Air, Soil and Water Research

ORIGINAL RESEARCH

Sediment and Nutrient Contributions from Subsurface Drains and Point Sources to an Agricultural Watershed

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Abstract: Excess sediment and nutrients in surface waters can threaten aquatic life. To determine the relative importance of subsurface drainage as a pathway for movement of sediment and nutrients to surface waters, loading from various tile systems was compared to that from sewage treatment plants (STP) within the same watershed. Movement through tiles comprised 1 to 8% of estimated total (overland plus tile) annual sediment loading from the respective areas drained by the tile. Load during the growing season from five *closed* drainage systems without surface inlets averaged 5 kg sediment/ha, 0.005 kg dissolved reactive P (DRP)/ha, 0.003 kg NH_4-N/ha, and 3.8 kg NO_3-N/ha; and from two *open* drainage systems with surface inlets averaged 14 kg sediment/ha, 0.03 kg DRP/ha, 0.04 kg NH_4-N/ha, and 3.1 kg NO_3-N/ha. The eight STP contributed about 44 530 kg suspended sediments, 3380 kg total P, 1340 kg NH_4-N, and 116 900 kg NO_3-N to the watershed annually. Drainage systems added less NH_4-N and P, but more NO_3-N and suspended solids to surface waters than STP. Tile drainage pathways for NO_3-N, STP in the case of P, and overland pathways for sediment are indicated as targets to control loading in artificially drained agricultural watersheds.

Keywords: phosphorus, nitrogen, sewage treatment, tile

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Introduction

Nutrient and sediment loading to surface waters can threaten aquatic species.[1,2] Enrichment of P and N depletes oxygen and the NH_3-N form of N can be directly toxic to aquatic life.[3] Turbidity is believed to be the main factor limiting the distribution of rare aquatic species in the Sydenham watershed of Ontario, Canada.[2] Within this river system reside: five species that are considered endangered; three classified as threatened; and six of special concern, according to the Committee on Status of Endangered Wildlife in Canada; along with an additional 19 rare species.[2] To protect species at risk in the Sydenham River, a Recovery Team was formed. Determination of the major pathways of sediments and nutrients to the river was a research and monitoring priority that they identified. This information was deemed necessary to develop and implement their Recovery Strategy.

Subsurface drainage may be an important pathway for sediment and nutrient movement to surface waters in agricultural watersheds that are extensively tile drained such as the Sydenham.[2,4,5] In the humid regions, approximately 8 M ha in Canada and 44 M ha in the US are artificially drained.[6,7] Nutrient movement to tiles is affected by nutrient application rate[8,9] and timing,[10] soil characteristics[11] and crop.[12] Sediment movement to tiles may be affected by management, but the trends and quantities reported are inconsistent.[13–16] Sediment movement to tiles increased with tillage in some studies,[13,14] but decreased with tillage in others.[15] Loads vary widely within and across studies e.g. from 97 kg/ha/yr under bluegrass sod (*Poa pratensis* L.) to 407 kg/ha/yr under continuous corn (*Zea mays* L.) in clay soil,[16] and from 13 kg/ha under conventional till to 61 kg/ha under ridge till in clay loam.[15] Discrepancies in quantities and management effects on sediment movement through tile might arise due to site-specific differences such as soil texture, or to imprecision (error >mean) in load calculations, which were often derived from estimated flows and/or concentrations.[13] The general lack of load data from tiles can be partly attributed to the difficulty in obtaining reliable measurements during large episodes when drain outlets tend to be submerged.[17]

In addition to diffuse sources such as agricultural lands, point sources such as sewage treatment plants contribute with suspended solids and nutrients to surface waters.[4] Knowledge of the relative inputs from point and non-point sources was also required for planning recovery actions.

Objectives

Our study was undertaken to compare sediment and nutrient contributions to surface waters from both STP and tiles draining fields with varied landscapes and management practices. Identification of sediment and nutrient sources and improved understanding of management effects can inform mitigation strategies. For example, if tiles are a major pathway, movement is primarily under rather than over the riparian zone, and buffers will be less effective to control movement.[2,18,19] Such information can be helpful in targeting resources to major pathways and practices that reduce turbidity and nutrient loading, and ultimately, improving habitat for aquatic species.

Materials and Methods

Diffuse sources

Sites and sampling

Within the Sydenham watershed, which has 85% agricultural land use,[2] tile drainage outlets located on five different farms ranging 81° 50–59' W, 42° 45–53' N (Fig. 1, Table 1) were monitored in 2003 and 2004. In 2003 at each of three selected farms, monitored outlets (designated A, B, C) were from *closed* systems without surface inlets draining land under annual crop production. The area drained by the systematic tile network or tile-shed, connected to Outlet A had complex slopes of up to 5%, and was planted with corn harvested for silage at the end of September 2003. Outlet A drained into a stream which flowed into a tributary called Haggerty Creek. A mowed grass buffer existed between the field and the stream. The tiles connected to Outlet B drained a field having slopes up to about 3%, which was cropped with soybean [*Glycine max* (L.) Merr.]. Water from Outlet B flowed into a roadside ditch and then through a culvert under a gravel road into a small creek. The creek was flanked by trees on one side and shrubs and grasses on the other for a distance of 64 m where it then emptied into the Sydenham River. The third outlet (C) drained a field with slopes of <1% and flowed into a municipal drainage ditch (Table 1). Monitoring at Outlet C began late (July, Fig. 2) because the sampling equipment was initially positioned at a different outlet which produced no flow throughout spring 2003.

