Relationships between Soil-test Phosphorus and Runoff Phosphorus in Small Alberta Watersheds

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ABSTRACT

Agricultural sources of phosphorus are associated with the eutrophication of surface waters in Alberta. Field-scale relationships between soil-test phosphorus (STP) and flow-weighted mean concentrations (FWMCs) of dissolved reactive phosphorus (DRP) and total phosphorus (TP) in runoff are essential for modelling phosphorus loss. However, there are limited field-scale data for determining STP and runoff phosphorus relationships in Alberta. A 3-yr field study was carried out to study the relationship between STP and phosphorus in runoff. The degree of phosphorus saturation (DPS) was also explored as a means of predicting phosphorus in runoff. Eight field-scale microwatersheds (2 to 248 ha) throughout Alberta were instrumented with circular flumes and automated water samplers and runoff was monitored for a 3-yr period. Soils were sampled from three incremental layers (0 to 2.5 cm, 2.5 to 5 cm and 5 to 15 cm) each spring and fall using a stratified landform-based approach. Soil samples were analyzed for STP content and the results were calculated for the 0- to 2.5-cm, 0- to 5-cm, and 0- to 15-cm soil layers. Five representations of STP were calculated for comparison with the DRP and TP FWMCs, including site mean, landform area-weighted mean, and means of subsamples of representative random samples, random samples, and the runoff contributing area. Average STP in the 0- to 15-cm layer ranged from 3 to 512 mg kg^{-1}, and DPS ranged from 5 to 91%. The majority of runoff was generated from spring snowmelt runoff. Seasonal FWMCs ranged from 0.01 to 7.4 mg L^{-1} DRP and 0.1 to 8.0 mg L^{-1} TP. Strong linear relationships (r^2 = 0.87 to 0.89) were found between the site average STP and the FWMCs of DRP and TP. The relationships had similar extraction coefficients (slopes), intercepts, and predictive power among all three soil layers. Relationships between the four other STP representations and DRP and TP FWMCs showed no significant improvement compared to those of the site mean STP. Extraction coefficients (0.013 to 0.014) were within the range of those reported for other studies in Alberta, but were greater than those for other rainfall simulation studies at laboratory- and field-plot scales in Alberta and in the United States. The DPS showed similar predictive ability to STP; however, the relationship was non-linear. The field-scale STP relationships should provide the basis for modelling phosphorus loss in runoff from agricultural land in Alberta.

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INTRODUCTION

Phosphorus is an essential nutrient for plant growth, and is often the limiting nutrient in soil and water. Since it is often not readily available for plant uptake, it is applied as a fertilizer in agricultural production systems. However, the intensification of livestock production has led to concentration of phosphorus in small areas, as more nutrients are imported than are exported in livestock products, including manure. The result is a net accumulation of phosphorus in soil. Excess phosphorus in soil is vulnerable to transport to surface waters via surface runoff, and this can cause degradation in water quality by accelerating eutrophication.

Eutrophication is the excessive growth of aquatic plants and algae due to the enrichment of surface waters with nutrients, which in turn cause oxygen depletion, taste and odour problems, loss of biodiversity, and loss of aesthetic and recreational value (Carpenter et al. 1998). Stimulation of plant and algal growth occurs at much lower phosphorus concentrations in aquatic environments than on land. Nutrient additions that cause eutrophication have been identified as "one of the most significant forms of river pollution" by the United Nations (UNESCO 2002) and one of most common causes of impairment in fresh waters in the United States (US EPA 1998). Diffuse losses from agriculture have been identified as the largest non-point source of phosphorus to water bodies in the United States (US EPA 2002) as well as impacting water bodies in many parts of Canada (Chambers et al. 2001).

The prediction of phosphorus losses from land has been a major focus of agricultural and limnological researchers during the past decade. Many researchers have reported a direct linear relationship between phosphorus concentrations in soil and levels of dissolved phosphorus in runoff (Sharpley et al. 1977, 1978; Daniel et al. 1994; Pote et al. 1996; Torbert et al. 2002). However, most of these relationships have been derived from rainfall simulations at laboratory or small-plot scales and may not adequately represent relationships from natural rainfall at field, catchment, or watershed scales since variables are site and soil specific (Young and Mutchler 1976; Mannaerts 1992). Research over large areas is complex, expensive, often unrepeatable, and may not necessarily improve the understanding of the processes resulting in phosphorus transfer (Doughtery et al. 2004). As such, plot and simulated rainfall studies are practical and necessary to illuminate fundamental processes that govern the relationship between soil-test phosphorus (STP) and dissolved phosphorus (DP) in runoff. However, lab- and plot-derived relationships must be validated at field scales so the processes and relationships identified can be extrapolated to scales where management is applied, and to larger watershed areas where management changes can be evaluated. Furthermore, while many relationships have been developed to predict losses of DP, water quality guidelines are based on total phosphorus (TP) as dissolved and particulate forms contribute to eutrophication.

Scale Dependency of Phosphorus Loss

Complex hydrological processes influence the amount and forms of phosphorus loss, and these processes may be scale dependent (Bloschl et al. 1995). Nash et al. (2002) identified detachment and dissolution as the primary processes of phosphorus mobilization in agricultural systems. In small-plot-scale studies, the comparatively high kinetic energy of overland flow and the detachment of soil particles are the most important mobilization mechanisms, causing greater proportions of particulate phosphorus (PP) to be exported relative to DP. Greater proportions of PP may also be lost from larger areas where run-on from other fields can increase water depth and the velocity of overland flow, favouring the detachment process. McDowell and Sharpley (2002) reported that flow path length at the field-plot scale influenced the forms and amounts of phosphorus lost during overland flow, with increased selective erosion of finer particles with increasing flow path length. Sharpley et al. (1991) and Smith et al. (1991) found that as erosion from natural rainfall increased, phosphorus transport in overland flow from four grassed and cropped watersheds showed a non-linear increase in percent of PP transported as TP in surface runoff, with a corresponding decrease in percent of dissolved reactive phosphorus (DRP).

For larger field-sized areas, hydrographs are longer and less peaked (Dingman 1994), favouring dissolution; hence, greater proportions of DP fractions are measured. Nash et al. (2002) also suggested that under natural rainfall, peak flows per unit area from small plots are disproportionately greater than from a field due to the shorter hydrologic response time for water to travel from the furthest point of the plot to the monitoring point. They also noted that equilibrium conditions, where phosphorus sorption processes to and from sediment are equal, are more likely to occur in catchment-scale studies than in small-plot- or lab-scale studies, due to the greater contact time between sediment and runoff water and the longer hydrologic response time in much larger areas. This leaves deposition and dilution as the major factors affecting the loss of DP. The results of other studies (Bloschl et al. 1995; Le Bissonnais et al. 1998; Gascho et al. 1998) support these concepts of scale-dependent hydrology.

Although strong relationships between STP and runoff phosphorus have been observed at the lab and plot scales, results from field-scale research have been confounded by variation in soil types, hydrology, and management. The relationship between STP and DRP in overland flow at the field scale varied with soil type, management, and runoff episodes (Sharpley et al. 1996; Sibbesen and Sharpley 1997). Relationships for cultivated land tend to have larger slopes, or extraction coefficients (i.e., lose more phosphorus per unit of STP) compared to grasslands (Sharpley et al. 2002; Kleinman et al. 2004). Sharpley et al. (2002) also found that the extraction coefficient increased with greater erosion or reduced soil cover, resulting in greater interaction between soil and overland flow and greater phosphorus release. Sharpley (1995) and Sharpley et al. (1996) concluded that field-scale coefficients are too variable to allow the use of a single or average relationship for all soils under the same management due to the inherent variability between soils and to the soil-specific nature of soil phosphorus release to overland flow. Sharpley et al. (2002) stated that the effect of land management on soil phosphorus release may be more accurately represented as a function of erosion. However, Vadas et al. (2005a) proposed that a single extraction coefficient could be used to approximate DRP release from soil to runoff, based on lab-scale and plot-scale results from 30 soil types. Since extraction coefficients from field-scale studies are not well documented, an understanding of the relationship between STP and phosphorus in runoff in conditions representing local climate, soil type, land use, and management is needed.

Total phosphorus is used for defining water quality guidelines in Alberta (Alberta Environment 1999), and recently proposed draft guidelines for Canadian freshwaters are also based on TP concentrations (CCME 2003). However, comparison of the results of field-scale studies of TP literature is difficult as few studies report TP results and these are not often coupled with measurements of STP. Schroeder et al. (2004) found stronger relationships between STP and TP (r^2 = 0.69) than between STP and DRP (r^2 = 0.56) at the field-plot scale, but most other researchers have reported poor relationships between STP and TP at the field-plot scale (Andraski and Bundy 2003; Kleinman et al. 2004). Kleinman et al. (2004) reported similar relationships between STP and DRP from rainfall simulations on 0.2-m^2 packed boxes of bare soil and at the field-plot scales under rainfall simulation, suggesting that similar processes operate at lab- and field-plot scales. However, they found that the packed boxes yielded increased concentrations of TP relative to the grassed field plots and this was attributed to reduced infiltration, increased surface flow, bare soil conditions, and increased erosion from the lab-scale packed boxes relative to the field-plot scale results. Erosion of PP is much more scale-dependent than dissolution of DRP and is therefore difficult to replicate at the lab scale.

Characterizing Soil Phosphorus at Field and Watershed Scales

The distribution of phosphorus concentrations can be variable within a landscape or farm fields (Schepers et al. 2000). Furthermore, not all areas of the field contribute equally to runoff. Gburek and Sharpley (1998) found that only small, saturated portions of a field contribute runoff during rainfall events. Their critical source area concept hypothesizes that near-stream STP content has a greater influence on phosphorus export from a watershed than does STP from the entire watershed. Pionke et al. (1996) reported that nearly all of the biologically available phosphorus originated from less than 10% of the contributing land area. Zollweg et al. (1997) and Weld et al. (2001) reported that more than 80% of phosphorus exported from a watershed came from small areas along stream channels during a few, rapid storm events, and that phosphorus export from these near-stream regions was large because the regions of high runoff potential coincided with areas of high STP. Therefore, characterizing STP using agronomic sampling methods or site averages may not adequately represent phosphorus available to runoff at the watershed scale.

Heathwaite and Dils (2000) and Cornish et al. (2002) observed that runoff phosphorus concentrations from small plots increased with distance downslope, which they attributed to greater STP levels from the downslope movement of phosphorus-rich material for extended periods of time. Although it is unknown whether this pattern of STP distribution is common or not and its influence on water quality is unknown (Doughtery et al. 2004), this trend has also been noted in Alberta (Nolan et al. 1999; Penney et al. 2003) and elsewhere (Mulla 1993; Campbell et al. 2003). These results suggest that the use of a typical random agronomic soil sampling scheme within farm fields may not adequately represent STP levels that can impact water quality, if high STP levels are in direct proximity to drainage areas.

In addition to within-field variability of STP, sample depth may influence runoff phosphorus. The interaction between soil and rainfall to generate runoff is generally restricted to the surface few centimetres of soil (Sharpley et al. 1978). The degree of interaction decreases exponentially with depth (Ahuja et al. 1981), and varies with slope, rainfall intensity, and kinetic energy of rain (Sharpley 1985). Plants also increase the depth of interaction by reducing runoff velocities and providing less dense topsoil with greater hydraulic conductivity (Ahuja and Lehman 1983). Guertal et al. (1991) showed that phosphorus levels in the upper 2 cm of no-till fields may be three times higher than at 8 cm. Torbert et al. (2002) reported that the strength of the relationship between STP and phosphorus in runoff from pasture plots with surface-applied manure was significantly reduced when sampling depth increased from 0 to 5 cm to 0 to 15 cm. Although these studies showed that a typical agronomic sample depth of 0 to 15 cm may not best represent STP levels from an environmental perspective, recent work by Schroeder et al. (2004) reported that STP levels from soil sampling depths of 0 to 2 cm or 0 to 10 cm had no effect on the relationship between STP and phosphorus in runoff. As well, Andraski and Bundy (2003) concluded that agronomic tests were as effective as environmental tests (at 0 to 2 cm, using different extraction techniques) for predicting DRP in runoff in rainfall simulation studies. It is unclear what depth of soil sample is best related to phosphorus concentrations in runoff, particularly at field scales.

The degree of soil phosphorus saturation (DPS) may also be an important means of understanding the relationship between STP and runoff. The DPS is a measure of how saturated soil sorption sites are with phosphorus, and is influenced by a number of variables, including aluminum, iron, calcium, clay, organic matter, pH, and carbon-to-phosphorus ratio. Sharpley (1995), Pote et al. (1996), and Hooda et al. (2000) reported that soils with similar STP levels have yielded different amounts of runoff phosphorus due to differences in phosphorus sorption capacity (PSC). Vadas et al. (2005a) found a split-line relationship where DRP rapidly increased at DPS values greater than 12.5% for noncalcareous soils. The question of whether an environmentally-oriented soil sampling method is more appropriate for understanding the relationship between STP and phosphorus in runoff has not been resolved, particularly for Alberta conditions.

Study Objectives

The main objective of this study was to determine the field-scale relationship between STP and runoff TP and DRP from field-sized catchments or “microwatersheds” under spring snowmelt and summer rainfall conditions in Alberta. This relationship was compared with the Edge-of-field Phosphorus Export Model (EFPEM) for DRP that was developed using only three field-scale catchments and the results of laboratory rainfall simulations on bare soil in packed boxes for 38 Alberta soils (Wright et al. 2003). We also examined whether a variety of depths and spatial representations of STP improved the prediction of phosphorus loss and explored the use of the DPS as an alternate method of predicting phosphorus export at the field scale.

MATERIALS AND METHODS

Site Description

Field-scale microwatershed sites were selected from watersheds that had high intensity agricultural use and existing water quality data. Each microwatershed had high runoff potential, good drainage, uniform management, no farmyard or non-agricultural influences, and good

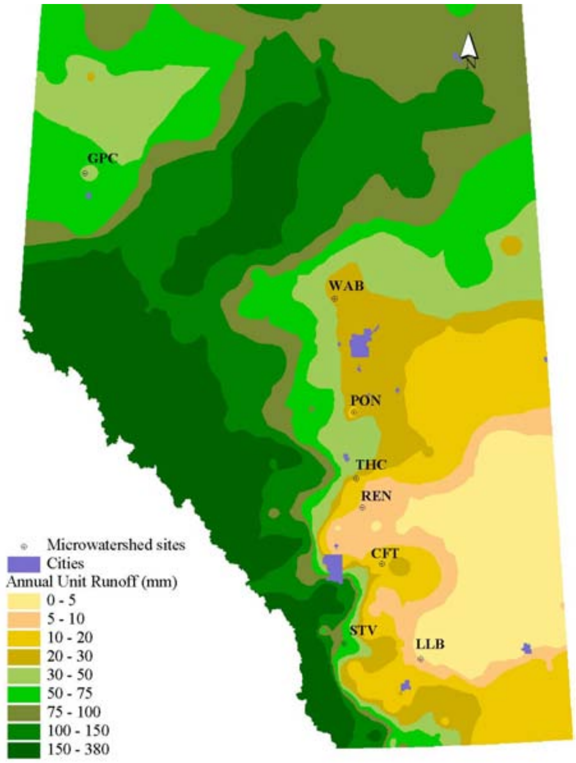

Fig. 1. Microwatershed sites on map of estimated annual runoff depths (Jedrych et al. 2006) within agricultural regions of Alberta.

access. Eight sites were selected for the study throughout the agricultural zone of Alberta (Fig. 1). The sites included one ungrazed grassland site (STV) west of Stavely; five cultivated, non-manured sites near Crowfoot Creek (CFT), Grande Prairie Creek (GPC), Renwick Creek (REN), Threehills Creek (THC), and Wabash Creek (WAB); and two cultivated, manured sites near Ponoka (PON) and Lower Little Bow River (LLB). Two sites were adapted from existing studies (PON, STV), while the rest were new sites.

The sites represented a range of precipitation and runoff potential within the agricultural area of Alberta (Table 1). Management characteristics of the cultivated sites were typical for Alberta and ranged from no-till at the CFT and THC sites, to reduced tillage at the REN site, and conventional tillage at the WAB, GPC, LLB, and PON sites (Table 1). The CFT, LLB, and GPC sites had multiple, but similarly managed fields. The PON site received high rates of cattle manure, whereas the LLB site received moderate rates of cattle manure. The STV site had not been grazed by cattle for at least 15 yr prior to the start of the study and had minimal grazing on the site since 1949; however, wildlife, such as deer and elk, are prevalent in the area.

Digital elevation models (DEMs) derived from photogrammetry at each site were used to identify microwatershed boundaries, contributing areas, and areas where flow and deposition were likely to occur. The $\ln(\alpha/\tan \beta)$ topographic or wetness index, where α is the upslope contributing area and β is the local slope, has been used as an indicator for surface runoff

Table 1. Characteristics and management information of the fields closest to the drainage outlet at the eight microwatershed sites.

Site	Area (ha)	Annual precipitation [z] (mm)	Est. annual runoff potential [y] (mm)	Management [x]	Type of phosphorus application	Added phosphorus (kg ha^{-1})
			Ungrazed grassland site			
STV	2	500-550	69			
			Non-manured sites			
CFT	248	350-400	18	NT	Banded with seed	17 – 22
GPC	62	450-500	50	CT	Banded with seed	10 – 21
REN	26	400-450	13	RT	Banded with seed	22-28
THC	51	450-500	25	NT	Banded with seed	15 – 25
WAB	33	500-550	27	CT	Banded with seed	15 – 17
			Manured sites			
LLB [w]	88	350-400	7	CT	Manure every 3 yr	
PON	30	500-550	19	CT	Manure 1-2x per yr	

[z] Chetner and Agroclimatic Atlas Working Group (2003).
[y] Jedrych et al. (2006).
[x] CT = conventional tillage, RT = reduced tillage, NT = no tillage before seeding.
[w] Irrigated.

contributing areas (Page et al. 2005) and as the basis for the rainfall-runoff model TOPMODEL (Beven and Kirkby 1979). As an example, Fig. 2 illustrates the PON microwatershed catchment boundaries with the products developed using the DEMs to identify upper, mid, and lower landform positions (MacMillan et al. 2000), to calculate a wetness index of Quinn et al. (1995), and to measure the length of the flow path. Maps for the other sites (except STV) can be found in Appendix 1.

Soil Sampling and Analysis

Soil sampling. A soil sampling strategy stratified by landform position was used at each of the cultivated sites. A minimum of six, three-point transects were selected on the classified DEM at each site according to landform position (upper, mid, and lower). Additional points were identified according to wetness index and proximity to outlet (Fig. 2) to ensure that STP was measured where flow and deposition were most likely to occur. Points were identified at a density of one sample per 1 to 6 ha (n = 22 to 48). The exception was the 2-ha STV site where only three sampling points were selected. A Differential Global Positioning System (DGPS), accurate to less than 1 m, was used to identify sampling points for repeat sampling.

A frame-excavation method was used to obtain representative portions of fertilizer bands or manure and soil. In the fall of 2002 and spring of 2003, a 19- by 50-cm steel frame was placed diagonal to crop rows to capture variations in tillage direction. A 5-cm deep frame was used in reduced-till fields, while a 10-cm deep frame was used in tilled conditions. The size of the frame was changed to 11 by 60 cm for sampling in the fall of 2003. The length of the new frame was adjustable to two times the fertilizer band width to improve the representation of banded phosphorus in the soil sample. The modified frames were placed perpendicular to the seed row

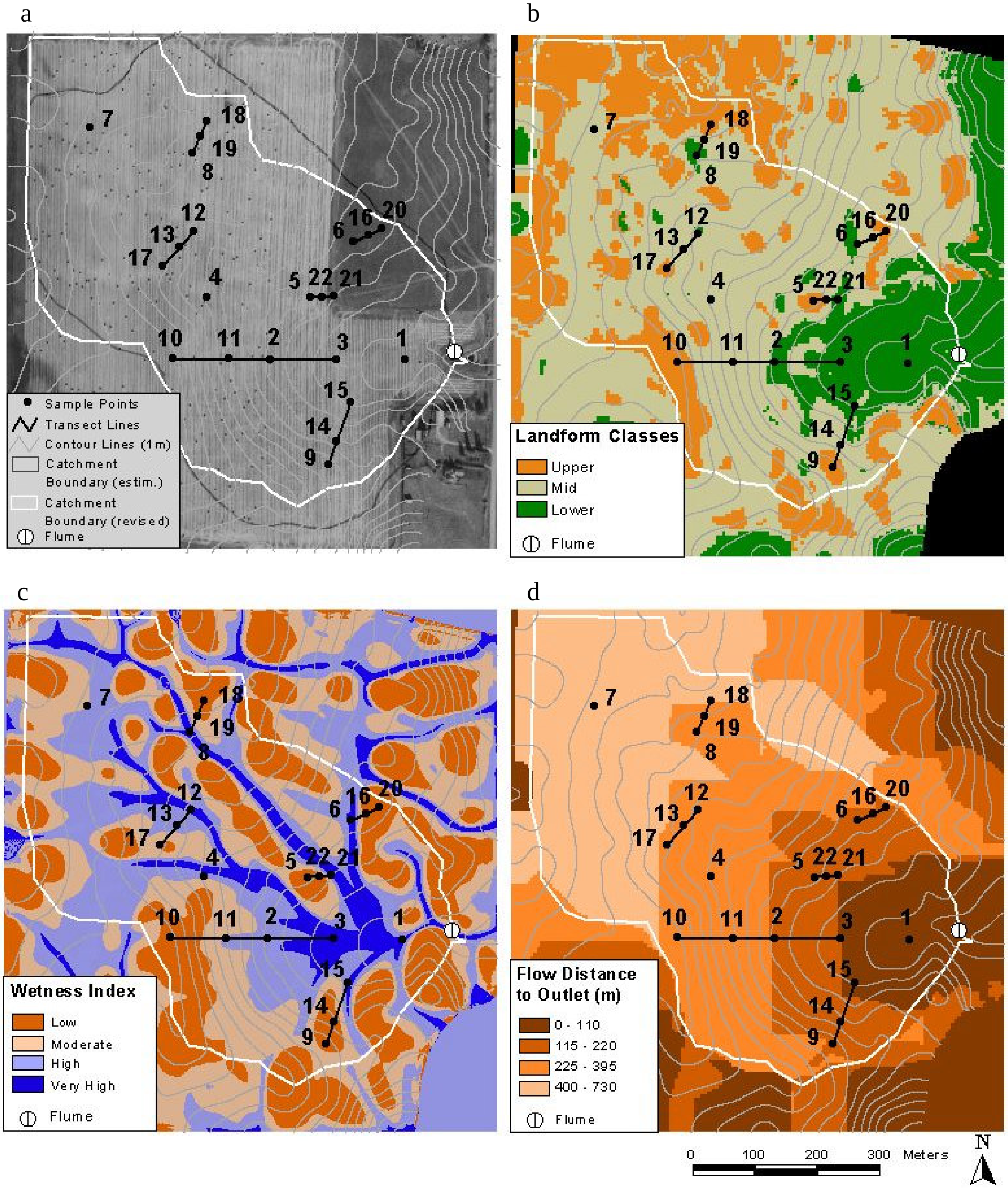

Fig. 2. Microwatershed boundary in relation to (a) air-photo features, (b) landform classes (MacMillan et al. 2000), (c) wetness index (Quinn et al. 1995), and (d) distance to outlet at the PON site. Numbers on transect lines are soil sampling points.

and fertilizer band. Soil samples were excavated from the 0- to 2.5-cm, 2.5- to 5-cm, and 5- to 15-cm layers using a 2.5-cm deep scoop, a 5-cm deep scoop, and a shovel, respectively. A comparison of the frame-excavation method with other soil sampling methods was described by Nolan et al. (2006). One frame per sampling point was used for non-manured fields and two frames per sampling point were used at the manured sites and at the ungrazed grassland site (STV). The excavated soil in each layer was well mixed in the field, and a 500-g subsample was shipped in coolers with ice packs to the laboratory.

