

# **Impact of Soil Phosphorus Loading on Water Quality in Alberta**

A Review

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## Summary

This study has four primary conclusions:

- , **research has increasingly identified phosphorus pollution from municipal, industrial and agricultural sources as a major threat to water quality**
- , **pollution risk increases with higher levels of soil phosphorus, however site specific soil, hydrologic and management characteristics also have considerable influence on water pollution risk**
- , **surface waters in Alberta are extremely sensitive to further phosphorus enrichment**
- , **a desirable long-range goal for Alberta is a balanced nutrient management approach to land application of nutrient-bearing materials, but a site specific approach to soil phosphorus management would be suitable for addressing the more immediate need to protect water quality**

An increasing amount of research has identified phosphorus (P) pollution from a variety of sources, including municipal, industrial and agricultural, as a major threat to water quality. The Canada-Alberta Environmentally Sustainable Agriculture Agreement (CAESA) water quality assessment showed that phosphorus runoff from agricultural lands in Alberta is an issue that must be addressed. There is growing support to develop a strategy to manage phosphorus concentrations in the soil profile to minimize the risk of movement of phosphorus into surface and groundwater. There have been concerns raised, however that there is not enough science-based evidence to justify the implication that high soil P levels increase water pollution risk. This report examined the science-based evidence for the relationship between soil phosphorus and water pollution risk. It also investigated some ways soil P loading can be managed to minimize water pollution risk.

Concerns about the contribution of agricultural operations to nutrient enrichment of water bodies in Alberta has increased as the industry has intensified. Nitrogen and phosphorus often exceeded water quality guidelines for protection of aquatic life in streams in areas of high and moderately intensive agriculture. Phosphorus concentrations often exceeded water quality guidelines for protection of aquatic life in small lakes and irrigation canals in high intensity areas.

Alberta lakes, streams and rivers differ from water bodies elsewhere in Canada or the United States. Many lakes and other surface water bodies in Alberta are naturally highly productive (eutrophic) and experience prolific algae and weed growth. Phosphorus is the primary plant nutrient controlling the level of phytoplankton growth in western Canadian lakes. Consequently, surface waters in Alberta are extremely sensitive to further P enrichment by internal or external sources. Further nutrient enrichment of surface waters in Alberta can have significant economic and environmental implications on public and animal health, recreation, tourism and possibly even the agriculture industry.

The risk of P transport from land to water increases with P loading of soil. Research has shown that the concentration of dissolved P increases with higher soil test P levels. Soil P levels that would be common for many agricultural soils can, in some situations, produce dissolved P concentrations in runoff that are high enough to exceed water quality guidelines. Although the soil P level is a major factor in the risk of P export to the water system, other factors, such as the site specific nature of the soil, the type of management and the hydrologic nature of the site are critical to P export risk.

The most practical way to reduce high P levels in soils is through crop uptake, however this can be a slow process. The rate of decline varies with soil type, crop, and management practices. Research from the Northern Great Plains shows the rate of decline of soil P ranges from 0.6 to 30 ppm per year. Soils in the County of Lethbridge, Alberta that have undergone repeated heavy manure applications can typically have soil test P levels high enough that several decades could be needed to reduce P levels to moderate values.

A desirable long-range goal is to achieve balanced nutrient management planning, which requires that inputs be balanced with crop uptake. This is a strategy to manage loading from all nutrients and is comprehensive enough to address all nutrient loading practices, including those associated with manure application, food processing effluent application, and cropping practices. The United States is moving toward a comprehensive nutrient management planning approach to agricultural production and many states have started to address phosphorus loading. We have their experience to learn from as well.

It is recognized that the transition from current practices to managing inputs to balance crop use will be complex and require time. The process to regulate the intensive livestock industry, and the expansion of the food processing industry in Alberta have resulted in a need to develop P-based standards for Alberta soon. As an interim process, a site specific approach to soil phosphorus management such as the P index, as described in Section 4, appears to be the most suitable. The P index is a practical method that will provide the most flexibility for operators, while minimizing risk to the environment. It must be designed so that it can be integrated into a watershed scale assessment.

## **1. Introduction**

Alberta's agricultural industry has expanded significantly in the last 25 years. Projections for growth were to reach 10 billion dollars in the primary and as much as 20 billion dollars in the value-added sector by the year 2005. Much of the value-added sector contribution has been in the intensive livestock industry.

Concerns about the contribution of agricultural operations to nutrient loading of water bodies in Alberta have increased as the industry has intensified. A recent water quality assessment (Canada-Alberta Environmentally Sustainable Agreement 1998) showed that nitrogen (N) and phosphorus (P) often exceeded water quality guidelines for protection of aquatic life in streams in high and moderately intensive agriculture areas. In addition, phosphorus often exceeded water quality guidelines for protection of aquatic life in small lakes and irrigation canals in high intensity areas.

One of the primary concerns regarding nutrient loading has been disposal of manure and food processing effluent. Food processing and intensive livestock operations require an adequate land base for disposal and it is often not economical to broaden the land base by hauling manure and effluent long distances. Runoff from agricultural land, particularly those areas enriched by nutrients from long-term manure application, has been identified as one of the major sources of non-point source pollution in other agricultural areas. Concern has particularly focused on phosphorus levels in runoff because of the sensitivity of aquatic systems to phosphorus concentrations.

In Alberta, the intensive livestock regulatory process and the expansion of the food processing industry have resulted in a need for leadership by Alberta Agriculture, Food and Rural Development in establishing acceptable guidelines and standards for organic-based P materials. There is a need to develop a plan to manage loading phosphorus in the soil that would allow intensive livestock operations, food processing and other industries to plan future development and expansion in a manner that will protect the quality of surface water.

The objective of this report was to examine the science-based evidence for the relationship between soil phosphorus and water pollution risk, and to present options for development of a soil P management strategy in Alberta.

### **1.1 Terms Used in this Report**

Phosphorus concentrations can be expressed in several ways, and no standard for the terms of expression has been adopted. Phosphorus is usually expressed as concentration of elemental phosphorus in soils and water, and as  $P_2O_5$  in commercial fertilizers. It can also be expressed as orthophosphate ( $PO_4^{-3}$ ). Phosphorus concentrations in soil are often expressed as  $mg\ kg^{-1}$  of soil, which is the same as parts per million (ppm). Phosphorus content in water, because of the sensitivity of water to low concentrations, is often reported as  $\mu g\ L^{-1}$ , which is the same as parts per billion (ppb).

In this report phosphorus will be expressed as ppm of elemental P. The relationship of some of the common reporting terms are presented in Table 1.1.

**Table 1.1 Conversions factors for some common phosphorus reporting terms**

$\mu\text{g L}^{-1}$	= ppb
$\text{mg L}^{-1}$	= ppm
$\text{mg kg}^{-1}$	= ppm
ppb	= 1000 * ppm
$\text{P}_2\text{O}_5$	= P * 2.2914
P	= $\text{P}_2\text{O}_5$ * 0.4364

Nutrient concentrations in soil are often expressed in terms of  $\text{kg ha}^{-1}$  for a given depth within the soil profile. To determine the nutrient concentration, the soil depth of interest, and the soil bulk density must be known or have assumed values. A common soil depth for nutrient analysis is 15 cm. For example, converting 100 ppm of P to a nutrient content in the top 15 cm (0.15 m) of the soil would require the following steps.

1. Soil bulk density assumed to be  $1.4 \text{ g cm}^{-3}$
2.  $100 \text{ ppm} = 100 \text{ mg kg}^{-1}$
3.  $(100 \text{ mg kg}^{-1} * 10^{-6} \text{ kg mg}^{-1} * 1.4 \text{ g cm}^{-3} * 10^{-3} \text{ kg g}^{-1} * 10^6 \text{ cm}^3 \text{ m}^{-3} * 10^4 \text{ m}^2 \text{ ha}^{-1} * 0.15 \text{ m depth})$

$$= 210 \text{ kg ha}^{-1}$$

## 2. Phosphorus in Soil, Manure and Organic Waste

### 2.1 The Phosphorus Cycle

Phosphorus exists in inorganic forms and organic forms within the soil. Phosphorus content varies with soil parent material, texture, and the management system used to input and remove P from the soil. Inorganic forms of P tend to dominate agricultural soils, typically at 50-75% of total P, but can range from 10-90% (Sharpley and Reikolainen 1997). Inorganic forms are found either adsorbed or absorbed to mineral fractions dominated by compounds of either aluminum (Al) or iron (Fe) in acidic soils, or calcium (Ca) in calcareous, alkaline soils. Organic phosphorus is generally in relatively labile nucleic acids, phospholipids, inositols and fulvic acids or in less soluble humic acids.

Figure 2.1 is a conceptualized soil phosphorus cycle (Chauhan et al. 1981). The cycle illustrates the relationships of “slow” (slowly available) inorganic and organic forms of soil P with the “rapid” (readily available) forms. Slow inorganic forms include primary and secondary minerals (e.g. calcium, iron and aluminum phosphates), as well as unavailable (occluded) forms. The slow organic fraction consists of chemically and physically protected organic P compounds that are usually adsorbed onto Al, Fe and Ca surfaces. They can be mineralized either as soil organic matter is mineralized, or by specific enzyme activity as a response to demand by microbes and plants.

Primary minerals in the soil parent geologic material dissolve through weathering and enter the solution pool. The solution P maintains an equilibrium with the labile inorganic P ( $P_i$ ) fraction such that, in any one soil a capacity factor, a ratio of solution P:labile P, is maintained for concentrations of P that are normal for cultivated soils. The capacity factor varies among different soils

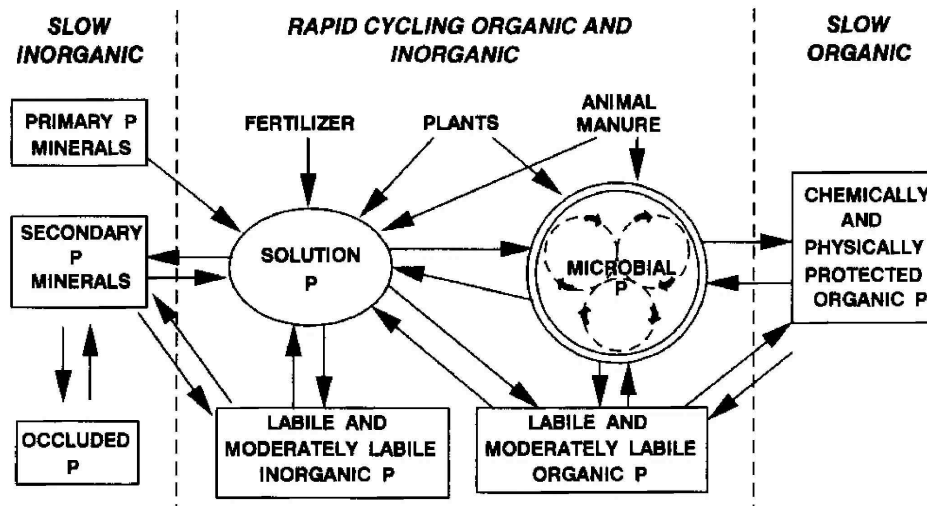


Figure 2.1 The soil phosphorus cycle and its measurable components (adapted from Chauhan et al. 1981; Stewart and Sharpley 1987).

depending on the physical and chemical properties of the adsorbing colloids, and can range from  $<10$  to  $> 1000$  (Stewart and Sharpley 1987). Concentrations in excess of the capacity factor result in precipitation of secondary minerals. In highly weathered soils, such as tropical soils, secondary minerals can become occluded.

Plants take up phosphorus directly from the solution pool (Barber 1984). Any depletion in the solution pool is immediately replenished from the labile fraction. If both the labile inorganic and solution pools are depleted, the secondary P minerals can move into the labile inorganic or the solution pools.

Microbial P, represented by the circling movements in Figure 2.1, originates from microbial uptake of P from the solution pool. This is stimulated by additions of substrates, such as plant residues. When microbial cells are ruptured, either through lysis or predation, P is released in both solution and labile organic ( $P_o$ ) forms, including RNA, phospholipids, acid soluble  $P_i$  and  $P_o$ , and DNA (Stewart and McKercher 1982). They can react with mineral and organic soil components to create a wide variety of different P compounds of different solubility or susceptibility to mineralization (Stewart and Sharpley 1987).

Phosphorus amendments, either organic or inorganic, are needed to maintain adequate available P in modern agricultural systems. A continuous drain on solution P from cultivation and crop removal will ultimately deplete the labile  $P_i$  and  $P_o$  pools. For organic amendments, the microbial activity has a central role in the P cycle, particularly by maintaining the labile P supply by replenishing the  $P_o$  pool. It is represented in Figure 2.1 as a “wheel” rotating in response to carbon (C) inputs. Should the wheel be stopped due to lack of C inputs or partial soil sterilization, the supply of P to plants becomes limited by the size of the existing labile  $P_i$  and  $P_o$  pools (Stewart and Sharpley 1987).

Further evidence of the important role microbes play in P cycling dynamics was presented when fumigation-extraction techniques were developed to measure soil microbial biomass. Brookes et al. (1984) measured microbial P fluxes in soils in continuous wheat and in grassland. In the continuous wheat, the microbial P fluxes were comparable to the P uptake by the crop, and in grasslands, the fluxes were higher than the P uptake by the grass. Sharpley and Reikolainen (1997) suggested that understanding the microbial contribution to soil P dynamics may impact management practices. They cited that the practice of maximizing organic matter build-up in the fall, which may result in short-term tie-up of available P for crops in the following spring. However this practice may result in enrichment of the surface layer with P, increasing the potential for higher P concentrations in spring runoff.

