Risk of Phosphorus Desorption from Canadian Agricultural Land: 25-Year Temporal Trend

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Phosphorus (P) use in excess of crop needs may impact surface water quality and contribute to eutrophication. However, P loss from agricultural land to water has never been estimated at the Canadian national scale. In this paper, the risk of P desorption from Canadian agricultural land is assessed by the source component of the indicator of risk of water contamination by P (IROWC-P). The IROWC-P source component (P source) characterized the mobilization potential of soluble P and integrated four models of P desorption by water for dominant agricultural soil series of Canada on the soil landscape of Canada polygon scale (1:1,000,000). The objective of our study was to describe and evaluate a standardized method for deriving the P source component. The P source was assessed over 5-yr intervals from 1981 to 2006 for scientifically based knowledge by relating annual P balance values, soil test P (STP) analyses, soil P saturation index, and Self-Davis water extractable P extraction values. Results show trends of soil P enrichment for most Canadian provinces over the 25-yr period but also an increased percentage of farmland classified above the water extractable soil P environmental threshold of 4 mg P kg⁻¹. The Canadian Prairies and Ontario showed small P_source values and almost no farmland above the environmental threshold. Quebec and the Atlantic Provinces had P source values that exceeded the environmental threshold in 2006; more than 33% of farmland is classified above the environmental threshold value.

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HOSPHORUS USE in Canadian agricultural systems con-tributes to the economic viability of agricultural enterprises and helps to maintain a secure food supply because it is an essential nutrient for plant growth. Phosphorus addition to the soil through inorganic P fertilizers, manures, and other wastes sustains and improves crop yields. Phosphorus fertilizers have been used extensively since the last century; availability was limited during World War II, but fertilization became more intensive in the 1950s and thereafter (MacDonald and Bennett, 2009). Phosphorus is also an essential nutrient for animals. The most widespread of all mineral deficiencies affecting livestock production is related to P. Since the late 1950s, researchers in animal nutrition have recommended P supplements to promote livestock production (Gueguen, 2005). In regions of intensive production, this trend has increased the nutrient surplus and, consequently, the risk of P transport from agricultural fields to surface water bodies.

Phosphorus is transported from agricultural land to surface water through surface runoff and from infiltrating water or tile drainage. In natural freshwater systems, P occurs in very low concentrations but may vary significantly with stream size and ecosystem characteristics (Chambers et al., 2008). Excessive amounts of P in surface fresh water contribute to the eutrophication of rivers and lakes and to cyanobacteria blooms (Carpenter et al., 1998). The risk of water contamination is considered greatest when soil tests show high P levels (Sharpley et al., 1996; Salvano et al., 2009), the ability of soils to retain P is low (Pellerin et al., 2006; Beauchemin et al., 2003; Khiari et al., 2000), P input and solubility are high (McDowell et al., 2001; Kleinman and Sharpley, 2002; Shober and Sims, 2007), the susceptibility for runoff is high and the macropore

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Abbreviations: ASAE, Society of Agricultural Engineers; CoA, Census of Agriculture; DSPS, degree of soil P saturation; FEMS, Farm Environmental Management Surveys; IROWC-P, indicator of risk of water contamination by P; LFPS, Livestock Farm Practices Survey; PSC, P sorption capacities; PSI, soil P saturation index; Pw, water extractable P; Pw Self-Davis, Self-Davis water extractable P analysis; SLC, Soil Landscape of Canada; STP, soil test P.

flow is high (Jarvis, 2007), soil erosion is high (Buczko and Kuchenbuch, 2007), and the connections to surface water and artificial drainage are dense (Gburek and Sharpley, 1998). The differences in P flux magnitudes between locations depend on cropping practices (Sharpley et al., 2001), soil properties (Little et al., 2007), and climate (Sharpley et al., 2008).

Since 2003, Agriculture and Agri-Food Canada, through its National Agri-Environmental Health Analysis and Reporting Program, has worked on developing an indicator of risk of water contamination by P (IROWC-P) for Canadian agricultural land. The IROWC-P has been developed to estimate the risk of water P contamination from agriculture in Canada and to assess how this risk has changed over 25 yr (1981–2006) within the agricultural watersheds. The goal is to integrate the most important sources and transport factors identified in the literature where national data are available (van Bochove et al., 2005, 2006, 2007, 2010a, 2010b). The spatial (Soil Landscape of Canada v.3.1; Soil Landscapes of Canada Working Group, 2006) and the temporal (calendar-year) scales on which the P source component was assessed greatly limit the number of databases that covered, on a national basis, the factors affecting soil P levels, such as the time of year when manure is applied and the type of mineral P fertilizer used. Soil P level is considered to be a good indicator because it integrates most of the agricultural management practices that affect P_source values. Thus, the development of the source component of IROWC-P was based on estimated soil P concentrations in Canadian agricultural soils. While soil P data are essential to identify critical sources of P contamination, documentation at the national level is poor. Some explanations for this poor documentation include the gradual transfer of soil-testing capacities from governmental to private laboratories since the 1990s in most provinces; identifying soil samples by postal code and not including localization, soil series, and crop

information from analytical laboratories; and confidentiality issues of agricultural producers.