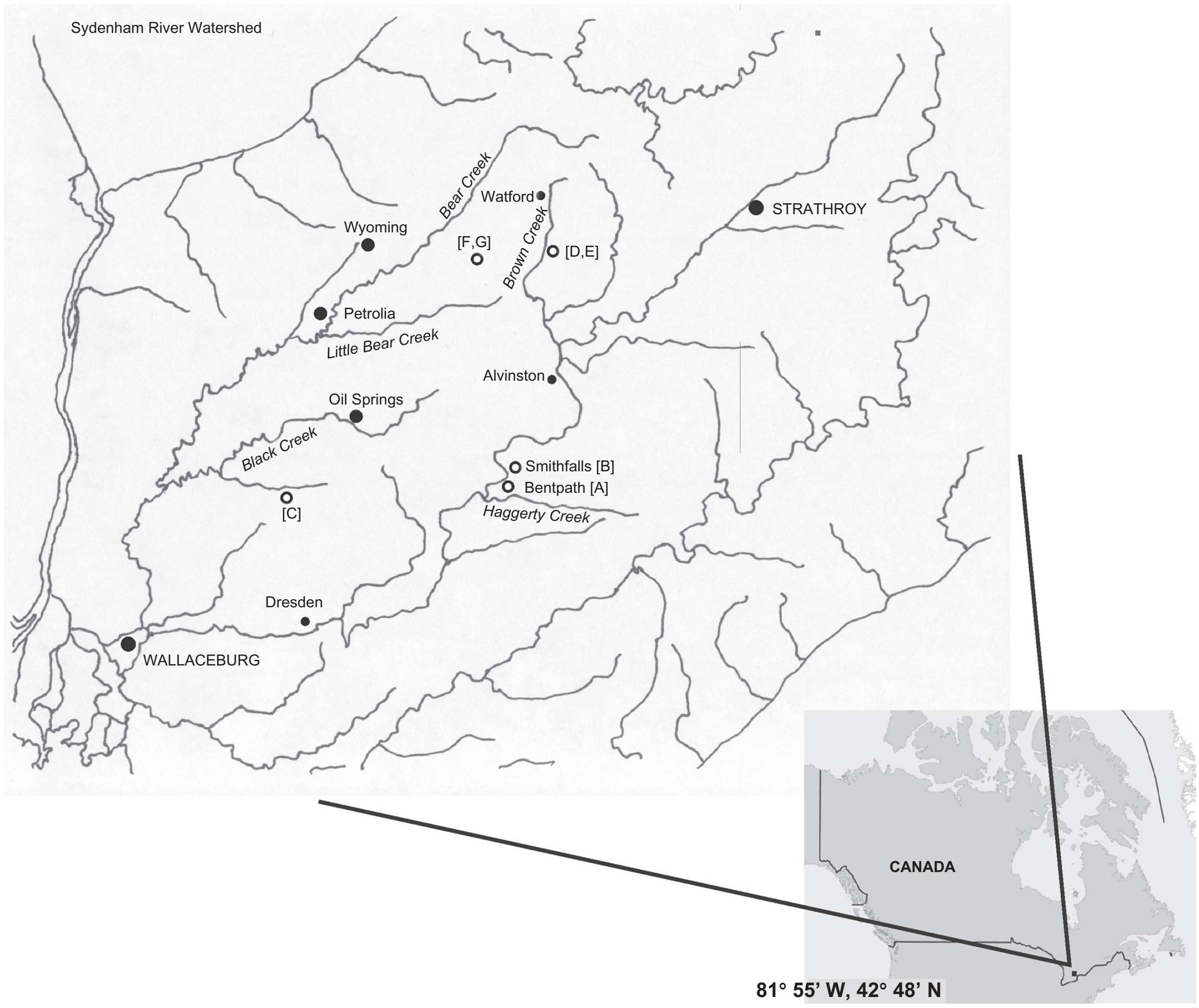

Figure 1. Sydenham watershed in Ontario, Canada with sampling Outlets A to G indicated with open circles. Sewage treatment plant locations and associated towns are indicated with filled circles.

In 2004 within each of two fields, one outlet draining a *closed* system and one outlet from an *open* system with surface inlets were monitored. One field was in perennial (Outlets D, E) and the other in annual crop production (Outlets F, G, Fig. 1, Table 1). The perennial forage field was well-established, mainly alfalfa (*Medicago sativa* L.). The *open* drainage system tile-shed (D) included a grass waterway, complex slopes up to 5% and two catch basins buffered by forage, with one basin located near a laneway. The *closed* tile-shed area (E) in the alfalfa field had <1% slope. Water exiting Outlets D and E flowed overland through grasses and trees to a tributary called Brown Creek. At the annual crop field, solid poultry manure was broadcast after winter wheat harvest in 2003 and soybeans were planted early June 2004. Both tile-sheds had <1% slope. The *open* system (F) included two hickenbottom surface inlets which are perforated risers, that were edged with gravel but had no vegetated buffer. Both the *closed* (G) and *open* (F) outlets drained into a ditch which flowed into a tributary called Little Bear Creek.

Outlet flows were determined by automated measurement of the depth of water flowing over a weir in a plywood box anchored below the outlet. Data were recorded every 5 min using a UL16 data logger and float installed on a pulley (FS 15, Lakewood Systems Ltd., Edmonton, AB) above a stilling well. Flow (Q) in m^3/s was calculated using:

$$Q = Cw \cdot h^x \quad (1)$$

where Cw equals the weir coefficient, which is 2.5 for a 90° V-notch; h equals head or height in m above the V from the float; and x is a constant. For each

Table 1. Land management and some physical and chemical topsoil properties in tile-sheds with *closed* drains exclusively below ground or *open* drains including surface inlets.

Outlet	Crop	Year	Drain system	Tillage	Area[1]	Topsoil characteristics					
						Texture	Silt[2]	Clay[2]	Organic matter	pH	P[3]
					(ha)		---------- (g/kg) ----------				(mg/L soil)
A	silage corn	2003	*closed*	no-till	10.0	sandy loam	170	140	25	7.2	13
B	soybean	2003	*closed*	no-till	3.7	loamy sand	90	60	30	7.9	19
C	grain corn	2003	*closed*	conventional	9.9	silty clay	390	460	42	7.8	50
D	alfalfa	2004	*open*	no-till	4.1	silt loam	550	230	46	7.3	10
E	alfalfa	2004	*closed*	no-till	1.1	silty clay loam	530	310	43	7.0	30
F	soybean	2004	*open*	conventional	14.2	silty clay	460	450	48	7.4	53
G	soybean	2004	*closed*	conventional	7.3	silty clay	470	410	44	7.7	50

[1]Area drained by the network of tile, flowing to the monitored outlet.
[2]Hydrometer method.[20]
[3]Bicarbonate extractable.

weir box, x was obtained by measuring heights of known flows. Water from the outlets was collected using automated samplers (Isco Inc., Lincoln, NE) with hoses positioned inside the weir boxes. When flow was continuous, samplers were usually programmed for collection every 8, or sometimes 12 h, but for more frequent collection, e.g. every 3 h, if rain was imminently forecast. If tiles were not flowing, liquid level actuators (Isco Inc., Lincoln, NE) positioned inside the boxes triggered sampling to commence along with flow. For start-up flows, samples were collected with decreasing frequency over time, i.e. six samples every 15 min, followed by two every 30 min, seven every 1 h, one every 3 h and every 6 h thereafter until the next site visit or until all 24 bottles were filled.