Fall sampling was completed after the landowners had completed crop harvesting, fertilization, manure application, and tillage were completed in order to characterize STP levels for spring runoff events. To characterize the STP levels for the summer runoff events, a subsample of points in runoff contributing areas identified by a high wetness index (Fig. 2c) were sampled after seeding and fertilizing had been completed. These points represented 20% of the points that were sampled in the fall (n = 5 to 10). The smaller number of samples was chosen to minimize crop disturbance and to characterize the critical source areas of the field that were most likely to produce runoff during summer precipitation events. All sites were sampled each spring, except for the STV site where no fertilizer was applied.

Representations of soil-test phosphorus. The fall sampling points were used as a basis for calculating different spatial representations of STP within the seven cultivated microwatersheds to determine the best representation related to phosphorus in runoff. Five STP representations were calculated for each soil layer:

- **Site mean.** All sampled points were used to calculate the site mean.
- **Landform area-weighted mean.** The DEM data were used to quantify the spatial extent of the upper, mid, and lower landforms within each catchment (Table 2, Appendix 1). The proportion of each landform position was used as a multiplier with mean STP levels measured within each landform class to calculate a single representation of STP within each microwatershed.
- **Runoff contributing area.** The wetness index developed by Quinn et al. (1995) was used to select a subsample of points close to the main drainage through the microwatershed (Fig 2c, Appendix 1). The subsample represented 20% of all points, for a total of 5 to 10 points per site (Appendix 2). The means of the subsampled points within each site were used in this representation of STP.
- **Representative random.** The Tri-Provincial Manure Application and Use Guidelines (The Prairie Provinces' Committee on Livestock Development and Manure Management 2004) recommends that STP be characterized using a representative random composite soil sampling strategy at a density of three to four samples per hectare. It is recommended that samples be taken in areas of the field where average yields are expected, by sampling in mid-slope positions in hilly fields and avoiding sources of unusual variability (e.g. knolls, saline areas). For the representation of STP in this study, the results of points sampled in depressions and on upper slopes were omitted and a random selection was made from the remaining points in mid and/or lower landform positions for a minimum of 15 points per site or one sample per 3.5 ha. The mean value of this set of subsamples was calculated for a single representation of STP. The points used for this representation of STP are listed in Appendix 2.

- **Random.** The mean of a random subsample of 15 points per field was used for this representation of STP (Appendix 2).

Table 2. Proportion of landforms within each microwatershed site (%).

Site	Upper	Mid	Lower
Ungrazed native grassland			
STV	33	34	33
Non-manured sites			
CFT	9	55	36
GPC	17	65	17
REN	32	52	15
THC	24	49	26
WAB	7	73	20
Manured sites			
LLB	30	59	11
PON	25	58	17

Soil characterization. The soil at the upper, mid, and lower landform positions of a transect was described for each microwatershed according to the Canadian System of Soil Classification (Soil Classification Working Group 1998). Soils at each landform position were sampled by horizon and analyzed for organic matter (loss-on-ignition method, McKeague 1978), and texture (hydrometer method, Day 1965), as well as pH and electrical conductivity in a 1:2 solution of soil:water (McKeague 1978).

Soil-test phosphorus. Soil samples were dried and ground to pass through a 2-mm sieve, and a 5-g subsample was removed for STP analysis. Samples taken in the fall of 2002 and the spring of 2003 were analyzed using the modified Kelowna extraction method of Ashworth and Mrazek (1995) and the remaining samples were analyzed using the modified Kelowna extraction method of Qian et al. (1991). The samples collected in the fall of 2002 and the spring of 2003 were analyzed at a private laboratory (Lab 1). Samples from the fall of 2003 and beyond were sent to a second private laboratory; however, the samples from the fall of 2003 were analyzed at the lab's Calgary location (Lab 2), while the remainder were analyzed at the Saskatoon location (Lab 3).

A large volume (20 to 25 L) of soil from the 0- to 15-cm layer was collected at each site during the summer of 2002. These samples were air-dried, ground, and mixed well and used as reference samples. Ten subsamples per site were analyzed at each laboratory used in the study for a lab reference (Table 3). A subsample of the reference sample was also submitted with each batch of site samples. Soil-test phosphorus values were standardized to the Lab 3 reference sample results to account for differences in methodology among laboratories (Table 3).

Table 3. Mean values of soil-test phosphorus (STP) for the reference samples analyzed at three private laboratories and adjustment factors used to standardize to Lab 3 results.

	Mean STP [z,y]			Adjustment factor	
Site	Lab 1 (mg kg^{-1})	Lab 2 (mg kg^{-1})	Lab 3 (mg kg^{-1})	Lab 1	Lab 2
STV	6.82a	4.79b	7.20c	1.06	1.50
CFT	19.17a	15.04c	17.60b	0.92	1.17
GPC	28.76b	27.80c	35.20a	1.22	1.27
REN	24.06a	18.59b	24.20a	1.01	1.30
THC	18.12a	13.34b	17.90a	0.99	1.34
WAB	27.10a	20.18b	28.30a	1.04	1.40
LLB	119.37a	87.00c	104.44b	0.87	1.20
PON	211.06a	182.40c	196.70b	0.93	1.08

[z] Lab 1 – fall 2002, spring 2003; Lab 2 – fall 2003; Lab 3 – spring 2004, fall 2004, spring 2005.
[y] Mean values in each row followed by the same letter are not significantly different at $P \leq 0.05$.

Degree of phosphorus saturation. The phosphorus sorption capacity was characterized at six transects per site for the 0- to 2.5-cm layer using samples taken in the fall of 2003. A calcium chloride ($CaCl_2$) method was used to measure the phosphorus sorption index (PSI) of each soil. For more details on the method and samples refer to Casson et al. (2006). The degree of phosphorus sorption (DPS) was determined as the ratio between STP to PSI plus STP (Indiati and Sequi 2004). A subsample of six points sampled in the fall of 2002 and the fall of 2004 at each of the manured sites was also analyzed to determine if there were changes in the $CaCl_2$-PSI at the manured sites with time.

Site Instrumentation

Flumes. The six new sites were instrumented with circular flumes (Samani et al. 1991), which consisted of a 0.273-m internal diameter (ID) high-density polyethylene (HDPE) pipe installed vertically inside a 0.9-m ID HDPE horizontal pipe (Fig. 3). The lengths of the horizontal and vertical pipes were 3 m and 2.1 m, respectively. The vertical column was located 0.9 m from the inlet of the flume and was slotted with 10, 8-mm ID holes spaced at 10-mm intervals. Due to site restrictions, the circular flume at the LLB site was shortened to 1.83 m in length, while the CFT site had a smaller version of the circular flume (0.61-m ID by 1.83-m long). The PON site was initially instrumented with a 0.61-m H-flume, which was replaced with a circular flume in June 2003. The STV site was bordered on the down-slope edge with a trough, which directed runoff water into a 0.15-m trapezoidal flume.

Installation of the flumes was site-specific. The LLB, CFT, GPC, and WAB sites had earthen berms of 20 to 48 m in length constructed to direct flow towards the inlet of the circular flume. At the REN and GPC sites, a wooden drop box was initially constructed in front of the flume to direct flow into the flume. In July 2003, the drop box at the REN site was removed and the flume

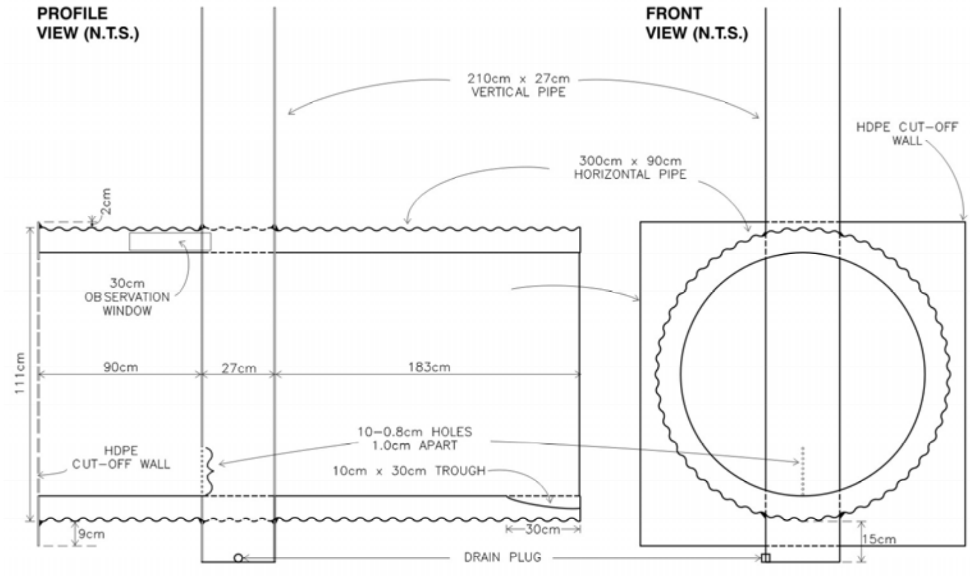

Fig. 3. Profile and front view of a typical circular flume.

was reinstalled following the washout of the flume during the spring 2003 runoff. The last 5 m of the approach channel to the re-installed flume was re-shaped and reinforced with erosion control matting and seeded to grass. The drop box at GPC was also reinforced with an impermeable geotextile liner prior to runoff in the spring of 2003 to prevent a similar washout.

Flumes at the REN and THC sites were attached directly to the inlet of the downstream road culverts, while flumes at the remaining sites were freely drained. The flumes were oversized in relation to the road culverts to compensate for the restricting effect of the vertical column on the road culverts. Settling of the flumes at some sites caused the outlet of the flume to be lower than the entrance to the culvert. This inactive head was taken into account when flow volumes were calculated.

Other instrumentation. Each site was equipped with a float potentiometer placed within the vertical column to measure head (or stage). Staff gauges were mounted on the exterior of the vertical column for manual flow measurements during site visits. Sites were also equipped with Lakewood TP10K5 thermistors and Davis tipping bucket rain gauges, which were replaced with Texas tipping bucket rain gauges in May 2004. Sites were powered with two, 15-W solar panels, and rechargeable 12-V batteries (Fig. 4). A second float potentiometer and Lakewood datalogger were installed at most sites in 2004 for backup collection of flow data.

Each site was equipped with ROM Communications Microcom units, except for the STV site. These units were integrated dataloggers with analog cellular communications technology that allow real-time monitoring of site conditions. The units were programmed to monitor head, temperature, precipitation, and battery voltage every 30 s. If flow or precipitation was detected, data would be recorded in the datalogger and reported on the website every 15 min and alarms

Fig. 4. Instrumentation at the LLB site showing the plywood-reinforced berm and flume.

would be sent via pagers and emails to team members. A technician and a continuously-monitored meteorological station were permanently on-site at the STV site. Therefore, the ROM Communications system was not needed at this site. Instead, the site was equipped with a Lakewood Ultralogger and a float potentiometer to record head.

Water samples were taken by ISCO 6700 automated water sampling devices, equipped with 24, 1-L ProPaks™ and disposable polyethylene inserts. The ISCO intakes were either 19 or 38 mm in diameter and were attached to the unit via a PVC suction line. Most of the ISCO intakes were attached in a trough near the outlet at the bottom of the flume. At the GPC site, the intake was moved to the front of the flume and raised to a 10-cm height to avoid sampling standing water due to the settling of the flume. Prior to spring runoff in 2004, the intakes at the REN and THC sites were also raised (3 cm at REN; 6 cm at THC) and placed within the mixing zone downstream of the vertical pipe.

The ISCO samplers were programmed to take a 150-mL sample every 15 min for a total volume of 900 mL or six samples per bottle. Changes in head were used to trigger the ISCO via a ROM Communications Microcom unit whenever flow volumes reached the minimum criteria set for each microwatershed.

A natural gas company constructed an earth road through the REN site in January 2004 that bisected the natural runoff pattern. In May 2004, two culverts were installed under the new road at low points to allow runoff to move through the microwatershed to the flume. Unfortunately, the road affected the 2004 runoff due to increased erosion at the site and because some of the runoff flowed over the county road northeast of the flume before entering the rear of the flume. A vertical pipe was installed on the edge of the road culvert in August 2004 to divert runoff from the road directly into the culvert.

Water Sampling and Analysis

Water samples were collected daily during runoff events and then immediately transported in coolers to the nearest Envirotest Laboratory location in Calgary, Edmonton, or Grande Prairie. Subsamples were filtered upon arrival and analyzed within 24 h for pH and electrical conductivity, within 48 h for DRP (ascorbic acid method; Murphy and Riley 1962), and within 30 d for dissolved phosphorus (DP) and TP (persulphate digestion). The first, middle, and last samples of the daily hydrographs were analyzed for additional parameters, including total suspended solids and total dissolved solids. Blanks filled with deionized water, as well as a prepared standard of known phosphorus concentrations, were submitted to the lab with each batch of samples as part of a quality assurance/quality control program.

Data Analysis

Flow measurements. Prior to head (or stage) data being converted to flows, values were corrected for the offset or zero value of the flume. The flumes were then calibrated using the Water Ware software program developed by Samani et al. (1991). The resulting calibrations were then plotted in TableCurve 2D, version 3 (Jandel Scientific Software 1994), to fit an appropriate curve to the data. Once a curve was selected and applied to the heads, a correction factor was applied to account for the slope of the flume and for any inactive head in the flume.

Flows in the 0.9-m circular flumes were best described by the following power function.

$$y = 0.0702\ x^{2.093} \tag{1}$$

Where:
y = flow (L s^{-1})
x = head (cm)

Flows in the 0.61-m circular flume at the CFT site were described by the following power function.

$$y = 0.0000673\ x^{2.072} \tag{2}$$

Runoff phosphorus calculations. To calculate flow-weighted mean concentrations (FWMCs), water chemistry data were linearly interpolated to 1-min intervals using Proc Expand in SAS (SAS Institute Inc. 2003). The expanded concentration data were then matched to the flow data and instantaneous loads were calculated for matching values by multiplying flow and concentration data. The area under the curve was then integrated to estimate total loads and flow volumes using a SAS area macro. Seasonal FWMCs were then calculated by dividing the total load for all events by the total flow volume for all events during the season.

Missing flows were accounted for using three methods depending on the period of time for which data were missing. If the time period for which the flow data were missing was less than 1 d, flows were linearly interpolated with missing head values supplemented by manual field measurements, where possible. If the time period was greater than 1 d, FWMCs were calculated for days with flow, and mean daily concentrations (not flow-weighted) were used for days without flow. The daily averages were then averaged for the whole event to determine the concentration. If no flow data were available, mean concentrations of the parameters were used.

Statistical analysis. Analyses of the soil and water samples were completed using SAS version 9.1 (SAS Institute Inc. 2003). The Univariate procedure was used to test the distribution of the data, and the Means and Summary procedures were used to generate descriptive statistics. Differences between means were tested using the Least Squared Means test in the Mixed procedure with variance components as the variance structure, the repeated and pdiff options, and Tukey-adjusted ***P*** values. The Corr procedure was used to calculate Pearson correlation coefficients. The REG procedure was used to relate measures of STP to runoff phosphorus and the Type III sums of squares in the GLM procedure was used to determine if there were significant differences in slopes and intercepts between regression equations. A significance level of 0.05 was used throughout this study.

RESULTS AND DISCUSSION

General Soil Characteristics

All sites had similar soil surface texture (loam), except for the GPC site, which was a clay loam (Table 4). Clay content in the surface horizons of the mid landform positions ranged from about 15% at the PON, REN, and STV sites to 29% at the GPC site (Table 4). Percent clay in the surface horizons of the upper and lower landform positions was within 10% of the mid landform values at all sites (Appendix 3). Slopes at most of the cultivated sites were similar, with lower slopes at the LLB site and greater slopes at the REN and THC sites. The STV site had steep slopes as it was located in the foothills. Organic matter content ranged from 14% at the STV site to 4.3% at the WAB site. Levels of organic matter in the mid landforms increased by at least 2% from upper to lower slope position at the STV, REN, THC, and PON sites, decreased in the lower landform relative to the upper landform at the LLB site, and was within 2% of the mid landform value at all other landform positions at the CFT, GPC, and WAB sites (Appendix 3). The pH in the mid landform positions was within the optimum range of nutrient availability at most sites, but was slightly alkaline at the manured LLB site. The pH was alkaline in the upper slopes at the CFT site, in all landform positions at the LLB site, and in the lower landform position at the PON site (Appendix 3).

Soil-test Phosphorus

Mean values of STP in the 0- to 15-cm layer ranged from 3 to 512 mg kg^{-1}, with the lowest value at the ungrazed grassland (STV) site and the highest value at the heavily manured PON site (Table 5). The range of mean STP values at the five cultivated, non-manured sites was only 19 mg kg^{-1} for all 3 yr. Ranges within sites were wider, varying from 24 mg kg^{-1} at the REN site in 2004 to 207 mg kg^{-1} at the CFT site in 2003, where manure had previously been applied to a knoll. At the manured sites, mean STP levels were an order of magnitude greater than at the non-manured sites, with levels at the moderately manured LLB site (269 mg kg^{-1}) close to half of the STP value at the heavily manured PON site (512 mg kg^{-1}) in 2002. The range of STP at each of the manured sites was close to 450 mg kg^{-1}. These results demonstrate the large increases in STP that can occur when manure is added to soil.

Table 4. Summary of surface soil and landform characteristics in the microwatersheds.

Site	Classification [z]	Landform type [y]	Slope (%)	Texture (surface / subsurface) [x]	Organic matter [x] (%)	Clay [x] (%)	pH [x]
				Ungrazed grassland site			
STV	O.BLC	M1h	6 – 25	L / CL	14.0	14	6.5
				Non-manured sites			
CFT	O.DBC	U1h	1 – 4	L / SiL	5.3	21	6.4
GPC	SZ.DGC	U1h-M1l	1 – 4	CL / C	7.5	29	6.0
REN	O.BLC	M1m	1 – 8	L / SL	6.6	15	5.7
THC	O.BLC	M1m	0 – 6	L / L	10.0	23	6.0
WAB	O.DGC	U1h	1 – 4	L / CL	4.3	20	5.9
				Manured sites			
LLB	O.DBC	U1l	1 – 2	L / CL	4.5	26	7.7
PON	E.BLC	H1l	0 – 5	L / CL	9.6	12	6.5

[z] Classification symbols follow Canadian System of Soil Classification: O = Orthic; SZ = Solonetzic; E = Eluviated; DB = Dark Brown; BL = Black; DG = Dark Gray; C = Chernozem.
[y] Landform symbols follow the AGRASID convention: U = undulating; M = rolling; H = hummocky; l = low relief; m = moderate relief; h = high relief (relative for each landform).
[x] Measured at the mid landform position.

Table 5. Soil-test phosphorus in the 0- to 15-cm layer.

		Fall 2002				Fall 2003				Fall 2004			
		Min.	Max.	Mean	S.E.	Min.	Max.	Mean	S.E.	Min.	Max.	Mean	S.E.
Site	n	mg kg^{-1}											
						Ungrazed grassland site							
STV	3	2	3	3	0	5	6	5	0	5	6	5	0
						Non-manured sites							
CFT [z]	48	12	104	34	2	13	220	39	4	10	206	35	4
GPC [y]	22	17	44	33	2	3	52	35	2	8	45	27	2
REN	28	11	50	20	1	14	62	24	2	13	37	21	1
THC [x]	27	7	58	26	2	11	68	27	3	8	52	23	2
WAB	27	21	55	35	2	16	57	32	2	14	41	25	1
						Manured sites							
LLB	45	47	482	269	15	51	630	236	17	25	480	242	17
PON	22	302	797	512	29	240	786	446	27	226	554	366	20

[z] Includes a manured knoll.
[y] Six points added in 2003 and 2004 after field observations of artificially modified drained area draining to flume.
[x] Two points outside of field in a ditch were removed from the data set.

Standard errors were lower at the non-manured sites than at the manured sites, since manure application increases STP variability due the high variability of manure as a phosphorus source (Dou et al. 2001). Coefficients of variation (CV) ranged from close to 20% at the STV, WAB, and GPC sites to 48% at the CFT and THC sites, with the CV of the manured sites lying between these extremes. The range between maximum and minimum values was wider at the CFT site than at the other sites due to past manure additions to the field. The high CV at the THC site was related to a low value in one mid landform position that was confirmed by re-analysis and by results in subsequent years.

All of the STP data from the microwatershed sites are presented in Appendix 4.

Variation with time. Soil-test phosphorus mean values in the 0- to 15-cm layer were generally not significantly different among the six sampling times at each site (Table 6). There was a tendency for STP to be higher in the spring compared to the previous fall; however, a significant difference only occurred between the fall of 2002 and the spring of 2003 at the GPC site. The tendency of higher spring STP values were likely the result of fertilizer additions to the sites (Table 1). The tendency for STP to decline in the fall compared to the previous spring was likely due to crop removal and sorption of the fertilizer by the soil, although this was significant only at the GPC site between the spring and fall of 2003. None of the fall STP results differed among years at any of the non-manured sites. Lockman and Molloy (1984) also measured higher STP levels after spring fertilizer applications, with gradual decreases to the lowest values in early winter, and attributed the differences to crop removal and soil fixation.

At the moderately manured LLB site, manure was applied at different times to the two halves of the 88-ha site. The east half of the microwatershed, which contained the field outlet, received manure in the spring of 2002 and again in the fall of 2004. The west half of the microwatershed received manure in the fall of 2002 and the spring of 2005. A ripper disc was used to till the soil to 38 cm in the fall of 2003 and following manure application in the fall of 2004 and the spring of 2005. The STP decline measured in the 0- to 15-cm layer between the fall of 2002 and the fall of 2004 was 50 mg kg^{-1} (90 kg ha^{-1}, assuming a bulk density of 1.2 Mg m^{-3}), although this was not significant. This decline was close to the expected removal of 30 kg ha^{-1} yr^{-1} by the corn silage grown at the site (Canadian Fertilizer Institute 2001). No increase in STP was observed between the spring and fall of 2004. This was likely influenced by the observation that six of the nine points sampled in the spring did not receive manure additions in the fall of 2004 due to the wet soil conditions of the draw in which they were located. When the three manured points were considered separately, the increase was 49 mg kg^{-1}.

No manure was applied after the fall of 2002 to the previously heavily manured PON site. Levels of STP declined with every sampling interval after the spring of 2003. The decrease of up to 177 mg kg^{-1} (319 kg ha^{-1}, assuming a bulk density of 1.2 Mg m^{-3}) between the spring of 2003 and the fall of 2004 was significant and likely the result of many factors. Since the manure was applied late in the fall of 2002 and poorly incorporated, STP levels were extremely high. In the fall of 2003, the low areas of the field were paraplowed to 46 cm and the entire field was deep-tilled to 15 cm. The dilution of STP with lower horizons by tillage has been shown to reduce STP levels with time (Kleinman et al. 2002). In addition, a portion of the manure phosphorus would have been fixed by the soil, while approximately 50 kg ha^{-1} of phosphorus would likely have been removed by the corn silage and barley crops grown during the two growing seasons at the site (Canadian Fertilizer Institute 2001).

