cost/m ³	\$	3.88	0.92
	Total cost/tonne solids	\$	129.37	27.82
ALDEC Centrifuge Decanter	Solids removal	%	30	30
	Capital cost	\$	115,000	152,500
	Operating cost	\$/year	9175	16,241
	*Total Cost	\$	126,432	172,736
	Total Cost/m ³	\$	14.05	2.03
	Total cost/tonne solids	\$	468.26	61.69

*The total cost is based on net present worth assuming amortization of the solid separation units over 10 years at 8% discount rate.

Freeze et al. suggest that the fertilizer value of manure solids can range between approximately \$240-\$417 per tonne depending on nutrient composition. Based on this potential revenue it appears as though separation could be a potentially profitable venture under certain circumstances. It must be stressed that these are relatively approximate figures that ignore certain costs associated with transport of the treated manure to the user, and assume that an stable market for the solids would exist.

Although the total costs of centrifugation are relatively high, the increased efficiency produces solids that have significantly superior fertilizer value to other methods, which should lead to a higher value product.

Despite the fact that no commercial revenue streams are currently arising from benefits such as improving N:P ratio of the effluent, reducing odour potential, facilitating further processing (i.e. composting), and improving handling of the liquid fraction, these intangibles could add significant value to solid separation.

6.3 Anaerobic Digestion

6.3.1 Process Description

Anaerobic digestion (AD) is a natural process of micro-organisms breaking down organic materials in the absence of oxygen. AD reduces manure solids content and produces biogas consisting primarily of methane and carbon dioxide (Figure 6.3). Solid and soluble organic carbon compounds are ultimately converted into gaseous methane and carbon dioxide. Depending on system design, biogas can be collected and combusted to run a generator producing electricity and heat, or it can be burned as a fuel in a boiler or other burner.

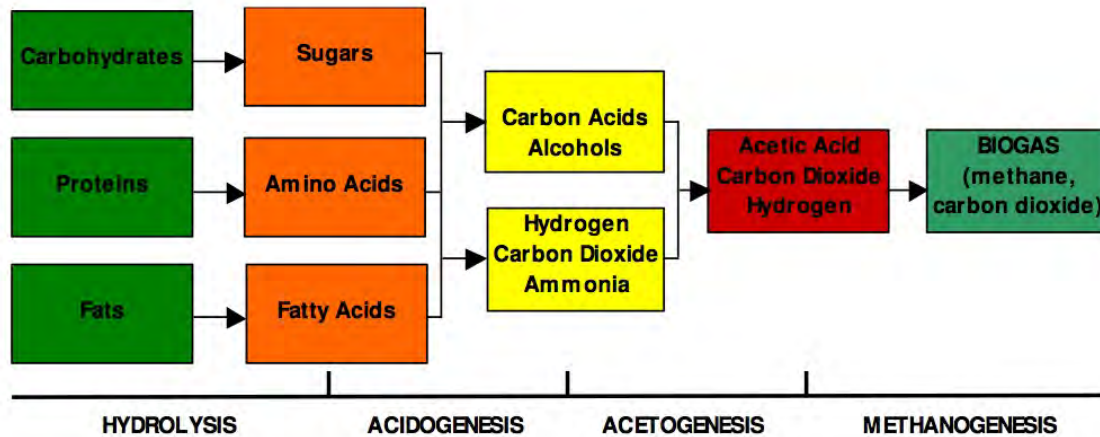


Figure 6.3 Basic anaerobic digestion process flow description (McNeill, 2005).

Figure 6.4 illustrates the basic components required for anaerobic digestion on a typical hog farm. Biogas can be used as a fuel source in nearly all devices intended for natural gas with minimal adjustment to account for the lower Btu content. Along with the biogas, the AD process also generates two other primary products:

- Fibre – can be used as nutrient-rich soil conditioner
- Liquor – can be used as liquid fertilizer

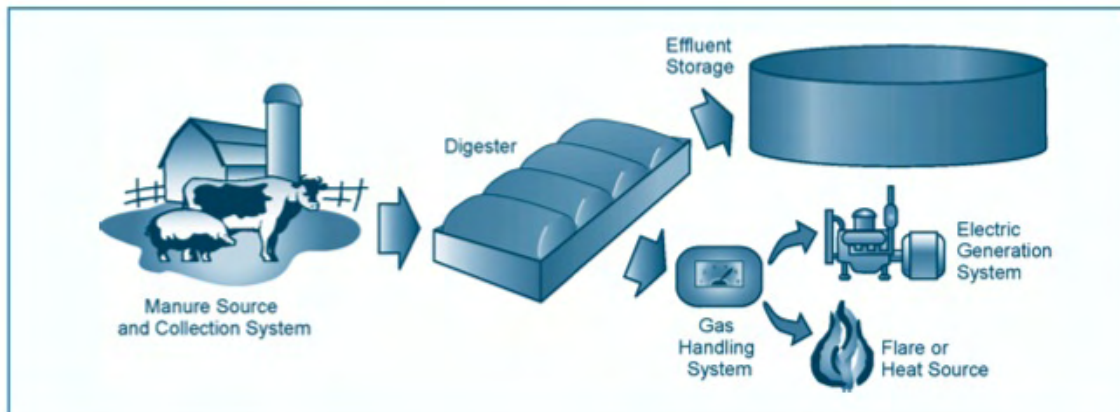


Figure 6.4 Typical anaerobic digestion equipment flow (US EPA, 2002)

6.3.2 Benefits

Anaerobic digestion shows promise in terms of nutrient management as it causes the breakdown of organic materials rich in nutrients, releasing them into the liquid phase of the slurry. This results in the improved fertilizing value of the slurry and makes nitrogen and phosphorous more available to the plant. The rheological properties of the treated liquid manure also lend themselves to easier handling for manure application on cropland (Burton 2003).

Biogas consisting mainly of methane and carbon dioxide has been shown to have heat values between 500 and 700tu/SCF (EPA 2006; Capstone 2007). Natural gas has a heat value of approximately 1000 Btu/SCF (Tiratsoo 1973). When organic carbon in manure is converted to biogas during AD and used to generate heat or electricity, fossil fuels previously used to provide heat and power can be displaced, reducing the greenhouse gas footprint of the farming operation. Additionally, manure that is left untreated and exposed the atmosphere will be a net emitter of greenhouse gases such as methane, carbon dioxide and nitrous oxide (Burton 2003).

Anaerobic digestion takes place in a sealed vessel where odourous gases are contained during the digestion process. The biogas containing the odourous gases such as hydrogen sulfide is either stored or used immediately on site. The liquid effluent, which is biologically stabilized by the digestion process, has reduced levels of offensive odour potential (Burton 2003)

If anaerobic digestion is conducted at high temperature (thermophilic) conditions of 55C and over, significant pathogen reduction is possible. This will reduce the potential public health risks associated with the application of manure on cropland and allow safe utilization of hog manure in vegetable growing operations and greenhouses.

6.3.3 Technology

An anaerobic digestion system on a farm operation would generally be made up of five elements: (1) the hog barn and manure source; (2) the anaerobic digester vessel; (3) the gas handling unit, including upgrading and storage; (4) the electricity and heat generating equipment; and (5) the effluent storage and handling systems.

In order to determine specifically what type of processing technology is appropriate, manure properties need to be characterized. Figure 6.5 illustrates how total solids content and manure classification determine handling options, suitability for biogas production and appropriate digester type. Table 6.5 summarizes technology options, relative cost, and suitability for hog manure. Generally covered lagoons and complete mix digesters are considered most appropriate and cost effective for hog manure.

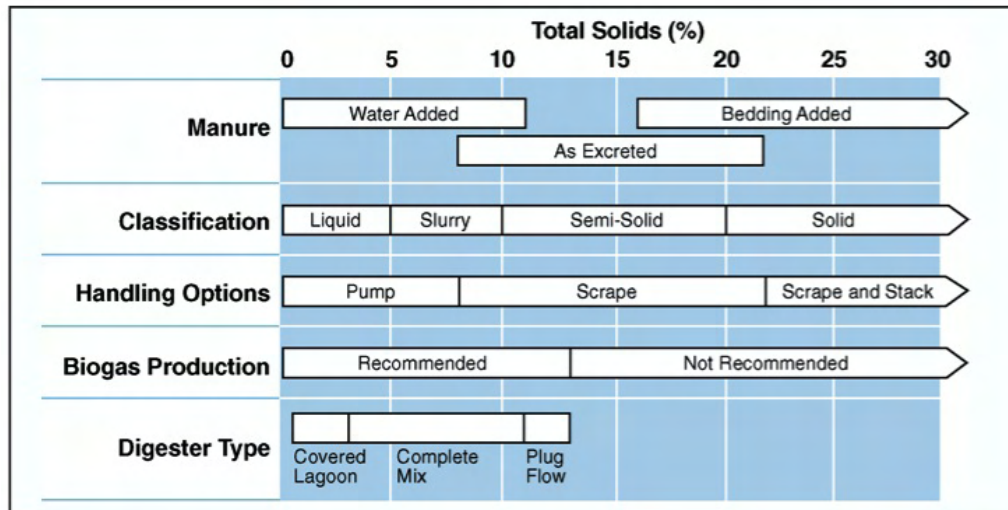


Figure 6.5 Summary of basic considerations for appropriate manure handling and processing based on solids content and fluid handling characteristics. (EPA-430-F-02-004)

With controlled anaerobic treatment processes, such as above ground complete mix anaerobic digesters, the temperature and nutrient levels of the manure are regulated so that only desirable gases and end products are produced. Whenever manure is stored in a pit, pile or lagoon, the manure decomposes anaerobically, but because the process is not controlled, many different gases can be formed at varying rates. Conventional storage lagoons operate on the same biological principle as anaerobic digesters, however they are not controlled and operated in the same fashion resulting in gas production rates that are low and unpredictable (US EPA 2002).

Table 6.5 Overview of basic technology options, their cost and suitability with respect to processing hog manure (US EPA 2002).

Technology option	Relative cost	Suitability for hog	Basic description
Covered lagoon	Low	High	Earthen pit with plastic cover
Complete mix	High	High	Reactor with agitator and controls
Plug flow	Med	Med to low	Concrete heated pit with plastic cover

Biogas composition and end products formed by the anaerobic decomposition will depend upon operating temperature and chemical characteristics of the manure. The most common anaerobic process is carried out at mesophilic temperatures (30-40°C), which allows rapid growth of methane-forming bacteria. Alternatively, thermophilic digestion is also used for biogas production, where temperatures are kept between 55 to 60°C. Thermophilic digestion processes have been reported to be less stable than mesophilic digestion processes (Burton 2003). More recently, bacteria strains were selected for a strong activity at psychrophilic temperatures (15 - 20°C) with liquid hog manure (Burton 2003).

Anaerobic digesters are designed and managed to optimize bacterial decomposition of organic matter under controlled conditions (mixing, temperature, manure loading, etc). One of the most common anaerobic reactors used for the treatment of swine manure is the covered lagoon digester. In this system, manure is added to the lagoon, allowing the effluent to overflow and be removed from the other end of the lagoon. European-based anaerobic digester manufacturers

build turnkey digesters coupled to electrical power generators. These systems take on a more classic fermentation vessel configuration and show promise to be the most efficient systems despite the relatively high initial cost.

Scale of AD Systems

Farm-based: These systems are typically designed for manure from a single farm or from several nearby small farms. They are lower cost and often involve a lower level of control and complexity than a large centralized system. Farm-based systems have been successfully operated throughout Asia, Europe, and parts of North America. Farm-based systems at large farms may come closer to approximating centralized facilities (DeBruyn 2004).

Centralized: Centralized AD systems are found in Europe. Manure from many farms is hauled to a centralized facility operating with a high biosecurity hauling process. Off-farm materials, such as food processing wastes, are often added to boost gas production. Treated manure is often transferred to remote field storages to allow for easier handling for land application (DeBruyn 2004).

Farm-based and centralized systems will require the farmer to procure new equipment and train on the use of new equipment and processes. Some specialized training will be required for the operation and maintenance of the digestion system, the gas train, and the heat and electricity generators.

6.3.4 Nutrient Fates

As shown in Figure 6.6, nitrogen, phosphorus, and potassium are transformed during anaerobic digestion, but these nutrients are not destroyed. Studies suggest that the total amount of N and P remains about the same in the influent and effluent of a digester; however, there is a shift from organic forms to inorganic forms. Studies show ammonia increase by 37% and soluble P increase by 26%. Thus anaerobic digestion increases the plant availability of N and P, but also increases the susceptibility to loss by leaching, run-off, and volatilization (Aldrich 2005). On the positive side, this means that the effluent will be more predictable in nutrient release, allowing the farmer to possibly reduce commercial fertilizer use more than they would have with raw manure. Conversely, the increased nutrient availability may increase nutrient loss if crops are not available for uptake.

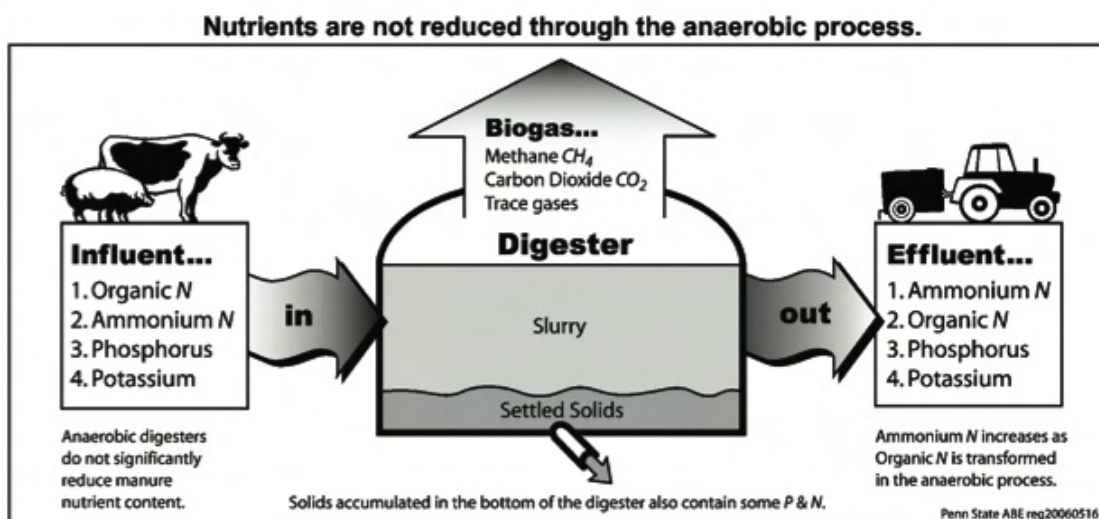


Figure 6.6 The Fate of Nutrients During Anaerobic Digestion of Dairy Manure (Topper 2006).

Digesters will not solve phosphorous run-off issues. Carbon is the only nutrient consumed in a digester, so total nitrogen and phosphorus concentrations are the same for raw and digested manure. However, separation systems and chemical amendments can be integrated into a digestion unit to capture phosphorus for other on or off-farm uses (MacLeod, 2004).

6.3.5 Anaerobic Digestion in Practice

Anaerobic digestion is an advanced processing technology with applications throughout many processing industries. Theory, equipment, and practical applications of anaerobic digestion for hog manure management are more established in countries outside of Canada.

Anaerobic digestion has been successfully coupled with combined heat and power plants in countries all around the world. China and Asia have several million small-scale biogas plants (Burton, 2003). In Europe there are several thousand AD installations and the increasing cost of energy is making AD and the resulting green energy more popular. More than 3,000 such plants have recently been built in Germany aided by a guaranteed premium price for the electricity produced (Burton 2003; German Biogas Association 2006). European companies are now considered world leaders in development of anaerobic digestion technology.

Total worldwide units currently in operation are estimated to exceed six million. Over 95% of small, farm based anaerobic digesters are located in Asia. Total worldwide electricity from biogas is estimated to be close to 6300 MW (installed capacity), with the majority of large-scale installation in Europe (i.e. Germany, Denmark, Italy, Austria, Sweden).

Experiences with Anaerobic digestion systems in Canada have been limited. The Ontario Government reports that all manure-based anaerobic digesters built in Ontario in the 1980s failed due to poor economic returns or operational difficulties. However, they also state that new technologies and control systems have seen the reconsideration of AD in agriculture. It was reported that there are currently two new agricultural digestion systems operating in Ontario with several proposals in the development or construction stages (DeBruyn 2004).

There are currently three Manitoba anaerobic digester installations, which are being used to study technical and economic feasibility of the technology. The anaerobic digester system at Cook Feeders in Teulon is in place and undergoing the finishing stages. Two others, at Topeaka Farms in Grunthal and Riverbend Colony near Carberry, are presently in the design and construction stages. A summary of the three sites is shown below:

Topeaka Farms, Grunthal, Manitoba

- 5500 head operation
- Thermophilic

Riverbend Colony, Carberry, Manitoba

- 1500 head operation
- Mesophilic or Thermophilic

Cook Feeders, Tuulon, Manitoba

- 6000 head operation
- Psychrophilic-BioTerre System

6.3.6 Feasibility in Manitoba Context

The feasibility of AD in Manitoba strongly depends on a regulatory framework to create an attractive environment for the promotion of the technology for renewable energy generation. One of the most significant barriers is the cold Manitoba climate. It is very difficult to justify the

investment in an anaerobic digester based only on the revenue received from natural gas displacement, electricity displacement or the sale of surplus electricity. The US EPA AgStar program suggests that anaerobic lagoon style digesters are only feasible for energy production south of the 40th parallel (Figure 6.7) (US EPA, 1997). More research needs to be done in terms of testing AD configurations adapted to extreme temperature fluctuations observed in Manitoba. Particularly the feasibility of innovative temperature control systems and improved insulation needs to be investigated.

QuickTime™ and a
TIFF (Uncompressed) decompressor
are needed to see this picture.

Figure 6.7 Feasibility of unheated covered lagoons in the (US EPA 1997).

Figure 6.7 suggests that AD installations above the 40th parallel would be used solely to flare biogas for the purpose of odor control and green house gas reduction, but not for year round energy production. However there are installations operating successfully in Canada in particular regions with similar climate challenges to Manitoba, such as the 1200 sow farrow-finish complete-mix AD in Saskatoon, SK (Figure 6.8).



Figure 6.8 Operational AD in Saskatoon, SK (Nova Scotia Agricultural College, factsheet)

The type of digester used can explain the difference. Most installations south of the 40th parallel are covered lagoons with little or no insulation or supplemental heat. The AD installation in Saskatoon is a complete-mix, above-ground, insulated digester with temperature control (MacLeod 2004). However, unless a premium price (minimum \$0.20/KWh) for electricity generated from biogas can be obtained by the AD operator, installations such as the one in Saskatoon, SK will remain economically unviable. Recent reports suggest that this facility has

now stopped operation primarily on the basis of economic conditions (not technical difficulties). The farm operation must be able to attribute some added economic value to the nutrient control, GHG reduction, and odor mitigation benefit in order to justify the additional cost it must bear for the installation and operation of an AD.

The active Manitoba pig population is estimated at 2.5 million and could produce as much as 37.5MW of electrical power based on estimated 15W per 100kg hog. This would be a significant addition of green power to the provincial energy portfolio and displace hydroelectric energy for the export market. This is, however, complicated by the low electricity rates in Manitoba and the lack of regulated subsidies for AD technology.

6.3.7 Economics

Table 6.6 presents an overall cost range for a variety of AD systems that can be applied to treat hog manure, while Table 6.8 present costs of installed AD systems identified in literature.

Table 6.6 General cost range for covered lagoons and AD systems (based on US EPA 2002)

Technology option	Cost range per 100 kg hog
Covered lagoon digesters with open storage ponds	\$30 to \$80
Heated digesters with open storage tanks	\$40 to \$80
Aerated lagoons with open storage ponds	\$40 to \$90
Separate treatment lagoons and storage ponds (2-cell systems)	\$40 to \$80
Combined treatment lagoons and storage ponds	\$40 to \$80
Storage ponds and tanks	\$5 to \$100

Table 6.8 Costs of installed and operational AD systems (US EPA, AgStar Program)

Location & Year Built	Size	AD type	Installed cost	End use	Annual savings	GHG reduction*
California, 1982	300 sow farrow to finish	NA	\$220,000	Electricity and hot air	unknown	741
Colorado, 1999	5,000 sow, 1,200 growers	Complete mix digester	\$368,000	Electricity	unknown	1482
Illinois, 1998	8,300 finishing hogs	Heated Mixed Covered Lagoon	\$140,000	Hot water and flare	unknown	2380
Iowa, 1998	3,000 nursery pigs	NA	\$15,000	Flare	unknown	1738
Iowa, 1998	5,000 sows farrow to wean	Complete mix digester	\$500,000	Electricity	unknown	1482
Mississippi 1998	145 pigs	NA	\$27,000	Flare	unknown	84
North Carolina, 1997	4,000 sows farrow to wean	Covered lagoon	\$290,000	Electricity and hot water	\$29,000	2317
Pennsylvania, 1985	4,000 swine	Complete mix digester	\$225,000	Electricity hot water; flare	unknown	3854
Virginia, 1993	600 sows farrow to feeder	Covered lagoon	\$85,000	Electricity	unknown	397
Corneche, Chile, 2002	102,000 finishers	Covered lagoon	\$700,000	F	odor	NA
Pennsylvania, 1986	17,000 farrow to finish	Complete mix digester	\$250,000	E, H, F	\$72,000	NA
Gypsy Hill, USA, 1983	16,000	Complete mix digester	\$289,500	E, H, F	\$29,000	NA

*metric tones of carbon equivalents. This measure is used to compare the emissions of different greenhouse gases based on the global warming potential (US EPA 2002), NA: Not Available

As is apparent from these tables, although information on costs is available, very little data could be identified on the savings and revenue streams arising from AD. Although no economic value can yet be put on greenhouse gas reductions in North America, considering that the current price of carbon in the European carbon market is fluctuating between €20-25 (\$28-35)/tonne of CO₂, significant revenues could be realized by AD through GHG reductions. Economically intangible benefits arising from odour removal or pathogen destruction could provide an added layer of value to AD projects.

6.4 Composting

6.4.1 Process Description

Composting is a biological process in which bacteria in the presence of oxygen convert organic matter to a soil-like substance referred to as compost. The bacteria consume oxygen and organic matter and in turn generate heat, carbon dioxide, and water vapour. The right conditions must be maintained in order to initiate an effective composting process. The key factors governing the efficiency of composting are nutrient balance, moisture content, temperature, and aeration (Buckley and Penn 2001).

In the case of composting, nutrient balance typically refers to the relative quantities of carbon and nitrogen available to bacteria. Bacteria require carbon and nitrogen for growth and there is an ideal balance to achieve good results. Excess nitrogen leads to significant amounts of ammonia being released to the atmosphere and excess carbon reduces the rate at which solids are converted to compost (Buckley and Penn 2001; MAFRI 2001). Ideally, moisture content of solids should be approximately 60 percent, thus slurry type hog manure (typically at a moisture content of 90% or higher) is not suited for composting directly. According to Paul (2005) composting high moisture manure directly is possible with the addition of bulking agents such as straw, corn stover or wood waste.

Composting is a thermophilic process, taking place at 40-65° C. Composting begins with a rapid increase in temperature to 50-60° C which lasts for multiple weeks with a gradual decrease to approximately ambient temperature as nutrients are depleted. Aeration refers to oxygen availability to the bacteria. During the composting process conditions must be maintained such that the mixture remains aerobic. During the initial stages of composting, bacteria require sufficient oxygen for rapid metabolism. If conditions become anaerobic, insufficient heat is generated and undesirable byproducts such as methane, organic acids, hydrogen sulfide and other odorous compounds are generated. Aeration also serves to remove heat, water vapour, and gases. Table 6.9 summarizes the recommended conditions for composting processes

Table 6.9: Recommended conditions for rapid composting (Adapted from Buckley and Penn 2001).

Condition	Reasonable Range	Preferred Range
Carbon/Nitrogen ratio	20:1 – 40:1	25:1 – 30:1
Water content	40 – 65 %	50 – 60 %
Temperature	43 – 65° C	54 - 60° C
Oxygen concentration	5 %	5 – 15 %
pH	5.5 – 9.0	6.5 – 8.0

6.4.2 Benefits

Composting can be used to achieve the following:

- Valuable end product (compost)
- Manure volume and mass reduction
- Improved manure handling characteristics
- Nutrient management
- Odour reduction/elimination

The primary end product of the composting process is compost, a valuable material that when applied to soil adds organic matter, improves soil structure, increases soil water holding

capacity, reduces inorganic fertilizer requirements, and minimizes potential of erosion (Buckley and Penn 2001). Quality compost is fine-textured, low moisture and odourless. There are numerous potential markets for compost including, but not limited to, home gardeners, landscapers, vegetable farmers, turf growers, golf courses, and ornamental growers (Buckley and Penn 2001).

From a soil amendment perspective, the rich microbial diversity and high organic matter concentration of compost are considered more beneficial than its nutrient content. There is currently no value recognized for these two aspects and until value is established through research, compost margins of return will be less than maximal (Buckley and Penn 2001).

Manure volume and mass can be reduced to up to 50% through composting (MAFRI 2001). This reduces transportation and fuel costs for land application and expands the radius of cropland economically accessible by the hog farmer. Compost can be transported further and the material can be applied at a lower rate (Paul 2005). If managed properly, compost has superior handling characteristics due to reduced moisture content, volume and weight (Buckley and Penn 2001).

Due to the aerobic conditions in the composting process, substantial microbial oxidation of odourous compounds or odour generating organic matter can be expected. Although some odour will be released during the process due to mixing, barring anaerobic conditions, excellent odour reduction compared to raw manure can be achieved.

6.4.3 Composting Strategies

6.4.3.1 Windrow

Windrow composting is achieved by piling raw material in long narrow rows typically 1 m high by 3 m wide. These “windrows” are turned with a mechanical device such as a specially equipped tractor as seen in Figure 6.9 or front end loader in order to provide aeration and rebuild bed porosity. Windrow composting typically takes 1 to 4 months depending on turning frequency (MAFRI 2001). For the windrow method to be used on a livestock farm, appropriate machinery will be required for windrow turning. The process will also require dedicated labour time, fuel, space (availability of land), and some monitoring.



Figure 6.9 Windrows with mechanical turning (Freeze et al.)

6.4.3.2 Aerated static pile

Aerated static piles are not mechanically mixed or turned during the composting process. Oxygen necessary for the process is provided directly with forced air systems to speed up the process (Manitoba Agriculture, Food and Rural Initiatives 2001). This is a less labour intensive and simpler method than the windrow process, but produces a less homogenous compost.



Figure 6.10 Aerated static piles for composting (University of Idaho)

6.4.3.3 Rotating Drum

Rotating drums or other “in-vessel” systems contain the raw materials in a building or container with some form of forced aeration and mechanical turning. For instance, the rotating drum pictured in Figure 6.11 features a cylindrical vessel where raw material is aerated with a fan and mechanically rotated. These systems are significantly more expensive than other methods yet require less labour and may lead to a superior compost product in reduced operating time.



Figure 6.11 Rotating drum composting vessel (Manitoba Agricultural Guidelines Development Committee, 2007)

6.4.4 Nutrient Fates

Since the nitrogen in compost is less likely to leach and losses through volatilization as ammonia are minimal, it can be stored, transported and applied at convenient intervals (Buckley and Penn 2001). This provides flexibility in nutrient management strategies. Compost also possesses carbon to nitrogen ratios more suitable for land application (Buckley and Penn 2001). Although some losses of nitrogen can be expected during the composting process, the carbon to nitrogen ratio is substantially lower in the compost than the raw manure. Unless there is significant leaching of water from the compost pile during the process, compost and raw manure do not exhibit substantially different phosphorus concentrations. The proliferation of microbial biomass during composting will result in uptake of phosphorus into bacterial cell mass, and somewhat reduce susceptibility of phosphorus mobilization from farmland. Table 6.10 summarizes the compositional changes after the composting of hog manure:

Table 6.10 Composition of raw manure/straw mixture (dry matter basis) prior to and following compost treatment at Brandon Research Centre (Buckley and Penn 2001)

Chemical (kg/Mg)	Constituent	Raw Manure	Compost
	Total Nitrogen	15.9	12.5
	Organic Nitrogen	15.8	12.1
	Ammonia Nitrogen	0.14	0.05
	C:N ratio	19:1	13:1
	Phosphate	10.9	10.2
Physical Properties			
	pH	8.6	8.5
	Moisture (%)	75	42

6.4.5 Composting in Practice

Currently composting of hog manure is not practiced in full-scale in Canada. There is a 5-acre experimental composting site at the Agriculture and Agri-Food Canada, Brandon Research Centre. This site uses a windrow scheme, which accommodates 8000 tonnes of raw manure.



Figure 6.12 Brandon Research Centre composting site (Buckley and Penn 2001)

Green Acres Colony in Manitoba is currently looking to construct a large-scale (60' x 9' diameter) rotating drum for co-composting of the solid fraction of hog manure, dead livestock, and organic solid waste created within the colony. This system is intended to be finished by November 2007 (Hofer 2007).

According to Paul (2005) there are two interesting examples of farms in Saskatchewan that successfully operate composting systems coupled with separation. Elite Stock Farms stores their manure during the winter and uses a screw press separator in the spring. Solids are then fortified with carbon-rich bulking materials and composted in windrows in the summer. The 2000 sow farrow to finish operation run by Outlook Pork separates and then composts their manure year round. They use a system referred to as agitated-bed, which apparently enables them to compost the solids without the addition of bulking agents.

Based on conversations with Katherine Buckley at AAFC, many operations are conducting "semi-controlled decomposition" rather than composting. This distinction is based on the fact that these operations are not strictly controlled or monitored. In the absence of solid-liquid

separation technology, composting is not a viable option for typical hog farms producing dilute manure slurry with relatively low solids concentration. Straw-bedding based systems could facilitate composting without the addition of separation technology but these operations do not represent a significant proportion of hog operations in Manitoba (Buckley 2007).

6.4.6 Feasibility in Manitoba Context

In the absence of a viable market for composted livestock manure, which is currently the condition in the Province of Manitoba, the costs of composting may not be recoverable in the sale of the final product (MAFRI 2001). Many hog farmers in Manitoba might not have a suitable site for composting, which must be developed to prevent runoff and leaching of nutrients. The composting site, raw material storage and finished compost storage can occupy significant land area (Buckley and Penn 2001). As composting relies heavily on low water content feedstock, without the wide-spread application of solid separation processes for slurry type hog manure, composting applications will remain limited in Manitoba. Integrated efforts such as those undertaken by Green Acres Colony, involving solid separation and utilization of a variety of waste products (food waste, livestock mortality, yard waste, etc.) along with manure hold significant promise for future application of composting.

6.4.7 Economics

The cost of establishing composting capacity is dependent on the compost strategy, amount and characteristics of the manure to be processed, labour requirements, cost of bulking agents, buildings, and getting the product to market (British Columbia Ministry of Agriculture, Food and Fisheries 1996). Further the British Columbia Ministry of Agriculture, Food and Fisheries (1996) suggests that transportation and handling absorbs approximately 35% of yearly sales revenue and packaging represents approximately 30% of sales revenue. Table 6.12 compares the relative costs for different composting strategies.

High transport costs have discouraged the use of manure as a valuable resource in favor of disposal (Freeze et al.). The additional cost associated with composting must be weighed against the value of manure or compost as a fertilizer (Freeze et al.).

Table 6.12 Comparison of capital costs for composting (British Columbia Ministry of Agriculture, Food and Fisheries 1996).

Composting Strategy	Costs	Relative cost
Windrow	<ul style="list-style-type: none"> • Appropriately equipped tractor for turning windrows • Land area 	Medium Low (if existing equipment is available)
Aerated static pile	<ul style="list-style-type: none"> • Aeration system (ducting, motors, fans and monitoring equipment) 	Medium
In-vessel system	<ul style="list-style-type: none"> • Vessel • Buildings • Agitation and aeration system • Monitoring equipment 	High

To penetrate potential markets, the compost must be priced competitively with existing inorganic fertilizers and soil conditioners (British Columbia Ministry of Agriculture, Food and Fisheries

1996). Table 6.13 shows the potential cost and value of composting. It must be recognized that the fertilizer value of compost is highly variable depending on the composition of the manure being composted and the available markets. Additionally, it is very challenging to predict composting costs in a universal manner and therefore it is of utmost importance to perform a financial analysis for each individual case.