Tipping buckets (TE525) and automated reflectometers (CS-615) connected to data loggers (Campbell Scientific Inc., Edmonton, AB) recorded rainfall and soil volumetric water content (θv). Reflectometers were calibrated using soil from the same field, and were installed in the top 0.3 m of soil. Two reflectometers and one tipping bucket were installed at location A from 16 Apr. to 5 Nov. 2003, at C from 15 July to 5 Nov. 2003, at D/E from 28 Apr. to 28 Nov. 2004 with reflectometers in the Outlet E tile-shed, and at F/G from 5 May to 28 Nov. 2004 with reflectometers in the Outlet G tile-shed. These data were not collected at Outlet B since it was only a few kilometers from Outlet A. A Com100 Motorolla cellular transceiver with Com200 modem (Campbell Scientific Inc., Edmonton, AB) was connected to one data logger each year to remotely determine the occurrence of rainfall and facilitate timely collection of samples. To characterize tile-shed areas, selected physical and chemical properties were determined in soil samples collected from the top 0.2 m (Table 1).

Samples of surface water into which outlets drained were collected periodically, either manually from creeks receiving Outlets A, B and C in 2003, or using an automated sampler from the creek receiving Outlets F and G in 2004. No surface water existed adjacent to the field drained by Outlets D and E. Water was also collected from the Sydenham River downstream of the creek receiving Outlet B. Creek flows were measured near Outlets A and B about 5 h after peak rain intensity on 3 Nov. 2003. Downstream of Outlet A, flow was determined from the height of water over a V-notch in a weir board which was 2.1 m long and installed across the stream. Downstream of Outlet B, flow was determined from time to capture known volumes of water flowing out of the culvert.

Water analyses and load calculations

Sediment concentrations were determined from weights obtained by filtering known volumes of water,

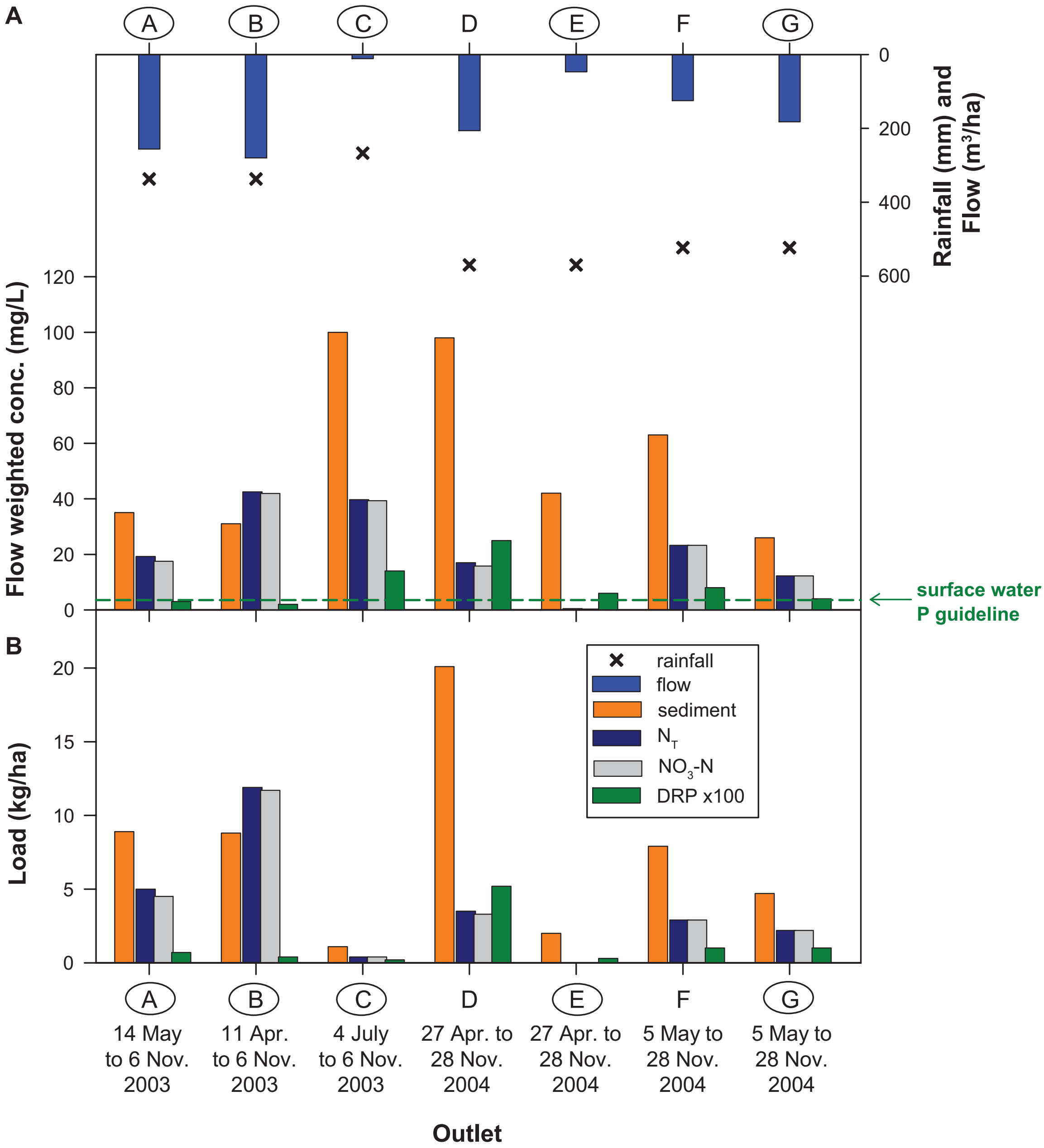

Figure 2. Seasonal rainfall, flow, and sediment and nutrient flow-weighted concentrations **A**) and load **B)** from outlets of *closed* drainage systems that were exclusively below ground (circled outlets) or *open* drains that included surface inlets. Rain amount for outlets F and G includes 132 mm during times outlets were flooded and flow data not available.
Abbreviations: N_T, total N; DRP, dissolved reactive phosphorus.

usually 200 ml, through 0.45 µm, then oven-drying at 90 °C. The filtrate and unfiltered water samples were then frozen until analyzed. Flow injection (Lachat Instruments, Milwaukee, WI) colorimetry was used for all analytes. In-line digestion was used to determine total P (Tot-P) and total N[21,22] in non-filtered samples. Concentrations of dissolved NO_3-N, NH_4-N, and molybdate-reactive phosphate (DRP)[22–24] were determined in filtered samples.