Table 6. Soil-test phosphorus measured in the 0- to 15-cm layer for the 20% subsample of points in the runoff contributing area after spring seeding and fall fertilizer application.

		Fall 2002		Spring 2003		Fall 2003		Spring 2004		Fall 2004		Spring 2005	
		Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Site	n	mg kg^{-1}											
Non-manured sites													
CFT	10	27 a [z]	4	44 a	5	29 a	4	37 a	3	25 a	4	34 a	4
GPC	5	24 b	2	38 a	3	23 b	4	23 b	3	15 b	3	29 ab	4
REN	6	24 a	6	39 a	5	28 a	7	35 a	4	28 a	3	34 a	3
THC	6	32 a	6	36 a	6	33 a	7	29 a	5	27 a	4	33 a	6
WAB	6	40 a	4	43 a	5	39 a	4	40 a	4	31 a	3	35 a	2
Manured sites													
LLB	9	216 a	34	212 a	30	158 a	37	169 a	28	166 a	41	178 a	35
PON	6	582 ab	51	614 a	36	527 ab	58	458 ab	38	437 b	36	498 ab	19

[z] Means within the same row followed by the same letter are not significantly different at $P \leq 0.05$.

Variation with depth of soil layer. In the fall of 2002, mean values of STP in the 0- to 2.5-cm layer were not significantly different from those in the 0- to 5-cm layer (Table 7), but both of these layers had significantly greater STP values than in the 0- to 15-cm soil layer at the GPC, REN, and THC sites. A tendency of lower STP values in the 0- to 15-cm layer was observed at the other four sites, but the differences were not significant. Soil-test phosphorus in the 0- to 2.5-cm layer was 1.9 times higher than in the 0 to 15-cm layer at the minimum-till REN site and 1.7 times higher at the no-tilled THC site (Fig. 5). Guertal et al. (1991) measured up to three times more STP in the 0- to 2-cm layer than in the 0- to 8-cm layer in no-till conditions. Selles et al. (1999) measured accumulations of plant-available phosphorus in the surface 5 cm of soil after 12 yr of no-till continuous wheat in western Canadian cropping systems. Andraski and Bundy (2003) and Butler and Coale (2005) also reported similar findings. Phosphorus tends to be more concentrated near the soil surface because of its limited mobility in soil (Sharpley et al. 1978; Sharpley 1985), especially under reduced- or no-till management (Sharpley et al. 1993). In reduced tillage and pasture systems, phosphorus fertilizers do not have an opportunity to become evenly distributed throughout the topsoil (Sharpley et al. 1993; Crozier et al. 1999).

Despite increased STP levels in the 0- to 2.5-cm layer, the range of STP in this layer did not increase relative to the 0- to 15-cm layer, except at the two manured sites. Soil-test phosphorus levels in the 0- to 2.5-cm layer were highly correlated with levels in the 0- to 15-cm layer (r^2 = 0.99, df = 32).

The variability of STP values was similar between the 0- to 2.5-cm and 0- to 5-cm layers at all sites, except for the PON site. Variability decreased in the 0- to 15-cm layer. Reasons for the higher variability of STP measurements in the surface horizons include the difficulty of obtaining an accurate sample at shallow surface layers, particularly when soil surfaces are rough due to tillage or surface manure application.

Table 7. Soil-test phosphorus measured in three soil layers in the fall of 2002.

	0 to 2.5 cm		0 to 5 cm		0 to 15 cm	
	Mean	SE	Mean	SE	Mean	SE
Site	mg kg^{-1}					
			Ungrazed grassland site			
STV	5 a [z]	2	4 a	1	3 a	0
			Non-manured sites			
CFT	43 a	3	43 a	4	34 a	2
GPC	45 a	3	44 a	2	33 b	2
REN	38 a	2	36 a	2	20 b	1
THC	44 a	3	38 a	3	26 b	2
WAB	37 a	2	37 a	2	35 a	2
			Manured sites			
LLB	316 a	17	307 a	16	269 a	15
PON	648 a	50	597 a	37	512 a	29

[z] Means within the same row followed by the same letter are not significantly different ($P \leq 0.05$).

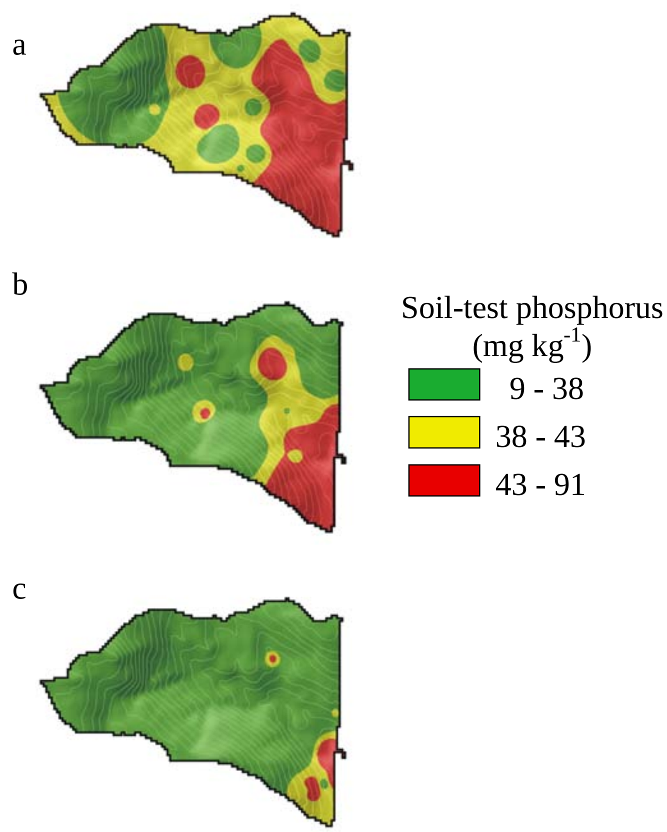

Fig. 5. Distribution of soil-test phosphorus in the (a) 0- to 2.5-cm, (b) 0- to 5-cm, and (c) 0- to 15-cm layers at the THC site in the fall of 2002.

Variation with landform position. Significantly greater STP levels in the 0- to 15-cm layer of the lower landform position were only observed at the THC site (Fig. 6). The increase in STP in the lower landform position compared with STP in the mid landform position at the THC site ranged from 13 to 15 mg kg^{-1} during the 3-yr study. Although STP levels were greater in the lower landform position at the THC site, other areas within the field had STP levels as high as those measured in the lower landform position (Fig. 7a). Significantly higher levels of STP in the lower landform position have been documented at the Soil Quality Benchmark sites in Alberta (Penney et al. 2003), as influenced by increases measured at 30% of the sites (Doug Penney, personal communication), and at other locations in Alberta (Nolan et al. 1999; Campbell et al. 2003), western Canada (Manning et al. 2001), and Minnesota (Mulla et al. 1993).

Different patterns of STP variation by landform position were observed at other sites. At the LLB site, unevenly distributed manure resulted in significantly lower levels of STP in the lower landform position than in the mid or upper landform position (Fig. 7b). There were no differences by landform position at the CFT, WAB, GPC, and PON sites (Fig. 6). These patterns were similar for all three soil layers in all 3 yr of the study.

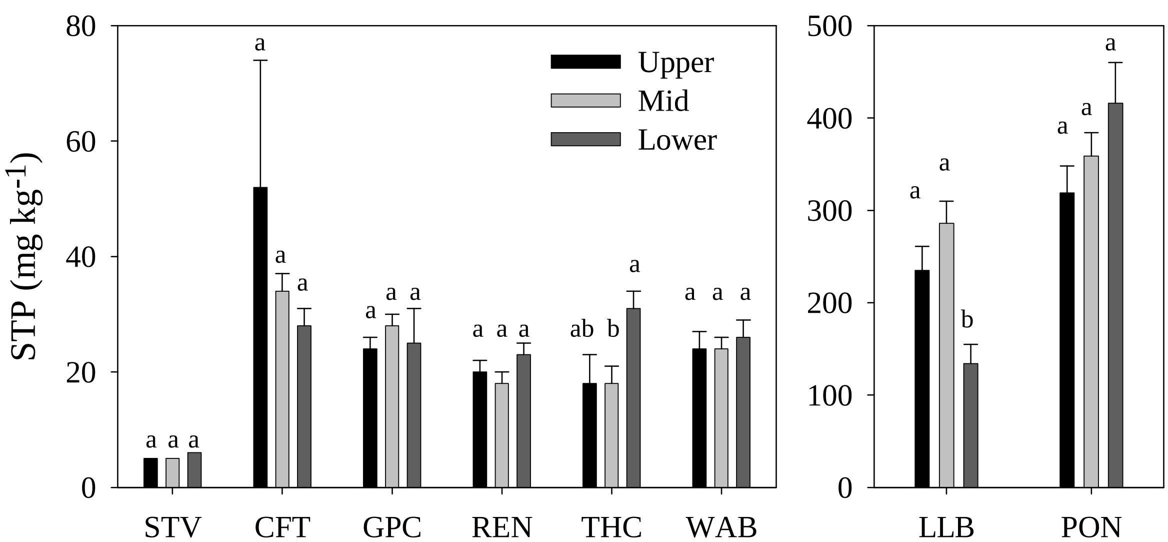

Fig. 6. Levels of soil-test phosphorus (STP) in the 0- to 15-cm soil layers at the different landform positions at the microwatershed study sites in the fall of 2004. Standard error bars are shown. Within each site, bars with the same letter are not significantly different at $P \leq 0.05$.

Weak, but significant relationships (r = 0.40 to 0.49) were found between STP in the 0- to 15-cm layer and the wetness index at all sites, except for the GPC site where the relationship was significantly negative (r = -0.44) and for the LLB and CFT sites where manure was differentially applied to the field and no relationship was found. For the 0- to 2.5-cm layer, a significant relationship between STP and the wetness index was only observed at the PON site. Page et al. (2005) concluded that the occurrence of "hot spots" obscured any strong relationships that might

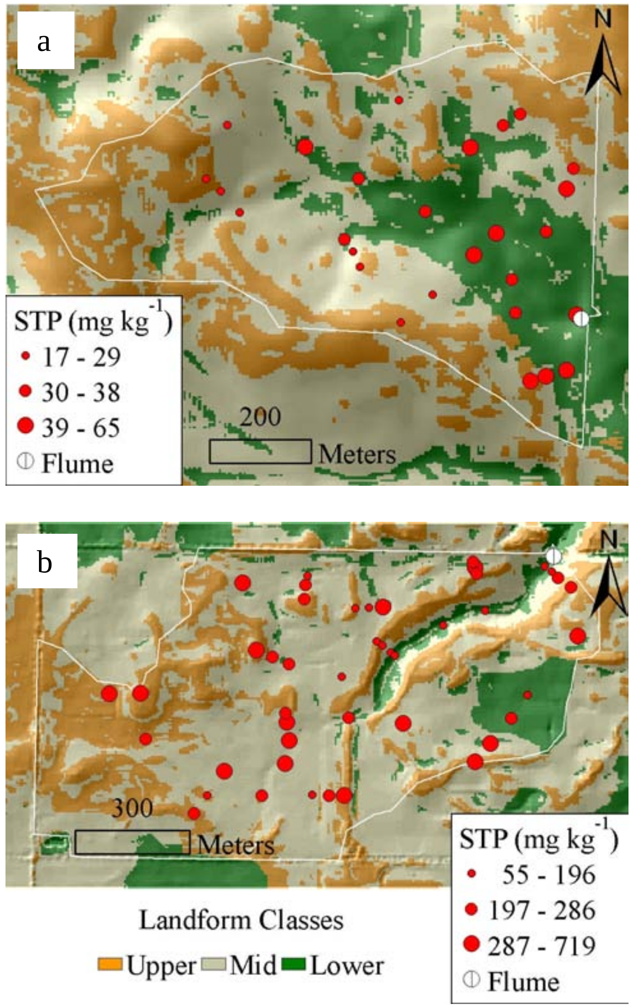

Fig. 7. Spatial representation of soil-test phosphorus (STP) levels mapped over the landform classification scheme at the (a) THC and (b) LLB sites. White line outlines the catchment boundary.

occur between soil phosphorus status and topographic index. They identified hot spots as areas related to differential management practices, such as manure application to selected areas. Similar observations of "patchy" distributions of STP within farm fields due to applications of manure and municipal sludge were also made by Cambardella and Karlen (1999). Gburek and Sharpley (1998) and Weld et al. (2001) also indicated that soil phosphorus status can be influenced by land use and field boundaries. These results suggest that management effects on the distribution of STP within a field, such as non-uniform manure or fertilizer application, grazing, or intensive tillage may override accumulations of STP in the lower landform position due to erosion processes (Cabot et al. 2004; Page et al. 2005).

Representations of soil-test phosphorus. There were few significant differences among the five STP representations in the 0- to 2.5-cm and 0- to 15-cm soil layers in the fall of 2002 (Table 8). This was also true for the 0- to 5-cm layer (data not shown). Significant differences were only observed in the 0- to 15-cm layer at the GPC site in the fall of 2002 where the runoff

Table 8. Mean values of various representations of soil-test phosphorus in the 0- to 2.5-cm and 0- to 15-cm soil layers at microwatershed sites in the fall of 2002.

Site	Site mean	Landform area weighted [z]	Runoff contributing area	Representative random	Random
	----------	----------	mg kg^{-1}	----------	----------
			0 to 2.5 cm		
CFT	43 a [y]	41	40 a	39 a	35 a
GPC	45 a	45	42 a	44 a	44 a
REN	38 a	36	43 a	39 a	39 a
THC	44 a	42	51 a	43 a	43 a
WAB	37 a	38	41 a	35 a	37 a
LLB	316 a	327	269 a	309 a	276 a
PON	648 a	654	680 a	662 a	673 a
			0 to 15 cm		
CFT	34 a	34	27 a	33 a	30 a
GPC	33 a	32	24 b	32 ab	33 a
REN	20 a	20	24 a	20 a	22 a
THC	26 a	24	32 a	26 a	25 a
WAB	35 a	34	40 a	32 a	34 a
LLB	269 a	279	216 a	265 a	257 a
PON	512 a	512	582 a	522 a	532 a

[z] Calculated using an area weighted mean, and as a result could not be statistically compared to the other STP representations.
[y] Means within the same row followed by the same letter are not significantly different at $P < 0.05$.

contributing area representation was significantly less than the site mean and random representations and in the 0- to 2.5-cm and 0- to 5-cm layers at the LLB site in the fall of 2004 where the runoff contributing area representation was significantly lower than the site mean and the representative random samples (data not shown) because manure was not applied in the wet lower landform positions in the fall of 2004. The differences among the five STP representations for the non-manured sites ranged from 3 to 9 mg kg^{-1} in the 0- to 2.5-cm layer and 4 to 9 mg kg^{-1} in the 0- to 15-cm layer (Table 8). The differences among the STP representations for the manured sites were 58 mg kg^{-1} at the LLB site and 32 mg kg^{-1} at the PON site in the 0- to 2.5-cm layer, and 63 mg kg^{-1} at the LLB site and 70 mg kg^{-1} at the PON site in the 0- to 15-cm layer.

Despite significantly higher levels of STP in the lower landform position at the THC site (Fig. 6), the relative area represented by the lower landform position at the site was less than 30% (Table 2). As such, the resulting mean of the landform area weighted calculations was slightly less than the THC site mean calculated using all sample points, for the 0- to 2.5-cm and 0- to 15-

cm soil layers (Table 8). Similar results were observed at the moderately manured LLB site, where despite significantly lower levels of STP in the lower landform positions (11% of the area) compared with the midslope position (59% of the area), landform area-weighted means were only about 10 mg kg^{-1} higher than the mean of all sampling points. Since the landform area weighted values were calculated as area weighted means, they could not be statistically compared to the other four STP representations, which were calculated as arithmetic means. Although these values could not be statistically compared, the landform area weighted values were similar to the other STP representations.

Page et al. (2005) noted that important information on the variability and spatial distribution of STP for a given sample area can be lost when samples are averaged. However, Daniels et al. (2001) concluded that when sampling soil phosphorus in pastures, current sampling strategies for agronomic soil tests can adequately account of spatial variability to produce a single, appropriate estimate of STP, if the recommendations are followed with respect to the number of samples. Similarly, Needelman et al. (2001) concluded that field mean STP in hog and poultry manure-amended fields could be used to characterize STP for applications that are not sensitive to small errors in STP estimates. In a study of manured and non-manured soils in Manitoba, Slevinsky et al. (2002) reported that there were no differences in STP levels measured in the 0- to 15-cm layer using a composite of 15 random points per field or using the average of four representative benchmark samples per field.

Degree of soil phosphorus saturation. In the fall of 2003, the PSI in the 0- to 2.5-cm layer was significantly higher at the GPC site than at any other sites (442 mg kg^{-1}, Fig. 8). The GPC site was the most recently cultivated and had the highest percentage of clay (Table 4). The range among PSI values at the non-manured sites (393 mg kg^{-1}) was much larger than the range among STP values (15 mg kg^{-1}). The PSI at the heavily manured PON site (49 mg kg^{-1}) was significantly lower than at all other sites with a range from 0 to 160 mg kg^{-1} (Appendix 5). There were no significant differences in PSI among years at either of the manured sites (Appendix 5), even though no manure was applied to the LLB site in 2003 or to the PON site in the fall of 2003 or 2004. These findings are similar to those of Carefoot et al. (2000), where the phosphorus sorption capacities of soils that had received manure for 16 yr did not increase even 9 yr after manure applications were discontinued.

The DPS in the 0- to 2.5-cm layer in the fall of 2003 was significantly lower at the ungrazed grassland STV site (5%) than at any other sites. The GPC site (10%) had significantly lower DPS than the other cultivated sites. Values of DPS were not significantly different among the remaining non-manured sites. The DPS was significantly higher at the heavily manured PON site (91%, Fig. 8) than at all other sites. Comparison of the DPS in soils sampled in the fall of 2002, 2003, and 2004 at the two manured sites showed no significant differences among years. Relative changes in PSI and DPS with increased depth of soil layer are described by Nolan et al. (2005).

An environmentally significant change point, where the rate of phosphorus release from Alberta soils increases from a linear to a quadratic slope, was found to be at DPS values of 47% (using the WEP desorption method, Casson et al. 2006) and 59% (using the $CaCl_2$-P desorption method, Casson et al. 2006). Casson et al. (2006) found similar change points in a study of 13

Alberta soils, which included the microwatershed soil samples. These change points correspond to STP values that are close to the agronomic threshold of 60 mg kg^{-1} (Howard 2006). The STP levels at the two manured sites exceeded these change points.

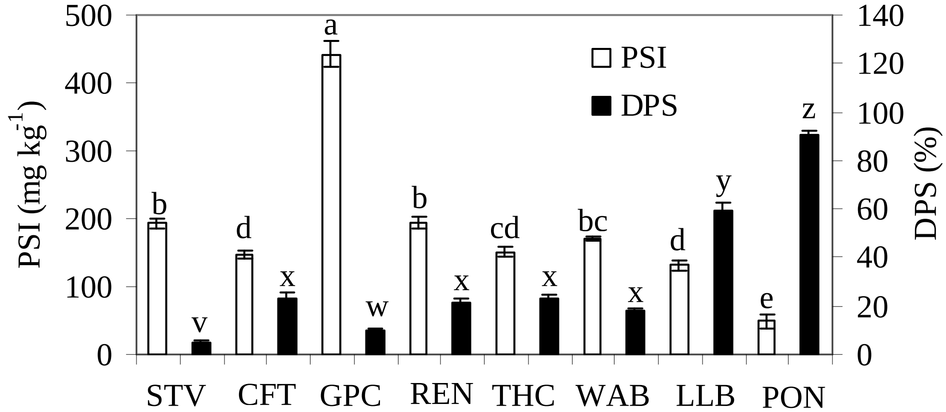

Fig. 8. Phosphorus sorption indices (PSI) and degree of soil phosphorus saturation (DPS) in the 0- to 2.5-cm soil layer at the microwatershed sites in the fall of 2003. Standard error bars are shown. For each parameter (PSI or DPS), bars with the same letter are not significantly different ($P \leq 0.05$).

Runoff Results

Hydrographs. Spring runoff occurred at all sites in 2003, at all sites except the LLB and PON sites in 2004, and at all sites except the LLB and STV sites in 2005 (Table 9, Appendix 6). Winter precipitation in 2003 ranged from more than 20% below normal at the STV and LLB sites to more than 70% above normal at the GPC, PON, and THC sites. In 2004, winter precipitation was well below normal at most sites and around normal for the THC and WAB sites. Winter precipitation in 2005 was near normal at most sites, except for the LLB site (-45%) and the THC site (+50%) (Table 10). Spring runoff started as early as mid-February in 2004 and as late as early April in 2003, starting at the southernmost sites and moving progressively north. Spring runoff was continuous at most sites in 2003, but had two or three phases at most sites in 2004 and 2005 due to intervening cold periods.

Total and mean flow volumes for the spring runoff period were least at the STV site for all 3 yr, while the greatest total volumes were observed at the CFT site in 2003, the REN site in 2004, and the GPC site in 2005. The four northernmost sites (THC, PON, WAB, and GPC) had their greatest flow volumes in 2005, while the three southernmost sites (LLB, STV, and CFT) had their greatest flow volumes in 2003. Nearly all sites had their lowest flow volumes in 2004, except for the STV site, which had no spring runoff in 2005, and the REN site, which was disturbed by the construction of a natural gas well. However, measurements at the REN site in 2004 were likely overestimated due to water entering the rear of the flume following road construction within the site. Flows were underestimated at the REN site in 2003 due to the

Table 9. Hydrological characteristics of the spring and summer runoff events.

Site	Duration of runoff (days)	Start of runoff	Mean flow ($m^3 s^{-1}$)	Total flow volume (m^3)	Runoff depth (mm)	Number of samples	Duration of runoff (days)	Number of events	Mean flow ($m^3 s^{-1}$)	Total flow volume (m^3)	Runoff depth (mm)	Number of samples
	Spring 2003						Summer 2003					
					Ungrazed grassland site							
STV	4	14-Mar	0.0009	406	20.3	13						
					Non-manured sites							
CFT	12	14-Mar	0.0760	59 320	23.9	139						
GPC	13	13-Apr	0.0530	25 858 [z]	41.7	157						
REN	12	22-Mar	0.0140	4 655 [y]	17.9	112	1	1	NA [x]	NA	NA	2
THC	10	23-Mar	0.0060	6 503	12.8	99						
WAB	9	11-Apr	0.0070	4 517	13.7	66						
					Manured sites							
LLB	7	15-Mar	0.0025	1 140	1.3	78	24	12	0.0027	1 828	2.1	141
PON	12	7-Apr	0.0025	2 044	6.8	118						
	Spring 2004						Summer 2004					
					Ungrazed grassland site							
STV	1	18-Mar	$3x10^{-6}$	0.23	0.01	2						
					Non-manured sites							
CFT	13	9-Mar	NA	NA	NA	141						
GPC	6	3-Apr	0.0070	5 139	8.3	50	19	5	0.0101	12 766	20.6	232
REN	13	25-Feb	0.0080	9 708	37.3	130	4	3	0.0023	124	0.5	7
THC	18	12-Mar	0.0060	7 457	14.6	196						
WAB	4	3-Apr	0.0085	2 321	7.0	49						
					Manured sites							
LLB	12	22-Feb	$6x10^{-5}$	253	0.02	127	12	9	0.0012	548	0.6	120
PON						0	2	2	0.0010	143	0.5	9
	Spring 2005						Summer 2005					
					Ungrazed, grassland site							
STV						0	3	2	NA	NA	NA	3
					Non-manured sites							
CFT	22	26-Feb	0.0056	9 740	3.9	215						
GPC	18	8-Mar	0.0310	37 934	61.2	170						
REN	13	1-Mar	0.0029	6 882	26.4	112	7	5	0.0054	772	3.0	31
THC	14	4-Mar	0.0085	10 640	20.9	169	2	2	0.0013	98	0.2	11
WAB	16	5-Mar	0.0070	6 025	18.2	174						
					Manured sites							
LLB						0	19	7	0.0122	11 238	6.1	122
PON	15	7-Mar	0.0204	21 521	71.7	153	1	1	0.0022	40	0.1	1

[z] Includes data up until 23 Apr 2003, though flow continued after this date.
[y] Includes data up until 30 Mar 2003, though flow continued after this date.
[x] Not available.

