Evaluating management practices to limit phosphorus losses from agricultural fields in the Castor watershed using the WEND model

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Abstract

The objective of this study was to apply the WEND model, a nutrient mass balance model, to the Castor watershed in southern Quebec to evaluate phosphorus movement, storage and export over time. The WEND model was customized to run on a field-scale and then individually applied to 266 fields on the watershed for a 30-year simulation period. Field-specific information for the period of 1997-1999, was used as basic inputs to the model. Climatic information was obtained from local sources. The additional information required to run the model was derived from the literature. Model outputs were analysed at three different levels: (i) the overall watershed impacts, (ii) by cropping system, and (iii) for field management practices presenting a high risk of P losses. Specific outputs examined were: soil test Mehlich-III P, soil P saturation with aluminium, RUSLE soil loss potential and TP export.

The model was used to examine the impacts of crop rotations, fertilizer application and tillage management on TP export. For the Castor watershed, the soil test P increased at a mean rate of 3.71 kg Mehlich-III P ha^{-1} yr^{-1}, equivalent to a mean input of about 32 kg P_2O_5 ha^{-1} yr^{-1} in excess of plant requirements, assuming current field management practices remain constant.

If TP export is considered the most important parameter in terms of P contamination, crop rotations are a good alternative to continuous corn monocropping under which losses could reach as high as 3.36 kg TP ha^{-1} yr^{-1}. Crop rotations were shown to be an important management practice that should be more carefully examined when establishing field management practices. Just one year of grassland within a rotation can greatly improve the overall environmental health of a watershed. The management of P inputs is also an important target for improvement, as fertiliser inputs often surpassed plant requirements by two or three-fold.

Résumé

L'objectif de ce projet était d'appliquer le modèle WEND, un modèle de bilan de masse, au bassin versant du Ruisseau Castor dans le sud du Québec afin d'évaluer le transport, le stockage et l'exportation de phosphore dans le temps. Le modèle WEND a été adapté afin d'utiliser la parcelle comme unité de base et ensuite il a été appliqué individuellement aux 266 parcelles du bassin pour une simulation d'une durée de 30 ans. Les données relatives aux champs, pour la période de 1997-1999, ont été utilisées comme données de base au modèle. Des données climatologiques locales ont été acquises. Les informations additionnelles requises afin d'opérer le modèle ont été dérivées de la littérature. Les extrants du modèle ont été analysés (i) à l'échelle du bassin versant, (ii) en fonction de certains types de gestion des cultures, et (iii) pour les champs dont les pratiques représentent un risque important de pertes de phosphore. Les extrants analysés sont: les teneur en P du sol (Mehlich-III P), la saturation du sol (P/Al), le potentiel de perte de sol (RUSLE) et les pertes de phosphore total (PT).

Le modèle a été utilisé afin d'évaluer l'impact des rotations de culture, de l'application de fertilisants et des méthodes de travail du sol sur les pertes de PT des parcelles. Pour le bassin versant du ruisseau Castor avec les pratiques actuelles demeurant constantes, la teneur en P du sol augmente à un taux moyen de 3.71 kg Mehlich-III P ha^{-1} an^{-1}, soit l'équivalent d'un ajout moyen d'environ 32 kg P_2O_5 ha^{-1} an^{-1} supérieur aux besoins des plantes.

Si les pertes de PT des parcelles sont considérées comme le paramètre le plus important en terme de contamination par le phosphore, les rotations de cultures sont une bonne alternative à la monoculture de maïs, sous laquelle les pertes atteignent 2.52 kg TP ha^{-1} an^{-1}. Les rotations de cultures démontrent qu'elles sont une pratique de gestion efficace et devraient être utilisées plus fréquemment. Une seule année de prairie dans une rotation peut grandement améliorer la santé environnementale d'un bassin versant. La gestion des intrants en phosphore est une cible importante dans ce processus d'amélioration, étant donné que ceux-ci dépassent souvent les besoins des plantes par deux ou trois fois.

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Table of contents

List of tables

List of figures

List of abbreviations

AEFP: Agro-environmental Fertilization Plan

AGNPS: Agricultural Non-Point Source Pollution Model (Yoon et al., 1993)

AnnAGNPS: Annualized Agricultural Non-Point Source Pollution Model (Bingner et al., 1998)

ANSWERS-2000: Areal Non-point Source Watershed Environment Response Simulation (Bouraoui & Dillaha, 1996)

a.u.: animal unit

BOD_5: biochemical oxygen demand during decomposition occurring over a 5-day period

CREAMS: Chemicals Runoff and Erosion from Agricultural Management Systems (Kinsel, 1985)

ECAF: European Conservation Agriculture Federation

GIS: geographic information system

IQBP: bacteriological and physico-chemical index (IQBP)

MENV: Ministère de l'Environnement du Québec

P-Index: phosphorus index (Lemunyon & Gilbert, 1993)

REA: Règlement sur les exploitations agricoles

RUSLE: Revised Universal Soil Loss Equation

RUSLEFAC: Revised Universal Soil Loss Equation for Application in Canada

TP: Total phosphorus

WEND: Watershed Ecosystem Nutrients Dynamics (Cassell & Kort, 1998).

1 Introduction

1.1 Problem definition

Over the past 20 years, surface water quality in many areas of Canada and Quebec has been improving (Bolinder et al., 2000) because many industrial and urban waste water treatment systems have been upgraded (Painchaud, 1997; Simard, 2000). Unfortunately, however, in many regions of Quebec surface water quality remains unacceptable, due in most part to non-point source P pollution (Bolinder et al., 2000). Such P pollution occurs in surface runoff and subsurface flows from urban, agricultural and forested lands. To meet water quality guidelines it will, in many cases, be necessary to reduce the magnitude of non-point source phosphorus pollution. It is widely believed that our ability to reduce non-point source pollution will be contingent upon how well we manage our agricultural and urban lands in the future.

Non-point source phosphorus pollution is of significant concern as, among other things, it contributes suspended sediment and nutrients to receiving water bodies. This accelerates natural eutrophication and deoxygenation processes that control wildlife populations and the overall usage of our water resources (Harker et al., 1998; Bolinder et al., 2000). Eutrophication is a normal aging process of water bodies that can transform a lake into a wetland over the period of centuries, but with the input of additional sediment and phosphorus this transformation may occur in a few decades (Sharpley, 1995; Sharpley et al., 1999; Aertebjerg et al., 2001). A level of 0.02 mg L^{-1} phosphorus in water is sufficient to increase the stimulation of eutrophication. This level is exceeded in the waters of most rivers in southern Quebec (Giroux & Tran, 1996).

Agricultural non-point source pollution is a major contributor to phosphorus loads in the surface waters of Quebec (Painchaud, 1997). Pollutants arising from the application, storage and treatment of agricultural inputs, including organic and inorganic fertilizers, have been detected in the surface and ground-waters of Quebec (Ministère de l'Environnement (MENV), 1993; Painchaud, 1997), especially in areas of high livestock densities (Painchaud, 1997; Harker et al., 1998; Simard, 2000). The level of agricultural non-point source pollution is not only a function of agricultural input management, but also depends on soil type, topography, hydrologic events, the phosphorus contents of the surface soils and water soluble P levels in the subsoil (MENV, 1993; Simard et al., 1995).

In the Missisquoi bay region in southern Quebec, where the Castor watershed is situated, agricultural soils are typically high in phosphorus. Consequently, erosion, enhanced by tillage and cultivation practices as well as high animal densities, contributes to high phosphorus concentrations in the local surface waters (MENV, 1999a). These conditions cause eutrophication of Missisquoi bay. The environmental conditions persisting in Missisquoi bay raise concerns on both sides of the border. Since 1991 a committee of scientists, the Lake Champlain Conference, have been meeting to consider the environmental conditions of Lake Champlain. The government of Quebec and those of the states of Vermont and New York agreed to work on an action plan to develop knowledge on the lake's watersheds, involve citizens, and determine methods to reduce point and non-point source phosphorus inputs. In Quebec, such a plan is also in place, and seeks to identify the main actions necessary in assessing the problem. Many field-level actions have been undertaken by local committees and joint efforts of the Quebec ministries of the Environment, of Municipal Affairs, of Health and Social services, of Natural Resources, Wildlife and Parks, and of Agriculture.

Rates of manure applications on agricultural lands have been based either on the disposal of all manure production on available field areas, or its application according to the crop nitrogen needs (Caumartin & Vincent, 1994; Simard et al., 1995; Bolinder et al., 2000; Cassell et al., 2000; Simard et al., 2001). Both practices have increased soil phosphorus content and phosphorus saturation levels in many Canadian soils (Simard et al., 1995), and appear to influence the quality of surface water in Eastern Canada (Simard et al., 2001). Miller (2001) indicated that farm profitability could be reduced by the over-application of phosphorus when soil test P is already high.

Water quality models can be used to identify areas of varying environmental risk in a watershed and to simulate the impacts of changes in management practices on the export of P from agricultural lands. Many such models have been developed and are discussed in section 2.4. The WEND model (Cassell & Kort, 1998), a phosphorus mass balance model, strategically evaluates the impacts of management practices on P loads in the watershed over the long-term, within a dynamic simulation-modeling framework. In the present study, the WEND model simulated phosphorus movement and storage within the context of the soils and agricultural management activities that are peculiar to the Castor watershed. Unique algorithms were developed to define the conditions that exist in the Castor watershed. It was also applied at the farm level, not as a watershed model, so that it can provide an analysis of field-level practices and an evaluation of long term effects of P loads.

1.2 Objectives

The notions of dynamic simulation modelling and dynamic mass balance analysis are used in this study to evaluate the components of soil P concentration, movement and transport over time, and to assess P imports to the watershed and exports to surface and ground waters. Since the WEND model was developed for a watershed-scale study of phosphorus pollution, its application at the field scale required some adjustments. The Castor watershed was chosen as a case study because of the quantity of data available on the watershed and the research already undertaken to evaluate agronomic impacts of phosphorus in the area. The great concern regarding P pollution in Missisquoi bay, the Quebec portion of Lake Champlain, raises the importance of understanding what goes on in the entire watershed. Consequently, the objectives of the present research were to:

(i) Customize the WEND model, by stipulating the equations characterizing the geophysical and geochemical characteristics of the fields located in the Castor watershed;

(ii) Identify, with soil P, TP export, soil saturation level and soil loss potential given by the model, characteristics of existing management practices and long-term management solutions that could be effective in reducing phosphorus export for the Castor watershed and for the different cropping systems;

(iii) Use the MENV codes and regulations on phosphorus application as fertilizer recommendations, in the WEND model, to asses the impacts of P loss in high risk zones.

1.3 Scope

The model introduced in the present study is developmental and intended to provide understanding of the dynamics of phosphorus in the Castor watershed and cannot be applied on another watershed without being modified. This model should be used with caution as running it with input values outside realistic boundaries can lead to incorrect results. Water samples taken throughout the Castor watershed from 1997 to 1999 were used to calibrate and validate the model.

2 Literature review

The MENV (2002a) has established a water quality standard of 0.03 mg total P (TP) L^{-1} for Quebec watercourses. Most rivers in southern Quebec exceed this P concentration guideline (Giroux & Tran, 1996). Non-point source pollution, mainly from agricultural sources, has been identified as being responsible for phosphorus pollution in these watercourses (MENV, 1993; Jamieson et al., 2001). A considerable quantity of P entering surface waters from agricultural areas is a result of over-application of fertiliser. While the MENV had set a June 2003 deadline for the deposition of farm-level P mass balance reports, by June 2004, 40% of the P mass balances were still unaccounted for.

Every farm producing over 1600 kg P yr^{-1} must produce an "Agro-environmental Fertilization Plan (AEFP)" for every field to which fertilizers are applied. The timeframe for fertilizer application differs according to the type of crop cultivated. Developed with an agronomist, the AEFP is prepared for a three- to five-year period, during which they are reevaluated. Target P concentrations for fields can be found in the "Règlement sur les exploitations agricoles (REA)" produced by the MENV (2003). The use of such plans to improve nutrient management on fields is encouraged by the government. The aim is to generate a sustainable production of quality crops and to protect natural resources. The guidelines levels for P applications are presented in Table 2-1 and 2-2.

Table 2-1: Recommended phosphorus application rates for corn (*Zea mays* L.) (MENV, 2003)

Soil phosphorus content (kg P ha^{-1})	Soil phosphorus saturation P/Al %	Corn yield (Mg ha^{-1} 15% humidity)		
		< 7	7 to 9	9 +
0 - 30		140	150	160
31 - 60		130	140	150
61 - 90				
121 - 150		120	130	140
151 - 250		110	120	130
	< 5	100	110	120
	5 to 10	90	100	110
251 - 500	>10	75	85	95
	≤10	50	60	70
501 +	>10	65	75	85
		50	60	70
		40	50	60

Table 2-2: Recommended phosphorus application rates for cereals, soybeans [*Glycine max* (L.) Merr.] , forages (MENV, 2003)

Soil phosphorus content (kg P ha^{-1})	Soil phosphorus saturation P/Al %	Crop yield (Mg ha^{-1} 15% humidity)		
		< 2.5[1] < 5[2]	2.5 to 3.5[1] 5 to 7[2]	> 3.5[1] > 7[2]
0 - 30		120	130	140
31 - 60		110	120	130
61 - 90				
121 - 150		100	110	120
151 - 250		90	100	110
	< 5	80	90	100
	5 to 10	70	80	90
251 - 500	>10	55	65	75
	≤10	30	40	50
501 +	>10	45	55	65
		30	40	50
		20	30	40

[1] This line corresponds to cereals and soy beans yields
[2] This line corresponds to prairies and pasture yields

2.1 Phosphorus behaviour in soil

2.1.1 Phosphorus in soil solution

Both organic and inorganic fractions of the soil P pool can occur under a number of different forms (Sharpley & Beegle, 1999; Sharpley et al., 1999). The highest P concentrations are found in the thin surface layer of the soil (Simard et al., 2001; Culleton et al., 2002), where it is strongly fixed, preventing it from moving through the different layers of the subsoil. The first few millimetres of the soil are termed the mixing zone, a site where many biological processes take place (Baker et al., 2001). Organic P is present in organic matter, plant residues and microbes (Schlesinger, 1997). Phosphorus can be transformed from available to immobile forms depending on its chemical composition. In contrast, inorganic P is associated with other minerals such as Al, Fe and Ca (Schlesinger, 1997) and can easily be taken up by plants or stably fixed, according to soil type, pH, soil parent material, texture and field management practices (Sharpley & Beegle, 1999; Sharpley et al., 1999). Inorganic P is also found in commercial fertilizers, and once applied to the soil is almost totally fixed within two to four weeks (Foth & Ellis, 1988; Sharpley & Beegle, 1999). Application methods and soil characteristics, such as soil minerals, pH, moisture and temperature control the behaviour of phosphorus fertilizers and the quantity of available P (Fenton et al., 2001).

Phosphorus can move through the soil in two main forms: sediment-bound (particulate) or dissolved (soluble). Soluble P is mainly composed of phosphate ions (HPO_4^{2-} and $H_2PO_4^-$) and of some organic compounds (Brady & Weil, 2002). Particulate P can represent 60% to 90% of total phosphorus in surface runoff from a precipitation event (Sharpley et al., 1992; Sharpley & Beegle, 1999). It is not available for plant uptake, but when suspended in water remains a long term source of P for algae (Sharpley, 1993; Ekholm, 1994; Sharpley et al., 1999). The ratio between particulate and soluble phosphorus in soil is influenced by soil type, composition and leaching characteristics (Sharpley & Syers, 1979; Bengston et al., 1992; Sharpley et al., 1999). The more P is fixed, the less will leached.

2.1.2 Phosphorus cycle

Phosphorus exits the soil system in one of three ways: plant uptake, with suspended sediment removed by erosion (particulate P) and as dissolved P in surface runoff (Brady & Weil, 2002; Figure 2-1). Plant uptake of P is, in some cases like corn (*Zea mays* L.), enhanced by mycorrhizal fungi and by the slow movement of ions to the root surface (Brady & Weil, 2002). The monovalent anion form ($H_2PO_4^-$) seems to be the preferred form for root absorption, in a proportion of 10 to 1 over the divalent anion (HPO_4^{-2}) form. However, since interconversion occurs rapidly, this plant preference has little impact (Foth & Ellis, 1988). On the other hand, some variety of fungi (hypnea) can absorb P ions and even strongly bound forms of P and transfer it to the roots (Brady & Weil, 2002). All forms of phosphorus exist in equilibrium, with different soil biological processes like mineralization and immobilization maintaining it (Sharpley & Beegle, 1999). Mineralization happens when bio-available organic P is converted to inorganic soluble P and immobilization occurs when organic P becomes more strongly fixed (Sharpley & Beegle, 1999).

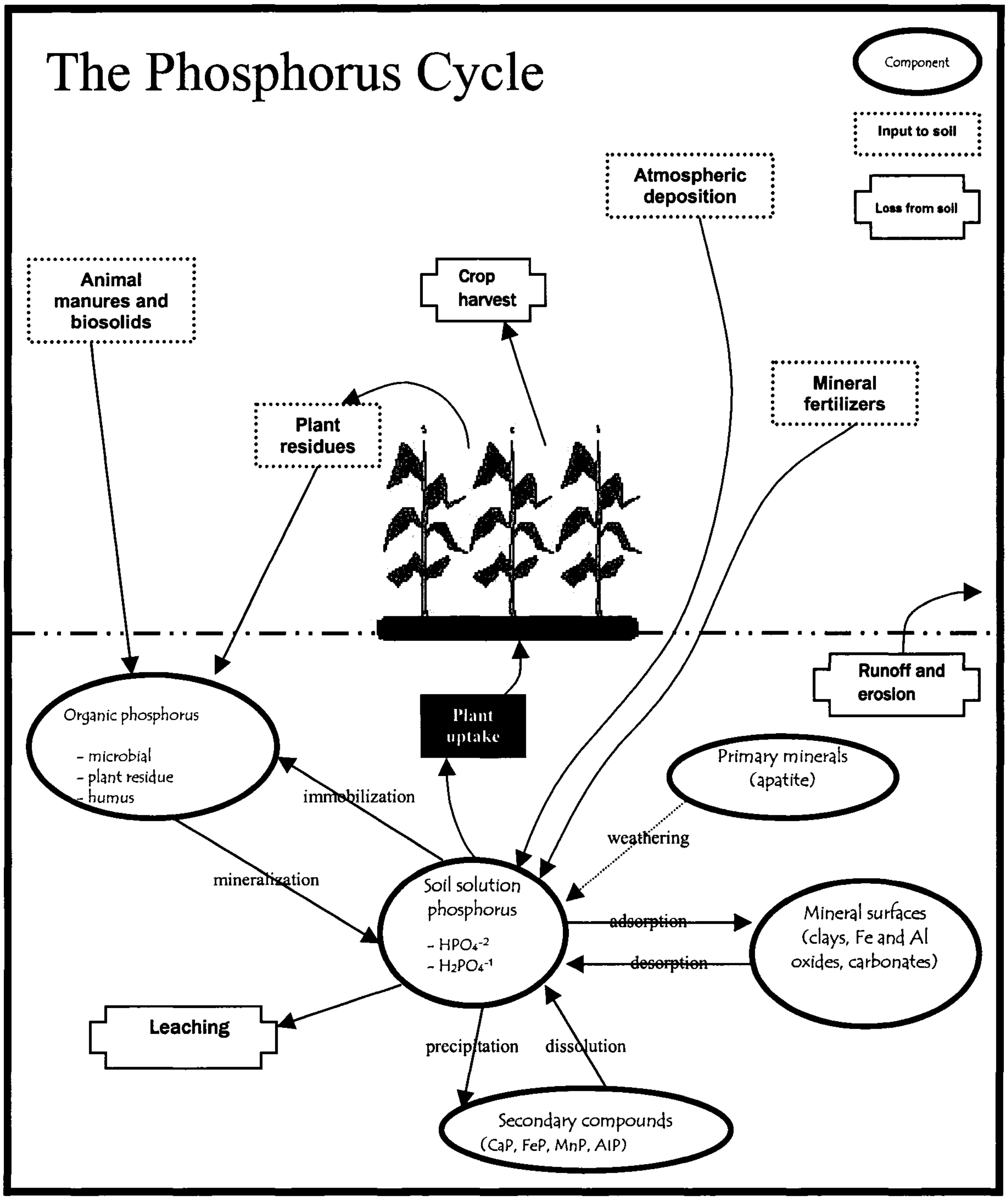

Figure 2-1: Phosphorus cycle (modified from Schlesinger, 1997)

Inorganic phosphorus also undergoes transformations within the soil P pool, including fixation and sorption (Sharpley & Beegle, 1999). The former process transforms phosphorus into a rather unavailable form which prevents plants from using it; however, it can be seen as a benefit, as it prevents some P from entering surface water channels (Brady & Weil, 2002).

The phosphorus cycle is also affected by human activities. According to Zheng et al. (2001), cropping systems and nutrient sources influence the behaviour of the different forms of P in the soil P pool in fine-textured Gleysolic soil. They found that the rate of P surplus increase is 3-fold greater in fields receiving liquid dairy manure applications than those receiving mineral fertiliser applications. They also underlined that climatic conditions have an impact on annual labile C inputs, which can subsequently result in large variations in organic P fractions. Phosphorus movement through the soil system is complex and not completely understood.

2.1.3 Measurement of soil P saturation

Phosphorus saturation level in the presence of aluminium is used as an environmental indicator to evaluate environmental risk potential for phosphorus contamination (MENV, 2003). This method was developed in Holland by Breeuwsma and Silva (1992) and is getting growing recognition internationally. This value is used to evaluate the phosphorus fixation capacity of the soil. In Quebec, the soil phosphorus and aluminium contents are measured using the Mehlich-III procedure (Mehlich, 1984). The saturation level is achieved by dividing the phosphorus content by that of aluminium. Giroux and Tran (1996) showed a significant linear mathematical relationship between Mehlich-III aluminium level and the soil phosphorus fixation capacity of Quebec soils.

Recently, soil P saturation thresholds were identified by Pellerin (2003) for both environmental and agronomic assessments. She found that the relation between the phosphorus available in the water solution and soil saturation is greatly improved when taking into consideration the proportion of clay in the soil. A threshold of 9.7 mg P L^{-1} was chosen to set soil saturation threshold values for soils with different clay contents (Table 2-3).

Table 2-3: Environmental critical threshold values for potato (*Solanum tuberosum* L.) and grain corn crops (Khiari et al. 2000; Pellerin, 2003)

Soil clay	Crop	Soil P saturation
%		%
	potato	
≤ 20		15
	grain corn	
≤ 30		13.1
> 30		7.6

From an agronomic standpoint, the assessment of the plant response to the fertiliser application and soil test are the main elements of a recommendation. As shown in Table 2-4, there is a minimum and a maximum agronomic threshold.

Table 2-4: Agronomic critical threshold values for potato and grain corn crops (Khiari et al. 2000; Pellerin, 2003)

Soil clay	Crop	Minimum Soil P saturation	Maximum Soil P saturation
%		%	%
	potato		
≤ 20		8.2	35
	grain corn		
≤ 30		4	21.7
> 30		2.5	21.4

According to the values of Tables 2-3 and 2-4, it is possible to assert that plant needs cans be met without applying excessive amounts of P to fields since environmental critical threshold values are greater than the minimum agronomic critical threshold values.

2.1.4 The P-Index

The P-index was developed at the U.S. Department of Agriculture by Lemunyon and Gilbert (1993) to identify critical source areas of phosphorus pollution, i.e. where the soil P concentrations are high and the erosion potential is significant (Sharpley & Beegle, 1999). The evaluation of source factors (soil P concentration, rate, timing and method of P application and form of P applied) and transport factors (erosion and runoff potential) form the basis of the P-index (Sharpley, 1995). These factors are evaluated according to their role in phosphorus pollution in diverse areas (Sharpley, 1995; Sharpley et al., 1999), giving the index flexibility and making it applicable under a variety of conditions.

Source areas of significant phosphorus pollution vary considerably over time. They are influenced by precipitation intensity and duration, soil moisture and temperature, soil type and characteristics, slope and ground water level (Sharpley & Beegle, 1999). Accelerated eutrophication frequently occurs far from the original source of phosphorus (Sharpley & Beegle, 1999) and a long period of time can elapse before any sign of environmental deterioration appears. Therefore source factors of P and transport factors of P need to be identified to assess non-point source phosphorus pollution and to apply management solutions at both levels (Sharpley et al., 1999). Research done in the United States has recognized animal feed on farms as the main source of excess phosphorus (Sharpley et al., 1999). Other research in Eastern Canada has identified the long term application of manure as a key factor in surface water quality (Simard et al., 2001). Many solutions have been identified to reduce soil loss (transport factors) from agricultural land; but while they reduce the amount of particulate P, they have a limited effect on reducing soluble P (Sharpley & Beegle, 1999). The reduction of phosphorus source factors, compared to transport factors, will thus have a greater effect on the overall reduction of P losses (Heatwole et al., 1987; Sharpley, 1995; Prato & Wu, 1991; Enright & Madramootoo, 2001). Several high livestock density watersheds in the St.Lawrence lowlands were recognized as critical source areas, showing high P saturation levels in soil and significant amounts of suspended sediment in surface waters. Reducing soil P content in these watersheds will take quite a long time (Simard et al., 1995; Simard, 2000).

