The Risk of Phosphorus Transfer to Water from

Manure Application onto Agricultural Land[1]

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Abstract
Much of Manitoba's agricultural land is naturally low in phosphorus (P) for crop production and benefits from the application of P in livestock manure. However, if manure is applied improperly or at an excessive rate, significant amounts of P can move off the land or through the soil and into surface water bodies where the increase in P results in eutrophication. The main pathways for P movement to surface waters are release of soil-bound P from eroded soil entering water bodies and the flow of soluble P dissolved in surface runoff or P from groundwater contaminated by P leaching. To minimize the movement of P from agricultural fields to water bodies, two groups of factors must be considered and managed accordingly: source factors, such as nutrient form, application and removal rates, the soil's capacity for nutrient retention and degree of saturation, and transport factors, such as risk of soil erosion, surface runoff and leaching. The ideal tool for managing or regulating P application to land incorporates all of these factors.

1 Introduction

Phosphorus contamination of surface water bodies can damage quality by the process of eutrophication (increasing growth of algae, surface scums, followed by the depleted oxygen concentrations, foul odours, sedimentation, fishkills and release of algal toxins). The P responsible for eutrophication originates from a variety of sources, not only from livestock and crop production activities, but also from natural ecosystems and discharge of human and industrial waste.

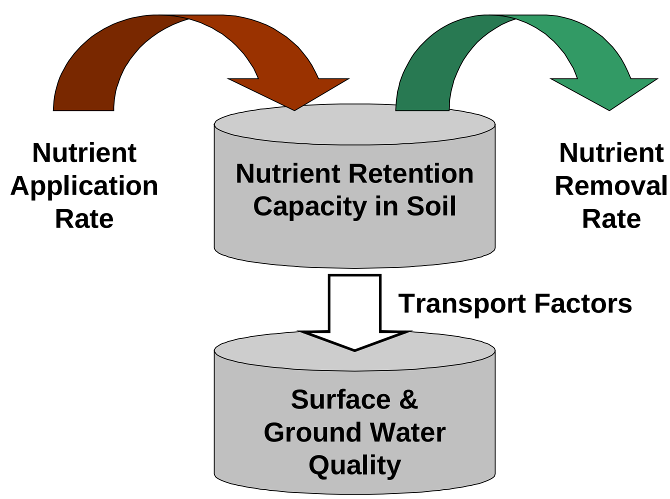
Eutrophication occurs at very low concentrations of P in water (Correll 1998). Such small amounts of P loss are not agronomically significant, but are very significant from a environmental perspective. In addition, the environmental impacts of these small losses of P occur many miles from the P source (e.g., in Lake Winnipeg) and reflect the cumulative impacts of all sources. As a result, where water quality guidelines for P are established, they are usually set at very low concentrations. Manitoba Conservation is in the process of developing water quality standards, objectives and guidelines (Manitoba Conservation 2001). Part of the reason for developing such standards is due to concerns over rising concentrations of P in many Manitoba streams (Jones and Armstrong 2001).

The main sources of P contamination in surface water include point sources from urban and industrial activities and non-point sources from agricultural activity and natural processes such as erosion and wildlife. Small amounts of P may also be deposited from rainfall, especially in areas where atmospheric pollution is a concern. For example, in a P balance study in France, scientists found measurable concentrations of P from atmospheric deposition, but the deposition rates were less than 0.5 kg P ha^{-1} $year^{-1}$ (Pommel and Dorioz 1997). Under natural conditions, the release of inorganic P from rocks and minerals is very slow due to the low solubility of such minerals. Studies of the natural weathering of such minerals in virgin watersheds in Canada showed that P export was greater from watersheds developed on sedimentary rock than from those developed on igneous rock. However, diatom studies have shown that the introduction of urban and agricultural activity into a region causes a substantial increase in accumulation of P in water (Dillon and Kirchner 1975). In Manitoba, as well, most of the streams where P concentrations are increasing are situated in Southern Manitoba, where urban and agricultural activities are most intense (Jones and Armstrong 2001).

Aquatic loading of P due to agricultural activity is a significant source of water quality impairment in Canada (Chambers et al. 2001) and the U.S. (Parry 1998). The main sources of P that contribute to these losses are synthetic fertilizers, crop residues, and livestock manure. For the latter source, areas that export P into surface water include pastures, manured hayland, manured cultivated fields, wintering areas, feedlots, manure storage systems, and livestock handling yards. As a result of the important role played by agriculture in increasing the concentration of P in water bodies, management of agricultural sources of P contamination has attracted considerable attention from the scientific community in North America and Europe (please refer to Sharpley and Menzel 1987, Daniel et al. 1998, Higgs et al. 2000, Sharpley et al. 1994, Sharpley and Rekolainen 1997, Sharpley et al. 2000, Sharpley and Tunney 2000, and Van der Molen et al. 1998 for some excellent reviews on this topic).

The factors affecting the risk of P transfer from agricultural land are similar to the typical source-receptor-pathway model that applies to most types of environmental contamination. In the case of nutrients applied onto agricultural land, the source factor can be subdivided into several subfactors including form of nutrient and rate, method of application; rate of nutrient removal by crops and livestock; and the soil's capacity to retain nutrients. Pathway factors relate to the transport of nutrients in particulate and dissolved forms. And, of course, the receptor factor relates to surface or groundwater, where a defined level of water quality is desired. All of these factors interact in the general model described in Figure 1.

Figure 1.1. Factors affecting the risk of nutrient transfer to water



With respect to P, much of the agricultural land in Manitoba is naturally low in phosphorus (P) and benefits from the application of P in livestock manure. However, if an excessive rate of manure is applied or if manure is applied improperly, significant amounts of P can move off the land or through the soil and into surface water bodies. The main pathways for P movement to surface waters are:

- release of soil-bound precipitated or adsorbed P from eroded soil entering water bodies
- soluble P dissolved in surface runoff
- flow of P from groundwater contaminated by P leaching

To minimize the movement of P from agricultural fields to water bodies, two groups of factors must be considered and managed accordingly: factors that affect accumulation of high concentrations of P at the soil surface (source factors, such as nutrient form, rates of nutrient application and removal and the soil's capacity to retain nutrients and degree of saturation) and factors that affect transport of that P to water bodies (transport factors, such as risk of soil erosion, surface runoff and leaching).

Environmental problems with P loss do not occur unless source and transport factors occur together simultaneously. For example, a soil with a high concentration of P and little opportunity for soil and water movement to a water body is not considered a major environmental hazard. Similarly, the environmental threat of a soil with a low concentration of P and high potential for movement to a water body is also low. However, a P enriched soil in a landscape and climate where movement to water is likely, will pose a significant threat to water quality. As a result of the interaction between these source and transport factors, a high proportion of the P loss from agricultural land is often confined to a relatively small area (Sharpley et al. 1999).

2 Phosphorus Content of Manures and Application Rates

Over the short term, applying P in modest excess above crop removal will improve the productivity of many soils in Manitoba, with little risk of environmental harm because many Manitoba soils are low in P. However, soil does not have an infinite capacity to retain P (Sharpley et al. 1999) without the risk of significant environmental harm. Therefore, over the long term, manure P application rates will have to be balanced with crop removal (Daniel et al. 1997) and P-based manure application guidelines and/or regulations will eventually be required. As a result, balancing additions and withdrawals of nutrients is an important exercise for regional governments and for individual farmers. For example, a regional P balance is an important tool for assessing the opportunities and limitations of land application of livestock manure P for a municipality and field scale P balances are important tools for managing nutrients in an agronomically and environmentally responsible manner within a specific farm.

2.1 Role of P in Livestock Production

Phosphorus plays a very important biological role in livestock production and has more known functions than any other mineral nutrient (Lynch and Caffrey 1997). One of the most important roles of P is as a structural element in bones, teeth, and eggshells; however, P also plays an important role in many metabolic processes and is an important ingredient for milk production in dairy cattle and nursing beef cows. Therefore, animal products contain significant amounts of P that must be considered when examining the balance between imports and exports of P in a livestock production system (Table 1.1).