Several methods are used to assess soil P levels, including soil test P (STP) (Olsen et al., 1954; Mehlich, 1984; van Lierop, 1988), degree of soil P saturation (DSPS) (Breeuwsma and Silva, 1992), and soil P saturation index (PSI) (Sharpley, 1995; Giroux and Tran, 1996). Many factors affect the sorption-desorption kinetics of soil P: soil chemical properties; fertilizer and biosolid applications; crop and tillage management practices; organic matter content; exchangeable Al, Fe, and Ca; pH; clay content and mineralogy; initial soil P; and soil P sorption capacity. The DSPS and PSI (P/Al Mehlich-III) methods are well defined for acidic soils because their P sorption capacities (PSC) (Bache and Williams, 1971) are well characterized. In contrast, although calcareous soils are common on Canadian agricultural land (Fig. 1), their PSC is not well defined and the assessment of their DSPS, with or without a PSI, is challenging and controversial (Ige et al., 2005). Olsen and Kelowna STP (Olsen et al., 1954; van Lierop, 1988) are currently used for assessing soil P levels of calcareous soils in Canada. Well-defined relations between STP and P inputs (manure and fertilizers) are widely reported in the scientific literature, so annual P balance values can be used to estimate annual soil P enrichments on a STP basis (e.g., Giroux et al., 2002; Zhang et al., 2006; Messiga et al., 2010). However, the need to merge all available soil P information into a single and integrated indicator, referred to as P source, was essential to develop a national P_source component of the IROWC-P. The Self-Davis water extractable P analysis (Pw Self-Davis) was chosen because in addition to being a relatively simple extraction analysis that can easily be inserted into a soil analysis program, it correlates well with STP and PSI (van Bochove et al., 2010a) and P concentration values in drainage waters by surface infiltration (Self-Davis et al., 2000) and showed a strong relationship with dissolved P concentration collected runoff waters with Ontario



Fig. 1. Agricultural Soil Landscape of Canada v.3.1 (colored area) (Soil Landscapes of Canada Working Group, 2006) of the 10 Canadian provinces classified into four soil groups as a function of the soil parental material and the predominant soil test P analyses (Olsen or Kelowna) or soil P saturation index (P/Al Mehlich-III).

soils (Wang et al., 2010). Also, Pw Self-Davis shows a strong correlation with the water extractable soil P (Pw) Sissingh analysis (Sissingh, 1971), which relates well with the PSI (P/Al Mehlich III) (van Bochove et al., 2010a).

The objective of this paper is to describe the techniques used to predict the source component for assessing the risk of soluble P loss (P_source) from agricultural soils to runoff and drainage waters and to validate these predictions against median soil test values for each province at the soil landscape of Canada polygon scale from 1981 to 2006. The paper also identifies critical regional areas across the country on the Soil Landscape of Canada (SLC) scale where the risk of P desorption is high and where further investigation is required to protect surface water quality from P contamination by agriculture.

Materials and Methods

This paper deals with the P source, which is one of the two IROWC-P components, the other being the transport-hydrology component (van Bochove et al., 2010a). The P_source is defined as the quantity of soluble P that can potentially be desorbed from agricultural soils in runoff and leaching waters and ultimately reach surface water bodies. The P_source is expressed as a Pw Self-Davis value (mg P kg⁻¹) and integrates the determined initial STP values, cumulative P balances, and P sorption capacities of soils. Natural levels of STP values are rarely known. Giroux et al. (2008) published estimates of natural Mehlich-III levels for three broad textural categories for Quebec acidic soils. For calcareous soils, values representing the very low fertility class in fertilization grids of Alberta, Ontario, and Manitoba were assumed to be close to natural soil P levels. More information can be found in van Bochove et al. (2010a). The initial periods, defined as the early stage of intensive agriculture, were fixed according to the historical data available: 1966 for the Atlantic Provinces, Quebec (QC) and Ontario (ON); and 1976 for Manitoba (MB), Saskatchewan (SK), Alberta (AB), and British Columbia (BC).

Data Sources

The use of homogenous national databases was critical to prevent regional distortion and to obtain the most realistic risk rating results on the national scale. Therefore, soil and landscape attributes (sand, silt, and clay percentages, pH, soil order classification, and parent material) were obtained from the SLC (SLC v3.1) database (1:1,000,000 map scale) (Soil Landscapes of Canada Working Group, 2006). The area covered by the 2780 SLC agricultural polygons corresponds to approximately 1,270,646 km² and is shown in Fig. 1. The SLC polygons are characterized by climatic and soil factors and hence differ in their areas. The mean area of all SLC agricultural polygons is 457 km²; 10% have an area smaller than 73 km² and 10% have an area larger than 932 km². A P balance assessment at the SLC scale was made using data and information coming from different sources. Briefly, land use and number of animals were extracted mainly from the 1981 to 2006 5-yr Census of Agriculture (CoA) reported on the SLC scale. Land and farm management practices such as number of pasturing animals, periods of pasture, manure management, and time of manure application were extracted from the 2001 and 2006 Farm Environmental Management Surveys (FEMS) (Statistics Canada, 2008), and the 2005 Livestock Farm Practices Survey (LFPS) (Sheppard et al., 2009). The FEMS is reported on the provincial scale, whereas LFPS is reported on an ecoregion scale. An extensive description of the P balance assessment is available in van Bochove et al. (2010a). The reporting of the CoA variables on SLC polygons represents an aggregation of the data at a finer scale but could include spatial errors when a census division is overlaid by more than one polygon. On the other hand, the management information extracted from the FEMS and LFPS surveys are used uniformly at a broad spatial scale and may not represent regional or local realities. Thus, the interpretation of the results at a scale larger than the provincial scale has its limitations and should be performed cautiously. The use of the phytase enzyme in monogastric animal diet is an animal feed practice that enhances animal P uptake and lowers the P content of manure. Because the extent of this practice is not uniformly reported on the national scale, phytase use is estimated for each Canadian province based on a survey sent to animal feed companies (van Bochove et al., 2010a). Briefly, the estimated swine herd fractions receiving phytase in their feed in Canada were 0, 25, and 50% in 1996, 2001, and 2006, respectively, except for QC, where they were 12, 83, and 90%. For poultry herd fractions, values were 0, 0, and 25% during the same year intervals, except for QC, where they were 0, 3, and 54%. A series of soil analyses including STP (Mehlich-III, Olsen or Kelowna), P/Al Mehlich-III, and Pw Self-Davis was applied to 443 Ap soil horizon samples (Soil Classification Working Group, 1998) or 0- to 15-cm-depth soil samples obtained from regional soil collections covering a wide range of agricultural soils from 8 of the 10 provinces; New Brunswick (NB) and Newfoundland and Labrador (NL) were not represented. These data were used to derive STP and PSI values and Pw Self-Davis correlations. Samples from soil collection had unspecified cropping or management history. The time lag between the collection of soil samples and their subsequent analysis indicates that they were not influenced by recent fertilizer and manure applications.