The P-index was adapted to Quebec conditions (Giroux et al., 1996; Bolinder et al., 1998; Beauchemin & Simard, 1999), but was never applied to fields within the province until 2001, when modifications in environmental legislation occurred (Enright & Madramootoo, 2001). According to the modified P-index, corn presents a greater threat of soil P transfer than forages; in both cases the spring application of fertilizer resulted in lower P values (Simard et al., 2001). Agriculture and Agri-Food Canada, using the modified P-index (Giroux et al., 1996; Beauchemin & Simard, 1999) sought to classify sites according to their risk of phosphorus contamination (Bolinder et al., 1998). Seven parameters were established to classify sites according to their P contamination risk. Transport factors included: soil loss and runoff potential. Soil P saturation level and relative availability of phosphorus were formulated as primary factors of site classification. The annual phosphorus balance is primarily influenced by three parameters: residues, manure and commercial fertilizer (Bolinder et al., 1998). The use of the P-index resulted in the conclusion that for the southern region of the province the risk of P contamination was medium to high. Because of the intensive agriculture occurring in the region, these conclusions were not unexpected. The P-index study in Quebec revealed many regions with high livestock density, where manure applications had resulted in higher phosphorus levels in the soil than the crop could remove. On the other hand, high soil phosphorus concentrations are also found where intensive agriculture is practiced. Areas where corn and soybeans [*Glycine max* (L.) Merr.] are grown for commercial purposes are also areas where large quantities of fertilizers are applied. A phosphorus mass balance must be promoted, such that when soil levels of P are already high, the addition of more phosphorus should be limited: a balance between phosphorus input and output should be possible (Bolinder et al., 2000). The nutrient demands of crops should be the cornerstone of manure management strategies (Granstedt, 2000). The soil P concentration is the main element jeopardizing surface water quality. The management of agricultural inputs directly influences P contamination risk (Bolinder et al., 2000).

In other research conducted to evaluate the effectiveness of the P-index in identifying phosphorus problem areas in small agricultural watersheds in Quebec, Enright and Madramootoo (2001) conducted water sampling on sub-basins of the St. Esprit watershed. They found that Total P concentrations in water were significantly correlated with suspended sediments when sediment concentrations were less than 0.1 g L^{-1}. They also found that measured TP concentrations increased as the percentage of the sub-basin used for agriculture increased. At the sub-basins level, it was found that the percentage of the soils classified as clay increased as the percentage of the area used for agriculture increased. So, by extension, TP concentrations measured in water were higher on those sub-basins where clay soils were dominant.

2.2 Phosphorus behaviour in water

2.2.1 Forms of agricultural phosphorus in surface runoff and subsurface flow

Both particulate and dissolved forms of phosphorus are found in surface runoff. The quantity of particulate P present in surface runoff is greatly influenced by the magnitude of soil loss from agricultural fields; which mainly affects the finer and less dense particles (Baker et al., 2001). Particulate P losses can be increased by some tillage practices (Brady & Weil, 2002). When introduced to surface waters, particulate matter can act as a reservoir for phosphorus, gradually allowing bound P to become available for aquatic plants (ECAF, 2000). Under ideal conditions, to preserve the equilibrium, phosphorus is permanently bound to inorganic sediments at the same rate as P is entering surface waters (Downing et al., 2000). Dissolved phosphorus can be transported to surface water with runoff in organic and inorganic forms, arising predominantly from unincorporated fertilizer and manure applied to agricultural fields (Brady & Weil, 2002). Such forms of phosphorus are immediately available when they interact with water. In subsurface flow the proportions of different P forms is different; there is a much smaller amount of particulate P than soluble P, because there are fewer clay particles in the solution capable of binding phosphorus.

2.2.2 Transport of agricultural phosphorus

There are three pathways by which phosphorus travels to surrounding watercourses. It can be dissolved (soluble P) in surface runoff, sediment bound (particulate P) in surface runoff or dissolved in leaching water (Baker et al., 2001). Precipitation patterns, soil characteristics and field management are the primary parameters influencing the transport of phosphorus (Baker et al., 2001). In surface runoff, the total phosphorus concentration is more influenced by the soil P concentration than by the total soil loss (Jamieson et al., 2001). The association between soil P concentration and surface runoff P concentration was also corroborated by Sharpley (1999). Runoff also transports particulate P; because of the soil particles smaller size near the surface, particulate P has a higher concentration in the few first soil centimetres (Culleton et al., 2002). When precipitation and runoff occurs, dissolved P is removed and particulate P is eroded from the mixing zone. Corroborating previous studies, the P concentration in the mixing zone directly influenced the dissolved P in runoff water and particulate P bound to sediment (Baker et al., 2001). Because soils tests are designed to evaluate mostly plant available P, or agronomic P, an incomplete portrait of phosphorus in surface runoff is made (Sharpley et al., 1999). The quantity of phosphorus in surface runoff is also influenced by management practices (Sharpley et al., 1996).

The concentrations of particulate P and of soluble P in leaching water are directly related to the level of phosphorus present in the surface soil. The concentration of P dissolved in the mixing zone decreases following precipitation (Baker et al., 2001). When the soil P concentration is low, P in the water leaching through the soil gets adsorbed to soil particles lowering the P concentration in surface runoff. On the other hand, when soils are saturated, dissolved P in leaching water is more significant (Baker et al., 2001). Researchers (Jamieson et al. 2001; Enright & Madramootoo, 2004) measured phosphorus levels in tile drains of two farms of the Rivière-aux-Brochets watershed in southern Quebec. The total phosphorus concentrations in subsurface runoff were higher than the MENV guidelines for water (0.03 mg of TP/L) (MENV, 2002a). The P load in subsurface runoff was significantly influenced by the concentration of dissolved P. Similar results were also observed in other experiments (Enright et al., 1998). Subsurface drainage accounted for 40% of the total P loads loss from the monitored fields (Enright & Madramootoo, 2004). This reinforced the idea that subsurface runoff may contribute significantly to the total annual phosphorus loads loss to watercourses from agricultural fields. Usually, the P concentration in deep percolating water is low because phosphorus gets fixed in the subsoil layers lowering the soil phosphorus saturation level. This level controls the ratio between the phosphorus leaching and the phosphorus being fixed (Wossink et al., 1999). The fixation of phosphorus present in percolating water is closely related to the drainage capacities of the subsoil. The faster the drainage process, the lower the fixation of phosphorus to the soil (Sharpley & Syers, 1979; Bengston et al., 1992; Sharpley et al., 1999). However, according to Enright and Madramootoo (2004), other elements should be taken into account when assessing soil P pollution, soil test P and soil saturation alone being inadequate indicators. Surface and subsurface ratio of drainage water and soil preferential pathways also significantly influence the P losses from a field.

2.3 State of global and regional water quality with regard to phosphorus

The MENV (2002a) water guidelines are surpassed in many Quebec rivers, especially in terms of standards for phosphorus. Different links between water quality and soil test P have been made in watersheds with intensive agriculture activities (Patoine & Simoneau, 2002). The median TP concentration of such rivers varies between 0.11 and 3.74 mg/L (Patoine & Simoneau, 2002). The water quality is better for rivers on the north shore of the St-Lawrence River than on the south shore, where most agricultural activities are concentrated. A highly significant direct relationship was noted between P in agricultural fields alone or those situated within forested land and the median TP concentration of the rivers (Patoine & Simoneau, 2002). Patoine and Simoneau (2002) also calculated a mean threshold value of 30 kg Mehlich-III P ha^{-1} for agricultural and forested soils, for the phosphorous standard of 0.03 mg L^{-1} to be respected in associated waterways. This value clearly exceeds phosphorous levels in most of Quebec's agricultural soils and particularly those in the Montérégie region.

The MENV has 37 monitoring stations in the Montérégie region (MENV, 2002b). Many water quality parameters are measured and a bacteriological and physico-chemical index (IQBP) used to compare the water quality of the different watercourses. Thirty percent of them were good to satisfactory in quality; 16% were poor, and 54% bad to extremely bad. The upper reaches of rivers often show the best result in terms of the IQBP. The worst results are found in rivers where non-point source pollution from agricultural land is predominant. The Montérégie region is greatly influenced by agriculture, often over 60% (areal basis) of townships' are under cultivation. Grain corn is the main crop in this region. This non-point source pollution is influenced by the duration and intensity of precipitation events, during which most surface and subsurface runoff as well as erosion occurs. When looking at the median of different parameters of the IQBP, some trends emerge. Concerning phosphorus concentration 46% of the watercourses of the region show good to satisfactory quality, compared to 62% for nitrates, 86% for suspended sediment and 95% for fecal coliforms. The 90 percentile of phosphorus parameters illustrates that only 11% of stations show good water quality, while 84% of them have, at some time, poor, bad or extremely bad water quality. Even if agricultural activities are not responsible for the overall state of water quality with respect to phosphorus, the geographic location of stations suggests that they play an important role in it. The Castor watershed, situated where the agricultural activities in the Montérégie region are most concentrated, is therefore a good site for a case study to assess the effects of farm management decisions on reducing P losses from fields.

2.4 Water quality modelling

When operating a model, the user must always keep in mind the specific water resource scenarios intended by the developer (Parsons et al., 2001). With regard to the input data sets, it is also important to consider missing information; the particular sensitivity of the model to certain inputs can lead to an impaired diagnosis or to irrelevant work. On the other hand, the analysis of the outputs can also be a problem, especially when the model is applied to new situations. Achieving research goals by identifying particular responses can sometimes be complicated by the large quantity of output parameters produced by some models. Moreover, the size of areas intended to be modeled (i.e. plot-scale, field-scale, watershed-scale) must be respected unless otherwise specified by developers. Most of the time, calibration of the model is necessary before running any simulations.

Different models commonly employed in such studies will be described in the following section; they are: (i) AGNPS: Agricultural Non-Point Source Pollution Model (Yoon et al., 1993), (ii) AnnAGNPS: Annualized Agricultural Non-Point Source Pollution Model (Bingner et al., 1998), (iii) CREAMS: Chemicals Runoff and Erosion from Agricultural Management Systems (Kinsel, 1985), (iv) ANSWERS-2000: Areal Non-point Source Watershed Environment Response Simulation (Bouraoui & Dillaha, 1996), (v) DRAINMOD (Skaggs, 1980; Fernandez et al., 1998) and (vi) WEND: Watershed Ecosystem Nutrients Dynamics (Cassell & Kort, 1998). Most of these are designed to evaluate the water quality of water courses. The WEND model allows the exploration of long term P management strategies in fields. This model was identified as the best tool to evaluate various farm management strategies to reduce phosphorus export from the watershed.

2.4.1 AGNPS and AnnAGNPS models

The Agricultural Non-Point Source Pollution Modelling System (AGNPS) (Yoon et al., 1993) is single-event model designed to study and evaluate runoff water quality within an agricultural watershed. It can simulate surface runoff, sediment, nutrients and pesticide movement. The spatial variability of the watershed is accounted for by hydrologic responses units. Each cell is a homogenous copy of the area inside its boundary. Spatial representation of factors influencing pollutant movement can be added to the model. These movements can be followed in the watershed setting on a daily time frame. Because of its flexibility and relative precision AGNPS is broadly use for water quality assessment. However, its single-event characteristic being a major limitation, attempts were made to create an annualized continuous simulation model (Bosh et al., 2001).

AGNPS 2001 is the new generation model used to evaluate the consequences of management practices on a watershed. Different geographic information system (GIS) procedures are used for the development of the inputs, including a synthetic weather generator (GEM), pollutant loading simulation (AnnAGNPS), in-stream submodel and salmon progress submodel. The use of larger watersheds brings more accurate outputs.

AnnAGNPS (Bingner et al., 1998) adds risk analysis of non-point source pollutants for the specific agricultural watershed. The basic principles of AGNPS are maintained in this submodel, but they are applied on a continuous time-frame. It is a multi-event modification of a single-event model (Bosh et al., 2001). The shape of the cells can be modified from their basic shape to a square cells grid with homogeneous hydrological data. Cells also enclose soil type, land use and management practice parameter data. This model includes more detailed inputs than AGNPS alone, resulting in a better representation of the pollutant movement in the watershed. Different forms of point source pollution can be considered in the development of the model. The soil is divided into two layers, allowing different characteristics to exist in each of them. A daily mass balance of N, P and organic carbon (OC) can be produced for every single cell.

Different nutrient parameters are taken into account: plant uptake, fertilizer application, residue cover effects and nutrient processes. The curve number was identified as the most sensitive parameter of the submodel because of its effect on hydrologic processes. The different categories of inputs can be sorted into four main categories: climate conditions, land characteristics, chemical properties and broad field management inputs. The site-specific outputs are based on user choices for different events. This model is subject to different structural limitations: (i) overall runoff and its chemical loads are evacuated every day of the simulation, despite the number of days which would be necessary in the field, (ii) no mass balance calculations are completed for water inflow and outflow, (iii) point source loads need to be constant for the simulation, and (iv) precipitation patterns are the same across the entire watershed (Bosh et al., 2001).

AGNPS was tested on the St-Esprit watershed (26.9 km^2) in Quebec. After calibration, the model produced an accurate forecast of surface runoff and sediment yield in the watershed, but gave poor estimates of peak flow (Perrone, 1997). Poor simulation accuracy occurred under long cold weather periods (Perrone, 1997), as expressed in the limitations of the model. For AGNPS to give accurate results where colder climatic conditions occurs, data representing seasonal factors must be carefully chosen (Perrone & Madramootoo, 1999).

2.4.2 CREAMS model

The Chemicals Runoff and Erosion from Agricultural Management Systems (CREAMS) (Kinsel, 1985) is a field-scale model with homogenous characteristics of land use, soil, climatic conditions and management practices. According to the type of precipitation data, surface runoff is estimated differently, either by SCS curve number when daily precipitation is available or with an infiltration-based model when hourly precipitation data are available. Sediment transport with P and N enrichment ratios, along with soil loss are also calculated using, among other things, the RUSLE equation. Several nitrogen processes are estimated, including mineralization, nitrification and denitrification as well as plant N-removal and percolation. Some pesticides' movements and interactions can also be simulated. The fact that CREAMS is not user-friendly is one downside to this model (Dunn, 1994). The sensitivity of this model to the RUSLE K factor was demonstrated by Loch et al. (1989) in some types of clay soils, where the K factor was underestimated, affecting the soil loss parameter.

2.4.3 ANSWERS-2000 model

ANSWERS 2000 is the latest version of the ANSWERS (Aereal Nonpoint Source Watershed Environment Response Simulation) model (Bouraoui & Dillaha, 1996). It was developed to assess the efficiency of management practices used to decrease soil loss, nutrient contamination of surface waters and nitrification risks arising from nitrogen leaching. The simulation can be done either at the watershed or at the farm scale. It is a short or long-term continuous distributed parameter simulation model, with no topographical constraints. A grid system of up to 1 hectare in size is applied to an area where parameters are considered to be uniform. The data management is facilitated by an ArcInfo GIS (geographic information system) interface. Data from farm-level monitoring within the watershed are not necessary because the model uses breakpoint precipitation information as the basis for its simulations. Throughout runoff events hydrologic processes are simulated on a 30 sec. time-frame which extends to 24 hours between events. A great deal of information can result from a simulation of interception, infiltration, surface water holding capacity and runoff, leaching and sediment characteristics and movement. The model also produces crop development and nutrient removal parameters. Furthermore, N and P dynamics are combined with the simulation of nitrification, N percolation, total Kjeldahl nitrogen and P in surface runoff, as influenced by soil characteristics, vegetative cover, nutrient and hydrologic conditions. Such interactions are possible because of the presence and equilibrium of four N pools and four P pools duplicating many processes undergone by those nutrients in soil, such as mineralization or plant removal (Dillaha et al., 2001).

ANSWERS-2000 was successfully applied to watersheds in the US to simulate runoff (especially for larger storms), sediment yield, nitrate-related processes, dissolved ammonium, sediment-bound T, K, N and dissolved P losses. The calculated sediment-bound ammonium losses were appreciably different from measured ones (Dillaha et al., 2001). Recently, a groundwater constituent was developed for the model. So far, it can successfully predict drainage below the root zone, evapotranspiration based on vegetation type as well as piezometric levels and tendencies in the watershed (Bouraoui et al., 1997). One of the model's major limitations concerns its applicability; a strong knowledge of hydrologic, soil and crop processes is needed to use it. Limitations related to the model structure concern the sediment detachment submodel, the ammonium losses in surface runoff submodel, the procedures related to fertilizer applications, a non-significant baseflow simulation, non-existing nutrient cycles and fate in receiving waters simulations, and non-existant winter conditions. The model is particularly sensitive to soil clay and silt content because of their influence on hydrology (Dillaha et al., 2001).

2.4.4 DRAINMOD model

DRAINMOD (Skaggs, 1980; Fernandez et al., 1998) is a field scale model developed to simulate different water management practices influencing surface and subsurface water flows in poorly drained soils with high water tables, as well as infiltration and evapotranspiration. Many different weather conditions can be created in a simulation with a time frame of 20 years or more. Simulations of the effects of different combinations of surface drainage, subsurface drainage, controlled drainage and subirrigation on water tables are achieved with a one-dimensional water balance. This water balance is situated at the midpoint between parallel drains, such as open ditches and subsurface tiles (Parsons et al., 2001). Water stress can be applied on different crops based on soil water conditions, permitting the estimation of crop yields. DRAINMOD's capacities were improved to simulate nitrogen leaching under shallow water table depths, providing N concentrations in the different soil horizons and in surface and subsurface drainage waters (Breve et al., 1997). The nitrate pool is balanced according to fertilizer dissolution, denitrification, mineralization and plant removal. The model inputs needed are soil characteristics, field management and long-term climatic data. Additional inputs concerning N transformation rates are required in the nitrogen submodel. DRAINMOD outputs can be generated on a daily, monthly or annual basis and consist of drainage volume, runoff, infiltration, evapotranspiration, water table depth, crop water stresses and numbers of work days based on soil air volume (Parsons et al., 2001). The model is sensitive to saturated hydraulic conductivity, water balance in the unsaturated zone and soil properties associated with evapotranspiration (Parsons et al., 2001). Field specific input data sets can generate water table depth with standard errors of less than 0.2 m, even if the model was not calibrated (Skaggs, 1982). Field and watershed water management sites have been used to broadly assess this model.

2.4.5 The WEND model

The Watershed Ecosystem Nutrient Dynamics model (Cassell & Kort, 1998) is a watershed-scale model developed to study phosphorus progression within watershed boundaries and outside of them. A dynamic simulation modelling framework supports all the mass balance calculations underlying phosphorus cycling processes. These processes include import, export, internal cycling and storage of phosphorus, illustrating the watershed infrastructure. Watersheds are analysed for P inputs and outputs over several decades (Cassell et al., 2000). The model is composed of sectors such as urban, agriculture, forestry where P is processed or stored. Other sectors included in the model are long term storage and output of P from drainage.

The model needs to be calibrated or even customized when applied to a different watershed than the one it was originally intended for. This raises one of the most important limitations of the model, even when used with the source watershed it has to be done by a person with a strong knowledge of P processes taking place in the watershed. Three different WEND models have been developed according to different agricultural productions types: dairy, poultry-grain and swine. With some calibration, these models can be applied to watersheds with similar agriculture production. Another limitation concerns the amount of inputs needed to run this type of model. Because it virtually mimics every sector of a watershed, inputs must range from farm management to urban wastewater treatment plant P loads. This model is not designed for event-base simulation or to explore site-specific methods. Diverse P outputs can be analysed according to the goal intended by the researchers.

The phosphorus processes in a watershed are multifaceted and complex. The unique characteristics of the WEND model appear to make it the best tool to strategically assess long-term management strategies for reducing P pollution on intensive agricultural watersheds in Quebec. The dynamic simulation-modeling framework permits the simulation of many different cropping systems and considers geographic characteristics that are peculiar to the Castor watershed. The model was applied at the farm level, not as a watershed model, so that it could provide an analysis of field-level practices and an evaluation of their long-term effects on P loads. The structure of the model as well as different environmental trend calculations are discussed in greater detail in chapter 3, section 2.

2.5 WEND applications

The first WEND-1 model (Cassell & Kort, 1998; Aschmann et al., 1999) was developed for the Winooski River watershed in Vermont. The watershed covers an area of 2753 km^2 draining into Lake Champlain, and contributing 20% of the total P loads to the lake. This watershed is made up of 74% forested land, 13% principally dairy agriculture, and 8% urban land, by area. Numerous monitoring data were collected on the Winooski River during a one-year period. Three different management scenarios were considered for the basin. The base-line scenario assumed that present-day conditions were maintained during the simulation period. The enhanced development scenario stipulated rate off economic development increased between years 20 and 30. The resources sustainability and environmental protection scenario led to the assessment of environmental management decisions for the different sectors of development in the watershed. The WEND-1 model showed that any changes in the management of the watershed influenced the annual P loads in soil and water.

The WEND-IBW (Cassell & Meals, 1999; Cassell et al., 1999) model was customized from the WEND-1 model for the Inland Bays watershed in Delaware. Half of the 769 km^2 watershed was devoted to poultry-grain oriented agriculture, with production rates of about 95 million birds *per annum.* The tourism industry was also an important economic input with an average of 5 million visitor-days yr^{-1}. The agricultural sector generates 53% of total phosphorus exports *per annum,* and urban activities 36%. Model outputs showed increases in phosphorus in the soils of agricultural lands. Because of the inevitable growth of anthropogenic activities, the phosphorus concentration will continue to increase in both the short- and long-term. If present-day conditions persist, the rate of increase is evaluated to be 124% over 20 years. Considering the different management scenarios, it was apparent that long-term reduction of phosphorus is only possible with the application of extreme urban and agricultural management solutions.

The WEND-LCRW model (Cassell et al., 2000) describes phosphorus behaviour in the Little Cobb River watershed in Minnesota. It is a large swine production-oriented agricultural watershed with 17 000 swine animal units supported on an area of 238 km^2. There is obviously significant manure production that has to be conscientiously managed. The P cycling in this watershed is mainly driven by the swine production. An increase in P concentrations in tile drainage was detected even if the soils immobilize P relatively easily. The WEND-LCRW model predicted an export of P from the watershed on the order of 26 682 kg P yr^{-1}. Given that the subsoil had a high P sorption capacity, only small changes were expected in the composition of leaching water in the next 40 years, even in the worst case scenario. An analysis of model outputs suggested that an implementation of conservation practices to agricultural lands with the objective of reducing the magnitude of annual soil loss would decrease P export from the watershed.

2.6 WEND model description and structure

The first WEND model (Cassell & Kort, 1998) was developed in 1998 by Alan Cassell, professor at the School of Natural Resources of the University of Vermont and Robert Kort of the Natural Resources Conservation Service in Winooski, Vermont. It considers the watershed as a working ecosystem in which people live, work and play (Cassell & Kort, 1998; Cassell et al., 1998; Aschmann et al., 1999; Cassell & Meals, 1999; Cassell et al., 1999; Cassell et al., 2000). This mass balance model is based on three principles (Figure 2-2).

1. Phosphorus must enter (input) or leave (output) the ecosystem by crossing the boundary, in this case the watershed limits.
2. Phosphorus must be stored in compartments. A watershed is composed of many compartments such as forest, fields, cities and industries.
3. Phosphorus can travel through the different compartments of the watershed.

To accurately represent reality, the model is divided into five sectors based on the organization of the watershed studied: forestry, agriculture, urban, drainage net and long-term storage (Figure 2-3). The forestry, agriculture and urban sectors consist of all the human and animal activities in relation with phosphorus within those different environments. The model's agriculture sector, which is that of most interest in this study, will mimic the different field-level P processes on a single field (Figure 2.1). The long-term storage sector includes the P no longer available from the previous sectors, for example, the phosphorus entering the deep regional groundwater or landfills. This P is no longer considered as traveling in the ecosystem, therefore it is removed from the model. Unfortunately, that storage sector is not completely hermetic; consequently some phosphorus can leave by the drainage net sector. The drainage net sector represents the watercourses and their associated channels and floodplains, the main pathway of P transportation throughout the watershed (Cassell & Kort, 1998; Cassell et al., 1998; Aschmann et al., 1999; Cassell & Meals, 1999; Cassell et al., 1999; Cassell et al., 2000).