Table 6.13 Cost and Value of compost from hog manure

Production Cost of Compost	Fertilizer value*
\$36-70 ^a	\$240-417 ^b

*Based on equivalent value of commercial fertilizer with similar nutrient composition

^a(British Columbia Ministry of Agriculture, Food and Fisheries 1996); ^b(Freeze et al.)

6.5 Conclusions & Recommendations

Solid Separation

- Solid separation technologies show promise in terms of producing an improved N:P ratio in the liquid phase, addressing phosphorous surplus issues, extending the range of farmland economically accessible for manure application, odour abatement, and as a pretreatment for manure solids composting. Their economic feasibility have not yet fully evaluated in the Manitoba marketplace.
- Phosphorous regulations will drive adoption of solid separation. The more-advanced knowledge base and implementation in Quebec was driven by regulations (1981 Nitrogen regulation implemented, 1997 Phosphorous regulation implemented, 2002 Revised Nitrogen and Phosphorous regulation). It was not until 2002 when Quebec really started looking at treatment systems to address nutrient concerns. An additional driver was the increased livestock intensity and diminishing land base. Significant similarities exist between the conditions of Manitoba and Quebec on these trends
- Centrifugation exhibits potential to concentrate a high degree of phosphorous in solids without using chemical amendments. Avoiding chemical amendments could have significant benefits in terms of cost and off-farm marketing/sales of solids. Regardless of the type of technology selected by the operation, the design must be case-based and configured for individual farm manure characteristics. Ease of operation and maintenance could play a crucial role in the adaptation rate of any one system.
- A careful plan needs to be developed for use or marketing of the separated solids prior to implementing separation. In Quebec, due to the high pressure to comply with new regulations, some operations did not effectively establish a strategy for the separated solids.
- For solid separation technology to succeed, significant attention should be paid to supporting systems such as solids and liquids storage infrastructure, pumps, piping, etc. Homogenization (agitation) of the raw manure prior to solid separation will yield more predictable results.
- Using less water in the operation of the livestock facility will have a large impact on the effectiveness of the solid separation technology. The economics of the treatment system is closely related to the volume and solids content of the manure. For instance concentrating manure from 2% solids to 8% (both within the range of typical Manitoba hog manure) through better management will reduce the size of the solids separation system by 4 fold
- Separation technologies presented in this section of the report are the most common but not the only options. There are additional options that have not been explored here and may yet prove superior in the future.

Anaerobic Digestion

- AD is used successfully as an energy producer and manure treatment process in some countries around the world, in particular where the price of energy prices are high and governmental subsidies are available for both capital investments and green energy sales. AD may not be financially viable in Manitoba unless hydro or natural gas become more expensive, or a more energy efficient process is developed for the Manitoba climate. Financial incentives from the Federal and Provincial Governments for renewable energy and greenhouse gas displacement could substantially improve the economic viability of AD in Manitoba.
- A significant knowledge gap exists on the strategies to improve the energy balance of AD in cold-climate conditions. Research on AD in Manitoba should focus on developing a more energy efficient system to counter heat losses and associated reductions in methane production during cold winter months. Amending AD with high-energy feedstock such as food-processing wastes, waste alcohols, or glycerol (a by-product of the emerging bio-diesel industry), could boost methane yields and improve the overall energy balance of the process.
- AD mitigates greenhouse gas production by harnessing otherwise lost methane produced during natural anaerobic digestion of stored manure or undigested field applied manure. If a carbon market, similar to the one in Europe, comes into effect in North America, the carbon credits could equal or exceed the revenue generated by biogas production.
- AD changes nutrient characteristics of liquid manure, making N and P more plant available but it does not reduce the overall content of phosphorous in manure. The more soluble and inorganic form of phosphorus and nitrogen present after AD lend themselves to additional separation and concentration (chemical precipitation) steps. The recovery of these nutrients in a concentrated might displace commercial inorganic fertilizer and present a net revenue stream to the farmer. However the extraction and recovery steps will further add costly processing equipment and increase operating costs and labour needs. A centralized AD facility receiving manure from a large number of livestock facilities might be able to justify this investment.
- AD reduces odor associated with undigested manure storage. Some of the costs associated with installations required to address odor could be used to off set capital associated with AD installations.
- Manitoba currently has approximately 1300 hog operations with an average of 2300 head of pig per operation. The average operation of this size could theoretically yield 30kW of electrical power, or on a province wide scale, from 1300 operations approximately 40MW. Manitoba Hydro has a total production capacity of 5461 MW in the Province. AD could make up 0.7% of Manitoba Hydro production capacity.

Composting

- The economic value of compost as a fertilizer and soil conditioner in the bulk and retail markets must be researched and established in the Manitoba context.
- Typical Manitoba manure slurry is not suitable for composting directly and therefore is not viable without solid separation as a pre-treatment. If separation is implemented to enable composting, the potential increase in market value of the pre-composted solids and composted solids should be evaluated. Separating the liquid from hog manure for the sole

purpose of composting the solids may not be viable because of the high cost of separation.

- Composting offers benefits such as volume reduction, odour abatement but may require significant land area, some added equipment and operator attention
- Research systems that promise the ability to compost dilute manure slurries directly (“liquid composting systems”) might be of great value and should be pursued. There is also research potential for low cost, high-carbon bulking agents that might be suitable in the Manitoba
- More research is needed to quantify the agronomic and environmental benefits of compost. Its marginal value will be higher on eroded lands or when applied to high value crops. As prices of inorganic fertilizer are projected to increase and as soils continue to erode, the qualities of compost will gain increasing importance.
- As surplus nutrients must be moved further away from intensive livestock production facilities of origin, transportation costs and associated greenhouse gas emissions might provide the appropriate economic and regulatory incentives to perform separation and or composting on a wide-scale in Manitoba in the near future.

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CHAPTER 7 Energy requirements for surface application of liquid hog manure in southern Manitoba

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7.1 Executive Summary

To determine the energy used to surface-apply liquid hog manure, a thorough accounting was done of all energy consuming activities required to transfer liquid pig manure from storage to field. Energy consumption was estimated by gathering data from local commercial manure application companies. Energy consumption was compared between the drag hose and the slurry wagon manure application systems. Single annual manure application and twice-annual manure application regimes were also compared. The single annual application of liquid pig manure applied at $81.49 \text{ m}^3 \text{ ha}^{-1}$ and transported 1.81 km from storage to field consumed 2180 MJ ha^{-1} with the drag hose system and 2185 MJ ha^{-1} with the slurry wagon system. The twice-annual manure application regime used 2726 and 2209 MJ ha^{-1} for the drag hose and slurry wagon systems, respectively. When energy use was calculated on the basis of MJ per kg of available N, liquid pig manure applied once annually with the slurry wagon system provided N at 17.76 MJ kg^{-1} of available N, which was 33% of the energy cost of N from anhydrous ammonia and 23% of the energy cost of N from urea. Manure transport distance could be increased to 8.4 km before the energy cost per kg of available N was equivalent to anhydrous ammonia, and up to 12.3 km before the energy cost of manure N was equivalent to urea N. Despite the high energy cost to deliver liquid pig manure from storage to field, the much lower cost per kg of available N compared to inorganic fertilizer N highlights the opportunities that exist for improving the energy efficiency of industrial agriculture by replacing inorganic fertilizers with manure.

7.2 Introduction

In agriculture, as in most economic activities, there are both financial and environmental reasons to improve energy efficiency. First, the vast majority of energy used in agricultural systems is from fossil fuels and these fuel costs are increasing. From an environmental perspective, energy use is associated with carbon dioxide emission which has serious implications for global climate change (IPCC 2001). Energy consumption is also indirectly

responsible for negative impacts on indigenous communities and natural ecosystems during fossil fuel extraction (Goddard, 1991; Griffiths et al. 2006).

In agriculture, the availability of large quantities of liquid manure from hog production facilities presents an opportunity for some farmers to dramatically reduce energy consumption by replacing energy-intensive synthetic fertilizer with manure. Fifty to 70% of the energy used for grain production may be embodied in the manufacture of chemical fertilizer (Swanton 1996; Nagy 2001; Hoepfner et al, 2006). Hog manure can be viewed as a waste product of pork production and therefore free of any energy costs for its production.

The goal of this report is to determine how the energy balance for supplying N to crops is changed when the N is supplied using liquid pig manure compared with inorganic fertilizer N that is produced using fossil-fuel energy.

7.3 Scope of research conducted elsewhere in Canada and Internationally

McLaughlin et al. (2000) calculated an energy saving of between 36 and 52% if liquid manure were used instead of inorganic fertilizer in corn production in southern Ontario. Where corn was produced with inorganic fertilizer, between 33 and 54 % of the total energy use was for the manufacture of inorganic fertilizer (McLaughlin et al. 2000). In contrast, corn produced with manure used between 3.5 and 6.3 % of total energy for manure application; however, no energy values were calculated for manure transportation or for energy embodied in machinery.

A major limitation to the use of liquid manure as fertilizer is its high water content and associated high transport costs; manure transportation is expensive both financially and energetically. An important question is whether the energy expended to transport manure negates the energy saved by replacing synthetic fertilizers. Previous work estimated that the energy required to manufacture the equivalent amount of N in the form of urea as is contained in 1 t of liquid pig manure would be sufficient to transport 1 t of liquid pig manure up to 66 km (Ceotto 2005). Energy required for manure transportation will vary depending on the nutrient concentration of the manure. In other words, manure with high N concentration will require less transportation energy than low N-concentration manure to achieve the same N application rate. Similarly, crops with high nutrient demands will enable high manure application rates and therefore low energy requirements for manure transportation.

Energy use is often described as the sum of direct (e.g. diesel fuel consumption) and indirect (e.g. energy for production of fertilizer or machinery) energy use (Dalgaard et al. 2000). In the case crops fertilized with liquid hog manure, direct energy use includes diesel fuel used for activities such as manure agitation, transportation and application while indirect energy use includes energy used to produce tractors, slurry wagons, drag hoses, etc.

The present study offers a detailed breakdown of the energy costs of applying liquid hog manure. The objectives of this study were to examine surface-application of liquid hog manure in order to determine: 1) the amount of energy used; 2) the allocation of energy consumption to the various activities of manure application; and 3) the nitrogen energy cost of manure vs. inorganic fertilizer N.

7.4 Research conducted in Manitoba

7.4.1 Methods

An experiment conducted near La Broquerie, Manitoba (49°31 N, 96°30 W) to measure the productivity of perennial forage treated with or without liquid hog manure was the basis for the energy analysis. The experiment included a Split manure application (50% applied in spring and 50% applied in fall), and a Full manure application (100% applied in spring). The two manure application treatments were compared in the original experiment in order to investigate the nutrient use efficiencies of the contrasting application regimes; however the standard practice in southern Manitoba is to apply the entire annual volume of manure at one time. Liquid hog manure was surface-applied to forage stands at a rate calculated to supply a total of 123 kg of N/ha to the forage crop. Energy use for manure application was estimated by investigating typical liquid hog manure application systems in Manitoba.

7.4.1.1 Energy Inputs

Energy use was investigated for two different manure application methods. The drag hose manure application system is becoming more common in Manitoba and therefore is likely to be used when applying manure to forage or crop land. The slurry wagon system was included because it is also common and is easier to use in grazing systems where fences cause problems for the set up of the feeder hoses for the drag hose system.

Efforts were made to account for all energy use in applying liquid hog manure from an earthen manure storage structure to farmland. Energy coefficients for the various raw materials used in the machinery were based on work done by Baird et al.(1997). Coefficients for embodied energy of tractors, tractor fuel consumption, and tractor lubrication energy are from Nagy (1999), with some modifications made to fuel consumption coefficients based on local conditions (Gallup, 2006 – personal communication). For each operation energy consumption is broken down into three categories: fuel energy, machine energy, and lubrication energy. Local operators of commercial manure application equipment provided estimates of the amount of time required for each operation (Gallup 2006 – personal communication; Penner 2006 – personal communication).

7.4.2 Results and Discussion

7.4.2.1 Energy use for manure application

Energy calculations for manure handling activities were based on a typical feeder hog scenario in southern Manitoba (Table 1). A total of 29.56 million litres of manure were produced annually from a barn producing 25000 finished feeder hogs per year. The nitrogen content of the manure entering storage was 3.88 kg/1000 litres (Table 7.1). Volatilization N loss while manure was in the earthen storage was assumed to be 30% (Sutton et al. 2001). At the time of application the manure in the earthen storage was assumed to contain 61% of its N in the NH_4^+ form and 39% in the organic form (Prairie Provinces Committee on Livestock Development and Manure Management). Assuming 25% of the NH_4^+ proportion is lost due to volatilization at the time of application (Prairie Provinces Committee on Livestock Development and Manure Management), and assuming 25% of the organic-N is available in the year of application (Prairie Provinces Committee on Livestock Development and Manure Management) the total available-N provided to the crop was 1.51 kg N/1000 L of manure. The rate of manure application was $81.49 \text{ m}^3 \text{ ha}^{-1}$, to achieve the desired N application rate of 123 kg available-N ha^{-1} .

Table 7.1. Assumptions made regarding liquid hog manure application to perennial forage

#	<u>Assumption</u>	<i>Manure production</i>
1.	Feeder hogs on site: 10 000	
2.	Litres of manure produced per hog per day: 5.1	
3.	Increase factor for spillage, wash water, and precipitation: 1.588	
4.	N produced per feeder hog per day (including feed wastage) (kg): 0.0403 (Fabian et al. 2004)	
5.	Number of days pigs are in the barn: 114	
6.	Number of batches of pigs per year: 2.5	
7.	Number of feeder hogs produced per year: 25000	
<i>Manure storage, N loss and N availability</i>		
8.	Total manure entering storage in one year (million litres): 29.56 (#2 x #3 x #5 x #6)	
9.	Total N Entering Storage in one year (kg/yr): 114,855 (#1 x #4 x #5 x #6)	
10.	N concentration of manure entering storage (kg/1000 l): 3.88 (#9 ÷ #8)	
11.	Proportion of N in storage in ammonium form: 61% (Prairie Provinces Committee on Livestock Development and Manure Management)	
12.	Percentage of N lost in earthen storage: 30% (Sutton et al. 2001) (note: it is assumed that manure is surface-applied to a forage crop in early spring or late fall (i.e. cool weather) regardless of which application system is used)	
13.	N loss during surface application using either system: Ammonium-N: 25%; Organic-N: 0% (Prairie Provinces Committee on Livestock Development and Manure Management)	
14.	N availability in year of application using surface application with either system: Ammonium-N: 100%; Organic-N: 25% (Prairie Provinces Committee on Livestock Development and Manure Management)	
15.	Available N after storage and application losses (kg/1000 l): 1.51	
<i>Manure application</i>		
16.	Desired N application rate (kg/ha): 123	
17.	Manure application rate (m ³ /ha): 81.49	
18.	Total land area covered with manure (ha): 362.8	
19.	Distance from storage to field (km): 1.81	
<i>Drag hose System Energy Use</i>		
20.	Area covered in a set during application with Drag hose System (ha): 16.19	
21.	Pumping rate for Drag hose System (m ³ per minute): 3.77	
22.	Application width in Drag hose System (m): 9.14	
23.	Maximum field speed for application tractors in Drag hose System (km/hour): 11.27	
24.	Tractor(s) for lagoon agitation: 1 tractor; 165 hp	
25.	Tractor(s) for manure application: 1 tractor; 200 hp	
26.	Tractor(s) for used for hose reel(s): 1 tractor; 200 hp	
27.	Manure pump and engine: 1 unit; 250 hp	
28.	Booster pump and engine: 1 unit; 250 hp (booster pump was used when distance was greater than 1.6 kilometers)	

29. Tractor and engine energy use: see Table 2 for operation time, fuel energy, machine energy, lubrication energy, total energy use, and energy use/ha
30. Additional Machinery: see Table 3 for equipment quantity, material, weight, embodied energy, lifespan, total energy use, and energy use/ha

Slurry Wagon System Energy Use

31. # and capacity of tanks used in Slurry Wagon system: $5 \times 28 \text{ m}^3$
 32. Total capacity of wagons: = 140 m^3 (sum of tank capacity) * 0.94 (assuming the tanks are filled to 94% of capacity)
 33. Loading rate for tankers in Slurry Wagon System (m^3 per minute): 9.09
 34. Application width in Slurry Wagon System (m): 18.29
 35. Unload rate for tankers in Slurry Wagon System (m^3 per minute): 9.09
 36. Maximum road speed for Slurry Wagons (km/h): 32
 37. Maximum field speed for tractors in Slurry Wagon System (km per hour): 14.48
 38. Travel time for tankers (minutes/km): 2.18 (less than 32 km/h to account for stops, corners, acceleration and deceleration)
 39. Loading time for tankers (minutes/load): 3
 40. Land covering time for tankers (minutes/load): 3
 41. Tractor(s) for lagoon agitation manure pumping: 1 tractor; 250 hp
 42. Tractor(s) for manure application: 5 tractors; 250 hp
 43. Tractor energy use: see Table 5 for operation time, fuel energy, machine energy, lubrication energy, total energy use, and energy use/hectare
 44. Additional Machinery: see Table 6 for quantity, material, weight, embodied energy, lifespan, total energy use, and energy use/hectare
-

At this application rate, the land base required for spreading was 362.8 ha, and the average distance from the manure storage to the field was assumed to be 1.81 km.

7.4.2.1.1 Drag hose manure application

Energy consumption by the tractors and pump engines in the drag hose system is described in Table 7.2. Energy consumption was divided into the three main components: fuel energy, machine energy, and lubrication energy. Hourly fuel consumption by each engine was multiplied by the hours of operation in order to calculate total fuel use. Total fuel use was multiplied by an energy coefficient of 43.99 MJ L^{-1} (Coxworth et al. 1994). The total amount of time required to apply the yearly production of 29.56 million litres of liquid pig manure with the drag hose system was 149.9 hours for the Full application treatment and 166.7 hours for the Split application treatment. The Split application treatment required the application tractors to be running for a longer time than the Full application treatment because all equipment had to be set up and moved across the fields twice (once in spring and once in fall); also, 15 extra hours were required for agitation (Table 7.2).

Table 7.2 Tractor and pump engine energy use for manure application under the Full and Split manure application treatments with the drag hose system, with breakdown for total hours of operation, fuel use/hour, fuel energy, machine energy, and lubrication energy.

Tractor or Engine	#	HP ^a	Hours of use at this job ^a				Total Fuel Use (l/h) ^a	Total Fuel Use (l)	Total Fuel Energy ^b (MJ)	Embodied Machine Energy ^c (MJ/h)	Total Machine Energy (MJ)	Lube Energy ^c (MJ/h)	Total Lube Energy (MJ)	Total Annual Fuel, Machine, and Lube Energy (MJ)	Total Annual Energy Use (MJ/ha)
			Set up	Moves	Operation	Total									
Full manure application treatment															
Agitation Manure app.	1	165	5	---	55	60	32	1920	84461	50.8	3048	6.0	360	87869	242.2
Hose reels	1	200	11	5.75	133.2	149.9	16	2398	105506	72.7	10898	6.2	929	117333	323.4
Pump engine	1	250	---	---	133.2	133.2	45.9	6112	268849	80.1	10665	6.3	839	280353	772.7
Booster pump	1	250	---	---	66.6	66.6	45.9	3056	134424	80.1	5333	6.3	419	140176	386.4
Totals	5							15884	698745		40841		3477	743063	2048.1
% of total									94.0		5.5		0.5		100.0
Split manure application treatment															
Agitation Manure app.	1	165	10	---	70	80.0	32	2560	112614	50.8	4064	6	480	117158	322.9
	1	200	22	11.5	133.2	166.7	30.7	5116	225060	72.7	12115	6.2	1033	238208	656.6

The total energy consumed by the tractors and pump engines to apply the total volume of manure was 743063 MJ or 2048 MJ ha⁻¹ for the Full application treatment and 906340 MJ or 2498 MJ ha⁻¹ for the Split application treatment (Table 7.2). Machine and lubrication energy were calculated by multiplying hours of operation by tractor energy coefficients (MJ h⁻¹) from Nagy (1999). Machine and lubrication energy consumption represented 5.5% and 0.5% respectively of total tractor and pump engine energy use for the Full application treatment, and 4.9% and 0.4% respectively for the Split application treatment (Table 7.2).

By far the largest component of energy consumption was fuel use (Table 7.2). Fuel use energy by the tractors and pump engines equaled 698745 MJ or 1926 MJ ha⁻¹ for the Full application treatment and 858242 MJ or 2366 MJ ha⁻¹ for the Split application treatment, which represented 94.0% and 94.7% of the tractor and pump engine energy consumption for the Full and Split application treatments respectively (Table 7.2). The 440 MJ of added fuel energy consumption by the Split application treatment compared to the Full application treatment is a result of two factors: 1) the extra running time in the Split application treatment; and 2) the higher fuel consumption rate by the application tractors (Table 7.2) since they were required to drive at double the speed during the Split application in order to apply manure at half the application rate compared to the Full application treatment. The largest consumer of fuel was the stationary engine that powered the main pump (6112 L or 38.5% of total fuel use (Table 7.2)).

In addition to the embodied machine energy of the tractors and engines there was embodied energy in the miscellaneous machinery (e.g. drag hoses, hose reels, couplers, etc.) used in the drag hose manure application system. The total embodied energy of the miscellaneous machinery used in the drag hose system was 132 MJ ha⁻¹ for the Full application treatment and 228 MJ ha⁻¹ for the Split application treatment (Table 7.3). The 96 MJ of additional energy embodied in miscellaneous machinery for the Split treatment was mainly because of higher replacement rates of drag hoses since the hoses were being dragged across twice the land area as compared to the Full application treatment (Table 7.3).

When the totals for manure application energy use from Tables 7.2 and 7.3 were combined the total energy consumed in manure application with the drag hose system was 2180 MJ/ha for the Full application treatment, and 2726 MJ/ha for the Split application treatment (Table 7.4). The higher machine energy of the Split application treatment that was seen with the miscellaneous equipment energy (Table 7.3) was still evident when the total energy consumption was separated into the various components (Table 7.4). The fuel, machine, and lubrication proportions of total manure application energy consumption are 88.3%, 11.2% and 0.4% respectively for the Full application treatment, and 86.8%, 12.8%, and 0.4% respectively for the Split application treatment (Table 7.4).

7.4.2.1.2 Slurry wagon manure application

The total amount of time spent for manure transport and application with slurry wagons was 51.12 hours (Table 7.5) for both the Full and Split application treatments, approximately one third the amount of time required for manure application with the drag hose system. For the Full application treatment, the slurry wagon system used a similar amount of total energy as the drag hose system (2185 MJ ha⁻¹ for slurry wagons vs. 2180 MJ ha⁻¹ for the drag hose method) (Table 7.4), despite the shorter application time.

Table 7.3 Embodied energy in miscellaneous machinery used for manure application in the Full and Split manure application treatments using the drag hose system.

Machine	Unit	Weight/unit (kg)	Material ^a	Energy Coefficient (MJ/kg)	Total Embodied Energy (MJ) ^b	Lifespan (years)	Proportion of yearly work dedicated to job	Total Energy for job (MJ)	Total Energy Use per area (MJ/ha)
Full manure application treatment									
Agitator	1	3400	steel	32	125120	15	0.0578947	482.92	1.33
Rubber hose (main line)	3.22 km	13028	rubber	110	1648042	4	0.066327	27327.42	75.32
Drag hose	792 m	4482	rubber	110	566973	2.5	0.066327	15042.25	41.46
Applicator	1	907	steel	32	33378	15	0.066575	148.14	0.41
Hose reels	2	7711	steel	32	567530	10	0.066327	3764.25	10.38
Couplers	74	6.8	aluminum	191	110528	10	0.066327	733.10	2.02
Fuel Trailer	1	1134	steel	32	41731	15	0.066327	184.53	0.51
All Terrain Vehicle (ATV)	1	290	steel	32	10672	10	0.066327	70.78	0.20
Hose Humper	1	272	steel	32	10010	10	0.066327	66.39	0.18
Total								47864.00	131.81
Split manure application treatment									
Agitator	1	3400	steel	32	125120	15	0.0842	702.4	1.94
Rubber hose (main line)	3.22 km	13028	rubber	110	1648042	3.5	0.0737	34719.1	95.70
Drag hose	792 m	4482	rubber	110	566973	1	0.0737	41805.2	115.23
Applicator	1	907	steel	32	33378	15	0.0666	148.1	0.41
Hose reels	2	7711	steel	32	567530	10	0.0737	4184.6	11.53
Couplers	74	6.8	aluminum	191	110528	10	0.0737	815.0	2.25
Fuel Trailer	1	1134	steel	32	41731	15	0.0737	205.1	0.57

All Terrain Vehicle (ATV)	1	290	steel	32	10672	10	0.0737	78.7	0.22
Hose Humper	1	272	steel	32	10010	10	0.0737	73.8	0.20
Total								82732.0	228.04

a Energy coefficients are from Baird et al. 1997

b Total embodied energy = coefficient for material multiplied by the machinery mass plus 15% for manufacturing energy

Table 7.4 Annual energy inputs (MJ/ha) for liquid hog manure applied with the drag hose and slurry wagon application methods.

		Fuel	Machine	Lubrication	Total
Drag hose	Full application	1926.0	244.5	9.6	2180
	% of total	88.35	11.22	0.44	100
	Split application	2365.6	350.1	10.5	2726
	% of total	86.77	12.84	0.38	100
Slurry Wagon	Full application	2064.5	115.0	5.4	2185
	% of total	94.49	5.26	0.25	100
	Split application	2087.2	115.9	5.4	2209
	% of total	94.51	5.25	0.25	100

Table 7.5. Tractor energy use for manure application under the Full and Split manure application treatments using slurry wagons, with breakdown for total hours of operation, fuel use/hour, total fuel energy, machine energy, and lubrication energy.

Tractor use	#	HP ^a	Hours of use at this job ^a		Fuel Use (l/h) ^a	Total Fuel Use (l)	Total Fuel Energy ^b (MJ)	Embodied Machine Energy ^c (MJ/h)	Total Machine Energy (MJ)	Lube Energy ^c (MJ/h)	Total Lube Energy (MJ)	Total Fuel, Machine, and Lube Energy (MJ)	Total Annual Energy Use (MJ/ha)	
			Set up	Operation										
Full manure application treatment														
Agitation and pumping	1	250	1	51.12	52.12	45.9	2392.308	105238	80.1	4174.812	6.3	328.356	109741	302.5
Transport and app.	5	250	0.5	51.12	51.62	283.5	14634.27	643762	400.5	20673.81	31.5	1626.03	666061	1835.9
Totals	6					329.4	17027	748999	481	24848.622	37.8	1954.386	775802	2138.4
% of total								96.55		3.20		0.25		100.0
Split manure application treatment														
Agitation and pumping	1	250	2	51.12	53.12	45.9	2438.208	107257	80.1	4254.912	6.3	334.656	111846	308.3
Manure app.	5	250	1	51.12	52.12	283.5	14776.02	649997	400.5	20874.06	31.5	1641.78	672513	1853.7
Totals	6						17214	757254		25128.972		1976.436	784359	2162.0
% of total								96.54		3.20		0.25		100.0

a Horse power of tractors, hours of use, and fuel use per hour is from Gallup 2006

b Total fuel energy is derived by multiplying total fuel use by 43.99 which is the energy coefficient for diesel fuel from Coxworth et al. 1999.

c Coefficients for embodied machine energy and lube energy are from Nagy 1999.

However, for the slurry wagons the Full treatment consumed more energy as fuel than did the Full treatment with the drag hose system, while the drag hose system consumed a higher proportion of total energy as energy embodied in machinery (Table 7.4). The Full application treatment applied with slurry wagons required 17027 L of diesel fuel (Table 7.5), while the same treatment applied with the drag hose method consumed 15884 L of diesel fuel (Table 7.2), a difference of 1143 L of diesel fuel.

In contrast to the drag hose system, total energy use in the slurry wagon changed very little from the Full (2185 MJ ha⁻¹) to the Split (2209 MJ ha⁻¹) application treatments (Table 7.4). Only minimal additional time was required for set up for the Split application compared to the Full application treatment in the slurry wagon system: 1 extra hour for agitation and 0.5 additional hours of set up for each slurry wagon tractor (Table 7.5). Fuel consumption rates did not change from the Full to the Split application treatments when using slurry wagons as they did with the drag hose system (Tables 7.2 and 7.4).

Machine energy of the slurry wagon system differed very little between the Full and Split application treatments (Table 7.4). The similar machine energy costs were because of the similar operation time between the Full and Split treatments, and because there was no additional equipment replacement as there was in the case of the drag hose system. The largest portion of machine energy was the energy embodied in the slurry wagons (41.4 MJ ha⁻¹) (Table 7.6). Machine and lubrication energy comprised 5.3% and 0.2% of total energy consumption respectively, while the fuel energy portion was 94.5% (Table 7.4).

It is notable that the energy requirements for manure application in southern Manitoba are similar to the energy requirements calculated for manure application in Denmark (Dalgaard 2001). The total energy cost of manure application calculated for southern Manitoba was 2180 MJ ha⁻¹ for Full application treatment with the drag hose system, and 2185 MJ ha⁻¹ for Full application treatment with the slurry wagons system (Table 7.4). Dalgaard et al. (2001), working in Denmark, reported the energy requirements for manure spreading to be 0.3 L of diesel fuel per tonne of slurry, plus 0.2 L of diesel fuel per tonne of manure per km of transport. When the fuel use rates from Dalgaard et al. (2001) are multiplied by the amount of manure applied in our study (81.49 m³/ha) and the hauling distance (1.81 km), and converted to MJ using 43.99 MJ L⁻¹ of diesel fuel (Coxworth et al. 1994), the total energy consumption to apply manure is 2373 MJ ha⁻¹; only 8.6% higher than the energy consumed by the slurry wagon system in our study. However, manure agitation and loading is not included in these two values offered by Dalgaard et al. (2001) and so the final energy cost of getting liquid pig manure from storage to field in Denmark is likely still higher than the 2373 MJ ha⁻¹ calculated here.

7.4.2.2 Nitrogen energy costs of manure versus inorganic fertilizer N

From an energy perspective, our study showed that manure N is much more economical than inorganic fertilizer N. We assumed that the N application rate was 123 kg available N ha⁻¹ and the energy cost to apply manure to achieve this available N rate was 2180 MJ ha⁻¹ using the drag hose delivery method and 2185 MJ ha⁻¹ with slurry wagons (Table 7.4). To determine the energy cost per unit of available-N we divided the energy cost of manure application by the amount of N applied, i.e. 2180 MJ ha⁻¹ / 123 kg N ha⁻¹, to arrive at values for energy cost per unit of available N, i.e. 17.72 MJ kg⁻¹ in the case of the drag hose system, and 17.76 MJ kg⁻¹ with the slurry wagon system.

Table 7.6 Embodied energy in miscellaneous machinery used for manure application in the Full and Split manure application treatments using slurry wagons at 1.81 km transport distance.

Machinery	Quantity	kg/unit	Material	Energy Coefficient ^a (MJ/kg)	Total Embodied Energy (MJ) ^b	Lifespan (years)	Hours at this job	Annual hours in operation	Proportion of yearly work dedicated to this job	Total Energy used for this job (MJ)	Total Energy Use per area (MJ/ha)
Full manure application treatment											
Agitator/pump	1	4536	steel	32	166925	5	52.12	1223	0.04262	1422.8	3.92
Slurry Wagon	5	7665.8	steel	32	1410507	4	51.12	1200	0.04260	15021.9	41.41
Spreadplates	5	453.6	steel	32	83462	15	51.12	1200	0.04260	237.0	0.65
Fuel Trailer	1	1134	steel	32	41731	15	52.12	1223	0.04262	118.6	0.33
All Terrain Vehicle	1	290	steel	32	10672	10	52.12	1223	0.04262	45.5	0.13
Total										16845.7	46.43
Split manure application treatment											
Agitator/pump	1	4536	steel	32	166925	5	53.12	1223	0.04343	1450.0	4.00
Slurry Wagon	5	7665.8	steel	32	1410507	4	51.12	1200	0.04260	15021.9	41.41
Spreadplates	5	453.6	steel	32	83462	15	51.12	1200	0.04260	237.0	0.65
Fuel Trailer	1	1134	steel	32	41731	15	53.12	1223	0.04343	120.8	0.33
All Terrain Vehicle	1	290	steel	32	10672	10	53.12	1223	0.04343	46.4	0.13
Total										16876.2	46.52

a Energy coefficient for steel is from Baird et al. 1997

b Total embodied energy = coefficient for steel multiplied by the machinery mass plus 15% for manufacturing energy

Energy coefficients for inorganic N are available in the literature. Energy costs of anhydrous ammonia and urea were derived from Nagy (1999), who reported that the energy coefficients for anhydrous ammonia and urea are 52.21 and 75.63 MJ kg⁻¹ of N, respectively. These values represent the total energy required to produce this fertilizer and transport it to the farm. A small additional amount of energy is required to apply this fertilizer (Nagy 1999), bringing the total energy cost per kg of N to 54.50 and 76.37 MJ kg⁻¹ for anhydrous ammonia and urea, respectively.