Annual Phosphorus Balance

Long-term experiments using excessive P fertilization show soil P enrichment (McCartney et al., 1998; Giroux et al., 2002; Chang et al., 2005; Heming, 2007). Therefore, we estimated the annual amount of residual P (P_Balance) that contributed to the soil P enrichment for each SLC agricultural polygon using the following equation:

$$P_Balance = P_Man + P_Min - P_Rem$$
[1]

where P_Man and P_Min are applied P (kg P) from animal manure and mineral fertilizers, respectively, and P_Rem is P removed (exported) by harvests (kg P). The detailed methodology used to calculate the P balance value is thoroughly described in van Bochove et al. (2010a) and summarized below.

The P_Man data are reported by the CoA at the SLC polygon scale. The P_Man data were estimated from the number of heads for major livestock categories in Canada (bovine, porcine, poultry, ovine, and equine), and manure and nutrient production coefficients from the American Society of Agricultural Engineers (ASAE, 2003). The use of phytase enzymes in monogastric animal feed has been a common practice in Canada since 1996 (BPR Inc., 2005). Phytase influences the P content of manure, but this practice was not integrated in the ASAE coefficients. Therefore, reduction factors of 30 and 25% were applied on manure P content for the porcine (Grandhi, 2001; Knowlton et al., 2004) and poultry (Lefrançois and Ouyed, 2004) herd fractions, respectively, fed with phytase. National and regional values of P_Man are provided in Table 1.

Provincial estimates of mineral P fertilizer use for 1981 to 2001 were obtained from Korol (2002) and from the Canadian Fertilizer Institute for 2006 (Canadian Fertilizer Institute, 2007). These estimates were downscaled to the SLC polygon level with a series of adjustments based on the following four general assumptions:

- 1. Phosphorus requirements for all major crops—grain cereals, corn (*Zea mays* L.), soybean [*Glycine max* (L.) Merr.], hay, vegetables, and potato (*Solanum tuberosum* L.)—at the SLC polygon scale follow provincial fertilization grids.
- 2. Manure is used first to meet crop nutrient (N and P) requirements, except for potato and vegetable crops where no other source of nutrients other than mineral fertilizer were used. The manure use was based on available N and P. The N and P availability coefficients integrated a series of parameters (van Bochove et al., 2010a) from which the most important ones were the animal category, the manure state (solid or liquid), the spreading period, the incorporation delay, and the soil textural class.
- 3. The maximum quantity of manure that can be applied corresponds to 75% of crop N fertilizer requirements.
- 4. The amount of mineral P fertilizer (P_Min) used in a given SLC polygon is derived as the product of the provincial estimate of mineral P fertilizer use and the "agronomic" ratio of the SLC polygon, given by the amount of P necessary to complete the crop P requirement of the SLC polygon divided by the total amount of P necessary to complete the crop P requirements of all SLC polygons of the province.

National and regional values of P_Min are shown in Table 1. The P_Rem component was calculated using the cropland areas given by the CoA, the crop yield data aggregated at the SLC polygon scale, and the Canadian Fertilizer Institute nutrient

removal coefficients at a regional scale (Canadian Fertilizer

Cumulative Phosphorus Balance

Institute, 2001).

A cumulative P balance was determined for each SLC agricultural polygon by summing its annual P balance values. The cumulative P balance calculation started in 1966 for eastern Canada since annual P surplus values were available from the

International Plant Nutrition Institute (2010), and in 1976 for BC, AB, SK, and MB, for which no earlier data were estimated. The P balance annual values for intervals between CoA years were estimated by means of linear interpolation.

Soil Phosphorus Enrichment

The national scale of this project necessitated the grouping of soils on a provincial basis according to the most prevalent STP used for crop fertilization management. A literature review covering the spectrum of Canadian soils and common STP methods was conducted to assess the general trend of soil P enrichment following repeated P applications (van Bochove et al., 2010a). Two relationships tested against published STP data were retained for this study: calcareous soils displayed an enrichment coefficient ratio with a median value of 5.8 mg kg⁻¹ P-Olsen (n = 15, coefficient of variation = 0.73) per 100 kg P applied in excess to crop needs, and neutral-to-acidic soils had a median value of 12.1 mg kg⁻¹ P-Mehlich-III (n = 31, coefficient of variation = 0.95). These enrichment coefficients were used to estimate the STP enrichment as a function of the cumulative P balance. Because of the scarcity of studies relating STP enrichment values with excess P inputs on soils, analyzed with the Kelowna method for managing crop P fertilization, no enrichment coefficient ratio could be assessed in the same way as described for the soil tested with Olsen and Mehlich-III. However, the Mehlich-III enrichment coefficient ratio of 12.1 mg kg⁻¹ per 100 kg P was transferred into a Kelowna enrichment coefficient ratio of 8.3 mg kg⁻¹ per 100 kg P using the relationship established by Kowalenko (2007).