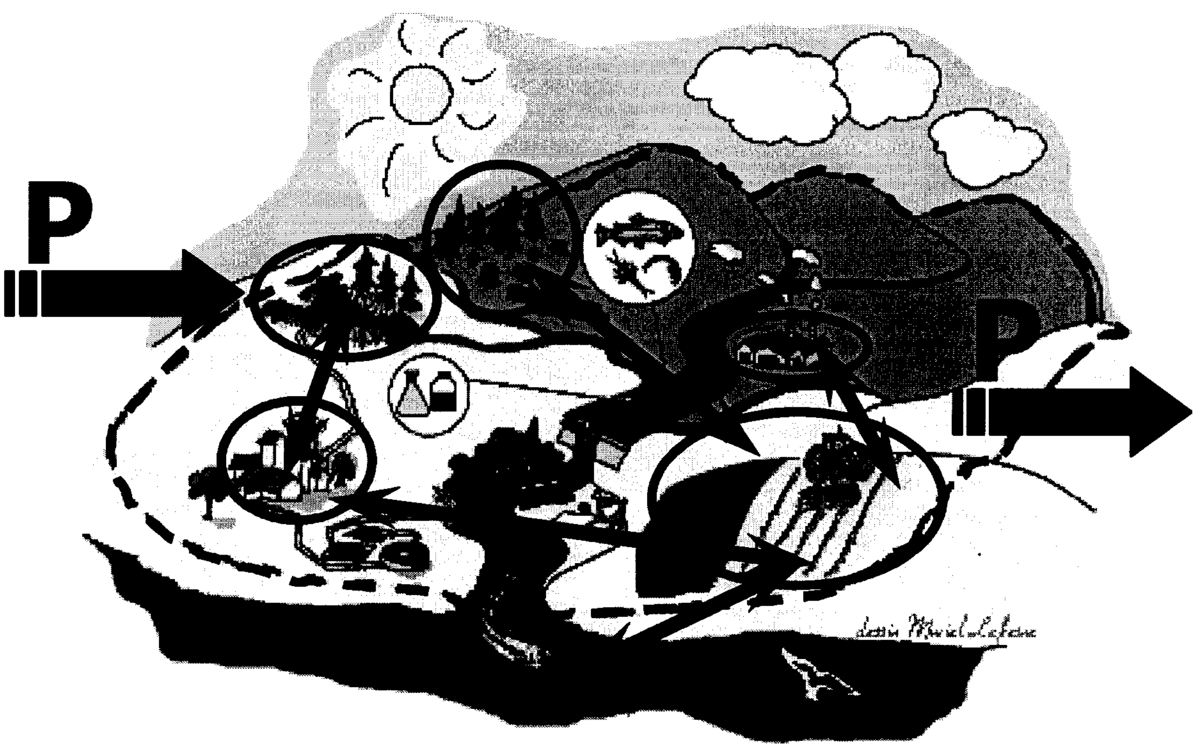

Figure 2-2: Mass balance principles of the WEND model (modified from http://www.menv.gouv.qc.ca/jeunesse/bassin_versant/index.htm).

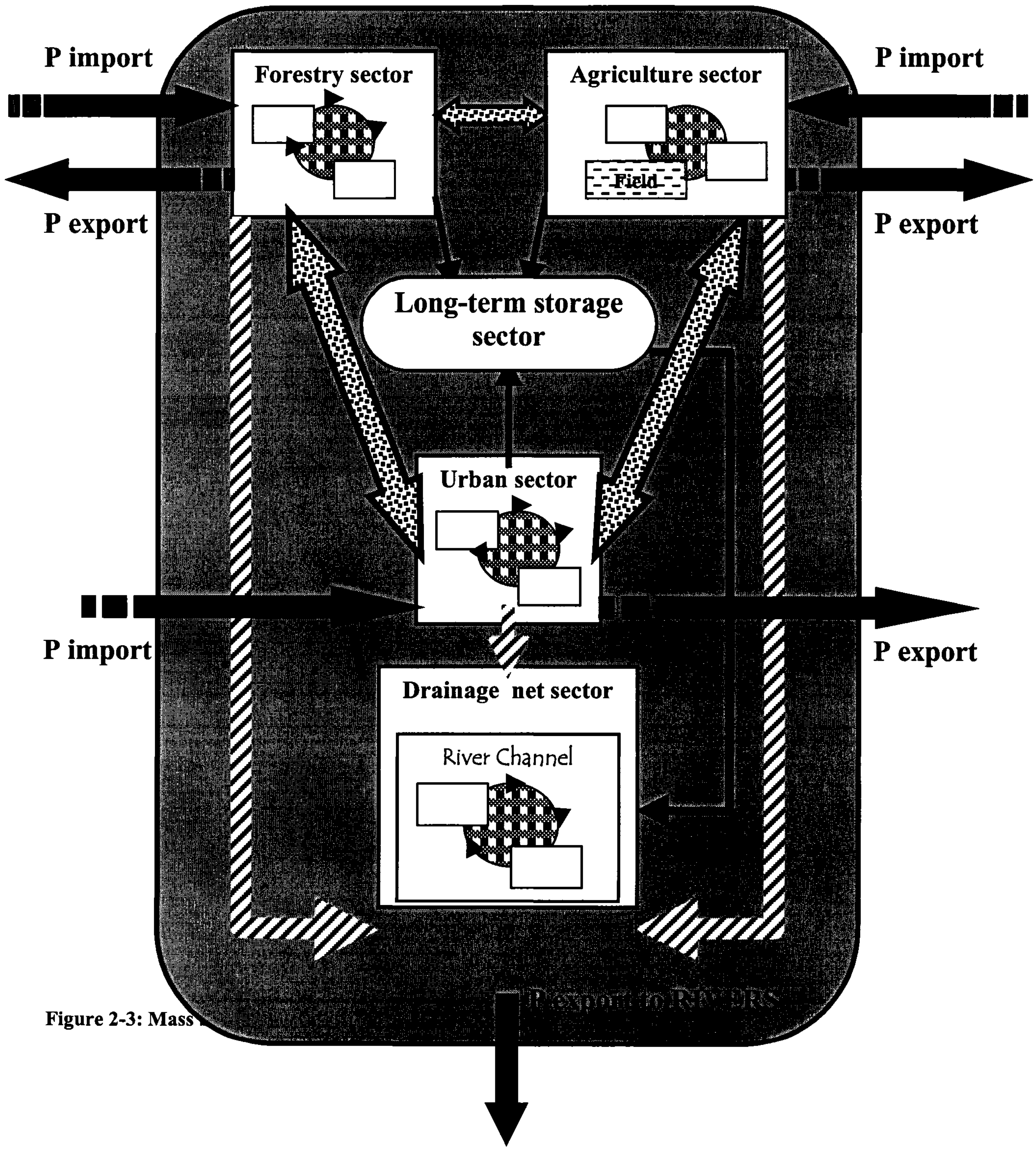

Figure 2-3: Mass

WEND model development is conducted in the Stella (Richmond, 2001) a system-thinking software. This interactive object-oriented dynamic simulation programming environment permits the conception of complex models of linear and non-linear active ecosystems (Cassell & Kort, 1998). When the different P movement and storage processes, including long term storage, are identified, they must provide a complete mass balance within each of the compartments. The total P inputs and outputs crossing the watershed and subwatersheds limits are then evaluated at every user-defined time interval determine (Cassell & Kort, 1998). The movement of phosphorus in each compartment is based on the following mass balance equation:

$$\sum (TP_{in} - TP_{out} - TP_{sto})\Delta t = \Delta(TP_{bm} + TP_{oth}) \qquad 2.1$$

where:

TP_{in} = total phosphorus inputs

TP_{out} = total phosphorus outputs

TP_{sto} = total phosphorus stored

Δt = (delta time) interval of time between calculations in the model

TP_{bm} = phosphorus stored in biomass

TP_{oth} = phosphorus stored in non-biomass forms

When $TP_{in} = TP_{out}$, the storage of phosphorus is remains unchanged (steady-state); when $TP_{in} < TP_{out}$ the storage of P is reduced; when $TP_{in} > TP_{out}$, the storage of P is increased. The mass balance theory is the primary concept subtending the WEND model. Lack of mass balance in the cultivated areas is also considered as one of the major causes of environmental problems (Bolinder et al., 1999).

3 Materials and methods

3.1 Site description

The Champlain Lake is one of the larger water bodies of the American north-east. On the Quebec side of the border it receives waters through Missisquoi bay, which has an area of 390 km^2 representing 3% of the Lake (Figure 3-1). This bay is the site of a number of recreational activities including swimming, navigation, holiday resorts, water sports, fishing and waterfowl hunting. It also serves as a source of fresh water for the municipalities of Bedford and St-Armand (MENV, 1999a). Unfortunately, eutrophication problems jeopardize tourism in the region. Over the past decade, water quality in the bay has noticeably declined. The quantity of suspended solids and real colour value surpassed both indicator and standard levels (Caumartin & Vincent, 1994). Elevated phosphorus loads have been observed to accelerate the natural eutrophication process. The physical characteristics of the bay, such as shallow waters, significant and constant wind, and continuous sunshine during the summer period, have facilitated the proliferation of algal blooms in the past decade. Most point sources of phosphorus pollutions have been taken care of, leaving the challenge of reducing non-point source pollution. The Quebec watersheds emptying into the bay support significant agricultural activities, contributing 79% (MENV, 2000) of the Bay's non-point source phosphorus pollution. A major environmental study with regards to water quality in the Rivière-aux-Brochets river emptying into the bay, concluded that, overall, tributaries exhibited poor water quality, with phosphorus indicators exceeding standards imposed by environmental legislation. The turbidity and suspended solids observed in a number of rivers like the Castor are likely a consequence of erosion problems.

In the Quebec portion of the watershed, positive efforts within the agricultural sector in terms of farm management practices have been overwhelmed by the impact of modifications in on-farm activities. In the past 15 years, areas devoted to large row crops has increased to the detriment of forages. In addition, between 1991 and 1996 the animal units on the watershed rose from 28 000 to 44 000. In 2001, there were 1.35 animal units by hectare of cropland (Statistics Canada, census 2001). Those changes have been mostly felt in the western half of the Quebec side of the Missisquoi bay watershed due to the important topographical variations in the eastern half. This is where the case study of this project, the Castor River, is situated. It is a tributary of the Rivière-aux-Brochets which flows into the Missisquoi bay. The physical characteristics and water quality data of the Castor River will be reviewed in detail in the next section.

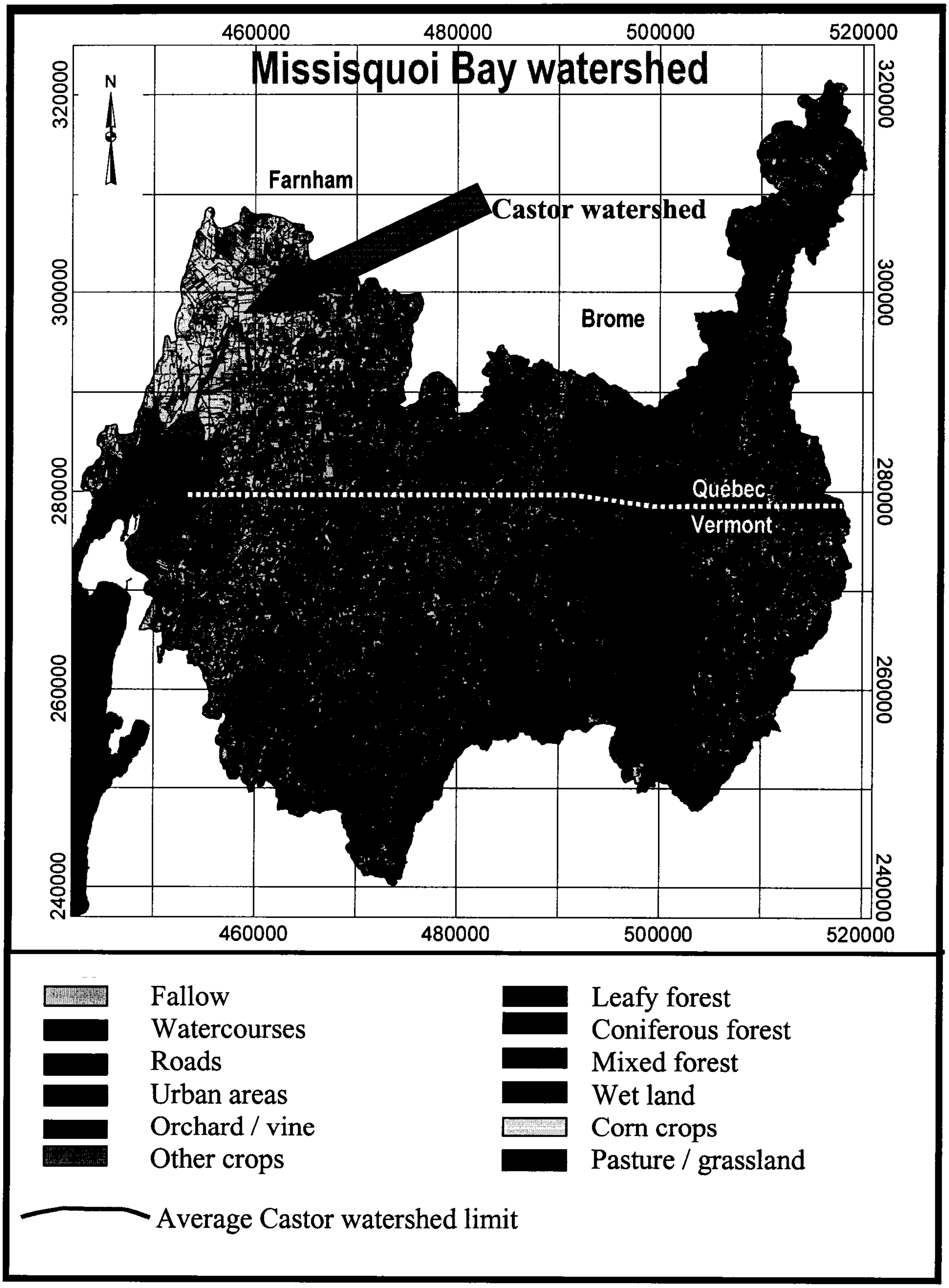

Figure 3-1 : Missisquoi Bay watershed area land use

3.1.1 Castor watershed

The Castor watershed landuse area is mainly devoted to agricultural activities. The areas covered by forested and urban land are minimal and not taken into account in the Castor WEND model (C-WEND). The database for the agriculture sector was obtained through a confidentiality agreement with the producers of the Castor watershed. Therefore, some data types, like maps, cannot be produced on paper, limiting the presentation of results to statistical data. Nevertheless, those data are expected to be quite telling.

The twenty-four farms within the 11 km^2 watershed each account for a mean area of 42 ha. However, some farms only represent a very small area as only one or two of their fields are situated within the watershed. Therefore, considering only farms with at least four fields located within the watershed, the mean area become approximately 47 ha and gives a more accurate portrait of farm sizes. There are four large farms covering together some 409 ha, and representing 44% of the cropland area in the watershed. They are believed to be animal producers due to the important pasture and forages area they own. The animal unit density in Bedford County is evaluated at 1.57 a.u. per ha of cropland (Statistics Canada, census 2001), three quarters of which is devoted to swine production. This is slightly higher than the overall Missisquoi bay watershed value of 1.35 a.u. per ha of cropland (Statistics Canada, census 2001). However, the area of Bedford County is greater than that of the Castor watershed. Consequently this value cannot be directly reported to the watershed, but gives a fair evaluation of what goes on in it. The 266 fields have a mean area of 3.48 ha. Large row crop like grain corn, cereals and soybeans are the principal crops (Table 3-1). The Castor watershed has 11 tributaries which have essentially been altered to serve as surface drainage ditches.

Table 3-1: Crop area of the Castor watershed (Adapted from Michaud et al., 2002a, 2002b, 2003)

Crop	% of total watershed area
Grain corn	46
Cereals	12
Silage corn	2
Soybeans	3
Grasslands	30
Pasture	6

A large database exists of surface and subsurface water quality parameters from different sampling points in the Castor watershed (Jamieson (2000), Enright, personal communication; Michaud et al. 2002a, 2002b, 2003; Enright & Madramootoo, 2004). Phosphorus analyses were conducted on a large number of water and soil samples. Precipitation, flows and volume of water were also measured on the river. Additionally, Michaud (2002a, 2002b, 2003) monitored nutrient balances on the farms of the watershed. These data will be discussed in upcoming sections.

3.1.1.1 Soils in the Castor watershed

The main soils of the watershed are of the Bedford, Richelieu, St-Brigide, St-Damase, St-Sebastien, Ste-Rosalie and Suffield soil groups (Figure 3-1). Alongside the Castor River the land is relatively flat with clayey soil (Ste-Rosalie, St-Brigide). In the higher elevations of the eastern portion of the watershed the land is more undulating and the soil is less clayey (Bedford, St-Sebastien soil types).

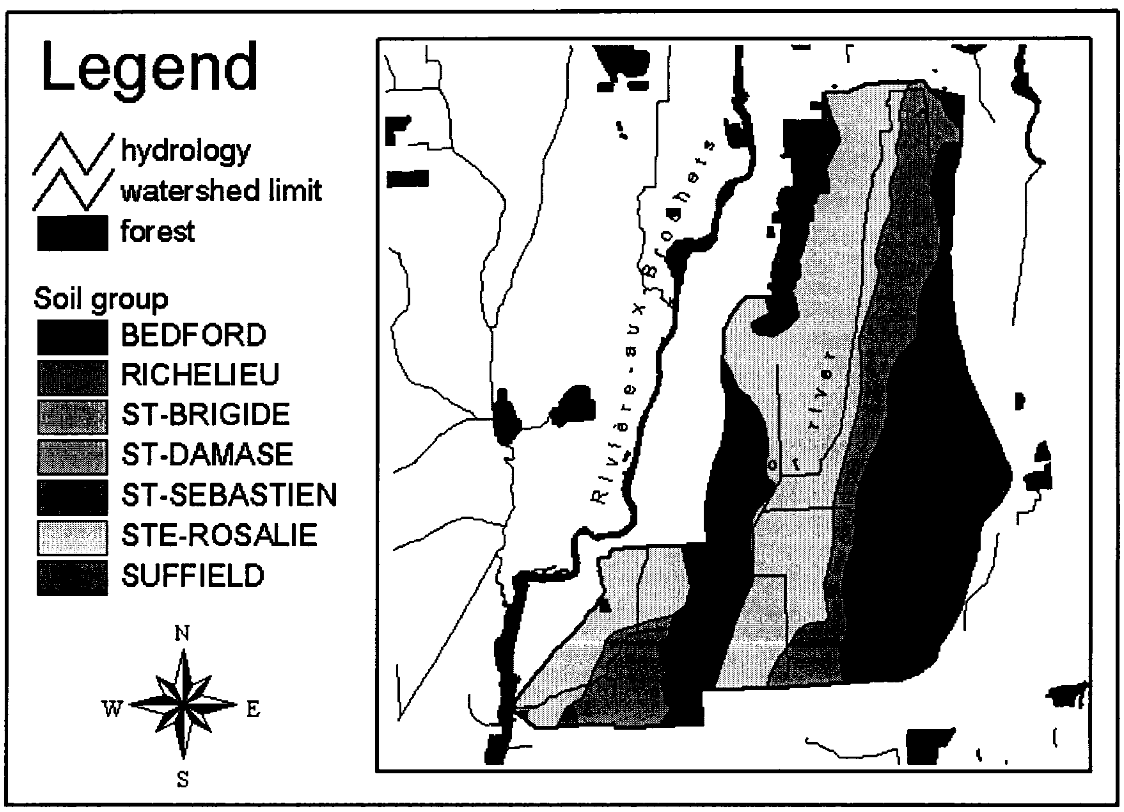

Figure 3-2: Soil groups of the Castor watershed (Michaud et al., 2002a, 2002b, 2003)

Soil tests were conducted on numerous samples from the different fields on the watershed (Michaud et al., 2002a, 2002b, 2003). The Mehlich-III protocol (Mehlich, 1984) was used to determine the level of agronomic P of the soils. Other elements also evaluated were: Al, B, Ca, Cu, Fe, K, Mg, Mn and Zn. The soil aluminium content is used to calculate the phosphorus soil saturation in the watershed fields.

3.1.1.2 On-farm phosphorus balance

In Quebec, the soil phosphorus saturation ratio serves as an environmental indicator of the solubility and desorption of phosphorus. The phosphorus fixation to soil particles depends on the concentration of reactive Al, Fe and Mn ions in the soil system (Giroux & Tran, 1996).

Commercial fertilizer and manure are the most important phosphorus inputs to the fields with a mean input of 27.55 kg TP ha^{-1} yr^{-1} for the watershed. The mean watershed phosphorus removal by harvested grains and plant materials is of 16.8 kg TP ha^{-1} yr^{-1} (Michaud et al., 2002b). These phosphorus inputs and outputs are compared using a mass balance approach and the results are presented in an **Annual Phosphorus Balance** (Table 3-2). Whenever the annual mass balance is positive phosphorus accumulates in the soil. Conversely, when the balance is negative the soil phosphorus concentration decreases.

Table 3-2: Crop area and Annual Phosphorus Balance of surface soil in the Castor watershed for 1997-1999 (Adapted from Michaud et al., 2002b)

Year	Crop removal (harvest)	Manure application	Commercial fertilizers application	Annual Phosphorus Balance
	kg-P/ha-yr	kg-P/ha-yr	kg-P/ha-yr	kg TP /ha-yr
1998	16.9 *	13.1	16.0	12.2
1999	16.7 *	12.5	13.5	9.3
Mean	16.8	12.8	14.75	10.75

* Phosphorus removed from fields as grain, with the remainder of the plant returning as residue.

For the period 1997-2000, the mean annual phosphorus balance for the Castor watershed was 10.75 kg TP ha^{-1} yr^{-1} (Michaud et al., 2002a, 2002b, 2003). Assuming that every 3.75 kg TP ha^{-1} yr^{-1} of excess phosphorus (computed as annual phosphorus balance) results in an increase of 1 kg ha^{-1} of Mehlich-III soil test (Giroux et al., 1996), with an annual phosphorus balance of 10.75 kg TP ha^{-1} yr^{-1}, a mean increase of 2.9 kg Mehlich-III P ha^{-1} yr^{-1} soil test P over the period 1997-1999 could be expected. On average it appears that agricultural fields' soil test P in the Castor watershed is increasing.

3.1.1.3 Water quality data

The Castor River has the lowest water quality of the Rivière-aux-Brochets' tributaries, with high levels of turbidity, suspended solids, poor color and high suspended phosphorus loads (Caumartin & Vincent, 1994). These conditions are believed to result from the highly fertilized soils that enter and are transported through the drainage network of the watershed (Caumartin & Vincent, 1994). The water quality of the Castor River does not seem to have improved over recent years. The MENV's summer median value for IQBP for the period of 1999-2001 was qualified as extremely poor (MENV, 2002b). The Castor River also bears high BOD_5 and fecal coliform levels, thought to be the result of manure applications to the fields (Caumartin & Vincent, 1994). The TP 90 percentile exceeded 0.2 mg L^{-1}, meaning that 90% of the time sample TP levels exceeded this threshold (MENV, 2002b). This value is also clearly higher than the MENV water quality standard of 0.03 mg total P L^{-1} for Québec watercourses. The 90 percentiles for suspended sediment, nitrate, and fecal coliforms, respectively, exceeded 41 mg L^{-1}, 75 mg L^{-1} and ranged between 200 and 1000 UFC $(100\ ml)^{-1}$ (MENV, 2002b). Except for fecal coliforms, these values are the highest for the entire Montérégie region, suggesting that the animal density for the Castor River watershed may be lower than in other regions. However, the data still imply that there is an over application of P and N fertilizer. The nitrate levels are above standards for the protection of aquatic life (40 mg L^{-1}), and consequently above those for human consumption (10 mg L^{-1}) however awareness is raised when the levels reach 1 mg L^{-1} and important actions are taken when it reaches 3 mg L^{-1}. The fecal coliform loads were also above MENV standards (2002a) [200 UFC $(100\ ml)^{-1}$ for swimming and 1000 UFC $(100\ ml)^{-1}$ for consumption after a complete water treatment]. In addition, the erosion is a serious problem, a probable consequence of the predominance of large row crops, especially grain corn.

Industrial activities that produce wastewater are limited to a few businesses. The Champlain Ltd. plant, long thought to be primarily responsible for the poor water quality of Castor River, closed in 1991 (Caumartin & Vincent, 1994). Since 1991, the city of Bedford water treatment plant treats the effluent from the Exeltor and Torrington metallurgy plants. Since 1992, Les Aliments Carrière Inc. factory treats it wastewater by anaerobic digestion before discharging it into the city wastewater treatment plant system. There are no other important contributors of industrial wastewater in the watershed (Caumartin & Vincent, 1994). Many of the residences on the watershed use a septic tank to clean their wastewaters. These systems can be quite efficient in removing P when use properly and maintained in good working conditions. However, systems installed many years ago do not respect current regulations and can allow a significant load of phosphorus to reach waterways. There is no inventory of the quantity and the condition of such systems for the area, but it is a well known fact that they contribute to P loads.

Michaud et al. (2002a, 2002b, 2003) divided the Castor watershed into 10 subwatersheds (Figure 3-3). Some 175 water samples were taken from 10 sampling stations covering the subwatersheds over the period 1997-1999. These samples were analysed for soluble, particulate and bioavailable phosphorus using complex protocols (Murphy & Riley 1962; Sharpley et al. 1991).

Michaud et al. (2002a, 2002b, 2003) also measured precipitation as well as surface flow of the Castor River at the outlet of the watershed. They found that on average 40% of total precipitation was exported from the basin. Subsurface drainage accounted for 52% of this export (Michaud et al. 2002a, 2002b, 2003).