Sediment and nutrient loads from outlets were calculated for each time increment between water samples and summed over time:

$$\text{Load} = \Sigma_0^t S \cdot V \qquad (2)$$

where S is measured constituent concentration (mg/L) at time t, and V is flow volume since the previous sample in L/time interval, as calculated from Q in equation (1). The amounts summed over the season were converted to a ha basis according to the tile-shed areas listed in Table 1. At the 2004 soybean field, load could not be calculated when excessive rains of 89 mm from 21–23 May and 43 mm on 9 June 2004 raised the water level in the ditch receiving Outlets F and G by about 3 m, above the level of outlets and monitoring equipment.

Overland sediment delivery to surface water was estimated for each tile-shed area using the Universal Soil Loss Equation with site specific parameters and a calibrated sediment delivery ratio,[25] which gives long term annual average. Overland sediment delivery to surface water from the sub-watershed upstream of Outlet A was similarly estimated. This long term annual average, along with measured outlet and creek sediment load on 3 Nov. 2003, was used to estimate sediment movement through tile over the winter period, which was not monitored, by assuming a constant contribution from Outlet A to creek load throughout the year. Creek sediment loads were calculated by multiplying measured flow volume with concurrent sediment concentrations.

Point sources

The eight STP located within the watershed (Fig. 1) provided information on treatment processes and data on flows and N and P concentrations, from which daily and annual loads were calculated. For Strathroy, the calculations were based on July 2002 data. Not all constituents were measured at every plant. The calculated values underestimate total nutrient flows since instances of flooding (overflow) are not included. For comparison to point sources, load from tiles on a watershed basis was estimated using the measured seasonal average from the five monitored *closed* drain systems and the land area of the Sydenham watershed that is tile drained of 163 500 ha, which is 60% of the total area.[2]

Results

Diffuse sources

Flow responses

On wet soil, tiles generally flowed within a few hours of ≥10 mm rain. For example, Outlet A flowed with 13 mm rain over 4 h on 2 Nov. 2003 (Fig. 3); Outlet B

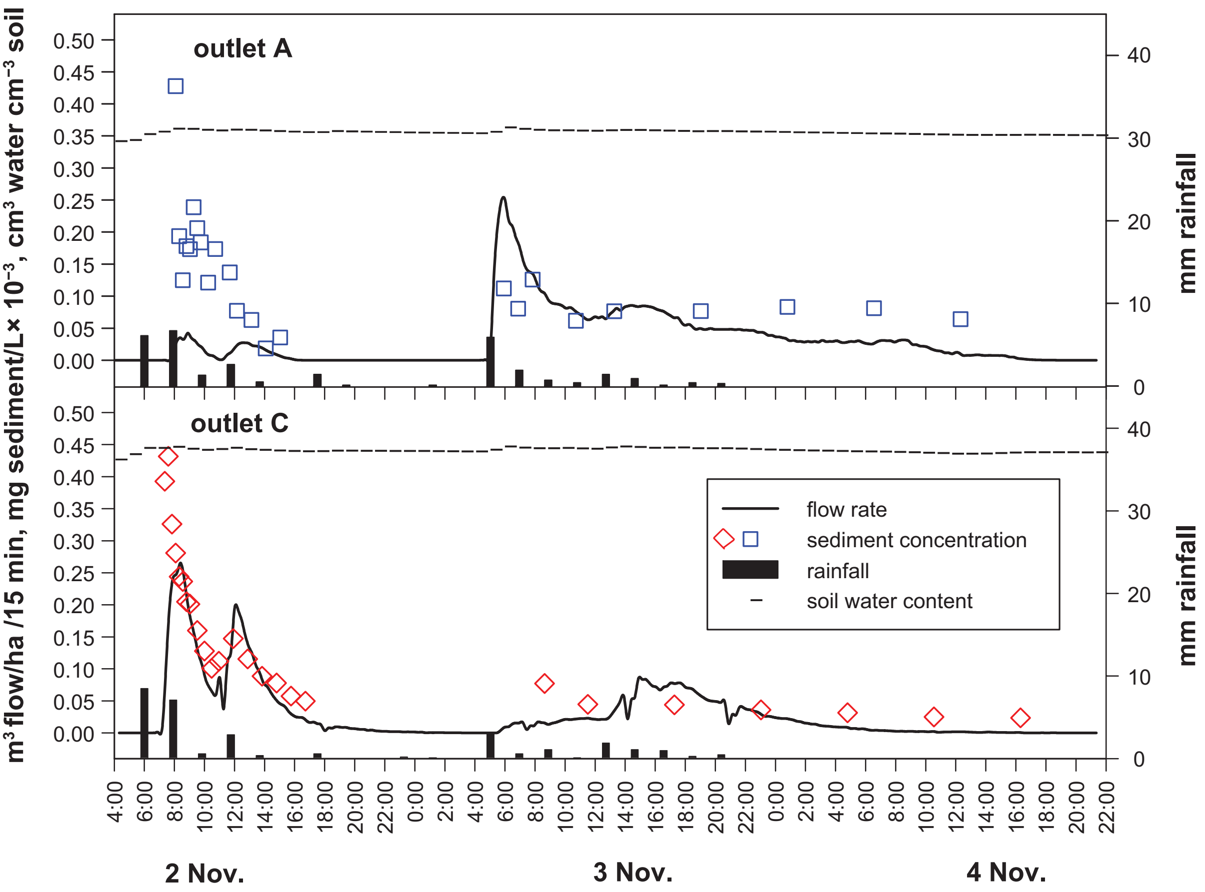

Figure 3. Soil water content, storm hydrograph and sediment concentration in tile water draining fields cropped with corn, Outlets **A** and **C**, in November 2003. Rainfall is shown in 2 h intervals.