Table 10. Precipitation differences from 30-yr normal data for each microwatershed site.

Site	Winter difference from normal precipitation (%) [z] 2003	2004	2005	Summer difference from normal precipitation (%) 2003	2004	2005
			Ungrazed grassland site			
STV	-38	-61	-14	-42	74	88
			Non-manured sites			
CFT	14	-21	-6	-30	-17	35
GPC	91	-5	-1	-44	77	-11
REN	8	-44	-9	-55	-5	19
THC	73	11	50	-59	23	5
WAB	64	3	-7	-47	47	-6
			Manured sites			
LLB	-23	-42	-45	-75	-1	74
PON	86	-43	-3	-48	-28	-16

[z] Positive values are percent greater than the 30-yr normals and negative values are percent less than the 30-yr normals for each site. Data were provided by Environment Canada (2005).

washout of the wooden drop box surrounding the flume. Datalogger failure resulted in missing or underestimation of spring flows at the CFT site in 2004, the GPC site in 2003, and the STV site in 2005.

The lack of spring runoff at the PON site in 2004 was due to a combination of lower precipitation levels in the winter of 2004 and deep tillage in the fall of 2003 that increased snowmelt infiltration. Deep tillage involved ripping the soil in the low lying areas of the field to 46-cm depths using a Paraplow, followed by deep cultivation to a 15-cm depth that left the soil surface very rough. At the LLB site in 2004, flows were minimal and were generated exclusively by a snowdrift along the plywood berm at the edge of the field. It is unlikely that any runoff would have been generated without the berm, which was installed with the site instrumentation (Fig. 4). No runoff was observed at the LLB site in the spring of 2005 due to lack of snow cover. Much of the snow at the LLB site likely sublimated in all 3 yr due to the low snowfall levels and the rapid temperature increases due to the Chinook winds that are prevalent in the region.

Although winter precipitation was below normal at the STV site in all 3 yr, the low runoff volumes were surprising given the steepness of the site. In comparison with grazed watersheds near the site, this site produced 10% or less of the runoff produced from the grazed watersheds (Mapfumo et al. 2002), even though this site had more snow accumulation than the grazed sites. The increased snow accumulation was attributed to the abundant litter cover, which was more heterogeneous and five to seven and a half times greater in mass than at the grazed sites (W. Willms, Agriculture and Agri-Food Canada, personal communications). The litter cover may have also increased infiltration; however, it is unlikely that it would account for that much of the observed decrease.

Runoff volumes were very high at the GPC and PON sites in 2005, due to average snow accumulation volumes and a rapid melt. Unfortunately, the peaks of the hydrograph were missed from these sites due to the binding of the float potentiometer within the vertical pipe at high flows. Therefore, the total flow volumes were underestimated at both of these sites in the spring of 2005. Reports from the local newspaper suggest that the runoff observed at the GPC site caused local flooding (Daily Herald Tribune – March 11, 2005) and field observations indicated that the water depth at the site was at least 1 m deep. The rapid temperature change was likely responsible for the large volumes as precipitation levels were around normal at both sites in 2005.

Summer events were much less common than spring events, occurring at only two sites in 2003, four sites in 2004, and five sites in 2005. Summer precipitation was below normal at all sites in 2003, but was above average at the northernmost sites (GPC, WAB, THC) and the STV site in 2004, and at the southernmost sites in 2005 (LLB, STV, CFT, REN) (Table 10). Only the PON site had below average summer precipitation in all 3 yr.

Most runoff events at the LLB site were of short duration (less than 12 h), had low flow rates, and were generated as a result of irrigation with a center-pivot sprinkler system. Although the size of the irrigation events was relatively small, they generated the majority of runoff in 2003 and 2004. In addition, summer runoff volumes at the LLB site were greater than the volume of spring snowmelt runoff in all 3 yr. Irrigation accounted for 57% of the runoff in 2003, 68% of the runoff in 2004, but only 8% of the runoff in 2005. Summer rainfall runoff was very high in 2005 when two greater than 1-in-100 yr rainfall events were recorded in the LLB region. More than 250 mm of rain was recorded in June compared with the 30-yr normal value of 53 mm, with another 137 mm recorded in September compared with the 30-yr normal value of 38.3 mm. These two events from June 5 to 9 and September 10 to 14 accounted for 85% of the runoff volume at the LLB site in 2005.

Summer events accounted for 71% of the total runoff at the GPC site in 2004; however, at the remainder of the sites, summer runoff was relatively minor, ranging from 0.18% at the PON site in 2005 to 10% at the REN site in 2005. Two sites (CFT and WAB) generated no summer runoff. Overall, summer events accounted for slightly less than 10% of all runoff during the 3-yr study. The relatively minor contribution of summer precipitation events to the phosphorus exported by overland flow compared to spring runoff is typical of cold climates in the western Canadian prairies. Nicholaichuk (1967) estimated that 80% of the runoff from two small watersheds in Saskatchewan was generated by spring snowmelt. In Alberta, total yearly runoff from small agricultural watersheds tends to be dominated by snowmelt (Gill et al. 1998; Wuite and Chanasyk 2003; Ontkean et al. 2005). For the majority of the microwatersheds, spring snowmelt runoff was still the predominant contributor to runoff.

Spring phosphorus concentrations. In 2003, the DRP FWMCs at the non-manured sites ranged from 0.01 mg L^{-1} at the GPC site to 0.63 mg L^{-1} at the THC site (Table 11). Although there was some variability in the concentrations among years, the ranking of DRP FWMCs was consistent each year, with the GPC site having the lowest DRP FWMC and the THC site the highest DRP FWMC. The low DRP FWMC at the GPC site may have been due to the low DPS (Fig. 8). This site had high clay content and may therefore have had more exchange sites available to bind phosphorus. The high DRP FWMC at the THC site may be due to significantly higher concentrations of STP in the lower landform positions.

Table 11. Minima, maxima, and flow-weighted mean concentrations (FWMC) of dissolved reactive phosphorus (DRP) and total phosphorus (TP) from all sites.

Site	DRP						TP					
	Min. (mg L^{-1})	Max. (mg L^{-1})	FWMC (mg L^{-1})	Min. (mg L^{-1})	Max. (mg L^{-1})	FWMC (mg L^{-1})	Min. (mg L^{-1})	Max. (mg L^{-1})	FWMC (mg L^{-1})	Min. (mg L^{-1})	Max. (mg L^{-1})	FWMC (mg L^{-1})
	----- Spring 2003 -----			----- Summer 2003 -----			----- Spring 2003 -----			----- Summer 2003 -----		
	Ungrazed grassland site											
STV	0.08	0.28	0.18				0.27	0.70	0.52			
	Non-manured sites											
CFT	0.06	0.87	0.24				0.13	1.20	0.38			
GPC	<DL [z]	0.09	0.01				0.05	0.66	0.20			
REN [y]	0.05	0.38	0.21	0.04	0.05	0.05	0.13	0.93	0.41	0.49	0.50	0.50
THC	0.21	1.31	0.63				0.44	1.63	0.77			
WAB	0.12	0.29	0.20				0.25	0.86	0.58			
	Manured sites											
LLB	1.43	4.73	3.44	0.22	3.65	2.15	2.03	8.24	3.94	1.52	5.29	2.86
PON	9.42	24.00	16.5				17.00	42.2	24.0			
	----- Spring 2004 -----			----- Summer 2004 -----			----- Spring 2004 -----			----- Summer 2004 -----		
	Ungrazed grassland site											
STV	0.09	0.09	0.09				0.18	0.20	0.19			
	Non-manured sites											
CFT	0.03	0.40	0.17				0.12	0.91	0.30			
GPC	0.03	0.15	0.08	<DL [z]	0.20	0.09	0.23	0.55	0.34	0.08	1.35	0.36
REN [y]	0.07	0.69	0.16	0.02	0.08	0.03	0.20	1.84	0.74	0.49	2.91	1.78
THC	<DL [z]	3.07	0.31				0.26	5.45	0.52			
WAB	0.12	0.25	0.18				0.21	0.39	0.30			
	Manured sites											
LLB	0.50	2.53	0.66	0.57	2.54	0.87	0.80	2.89	1.15	0.97	3.38	1.84
PON				5.43	7.10	6.28				5.63	7.51	6.59
	----- Spring 2005 -----			----- Summer 2005 -----			----- Spring 2005 -----			----- Summer 2005 -----		
	Ungrazed grassland site											
STV				0.04	0.09	0.06				0.09	0.15	0.10
	Non-manured sites											
CFT	0.07	1.34	0.30				0.26	2.38	0.53			
GPC	<DL [z]	0.35	0.19				0.14	0.98	0.41			
REN [y]	0.23	1.61	1.04	0.05	0.46	0.22	0.67	1.92	1.54	0.28	7.02	1.38
THC	0.18	1.76	0.53	0.03	0.26	0.10	0.41	2.89	0.86	0.67	2.87	1.57
WAB	0.06	0.18	0.14				0.23	0.52	0.41			
	Manured sites											
LLB				0.80	3.74	2.63				1.13	5.19	3.54
PON	4.74	11.6	7.39	5.25	5.25	5.25	5.3	12.9	8.00	5.26	5.26	5.26

[z] Below laboratory detection limits.
[y] Site contaminated by gas well access road construction prior to runoff in the spring of 2004.

Levels of DRP were greatest at the manured sites in the spring of 2003 following the manure applications in the fall of 2002. Individual values at the PON site in the spring of 2003 were as high as 24 mg L^{-1}, with a FWMC of 16.5 mg L^{-1} DRP. These extremely high values can be attributed to the application of manure close to freeze up in the fall of 2002. The manure was very poorly incorporated and visible on the surface at the time of soil sampling. In addition, the PON site had a very high DPS, suggesting that it had little capacity to bind phosphorus and was more likely to release phosphorus to overlying water. In contrast, the LLB site had manure applied to the portion of the microwatershed nearest the outlet in the spring of 2002, which allowed greater opportunity for phosphorus to be adsorbed by soil and mixed with the subsurface soil by intensive tillage following spring manure application and fall harvest. As such, the DRP FWMCs values at the LLB site were an order of magnitude lower than at the PON site. Previous studies have indicated that when soils have received surface applications of manure, the manure phosphorus overwhelms the soil phosphorus and becomes the major source of phosphorus to runoff instead of the soil (Pierson et al. 2001; Kleinman et al. 2002). Therefore, STP of freshly manured soil is often not an accurate representation of runoff-available phosphorus. However, the differences between amended and un-amended soils are much less if the manure has been incorporated (Kleinman et al. 2002) or has had time to equilibrate with the soil (Eghball et al. 2002).

The DRP FWMC from the STV site in 2003 was within the range of the non-manured sites, despite an STP level that was about one-third of the non-manured sites. Concentrations were comparable to those reported by Timmons and Holt (1977) from native grasses in Minnesota. The relatively high values may be due to leaching of DRP from the large amounts of vegetation cover and surface thatch at this site. In addition, freezing and thawing of plant material dramatically increases the amount of nutrients that can be leached (Timmons et al. 1970; Bechmann et al. 2005). In 2004, the DRP concentration at the STV was much lower, possibly due to the much smaller volume of runoff and decreased interaction with the vegetation cover.

The TP FWMC of the non-manured sites ranged from 0.20 mg L^{-1} at the GPC site in 2003 to 1.54 mg L^{-1} at the REN site in 2005. Although it was hoped that phosphorus and sediment concentrations in the runoff at the REN site would stabilize following the construction of the gas well access road in 2004, minimum TP values in 2005 were higher than the FWMC in 2003. Consequently, the water chemistry results from the REN site in 2004 and 2005 were deemed to be outliers. After these points were removed from the dataset, the TP FWMC showed a similar pattern to the DRP FWMCs, with the highest concentrations observed at the THC site in 2005. Not surprisingly, DRP and TP FWMCs are highly correlated ($r^2 = 0.99$, df = 19).

The TP FWMC at the heavily manured PON site was extremely high in the spring of 2003, with some individual TP values exceeding 40 mg L^{-1}. Total suspended solids (TSS) concentrations were also elevated and accumulation of sediment in the flume was observed during field visits, indicating that selective sampling of sediment from the H-flume may have been an issue at the PON site. Therefore, samples with extreme TSS concentrations and TP:DP ratios greater than 10 were deemed to be outliers and removed from the dataset. The replacement of the H-flume in the spring of 2003 mitigated these concerns. Even with these extreme values removed, the spring 2003 TP FWMC was still three times greater than from any other runoff event. Given the high TP FWMC and the poor incorporation following the recent manure application, it is likely that the STP was overwhelmed by the phosphorus content of the manure and was not representative of soil phosphorus conditions.

The TP FWMCs at the STV site were quite high in 2003 and much lower in 2004; however, the 2003 results were comparable to another study on native range in Minnesota (Timmons and Holt 1977).

Flow-weighted mean concentrations of DRP and TP were similar to the mean concentrations. Differences between the DRP mean and DRP FWMC averaged 0.04 mg L^{-1} at the non-manured sites and 0.26 mg L^{-1} at the manured sites, while TP differences averaged 0.08 mg L^{-1} at the non-manured sites and 0.42 mg L^{-1} at the manured sites. Although several sites had elevated TP concentrations at the beginning of the runoff event when flows were lower (Appendix 7), the differences between FWMCs and means were minor at most sites because flow and concentration were unrelated (data not shown). The exception was the CFT site in 2003 where DRP and TP FWMCs were reduced by 40% and 30%, respectively, compared with means. At this site, there was an inverse correlation between concentration and flow.

Ratios of DRP:TP in snowmelt varied widely at the non-manured sites, ranging from 0.08 at the GPC site in 2003 to 0.81 at the THC site in 2003 (Table 12). The low proportion of DRP at the GPC site may be due to the low DPS that is related to higher clay contents at this site. Low ratios were also observed at the REN site in 2004, which was due to high losses of PP caused by the recent construction of a service road at this site. Mean values for snowmelt runoff at the non-manured sites were about 0.45, with higher proportions of DRP observed in snowmelt runoff at the manured sites (0.79).

Very little phosphorus was in particulate form at the manured sites, with the exception of the PON site in 2003, despite intensive cultivation at these sites. This result was likely due to the low flow rates observed at the LLB site in conjunction with the application of manure, which promotes infiltration and reduces erosion. At the REN and THC sites, the ponding of water in front of the flume due to flow restriction by the downstream culvert may have caused PP to settle out before it reached the flume. This problem was minimized after 2003 by adjusting the placement of the ISCO intakes. At the CFT site, ponding of water in front of the flume occurred at the start of runoff in 2003 and 2004, but was resolved when a downstream culvert that restricted flow was cleared of ice and snow. The ponding of water at these sites may have increased the DRP:TP ratio in these years; however, differences in the DRP:TP ratio were only observed at the REN and THC sites, and these sites were likely also impacted by management changes (i.e., road construction at the REN site).

Since erosion is the dominant process in the transport of PP, many factors other than flow volume or flow rate can play a role in its transport in comparison with dissolved fractions. Recent tillage can increase the amount of PP transported by runoff, as well as increased slope. However, soil factors were likely minimized due to frozen conditions at the onset of spring-melt runoff, which would restrict infiltration as well as the zone of interaction between the runoff and the soil (Hansen et al. 2000). In addition, the low energy of the snowmelt runoff may not move large quantities of PP (Hansen et al. 2000). Higher PP concentrations occasionally occurred towards the end of the runoff period when greater contact between runoff water and unfrozen

Table 12. Ratio of dissolved reactive phosphorus (DRP) flow-weighted mean concentration (FWMC) to total phosphorus (TP) FWMC by runoff type.

Site	Snowmelt 2003	Rainfall 2003	Snowmelt 2004	Rainfall 2004	Snowmelt 2005	Rainfall 2005
			Ungrazed grassland site			
STV	0.35		0.47			0.55
			Non-manured sites			
CFT	0.63		0.56		0.56	
GPC	0.08		0.21 [z]	0.18	0.23	
REN	0.51	0.10	0.22 [y]	0.03 [y]	0.68	0.21
THC	0.81		0.60		0.62	0.07
WAB	0.34		0.58		0.34	
			Manured sites			
LLB	0.86 [z]	0.76 [x]	0.57	0.45 [x]		0.76 [x]
PON	0.58			0.95	0.92	1.00

[z] Includes spring runoff plus additional snowmelt events.
[y] Site contaminated by gas well access road construction prior to runoff in the spring of 2004.
[x] Includes irrigation events.

soils may have occurred (CFT – 2003, 2004; GPC – 2003, 2005; THC – 2005; PON – 2005; Appendix 7). At the manured sites, manure application may protect soils from erosion by increasing organic matter levels and thus infiltration.

Summer phosphorus concentrations. At the non-manured sites, DRP FWMC ranged from 0.03 mg L^{-1} at the REN site in 2004 to 0.20 mg L^{-1} at the REN site in 2005 (Table 11). Compared with spring DRP FWMCs from the same site, summer DRP FWMCs were similar or lower. Dissolved reactive phosphorus FWMCs at the manured sites were much more variable, with individual event FWMCs ranging from 0.84 to 3.01 mg L^{-1} at the LLB site and from 5.25 to 6.19 mg L^{-1} at the PON site. Comparisons between spring and summer events were difficult to make at the manured sites because only the LLB site in 2003 and PON in 2005 had spring snowmelt runoff and summer runoff. In both of these cases, summer DRP FWMCs were lower than spring values. Summer 2003 and 2005 runoff values from the LLB were much greater than in 2004 as the portion of the site nearest the outlet was manured in the spring of 2002 and the fall of 2004.

In contrast with the DRP FWMCs, TP FWMCs were higher in summer runoff at the non-manured sites than in spring runoff, with values ranging from 0.33 mg L^{-1} at the GPC site in 2004 to 2.80 mg L^{-1} at the REN site in 2005 (Table 11). Phosphorus in summer runoff was dominated by particulate fractions, with DRP:TP ratios ranging from 0.07 to 0.27 at the GPC site (Table 12). This may be related to greater sediment losses from the increased erosion of unfrozen soils and/or greater precipitation intensity from rainfall compared to snowmelt.

At the manured sites, summer TP FWMCs were lower than those recorded during spring runoff and the LLB and PON sites were dominated by dissolved fractions, with DRP:TP ratios averaging 0.73 and 0.96, respectively. The summer TP FWMC in 2004 represented only 27% of the TP measured at the PON site during the 2003 spring runoff; however, no data were available from spring 2004 for comparison due to the lack of runoff. This decline was directly related to decreased levels of STP in the spring of 2004 relative to the fall of 2002 due to the equilibration of the manure with the soil, dilution by tillage, and crop uptake. As well, the greater tillage depth and intensity at the PON site in the fall of 2003 and the lower flow volumes in summer events may have resulted in less transport of PP, as suggested by the large proportion of DRP relative to TP (Table 12).

The LLB site showed declines in TP FWMCs following spring runoff in 2003 until further applications of manure to the east half of the field in fall 2004 and the west half of the field in spring 2005. There were also differences between types of runoff, with rainfall events having higher DRP FWMCs than irrigation events, but similar TP FWMCs. A greater proportion of TP was DRP (DRP:TP ratio = 0.93, n = 7) in the rainfall runoff compared with irrigation events (DRP:TP ratio = 0.67, n =19) ($t = -5.12$, $df = 24$, $P<0.001$).

At the REN site, summer events yielded TP FWMCs that were similar to those at the manured LLB site, suggesting that summer events were also impacted by the road construction in 2004 (Table 11). Total phosphorus concentrations were greater in 2005 than in 2004. Fertilizer that could not be incorporated due to soil compaction was observed adjacent to the road and may have contributed to increased phosphorus concentrations in runoff.

Lower FWMC of phosphorus in runoff from summer events compared with spring snowmelt events have been noted in other studies, especially for dissolved nutrient fractions (Burwell et al. 1975; Wright et al. 2003). This trend may be due to reduced interaction of the runoff with the soil in the spring runoff event (i.e., a dilution effect). This effect was noted at the non-manured sites, except at the GPC site where DRP FWMC concentrations were similar in the spring and summer. Hansen et al. (2002) reported a reduced proportion of TP as DRP in summer rainfall runoff relative to spring runoff, and they attributed this to the decreased flow energy of spring runoff to detach and transport smaller amounts of PP. The frozen soil surface in spring conditions would also reduce levels of PP in runoff since detachment and entrainment of soil particles is likely minimized due to decreased infiltration and increased contact with snow pack rather than soil (Hansen et al. 2000). Greater TP FWMCs from summer events compared with spring runoff events were observed at most sites, except for the manured sites. The elevated spring runoff concentrations at the manured sites may have been due to the recent applications of manure (mostly in the fall), and the subsequent decreases in runoff phosphorus concentrations were likely due to the longer period of time for manure to equilibrate with the soil. These changes likely obscured any seasonal differences. Although direct comparisons with spring results were not always possible at the LLB and PON sites due to the lack of spring runoff, the high proportion of DRP in summer runoff compared with TP was also not consistent with findings at the other sites. The lower flow volumes, which were likely influenced by the deep tillage at the PON site, may have reduced PP losses, while the increased organic matter may have promoted infiltration.

As for the spring events, FWMCs of DRP and TP were similar to mean concentrations. On average, concentrations of TP differed from TP FWMCs by 0.03 mg L^{-1} at the non-manured sites and 0.21 mg L^{-1} at the manured sites, while differences in DRP were 0.02 mg L^{-1} at the non-manured sites and 0.26 mg L^{-1} at the manured sites. Although correlations between TP and flow were observed for a few events, these were not consistent among sites (data not shown).

Regardless of how runoff was generated, concentrations of TP exported from the microwatersheds exceeded the in-stream Alberta water quality guideline of 0.05 mg L^{-1} TP for the protection of aquatic life (Alberta Environment 1999). It should be noted that this guideline was developed for third- and fourth-order streams, which are much larger than our ephemeral first-order streams and may not be directly applicable to field scales. The TP in runoff from the non-manured sites exceeded the guideline by 4 to 16 times, while TP in runoff from manured sites was several orders of magnitude above this guideline. Even runoff from the ungrazed native prairie site exceeded the 0.05 mg L^{-1} TP guideline by 2 to 10 times in all years of the study. Anderson et al. (1998) reported that streams within watersheds with low intensities of agricultural development have phosphorus concentrations above the 0.05 mg L^{-1} guideline. As such, it appears that applying the 0.05 mg L^{-1} TP water quality objective to field scales may not be appropriate.

The TP FWMC values from the non-manured sites were close to previously reported median values of 0.5 mg L^{-1} (Anderson et al. 1998) and 0.4 mg L^{-1} (Depoe 2004) in first-order streams that drain high intensity agricultural watersheds in Alberta. The TP FWMCs from the non-manured sites were less than the 1.0 mg L^{-1} TP permitted from federal waste-water treatment facilities (Environment Canada 2000). However, levels of TP from the manured sites exceeded this guideline by 2 to 24 times.