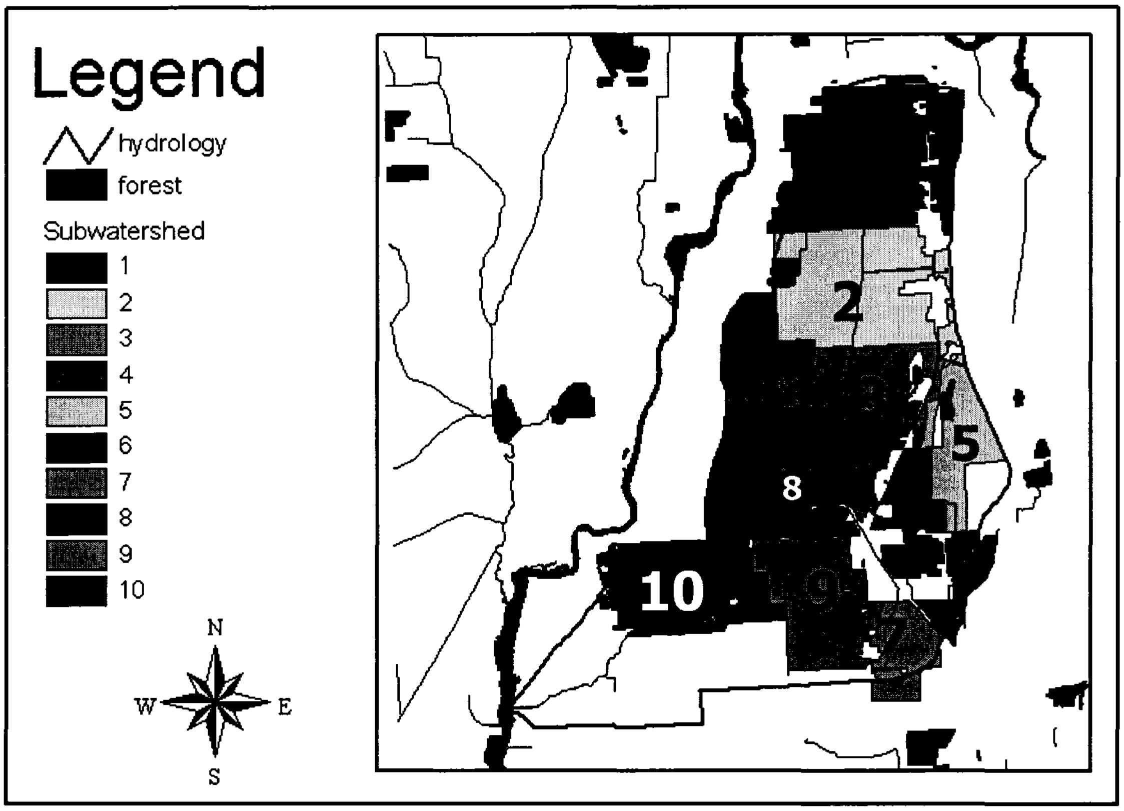

Figure 3-3: Subwatersheds of the Castor River used for sampling (Michaud et al., 2002a, 2002b, 2003)

Water moves over soil surfaces whenever the soil infiltration capacity is exceeded by the rate of precipitation or snow melt (Viessman & Lewis, 1995). In the Castor watershed 40 mm of precipitation over five days is required to noticeably increase stream flow. Consequently, only during large precipitation events do large quantities of phosphorus tend to be exported from the watershed (Deslandes et al., 2002; Michaud et al., 2002b). The phosphorus load (flow × P concentration) exported from the Castor watershed for the period 1997-1999 was 1.54 kg yr^{-1} (Michaud et al., 2002b). Of this 1.54 kg yr^{-1} total phosphorus load, 0.53 kg (P) and 1.01 kg (P) represent soluble and particulate phosphorus, respectively (Michaud et al., 2002b). The ratio of soluble to particulate P is approximately 0.3. The load of exported bioavailable phosphorus was 0.93 kg yr^{-1}, almost equally divided between soluble and particulate phosphorus (Michaud et al., 2002b).

Surface runoff and tile drainage was monitored on two farms of the watershed (Jamieson, 2000; Jamieson et al., 2001; Enright, personal communications; Enright & Madramootoo, 2004). The water samples were analyzed for, among other parameters, TP, ortho-phosphates and total dissolved phosphorus, to assess the ratio of the TP exported in tile drainage to TP lost in surface runoff. This ratio varied from 0.4 to 1.7 during the study period, averaging 0.68.

3.2 WEND modelling of the Castor watershed

In this study, the WEND model was customized to use a field as the basic unit. This led to the creation of a WEND model for each farm which included the same number of field submodels as there were fields on the farm. The outputs values of all the fields were subsequently averaged in a spreadsheet.

The C-WEND model is designed to provide the following outputs: soil test P (kg TP ha^{-1}), soil saturation with Al (%), potential soil loss (RUSLE, kg ha^{-1}) and total phosphorus export (kg TP ha^{-1}). The simulation was run over a 30-year modelling period and the different outputs obtained for each field (266), for every year of the simulation. These outputs were entered in a spreadsheet to give appropriate weighted averages for the watershed and then analyzed to underline the long term watershed P movement and storage trends. These same field outputs were aggregated by crop type and field management practices so they could be analysed according to the different cropping systems, thus allowing the identification of which farming practices were responsible for the worsening phosphorus contamination. Then, the field with the highest risk for P loss was identified and different simulations of crop rotation or management practices done to highlight which management strategies could help reduce field TP export. These outputs could be represented in a GIS to visualize the changes in phosphorus conditions over time, for oral presentations purposes, the confidentiality agreement restricting the printed diffusion of such maps.

3.2.1 Detailed description of the WEND model

3.2.1.1 Description of modelling environment

The C-WEND model was developed within the object oriented system-thinking software Stella (Richmond, 2001). All Stella models are produced by creating unique structures from three types of building blocks or objects (stock, flow and converter). Figure 3-4 shows a typical model structure consisting of these building blocks and describes their functions. This simple model indicates that material flows into the **stock** through the **flow in** and flows out of the stock through the **flow out**. The rate of flow in is determined by the converter **input** which is connected to the flow controller **in flow**. If **flow in** exceeds **flow out**, the increase will accumulate in the **stock** over time. The reverse occurs **if flow** out exceeds **flow in**. A wide range of mathematical, statistical or logical functions are built-in to the software and can be added to any converter. A flow controller has an infinite variety of specific algorithms unique to the model being constructed.

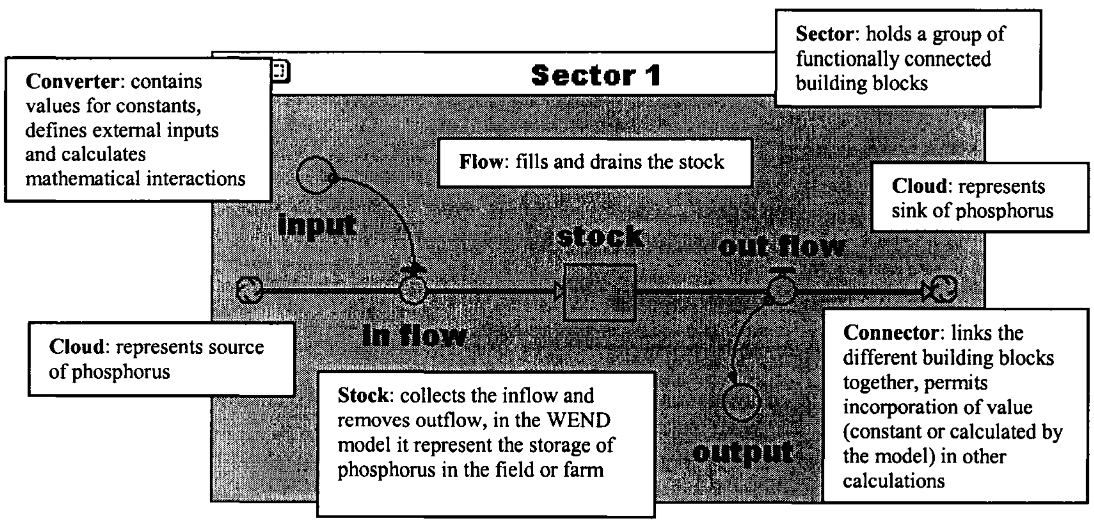

Figure 3-4: Diagram representing typical functionally related elements

Figure 3-5 represents the modelling structure that emulates annual phosphorus mass balance on a typical field in the Castor watershed. Since there are 266 fields in the Castor watershed, there are 266 such structures in the C-WEND model. The basic field phosphorus mass balance is composed of three different phosphorus inputs: ***atmospheric deposition input, commercial fertiliser input*** and ***manure input***. They flow into the ***field soil P*** which represents the soil P pool. There are also three phosphorus outputs from each field: ***plant output***, ***surface runoff output*** and ***tile drain output***. If the inputs sum exceeds the outputs sum the soil P concentration increases. The reverse is true if the outputs exceed the inputs, the soil P concentration decreases. In the models, field soil P is expressed as kg (TP) ha^{-1} while all flows into and out of the field soil P stock are expressed as kg (TP) ha^{-1} yr^{-1}.

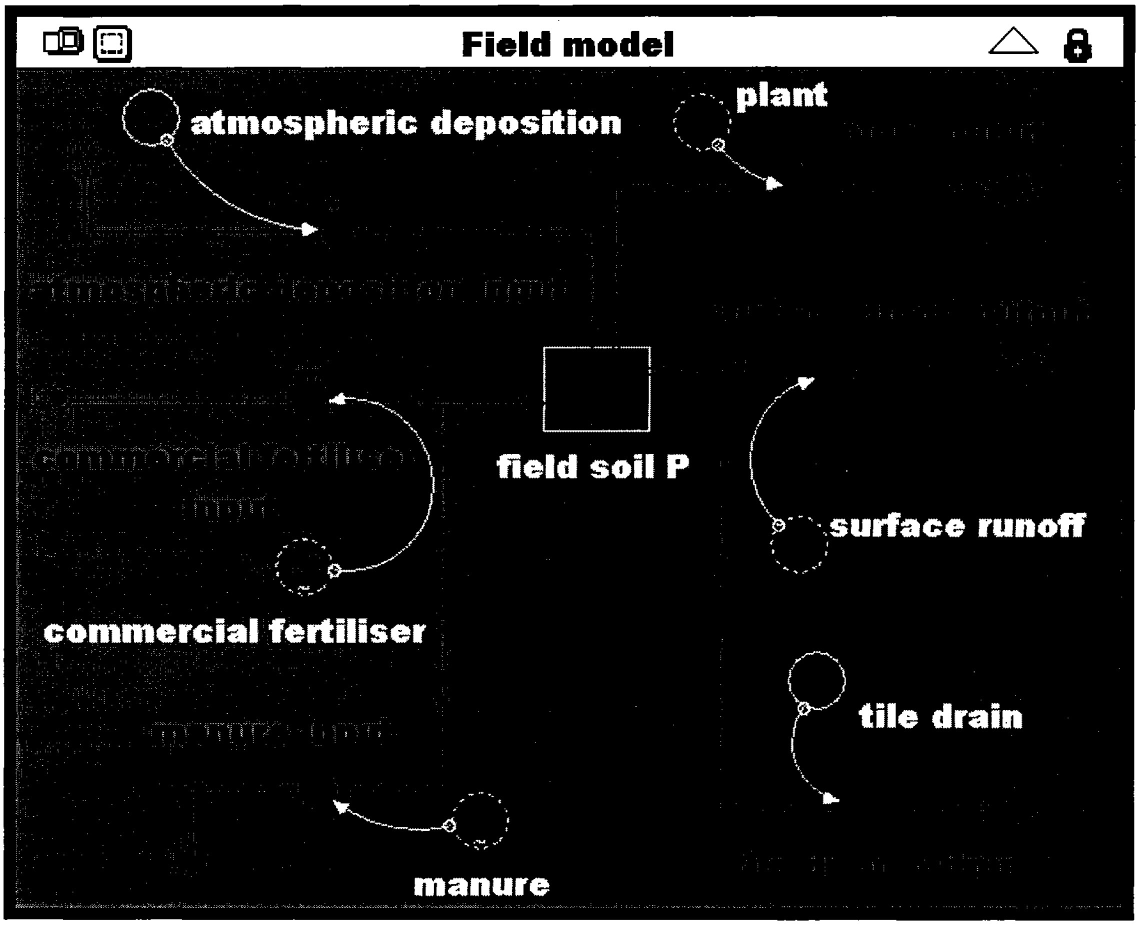

Figure 3-5: Stella structure to compute field-scale Annual Phosphorus Balance of a field

In an existent farm model, the field phosphorus mass balance is composed of many modules such as: field inputs, plant output, bulk density, surface runoff calculations, RUSLE inputs, etc. The steps of the C-WEND model mass balance calculations are discussed in the following section.

3.3 Important algorithms underlying the model

There are a number of algorithms that control computations within this mass balance model. They are derived from monitoring data, values from the literature and local conditions. They were chosen to evaluate the different P inputs and outputs from a field (Figure 3-4). To run the C-WEND model requires the input of 26 different parameters, of which 10 are regional inputs and must be entered once for all the models and 16 are field-specific inputs and must be manually entered for every field. The numerical values of the regional inputs parameters are derived from monitoring data, literature values and local conditions. The field inputs values are based on the data available for each farm of the watershed. The software permits the inputs to be easily changed before each simulation. However, a modification in regional inputs is more time consuming because there is a copy of the model developed for each farm and modifying one does not modify all of them.

3.3.1 Regional inputs

The regional inputs are the same for every field and include precipitation ratio iteration, soil depth, Mehlich P to TP conversion factor, cumulative time system inputs, atmospheric deposition, soluble P to particulate P ratio and RUSLE R and P factors.

3.3.1.1 Precipitation ratio iteration calculations

Michaud et al. (2002a, 2002b, 2003) and others (Deslandes et al., 2002) have underscored the importance of precipitation intensity and duration on phosphorus export from a field. The model was customized to respond to the precipitation pattern because of its manifest influence on surface runoff and tile drain flow. Therefore, special care was taken in the creation of a random iterative system generating the annual precipitation pattern. This module was based on a random function of the normal distribution, generating different precipitation ratios, based on precipitation patterns of past three decades at the Phillipsburg station 17 km south-west of Bedford (Figure 3-6). The mean annual precipitation is 1088 mm, as shown by the black bar on Figure 3-6. The mean annual precipitation produced by the model was 1087 mm. This generated function gives a good representation of the precipitation pattern over time.

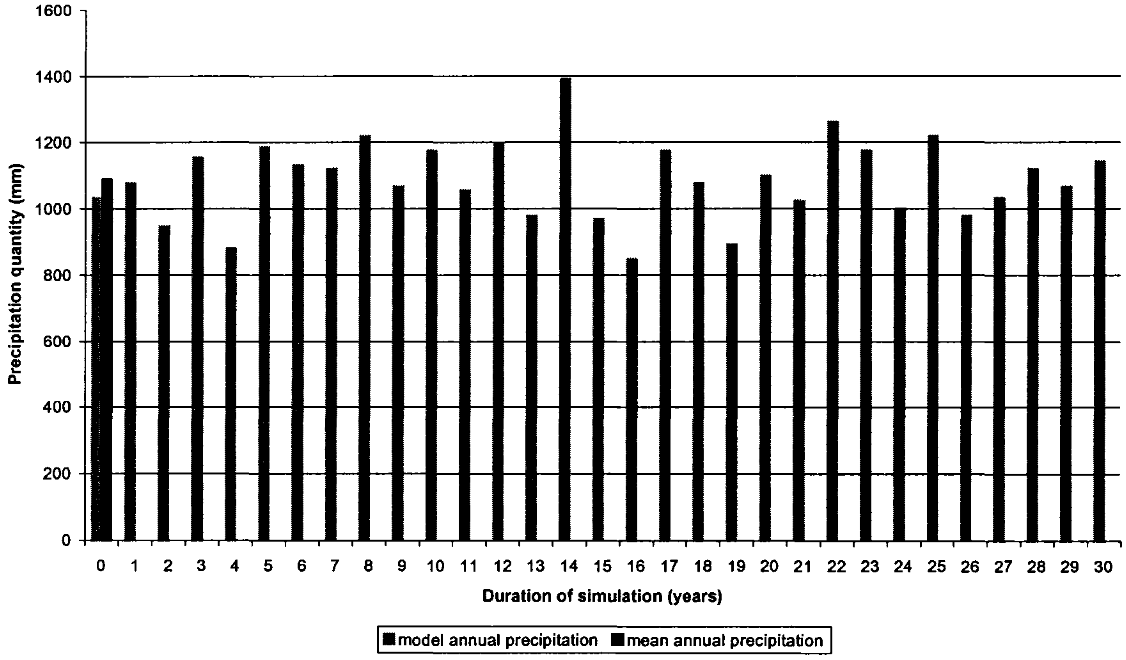

Figure 3-6 : Model-derived annual precipitation (mm) compared to measured mean annual precipitation for the 30-year simulation period

In the model however, the precipitation pattern is not used as the annual quantities in mm but as a ratio corresponding to the actual annual precipitation divided by the mean annual precipitation. The normal distribution has a mean of 1 and a standard error of 0.12 (MENV, 1999b). The first five years (1997 to 2002) are taken directly from the data obtained from the station in Phillipsburg. This permits the model to represent exactly what happened on the basin during the period monitored by Michaud et al. (2002a, 2002b, 2003). The same precipitation ratio pattern as shown in Table 3-3 is applied to every model.

Table 3-3: Precipitation ratio values

Years of simulation	Precipitation ratio	Years of simulation	Precipitation ratio
0	0.95	15	0.89
1	0.99	16	0.78
2	0.87	17	1.08
3	1.06	18	0.99
4	0.81	19	0.82
5	1.09	20	1.01
6	1.04	21	0.94
7	1.03	22	1.16
8	1.12	23	1.08
9	0.98	24	0.92
10	1.08	25	1.12
11	0.97	26	0.90
12	1.10	27	0.95
13	0.90	28	1.03
14	1.28	29	0.98
		30	1.05

3.3.1.2 Plough layer soil depth

The soil depth value used by the MENV when doing soil testing is the plough layer which was set at 0.169 m. It was used in different calculations such as soil P saturation and weight of dry soil.

3.3.1.3 Conversion of soil Mehlich-III P soil test value to total phosphorus value

Since the phosphorus concentration of the field soils was carried out according to the Mehlich-III protocol (Mehlich 1984) and the C-WEND model carried out mass balance concentration based on TP, it was necessary to convert the agronomic Mehlich-III soil P concentration to soil TP concentration. A coefficient of 3.75 was identified by Giroux & Tran (1996) for all soil types and levels of soil P richness and supported by the MENV to convert Mehlich-III P to TP. Basically it means that 1 kg Mehlich-III ha^{-1} represent 3.75 kg TP ha^{-1}. This value was used to modify the soil test P data available from each field into TP soil concentration.

3.3.1.4 Cumulative time system

The availability of field data made it possible to have a three-year rotation pattern. The cumulative time system was used to ensure the stability of the model with this three year rotation system. There are two inputs to this module; the model rotation time and the model rotation time adjust. The model rotation time corresponds to a three-year crop and fertilizer rotation. Therefore the rotation time was set to three years for the Castor watershed. Accordingly, the model rotation time adjust was the value equal to the rotation time required to assure model stability, i.e. three.

3.3.1.5 RUSLE R and P factors

The RUSLE soil loss potential calculation is used in the model to evaluate the soil loss potential of a field. Because some factors are constant they were included in the regional inputs module. The P factor is the supporting practice factor. Since there were no supporting practices in the Castor watershed the value was set to 1. The R factor is the precipitation and runoff factor. It has been evaluated at 1675 MJ mm ha^{-1} hr^{-1} by Madramootoo (1988) and includes winter conditions and critical and severe cases of precipitations.

3.3.1.6 The particulate P to soluble P ratio

The particulate P to soluble P ratio was based on Castor watershed monitoring data obtained by Michaud et al. (2002a, 2002b, 2003). The value was set to 0.3.

3.3.1.7 Ratio of tile drain to surface runoff

Identified in the model as the coefficient ratio, the ratio of tile drain loads to surface runoff loads was obtained by analysis of the continuous monitoring data of two farms in the Castor watershed (Jamieson, 2000; Jamieson et al., 2001; Enright, personal communications; Enright & Madramootoo, 2004). The value of the coefficient ratio was consequently set at 0.68.

3.3.1.8 Atmospheric deposition input

The atmospheric deposition is determined from regional atmospheric phosphorus deposition data. It represents the level of total phosphorus deposited on the soil of the Castor watershed region based on data from USGS Water Resources Investigations (Sherwood, 1999). This deposition constant is set to 0.049 kg (P) ha^{-1} yr^{-1}.

3.3.2 Field management inputs

The field inputs are different for each field of the watershed and are composed of field inputs, fertilisation inputs, crop rotation and management practices inputs. The majority of those values were available from the database collected by Michaud et al. (2002a, 2002b, 2003).

3.3.2.1 Field physical inputs

The field inputs module regroups the physical elements of the field, such as Mehlich-III soil test P which represented the agronomic soil phosphorus concentration of the field. Other values such as field area (ha) and soil Al content (ppm) were also included. An example of typical physical inputs for a field is presented in Table 3-4. The soil series identified the soil group present in the field from the following groups: Ste-Rosalie (silty clay), Ste-Brigide (sandy loam), St-Sébastien (loam), Richelieu (loam), St-Damase (sandy loam), Suffield (silty clay) and Bedford (loam). This input was used to evaluate field bulk density.

Table 3-4: Example of physical inputs to the field

Soil test P	Field area	Aluminium level	Soil series
kg Mehlich-III P ha^{-1}	ha	ppm	
150	7	1033	1 (Ste-Rosalie)

3.3.2.2 Fertilisation inputs

The fertilisation inputs were made up of two values: commercial fertilizer and manure applications. The model assumes that manure and/or commercial fertilizer is added on a three-year rotation, based on the specific conditions of the field being modelled. The unit used is kg TP ha^{-1} yr^{-1} and corresponds to the total quantity of fertilizer applied on the field. To obtain this value the fertilisation value in P_2O_5 was multiplied by 2.29. An example of typical commercial fertiliser and manure application rates are presented in Table 3-5.

Table 3-5: Example of commercial fertilizer and manure application to the field

Rotation year	Manure	Commercial fertilizer
	kg TP / ha	kg TP / ha
1	66	88
2	163	21
3	163	21

3.3.2.3 Crop and management practices inputs

As shown in Table 3-6, the RUSLE K and LS factors, which vary from field to field but are constant for the simulation period, were included in this module. The LS factor stands for the field slope, length and steepness and is unitless. The K factor represents the soil erodability and is expressed in Mg ha hr MJ^{-1} mm^{-1} ha^{-1}. The crop rotation and management practices input are based on the three-year rotation period. They identify the crops cultivated in the field and the management practices used. The three-year crop rotation system was incorporated into the model to more realistically simulate agricultural practices in the Castor watershed. The model was built this way to compensate for the lack of information on crop rotations, because crop rotations are usually based on a 5- to 7-year duration. An example of management practice inputs is presented in Table 3-6.

Table 3-6: Example of crop and management practice inputs

LS factor	K factor	Management practices	Crop rotation
unitless	in t ha h MJ^{-1} mm^{-1} ha^{-1}		
0.2	0.0253	1 (conventional) 2 (conservation) 3 (no till)	1 (grain corn) 2 (pasture) 3 (cereals)* 4 (silage corn) 5 (grasslands with alfalfa)* 6 (grasslands) 7 (soybeans)

**Triticum æstivum* L., *Hordeum vulgare* L., *Avena sativa* L.
***Medicago sativa* L.

3.3.3 Plant output calculations

The plant output represents the amount of P taken up annually from the field by plants. It is calculated by multiplying the plant uptake by the crop yield, determined from the crop rotation input (Table 3-7). In the model, logical built-in equation and graphical function identify the value to use in the multiplication base on the crop rotation input. The plant output is subtracted from the soil test P value. Because the rotation pattern is on a three-year period, there can only be three-different plant output values. However, if the crop rotation doesn't change, the plant output value can be the same for the 30 years simulation period.

Table 3-7: Example of field-scale plant output calculations over a three year simulation

Crop rotation input	Crop yield Mg (TP) ha^{-1}	Plant uptake at harvest kg (TP) Mg^{-1}	TP plant output kg (TP) ha^{-1}
1 (grain corn)	7	3	21
2 (pasture)	3.5	0	0
3 (cereals)*	3.5	4	14
4 (silage corn)	13	2.5	32.5
5 (grasslands with alfalfa)*	3.5	3	10.5
6 (grasslands)	5	2.5	12.5
7 (soybeans)	3.5	6	21

**Triticum æstivum* L., *Hordeum vulgare* L., *Avena sativa* L.
***Medicago sativa* L.

3.3.4 Surface runoff and tile drain outputs

To achieve the surface runoff P load calculations a number of elements need to be in place (Figure 3-7). Those in white have been explained in previous sections. The more pointers there are towards a converter (O) the more elements enter into the calculations effected there. To facilitate the understanding of all the equations needed for the evaluation of P loads exported from a field, elements will be discussed in the order of the number of arrows going to them. The simpler elements are often the results of previous calculations reported in this module.

The surface runoff output is based on the RUSLE equation and the soil phosphorus concentration identified in field soil P. It is calculated annually according to the addition of soluble and particulate phosphorus.