When P is applied to soil, there is generally an increase in the available P content, although the fate of applied P is not always the same as that of applied N. The magnitude of the increase is a function of soil properties such as clay content, organic C, Fe, Al, and calcium carbonate ( $CaCO_3$ ) content. Applied P can be taken up by the crop or converted to the tissue component of soil microflora and microfauna. However most of the applied P is either weakly or strongly adsorbed on to Al, Fe or Ca

compounds (Syers and Curtin 1988). Once the initial adsorption reaction has occurred, added P is gradually fixed (absorbed) to the compounds, resulting in a portion of the fixed P being unavailable for plant uptake (Sharpley and Reikolainen 1997). Although fixation was once considered irreversible, it is now known that P can be slowly released back into solution when available P levels are depleted. An estimated 25% of applied P is available to plant roots, whereas the rest is quickly adsorbed by the soil components and over time released for plant use (Morgan 1997).

Phosphatase enzymes associated with plant roots and microorganisms play a key role in P availability, particularly in the hydrolysis of organic P. There is a wide range of soil microorganisms that have the capability, through their phosphatase activities, to mineralize all known organic phosphates of plant origin. The phosphatase activity is strongly influenced by temperature. It is also influenced by moisture, pH and organic carbon (Tisdale et al. 1985).

Continued application of P in excess of crop demand will increase P levels to a point of environmental concern and possibly to agronomic concern. In the case of commercial fertilizers, economic factors dictate that care be taken to ensure that application rates do not exceed crop demands. In manure application however, the philosophy is often to use the land for manure disposal. In these cases, to reduce the transport costs, manure is often repeatedly applied to the same land base at rates where nutrients consistently exceed crop demand, and nutrient accumulation can lead to environmental risk.

## **2.2 The Phosphorus Content of Manure and Organic Waste Applied to Land**

Agricultural land is a sink for the application of a variety of materials, including production inputs (commercial fertilizers, pesticides, and irrigation water), and waste by-products (livestock manure, industrial and municipal wastewater, and sludge).

### **2.2.1 Manure**

Recovery of phosphorus is relatively efficient in crop production compared to animal production, with 56-76% of the input P recovered in crops, relative to 10-34% in animals (Sharpley and Reikolainen 1997). A variety of factors can affect the nutrient content of manure such as animal type, manure management system (solid or liquid), storage methods, amount and type of bedding used, application method, animal diet, composting techniques, and climate. As a result, the nutrient content and physical properties of animal manure are highly variable (Table 2.1). For example, DeLuca and DeLuca (1997) reported that the P content of beef feedlot manure, from values cited in the literature, ranged from 0.1 - 0.8%, averaging 0.4%.

There are no standard methods for testing the P content of solid animal waste; however a method proposed to the United States Department of Agriculture is a modification of the Olson method for soil P testing (Self-Davis and Moore 1998).

### **2.2.2 Municipal Wastes**

Municipal wastes include storm-drain water, liquid sewage with suspended solids, and solid material generated by households and businesses. Storm-drain water is usually diverted directly into a watercourse, such as a river. Solid material from households and businesses is taken to land-fill sites. Municipal sewage is treated before release into the environment.

Municipal sewage is about 99% liquid with suspended and dissolved organic and inorganic materials, and a large number of microorganisms. Sewage treatment facilities remove floating matter and grit, and reduce suspended solids, oil, grease, dissolved organic matter, dissolved nutrients, and microorganisms. Sewage treatment facilities generate two types of wastes: treated effluent and sewage sludge. The latter is also referred to as biosolids. The treated effluent is discharged to either

**Table 2.1 Nitrogen and phosphorus content (kg tonne<sup>-1</sup>) of manures from various type of livestock (Alberta Agriculture, Food and Rural Development 1999a; Intensive Livestock Operations Committee 1995).**

Type of Livestock		Moisture %	Total N	Total P	N/P	% P
Beef	Open Lot	50	10.0	2.4	4.2	0.24
Dairy	Free Stall	92	4.0	0.9	4.4	0.09
	Tie Stall	80	5.0	0.9	5.6	0.09
Hogs	All	96	3.5	1.1	3.2	0.11
Poultry	Layers - solid	60	16.0	12.2	1.3	1.22
	Layers - liquid	90	9.6	7.3	1.3	0.73
	Broilers	35	18.6	6.5	2.9	0.65
	Breeders	35	19.7	13.1	1.5	1.30
Turkey	All	35	18.6	6.5	2.9	0.65
Sheep	All	50	7.0	2.0	3.5	0.20
Horse			6.0	1.3	4.6	0.13

Values are based on averages from a variety of sources, and actual farm values may vary greatly.

land or water. Land application methods generally include irrigation (Alberta Environmental Protection 1997a), rapid infiltration (Alberta Environmental Protection and City of Red Deer 1981), and wetland disposal (Alberta Environmental Protection, date unknown). Municipal effluent generally contains between 2 and 6 mg L<sup>-1</sup> total P, 10 to 20 mg L<sup>-1</sup> total N, and 5 to 40 mg L<sup>-1</sup> total potassium (K) (Alberta Environmental Protection 1997a). Biosolid material, which is collected from settling ponds, typically contains about 10% solids and 90% water. Total P, total N, and total K content are about 4%, 5.5%, and 0.35% on a dry-weight basis, respectively (Barl and McKenzie

1995). The most common biosolid disposal methods are sanitary landfilling, incineration, permanent lagoon storage, and land application (Alberta Environmental Protection 1997b). Land application of biosolids is usually carried out by injection.

### **2.2.3 Food Processing Wastewater**

Food processing facilities often require large quantities of water of which a major portion becomes wastewater. This type of wastewater contains significant amounts of nutrients and organic material. Barl and McKenzie (1995) reported 42 to 72 mg L<sup>-1</sup> total P, 75 to 211 mg L<sup>-1</sup> total N, 466 to 707 mg L<sup>-1</sup> total K, and 1% total solids in potato processing wastewater. Wastewater can be diverted into existing municipal sewage treatment facilities if processing plants are within or near municipal infrastructures. However, food processing plants that do not have access to municipal services must develop alternative methods for handling wastewater. Land application is generally the most effective method of disposal. Wastewater is temporarily held in holding ponds, followed by irrigating onto selected parcels of land. An alternative method is to divert the wastewater to wetlands for processing.

## **2.3 Methods of Phosphorus Analysis**

Analytical techniques are targeted to identify five fractions of phosphorus: total P (P<sub>t</sub>), dissolved P (P<sub>d</sub>), particulate P (P<sub>p</sub>), biologically-available P (BAP), and plant available P. Water analysis techniques were generally designed to identify P<sub>t</sub>, P<sub>d</sub>, P<sub>p</sub>, or BAP. Soil analysis techniques were generally designed to identify either P<sub>t</sub> or plant available P, however BAP, P<sub>d</sub>, P<sub>p</sub> have become important factors when assessing agricultural P contributions to water quality.

When assessing potential P transport from runoff, variability in runoff volume as a result of climate, soils and agronomic factors plays a larger role in determining P loss than does a soil test (Sibbsen and Sharpley 1997). Care should be taken that soil testing is not the only criterion used when determining the potential for P transport from runoff (Haygarth and Jarvis 1999).

### **2.3.1 Analysis of Phosphorus in Soils**

Relating soil chemical and physical properties to the potential for P transport from surface runoff requires a different approach to sample collection than that required for soil fertility analysis. For most agricultural soils, samples collected to a depth of 2 cm would accurately define the effective depth for interaction between surface soil and runoff for rainfall intensities of less than 50 mm hr<sup>-1</sup>. For medium to coarse textured soils on slopes greater than 12% that are subject to high intensity rainfall (>100 mm hr<sup>-1</sup>), the sampling depth should be 4 cm (Coale 1998).

#### **Total P**

Total P (P<sub>t</sub>) includes all inorganic and organic forms of P in the soil. Analysis of P<sub>t</sub> is accomplished by converting all forms of P to orthophosphate (PO<sub>4</sub><sup>-3</sup>), and then analyzing the PO<sub>4</sub><sup>-3</sup> content using colorimetric or inductively coupled plasma (ICP) spectroscopic techniques. Conventional methods

use strong reagents and high temperatures to digest and dissolve all the soil P, then the sample is filtered, and the leachate is analyzed. Common methods include the perchloric acid digestion method (Olson and Sommers 1982), and the sulfuric acid / hydrogen peroxide / hydrofluoric acid digestion method (Bowman 1988).

### Available P

Most soil P analytical methods are used to identify P levels available for crop use. The extractants used are weaker than the digestion reagents used for total P. They are targeted at potentially-available P, which is only a portion of the total P in the soil sample. A typical extraction procedure involves the shaking of a known weight of soil (e.g. 5 g) with a known volume of extractant (e.g. 50 mL) for a fixed period of time (e.g. 30 minutes) at room temperature. The mixture is then filtered and the filtrate is analyzed for extractable P ( $\text{PO}_4^{3-}$ ) content. The tests are differentiated by the extractant used. A summary of the more common tests is presented in Table 2.2.

The numerous extraction methods all have some advantages and limitations depending upon the soil type for which they are targeted. Their suitability relies on long-term correlation studies that have established relationships between the extraction method, the soil, and crop response. Therefore some

**Table 2.2. Summary of phosphorus analysis methods.**

Analysis Method	Extractant	Comments
Olson	0.5M $\text{NaHCO}_3$ @ pH 8.5	-common method for assessing crop-available P in U.S. -applies best to soils with pH > 7
Mehlich-1	0.1N HCl 0.025N $\text{H}_2\text{SO}_4$	
Mehlich-3	0.2M $\text{CH}_3\text{COOH}$ 0.25M $\text{NH}_4\text{NO}_3$ 0.015M $\text{NH}_4\text{F}$ 0.013M $\text{HNO}_3$ 0.001M EDTA <sup>1</sup>	-common method for assessing crop-available P in U.S.
Bray I	0.03N $\text{NH}_4\text{F}$ 0.025N HCl @ pH 3.5	-not suited for calcareous soils
Bray (strong)	0.03N $\text{NH}_4\text{F}$ 0.1N HCl @ pH 1.0	
modified Kelowna <sup>2</sup>	0.015M $\text{NH}_4\text{F}$ 0.25M ammonium acetate 0.25M acetic acid	-best method for a wide range of soil pH levels -standard soil P test for the Prairie Provinces

Kelowna	0.015M NH <sub>4</sub> F 0.25M ammonium acetate	
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<sup>1</sup> EDTA is ethylene diamine tetraacetic acid

<sup>2</sup> there are two versions of the modified Kelowna, with slightly different strengths of extractants

extraction methods are favored more in some parts of the world than others. Soil pH, mineralogy, and organic matter content are major factors in accuracy. For example, the Bray I, which is not suitable for calcareous soils, is accurate for non-calcareous California soils (Vaughan and Jones 1980; referenced in McKenzie 1989). Care must be taken to make sure the extraction method is suitable for the soil conditions.

The most common extraction method worldwide is likely the Olson method (Tiessen and Moir, 1993). In Alberta the most common method is the modified Kelowna method, because of its suitability over a wide range of soil pH values. While efforts have been made to adopt standard tests, there is no consistency among regions. Similarly there is no standard on what constitutes a high soil test P value, and values will depend on the type of test used.

Although most tests are correlated to crop response, there is little direct correlation to environmental factors such as surface runoff. While these tests have been used to describe the relationship between available P in the soil and P<sub>d</sub> in the runoff, they have limitations. First, they represent the P<sub>d</sub> in the runoff, and while P<sub>d</sub> is an important water quality parameter, there is a considerable amount of P<sub>p</sub> in runoff that would be undetected (Daniel et al. 1998). This fraction can potentially become available for aquatic plant growth. Secondly, the Mehlich-3 test, which uses strong acids, removes not only the readily available P fraction, but also portion of the slowly available P fraction (Self-Davis et al. 1998). The test would therefore tend to overestimate the P<sub>d</sub> fraction in runoff. Sharpley et al. (1996) suggested that for an environmental test, distilled water or a dilute salt solution (e.g. 0.01M CaCl<sub>2</sub>) would be better-suited to assess the P<sub>d</sub> fraction. They stated that such methods would need further evaluation before they could be recommended as a standard procedure.

### **Biologically available P**

Biologically available P (BAP) has been defined as “the amount of inorganic P that a P-deficient algal population can utilize over a period of 24 hours or longer” (Sonzogni et al. 1982). It includes dissolved P and a portion of the particulate P that is available for algal uptake (Sharpley and Smith 1993). Algal uptake has been closely related to amounts of P extracted from soils or lake sediments by iron-oxide impregnated paper strips (Sharpley, 1993). It requires a 16 hour extraction time, which makes it difficult for labs to process large numbers of samples quickly. Sharpley et al. (1994) suggested that some routine soil tests are well correlated with the iron-oxide measure and could be used as a preliminary assessment. The iron oxide method could then be used as a special test for more intensive management cases.

### **2.3.2 Analysis of Phosphorus in Freshwater**

Inorganic phosphorus is found either in the inorganic or organic form in freshwater systems and either can contribute to the  $P_d$  or  $P_p$  fraction. The form and fraction of phosphorus, as well as the physical, biological and chemical characteristics of the receiving water, govern the availability of P for growth of algae (Sonzogni et al. 1982).

Phosphorus is the nutrient controlling the level of phytoplankton productivity in western Canadian lakes (Prepas and Trew 1983). Algal growth (primary production) in freshwater systems is dependant on the biological availability of P. Much of the phosphorus in runoff from agricultural lands in Alberta is in the dissolved and dissolved reactive fraction (Anderson et al. 1998; Cooke and Prepas 1998), the forms of phosphorus that are more readily available for primary production. Consequently, high dissolved P concentrations in runoff from agricultural lands likely increase the severity of primary productivity in P-sensitive water bodies in Alberta.