The preceding calculations show that despite the high energy cost of applying manure at an average hauling distance of 1.81 km, it is three and four times more energy efficient to use liquid hog manure than it is to use inorganic fertilizers to provide N to crops. Our estimate of 17.76 MJ kg⁻¹ for manure-derived N is 33% of the energy cost of anhydrous ammonia N and 23% of the energy cost of urea N. In addition to providing energy efficient nitrogen, liquid pig manure also provides P, K, S, and micronutrients, as well as organic matter capable of delivering nutrients to several subsequent crops.

7.4.2.3 Effect of hauling distance on energy cost of manure N

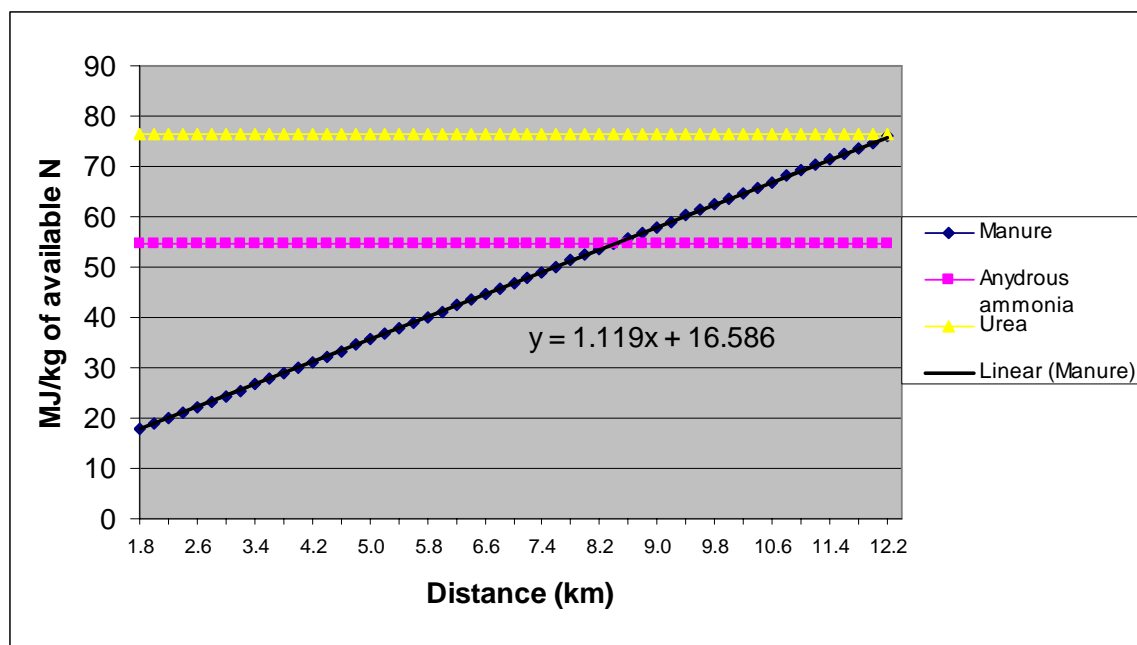
Since pig manure is less portable than inorganic fertilizer N a proper comparison of energy costs of manure N and inorganic fertilizer N requires consideration of distance between pig barn and field. In our example, the assumed average hauling distance for manure was 1.81 km, a common average hauling distance for liquid pig manure in southern Manitoba (Gallup, 2006 – personal communication). However, newly introduced provincial legislation regarding phosphorus application rates may necessitate longer hauling distances in order to prevent excessive P accumulation in soils of fields surrounding hog barns (Akinremi et al. 2007). Hauling manure longer distances will increase the energy cost of manure derived N. Based on the assumptions used for operation time and hourly fuel use (Tables 7.1, 7.5, 7.6), we were able to calculate how hauling distance affected energy consumption with the slurry wagon system at a constant application rate of 81.49 m³ ha⁻¹ (Figure 7.1).

Given that travel time with the slurry wagons was 2.18 minutes km⁻¹ (Table 7.1) and hourly energy consumption with the slurry wagon system was 15339 MJ h⁻¹ (tractors: 15009 MJ h⁻¹ – calculated from Table 5; miscellaneous equipment: 330 MJ h⁻¹ – calculated from Table 7.6), the additional energy cost for each added km of transport distance is approximately 688 MJ ha⁻¹ or 5.59 MJ kg⁻¹ of N (2.18 min km⁻¹ * 2 km = 4.36 minutes per round trip; 4.36 min * 224 trips per wagon to transport 29,560 m³ of manure = 16.28 hours for each km that manure is transported from the storage site; 16.28 h * 15339 MJ h⁻¹ = 249719 MJ; 249719 MJ / 362.8 ha = 688 MJ ha⁻¹; 688 MJ ha⁻¹ / 123 kg N ha⁻¹ = 5.59 MJ kg⁻¹ N). Recalling the energy costs of synthetic N fertilizers, manure could be hauled about 8.4 km before the energy cost per kg of available N is equivalent to anhydrous ammonia derived N (Figure 7.1). Average hauling distance could be increased to 12.3 km before manure N is equivalent in energy cost to urea N (Figure 7.1).

Previous work in Italy on the energy costs of manure hauling has shown that liquid pig manure can be transported up to 66 km before the energy per kg of N is equivalent to urea derived N (Ceotto 2005). Ceotto (2005) used manure with 3 kg N t⁻¹ for his calculations, whereas the manure in the current study contained only 1.51 kg available N t⁻¹. The lower N concentration of manure in the current study compared to the work by Ceotto (2005) partly explains why manure hauling required more energy per kg of N in the current study. However, if the same calculations made by Ceotto (2005) are conducted using manure containing 1.51 kg available N t⁻¹ (i.e. 1.51 kg N t⁻¹ x 76.3 MJ kg⁻¹ of urea-N = 115.2 MJ to produce the equivalent amount of urea-N as is contained in one tonne of pig slurry; 115.2 MJ t⁻¹ / 3.47 MJ t⁻¹ km⁻¹ hauling energy =

33.2 km), the distance which manure can be hauled before the energy cost is equivalent to urea-N is 33 km, still much longer than the 12.3 km we calculated in our study (Figure 7.1).

Figure 7.1 Effect of hauling distance on energy consumption for manure application with slurry wagons



The remaining discrepancy between our results and those of Ceotto (2005) may be explained by two factors: 1) our energy budget included activities other than just transportation, such as agitation and application; and 2) our higher calculated energy costs for manure transportation. Ceotto (2005) uses an energy value of $3.47 \text{ MJ t}^{-1} \text{ km}^{-1}$ for manure transportation. In our study only 56.66% of the operation time for the transport and application tractors was in road travel. Therefore, in our study, 56.66% of $1835.9 \text{ MJ ha}^{-1}$ for tractor energy use (Table 7.5) equals 1040 MJ ha^{-1} for manure transportation, or 12.76 MJ t^{-1} , which, when divided by our average hauling distance of 1.81 km results in $7.04 \text{ MJ t}^{-1} \text{ km}^{-1}$, which is just over twice the value for manure transportation which Ceotto (2005) cites (i.e. $3.47 \text{ MJ t}^{-1} \text{ km}^{-1}$), suggesting that the energy consumption for transporting liquid pig manure in southern Manitoba with slurry wagons pulled by tractors consumes twice as much energy as estimated by Ceotto (2005). Alternatively, our results may be twice as high as those of Ceotto due to our inclusion of the energy expended on the empty return trip of the slurry wagons. The 1.81 km distance is the distance of manure transport from the manure storage, not the round trip distance covered by the slurry wagons, which always come back empty. Therefore, if we include the round trip distance of 3.62 km the calculated the energy cost is $3.52 \text{ MJ t}^{-1} \text{ km}^{-1}$, very close to value used by Ceotto; however, using the entire round trip distance in the calculation means that the manure is only being moved half of this distance from the storage site.

Our original calculation for manure transport energy of $7.04 \text{ MJ t}^{-1} \text{ km}^{-1}$ corresponds well with the value of 0.2 L of diesel fuel per tonne of manure per km of transport reported by Dalgaard et

al. (2001). When 0.2 L is multiplied by 40.5 MJ L⁻¹, Dalgaard's (2001) energy coefficient for diesel fuel, the resulting energy consumption for manure transportation is 8.18 MJ t⁻¹ km⁻¹. However, both our study and that of Dalgaard et al. measured fuel consumption for transportation together with fuel consumption for loading the manure, and then subtracted the energy consumption estimated to be attributed to loading; therefore, these values for manure transportation are estimates.

In summary, our analysis of the energy cost of manure derived N shows that liquid pig manure can be transported up to 12.3 km before energy consumption per kg of crop-available N approaches that of synthetic urea nitrogen fertilizer.

7.5 Conclusions

The energy consumed for hog manure application was 2180 MJ ha⁻¹ with the drag hose system and 2185 MJ ha⁻¹ with the slurry wagon system when manure was applied as the Full treatment single application of 81.49 m³ ha⁻¹ to provide 123 kg ha⁻¹ available N to the following crop. Very similar amounts of energy were consumed by the two manure application methods but the ways in which the energy was consumed differed. The slurry wagon system used a higher proportion of total energy as fuel (94.5%) than did the drag hose system (88.4%), and the drag hose system accrued higher machine energy (11.2%) than the slurry wagon system (5.3%) because of drag hose replacement and also because of the longer application time with the drag hose system (149.9 hours) than the slurry wagon system (51.1 hours).

Energy use increased to 2726 MJ ha⁻¹ and 2209 MJ ha⁻¹ for the drag hose and slurry wagon systems, respectively, if manure was applied as a split spring/fall application with 50% applied at each application. The large increase in energy use from the Full to the Split application treatment with the drag hose system was due to higher replacement rates for drag hoses and a higher fuel use rate for the application tractor.

The energy cost per kg of available N from manure was 3 to 4 times less than the energy cost of N from inorganic fertilizer. One kg of crop-available N from liquid pig manure was delivered with 17.72 MJ and 17.76 MJ with the drag hose system and the slurry wagon system respectively. In comparison, the energy costs of N from anhydrous ammonia and urea are 52.21 and 75.63 MJ/kg of N, respectively. Such information is important in an age of increasing fossil fuel costs.

Transportation of liquid pig manure was found to consume 17.04 MJ per tonne per km. The distance at which hauling liquid pig manure expended the equivalent energy as the use of urea fertilizer was 12.3 km. In southern Manitoba a typical hauling distance for liquid hog manure is currently 1.81 km but may soon increase with newly introduced nutrient loading regulations. These results suggest that changing from N to P-based manure regulations in Manitoba will not seriously compromise the superior energy efficiency of manure and a nutrient source for crop production.

With short hauling distances, the superior energy efficiency of N from liquid pig manure over N from inorganic fertilizers offers opportunities for farmers to improve the energy efficiency of industrial agricultural production by substituting manure for inorganic fertilizers.

This study provided the energy balance of pig manure transportation and distribution but did not take into account the energy cost of delivering the feed to the pig barn. Future research that considers the entire crop-livestock system should be conducted.

7.6 References

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CHAPTER 8 Pathogens and Antibiotic Resistance

Assessment of the effectiveness of current Manitoba policy and manure management practices in addressing pathogen contamination and an assessment of the contribution of animal agriculture to antibiotic resistance in human medicine

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8.1 Executive Summary

Food and waterborne disease is a major economic and social concern in Canada where about 25% of people suffer gastrointestinal illness each year with associated costs exceeding \$3.7 billion. Most illness, except for significant outbreaks, is unreported, and in only about half of the reported cases is the agent responsible identified. Difficulties associated with illness case identification are compounded in Canada by a surveillance system that contains gaps and inconsistencies in its ability to acquire relevant data. Also, there is no systematic approach to tracking antibiotic resistance organisms in Canada. There are programs in certain provinces, but not all, and Manitoba is one of the exclusions. Most effort has been put into pharmacy prescribing practices, less into veterinary medical practice, and even less for feed mill purchases and distribution. There is currently no comprehensive monitoring system of sub-therapeutic antibiotic use in swine facilities in Manitoba, and municipal water supplies are not systematically assessed for a range of organic pharmaceuticals.

Nonetheless, the Canadian system is capable of rapidly identifying changes in frequencies of reportable illnesses caused by specific disease agents (eg. *Salmonella*, *Escherichia (E.) coli* O157:H7 and some viruses like Hepatitis A). It is not surprising that there are no data available in Canada which firmly link food and waterborne disease to livestock industry activities. However, there are data available in Canada which suggest there is a link (eg. the Walkerton outbreak) and there are data in other countries where the connection is clearer. This is evident in the reports from the U.S. which have involved Canadian consumers where large numbers of people have become ill from consumption of fresh produce contaminated directly or indirectly by livestock operations.

Enteric pathogens which cause 90% of food and waterborne illness in humans are resident in the gastrointestinal tract of animals, but the animals are largely unaffected by their presence. Thus, detection of these “zoonotic” organisms requires laboratory analysis and confirmation. At present the major organisms of concern from a human health perspective include the bacteria

E. coli O157:H7, *Salmonella*, *Campylobacter*, *Yersinia* and *Shigella* as well as the protozoans *Giardia* and *Cryptosporidium* which cause the majority of cases of gastroenteritis from the consumption of contaminated food and water. Enteric viruses from domestic food animals are not a major human health concern. The risk is significant that an outbreak of illness traceable to livestock farming operations occurs in Canada because manure contains these pathogens when it is applied to pasture and other arable land.

Most zoonotic pathogens die slowly in stored manure slurry. Holding manure for 90d at moderate (25°C) temperature has the potential to significantly reduce bacterial pathogen viability; however, continuous addition of new manure, which is normal practice, will stabilize pathogen levels in the storage. Thus, as currently practiced in Manitoba, manure storage will not eliminate pathogens of human health concern. Given the frequencies with which zoonotic pathogens occur in hogs it is very likely that most manure when spread on land in Manitoba is contaminated with some of the above organisms.

The *Campylobacter* species in hogs, unlike the predominant species in poultry, is not a concern for human health, however, *Salmonella* are an ever present risk and toxigenic *E.coli* (like O157:H7) plus *Yersinia* (which grows at cold temperatures) may be in the future. Transmission of *Shigella* and enteric viruses affecting humans is largely unrelated to hog operations. Both *Giardia* and *Cryptosporidium* survive better than the bacterial pathogens in the environment, and although human health risk from *Giardia* appears higher than from *Cryptosporidium*, *Giardia* cysts are more susceptible to lethality from freezing. This suggests there may be some benefit to fall manure application where *Giardia* and *Cryptosporidia* are more likely to become frozen in the field than in the manure storage reservoir.

Survival of zoonotic organisms after manure application to soil is variable but shorter than when in storage, and at levels likely present in soil will persist for 30d in spring weather. It is ironic that incorporation of manure into soil reduces environmental spread but prolongs pathogen survival in soil. Sunlight (UV), desiccation, low moisture, low organic matter and warm temperatures accelerate pathogen decline.

Recognition is growing in the European Union (EU) that animal manures should be treated to eliminate zoonotic pathogens (eg. 70°C for 60 min) before manure is used in the external environment. Such regulations are in place there for transported manure which is traded, and they exist in both the EU and North America for the treatment of municipal sewage but not manure, even though sewage and manure slurry are very similar in terms of the potential for pathogens to be present.

Recommended best management practices for Manitoba hog farmers (MB-1) will reduce the extent of pathogen distribution in the environment, but cannot be expected to interrupt their recycling among animals or prevent pathogen distribution from manure application to land.

Among seemingly attractive alternatives to thermal or tertiary treatment (as is done for municipal sewage) to reduce risk from pathogens in animal manure, adoption of multi (>2) cell lagoons, as well as addition of 0.5% ammonia or 2% urea directly to slurry should be explored in Manitoba. These three methods can eliminate zoonotic pathogens from manure.

Environmental risk posed by manure borne pathogens is not specifically addressed by measures taken to prevent chemical contamination and accumulation in soil by agricultural activity. It is unknown whether manure application rates and setback distances prescribed will prevent pathogen distribution in the environment. It is unlikely that present regulations

prescribing the time of manure application will eliminate the risk of recycling *Salmonella*, *E.coli*, *Yersinia* and protozoans in the environment.

The first part of this chapter will assess the importance of zoonotic pathogens in animal agriculture (hog production) and its relationship with food and waterborne illnesses in humans. The contribution made by animal production practices to the development and persistence of antibiotic resistance in zoonotic pathogens will be discussed in a separate section. The effectiveness of current hog production/management practices and policy in Manitoba will be evaluated in terms of their ability to minimize contamination by pathogens associated with intensive hog operations. Emphasis will be placed on strategies used to reduce levels of pathogenic organisms in hog manure in Manitoba and these will be evaluated in terms of experience reported in other jurisdictions in the management of animal wastes.

In the second section of the document we will discuss the potential link between antibiotics in livestock production and natural waters which included surface water and groundwater. We will also discuss in some detail the evidence that exists to support, and refute, the role that animal agriculture has in increased resistance to antibiotics currently seen worldwide in antibiotic resistance. Furthermore we will discuss the link between natural water bodies (lakes, rivers, streams, ground water etc.) and the likelihood of an increase in antibiotic resistance in human medicine. From a scientific point of view it is important to understand how a link between the two can be verified. The issues are not that simple. On the one hand there are those that believe that antibiotic usage in livestock, and particularly swine, is the major contributor to antibiotic resistance in human medicine. There are those who believe that the opposite is true and that it is difficult, or even impossible to point a finger at livestock. What we will attempt to do is to look at the evidence and formulate a balanced view based on the strength of the evidence. We will then extrapolate, where appropriate, to the Manitoba situation, and finally make recommendation as to where the gaps in knowledge lie.

8.2 Relevance of Issue

Foodborne illness is a major economic and social issue in Canada and the United States. In the U.S. it is estimated that each year there are 76 million illnesses, 325,000 hospitalizations and 5,000 deaths resulting from foodborne disease (Mead et al., 1999). The leading bacterial causes of illness are *Campylobacter* and *Salmonella*, and while *Campylobacter* cause a larger number of illnesses, *Salmonella* infections cause greater mortality (Table 8.1). *Shigella* is an important bacterial group causing gastroenteritis, largely through consumption of contaminated water or food which has come in contact with poor quality water. Other bacteria including *Listeria monocytogenes*, toxigenic *Escherichia coli* (like *E.coli* O157:H7) and *Vibrio* cause far fewer illnesses than *Campylobacter* and *Salmonella*, but case/fatality rates are significantly higher (Potter et al., 2002). Viruses, particularly the Norwalk-like or caliciviruses (often called noroviruses), are believed to cause as many as 65% of acute cases of gastrointestinal (GI) illness with low mortality (Mead et al., 1999). However, better analytical methods are needed to establish the true magnitude of their involvement in foodborne illness. Parasitic protozoans (*Toxoplasma*, *Giardia*, *Cryptosporidium* and *Cyclospora*) cause far fewer illnesses, but it is notable that in the U.S., *Toxoplasma* are endemic in companion animals (particularly house cats), and are believed to cause significant mortality in humans (Table 8.1). Unlike most bacterial pathogens of concern in food, viruses and parasitic protozoans are environmentally inert and the cyst/oocyst forms of the protozoans are resistant to most forms of environmental stress as well as many of the methods used to control bacterial pathogens, such as chlorine treatment.

Table 8.1. Causes and frequency of foodborne illness in the United States

Agents	Cases/100,000	Deaths
Bacteria		
<i>Campylobacter</i>	17.8	99
<i>Salmonella</i>	14.8	553
<i>C.perfringens</i>	2.5	7
<i>S.aureus</i>	1.9	2
<i>Shigella spp.</i>	5.0	14
<i>Yersinia enterocolitica</i>	0.8	2
<i>E.coli O157:H7</i>	2.1	52
<i>B.cereus</i>	0.3	0
<i>Vibrio spp.</i>	0.06	31
<i>L.monocytogenes</i>	0.5	499
<i>Brucella spp.</i>	0.008	6
<i>C. botulinum</i>	0.0006	4
Protozoa & Parasites		
<i>Giardia lamblia</i>	2.0	1
<i>Toxoplasma gondii</i>	1.1	375
<i>Cryptosporidium parvum</i>	0.3	7
<i>Cyclospora cayetanensis</i>	0.2	0
Viruses		
Norwalk-like viruses	92.0	124
Hepatitis A	0.04	4
Potter <i>et al.</i> (2002)		

The most common human enteric protozoan infections in the U.S. are caused by *Cryptosporidium* and *Giardia* with some instances of illness outbreaks being caused by *Cyclospora*-contaminated fruit. Over the last 25 years in the U.S. enteric protozoans have been the leading cause of waterborne disease outbreaks for which a cause is known. Community-wide outbreaks caused by parasitic protozoans are often due to the use of unfiltered surface waters or water supplies which are inadequately flocculated or filtered (Potter *et al.*, 2002). Two recent relatively large outbreaks of *Cryptosporidium* caused by contaminated municipal drinking water in the U.S. involved nearly 0.5 million people (Potter *et al.*, 2002).

Farm runoff/untreated sewage could be implicated in future similar outbreaks. Both food and waterborne (gastroenteritis) illnesses in Canada are of significant concern and unlike the U.S. data presented (Table 8.1) where only foodborne illness is represented and waterborne illness is considered separately (Lynch *et al.*, 2006), Canadian data available combine illness from both food and water. This explains why cases of illness caused by parasites (protozoans) in Canada (Table 8.2) are much higher than in the U.S. (Table 8.1).

In Canada it is estimated that each year there are 2.5 to 11 million cases of GI illness (1 in 4 people) costing between 2.2 to 3.7 billion dollars. Majowicz *et al.* (2006) reported an in-depth study of gastroenteritis in Hamilton ON and found average costs/case were \$1089. This is likely an underestimate since the figure does not include surveillance costs and the cost associated with care of individuals who developed chronic illness lasting beyond several weeks.

Table 8.2. Major enteric pathogens isolated from humans (laboratory confirmed) in Canada, 2002-2004^a

	no. isolations		illness cases/100,000 persons	
	2000	2004	2000	2004
<i>Campylobacter</i>	12,000	≥11,000 ^b	39.1	31.4 ^b
<i>Salmonella</i>	4,500	5,500 ^c	20.5	17.2
Parasites ^d	5,800	5,500 ^c	18.5	17.3
<i>Shigella</i>	1,900	900	3.7	2.3
<i>Yersinia</i>	700	600	2.6	1.9
<i>E.coli</i> O157	1,900	1,100	5.9	3.4

a) PHAC (2007)

b) data incomplete for 2004

c) *Salmonella* serovars Typhimurium, Heidelberg, Enteritidis were 56.4% of isolations

d) *Cryptosporidium* (10.6%), *Cyclospora* (2.6%), *Entamoeba* (12.1%), *Giardia* (74.7%)

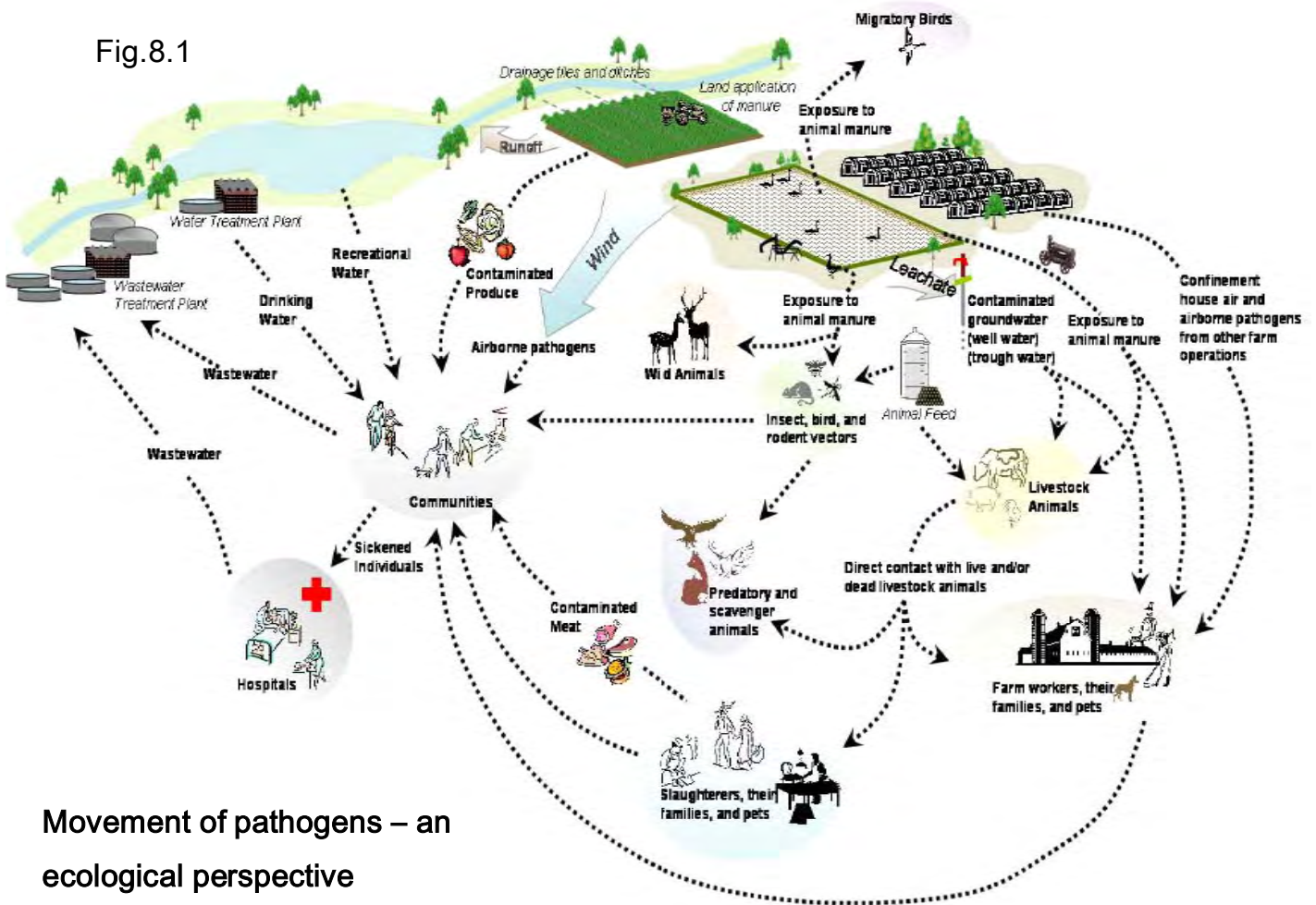
Data on food and waterborne gastric illness in humans in Canada is collected by two almost separate, independent systems, the National Notifiable Disease Reporting System (NDRS) which compiles data on mandatory notifiable disease agents from provincial/territorial health units (an epidemiological data base), and through the National Enteric Surveillance Program (NESP) which is a laboratory data base of identified organisms isolated from humans, animals and the environment. There are significant inconsistencies and gaps in the reporting systems with most (8) provinces/territories reporting data on a case by case basis, while the remaining 5 report only aggregate data. In addition, it is known that only 13% of the 1 in 5 people who seek care for acute GI illness actually submit a sample for analysis. Thus, current data describing causes of GI illness in Canada (>24,000 cases in 2004, Table 8.2) are believed to represent only a fraction of the actual human cases that occur (PHAC, 2007). The incompleteness of the information is illustrated by other data in the same report that note there were only 82 records of outbreaks of GI illness among Canadians which yielded reports of only 737 cases of illness in 2004. Inadequacies in the systems (recognized by the Public Health Agency), consequently limit the value of the data presented in the annual report (PHAC, 2007) to establishing changes in illness trends from year to year and identifying organisms most likely causing the most frequent cases of foodborne/waterborne illness in humans (Table 8.2).

Given the above limitations, the organisms of major concern in Canada which cause the most cases of GI illness are *Campylobacter*, *Salmonella*, parasites (*Giardia*), *Shigella* (from water/foreign travel), *E.coli* O157 and *Yersinia*. Absent from the list are *Listeria* and viruses. *Listeria* cause a small number of cases of illness each year but the mortality rate is about 15% (Table 8.1, Lynch et al., 2006) much higher than for the organisms listed in Table 8.2. Satisfactory methods for virus isolation are not yet available and when developed will likely show that more than half of the acute cases of intestinal illness are caused by these organisms (Potter et al., 2002).

Enteric pathogens which are responsible for causing 90% of food and waterborne illnesses in humans reside in the GI tract of animals, and the latter hosts are for the most part unaffected

(Rogers and Haines, 2005). Without clinical symptoms, animals shedding pathogens in their environment remain undetected. There is continuous re-infection among animals in herds as well as exchange of pathogens between them and the environment. These zoonotic pathogens are more frequently detectable in animals and animal feces during periods of stress (eg. transport) with some animals being more susceptible at younger ages. Strains of some of the above pathogens can cause clinical disease in animals which facilitates their detection. Livestock farm surveys (oral, fecal, environmental samples) are used to understand the movement and distribution of these organisms (Fig. 8.1), and implement improved management practices to reduce the frequency of pathogen-positive animals. However, more needs to be done and this should include a concerted effort to consistently eliminate zoonotic pathogens (eg. *Salmonella* and toxigenic *E.coli*) from animal feed which can be contaminated with frequencies $\leq 10\%$ (Rogers and Haines, 2005).

Fig.8.1



Movement of pathogens – an ecological perspective
Rogers & Haines (2005)

Salmonella, toxigenic *E.coli* and *Shigella* are capable of causing large outbreaks of GI illness in humans through contaminated water/food. The size of outbreaks is often expanded by transmission of these organisms from human to human. Virtually all poultry and many hogs carry *Campylobacter* asymptotically. However, the predominant species resident in hogs, *Campylobacter coli*, is rarely pathogenic to humans. Poultry are the major reservoir of the human pathogen *Campylobacter jejuni*. Species of both *Campylobacter* and *Salmonella* are endemic (commensal) in poultry, cattle and hogs.

There are 2,700 serotypes of *Salmonella* and about 30 of these cause most disease in humans. *Salmonella* Enteritidis, *S. Typhimurium* and *S. Heidelberg* are the most frequent causes of human illness in Canada (Table 8.2). *Salmonella Pullorum* and *S. Gallinarum* cause illness in poultry and are normally absent from flocks. Multiple-drug resistant *S. Typhimurium* DT-104 kills cattle and makes hogs ill. *Salmonella* serovars are found in farmed hogs with frequencies of <50-83% of herds and 14-47% of hogs in the U.S., Canada, Denmark and The Netherlands. The predominant serovar in U.K. hogs appears to be *S. Typhimurium* while *S. Derby* is most frequent among hogs in North America. Hog carcasses are contaminated with *Salmonella* at rates of 0.1 to 70%, with about 1 to 3% of raw pork samples being routinely found *Salmonella* positive (Hald et al., 2003; Kranker et al., 2003; Valdezate et al., 2005).

Toxigenic *E.coli* O157:H7 are intermittently carried by ≥8% of cattle, most often without clinical effect (calves are sometimes ill). Poultry do not serve as a reservoir for this organism and the same was believed true for hogs until recently when Doane et al. (2007) reported *E.coli* O157:H7 occurrence in hogs on 4 farms in different U.S. states at about 9% of animals. In other studies *E.coli* O157:H7 was found in 2-10% of hogs tested in Canada, Japan, The Netherlands, Norway, Germany and Chile (Gyles et al., 2002; Caprioli et al., 1994). It is now evident that hogs are able to serve as a reservoir for *E.coli* O157:H7 (Cornick and Helgerson, 2004), and this should signal a continuing need to ensure pork is thoroughly cooked before consumption. Other toxigenic *E.coli* serotypes have also been found in hogs and their environment (Rogers and Haines, 2005).