Canadian agricultural soils were classified into four groups based on principal soil analysis used to manage crop P fertilization in Canada using soil name attributes (localization and parent material chemical property): Olsen- and Kelowna-STP analyses and the soil P saturation index (PSI), P/Al Mehlich-III (Fig. 1). The first group of soils (soil group 1) consists of acidic-to-neutral soils of QC, NB, Nova Scotia (NS), Prince Edward Island (PE), and NL (PSI = P/Al Mehlich-III); the second group (soil group 2) consists of calcareous soils of MB, ON, and QC (STP = Olsen-P); the third group (soil group 3) consists of acidic-to-calcareous soils of BC, AB, and SK (STP = Kelowna-P); and the fourth group (soil group 4) consists of acidic soils of BC (PSI = P/Al Mehlich-III). The two groups of soils using Mehlich-III extracts (groups 1 and 4) were kept separated, as is explained more thoroughly below, based on their significantly different relationship with water extractable soil P (P_w). The initial STP was assigned to low class values from respective provincial P fertilization tables, except for QC, where natural P level estimates were available for different soil

Table 1. Average amounts of mineral and manure P applied per hectare of cropland in Soil Landscape of Canada polygons with more than 5% agriculture (SLC v3.1; Soil Landscapes of Canada Working Group, 2006).

Car	nada	Eastern	Canada	Western Canada		
Mineral	Manure	Mineral	Manure	Mineral	Manure	
		kg P	ha ⁻¹			
4.99	4.85	10.68	13.57	3.93	3.22	
5.46	4.49	11.76	13.81	4.43	2.97	
4.36	4.66	9.57	13.46	3.56	3.30	
4.97	5.17	7.56	13.35	4.57	3.90	
4.63	5.22	7.51	13.03	4.21	4.07	
4.17	5.23	6.11	12.79	3.89	4.15	
	Car Mineral 4.99 5.46 4.36 4.97 4.63 4.17	Mineral Manure 4.99 4.85 5.46 4.49 4.36 4.66 4.97 5.17 4.63 5.22 4.17 5.23	Canada Eastern Mineral Manure Mineral 4.99 4.85 10.68 5.46 4.49 11.76 4.36 4.66 9.57 4.97 5.17 7.56 4.63 5.22 7.51 4.17 5.23 6.11	Canada Eastern Canada Mineral Manure Mineral Manure 4.99 4.85 10.68 13.57 5.46 4.49 11.76 13.81 4.36 4.66 9.57 13.46 4.97 5.17 7.56 13.35 4.63 5.22 7.51 13.03 4.17 5.23 6.11 12.79	Canada Eastern Canada Western Mineral Manure Mineral Manure Mineral 4.99 4.85 10.68 13.57 3.93 5.46 4.49 11.76 13.81 4.43 4.36 4.66 9.57 13.46 3.56 4.97 5.17 7.56 13.35 4.57 4.63 5.22 7.51 13.03 4.21 4.17 5.23 6.11 12.79 3.89	

textural groups (Giroux et al., 2008). This assumption may well represent the intensification of the agriculture in the Prairies but also underestimates historical P loads in the St. Lawrence River Basin, where some regional imbalance appeared in the 1950s (MacDonald and Bennett, 2009). Hence, the legacy of STP in a given SLC polygon is assessed using the following equation:

$$STP_{t} = X_{0} + (PER \times CumulativeP_Balance_{t})$$
[2]

where STP_t is the estimated STP value at year t, X_0 is the estimated natural or initial level of P in the soil (mg kg⁻¹), PER is the P enrichment ratio (P enrichment coefficient from the literature), and the CumulativeP_Balance_t is the sum of the annual P balance values from 1966 or 1976 up to year t.

For soil groups 1 and 4 that are characterized by Mehlich-III extracts, STP, values were converted to PSI, values using empirical conversion factors specific to the soil textural class (van Bochove et al., 2010a). This conversion is supported by studies indicating that PSI, both an agronomic and an environmental indicator, displays a better correlation with P_w than does P Mehlich-III (Khiari et al., 2000; Sims et al., 2002; Nair et al., 2004; Pellerin et al., 2006). The conversion is performed with the following equation:

$$PSI_{t} = STP_{t} \times ECF$$
 [3]

where PSI_t is the estimated PSI value at year *t*, STP_t is the value obtained with Eq. [2] for soil groups 1 and 4, and ECF is the empirical conversion factor specific to the soil textural class. Briefly, ECF values of 0.206, 0.171, and 0.146 were used for fine-textured, medium-textured, and coarse-textured soils, respectively (van Bochove et al., 2010a).

Soil Phosphorus Potentially Desorbed by Water (Pw)

The risk of P transport is a function of soil P richness, which is generally expressed as STP values but also as a function of specific soil properties affecting soil P sorption capacity, such as clay content (Cox and Hendricks, 2000), the Ca and Mg contents of calcareous soils (Bache and Williams, 1971; Kleinman and Sharpley, 2002), and the Fe and Al contents of acidic soils (Breeuwsma and Silva, 1992; Khiari et al., 2000). Other studies showed a strong link between Pw Self-Davis concentrations and runoff P in a variety of landscapes (Pote et al., 1996; Hooda et al., 2000; Davis et al., 2005; Vadas et al., 2005). Moreover, strong relationships were found between the STP methods commonly used in Canada, namely, Olsen-P and Kelowna-P extraction methods, as well as the P saturation index (P/Al Mehlich-III) and the Pw Self-Davis (van Bochove et al., 2010a). The use of an environmentally relevant indicator, such as water extractable P concentration, as a proxy of soil P potentially transportable to surface water, was needed to integrate the different STP and P saturation index analyses used in Canada into a P desorption model.