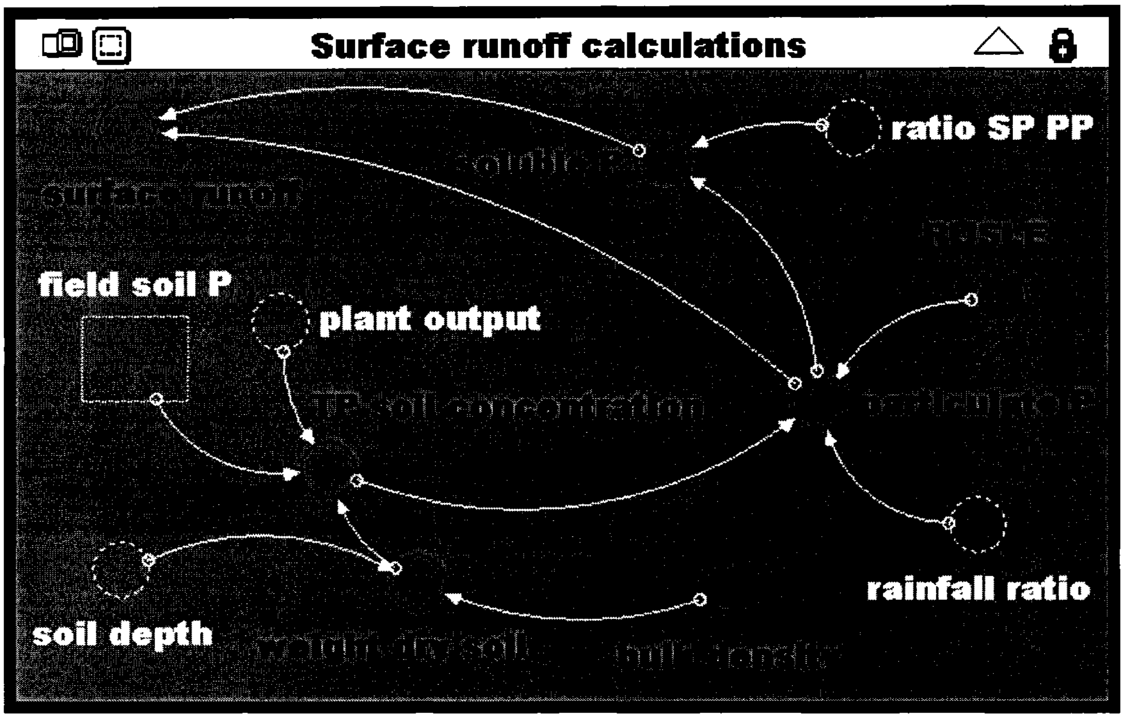

Figure 3-7: Stella structure for field-scale surface runoff calculations

3.3.4.1 Bulk density calculations

The bulk density of a soil represents the mass of dry soil per unit of bulk volume including air and water space. The greater the bulk density the less compact the soil is. Accordingly, soils with higher sand content have greater bulk densities than soils with higher clay content. In the model, a graphical function identifies the value to use in the multiplication, based on the soil series input. An example of typical bulk density and soil clay content values associated with soil series and related nomenclature are presented in Table 3-8.

Table 3-8: Bulk density according to soil series

Soil series	Soil nomenclature	Bulk density Mg m^{-3}	Clay content %
1	Ste-Rosalie	1.36	51
2	Ste-Brigide	1.64	12
3	St-Sébastien	1.5	19
4	Bedford	1.4	18
5	Richelieu	1.45	35
6	St-Damase	1.6	8
7	Suffield	1.35	26
8	Other	1.39	20

3.3.4.2 RUSLE calculations

The Revised Universal Soil Loss Equation (RUSLE) evaluates the potential soil loss from a field. The multiplication of all five factors (R, P, LS, K and C) give the A factor which is the quantity of soil loss from the particular field in Mg ha^{-1}. As seen in previous section the R (precipitation and runoff factor) and P (supporting practices factor) factors are set for the whole watershed in the regional input module. The R factor was 1675 Mg ha hr MJ^{-1} mm^{-1} ha^{-1} and the P factor was 1. The LS and K factors were calculated for each field according to Michaud et al. (2002a, 2002b, 2003) and are inserted in the field input module. C factor value is set by RUSLEFAC (Pringle et al., 1995) and is based on the crop rotation and management practices; consequently it can vary from field to field. In the model, logical built-in equations and graphical functions identify the value to use in the multiplication based on the crop rotation and the management practices (Table 3-9). The A value is then transferred to the particulate P calculations as soil loss potential value in kg ha^{-1} yr^{-1}.

Table 3-9: Example of field-scale RUSLE calculations

Crop rotation	Management practices	C factor	LS factor	K factor	A Mg ha^{-1} yr^{-1}
1 (grain corn)	1 (conventional)	0.37	0.12	0.0253	**1.88**
	2 (conservation)	0.32		0.0250	**1.63**
	3 (no till)	0.15		0.0177	**0.76**
2 (pasture)	1 (conventional)	0.004	0.15	0.0253	**0.025**
	2 (conservation)	0.004		0.0250	**0.025**
	3 (no till)	0.004		0.0177	**0.025**
3 (cereals)*	1 (conventional)	0.41	0.45	0.0253	**5.47**
	2 (conservation)	0.36		0.0250	**4.8**
	3 (no till)	0.15		0.0177	**2**
4 (silage corn)	1 (conventional)	0.51	0.12	0.0253	**2.59**
	2 (conservation)	0.44		0.0250	**2.34**
	3 (no till)	0.21		0.0177	**1.07**
5 (grasslands with alfalfa)**	1 (conventional)	0.02	0.15	0.0253	**0.13**
	2 (conservation)	0.02		0.0250	**0.13**
	3 (no till)	0.02		0.0177	**0.13**
6 (grasslands)	1 (conventional)	0.004	0.45	0.0253	**0.053**
	2 (conservation)	0.004		0.0250	**0.053**
	3 (no till)	0.004		0.0177	**0.053**
7 (soybeans)	1 (conventional)	0.46	0.12	0.0253	**2.34**
	2 (conservation)	0.4		0.0250	**2.03**
	3 (no till)	0.28		0.0177	**1.42**

**Triticum æstivum* L., *Hordeum vulgare* L., *Avena sativa* L.
***Medicago sativa* L.

3.3.4.3 Weight of dry soil

The weight of dry soil is the weight of a surface of one hectare of soil according to its particular bulk density and to a depth of 0.169 m. An example of values of weight of dry soil is presented in Table 3-10. It is used in the calculations of TP soil concentration and is based on the original model composition.

Table 3-10: Example of field-scale weight of dry soil calculations

Soil series	Bulk density Mg m^{-3}	Weight of dry soil Gg ha^{-1}
1 (Ste-Rosalie)	1.36	**2.30**
2 (Ste-Brigide)	1.64	**2.77**
3 (St-Sébastien)	1.5	**2.54**
4 (Bedford)	1.4	**2.37**
5 (Richelieu)	1.45	**2.45**
6 (St-Damase)	1.6	**2.70**
7 (Suffield)	1.35	**2.28**

3.3.4.4 TP soil concentration

The TP soil concentration is the concentration in mg of phosphorus per kg of soil. It is used in the particulate P calculations. To be computed plant output is subtracted from field soil P and divided by the weight of dry soil. An example of typical values of soil TP concentration is presented in Table 3-11.

Table 3-11: Example of field-scale TP soil concentration calculations

Weight of dry soil Gg ha^{-1} (see Table 3-10)	Field soil P kg TP ha^{-1}	Plant output kg ha^{-1} (see Table 3-7)	**TP soil concentration mg P kg^{-1}**
2.30	472.5	21	**196.44**
2.30	649.5	32.5	**268.45**
2.30	625.2	14	**265.92**

3.3.4.5 Particulate and soluble P calculations

The particulate P value represents the particulate P load in surface runoff. It is obtained by multiplying the TP soil concentration with the soil loss potential value to identify the phosphorus quantity potentially eroded from a field and then it is also multiplied by the precipitation ratio. An example of the values used in the computation of surface runoff is presented in Table 3-12. The model was customized to respond to the precipitation pattern because of its manifest influence on surface runoff and tile drain flow. To obtain soluble P, the particulate P value was multiplied by a ratio of 0.3. This ratio was calculated by Michaud et al. (2002a, 2002b, 2003) for the Castor watershed. Surface runoff load is then attained by adding those two values.

Table 3-12: Example of field-scale surface runoff calculations

TP soil concentration mg P kg^{-1}	Precipitation ratio	RUSLE kg ha^{-1}	Soluble P kg ha^{-1}	Particulate P kg ha^{-1}	**Surface runoff kg ha^{-1}**
196.44	0.95	2670	0.15	0.5	**0.65**
268.45	0.99	1510	0.12	0.4	**0.52**
265.92	0.87	1080	0.07	0.25	**0.32**

3.3.5 Soil saturation calculations

The soil P saturation was calculated according to Giroux and Tran (1996) and was expressed as a percentage. In the model, two equations were built in different modules to make the calculation more field-specific (Table 3-13). The soil P saturation equation divides the field soil Mehlich-III P concentration (column 6) by the conversion coefficient (column 5) and multiplies it by 100 to obtain a percentage (column 7). The conversion factor (column 5) translates the Al soil concentration in ppm (column 1) to kg ha^{-1}. To achieve that Al, is multiplied by soil depth (column 2), bulk density (column 3) and a conversion coefficient (column 4).

Table 3-13: Example of field-scale soil P saturation calculations

1	2	3	4	5	6	7
Al ppm	Soil depth m	Bulk density Mg m^{-3}	Conversion coefficient to kg Al ha^{-1}	Conversion factor kg Al ha^{-1}	Field TP convert into Mehlich-III P kg ha^{-1}	**Soil saturation** **%**
1027	0.169	1.36	10	2360.46	126	**5**
1027	0.169	1.36	10	2360.46	173	7
1027	0.169	1.36	10	2360.46	168	7

4 Results and discussion

The WEND model was customized to run on a field-scale rather than on a watershed-scale. It was then individually applied to 266 fields on the watershed over a 30-year simulation period. To customise the WEND model to an agricultural-only interface, the following developments were made: (i) integration of an iterated precipitation routine, and (ii) a combination of crop rotation and management practices.

Field specific information for the 1997-1999 period was available for the fields of this watershed (Michaud et al., 2002a, 2002b, 2003) The monitoring data used as basic inputs for the model were collected in previous studies (Jamieson, 2000; Jamieson et al., 2001; Enright, personal communications; Michaud et al., 2002a, 2002b, 2003; Enright & Madramootoo, 2004). Climatic information was inferred from local sources. The additional information required to run the model was derived from the literature.

An elaborate comparison between monitored and predicted data would have definitely been interesting. However, many reasons make this unfeasible: (i) as in the case of many model applications, the lack of field data equivalent to the outputs of the model, preclude a direct comparison, (ii) the confidentiality of the data made it complicated to produce any logical comparison between predicted and monitored values; given the size of the watershed a comparison of aggregate values would have been virtually meaningless, and (iii) the only available monitoring data were used to build the model, so the assessment of shifts in P parameters over the modelling time scale were not possible.

Results of the modelling analysis are presented here for three different scales: (i) overall watershed impacts, (ii) cropping system, and (iii) for a field situation showing high risk for P loss. All the simulations were done for a 30 year period on the 266 individual fields. The selected outputs are soil test Mehlich-III P, soil P saturation, potential soil loss and TP export. The MENV (2003) codes and regulations on phosphorus application will be used as fertilizer application recommendations for the fields at high risk of P losses.

4.1 Castor watershed impacts

A first set of simulations investigated what would happen if current practices continued to be used in the long term. Area-weighted means were determined by computing means of values obtained for individual fields. This is not a "true" complete watershed analysis because the WEND simulations do not consider forest, industrial and urban sources of P. However, the Castor watershed is very small and these areas can be considered negligible. The type of crop and the management practices have an important effect on the amount of P lost from a field by erosion. Thus, the area that a particular crop occupies on the watershed and/or the area on which a particular management practice is used will give one a fair indication of the trends one is likely to observe in a the long term simulation. The distribution of crops was similar over the three-year rotation. Most of the cultivated area of the watershed was cropped to corn. The area dedicated to corn went down in the 2nd and 3rd years of the rotation, while the area cropped to cereals and forages increased (Table 4-1).

Table 4-1: Percentage of area by crop type for the three year rotation period

CROPS	Percent area by year of rotation		
	Year 1	Year 2	Year 3
Grain corn	54	46	45
Pasture	6	6	5
Cereal (wheat, barley, oat)*	8	15	17
Silage corn	2	2	3
Grassland with alfalfa**	5	1	0
Grassland	24	28	28
Soybean	1	1	2

**Triticum æstivum* L., *Hordeum vulgare* L., *Avena sativa* L.
***Medicago sativa* L.

As for management practices (Table 4-2), conventional practices are more prevalent in the first year of rotation and they decrease slightly in the second and third year to the benefit of no-till practices. It is important to note that Table 4-2 describes the entire watershed area, including forages. Since the forages are perennial crops, the data illustrates that conventional tillage is by far, the dominant practice. In fact, the percentage of the watershed area in no-till in Table 4-2 (38%) is almost the same as the percentage in forages in Table 4-1 (34%). As less corn is grown in years 2 and 3 of the rotation, the percentage of the watershed area not tilled increases by 4%. This implies that some of the cereals and all of the forages are being grown using conservation tillage techniques. With this data (Tables 4-1 and 4-2), it is expected that the potential soil loss and TP exports will decrease in the 2nd and 3rd year of rotation.

Table 4-2: Percentage of area by management practice for the three-year rotation period

Management practice	Percent of area by year of rotation		
	Year 1	Year 2	Year 3
Conventional	59	56	56
Conservation	3	3	2
No-till	38	42	42

Tables 4-3 illustrate the mean quantity of fertiliser in TP applied by crop for the three years of rotation. There is a larger application for the first year compared to the second and third year. Such levels are mainly due to the higher proportion of corn in the first year of rotation (Table 4-1).

Another important element to consider is the greater quantity of commercial fertiliser and manure applied on cereals for the three years of rotation. It is mainly a consequence of late summer application of manure on cereal crop fields. Silage corn also receives a great amount of fertiliser especially for the first year of rotation, but due to the small watershed area covered (2%, 2%, 3% by year of rotation) its impact is minor.

Table 4-3: Mean quantity of fertiliser applied according to each crop for the three year rotation period

CROPS	Qty kg TP/ha Year 1		Qty kg TP/ha Year 2		Qty kg TP/ha Year 3	
	manure	com fert	manure	com fert	manure	com fert
Grain corn	10	32	10	20	10	20
Pasture	3	7	0.6	4	0.7	4
Cereal (wheat, barley, oat)	57	68	38	10	34	9
Silage corn	45	63	0.4	19	0.3	15
Grassland with alfalfa	0.7	10	0	14	0	0
Grassland	7.4	16	7.4	8	7.4	8
Soybean	0	17	0	18	0	15

It should be underlined that manure is not an effective fertiliser for soybeans. However, some people consider it produces excellent results when apply in the fall prior to soybean crop cultivation. Therefore, there was no application of manure for this crop during the simulation. Commercial fertilizers are applied on the fields in this watershed in considerable amounts. However, there is probably enough manure in the region to cover the plant requirements.

4.1.1 Soil loss potential

Predicted soil loss values varied slightly over time: in the first year of rotation the soil loss potential was 2085 kg ha^{-1} yr^{-1}; for the second year, 1973 kg ha^{-1} yr^{-1}; and for the third year, 1932 kg ha^{-1} yr^{-1} (Figure 4-1). There was no increase or decrease over the 30-year simulation period because RUSLE calculations are based upon a constant R value. As well, since the precipitation pattern is not used in the RUSLE calculations, they obviously cannot have any influence over their predicted values. The repeating pattern in Figure 4-3 is a consequence of the three year rotation system. The slight decrease in soil loss potential over each 3-year period is due to a lower presence of corn and conventional management practices over the watershed in the last two years of the rotation (Tables 4-1 & 4-2).

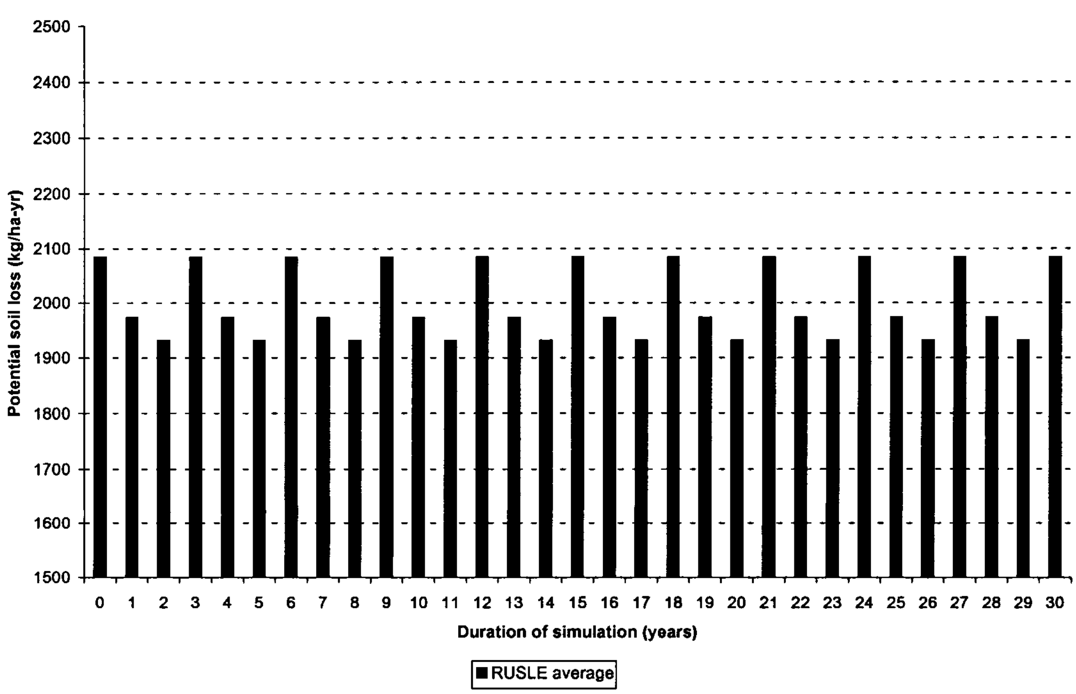

Figure 4-1: Mean predicted soil loss from agricultural fields in the Castor watershed

It is difficult to compare soil loss potential calculated with RUSLE with actual field monitoring. RUSLE has a number of limitations; the equation does not consider gully erosion, streambank erosion or mass soil failure (Yoder et al. 2001). Further, RUSLE calculates what can potentially be eroded from the fields, but does not consider the part of the potential soil loss which is redeposited elsewhere on the field, at the edge of the field or in the stream network. Consequently, the difference between the potential soil loss and what is measured at the outlet can be quite large. Adding some type of sediment yield ratio and sediment delivery ratio estimates to the model can improve its efficiency. The sediment yield ratio represents the amount of sediment passing a given point in a watershed, usually the outlet, over a certain period of time, expressed in kg yr^{-1} (Novotney & Olem, 1994). The sediment delivery ratio represents the ratio of the amount of sediment which leaves a watershed to the gross amount of sediment that was originally eroded from the soil mass in the watershed (Novotney & Olem, 1994). Coupling the soil loss potential calculations with the precipitation pattern could also improve the estimation of soil loss especially in the long-term. However, a mean value of 2000 kg ha^{-1} yr^{-1} for soil loss potential of the watershed's fields is still a reasonable estimation of what could be expected to be lost from these fields.

4.1.2 Soil test P

Soil test P values are reported in Mehlich-III P kg/ha. Phosphorus enrichment of soil has been observed in other areas of Quebec where similar agronomic production systems are implemented. Given that annual P applications exceeded crop requirements on this watershed, it was anticipated that the simulation model would predict soil enrichment on the Castor watershed, which is indeed the trend which appears in Figure 4-2. This also shows the influence of the three year rotation in crop type and field management practice. As expected, annual variations in precipitation (Figure 3-6) did not appear to affect long-term soil enrichment. Since, as demonstrated in later sections, variation in precipitation simply affected annual losses, whereas soil P enrichment was driven by annual fertilizer applications exceeding plant requirements.

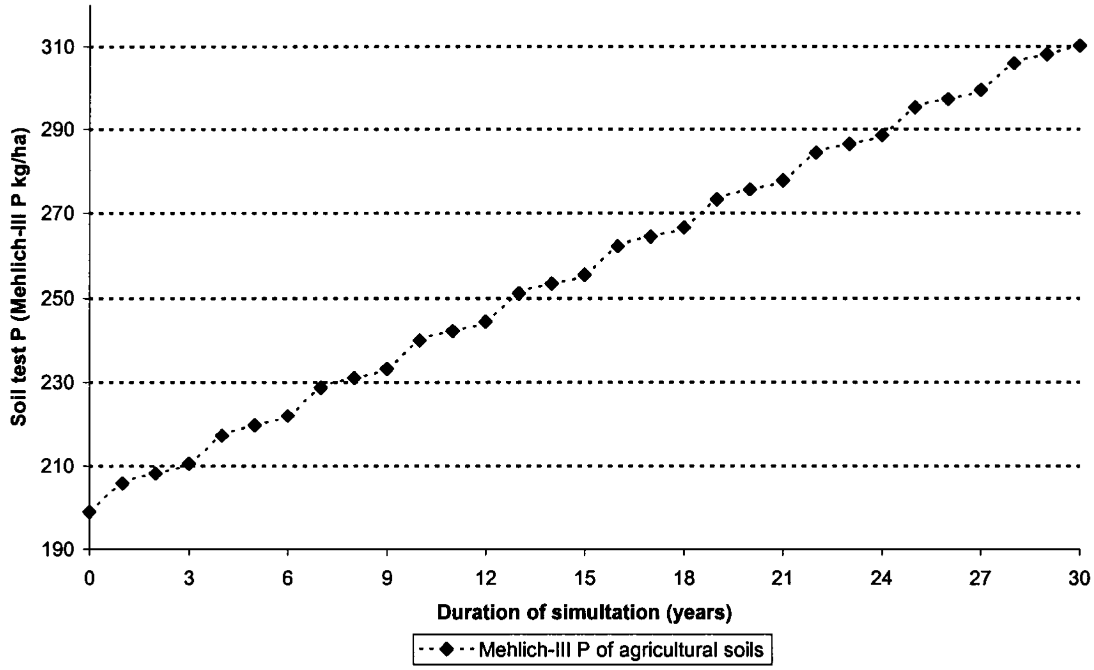

Figure 4-2: Predicted Mehlich-III P concentration of Castor watershed agricultural soils (The data represented on the graph are punctual; the dotted line between symbols serves to better illustrate the trend.)

Further results for the watershed soil test P values are presented in Table 4-4. Were management practices and crop rotations to remain unchanged for a 30-year period, one would expect the watershed soil P concentration to experience a rise of similar magnitude to that predicted by the model: 2.1-6.8 kg Mehlich-III P ha^{-1} yr^{-1}. The variability in annual P enrichment is due to the variation in fertilizer inputs linked to the different management practices and crop rotation.

The mean soil test P increase predicted by the model was 3.71 kg Mehlich-III P ha^{-1} yr^{-1}, representing, over the whole watershed, a mean P input that would, if current field management practices remained unchanged, exceed plant requirements by roughly 32 kg P_2O_5 ha^{-1} yr^{-1}. Over the 30-year simulation period, this corresponds to an input of about 960 kg P_2O_5 ha^{-1} over the plant requirements for the whole watershed.

Table 4-4: Watershed predicted mean soil test P compared with measured soil test P values

Model predictions for watershed soil test P					Measured soil test P increase 1997-1999 *
Year 0	Year 30	Rate of increase			
		1997-1999	30-year mean	Annual mean	
kg Mehlich-III P ha^{-1}	kg Mehlich-III P ha^{-1}	kg Mehlich-III ha^{-1} yr^{-1}	kg Mehlich-III ha^{-1} yr^{-1}	%	kg Mehlich-III ha^{-1} yr^{-1}
199	310	4.6	3.71	1.4	2.9

*(Michaud et al, 2002b)

Input parameters for the model were based on the data collected for the period 1997-1999 by Michaud et al. (2002a, 2002b, 2003). Michaud also conducted annual soil sampling to assess soil enrichment and P export. They noted a 2.9 kg Mehlich-III ha^{-1} yr^{-1} increase in soil test P from 1997 to 1999. For the same timeframe, the simulation model predicted 4.6 kg Mehlich-III P ha^{-1} increase in soil test P. The difference can be attributed to the simplistic structure of the model, or to climatic effects or both. The model is not a perfect mirror of reality; it is only meant to mimic the most important known phosphorus processes. Therefore, some processes that can have a real effect on soil P concentration may not be described at all or may be inadequately described. On the other hand, the period during which the soil tests were taken can affect the values generated in the test. The model gives a mean annual value, but the soil test is a one-time spot measurement. This may explain the small difference between the predicted and measured values in 1997-1999 change in soil test P (Table 4-4).

4.1.3 P saturation

The soil P concentration is an important constituent of the soil P saturation calculation: P Mehlich-III / Al Mehlich-III. Al concentration is unlikely to change; therefore, P saturation trends should follow soil test P trends. The linear increase observed in the in P saturation with time (Figure 4-3), is directly linked to the increase in soil P test. The effect of the crop rotation and management practices is also apparent on the P saturation values. Again, the influence of annual precipitation is not present in the trendline.