Yersinia enterocolitica is another bacterium which is a major cause of enteric illness in Canada (Table 8.2), the U.S. and Japan. It is a significantly more important cause of human illness in Europe, particularly in Denmark, Holland, Belgium and Norway where pork consumption and yersiniosis are linked (Nesbakken, 2006). This is unlike the situation in Canada where pork is more thoroughly cooked before consumption. *Yersinia enterocolitica* has been found in Europe and the U.S. in 13-25% of market-age hogs and in 6-16% of fresh pork. In Canada 21% of hogs tested were *Yersinia*-positive. Normal healthy hogs carry the organisms on the tonsils and in the GI tract, and are its primary reservoir, with lambs being occasional carriers and cattle sometimes being positive (Bowman et al., 2007; Nesbakken, 2006).

Humans are usually the source of *Shigella*, noroviruses and the Hepatitis A virus which are significant causes of acute gastroenteritis. These organisms as well as the protozoans *Cryptosporidium* and *Giardia* are transmitted most often among and to humans by contaminated water.

In Canada, cattle appear to be the primary reservoirs for *Cryptosporidium* oocysts and *Giardia* cysts, but municipal sewage is also a significant source in the environment. In an analysis of a single hog and two dairy farms in Alberta, *Giardia duodenalis* (synonymous with *G.lamblia*, a human pathogen) was found on both types of farms (18% of hog samples were positive), but the human pathogen *Cryptosporidium parvum* was absent from the hog farm samples (Heitman

et al., 2002). Although *Cryptosporidium suis* was present in Alberta hogs, the organism is not pathogenic for humans (Guselle and Olson, 2005). In contrast, Rogers and Haines (2005) reported results from a Centers for Disease Control (CDC, US) study showing that *Cryptosporidium parvum* oocysts were present in 9 confined animal feeding operation (CAFO) waste lagoons in Iowa (at levels of <2.3 oocysts/ml), in well water near farms, and in a river adjacent to one hog farm. In a separate study, Beuchat (2006) reported that 60% and 36% of samples of irrigation water used for produce production in the U.S., Panama, Mexico and Costa Rica were contaminated with *Giardia* and *Cryptosporidium*, respectively.

All of the above organisms, but rarely *Campylobacter* (which does not survive well outside human/animal hosts) can be transmitted to fresh produce through contaminated water or by direct application of contaminated animal manures used as fertilizer (Beuchat, 2006). In addition, wildlife can serve as reservoirs for some human enteric pathogens (deer - *E.coli* O157:H7, beavers - *Giardia*) (Rogers and Haines, 2005).

Foodborne illness data in the U.S. over the past 25 years indicate that the frequency with which fresh produce (fruits, juices, vegetables) was implicated in causing human illness has increased from about 5% to 56% of cases of known etiology for the period 1990-2004 (Table 8.3). Reasons for the change include a doubling of fresh produce consumption per capita since 1980, increased importation of produce from countries with uncertain quality/safety standards, as well as the distribution of contaminated domestic produce. It is significant that produce-related outbreaks of illness now involve larger numbers of individuals (cases) than outbreaks caused by any other single food commodity (Table 8.3), reflecting both the large size of production lots and the wide geographic distribution of produce within the country. Whether contaminated pre- or post-harvest, most produce cannot be treated to eliminate contaminating pathogens before sale without destroying desirable organoleptic quality (Beuchat, 2006). Organisms (bacteria, protozoans, and viruses) commonly found to be responsible for illness outbreaks associated with produce are almost exclusively of animal origin. The potential presence of these organisms in animal waste and irrigation water used on arable land raises concern with respect to the microbial safety of produce which is generally consumed raw. There are now a large number of reports documenting the contamination of produce following application of animal manure to fields used for crop production and by runoff from manure-treated fields (Rogers and Haines, 2005). It is not only of interest that *Salmonella* were responsible for 25% of produce-related foodborne illness outbreaks in the U.S. (Table 8.3), but also that of 51 outbreaks involving 6 broad categories of produce, *Salmonella* serovars commonly isolated from hogs were present in each category of produce responsible for human illness, although the origin of the pathogens is unclear (Table 8.4).

Movement of zoonotic pathogens within and beyond the livestock environment involves a series of complex inter-relationships among the animal host, the environment, the pathogen (infectivity, viability) and humans (activity, susceptibility). Some of these relationships are shown in Fig. 8.1 which was developed for CAFO in the U.S. (Rogers and Haines, 2005). The diverse nature of routes by which distribution of zoonotic pathogens may occur is evident. Not only is contamination of produce by pathogens present in animal manure significant in terms of its likely impact on public health, but also consideration must be given to the impact that pathogen recycling may have upon livestock. It could be expected that the frequency with which animals test positive for zoonotic pathogens may change as animals are continuously exposed to contaminated feed/water in the field or barn. Consolidation of more concentrated foci for re-infection of animals that may result, could have a significant effect on the proportion of zoonotic pathogen-free food animals. (Also see the discussion about antibiotic resistance below.)

We have learned that waste from municipal sewage treatment plants should be pathogen-free before release in the environment, but are only learning that it may be equally important to ensure animal wastes do not contain pathogens before they are used on agricultural land.

Table 8.3 Commodity-related foodborne illnesses (US)^a

Source	Outbreaks	Cases	Cases/Outbreak
Seafood	984	9,969	10.1
Produce	639	31,496	49.3
Poultry	541	16,280	30.1
Beef	467	13,220	28.3
Egg	341	11,027	32.2
Pork	188	6,081	32.2
Produce (salmonella) ^b	51 ^c	4,664	91.5

^a CSPI (2006) Outbreak Alert (1990-2004)

^b CSPI leading bacterial pathogen on produce (25% of outbreaks)

^c CDC (2007) (1990-2004 data). 38 outbreaks since 2000

Table 8.4. Produce-related human illnesses caused by *Salmonella* (US)^a

No. Outbreaks	Source	<i>Salmonella</i> serovar ^e
13	Tomato (roma, grape)	Bar, Jav, Mont ^{bc} , New, St.p. Typ ^{bcd} , Thom, Vir
12	Alfalfa, clover, mung sprouts	Ana ^{bcd} , Bov, Ches, Hav, Mba ^{bc} , Mel, New, St.p., Sen ^{bc} , Sta
9	Melons (cantaloupe, honeydew, watermelon)	Ber, Jav, Hei ^{bcd} , Mue, Poona
9	Lettuce, parsley, mustard greens, coleslaw	Ent ^{bcd} , Jav, Joh, New, St.p., Typ ^{bcd}
4	Almonds, mango, mushroom, mixed vegetables	Ent ^{bcd} , St.p., Typ var. Copenhagen
4	Fruit juice, orange juice	Ent ^{bcd} , Gam, Hart

a) CDC data 1990-2004 (CDC, 2007)

b) Common cattle isolate

c) Common chicken isolate

d) Common swine isolate

e) *Salmonella* serovars: Anatum, Barenderup, Berta, Bovismorbificans, Chester, Enteritidis, Gaminara, Hartford, Havana, Heidelberg, Javiana, Johannesburg, Mbandaka, Meleagridis, Montevideo, Muenchen, Newport, Poona, Saintpaul, Senftenberg, Stanley, Thompson, Typhimurium, Virchow

8.3 Manure Management

The effectiveness of a variety of approaches for reducing pathogen viability during handling of animal manure is described with specific detail in Section 8.6. In this section the effectiveness of major methods used to handle manure will be described and include those which show promise for use in this region.

In a generic sense the systems used in animal agriculture for manure handling are either passive or active (Rogers and Haines 2005). Passive systems include lagoons, other types of storage before disposal, segregation of differently aged animals, vegetative buffer strips, constructed wetlands and land application. Active systems include composting, anaerobic or aerobic digesters as well as actively operated lagoons.

Lagoons which may be single or multi-celled are essentially anaerobic because of the large input load of organic matter and the limited oxygen which is only available through surface diffusion. Multi-cell facilities can be designed to have an anaerobic cell(s) followed by increasingly aerobic cells as the organic content of the waste stream decreases by settling and microbial degradation. Lagoons can be modified by installation of covers to collect methane or if a permeable cover is used it may serve to oxidize ammonia (NH_3) to nitrate (NO_3), which in turn is reduced to nitrogen (N_2) in the cover microenvironment. Active systems such as composting and anaerobic/aerobic digestion require considerably more labour input than is necessary for passive systems. Description of anaerobic/aerobic system designs, advantages and limitations, geographic regions of common use and characteristic biochemical reactions in different phases of manure treatment processes are given in Guan and Holley (2003b) and Burton and Turner (2003).

In temperate climates, microbial reductions of 90% occur simply by holding manure for >90d without treatment. These reductions occur faster (60d) in summer, and regardless of seasonal history, an interval of 30d was suggested following slurry application to pasture (to further reduce pathogen numbers) before allowing grazing by animals (Burton and Turner 2003). Two problems are evident with this advice. In most regions, including Manitoba, fresh manure is almost continuously added to storages and a 90% reduction in microbial numbers is insufficient to eliminate zoonotic pathogens from the stored manure. In addition, low ambient temperatures typical of Manitoba extend the period of pathogen survival in hog manure slurry to reach periods as long as or longer than those normally used to fill storage reservoirs having the minimum legislated capacity in Manitoba (Arrus et al., 2006). Multi-cell lagoons (2-3 cells) in series can each reduce bacterial numbers by 90% and thus have potential to exert significant control over pathogen distribution.

With other passive systems, such as properly constructed vegetative buffer strips (30m wide, <8% slope and >90% plant cover), even though a 50-99% reduction in bacterial pathogen/protozoan numbers can be obtained, viable organisms remaining are still greater in number than most regional water guidelines will allow for direct discharge. Similar pathogen reductions have been reported for constructed wetlands in the U.S. (Rogers and Haines, 2005). Segregation of animals into groups by age may protect animal health, but separation of waste streams from age-segregated barns appears to contribute little to reduced numbers of zoonotic pathogens in slurry.

In actively operated systems - whether composting, aerobic or anaerobic digestion is used, if maximum temperatures reached are <50° C the maximum pathogen reductions achievable are <2 log colony forming units (CFU)/ml (or g) unless several treatment reactors are used sequentially. However at higher, thermophilic temperatures (whether aerobic or anaerobic),

pathogen reductions in the order of 4 log CFU/ml (or g) are achieved in municipal waste treatment systems. Data available for animal waste treatment are insufficient to directly reach the same conclusion for animal waste, however, the situation is likely to be very similar (Sobsey et al., 2006; Rogers and Haines, 2005).

Physical separation of solids from liquid waste separates pathogens approximately equally in each phase and thus is not useful as a complete treatment by itself for pathogen reduction. Drying to <1% moisture can provide a 4 log CFU/g reduction, but drying to 5% moisture yields only a 1 log reduction. High pH (>9) through alkaline lime stabilization can generate 3 to 4 log reductions depending on reaction time (Sobsey et al., 2006; Rogers and Haines, 2005). Ammonia or urea treatment can be just as lethal as lime (Ottoson et al., 2007).

Among other active treatment systems, fractionation plus sequencing batch reactor systems (multi-stage, single reservoir) have proven effective for reducing bacterial pathogens/protozoans to undetectable levels in swine slurry (Coté et al., 2006b), although not all are entirely satisfactory (Pourcher et al., 2006). A number of environmentally superior technologies (EST) have been evaluated in North Carolina for hog slurry management and among these one multi-stage system was recommended (Vanotti et al., 2007). A Canadian system by "ATD" is in final development which yields an essentially sterile effluent (ATD), but these systems are more economic for large operations and require complex equipment installation. Recent work in Sweden suggests that alkaline ammonia or simple urea treatment of hog slurry has the ability to satisfactorily reduce pathogen viability and retain manure usefulness as a fertilizer (Ottoson et al., 2007). These systems may be readily adopted for use in Manitoba without expensive construction requirements.

Although hog manure slurry in storage can represent substantial reservoirs for environmental contamination by zoonotic pathogens for many months, particularly in Manitoba with its cool climate, once manure is applied to soil the death rate of most pathogens is accelerated. Lethality is greater, generally, at higher soil temperature, in the presence of UV irradiation from sunlight, desiccation, and at low soil moisture as well as low organic matter and clay content (Holley et al., 2006; Sobsey et al., 2006).

While surface application of contaminated manure slurry accelerates pathogen reduction, injection reduces the risk of surface water pathogen contamination by rainfall-enhanced runoff from manure-treated fields. Since the interaction of environmental factors upon pathogen viability in soil treated with hog slurry containing zoonotic pathogens is complex, it becomes almost impossible to meaningfully evaluate pathogen response to individual environmental factors experimentally. Consequently, the optimal approach would be to treat manure before its application to soil to ensure the absence of viable zoonotic pathogens. Alternatively, delay between manure application and use of the land for crops or grazing (≥ 30 d) or between manure application and harvest (≥ 180 d) are often followed to provide time for further pathogen reductions (Ingham et al., 2005). Observations that horticultural crops can be internally colonized by zoonotic pathogens after contact with contaminated manure (increasing pathogen environmental persistence and potential for human illness) is an unresolved issue (Beuchat, 2006).

8.4. Current production practices in Manitoba

Potentially all practices during hog production and manure management can influence the presence, survival and transport of pathogens. Since large concentrations of pathogenic microorganisms can be naturally present in fresh livestock manures, practices for managing hog

wastes can have the most significant impact on releasing pathogens into the environment. Management includes the handling, collection, storage, treatment and use of hog manure.

Commercial hog barns in southern Manitoba are specialized and separated according to different production phases which include sow, feeder, nursery or farrow-finish barns. Housing a large number of hogs in a small enclosed area (ie. confinement) can increase the potential for the spread of infectious diseases. To prevent or treat infected herds, antibiotics such as amoxicillin, neomycin, oxytetracycline, and penicillin may be routinely fed to pigs. This practice promotes the persistence of antibiotic-resistant bacteria that can be transmitted to humans in food, water and even air from barn dust. Hogs in Manitoba are typically raised indoors. Hoop-shelters, for example, are generally used to house grower/finisher pigs (*MB-1*). Hoop shelters are built on floors that are part concrete and part deep straw bedding where the latter is used to absorb most of the liquid manure excreted. Buildings with slotted floors, on the other hand, collect all the manure in a pit underneath stalls to be later flushed to covered concrete tanks. Because the type of shelter can determine the way hog wastes and manure will be collected and the total amount of manure removed from the hog barns, the choice of shelter also has an important influence on the potential for disease to be spread within a herd.

In Manitoba, pig manure is typically in a liquid form (<5% solids) and handled as slurry (*MB-1*). Liquid hog manure is stored in outdoor earthen lagoons or concrete steel structures installed at ground level or above, respectively (*MB-1*). Liquid manures stored in lagoon reservoirs can leak, seep or otherwise overflow and lead to uncontrolled release of concentrated waste. In Manitoba, as in most provinces in Canada, the main method of disposing of livestock manure is to apply it to pasture or cropland (grain and cereal) as an organic fertilizer usually in the spring and/or fall. Stored liquid hog manure may be surface applied (high or low-level broadcasting) or injected with or without subsequently being incorporated into the soil (*MB-1*).

Treating manure by physical, biological or chemical means has been shown to reduce the level of pathogens in manure as described previously in Section 8.3. Treatment of stored hog manure, however, is uncommon in Canada. According to a Farm Environmental Management Survey conducted by *Statistics Canada* in 2001, about 53% of stored hog manure in Canada did not undergo any treatment prior to land application (*FEMS, 2001*). Nonetheless, a variety of methods to treat hog manure are still used. Of the Manitoba farms that raise livestock, the majority claimed they used "composting" to treat stored manure (51%), while others used no treatment (35%) or heat drying (19.4%) (*FEMS 2001*). The data for composting was considered an overestimate since some farmers called conventional solid manure storage "composting". Instead, the term should be limited to a controlled process involving turning manure to allow complete aerobic decomposition of organic matter into humus (*FEMS, 2001*). Essentially, the way hog manure is handled on farms in Manitoba has the potential to cause adverse environmental effects through pathogen contamination of plants and water.

8.5 Environmental Protection Measures (EPM)

There are a number of protection measures or Best / Beneficial Management Practices (BMPs) recommended and used by farmers to handle manure in the most efficient and environmentally sound means possible. Accepted manure BMPs can be used to protect humans, animals, plants and water by reducing the risk of pollutant release generated by hog operations. All aspects of manure production and handling (ie. from pig to end use by crops) can be controlled in some way to reduce the risk of pathogen release to the environment. Hog waste management practices can vary greatly among jurisdiction due regional climate, land available for manure disposal, herd size and other variables such as the level of manure applied. Because very

similar protection measures have been adopted by the international hog farming community, BMPs will be summarised in the Manitoba context. Differences in protection measures used in other jurisdictions will be identified when discussing specific policies and mitigation strategies (Section 8.5.1). It is important to understand that many of the protection measures that will be described in this section are primarily designed to manage the distribution of nutrients, but they will be assessed for their effectiveness in controlling pathogen contamination.

8.5.1 Protection measures in Canada for reducing pathogen risk

The Canadian Code of Practice for Environmentally Sound Hog Production developed by the Canadian Pork Council provides provincial and local governments with mandatory and optional production practices to help address environmental concerns. The Code focuses on several major concerns including protecting land, water and air from contaminants, particularly from the handling, storage and use of hog manure. In Canada, such BMPs are meant to encompass both nutrient and pathogen release to surface and groundwater. According to this Code, “environmentally sound manure management” incorporates several considerations: long term manure storage, incorporation of manure below the soil surface, prevention of water contamination, odour control, limitations on manure application rates and adequate land area for applied manure (*CCP ESHP, 1996*). Specific recommendations in the Code suggest spreading manure only during the spring or summer, construction of manure storage facilities that can contain manure for 6-8 months and include runoff collection structures that can hold contaminated runoff from a 25-year storm event (*CCP ESHP, 1996*).

8.5.1.1 Manitoba

As will be discussed in Section 8.5.2, production practices in Manitoba rely on proper siting and construction of manure storage structures to prevent pollution of surface and ground waters. In addition to site selection, good hygiene and guidelines for handling, storing and applying manure should be followed to reduce public health risk by preventing contamination of food and water (*MB-1*). A variety of BMPs can be implemented to minimize potential pollution or safety hazards from nutrients in manure as well as odour and gas emissions. However, only those known or suspected to be relevant in minimising pathogen distribution will be cited and discussed (Table 8.5).

Table 8.5: Recommended Best Management Practices to minimise the impact of land-applied hog manure on pathogen distribution in the environment throughout the manure management chain.

Stage in Manure Management	Protection Measures / BMPs*
Site Selection	<ul style="list-style-type: none"> • Maintain good hygiene standards in buildings. • Comply with applicable local, provincial, and federal environmental regulations and zoning laws. <ul style="list-style-type: none"> • Establish setback distances for locating operation facilities from residences. • Locate to contain runoff from open lots or manure storages and accommodate for adequate land application of manure.
Manure Production	<ul style="list-style-type: none"> • Control volume of wastes produced. • Formulate diets to reduce manure production.

Collection and Storage	<p>To control potential water and odour pollution:</p> <ul style="list-style-type: none"> • Locate storage facilities at adequate distances from water ways to avoid pollution. • Provide adequate storage capacities. • Prevent rainwater from overflowing storages. • Cover storage structure when economical.
Treatment	<p>Although the treatment of hog manure is not common on Manitoba pig farms, in some instances the following practices can reduce the environmental impact:</p> <ul style="list-style-type: none"> • liquid-solid separation systems • composting • anaerobic digestion • aerated storage • vegetative filter strips and constructed wetlands • chemical additives
Application	<p>Although primarily designed to ensure efficient nutrient use by crops, these practices may also help decrease pollution potential from pathogens as well:</p> <ul style="list-style-type: none"> • Develop a manure management plan: <ul style="list-style-type: none"> • Apply manure at uniform and agronomic rates to meet crop nutrient requirements (timing). • Incorporate or inject manure into the soil to reduce ammonia volatilisation. • Protect surface and ground waters: <ul style="list-style-type: none"> • No direct discharge of wastes into any water course. • Establish appropriate setbacks and buffer strips between land application areas and water courses (eg. vegetative buffers) • Avoid applying manure on frozen soil, snow-covered ground or under otherwise wet conditions. • Avoid spreading manure near surface waters or waterways. • Avoid spreading manure on land that will not be growing crops/vegetation. • Restrict animal access to surface water (eg. construct a fence to separate livestock from streams). • Avoid using fresh, untreated manure to fertilize crops. • Delay or waiting periods are also recommended to limit the exposure of people and livestock. • Take precautions to reduce runoff and soil erosion: <ul style="list-style-type: none"> • Adopt reduced tillage which allows for incorporation of manure. • Practice contour tillage. • Maintain adequate cover on the soil surface using crop residue or cover crops. • Establish settling basins where erosion rates are high

**Adapted from Manitoba Guidelines: Tri-Provincial Manure Application and Use Guidelines and the Farm Practices Guideline for Pig Producers in Manitoba)*

In addition to the measures summarised in Table 8.5, it is also important to protect wells and routinely monitor drinking water quality. This can be done by ensuring proper well construction, and treating and testing domestically used surface and groundwater. Limits, thresholds, and other criteria described in the manure BMPs as well as the basis for these can be found in the Farm Practices Guideline for Pig Producers in Manitoba (*MB-1*). The regulations and standards, however, are defined only for nutrient loading of nitrogen and phosphorus in soils (*Livestock Manure and Mortalities Management Regulation, LMMMR*). Specific criteria will be discussed in further detail under Manitoba regulations (Section 8.5.2.1.1).

8.5.2. Public Policy & Mitigation Strategies

There are a number of current and proposed public policies and mitigation strategies used to encourage the implementation of the protection measures previously described. These strategies also reflect the individual needs of each jurisdiction. For instance, regulated limits for setback distances, storage periods and application rates vary among each jurisdiction based levels of production, geographic and climate conditions. All jurisdictions do, however, prohibit the direct discharge of livestock manure into any waterways and regulate to some extent the spreading of manure during the fall/winter season. Insight into environmental measures and emerging agri-environmental policy in other domestic and international areas of intensive pig farming will hopefully provide possible directions for how Manitoba's hog industry should address public health concerns.

8.5.2.1 Canada

At the Federal level, Canada has several public policies under which environmental protection from agricultural operations are addressed, including the Environmental Quality, Agricultural, Fisheries and Water Resources Acts. The Fisheries Act is the primary Federal Act that deals with pollution concerns resulting from farming operations. Unlike the U.S., however, no Federal agency in Canada has defined rules specific to the establishment and operation of intensive livestock operations (*OMAFRA, 2003*). Instead, each province in Canada has its own legislation to meet specific environmental standards. The practices recommended in the Canadian Code of Practice for Environmentally Sound Hog Production, for instance, provide ways for hog producers to meet objectives of legislative measures to protect the environment set by Federal and Provincial governments (*CCP ESHP, 1996*). It is important to note that specific antibiotic, hormone and pathogen issues are beyond the bounds of current environmental regulation. Biowaste composting standards for pathogens are provided by the Composting Council of Canada (*CCC*). Specified limits are <1000 MPN/g dry weight for fecal coliforms and absence of salmonellae (<3 MPN/4 g total solids). According to the Canada-Manitoba Farm Stewardship Program (*CMFSP*) for product waste BMPs, funding for on-farm storage is given only for use of compost systems that meet all standards and regulations set by the CCC. Manure treatment BMPs for composting manure and mortalities are also covered in this program.

Canadian producers have responded to environmental concerns about their production practices by developing voluntary initiatives, such as Environmental Farm Plans (EFP). For instance, the embodiment of EFPs in the CMFSP occurred as a result of the Federal-Provincial Agricultural Policy Framework agreement which encouraged farmers to evaluate their operations, develop practical environmental action plans and adopt BMPs. Program priorities include controlling manure nutrient addition to soils as well as pathogen and pesticide losses to the environment and promoting use of tools and techniques to manage associated risks (*EFP*). EFP programs are available across Canada. Environmental cost-share programs are available to assist in implementing projects.

8.5.2.1.1 Manitoba

The pork industry in Manitoba follows a number of Acts, Regulations and By-Laws intended to ensure good farming practice and protect the environment. Provincial legislation for manure management is defined in the Livestock Manure and Mortalities Management Regulation 42/98 (LMMMMR) that was created in 1998 under the Environment Act. Several amendments were made in 2004 and 2006, with the most recent change being made to include phosphorus in manure for application in the guideline. The LMMMMR focuses on issues related to manure storage facilities, manure handling and application. It also describes an Intensive Livestock Operation (ILO) as one with greater than 300 animal units (AU). Manitoba defines an AU as the number of animals of a particular species that collectively excrete 75 kg of total nitrogen in a 12-month period. For example, 300 AUs are equivalent to 240 farrow-to-finish sows (*MB-1*). Key points of *LMMMMR 42/98* that pertain to ILOs are as follows:

- A permit from Manitoba Environment is required for siting and construction of all manure storage facilities (new or modified). There are strict design and construction requirements for earthen manure storages.
- As of 2006, permits specific to hog operations were required in order to construct, modify or expand a confined livestock area that can house 10 or more AUs with several exceptions. The Director can issue permits for manure storage facilities if stored manure undergoes anaerobic digestion or a similar environmentally sound treatment.
- ILOs (> 300 AU) must prepare and register an annual manure management plan for the handling, storage, disposal and application of manure.
- The regulation specifies a minimum setback distance of 100m between manure storage facilities and wells, surface watercourses, sinkholes and springs.
- Winter spreading between November 10 and April 10 for farms with >300 AU is prohibited with several exceptions. Those exempt (operations with <300 AU) must still meet minimum setback requirements from water courses, sinkhole, springs and wells during manure application. Setback distances range from 150m to 450m for land with <4% and 6-12% mean slope, respectively.
- Fall spreading of manure to land located in an area sensitive to flooding is prohibited between September 10 and November 10 of any year. A farm is exempt if the manure is incorporated into the soil within 48 hours after application or injected into the soil, or if manure is applied on land with established perennial forages and the soil is not disturbed, or if there is adequate crop residue to control erosion.
- Manure application is regulated on the basis of both residual nitrogen concentrations for crop requirements and phosphorus levels as of the 238/2006 amendment. Application rates cannot result in more than 140 pounds per acre (157.1 kg/ha) nitrogen for Class 1, 2 and 3 soils; 90lbs/acre (101 kg/ha) for Class 3M, 3MW and 4; and no more than 30lbs/acre (33.6 kg/ha) for Class 5 soils.
- In all cases, manure cannot be applied if phosphorus levels in the soil are 180 ppm or greater. Phosphorus concentrations for manure application rates are regulated based on soil test phosphorus levels and the annual crop removal rate of P₂O₅.
- Manure should only be applied as fertilizer to land on which a crop is or will be grown and application to land with a slope greater than 12% is restricted.
- The direct discharge of manure into surface and ground waters is strictly prohibited.

The Farm Practices Guidelines for Pig Producers in MB discussed earlier (*MB-1*) provides recommendations for Best Management Practices (BMPs) and outlines requirements for manure management in order for Manitoba hog producers to meet the regulations set out in the

LMMMR. The LMMMR recognizes the progress in research and innovation related to livestock production, therefore requirements of the regulation may vary where innovative or environmentally sound practices/procedures are proposed. With respect to enforcement, there are protocols for reviewing and inspecting manure storage engineering designs. New construction and manure management plans are reviewed and registered. Regular inspections, ongoing education, research, and farmer assistance also continue to be practiced. Methods for dealing with violations range from warnings to legal prosecution. Bylaws enacted by municipalities to regulate livestock production, for the most part, coincide with provincial legislation.

8.5.2.1.2 Quebec

Quebec uses protection measures for hog production similar to those described for Manitoba. Under the statute of the Environmental Quality Act, the Agricultural Operations Regulation, R.Q. c. Q-2, r.11.1 (*CanLII*) includes prohibitions, obligations and standards related to the handling, storage, disposal, spreading and treatment of livestock wastes (Chapter II – III), as well as financial penalties for offences under certain provisions of the Regulation (Chapter IV). The Quebec Ministry of the Environment also endorses a number of these laws and regulations. Manure management practices are largely based on annual phosphorus production. The main points in Quebec legislation that pertain to ILOs (*CANLII*) include:

- Manure storage facilities should be watertight (ie. to prevent leakage and contact with precipitation), have a minimum of 250 days manure storage, and be located at least 150m from watercourses, lakes, swamps, marshes or ponds.
- Application areas should be at least 300m from property boundaries.
- The spreading of manure is banned between October 1 and March 31 of the next year with some exceptions. No aerial spreading (with an irrigation gun or cannon) is permitted.
- Agro-environmental fertilization plans for the spreading of manure are required for operations with >40 AU (one AU = the number of animals that generate 500 kg live body weight). Penalties are also in place for violations.

8.5.2.1.3 Ontario

In Ontario, manure management practices for hog operations are primarily found in the Canadian Code of Practice for Environmentally Sound Hog Production (*CCP ESHP, 1996*). Aside from regulations through its Environmental Protection Act, Water Resources Act, and Nutrient Management Act 267/03 amended in 2006 (Reg. 474/06) that control nutrient release from municipal and agricultural sources, Ontario does not have any specific formal regulations regarding swine waste management. This is because local governments are primarily responsible for regulating ILOs in Ontario. An ILO is defined as having >150 livestock units (LUs). For swine, 5 boars/sows, 4 feeder hogs (<120 kg), 20 weaners (4-30 kg) each represent one LU (*OMAFRA, 2003*). Under the Ontario Planning and Development legislation, municipalities are granted the authority to establish standards for minimum separation distances, siting, nutrient management plans and manure storage (*Speir et al., 2003*). Ontario farmers have also participated in developing EFPs since 1993, and this has been reported as being very successful in helping Ontario farmers adopt more environmentally sustainable practices (*OMAFRA, 2005*).

8.5.2.2 United States

Similar protection measures for hog operations have been implemented in the United States: these include requirements for manure management plans, defined manure application rates,

minimum separation distances from watercourses, and requirements for building/facility construction and expansion permits. In general, hog manure in the United States is managed to reduce the risk of pathogens by a variety of approaches: manure storage prior to disposal, use of vegetative buffer strips, restrictions for wetland application, construction standards, separating animals of different ages as well as active manure treatment systems that require additional labour and capital cost such as composting, aerobic and anaerobic digestion (*Rogers and Haines, 2005*).

The U.S. Environmental Protection Agency (EPA) oversees most of the environmental issues associated with agricultural activities for each State. The majority of animal agriculture in the U.S. is industrialized and livestock are raised in concentrated animal feeding operations (CAFOs). A very comprehensive review of the environmental impact of manure-borne pathogens from CAFOs is available from the US EPA National Risk Management Lab (*Rogers and Haines, 2005*). The report discusses emerging technologies to monitor pathogens in the environment, methods for microbial source tracking, manure treatment technologies and management practices plus ongoing research being conducted by the EPA and other Federal Agencies. The EPA specifies management measures (practices) to protect coastal waters from agricultural animal sources of waste which are considered non-point pollution. Pathogens are included under “animal wastes” as one of the major non-point source pollutants in addition to nutrients, sediments, pesticides and salts (*EPA-1*). Although State programs are required to specify management measures that conform to regulations, they are not required to use the specific management practices described by the EPA (*EPA-2*). Production areas for large swine CAFOs must also be able to contain all manure plus moisture runoff from a 25-year, 24-hour rainfall event (*NPDES*).

The Unified Strategy for Animal Feeding Operations (AFOs) developed by the USDA and US EPA in 1999 focuses on minimising water quality and public health impacts from improperly managed animal manure from large AFOs (*EPA-4*). Under the strategy, a final rule for the National Pollutant Discharge Elimination System (NPDES) effluent limitation guidelines and standards was enacted on February 12, 2003 and required: establishment of BMPs, permits for all owners and operators of swine CAFOs, and a comprehensive nutrient management plan for all CAFOs covered by a permit. A new zero discharge requirement for manure and other wastewater pollutants became enforceable for large swine, veal and poultry operations (*NPDES*). Nutrient management plans were recommended, but not required for smaller AFOs (<1000 AU or 2500 swine). Since pollution from agricultural activity is classified as a non-point source (surface runoff from fields or feedlots), regulations that control this type of pollution are primarily governed by individual States which are given authority by Federal law to manage their own programs and grant permits (*OMAFRA, 2003*). However, most regulations regarding agricultural practices and environmental protection at the State level are addressed at the Federal level by the USDA or EPA.