Relating Phosphorus Concentrations in Soil and Runoff

Soil-test phosphorus and runoff phosphorus relationships. Results from spring and summer runoff events were included for analysis of the relationship between site mean STP and FWMCs of DRP and TP. Seasonal flow-weighted averages were calculated by summing the loads from all spring or summer events and dividing them by the flow during that period. Spring snowmelt runoff results were related to the soil sampling results from the previous fall, while summer runoff events were related to the soil sampling results from the spring of the same year. For comparison of the five STP representations, only fall STP values and spring runoff events were used since the subsets of spring soil sampling points were too small for the application of the STP representations. The 2004 and 2005 data from the REN site were excluded since comparison of the results with those measured in 2003 indicated that the gas well access road construction in spring 2004 caused abnormally high concentrations of TP in the spring and summer runoff. Data from spring snowmelt in 2004 at the LLB site were also excluded as runoff was generated exclusively from a snowbank at the edge of the field and would not have occurred without the berm constructed to direct runoff into the flume. Results from the spring runoff in 2003 at the PON site were also excluded due to the recent application of manure that was poorly incorporated due to the frozen soil conditions and the selective sampling of sediment by the ISCO due to sediment accumulation in the H-flume.

Table 13. Slopes, intercepts, and coefficients of determination for the relationships between the soil-test phosphorus (STP) representations in the 0- to 15-cm layer and the flow-weighted mean concentrations (FWMCs) of dissolved reactive phosphorus (DRP) and total phosphorus (TP) in snowmelt runoff from the seven cultivated sites.

	DRP FWMC			TP FWMC		
Representation of STP	Slope	Intercept	r^2	Slope	Intercept	r^2
Mean of all point data	0.019	-0.340	0.94	0.020	-0.154	0.95
Landform area weighted	0.018	-0.315	0.93	0.020	-0.129	0.93
Runoff contributing area	0.017	-0.249	0.99	0.018	-0.055	0.99
Representative random	0.018	-0.311	0.96	0.019	-0.123	0.96
Random	0.019	-0.316	0.96	0.020	-0.129	0.96

Strong linear relationships (r^2 = 0.93 to 0.99) were found between all representations of fall STP values in the 0- to 15-cm layer and DRP and TP FWMCs in spring runoff (Table 13). Coefficients of determination improved slightly using the runoff contributing area mean, but were lower from the landform area-weighted mean. Reports in the literature have suggested that soils in the lower landform positions can have greater influence on phosphorus loss in runoff than soils in other landform positions (Gburek and Sharpley 1998); however, in our study, there were no significant differences among the regression equations for all STP representations. This was partly due to the observation that few differences in STP by landform position were detected and that variable management practices, such as the uneven distribution of manure at the LLB site or conventional tillage at the GPC and WAB sites, may have obscured differences in STP by landform position. The similarity between regression equations may also be partly attributed to the observation that spring runoff was generated from a greater proportion of the field due to the restricted infiltration on frozen soils compared with summer precipitation events. Regressions using representative random sampling and a random subset of samples also produced similar results to using the site mean. Results of the regression analysis among STP representations were similar for all three soil layers (data not shown). Differences among the five STP representations were minimal (Table 8), and this was reflected in the similarity among regression results. Because there were no differences among the STP representations, it was decided that only the site mean STP values be used for further analysis of the STP and runoff phosphorus relationships.

Strong linear relationships were also found when spring and summer runoff results were combined for the comparison with site mean STP (Fig. 9). Although regression slopes and coefficients of determination were lower compared to values in Table 13, the distribution of the data improved when spring and summer events were included, especially for the higher values of STP.

Although previous studies have found that surface runoff interacts with only a very shallow depth of soil (Sharpley et al. 1978; Sharpley 1985), the relationships developed in our study had similar predictive ability among the soil layers (Fig. 9). Statistical comparisons of the relationships indicated that the slopes and intercepts of the relationships for all three layers were not significantly different, although slopes and intercepts tended to increase with increasing

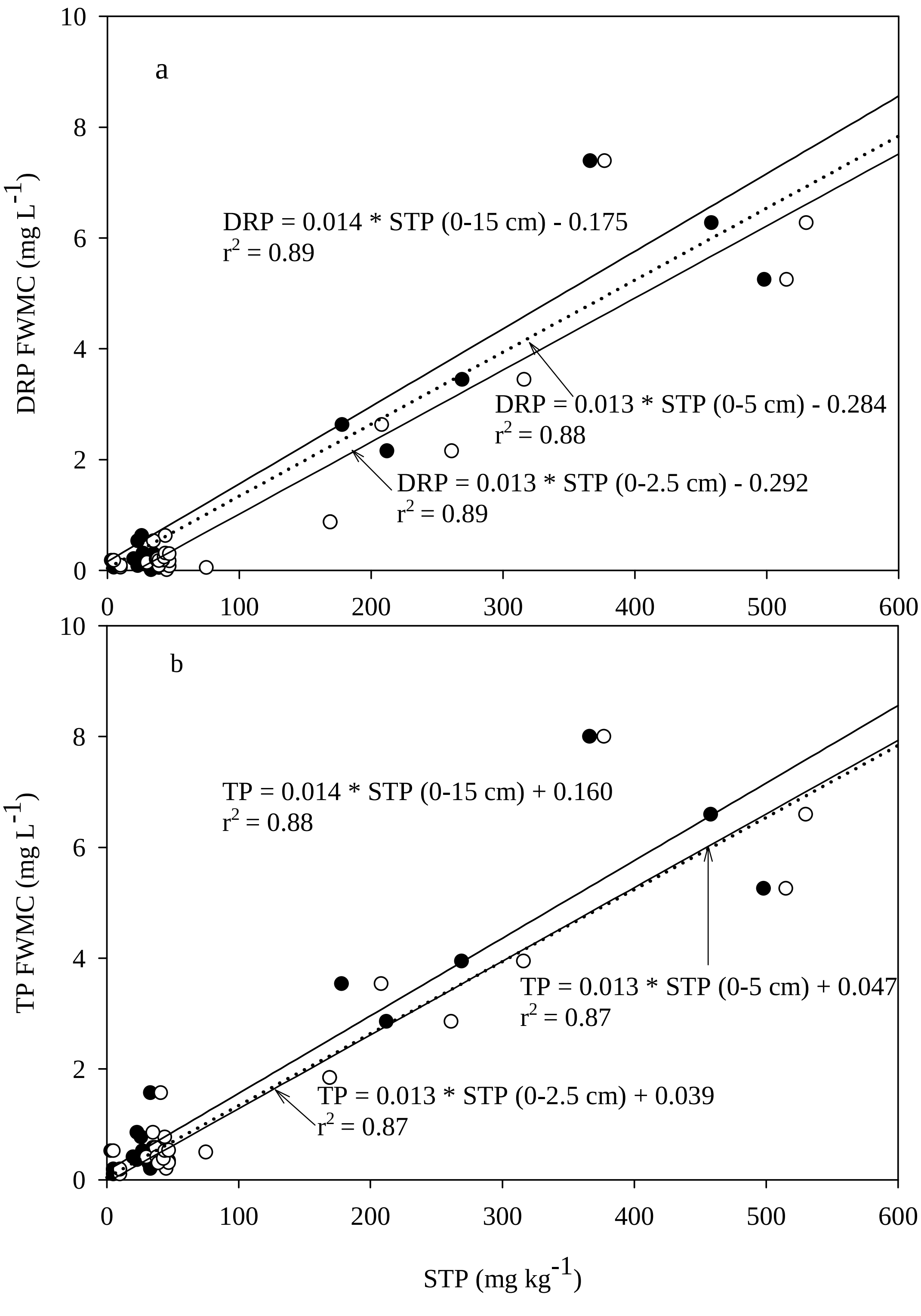

Fig. 9. Relationships between soil-test phosphorus (STP) and the flow-weighted mean concentration (FWMC) of (a) dissolved reactive phosphorus (DRP) and (b) total phosphorus (TP) from the microwatershed sites. Open circles are 0- to 2.5-cm STP values, while closed circles are 0- to 15-cm STP values. The 0- to 5-cm STP relationship is represented by the dotted line.

depth. It was anticipated that STP from shallower sampling depths may have a stronger relationship with runoff phosphorus because the majority of runoff occurred during spring snowmelt when frozen soil restricts infiltration and minimizes the interaction between runoff and soils. However, given that STP results among all three layers were highly correlated in our study, it was not surprising they predicted runoff phosphorus equally well. Andraski and Bundy (2003) also reported increased slopes for the relationship between STP in the 0- to 15-cm soil layer and DRP in simulated rainfall runoff compared with the 0- to 2-cm soil layer, but concluded that taking account of increased STP levels in the shallow layers did not improve relationships with DRP compared to those measured in the 0- to 15-cm layer. Vadas et al. (2005b) combined data from rainfall simulator studies representing 30 soil types throughout the United States at 0 to 5 cm, 0 to 15 cm and 0 to 20 cm and found that STP measured from shallow samples in phosphorus stratified soils gave a similar assessment of STP available to DRP in runoff as deeper samples in well-tilled soils.

Due to the relatively narrow range of STP among the non-manured sites, the manured sites drive the relationships, regardless of the representation of STP. The distribution of values improved with time since manure was not applied to the PON site after the fall of 2002 or between the fall of 2002 and the fall of 2004 at the LLB site, which resulted in lower STP values from these sites as the manure was incorporated into the soil by tillage and manure phosphorus became equilibrated with the soil. Additional data from summer runoff events that corresponded with increased STP levels at the non-manured sites also helped to improve the distribution of points. However, there were no observations within the STP range of 75 to 150 mg kg^{-1}, as even a single manure application can rapidly increase the STP levels in soil (Ontkean et al. 2006). Corresponding changes in STP and DRP and TP FWMCs at the manured sites support that the relationship is linear and there is no reason to suspect that there would be any deviation from a linear relationship within the range of 75 to 150 mg kg^{-1} of STP.

Analysis of the residuals for the 0- to 2.5-cm STP versus TP equation indicated that four of the residuals were outside the 95% confidence intervals of the regression (data not shown). The regression equation underestimated runoff TP FWMCs at the LLB, PON, and THC sites in the summer of 2005, and overestimated the snowmelt runoff TP FWMC at the PON site in 2005. The summer event at the THC site in 2005 had a very high proportion of PP and therefore, a high TP FWMC. This finding suggests that high-intensity, short-duration summer events may not be well predicted by the model. However, average residuals were similar between rainfall and snowmelt events, suggesting that the equation can be applied to both types of runoff events. Furthermore, given that spring runoff accounts for the majority of runoff in Alberta, the relationship between STP and TP is likely adequate for predicting phosphorus losses during most runoff events. To ensure that the events with large residuals were not unduly influencing the relationships, the regressions were also run without the events with the two largest residuals (from the PON site in 2005). Removal of the two events had only a minor effect on the regression equations, which were not statistically different from the regression with all points. Since data were limited from the manured sites, these points were kept in the dataset.

There have been many studies that have reported strong linear relationships between STP and DRP in simulated runoff at lab and field scales (Wright et al. 2003; Vadas et al. 2005a). However, very few have developed relationships with TP (Schroeder et al. 2004), which combines dissolved and particulate fractions. Particulate phosphorus concentrations can be impacted by several additional factors related to erosion, including tillage (Zhao et al. 2001), event size (Quinton et al. 2001), crop cover, and clay content of the soils (Calhoun et al. 2002). These factors are often difficult to evaluate using lab- or plot-scale rainfall simulation studies because erosion processes operate at larger scales. However, incorporation of an erosion factor to account for PP was not necessary in our study, since 90% of the runoff was generated by spring snowmelt from frozen soils and PP was only a minor component in runoff from manured sites in summer runoff.

The linear relationships can be used to predict the phosphorus concentration in runoff water from given STP values, or determine a required STP level needed to achieve a certain phosphorus concentration in runoff water at the edge of field. For example, Manunta et al. (2000) examined more than 56 000 STP records from the period 1993 to 1997 and found that the majority of ecodistricts in Alberta had a mean STP value between 25 and 30 mg L^{-1} in the top 15 cm of soil. Therefore, using the TP equation for the 0- to 15-cm layer (Fig. 9b), the predicted TP concentration in runoff would be 0.52 mg L^{-1} TP from a field with an STP value of 25 mg kg^{-1}. If the soil contained the agronomic threshold of 60 mg kg^{-1} STP, then the predicted TP in the runoff would be 1 mg L^{-1}.

Degree of phosphorus saturation and runoff phosphorus relationships. The DPS in the 0- to 2.5-cm layer was also related to phosphorus concentrations; however, the relationship was not linear. The relationship was described by a simple exponential equation (Fig. 10), and explained a similar amount of variation as the STP representations (Table 13). Andraski and Bundy (2003) reported that phosphorus saturation explained similar amounts of variability in runoff phosphorus concentrations at one site, but explained less variability than STP at two other sites.

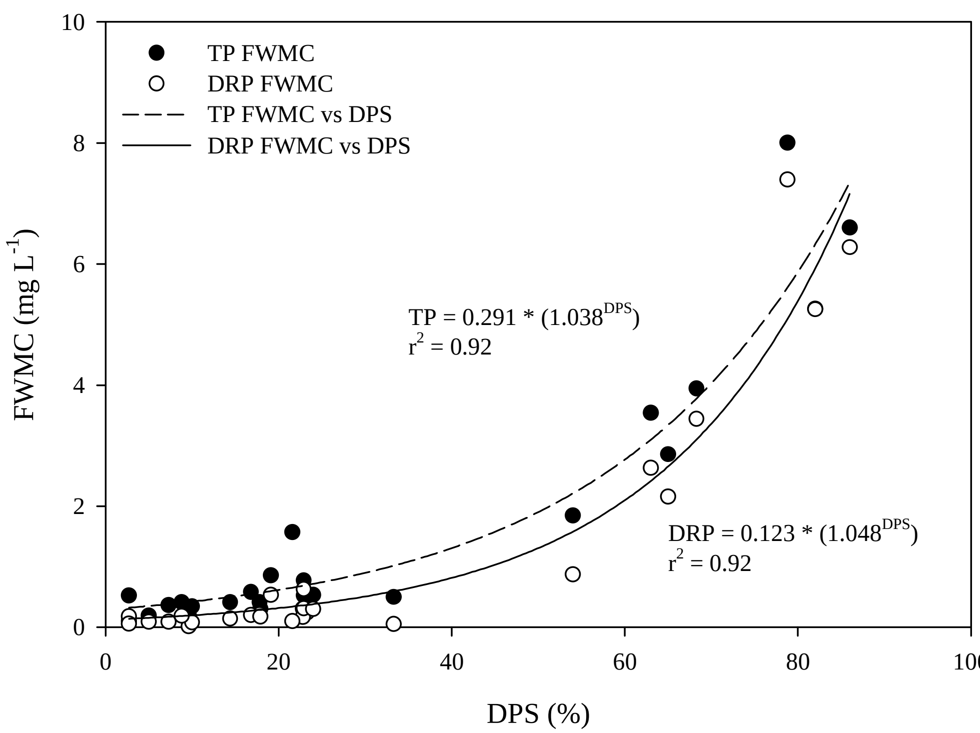

Fig. 10. Relationship between the degree of phosphorus saturation (DPS) and the flow-weighted mean concentration (FWMC) of dissolved reactive phosphorus (DRP) and total phosphorus (TP).

These results support the conclusions of Vadas et al. (2005a) and Andraski and Bundy (2003) that the relationship between STP and DRP concentrations in runoff was not improved using alternative STP extraction methods compared with agronomic sampling methods currently in use. Our results suggest that this may also be true for TP concentrations in runoff.

Comparison with Phase 1 and 2 Results

One of the objectives of this study was to compare the results to the Edge-of-Field Phosphorus Export Model (EFPEM) developed by the Phosphorus Mobility Study during Phase 1 of the Soil Phosphorus Limits Project (Wright et al. 2003), as well as to other rainfall simulation studies from Phase 2. The Phase 1 study included extensive laboratory and field rainfall simulations, as well as limited catchment (natural runoff) monitoring at field scales. Wright et al. (2003) used their catchment data to calculate a scaling factor of 5.9, which they then applied to the laboratory relationship (Equation 3; Wright et al. 2003, 2006) between STP and DRP to derive the EFPEM (Equation 4; Wright et al. 2003).

$$DRP = 0.003 \times STP - 0.044 \qquad (3)$$
$$DRP = 0.018 \times STP - 0.258 \qquad (4)$$

Where:
DRP = flow-weighted mean concentration in equilibrium runoff (mg L^{-1})
STP = soil-test phosphorus concentrations in the 0- to 5-cm layer (mg kg^{-1})

Although the Phase 1 results used STP values for the 0- to 5-cm soil layer, results for the 0- to 2.5-cm and 0- to 5-cm layers were similar in the microwatershed study; therefore, results from the 0- to 2.5-cm layer at the microwatershed sites were used in the comparison.

The STP versus DRP FWMC relationships for the Phase 1 laboratory rainfall simulations and the microwatershed study were strong, with r^2 values equal to or greater than 0.89 (Fig. 11). However, the extraction coefficient for the microwatershed study was significantly greater (0.013; Table 13) than the Phase 1 laboratory rainfall simulation coefficient (0.003; Equation 3). Statistical comparisons could not be made between the microwatershed study and the EFPEM because the EFPEM was derived from the laboratory simulation data; however, the calculated EFPEM coefficient (0.018; Equation 4) was greater than the coefficient for the microwatershed data (Fig. 11). These results suggest that the scaling factor used by Wright et al. (2003) may have been overestimated because of limited and varied field data.

The extraction coefficient for TP for the Phase 1 laboratory rainfall simulation results (0.011; as calculated from the results of Wright et al. 2003) was close to that of the microwatershed study relationship (0.013, Table 13), although the relationship was weaker ($r^2 = 0.27$) and the intercept was significantly greater at the laboratory scale. The DRP fraction was also a very small proportion of TP (0.08) for the laboratory rainfall simulations compared to the DRP:TP ratio of 0.55 for the microwatershed results. Although the bare soil conditions of the laboratory simulations may have contributed to the large proportion of PP, the Phase 1 plot-scale rainfall simulations in cropped conditions also yielded a relatively low ratio of DRP:TP (0.16), as did

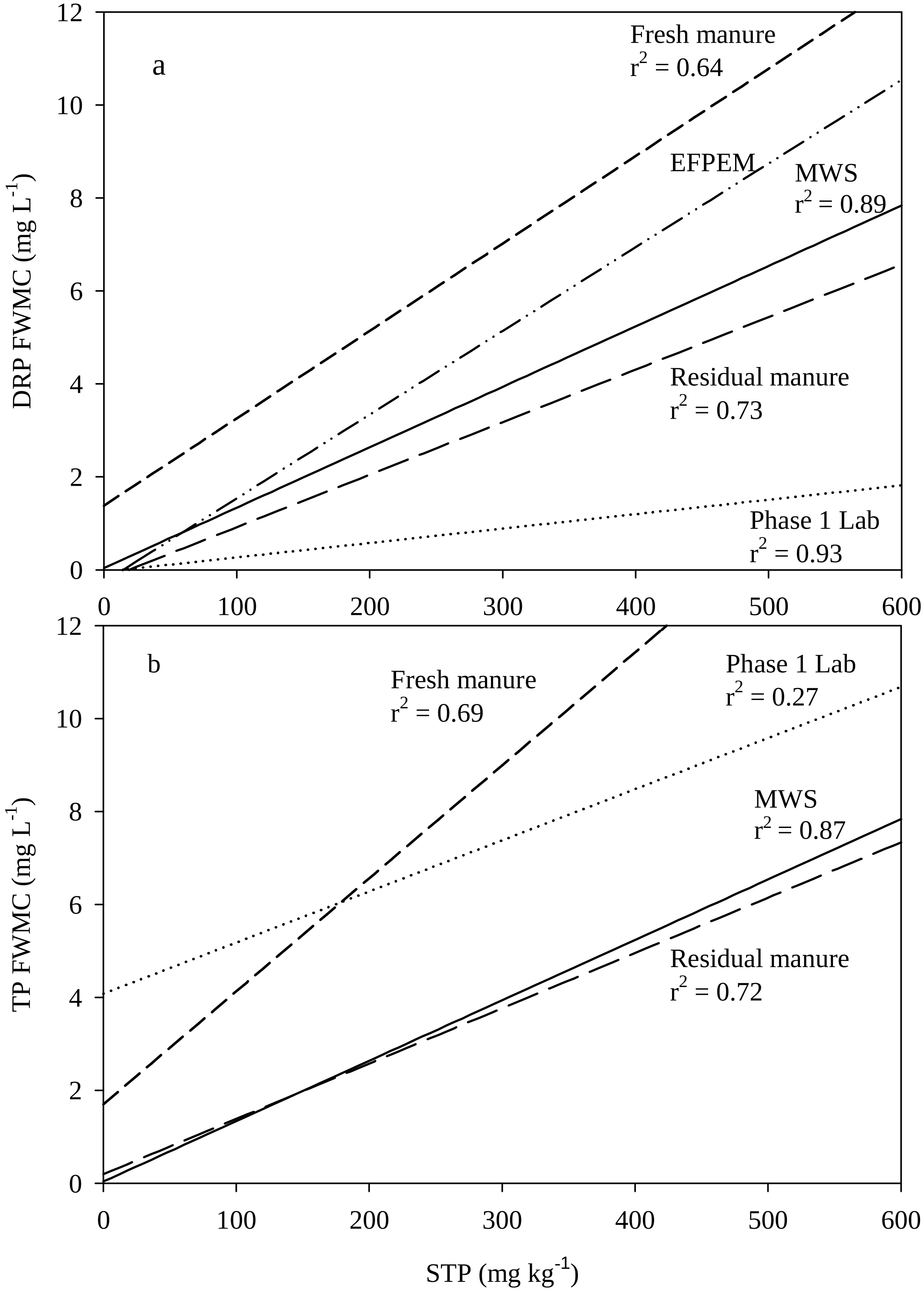

Fig. 11. Comparison of relationships between soil-test phosphorus (STP) and (a) dissolved reactive phosphorus (DRP) flow-weighted mean concentration (FWMC) and (b) total phosphorus (TP) FWMC from the microwatershed (MWS) study, Phase 1 lab results and EFPEM (Wright et al. 2003), and Phase 2 fresh manure and residual incorporated manure rainfall simulations (Ontkean et al. 2006).

simulations conducted by Andraski and Bundy (2003). Conversely, Ontkean et al. (2006) reported average DRP:TP ratios of 0.70 from plot-scale rainfall simulations, although ratios were greater from manured sites than non-manured sites and were lower immediately following manure application than 1 yr after application.

Possible explanations for the higher proportion of DRP measured at the microwatershed scale relative to the laboratory and small-plot scales include the longer time that runoff is in contact with soil at the field scale compared to the plot scale (Nash et al. 2002), and this may increase concentrations of dissolved phosphorus in runoff water. The PP fraction of TP tends to be favoured in small plot-scale and lab-scale studies, due to the comparatively high kinetic energy of overland flow that increases the detachment of soil particles (Nash et al. 2002). Variations in topography at field scales also offer greater opportunities for the deposition of PP than at plot scales. Manure application and incorporation may have increased infiltration and reduced detachment in the Phase 2 rainfall simulations (Ontkean et al. 2006), resulting in higher DRP:TP ratios than other small plot-scale studies. In addition, snowmelt runoff, which accounted for the majority of runoff in the microwatershed study, was not measured at the plot or lab scale. Snowmelt tends to have higher proportions of DRP since frozen soils reduce the detachment of soil particles (Hansen et al. 2000). Higher ratios of DRP to TP in snowmelt were observed at the non-manured sites, but not at the manured sites. A greater proportion of rainfall runoff events were observed during Phase 1 than were observed during the microwatershed study. Although the higher erosivity of the rainfall may increase PP fractions, the Phase 1 catchment results contained a similar proportion of DRP (59%) compared to the microwatershed results (55%).