In the feed grains portion of a typical livestock diet, 40-70% of the P is present as phytin, a very stable ring-type chemical form of P that is not available to animals unless it is broken apart by the phytase enzyme (Lynch and Caffrey 1997, Mikkelson 1997). The digestive tracts of non-ruminants such as hogs and poultry, have small quantities of phytase and are not capable of using organic P efficiently. As a result, typically, more than 65-90% of P ingested by hogs is excreted as feces and urine (Lynch and Caffrey 1997, Mikkelson 1997, Brumm 1998) and approximately 20% of that excreted P is in the form of intact phytate (He and Honeycutt 2001). Presumably, this problem of poor utilization of P by monogastric animals is at least partly responsible for the observation that poultry and hog farms tend to have higher surpluses of P on their farms than dairy and beef producers (Sims et al. 2000).

Table 1.1. Export of P in animal products (adapted from Lynch and Caffrey 1997)

Animal Product	**P Concentration ($g\ P\ kg^{-1}$)**
Cattle (live basis)	7-9
Milk	0.9
Chickens (live basis)	5
Eggs	2
Pigs (live basis)	5

In order to ensure that inadequate P nutrition does not limit livestock production, livestock feed is frequently supplemented with concentrations of additional P that provide a substantial "safety" margin to ensure sufficiency (Poulsen 2000). For example, in swine feed supplements, P is frequently added as monocalcium or dicalcium phosphate, forms of P that are relatively available to the animal.

Researchers and commercial producers are attempting to modify animal feeding strategies to increase the efficiency of feed utilization and decrease the P content in manure (Valk et al. 2000). Ruminant livestock are assumed to be capable of using organic P because the rumen microbes provide ample quantities of phytase enzyme (Van Horn and Hall 1997). However, many dairy producers overfeed P relative to NRC recommendations and reducing P to NRC recommendations would reduce P excretion per cow by approximately 20% or more (Van Horn and Hall 1997). Beef cattle nutritionists are also experimenting with reduced rates of supplemental P in diets and finding no loss in livestock performance (Erickson et al. 1999). Adding phytase as an enzyme supplement to feed can increase P availability to monogastric livestock and lower P requirements in the feed (Brumm 1998). In addition, plant breeders are attempting to develop varieties of Highly Available Phosphorus (HAP) feed grains, which have low concentrations of phytin (Ertl et al. 1998). For example, the Crop Development Centre at the University of Saskatchewan is incorporating HAP characteristics into some new barley breeding lines. However, the net environmental benefit of these practices is unknown because the impact of improved availability of dietary P to the animal on the losses of P from their manure is still being investigated (McGrath and Sims 2000).

Utilization of digested P must also be considered. Once absorbed by the animal's digestive tract, P in excess of the animal's requirements is excreted, mainly as urine by monogastric animals and mainly as feces by ruminants (Lynch and Caffrey 1997, Eghball and Power 1998). For example, slaughter hogs and sows excrete approximately 10% and 32% of dietary P, respectively, via urine (Poulsen 2000).

2.2 Forms of Phosphorus in Manures

The balance of various forms of P varies substantially with species, diet composition, production system and manure management system (He and Honeycutt 2001). Therefore, estimates for the composition of P in manure are also highly variable. For example, Brookes et al. (1997) estimated that inorganic P comprises approximately 80% of the P in liquid hog manure and organic P comprises only 20% of the P in liquid hog manure. This figure agrees with the figure of 70-90% cited by Schoumans and Groenendijk (2000). However, Mikkelson (1997) estimated that, generally, up to 50% of the total P in swine manure is in the organic form, compatible with the 49% organic P in swine manure measured by He and Honeycutt (2001). He and Honeycutt (2001) also examined beef cattle manure and measured 44% organic P in that source. Within the organic fraction of manures, the forms of P are diverse, including: simple P monoesters, phytate, nucleic acids, organic pyrophosphates and unidentified residual organic P (He and Honeycutt 2001).

2.3 Effect of Livestock Species and Stage of Growth on P Content in Manure

Poultry manure tends to have a higher concentration of P than for manure from other farm animals (Table 1.2). The high proportion of soluble P in poultry manure further enhances the mobility of poultry manure P, resulting in runoff of P from fields receiving poultry manure, even when best management practices are followed (Moore et al. 1998).

The N:P ratio in manure (Table 1.2) is generally lower than the N:P ratio in crops (Table 1.6). One fundamental reason for the lower N:P ratio in manure than in crops is that livestock are relatively more efficient at utilizing N in the crops they are fed, compared to P. As a result, the N:P ratio of plant material drops as the material passes through the animal's digestive system. Another reason for the relatively low N:P ratio of manure is the addition of P supplements to livestock feed and the relatively high losses of N due to volatilization and denitrification during collection, storage and application of manure (Table 1.2).

The concentration of P in manure and the N:P ratio in manure also varies with stage of growth, especially if different livestock housing and manure handling systems are used in different stages (Table 1.2.). However, if the housing and manure handling systems are similar, for example in confined feeding of hogs for slaughter, the N:P ratio remains fairly consistent (Table 1.3).

Table 1.2. Typical manure production values per 1000 lb animal unit for various types of livestock (Source: adapted from USDA NRCS figures in Kellogg et al. 2000).

Livestock type	Total Manure (lb/day/AU)	N (lb/day/AU)		P[1] (lb/day/AU)		N:P Ratio	
		As excreted	After losses	As excreted	After losses	As excreted	After losses
Fattened beef cattle	58	0.32	0.13	0.10	0.08	3.26	1.53
Beef calves to 500 lb	62	0.26	0.08	0.07	0.06	3.66	1.29
Beef heifers for replacements	66	0.20	0.06	0.04	0.04	4.66	1.65
Beef breeding cows and bulls	63	0.34	0.10	0.12	0.10	2.89	1.02
Beef stockers and grass fed for beef	62	0.26	0.08	0.07	0.06	3.66	1.29
Dairy cows	83	0.45	0.18	0.08	0.07	5.57	2.61
Dairy calves to 500 lb	66	0.20	0.06	0.04	0.04	4.66	1.65
Dairy heifers for replacements	66	0.20	0.06	0.04	0.04	4.66	1.65
Dairy stockers and grass fed for beef	66	0.20	0.06	0.04	0.04	4.66	1.65
Hogs, breeding	33	0.22	0.06	0.07	0.06	3.10	0.92
Hogs, slaughter	80	0.45	0.11	0.13	0.11	3.43	1.01
Chickens, layers	63	0.84	0.58	0.31	0.27	2.70	2.17
Chickens, pullets	46	0.62	0.31	0.24	0.20	2.58	1.52
Chickens, broilers	82	1.10	0.66	0.32	0.27	3.44	2.44
Turkeys, breeding	50	0.56	0.28	0.33	0.28	1.70	1.00
Turkeys, slaughter	45	0.68	0.36	0.26	0.23	2.57	1.61

[1]Expressed on elemental P basis (not as P_2O_5)

Table 1.3. Manure production rates and characteristics for hogs at various stages of growth

Stage of Development	Weight	Nitrogen	Phosphorus[1]	N:P Ratio
	kg/animal	------ kg/day/animal -------		
Weanling	16	0.007	0.002	3.21
Grower	41	0.019	0.006	3.11
Finisher	79	0.036	0.012	3.05
	102	0.046	0.015	3.10
	113	0.052	0.017	3.05
Gestating Sow	125	0.028	0.010	2.91
	148	0.033	0.011	2.91
	181	0.041	0.014	2.93
Sow & Litter	136	0.083	0.028	3.02
	181	0.111	0.037	3.03
Boar	159	0.035	0.012	2.97

[1]Expressed as elemental P (not as P_2O_5)

(Source: adapted from American Society of Agricultural Engineers Standard ASAE D384 and Midwest Plan Service MWPS-18, Livestock Facilities Handbook, as reported in Appendix J of Farm Practices Guidelines for Hog Producers in Manitoba, Manitoba Agriculture 1998).

2.4 Bedding, Manure Handling and Storage Systems

Within a production system for a given livestock species, such as hogs, the concentrations of N and P in "fresh" manure are highly variable (Table 1.4) and are influenced by factors such as type of barn, type of manure storage and location of sample within the manure storage system (Table 1.5). For example, in a recent study of hog barns in Manitoba, the average N:P ratio of nursery barns was much lower than the overall average, while the N:P ratio of feeder barns was slightly higher than average. In addition, manure samples collected from the bottom third of agitated liquid storage facilities had lower N:P ratios than those from the top third (Fitzgerald and Racz 2001).

Table 1.4. Summary of nutrient analyses from 145 liquid hog manure samples collected in Manitoba during 1998-1999 (Source: Fitzgerald and Racz 2001).