The Voluntary Environmental Quality Incentive Program (EQIP) of the USDA Natural Resources Conservation Service is designed to help CAFOs meet the manure application standards proposed by the EPA. This includes providing technical and financial assistance for nutrient management plans, waste management structures and incentives for environmental farm improvements. The three major hog-producing States (Iowa, North Carolina and South Carolina) have each adopted EQIP.

With respect to education and research, the Swine Odour and Manure Management Research Unit (SOMMRU) of the Agricultural Research Service (USDA) located in the National Swine Research and Information Center at Iowa State University is in charge of current and ongoing

research projects related to swine production. There is also the National Center for Manure and Animal Waste Management. In 2007, the USDA developed a protocol for quantifying and reporting the performance of anaerobic digestion systems for livestock manures (*Protocol, 2007*)

8.5.2.2.1 Iowa

Each State also has its own standards based on the Clean Water Act legislation for CAFOs. Iowa, for the most part, follows this Federal Act but legislation specific to Iowa hog farming practices can be found through its Animal Centre (*Iowa DD*). Most of Iowa's laws relating to hog confinement regulate the way in which manure should be handled, stored and disposed. Moreover, the extent to which a particular facility is regulated by the State depends on the size of the operation (number of hogs housed) and the method of manure storage used. State regulations set out minimum manure control requirements for all animal confinement operations, such as land application and separation distances. There is also a nuisance law and tax exemptions to encourage farmers to adopt practices that reduce air and water pollution.

8.5.2.2.2 North Carolina

North Carolina follows the NPDES guidelines set by the EPA, but unlike Iowa the North Carolina livestock industry is regulated largely at the State level. The North Carolina Department of Environment and Natural Resources (NCDENR) administers legislation focusing on large-scale hog farms. Additional legislation to the 1993 *Waste Not Discharged to Surface Waters* 0.0200 rules focus primarily on training/certification, hog production, problems with lagoon and sprayfield systems as well as flood plain building for livestock systems (*NCPC*). Environmental rules and regulations that apply to hog farmers in North Carolina include, for instance, requirements for nutrient soil testing and prohibiting the building of lagoon storages in a 100-year floodplain (*NCPC*). There are also regulations for the setback distances of lagoon and sprayfield systems from residences, property boundaries and water systems as well as standards for vegetative buffers and maximum manure application rates to meet nutrient crop requirements and reduce runoff.

Incentive in the late 90's to assess alternative manure treatment technologies led to the development of a voluntary plan by Frontline Farmers (a non-profit organization of swine producers and their families) and Environmental Defense to replace traditional lagoon/sprayfield systems with more effective methods of waste disposal (*Env Def*). In the fall of 2000, the Attorney General of NC and Smithfield Foods funded a six year research project to improve hog manure storage and treatment, known as the Smithfield Agreement (*Smithfield*). Results and details of the projects are available through North Carolina State University's (NCSU) Waste Management Programs (*NCSU*). In 2007, an incentive-based Early Adoption Program in the form of a Bill was developed that provides \$50 million to help farmers convert to more environmentally friendly waste disposal systems (*Rawlins, 2007a*). These "Environmentally Superior Technologies" (ESTs) identified under the Smithfield Agreement were designed to meet certain environmental and economic criteria. In addition to eliminating the discharge of waste to waters as well as ammonia, odour, nutrient and heavy metal emissions, one of the performance standards for an acceptable EST was that disease causing vectors and airborne pathogens substantially be eliminated. Bill 03/28 "Phase Out Lagoon and Sprayfield Systems" was unanimously passed by the Senate in April 2007, providing farmers with access to the above-mentioned grants (*House Bill*). As of September 1, 2007 new hog waste lagoons will be banned in the state of North Carolina (*Rawlins, 2007b*) because of regional geographic limitations (*Hill and Sobsey, 2003; Humenik et al., 2004*).

8.5.2.2.3 South Carolina

Complete up-to-date regulations specifically regarding confined swine feeding operations can be found in Chapter 20 of the SC Code of Laws (2006). A very extensive set of regulations regarding hog production is found in Part 100: Swine facilities (*SC Code of Laws*). Setback distances for swine operations, manure storage and treatment facilities from residences, property lines and water courses defined by South Carolina are not as restrictive as those set by either Iowa or North Carolina.

8.5.2.3 Europe

Livestock farming (especially pig production) in Western Europe is concentrated in several regions where large intensive farms are common. Also see below in relation to antibiotics. Often there is not enough land to effectively handle the manure produced and consequently the manure is treated as waste rather than a resource. The practice of transporting manure off site to biogas plants (BGP) for energy production or treating it to produce a safe product that can be recycled as fertiliser is widely adopted in European countries. Measures taken to deal with manure problems include development of guidelines for Good Farming Practices (GFPs) and regulations describing acceptable manure management methods. Since the environmental issues associated with livestock production are more consistent throughout Europe than in North America, regulations developed by the central government (EU) are more relevant regionally.

The European Environment Agency (EEA) under the European Commission oversees many of the current environmental protection measures and regulations for Europe, many of which are based on protecting water quality. In particular, the European Nitrate Directive and Common Agricultural Policy (CAP) deals with environmental protection from livestock production. The Nitrate Directive is targeted towards monitoring and controlling the release of nitrates into waters. Under this Directive, codes of “good agricultural practices” became mandatory (*EC Nitrates*). The Nitrate Directive promotes requirements similar to those in Manitoba, including use of fertilisers, manure, application methods and timing based on crop nitrogen requirements and sufficient manure storage. The rate of manure application should be <170 kg organic nitrogen per hectare per year (*Ondersteijn et al., 2002*). By 2003, the CAP was updated to integrate more environmental protection measures into a single incentive payment system. Under this new system, incentives for having intensive livestock operations were reduced and only those who met *Good Farming Practice (GFP)* and minimum environmental standards qualified for payments (*CAP*). Following GFPs includes, at the very least, complying with European Community and national environmental legislation, such as the Nitrates Directive (*CAP*).

EC legislation 1774/2002 strictly regulates the treatment of biowaste if it includes animal by-products (ABP) or manure and is mandatory in all member EC states. Category 2 materials, which include livestock manure, are those that pose a risk of contamination with a disease other than TSE or are at risk of veterinary drug residues. EC legislation 1774/2002 and 208/2006 specify processing standards for biogas and composting plants as well as requirements for reducing pathogens and indicator organisms in livestock manure and other ABPs being applied on agricultural land. Category 2 material can be recycled onto land if a separate heat treatment of at least 70°C for at least 60 minutes is used to achieve the required 5 log reduction in microbial contaminants in manure (*EC 1774/2002*). Additional legislation 208/2006 to be implemented in 2007 permits alternative treatments to pasteurisation, once individual member states have validated that such treatments have an equivalent hygienic effect. However, manure can be processed in biogas plants without pre-treatment (*Paavola et al., 2006*). The

effectiveness of the proposed heat treatment (70°C for 60 min) in the regulation was evaluated in a 2005 Environmental Food Safety Authority (EFSA) report.

Nation-wide *Salmonella* surveillance and control programs in both Denmark and Finland approved by the EU ensure that the most adequate measures are taken to reduce the risk of disease spread among swine herds. This includes monitoring and identifying infected herds and determining the level of infection. Under such programs, operations with infected herds must follow certain restrictions based on their infection status, including special slurry handling requirements. For instance officials in that country suggest that slurry from positive Danish *Salmonella* Typhimurium DT104 herds be deposited with a hose applicator and immediately ploughed (Boes *et al.*, 2005). Slurry from *S. Typhimurium* DT-104 infected Danish pig herds is also prohibited from being spread on pasture (Boes *et al.*, 2005). Details on the Danish *Salmonella* surveillance and control program are well documented and publicized (Nielsen *et al.*, 2001; Alban *et al.*, 2002; Wegener *et al.*, 2003).

The *Zoonosis Directive 2003/99/EC* is a system for monitoring pathogenic microorganisms from livestock and food of animal origin. Specifically for the control of *Salmonella* and other food-borne zoonotic agents, EC Regulation No. 2160/2003 was created. Most recently, it was announced that the European Food Safety Authority will launch a study that will assess the public health risks posed by *Salmonella* in pigs (Anon, 2007a). The microbiological risk of factors and effects of proposed measures will be assessed in addition to analysing whether reducing *Salmonella* in pigs will reduce its prevalence and the number of associated food poisoning cases (Anon, 2007a). The assessment will also help put targets and controls in place for *Salmonella* in the hog industry.

8.5.2.3.1 The Netherlands

There are basic measures applied to all hog farms in the Netherlands: no manure spreading is permitted in both autumn and winter; covers are required for stored manures and techniques must be used to minimise ammonia losses at spreading. Manure holding facilities should also have a 6 month storage capacity (Prairie Swine Centre). Due to stringent rules in the Netherlands on manure handling, the country seems to be constantly evaluating new technologies for the treatment of pig manure (Melse and Verdoes, 2005). Several practices have been proposed and studied in the Netherlands such as the concept of feed management to minimise the amount of animal waste produced and housing techniques to reduce ammonia emissions. Manufactured phytase, for instance, has been used as a feed additive to reduce phosphorus emissions, while the Green Label housing certification system serves as an incentive to farmers to reduce ammonia emissions by 40-60% (van der Peet-Schwering *et al.*, 1999).

The Netherlands was reported as having the “most stringent manure regulations in the world” (Blankenship, 1996), but this referred mainly to the control of mineral nutrient release from animal production systems. At the national level, the Dutch Minerals Accounting System (MINAS) places more stringent standards for maximum allowable nutrient loss or “loss standards” to better prevent water pollution by excess nitrogen and phosphorus from farms with high livestock density. By 2008-2010, the loss standards for phosphorus will be reduced from 40 to 20 kg P₂O₅ per hectare (Jongbloed *et al.*, 1999). Because of stricter MINAS loss standards to restrict the load of nutrients in soil and water, the Netherlands has requested use of a limit of 250 kg N/ha from manure applied on grassland as opposed to the EU Nitrates Directive limit of 170 kg N/ha (Neth-1).

8.5.2.3.2 Denmark

The Danish Ministry of Food, Agriculture and Fisheries is the main governing body overseeing research, education, regulations and financial incentives related to livestock production in Denmark. Of the EU countries, Denmark has been a leader in the research and production of biogas. There have been a number of government financial subsidy programs in the past which resulted in the almost 20 biogas plants in operation today (*Amon et al., 2006; Gooch, 2006*). On March 27, 2007 new fertilizer legislation was passed to restrict the sale and use of ammonium nitrate fertiliser in order to prevent terrorism and protect the environment (*Anon, 2007b*).

In Denmark the Knowledge Center of Manure and Biomass Treatment Technology leads research and education. The Center deals with specific issues regarding research and technology in livestock waste production, from feeding processes to environmental impact. Different manure handling technologies have been evaluated by Danish researchers for their feasibility compared to existing technologies (*Sorensen et al. 2003*). Specific requirements for the handling of livestock manure include a maximum 170kg/ha N application rate, a spreading ban between harvest and February, a 9 month manure storage capacity and covers for storage tanks (*OMAFRA, 2003*).

8.5.2.3.3 France

Compared to the Netherlands and Denmark, France supports the largest average pig herd sizes on about ten times more agricultural farming area (*Jongbloed et al., 1999*). In France, the amount of N from applied animal manure should not exceed 170 kg N/ha, with specific limitations around wells and rivers. No specific provisions of any recent regulations for hog production in France could be found, however, they appear to be mainly based on EU regulations.

8.5.2.3.4 Finland

An organization called Evira is one of the main food safety authorities in Finland. In collaboration with other safety and regulatory bodies, Evira has established a national environmental health control program for 2007. This program's goal is to coordinate the operations of various other programs, such as environmental health care, animal disease and welfare plus food control, in order to ensure that basic operating principles are consistent (*Evira, 2006*). *Siekkinen et al. (2006)* reported results from a survey of Finnish farms that related the influence of management activities, farm size, pest monitoring, feeding procedures, and hygiene to changes in the numbers of *Yersinia* and *Listeria* on the farm. The authors felt that this type of survey could work well in a variety of production systems to measure compliance with hygiene requirements. In an overview of EU agricultural activities, it was reported that the Common Agricultural Policy now pays more attention to hygiene, food quality and animal welfare concerns in response to the rise of more intensive farming while still finding ways to accommodate farmers. The Finnish National *Salmonella* Control Program, initiated in 1995, has been successful in reaching its objectives. In particular, risk assessments confirm a low incidence of *Salmonella* in foods of animal origin in Finland (*Evira, 2007*).

8.5.2.4 Asia

The ASIA-PRO-ECO project Chicken Manure Treatment and Application (CHIMATRA) is a joint project between Universities in Germany, Malaysia and the Netherlands which is funded by the EU (*Korner et al., 2006*). There are a number of ongoing activities within the project, including a treatment method involving agglomeration, hygienisation and drying to produce a pelleted chicken manure product. The ASIA PRO ECO project in Northern Vietnam is a 5-year EU initiative that began in 2002 to adopt policies, technologies and practices that promote clean and sustainable solutions to environmental problems in urban and rural Asia (*EuropeAid*). As part of this initiative, the Thai Binh project will assess animal waste surpluses, nutrient

requirements of crops and fish ponds, and define more reliable manure management and technology options (*Pig Trop*). It also gives guidance to agricultural policy makers at the national and provincial level mainly for the pig, crops and aquaculture production sectors.

8.6 Assessment of Environmental Protection Measures

The effectiveness of protection measures and public policy for mitigating the distribution of pathogens at each stage of the manure management chain will be discussed. From the discussion of public policies related to hog production, it is apparent that the environmental risk posed by manure-borne pathogens (MBP) is not specifically addressed by measures to prevent or minimise soil and water contamination by agricultural pollutants. The assessment here will evaluate the ability of current Manitoba strategies and policy to deal with the recycling of pathogens to the environment as a result of hog manure handling. Specifically, the effectiveness of storage, application and treatment practices in reducing levels of indicator and pathogenic organisms such as *E. coli*, *Salmonella* spp., and *Cryptosporidium* will be discussed.

8.6.1 Effect of manure management practices on relevant pathogenic microorganisms in hog manure

8.6.1.1 Manure Production

Production phase and animal age have been suggested as important factors influencing the survival of pathogens. The practice of separating animals based on their production phase or age group, for instance, has been cited as being effective in reducing specific pathogens, especially *Cryptosporidium parvum* (Rogers and Haines, 2005). Younger animals are known to frequently shed larger numbers of oocysts than older animals. There is some indication that *Salmonella* can not only be found at higher initial levels, but may also survive better in manures from hog nursery barns compared to phases of production involving older animals (Arrus et al., 2006; Pourcher et al., 2006). On the other hand, there was no difference in the rate of pathogenic bacterial decline between the slurries of breeder and finishing pigs (Hutchison et al., 2005a). Studies have also confirmed that dry matter content of hog or cattle slurry is either unrelated to or has no significant effect on how long pathogens can survive (Arrus et al., 2006; Nicholson et al., 2005; Hutchison et al., 2005a). Separating hogs and their manures by different age groups is not practiced nor regulated in Manitoba. However, the microbiological risk of collecting and handling all types of hog manure together in the same manner appears to be low.

8.6.1.2 Manure Storage

Since manures removed from hog barns should not be directly applied on land, storage reservoirs are used to accommodate the delay before manure is applied on land. In fact, no further treatment of stored manure is required by Manitoba regulations prior to land application. It is important, then, to examine the impact of storage systems on pathogen viability. Results of recent on farm studies support the ability of extended storage in aboveground or earthen reservoirs to reduce the survival of pathogenic bacteria such as *Salmonella* spp. in liquid hog manure (Arrus et al., 2006; Côté et al., 2006a; Hutchison et al., 2005a; Grewal et al., 2007). Storage, however, has a limited or no effect against potentially harmful *Clostridium* spores (Hill and Sobsey, 2003) or parasites such as *Cryptosporidium* spp. (Hutchison et al., 2005a).

8.6.1.2.1 Size (Capacity) and Operation

Manitoba policy requires that hog producers ensure sufficient storage capacity for the winter months when spreading is prohibited to allow spreading at more ideal times and at suitable rates for the efficient use of nutrients by growing plants (LMMMR). Recent studies have shown that not only can indicator and pathogenic bacteria be present in stored manures during the spreading period between spring and fall (Côté et al., 2006a), but that the manure storage

temperatures observed under typical Manitoba conditions could allow them to survive over winter to contaminate the land when applied in the spring (Arrus et al., 2006). These studies also recommended holding hog slurry for at least 60 or 54-114 days at ambient temperatures in the spring before application to reach undetectable levels of *Salmonella* and *E. coli* (<10 colony forming units per ml) (Arrus et al., 2006; Côté et al., 2006a). Guan and Holley (2003a) hypothesized that it should be possible to eliminate the major bacterial and protozoan pathogens from liquid manure holding systems with 3 months (90 days) storage at 25°C. This, however, was largely based on cattle manure data. Based on recent findings, it appears that this hypothesis can be made for swine manure systems provided certain conditions exist during storage. For instance, the recommendations for minimum storage periods were based on the assumption that storage would be conducted as separate batches that would not receive constant additions of fresh manure. The practice of adding fresh manure to stores can stabilize the level of pathogens even after several months of storage (Peu et al., 2006; Hutchison et al., 2005b). So, both manure storage capacity and method of storage operation are important factors determining when and how often stored manures can be safely applied to land. For operations prohibited from winter application, a minimum manure storage capacity of 200 days is required under the LMMMR and at least 400 days capacity is recommended for earthen manure storage structures to ensure that manure is not applied during the banned winter period. This is not much different from the 250 day minimum storage period required in Quebec, where the manure spreading ban is from Oct. 1 to Apr. 1 (CANLI). In comparison, Spain, Denmark and Norway require storage capacities of 8, 9 and 10-12 months, respectively (OMAFRA, 2003; Prairie Swine Centre). Under these regulations, it would appear that such storage structures are capable of meeting the holding periods previously cited. Recall that hog operations in Manitoba with <400 AU are exempt from the winter spreading ban. So storage capacities on these farms would likely be insufficient to hold more than a few months of waste. Like in many European countries and the United States, in Manitoba it is not practical to store manure in slurry tanks and earthen manure storages in which fresh waste is not continuously added (Hutchison et al., 2005b). Although Manitoba regulates the capacity of a manure storage facility, there are no recommendations describing minimum periods that manure must be held outside of the winter season to reduce the risk of spreading pathogens onto land.

Aerating liquid hog manure by mechanical pumping or stirring during storage can promote pathogen reduction (Hutchison et al., 2005a, 2005b; Peu et al., 2006; Grewal et al., 2007). A study conducted in Ohio reported that aerated liquid lagoon storage resulted in a 3 to 4 log most probable number/g (MPN/g) and 2 log MPN/g reduction of *Listeria monocytogenes* and *S. Typhimurium*, respectively, after 3 days compared to only a 1 log decrease of these pathogens by day 7 in an unaerated (anaerobic) lagoon (Grewal et al., 2007). As mentioned in Section 8.4., periodic mechanical agitation by pumping occurs in manure storages in Manitoba. However, constantly aerating manure storages is considered expensive (Manitoba Farm Practices Guideline, MB-1).

8.6.1.2.2 Design and Construction

As mentioned earlier, production practices in Manitoba rely on proper siting and construction of manure storage structures to prevent pollution of surface and ground waters. If we look at the situation in the southern United States, water and air pollution due to inadequately maintained storage structures is an important concern that supports Manitoba's strict policy for the design and construction of manure storages. In addition to problems of odour and ammonia loss, overflow and leakage commonly occur with the use of outdoor lagoons to store swine wastes. Recall that swine waste lagoon/sprayfield systems are no longer permitted in North Carolina as a result of the environmental problems they have been consistently reported to cause. A survey conducted by Richard and Hinrichs (2002) also found that the majority of earthen manure

structures in Iowa, for instance, have experienced problems with spills and leakages before, during and/or after storage. Therefore, if stored manure is not treated before land application, routine careful monitoring and maintenance of earthen manure storage structures may be important preventative measures. After evaluating the effectiveness of lagoon systems in reducing indicator and pathogenic organisms, it was recommended that either a series of multiple lagoon cells or alternative treatments be used to minimise the impact of pathogens in swine waste (Hill and Sobsey, 2003). Manure can be held in one cell for a prescribed minimum period of time, while the other cell(s) receives fresh waste. Controlling the release of manures based on their “age” allows the oldest supply to be applied first and avoids the spreading of fresh manure onto farmland. In fact, manure age was found to be one of the most significant factors influencing *E. coli* numbers in runoff (Meals and Braun, 2006). It was suggested that applying the oldest manures of at least 90 days from stores could significantly reduce pathogen distribution to land. Design and construction requirements for two-cell storage structures are provided in the MAFRI guideline (MB-1), but are not given any preference over single-cell structures.

Since many hog farms in Manitoba do not treat stored manure prior to land application, it appears important to identify minimum holding time requirements for storage systems to effectively minimise the risk of transferring pathogens from manure to the environment. This could include defining minimum storage periods, promoting multiple compartment storage structures or batch processes over continuous operations or providing financial support for farmers to aerate manure storages before land application.

8.6.1.3 Manure Application

In general, studies have shown that pathogens present in storages at manure spreading period can also persist in the soil. The 2006 amendment to the LMMMR was created in response to evidence that manure application based only on nitrogen content resulted in increased concentrations of phosphorus in some soils. In this chapter an attempt will be made to assess whether a similar concern is warranted for pathogens as for other environmental pollutants. In several of the plot and on-farm studies examined during preparation of the present report, *E. coli* was confirmed to be a better indicator of fecal pollution compared to other fecal coliforms (Gessel *et al.*, 2004, Boes *et al.*, 2005, Kjaer *et al.*, 2006). Fecal coliforms have limited value as indicators because they are naturally present at low levels in most soils and so they cannot specifically reflect manure use on land.

8.6.1.3.1 Rates

In most countries, manure management plans consider nitrogen and/or phosphorus for calculating manure application rates. The availability of manure nutrients can facilitate the survival of microorganisms in the soil environment (Saini *et al.*, 2003; Holley *et al.*, 2006). Although not detected on pickling cucumbers, *E. coli* and *Salmonella* were found in some soil samples at harvest two months after spring application of manure at a rate of 115 kg/ha nitrogen (Côté and Quessy, 2005). Little evidence has been found to confirm whether or not prescribed nutrient-based manure application rates in Manitoba and other jurisdictions are also effective in minimizing pathogen contamination. However, there is some indication that application rates above approved agronomic levels can increase the persistence of some pathogens in the soil. Although Minnesota researchers found no relation between the persistence of *Salmonella* in soil from liquid hog manure at application rates 0.5, 1 or 2 times the agronomic rate (Gessel *et al.*, 2004), the persistence of somatic coliphages did increase with higher application rates. Somatic coliphages can indicate the presence of enteric viruses. Findings of this on-farm study are relevant to Manitoba since the hog farming conditions and manure management practices used were within the regulated criteria for Manitoba hog farms (i.e., fall spreading of liquid manure

allowed by broadcast application with immediate incorporation into loamy soil cropped to corn and grain, and application rate based on soil test results) (Gessel *et al.*, 2004). Although the agronomic rate was reported in terms of the volume of liquid hog manure applied per hectare rather than based on crop nitrogen requirements, it can still be inferred that the risk of pathogen transport in runoff from fields treated with liquid pig manure can be controlled by acceptable application rates.

During a recently conducted two year study at a La Broquerie, MB field site, hog manure naturally contaminated with *Salmonella* Derby was surface spread at 16-20,000 L (110 kgN)/acre in single spring or in split spring/fall applications. *Salmonella* Typhimurium was recovered from some treated vegetation samples but none of the >200 cattle grazed on the site were found to be *Salmonella* positive (Walkty, 2007). However, results from two earlier studies indicated that *Salmonella* could be transferred to cattle from fields contaminated with *Salmonella*-containing manure (Jack and Heppler, 1969; Taylor and Burrows, 1971), but in these latter studies manure application rates were 2 to 5 times higher than in the Manitoba study which followed Provincial manure application rate guidelines.

8.6.1.3.2 Methods

There has been pressure to control ammonia and odour emissions by promotion of injection or incorporation for hog slurry application. The effect of incorporating manure into soil on the persistence of pathogens has been examined by several researchers. Although injection or incorporation can lower the risks of causing air and water pollution (Boes *et al.*, 2005), they may also increase the potential for pathogens to survive in the soil because the slurry is protected from drying and UV radiation from sunlight that would typically destroy pathogens on the surface (Gessel *et al.*, 2004; Nicholson *et al.*, 2005). Pathogens have been found to survive within the soil for several months after manure incorporation (Côté and Quessy, 2005; Holley *et al.*, 2006). In another study, incorporation increased the survival of *E. coli* strain RS2G compared to till and no-till methods (Saini *et al.*, 2003). In other words, incorporation can reduce pathogen distribution to surface water, but can prolong pathogen survival in the environment. Researchers in Denmark suggested that spreading manure according to recommended practices (injection and ploughing) poses a potential contamination risk to water courses with fecal bacteria (Kjaer *et al.*, 2006). During a controlled experiment, drainage water samples were found to contain *E. coli* at levels of up to 6700 colony forming units/ml (CFU/mL) between 50-211 days after fresh pig slurry was injected and immediately ploughed (Kjaer *et al.*, 2006). To put this in the context of acceptable drinking water quality, the *E. coli* levels observed in the Danish study exceeded the maximum acceptable concentration of zero *E. coli* per 100 mL under Canadian drinking water standards (Health Canada). On the other hand, *E. coli* was not found in water samples from another site in the same study after being either injected or immediately ploughed after hose application (Kjaer *et al.*, 2006). Incorporating manure into soil by ploughing and harrowing can also reduce the risk of pathogens in runoff. A study in Denmark found no *Salmonella* in soil samples (< 1 log CFU/g) 2 weeks after swine slurry application by either ploughing or injection, and no *Salmonella* on winter wheat seedlings within 24 hours of application (Boes *et al.*, 2005). Ploughing is also considered to be a suitable method of applying pig slurry when contamination levels are high (≥ 4 log CFU/g), but any application method might be used without major hazard based on the low levels and short survival times observed in subclinical *S. Typhimurium* DT104 Danish pig herds (Boes *et al.*, 2005). These findings support the LMMMR recommended practice of applying manure either by injection or immediate incorporation after surface application in order to reduce pathogen transport and animal exposure (LMMMR). Moreover, the results emphasize the importance of restricting fresh manure deposition on land to minimize pathogen distribution.

8.6.1.3.3 Timing

Sufficient delay or waiting periods between manure application and land use are recommended to limit the exposure of people and livestock to pathogens. Minimum delay periods are not specified in Manitoba regulations, but guidelines do recommend a minimum delay of 30 days between application of manure and introducing animals to graze on treated pastures (*MB-1*). This agrees with the minimum 30 day waiting period suggested to minimise the potential recycling of *Salmonella*, *E. coli* O157:H7, and *Campylobacter* between animals and crops (*Holley et al., 2006; Nicholson et al., 2005*). On the other hand, Quebec researchers *Côté and Quessy (2005)* suggested a 100 day delay between liquid hog manure application and harvest to avoid contamination from *E. coli* and *Salmonella* in sandy loam soils. Safe delay periods for other soil types have yet to be determined.

Regulating manure spreading in certain seasons is also an important way of reducing pathogen contamination. The ban on winter spreading in Manitoba and other jurisdictions means that manure must be applied between the spring and fall. But there is evidence to show that temperatures in the summer and fall would make it possible for pathogenic bacteria to survive for long periods. Using current production practices and limits (115 kg/ha N spread and incorporated to 20 cm depth) recommended for Manitoba hog farmers, *Holley et al. (2006)* reported in a laboratory study that *Salmonella* Albany (which was naturally present in the hog manure used to treat the soil) was not found 7 days after application, but when other strains of *Salmonella* were added at >5 log CFU/g, they survived >180 days. This suggests that if manure were contaminated at levels greater than 100,000 or 1,000,000 organisms per mL (normally levels are <350/ml, *Hill and Sobsey, 2003; Rogers and Haines, 2005; Fablet et al., 2006*), storage alone and the current measures that prescribe the time of manure application would not eliminate the risk of recycling *Salmonella* back into the environment over the summer and winter. Therefore, application at early spring may be best. Not only did over 67% of Manitoba farms report that they apply manure during the fall season, but at least 50% of farms with livestock incorporated manure after more than 7 days or just left it on the soil surface (*FEMS, 2001*). The new LMMMR restrictions on fall spreading of manure described in Section 8.5.2 which end the practices of surface application of manure in the fall without immediate or any incorporation may reduce pollution. Thus, currently recommended strategies for hog manure application should minimise pathogen distribution in the environment. Moreover, the dependence of delay/waiting periods on initial pathogen levels implies that testing manure and soils for fecal indicator microorganisms when planning application may be important. Biosolids, for instance, can be exempt in Ontario from waiting periods between application and land use if no pathogens are detected in the waste (*Guidelines, 1996*)

The limits and levels used in strategies to optimize crop nutrient requirements do not consider the maximum amount of manure that soils can handle for pathogen reduction and prevention of water contamination. Although concentrations of *E. coli* RS2G in soil were unaffected by rainfall, the levels in the leached waters declined with longer intervals between application and the first rainfall (*Saini et al., 2003*). Longer periods between application and a runoff event have been shown to reduce *E. coli* in runoff by about 50% (*Meals and Braun, 2006*). Farmers obviously cannot control rainfall, but they can time manure application to avoid forecasted rainfall events to reduce the risk of water pollution.

8.6.1.4 Pathogen transport and best management practices

The greatest risk of contamination by pathogenic microorganisms occurs when strong or persistent rainfall results in runoff from agricultural land that has been recently treated with manure. Consequent well-water contamination, in particular, can expose humans and animals to disease. Although faster rates of pathogen death can be achieved when manure is applied

closer to or on the surface of the soil, the potential for long periods of survival in soils may not justify surface application where manure can be carried in runoff and pollute waterways. Studies of runoff from land treated with fresh manure show there is significant potential for pollution of runoff water by pathogens at concentrations greatly exceeding acceptable safe levels (*Thurston et al., 2005; Meals and Braun, 2006*). Therefore the practice of prohibiting the direct discharge of wastes into any water course is critical to protecting animals and humans from infection.

8.6.1.4.1 Crop Cover

Recall that the LMMMR only allows manure application on land that is or will be growing crops, and that the new fall restrictions make an exemption for manure that is applied on land with established perennial forages or undisturbed soil. This suggests it is considered that there is a low hazard related to applying manure on crop-covered soil (*LMMMR*). Following is an examination of whether there is evidence to support this assumption. Data on the transport of indicator and pathogenic microorganisms in runoff are largely limited to results from experiments using cattle manure as the soil amendment (*Davies et al., 2004, Pachepsky et al., 2006, Stout et al., 2005*). Not only is the volume of runoff reduced on land with established vegetation (*Roodsari et al., 2005*), but the transport and number of fecal coliforms on the surface of soil with vegetation was decreased compared to bare soils. In addition, 100% of the fecal coliforms released from applied manure were found in bare clay loam soil but only 25% were found in bare sandy loam soil (*Roodsari et al., 2005*), suggesting that there is a greater risk of surface runoff from sandy loam soils or that bacteria penetrated below the 60 cm sampling depth. Evidence from other studies also supports adoption of precautions during the application of manure on coarsely-textured solids (eg. sand) to prevent pathogen transport (*Côté and Quessy 2005; Nicholson et al., 2005*). In another plot study, 20 times more *Cryptosporidium* oocysts in fresh cow pats from bare compared to crop-covered loam soil, were transported in the runoff over 1m during a controlled rainfall experiment (*Davies et al., 2004*). In general, these studies support use of adequate soil cover (eg. vegetation) to control runoff, as well as highlight the importance of considering climatic and land (soil) conditions when deciding how, when and where to apply hog manure.