Scale-related differences between the field-plot scale rainfall simulations (Ontkean et al. 2006) and the microwatersheds were not as evident as those in Phase 1. Ontkean et al. (2006) measured DRP and TP FWMCs in runoff from three sites (Wilson, Lacombe, and Beaverlodge) with a range of manure application rates and with and without incorporation. The rainfall simulations were completed immediately after manure application and incorporation and then repeated 1 yr later. The equation derived from the microwatersheds had an extraction coefficient that was not significantly different from those found by Ontkean et al. (2006) 1 yr after manure application for all incorporated treatments (Fig. 11) at the Lacombe and Wilson sites (0.012 to 0.015), but was lower for the Beaverlodge site, which had a narrower STP range and lower infiltration rates. The relationships derived immediately after manure application had greater extraction coefficients (DRP: 0.013 to 0.032; TP: 0.024 to 0.11) than the microwatershed study (Fig. 11), although differences between the DRP extraction coefficient for the microwatersheds and the combined data from the Lacombe and Wilson sites were not significant. Interestingly, the regression equations derived from simulations immediately after manure application from the sites with similar STP ranges had greater accuracy in predicting the PON spring 2003 DRP and TP FWMCs (Table 11; DRP: 13.0 to 19.9 mg L^{-1}; TP: 17.4 to 18.2 mg L^{-1}) than the microwatershed regression equations. The PON spring 2003 values were not included in the microwatershed equation because they were generated following a heavy and poorly incorporated manure application that led to greater DRP and TP FWMCs than any other site. This finding suggests that the PON spring 2003 values were closer to those from freshly applied manure than to those from residual manure that had equilibrated with the soil.

The microwatershed study results were based on field-scale data from Alberta in three variable climatic years. Strong relationships were found between STP and DRP and TP FWMCs under snowmelt and rainfall precipitation. Since this is the most extensive dataset available for Alberta, it may provide the basis for estimating phosphorus losses in runoff from agricultural land in the province.

CONCLUSIONS

Strong linear relationships between STP and phosphorus in runoff from eight field-scale microwatershed sites in Alberta were developed in this study. Relationships were developed for FWMCs of DRP and TP. Reduced levels of STP following the cessation of manure application corresponded directly with reductions in runoff phosphorus. This helped to clarify the linear nature of the relationship between STP and the phosphorus in runoff at the field scale, and helped to fill gaps in the wide range of STP measured in 2002. Although a data gap exists between STP values of 75 to 150 mg kg^{-1}, there is no reason to suspect that the linear relationship observed would deviate within this STP range.

Although a number of different landform STP representations were examined to predict spring runoff phosphorus concentrations, a simple average of all soil sampling points was as good a predictor of runoff phosphorus concentrations as a landform area-weighted mean representation and a subsample of points within the runoff contributing area. A random subset of samples and representative random samples also produced similar results. There was no significant difference in the slopes or intercepts in any of the relationships.

Previous studies have found that STP in the 0- to 5-cm layer is the best predictor of phosphorus in runoff. However, there were no significant differences in predictive ability for any of the three soil layers examined in our study. Therefore, it is likely that, in most cases, an agronomic soil sampling depth of 0 to 15 cm can be used to predict phosphorus in runoff from agricultural land in Alberta.

Spring runoff accounted for the majority of runoff at nearly all of the sites, except for the irrigated LLB site and the GPC site in 2004. Snowmelt runoff accounted for 90% of the runoff volume from the eight sites during the 3 yr of the study. At the non-manured sites, TP FWMCs were greater during summer precipitation events than during spring runoff events and predicted TP FWMCs were underestimated for some summer events. However, the relationship between STP and TP is likely adequate for predicting phosphorus concentrations in most runoff events, given that the vast majority of runoff in Alberta occurs in the spring.

Strong relationships were found between DPS and the FWMCs of DRP and TP; however, the relationships were not linear. Predictive abilities were similar to those observed for STP. Change point values corresponded to STP values that were near the agronomic threshold of 60 mg kg^{-1}. Although this method holds promise for predicting runoff and leaching losses of phosphorus, modified Kelowna STP is the standard for agronomic sampling in Alberta and our results suggest there is no strong imperative to move toward another soil test.

Field-scale concentrations of TP measured from all non-manured sites exceeded the watershed-scale in-stream Alberta water quality guideline of 0.05 mg L^{-1} TP for the protection of aquatic life by 3 to 16 times in all 3 yr. Concentrations of TP in runoff from the ungrazed native prairie site also exceeded this guideline by 3 to 10 times. Concentrations of TP from the non-manured sites were similar to watershed-scale values of TP measured in first-order streams that drain high intensity agricultural watersheds in Alberta. Concentrations of TP from the non-manured sites were generally less than the 1.0 mg L^{-1} TP permitted from federal wastewater treatment facilities; however, TP from the manured sites exceeded this guideline by 2 to 24 times.

While several studies have examined the relationship between STP and DRP, few have reported relationships between STP and TP. In comparison with other Alberta studies, slopes of the lines for DRP were greater than lab-derived values, but were lower than those employed in the Edge-of-Field phosphorus export model (EFPEM). Since our study was based on field-scale results from Alberta, rather than a modification of laboratory data, these relationships should supersede the EFPEM and provide the basis for phosphorus modelling in Alberta. Based on our TP relationship for the 0- to 15-cm soil layer, the predicted TP in runoff from the edge-of-field would be 1 mg L^{-1} if the STP was 60 mg kg^{-1}, which is generally considered an agronomic STP threshold for Alberta soils.

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APPENDICES

Appendix 1. Microwatershed boundaries with spatial features and sampling points.

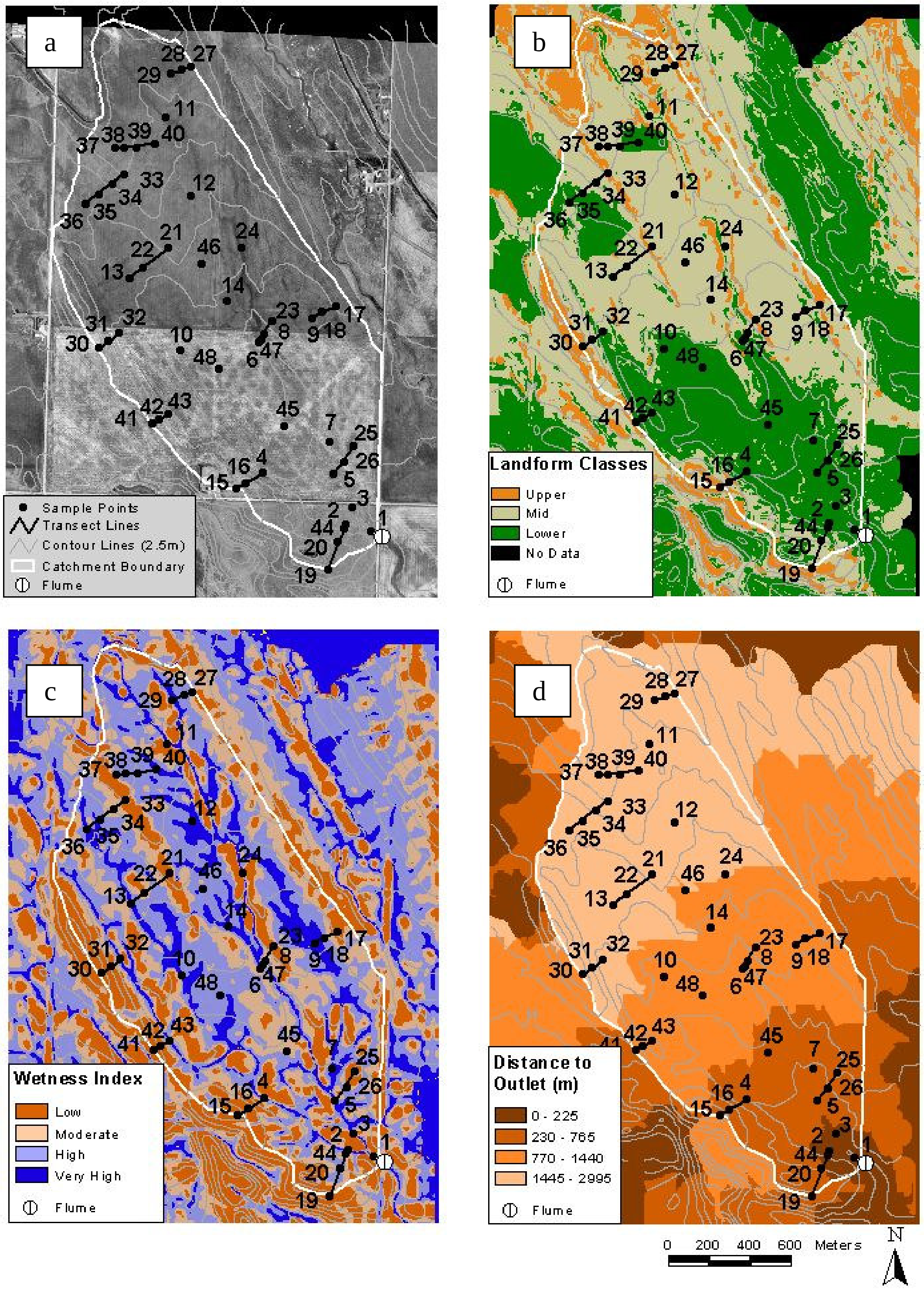

Fig. A1.1. Microwatershed boundaries and soil sampling points at the Crowfoot Creek site overlaying (a) air photo features, (b) landform classes, (c) wetness index, and (d) distance to outlet.

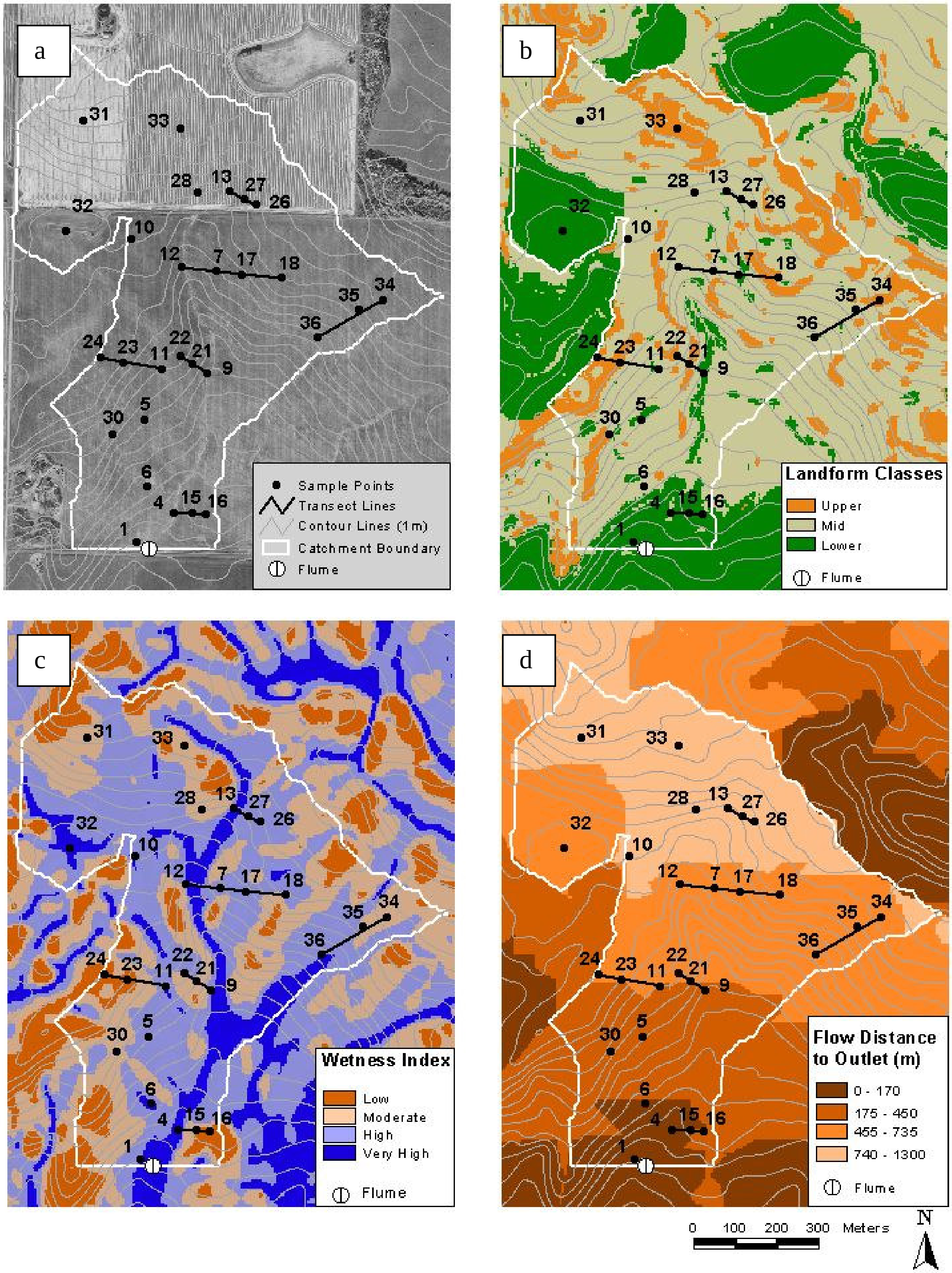

Fig. A1.2. Microwatershed boundaries and soil sampling points at the Grande Prairie Creek site overlaying (a) air photo features, (b) landforms, (c) wetness index, and (d) distance to outlet.

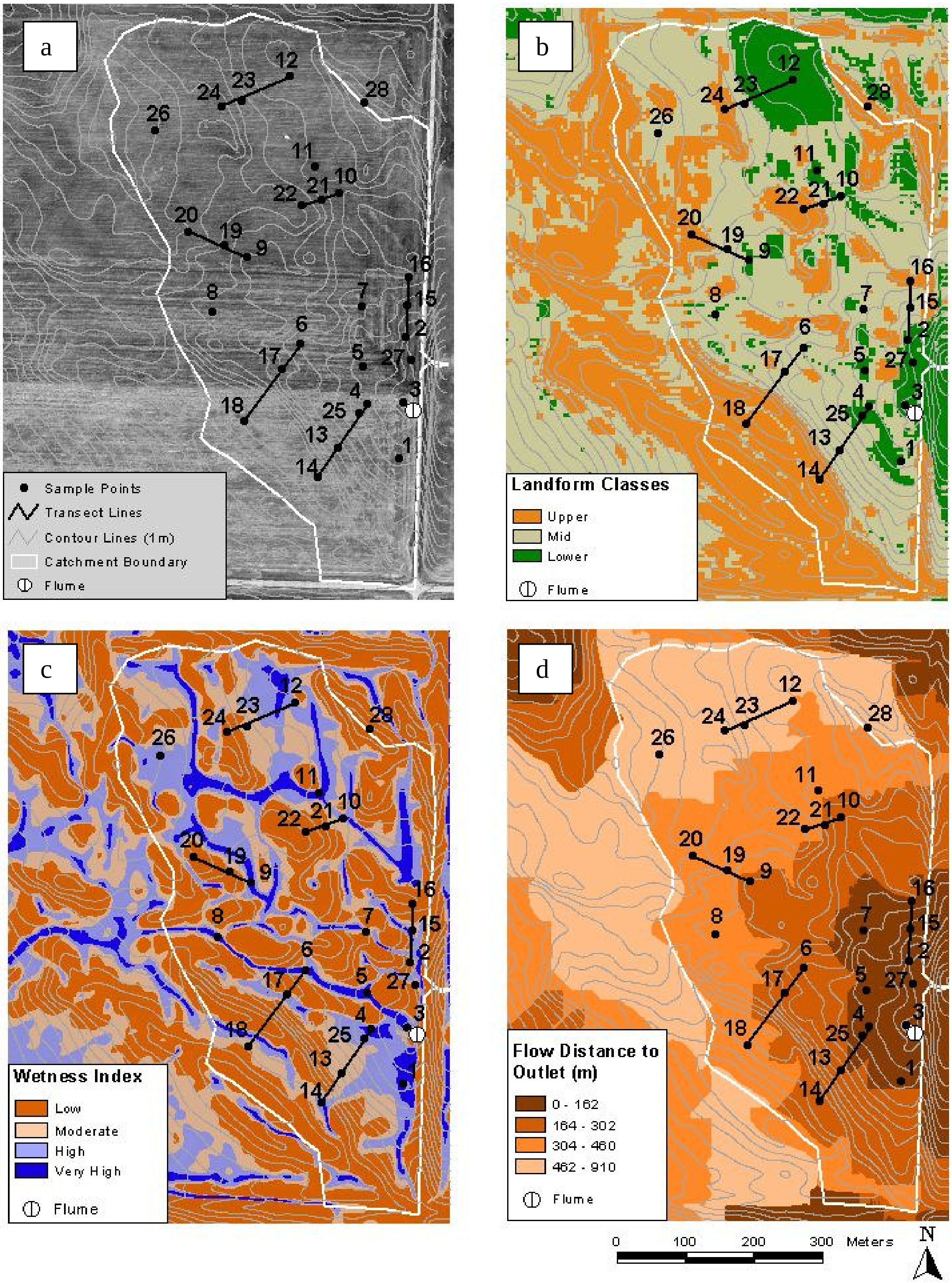

Fig. A1.3. Microwatershed boundaries and soil sampling points at the Renwick Creek site overlaying (a) air photo features, (b) landforms, (c) wetness index, and (d) distance to outlet.

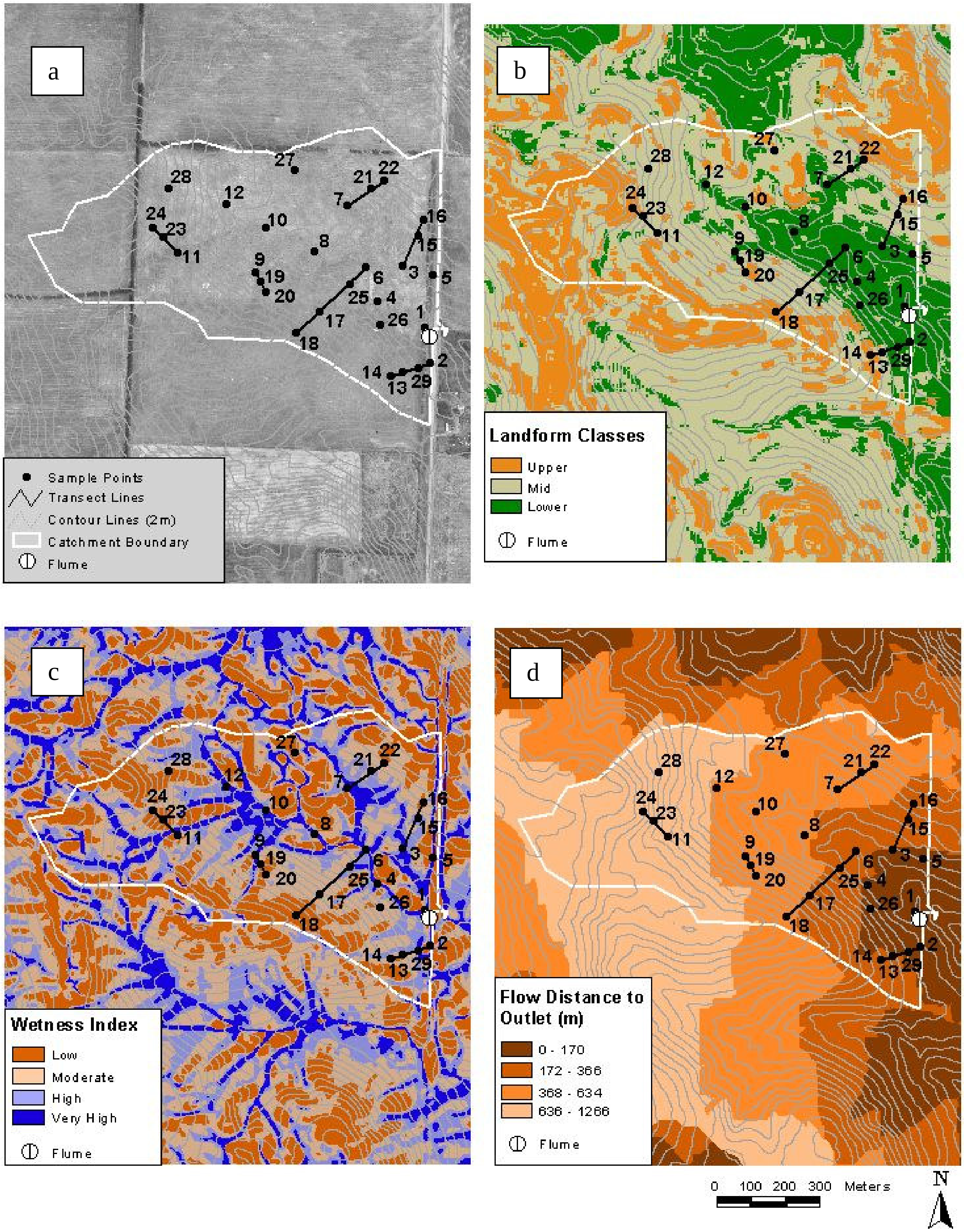

Fig. A1.4. Microwatershed boundaries and soil sampling points at the Three Hills Creek site overlaying (a) air photo features, (b) landforms, (c) wetness index, and (d) distance to outlet.