### **Total P, dissolved P and particulate P**

Total P ( $P_t$ ) is the summation of both  $P_p$  and  $P_d$ , and it accounts for all inorganic and organic phosphorus forms. Digestion and detection principles are similar to soil  $P_t$  analysis, but the reagents are different. Potassium persulfate digestion as described by Menzel and Corwin (1965) and modified by Prepas and Rigler (1982) is the routine chemical digestion method for the determination of  $P_t$  in freshwater.

Dissolved P is the fraction of P that is not retained by a 0.45  $\mu\text{m}$  pore membrane filter. It is comprised of organic and inorganic fractions. Much of the dissolved organic phosphorus is inert, while dissolved inorganic phosphorus is more readily available for biological use. However, the measurement of dissolved inorganic phosphorus is not a direct measure of the bioavailability of phosphorus in aquatic systems.

Particulate P is the P fraction that is adsorbed onto particulate materials including organic debris or soil particles that contain Fe- or Al- hydrous oxides, or Fe-P, Al-P, or Ca-P minerals (Sonzogni et al. 1982). In standard water analysis,  $P_p$  concentrations are not measured directly, but are calculated as the difference between  $P_t$  and  $P_d$  concentrations. While  $P_d$  is, for the most part, immediately available for biological uptake, the availability of  $P_p$  to aquatic organisms is limited. It can however, provide a variable, but long-term source of P in lakes (Sharpley et al. 1992).

### **Biologically available P**

Although  $P_t$  represents a variable, long-term source of P to aquatic systems, the biological availability of P is critical in evaluating the risk of P loading and subsequent eutrophication of surface waters. Dissolved inorganic phosphate (mainly as  $\text{H}_2\text{PO}_4^-$  or  $\text{HPO}_4^{2-}$  in the pH range of most natural waters) and orthophosphate ( $\text{PO}_4^{3-}$ ) are the inorganic forms of P most readily available to aquatic plants for biological production. Other forms of P ( $P_d$  and  $P_p$ ) can be converted to  $\text{PO}_4^{3-}$  through natural biological processes or hydrolysis (Pote and Daniel 1998).

Dissolved reactive phosphorus (DRP), as determined by the molybdenum blue reaction on undigested samples, is an estimate of orthophosphate in fresh water systems (Murphy and Riley 1962). The chemical extraction of DRP in freshwater tends to overestimate the biological availability of phosphorus (Sonzogni et al. 1982). A true measure of the potential biologically available (bioavailable) phosphorus (BAP) in freshwater is by evaluating the growth response of test organisms (e.g. algae; *Selenastrum* sp.) to phosphorus in labour-intensive bioassays. This method is very time consuming and costly. Therefore, the chemical extraction of DRP is a preferred method to estimate BAP.

Analysis of BAP and  $P_i$  have distinct roles to play in assessing accelerated eutrophication risk. When analyzing water samples, determination of BAP is critical for assessing the immediate risk of P loading to the water system. When assessing the long-term risk of a water system, many researchers and watershed managers want to know the  $P_i$  content (Pote and Daniel 1998).

### **3. Impact of Excess Phosphorus Levels**

#### **3.1 Soil Phosphorus and Crops**

##### **3.1.1 Impact of High Soil Phosphorus on Crop Response**

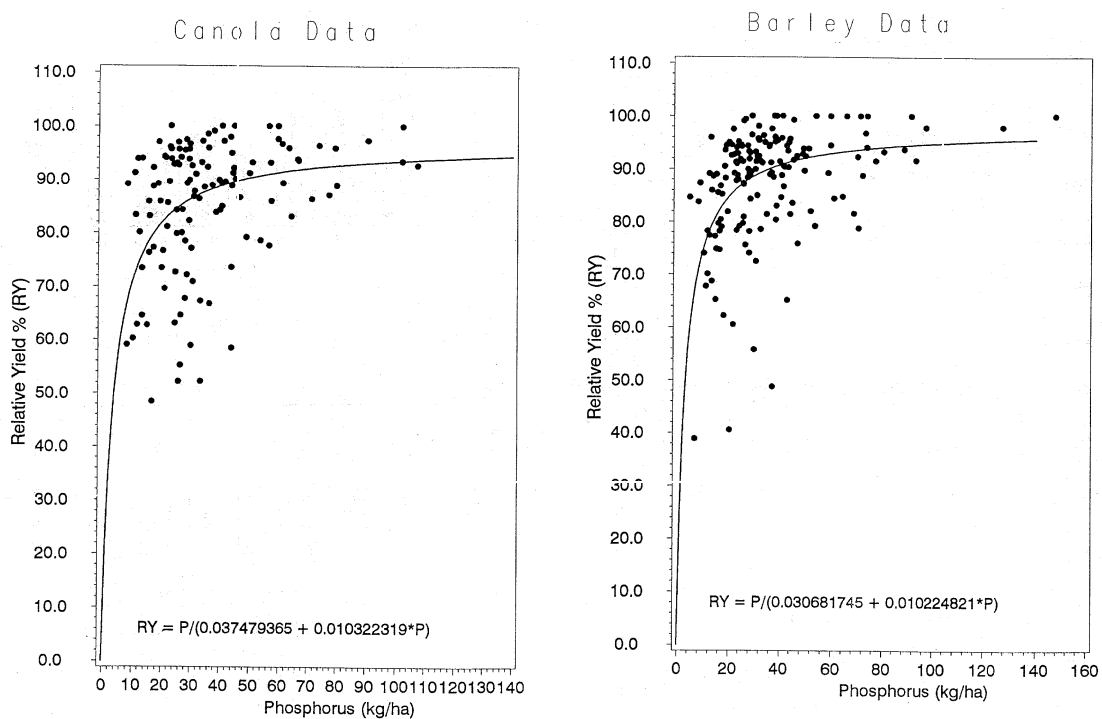
The mobility of P is relatively low compared to N, therefore plant roots have a lower probability of contacting P. During active P uptake periods, plant roots get most of their P from within 2 mm of the root surface (Nye and Tinker 1977, referenced in Sibbsen and Sharpley 1997). Also, agricultural crops take up only 5-10% of the applied fertilizer in the first year (Greenwood et al. 1980) with the remainder originating from the residual soil P (Sibbsen and Sharpley 1997). Consequently, agricultural soils must have a pool of residual P that is larger than the P uptake requirements for any single crop, in order to ensure adequate crop nutrition. To ensure optimum yields, managers tend to apply P fertilizer at rates in excess of crop demand and as a result, soil P levels have increased from very low to medium and high during this century (Sibbsen and Sharpley 1997).

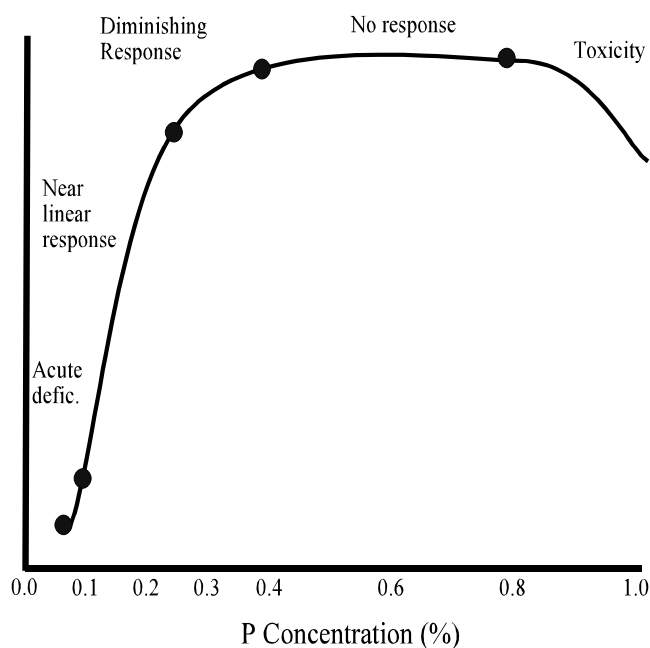
Crop yield response to P levels will increase to a critical level, beyond which yields do not appreciably increase regardless of how much P is applied (McKenzie et al. 1995; Johnston and Poulton 1997) (Figure 3.1).

**Figure 3.1 The relationship of soil test P to yield of barley and canola in Alberta (adapted from McKenzie et al. 1995).**

The information available does not present a clear trend about whether the impact of excessive soil P on crop yield will decrease solely in response to soil P. Tinker (1980) discussed the influence of high levels of available soil P on reduced mycorrhizal infection in plants, causing reduced uptake of zinc and copper. Janzen et al. (1993) also reported that increased levels of fertilizer P interfered with the uptake of zinc and copper, however, similarly-high rates of P from manure applications did not appear to interfere with zinc and copper uptake (Chang et al. 1994). The high availability of micronutrients in manure may prevent the uptake suppression effects observed from high rates of fertilizer P (Dormaer and Chang 1995).

Ozanne (1980) examined yield response to P concentrations in plant tissue and suggested that when P concentrations exceed 0.8%, toxicity may occur (Figure 3.2). Plant tissue concentrations may provide a more precise relationship with plant response since they are independent of soil factors.





**Figure 3.2** General relationship between yield and the concentration of P in plant tissue (adapted from Ozanne 1980).

Continued accumulation of P in the soil will result in an increased risk to the environment before agronomic problems appear. It is difficult to single out soil P levels alone in assessing agronomic problems related to manure application, because other components in the manure can potentially act to enhance or reduce crop performance. When considering land application of manure, municipal sewage, and food processing effluent, all nutrients and their potential impact on the soil, water, crops and air should be considered.

### 3.1.2 Using Crops to Reduce High Soil Phosphorus Levels

Using crops to remove P has been proposed as the most practical way to reduce P in soils with high P levels (Dormaer and Chang 1995; Daniel et al. 1998). The rate of decline of P in soils with P levels well above those required for optimum crop growth varies with soil type and management. Sharpley and Reikolainen (1997) presented research results showing a range in the rate of decline from 0.6 - 30 ppm yr<sup>-1</sup> (Table 3.1) for soils in the Northern Great Plains. At these rates several decades would be needed to reduce P to acceptable levels in fields where soil test P levels are in excess of 400 ppm.

Crop utilization of P is enhanced when a balance of nutrients is available. Nitrogen, in particular, enhances P uptake by increasing top and root growth, altering the plant metabolism and increasing the solubility and availability of P (Tisdale et al. 1985). Crop uptake of P also increases with

increasing soil temperature, moisture, aeration, and soil biological activity. Phosphorus availability to crops is most favorable at a pH range of 6.0 to 6.5. Availability is decreased at low pH levels by oxides of iron and aluminum and at high pH by calcium and magnesium.

**Table 3.1 Decrease in available soil P following phosphorus application (adapted from Sharpley and Reikolainen 1997).**

Soil <sup>1</sup>	Crop	Time (years)	Available Soil P				Reference <sup>2</sup> and Location
			Method	Initial (ppm)	Final (ppm)	Decline (ppm yr <sup>-1</sup> )	
Thurlow, l	Small grains	9	Olsen	13	4	1.0	Campbell 1965; Montana
		9		20	4	1.8	
		9		60	6	6.0	
Haverhill, cl	Wheat/fallow	14	Olsen	40	25	1.1	Cox et al. 1981; Saskatchewan
		14		74	33	2.9	
		14		134	69	4.6	
Portsmouth, fsl Sceptre, c	Small grains	8	Mehlich-1	23	18	0.6	Cox et al. 1981; Saskatchewan
		9		54	26	3.1	
	Wheat/fallow	Olsen	8	45	18	3.4	
			8	67	18	6.1	
			8	147	40	13.4	
Williams, l	Wheat/barley	16	Olsen	26	8	1.1	Halvorson and Black 1985; Montana
		16		45	14	1.9	
Carroll, cl	Wheat/flax	8	Olsen	71	10	7.6	Spratt et al. 1980; Manitoba
		8		135	23	14.0	
		8		222	50	21.5	
Waskada, l	Wheat/flax	8	Olsen	48	9	4.9	Spratt et al. 1980; Manitoba
		8		88	23	8.1	
		8		200	49	18.9	
Waskada, cl	Wheat/flax	8	Bray	140	50	11.3	Wagar et al. 1986; Manitoba
		8		320	80	30.0	

<sup>1</sup>. cl - clay loam; fsl - fine sandy loam; l - loam; c - clay.

<sup>2</sup>. Referenced in Sharpley and Reikolainen 1997.

In areas with excessive soil test P levels, withholding P application altogether may only produce short-term success. Withers et al. (1994, referenced in Sharpley and Reikolainen 1997) found that after three years of no P application to high P soils, crop yields began to decline. Rather than no P application, it may be better to use small amounts of P. Use of small amounts of P as a starter fertilizer has successfully reduced the need for larger P applications in vegetable crops, and there is potential for this practice to apply to other crops as well (Sharpley and Reikolainen 1997). Tisdale et al. (1985) stated that application of a small amount of P in the seedbed helped early crop establishment under cold, wet conditions. Management practices that improve crop performance will result in improved utilization of P.

### **3.2 Impact of Soil Phosphorus Levels on Phosphorus Mobility and Water Quality**

Water is highly sensitive to P accumulation. Eutrophication of lakes can become greatly accelerated when P levels in the water are between 0.01 - 0.02 ppm. This is an order of magnitude more sensitive than the soil levels for P considered critical for plant growth, which range from 0.2 - 0.3 ppm (Daniel et al. 1998).