Therefore, 10 soil P desorption models were developed to relate STP (Kelowna and Olsen) analyses and P saturation index (P/Al-Mehlich 3) values to Pw Self-Davis using the 443 soil samples classified in the four soil groups with respect to soil textural classification. Previous studies showed that soil textural classification is a good indicator of P sorption and desorption characteristics (Pellerin et al., 2006; Leclerc et al., 2001). All the modeled relationships were forced through the origin provided that the water extractable soil P only represents a fraction of the P readily available to crops (estimated by STP analysis). The P_source expressed as water extractable soil P (mg kg⁻¹) was estimated using the following equation:

$$P_Source_{t} = [STP_{t}or_PSI_{t}] \times m$$
[4]

where P_Source_t is the potentially available P for desorption at time t, STP_t is the estimated STP value (Olsen or Kelowna) at year t, PSI_t is the estimated P saturation index value (P/ Al Mehlich-III) at time t, and m is the slope of the linear relationship between Pw Self-Davis and P-Olsen, P-Kelowna, or P/Al-Mehlich-III. The omission of the intercept was made to simplify the calculation and because we hypothesized that no water extractable P should be found when no STP is extracted (rejecting the case of a positive intercept) and that a minimal inherent risk exists at low STP values (rejecting the case of a negative intercept).

The P_source value was calculated for each soil series of every agricultural SLC polygon and reported as an area-weighted mean on either the SLC polygon or the provincial scale. A P_source classification was established in proportion to an environmental threshold value. Based on the reliable critical environmental value of 9.7 mg P kg⁻¹ soil of water extracted P using the Sissingh method (Sissingh, 1971; Khiari et al., 2000; Pellerin et al., 2006) and the strong correlation (R^2 , 0.91) between the Pw Sissingh and the Pw Self-Davis assessed with 50 mineral soils from across Canada (van Bochove et al., 2010a), a critical environmental threshold value of 4 mg P kg⁻¹ for Pw Self-Davis was retained for this study.

Results and Discussion

P-Balance Trends over 25 Years (1981–2006)

In Western Canada, temporal trends in annual P balance show that average P surpluses increased regularly during the 25-yr period in the provinces of AB, SK, and MB, compared with BC, where P surpluses showed no clear temporal variation (Fig. 2a and 2b). In 2006, all four provinces showed an average P surplus of 5 kg P ha⁻¹. In both AB and SK, the total agricultural area increased, but bovine and porcine productions increased at a faster rate. In 1981, the porcine industry in the provinces of AB, SK, and MB counted approximately 1.2 million, 0.6 million, and 0.9 million animal heads, respectively, while these figures were up to 2.1 million, 1.4 million, and 2.9 million in 2006. In these provinces, livestock population in the bovine industry increased by about 50% during the same period. These increases resulted in larger manure-P amounts, while the fertilizer sold in each province and the exported P in harvested crops remained relatively stable, thus increasing the P balance. In BC, the number of heads in 2006 for porcine production declined to almost half that of 1981, whereas the bovine population remained nearly constant. Comparatively, the poultry industry encountered a substantial increase of 80% between 1981 and 2006.

The P balance values from Eastern Canada show a net decrease after 1991 in QC and ON, but the temporal trend was less evident for the Atlantic Provinces (NB, NS, PE, and NL) (Fig. 2b and 2c). The decrease in QC coincided with the implementation of efficient fertilizing regulations and use of phytase in pig and poultry feeding. In ON, with the exception of year 2001, the temporal trend followed the same pattern as



Fig. 2. Box-and-whisker diagrams of estimated annual P balance values per hectare of cropland for each agricultural Soil Landscape of Canada polygon in the provinces of Canada (1981–2006): (a) British Columbia (BC), Alberta (AB), and Saskatchewan (SK); (b) Manitoba (MB) and Ontario (ON); and (c) Quebec (QC) and New Brunswick (NB), Nova Scotia (NS), Prince Edward Island (PE), Newfoundland and Labrador (NL). Dashed lines represent the mean value; the whiskers, the 10th and 90th percentiles; × symbols represent the 5th and the 95th percentiles.

for QC due to environmental concerns that led to establishing provincial regulations. The year 2001 was an exceptionally dry year, and crop yields were seriously affected; the lower-thanexpected fraction of P exported by crops could explain the relatively high value of the P balance. Extensive efforts through governmental programs and regulations in recent years in QC and ON have been aimed at reducing agricultural P point source contamination from manure storage structures as well as diffusing the P_sources of contamination during manure spreading. In these two provinces, P is currently part of nutrient management plans developed for farming operations to reduce the risk of contaminating adjacent surface water bodies (implemented in 1997 in QC with the Regulation respecting the reduction of pollution from agricultural sources, Q.c.Q-2,r.18.2, and in 2002 in ON with the Nutrient Management Act). In 2006, QC and ON showed average P surpluses of 5 kg P ha⁻¹ and 2 kg P ha⁻¹, respectively. The pressure on agricultural land remained very high in the Atlantic Provinces, with a P balance ranging from 15 to 20 kg P ha⁻¹ over the 25-yr period under investigation. The expansion of porcine and poultry industries combined with a 10% decrease in total agricultural area partially explains the remaining high P balance values.

Some annual P balance mean values shown in Fig. 2 compare well with results from other studies. The 2001 annual P balance for the province of QC (8 kg P ha⁻¹, Fig. 2c) compares positively with MacDonald and Bennett (2009), who used similar data sources (e.g., Census of Agriculture, ASAE manure coefficients, FEMS data) and a similar surface balance approach (P inputs of manure and fertilizer minus P outputs by harvested crops) for 76 subwatersheds of the St Lawrence subbasin scale. The mean value of the 2001 annual P balance for MB (5 kg P ha⁻¹, Fig. 2b) is also in the same range as other results $(2.8 \text{ kg P ha}^{-1})$ calculated using comparable data sources (crop insurance reported values at the county level, International Plant Nutrition Institute STP and P balance values) at the Manitoba agricultural region scale (T. Jensen, personal communication, 2000). Slight variances in P balance mean values originate from factors such as dissimilarity in the manure excretion coefficients, crop uptake coefficients, yield data sources, and fertilizer use estimations. Still, limitations in available data for a specific year may result in discrepancies between assessed and real values, although these effects would be less pronounced in the long term as more data becomes available.