8.6.1.4.2 Vegetation Buffers

Studies generally support the use of vegetation buffer or filter strips (VFS) to minimise the potential contamination of nearby drinking and irrigation water supplies by pathogens present in applied manure. Vegetation buffers were found to be effective in retaining manure-borne indicator (*E. coli*) and pathogenic organisms (*Cryptosporidium parvum* oocysts) carried in runoff from fields treated with cattle manure (*Davies et al., 2004; Atwill et al., 2006; Mankin et al., 2006*). Evidence shows that vegetation covers are particularly important for managing *Cryptosporidium* transport within watersheds. For instance, vegetation setbacks as little as 1m were sufficient in slowing the transport of oocysts (*Davies et al., 2004*). In another study, however, it was reported that regardless of a vegetation buffer, *C. parvum* was still retained in the applied fecal material during storm events (*Atwill et al., 2006*). Grass hay VFS reduced the average concentrations of *E. coli*, total N and total P in runoff by about 83%, 67% and 66%, respectively (*Mankin et al., 2006*). This suggests that vegetation buffers may be more effective in reducing losses of microbial contaminants compared to nutrients. In fact, Manitoba studies have shown VFS to be ineffective in reducing P losses because most losses occur during snowmelt when there is no vegetation and the ground is frozen (*MAFRI-1*). On the other hand, if soil erosion in the spring and summer is the primary cause for P loss, then like the studies cited, buffer strips would be effective (*MAFRI-1*).

Practices to specifically control the release and transport of microorganisms in agricultural runoff to surface waters have not been as widely adopted or tested as those designed to control

sediment and nutrients (*Meals and Braun, 2006*). Not only is there very little data available, but measuring manure borne pathogen (MBP) transport can also be very costly. Therefore, it may be feasible to use a natural tracer that would have similar transport patterns to those of MBPs (*Pachepsky et al., 2006*). Recently, similarities in transport of manure-borne phosphorus and fecal coliforms have been shown in controlled experiments for relatively short (1–6 m) transport distances (*Stout et al., 2005*). Whether or not this relationship will be the same for longer distances still needs to be verified. Still, information available on the transport of manure-borne P is much greater compared to that on MBP. Since many of the BMPs to minimise phosphorus in runoff are also used to control pathogen losses, using P to model pathogen transport may be informative from studies done after snowmelt.

In general, the studies on vegetation buffers indicate the ability of soil surface cover or plants to slow the transport of manure borne pathogens in runoff by allowing manure to be absorbed into the soil. Taller vegetation may even be beneficial in some cases (*Meals and Braun, 2006*). Certain crops/residues will differ in the ability to retain pathogens and prevent their transport in runoff. For example, indicator organisms (*E. coli* and enterococci) were not significantly affected by corn residue on manure-treated soil (*Thurston-Enriquez et al., 2005*). The effect of crop residues on the transport of pathogens from swine slurry, however, requires further investigation.

There is no evidence to support the adequacy of regulated setback distances between sites, storage facilities or application areas and water courses to prevent water contamination by pathogens. In general, maximum concentrations of microbial indicators decrease with distance from the location of applied manure because of dilution. Nonetheless, setback distances to waterways from application areas, soil cover and soil texture play important roles in controlling transport of pathogen pollutants in runoff (*Davies et al., 2004; Roodsari et al., 2005*). By limiting application to land with crop cover or vegetation with specific seasonal and soil fertility restrictions, current Manitoba policies may not completely address the recycling of pathogens in the environment but are as comprehensive as one might expect, given the uncertainty concerning pathogen survival in the manure-soil environment.

8.6.1.5 Manure Treatment

Unlike human or municipal wastes, animal manures are not required by law in most countries, including Canada and the U.S., to undergo any treatment to reduce pathogen content before land application. The U.S. EPA 40 CFR Part 503 Biosolids Rule places microbial limits (Class A and B pathogen requirements) on products of composting or anaerobic digestion destined for land application (*Foulkes et al., 2006*). It is ironic, then, that similar safety standards for treatment processes and pathogen reduction are not required for the agricultural use of livestock manures which might also contain a variety of pathogens at variable levels (*Rogers and Haines, 2005*). Still, in 2001, Manitoba treated stored manure to the greatest extent (64.7%) compared to Quebec (22.3%) and Ontario (42%). Treatments used in Manitoba included aeration, composting or drying. In Quebec and Ontario <3% of livestock farms used additives and <1% used constructed wetlands (filtering marshes). A small percent of farms used “other treatments” that may have included anaerobic digestion or solid-liquid separation (*FEMS, 2001*).

Current technologies available to Manitoba hog farmers (both regulated and recommended) as well as some proposed technologies will be briefly assessed for their effectiveness in reducing pathogen viability. Recall that storage and application can be effective in controlling the levels and distribution of pathogens if certain precautions are taken. But it may be more efficient to adopt technologies where sanitation can be monitored according to specified standards prior to manure application. Three different treatment methods for producing hygienically safe end

products are analysed: composting, aerobic and anaerobic digestion, and alkaline treatment. Keep in mind that unlike European jurisdictions, Manitoba has little to no sanitation or hygienic safety standards for treated manure intended to be used as fertiliser.

8.6.1.5.1 Composting (Aerobic digestion)

As long as composting meets conditions of specified time and temperatures for adequate disinfection which are defined by the Canadian Compost Council, composting can be a very effective means of reducing pathogens in manures. The process limits for composting are $\geq 55^{\circ}\text{C}$ for at least 3 days in piles or 15 days for the windrow (turning) process (CCC). Composting, however, is primarily a process used to handle solid manure and is not practical for liquid swine wastes that are typically in liquid form. Nonetheless, recently conducted studies on the composting of swine manure support the effectiveness of composting at elevated temperatures ($\geq 55^{\circ}\text{C}$) to reduce pathogenic bacteria (*Nicholson et al., 2005; Ros et al., 2006; Grewal et al., 2007*). Compared to liquid storage or low temperature manure packing at 25°C , high temperature composting destroys pathogens (*E. coli* O157:H7, *Salmonella* and *Campylobacter*) more rapidly, but may not be as suitable for eliminating *Listeria* which are more heat resistant (*Nicholson et al., 2005; Grewal et al., 2007*). Pathogen inactivation was reported as being significantly faster in liquid slurry plus wood shavings (within the first week) compared to that in the slurry solid fraction (after 2 weeks) at high temperatures (*Ros et al., 2006*). Quebec researchers proposed that self-heating high temperature aerobic digestion ($>50^{\circ}\text{C}$) or “liquid composting” is a valuable option for treating liquid manure because it produced organic fertiliser of comparable quality to solid compost in terms of its near absence of pathogens (*Juteau, 2006*).

8.6.1.5.2 Anaerobic Digestion

Anaerobic digestion is widely adopted for biogas production in European countries to remove excess nutrients and generate energy. Hygienisation (sterilisation at 133°C for 20 min or pasteurisation at 70°C for 60 min) prior to composting or anaerobic digestion in biogas plants has proven effective for sanitising animal manures and other organic wastes (≥ 5 log reduction of relevant indicator and pathogenic organisms) (*Angelidaki and Ellegaard, 2003; Bagge et al., 2005; Paavola et al., 2006*). As was the case with manure storages, most large-scale BGPs use a continuous process, which is less reliable than a batch system in terms of its effect on sanitation (*Bagge et al., 2005; Juteau, 2006*). A separate heat treatment step before anaerobic digestion is reliable, but digested residues can be recontaminated after the process (*Bagge et al. 2005*). If a pasteurisation step is not used, BGPs must rely on sanitation during the digestion process. However, spore-forming bacteria are not affected by pasteurisation at $<80^{\circ}\text{C}$, so some jurisdictions including Sweden recommend that digested residues from biogas plants only be used on arable land and not pasture (*Bagge et al., 2005*). Although Category 2 ABPs should be sterilized prior to anaerobic digestion, animal manures are exempt from pretreatment prior to anaerobic digestion in a biogas plant (*EC 1774/2002; Paavola et al., 2006*). This suggests that, like Canada, a similar level of hygiene for livestock manures is not required in other jurisdictions either. Therefore, the mesophilic digestion process ($20\text{-}40^{\circ}\text{C}$) recommended by Manitoba guidelines as an alternative for treating liquid hog manures (*MB-1*) might be sufficient for minimising the risk of pathogens, however more data are required here.

In general, mesophilic digestion at $35\text{-}38^{\circ}\text{C}$ for 20-25 days has been found to be unreliable for destroying *Salmonella*, while thermophilic anaerobic digestion ($>50^{\circ}\text{C}$ for 15 days) was much more effective in reducing indicator and pathogenic bacteria in biowastes (*Paavola et al., 2006*). In contrast, Quebec researchers reported that low temperature (psychrophilic) anaerobic digestion (PAD) was effective in eliminating pathogens including protozoa (*Cryptosporidia* and *Giardia*) in swine slurries of different initial microbiological and physicochemical properties (*Côté*

et al., 2006b). The PAD process at 20°C for 20 days in a sequencing batch reactor was confirmed and recommended as an effective process for reducing levels of naturally-occurring indicator and pathogenic populations of *E. coli* by 99.67-100% and *Salmonella*, *Cryptosporidium* and *Giardia* to undetectable levels in swine manure slurry (Côté *et al.*, 2006b). These processing conditions are consistent with Winnipeg's wastewater requirement of exposure at >20°C for 30 days (*Environ Act License*). These data suggest that alternative time/temperature treatment combinations can achieve equivalent pathogen reductions (>4 log units) for efficient sanitation of digested wastes. After assessing the effectiveness of the heat treatments prescribed by EC Regulation 1774/2002, the European Food Safety Authority concluded that the risk of pathogen spread must be taken into account only if the manure produced was to be marketed outside an "epidemiological geographic area"; otherwise, non-heat treated manure applied to land was considered to pose little problem to human and animal health (EFSA, 2005). In other work, of three patented Canadian and French technologies evaluated (*Biofertile*, *Biosor* and *Bioterre*), only the *Bioterre* system which used a single anaerobic digestion step had detectable *Cryptosporidium* levels at the end of the treatment (Pourcher *et al.*, 2006). The other two systems (*Biofertile*-solids separation by sieve and screw press, aerobic digestion and electrochemical treatment; *Biosor*-separation by flocculation/coagulation followed by biofiltration) performed well in terms of pathogen reductions. However, the presence of *Cryptosporidium* oocysts in the final *Bioterre* by-product indicates that all currently available commercial manure treatment technologies cannot achieve complete sanitation of manure intended for land application (Pourcher *et al.*, 2006).

8.6.2 Proposed technologies for manure treatment

8.6.2.1 Multi-stage systems

Proposed new technologies for treating swine waste are also based on a multi-step principle combining physical, biological and chemical processes to optimise performance in reducing pathogens. Of five systems identified as ESTs under the Smithfield agreement in North Carolina, the only one which treated both the liquid and solid fraction and involved solid-liquid separation, nitrification/denitrification and soluble phosphorus removal was evaluated by Vanotti *et al.* (2005, 2007). This method was found to reduce suspended solids, total N and P and odours by ≥98% (Vanotti *et al.*, 2005). Each step in the process was also found to consistently reduce the levels of pathogens and microbial indicators. Solid-liquid separation was not as effective on its own (only 0.5-1 log reduction) as the other two processes which reduced pathogen and indicator bacteria to undetectable levels (<0.3 log CFU/mL) (Vanotti *et al.*, 2005). Not only were ammonia emissions effectively managed by nitrification/denitrification, but a sanitised or disinfected material was produced after the alkaline treatment (Vanotti *et al.*, 2005, 2007). In addition, a Canadian "ATD" Manure Treatment System also uses solids separation plus nutrient and ammonia extraction by membrane filtration, followed by liquid sterilisation and drying to convert manures into an environmentally-friendly, dry pelleted fertilizer in a "closed-loop" system (ATD). Compared to anaerobic digestion or composting, ATD is claimed to produce solids and liquids that are pathogen-free and eliminates the need for lagoons or digesters.

8.6.2.2 Ammonia Treatment

In the 2001 Canadian farm survey, Manitoba livestock farmers reported not using any additives for treating stored manure (FEMS, 2001). It is of interest that a Swedish study provides evidence that alkaline ammonia treatment is effective as a disinfectant for cattle manures (Ottoson *et al.*, 2007). Specifically, treatment during storage at 14°C with 0.5% aqueous ammonia for 1 week or 2% urea for 2 weeks, as well as 2% urea for one month storage at 4°C were predicted to be able to achieve a 5 log reduction of *Salmonella* in manure (Ottoson *et al.*, 2007). Furthermore, it was suggested that the destruction of microorganisms in swine slurry

may be faster than reported for cattle manure because of the lower total solids content (<5%) in the former. Lower total solids reduces the buffering capacity and allows for a higher pH to be reached from the free ammonia exposure (the disinfecting agent). Compared to liming (Sasáková *et al.*, 2005; Maguire *et al.*, 2006), ammonia treatment requires very low concentrations to ensure adequate disinfection of stored manure. Treated manure can serve to recycle nitrogen as a high value fertiliser and can maintain its high pH (9.0) to prevent pathogens from recontaminating the treated product.

Manitoba could adopt this practice and substantially shorten the period of storage needed to eliminate pathogens. Furthermore, these recommended treatments may be well-suited to Manitoba farms without the need to adopt completely new systems and build treatment facilities. The temperatures used in the study by Ottoson *et al.* (2007) also correspond well to the temperatures experienced in commercial Manitoba hog farm manure storages (4°C and 14°C) reported by Arrus *et al.* (2006). It is important to note that pathogenic bacteria have been found to recontaminate manures during transportation (Bagge *et al.*, 2005). Because of its continuous alkaline influence, ammonia treatment may also be an important strategy for preventing recontamination and regrowth of inhibited pathogenic microorganisms when manure is transported. To control the increased ammonia emissions during this process, storage covers can be used.

8.7 Conclusions: pathogens and manure management

Currently a correlation between reduced levels of pathogens in manure applied to land and reduced incidences of food and waterborne disease cannot be established and requires further investigation. Most often the source of waterborne disease cannot be identified or the connection to livestock wastes is unclear. Therefore, it is difficult to draw conclusions about the effectiveness of manure management practices in preventing food or waterborne disease outbreaks as a result of manure-borne pathogens. There is no doubt, however, that even small reductions in the microbial numbers in manure can reduce the potential risk of food and water contamination by pathogens. Data from recently conducted studies suggest that current environmental protection measures taken during the storage, application and treatment of hog manure in Manitoba should reduce the environmental impacts of pathogens from hog operations. Of the regulated practices, long term storage, manure incorporation, application in the spring, soil crop cover and composting have each been shown to reduce pathogen numbers and the extent of their distribution. However, additional treatments are required to produce sanitised material.

Aside from improvements that can be made to current production practices (storage and application), no new evidence has been found to show that the BMPs currently being recommended are incapable of reducing the frequency of pathogen recycling. There are, however, some shortcomings in the policies used to enforce the implementation of these measures. For one, it is difficult to assess the effectiveness of practices in controlling manure-borne pathogens when neither the province nor Canada have defined pathogen reduction or hygiene standards for animal manures prior to land application. Findings from other jurisdictions indicate that pathogens can be more effectively controlled by technologies and processes that are not widely used in this province such as thermophilic anaerobic digestion. The U.S. and Europe, with much more intensive hog operations compared to Manitoba, have taken initiatives towards promoting and regulating the treatment of livestock manure to substantially reduce pathogens to safe levels for agricultural use. Situations in Europe have even prompted nationwide *Salmonella* control programs for monitoring and identifying herd infection levels which help determine the safest methods for handling infected animals and their outputs. So,

compared to other jurisdictions it appears that a lot more could be done to promote newer technologies to treat manure and define pathogen reduction standards for manure to be applied on farm land.

Of the alternative practices discussed, the most promising appears to be the use of natural additives such as urea or ammonia to eliminate pathogens from liquid hog manure. With other methods, a single system used on its own, may not be effective against all pathogens. It appears that higher levels of reduction are possible when multiple strategies are combined. Overall, managing manure on a “whole-farm” basis can decrease pathogen transfers from hog operations to land, water and food.

8.8 Recommendations: pathogens and manure management

1. The contribution made by the spreading of manure containing zoonotic pathogens on arable land to human illness in Canada is unknown, yet data from other countries shows the risk to human health is significant. Since most human gastroenteritis is caused by zoonotic pathogens and since fresh produce, once contaminated with manure cannot be freed of contamination, horticultural crops should not be treated with manure or irrigated with agricultural waste water which contain these pathogens. Large outbreaks of foodborne illness from contaminated produce have occurred in the U.S. can be avoided by following this recommendation.
2. Work should be undertaken to determine whether N/P limits governing manure application in Manitoba protect the environment, animals and humans from pathogen distribution and recycling.
3. Regular intermittent addition of fresh manure to storage reservoirs restores pathogen numbers to those characteristic of unstored manure. Each cell in a multi-cell lagoon reduces bacterial numbers by about 90%. Lagoons containing 3 or more cells in series should be examined in Manitoba for their ability to reduce zoonotic pathogens to safe levels.
4. Swedish work with cattle manure has shown that zoonotic bacterial and protozoan pathogens were eliminated from cattle manure by treatment with 0.5% ammonia or 2% urea at 4 to 14° C in less than a month. These agents should be evaluated in Manitoba using hog manure slurry for their ability to eliminate pathogens.
5. Hog manure slurry in Manitoba should be examined for veterinary drug residues including antibiotics and antibiotic resistant bacteria.

8.9. Antimicrobial resistance

8.9.1. Antimicrobial resistance within context of pathogen contamination and manure management.

The discussion below to a large extent falls within the context of pathogen contamination and manure management. In one sense the analysis of antibiotic resistance could have been incorporated into the text on pathogens but this has not been done for a number of reasons: These reasons include the following:

- A discussion of antibiotics must be placed within a framework which includes current clinical practice in human medicine.
- There are a number of regulations, both federal and provincial, that are of specific application to veterinary and human drugs that require a separate discussion.
- The role of animal husbandry practices has received a lot of blame for the rise in antibiotic resistance in human medicine and deserves a separate discussion to draw the points of interest out.
- Management options at government level for food pathogens and antibiotic resistance are not necessarily the same.

8.9.2 Background to antimicrobial resistance

In this section of the document we will discuss the potential link between antibiotics in livestock production and natural waters which included surface water and groundwater. Furthermore we will discuss the link between natural water and the likelihood of an increase in antibiotic resistance in human medicine. From a scientific point of view it is important to understand how a link between the two can be verified. The issues are not that simple. On the one hand there are those that believe that antibiotic usage in livestock, and particularly swine, is the major contributor to antibiotic resistance in human medicine. There are those who believe that the opposite is true and that it is difficult, or even impossible to point a finger at livestock. What we will attempt to do is look at the evidence and draw conclusions based on the strength of the evidence. We will then extrapolate, where appropriate to the Manitoba situation, and finally make recommendation as to where the gaps in knowledge lie.

The basis of the problem with antibiotic resistance is that in human medicine over the last 20 years there has been a dramatic increase in the resistance of pathogens to antibiotics (*Phillips et al. 2004*). This has significant implications for public health because it is becoming more difficult and more costly to treat infections. Just after the Second World War, at about the time that penicillin was first being used extensively, relatively low levels of penicillin could be used to treat a wide range of infections (*Bud, 2007*).

The use of penicillin had a dramatic influence on mortality rates of soldiers who very often would succumb to gangrene that developed in wounds. Even during the latter parts of the war it was noticed that penicillin was becoming less effective and that more had to be used to obtain a positive outcome. After the war as more extensive use was made of penicillin resistance rose quickly so that by the 1950's new antibiotics were being introduced into human clinical practice (*Bud, 2007*).

It should be noted that in the 1940's and 1950's antibiotics were not used extensively in animal production. This was primarily because of the high cost of antibiotics and it was only in the early 1950's that antibiotics were used therapeutically to treat some serious infections in animals. Thus at this stage resistance in human medicine could not have been blamed on animal agriculture because antibiotics were not yet been extensively used in animals (*Cromwell, 2002*).

As the years progressed the ability to produce antibiotics at low cost became possible. During the 1960's antibiotics were included in the feed or water of animals to treat frank infections on a herd basis. It was noticed that there was a growth performance response in young animals even though they were not ill, or likely even clinically ill. This then became the beginning of an era where antibiotics were included in feed and water of farm animals at sub-therapeutic levels as growth promotors (*Cromwell, 2002*).

There is still not clarity as to what the mode of action of antibiotics in growth. In Table 8.6 a summary of what is known about the growth promoting properties of antibiotics is outlined

Table 8.6. Potential physiological, nutritional and metabolic effects of growth promoting antibiotics

Physiological	Nutritional	Metabolic
----- <i>Increase</i> -----		
Nutrient absorption	Energy retention	Liver protein synthesis
Feed intake	Nitrogen retention	Gut alkaline phosphatase
	Vitamin absorption	
	Trace element absorption	
	Fatty acid absorption	
	Glucose absorption	
	Calcium absorption	
	Plasma nutrients	
----- <i>Decrease</i> -----		
Food transit time	Gut energy loss	Ammonia production
Gut wall diameter	Vitamin synthesis	Toxic amine production
Gut wall length		Aromatic phenols
Gut wall weight		Bile degradation products
Fecal moisture		Fatty acid oxidation
Mucosal cell turnover		Fecal fat excretion
		Gut microbial urease

Gaskins et al., 2002

(*Gaskins et al., 2002*). If one looks at the table carefully one will note that the modes of action attributed to antibiotics are non-specific. For example, the reduction in ammonia; there are many ways of reducing ammonia. Currently, research is lacking as to how antibiotics as growth promotors work (*Cromwell 2002; Gaskins et al., 2002*). This is important for at least two reasons. Firstly, compounds that can give growth promoting effects in swine are important. In Denmark, the removal of antibiotics as growth promotants has led to an increase in therapeutic use of antibiotics, the net result being that as much, or even more antibiotics are used in swine (*Fluit et al., 2006*). These antibiotics, because they are used therapeutically are in categories III and IV above. Secondly, when compounds such as copper sulfate have been used as an alternative, co-resistance becomes a problem. That is, the selective pressure put on microorganisms by copper sulfate at the same time selects for microorganism resistant to an antibiotic.

It was really only in the mid-1960's and early 1970's that antibiotics were cheap enough that they could be extensively use in swine rations, as well as other livestock, particularly poultry. During the 1980's when the problem with antibiotic resistance in human to antibiotics increased there was a concern that antibiotics in livestock production may be related to antibiotic

resistance in humans. But as noted before, resistance was already occurring in clinical medicine in the 1940's before antibiotics were even being used in animal feed. That is not to say that animal related antibiotic use should be carefully regulated, because there are obvious reasons for doing everything possible to ensure that drugs remain effective in clinical medicine. The big concern is that hog numbers in Manitoba have risen dramatically over the last 10-years to current levels of approximately 8 million (MPC, 2007). These hog operations produce manure, which is often spread on annual and perennial crop land, as a waste disposal method as well as a means of fertilizing the pasture to increase its productivity. The latter practice, for example, also has led to an increase in the backgrounding of cattle and improved revenues for the rural economy in Manitoba. As we will discuss below, the manure and waste water from hog operations either contains microorganism that are resistant to antibiotics which can transfer to humans, or the manure or wastewater contains antibiotics which get into the environment.

8.10 Ecology of antibiotic resistance

The link, or path, between antimicrobials between livestock and animals is not simple. The inset (Fig 8.2) shows a highly simplified set of associations between livestock and humans. Human medical practice is the major user of antimicrobials. Antimicrobials are partly consumed by patients or are waste (eg. flushed down the toilet). Relatively little is properly destroyed (that's why pharmacies have patient disposal programs) and a large proportion reaches municipal wastewater plants. Once the water is treated, it is released into rivers, lakes, and streams. The water that is released is not pure in the sense that there are no microorganisms in the water and that all chemicals have been removed. As such antimicrobial can reach the environment (Kolpin et al., 2002). In fact the amount released into the environment from urban and industrial sources may be substantial United States Geological Survey (USGA, 2007). One of the problems with agricultural uses of antibiotics is that there are suggestions that these compounds persist in the environment and may not be easily degradable in soil or water. The USGA and several European sources suggest that appropriate treatment of municipal water is in some instance lacking.

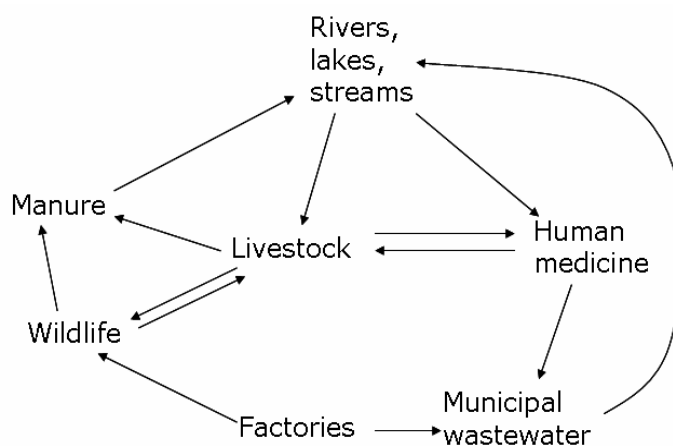


Figure 8.2. Potential flow of antibiotics, between animals, people, and the environment

On the livestock side, subtherapeutic antimicrobial can be included in animal feed, and residual antibiotics and/or microorganisms are released into wastewater. This wastewater is generally not treated and is released into the environment as manure, or in surface or groundwater. Bacteria that are resistant to microorganisms can reach humans via contamination of food products. Once again, food is not sterile and there are a variety of microorganisms that can reach foods consumed by people.

One of the dangers with contamination of human food with microorganisms from animals is that the types of organisms found in the animals digestive tract is typically the same as those found in the human digestive tract. So for example, the major causes of gastroenteritis in the province

of Manitoba are caused by *Salmonella* spp., *Escherichia coli*, *Shigella* spp., *Campylobacter* spp., *Giardia* spp., and *Cryptosporidium* spp., all of which occur in the animal digestive tract. This aspect has been extensively discussed in other sections.

Data availability on the amounts and types of organic compounds in Manitoba is not complete. A more comprehensive survey should be conducted to determine the sources of organic compounds in water particularly, but should not simply be restricted to agricultural waters by also to water sources in the Red River Valley. This survey should obviously include antibiotics.

8.11 Consumption of antibiotics in Manitoba

It is worth knowing how much antibiotics are consumed in Manitoba. Unfortunately very little information is available for Manitoba and it is therefore necessary to extrapolate from other studies. The Canadian Integrated Program for Antimicrobial Resistance Surveillance (CIPARS) in 2002 estimated that between April 2000 and March 2001 approximately 203,136 kg of antibiotics were dispensed through selected pharmacies for human consumption (CIPARS, 2002). This survey included those from Manitoba. This included all classes of antibiotics (Table 8.7). A study in British Columbia indicated that antibiotics dispensed through veterinarians and to feed mills amounted to approximately 46,000 kg in 1998. The antibiotics used were primarily tetracycline, virginiamycin, penicillin, bacitracin, and sulphonamide (Fraser *et al.*, 2004). Based on these figures one would assume that the volume of antibiotics in animals use exceeds that used in human medicine, but of course the BC study shows they are not category I or II antibiotics.

Similar studies have not been done in Manitoba, although antidotal evidence from Manitoba Pork Council suggests that the amount of sub-therapeutic antibiotics used in feed has been declining. However, no formal study has been done for the Manitoba Pork Industry. Generally speaking the antibiotics used in veterinary medicine and the feed industry are regarded as medium or low importance in the human medicine (CIPARS, 2002). In other words, Health Canada has mandated that antibiotics used in humans should in general not be used in animals but there is some overlap at the lower end of the scale with the penicillins and tetracyclines.

Table 8.7. Categories of antibiotics according to importance in human medicine

Category	Human importance	Type of antibiotic
I ^a	Very high	Fluoroquinolones, Glycopeptides, Carbapenems, 3rd and 4th Generation Cephalosporins, Streptogramins, Newer Generation Antimicrobial Drugs
II ^b	High	Penicillins Group 1 (Beta-lactamase resistant penicillins, extended spectrum penicillins), Aminoglycosides, Macrolides, Lincosamides,
III ^c	Moderate	1 st and 2 nd Generation Cephalosporins, Penicillins Group 2 (natural penicillins, aminopenicillins), Tetracycline, Sulphonamides
IV ^d	Low	Zinc Bacitracin, Polymyxin B, Colistin, Quinoxalines, Flavophospholipols, Ionophores

^aFor the treatment of life-threatening bacterial infections in humans. Potentially no alternative antimicrobials in case of emergence of resistance to these agents. Should not be used in feed but are used as veterinary therapeutics.

^bCan be used to treat human infections caused by bacteria that are resistant to category III antibiotics. Generally not used in animals and not included in feed.

^cUsually the first to be used to treat human infections. Also used to treat animals. Usually not used in feed but commonly used in as a veterinary therapeutic.

^dOf limited use in human medicine. Ionophores, are not ever used in human medicine.

Adapted from **CIPARS, 2002**; annotations are the authors and not CIPARS

8.12 Antibiotic resistance in the Province of Manitoba

The pattern of human related antibiotic prescription practices and resistance in Manitoba is very similar to that found in the rest of North America and Europe. In summary, since the 1990's antibiotic prescriptions indicate a decline in volume, and a reduction in resistance of human isolates for most, but not all antibiotics (*Kozyrskyj et al., 2004; Carrie et al., 2000*). This is substantially the same picture in animals except that it is not as well documented. In particular there seems to be an increase in macrolide use and likely resistance to this class of antibiotics (*Kozyrskyj et al., 2004; Carrie et al., 2000*).

One antibiotic and bacterial resistance pattern of particular concern is methicillin-resistant *Staphylococcus aureus*. This bacterium is by far the most important staphylococcal infection and is associated with a range of infections. *S. aureus* may be present as part of the normal flora of the skin, vagina, urinary tract, and digestive tract and results in infection when the epithelial barrier of any of these sites is breached, either mechanically, chemically, or by another infectious agent. Over the last several years in Manitoba there has been a rapid increase in methicillin resistance (*Larcombe et al., 2007*). This is not only the case in Manitoba but is a phenomenon seen in the rest of Canada, USA, and Europe. Both methicillins and macrolides are classified by Health Canada as category II, high importance antibiotics which should only be used in humans (Table 8.7) (*CIPARS, 2002*). Thus the argument could be made that the increase in methicillin and macrolide resistance may not be the result of animal use, but it cannot be excluded either.

8.13 More detailed analysis of resistance in Canada

It is beyond the scope of this document to provide detailed data on the resistance surveillance of all pathogens of human importance and we present data, where possible, for only *Escherichia coli*, *Shigella*, and *Salmonella* spp. As a note to the reader, *E. coli* and *Shigella* are often lumped together in survey data because taxonomically the difference is quite small. There

are a number of authorities who dispute whether *E. coli* and *Shigella* are in fact separate species. The most comprehensive data for Canada is from **CIPARS (2002)** but does not include data from Manitoba. Data from Alberta, Newfoundland and Labrador, Ontario, Prince Edward Island, and Saskatchewan only were included.