Alberta Environmental Protection water quality criteria to control eutrophication in surface water in Alberta use a maximum concentration of 0.05 ppm of P. The United States Environmental Protection Agency (USEPA) has guidelines that P concentration should not exceed 0.05 ppm for streams entering lakes and reservoirs, and 0.025 ppm for the lakes and reservoirs themselves. For the prevention of plant nuisances in streams and other flowing water not directly entering lakes and reservoirs the P concentration should not exceed 0.10 ppm (Daniel et al. 1998).

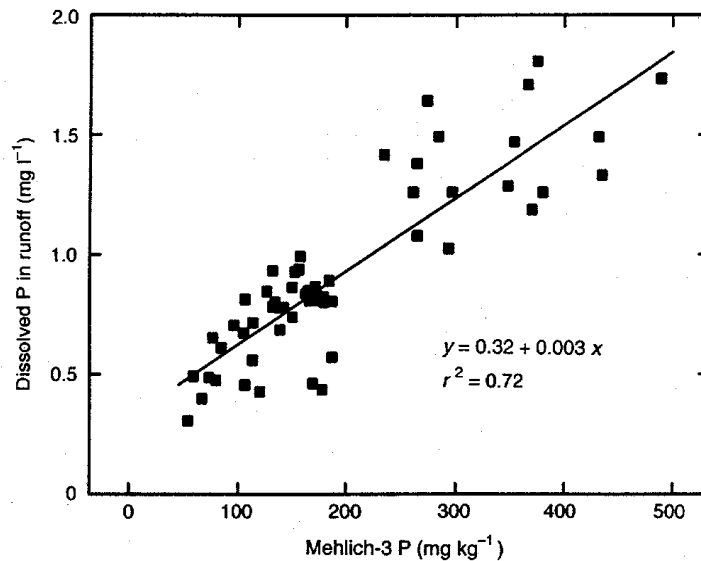
Eutrophication is the process of nutrient enrichment of surface waters. Excessive nutrients, like nitrogen and phosphorus, in freshwater promote the growth of aquatic plants and phytoplankton (algae). While both nitrogen and phosphorus are essential elements for aquatic plant and algal growth, certain species of phytoplankton can use or 'fix' nitrogen from the atmosphere. Therefore, phosphorus tends to be the limiting nutrient controlling phytoplankton productivity in aquatic systems in western Canada (Prepas and Trew 1983). Biological productivity in aquatic systems requires a relatively low concentration of P relative to N. Uhlmann and Albrecht (1968, referenced in Gibson, 1997) applied a C:P:N ratio of 105:15:1 to freshwater biota. This implies that freshwater organisms are 15 times more susceptible to changes in P levels than to changes in N.

Nutrient enrichment of surface waters promotes biological productivity and disrupts the natural balance of biological systems. Once nutrient supplies are exhausted, oxygen levels in the water are depleted as a result of the death and decomposition of the biota. This results in the suffocation of more sensitive or commercially important aquatic organisms, particularly fish. Additional effects of excessive biological productivity include increased water temperatures, impede water flow and navigation, increased rate of water loss through evaporation, and increased sedimentation which decreases the water body volume. Highly eutrophic waters used for drinking water supplies can also develop unacceptable taste and odour problems in the finished water. Blooms of cyanobacteria (blue-green algae) contribute to summer fish kills, unpalatability of drinking water and formation of

trichloromethane during water chlorination. Consumption of the cyanobacterial blooms, or the water-soluble neuro- and hepato-toxins that they release when they die, can be harmful to humans and livestock (Lawton and Codd 1991; referenced in Daniel et al. 1998). Periodic blooms of cyanobacteria have occurred in lakes and dugouts in Alberta (Kotak et al. 1993; Kotak et al. 1994). Therefore, the eutrophication of surface waters can be potentially catastrophic (Haygarth and Jarvis 1999) with significant economical and environmental impacts on society.

Non-point source pollution from agriculture was identified by the USEPA to the United States Congress as the major source of contamination of streams and lakes. Specifically, eutrophication was identified as the critical problem in those surface waters with degraded quality, with agriculture affecting 60% of the river contamination, 50% of the lake contamination, and 34% of estuary contamination (Parry 1998). Nitrogen and phosphorus were the leading contaminants in lakes and estuaries, and the third leading contaminant in rivers.

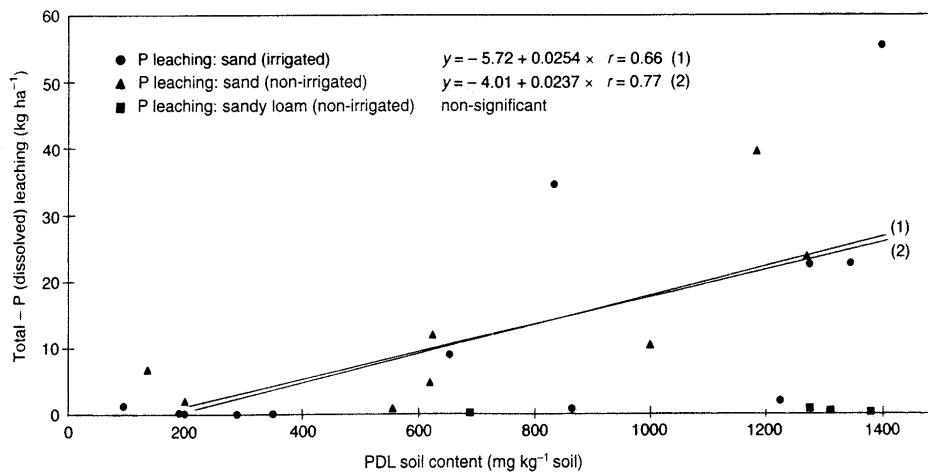
The risk of P export from land to water increases with increasing soil test P levels, primarily from increased dissolved P ( $P_d$ ). Figure 3.3 shows a linear increase in the  $P_d$  concentration in runoff as soil test P increases. Data from Europe suggest that at higher soil test P levels the amount of  $P_d$  in runoff increases exponentially (Sibbsen and Sharpley 1997).



**Figure 3.3** Relationship between soil test phosphorus content of surface soil (0-2cm) and dissolved phosphorus concentration of surface runoff from fescue in Arkansas (adapted from Pote et al. 1996).

Dissolved phosphorus concentrations in leachate have been shown to increase proportionally with soil test P levels. Figure 3.4 presents data from Germany relating soil test P, measured as percentage double-lactate soluble P (PDL), to dissolved P in lysimeters.

Soil P test levels alone are not necessarily a reliable predictor of P export to water systems. Relationships between soil test P and  $P_d$  vary as a function of management and soil type. Sharpley et al. (1996) reported that on the same soil type, regression slopes between  $P_d$  and soil test P were lower for grass than for cultivated land, suggesting that the vegetative cover was protecting the soil, and reducing  $P_d$  concentrations in the runoff. In addition to the level of P in the soil, the potential loss of P from agricultural watersheds is a function of local hydrologic and soil factors and management.



**Figure 3.4 Results of lysimeter experiments of phosphorus leaching as a function of phosphorus soil content as a result of increased slurry application (adapted from Meissner et al. 1997).**

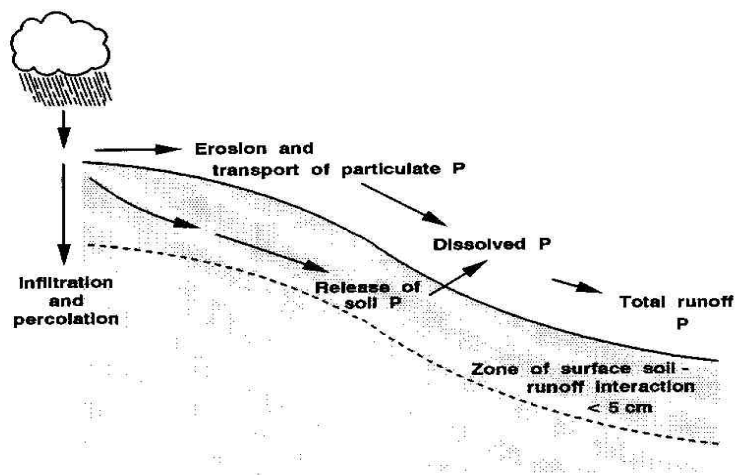
### 3.3 How Different Soil, Landscape Management and Environmental Factors Affect P Mobility

While much research has been directed toward understanding the dynamics of P in the soil, the hydrologic controls that link the spatially variable P sources, sinks, temporary storages, and transport processes within a watershed are less well understood (Gburek and Sharpley 1998). Effective control of P losses requires a more thorough understanding of the hydrologic processes within a given watershed. Haygarth and Jarvis (1999) consider hydrology the most important factor in the P transport regime.

While off-site movement of phosphorus may have little or no financial impact on the farmer, its impact may be felt many kilometers away and after a buildup over a considerable time period. By

the time the impact is felt, the farming practices that contributed to the problem may have been in place for generations and may be difficult to rectify.

It is generally considered that the dominant process for P export from watersheds is surface runoff, and contributions from subsurface flow and leaching are of relatively less concern (Sharpley and Syers 1979; Sharpley and Reikolainen 1997; Sims et al. 1998; Daniel et al. 1998). Surface runoff includes both P attached to particulates ( $P_p$ ) and P dissolved in the water ( $P_d$ ). The highest concentrations of P are typically found in surface runoff (Lennox et al. 1997), although this does not necessarily result in the highest amount of P transport. Quantification of P transport by runoff is complex and controlled by temporal and spatial factors, including climate.



**Figure 3.5 Phosphorus transport processes (adapted from Sharpley et al. 1996).**

In humid climates, runoff is generated from only a small number of source areas. These areas may expand and contract with time, varying rapidly during a storm as a function of temperature, soil, topography, groundwater and antecedent moisture status (Gburek and Pionke 1995). Runoff from these areas is limited by moisture storage, rather than by infiltration capacity, since antecedent soil moisture levels tend to be high.

In more arid areas, runoff is limited by infiltration rate rather than moisture storage. Areas within the catchment can alternate between sources and sinks for surface flow as a function of soil properties, rainfall intensity and duration, and antecedent moisture conditions. If we assume surface runoff is the primary means of P export, then in these cases if surface runoff does not occur, P exports are negligible (Brookes et al. 1997).

The export of P by subsurface flow or leaching has not been considered of major importance. This is because in most soils the P concentration of water has been thought to decrease with depth as it percolates through the profile, a result of adsorption of P by P deficient subsoils (Stamm et al. 1998). Exceptions include peaty or acid soils, sandy soils and waterlogged soils (Daniel et al. 1998). Several cases have been identified, however, where the contribution of dissolved P from subsurface flow may be substantial (Ryden et al. 1973; Hergert et al. 1981a, b; Culley et al. 1983; Heckrath et al. 1995; Hawkins and Scholefield 1996, all referenced in Stamm et al. 1998 ). Ryden et al. (1973) cited eight cases where P concentrations in tile drain water were higher than levels usually associated with eutrophication. They identified two cases where the P export in subsurface flow equaled or exceeded values typically found in surface runoff. In cases where the transport process was identified, P export was by preferential flow through macropores was a major factor.

Xue et al. (1998) reported that tile drains made significant contributions of dissolved P to the total river load. They also showed that the tile drain load was strongly event-driven, with the highest loads corresponding to heavy rain events. Stamm et al. (1998) showed that preferential flow through macropores was contributing to high soluble P levels in tile drain water under manured grasslands. Phosphorus leaching by preferential flow was also demonstrated by Jensen et al. (1998) in laboratory soil column tests.

Leaching of P has been observed in organic soils where adsorption capacity is low due to low pH, low clay content, or complexing of Al and Fe. Also coarse textured soils, and areas prone to shallow water tables, are at higher risk of having P transported by subsurface flow, as are soils with high P levels due to over-fertilization or excessive manure applications (Sims et al. 1998).

Use of water quality guidelines alone should not be the only criterion to control P levels in runoff. Daniel et al. (1998) identified cases where runoff from non-manured, non-fertilized land exceeded the 0.100 ppm criteria, creating a situation that was very difficult to manage. They suggested considering the relationship between P loading and watershed characteristics on a site specific and water use basis. For example, lakes used for water supply or swimming will benefit from low P loading, whereas lakes used for fish production may benefit from a more moderate level of P loading. The factors that should be considered include proximity to P-sensitive water, land use, runoff and leaching potential, mean depth of water bodies and hydraulic residence time.

## 4. Controlling Phosphorus Levels in the Environment

### 4.1 Single Limit Approach

Several American states have adopted or proposed guidelines for maximum phosphorus levels in the soil, as shown in Table 4.1. The single limit approach offers a quick method that requires soil testing as the basis for establishment. Land managers can easily plan the land base needed for their application needs.

A major difficulty in the acceptance of a single limit approach is the scientific basis for establishment of a soil test P level that represents a water hazard. The database relating soil test P to P in runoff is limited to a few soils and cropping systems. In addition the risk of P loss to the water system as a function of soil test P level, varies with soil type (Sharpley 1995) and hydrologic parameters (Sharpley et al. 1999). Also soil tests would need to be standardized in order to apply meaningful limits.