Cumulative Phosphorus Balance

Cumulative P balance trends for each province are shown in Table 2. All provinces show positive trends between 1981 and 2006, with the exception of AB and SK, where positive trends began in 1986 and 1996, respectively. The environmental pressure of P accumulation in agricultural soils in the Atlantic Provinces (NB, NS, PE, and NL) is high due to elevated P application rates related to potato and horticultural productions and their relatively small agricultural areas compared with other provinces. Moreover, their cumulative P balance trends showed no sign of reduction during the last 25 yr. The province of QC also has a high cumulative P balance value (430 kg P ha⁻¹); however, the rate of P build-up in agricultural soils decreased after 1996 as a result of the annual P balance decrease discussed previously. The province of ON had the highest cumulative P balance in 1981 because its agricultural land (approximately 4.4×10^6 ha) covered twice as much area as that of QC. Moreover, the annual P balance values of ON were approximately half those of QC, explaining why its cumulative P balance values expressed on an agricultural area basis have increased more slowly than those of QC. Cumulative P balance values of BC, AB, SK, and MB for year 1981 were lower than those of other provinces. This was partly due to the lack of historical data that artificially imposed P accumulation estimates from 1976 instead of 1966. Moreover, AB, SK, and MB had negative annual P balance values or values at equilibrium in 1981.

Soil Phosphorus Enrichment

In QC, long-term studies on acidic soil experimental plots have shown enrichment coefficient values ranging from 6 to 22 mg P-Mehlich-III per kg of soil for each 100 kg P applied in

Table 2. Temporal trends in cumulative P balance for the provinces of Canada (1981–2006). Values are given on an agricultural land area basis (kg P ha⁻¹) and as the total amount per province (Mg P).

Province	Unit	1981	1986	1991	1996	2001	2006
BC†	kg P ha⁻¹	15	47	77	107	139	167
	Mg P	25,823	81,320	134,570	197,498	266,336	361,854
AB†	kg P ha⁻¹	-3	-5	1	11	30	55
	Mg P	-49,295	-78,604	16,473	198,714	559,615	1,025,161
SK†	kg P ha⁻¹	-2	-5	-8	-10	-4	8
	Mg P	-43,365	-101,587	-161,278	-211,257	-81,839	172,524
MB†	kg P ha⁻¹	0	6	17	30	47	69
	Mg P	3,011	36,945	116,772	197,791	315,305	463,820
ON‡	kg P ha⁻¹	78	104	126	140	156	167
	Mg P	394,172	468,682	555,122	636,641	701,529	733,799
QC‡	kg P ha⁻¹	143	214	287	351	398	430
	Mg P	355,586	492,682	645,215	781,006	871,091	945,724
NB‡	kg P ha⁻¹	251	334	425	501	566	634
	Mg P	45,956	52,925	65,996	80,835	95,535	104,559
NS‡	kg P ha⁻¹	264	354	456	538	613	688
	Mg P	44,609	51,704	64,955	82,332	92,281	98,426
PE‡	kg P ha⁻¹	234	307	392	491	605	717
	Mg P	48,858	59,705	74,202	97,106	121,228	139,619
NL‡	kg P ha⁻¹	511	684	841	969	1,074	1,207
	Mg P	7,066	7,288	12,518	9,340	12,324	15,002

+ Summation of annual P balance values starting in 1976.

‡ Summation of annual P balance values starting in 1966.

excess (Giroux et al., 2002). The variability of the enrichment coefficients increased with high P-enriched soils where P stabilization on soil particles is observed (Giroux and Royer, 2007). Another study (Tabi et al., 1990) and an extensive survey from QC, between 1998 and 2002 (Beaudet et al., 2003), provide accessible and identifiable soil P data. Two other studies published Canadian P budget data that are complementary to soil P data (MacDonald and Bennett, 2009; Potash & Phosphate Institute, 2005). Figure 3 presents the temporal trends in the distribution of annual soil P enrichment estimated using Eq. [2] and [3] for the four groups of soils of Canada (Fig. 1). Not surprisingly, all four soil groups show an increasing trend in their P enrichment medians. However, their annual 10th percentile values are relatively constant. Areas of intensive agricultural practices that present recurring high annual P balance explain the expanding deviation between the median and the 95th percentile values observed in soil groups 1, 3, and 4. Soil group 2 (Fig. 3b) from MB and ON seems less affected by these areas.

Comparisons of the estimated provincial STP medians using the 2006 cumulative P balance values with the median of measured provincial STP values collected by the Potash & Phosphate Institute (2005, now renamed the International Plant Nutrition Institute), show good similarities between the two datasets (Fig. 4). These results indicate that the three soil P enrichment ratios used for the temporal soil P enrichment assessment, as well as the cumulative P balance estimates, are reliable as shown by a relatively low root mean square error value of 22.7 mg P kg⁻¹ soil and a correlation coefficient of 0.74 on the national scale. Four points deviate substantially from the diagonal: estimated values for NB, ON, and BC are lower than their respective measured values, and the estimated value for QC is greater than the measured value. The higher measured median STP values for BC and NB can be explained by a sampling bias

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coming from intensive agricultural regions of these provinces where P-enriched soils are tested for environmental concerns. The highest estimated median STP value for QC (Fig. 4) may be overestimated by the soil P enrichment model, which does not account for decreasing available soil P contents—i.e., precipitation of P in more insoluble forms—where P balance



Fig. 3. Box-and-whisker diagrams of annual estimates of soil test P (STP) and P saturation index (PSI) derived from calculated cumulative P surplus values for the four soil groups: (a) Soil group 1—PSI for Quebec (QC), New Brunswick (NB), Nova Scotia (NS), Prince Edward Island (PE), and Newfoundland and Labrador (NL); (b) Soil group 2—STP for Manitoba (MB) and Ontario (ON); (c) Soil group 3—STP for British Columbia (BC), Alberta (AB), and Saskatchewan (SK); (d) Soil group 4—PSI for BC. Dashed lines represent the mean value; the whiskers represent the 10th and 90th percentiles; × symbols represent the 5th and the 95th percentiles.