	Dry Matter %	Total N	Available N	Total P[1]	N:P Ratio
		---------- Lbs/1000 gallons ---------			
Maximum	38.6	65.1	46.9	55.2	32.00
Minimum	0.0	6.0	5.4	0.3	0.89
Average[2]	3.5	28.9	23.6	9.2	3.13
Median	1.7	25.0	22.4	4.6	5.00

[1] Expressed as elemental P (not P_2O_5 equivalent)
[2] Average N:P ratios are calculated from overall average concentrations of N and P, not from ratios for individual samples. For mathematical reasons, the calculation of average ratios from ratios for individual samples results an artificially high N:P ratio that is not coherent with the raw data.

Table 1.5. Summary of average nutrient analyses for different sample depths and types of barn and manure storage for 145 samples of liquid manure collected from hog farms in Manitoba (Fitzgerald and Racz 2001).

	Dry Matter %	Total N	Available N	Total P[1]	N:P Ratio[2]
		------- lb/1000 gallons --------			
Sample Depth					
Top (n=62)	2.3	26.0	22.3	6.1	4.2
Middle (n=30)	3.7	28.0	22.9	8.8	3.2
Bottom (n=53)	4.6	32.8	25.4	13.1	2.5
Type of Barn					
Farrow to Finish (n=5)	2.1	27.6	25.0	10.6	2.6
Nursery (n=11)	3.1	27.2	21.7	11.7	2.3
Feeder (n=92)	3.7	33.9	27.8	10.0	3.4
Sow (n=37)	3.0	17.3	13.6	6.5	2.7
Type of Manure Storage					
Open Earthen (n=114)	3.3	29.7	24.2	8.9	3.3
Open Earthen - Primary (n=9)	8.0	32.1	23.4	13.6	2.4
Open Earthen - Secondary (n=14)	1.7	20.5	16.5	7.9	2.6
Slurry (n=8)	2.9	28.7	26.7	11.6	2.5

[1] Expressed as elemental P (not P_2O_5 equivalent)
[2] Average N:P ratios are calculated from overall average concentrations of N and P, not from ratios for individual samples

Composting of livestock manure generally results in losses of C, N and water. As a consequence, composting generally increases the P concentration, decreases the N:P ratio and, in most cases except poultry manure, also decreases the extractability and plant availability of P in the manure due to immobilization (Gagnon and Simard 1999). However, the extent of these effects varies greatly with the source of compost material and with the method of compost management.

2.5 Effects of Manure Application Methods and Timing on Phosphorus Loading Into Soil/Water System

Surface application of livestock manure greatly increases the concentration of dissolved P in surface runoff (Sharpley et al. 1994). Conversely, subsurface placement or incorporation of manure greatly reduces the loss of dissolved P; however, the tillage operation associated with such methods of application usually result in increased risk of P loss in the particulate form.

Timing has several effects on the risk of P loss. Over the short term, the risk of losing dissolved and particulate P declines rapidly within a few hours or days of manure application (Pionke et al. 1997, Sharpley et al. 1998). Environmental conditions also play a large role in the control of P loss. For example, in a simulated rainfall experiment, Pote et al. (1999) found that soil sampled in August (dry season) lost nearly twice as much dissolved reactive P as samples taken in May (wet season). Another seasonal issue to consider is that the ideal time for applying manure is when the crop is actively growing, protecting the soil surface and utilizing nutrients. Manure should also be applied when the risk of runoff is low. Therefore, fall or winter application of manure prior to the runoff events that usually occur during spring snowmelt is not desirable (for more detailed information on the effects of manure application and timing on P losses to water, please refer to section 5).

3 Phosphorus Removal by Crop Uptake

Currently, the main objective of Manitoba's manure application guidelines and regulations is to prevent excessive accumulations of N that might otherwise cause problems with nitrate contamination of groundwater due to leaching. However, many studies in other parts of the world (e.g., the U.S. and Europe) have clearly documented that this N-based management strategy will eventually create excessive accumulations of P (Sims 1997, Sims et al. 2000) because, as mentioned previously, the N:P ratio in most livestock manures is lower than the N:P ratio in the harvested portion of crops. In Manitoba, the ratio of N:P in the harvested portion of crops is approximately 5:1 and 10:1 for common annual crops and alfalfa, respectively (Table 1.6). Therefore, as mentioned earlier, the N:P ratio in the harvested portion of crops is significantly greater than the N:P ratio of 3:1 or less in most livestock manures (Table 1.2).

In Manitoba, the amount of P removed by the harvested portion of above average yields of typical crops is approximately 15 lb P/ac for common annual crops (e.g., wheat, oats, barley and canola) and 25 lb P/ac for alfalfa (Table 1.6). Therefore, over the long term, these rates of P removal should be used to establish maximum rates of manure P application rates (Sharpley et al. 1999). Failure to do so will result in a build-up of P to very high concentrations in areas of intensive livestock production, as already recorded in areas of Delaware (Sims 1997), the Netherlands (Sharpley et al. 1998, Sims et al. 2000) and Alberta (Whalen and Chang 2001).

Furthermore, although the rate of P buildup in soil may be rapid, the rate of P depletion by crops is relatively slow (Sharpley et al. 1998). Therefore, a soil with a high concentration of P requires many years of crop production to deplete (McCollum 1991).

Table 1.6. Nutrients removed by typical Manitoba crops (Source: Adapted from Manitoba Agriculture and Food Soil Fertility Guide and Manure Management Facts: Crop Rotations and Timing of Manure Applications)

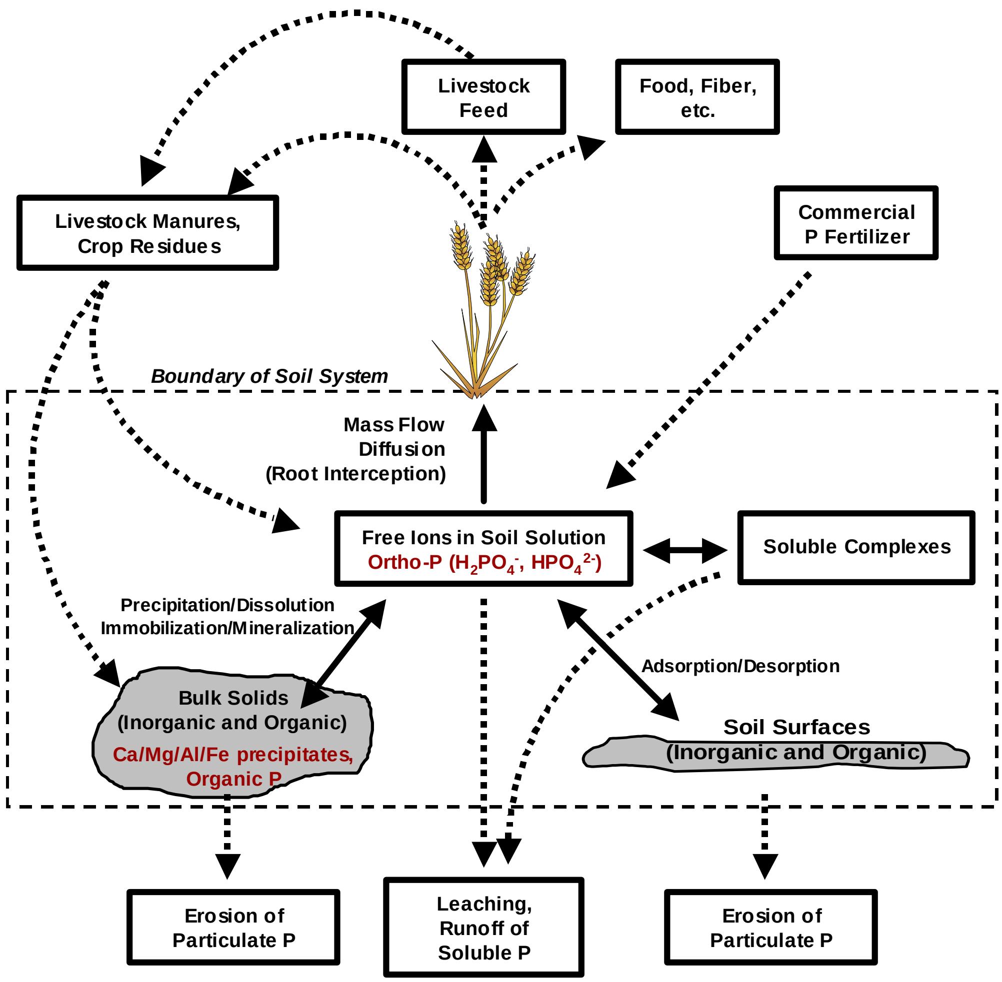
Crop	Yield	Crop Part	Nitrogen	Phosphorus[1]	N:P Ratio
			------ Lbs/acre ------		
Spring Wheat	40 bu/ac	seed	60	10	6.0
		straw	25	4	6.4
		total	85	14	6.1
Winter Wheat	50 bu/ac	seed	52	11	4.6
		straw	15	2	6.9
		total	67	13	5.0
Barley	80 bu/ac	seed	78	15	5.3
		straw	28	4	7.2
		total	106	19	5.7
Oats	100 bu/ac	seed	61	11	5.4
		straw	45	7	6.9
		total	106	18	5.9
Rye	55 bu/ac	seed	59	11	5.4
		straw	33	9	3.6
		total	92	20	4.6
Corn (grain)	100 bu/ac	seed	97	19	5.1
		straw	56	8	6.8
		total	153	27	5.6
Corn (silage)	6 t/ac	total	156	27	5.7
Canola	35 bu/ac	seed	68	18	3.8
		straw	44	7	6.0
		total	112	25	4.4
Flax	24 bu/ac	seed	51	7	7.8
		straw	14	1	10.7
		total	65	8	8.3
Sunflowers	2000 lb/ac	seed	53	7	7.6
		straw	21	4	4.8
		total	74	11	6.5
Peas	50 bu/ac	seed	117	15	7.7
		straw	36	3	10.4
		total	153	19	8.2
Soybeans	25 bu/ac	seed	100	9	11.5
		straw	32	2	18.4
		total	132	10	12.7
Potatoes	300 cwt/ac	tubers	96	12	7.9
		vines	75	10	7.8
		total	171	22	7.9
Alfalfa (forage)	4 t/ac	total	232	24	9.7
Clover (forage)	4 t/ac	total	215	24	9.0
Grass @ 10% CP	3 t/ac	total	103	13	7.9
Grass @ 20% CP	3 t/ac	total	200	13	15.3