**Table 4.1 Guidelines for maximum levels of phosphorus in soils (adapted from Sibbsen and Sharpley 1997).**

Location	Critical Level for Soil P (ppm)	Management Recommendations (when at or above limit)
Arkansas	150	<ul style="list-style-type: none"> <li>• apply no P from any source.</li> <li>• minimize erosion by using constant cover.</li> <li>• provide buffers next to streams.</li> <li>• overseed pastures with legumes to aid P removal.</li> </ul>
Delaware	120	<ul style="list-style-type: none"> <li>• apply no P from any source.</li> </ul>
Ohio	150	<ul style="list-style-type: none"> <li>• reduce/eliminate erosion.</li> <li>• reduce/eliminate P additions.</li> </ul>
Oklahoma	130; 200	<ul style="list-style-type: none"> <li>• 130-200 ppm, half rates of P on all soils.</li> <li>• &gt; 200 ppm, P rate not to exceed crop removal.</li> </ul>
Michigan	75; 150	<ul style="list-style-type: none"> <li>• 75-150 ppm, P rate not to exceed crop removal.</li> <li>• &gt; 150 ppm, no P addition from any source.</li> </ul>
Texas	200	<ul style="list-style-type: none"> <li>• P rate not to exceed crop removal.</li> </ul>

Wisconsin	75; 150	<ul style="list-style-type: none"> <li>• 75-150 ppm: rotate to P-demanding crops and reduce manure application rates.</li> <li>• &gt; 150 ppm, no manure applications.</li> </ul>
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Because of the economic implications brought about by limiting manure applications to a given land base, industry is reluctant to accept imposition of a single limit and is vigorously challenging their scientific basis (Daniel et al. 1998). The impact of P pollution on other industries has been a factor driving regulation in some states. For example, the protection of marine-based industries, such as the shellfish industry, from the effects of P contamination in coastal estuaries was a major factor in the recent enactment of manure regulations in Virginia.

#### **4.2 Phosphorus Index**

The Phosphorus Index was introduced by the USDA as a tool to assess the various landforms and management practices to determine a risk for P export to the water system (Lemunyon and Gilbert 1993). A ranking with the Phosphorus Index shows the relative risk of P movement compared to other sites. When the parameters of the Index are analyzed it will be apparent that some factors are disproportionately high. These factors can then be addressed as the basis for planning corrective soil and water management. For example, adjacent fields having similar soil-test P levels but different susceptibilities to runoff and erosion, based on soils, topography, and/or land use, would have different P application strategies.

At its initiation the Phosphorus Index was in a simplified form, using a limited number of landform characteristics that required readily-available field data. It was intended to evolve into a more complex form in which a computer model would relate landform characteristics to the risk of P loss from the site (Lemunyon and Gilbert 1993). The original index had eight site characteristics and five classes of P loss for each characteristic. Each characteristic and class was assigned a weighting factor. The class factor doubled for each increment in severity (Table 4.2)

The weighted site characteristic H class products were summed to provide a rating for the vulnerability of the site. The higher the rating the more vulnerable the site is to significant off-site P transport. Site vulnerability ratings were divided into four categories (Table 4.3).

The index was intended for use by field staff who work with producers. It can reveal the degree of P loss risk from management practices on specific landforms. It can also be used by planners working at regional or watershed levels to identify high risk areas that can be targeted for priority action. The index can also be used to assess the relative impact of conservation practices on a number of sites.

Stevens et al. (1993) found that the P index was very sensitive to fertilizer P application rates and methods but not sensitive enough to differentiate sites based on site factors controlling erosion and runoff. Beyond this there has been very little field validation of the P index, with individual states proceeding on their own to validate it. Heathwaite (1997) argued that a P index does not take into

account watershed catchment characteristics, notably hydrologic parameters and the nature of the drainage network, and cautioned that care should be taken that it is used to identify and prioritize sensitive sites within a watershed rather than as a watershed assessment tool.

Sharpley et al. (1999) modified the P index to apply to a watershed scale. They separated source factors (fertilizer and organic applications) and transport factors (runoff and erosion) for site vulnerability classification within a watershed (Tables 4.4 and 4.5). To better represent the relationship of surface runoff and soil P, as well as the potential for surface runoff to contribute to stream flow, they multiplied the transport factors, and multiplied the final transport and source characteristics for a site. Under this method, both the transport and source factors must coincide to identify a site as highly vulnerable to P loss. In the original index, a site could be identified as highly vulnerable to runoff based on high source factors alone, even if no runoff occurred.

Sharpley et al. (1999) used two site assessment indices, the modified P index coupled with an N index, to assess watersheds for nutrient management sensitivity. The structure of the N index and site vulnerability classification (Tables 4.6 and 4.7) was derived from the P index, and the principles of N leaching described in Kissel et al. (1982). An overall site vulnerability rating (Table 4.8) was derived by coupling the modified P index and the N index.

**Table 4.2 Phosphorus Index, initial version (adapted from Lemunyon and Gilbert 1993).**

	<b>Phosphorus Loss Rating (Value)</b>				
<b>Site Characteristic (Weight)</b>	<b>None (0)</b>	<b>Low (1)</b>	<b>Medium (2)</b>	<b>High (4)</b>	<b>Very High (8)</b>
Soil erosion (1.5) Irrigation erosion (1.5)	Not applicable Not applicable	< 5 tons/acre Tailwater recovery or QS < 6 for very erodible soils or QS < 10 for other soils	5-10 tons/acre QS > 10 for erosion resistant soils	10-15 tons/acre QS > 10 for erodible soils	> 15 tons/acre QS > 6 for very erodible soils
Runoff class (0.5) Soil P test (1.0) P fertilizer application rate (0.75)	Negligible Not applicable None applied	Very low or Low Low 1-30 P <sub>2</sub> O <sub>5</sub> lb/acre	Medium Medium 31-90 P <sub>2</sub> O <sub>5</sub> lb/acre	High High 91-150 P <sub>2</sub> O <sub>5</sub> lb/acre	Very high Excessive > 150 P <sub>2</sub> O <sub>5</sub> lb/acre
P fertilizer application method (0.5)	None applied	Placed with planter deeper than 2 in.	Incorporated immediately before crop	Incorporated > 3 months before crop or surface applied < 3 months before crop	Surface applied > 3 months before crop

Organic P source application rate (1.0)	None applied	1-30 P <sub>2</sub> O <sub>5</sub> lb/acre	31-60 P <sub>2</sub> O <sub>5</sub> lb/acre	61-90 P <sub>2</sub> O <sub>5</sub> lb/acre	> 90 P <sub>2</sub> O <sub>5</sub> lb/acre
Organic P source application method (1.0)	None	Injected deeper than 2 in.	Incorporated immediately before crop	Incorporated > 3 months before crop or surface applied < 3 months before crop	Surface applied to pasture, or > 3 months before crop

**Table 4.3 Phosphorus index, site vulnerability ratings (adapted from Lemunyon and Gilbert 1993).**

Rating from Total of Weighted Products	Site Vulnerability Category
< 8	Low
8-14	Medium
15-32	High
> 32	Very High

**Table 4.4 Modified P index (adapted from Sharpley et al. 1999).**

Factors (Weight)	Phosphorus transport potential (Value)				
	None (0.6)	Low (0.7)	Medium (0.8)	High (0.9)	Very High (1.0)
Soil Erosion <sup>1</sup>	Negligible	< 10	10 - 20	20 - 30	> 30
Irrigation erosion (1.0)	Negligible	Infrequent irrigation on well-drained soils	Moderate irrigation on soils with slopes <5%	Frequent irrigation on soils with slopes of 2 to 5%	Frequent irrigation on soil with slopes >5%
Runoff Class (1.0)	Negligible	Very low or low	Medium	High	Very High
Contributing distance, m (1.0)	None (0.2)	Low (0.4)	Medium (0.6)	High (0.8)	Very High (1.0)
	>170	170 - 130	130 - 180	80 - 30	<30
	Phosphorus source potential (Value)				
	None	Low (1)	Medium (2)	High (4)	Very High (8)
Soil Test P (1.0) <sup>2</sup>	<10	10 - 30	30 - 100	100 - 200	>200
Fertilizer P rate (0.75)	None	1 - 15	16 - 45	46 - 75	>75
Application method for fertilizer (0.5)	None	Placed with planter or injected deeper than 5 cm	Incorporated immediately before crop	Incorporated >3 months or surface applied <3 months before crop	Surface applied >3 months before crop
Organic P rate (1.0) <sup>3</sup>	None	1 - 15	16 - 30	30 - 45	>45

Application method for organic source (1.0)	None	Placed with planter or injected deeper than 5 cm	Incorporated immediately before crop	Incorporated >3 months or surface applied <3 months before crop	Surface applied >3 months before crop
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<sup>1</sup>Units for soil erosion are Mg ha<sup>-1</sup>

<sup>2</sup>Units for Mehlich-3 soil P are mg P kg<sup>-1</sup>

<sup>3</sup>Units for P application are kg P ha<sup>-1</sup>

**Table 4.5 Site vulnerability based on the modified P index (adapted from Sharpley et al. 1999).**

Site Vulnerability	Total P index Rating Value <sup>1</sup>
Low	< 5
Medium	5 - 9
High	10 - 22
Very High	> 22

<sup>1</sup>P index rating = (erosion X runoff X return period) X 3(source characteristic X weight)

**Table 4.6 N index (adapted from Sharpley et al. 1999).**

Factors	Leaching Potential from Transport Factors (Value)				
	None (0)	Low (1)	Medium (2)	High (4)	Very High (8)
Texture	Clay	Clay loam to silty clay loam	Loam to silt loam	Loamy fine sand to coarse sandy loam	Sand
Permeability <sup>1</sup>	<1.5	1.6 - 4	4.1 - 14.9	15 - 50	>50
	Leaching Potential from Source Factors (Value)				
Fertilizer N <sup>2</sup>	None	1 - 50	51 - 150	151 - 300	>300
Application method for fertilizer	None	Placed with planting	Incorporated immediately before crop	Surface applied >3 months before crop or incorporated <3 months before crop	Incorporate d >3 months before crop
Manure N <sup>2</sup>	None	1 - 50	51 -150	151 - 300	>300
Application method for manure	None	Placed with planting	Incorporated immediately before crop	Surface applied >3 months before or incorporated <3 months before crop	Incorporated >3 months before crop

<sup>1</sup>Units for permeability are cm hr<sup>-1</sup>

<sup>2</sup>Units for N application are kg N ha<sup>-1</sup>

**Table 4.7 Site vulnerability based on the N index (adapted from Sharpley et al. 1999).**

Site Vulnerability	Total P index Rating Value <sup>1</sup>
Low	< 3
Medium	3 - 8
High	9 - 18
Very High	> 18

<sup>1</sup>P index rating = (erosion H runoff H return period) H 3(source characteristic H weight)

**Table 4.8 Site vulnerability rating, based on the coupled P index and N index (adapted from Sharpley et al. 1999).**

Index Rating		Generalized Interpretation of the P index
P	N	
< 5	< 3	<b>Low</b> potential for P loss. If the current farming practices are maintained there is a low probability of adverse impacts on surface waters
5-8	4-8	<b>Medium</b> potential for P loss. The chance for adverse impacts on surface waters exists and some remediation should be taken to minimize the probability of P loss
9-22	9-18	<b>High</b> potential for P loss and adverse impacts on surface waters. Soil and water conservation measures and nutrient management plans are needed to minimize the probability of P loss
> 22	> 18	<b>Very High</b> potential for P loss and adverse impacts on surface waters. All necessary soil and water conservation measures and nutrient management plans must be implemented to minimize the probability of P loss

Sites were selected within a watershed, and soil samples were taken on a 30 m grid to determine soil P levels. Transport factors were derived from soil survey data, and source factors were determined from farmer surveys in the watershed. The N and P indices were then applied at a 5 m cell scale and integrated over the entire watershed. The result was a sensitivity map of the watershed for N and another map for P based on the criteria in Tables 4.5 and 4.7.

Sharpley et al. (1999) demonstrated that the modified indices provided greater flexibility in managing sites within a watershed. In the test watershed, use of a single limit of 200 ppm would have resulted in 33% of the watershed (67% of the cultivated land) being ineligible for further manure application. With their index approach, only 1% of the watershed presented a high risk to P loss from manure application.

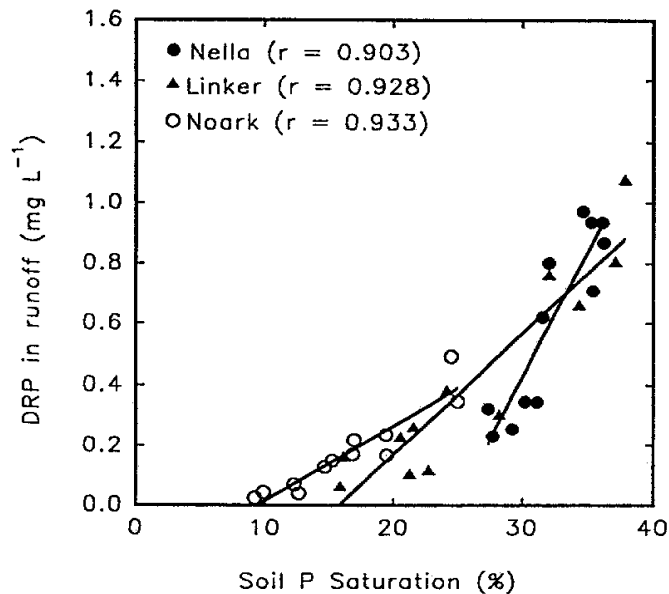
### **4.3 Percent Saturation Approach**

Sibbsen and Sharpley (1997) presented an approach to P management used in the Netherlands based on soil P-saturation characteristics. The Netherlands has a national strategy to limit P entry into surface water and groundwater. One of the ways to attain this goal has been to identify a soil P level above which P inputs must be matched by crop removal. Rather than a single limit for all soils based on soil test P, the limit is based on the relationship between soil P level and the potential for dissolved P ( $P_d$ ) to enter the water system. Specifically, as the amount of phosphorus sorbed to soil components increases,  $P_d$  concentration increases. An index for the soil P, called sorption-saturation, is calculated as the percent of  $P_d$  extracted from the soil relative to the P sorption capacity. The index is relatively independent of soil type. In the Netherlands a sorption-saturation value of 25% has been established as the critical limit, above which P concentrations in leachate and runoff become unacceptable.