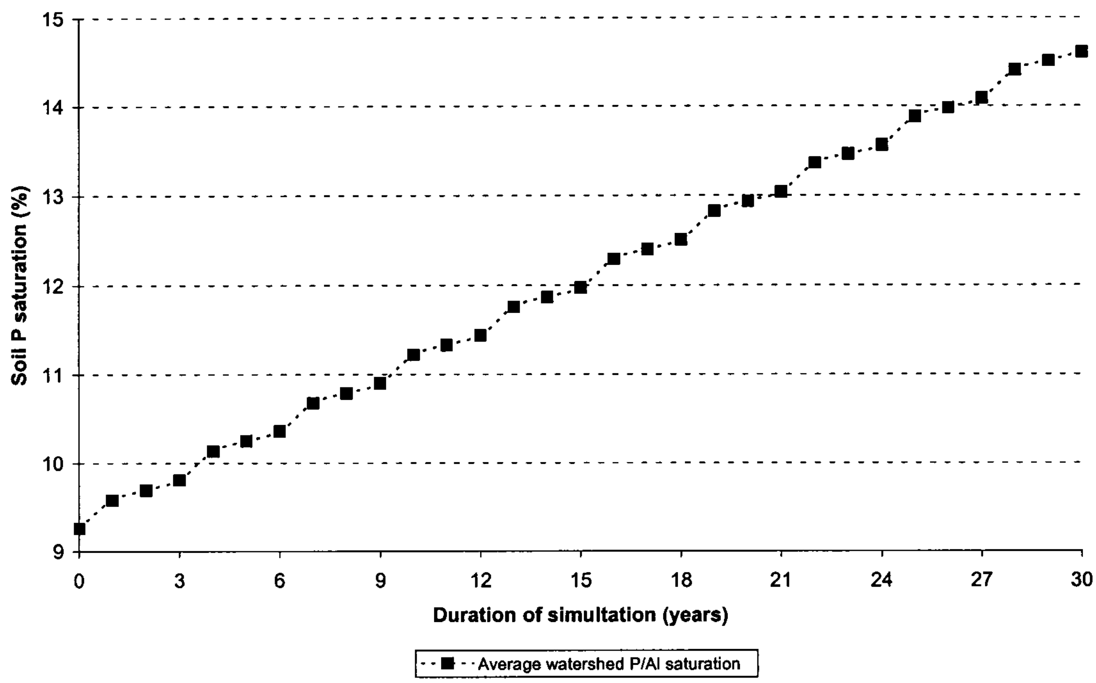

Figure 4-3: Predicted variation in P/Al soil P saturation for the fields within the Castor watershed
(The data represented on the graph are punctual; the dotted line between symbols serves to better illustrate the trend)

The model-predicted increase in year to year P saturation is easier to analyse than the equivalent value for soil test P because the variation range of P saturation is already on a normalized scale. We can then more clearly observe the greater rise which occurs in the first year of the rotation, than those more similar rises occurring in the second and third years of rotation. Consequently, a link can be made between this variation and the three year crop and management practices rotation. It is also clear that particular types of rotations, because of their inherent differences in fertilizer inputs, would have distinctly different impacts on soil test P values, and by extension on increases in P soil saturation.

Table 4-5: Mean model-predicted P saturation for the Castor watershed

Watershed P-saturation (%)		Watershed mean rise in P-saturation (% yr^{-1})
Year 0	Year 30	
9	15	0.18

The total model-predicted increase in soil test P (Table 4-4) is equivalent to 111 kg Mehlich-III P ha^{-1}, giving rise to a 6% increase in P saturation. Therefore, it would take 5 years for the soil P saturation to rise by 1%. This increase in kg Mehlich-III P ha^{-1} corresponds to an input of 178 kg P_2O_5 ha^{-1} in excess of crop requirements.

The environmental threshold for soil P saturation varies according to the clay content of the soil (Table 2-3; Khiari et al., 2000; Pellerin, 2003). The first column of Table 4-6 expresses the number of times the soil saturation value was surpassed according to its soil type. A value of 1 means that the soil saturation value has reached the environmental threshold value for this type of soil. Based on the data available for the Castor watershed, currently (year 0) 24% of the fields corresponding to 32% of the area have soil P saturation value over their environmental thresholds. After the 30 years of simulation, 39% of the fields corresponding to 44% of the watershed area would surpass their environmental threshold. This data is significant because there is no agronomic or economic advantage to bring the soil saturation over the threshold (Tables 2-3 & 2-4).

Table 4-6 : Percent of fields and of watershed area exceeding Environmental Thresholds in soil P saturation at years 0 and 30

No. times P sat. > Environ. Threshold	Year 0		Year 30	
	% fields	% area	%fields	% area
0 – 1	76	68	61	61
1 - 2	21	29	24	27
2 - 3	2	3	7	8
3 – 4	0.75	0.11	3	4
4 - 5	-	-	0.75	0.5
5 - 10	-	-	3	3
10 +	-	-	1	0.8

After 30 years, according to the simulations, 15% of fields or 17% of the watershed area would have soil P saturation levels at least twice their environmental threshold. By maintaining current field management practices, the number of fields and percent of watershed area would rise, respectively, by 5- and 6-fold respectively, over the 30 years.

4.1.4 TP export

TP export was computed by summing TP in surface runoff and that in tile drain flow. The precipitation pattern plays an important role in the model's calculations of TP export. The C-WEND model simulations over the 30-year period indicate that predicted TP exports closely parallel precipitations patterns (Figure 4-4). However, the model does not address the variation in TP export within a year, under precipitation events of different intensities and durations.

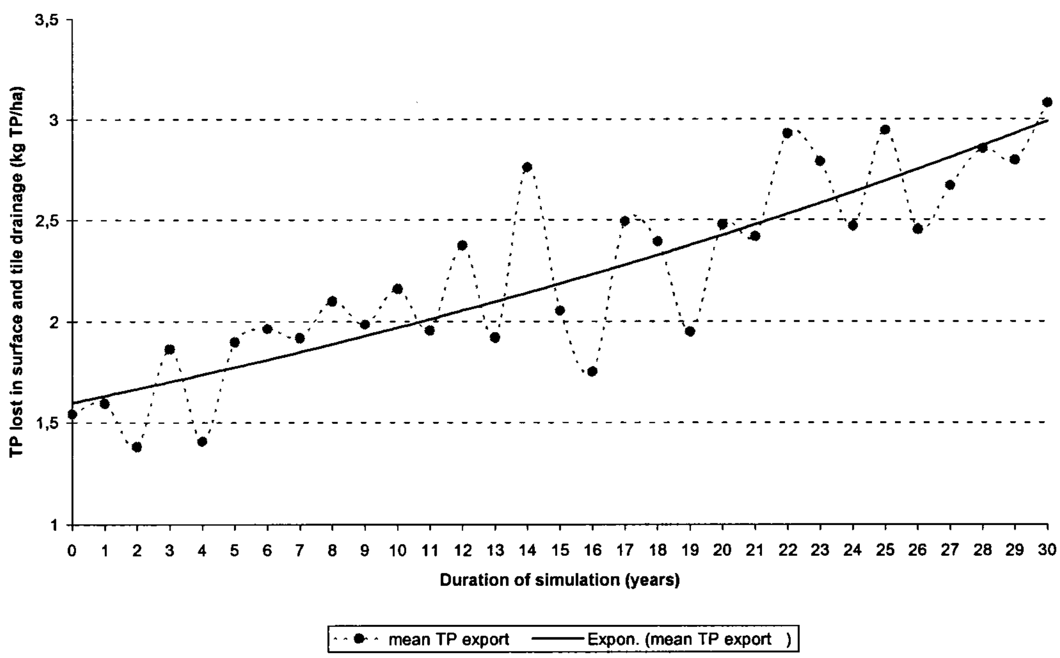

Figure 4-4: Predicted TP export from Castor watershed agricultural fields (The data represented on the graph are punctual; the dotted line between symbols serves to better illustrate the trend.)

Over 30 years, a slight increase in TP export values can be observed in the trendline. It is due to the increase in soil test P. Therefore, the factors which influence variations in soil test P also have an indirect effect on TP exports. Over-application of P probably has a greater influence on TP exports than either crop rotation or other management practices. However, these do have an effect on soil loss and soil test P, and therefore also influence TP export to some degree.

Table 4-7: Watershed TP export values

Model-predicted values							Measured values		
TP exports (kg ha^{-1} yr^{-1})						Mean increase (% yr^{-1})	TP exports (kg ha^{-1} yr^{-1})		
Mean	Min	Max	Years				Years		
			1997	1998	1999		1997	1998	1999
2.24	1.38	3.08	1.54	1.59	1.38	2.31	1.2	1.3	2.13

Model-predicted values of TP exports for the period of 1997-1999 were higher than values measured at the river outlet for the first two years and less for the third year (Table 4-7). There are a number of elements to consider when comparing data series produced by a model and watershed monitoring data. An important climatic event occurred in the winter of 1998, affecting the monitoring apparatus and consequently the data it collected. Also, the C-WEND model simulates edge of field loads and does not consider sedimentation in the watercourse network. In future development of the model, the addition of an enrichment ratio calculation could help resolve this situation. This ratio refers to the modification, due to the preferential transport of fine soil particles and the preferential deposition of larger ones, of the soil particle size distribution compared to what it was before erosion occurred. It is also difficult to compare the measured values with the model values, knowing the influence of precipitation pattern, because the relationship between loads and precipitation intensity pattern is not expressed in the model, but definitely experienced on the fields. In addition, the C-WEND model was expected to produce better long-term than short-term predictions.

4.2 Cropping system analysis

In the following sections, the results will be analysed at the field scale according to the type of crop cultivated with current practices. Averages have been made by computing the values obtained for each field with regard to 6 different cropping systems. Soil Mehlich-III P, soil P saturation with aluminium, potential soil loss values calculated with RUSLE and total export of P are presented. In Table 4-8, the most common cropping systems selected are identified as well as the percentage of area they cover in the watershed.

Table 4-8: Cropping system percentage in the Castor watershed

Cropping system	% AREA
Continuous corn	33.1
Continuous grassland	5.1
Pasture	19.5
Rotation with corn	26.9
Rotation without corn	11.5
Rotation with soybeans	3.9

The cropping systems are:

1. Continuous corn: Corn is the only crop cultivated during the entire simulation period. The mean initial soil Mehlich-III P 249 kg ha^{-1} and a soil P saturation of 11%. Conventional tillage constitutes the most common management practices but conservation tillage practices are applied on some fields at one point in their rotation. Mean manure and commercial fertilizer inputs were 17 kg P_2O_5 ha^{-1} and 61 kg P_2O_5 ha^{-1}, respectively.
2. Continuous Grassland: Fields support grassland and grassland combined with alfalfa. The rotation does not include other crops. The initial soil Mehlich-III P for this cropping system is 150 kg ha^{-1} with a soil P saturation level of 7%. There is no tillage done on these fields. Mean manure and commercial fertilizer inputs were 7 kg P_2O_5 ha^{-1} and 15 kg P_2O_5 ha^{-1}, respectively.

3. Pasture: For this cropping system only pasture areas are considered. The initial soil Mehlich-III P is 224 kg ha^{-1} with a soil P saturation level of 10%. No tillage is applied to these fields. Mean manure and commercial fertilizer inputs were 4 kg P_2O_5 ha^{-1} and 6 kg P_2O_5 ha^{-1}, respectively, ignoring herd contribution.
4. Crop Rotation with Corn: This cropping system includes at least one year of grain or silage corn. The other crops that are part of the rotation can be cereals, grassland and grassland with alfalfa. The initial soil Mehlich-III P for this type of field is 178 kg ha^{-1}, with a soil P saturation level of 8%. Two management practices are applied on the fields, no-till or conventional tillage, the latter being more spatially and temporally prevalent. Mean manure and commercial fertilizer inputs were 42 kg P_2O_5 ha^{-1} and 48 kg P_2O_5 ha^{-1}, respectively.
5. Crop Rotation with soybeans: This cropping system regroups all the fields with at least one year of soybeans. The second most common crop in the rotation is corn, but occasionally cereals are also present. The initial soil Mehlich-III P for this type of field is 148 kg ha^{-1} with a soil P saturation level of 6%. Three types of tillage practices, conventional, conservation and no-till, are employed in this cropping system. Mean manure and commercial fertilizer inputs were 17 kg P_2O_5 ha^{-1} and 29 kg P_2O_5 ha^{-1}, respectively.
6. Crop Rotation without Corn: This cropping system is composed of pasture, cereal, grassland or grassland with alfalfa and excludes corn cropping. The initial soil Mehlich-III P for this type of field is 193 kg ha^{-1} with a soil P saturation level of 10%. The management practices employed are no-till and conventional tillage, the former being the most common. Mean manure and commercial fertilizer inputs were 84 kg P_2O_5/ha and 49 kg P_2O_5/ha, respectively.

4.2.1 Soil loss potential

Soil loss under the different cropping systems was similar because inputs values were the same for the 3-year rotation period (Figure 4-5). "Continuous corn", "continuous grassland" and "pasture" fields showed negligible variation of their soil loss potential during the three year rotation pattern because there was only a small variation in their cropping system. Comparatively, "crop rotation with corn", "crop rotation without corn" and "crop rotation with soybeans" showed a more significant variation during their 3-year rotations because of the changing combinations of their crop types and management practices. Years with the greatest presence of forages showed the lowest soil loss potential. Most of the soil characteristics for the croplands of the Castor watershed, when separated by cropping systems, are quite similar (Table 4-9). The LS factor (≈ 0.2) is low, meaning that the average fields are relatively flat. Similarly, their clay contents are similar, ranging from 23% to 33%. The mean K factor and bulk density values show similar tendencies: soil has a medium level of compaction and a relatively high rate of water infiltration. All these characteristics help to reduce the risk of erosion and result in potential soil loss being more likely to vary on a field basis than by cropping system.

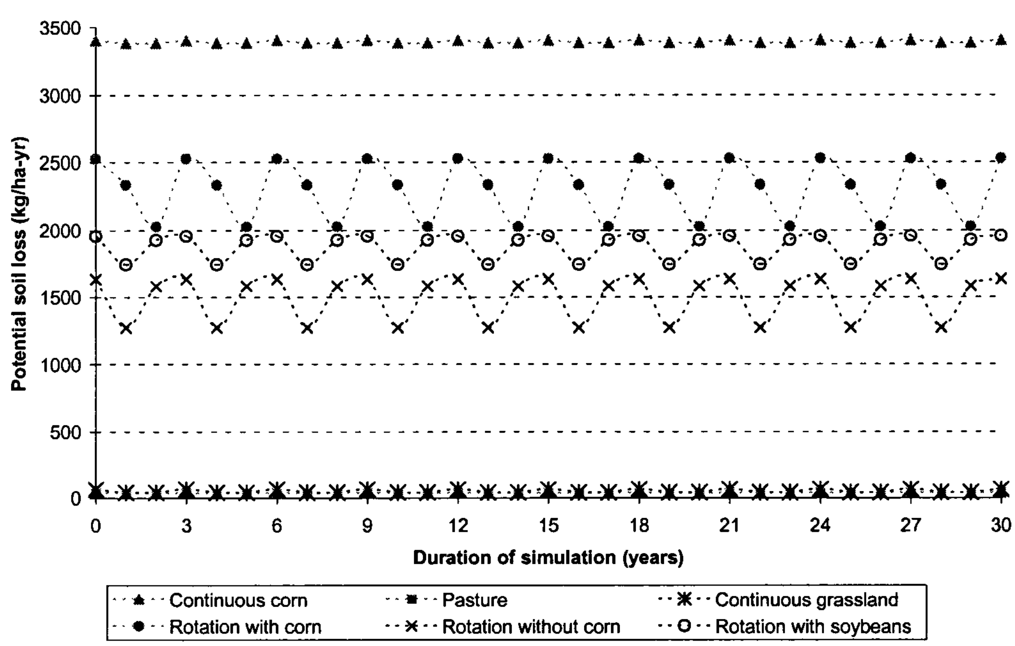

Figure 4-5: Mean predicted variation in soil loss in the Castor watershed by cropping systems (The data represented on the graph are punctual; the dotted line between symbols serves to better indicate trends for each cropping system.)

The "continuous corn" showed the greatest soil loss potential, with values ranging between 3383 kg ha^{-1} yr^{-1} and 3400 kg ha^{-1} yr^{-1}. The 170 kg ha^{-1} yr^{-1} fluctuations in soil loss are attributable to the different management practices applied on the fields over time. Conservation management practices lowered soil loss potential values, whereas corn crop under conventional tillage resulted in a higher C factor value (0.37), which contributed to greater erosion. The soil type also has an influence on the quantity of soil loss from a field (Table 4-9). The mean LS factor of 0.21 represented a relatively flat landscape. The considerable presence of clay (30%), the intermediate K factor and bulk density values also helped to reduce the risk of erosion in fields continuously cropped to corn.

The "continuous grassland" fields exhibited a soil loss potential between 39 kg ha^{-1} yr^{-1} and 63 kg ha^{-1} yr^{-1}. The 24 kg ha^{-1} yr^{-1} fluctuation in soil loss is attributable to the rotation between grassland and grassland with alfalfa; the latter having a higher C factor than grassland alone. The relatively flat landscape was represented by a mean LS factor of 0.21. The typical soil type for continuous grassland had a mean clay content of 33%, an intermediate K factor and bulk density values which also helped to reduce the risk of erosion (Table 4-9).

As there was no harvest and tillage done on those fields, the pasture fields also exhibited a low soil loss potential: 36 kg ha^{-1} yr^{-1}. It remained constant because the crop and the management practices remained constant. The mean clay content of this soil type (23%) was lower that under other cropping systems, but would not have had an important effect on erosion (Table 4-9).

Table 4-9 : Factors influencing soil loss potential from a field according to cropping system

	Soil and RUSLE parameters			
CROPPING SYSTEMS	LS factor	K Factor Mg ha hr MJ^{-1} mm^{-1} ha^{-1}	Bulk density Mg m^{-3}	Clay content %
Continuous corn	0.21	0.0270	1.46	30
Pasture	0.21	0.0273	1.48	23
Continuous grassland	0.21	0.0264	1.45	33
Crop rotation with corn	0.18	0.0274	1.45	28
Crop rotation without corn	0.22	0.0279	1.47	23
Crop rotation with soybeans	0.16	0.0230	1.42	28

The "crop rotation with corn" cropping system showed a high soil loss potential with values ranging between 2025 kg ha^{-1} yr^{-1} and 2528 kg ha-1 yr^{-1} of soil loss. The higher soil loss values were associated with conventional management practices in corn fields (Table 4-10). Soil loss potential went down as the proportion of corn with conventional tillage decrease. An increase in the area of cereal crops under conservation tillage during the rotation helped to decrease soil loss potential as well as the other pertinent soil characteristics (Table 4-9).

Table 4-10: Percent of area by crop for rotation with corn

Cropping system	Percent of area		
	Year 1	Year 2	Year 3
Corn	66	43	37
Cereal	4	32	27
Silage corn	8	9	11
Grassland with alfalfa	2	0	0
Grassland	21	15	23

For the "crop rotation with soybeans" values ranges between 1744 kg ha^{-1} yr^{-1} and 1956 kg ha^{-1} yr^{-1} of soil loss. In this rotation, corn is always grown under conventional tillage which makes the soil loss potential greater. As for the soybeans and cereal crops, they are grown under conservation and no-till management practices, bringing the soil loss potential values down. As presented in Table 4-11, the area under corn is greater for the first year of rotation, which influences the soil loss potential values (Figure 4-5). On the other hand, the lower LS and K factors help to reduce the risk of erosion from those fields.

Table 4-11: Percent of area by crop for rotation with soybeans

Cropping system	Percent of area		
	Year 1	Year 2	Year 3
Corn	65	52	55
Soybean	35	36	45
Cereal	0	12	0

The "crop rotation without corn" fields showed soil loss potentials ranging from 1273 kg ha^{-1} yr^{-1} to 1632 kg ha^{-1} yr^{-1}. The importance of forages in the rotation and the use of conservation and no-till management practices kept the soil loss potential values to a minimum. The second year of rotation has the lowest soil loss potential value because there were fewer cereals and more grassland in the area (Table 4-12). However, the lower clay content (23%) makes the fields more prone to erosion (Table 4-9).

Table 4-12: Percent of area by crop for rotation without corn

Cropping system	Percent of area		
	Year 1	Year 2	Year 3
Cereal	58	54	82
Silage corn	5	0	5
Grassland with alfalfa	11	11	0
Grassland	17	28	13
Pasture	9	6	0

4.2.2 Soil test P

Model-predicted Mehlich-III P concentrations under continuing current practices are different for each field. However, trends can be observed for the different cropping systems. Of the six cropping systems, four have an increasing progression (rotation without corn, continuous corn, pasture and rotation with corn) and the two a decreasing one (rotation with soybeans and continuous grassland) (Figure 4-6).

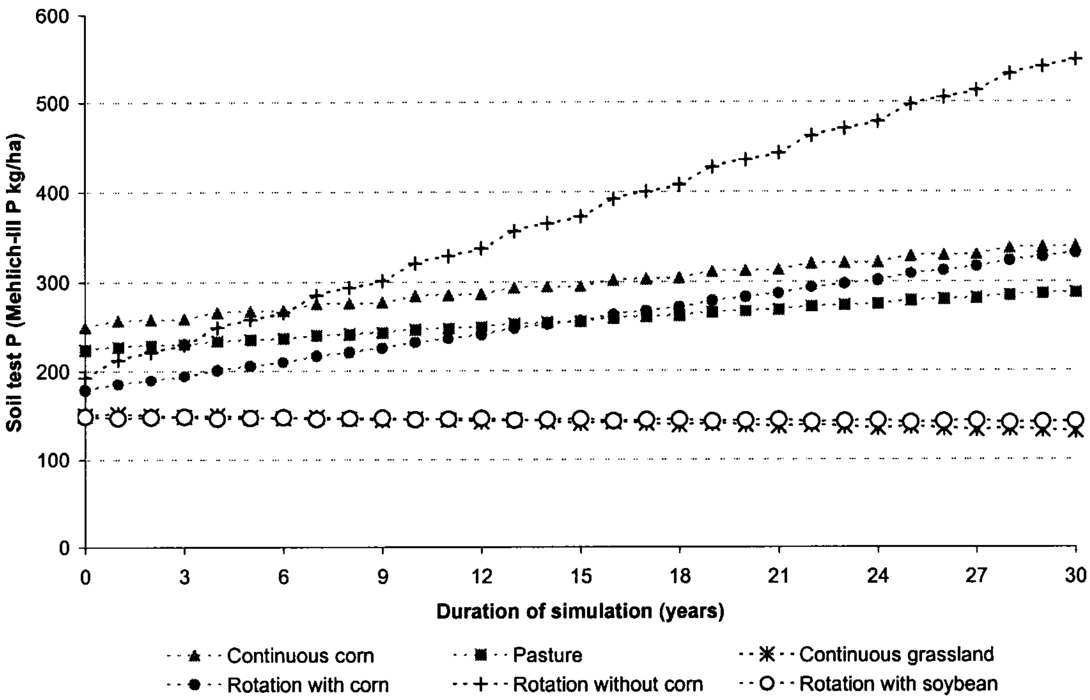

Figure 4-6: Predicted Mehlich-III P of agricultural soils of the Castor watershed by cropping system
(The data represented on the graph are punctual; the dotted line between symbols indicates the trend for each cropping system.)

Table 4-13 presents mean Mehlich-III P concentrations for the different field types. The mean increase in Mehlich-III measured by Michaud et al. (2002b) for the period of 1997-1999 was 2.9 kg Mehlich-III-P ha^{-1} yr^{-1} and the mean predicted increase in Mehlich-III was 3.71 kg Mehlich-III-P ha^{-1} yr^{-1}.

Table 4-13 : Mehlich-III P values by cropping system

CROPPING SYSTEM	Predicted soil test P (kg Mehlich-III P ha^{-1})				Predicted mean annual increase in soil test P
	Year 0	Year 30	Change	Mean	kg Mehlich-III P ha^{-1} yr^{-1}
Continuous corn	249	339	91	297	3.01
Pasture	224	288	64	256	2.14
Continuous grassland	150	131	-19	141	-0.63
Crop rotation with corn	178	332	153	256	5.12
Crop rotation without corn	193	547	354	375	11.81
Crop rotation with soybeans	148	142	-6	144	-0.21

The mean annual increase for "continuous corn" was 3.01 kg Mehlich-III P ha^{-1} yr^{-1}, corresponding to a P input which exceeded plant requirements by 26 kg P_2O_5 ha^{-1} yr^{-1}. This annual increase for corn fields seems low for this type of crop; it may be a result of the high soil loss potential generating greater TP exports. This trend is clear in Table 4-14, where the annual mean mass balance (mean inputs - mean outputs) gives 30 kg P_2O_5 ha^{-1} yr^{-1} and the annual mean increase predicted by the model is 26 kg P_2O_5 ha^{-1} yr^{-1}. The difference of 4 kg P_2O_5 ha^{-1} yr^{-1} (1.75 kg TP ha^{-1}) is loss in export.