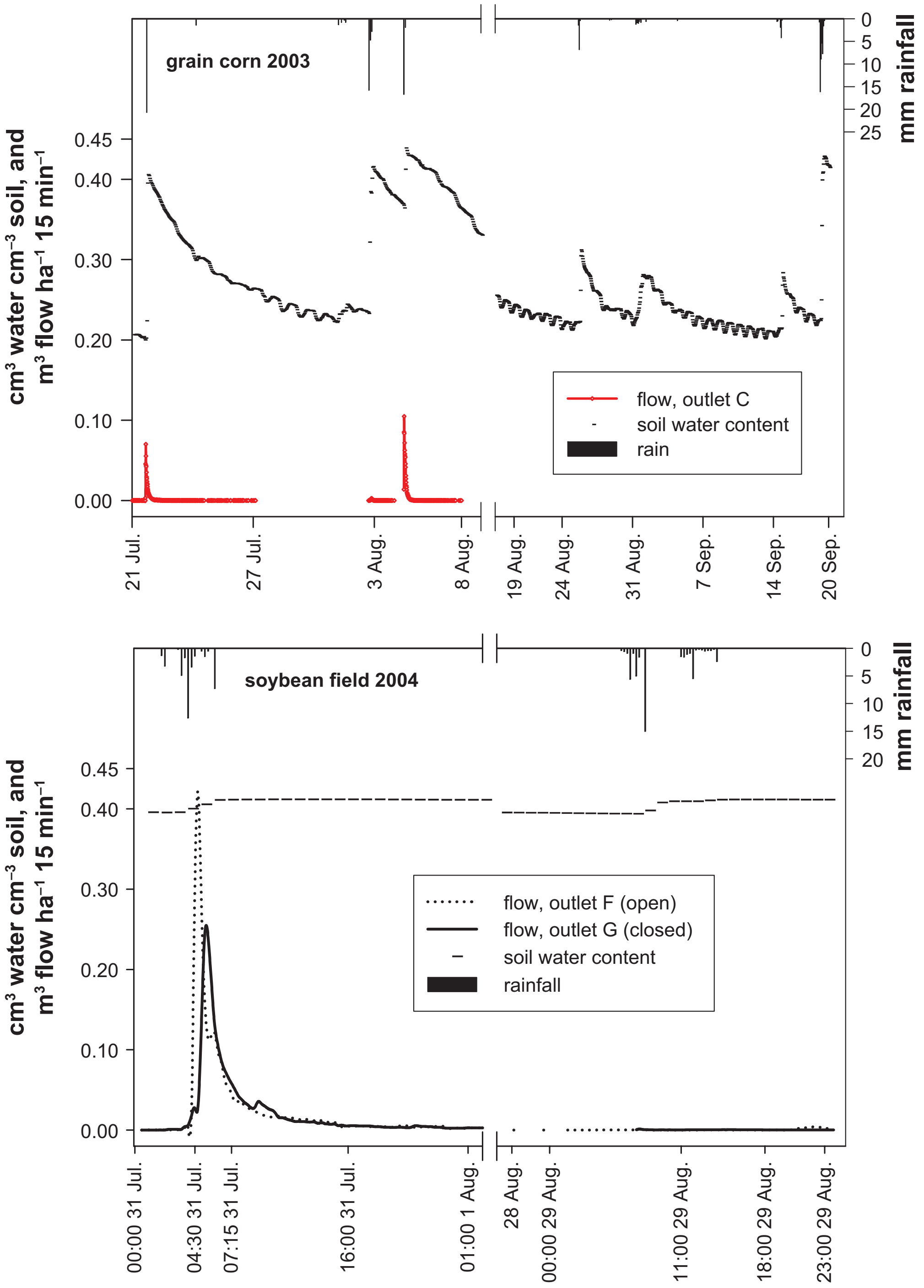

Figure 4. Storm hydrograph and soil water content in fields cropped with corn, Outlet C, from July to September 2003, and with soybeans, Outlets F and G, from July to August 2004. Rainfall is shown in 2 h intervals.

flowed following 6 mm in 1 h on 20 May and 12 mm over 12 h on 12 June, but not following 12 mm in 1 h on 5 July with dry soil, or 32 mm over 14 h on 14 Oct.; Outlet C flowed after 21 mm in 2 h on 21 July, 30 min after 17 mm in 2 h on 5 Aug. (Fig. 4) and after 14 mm over 3 h on 2 Nov. (Fig. 3), but not with 41 mm over 12 h on 19 Sept.; Outlet D (*open*) flowed with 12 mm in 30 min on 14 June 2004 (Fig. 5); Outlets F and G flowed after 11 mm rain in 8 h on 15 July (Fig. 6), but slowly following 23.5 mm over 8 h on 29 Aug. (Fig. 4). After 19 mm rain in 3 h on 31 July flow from Outlet F (*open*) peaked 45 min earlier than peak flow from Outlet G (*closed*) (Fig. 4). The *closed* system draining alfalfa (E) flowed less than the other