Sharpley (1995) measured the relationship of soil test P (Mehlich-3) to  $P_d$  concentrations in runoff for ten soils ranging from coarse to fine texture. Comparing the soil test P to  $P_d$  concentration revealed no consistent relationship from one soil to another. A 200 ppm soil test level corresponded to a  $P_d$  concentration of 0.28 ppm in one soil and 1.36 ppm in another. When the sorption saturation for each soil was calculated, a clear relationship describing  $P_d$  as a function of sorption saturation emerged.

Once calibrated for a specific soil test, the P sorption-saturation approach has the potential to describe the capability for wide range of soils to release P into the water system. It also offers an advantage in indicating how close a soil is to being an environmental risk from P loading. For example, soils with sorption-saturation levels near a critical limit, (e.g. 25%), should not receive P inputs, whereas those with lower sorption-saturation levels could receive more P inputs. Therefore the approach becomes a sliding scale for P inputs based on the soils capability to release P into the water system.

Prediction of P movement into water systems should not rely on soil factors alone, however. Pote et al. (1999) tested the percent saturation approach to predict P concentrations in runoff in field plots. The relationship between percent saturation and  $P_d$  concentrations in runoff within a soil series was highly correlated, but it was not consistent across different soil series (Figure 4.1). The variability in runoff volumes on some soil series accounted for much of the inconsistency.



**Figure 4.1 Relationship between P saturation (oxalate method) of surface soil and dissolved reactive phosphorus (DRP) in runoff in three Ultisols (adapted from Pote et al. 1999).**

#### **4.4 Nutrient Management Planning**

Nutrient management planning is being used by every state in the United States with noteworthy programs in Maryland, Pennsylvania, and Wisconsin (Daniel et al. 1997). It offers advantages by being a means of cutting costs and preserving water quality without jeopardizing yields, and is generally thought to present a win-win situation to both animal feeding operations (AFO's) and the public. The United States Draft National Strategy for AFO's (USDA/USEPA 1998) proposes setting a national performance expectation that all AFOs develop a technically sound and economically stable Comprehensive Nutrient Management Plan (CMNP) for their operation.

The technical components and details of nutrient management planning will be site specific and therefore vary from one location to another; however there are fundamental components that should be common to all planning. The most fundamental component is the concept that the farm planning is based on a nutrient budget, where there is a storage of nutrients on the farm in livestock, crops, and soil. Nutrients are imported into the farm in various forms including fertilizer, feed, and even atmospheric (through fixation of N). Forms of nutrients leaving the farm include livestock and livestock products (e.g. milk and eggs), crops, manure, and crop residues. Nutrients can also leave the farm by volatilization to the atmosphere, erosion, runoff, and leaching. The goal is to balance nutrient inputs and outputs to keep soil storage levels and outputs through volatilization, erosion, runoff and leaching to target levels that are safe to the environment. Daniel et al. (1997) breaks the components into two sets, institutional and technical.

Institutional components are those directed at planning, education, and infrastructure developments that ensure an effective result. An example is that in areas with intense AFO activity, management of large manure volumes may be improved by more effective transportation systems. From a planning perspective, institutional approaches are needed to manage watersheds, set water quality goals, inform the public, integrate the variety of agencies, industries, and institutions that are stakeholders in the watershed, and develop approaches to prioritize those watersheds in most need of nutrient management planning. An example is the Wisconsin priority watershed program which focuses a limited supply of resources where they can get the most results. Nutrient Management Plans are being developed by the private sector and are reimbursed at no cost to the landowner.

Institutional components also include a way of ensuring compliance from AFOs. This could involve development of both monitoring and enforcement strategies. For example, Maryland has a program to assist farmers with construction of manure management facilities. To qualify, the farm must have a CNMP in place.

When developing plans, the institutional framework should include ways of integrating private sector and public sector activities. Generally the private sector has advantages in doing the planning, given the proper direction and supervision. The public sector should retain the role of determining who can do the planning (e.g. through licencing or accreditation) and which plans are sound.

The technical components recommended by Daniel et al. (1997) to be incorporated into nutrient management planning include quantity of manure produced, nutrient content of the manure, and nutrient availability. Nutrient availability is tied to land application practices and has potential to offset some fertilizer costs. Good application practices can result in more nutrients available for cropping in the following year or two. Other technical components include soil testing for both N and P, identification and avoidance of environmentally sensitive areas, setting realistic yield goals, and timing of manure application.

The CNMPs proposed by the US National Strategy for AFOs must meet clearly defined nutrient management goals and identify the measures and schedules for attaining these goals (USDA/USEPA, 1998). They should be site specific and written to meet the needs of the AFO owner/operator and conditions of the farm. They should address, at a minimum, feed management, manure handling and storage, land application of manure, land management, record keeping, and management of other manure utilization options. The other options would apply to cases such as vulnerable watersheds, and include composting, sale, and use of manure for power generation. Plans should also address other potential pollutants such as pathogens. Plans should also be periodically reviewed and revised in cases where expansion is planned, or changes in manure management or in the AFO operation are considered. Development of the CNMP will be the responsibility of the operator, with assistance as needed from certified industry staff, government specialists, private consultants and other qualified vendors. The recommended primary technical reference is the National Resource Conservation Service (NRCS) Field Office Technical Guide.

The National Strategy proposes using voluntary programs as the principal means of implementation. The implementation strategy would focus on locally-led conservation efforts, environmental education, and technical and financial assistance. The AFOs would need to implement CNMPs before qualifying for financial assistance.

## **5. Factors Influencing the Alberta Situation**

Alberta lakes, streams and rivers differ from water bodies found elsewhere in Canada or the United States. Climate, landscape (including soils, topography and ground cover), hydrology and the inherent water chemistry influence the complex and dynamic freshwater systems in Alberta. Many lakes and other surface water bodies in Alberta are naturally highly productive (or eutrophic) and experience prolific phytoplankton (algae) and macrophyte (plant) growth. Consequently, surface waters in Alberta are extremely sensitive to further P enrichment either by internal or external sources. Further nutrient enrichment of surface waters in Alberta can have significant economic and environmental implications on public and animal health, the recreation and tourism industries as well as detrimental effects to aquatic life. Careful P management on the landscape is critical to maintain or improve water quality in Alberta.

### **5.1 Industrial Development and Expansion**

Lamb-Weston, which has recently constructed a \$100 million potato processing plant near Taber, has requested soil phosphorus loading guidelines in order to assess land area requirements for irrigating with wastewater effluent. McCain's is building a similar plant in the region and plan to use the effluent for irrigation. Several other industries in Lethbridge have also approached Alberta Agriculture, Food and Rural Development for phosphorus loading guidelines.

Alberta Environment, in response to industry pressure, is looking to Alberta Agriculture, Food and Rural Development for leadership in establishing safe phosphorus levels. As a temporary working solution, a 200 ppm upper limit for phosphorus in the top 15 cm of the soil was proposed. Lamb-Weston has accepted this value, and developed its wastewater irrigation system accordingly. The other processing industries are also expected to accept this value.

There is an opportunity for the agri-food sector in Alberta to grow to \$20 billion in value-added shipments and \$10 billion in farm cash receipts by the year 2005 (Alberta Agriculture, Food and Rural Development 1999b). Development and expansion of intensive livestock operations and food processing companies will be a key factor in the growth of the agri-food sector. Without clear guidelines regarding maximum soil loading of nutrients such as nitrogen and phosphorus, producers must guess at the land base required for their operations. Governments who make decisions on processing and intensive livestock operation (ILO) development applications, also need consistent guidelines on which to base their decisions. The Livestock Regulatory Stakeholder Advisory Group (LRSAG), which developed the regulation of the intensive livestock industry in Alberta has accepted N-based guidelines for manure application, but has recognized that P-based standards for manure application will be needed in the near-future. The Expert Technical Committee, which was commissioned by the LRSAG to develop the industry operating standards as part of the regulatory process, has requested that P-based standards for manure application be developed by the year 2001.

### **5.2 Municipal Waste Disposal**

Municipal wastewater that is not discharged into water is applied to land. Guidelines for application of municipal wastewater have been developed by Alberta Environmental Protection (Alberta Environmental Protection 1997c). Draft guidelines have been developed specifically for the irrigation method of disposing municipal wastewater (Alberta Environmental Protection, 1997a). These draft guidelines will replace previous guidelines prepared in 1984 (Alberta Environmental Protection 1984). Similar approaches are used as required for natural irrigation water such as determining the suitability of the wastewater and land for irrigation. The major difference between municipal wastewater and high quality irrigation water is that wastewater has higher concentrations of living and nonliving organic material, N, P, and in some cases, higher concentrations of sodium (Na) and salts. Low concentrations of grease, oil, detergents, and certain metals may also be present, but these are generally at concentrations that do not adversely impact crops or land if applied by irrigation methods (Alberta Environmental Protection 1997a). Typical P, N, and K contents of municipal wastewater can range from 2 to 6, 10 to 20, and 4 to 40 mg L<sup>-1</sup>, respectively (Alberta Environmental Protection 1997a). If the nutrient content of a wastewater sample was assumed to be 4 mg L<sup>-1</sup> P, 15 mg L<sup>-1</sup> N, and 22 mg L<sup>-1</sup> K, then an application of 15 cm of wastewater would deliver 6 kg ha<sup>-1</sup> of P, 22.5 kg ha<sup>-1</sup> of N, and 33 kg ha<sup>-1</sup> of K. A large amount of wastewater would be required in order to deliver a substantial amount of P, based on these values.

Specific draft guidelines have also been developed for the application of municipal biosolids to agricultural land (Alberta Environmental Protection 1997b). These guidelines will replace previous guidelines developed in 1982 (Alberta Environmental Protection 1997b). As mentioned in Section 2.2.2, there are several methods used to dispose of municipal biosolids. However, there are major programs in Alberta that use the land application method, such as the City of Edmonton's Nutri-Gold program and the City of Calgary's CALGRO program. Both programs solicit the farming community for land that could be used to receive biosolids. CALGRO has been operating since 1983. The biosolid material is injected by CALGRO at a rate of 14 tonnes ha<sup>-1</sup> on a dry-weight basis. The biosolid material contains about 4% P, 5.5 % N, and 0.35% K on a dry-weight basis and about 10% solid material (Barl and McKenzie 1995). At this rate, about 560 kg ha<sup>-1</sup> of P, 770 kg ha<sup>-1</sup> of N, and 49 kg ha<sup>-1</sup> K would be applied. The P and N rates are certainly well above the nutrient requirements of any crops. CALGRO applied an average of 18,950 tonnes (dry-weight basis) of biosolids onto 1,400 ha of farm land each year from 1984 to 1992 (Barl and McKenzie 1995). Biosolid material was applied only on the same parcel of land once every three years, provided the land continued to meet the application criteria.

### **5.3 Food Processing Wastewater Disposal**

Guidelines were developed by Alberta Environmental Protection for the approval of land treatment facilities for the treatment of industrial wastes including food processing wastes (Alberta Environment 1988). The approval of land treatment facilities was under the provisions of the Clean Water Act, which was consolidated with other acts in the Environmental Protection and Enhancement Act in 1993.

The nutrient content of wastewater processing plants tends to be higher than for municipal wastewater. Barl and McKenzie (1995) reported 42 to 72 mg L<sup>-1</sup> total P, 75 to 211 mg L<sup>-1</sup> total N, 466 to 707 mg L<sup>-1</sup> total K for wastewater from a potato processing plant. If the nutrient content of a wastewater sample

was assumed to be 55 mg L<sup>-1</sup> P, 150 mg L<sup>-1</sup> N, and 580 mg L<sup>-1</sup> K, then an application of 15 cm of wastewater would deliver 82.5 kg ha<sup>-1</sup> of P, 225 kg ha<sup>-1</sup> of N, and 870 kg ha<sup>-1</sup> of K. The amount of food processing wastewater required to meet nutrient requirements of crops would be less than for municipal wastewater.

#### **5.4 Manure Disposal on Land**

In Alberta, studies of soils that have received manure at rates of 34 tonnes ha<sup>-1</sup> (15 tons acre<sup>-1</sup>) or higher for more than 15 years are showing very high phosphorus levels in the upper soil profile (top 15 cm). In the County of Lethbridge, Alberta, soil test phosphorus levels in the top 15 cm of manured soils are typically 100-400 ppm but some soil P levels in excess of 675 ppm, which is equal to 6370 kg ha<sup>-1</sup> of 11-51-0 fertilizer, have been measured in the area.

Current manure application guidelines in Alberta use nitrogen as the basis for nutrient loading. Implementing maximum loading levels based on phosphorus would have complications. There has been considerable experience in relating N-based applications to crop requirements. In southern Alberta, N availability has been based on a decay series by Pratt et al. (1973). This series estimates the available N as 35, 15, 10, 7.5, 5 and 4 percent of the total residual N in the first, second, third, fourth, fifth and sixth years following application. Little is known about how much P would be available for crop use in a given year from manure application, and estimates have varied from 25% to 50%. Since not all of the P would be available immediately, application based on the total P content in the manure would potentially result in insufficient P being available for the crop. Application rates based on available P would result in a continual loading of P in the soil from the residual, unavailable P. Operators could require as much as four to six times the land base that is now required for N-based manure disposal (McKenzie et al. 1999).