[1] Expressed as elemental P (not as P_2O_5 equivalent)

4 Retention of Manure Phosphorus in Soil

Although P is generally regarded as relatively insoluble and strongly attracted to soil particles, a very small amount of P movement into water bodies can cause significant environmental harm. Therefore the processes that affect transformation, retention and release of P are very important for assessing the risk of environmental harm from application of manure P (Figure 1.2).

Figure 1.2. Overall nutrient cycle for P in crop and livestock production systems



4.1 Mineralization and Immobilization of P

The phosphorus in manure is initially present in both organic and inorganic forms and the balance between the two forms varies substantially with source of manure. The organic P fraction is not immediately available to crops, but becomes available as a result of mineralization, the process whereby microorganisms convert the organic P into inorganic P. One of the dominant forms of residual organic P from livestock manure is phytic acid. As mentioned previously, phytic acid is a major component of the P content in animal feed grains and is very stable, resisting decomposition in the digestion process and persisting in the soil, as well.

The microorganisms that mineralize organic P and immobilize inorganic P are active in soil and in manure storage systems. Therefore, the fraction of manure P present in the organic form changes with time in both environments. However, the rate of transformation is highly variable, depending on the form of P in the manure and the environmental conditions. Immobilization of inorganic P from livestock manure into organic forms may be a significant mechanism of P retention on land in which soil organic matter concentrations are increasing due to a change in cropping system or management (Tunney et al. 1997). For example, this type of retention may occur during the initial period after livestock manure is applied onto nutrient depleted pasture or hayland, as soil organic matter concentrations rise.

4.2 Precipitation and Dissolution of P

The availability of inorganic P from soil, fertilizer, or manure also varies with the form of the inorganic P and the capacity of the soil to precipitate P. The dominant form of P precipitation in acid soils ($pH<5$) is assumed to be with Fe and Al to form relatively insoluble Fe and Al phosphate compounds such as strengite and variscite, respectively. In neutral and calcareous soils, such as those that predominate in Manitoba, the dominant form of P precipitation is assumed to be with Ca and Mg to form compounds such as dicalcium phosphate dihydrate (DCPD) and octacalcium phosphate (OCP) (Havlin et al. 1999). For example, in a study in Saskatchewan, Qian and Schoenau (2000) reported that a large proportion of the P in a single application of liquid hog manure was precipitated as Ca-P. However, in other studies with P in calcareous soils, Fe and Al have been identified as playing an important role in P retention (Beauchemin and Simard 1999).

Across a wide range in soil pH, P can be precipitated by free Fe, Al, Ca and Mg that may originate from the soil's cation exchange. Therefore, the capacity of a soil to precipitate P may, in fact, be related to its cation exchange capacity (CEC). For example, Leclerc et al. (2001) used multivariate analysis to show that, indeed, P retention capacity was strongly related to soil CEC for calcareous and non-calcareous soils in Quebec. However, the exact mechanism responsible for this relationship was not determined and was ascribed simply to "sorption." Overall, in the dataset used by Leclerc et al., the soil's CEC was the best predictor of P sorption capacity, accounting for 52% of the variability in the P sorption index values for the calcareous soils (Simard 2001).

4.3 Adsorption and Desorption of P

Regardless of soil pH, all mineral soils contain aluminum and iron oxides/hydrous oxides that occur as discrete particles or surface coatings on other particles (Morgan 1997). These compounds are very effective at adsorbing P from soil solution. Since adsorption is a surface reaction, soils with a high specific surface, such as those with high concentrations of clay and organic matter are likely to have a high capacity to retain P. Such soils are also likely to have high CEC, offering another possible explanation for why CEC was so highly correlated with P retention capacity in recent studies by Leclerc et al. in Quebec (2001).

In addition to the oxides, calcium carbonates (lime) can also provide a suitable surface for adsorption of P. Therefore, soils with high concentrations of calcium carbonates, such as those that are common in Manitoba, are often assumed to have a high capacity for P retention (James et al. 1996, Manitoba Agriculture 1998). However, studies at the University of Manitoba in the early 1960s showed that calcium carbonate played a minor role in the retention of added fertilizer P (Weir and Soper 1963; Soper and El Bagouri 1964). Using the Chang and Jackson fractionation procedure, Soper and El Bagouri also showed that 47-88% of the P fertilizer that was mixed into the soil was extracted by ammonium fluoride and, therefore, appeared to be retained by aluminum and that calcium carbonate content did not appear to affect the extractability of recently added P. Scientists at the Brandon Experimental Farm also reported that the largest increase in extractable P from annual applications of fertilizer P was found in the Al-P fraction, extracted by ammonium fluoride (Research Branch - Canada Department of Agriculture 1964). However, in a later review, Art Toews noted that one of the reasons for the apparently high proportion of so-called "Al-P" extracted by ammonium fluoride in El Bagouri's studies may have been because El Bagouri did not use ammonium chloride to remove the most water soluble and loosely bound P from the soil prior to extraction with ammonium fluoride (Toews 1965). Also, as noted by Nelson and Logan (1983), the assumption that a specific extractant can selectively remove a particular form of P from soil is highly suspect, given the complexity of P chemistry and mineralogy in soils.

Although the issue of whether or not the P was retained as Al-P in El-Bagouri's and Weir's studies is not clear, their observation of the minor role of carbonates in P retention has been recently duplicated. For example, for soils in Quebec, Leclerc et al. (2001) found that the P sorption index values in carbonated soils were similar or lower than the values in non-carbonated soils within the same textural class. In another Quebec study, Mnkeni and MacKenzie (1985) found that adding calcium carbonate to topsoil reduced retention of polyphosphate and had no effect on orthosphosphate retention.

4.4 Measuring P retention capacity in soil and degree of saturation

In general, soils with low concentrations of clay, organic matter, carbonates, and Al/Fe oxide have low retention capacity for P. A group of soils that may differ from this general trend is organic soil, where very high concentrations of organic matter resulted in very high concentrations of P in drainage water (Miller 1979).

Many scientists have attempted a simple approach to predict the risk of P movement using a single measure of soil test P (e.g., by agronomic soil test extractants with acid, base, water, or weak calcium chloride solution). However, although the relationship between soil test P and P loss may be highly correlated within a single type of soil, the relationship breaks down when several different soils are included in the analysis, resulting in a series of correlations with different slopes and intercepts (Sharpley 1995, Sharpley et al. 1999). The main problem with using the simple soil test P approach is that only labile P is measured and the soil's overall capacity for P retention is not considered.