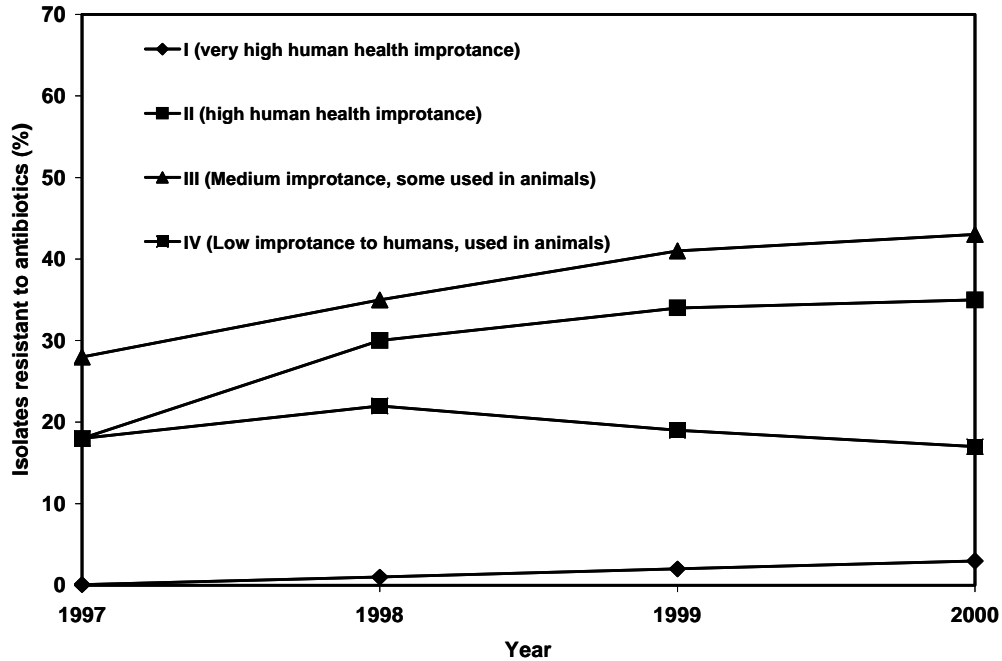


Figure 8.3. Antimicrobial resistance of human isolates of *Salmonella* in Canada (CIPARS 2002)

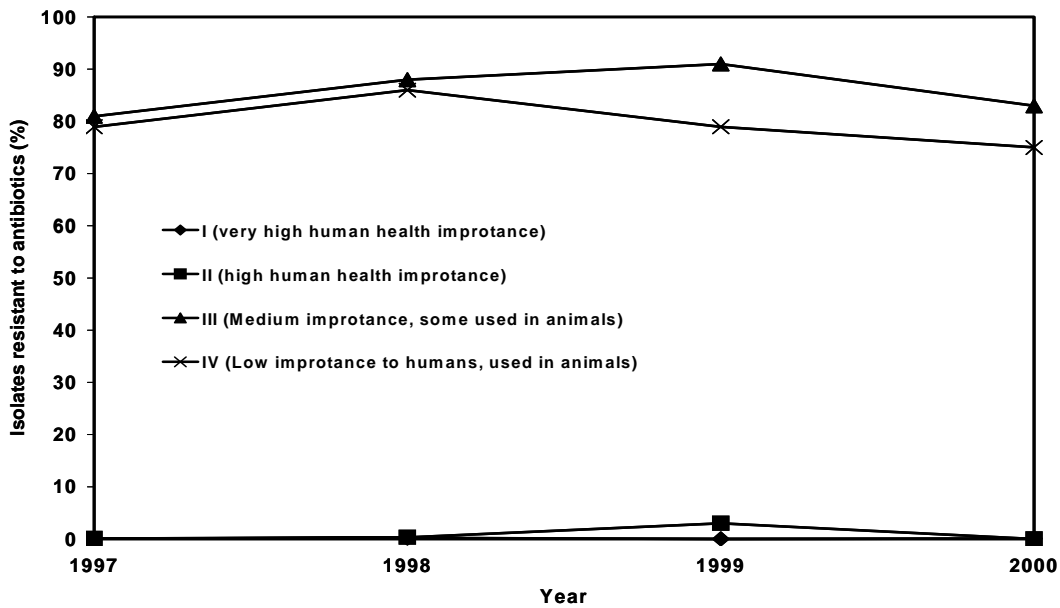


Figure 8.4 Antimicrobial resistance of human isolates of *Shigella* in Canada. (From CIPARS 2002)

In general there was a higher level of resistance to a broader range of antibiotics in *Salmonella*, than *Shigella*. In human isolates the level of resistance was below 4% for category I antibiotics which are of very high importance in human medicine. The flip side is that there was an increase in resistance in category II (high importance) and some category I antibiotics. There was a particular increase in aminoglycosides which include amikacin, gentamicin, kanamycin, neomycin, netilmicin, paromomycin, streptomycin, tobramycin and apramycin. The other antibiotics of concern are the folate pathway inhibitors. These antibiotics of concern are generally not used in animals.

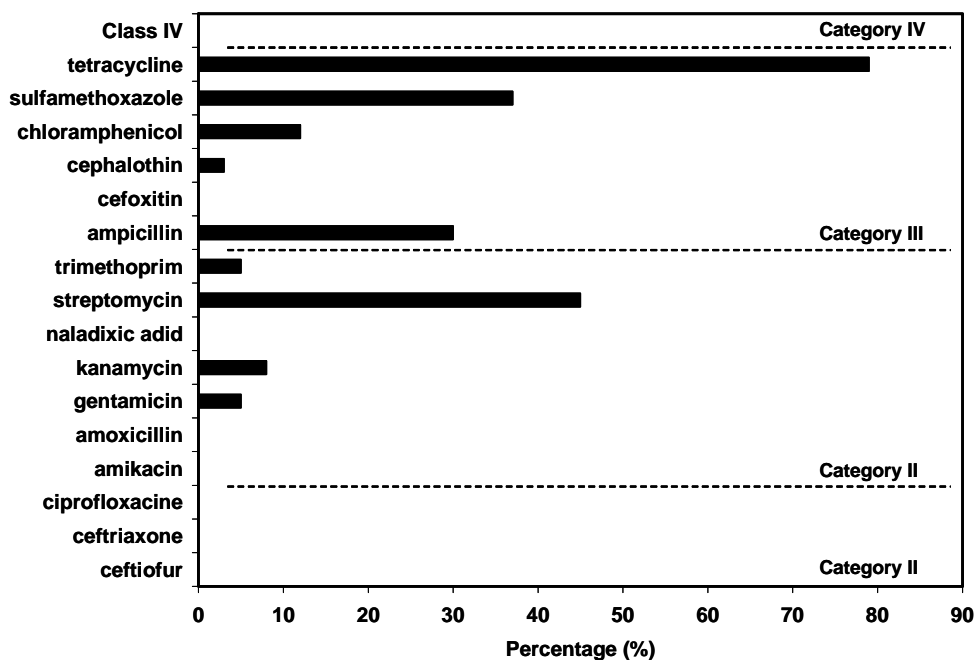


Figure 8.5. Prevalence of antibiotic resistance in *E. coli* obtained from cecal contents of pigs at abattoirs in 2002, From CIRARS 2002

When cecal *E. coli* from pigs slaughtered were assessed it was found that 21% of isolates were susceptible to all antibiotics tested. All isolates were susceptible to category I antibiotics. About 79% were resistant to tetracycline which is of medium importance (category III) to humans and is used in animals. There was quite high resistance to streptomycin which is used therapeutically in animals but not in feed.

When a similar survey was done on *Salmonella*, but from cecal isolates as well as those recovered from fresh pork a slightly different pattern emerged. About 55% of cecal isolates were susceptible to all antibiotics and in general there was more resistance in that with *E. coli* and in categories of antibiotics of more important to humans. There was also a tendency of higher resistance in fresh pork isolates. These fresh pork isolates could well be community acquired and potentially reflects resistance in the human population but not necessarily in animals.

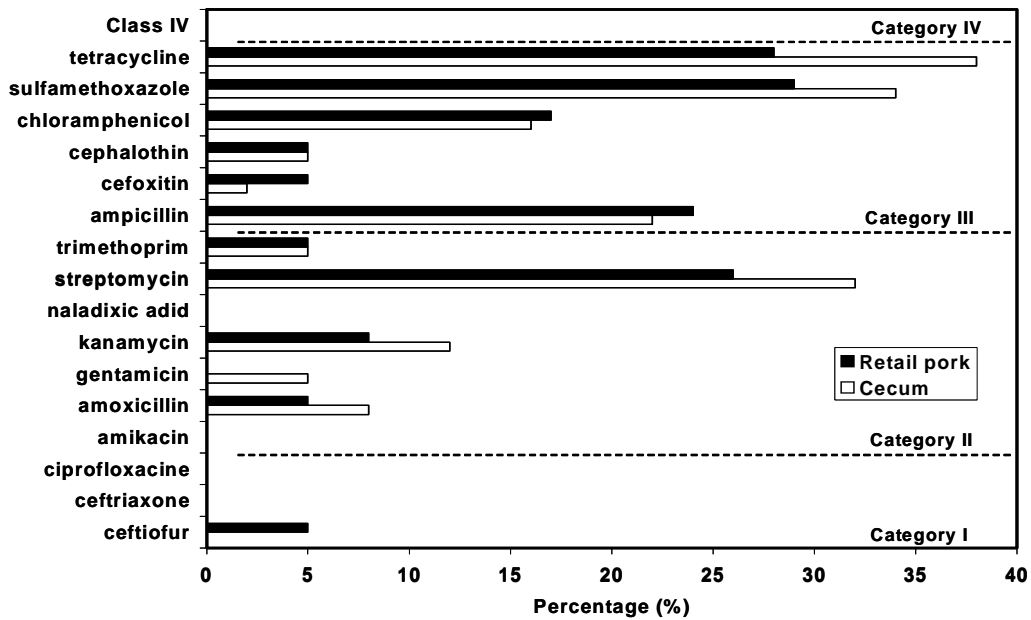


Figure 8.6. Prevalence of antibiotic resistance in *Salmonella* obtained from cecal contents of pigs at abattoirs and fresh pork in 2002, From CIRARS 2002

What this data shows us is that different bacterial strains have different levels of resistance and exhibit different patterns of susceptibility. Thus data from one set of bacteria cannot simply be extrapolated to other species of bacteria without validation studies being carried out. Also these data have limitations, the biggest being that it does not include Manitoba. Secondly, the numbers of isolates tested were not as numerous as it could be and recent longitudinal data was not used. Clearly an important need is Manitoba data in a variety of forms.

8.14 Rise of antimicrobial resistance from the pre – to post-antibiotic era

We have already discussed some of the issues in relation to observation just after the Second World War (*Budd, 2007*). The link between animal use of antimicrobials and human medicine is not clear (*Cromwell, 2002*). Data that link the two are based on associative studies. An increase in antimicrobial resistance in human pathogens has more or less happened at the same time that feed antibiotics have occurred in animal feed. This does not actually mean that the two are directly linked, and to make that link one would have had to conduct monitoring studies pre- and post – the antibiotic era. Of course this is not that easy.

Houndt and Ochman (2002) conducted a rather interesting study to try and answer this question. Some types of bacteria, like *E. coli* were studied for various reasons before the widespread use of antibiotics in the mid 1940s. Typically scientists will store bacteria they have newly isolated and preserve them at low temperatures. What *Houndt and Ochman (2002)* did was to go to a collection of *E. coli* that was archived before the 1940s and compare them to *E. coli* that was stored after the 1940s. By comparing the antibiotic resistance, before and after the 1940s, they thought would give an indication of the level of resistance.

All of the *E. coli* that were collected in the pre-antibiotic era were susceptible to high levels of antibiotics, but 20% of strains from more recently collected populations of *E. coli* displayed high-level resistance to at least one of the antibiotics. In addition to the increase in the frequency of high-level resistance, background levels, conferred by genes providing nonspecific low-level

resistance to multiple antibiotics, were significantly higher among contemporary strains. A point to clarify here is that they were looking at high levels of resistance. This does not mean there was no resistance in the pre-antibiotic era, but rather that high level resistance was not there. So from this study one would conclude that antibiotics have had an impact on resistance in the community. As we have noted, the major driving force in making bacteria more resistant has not clearly been defined; is it agriculture, is it pharmaceuticals, or is it industrial?

Bacteria develop resistance to antibiotics very quickly (*Phillips et al., 2004*). Tetracycline resistant bacteria occurred in 1956, four years after its introduction to clinical use and only eight years after discovery. The time between introducing an antibiotic into the clinic and the occurrence of resistant bacteria was 15 years for vancomycin, 4 years for nalidixic acid, 3 years for gentamicin, 3 years for fluoroquinolones, one year for erythromycin, and less than one year for streptomycin (*Harbottle et al., 2004*). Although the latent period between the introduction of an antimicrobial and the emergence of resistance may vary, once the prevalence of resistance in a population reaches a certain level, reversal of the problem may be extremely difficult (*Swartz 2002*). For example, fluoroquinolone-resistant *Campylobacter* have persisted even though the antibiotic has been withdrawn (*Price et al., 2007*).

The movement of bacteria that carry antibiotic resistant microorganism between livestock and off-farm is complex. In an older study Herriott et al., (1996) tested twelve herds as well as feed, water, etc and wild animals in Washington State for the presence of *E. coli* O157:H7. They could find *E. coli* O157:H7-positive cattle in all herds at around 1 – 5 % as well as in water troughs, in horses, dogs, bird manure, and even in flies. The water, flies, and bird are all vectors for the transfer of antimicrobial resistant bacteria.

Much of the evidence for the transfer of antibiotic resistance comes from data on *E. coli* and *Salmonella*, which as we have noted above cannot be used to extrapolate to other situations and other bacteria. There is even doubt that one can extrapolate to other geographical regions. For example the population density and co-proximity between human and animal population in Manitoba and Western Europe are very different. This is actually one of the problems. The most comprehensive datasets on antibiotic resistance are from Europe and particularly the Danish government and these data are often extrapolated to Canada. Can we do that? Can we do that in Manitoba, which has a different population density, has different animal production systems, different weather, and different water distribution systems as well as soil? Frankly, we don't know.

Furthermore *E. coli* from hogs, cattle, soil and groundwater are not generally the same ones that cause disease in humans with exception of serotypes of bacteria like *E. coli* O157:H7. The epidemiology and transfer of disease causing bacteria is highly complex. In our own (Krause et al., 2007; unpublished) studies in La Broquerie, Manitoba, we found that *E. coli* resistant to various antibiotics isolated from hog manure did not survive for more that 6 weeks on soil, and when cattle grazed the pastures spread with the manure they did not pick up the *E. coli* that was in the manure. The La Broquerie study provides some of the first data directly relevant to Manitoba but should be considered as preliminary data.

In our study of the groundwater at La Broquerie we enriched groundwater samples take from the site on a range of antibiotics. Once we had the enrichment we then characterized the bacteria cultured using broad phylogenetic analysis using molecular based techniques and assigned isolated to groups at a phylum level. What we found was quite interesting. In the biosphere there are 27 official phyla, and 27 candidate phyla, but the important point is that only 8 of those phyla are typically found in the digestive tract of animals. We could therefore use the molecular signature of the bacteria in the enrichment with antibiotics to determine if they came

from a digestive tract or not. What we found was that only 10-15% of the bacteria that grew on the antibiotic enrichment likely came from the digestive tract of animals and that the majority of antibiotic resistance bacteria in the culture came from non-digestive tract sources.

At first this may seem surprising, but on reflection it is not. Most of the antibiotics we use today come from the *Actinomyces* and *Streptomyces*, which are both soil microorganisms. If one looks at the bacteria that live in soil there are a huge number of other species of bacteria there. These bacteria have learnt to live in the presence of the antibiotics produced by *Actinomyces* and *Streptomyces*. Thus, antibiotic resistance in the environment should not be surprising. The question we still need to ask at the La Broquerie site is how the numbers, not the presence or absence, of antibiotic resistant bacteria change in the presence of manure.

8.15 Ecology of antibiotic resistance

There is virtually no data available from Manitoba on the topic of the ecology of resistance in the swine sector and the little that is available comes mainly from our laboratory as described above. There is an associative link between animal use of an antibiotic and resistance in human medicine but there are many different factors that affect resistance and it is not a simple cause and effect situation. When antibiotics are used in animal feed, in pet foods, or as therapeutics resistant bacteria are likely to be selected for in both pathogenic bacteria as well as normal flora. As a consequence these resistant bacteria will increase in frequency, but again there are also other factors.

Virginiamycin is a streptogramin antibiotic and is classified by Health Canada as a category I antimicrobial. It is still used in the USA as a growth promotor and there is significant streptogramin resistance in animal *Enterococcus faecium* (referenced in Phillips et al., 2004). On the other hand, avoparcin is a glycopeptide antibiotic, was banned because of the concern about vancomycin resistance in human gram-positive infections, and resistance to vancomycin is low in animal enterococci ((referenced in Phillips et al., 2004)).

One of two things can happen when antibiotics are removed. Firstly, the prevalence of the organism diminishes because the selective pressure provided by that antibiotic is no longer there and the organism disappears. Or, the resistant organisms declines, is replaced by other bacteria, but persists at low levels retaining residual resistance because of co-selection by another compound. There is evidence that both these processes occur. In Denmark, and most of the rest of the EU, avoparcin (and thus vancomycin) was banned in 1995 as a growth promotor and resistance pattern decline for vancomycin in human isolates. However, the same pattern does not always occur when other antibiotics are removed from animal use. From 1994 to 2001 the volume of antibiotics (not just avoparcin) used in feed has declined from 116 tonnes in 1994 to only 0.01 tonnes in 2001 (referenced in Phillips et al., 2004). Over that same period the use of antibiotics as therapeutics has remained at around 80 to 94 tonnes and there was an increase in non-gut infections particularly respiratory infections. So the problem was not actually solved, it was just relocated!

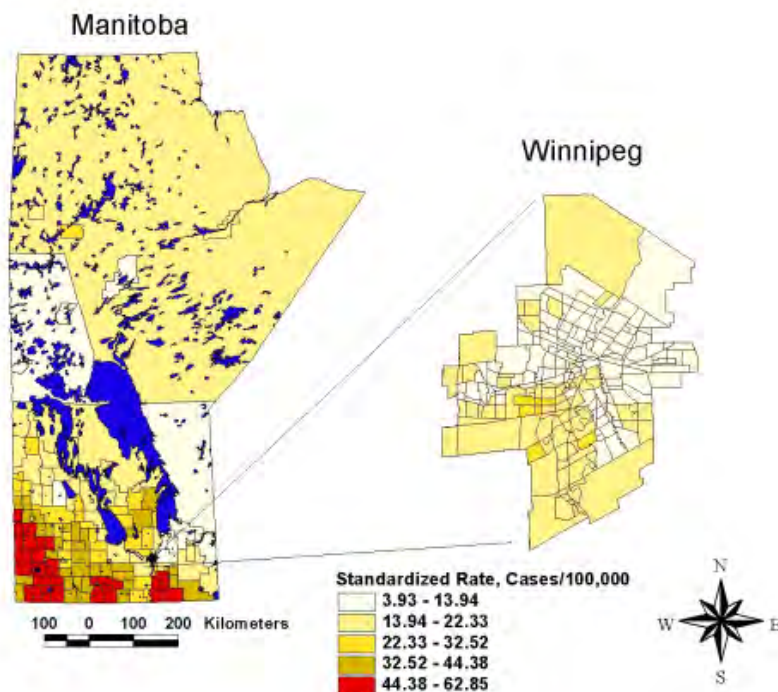
What is interesting about the data is that there was an initial decline in therapeutic uses in the range of 50%, but it has slowly crept back up to around 90 tonnes. This is probably an ecological phenomenon in which resistant strains were replaced by susceptible strains but as adaptation to various conditions has occurred resistance and thus therapeutic application has increased. For example, as the use of antibiotics has declined as a feed supplement the use of alternative non-antibiotic alternatives has increased. One of these has been copper sulphate. There is now evidence that copper sulphate selects for strains that are also resistant to

antibiotics. This is called co-selection. What happens is that the molecular machinery in the DNA of a bacterium that allows it to be resistant to an antibiotic is quite often in the same location, or carried on the same piece of DNA, as the molecular machinery that allows the bacterium to be resistant to some other compound. So when selection for one compound happens, inadvertently selection for another compound occurs.

Keeping in mind that avoparcin was removed from feed animal inclusion guidelines in 1995 there might be an expectation that vancomycin resistance (same as avoparcin) would decline. In Europe human isolates have stayed more or less the same, but in broiler feces and broiler meat the decline has been significant. So one sees a direct link between avoparcin in feed and resistance in animal isolates, but it is not as obvious in human clinical isolates where the decline in resistance is not nearly as pronounced and depending on how one looks at the data one might be inclined to say there is no difference (*referenced in Phillips et al., 2004*).

Virginiamycin is a streptogramin and when it was removed as a feed additive there was little decline in *Enterococcus faecium* resistance in broiler feces, a large decline in broiler meat isolates, a decline in resistance in human feces, but surprisingly an increase in streptogramin resistance in human clinical isolate resistance. The persistence of virginiamycin resistance in human clinical isolates has been explained by the use of other streptogramin antibiotics like pristinamycin and quinupristin/dalfopristin.

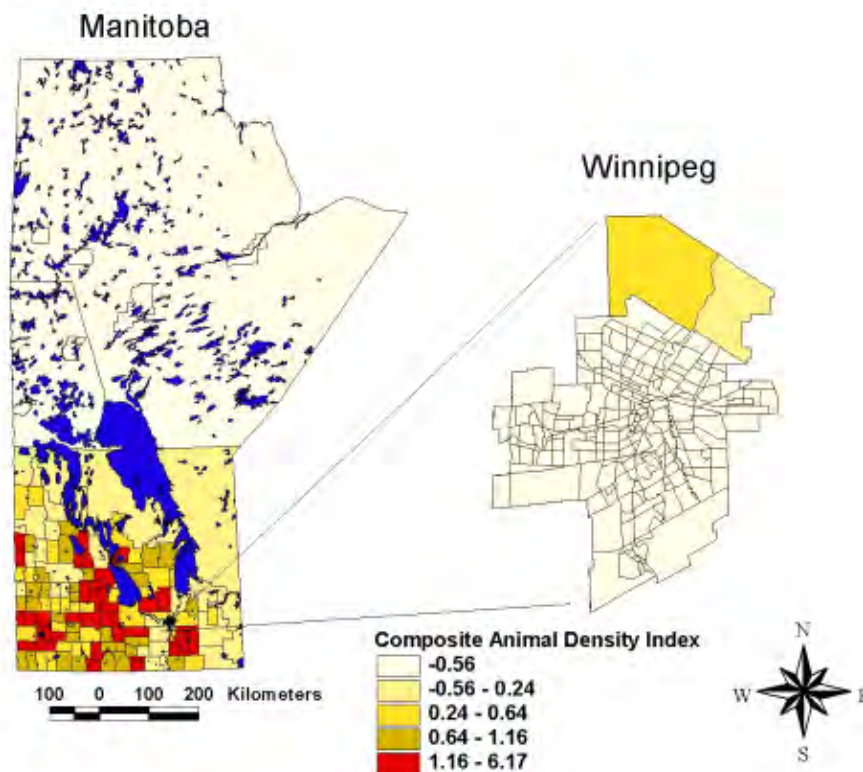
8.16 Distribution of hog operations and infection in Manitoba



Campylobacter incidence map, Manitoba, 1996–2004 (Green et al., 2005)

The location of livestock operations in close proximity to human populations is often used as an argument for explaining possible transfer of pathogens to humans. There is relatively little data on this in Manitoba, but a 2004 we (Denis Krause) set up a collaboration with Dr Chris Green (Public Health Branch, Manitoba Health) and Dr. John Wylie (Cadham Provincial Laboratory, Manitoba Health) to try and address these issues (*Green et al., 2006*). We asked whether there was any correlation between incidence of campylobacteriosis in humans and the location of animal population in Manitoba.

We used data derived from the Manitoba Health Public Health Branch communicable disease surveillance database, and applied spatial and ecological techniques to model campylobacter incidence for the years 1996 to 2004. There was a statistically significant geographic variability in the rates of *Campylobacter* incidence in Manitoba. Observation of the data show that in general there was a higher incidence of the disease in rural areas. However, if one looks at the incidence maps for Winnipeg, the highest animal density is the north end of the city but there is significant disease in the south end where there are few animals. Also, the incidence of disease is fairly high in Northern Manitoba, which has a very low animal density. We concluded that there were potentially three distinct mechanisms for the transmission of *Campylobacter* in Manitoba which all happen at the same time. One was extensive population exposure to a centralized food system which was infected with the *Campylobacter* organism, exposure to local



Animal density, Manitoba,
1996–2004 (Green et al., 2005)

factors like farm animals or contaminated water, and exposure to *Campylobacter* infection through foreign travel. Although this study obviously did not assess antibiotic resistance it

provides some idea of geographical distribution of potentially antibiotic resistant bacteria in the province. One issue we raised is that distribution of bacteria, and thus the ecological etiology of the disease needs to be more closely examined. Water sources are one of these factors and often thought of as the most important vector for disease transmission.

8.17 Antibiotics in the environment

One of the obvious sources of antibiotics in the food system is via contamination of waterways, rivers, and groundwater that is used as source of drinking water. A comprehensive survey was conducted by US Geological Survey (USGS, 2007). The primary objective of this study was to provide a US survey of 95 organic contaminants found in water bodies across the United States. These OWCs are associated with human, industrial, and agricultural water streams and include antibiotics, and several other compounds. The compounds tested were selected because they had the characteristics that enabled them to enter the environment in significant quantities.

A total of 139 streams in 30 states during 1999 and 2000 were sampled. The most common in the water were coprostanol (fecal steroid), cholesterol (plant and animal steroid), N,N-diethyltoluamide (insect repellent), caffeine (stimulant), triclosan (antimicrobial disinfectant), tri(2-chloroethyl)phosphate (fire retardant), and 4-nonylphenol (nonionic detergent metabolite). In particular antibiotics were detected in these waterbodies and included compounds that would fit into categories I through IV. Of concern is that category I antibiotics like fluoroquinolones were found in the water.

In general there were a large variety of compounds many of which have antibacterial activity (eg. phenol and triclosan). Put another way, there are significant quantities of non-antibiotic antimicrobials going into the water and this includes agricultural, industrial and urban waste water sources. Based on our discussion earlier it should be noted that antimicrobial resistance can be selected by one set of compound that results in co-selection of antibiotic compounds.

8.18 Antibiotic risk assessments

The most powerful way of determining the affect of antibiotics in animal agriculture is by conducting a quantitative risk assessment (QRA). This procedure gives a statistical probability that a particular event will occur. In statistical terms, we are asking what the probability is of an antibiotic creating resistance in an animal bacterium X (multiplied by) the probability that this bacterium will be transferred to humans X the probability that this bacterium will cause disease or transfer its resistance to another bacterium. An added uncertainty is that the bacteria and the mechanisms of human isolate resistance development are identical. How do we differentiate the two? Is it from animals, or is it from human?

For many pathogens the human use of antibiotics is clearly implicated. For example, the problem with methicillin resistant *Staphylococcus aureus* was created by the successive use of different classes of antibiotics. As *S. aureus* became resistant to one class of antibiotic, we moved on to the next one, and then the next one, until we got to the stage where we had painted ourselves into a corner, where there is not much left. The same can be said for penicillin- and macrolide-resistant *Streptococcus pneumoniae*, and macrolide-resistant *Streptococcus pyogenes*. It can be shown that normal oral streptococci have plenty of resistant genes that can be transferred to other bacteria and probably accumulate when oral antibiotics are administered (referenced in Phillips et al., 2004).

The converse is also true. In the USA, where virginiamycin is extensively used in feed, it has clearly led to resistance to streptogramins in animal *Enterococcus faecium* strains (referenced in Phillips et al., 2004). But where avoparcin has not been used resistance to glycopeptides in animal enterococci is minimal. Resistance declines when antibiotic use is stopped or decreased and individual strains are often replaced by susceptible strains when the selective pressure is removed. About 75% of *E. faecium* isolates from broiler chickens in Denmark were resistant to avoparcin (and consequently vancomycin) and about 65% resistant to virginiamycin (and also quinupristin–dalfopristin). About 75% were resistant to avilamycin which is not used in human medicine. In 2000, (post-growth-promoter ban), resistance rates were less than 5% for avoparcin and avilamycin, but stayed at about 30% for virginiamycin (referenced in Phillips et al., 2004).

Risk assessment involves a statistical probability analysis made up of (a) hazard identification, (b) exposure assessment, (c) exposure–response modelling, (d) risk characterization, and (e) uncertainty characterization. As far as we are aware no risk assessment has identified a distinct clinically pathway for antibiotic-resistant bacteria from poultry (or other animals) resulting in poor human health outcomes. The hazard identification step therefore has not been met for fluoroquinolone-resistant campylobacter or for streptogramin-resistant *E. faecium* in poultry, for which the most data are available (referenced in Phillips et al., 2004).

Travers and Barza have calculated that fluoroquinolone resistance, possibly acquired from antibiotics in food animals, leads to >400,000 excess days of diarrhea in the United States per year than if all of the isolates were susceptible. In contrast Cox & Popken (referenced in Phillips et al., 2004) have calculated that banning virginiamycin would prevent no more than 0.3 statistical mortalities in the entire US population over a 5 year period. The impacts on human health outcomes from fluoroquinolone-resistant campylobacters coming from cattle suggests that of 16,000 individuals who could be infected from ground beef, only 150 might be hospitalized and four might die. Clearly, the data analysis from different authorities does not agree and there is still considerable disagreement between authors.

8.19 Conclusion: antibiotic resistance

1. From the data reviewed it is difficult to conclude that antibiotic use in animals has a major effect on human health. At the same time it is not possible to conclude that it does not. The truth is probably somewhere in the middle; in some circumstance antibiotic use in animals can have a major impact and it likely depends on the type of bacterium, the production system, and the type of antibiotic.
2. Having said this Manitoba is clearly lacking in the kinds of information that makes it possible to assert that everything a reasonable person would expect to ensure reasonable safety is been done.
3. Surveillance of antibiotic resistance is routine for human clinical isolates and is primarily conducted at the Health Sciences Centre under the auspices of Manitoba Health, and by the provincial Cadam Laboratories.
4. There is however no routine surveillance of clinical strains from veterinary infections. Additionally, we do not know the resistance profile of fecal or carcass isolates of common food pathogens in the Province of Manitoba.
5. A survey should be conducted to determine which, and what level of feed antibiotics is in fact used on hog operations in Manitoba. Additionally, non-clinical isolates of gastrointestinal strains related to gastroenteritis should be collected and tested from farms and their local environment on a random basis to determine how resistance profiles change over time.
6. Given that water can be a major vector of antibiotic resistance organism major water sources, surface and ground water needs to be monitored to assess the level of

pharmaceutical organics present. In addition municipal sources of antimicrobial for the larger cities like Brandon and Winnipeg should be assessed.

8.20 Recommendations: antibiotic resistance

1. Specific information is required about which antimicrobials, and the volume of each antibiotic used and for which class of animal and for how long in the hog industry. It is not currently possible to compare the volume and class of antibiotic used in the swine versus hospitals/humans in Manitoba.
2. Microbial contamination of water and their resistance profiles have as far as we know only been done at the La Broquerie site. Other sites need to be considered.
3. The dynamics of microbial contamination in water sources needs to be done at the catchment level. This is because water flows are at the catchment level and from these sorts of studies a proper understanding can be developed about specific point in each catchment that can be measured. In general terms this means that water flows to the lowest point in a catchment and by measuring those point it should be possible to obtain a fairly good idea of how the catchment is been managed for water quality.
4. Within these catchments specific indicator organisms should be measured quantitatively and linked to their number in manure etc. From this data dynamic predictive models can be developed.

8.21 References

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CHAPTER 9 Integrating People, Economics and Science

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9.1 Developing policy to meet the long term needs of all stakeholders

After reviewing the previous chapters, it is clear that the interaction of the pig industry and other stakeholders is complex and only partially understood at the present time. The sector is important in terms of income generated and the positive connections to further processing and the large feed grain sector in Manitoba. The potential for negative impacts on public health, on water users and even rural residents are also important if not fully understood at this time. Successful policy will necessarily balance the needs of all stakeholders. This chapter will review some of the major stakeholders who would be interested in any policy regarding the sector and some of their likely concerns.

9.1.1 Stakeholders

9.1.1.1 Pig industry

The Manitoba pig industry is a significant economic force. Some 9 million pigs are produced for sale each year in Manitoba and pigs are now the single largest contributor to farm income in the province. Farms and further processing in Brandon and Winnipeg contribute significantly to Manitoba's employment.

9.1.1.2 Grain industry/Rural Residents

The grain industry in Manitoba benefits from the nutrients supplied from manure as inputs into their production process and hog feeders are important customers of feed grain growers. In fact, there is evidence that the livestock sector needs even more feed grain than currently supplied by Manitoba's grain farmers. Thus most farmers, grain and hog and other livestock producers, are affected by the growth of the hog sector.

Most rural communities in Manitoba rely on agriculture's contribution to local incomes. These means rural residents and municipal governments also are key stakeholders in regulation of this industry. In addition to importance of income from agriculture, some studies suggest rural residential property may be affected by proximity to intensive livestock operations (Ready and Abdalla, 2005). In circumstances where nuisance odour is driving a downward effect on property value, it may be possible for rural residents to offer incentives to pig producers to take mitigating actions like those suggested in section 4.6 of this report.

A study done in Manitoba (Royal LePage 2004) actually found positive correlations between rural residential markets and their proximity to intensive livestock operations. It is not clear if improved wages from this sector may be driving up housing prices or some other factor that improves land values in these regions may have been captured in the model.