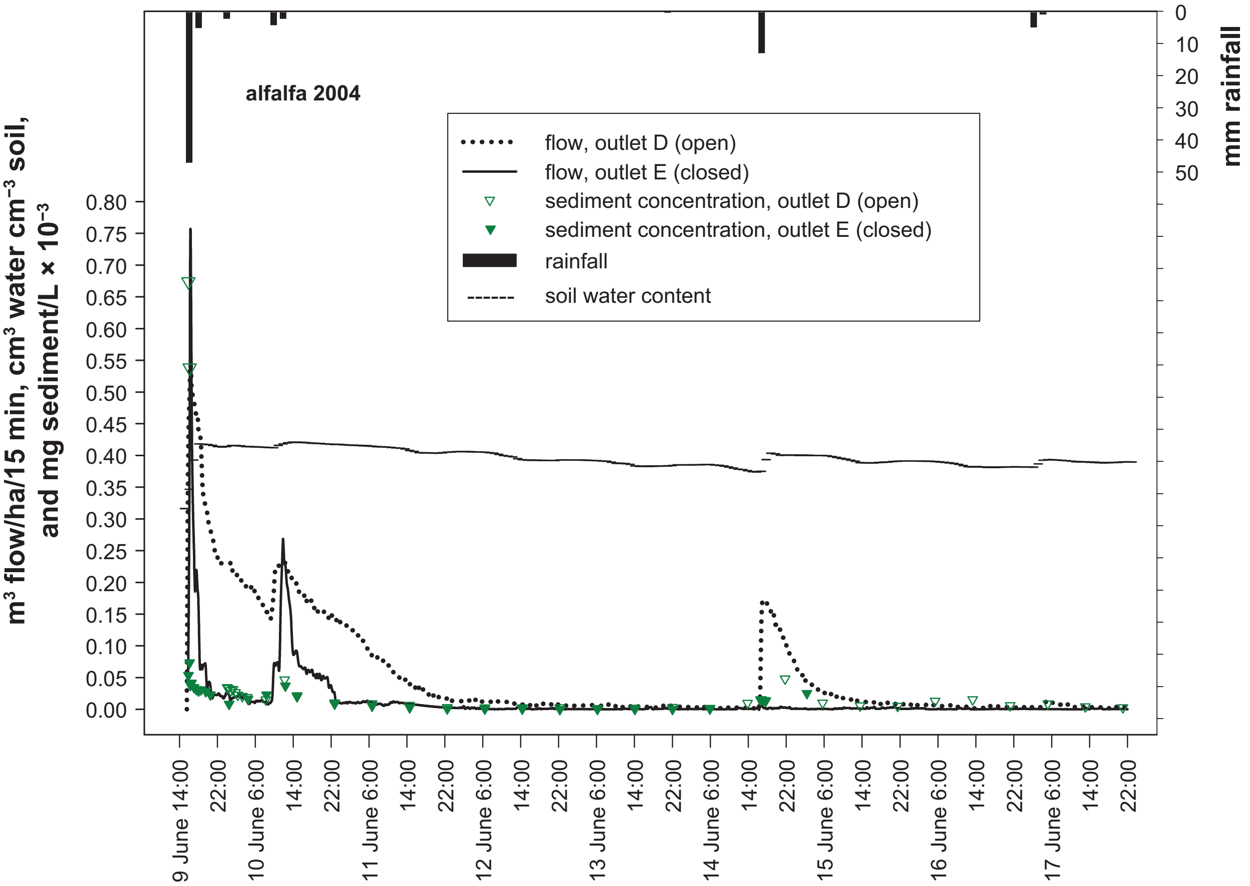

Figure 5. Soil water content, storm hydrograph and sediment concentration in tile water draining a field cropped with alfalfa, Outlets D and E, in June 2004. Rainfall is shown in 2 h intervals.

three outlets monitored in 2004 (Fig. 2), and flow was usually coincident with $\theta v > 0.40$ m³/m³ which is approximate field capacity (held at –2 kPa as determined in a pressure plate apparatus).

Sediment instantaneous and flow-weighted concentrations and load

Sediment concentration in tile drainage water was usually less than in adjacent receiving waters (creeks and river, Figs. 7, 8), regardless of flow condition or time since rain events. Sediment movement through tile was episodic, with greatest concentrations during the first few hours following intense rainfall on start-up of tile flow. For example, instantaneous sediment concentrations (data not shown) were: 442 mg/L at 14:25 on 5 Aug. and 432 mg/L at 7:40 on 2 Nov. 2003 from Outlet C; and 428 mg/L, which was the maximum observed, at 8:10 on 2 Nov. 2003 from Outlet A. Maximum observed sediment concentration from Outlet B was only 90 mg/L which occurred on 4 June at 2:47 and again on 14 June at 9:00, not coincident with greatest flow rates, unlike tendencies for the other outlets. Spikes in sediment concentrations were more dramatic in water collected from *open* than from *closed* systems within the same field (Fig. 8). For example, maximum sediment concentration was 1983 mg/L at 1:17 on 22 May from the *open* system D as compared with 83 mg/L at 7:30 on 23 May from the comparable *closed* system E. A similar pattern was observed on 9 June at 15:50 with 9-fold greater sediment concentration in water from Outlet D than E (Fig. 5), and on 14 July 2004 from the soybean field with greater sediment concentration in water from *open* Outlet F than *closed* Outlet G at 23:00 following 20.3 mm rain from 15:15 to 22:45 (Fig. 6). Seasonal flow-weighted sediment concentrations reflected this trend of more movement from *open* systems D and F than from *closed* systems E and G (Fig. 2).

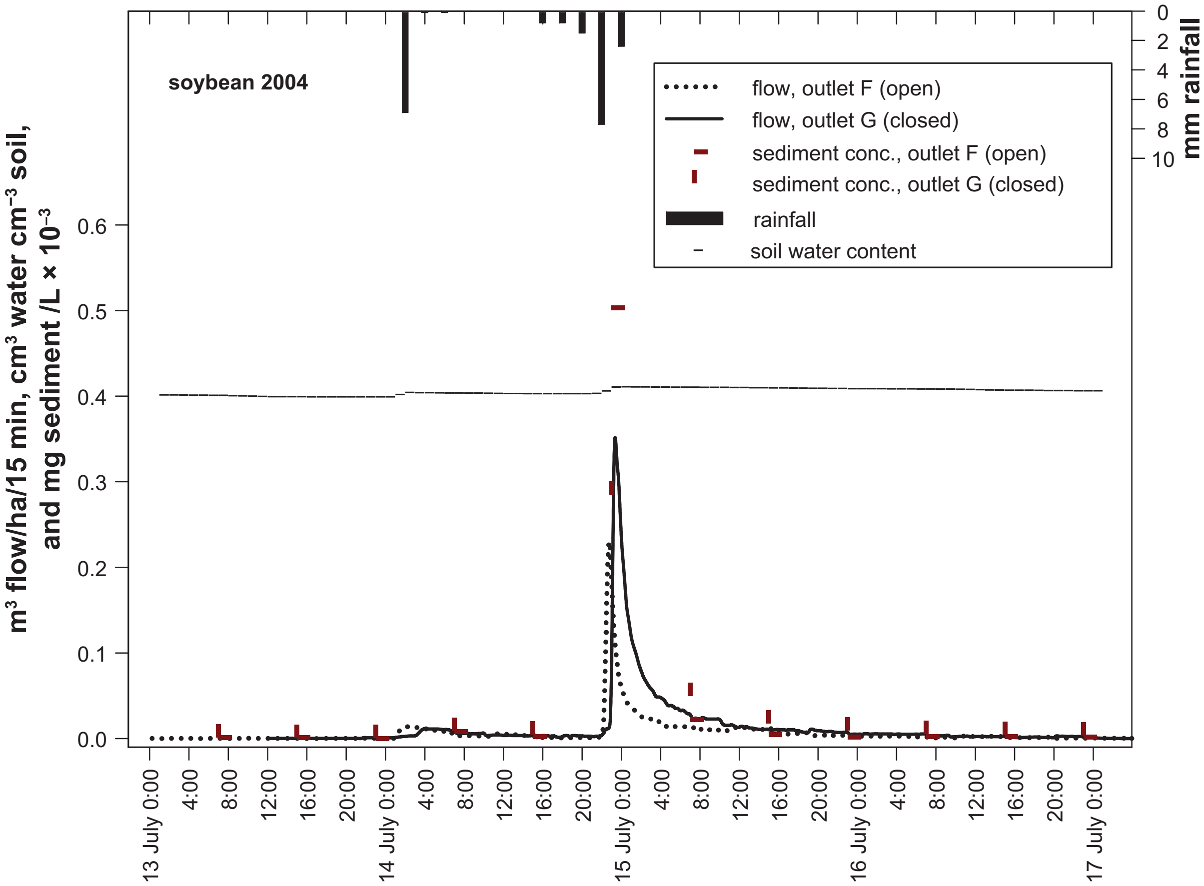

Figure 6. Soil water content, storm hydrograph and sediment concentration in tile water draining a field cropped with soybeans, Outlets F and G, in July 2004. Rainfall is shown in 2 h intervals.