Table 4-14 : Mass balance for a continuous corn cropping system

Parameter	Forms of phosphorus		
	kg TP ha^{-1} yr^{-1}	kg Mehlich-III P ha^{-1} yr^{-1}	kg P_2O_5 ha^{-1} yr^{-1}
Predicted mean annual increase	11	**3.01**	26
Plant requirements (corn)	**21**	6	48
Mean annual input (fertiliser)	34	9	**78**
Mean inputs – mean outputs	13	3	30

The mean increase for "pasture fields" was 2.14 kg Mehlich-III P ha^{-1} yr^{-1}, which represents a total increase of 64 kg Mehlich-III P ha^{-1} over 30 years. This enrichment is possible because of the low potential soil loss and also because the crop is not harvested, leaving P inputs in the field (Table 4-15). The input and output due to the presence of herd on the pasture is not evaluated in this mass balance, but it likely would contribute a further phosphorus input.

Table 4-15 : Mass balance for a pasture cropping system

Parameter	Forms of phosphorus		
	kg TP ha^{-1} yr^{-1}	kg Mehlich-III P ha^{-1} yr^{-1}	kg P_2O_5 ha^{-1} yr^{-1}
Predicted mean annual increase	8	**2.14**	18
Plant requirements (pasture)	**0**	0	0
Mean annual input (fertiliser)	4.4	1.2	**10**
Mean inputs – mean outputs	4.4	1.2	10

The "continuous grassland fields" showed a decreasing soil P concentration value of 19 kg Mehlich-III P ha^{-1} over the 30 year simulation period. This decrease for "continuous grassland fields" is a result of the low inputs of manure and commercial fertilizer and the yearly harvest of hay (Table 4-16). The outputs being greater than the inputs resulted in a decrease in soil P concentration. The small difference between mean annual mass balance (mean inputs - mean outputs) and predicted mean annual increase confirm that there are small quantities of P lost in export from continuous grassland fields.

Table 4-16 : Mass balance for a continuous grassland cropping system

Parameter	Forms of phosphorus		
	kg TP ha^{-1} yr^{-1}	kg Mehlich-III P ha^{-1} yr^{-1}	kg P_2O_5 ha^{-1} yr^{-1}
Predicted mean annual increase	-2.4	**-0.63**	-5.5
Plant requirements (grassland)	**12**	3.2	28
Mean annual input (fertiliser)	10	2.6	**22**
Mean inputs – mean outputs	2	-0.6	-6

The mean increase in soil test P under "crop rotation with corn fields" was 5.12 kg Mehlich-III P $ha^{-1}yr^{-1}$, which results in a yearly fertilizer input of 44 kg P_2O_5 ha^{-1} yr^{-1}. The inputs on such fields are twice the mean plant requirements for this rotation (Table 4-17). This increase for "crop rotation with corn fields" is closest, compared to other cropping systems, to the increase in Mehlich-III P measured for the watershed by Michaud et al. (2002a, 2002b, 2003). It is thus a good representation of what goes on in the watershed over time. The significant P inputs for corn and cereals promote a rise in soil P concentration in the fields. The difference between the mean annual mass balance (mean inputs - mean outputs) and predicted mean annual soil test P was 1 kg TP ha^{-1} yr^{-1}, loss in export.

Table 4-17 : Mass balance for crop rotation with corn cropping system

Parameter	Forms of phosphorus		
	kg TP ha^{-1} yr^{-1}	kg Mehlich-III P ha^{-1} yr^{-1}	kg P_2O_5 ha^{-1} yr^{-1}
Predicted mean annual increase	19	**5.12**	44
Plant requirements (corn rotation)	**19**	5	44
Mean annual input (fertiliser)	39	10.5	**90**
Mean inputs – mean outputs	20	5.5	46

The mean value for "crop rotation without corn fields" was 375 kg Mehlich-III P ha^{-1}. This cropping system underwent the most intensive increase in the watershed (Figure 4-6). The mean increase was 12 kg Mehlich-III P ha^{-1} yr^{-1}, resulting in a total increase of 354 kg Mehlich-III P ha^{-1} over 30 years (Table 4-18). This increase in phosphorus is attributable to the substantial P inputs in cereal crops, the management practices and the lower plant requirements, all contributing to the greater retention of P in fields. The fertilizer application were roughly 4.5-fold the mean plant requirements. This could be explained by the greater number of fertilizer applications on cereal crops because of its height.

Table 4-18 : Mass balance for crop rotation without corn cropping system

Parameter	Forms of phosphorus		
	kg TP ha^{-1} yr^{-1}	kg Mehlich-III P ha^{-1} yr^{-1}	kg P_2O_5 ha^{-1} yr^{-1}
Predicted mean annual increase	44	**12**	102
Plant requirements (rotation without corn)	**13**	4	30
Mean annual input (fertiliser)	58	15	**133**
Mean inputs – mean outputs	45	11	103

The mean value of soil test P for "crop rotation with soybeans" was 144 kg Mehlich-III P ha^{-1} (Table 4-19). This crop rotation showed a mean decrease of 0.21 kg Mehlich-III P ha^{-1} yr^{-1}, which results in a total decrease of 6 kg Mehlich-III P ha^{-1} over 30 years. This decrease for "crop rotation with soybeans" is a result of P inputs slightly lesser than plant requirements. However, a difference of this small a magnitude is not an agronomic threat for the crops.

Table 4-19 : Mass balance for crop rotation with soybeans cropping system

Parameter	Forms of phosphorus		
	kg TP ha^{-1} yr^{-1}	kg Mehlich-III P ha^{-1} yr^{-1}	kg P_2O_5 ha^{-1} yr^{-1}
Predicted mean annual increase	-0.79	**-0.21**	-1.8
Plant requirements (rotation with soybeans)	**21**	6	48
Mean annual input (fertiliser)	20	5.4	**46**
Mean inputs – mean outputs	-1	-0.6	-2

4.2.3 P saturation

The trends followed by soil P saturation (Figure 4-7) according to cropping systems are analogous to the trends in Mehlich-III P (Figure 4-6). This was expected since soil Mehlich-III P values are required to calculate soil P saturation. Based on the type of crop and the management practices, the soil saturation level increased or decreased differently. Like Mehlich-III P trends, of the six cropping systems, four exhibited an increasing trend (rotation without corn, continuous corn, continuous pasture and rotation with corn) and two a decreasing one (rotation with soybeans and continuous grassland) (Figure 4-7).

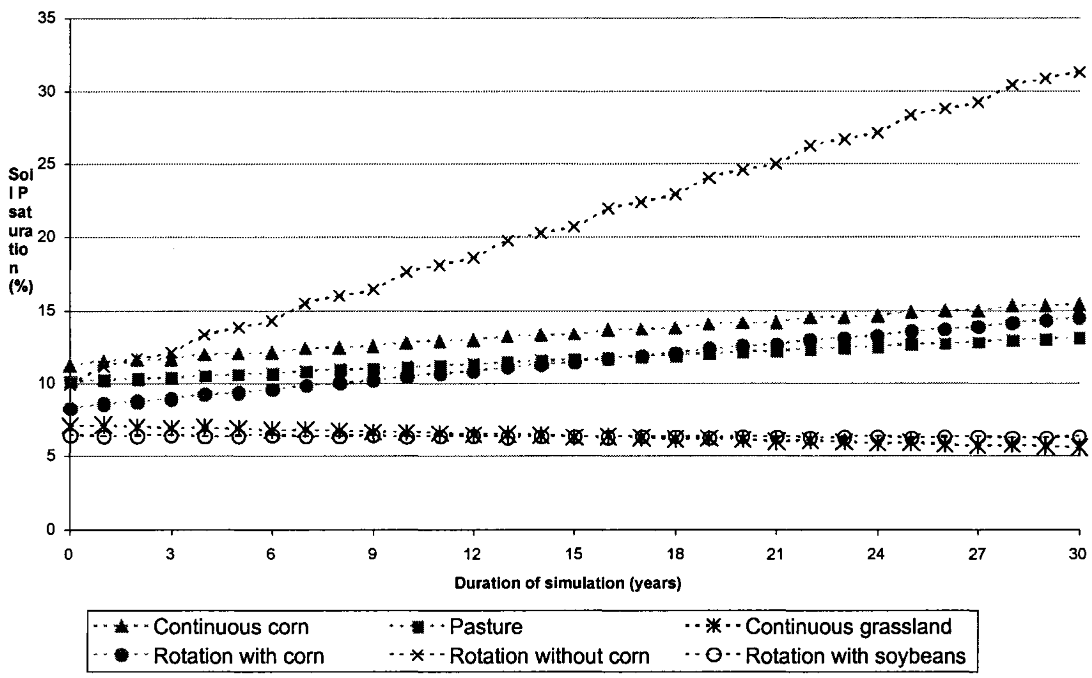

Figure 4-7: Comparison of predicted P/Al saturation levels under different cropping systems (The data represented on the graph are punctual; the dotted line between symbols better illustrates trends)

As with soil test Mehlich-III P, the greatest and most remarkable rise occurred under "crop rotation without corn" (Figure 4-6). The mean value for "crop rotation without corn" soil P saturation level is 21% (Table 4-20), with a mean annual increase of about 0.71%. Like for soil P concentration, the substantial increase is a result of the considerable amount of manure and commercial fertilizer applied on cereal crops as well as the management practices sustaining the presence of P in fields.

Table 4-20 : Soil P saturation values by cropping system

CROPPING SYSTEMS	Year 0	Year 30	Total variation over 30 years	Average annual increase model values
	%	%	%	%
Continuous corn	11.2	15.4	4.2	0.14
Pasture	10.1	13.1	3	0.1
Continuous grassland	7.1	5.6	-1.5	-0.05
Crop rotation with corn	8.3	14.5	6.2	0.21
Crop rotation without corn	9.9	31.3	21.4	0.71
Crop rotation with soybeans	6.5	6.3	-0.2	0

The "crop rotation with corn" increased soil P saturation, though less dramatically than did the "crop rotation without corn". The total change under "crop rotation with corn" was 6.2%, with a mean annual increase of about 0.21%. As with soil test P, the increase was a result of significant P inputs to the field.

"Continuous corn" cropping also resulted in an increase in soil P saturation: 4.2% over 30 years, representing an annual mean increase of 0.14%. As with soil test P, the small increase was a consequence of high TP exports causes by high soil loss potential.

Soil saturation in pasture fields rose 3% over the simulation period. As with soil test P, the minimal P output resulted in a soil test P increase over time.

The "crop rotation with soybeans" showed a negligible change (-0.2%) in soil P saturation over the 30 year simulation period. As with soil test P, the decrease was mainly due to manure and commercial fertilizer inputs being lesser than plant requirements. The significant soil loss potential due to the inclusion of corn fields and conventional management practices also contributed to the decrease in soil P saturation.

The "continuous grassland" cropping system also exhibited a negligible change (-0.05%) over the 30-year simulation period. Plant requirements exceeded fertiliser inputs, resulting in a small decrease in soil P saturation over time.

4.2.4 TP export

The TP export by cropping system represents the quantity of P exported from every field. It is apparent from Figure 4-8, The "continuous corn," "crop rotation with corn" and "crop rotation without corn" were clearly exporting the greatest quantities of P, and at increasing rates. For the other cropping systems ("pasture," "continuous grassland" and "crop rotation with soybeans") exports were lower and more constant over the long term.

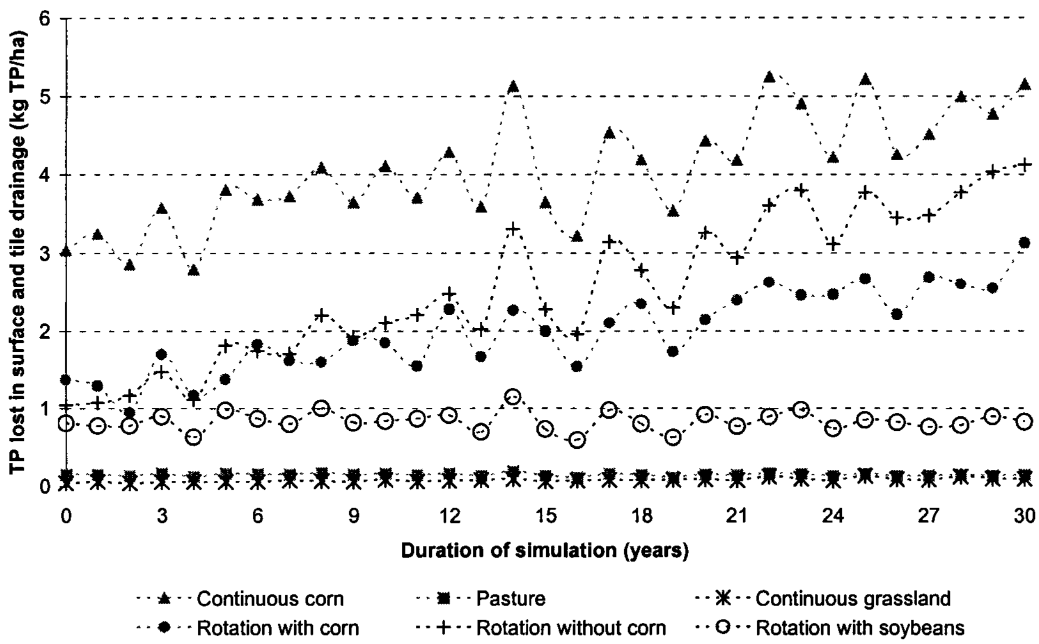

Figure 4-8: Comparison of predicted TP exports under different cropping systems (The data represented on the graph are punctual; the dotted line between symbols serves to better illustrate the trend for each cropping system.)

For fields under "continuous corn," the TP export values ranged from 2.8 kg TP ha^{-1} yr^{-1} and 5.2 kg TP ha^{-1} yr^{-1} (Figure 4-8). The mean TP export from corn fields was 4.1 kg TP ha^{-1} yr^{-1}. These high TP values were a result of significant potential soil loss combined with high soil test P. As expected, the TP export trend for continuous corn followed the pattern of the precipitation ratio. An increase in TP values with time resulted from increases in soil test P.

As shown in Figure 4-8, The "pasture" and "continuous grassland" fields showed very low TP exports: 0.11-0.19 kg TP ha^{-1} yr^{-1} and 0.04-0.13 kg TP ha^{-1} yr^{-1} , respectively. The mean TP export for pasture was 0.15 kg TP ha^{-1} yr^{-1} and 0.08 kg TP ha^{-1} yr^{-1} for continuous grassland. The low TP export values of those types of fields are mainly attributable to management practices, low P inputs and low soil loss potential. Despite the fact that TP exports for "pasture" and "continuous grassland" were low, they nonetheless followed precipitation patterns.

The "rotation with corn" TP exports ranged between 0.94 kg TP ha^{-1} yr^{-1} and 3.12 kg TP ha^{-1} yr^{-1} (Figure 4-8), with a mean of 2 kg TP ha^{-1} yr^{-1}. The manifest presence of corn fields with conventional management practices in the rotation was mainly responsible for raising TP export from such fields. Again, the TP exports were clearly influenced by the precipitation ratio pattern. In the case of "rotation without corn" the TP export values increased rapidly, ranging between 1.0 kg TP ha^{-1} yr^{-1} and 4.1 kg TP ha^{-1} yr^{-1} with a mean of 2.55 kg TP ha^{-1} yr^{-1}. This significant increase was caused by the considerable excess of P inputs over plant requirements (Table 4-18). These values also exhibited sensitivity to changes in precipitation ratio.

The pattern followed by the "crop rotation with soybeans" was different. The values ranged between 0.6 kg TP ha^{-1} yr^{-1} and 1.1 kg TP ha^{-1} yr^{-1}, with a mean of 0.83 kg TP ha^{-1} yr^{-1}. Fertilizer applications being slightly lower than plant requirements kept TP exports down, even if there was a considerable presence of corn bringing important soil loss potential.

4.3 Alternative management practices for reducing P

The previously discussed model-predicted data showed monocropping corn to be a high risk for P losses. Therefore, alternative field management practices were simulated to observe their effects on P reduction. It also illustrated that corn fields export large quantities of TP and are usually over-fertilised. According to Jamieson et al. (2001), large volumes of water exporting large amounts of sediment from low P concentrations soils contributes more P to the waterways than the converse. Since the development of best management practices (BMPs) for corn fields has been identified as a top priority in Quebec, the following section will outline how model simulations on a representative corn field were used to compare BMPs for corn.

A soil Mehlich-III P content of 200 kg ha^{-1} was entered for the representative corn field, which covered an area of 5 ha on a Ste-Brigide sandy loam. Its aluminium content was is 964 mg kg^{-1} (ppm); the LS and K factor of the RUSLE equation were set at 0.21 and 0.027, respectively. Five different cropping systems were applied to the field (Table 4-21). Three are invariant i.e. continuous corn under an invariant tillage practice (conventional, conservation, no-till). Two represent crop rotations based on the most common farm management practices. Fertiliser applications were based on the application recommendations from MENV regulations (2003) (Table 2-1 and 2-2).

Table 4-21 : Range of management practices applied in the model to a representative Castor watershed corn field

Cropping systems	Crop	Management practices
Conventional	Corn	Conventional
Conservation	Corn	Conservation
No-till	Corn	No-till
	Corn	Conventional
Cash crop rotation	Soybeans	Conservation
	Corn	Conservation
	Cereals	Conservation
	Corn	Conventional
Dairy rotation	Cereal	Conservation
	Grassland with alfalfa	No-till
	Grassland	No-till

4.3.1 Soil loss potential

For invariant crop rotations, no variation in soil loss potential occurred over the simulation period (Figure 4-9). Within these three corn field management regimes, the greatest soil loss potential existed under conventional tillage. Soil loss potentials for conventional, conservation and no-till tillage practices were 3514, 3039 and 1425 kg ha^{-1} yr^{-1}, respectively. The differing tillage practices clearly influenced soil loss potential. The C factor for the RUSLE calculations is greater for fields under conventional practices, resulting in a greater soil loss potential in such fields.

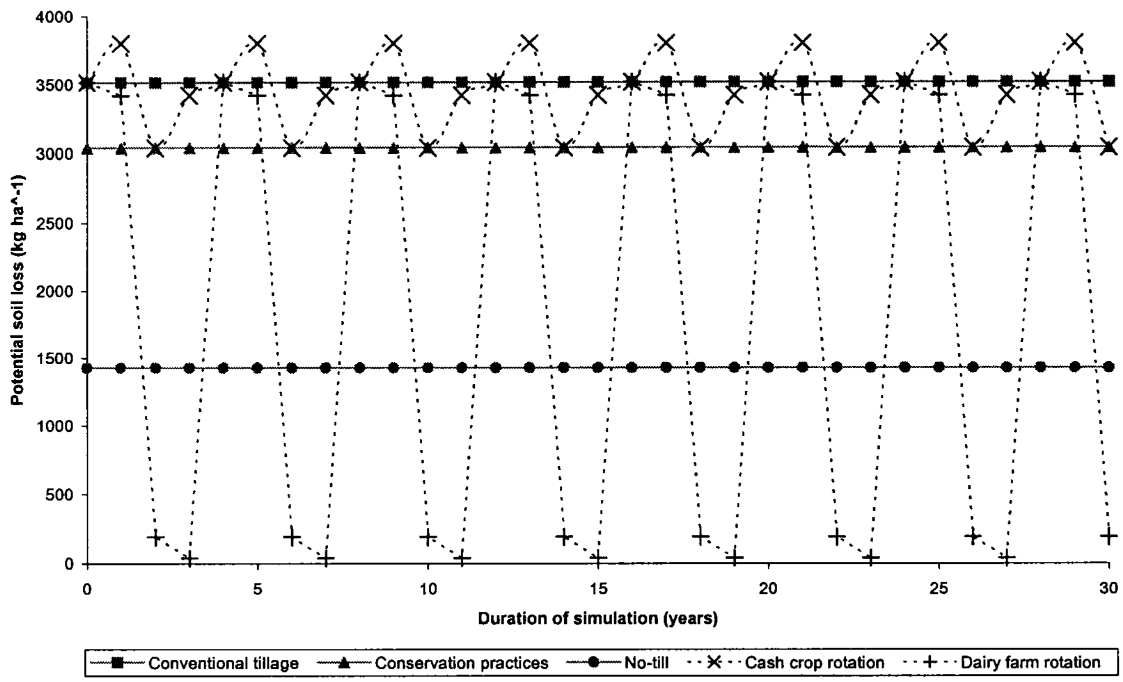

Figure 4-9 : Predicted soil loss potential of corn fields according to management practice (The data represented on the graph are punctual; the dotted line between symbols serves to illustrate the trends for each cropping system.)

The variations in soil loss potential for the cash crop and dairy farm rotations indicate that large variations in soil loss potential can occur, according to the type of crop and management practices applied. Values for the cash crop rotation field were 3039, 3419, 3514 and 3799 kg ha^{-1} yr^{-1} for the different years of rotation. Within the cash crop rotation, the least soil loss potential occurs for the corn field under conservation tillage, because the C factor of the RUSLE equation is lower. The greatest soil loss potential occurs for the soybean crop under conservation tillage; again it is the C factor that determines the variation in the soil loss potential.

As for the dairy farm rotation, values of soil P loss potential were 38, 190, 3419 and 3514 kg ha^{-1} yr^{-1} for the different years of rotation. The lowest soil loss potentials were attributed to grassland and grassland with alfalfa, in part because no work was done on such fields. The greatest potentials occurred for corn and cereal cultivation. It is evident that the best way to reduce soil loss potential in a rotation is to introduce no-till grasslands. In a four-year rotation, just one year of grassland can reduce soil loss by about 3000 kg ha^{-1}, whereas over a 30- year period it can retain up to 21 000 kg ha^{-1} of P laden soil in the field.

4.3.2 Soil test P

For all cropping systems, soil test P and soil P saturation rose under the basal P fertilization rate, until they exceeded critical levels of 251 kg Mehlich-III-P ha^{-1} soil test P and/or 10% soil P saturation suggested in MENV regulations (2003), at which point P fertilizer inputs were lowered (Figure 4-10 and 4-11).

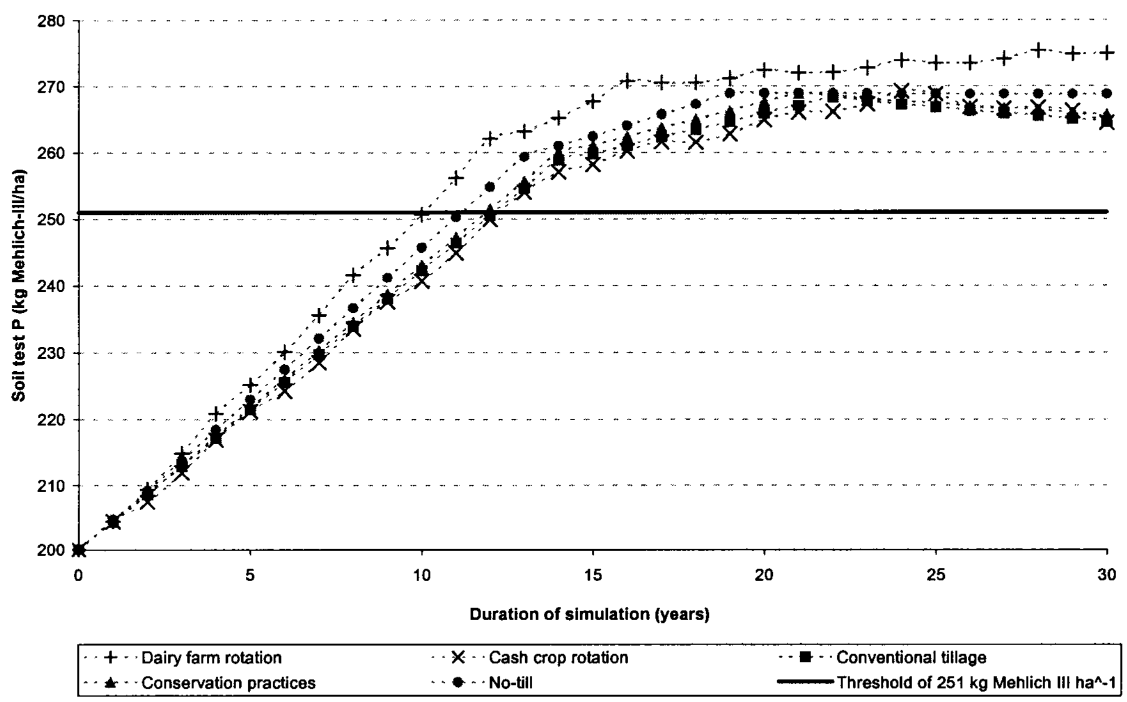

Figure 4-10 : Predicted soil test P of corn fields according to management practice (The data represented on the graph are punctual; the dotted line between symbols serves to illustrate the trends for each cropping system.)

Over 30 years of simulation, the mean values of soil test P for continuous corn with conventional, conservation and no-till practices were, 248, 249 and 250 kg Mehlich-III-P ha^{-1} yr^{-1}, respectively. The mean increase in soil test P for each of these cropping systems was 2.15, 2.19 and 2.29 kg Mehlich-III-P ha^{-1} yr^{-1}, respectively.