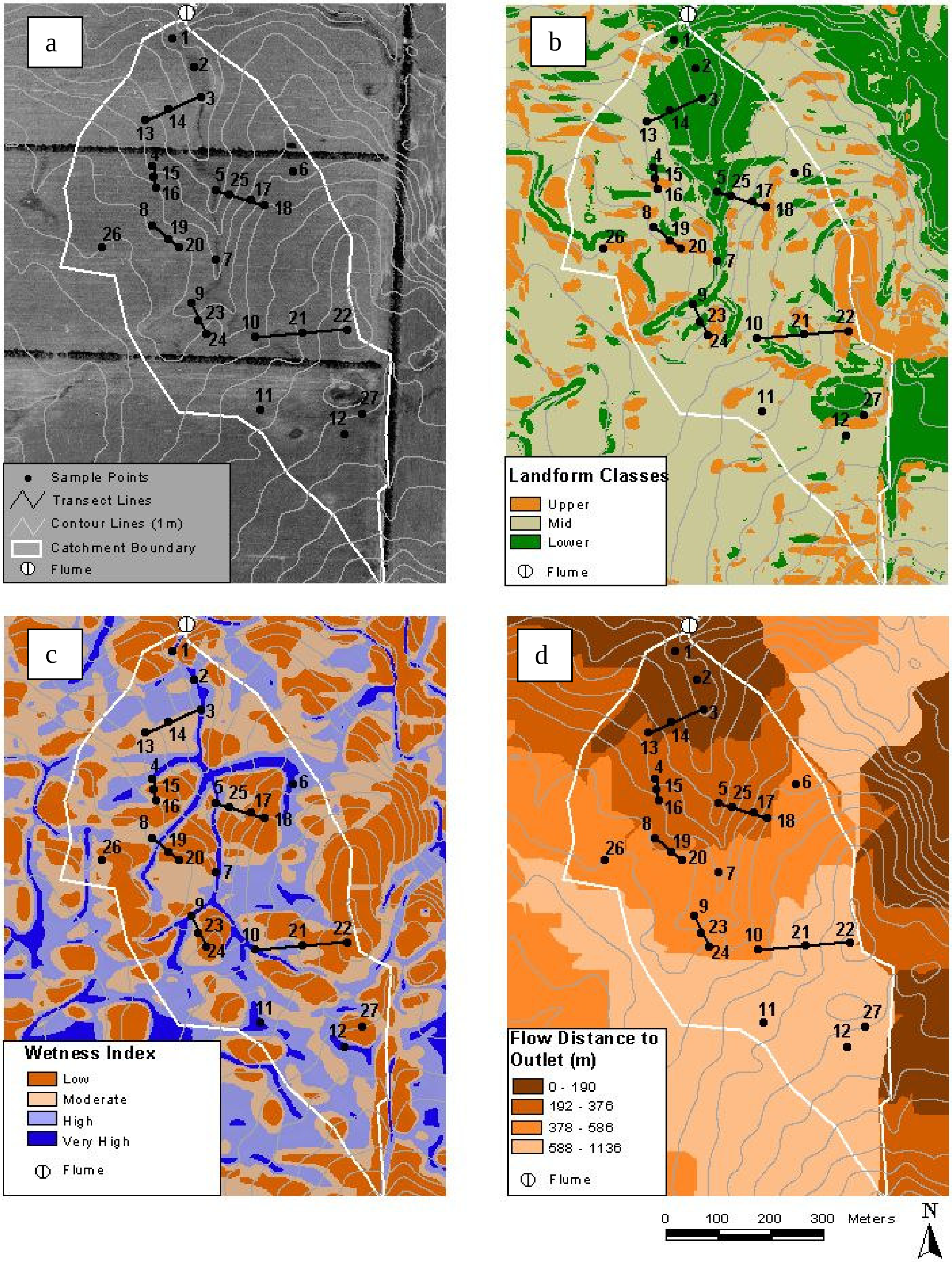

Fig. A1.5. Microwatershed boundaries and soil sampling points at the Wabash Creek site overlaying (a) air photo features, (b) landforms, (c) wetness index, and (d) distance to outlet.

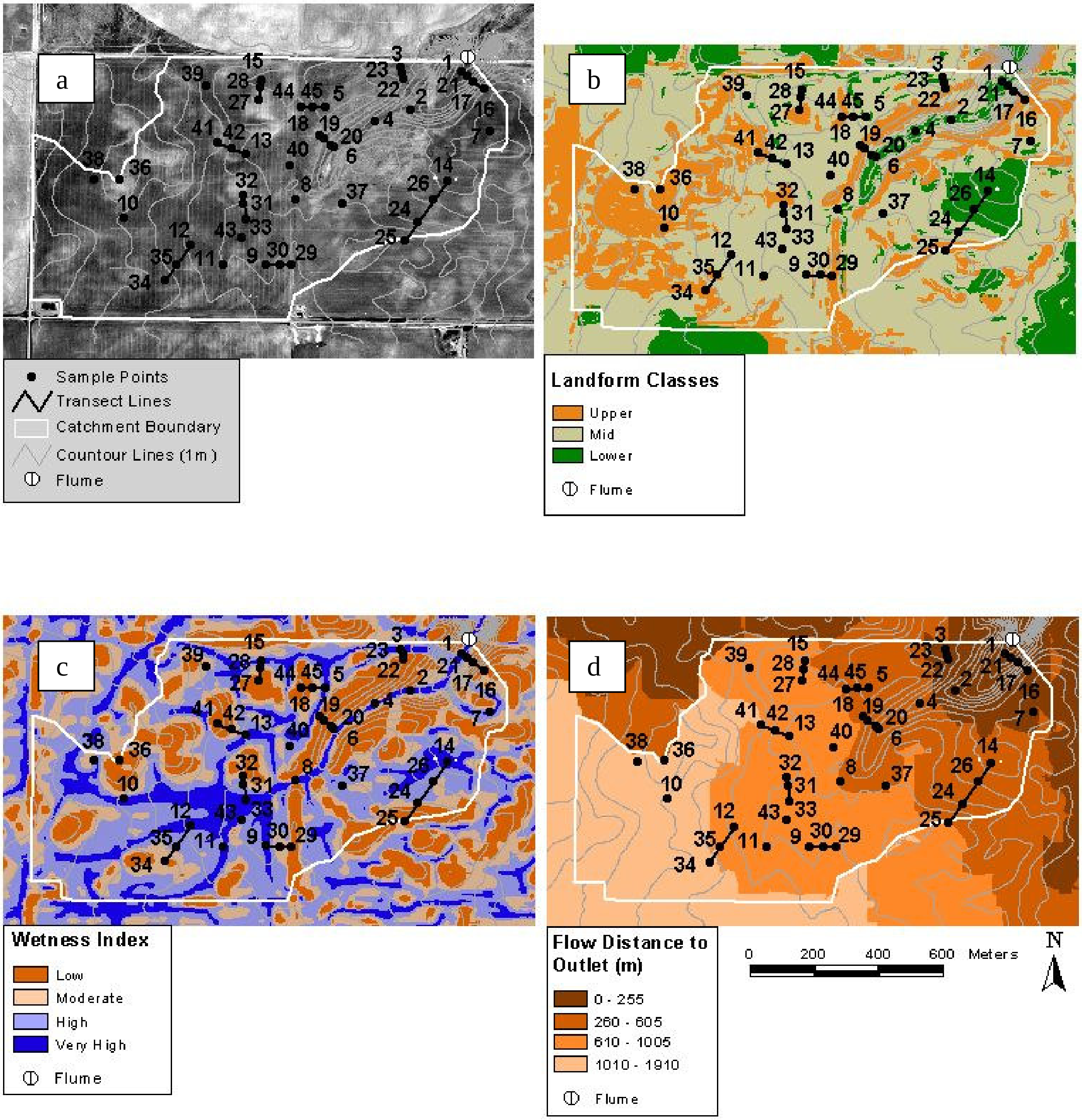

Fig. A1.6. Microwatershed boundaries and soil sampling points at the Lower Little Bow River site overlaying (a) air photo features, (b) landforms, (c) wetness index, and (d) distance to outlet.

Appendix 2. Soil sampling point attributes at the eight microwatershed sites.

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] $\ln(\alpha/\tan\beta)$	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
CFT	01	346447.49	5659409.67	905.17	8	L	L	L		
CFT	02	346324.83	5659441.23	905.60	11	D	D			
CFT	03	346353.77	5659524.51	906.02	12	D	D			
CFT	04	345923.03	5659694.08	907.02	9	L	L	L		
CFT	05	346265.88	5659685.26	912.91	11	D	D			
CFT	06	345901.32	5660325.34	911.19	11	L	L	L		
CFT	07	346248.31	5659840.92	918.57	7	L	L	L	L	
CFT	08	345928.24	5660364.71	915.46	9	M		M		
CFT	09	346162.71	5660446.16	914.67	9	M		M	M	
CFT	10	345517.88	5660291.40	913.66	4	L	L	L		
CFT	11	345451.05	5661419.57	915.76	3	M		M		
CFT	12	345569.47	5661039.40	913.01	6	M		M		
CFT	13	345272.20	5660638.96	912.36	3	M		M		
CFT	14	345746.79	5660530.63	915.86	4	M		M	M	
CFT	15	345791.64	5659613.91	913.73	3	U			U	
CFT	16	345839.65	5659643.34	915.35	6	M		M	M	
CFT	17	346278.54	5660505.25	907.38	5	U				
CFT	18	346208.59	5660474.30	924.37	4	M		M		
CFT	19	346239.44	5659217.96	923.66	9	U				
CFT	20	346287.23	5659355.18	920.93	3	M		M		
CFT	21	345463.35	5660787.42	916.93	4	M		M	M	
CFT	22	345337.30	5660689.62	920.57	4	M		M		
CFT	23	345964.40	5660434.08	917.90	9	U				
CFT	24	345818.27	5660788.13	920.88	10	M		M	M	
CFT	25	346358.98	5659826.02	911.82	3	L		L	L	
CFT	26	346317.85	5659748.48	909.52	8	L		L		

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
CFT	27	345573.98	5661670.17	909.20	5	U				
CFT	28	345529.62	5661654.66	916.43	6	M		M		
CFT	29	345474.93	5661633.02	911.08	7	M		M	M	
CFT	30	345128.27	5660304.16	910.40	2	U				
CFT	31	345173.67	5660337.18	907.54	12	M		M		
CFT	32	345221.75	5660372.64	915.88	3	L		L		
CFT	33	345246.02	5661145.43	910.04	12	U				
CFT	34	345190.18	5661100.66	911.25	5	M		M	M	
CFT	35	345127.73	5661050.10	910.42	5	L		L	L	
CFT	36	345058.12	5661001.61	914.24	2	M		M		
CFT	37	345203.34	5661270.34	914.26	7	M		M		
CFT	38	345250.04	5661271.39	905.87	14	M		M		
CFT	39	345305.82	5661276.64	909.74	5	M		M	M	
CFT	40	345398.17	5661291.94	906.76	4	D				
CFT	41	345381.85	5659930.19	918.23	3	U				
CFT	42	345415.92	5659949.87	922.65	3	L		L	L	
CFT	43	345462.14	5659976.28	923.75	5	L	L	L		
CFT	44	346316.05	5659422.07	923.13	5	L	L	L		
CFT	45	346028.04	5659920.52	922.12	6	L		L		
CFT	46	345622.73	5660709.06	923.90	6	M		M	M	
CFT	47	345914.50	5660340.24	920.75	9	L		L	L	
CFT	48	345705.52	5660196.35	918.45	6	L		L		
GPC	01	380178.41	6142131.76	758.16	7	L	L	L	L	
GPC	04	380266.68	6142201.90	759.53	12	M	M	M	M	
GPC	05	380196.15	6142422.34	765.07	6	M		M		
GPC	06	380204.58	6142264.14	761.29	7	M		M	M	
GPC	07	380368.27	6142778.29	769.65	6	M		M	M	
GPC	09	380345.73	6142533.81	763.84	6	L	L	L		

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
GPC	10	380166.86	6142853.08	772.02	6	M		M		
GPC	11	380237.35	6142543.90	766.92	7	M		M	M	
GPC	12	380285.14	6142785.32	768.99	7	M	M	M		
GPC	13	380399.56	6142965.92	772.74	8	M	M	M	M	
GPC	15	380312.41	6142199.47	760.25	4	L		L	L	
GPC	16	380342.36	6142197.91	760.72	4	L		L		
GPC	17	380430.12	6142765.67	770.64	6	L		L	L	
GPC	18	380521.59	6142759.73	773.03	5	M		M	M	
GPC	21	380312.33	6142554.78	766.09	5	U			U	
GPC	22	380283.81	6142573.50	767.22	5	M		M	M	
GPC	23	380145.92	6142558.90	769.82	8	U			U	
GPC	24	380091.68	6142569.54	770.92	4	U			U	
GPC	26	380462.80	6142933.95	773.76	5	M		M		
GPC	27	380434.20	6142949.18	773.24	5	M				
GPC	28	380322.54	6142964.19	773.06	4	M		M	M	
GPC	30	380122.04	6142386.92	766.38	4	M		M	M	
GPC	31	380051.26	6143136.40	778.33	4	M				Area added in 2003
GPC	32	380011.77	6142873.13	771.23	11	D				Area added in 2003
GPC	33	380281.62	6143104.46	777.48	4	U				Area added in 2003
GPC	34	380764.27	6142708.59	774.50	4	U				Area added in 2003
GPC	35	380707.23	6142684.46	772.75	4	M				Area added in 2003
GPC	36	380608.50	6142618.64	768.36	6	M				Area added in 2003
LLB	01	388412.64	5538625.25	830.23	11	L	L	L	L	
LLB	02	388256.66	5538509.37	834.84	11	L	L	L	L	
LLB	03	388226.85	5538638.87	838.37	9	M	M			
LLB	04	388147.44	5538471.34	837.46	13	L	L	L	L	
LLB	05	387989.67	5538518.62	840.51	9	L		L		
LLB	06	388020.23	5538392.27	838.91	5	L	L			

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
LLB	07	388500.52	5538443.46	839.37	9	M	M	M		
LLB	08	387900.73	5538230.17	841.49	12	M	M			
LLB	09	387804.67	5538028.50	842.73	12	M		M		
LLB	10	387368.33	5538175.17	845.87	9	U				
LLB	11	387672.58	5538025.53	843.04	9	M		M	M	
LLB	12	387574.40	5538089.37	843.80	8	M		M	M	
LLB	13	387744.05	5538371.53	841.87	9	M		M		
LLB	14	388368.60	5538288.83	838.73	12	D			D	
LLB	15	387791.51	5538601.80	841.80	8	M				
LLB	16	388482.85	5538571.54	837.83	3	U				
LLB	17	388448.50	5538597.32	833.50	3	M		M	M	
LLB	18	387972.91	5538430.35	841.38	3	U			U	
LLB	19	387987.49	5538418.84	840.95	3	U			U	
LLB	20	388009.52	5538400.06	839.34	4	L	L	L	L	
LLB	21	388431.32	5538611.58	830.92	4	L	L	L		
LLB	22	388235.89	5538606.82	838.75	3	U				
LLB	23	388231.53	5538622.33	838.53	4	M		M	M	
LLB	24	388230.59	5538114.86	841.13	4	M				
LLB	25	388271.65	5538161.79	842.70	3	M		M		
LLB	26	388327.39	5538229.26	839.91	5	L		L		
LLB	27	387784.72	5538540.19	842.90	3	U			U	
LLB	28	387787.65	5538578.33	842.06	4	M		M		
LLB	29	387887.31	5538025.84	843.40	3	U				
LLB	30	387849.25	5538026.86	843.19	4	M		M		
LLB	31	387744.28	5538169.67	842.33	10	M			M	
LLB	32	387734.54	5538243.12	843.21	4	M		M		
LLB	33	387737.79	5538217.53	842.90	5	M		M		
LLB	34	387495.72	5537979.98	845.49	4	U				

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
LLB	35	387528.77	5538026.50	844.46	6	M		M		
LLB	36	387354.88	5538294.06	847.94	3	U				
LLB	37	388043.00	5538215.50	841.95	5	M		M		
LLB	38	387273.00	5538292.50	846.84	6	U				
LLB	39	387621.33	5538582.17	843.50	5	M		M		
LLB	40	387881.67	5538336.50	841.59	6	M		M	M	
LLB	41	387657.94	5538406.37	844.51	2	U				
LLB	42	387700.85	5538389.02	842.66	4	M		M	M	
LLB	43	387733.00	5538109.78	842.56	6	M		M		
LLB	44	387917.86	5538516.88	842.18	3	U				
LLB	45	387952.51	5538517.85	841.19	4	M		M		
PON	01	322373.05	5851947.03	826.78	13	D	D			
PON	02	322158.81	5851948.39	827.21	5	L	L	L	L	
PON	03	322263.92	5851948.33	830.12	9	L	L	L	L	
PON	04	322057.54	5852047.44	831.73	6	M		M	M	
PON	05	322222.36	5852046.12	832.49	4	U			U	
PON	06	322292.40	5852135.64	824.97	9	M	M	M	M	
PON	07	321871.45	5852317.42	833.26	5	M		M	M	
PON	08	322035.56	5852276.47	827.46	9	L		L		
PON	09	322252.43	5851780.96	827.45	3	U				
PON	10	322005.12	5851949.42	829.37	3	U			U	
PON	11	322094.23	5851949.08	827.30	5	M		M		
PON	12	322038.19	5852152.04	826.62	9	L		L	L	
PON	13	322014.96	5852127.44	834.84	5	M		M	M	
PON	14	322265.40	5851817.06	835.58	4	M		M	M	
PON	15	322288.31	5851878.69	823.15	7	L	L	L	L	
PON	16	322315.65	5852146.57	836.69	4	M		M	M	
PON	17	321987.62	5852097.37	827.30	4	U				

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
PON	18	322058.69	5852326.99	835.17	4	U				
PON	19	322047.76	5852302.38	830.85	4	M		M	M	
PON	20	322336.49	5852156.82	834.13	3	U			U	
PON	21	322261.45	5852048.48	825.96	8	L	L	L		
PON	22	322242.87	5852047.48	824.50	5	M		M	M	
REN	01	325547.80	5731672.82	979.63	8	M	M	M		
REN	02	325558.39	5731847.93	976.55	4	L		L	L	
REN	03	325553.90	5731753.93	979.12	11	L	L	L	L	
REN	04	325501.03	5731751.80	979.48	9	L	L	L		
REN	05	325495.71	5731804.62	981.95	11	L	L	L		
REN	06	325405.02	5731838.18	985.44	10	L	L			
REN	07	325493.28	5731893.13	984.86	5	M		M	M	
REN	08	325277.13	5731885.73	983.24	10	L		L	L	
REN	09	325326.48	5731964.91	984.06	10	L			L	
REN	10	325460.34	5732057.73	984.32	7	L		L	L	
REN	11	325425.46	5732095.54	983.59	10	L				
REN	12	325389.84	5732226.58	986.67	7	D				
REN	13	325459.26	5731688.26	988.99	5	M		M	M	
REN	14	325428.96	5731644.28	986.86	8	U			U	
REN	15	325560.49	5731895.42	986.53	6	M		M	M	
REN	16	325561.77	5731934.62	988.51	3	U			U	
REN	17	325378.42	5731801.59	979.52	6	M		M	M	
REN	18	325322.35	5731725.59	990.05	5	U			U	
REN	19	325294.34	5731980.97	987.27	4	M		M		
REN	20	325242.58	5732001.68	988.31	4	U				
REN	21	325435.67	5732046.65	988.68	5	M		M		
REN	22	325406.80	5732039.88	984.81	4	U			U	
REN	23	325319.71	5732191.94	976.86	5	M		M		

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
REN	24	325290.78	5732184.45	983.62	4	U				
REN	25	325490.64	5731737.60	981.21	7	L			L	
REN	26	325193.89	5732149.34	981.89	7	M		M		
REN	27	325564.53	5731814.56	979.45	5	L	L			
REN	28	325498.56	5732189.39	977.31	3	U			U	
STV	01	292775.70	5562780.54	1312.41[v]		U		U	U	
STV	02	292714.52	5562768.66	1335.75[v]		M		M	M	
STV	03	292618.57	5562743.89	1353.87[v]		L	L	L	L	
THC	01	320023.67	5768391.63	942.04	11	D	D			
THC	02	320039.38	5768295.50	945.94	8	L				Ditch, removed point
THC	03	319964.83	5768556.08	947.73	8	L	L		L	
THC	04	319896.81	5768461.22	945.30	8	L	L	L	L	
THC	05	320045.07	5768532.67	947.66	9	L			L	Ditch, removed point
THC	06	319866.31	5768552.33	945.98	17	L	L		L	
THC	07	319814.75	5768720.37	951.14	8	L		L	L	
THC	08	319726.55	5768595.48	949.40	13	L	L	L		
THC	09	319566.98	5768540.79	958.00	8	L		L		
THC	10	319595.11	5768660.25	953.14	13	M		M	M	
THC	11	319359.84	5768592.38	967.57	5	M		M	M	
THC	12	319488.55	5768722.09	956.99	9	L		L	L	
THC	13	319963.19	5768270.08	951.09	5	M		M		
THC	14	319933.62	5768261.72	954.15	5	U			U	
THC	15	320004.72	5768639.92	952.84	6	M		M		
THC	16	320018.84	5768680.51	956.06	5	U				
THC	17	319739.76	5768431.69	959.40	5	M		M		
THC	18	319677.25	5768376.79	968.15	4	U			U	
THC	19	319583.21	5768515.56	960.03	5	M		M		
THC	20	319596.57	5768485.49	963.04	5	U				

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
THC	21	319879.69	5768764.58	954.52	6	M		M	M	
THC	22	319913.93	5768787.43	957.91	5	M		M	M	
THC	23	319321.91	5768634.82	971.30	4	M			M	
THC	24	319293.00	5768659.21	971.30	4	U			U	
THC	25	319821.27	5768508.80	948.55	7	L				
THC	26	319904.78	5768396.82	948.92	6	L	L		L	
THC	27	319674.28	5768813.65	956.14	4	M		M		
THC	28	319334.75	5768765.35	970.43	4	M		M		
THC	29	320004.61	5768282.28	947.81	7	L				
WAB	01	305422.62	5994648.44	661.62	9	L	L	L		
WAB	02	305462.08	5994593.65	661.78	12	D	D		D	
WAB	03	305474.89	5994539.33	664.72	12	L	L		L	
WAB	04	305384.18	5994407.12	663.74	8	L	L		L	
WAB	05	305505.12	5994358.95	665.84	10	L	L	L		
WAB	06	305651.11	5994396.86	666.50	9	M	M	M		
WAB	07	305503.07	5994228.78	668.56	7	L		L	L	
WAB	08	305384.13	5994293.72	670.71	7	M		M		
WAB	09	305458.49	5994146.28	672.27	11	M		M		
WAB	10	305578.92	5994082.73	664.54	10	M			M	
WAB	11	305587.63	5993941.81	665.57	10	M		M	M	
WAB	12	305747.52	5993895.68	666.03	9	M		M	M	
WAB	13	305369.32	5994493.21	666.41	6	M		M		
WAB	14	305414.24	5994515.21	667.03	7	L		L	L	
WAB	15	305385.71	5994387.13	667.36	5	M			M	
WAB	16	305389.82	5994365.61	670.32	7	M				
WAB	17	305569.69	5994340.50	667.33	5	M		M	M	
WAB	18	305597.31	5994332.45	668.28	4	U			U	
WAB	19	305414.42	5994267.73	664.71	5	M		M		

Table A2.1 Soil sampling point attributes of GPS coordinates, wetness indices, landform classes, and points used as subsamples to characterize runoff contributing area, representative random, and random representations of soil-test phosphorus.

Site	Sample point	Easting (m)	Northing (m)	Elevation (m)	Wetness index[z] ln(α/tanβ)	Landform class[y]	Runoff contributing area[x] subsample	Representative random[w] subsample	Random 15 point subsample	Notes
WAB	20	305434.40	5994250.82	668.04	5	U				
WAB	21	305670.14	5994088.88	671.95	6	M		M	M	
WAB	22	305753.86	5994094.68	661.35	5	U				
WAB	23	305471.30	5994112.45	663.48	6	M		M	M	
WAB	24	305484.63	5994088.37	667.85	5	U			U	
WAB	25	305530.75	5994352.28	666.66	6	L				
WAB	26	305286.82	5994251.84	666.65	6	M				
WAB	27	305780.83	5993936.17	672.22	5	L		L	L	

[z] Quinn et al. 1995.
[y] U = Upper, M=Mid, L=Lower, D=Depression (reclassed as L) landform classes as described in MacMillan et al. (2000).
[x] Sub-sample of 20% of points with high wetness index located in main drain of field, D reclassed as L.
[w] Based on recommendations of The Prairie Provinces' Committee on Livestock Development and Manure Management (2004).
[v] Used GPS coordinates rather than digital elevation model data.

Appendix 3. Soil characteristics of a representative transect within the microwatershed sites.

Table A3.1. Soil characteristics of a representative transect of upper, mid, lower landform positions within the microwatershed study sites.