Cumulative sediment loads on a drained area (ha) basis were similar among *closed* system Outlets A, B and G with season-long monitoring, but less from both Outlet C which was not monitored during spring, and Outlet E which drained alfalfa (Fig. 2). Ten-fold more sediment moved through the drain system that had surface inlets than through the comparable *closed* system at the alfalfa field and 2-fold more at the soybean field (Fig. 2). Greater sediment load through the *open* system at the alfalfa (D) than at the soybean (F) field (Fig. 2) may be in part attributed to missing data when sediment movement could not be quantified at the soybean field due to flooding during unusually wet May and early June weather. Rainfall in May 2004 was 159 mm, double the long-term May average of 79 mm. With flood dates excluded from the totals, sediment movement through the *open* system at the alfalfa field was 1.3 kg/ha, which was less than through the *open* system at the soybean field (Fig. 2).

Long-term average annual overland sediment delivery to surface waters from the tile-shed areas drained by Outlets A, B, C, D, E, F and G was an estimated 170, 230, 280, 1300, 60, 700, and 980 kg/ha/yr, respectively. Sediment load in the creek receiving Outlet B was 2.7 kg/h on 3 Nov. 2003, comprising surface and subsurface contributions from the 69 ha upstream sub-watershed, with no corresponding flow from Outlet B. Sediment load measured in the stream receiving Outlet A on this date was 12.3 kg/h, during which time sediment exiting Outlet A was equivalent to 1% of the stream load. Annual overland sediment delivery from the 187 ha upstream (of A) sub-watershed was an estimated 14 700 kg/yr (long term average). If the outlet contributes a constant 1% to stream load over time, annual sediment movement through this tile system including winter, when outlets were not monitored, is approximately 15 kg/ha/yr.

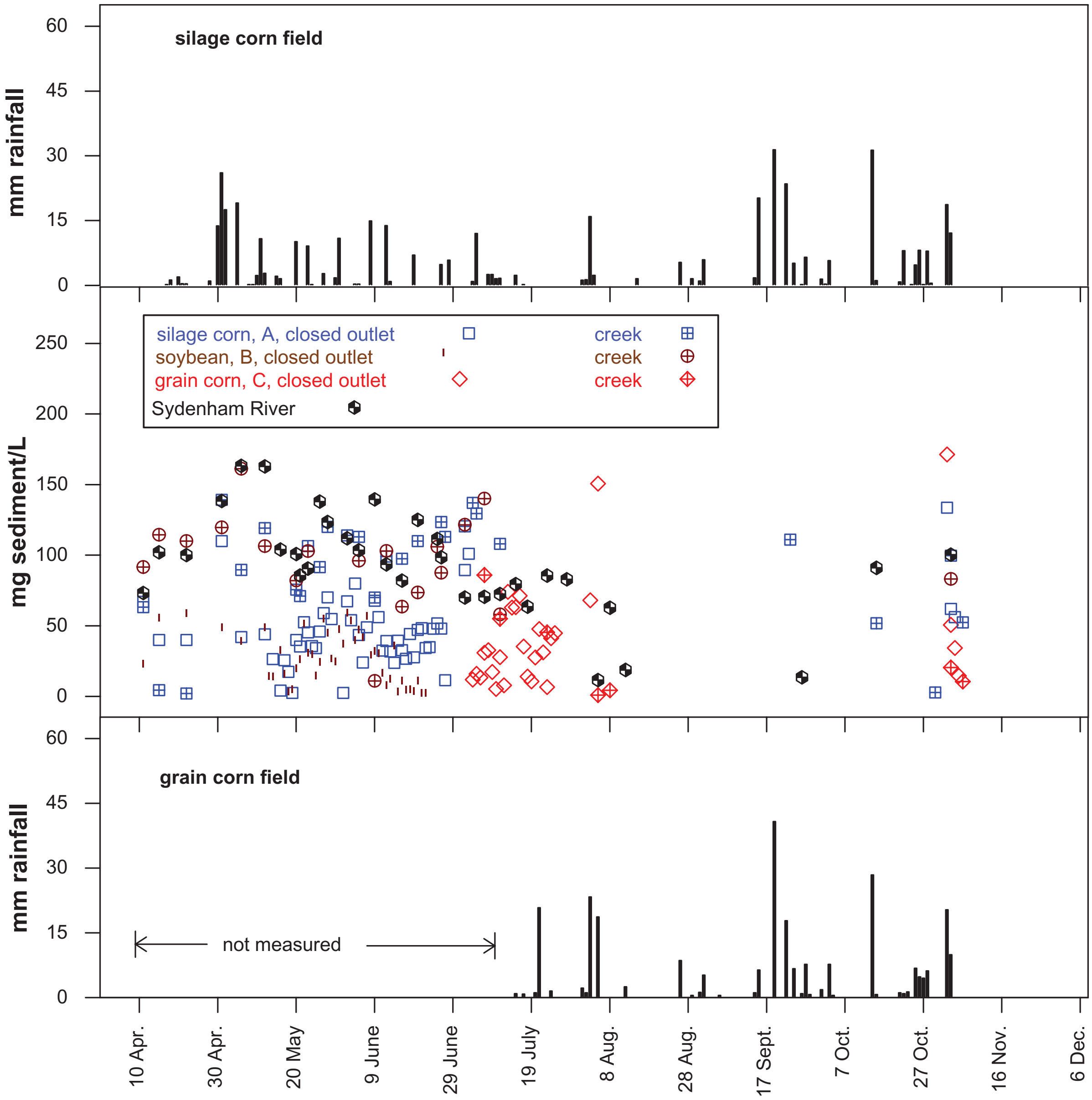

Figure 7. Rainfall and daily average sediment concentration in outflows from *closed* drainage system Outlets A (silage corn), B (soybeans), and C (grain corn), adjacent surface waters and the Sydenham River in 2003.