In the 1980s and 1990s scientists developed a practical tool to predict P loss at a point or small plot scale in a variety of soil types, based on the percentage of P adsorption sites that were occupied by P (Sharpley 1995, Sharpley and Rekolainen 1997). Dutch soils, for example, are considered to be saturated and susceptible to excessive P loss when more than 25% of their P sorption capacity is occupied (Sharpley et al. 1998). However, in Quebec soils, where the limit for P concentrations in surface water is lower than in the Netherlands, the critical degree of P saturation appears to be much lower, at approximately 9-15% (Beauchemin et al. 1996, Khiari and Parent 2000, Khiari et al. 2000).

Phosphorus saturation indices also vary in methods of determination. In the the non-calcareous soils of the Netherlands, the P saturation index is defined by the ratio between P and Al+Fe extracted by ammonium oxalate. However, in Quebec, the ratio of Mehlich III extractable P to Al has been developed as a suitable measure of P saturation index for a wide variety of soil types (Beauchemin and Simard 1999, 2000).

Some scientists have attempted to describe the relationship between P saturation in soil and concentrations of P in soil solution by using large number of measurements to develop quantity-intensity equilibrium models such as the Freundlich and Langmuir adsorption isotherm equations. However, these types of models are extremely laborious and expensive to develop and have also been criticized for describing the rapid process of P adsorption better than the relatively slow process of P desorption (Nelson and Logan 1983).

In the early 1970s, Bache and Williams (1971) adapted the concept of P sorption isotherms to create a relatively simple, single measurement technique to estimate P sorption capacity or the "P sorption index" or PSI. In this method, the quantity of P adsorbed from a relatively large quantity of P in solution P is measured, along with the concentration of P remaining in the solution at equilibrium. The P sorption index is calculated as the amount P that is sorbed divided by the log of the concentration of P in solution at equilibrium. Subsequent studies by Mozaffari and Sims (1994) showed that a slightly modified version of Bache and Williams original P sorption index had the potential to be a reliable and easily used method to rank soils with respect to P sorption capacity. Nair et al. (1998) have also successfully used a variation of this method. As a result, the P sorption index is frequently used to assess the capacity of a soil to retain P (Beauchemin and Simard 1999, Beauchemin and Simard 2000, Leclerc et al. 2001, Pautler and Sims 2001).

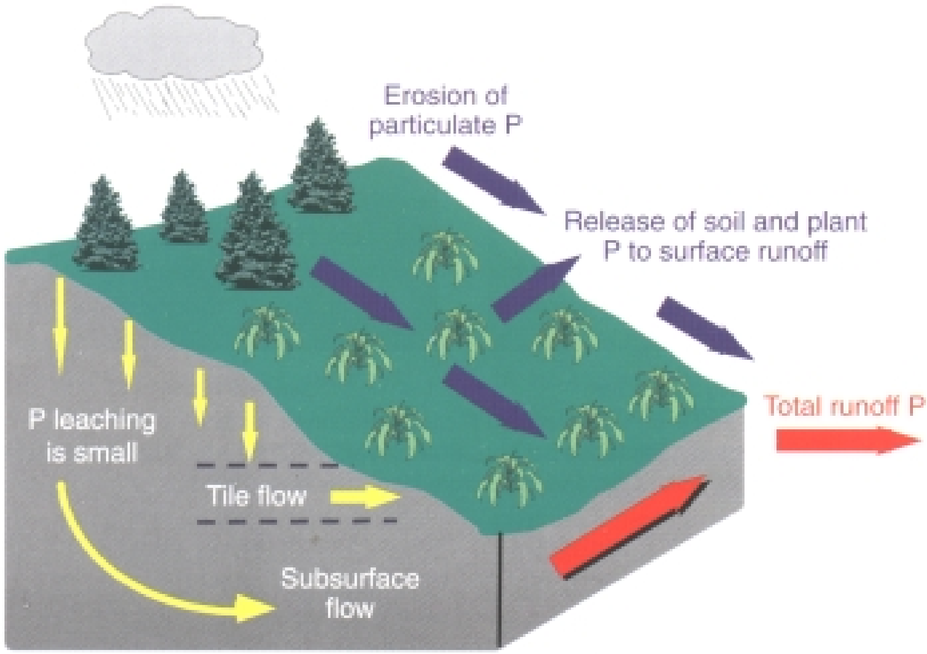
Regardless of the method that is used to characterize the P retention capacity or degree of saturation, however, it is important to remember that transport factors related to the landscape and climate must also be included in any regional assessment of P contamination risk (Sharpley et al. 1993, Sharpley et al. 1999, Sharpley et al. 2001, Sims et al. 2000) especially when attempting to predict P contamination at the watershed scale (McDowell et al. 2001).

5 Phosphorus Transport to Water

Haygarth and Jarvis (1999) have recently published an excellent detailed description of the mechanisms of P transport. However, from a simplified perspective, the main processes that move P into water bodies are as follows (Figure 1.3):

- release of P from eroded particulates that enter water bodies as sediment or suspended particles. When the P-enriched particles reach a water body, they release precipitated or adsorbed P because the concentrations of P in most water bodies is much lower than that in soil solution.
- soluble P dissolved in surface runoff. The runoff from manured fields can contain significant amounts of dissolved phosphorus, particularly when manures have not been injected or incorporated into the soil following application.
- flow of suspended particulate P or dissolved P from groundwater contaminated by P leaching. This pathway has received very little attention in the past due to the erroneous assumption that P was held so strongly by the soil that it was not leached. However, recent studies have shown that high concentrations of P in soil, especially in the form of organic P, create the potential for significant leaching of P into groundwater.

Figure 1.3. Phosphorus can be released from soil and plant material to surface and subsurface runoff water or lost by erosion. (from Sharpley et al. 1999).



Factors that increase the amount of P lost by erosion and runoff include:

- slow water infiltration rate due to fine soil texture and/or poor structure due to compaction from tillage or livestock
- climate and weather factors such as high rainfall amounts, intensities and durations; high snowfall and rapid snowmelt conditions
- soil and crop management factors such as intensive tillage systems, annual (vs. perennial) vegetation on fields and/or in riparian buffer zones
- landscape factors such as steep slopes and close proximity to surface waterways

The beneficial effects of livestock manure on water infiltration and reduction in water flow across the soil surface may help in some cases to reduce total P losses by erosion and runoff compared to land where similar amounts of P have been applied as synthetic fertilizer (Edwards and Daniel 1994, MacLean et al. 1983; Simard et al. 2001). Therefore, measurement of P dynamics in soil by chemical analysis, alone, is not a reliable predictor of the impact of P application on the risk of water contamination (Simard et al. 2001).

5.1 Erosion of Particulate Phosphorus

Rainfall and snowfall have negligible to low concentrations of P, but both provide a vehicle for moving P into surface water bodies either directly, through transport of dissolved P, or indirectly, through erosion and the transport of soil phase P that has precipitated or adsorbed to soil particles.

Particulate P is defined operationally as that form of P that cannot pass through a 0.45 μm pore diameter membrane filter (Nelson and Logan 1993). Some of the particulate P is intimately associated with soil and moves through the landscape by the processes of detachment and erosion; some of the particulate P may be intact fertilizer or manure that is moved by "incidental" processes if runoff occurs soon after application (Haygarth and Sharpley 2000). In the U.S., Sharpley and his co-workers estimate that erosion of particulate P accounts for 60 to 90 percent of P transport to surface waters from cultivated land (Sharpley et al. 1999). Conversely, most of the P transport from non-cultivated soils is in the form of soluble P (Kimmell et al. 2001, Sharpley et al. 1999).

The amount of P loading of waterways by particulate transport varies with the total concentration of P in the soil. However, transport of particulate P also varies with particle size. Small particles of mineral and organic matter move easier and further than large particles. Furthermore, smaller sizes of particles usually contain higher concentrations of nutrients, including P (Nelson and Logan 1983). Therefore, if all other environmental factors are similar, particulate P losses from fine-textured soils (e.g., clay or clay loam) will generally be greater than those from coarse-textured soils (e.g., sand). Another consequence of preferential behaviour according to particle size is that the P concentration in the eroded sediment is often two to six times greater than that in the original bulk soil (e.g., enrichment ratios of 2 to 6, as reported in Nelson and Logan 1983).