Sample Site/mo/yr/point	Landform class [z]	Soil horizon	Horizon depth (cm)	Organic matter (%)	Sand (%)	Silt (%)	Clay (%)	Texture	Sample depth (cm)	Initial [y] pH 1S:2W	Initial EC 1S:2W ($dS\ m^{-1}$)	Fall04 [x] pH 1S:2W	Fall04 EC 1S:2W ($dS\ m^{-1}$)
Ungrazed grassland site													
STV/10/02/03	Upper	Ah	0-70	10.0	34	48	18	Loam	0-70	6.3	0.2		
STV/10/02/03		Bm	70-85	2.1	25	46	30	Clay Loam	70-85	6.6	0.1		
STV/10/02/03		Bc	85-100	2.1	26	41	33	Clay Loam	85-100	6.8	0.2		
STV/10/02/02	Mid	Ah	0-33	13.9	40	46	14	Loam	0-33	6.5	0.2		
STV/10/02/02		Bm	33-60	4.1	29	34	37	Clay Loam	33-60	6.3	0.2		
STV/10/02/02		Ck	70-100	2.0 [w]	29	39	32	Clay Loam	70-100	7.9	0.5		
STV/10/02/01	Lower	Ah	0-25	12.2	38	47	16	Loam	0-25	6.3	0.2		
STV/10/02/01		Bm	25-50	3.5	30	37	33	Clay Loam	25-50	6.7	0.2		
STV/10/02/01		Ck	50-100	2.1 [w]	33	39	28	Clay Loam	50-100	7.9	0.5		
Non-manured sites													
CFT/10/02/17	Upper	Ap	0-15	3.6 [v]	33	48	19	Loam	0-15	8.0	0.4	8.0	0.2
CFT/10/02/17		Bm	15-25	2.1 [v]	23	56	21	Silt Loam	15-30	8.2	0.4	8.2	0.2
CFT/10/02/17		Cca	25-50	1.6 [v]	35	39	26	Loam	30-60	8.5	0.6		
CFT/10/02/17		Ck	50-100	1.1 [w]	27	48	25	Loam					
CFT/10/02/18	Mid	Ap	0-15	5.3	30	49	21	Loam	0-15	6.2	0.3	6.4	0.2
CFT/10/02/18		Bm	15-30	2.5 [v]	27	52	21	Silt Loam	15-30	7.8	1.9	7.9	2.8
CFT/10/02/18		Cca	30-60	1.6 [v]	26	53	21	Silt Loam	30-60	8.3	7.9		
CFT/10/02/18		Ck	60-100	0.9 [w]	25	44	31	Clay Loam					
CFT/10/02/09	Lower	Ap	0-15	5.4	29	51	20	Silt Loam	0-15	6.4	0.5	5.9	0.1
CFT/10/02/09		Bm	15-40	2.4	25	52	24	Silt Loam	15-30	7.6	0.8	7.6	1.2
CFT/10/02/09		Cca	40-80	1.2 [w]	21	54	25	Silt Loam	30-60	8.3	5.3		
CFT/10/02/09		Ck	80-100	0.6 [w]	23	49	28	Clay Loam					
CFT/10/02/05	Lower, Hi WI [u]	Ap	0-15	5.5	24	56	19	Silt Loam	0-15	6.5	0.3	6.5	0.2

Table A3.1. Soil characteristics of a representative transect of upper, mid, lower landform positions within the microwatershed study sites.

Sample Site/mo/yr/point	Landform class [z]	Soil horizon	Horizon depth (cm)	Organic matter (%)	Sand (%)	Silt (%)	Clay (%)	Texture	Sample depth (cm)	Initial [y] pH 1S:2W	Initial EC 1S:2W ($dS\ m^{-1}$)	Fall04 [x] pH 1S:2W	Fall04 EC 1S:2W ($dS\ m^{-1}$)
CFT/10/02/05		Bt	40-85	2.4	25	52	23	Silt Loam	15-30	7.1	0.2	6.8	0.1
GPC/10/02/24	Upper	Ap	0-15	7.5	27	47	26	Loam	0-15	6.5	0.4	5.9	0.2
GPC/10/02/24		AE	15-24	2.8	19	41	40	Silty Clay Loam	15-30	6.6	0.4	6.5	0.4
GPC/10/02/24		B	24-35	4.1	18	27	55	Clay	30-60	7.6	1.7		
GPC/10/02/24		C	50-100	2.0 [w]	23	27	50	Clay					
GPC/10/02/22	Mid	Ap	0-10	7.5	27	44	29	Clay Loam	0-15	6.3	1.5	6.0	0.3
GPC/10/02/22		AE	10-15	3.9 [v]	21	48	31	Clay Loam	15-30	7.7	5.7	6.7	0.9
GPC/10/02/22		B	15-30	4.3 [v]	15	28	57	Clay	30-60	7.9	10.4		
GPC/10/02/22		Bsa	40-60	2.4	23	31	46	Clay					
GPC/10/02/22		C	60-100	1.8 [w]	20	31	49	Clay					
GPC/10/02/09	Lower	Ap	0-10	7.6	27	47	26	Loam	0-15	6.0	1.0	6.1	0.5
GPC/10/02/09		AE	10-16	2.6	27	39	33	Clay Loam	15-30	6.4	2.5	6.6	0.8
GPC/10/02/09		B	16-45	4.0	17	28	55	Clay	30-60	7.1	10.2		
GPC/10/02/09		Bsa	45-70	2.8	16	27	57	Clay					
GPC/10/02/09		C	70-100	2.2 [w]	19	29	53	Clay					
REN/10/02/14	Upper	Ap	0-15	3.7	32	45	23	Loam	0-15	5.9	0.3	6.0	0.1
REN/10/02/14		Bm	15-55	1.9	32	44	24	Loam	15-30	5.4	0.2	6.9	0.1
REN/10/02/14		Cca	55-75	0.5	55	29	15	Sandy Loam	30-60	6.4	0.2		
REN/10/02/14		Ck	75-100	0.4 [w]	60	28	12	Sandy Loam					
REN/10/02/13	Mid	Ap	0-15	6.6	44	41	15	Loam	0-15	5.7	0.2	5.7	0.1
REN/10/02/13		Bm1	15-40	2.1	24	49	28	Clay Loam	15-30	6.3	0.2	6.5	0.1
REN/10/02/13		Bm2	40-80	0.5	61	25	14	Sandy Loam	30-60	6.8	0.1		
REN/10/02/13		Cca	80-100	0.4 [w]	65	23	13	Sandy Loam					
REN/10/02/25	Lower	Ap	0-15	7.0	42	43	16	Loam	0-15	6.3	0.2	5.7	0.1
REN/10/02/25		Bm1	20-60	1.9	24	51	25	Silt Loam	15-30	6.3	0.2	6.5	0.1

Table A3.1. Soil characteristics of a representative transect of upper, mid, lower landform positions within the microwatershed study sites.

Sample Site/mo/yr/point	Landform class [z]	Soil horizon	Horizon depth (cm)	Organic matter (%)	Sand (%)	Silt (%)	Clay (%)	Texture	Sample depth (cm)	Initial [y] pH 1S:2W	Initial EC 1S:2W ($dS\ m^{-1}$)	Fall04 [x] pH 1S:2W	Fall04 EC 1S:2W ($dS\ m^{-1}$)
REN/10/02/25		Bm2	60-90	1.0	62	23	14	Sandy Loam	30-60	7.1	0.5		
REN/10/02/25		Cca	90-100	0.8	67	19	14	Sandy Loam					
REN/10/02/04	Lower, Hi WI	Ap	0-15	7.1	48	39	12	Loam	0-15	5.9	0.3	6.1	0.1
REN/10/02/04		Ae	15-35	2.1	27	53	19	Silt Loam	15-30	7.7	0.4	6.8	0.1
REN/10/02/04		Bt	35-80	2.5	32	43	25	Loam	30-60	6.9	0.8		
REN/10/02/04		Cca	80-100	0.8 [w]	60	24	16	Sandy Loam					
THC/05/03/16	Upper	Ap	0-15	8.8	37	39	24	Loam	0-15	6.2	0.3	5.9	0.2
THC/05/03/16		B	30-60	1.9	52	26	23	Sandy Clay Loam	15-30	6.6	0.4	6.7	0.1
THC/05/03/16		C	90-115	1.0 [w]	49	30	21	Loam					
THC/05/03/15	Mid	Ap	0-20	10.0	26	51	23	Silt Loam	0-15	6.1	0.3	6.0	0.1
THC/05/03/15		A	20-40	7.5	30	47	23	Loam	15-30	6.7	0.5	6.5	0.1
THC/05/03/15		B1	40-60	1.8	32	42	26	Loam					
THC/05/03/15		B2	60-80	1.4	48	30	22	Loam					
THC/05/03/15		C	90-120	1.4 [w]	51	30	19	Loam					
THC/05/03/03	Lower	Ap	0-25	12.1	52	33	15	Sandy Loam	0-15	6.1	0.4	5.9	0.1
THC/05/03/03		A1	25-40	8.5	27	48	25	Loam	15-30	6.5	0.4	6.3	0.1
THC/05/03/03		A2	50-60	1.9	31	44	25	Loam					
THC/05/03/03		B	62-80	1.8	20	51	29	Silty Clay Loam					
THC/05/03/03		C	90-100	1.4 [w]	41	35	24	Loam					
THC/05/03/06	Lower, Hi WI	Ap	0-15	9.6	37	41	22	Loam	0-15	6.3	0.4	6.1	0.2
THC/05/03/06		A1	30-45	3.9	23	42	34	Clay Loam	15-30	6.6	0.5	6.6	0.3
THC/05/03/06		A2	60-90	1.4	32	44	25	Loam					
THC/05/03/06		B	90-110	1.4	45	31	23	Loam					
WAB/10/02/18	Upper	Ap	0-20	4.5	31	49	20	Loam	0-15	5.6	0.3	5.7	0.2
WAB/10/02/18		Bt	30-55	1.6	34	28	38	Clay Loam	15-30	6.9	0.3	6.3	0.1

Table A3.1. Soil characteristics of a representative transect of upper, mid, lower landform positions within the microwatershed study sites.

Sample Site/mo/yr/point	Landform class [z]	Soil horizon	Horizon depth (cm)	Organic matter (%)	Sand (%)	Silt (%)	Clay (%)	Texture	Sample depth (cm)	Initial [y] pH 1S:2W	Initial EC 1S:2W ($dS\ m^{-1}$)	Fall04 [x] pH 1S:2W	Fall04 EC 1S:2W ($dS\ m^{-1}$)
WAB/10/02/18		BC	55-90	1.0	42	28	30	Clay Loam	30-60	6.4	0.2		
WAB/10/02/17	Mid	Ap	0-20	4.3	29	51	20	Silt Loam	0-15	5.7	0.3	5.9	0.2
WAB/10/02/17		Bnt	35-55	1.6	34	29	37	Clay Loam	15-30	6.9	0.2	6.3	0.2
WAB/10/02/17		BC	80-100	1.1	36	29	35	Clay Loam	30-60	7.1	0.3		
WAB/10/02/25	Lower	Ap	0-20	4.5	26	47	27	Loam	0-15	6.2	0.4	6.1	0.1
WAB/10/02/25		AB	20-30	2.5	23	28	49	Clay	15-30	6.4	0.3	6.2	0.3
WAB/10/02/25		Bt	30-65	1.4	41	28	31	Clay Loam	30-60	6.6	0.3		
WAB/10/02/25		BC	65-85	1.1	44	28	28	Clay Loam					
WAB/10/02/05	Lower, Hi WI	Ap	0-20	6.1	26	49	25	Loam	0-15	6.3	0.4	6.2	0.2
WAB/10/02/05		Ah	20-40	6.3	25	49	26	Loam	15-30	6.9	0.3	6.8	0.1
WAB/10/02/05		Aeg	40-55	1.0 [v]	26	47	27	Loam	30-60	7.5	0.3		
WAB/10/02/05		Bg	55-90	0.9 [v]	31	42	27	Loam					
WAB/10/02/05		Ck	90+	2.1 [w]	20	24	56	Clay					
						Manured sites							
LLB/10/03/32	Upper	AP (K)	0-12	4.1 [v]	15	57	28	Silty Clay Loam	0-15	8.0	0.3	7.9	0.4
LLB/10/03/32		BM (k)	12-20	3.1 [v]	36	37	27	Clay Loam	15-30	8.4	0.3	8.2	0.5
LLB/10/03/32		Cca	20-50	1.2 [w]	36	36	28	Clay Loam	30-60	8.7	0.2		
LLB/10/03/32		Ck	50-85	1.0 [w]	39	34	27	Clay Loam					
LLB/10/03/33	Mid	AP (K)	0-13	4.5 [v]	38	36	26	Loam	0-15	7.8	0.4	7.7	1.0
LLB/10/03/33		BM (k)	13-25	3.5 [v]	33	39	28	Clay Loam	15-30	8.1	0.4	8.0	1.6
LLB/10/03/33		Cca	25-50	1.6 [w]	11	61	29	Silty Clay Loam	30-60	8.2	0.7		
LLB/10/03/33		II Ck	50-60	1.2	39	35	26	Loam					
LLB/10/03/31	Lower	Ap	0-20	3.2 [v]	48	32	20	Loam	0-15	7.9	0.3	7.6	0.5
LLB/10/03/31		BM	20-40	2.3 [v]	41	36	23	Loam	15-30	8.0	0.5	7.9	0.4
LLB/10/03/31		Cca	40-70	2.0 [w]	15	50	36	Silty Clay Loam	30-60	8.0	0.7		

Table A3.1. Soil characteristics of a representative transect of upper, mid, lower landform positions within the microwatershed study sites.

Sample Site/mo/yr/point	Landform class [z]	Soil horizon	Horizon depth (cm)	Organic matter (%)	Sand (%)	Silt (%)	Clay (%)	Texture	Sample depth (cm)	Initial [y] pH 1S:2W	Initial EC 1S:2W (dS m^{-1})	Fall04 [x] pH 1S:2W	Fall04 EC 1S:2W (dS m^{-1})
LLB/10/03/31		Ck	70-100	1.1	25	46	29	Clay Loam					
PON/05/03/10	Upper	Ap	0-15	8.0	50	35	15	Loam	0-15	6.5	0.6	5.9	0.3
PON/05/03/10		Ah	15-45	3.8	53	29	18	Sandy Loam	15-30	6.8	0.6	6.4	0.2
PON/05/03/10		Bt	55-90	1.2	35	32	33	Clay Loam					
PON/05/03/11	Mid	Ap	0-20	9.6	50	38	12	Loam	0-15	7.0	0.8	6.5	0.5
PON/05/03/11		Ah	20-75	8.4	50	41	10	Loam	15-30	6.9	0.6	5.9	0.5
PON/05/03/11		Bcg	75-100	0.6	37	42	21	Loam					
PON/05/03/03	Lower	Ap	0-20	10.0	42	42	16	L	0-15	7.5	1.9	7.5	0.5
PON/05/03/03		Ah	30-45	9.8 [v]				L	15-30	8.0	1.7	7.9	0.7
PON/05/03/03		Bg	55-90	0.6				CL					
PON/05/03/02	Lower, Hi WI	Ap	0-15	8.8	51	33	16	L	0-15	7.7	0.9	7.1	0.4
PON/05/03/02		Ah	15-35	8.4 [v]				L	15-30	8.0	1.3	7.9	0.5
PON/05/03/02		Bg	35-60	0.8				CL					
PON/05/03/02		Cg	60-90	0.6 [w]				CL					

[z] Landforms calculated from digital elevation data using method of MacMillan et al. (2000).
[y] Measured at sample date.
[x] Agronomic sample depths of 0- to 15-cm and 0- to 30-cm used for the cultivated sites.
[w] Organic matter levels in Ck horizons are suspect, since loss on ignition method may include CO_2 loss from carbonates. Value may be zero.
[v] Organic matter levels where pH 7.7 – 8.3 are suspect, since loss on ignition method may include CO_2 loss from carbonates. Value may be 1% lower than reported.
[u] Additional samples taken from lower landforms with high wetness index (Hi WI, Quinn et al. 1995).

Appendix 4. Soil-test phosphorus results from the microwatershed sites.

Table A4.1. Fall 2002 reference-corrected soil-test phosphorus (STP) results from the microwatershed sites.

Site	Sample point	Date	STP 0 - 2.5 cm (mg kg^{-1})	STP 2.5 - 5 cm (mg kg^{-1})	STP 5 - 15 cm (mg kg^{-1})	Notes
CFT	1	10/17-18/2002	43	39	15	
CFT	2	10/17-18/2002	40	34	15	
CFT	3	10/17-18/2002	44	50	16	
CFT	4	10/17-18/2002	34	32	10	
CFT	5	10/17-18/2002	39	35	28	
CFT	6	10/17-18/2002	40	34	25	
CFT	7	10/17-18/2002	52	63	49	
CFT	8	10/17-18/2002	38	28	40	
CFT	9	10/17-18/2002	32	33	21	
CFT	10	10/17-18/2002	24	24	9	
CFT	11	10/17-18/2002	40	40	36	
CFT	12	10/17-18/2002	22	17	10	
CFT	13	10/17-18/2002	27	24	30	
CFT	14	10/17-18/2002	29	29	12	
CFT	15	10/17-18/2002	58	59	33	
CFT	16	10/17-18/2002	29	33	7	
CFT	17	10/17-18/2002	53	54	25	
CFT	18	10/17-18/2002	38	36	27	
CFT	19	10/17-18/2002	166	184	67	Rerun confirmed, manured
CFT	20	10/17-18/2002	47	56	29	
CFT	21	10/17-18/2002	34	29	43	
CFT	22	10/17-18/2002	46	37	47	
CFT	23	10/17-18/2002	39	37	15	
CFT	24	10/17-18/2002	28	26	16	
CFT	25	10/17-18/2002	23	24	10	
CFT	26	10/17-18/2002	79	87	35	Rerun confirmed
CFT	27	10/17-18/2002	35	33	28	
CFT	28	10/17-18/2002	43	41	44	
CFT	29	10/17-18/2002	33	28	26	
CFT	30	10/17-18/2002	47	60	25	Rerun confirmed
CFT	31	10/17-18/2002	34	21	34	
CFT	32	10/17-18/2002	73	80	68	Rerun confirmed
CFT	33	10/17-18/2002	40	34	35	
CFT	34	10/17-18/2002	43	57	41	
CFT	35	10/17-18/2002	33	29	24	
CFT	36	10/17-18/2002	52	56	57	
CFT	37	10/17-18/2002	43	43	56	Rerun confirmed (5-15)
CFT	38	10/17-18/2002	28	29	34	
CFT	39	10/17-18/2002	31	26	32	
CFT	40	10/17-18/2002	31	25	37	

Table A4.1. Fall 2002 reference-corrected soil-test phosphorus (STP) results from the microwatershed sites.

Site	Sample point	Date	STP 0 - 2.5 cm (mg kg^{-1})	STP 2.5 - 5 cm (mg kg^{-1})	STP 5 - 15 cm (mg kg^{-1})	Notes
CFT	41	10/17-18/2002	54	59	25	
CFT	42	10/17-18/2002	14	17	10	
CFT	43	10/17-18/2002	49	51	27	
CFT	44	10/17-18/2002	32	29	5	
CFT	45	10/17-18/2002	49	47	27	
CFT	46	10/17-18/2002	32	22	35	
CFT	47	10/17-18/2002	50	54	39	
CFT	48	10/17-18/2002	62	63	38	
GPC	1	11/5-6/2002	34	34	26	
GPC	4	11/5-6/2002	66	33	10	Rerun confirmed (0-2.5)
GPC	5	11/5-6/2002	28	28	22	
GPC	6	11/5-6/2002	40	44	35	
GPC	7	11/5-6/2002	49	49	31	
GPC	9	11/5-6/2002	39	40	23	
GPC	10	11/5-6/2002	50	45	33	
GPC	11	11/5-6/2002	46	45	31	
GPC	12	11/5-6/2002	34	17	12	
GPC	13	11/5-6/2002	37	38	17	
GPC	15	11/5-6/2002	38	40	23	
GPC	16	11/5-6/2002	57	55	38	
GPC	17	11/5-6/2002	49	51	34	
GPC	18	11/5-6/2002	38	40	27	
GPC	21	11/5-6/2002	54	57	32	Rerun confirmed
GPC	22	11/5-6/2002	52	55	37	
GPC	23	11/5-6/2002	41	43	24	
GPC	24	11/5-6/2002	40	44	29	
GPC	26	11/5-6/2002	56	46	18	
GPC	27	11/5-6/2002	77	54	24	Rerun confirmed (0-2.5, 2.5-5)
GPC	28	11/5-6/2002	43	49	29	
GPC	30	11/5-6/2002	28	33	21	
GPC	31		.	.	.	New contributing area
GPC	32		.	.	.	New contributing area
GPC	33		.	.	.	New contributing area
GPC	34		.	.	.	New contributing area
GPC	35		.	.	.	New contributing area
GPC	36		.	.	.	New contributing area
LLB	1	12/3-4/2002	226	226	191	
LLB	2	12/3-4/2002	437	322	313	
LLB	3	12/3-4/2002	296	296	226	
LLB	4	12/3-4/2002	226	226	148	
LLB	5	12/3-4/2002	200	209	200	
LLB	6	12/3-4/2002	174	157	44	

Table A4.1. Fall 2002 reference-corrected soil-test phosphorus (STP) results from the microwatershed sites.

Site	Sample point	Date	STP 0 - 2.5 cm (mg kg^{-1})	STP 2.5 - 5 cm (mg kg^{-1})	STP 5 - 15 cm (mg kg^{-1})	Notes
LLB	7	12/3-4/2002	392	374	252	
LLB	8	12/3-4/2002	383	348	339	
LLB	9	12/3-4/2002	523	409	313	Replaced with rerun since initial > DL[z]
LLB	10	12/3-4/2002	461	475	331	
LLB	11	12/3-4/2002	287	313	296	
LLB	12	12/3-4/2002	287	531	512	Rerun confirmed
LLB	13	12/3-4/2002	518	313	322	
LLB	14	12/3-4/2002	218	226	157	
LLB	15	12/3-4/2002	313	287	261	
LLB	16	12/3-4/2002	339	458	183	Rerun confirmed
LLB	17	12/3-4/2002	348	348	218	
LLB	18	12/3-4/2002	209	200	165	
LLB	19	12/3-4/2002	183	209	174	
LLB	20	12/3-4/2002	131	139	148	
LLB	21	12/3-4/2002	157	96	70	
LLB	22	12/3-4/2002	365	400	357	Rerun confirmed
LLB	23	12/3-4/2002	365	374	313	
LLB	24	12/16/2002	539	503	400	
LLB	25	12/16/2002	331	331	261	
LLB	26	12/3-4/2002	261	244	218	
LLB	27	12/3-4/2002	374	331	313	
LLB	28	12/3-4/2002	497	392	322	
LLB	29	12/3-4/2002	270	235	200	
LLB	30	12/3-4/2002	244	226	200	
LLB	31	12/3-4/2002	426	374	313	
LLB	32	12/3-4/2002	270	235	148	
LLB	33	12/3-4/2002	442	287	270	
LLB	34	12/3-4/2002	296	322	191	
LLB	35	12/3-4/2002	235	191	183	
LLB	36	12/3-4/2002	305	322	278	
LLB	37	12/3-4/2002	418	400	383	
LLB	38	12/3-4/2002	458	418	331	
LLB	39	12/3-4/2002	392	400	400	
LLB	40	12/3-4/2002	165	139	139	
LLB	41	12/3-4/2002	374	357	313	
LLB	42	12/3-4/2002	252	252	226	
LLB	43	12/3-4/2002	339	305	305	
LLB	44	12/3-4/2002	218	148	104	
LLB	45	12/3-4/2002	96	74	28	
PON	1	11/12/2002	837	772	344	
PON	2	11/12/2002	391	466	419	
PON	3	11/12/2002	502	800	553	Rerun confirmed