9.1.1.3 Fishing/Tourism

Fishing and tourism are two important industries in Manitoba, and their reliance on the environment, suggests potential risk of suffering losses in revenues due to any environmental contamination. In 2002, the Manitoba commercial fishing industry ranked first in Canada for landing the most fresh water fish (15,555 tonnes or 39% of the total) and in landed value, as the landed fish were calculated to be worth over \$28 million CDN (49% of the total national value) (Stats Canada, 2002). In terms of the industry's export volume and value, Manitoba was second only to Ontario, selling 6,805 tonnes of fish worth almost \$50 million CDN (Fisheries and Oceans, 2002).

Tourism in Manitoba accounted for \$1.39 billion CDN in 2003, which comprised almost 4% of the provinces total gross domestic product with direct tourism receipts totalling \$858.1 million CDN (Travel Manitoba, 2003). Of the \$858.1 million CDN directly spent by tourists, fishing expenditures accounted for over \$97 million CDN, adventure activities (camping, boating, wildlife viewing, etc) contributed close to \$190 million CDN.

9.1.1.4 Healthcare

The health effects of toxic level of pollutants in the air or water or even nuisance levels that evoke psychological effects can lead to reductions in available labour due to sick days and increased health care costs to individuals. Estimates of costs associated with gastrointestinal illness each year exceed \$3.7 billion, but specific links between the pig sector and any such illness have not been established (or refuted). The potential costs require the inclusion of the health care sector as a stakeholder. Some studies, like the US study by the Environmental Protection Agency (EPA 2000) mentioned in the next section, have estimated the costs of contamination, but more evidence is required to verify any direct links between the management of manure and any illness in Manitoba.

9.1.2 Costs of Compliance and Social Cost-Benefit Analysis

Initial estimates of the cost of complying with new phosphorus regulation range from \$0.15 to \$3.50 per marketed hog depending on the availability of nearby land for spreading (Salvano et al., 2006). The higher cost estimates assumed manure would need to be processed using aerobic technology. Across the province total compliance costs were estimated between \$18 and \$28 million depending on the long run application rate being one or two times the plant removal rate (Mann and Grant, 2006).

In 2002 the Environmental Protection Agency (EPA) updated and revised the National Pollutant Discharge Elimination System (NPDES) regulations and the effluent limitations guidelines to maintain and improve upon the safeguards related to confined animal feeding operations (CAFOs) harming water quality (EPA, 2002). The justification for the rule changes developed by the EPA came from their economic analysis of the estimated annual compliance costs and the economic impacts that could occur if manure management was focused more on phosphorus levels than nitrogen (EPA, 2002). Compliance to proposed changes related to phosphorus discharge was estimated to have a total aggregate incremental cost to hog CAFOs of \$34.8 million US. Of that amount, large CAFOs (>1000 animal units) would pay \$24.9 million US, medium CAFOs

(300 – 1000 animal units) would face \$9.5 million in costs and the remaining \$0.4 million US would be paid by designated CAFOs.

Metcalfe (2001) notes that with increases in the stringency of environmental regulations there may be corresponding increases in compliance costs for producers. As a result, these increased compliance costs will lower the competitiveness of firms within the state, causing the state level of production to decrease as firms exit the state. However, the effect of the increased compliance costs may not be large enough to be economically significant and affect production level. To determine if regulation stringency affected production, Metcalfe (2001) employed a profit maximization model for hog production, while accounting for the endogeneity of pig inventory and the level of environmental regulations. Metcalfe was worried that most regulation occurred after inventory numbers had already peaked due to other factors. His results found that increasing stringency had a negative effect on the production of small farms, suggesting that increasing regulations would play a role in decreasing production on small farms. However, Metcalfe also found that there was no evidence that increased regulation lowered production from large operations. This would then imply two things. The first being that other factors, such as geography or other economic factors are important in influencing production levels; and that increased regulation hurts small operations more than the larger operations.

In terms of the total market and non-market impact, the EPA conducted an extensive cost-benefit analysis. Their study showed that the total social costs associated with the regulatory changes, with respect to phosphorus, would equal \$289 million US. The total benefits, including some estimates of reduced illness, were estimated to be between \$204.1 million and \$340.2 million US. This estimate did not include a number of potential social benefits, such as reduced eutrophication, as they were not monetized in the study. This analysis by the EPA and its findings strictly pertains to the U.S., but it does give an indication of what kind of benefits and costs should be examined in Manitoba.

9.1.3 Non-Market Costs of Pollutants

Wohl (1996) correctly points out that clean drinking water, clean air, and improved health are commodities that are not bought or sold in a proper market; however these goods influence market choices. These non-market goods have an economic value that is considered by consumers. In this instance, alternative means for monetizing value of these goods must be employed. Wohl notes several acceptable methods; including contingent valuation¹, adverting expenditures², and to estimate the lost revenues caused by pollution or damages to a particular resource. Another accepted means of pricing non-market goods, not mentioned by Wohl is hedonic estimation³.

¹ Field and Olewiler (2005) define contingent valuation as “Inferring a person’s willingness to pay for an improvement in environmental quality by presenting that person with a hypothetical scenario that asks the person to state their willingness to pay (or willingness to accept compensation).” (pg. 427)

² Wohl (2006) states that adverting expenditures are “...to observe how people spend to avoid exposure to contamination. This gives some indication of how much the amenity is worth to people.”

³Field and Olewiler (2005) define hedonic estimation as “An empirical technique that decomposes the prices of a market good into a number of different characteristics of the good. This allows one to estimate the demand for a specific characteristic (holding other characteristics constant).” (pg. 431) A common

Several studies surveyed by Wohl estimate the value of clean drinking water through willingness to pay and advertising expenditure evaluations. Athwal (1994) found that individuals in Abbotsford, BC would be willing to pay \$0.5 - \$0.6 million CDN/year (\$81 - \$97 CDN/year/household) to obtain higher quality water, whereas actual advertising expenditures were measured to be \$143 CDN/year/household. Hauser, van Kooten, and Cain (1994) investigated the Central Fraser Valley region in British Columbia and estimated that the area's willingness-to-pay for clean water was in the range of \$78 - \$284 CDN/year/household. This is contrasted by their finding that the region's advertising expenditures, based on bottled water purchases, was around \$0.4 million CDN/year. Abdalla (1992) calculated advertising expenditures, by residents in the borough of Perkasio in Pennsylvania, to avoid damages from polluted water. It was found that they ranged from \$60,000 - \$130,000 US or \$22 - \$47 US/household (\$71,000 - \$154,000 CDN or \$26 - \$56 CDN/household). Finally, Hanley (1989) estimated how a change in water policy in Eastern England that would guarantee that water supplies for that area would not exceed specified limits. This study determined that the average willingness-to-pay was in the neighbourhood of £13/year/household or \$25 CDN/year/household.

With respect to clean air, contingent valuation studies have also been conducted. Léger (1999) studied individuals' willingness to pay for a 50% reduction in ozone levels in Montreal, PQ through the Quebec Health Survey. It was found that, based only on the opportunity cost of the time spent consuming medical care, individuals on average would be willing to pay \$1.56 CDN/year for the improvement in air quality. When physical limitations and work leave are added to the second model, it was determined that the average willingness to pay was \$29.16 CDN/year. Léger (1999) surmised that if private opportunity costs of time, pain and suffering, medical expenses, reduced mortality and savings to the health care system are accounted for, the average willingness to pay would certainly exceed his estimate of \$29.16 CDN/year.

Gerking and Stanley (1986) also estimated individual willingness to pay for reduced ozone levels, basing their study in the city of St. Louis, MO. Their study found that the estimated willingness to pay ranged from \$18.45 to \$24.28 US/year when accounting only for the costs associated with medical care. Another study, by Tyndall and Colletti (2006), looked at willingness to pay for pork products that originated from farms that emphasized the use of shelterbelts to mitigate odor. Surveyed results from the states of Iowa, North Carolina and Washington indicated consumers would be willing to pay, on average, \$0.14 US/pound of pork. Interestingly, producers from the three States revealed that, on average, their willingness to pay for planting and maintaining shelterbelts was \$0.14 US/hog produced.

It is important to note, however, that there is an important critique of contingent valuation. As Field and Olewiler (2005) point out, there several issues surrounding this method that could very easily introduce bias in the results. These include how the survey is conducted/structured, small sample sizes, self-selection problems and the most crucial, being that there are incentives for those being surveyed to overstate their true willingness to pay. These incentives arise from the possibility that by overstating one's willingness to pay, and hoping others do the same knowing that their share of the

example is the value of a house with a view of the mountains. A researcher can measure the value of houses with a view and house without the view but similar in all other respects. The difference is the estimated value of the view even though there is no actual exchange or market for the view.

cost is relatively small, the item or policy will come to fruition. As such, contingent valuation studies should be taken with a large grain of salt.

Tegtmeier and Duffy (2004) discuss external costs to agriculture as an example of estimating the lost revenues or costs associated with pollution or environmental degradation. Two examples, out of the many given in their discussion, include microbial pathogens damaging water resources and manure discharge into open surface waters. Microorganisms found in livestock manure can cause a number of different diseases and health concerns for humans. Should these microbes find their way into water sources, treatment is necessary for the water to be safe for human consumption. Should these microbes find their way into water sources, treatment is necessary for the water to be safe for human consumption. It was found that the cost associated with livestock manure is \$118.6 million US/year.

Catastrophic manure spills, leaks and or dumping by animal feeding operations into surface waters cause significant damage to aquatic environments and to some degree, can be assessed by the number of fish killed. Citing a report by the Clean Water Network from 2000, entitled "Spills and Kills: Manure Pollution and America's Livestock Feedlots", that recorded information on animal feeding operation manure spills and related fish kills in 10 states from 1995 to 1998; Tegtmeier and Duffy (2002) state that more than 13 million fish died as a result of over 200 documented manure pollution events. Tegtmeier and Duffy estimate the annual cost of these spills at \$11.9 million US.

A number of studies have employed hedonic estimation to determine the value of clean air in the environment. Milla et al (2005) evaluated the impacts, especially odor, of hog farms on property values. For their study, a geographic information systems-based hedonic pricing model was employed, using Craven County, NC as the area of observation. It was determined that at a distance of 1 mile from a 5000 head hog farm, a home valued at \$114,000 would experience a decline in value of \$3,550 or 3.1%. Other studies on property values also yielded similar effects. Kim (2004) found that assessed property values in Craven County declined at different rates at different given separation distances. Those rates and distances were found to be \$0.47/hog at a distance 0.75 miles, and \$0.42/hog at a separation of 1.25 miles.

Herriges et al (2003) studied five different counties in Iowa, using several different mathematical functional form models. From their findings, it was approximated that there could be a 10% reduction in property values should a new livestock facility be located upwind and near an existing residence. Ready and Abdalla (2003) analyzed the effect of near-by land uses and the potential disamenities on residential property values in Berks County, PA. Their study concluded that at 0.5 miles separation distance from an animal operation, there was a 4.1% decline in property value.

9.1.4 Policy tools/Regulatory Options

Although there is significant value in the nutrients from manure, over application of the manure to one field or poor management of the manure somewhere in the flow from the hog barn to the field could lead to toxic levels of nutrients in the soil or water shed. Poor management in the barn can also lead to odour and air pollution. As well, pathogens which might find a friendly environment in a storage lagoon could be transferred to fields and water sheds.

To combat environmental threats, many jurisdictions have some kind of legislation in place to deal with water and air pollution. Whether or not a region has specific legislation to regulate pollution from animal agriculture seems to be dependant upon the size of industry and its potential impact on the environment as Metcalfe (2001) indicated. For jurisdictions that do regulate their livestock industries for the purposes of environmental protection, there are many different regulatory options available. However, regulations on manure application limits, storage, abatement procedures, and separation distances tend to be the most common.

9.1.4.1 Short-term and long-term education programming

One policy option that leads to little conflict is the clarification and extension of management practices that benefit both the profitability of farmers and help protect the environment. Two examples are the use of phytase in rations and the incorporation of manure into the soil. In the first example, phytase, an additive to feed rations, saves producers money by reducing the amount of phosphorus needed in a ration and at the same time reduces the total phosphorus in the manure. The second example, incorporating the liquid manure into the soil, improves the availability of nutrients for plants from manure while lowering the risks of contamination in runoff and nuisance odour.

When win/win practices are identified, the only need is the extension of these ideas to producers. Pig producers have quickly adapted phytase to the extent that even with increasing pig numbers the amount of phosphorus generated by the industry in Manitoba could be going down.

Other beneficial practices explored in this report include; multiple cell storage that lowers risks of pathogen survival as well as lowering the energy and other costs of transporting the manure; and formulating diets with lower protein to reduce feed costs and as well as lowering ammonia gas production.

Provincial investments into research that identify these innovations and into extension programs that promote the new ideas would be of clear benefit to all of the stakeholders identified.

9.1.4.2 Incentive programs/Credits/Discharge Permit Systems

In the long run, as nutrient supplies become scarce and energy becomes more expensive, market forces will move to balance some of the factors discussed in this report. Increasing commercial fertilizer costs will push grain and pig farmers to make better use of the nutrients in manure and lower nutrient losses to the environment. Increasing energy costs will lead to better choices in terms of the energy budgets to supply nutrients to crop land. The market can be a useful tool to sort out the best use of a resource as long as those affected can make effective contracts with the resource owner. It is possible for groups to get together and pay producers to lower odour or take steps to keep the water supply clean if those affected can insure they will benefit from such actions and that farmers will do what they agreed to do.

Regulators have already begun to use market forces to sort out the cheapest ways to reduce carbon emissions in the E.U. and a voluntary carbon emissions market has started in the U.S. These markets let the polluter with the most economic benefit from polluting trade with agents who can reduce carbon emissions cheaply. While these markets have a lot of appeal theoretically, they need strict enforcement of the contracts

that certify emission reduction and they need regulators who set initial emission levels that lead to real environmental change.

In economic jargon the markets discussed above are known as a transferable discharge permit systems. In this case, economic incentives to abate pollution are incorporated into a decentralized policy application, where an exchange market is set up for polluters to buy and sell pollution credits or transferable discharge permits. Field and Olewiler (2005, pg. 442) define transferable discharge permits as “A property right to emit a specific amount of pollution. The right is transferable; that is, it can be sold or given away”. Normally, each permit gives the holder the ability to discharge one unit of waste (as it is measured) into the environment over a certain period of time, as specified in the permit. Therefore, the total number of permits available in the market is equal to the total amount of pollution allowed to be discharged. Similar to an effluent tax, there is an upper limit applied to how much pollution is acceptable under the law, forcing those who would surpass the set limit or are unable to acquire more permits to take action to reduce their emissions. A functioning example of this is the permit program for SO₂ emissions among electricity producers in the U.S.

Initially, the pollution credit market is set up with a decision on the total number of credits that will be made available. Then, a selected number of permits are distributed to each of the firms that are responsible for the emissions. Afterwards transactions between the firms would occur. The key principal behind the trading is that a firm will decrease its emissions and sell the excess credits it owns if the market price is greater than or equal to its marginal abatement cost at the firm’s chosen level of discharge. A firm will buy credits if the price is less than or equal to its marginal abatement cost at its chosen level of emission. In essence, a firm’s marginal abatement cost curves are analogous to its demand curve (when buying) or supply curve (when selling). The market then acts in a similar manner to any other commodity market as those desiring more permits would trade with those who desire fewer permits until the market equilibrium is reached; where supply equals demand.

There are several advantages associated with a transferable discharge permit system. Like a standard based system, the credits guarantee that a target emission level is met. Similar to an environment tax, the permit system is a cost-effective policy as regulators do not have to determine each firm’s marginal abatement cost curve to find the correct price to reach cost-effectiveness. This is because the market reveals this information as, when the market clears, the permit price is equal to the marginal abatement cost of each polluter. There are also market-driven incentives to innovate and discover less costly ways to reduce emissions. This leads to greater investment in research and development and the creation of better technology for the industry.

Despite the upside of this system, there are also a number of issues that could very well lead to problems. The first is the initial rights allocation, as just about any method of allocation will have some inequalities or negative incentives associated with it. An example of this would be if regulators initially distributed permits evenly amongst all firms. This would seem fair, however, it does not account for the fact that firms can vary greatly in size and in output. As a result, firms of smaller size would be conferred an advantage over larger firms, essentially penalizing larger firms. Another issue is the transaction rule set up for the market. Rules must be set in place to determine who may trade, how the trades are to be conducted, and still insure that allow the market will work effectively. The goal is to leave the market to its own devices after the initial permit

allocation, rather than having public or environmental agencies trying to monitor and possibly influence the market's performance. Such an influence would increase transactions costs in the market and hinder its efficiency.

Thirdly, the degree of competition in the market is critical issue. A fundamental aspect required for any market to succeed is a sufficient amount of liquidity, driven by market competition. Should there be too few traders, market inefficiencies could arise from lack of any competitive pressure. Also should a small group of firms or a dominant firm exercise some level of control over the market, this would allow them to have a degree of economic control over the industry that they normally would not have. Therefore it is important to have a large number of relevant market participants to foster competition. Lastly, the ability to monitor pollution can also be an issue. This is because in many instances, pollution emissions are not discharged uniformly over time (daily, weekly, and seasonal rates may vary). Because of this, sophisticated monitoring would be required, adding to the existing administrative costs of enforcing the rules to prevent firms from cheating.

9.1.4.3 Disincentive programs

Other regulations as will be discussed below, focus on regulations that force specific behaviour under the rule of law. These may be the most effective when the beneficiaries of a behaviour do not form a cohesive group, they include society as a whole, or the need for control is too urgent to develop a market tool. While they are wide spread, these options suffer if enforcement is insufficient or the laws are not consistently applied. For example small hog farms in Manitoba may be able to do things larger ones cannot because they face a different regulatory regime.

9.2 Regulatory tools used in other jurisdictions

Many countries/states/provinces generally have some kinds of legislation in place to deal with water and air pollution. Whether or not a jurisdiction has specific legislation to regulate pollution from animal agriculture would seem to be dependant upon the size of industry and its potential impact on the environment as Metcalfe (2001) indicated. For jurisdictions that do regulate their livestock industries for the purposes of environmental protection, there are many different regulatory options available. However, regulations on manure application limits, storage, abatement procedures, and separation distances tend to be the most commonly found in legislation. With regard to manure application regulations, (for more information on storage regulations, see the Manure Storage Section), the following jurisdictions will be highlighted below: Saskatchewan, Alberta, Ontario, North Dakota, Minnesota, Iowa and North Carolina.

9.2.1 Saskatchewan

Saskatchewan's animal feeding operations are governed by two pieces of legislation, including The Agriculture Operations Act (Act) and The Agricultural Operations Regulations (Regs). Intensive livestock operations that: contain an earthen manure storage area or a lagoon; and/or involves rearing, confinement, or feeding 300 or more animal units; and/or involves more than 20 but less than 300 animal units which is within 300 meters of surface water or 30 meters of domestic water well that is not controlled by the operator of the livestock operation are required to have an approved waste storage plan and an approved waste management plan (Act S.19, Regs S.3). The required plans will only be approved if: all requirements of the Act and Regulations have been

satisfied; it is deemed that water pollution will not occur; and sufficient provisions have been made for managing animal manure (Act S.21).

Key stipulations within waste management plans that are required include: providing the annual volume of manure produced; estimating the nutrient levels in the manure for nitrogen, phosphate and potassium; specifying the form of the manure (solid or liquid); specify the method and the season for the application of the manure; specifying the expected crop nutrient requirements based on the crop production area or specific cropping practices; specifying the annual rate of manure application to meet estimated crop nutrient requirements based accepted manure utilization rates; and specifying the land area available for application of manure and providing a map to identify the location of the lands used for manure application (Regs S.8).

9.2.2 Alberta

In Alberta, the Agricultural Operation Practices Act (AOP Act), and the associated Standards and Administration Regulation (SA Regs) and the Agricultural Operations, Part 2 Matters Regulation (AOM Regs) and the Manure Characteristics and Land Base Code govern manure application practices. The first piece of legislation that application standards are built upon is that a person who applies manure, compost or composting materials (manure) must apply it in such a manner that does not disobey the regulations set out unless approval or authorization has been granted (Act S. 15). Section 23 of the Standards and Administration Regulation states that the owner/operator of an agricultural operation must manage manure in accordance with the nutrient management requirements found in Schedule 3 of the SA Regs.

Within the published application limits in Schedule 3, other stipulations also affect manure application. Manure can only be applied to arable land and if the land is cultivated, the manure must be incorporated within 48 hours of application (SA Regs S.24(1)). This is unless it is applied to forage, directly seeded crops or frozen/snow-covered land (SA Regs S.24(5)). Frozen/snow-covered application must receive approval or be deemed necessary due to weather conditions (SA Regs S. 24(6-7)). In any case, manure must not be applied within: 10 m of a common body of water if injecting; 30 m of a common body of water if the person is surface applying and then incorporating; and 30 m of a water well (SA Reg S.24(9)).

Soil must be tested at least every 3 years or else application is not allowed, unless a total of 500 tonnes or less is applied annually; application cannot increase the soil salinity by more than 1 decisiemens/m and if soil salinity is measure to be 4 decisiemens/m or greater, manure cannot be applied (SA Regs S. 25(1-4)). If the soil nitrate-nitrogen levels will exceed the limits outlined in Table 3 above after application of manure, application is then prohibited (SA Regs S. 25(5)). Finally, nutrient management plans are required for manure application, unless the owner/operator meets land base requirements set out in the applicable tables found in the Manure Characteristics and Land Base Code (SA Regs S. 24(2a) & 25(7)).

9.2.3 Ontario

Ontario has set in place the Nutrient Management Act (NM Act) and the Nutrient Management Act Regulations (NMA Regs) to legislate manure application within the province. The Act gives the right for regulations to be made that govern all pertinent aspects associated with materials containing nutrients from farm animals, such as manure management plans, application guidelines, licensing, etc (NM Act S. 6(1-3)).

Nutrient management plans apply to an agricultural operation carried out on a farm and the person who owns/controls the operation, such that nutrients are applied to the land on a farm, must adhere to the nutrient management plan without deviation (NMA Regs S. 9(1(a) & 14(1)). Nutrient management plans must optimize the relationship between land-based application of nutrients, management techniques and crop requirements; use land which maximizes the efficiency of on-site nutrient use; and minimize adverse environmental impacts (NMA Regs S.23).

Manure cannot be applied within 100 m of a municipal well, 15 m of a well with 15 m depth and a water-tight casing at least 6 m deep below ground, or 30 m of any other well (NMA Regs S. 43(1-3). Vegetated buffer zones are required to be in place, in addition to a separation distance of 13 m that must be observed from the top of the nearest bank of surface water, otherwise manure cannot be applied in fields adjacent to surface water areas (NMA S. 44(1,4)). During winter and other times when soil is frozen or snow-covered, manure application is generally prohibited (NMA S. 47(1)).

Land application rates for manure are to be calculated from soil and manure test analyses in which phosphorus and potassium are the key nutrients analyzed in the soil and nitrogen, phosphorus and potassium are the main nutrients analyzed in the manure (NMA S. 91(1-3). The maximum application rate allowed under the regulations is such that the total available phosphorus does not exceed the greater of: the crop production requirements per hectare for the 5 year period of the nutrient management plan plus 85 kg of phosphorus per hectare; and the phosphorus removed from the land per hectare in the harvested portion of the crop during the 5 year period plus 390 kg of phosphorus per hectare (NMA S. 92(2)).

Ontario has also influenced some behaviour change by offering beneficial rates to producers supplying low emission energy through biogenerators as discussed in the manure processing chapter.

9.2.4 North Dakota

The legislation governing North Dakota's animal manure applications include Chapter 33-16-03.1 Control of Pollution from Animal Feeding Operations found in the North Dakota Administrative Code (NDAC) and the North Dakota Livestock Program Design Manual (DM) which acts as an interpretation of the NDAC. By law, the operator of an animal feeding operation that is defined or designated as a concentrated animal feeding operation (CAFO) must obtain a North Dakota pollutant discharge elimination system permit; or be granted a no potential to pollute status (NDAC S. 5). Operations must develop and maintain a nutrient management plan, if manure from the operation is to be applied to croplands or grasslands (NDAC S. 8(3). Nutrient management plans must provide information on the land the manure is to be applied to, and information that demonstrates manure will be applied at agronomic rates (NDAC S. 8(3a)). The agronomic rate for nitrogen cannot be greater than the crop utilization rate for the cropping year, and phosphorus is not to be applied at rates exceeding the recommendations based on either the North Dakota phosphorus index, the North Dakota state university service soil tests or other approved assessment methods (NDAC S. 8(3a)). Also, precautions must be taken to prevent manure from reaching waters of the state and to minimize odors to residences and public areas during application and transport (NDAC S. 8(3c)).

9.2.5 Minnesota

Minnesota farmers that produce and spread manure are governed by Chapter 7020 of the Minnesota Rules entitled Land Application of Manure (MR). Overall, manure cannot be applied to land in such a manner that it results in discharge into state waters or causes pollution of state waters from manure contaminated runoff (MR S. 1A(1,2)). Manure is also prohibited from being applied into road ditches (MR S. 1B).

Owners and operators of animal feedlots that either: apply for a National Pollution Discharge Elimination System (NPDES) permit or other state permits (construction, etc) are required to file a manure management plan (MR S. 4A(1)). Owners of feedlots that have the capacity to house 300 or more animal units that do not yet have a permit and apply manure without certification must also be required to file manure management plans (MR S. 4A(2)). Also, manure management plans can be requested by the state authority or the local county feedlot pollution control officer (MR S. 4B(2)). After this, manure management plans are to be reviewed annually by the owner, and adjusted as necessary to account for any variations in production or practice that would affect the nutrients available or required for fields that receive manure (MR S. 4C).

Application rates in Minnesota are to be limited so that the estimated plant available nitrogen is not in excess of the expected nitrogen requirements of the crop (MR S. 3A). However, the estimated plant available nitrogen from organic sources in the field, including manure, is allowed to deviate by as much as 20 percent if due to field nutrient history, soil conditions, or weather cause nitrogen deficiencies (MR S. 3A(2)). If the deficiencies are measured or visible, corrective applications of nitrogen can go above the 20 percent deviation (MR S. 3A(2)). Fields that have manure spread on them from operations that house 300 or more animal units must be soil tested for phosphorus at least once every four years (MR S. 3C). Should the phosphorus be measured above specified limits, then the farmer must complete a manure management plan and apply for a NPDES or other state permit (MR S. 3C & 4B(1)). Manure that is to be applied in special protection areas, must not do so when the area is snow-covered or frozen (MR S. 6A). Manure application to protected areas must adhere to a number of state ordinances found in section 6 of Chapter 7020.2225.

9.2.6 Iowa

Iowa's laws are found in Chapter 459 of the Iowa Code, entitled as the Animal Agriculture Compliance Act (AAC Act). The first portion of this law that deals with application is found in Section 204 which states that there must be a separation distance of 750 feet from a residence not owned by the title holder of the land, commercial enterprise, religious institution, educational institution or a public use area when applying liquid manure to a field. The exceptions to this rule include the following: the liquid manure is injected or incorporated into the soil within 24 hours of the application; the titleholder of the land benefiting from the separation distance signs a waiver allowing the application; the manure originates from an operation stated to be small in size; manure is applied by irrigation using a central pivot system that does not disperse effluent at a height greater than 9 feet, with pressure being less than 25 psi and the manure is not applied within 250 feet of any building or area listed in S. 204 (AAC Act S. 205(4)).

Manure is not allowed to be discharged directly into waters of the state or into tile lines that drain into waters of the state and must be disposed of in such a way so as to not pollute surface or groundwater (AAC Act S. 331(1-2)). Owners of animal feeding operations, except for small operations as they are identified, must submit a manure

management plan on an annual basis and must receive approval in order to apply manure (AAC Act S. 312 (1(a), 3 & 5)). Manure application rates are to be determined by: soil and manure nutrient tests and calculations used to find the land area needed for manure application based on nitrogen use levels for obtaining optimum crop yields; and a phosphorus index that shall determine if application rates can be applied based solely on nitrogen in the soil and the manure or on phosphorus with the application rate based on phosphorus content inherent to the soil and manure (AAC S.331 (10 (1, 2)), Phosphorus Index).

9.2.7 North Carolina

North Carolina has used strict emergency measures including a total moratorium on expansion of the industry. Currently the Swine Farm Siting Act (SFS Act) and Chapter 143 of the North Carolina General Assembly entitled Animal Waste Management Systems (AWMS) govern the application of Manure in North Carolina. Under the Chapter 143, section 215.10C(a) of the Code, no one is allowed to construct or operate an animal waste management system without a proper permit. This is to ensure that animal waste management systems are designed, constructed and operated so as to avoid pollution of state waters, except in the case of a rainfall from a storm event more severe than a 25-year, 24 hour storm (AWMS S. 215.10C(b)). Animal waste management plans are required for animal operations, which must provide: a checklist of potential odour sources and a choice of site-specific; remedial action to minimize those sources; periodic testing of manure with 60 days prior to application; and periodic testing (at least annually) of soils found in fields in which manure is to be applied (AWMS S. 215.10C(e)(4,6)).

For field application nitrogen is to be the rate determining factor, with phosphorus, zinc and copper being evaluated based on nutrient management standards; in the event that phosphorus is in need of being limited in order to comply with standards, phosphorus shall be a rate determining factor (AWMS S. 215.10C(e)(6)). Waste utilization plans must ensure that a balance between nitrogen application rates and crop nitrogen utilization rates, while allowing for corrective action, should this balance not be achieved (AWMS S. 215.10C(e)(7)). Lime is to be applied to fields to maintain their soil pH in the optimum range for crop production (AWMS S. 215.10C(e)(7)). When applying manure to fields, the Swine Farm Siting Act stipulates that the outer perimeter of the field must be at least 75 feet from any property boundary having an occupied residence and from any perennial stream or river aside from an irrigation canal/ditch (SFS Act S. 803(a1)).

9.2.8 Stringency Comparisons

Although getting out of date, Metcalfe³ gathered U.S. state by state comparisons of environmental stringency for analysis of these effects on ILO locations and viability. There are some arguments that stringency normally follows intensive ILO growth, so regulations impact on growth was not clear in terms of causality.

9.3 Policy recommendations for Manitoba

While this report contains wide spread recommendations in each chapter the focus has been on scientifically proven links between a recommended change and the desired outcome, or on changes that seem to clearly benefit all of the stake holders.

9.3.1 Verifiable data collection systems

A basic lack of data for Manitoba was noted for many of the issues discussed in this report. Even where specific links between a negative affect and some nutrient level or pathogen level were established, the link between pig production and those nutrient or pathogen levels often could not be established. In other cases research is missing. For example, the basic soil mechanics of the flow of phosphorus needs further research. Funding to establish baseline data and trends for key parameters is a need. An example would be the monitoring of nutrients levels in drainage basins and soils.

9.3.2 Technology-based vs. consensus-based recommendations

Recommendations in previous chapters were made by people with expertise in the specific issue. The recommendations were based on the best science available and applied to Manitoba conditions. When there is not enough data to make a technical recommendation, or when various stake holders are worried about different risks, recommendations may be based on practices that would benefit most of the stakeholders involved.

It seems clear that some costs will be forced on the pig industry. To force changes without clear proof of the benefit, seems unfair unless those requesting the change are willing to pay incentives to the producers for the change. Even if there is proof of benefit, stakeholders receiving benefits should be willing to compensate those who face an increased cost.

9.3.3 Flexibility

Policy needs to be flexible if potential threats and solutions are still evolving. The current policy proposals for phosphorus based nutrient management are an example of a measure that allowed some flexibility in meeting a specific environmental target. There is often a need for flexibility as the science, technology, and data needs of an environmental problem are established. For example, without any regulatory push, hog farmers have already begun to reduce phosphorus generated through the use of phytase. There are various practices identified above that would change average nutrient loading (and removal) from liquid hog manure. By allowing some flexibility as the soils build up depleted (in some cases) phosphorus levels the industry has time to develop appropriate technology and make adjustments. At the same time, scientists have time to improve their models and regulators have time to collect important data to confirm the real effects of any proposed changes. Once a real threat is identified, flexibility might not be possible, but as a system as complex as the Manitoba hog industry learns more about its environmental footprint, some level of flexibility will allow the industry to develop new technology and practices to minimize the costs to meet environmental goals.

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