Fig. 4. Comparison between estimated provincial soil test P (STP) median values and measured provincial STP median values (Potash & Phosphate Institute, 2005). All values are shown in equivalent Bray-P units using PPI conversion factors. The diagonal represents a 1:1 relation. RMSE = root mean square error.

is at equilibrium (Giroux et al., 2002; Chang et al., 2005). The fact that the International Plant Nutrition Institute values covered most of ON agricultural soils may indicate that the soil P enrichment model underestimated the median STP value for 2006. Other limitations impacting the outputs of the soil-P enrichment model are the uncertainties related to the accuracy of the initial values of soil-P enrichment, of the estimated soil-P enrichment coefficients, and of the downscaling results of the mineral P values associated to the P balance assessment. Also, the plausible transport of organic fertilizer P from a SLC polygon to adjacent SLC polygons is not taken into consideration because of lack of data.

Soil Phosphorus Potentially Desorbed by Water

Table 3 provides the slopes (m) of all 10 derived models for predicting Pw Self Davis. Slopes were statistically tested for significant differences (P < 0.05) within each soil group. Limited data from coarse-textured soils for soil groups 3 and 4 preclude any relationship assessment. When soil textural classes within the same soil group had no significant different slopes, soil data from the respective textural classes were combined, and the STP–Pw and the PSI–Pw relations were recalculated

to obtain a new integrated slope coefficient. Consequently, STP-Pw or PSI-Pw relations were recalculated for soil group 1 (for combined coarse-medium textural classes), soil group 3 (for combined medium-fine textural classes), and soil group 4 (for combined medium-fine textural classes), giving new slope coefficients of 0.24, 0.11, and 0.41, respectively. These three combinations reduced the number of soil P desorption models from 10 to 7. The omission of the models' intercept did not have an important impact on the models' accuracies since five of the seven models did not have an intercept different from the origin. Two model slopes were influenced downward with the omission of a significantly negative intercept (soil group 2, medium class; and soil group 4, medium-fine class). Although STP-Pw or PSI-Pw relations of discrete textural classes within a given soil group were not significantly different, slope coefficients of soil group 1 had an inverse overall tendency from that observed for the other three soil groups (Table 3), where fine textural class soils showed a greater risk of soluble P desorption for a same PSI. These results are in agreement with other studies on soils from QC (Pellerin et al., 2006; Beauchemin et al., 2003). It is assumed that these regional differences are probably due to different parental soil material. This particular difference can impact the soil P sorption-desorption kinetics and may partly explain the inverse relationship. Some other regional differences may, however, be underestimated or even masked because of limited data from some regions, particularly in BC, AB, and SK (Table 3). This scenario may be the case for soil groups 3 and 4, for which only one relationship for all analyzed soil series was obtained. Because most soils in the prairies are in the medium to fine textural class range, these samples outnumber those of coarse-textured soils. The sole relationship found for the soil group 3 could also be related to the high organic matter and calcium content of the Chernozemic soils (Soil Classification Working Group, 1998) which may outweigh the textural influences.

National and provincial estimated P_source values are provided in Table 4. As anticipated, all provinces showed different rates of increase during the 25-yr interval of 1981 to 2006. By 2006, the Atlantic Provinces (NB, NS, PE, and NL) had seen their provincial P_source mean values reach and exceed the threshold value of 4 kg P kg⁻¹ (Pw Self-Davis). The province of NL, with a relatively small agricultural area, showed values

Model	Soil	Textural classification	n	Model slope (m)	95% Confidence interval	Adjusted R ²
Soil group #1 (P/Al	Neutral to acidic soils of QC; NB;	Coarse	26	0.22ª†	[0.179–0.260]	0.82
Mehlich-III)	NS and PE	Medium	24	0.28ª	[0.236-0.331]	0.86
		Fine	18	0.45 ^b	[0.378-0.518]	0.91
Soil group #2 (P-Olsen)	Calcareous soil of MB and ON	Coarse	54	0.18ª	[0.164–0.201]	0.88
		Medium	115	0.12 ^b	[0.104–0.127]	0.79
		Fine	98	0.09 ^c	[0.080-0.097]	0.80
Soil group #3 (P Kelowna)	Calcareous soil of BC, AB and SK	Coarse	1	_	-	
		Medium	18	0.11ª	[0.095–0.119]	0.95
		Fine	9	0.10ª	[0.096-0.110]	0.99
Soil group #4 (P/Al Mehlich-III)	Neutral to acidic soils of BC	Coarse	3	-	-	
		Medium	26	0.43ª	[0.375–0.494]	0.90
		Fine	11	0.32ª	[0.272–0.371]	0.95

Table 3. Statistics of linear relationships between soil test P or phosphorus saturation index and Pw Self-Davis on a soil textural basis for the four groups of agricultural soils.

 \pm Values of model slope followed by the same lowercase letter within each soil group are not significantly different at $p \le 0.05$.

Table 4. Pw Self-Davis concentration (P_source) trends for the provinces of Canada and proportion of farmland considered at high risk of P desorption (>4 mg P kg⁻¹ soil).