Runoff losses of particulate P are usually predicted by an equation or model developed for water erosion (e.g., the original or modified Universal Soil Loss Equation (USLE) or the Water Erosion Prediction Project model (WEPP)). The former equation has been used in a recent theoretical examination of P loss in Manitoba landscapes (Ecomatters Inc. 2000), while the latter is being used in studies of P loss in Alberta (Olson and Paterson 2001).

Snowfall has much less kinetic energy than rainfall; therefore, snowmelt might be expected to cause less erosion than rainfall. However, Finnish studies showed that the proportion of total annual erosion caused by snowmelt is greater than the proportion of total annual precipitation that comes in the form of snow (Rekolainen et al. 1997). The relatively high rate of erosion caused by snowmelt is probably due, in large degree, to the impermeable state of the frozen soil during the snowmelt period and the consequent high runoff volumes. Similar results have also been observed in North America, although the distinction between particulate P and dissolved P was not always specified. 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 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. Under Manitoba conditions, Green (1996) found that the surface application of manure to frozen soil resulted in increased phosphate concentrations in spring runoff relative to control fields. In another study in 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.

5.2 Runoff of Dissolved Phosphorus in Surface Water

Although erosion of particulate P has been traditionally regarded as the dominant form of P loss from cultivated fields, recent studies have shown otherwise. For example, recent studies in Alberta have shown that soluble P in runoff may be the dominant form of P loss in the Canadian Prairies (AAFRD Soil P Technical Working Group 2001). The balance between erosion of particulate P and runoff of dissolved P varies substantially from one situation to another, even for measurements on the same watershed area (Nelson and Logan 1983). For example, Heathewaite (1997) reported that dissolved P comprised 80% of total P losses from an initial fall rainstorm event and only 30% of the total P losses from the subsequent rainstorm event on the same catchment in Great Britain.

Dissolved P is defined operationally as that P that passes through a 0.45 μm pore diameter membrane filter at < 60 cm Hg and may include some fine particles of colloidal P (Haygarth and Sharpley 2000, Nelson and Logan 1983, Simard et al. 2001). In other words, the common term of dissolved P is inaccurate, since not all "dissolved" P is truly dissolved (Haygarth and Jarvis 1999).

The "dissolved P" exists in a variety of forms from a variety of sources and is highly, but not completely bioavailable. Some of the dissolved P is soluble inorganic P derived from inorganic minerals, fertilizers and mineralized organic P. Some of the dissolved P is soluble organic P from soil organic matter and organic amendments such as livestock manure. This dissolved organic P is generally regarded as more mobile than the inorganic forms of dissolved P (Brookes et al. 1997, Chardon 1997). The relatively high mobility of organic P may explain why several long term studies have shown deeper P leaching from manure than from synthetic fertilizers (MacLean et al. 1983, Eghball et al. 1996, Zheng et al. 2001) and why P leaching has been observed under feedlot conditions in Manitoba (Campbell and Racz 1975). The balance between the inorganic and organic portions varies substantially with the type of manure (Heathwaite 1997).

Most of the P in dissolved P is in the highly bioavailable form of orthophosphate, but some of the P that is measured in this fraction is also colloidal P or polyphosphate P that may be inadvertently hydrolyzed into orthophosphate P during the process of using the molybdate blue colorimetric analysis for P (Haygarth and Sharpley 2000, Nelson and Logan 1983). Therefore, dissolved P is often partitioned on an more exact basis into subfractions including reactive P (or Mo-reactive P), unreactive P and total P. In some cases the degree of biological availability may also be determined, using a bioassay, instead of a chemical analysis (e.g., algal available P (Haygarth and Sharpley 2000)). As a result of the diversity and sensitivity of P forms in the soluble fraction, no single chemical extraction or indicator perfectly describes the bioavailable portion, as it would behave under natural conditions (Rekolainen et al. 1997).

A significant portion of the dissolved P in runoff water also comes as aqueous leachate from living and dead plant material (White 1973, Sharpley et al. 1981, Nelson and Logan 1983). Dissolved P in snowmelt runoff is often particularly high, probably due to the solubilization of plant residue P during freezing and thawing and the extensive saturation period for extraction of P from surface soil (Rekolainen et al. 1997). The relatively high contribution of plant residue P to surface runoff has, therefore, been accounted for with relatively high weighting factors in the model used by Bolinder et al. (2000) for assessing the risk of P contamination in Quebec.

From a management perspective, losses of dissolved P are much more difficult to reduce than losses of particulate P. Traditional management practices for controlling particulate P such as establishment of grassland and perennial forage crops may not always be effective for controlling dissolved P. For example, permanent vegetative cover is generally excellent for reducing erosion of particulate P, but this practice probably has much less effect on reducing the loss of soluble P.

Due to the difficulty of controlling runoff, itself, the most effective management practices to reduce dissolved P lost in runoff are focused on preventing the excessive accumulation of soil P in areas of the landscape where runoff is likely to occur (Sibbeson and Sharpley 1997). Such measures include eliminating application of manure or fertilizer P onto steep slopes and/or frozen or snow-covered soil and careful selection of livestock watering and winter feeding areas, for examples.

5.3 Leaching of Phosphorus Below Root Zone

In the Canadian Prairies, loss of P by leaching is probably not extensive and is highly concentrated spatially and temporally, in a few areas of the landscape and during a few snowmelt or rainfall events. Until the 1980s the problem of P leaching did not attract much attention (Brookes et al. 1997; Nelson and Logan 1983), although Canadian researchers documented several cases of P leaching in the 1970s in Eastern and Western Canada (Bolton et al. 1970; Campbell and Racz 1975). More recently, Joan Rodvang and her colleagues found 0.39 mg P L^{-1} in shallow groundwaters in Southern Alberta, near an area of intensive livestock production and irrigated crop production (Rodvang et al. 2001). This concentration of P is nearly eight times the limit of 0.05 mg P L^{-1} recommended for rivers and streams in Canada, by the Canadian Council of Ministers of Environment. As a result of the increased awareness and concern of the short and long term impact of P leaching around the world, several excellent reviews have been published on this issue (Simard et al. 2000, Sims et al. 1998).

One mechanism for P leaching is due to preferential or bypass flow of water and P through tile drains, cracks in clay soils and old root channels, earthworm tunnels, and animal burrows (Simard et al. 2000, Sims et al. 1998, Stamm et al. 1998). In the case of preferential flow, the water flow is sufficiently rapid and the channels are sufficiently large that dissolved or suspended particulate P can avoid retention by the soil surfaces. The balance between dissolved and particulate P is variable; however, in recent studies in the UK, Heathewaite (1997) found that particulate P accounted for up to 90% of the leached P under intensive storm conditions. Subsequent research in Britain has confirmed the important role that particulate P may play in leaching, especially in sandy soils (Turner and Haygarth 2000). Tile drains are generally assumed to dramatically increase the risk of P transport to surface water by providing a rapid lateral path for flow of leachate. However, tile drains may also serve an important role as "early warning detectors" for a contamination problem that poses an undetected, but equally severe long term risk in undrained areas (Heathwaite 1997).

Another mechanism for P leaching is matrix flow of P due to accumulation of P in excess of the soil's capacity for retention, especially in sandy soils. In these situations, most of the leached P is assumed to be soluble P, present in relatively low concentrations due to the soil's ability to retain P, mostly via adsorption reactions. As a result, at least part of the relationship between phosphorus accumulation and risk of leaching can be predicted from knowledge of P adsorption or saturation behaviour (Schoumans and Groenendijk 2000). However, although retention generally keeps the concentration of P in the leachate is low, the system is highly buffered and, therefore, persistent. Therefore, the problem of P leaching requires a long time to develop or detect, but also a very long time to correct. As a result, some scientists regard any theoretical remedy for such a problem, once developed, as nearly impossible or impractical to implement (Brookes et al. 1997). The long delay between the time of application and the adverse environmental impact has led some researchers to conclude that leaching of P is, therefore, an environmental "time bomb" (Moore 1998).