The inputs are the same for continuous corn with conventional and conservation tillage, but because fields under continuous corn under conservation and no-till practices reach the threshold of 251 kg Mehlich-III-P ha^{-1} yr^{-1} and 10% soil saturation earlier than corn under conventional tillage, the fertiliser inputs are reduced earlier (Figure 4-10 and 4-11: between year 10 and year 15). The variation in the pattern on Figure 4-10 and 4-11, show the onsets of changes in fertiliser inputs (Table 4-22). With all the other conditions kept constant, only the soil loss potential would influence the soil test P of each field. Corn fields under conventional management practices are subject to greater erosion than corn fields under conservation tillage or no-till practices. A quantity of phosphorus travels with the soil lost from the more tilled fields, which, over time, bring the surface soil P content down faster than on fields where other less intensive tillage practices are applied. This explains why the soil test P and field saturation threshold are reached earlier in the continuous corn with no-till management and also why, after thirty years of simulation, the soil test P values are higher for no-till fields, followed by fields with conservation practices and conventionally tilled fields.

Table 4-22 : Variation in fertiliser input for the different cropping system of representative Castor watershed corn fields

Cropping systems	First modification Years	Second modification Years
Conventional	14	22
Conservation	14	21
No-till	13	19
Cash crop rotation	14	24
Dairy rotation	12	16

After 30 years of simulation the mean soil test P values for to the cash crop and dairy farm rotation cropping systems were 247 kg Mehlich-III-P ha^{-1} yr^{-1} and 254 kg Mehlich-III-P ha^{-1} yr^{-1}, respectively. The mean increase for these management systems were 2.15 kg Mehlich-III-P ha^{-1} yr^{-1} and 2.5 kg Mehlich-III-P ha^{-1} yr^{-1}, respectively. The presence of other crops and different management practices in the four-year rotation is detectable in the graph (Figure 4-10). It is not manifest in the first five years, but over time the dairy farm rotation shows a greater increase in soil test P than does the cash crop rotation. The return of a corn crop every two years under the cash crop rotation seems to makes this cropping system more sensitive to erosion, which has the effect of removing some phosphorus from the field. Under the dairy farm rotation, the same conditions occur, but with only one year of corn so the effect is less apparent.

4.3.3 P saturation

The trends followed by soil P saturation are similar (Figure 4-10 and 4-11) to the trends in soil test P values. According to Pellerin (2003) using soil P saturation to produce agronomic recommendations is safer than soil test P because it is less prone to spatial variability and aluminium levels are quite stable. As with soil test P, all cropping systems showed an increasing saturation at first, followed by a more or less slow decline after the reduction of fertiliser inputs. The field soil P saturation value at the beginning of the simulation (T=0) were identical (9.36%) for all management practices.

The soil P saturation for continuous corn fields under conventional, conservation, and no tillage practices reached 9.90, 9.94 and 10.06%, respectively, after 30 years, representing absolute increases of 0.54, 0.58 and 0.70% soil P saturation. The fact that the increases in soil P saturation were small is attributable to the modification in fertiliser inputs when threshold soil P and soil P saturation levels were reached. As with soil test P, soil P saturation for fields under no till or conservation tillage were greater than under conventional tillage, again as a result of the greater erosion potential under the latter management practice.

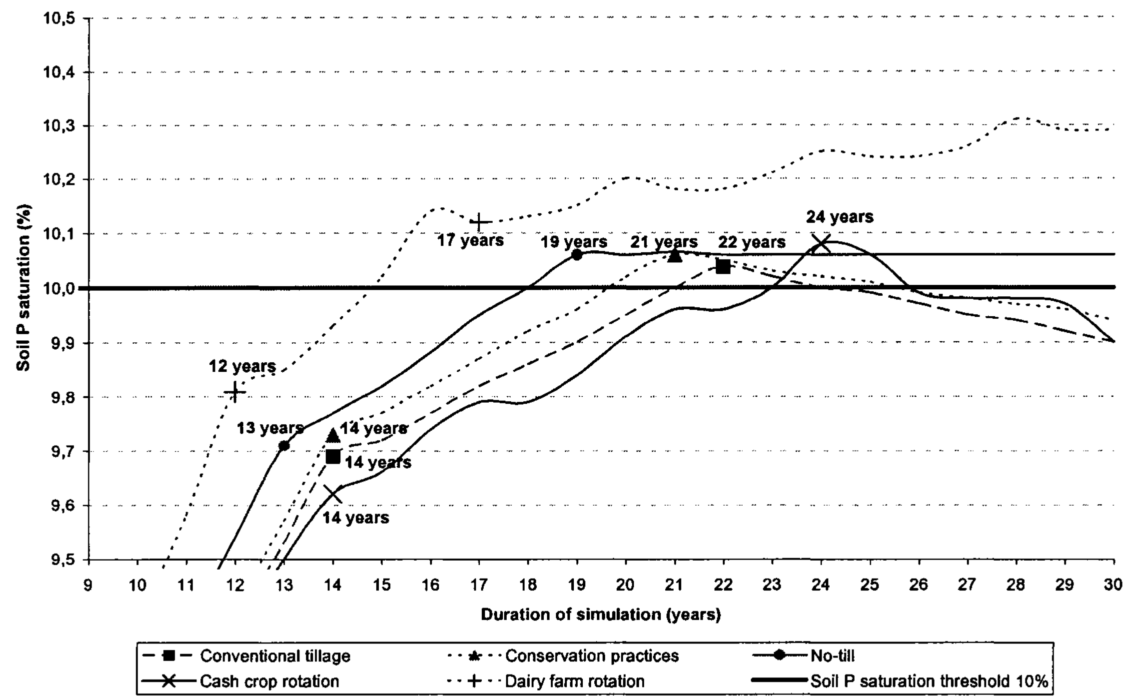

Figure 4-11 : Predicted soil P saturation level of corn fields according to management practice (The data represented on the graph are punctual; the dotted line between symbols serves to illustrate the trends for each cropping system.)

Soil P saturation for cash crop and dairy farm cropping systems both showed increases until fertiliser inputs were modified and, as with soil test P, a four-year cycle in cropping and management practices was detectable (Figure 4-11). The soil P saturation value for the cash crop rotation field reached 9.9% after 30 years, an increase of 0.54%, whereas it reached 10.29% for the dairy farm rotation, an increase of 0.93%. Again, the difference between these cropping systems was likely due to the greater presence of corn in the cash crop rotation, keeping the soil P saturation somewhat lower due to the greater soil loss potential.

4.3.4 TP export

The TP export pattern of each cropping system was a unique combination of soil loss potential (Figure 4-9) and soil test P (Figure 4-10). Continuous corn fields under conventional and conservation tillage showed very similar TP export levels and pattern (Figure 4-12). Corn under conventional management practices led to greater TP export values than corn under conservation management practices, because soil loss potential was greater under conventional tillage practices. Under this type of management practice, values ranged from 1.78 to 3.36 kg TP ha^{-1} yr^{-1}, with a mean of 2.52 kg TP ha^{-1} yr^{-1}. Under conservation management practices TP exports ranged between 1.54 and 2.91 kg TP ha^{-1} yr^{-1}, with a mean value of 2.18 kg TP ha^{-1} yr^{-1}. Under no-till practices, values ranged between 0.72 and 1.37 kg TP ha^{-1} yr^{-1}, with a mean of 1.03 kg TP ha^{-1} yr^{-1}. The lower values under no-till fields are attributable to such a management practice considerably reducing the amount of soil lost from the field and consequently reducing TP exports. With the constant soil loss potential of invariant crop rotations, it is easier to see the impact of the precipitation ratio pattern, which, in the absence of crop and management practice rotations, predominates. However, over time there still remains an increase in TP export as a direct consequence of the increase in soil test P as shown by the trendlines (Figure 4-12).

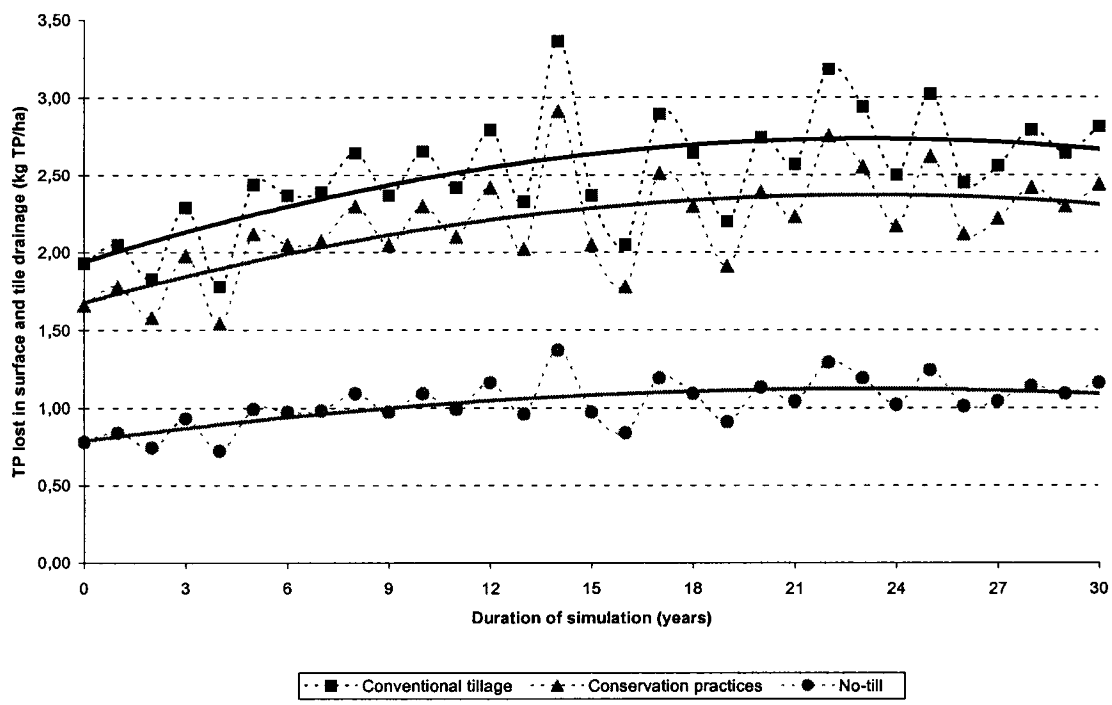

Figure 4-12 : Predicted TP export from corn fields under conventional, conservation or no-till management **(The data represented on the graph are punctual; the dotted line between the symbol is there to better understand the trend of each cropping system.)**

In the case of the evolution of TP export values for cash crop and dairy farm cropping systems, while the influence of precipitation ratio pattern is still present, the impact of the 4-year rotation system is also important, especially in the case of the dairy farm rotation (Figure 4-13). The impact of grassland with alfalfa or simple grassland on soil loss potential and soil test P has a direct consequence on the TP export pattern. The values of TP export for the cash crop rotation range between 1.57 and 3.29 kg TP ha^{-1} yr^{-1}, with a mean of 2.46 kg TP ha^{-1} yr^{-1}. As for the dairy farm rotation, values ranged between 0.02 and 3.04 kg TP ha^{-1} yr^{-1}, with a mean of 1.34 kg TP ha^{-1} yr^{-1}. The years when corn with conventional tillage was implemented, export values under the two rotations reached values seen under corn with conventional tillage.

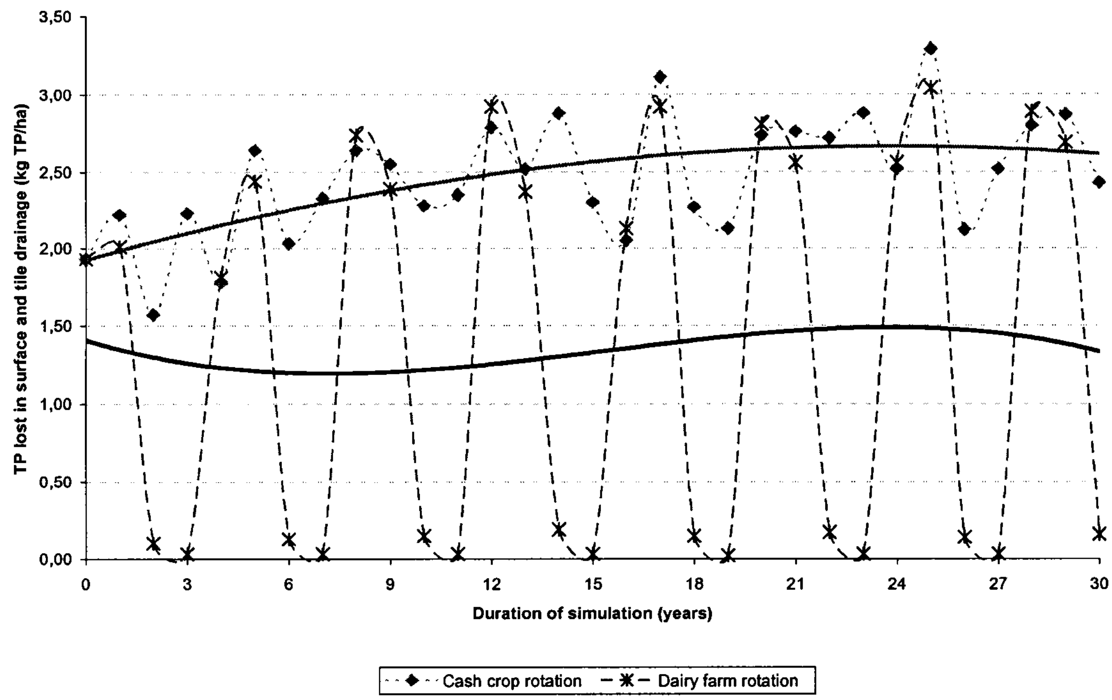

Figure 4-13 : Predicted TP export from corn fields according to cash crop and dairy farm rotations
(The data represented on the graph are punctual; the dotted line between symbols serves to illustrate the trends for each rotation system.)

5 Summary and conclusions

The objective of this study was to apply the WEND (Cassell & Kort, 1998) model to a small agricultural watershed in the southern part of Quebec to evaluate phosphorus movement, storage and export over time.

The model output for the 266 fields were analysed for three different scenarios: overall watershed impacts, cropping systems, and for a field situation showing high risk for P loss. Different outputs were evaluated: soil test Mehlich-III P, soil P saturation with aluminium, RUSLE soil loss potential and TP export. The output data were compared to the monitored data to evaluate the validity of the model.

5.1 Watershed impacts analysis

The input data for the overall watershed impacts showed some characteristics that influenced the simulated data. More than half of the area is planted to corn under conventional tillage practices. In general, fertiliser applications significantly surpass plant requirements. This trend was more significant for the cereal crops. While manure produced in the region provides sufficient phosphorus fertiliser inputs for the fields, large quantities of commercial fertiliser are imported into the basin.

As a result, the soil test P increased at a mean rate of 3.71 kg Mehlich-III P ha^{-1} yr^{-1}, equivalent to a mean input of about 32 kg P_2O_5 ha^{-1} yr^{-1} in excess of plant requirements for the whole watershed, assuming current field management practices remain constant. Over the 30-year simulation period, this corresponds to an input of about 962 kg P_2O_5 ha^{-1} in excess of plant requirements over the entire watershed. This, over time, reduces profits, because excess commercial fertiliser is being purchased. The resulting environmental impacts are going to be severe if agronomic actions are not implemented.

The impact of the 3-year rotation pattern is evident in yearly mean soil P saturation levels; two of the three years showed a similar annual mean increase, but the third year of the rotation showed a greater mean increase. This variation was attributable to the higher fertilizer input for that year in comparison to the others. The behaviour of phosphorus in the soil is different from one type of soil to another and also varies according to the type of crop cultivated. Therefore, a series of environmental thresholds have been identified for different soil types and crops. At the beginning of the simulation, more than 32% of the watershed cropland area surpassed its environmental threshold, and after thirty years the threshold was exceeded by more than 43% of the area.

Because of the preponderance of corn under conventional tillage practices, the soil loss potential is expected to be quite large. However, the presence of forages throughout the watershed helps to balance the potential soil loss, resulting in a mean watershed sediment loss of about 2000 kg ha^{-1} yr^{-1}.

Annual TP export is significantly influenced by the precipitation ratio pattern. Because of the increase in soil test P, there is also an increase in TP exports over time. Mean predicted TP exports for the watershed were 2.24 kg TP ha^{-1} yr^{-1}. The influence of the 3-year rotation pattern can also be noticed when the impact of precipitation ratio is removed.

5.2 Analysis of cropping systems

Six important cropping systems were analysed for the Castor watershed. They were continuous corn, continuous grassland, pasture, crop rotation with corn, crop rotation without corn, and crop rotation with soybeans. In general, the management practices including tillage practices, fertiliser application and crop rotation. All these practices were identified as having a predominant influence on the model outputs.

5.2.1 Analysis of continuous corn

Continuous corn cultivation showed a high soil loss potential as conventional management practices resulted in the export of significant quantities of P from these fields. Considerable P inputs coupled with high TP export and higher plant uptake slowed the accumulation of P in the soil. Fields under continuous corn exported an average of 4.1 kg TP ha^{-1} yr^{-1}, as part of the 3390 kg soil eroded per hectare per year. Such values were amplified at the watershed level because continuous corn fields represented one-third of the cropland area.

5.2.2 Analysis of continuous grassland

Continuous grassland cropping systems bring a balance to the watershed. Low potential soil loss coupled with low fertiliser inputs result in very low TP exports. The insertion of grassland in the rotation of any crop helps to slow the increase in soil P, thereby reducing TP exports as well as soil erosion.

5.2.3 Analysis of pastures

As was the case for continuous grasslands, pasture areas resulted in little soil loss or TP exports. Soil P concentrations increased slowly over the simulation period because of slightly higher P inputs than outputs. Again, the presence of pasture areas in a watershed helps to maintain a good phosphorus balance. However, the impact of a grazing herd on the pasture was not taken into account. A large herd on a small pasture area could quickly erode the soil and subsequently greatly contaminate surrounding watercourses, especially if the animals had easy access to them.

5.2.4 Analysis of crop rotation with corn

This type of field rotation covers 27% of the watershed cropland area. It is the second most common landuse in the watershed. The greater the proportion of corn cropping in the rotation, the greater the soil loss potential, and in turn the greater the TP exports. On the other hand, the inclusion of cereal crops with high P inputs, despite their lower soil loss potential, increases soil P concentration. This type of rotation is very representative of what goes on over the entire watershed. The mean soil P increase of 5.12 kg Mehlich-III P ha^{-1} yr^{-1} under such conditions occurs because of the combined effect of corn and cereal crops and their respective management practices. Thus, the presence of grassland in the rotation can help rebalance soil P levels in these cropping systems. This type of rotation is a good alternative to continuous corn monoculture. The soil loss and TP exports are still significant, but compared to continuous corn, the improvement is substantial.

5.2.5 Analysis of crop rotation with soybeans

Crop rotation with soybeans required relatively low commercial fertilizer and manure inputs. The distinct pattern in the evolution of soil P concentrations and soil loss was created by differences between the two main crops: soybean and corn. Soybeans result in less soil loss, mainly because in this watershed, they are grown under conservation or no-till tillage practices. For that reason, TP exports were lower during this part of the rotation. Within the rotation cycle, corn grown under conventional practices resulted in greater soil loss and TP exports. However, the decrease in soil P concentration was not apparent on the TP export trendline (Figure 4-8).

5.2.6 Analysis of crop rotations without corn

Crop rotations without corn were characterised by a rapid increase in soil P concentration. Substantial manure and commercial fertilizer inputs, specific management practices and the low plant requirements are collectively responsible for this trend. The importance of forages and conservation tillage in the rotation results in lower potential soil loss with a mean of 1500 kg soil ha-1 yr^{-1}. However, TP export loads are still quite high at 2.55 kg TP/ha-yr.

5.3 Analysis of alternative management practices for reducing P

Further analysis was done by applying different management practices to the worst case cropping system leading to highest phosphorus contamination. Because of its high TP export, continuous corn fields were recognized as the worst case in P pollution. Different types of management modifications were applied to the corn cropping system. When corn is part of a crop rotation, the model outputs decrease to more reasonable levels. The different management practices for continuous corn are discussed below.

5.3.1 Analysis of continuous corn with different management practices

Continuous corn cropping shows over time, an increase in soil test P. The increase is due to the fertiliser input and the soil loss potential. When fertiliser inputs were reduced or increased, the soil test P changed accordingly. The soil loss potential varied as a result of the management practices applied on the field. The more intense the tillage practices, the greater erosion which took place. As for TP exports, they resulted from a combination of soil test P and soil loss potential. However, the soil loss potential for continuous corn with conventional tillage was so great, that this cropping system produced the greatest TP loads. In this case, the soil test P increase was less important than the one achieved under continuous corn with no-till, because of its high TP export value. Over time, with the same P inputs, the continuous corn field with no-till gets richer faster, but from an environmental standpoint, overall TP loads are more detrimental in the short term than soil richness. Thus, continuous corn fields with conventional tillage still pose a greater threat. However, when P inputs approached plant requirements, the threat was reduced.

5.3.2 Analysis of cash crop and dairy farm rotation

The simulations for cash crop and dairy farm rotations indicated that these two cropping systems show an increase in soil test P values over time. The mean annual increase for the cash crop rotation was 2.15 kg Mehlich-III P ha^{-1} yr^{-1}, and 2.50 kg Mehlich-III ha^{-1} yr^{-1} for the dairy farm rotation. As expected, the cash crop rotation showed a greater soil loss potential than the dairy farm rotation. The best way to reduce soil loss potential in a rotation is to introduce one or more seasons of grassland with no tillage. In a four year rotation, just one year of grassland can reduce soil loss by roughly 3000 kg ha^{-1}. Over a 30-year period it can retain up to 21 000 kg soil ha^{-1} as well as the attached phosphorus.

If TP export is considered the most important parameter in terms of P contamination, crop rotations are a good alternative to continuous corn monocropping under which losses could reach as high as 3.36 kg TP ha^{-1} yr^{-1}. For example, under the dairy farm rotation, losses would be only 1.34 kg TP ha^{-1} yr^{-1}, while losses under the cash crop rotation (2.46 kg TP ha^{-1} yr^{-1}) would result in a small improvement. The value under the cash crop rotation is closer to that under corn with conventional tillage because the years where corn with conventional tillage is present in the rotation, the export values of the two rotations reach the values of corn with conventional tillage. However, over a thirty year period the improvements result in a load reduction of 3.3 kg TP ha^{-1}. In addition, the fertiliser inputs simulated in these five cases were closer to plant requirements, resulting in smaller loads in TP export than occurred in the watershed. However, even with high values of soil test P and soil saturation, the fertilizer application rates suggested by the MENV regulations may still result in increases in those two parameters.

5.4 Overall conclusions

This study demonstrates the applicability of the WEND model to the Castor watershed at the field-scale. The outputs obtained by the simulation are close to what has been measured at the outlet of the watershed. The model was used to examine the impacts of crop rotations, fertilizer application and tillage management on TP export. The model performance could be improved with some modifications and a greater amount of input data. Further development of this model can make it an excellent tool for researchers or agroenvironmental specialists to understand the effects of different management practices.

However, crop rotations seem to be an important management practice that should be more carefully examined when establishing field management practices. Just one year of grassland can greatly improve the overall environmental health of a watershed. The other important element relates to the management of P inputs. Fertiliser inputs often surpassed plant requirements by two or three-fold. Even after following MENV's regulations, which are considered severe in the agricultural sector, for applying fertilizers to soils that are already rich in P, soil test P and soil P saturation continued to rise.

6 Recommendations for future research

1. The model predictions can be greatly improved if the farm management data were available for a longer period of time. The crop rotation and management practices are usually applied over a period of five to seven years. Accessibility to these data would reduce the prediction of less extreme values by the model and permit better long-term model efficiency.

2. Soil loss in the WEND model is predicted by RUSLE, which is not sensitive to precipitation intensity. It is recommended that some mathematical relations that include precipitation intensity be inserted in the model, to improve prediction of the potential soil loss.

3. Moreover, the insertion of sediment yield calculations and a sediment delivery ratio would permit better estimation of real soil loss from fields compared with outlet measured values. The sediment yield calculations represent the amount of sediment passing a given point in a watershed, usually the outlet, over a certain period of time, expressed in $kg\ yr^{-1}$ (Novotney & Olem, 1994). The sediment delivery ratio represents the ratio of the amount of sediment which leaves a watershed to the gross amount of sediment that was originally eroded from the soil mass in the watershed (Novotney & Olem, 1994).

4. Because the soil types have different physical characteristics, enrichment ratio calculations could be included in the model to improve the phosphorus concentration evaluation. This ratio refers to the modification in soil particle size distribution of the original soil mass compared to the original composition before erosion occurred. This would then adequately represent the preferential transport of fine soil particles and the preferential deposition of larger ones. Adding this calculation could permit a better estimation of phosphorus in different types of soil.

5. The model could also be rendered sensitive to the amount of surface water loads. When loads were high, it could be a result of super rich soils or a lot of surface runoff with not much drainage. In contrast, small surface loads could be a result of low soil P concentration or soil with better drainage characteristics resulting in greater subsurface drainage.

6. It would also be interesting to see the impact of the amount and method of fertilizer application on P losses. This could show the influence of manure application *vs.* commercial fertilizer on soil composition and plant vigour.

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