## 6. Options to Achieve Safe Phosphorus Levels in the Soil

The following summary of the pros and cons of the P management strategies described in Chapter 4 is presented for consideration when developing a land application strategy to ensure environmentally safe levels of phosphorus.

### Single Limit Approach:

pros:

- easy to implement
- relatively easy to monitor
- simplifies long-range planning

cons:

- does not allow for site specific variation in risk of P movement, which is controlled by site specific factors such as soil type and landform. A single limit may be too high for one site but too conservative for another
- scientific validation has been challenged
- dependent on soil test methods that are not standardized

### Phosphorus Index:

pros:

- addresses the soil and landform factors controlling P movement into the water system
- allows sensitive areas to be identified and targeted for special management
- allows relative comparisons of the impact of management practices
- provides more flexibility than the single limit approach, and still minimizes risk
- provides extension opportunities that would assist producers and land managers

cons:

- needs to be fully developed as a tool
- weighting factors are best estimates
- little field validation information available
- field evaluations are more complex than the single limit or percent saturation approach

### Percent Saturation Approach:

pros:

- strong correlation with soil type
- independent of soil test procedure
- potential to apply to a wide range of soil types and existing soil P levels

cons:

- limited information or research available
- limited information on saturation capacity for Alberta soils

-does not address landform or hydrologic issues

### **Nutrient Management Planning:**

pros

- flexible, addresses a wide range of options for nutrient management and not tied to land application alone
- not specific to any one nutrient, looks at the whole farm and feeding operation
- can apply to wide variety of agricultural operations, including animal feeding, food processing, and cropping
- has been tried by some American states

cons:

- complex and needs considerable information for planning; however the principles are similar to those used for commercial fertilizer management
- feasibility of an operation attaining a nutrient balance is in question
- requires standards for nutrient loading. In areas where inputs are exceeding crop uptake, considerable time may be needed to arrive at a balanced nutrient management practice? Some form of P limit may be necessary as an interim measure, while land managers adopt this approach
- monitoring would be expensive

There is a growing consensus that the water quality problems now facing society can best be solved using a whole-watershed approach (Daniel et al. 1998). Watershed assessment and management generally requires a combination of water quality monitoring, spatial data collection, and modeling. The accuracy will depend on good experimental results that can be used for model validation, and reliable regional data on the factors governing P transport. The key data identified by Daniel et al. (1998) were land use and management practices, soil texture, and topography. Additional factors were identified by Cassell et al. (1998) as industrial activity, population and municipal waste technologies. Climate and hydrological factors play an important role. For example, P transport varies widely with individual storm events (Xue et al. 1998), and runoff can be variable even within a soil series (Pote et al. 1999). Methods that include assessment and integration of these factors, based on science wherever possible, are more likely to achieve the desired goals when applied in the field.

Nutrient management planning has been proposed by the United States Draft National Strategy for AFO's (USDA/USEPA 1998) as the way for AFO's to preserve water quality. Because it considers all nutrients, it can apply to all products applied to land, not just manure. It therefore has advantages in planning for food processing and municipal waste application, where P may not always be the nutrient of concern. Balanced nutrient management appears to be the only way to minimize the risk of excess nutrients in the soil or water, and therefore should be a long range goal Alberta.

The P index approach links landscape conditions, climate, management and soil factors at a site specific scale. It is a practical method that can provide a high degree of flexibility for operators, while minimizing risk to the environment. Because it provides the opportunity to work with producers and operators, it also has value as a tech transfer tool. To address the immediate issue of controlling P to minimize water pollution risk, a site specific approach, such as the P index, should be developed. The approach should be designed to be integrated into a watershed scale assessment.

## **7. Conclusions**

An increasing amount of research has identified phosphorus pollution, from both agricultural and non-agricultural sources, as a major threat to water quality. The Canada-Alberta Environmentally Sustainable Agriculture Agreement (CAESA) water quality assessment showed that phosphorus runoff from agricultural lands in Alberta is an issue that must be addressed. There is growing pressure to establish a strategy to manage phosphorus concentrations in the soil profile in order to minimize the movement of phosphorus into surface water resources.

Concerns about the contribution of agricultural operations to nutrient loading of water bodies in Alberta has arisen as the industry has intensified. The CAESA water quality assessment showed that nitrogen and phosphorus often exceeded water quality guidelines for protection of aquatic life in streams in high and moderately intensive agriculture areas. In addition, phosphorus also often exceeded water quality guidelines for protection of aquatic life in small lakes and irrigation canals in high intensity areas.

Alberta lakes, streams and rivers differ from water bodies found elsewhere in Canada or the United States. Many lakes and surface water bodies in Alberta are naturally highly productive (or eutrophic) and experience prolific growths of phytoplankton (algae) and macrophyte (weed). Phosphorus is the primary plant nutrient controlling the level of phytoplankton growth in western Canadian lakes. Consequently, surface waters in Alberta are extremely sensitive to further P enrichment either by internal or external sources.

The effects of P enrichment in aquatic systems in Alberta are complex. Eutrophication, has many negative effects on aquatic ecosystems that include killing of sport fish and other desirable aquatic species, increased water treatment problems, and decreased clarity and odour. These can have negative impacts on the recreation and tourism industry. Eutrophication also results in periodic massive growths of cyanobacteria, which poison water to the extent that it can kill livestock. Therefore, further nutrient enrichment of surface waters in Alberta can have significant economic and environmental implications on public and animal health, recreation, tourism and possibly even the agriculture industry.

The risk of P export from land to water increases as phosphorus loading of soil increases. There are clear relationships showing that the concentration of dissolved P in runoff and leaching increases with higher soil test P levels. Dissolved P concentrations can exceed Alberta Environment guidelines for runoff at soil test P levels that would be common for many agricultural soils.

Agriculture is not the only contributor. The City of Calgary has applied municipal biosolids to land at rates of over 500 kg ha<sup>-1</sup> of P. Even though application on the same piece of land was restricted to once every three years, the rates were still well above crop requirements.

The only practical way to reduce high P levels in soils is through crop uptake. The rate of decline varies with soil type, crop and management practices. Research results show that in the Northern Great Plains, the rate of decline of soil P ranges from 0.6 to 30 ppm per year. Soils in the County of Lethbridge that have undergone repeated heavy manure applications can typically have soil test P levels high enough that several decades could be needed to reduce P levels to moderate values.

There is currently enough scientific evidence to support development of a phosphorus management strategy in Alberta. Additional site specific information on P behavior in the landscape would greatly enhance the relevance of any strategy, however. Any phosphorus strategy must consider not only the current level of phosphorus in the soil, but also the site specific-nature of the soil, the type of management and the hydrologic factors associated with it.

Water quality is a complex issue of which agriculture is only one factor. Eutrophication is only one threat to water quality. While phosphorus may be the key factor in managing eutrophication, it is difficult to treat in isolation, especially when animal and organic wastes contain a variety of nutrients in forms that themselves have a range of complexities. Also while the application of manure and food processing wastes on to crop land are at the center of the issue, water quality can be at risk from the nutrient load in other crop land as well. In considering a strategy for managing land application of manure, food processing effluent and organic agricultural material, it should be part of larger overall strategy to address the broader issues of water quality.

A desirable long-range goal is to achieve a balanced nutrient management approach, however the factors needed to successfully implement this in Alberta must be defined. A timeline for change, and identification of interim steps and possible infrastructure support will be key components in the strategy. The United States is moving toward a comprehensive nutrient management planning approach to agricultural production and many states have started to address phosphorus loading. We have their experience to learn from as well.

The process to regulate the intensive livestock industry, and the expansion of the food processing industry in Alberta have accelerated the need to develop P-based standards for Alberta. Standards or guidelines would allow intensive livestock operations, food processing and other industries to plan future development and land application strategies in a manner that will protect water quality. The P index is a practical method that will provide the most flexibility for operators, while minimizing risk to the environment. It addresses the site specific factors that research has identified as critical to the accurate prediction of P export. A site specific approach, such as the P index, that can be integrated into a watershed scale assessment, appears to be the most suitable way to develop P-based standards for Alberta.

## References

- Alberta Agriculture, Food and Rural Development. 1999a. Addendum to the Code of Practice for the Safe and Economic Handling of Animal Manures. Alberta Agriculture, Food and Rural Development, Edmonton, Alberta. 21 pages.
- Alberta Agriculture, Food and Rural Development. 1999b. Ministry Business Plan 1999/2000 to 2001/2002. Alberta Agriculture, Food and Rural Development, Edmonton, Alberta.
- Alberta Environment. 1988. Guidelines for the land treatment of industrial wastes. Environmental Protection Services, Alberta Environment.
- Alberta Environmental Protection. (date unknown). Guidelines for the approval and design of natural and constructed treatment wetlands for water quality improvement.
- Alberta Environmental Protection. 1984. A practical guide to municipal wastewater irrigation - discussions and guidelines.
- Alberta Environmental Protection. 1997a. Guidelines for municipal wastewater irrigation (draft). Standards and Guidelines Branch, AEP. 22 pages.
- Alberta Environmental Protection. 1997b. Guidelines for the application of municipal wastewater sludges to agricultural lands (draft). Standards and Guidelines Branch, AEP. 29 pages.
- Alberta Environmental Protection. 1997c. Standards and guidelines for municipal waterworks, wastewater, and storm drainage systems. Standards and Guidelines Branch, EP. 358 pages.
- Alberta Environmental Protection and City of Red Deer. 1981. Rapid infiltration - a design manual.
- Anderson, A.-M., D.O. Trew, R.D. Neilson, N.D. MacAlpine and R. Borg. 1998. Impacts of agriculture on surface water quality in Alberta. Part I: Haynes Creek Study. Alberta Environmental Protection, Edmonton, Alberta.
- Barber, S.A. 1984. *Soil Nutrient Bioavailability*. Wiley-Interscience, John Wiley and Sons, New York.
- Barl, B. and R.C. McKenzie. 1995. Potato processing liquid effluent feasibility study of sustainable disposal alternatives. Alberta Agriculture, Food and Rural Development, Brooks, Alberta. CAESA Research Program project no. RES-044-93. 84 pages.
- Brookes, P.C., D.S. Powlson, and D.S. Jenkinson. 1984. Phosphorus in the soil microbial biomass. *Soil Biology and Biochemistry* 16: 169-175.
- Brookes, P.C., G. Heckrath, J. DeSmet, G. Hofman, and J Vanderdeelen. 1997. Losses of phosphorus in drainage water. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 253-272.
- Bowman, R.A. 1988. A rapid method to determine total phosphorus in soils. *Soil Sci. Soc. Am. J.* 52: 1301-1304.
- Canada-Alberta Environmentally Sustainable Agriculture Agreement. 1998. *Agricultural Impacts on Water Quality in Alberta; An Initial Assessment*. Alberta Agriculture, Food and Rural Development. Edmonton, Alberta. 95 pages.
- Cassell E.A. J.M. Dorioz, R.L.Kort, J.P. Hoffman, D.W. Meals, D. Kirschtel, and D.C. Braun. 1998. Modelling phosphorus dynamics in ecosystems: mass balance and dynamic simulation approaches. *J. Environ.Qual.* 27 (2): 293-298.
- Chang, C., H.H. Janzen and T. Entz. 1994. Long term manure application effects on nutrient uptake by barley. *Can. J. Plant Sci.* 74:327-330.

- Chauhan, B.S., J.W.B. Stewart and E.A. Paul. 1981. Effects of labile inorganic phosphate status and organic carbon additions of the microbial uptake of phosphorus in soils. *Can Jour. Soil Sci.* 61: 373-386
- Coale, F.J. 1998. Sample collection, handling, preparation and storage. In *Methods of Analysis for Soils Sediments, Residuals, and Waters. Draft*. G.M. Pierzynski ed. USDA-CSREES Regional Comm. SERA-IEG 17. Southern Cooperative Series Bull. TBA.
- Cooke, S.E. and E.E. Prepas. 1998. Stream phosphorus and nitrogen export from agricultural and forested watersheds on the Boreal Plain. *Can. J. Fish. Aquat. Sci.* 55:2292-2299.
- Culley, J.L.B, E.F. Bolton and V. Bernyk. 1983. Suspended solids and phosphorus loads from a clay soil: I. Plot studies. *J. Environ.Qual.* 12:493-498.
- Daniel, T.C., O.T. Carton and W.L. Magette. 1997. Nutrient management planning. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 205 - 224.
- Daniel, T.C., A.N. Sharpley and J.L. Lemunyon. 1998. Agricultural phosphorus and eutrophication: a symposium overview. *J. Environ.Qual.* 27 (2): 251-257.
- DeLuca, T.H. and D.K. DeLuca. 1997. Composting for feedlot manure management and soil quality *J. Prod. Agric.* 10:235-241.
- Dormaar, J.F. and C. Chang. 1995. Effect of 20 annual applications of excess feedlot manure on labile soil phosphorus. *Can. J. Soil Sci.* 75:507-512.
- Gburek, W.J. and H.B. Pionke. 1995. Management strategies for land based disposal of animal wastes: Hydrologic implications. In *Impact of Animal Manure and the Land-Water Interface*, K. Steele ed. Lewis Publishers, CRC Press, Boca Raton, Florida, pages 313-323.
- Gburek, W.J. and A.N. Sharpley. 1998. Hydrologic controls on phosphorus loss from upland agricultural watersheds. *J. Environ.Qual.* 27 (2): 267-277.
- Gibson, C.E. 1997. The dynamics of phosphorus in freshwater and marine environments. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 119-136.
- Greenwood, D.J., T.J. Cleaver, M.K. Turner, J. Hunt, K.B. Neindorf and S.M.H. Loquens. 1980. Comparison of the effects of of phosphate fertilizer on the yield: phosphate content and quality of 22 different vegetable and agronomic crops. *Jour. Agricultural Sci.* 95: 457-469.
- Hawkins, J.M.B. and D. Scholefield. 1996. Molybdate-reactive phosphorus losses in surface and drainage waters from permanent grassland. *J. Environ.Qual.* 25: 727-732.
- Haygarth, P.M. and S.C. Jarvis. 1999. Transfer of phosphorus from agricultural soils. *Adv. Agron.* 66:195-249.
- Heathwaite, A.L. 1997. Sources and pathways for phosphorus loss from agriculture. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 205 - 224.
- Heckrath, G., P.C Brookes, P.R. Poulton and K.W.T. Goulding. 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. *J. Environ.Qual.* 24: 904-910
- Hergert, G.W., D.R. Bouldin, S.D. Klausner and P.J. Zwerman. 1981a. Phosphorus concentration-water flow interactions in the effluent from manured land. *J. Environ.Qual.* 10:338-344