One of the most important factors governing whether or not significant leaching of P will occur as non-preferential flow is the degree to which the soil's P retention capacity has been saturated. Experiments in England showed that P leaching "broke through" the P retention capacity of a variety of clay loam and clay soils at approximately 60 mg Olsen extractable P per kg of soil (Brookes et al. 1997, Hesketh and Brookes 2000). Other factors that increase the risk of this type of P leaching include:

- high water infiltration rate and low water holding capacity due to sandy or peaty soils and loose structure
- climate and weather factors such as high rainfall amounts, intensities and durations; high snowfall and rapid snowmelt conditions
- soil and crop management factors such as intensive tillage systems, annual (vs. perennial vegetation) on fields and/or in riparian buffer zones, and irrigation management
- landscape factors such as steep slopes and poor surface drainage

As mentioned previously, soils with high concentrations of calcium carbonates are sometimes considered to have a high capacity for P retention (James et al. 1996, Manitoba Agriculture 1998). However, the conclusion that carbonated soils have a huge capacity for retaining P is not consistently supported in research data. For example, James et al. (1996) reached the conclusion that there was "no practical limit to the P-retention ability of these calcareous soils" in spite of the fact that they found elevated concentrations of soil test P to a depth of up to 210 cm on land with a history of heavy applications of beef or turkey manure.

Organic P in manure appears to pose a greater risk of leaching loss than inorganic P (James et al. 1996). The mobility of organic sources of P in Manitoba soils has already been documented in a beef feedlot situation (Campbell and Racz 1975). European researchers have also observed significant P movement through manured soil and have predicted a breakthrough of high P concentrations in groundwater within 20-30 years under continuous large loadings of manure (Smith et al. 1998). Long term studies at Rothamstead in the UK showed that P in manure appeared to be more soluble and more susceptible to leaching than comparable treatments with synthetic fertilizer P (Steen 1997, Johnston and Poulton 1997, Brookes et al. 1997). Similarily, recent studies by Zheng and Simard (2001) in Quebec showed that liquid dairy manure produced three times as much labile P per unit of P added in surplus of crop requirements than did mineral fertilizer. In addition, organic soils appear to be more susceptible to P leaching than mineral soils (Miller et al. 1979, Brookes et al. 1997).

Measures for control of P leaching include prevention of excessive accumulation of P in the soil, improved surface drainage, and planting crops that utilize large amounts of water and nutrients. Fortunately, many of these measures are similar to those that can be used to control erosion of particulate P and runoff of dissolved P. However, some practices that decrease leaching may increase surface losses (e.g., improved surface drainage).

6 Estimating the Overall Risk of Water Contamination by Phosphorus

As stated earlier, any long term strategy for sustainable livestock production must eventually aim to match application rates to crop removal of P. However, most of the agricultural land in Manitoba is naturally low in P and will benefit agronomically from the application of P in manure and a modest buildup of P to higher levels. Furthermore, many of Manitoba's soils are can probably accommodate a modest increase in P concentration without imposing a major threat to environmental quality. Therefore, over the short term, application of P in moderate excess of crop removal may not create environmental problems on most soils in Manitoba.

However, applying P in excess of crop withdrawals should not be allowed unless objective thresholds are determined for the environmentally responsible limits of manure P (or any other form of P) application rates onto land (Sims et al. 2000). A variety of tools have been developed to help agricultural producers and regulatory agencies in this task. Generally, these tools incorporate the two groups of factors that account for movement of P from fields to water bodies: <u>source</u> factors such as nutrient application rate and retention capacity in the soil and <u>transport</u> factors such as risk of soil erosion, surface runoff and leaching.

6.1 Site-Specific P Indexes

A practical, semi-quantitative approach is to use knowledge of the source and transport factors that affect P loss to develop a comprehensive "P Index" that can be used to target P thresholds and management strategies on a site-specific basis. This concept, originally developed by USDA scientists in the early 1990s (Lemunyon and Gilbert 1993), has been widely accepted as a practical method of assessing the risk of P loss and target preventive and remedial management practices. The index is a semi-quantitative numerical rating tool that varies from one jurisdiction (e.g. state) to another, depending on the local environment and agricultural practices. The index is used to identify and rank source and transport factors, to identify where the risk of P loss is very high, high, medium, or low.

Source factors that are typically considered include soil test P, rate and method of commercial P fertilizer application, and rate and method of manure P application. Crop residue components may also be considered due to the tendency of decaying plant material to release P (Bolinder et al. 2000, Nelson and Logan 1983). Transport factors include estimates of soil erosion, irrigation erosion, soil runoff and distance from a water body. Contemporary versions of the index also include leaching as a transport factor (Gburek et al. 2000, Sharpley et al. 2001). Each factor is selected and weighted according to conditions for that region. The numerical ratings for each factor are then summed or multiplied to generate an overall index of the risk of P loss. This index is then used to determine appropriate management practices for the area in question.

One of the reasons why the P index has become a popular tool is that it is reasonably practical for use by field personnel and livestock producers, as well as being reasonably sound from a scientific perspective, once the source and transport factors have been calibrated to regional conditions. Another reason that the P index is popular is that it has proven to be a more accurate predictor of P transport to water than soil test P thresholds, alone (Sharpley et al. 2001).

If a site-specific P index was desired for use in Manitoba, that index should not be imported directly from elsewhere and would have to be adapted to Manitoba's soils, landscapes and climate. Unfortunately Manitoba does not currently have objective, quantitative and scientific data to use for developing the source and transport criteria for a P index. Very little research elsewhere has focused on runoff water from seasonally frozen semi-arid soils such as those in the Canadian Prairies. Similarly, very little research elsewhere has been conducted in "pothole" topography with closed basins (e.g., many landscapes in Western Manitoba), coarse textured soils over shallow, pristine aquifers (e.g., in the Upper Assiniboine Delta, Interlake and Southeastern Manitoba) and in flat, fine-textured soils (e.g., in the Red River Valley).

Recent research at the University of Minnesota, for example, has shown that the original P index required modification to accurately predict P losses to water bodies in most of that state. Those modifications to the P index improved the accuracy of the index throughout the eastern and southern regions of that state, but the modified index continued to underpredict P losses in the flat landscapes of the Red River Basin (Birr and Mulla 2001). The authors speculated that the inaccuracy of the original and modified P index for these landscapes was due to the dependence of those indices on an inaccurate water erosion model for fine-textured soils in nearly level landscapes. However, the index may also fail to account for the slow rate of water flow and the consequently long contact time for surface water to extract soil P and transport that P in a dissolved form in these nearly level landscapes.

Bolinder et al. (2000) have published one of the only studies in Canada where the phosphorus index has been used to systematically identify the risk of P contamination of water on a regional basis. In their study, Bolinder and co-workers used a version of the P index developed by McFarland et al. (1998) that focuses primarily on surface transport of P through runoff and erosion. Soil Landscape of Canada information at the soil polygon scale was used to develop the transport factors. The Quebec government's detailed information about soil test P concentrations and P saturation indices was used to provide reasonably detailed information on P source factors. Bolinder et al. concluded that their regional index appeared to be reasonably sensitive to real differences in nutrient management practices between two census dates and various locations.

During a recent conference in Winnipeg, one of the co-investigators in the study by Bolinder et al. (2000), Regis Simard, proposed a modified P index for the Prairies (Simard et al. 2001). One of the major differences between the proposed index and most others is the separation of source factors into charge and management factors. Simard's proposed charge factors include soil test P concentrations and degree of soil P saturation; management factors include the rate and method of application for fertilizer and manure P added and grazing intensity. The transport factors proposed by Simard include the traditional factors of erosion, surface runoff, and distance to a waterbody, plus new factors such as wind erosion, leaching by preferential flow and the risk of incidental, surface transfer of intact manure/fertilizer particles. Simard also proposed that the charge, management and transport factors be multiplied rather than summed, similar to the recommendations of Gburek et al. (2000).

6.2 Quantitative, Process-Based P Mobility Model

Site-specific P indexes are usually developed relatively quickly, using professional estimates of how various source and transport factors affect the risk of P movement in a particular area. However, these indexes are fairly subjective, very approximate and must be readjusted for different environments. For example, as mentioned earlier, the version of the P index that worked well in most of Minnesota performed relatively poorly in the Red River Valley region (Birr and Mulla 2001).

The ideal tool for predicting the risk of P loss would be a process-based model that could predict the quantity P loading to water from runoff and eroded soil. Such a model would be built on a fundamental mathematical understanding of the chemical, physical and biological mechanisms that affect P accumulation and movement under local environmental conditions. Researchers in the U.S. and Europe have proposed such a theoretical model (Cassell et al., 1998). In a similar manner, Alberta Agriculture, Food and Rural Development has initiated a study to mechanically predict phosphorus mobility in soil (Olson and Paterson, 2001). In the Alberta project, studies on P retention and release are being coupled to the Alberta "Water Erosion Prediction Project" and the "Watershed Assessment Study" to estimate the movement of dissolved inorganic P and total particulate P into water bodies.