Province	P_Source					Proportion of farmland above 4 mg P kg ⁻¹						
	1981	1986	1991	1996	2001	2006	1981	1986	1991	1996	2001	2006
	mg P kg ⁻¹ soil						%%					
BC	1.36	1.63	1.88	2.13	2.40	2.64	0.0	2.0	4.3	5.3	5.7	6.1
AB	1.12	1.14	1.19	1.26	1.40	1.61	0.0	0.0	0.0	0.0	0.1	0.7
SK	1.11	1.13	1.14	1.16	1.20	1.28	0.0	0.0	0.0	0.0	0.0	0.0
MB	0.61	0.65	0.72	0.80	0.92	1.07	0.0	0.0	0.0	0.0	0.0	0.0
ON	1.18	1.37	1.52	1.63	1.75	1.84	0.0	0.0	0.0	0.0	0.0	0.0
QC	1.87	2.31	2.76	3.15	3.43	3.61	0.0	1.6	15.9	23.1	30.7	34.5
NB	2.14	2.56	3.03	3.41	3.74	4.09	2.2	9.3	12.2	20.3	30.3	34.4
NS	2.09	2.51	2.98	3.37	3.71	4.06	1.1	3.2	3.4	12.1	22.2	42.2
PE	1.94	2.26	2.62	3.04	3.53	4.02	0.0	0.0	0.0	21.5	29.9	43.6
NL	4.12	5.22	6.15	6.87	7.44	8.20	28.3	35.0	41.3	67.8	74.4	74.4
Canada	1.13	1.19	1.27	1.34	1.46	1.60	0.0	0.2	0.9	1.4	1.9	2.5

above the 4 mg P kg⁻¹ environmental threshold for the duration of this study, and almost three-quarters of its farmland had a Pw Self-Davis value above 4 mg P kg⁻¹ in 2001 and 2006. The provincial P_source mean value of QC followed closely with a value of 3.61 mg P kg⁻¹. The Atlantic Provinces and QC showed higher Pw Self-Davis values and had a higher percentage of farmland above the 4 mg P kg⁻¹ environmental threshold due to their intensive agriculture. These relatively high P_source values contrasted with the lower values for ON, MB, SK, AB, and BC. The Canadian Prairies are characterized by a relatively recent agriculture, and their agricultural land covers a large area, which partly explains its lower values. The contrast is more evident when restricted to the proportion of farmland exceeding the P_source value of 4 mg P kg⁻¹ (Table 4).

The 2006 P_source values across Canadian agricultural land (>5% agriculture) were mapped into five incremental classes of 1 mg P kg⁻¹ at the SLC scale (Fig. 5). Areas having P_source concentrations of more than 4 mg P kg⁻¹ are precisely located around Abbotsford (BC) and Lethbridge (AB), and in a more evenly distributed pattern in QC, NB, NS, PE, and NL.

The same limitations noted for the soil P enrichment assessment prevailed for the soil P desorbed by water estimates. The grouping of soils covered a series of chemical (e.g., Fe and Al oxides concentrations, calcium carbonates concentrations, and pH) and physical (e.g., soil textural classes, soil structure, and parent material) properties of marginal soils that directly impact soil P sorption capacity and, subsequently, also the Pw concentrations. Moreover, the P_source values were assessed on an annual basis and thus did not account for the timing of P inputs (manure spreading and mineral P fertilization) with water movements (storm events, surface runoff, and water leaching). Further development of the P_source assessment might explore the feasibility of using a seasonal, or even a daily, time scale to match it more closely with the transport component of the IROWC-P, which integrates daily water balance values. According to the limitations at this stage of development, the Canadian map of the P_source values (Fig. 5) should be interpreted on a comparative basis other than on an absolute Pw concentration basis.



Fig. 5. P_source (Pw Self-Davis, mg P kg⁻¹ soil) class distribution over Canadian farmland estimated with the 2006 Census of Agriculture data.

Conclusions

The scientifically based relationships between annual P balance values, STP, DSPS, PSI, and Pw were used to develop a methodology to assess a P source value. The P source values were assessed for all Canadian agricultural soils from 1981 through 2006 on a 5-yr CoA basis. Results show a strong positive trend in soil P enrichment during the 25-yr interval but also a strong positive trend in the percentage of farmland classified as above the environmental threshold of 4 mg P kg⁻¹ (Pw). Even though only 2.5% of the Canadian farmland was above the environmental threshold value, it is clear that some regions of the country were showing 35 to 75% of their cropland above the 4 mg P kg⁻¹ value. These regions were located essentially in Eastern Canada (QC and the Atlantic provinces) and to a lesser extent in BC. The intensity of the agriculture combined with the relatively less important agricultural area of these regions, compared to the Prairies provinces and ON, explains these high percentage values. Although the methodology presented in this study identifies regions of high risk of P desorption at a broad spatial scale, it will be necessary to carry out more detailed assessment studies in these identified regions to validate the risk and to determine which adjustments are needed to lessen the risk of P desorption. Moreover, objective information on the current environmental performance of the agricultural sector is crucial to determine whether this performance is satisfactory and how it is likely to behave in response to the decision-making processes. Developing a national STP database would greatly improve the precision of the P_source assessment because STP integrates soil and fertilizer managements. Many assumptions in this study, such as native soil STP and the homogeneous allocation of manure P and fertilizer P on agricultural soils in a given SLC polygon, would be integrated within the measured STP values, so no estimates—and their inherent errors—of these important attributes would be necessary. Moreover, further soil information obtained from reports and analysis in understudied regions could reveal some other soil characteristics than soil textural class that may well enhance the representativeness of the soil grouping with the genuine conditions prevailing on the landscape. Furthermore, the variability of the soil P enrichment ratio will have to be taken into account due to high soil P levels in some areas. We also recommend conducting more national surveys on soil and management practices to better integrate their trends of adoption by the farmers.

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