- Hergert, G.W., S.D. Klausner, D.R. Bouldin and P.J. Zwerman. 1981b. Effects of dairy manure on phosphorus concentrations and losses in tile effluent. *J. Environ.Qual.* 10:345-349
- Intensive Livestock Operations Committee. 1995. Code of Practice for the Safe and Economic Handling of Animal Manures. AGDEX 400/27-2. Alberta Agriculture, Food and Rural Development, Edmonton, Alberta. 38 pages.
- Janzen, H.H., M.J. Clapperton and A.M Johnston. 1993. Suppression of Zn and Cu uptake by phosphorus fertilization in long-term wheat rotations. In *Report of Progress of the Sustainable Cropping Systems Research Study*, R.C. Izurralde, H.H. Janzen and H.P. VanderPluym eds. Alberta Agricultural Research Institute, Edmonton, AB, pages 42-49.
- Jensen, M.B., P.R. Jorgensen, H.C.B. Hansen and N.E. Nielsen. 1998. Biopore mediated subsurface transport of dissolved orthophosphate. *J. Environ.Qual.* (5): 1130-1138.
- Johnston, A.E. and P.R. Poulton. 1997. Defining critical levels of available phosphorus for agricultural crops. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 441-445.
- Kissel, D.E., O.W. Bidwell and F.J. Klentz. 1982. Leaching classes of Kansas soils. *Kansas Experimental Station Bull. No. 6641*. Manhattan, Kansas.
- Kotak, B.G., S.L. Kenefick, D.L. Fritz, C.G. Rousseau, E.E. Prepas and S.E. Hrudey. 1993. Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. *Water Research* 27: 495-506.
- Kotak, B.G., E.E. Prepas and S.E. Hrudey. 1994. Blue green algal toxins in drinking water supplies: research in Alberta. *Lake Line* 14:37-40.
- Lawton, L.A. and G.A. Codd. 1991. Cyanobacteria (blue-green algae) toxins and their significance in UK and European waters. *Jour. of the Institute of Environmental Management* 5: 460-465.
- Lemunyon and Gilbert, 1993. The concept and need for a phosphorus assessment tool. *J. Applied Agriculture* 6 (4): 483-486
- Lennox, S.D., R.H. Foy, R.V. Smith and C. Jordan. 1997. Estimating the contribution from agriculture to the phosphorus load in surface water. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 430-431.
- McKenzie, R.H. 1989. Mechanisms controlling soil phosphorus dynamics in chernozemic and luvisolic soils. Ph.D. Thesis, Dept of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, 258 pages.
- McKenzie, R.H., L. Kryzanowski, K. Cannon, E. Solberg, D. Penney, G. Coy, D. Heaney, J. Harapiuk and N. Flore. 1995. Field evaluation of laboratory tests for soil phosphorus. Final Report. Alberta Agricultural Research Institute Project No. 90-M230. Alberta Agriculture, Food and Rural Development, Edmonton AB., pages 21-23.
- McKenzie, R.H., B.M. Olson, F.J. Larney and J.G. Davis. 1999. Applying manure based on either nitrogen or phosphorus content. In *Proc. 36th Annual Alberta Soil Science Workshop*, Calgary Alberta. Alberta Agriculture, Food and Rural Development, Edmonton AB.
- Meissner, R., H. Rupp and J. Seeger. 1997. Threat of phosphorus leaching from intensively farmed agricultural soils in the central reaches of the River Elbe. In *Phosphorus Loss from Soil to*

- Water, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 430-431.
- Menzel, D.W. and N. Corwin. 1965. The measurement of total phosphorus in seawater based on the liberation of organically bound fractions by persulfate oxydation. *Limnol. Oceanogr.* 10:280-282.
- Morgan, M.A. 1997. The behavior of soil and fertilizer phosphorus. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 137-150
- Murphy, J.H. and J.P. Riley. 1962. A modified single solution method for the determination of phosphorus in natural waters. *Anal. Chim. Acta.* 27:31-36.
- Nye, P. and P.B. Tinker. 1977. *Soil Movement in the Soil-Root System*. Studies in Ecology, Volume 4. Blackwell Scientific Publications, Oxford. 342pages.
- Olson, S.R. and L.E. Sommers. 1982. Phosphorus. In *Methods of Soil Analysis*. Part 2, 2nd Ed. Agronomy No. 9, American Society of Agronomy, Madison Wisconsin, 416 pares.
- Ozanne, P.G. 1980. Phosphate nutrition in plants- a general treatise. In *The Role of Phosphorus in Agriculture*, F.E. Khasawenh, E.C. Sample and E.J. Kamprath eds. American Soc. Agron., Crop Sci. Soc of Am. and Soil Sci. Soc. Am., Madison, Wisconsin. 910 pages.
- Parry, R. 1998. Agricultural phosphorus and water quality: A U.S. Environmental Protection Agency perspective. *J. Environ.Qual.* 27 (2): 258-261
- Pratt, P.F., F.E. Broadbent and J.P. Martin. 1973. Using organic wastes as nitrogen fertilizers. *California Agriculture* 27(6): 10-13.
- Prepas, E.E. and F.H. Rigler. 1982. Improvements in quantifying the phosphorus concentrations in lake water. *Can. J. Fish. Aquat. Sci.* 39:822-829.
- Prepas, E.E. and D.O. Trew. 1983. Evaluation of the Phosphorus-Chlorophyll Relationship for lakes off the precambrian shield in Western Canada. *Can. J. Fish. Aquat. Sci.* 40:27-35.
- Pote, D.H., T.C. Daniel, A.N. Sharpley, P.A. Moore Jr., D.R. Edwards and D.J Nichols. 1996. Relating extractable soil phosphorus to phosphorus losses in runoff. *Soil Sci. Am. J.* 60:855-859.
- Pote, D.H. and T.C. Daniel. 1998. Analyzing for total P and total dissolved P in water samples. In *Methods of Analysis for Soils Sediments, Residuals and Waters*. Draft. G.M. Pierzynski ed. USDA-CSREES Regional Comm. SERA-IEG 17. Southern Cooperative Series Bull. TBA.
- Pote, D.H., T.C. Daniel, D.J.Nichols, A.N. Sharpley, P.A. Moore Jr., D.R. Edwards. 1999. Relationship between phosphorus levels in three ultisols and phosphorus concentraion in runoff. *J. Environ Qual.* 28:170-175.
- Ryden, J.C., J.K Syers and R.F. Harris. 1973. Phosphorus in runoff and streams. *Adv. Agron.* 25:1-45.
- Self-Davis, M.L. and P.A. Moore, 1998. Determining water soluble phosphorus in animal manure. In *Methods of Analysis for Soils Sediments, Residuals and Waters*. Draft. G.M. Pierzynski ed. USDA-CSREES Regional Comm. SERA-IEG 17. Southern Cooperative Series Bull. TBA.
- Self-Davis, M.L., P.A. Moore and B.C. Joern. 1998. Determination of water and/or salt-extractable phosphorus. In *Methods of Analysis for Soils Sediments, Residuals and Waters*. Draft. G.M. Pierzynski ed. USDA-CSREES Regional Comm. SERA-IEG 17. Southern Cooperative Series Bull. TBA.

- Sharpley, A.N. 1993. An innovative approach to estimate bioavailable phosphorus in agricultural runoff using iron oxide-impregnated paper. *J. Environ. Quality* 22: 597-601.
- Sharpley, A.N. 1995. Dependence of runoff phosphorus on soil phosphorus content. *J. Environ. Qual.* 24: 920-926
- Sharpley, A.N. and J.K. Syers. 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea applications and grazing animals. *New Zealand J. Agric. Research* 22:127-131
- Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. *J. Environ. Qual.* 20:30-35.
- Sharpley, A.N. and S.J. Smith. 1993. Wheat tillage and water quality in the Southern Plains. *Soil and Tillage Res.* 30: 33-48
- Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. *J. Environ. Qual.* 23(3): 437-451.
- Sharpley, A.N.; T.C. Daniel; J.T. Sims and D.H. Pote. 1996. Determining environmentally sound soil phosphorus levels. *J. Soil Water Conserv.* 51(2): 160-166.
- Sharpley, A.N. and S. Reikolainen. 1997. Phosphorus in agriculture. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C. Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 1-53
- Sharpley, A.N., P. Kleinman, W. Gburek and G. Folmar. 1999. Developing and integrated nutrient management approach for land-applied manure at a watershed scale. In *Manure Management '99; Proceedings of a Tri-provincial Conference on Manure Management*. Saskatchewan Agriculture and Food, Saskatoon, Saskatchewan. pages 478-488
- Sibbsen, E. and A.N. Sharpley. 1997. Setting and justifying upper critical limits phosphorus in soils. In *Phosphorus Loss from Soil to Water*, H. Tunney, O.C. Carton, P.C. Brookes and A.E. Johnston eds. CAB International, New York. pages 151-176.
- Sims, J.T., R.R. Simard and B.C. Joern. 1998. Phosphorus loss in agricultural drainage: Historical perspective and current research. *J. Environ. Qual.* 27 (2): 277-293
- Sonzogni, W.C., S.C. Chapra, D.E. Armstrong and T.J. Logan. 1982. Bioavailability of phosphorus inputs to lakes. *J. Environ. Qual.* 11:555-563.
- Stamm, C., H. Fluhler, R. Gachter, J. Leuenberger and H. Wunderli. 1998. Preferential transport of phosphorus in drained grassland soils. *J. Environ. Qual.* 27 (3): 515-521.
- Stewart, J.W.B. and A.N. Sharpley. 1987. Controls on dynamics of soil and fertilizer phosphorus and sulfur. In *Soil Fertility and Organic Matter as Critical Components of Production Systems*, R.F. Follert, J.W.B. Stewart and C.V. Cole eds. SSSA Special Pub. No. 19, American Society of Agronomy, Madison Wisconsin. pages 101-121.
- Stewart, J.W.B. and R.B. McKercher. 1982. Phosphorus cycle. In *Experimental Microbial Ecology*, R.G. Burns and J.H. Slater eds. Blackwell Scientific Publications, Oxford, pages 221-238.
- Stevens, R.G., T.M. Sobecki and T. L. Spofford. 1993. Using the phosphorus assessment tool in the field. *J. Prod. Agric.* 6 (4): 487-492.
- Syers, J.K. and D. Curtin. 1988. Inorganic reactions controlling phosphorus cycling. In *Phosphorus Cycles in Terrestrial and Aquatic Ecosystems*, H. Tiessen ed. UNDP, Saskatchewan Institute of Pedology, Saskatoon, Saskatchewan, pages 17-29.

- Tiessen, H. and J.O. Moir. 1993. Characterization of available P by sequential extraction. In *Soil Sampling and Methods of Analysis*, M.R. Carter ed. Canadian Soc. of Soil Science. Lewis Publishers, Boca Raton, Florida. 824 pages (pages 75-87).
- Tinker, P.B. 1980. Role of rhizosphere microorganisms in phosphorus uptake by plants. In *The Role of Phosphorus in Agriculture*, F.E. Khasawneh, E.C. Sample and E.J. Kamprath eds. American Soc. of Agron., Crop Sci. Soc. of Am. and Soil Sci. Soc. Am., Madison, Wisconsin, 910 pages.
- Tisdale, S.L., W.L. Nelson and J.D. Beaton. 1985. *Soil Fertility and Fertilizers*. Chapter 6. McMillan Publishing Co. New York. 755 pages.
- Uhlmann, D. and E. Albrecht. 1968. Biogeochemische faktoren der eutrophierung van trinkwassentalsperren. *Limnologica* (Berlin) 6: 225-245
- USDA/USEPA. 1998. Draft National Strategy for Animal Feeding Operations. United States Department of Agriculture, United States Environmental Protection Agency, September 11, 1998 draft. 47 pages.
- Vaughan, C.E. and M.B. Jones. 1980. Soil phosphorus tests on California subclover-annual grass pastures. *Soil Sci.* 130: 307-313
- Withers, P.A., R.J. Unnein, J.P. Grylls and R. Kane. 1994. Effects of withholding phosphate and potash fertilizer on grain yield of cereals and on plant available phosphorus and potassium on calcareous soils. *European J. Agron.* 3: 1-8.
- Xue, Y., M.B. David, L.E. Gentry and D.A. Kovacic. 1998. Kinetics and modelling of dissolved phosphorus export from a tile-drained agricultural watershed. *J. Environ. Qual.* 27 (4): 917-922.