Although the goal of such a process-based mechanistic model is noble, that goal is also very ambitious and requires a very large investment in fundamental and applied research. Furthermore, such models are often extremely complex and very difficult for field personnel and producers to use. As a result, the development of a process-based, mechanistic model for Manitoba conditions is probably unrealistic in the short term.

7 Managing Manure P in Crop Production

7.1 Agronomic Value of Manure

Livestock production and land application of the manure produced by those livestock has the potential to significantly improve the sustainability of prairie agriculture. In many areas of the prairies, the organic matter and nutrient content of prairie soils has been declining as rates of removal under annual cropping and grain export have outpaced the return organic matter and nutrients to the soil. Suitable application of animal manure provides an excellent source of nutrients and organic matter to replenish the productive capacity of soil.

Manure contains all of the major crop nutrients (nitrogen, phosphorus and potassium) as well as many, if not all of the minor nutrients needed by plants. Manure has many additional agronomic benefits beyond its nutrient content. For example, manure contains many organic compounds that help to build and maintain soil structure which in turn helps to reduce soil erosion and maintain or improve soil tilth, water holding capacity, water infiltration rates, and soil aeration (Eghball and Power 1998).

In addition to providing agronomic benefits, manure may provide environmental benefits, as well. Manure application onto cropland recycles nutrients locally, reducing the need for synthetic fertilizer and the environmental impacts of manufacturing and transporting that fertilizer. The application of animal manure also increases the organic matter content in some soils, thus reducing net CO_2 emissions to the atmosphere (increasing C sequestration).

As mentioned earlier, efficient and effective manure management balances the amount and timing of nutrient supply with crop demand, applying the correct amount of manure, at the right time, to the most appropriate crop in the most effective manner. Failure to adopt such a strategy not only squanders the benefits that manure can provide but is also harmful to the environment. A typical example of such a failure has occurred in Western England, for example, where recent "specialization and intensification of land use has led to a concentration of large livestock production systems. The increasing number of freshwater pollution incidents traced to agricultural sources suggests that the rate of intensification has not been matched by measures to effectively utilize the increased production of livestock manure" (Heathwaite 1997).

7.2 Opportunities and Limitations of Manure Testing

The largest challenge in testing manure for nutrient content is the huge variation in nutrient content as a function of species, type of housing, manure storage system, and position within the storage unit. For example, in a recent study of liquid manure samples from hog barns in Manitoba, concentrations of total N ranged from 6 to 65 lb/1000 gallons and concentrations of total P ranged from 0.3 to 55 lb/1000 gallons (Table 1.4). According to the results of this study, one of the main causes of this variability is due to the difficulty of mixing stored manure uniformly for sampling and application (Table 1.5). The concentrations of N and P were lower and the ratios of N to P were higher for samples taken from shallow depths in storage compared to those taken from the bottom of the storage.

In addition to the variability in nutrient concentration within different samples and loads of manure, there is also variability in the time pattern of nutrient release from different manures. All manure contains some organic forms of nutrients that must be mineralized by soil microorganisms before becoming available to crops. Some types of manure also contain bedding that may stimulate net immobilization of nutrients by soil microorganisms. In both cases, factors such as carbon to nutrient ratio, soil moisture, soil temperature, and tillage system will have a profound effect on how quickly the nutrients are released or retained.

7.2.7 Manure Phosphorus Testing and Availability

The P content of manure is measured simply as total P, without distinction for any particular form (e.g., organic vs. inorganic P). This lack of attention to the availability of the P in manure is inappropriate, given the agronomic and environmental importance of P as a nutrient for crops and algae (Maguire et al. 2001).

In some areas of the world, the P content of livestock manure is regarded as being 100% as effective as synthetic fertilizer; in other areas, manure P is regarded as being only 50% as effective (Tunney et al. 1997). Long term studies at Rothamsted over a 117 year period showed that concentrations of 0.01 M $CaCl_2$ extractable P were much higher in plots that received 40 kg P ha^{-1} as manure P than in those that received 33 kg P ha^{-1} as fertilizer P (MacLean et al. 1983). These results imply that the water solubility of reaction products from manure sources of P may be higher than those from commercial fertilizers. Research in the UK has shown that manure P and inorganic fertilizer P should be regarded as being equivalent in efficiency (Smith et al. 1998). Similarly, Eghball and Power (1998) cite research in the US where 20% of cattle manure P was utilized by crops during the first year after application, a level of efficiency similar to optimum performance from inorganic fertilizer sources. However, in Manitoba, the agronomic availability of P in swine manure is estimated to be 50% of commercial, water-soluble fertilizer P within the first year after application (Manitoba Agriculture 1998). This figure was not developed from research data and is based on the assumption that manure P will not be placed in or near the seed row for maximum efficiency of P use, as is done with commercial P fertilizers (G. Racz, 2001). No comprehensive studies to determine the form and bioavailability of manure

P in Manitoba soils have been completed. Recently, however, Dr. Wole Akinremi and Dr. Geza Racz, at the University of Manitoba, have initiated a study to begin examining this issue.

As mentioned previously, the efficiency of crop utilization of P from synthetic fertilizer is usually less than 20%. Therefore, only a small fraction of the manure or fertilizer P that is accumulated in soil is available to a crop in a given year. As a result, after a soil has accumulated a high concentration of P, whether from synthetic fertilizer or manure, many years of crop removal are required to reduce P concentrations to acceptable levels, even after no further applications of P (McCollum 1991).

7.3 Opportunities and Limitations of Soil Testing

7.3.7 Soil Testing for Phosphorus

The magnitude of P loss from agricultural land is more strongly influenced by residual soil P content than the annual rate of P application (Heathwaite 1997). Therefore, analyzing the soil for residual P is an important component of a sound manure management program. However, in many cases, unique sampling and analytical procedures are required. For example, the standard array of agronomic soil tests for P should be expanded for environmental purposes by adding new tests such as water soluble P, easily desorbable P, P sorption index and degree of P saturation (Sims et al. 2000).

Soil test methods that have been developed for agronomic evaluation of P fertility for crop production (e.g., conventional soil testing lab procedures using the “Kelowna” test or “Olsen” test) may be of limited use in determining the relationship between P accumulation and potential for P loss. Although higher P concentrations in soil fertility tests can often be correlated to higher potential for P loss within a soil type, that relationship is not consistent across different soil types and landscapes (Sharpley 1995, Sharpley et al. 1999). Therefore, a soil fertility test value of 50 mg/kg may indicate little risk of P loss in one soil type and a high risk of P loss in another soil type, especially when the soils may react quite differently if eroded into an aquatic environment. As a result, specialized soil tests have been developed to determine the risk of P loss. These tests usually measure the overall capacity of the soil to retain P, in addition to the amount of P that has already accumulated in the soil (please refer to section 4.4 for more information on soil P analyses). An additional problem with soil test P extraction methods is that many methods do not efficiently extract P that has been recently applied and which remains biologically available (Tunney et al. 1997). As a result of this inaccuracy, the apparent capacity of a soil to retain P may be overestimated by some soil test methods.

In order to estimate the potential for P loss, soil should be sampled at relatively shallow depths (less than 7.5 cm) for several reasons. Most of the transport of P to waterways is a result of the interaction between water and the surface of the soil. Also, due to the topsoil’s ability to retain P that is released from decaying plant roots and shoots, P concentrations in soil are often highly concentrated toward the surface. This surface enrichment is especially pronounced where manure is applied onto permanent forage or other reduced tillage systems. However, in landscapes where gully erosion is a significant problem, shallow soil sample depths may not be representative for estimating P losses from land to surface water (Nelson and Logan 1983). In addition, periodic and systematic deep sampling for P may help to identify where leaching of P may be occurring.

Soil tests must also be interpreted differently when used for assessing environmental risk compared to crop production fertility because the calibration criteria for the two types of tests are usually quite different. Fortunately for agricultural crop production, soil test P concentrations that are optimum for crop production are frequently lower than those that cause unacceptable P loss (Sibbeson and Sharpley 1997, Sharpley and Tunney 2000). In Idaho, for example, the agronomic optimum soil test P concentration for non-vegetable crops, using the Olsen test is set at 12 mg P/kg while the environmental threshold is set at 50 to 100 mg P/kg for sandy and silt loam soils, respectively (Sharpley et al. 1999). However, it is important to consider that calibration of the environmental threshold is highly specific to the influence of soil type, climate and landscape on source and transport factors.

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