Phosphorus instantaneous and flow-weighted concentrations and load

In spring and summer, DRP concentrations in water from Outlets A and B were similar, usually <0.04 mg/L, except when it was 0.06 mg/L on 24 June from Outlet B for both daily average (Fig. 9) and instantaneous concentrations. In the fall however, when Outlet B had no flow, DRP concentrations in water from Outlet A rose to 0.78 mg/L instantaneous concentration at 9:00 (data not shown) and 0.48 mg/L daily average (Fig. 9) on 2 Nov. after no recorded flow since 4 July. From Outlet C, DRP concentration was greatest on 14 July with 0.27 mg/L at 18:00 and 0.21 mg/L daily average, and usually exceeded concentrations measured in the creeks and river in 2003 (Fig. 9). In 2004, maximum instantaneous DRP concentrations occurred: on 2 May for Outlet D with 0.98 mg/L at 14:14; on 9 June for Outlet E with 0.17 mg/L at 15:50; and on 5 Nov. for Outlet F with 0.32 mg/L at 16:00 and Outlet G with 0.10 mg/L at 3:00. Daily average and seasonal flow-weighted DRP concentrations were greater in water from *open* than *closed* drains within the same field (Figs. 2, 10).

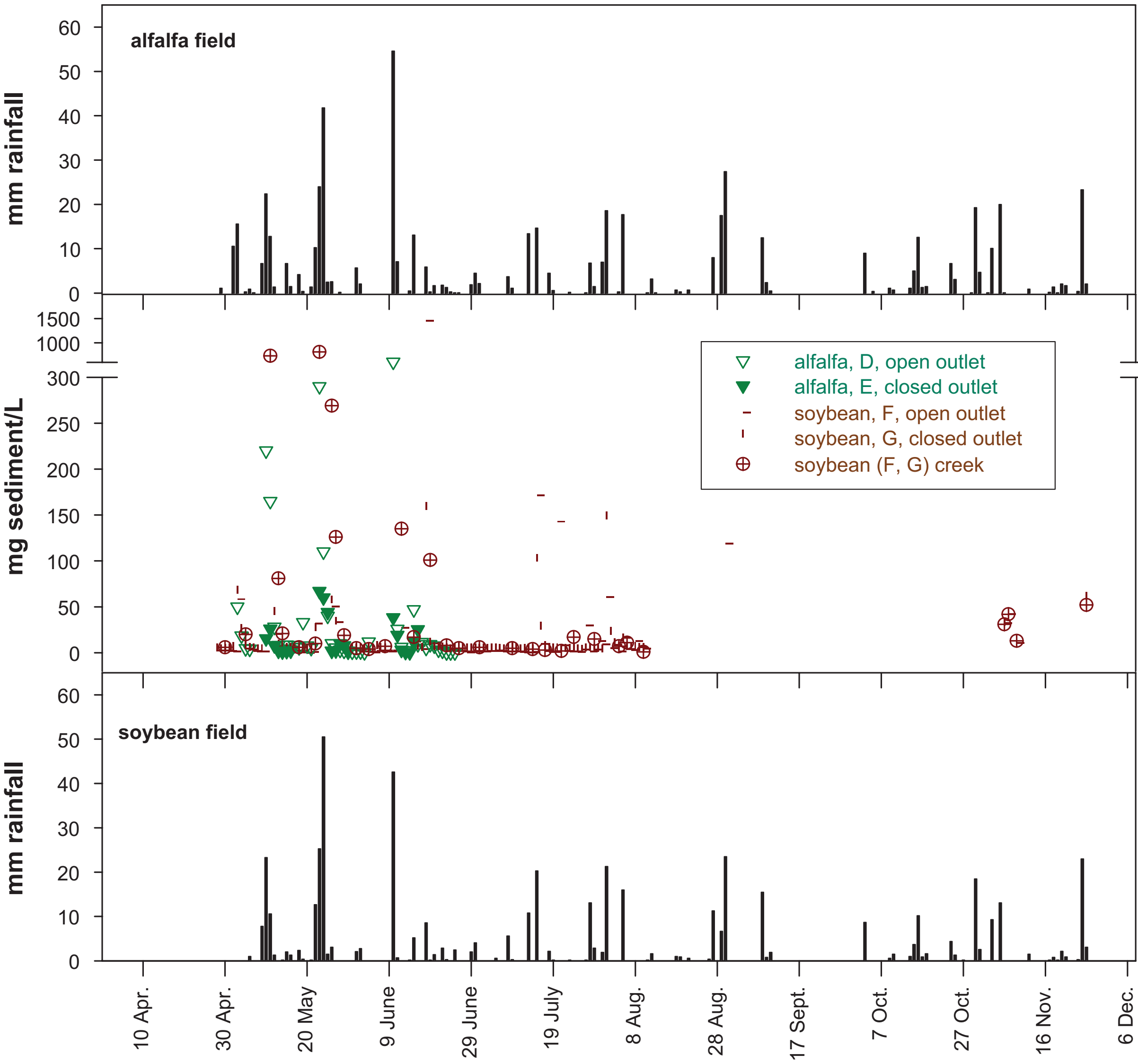

Figure 8. Rainfall and daily average sediment concentration in outflows from *closed* and *open* drainage system Outlets D and E (alfalfa) and F and G (soybeans), and in the adjacent creek at the soybean field in 2004.

Nitrogen instantaneous and flow-weighted concentrations and load

In tile drainage water, total N was comprised mainly of NO_3-N (Fig. 2). Greatest daily average NH_4-N concentrations were observed from Outlet B draining soybeans on 24 June 2003 (Fig. 11) and Outlet D draining alfalfa on 9 June 2004 (Fig. 12). Elevated concentrations of NH_4-N were also observed on 11 Apr. 2003 from Outlet A (Fig. 11) and on 29 June 2004 from the *open* Outlet (F) draining soybeans (Fig. 12). Seasonal NH_4-N load through tile totaled: 3.3, 1.9, 0.4, 50.3, 3.3, 3.5 and 4.2 g/ha from Outlets A, B, C, D, E, F and G, respectively. Concentration maxima of NH_4-N in surface waters occurred in spring or early summer—on 1 May in the creek receiving Outlet A and 7 July 2003 in the creek receiving Outlet C (Fig. 11) and on 23 May 2004 in the creek receiving Outlets F and G (Fig. 12) with an instantaneous concentration of 0.68 mg/L at 4:00.

Surface water NO_3-N concentration dynamics differed from that of NH_4-N, with increased concentrations in October or November relative to July and August e.g. in the Sydenham River and creeks receiving Outlets C and F/G (Figs. 13, 14). In receiving surface waters, NO_3-N concentrations were less than in water from tile drains except Outlet F on 5 and 25 Nov. 2004 (Fig. 14). Among all outlets, B and C had highest seasonal flow-weighted total N and NO_3-N concentrations (Fig. 2). Outlet C, cropped with corn, had high NO_3-N concentrations in November, whereas the high

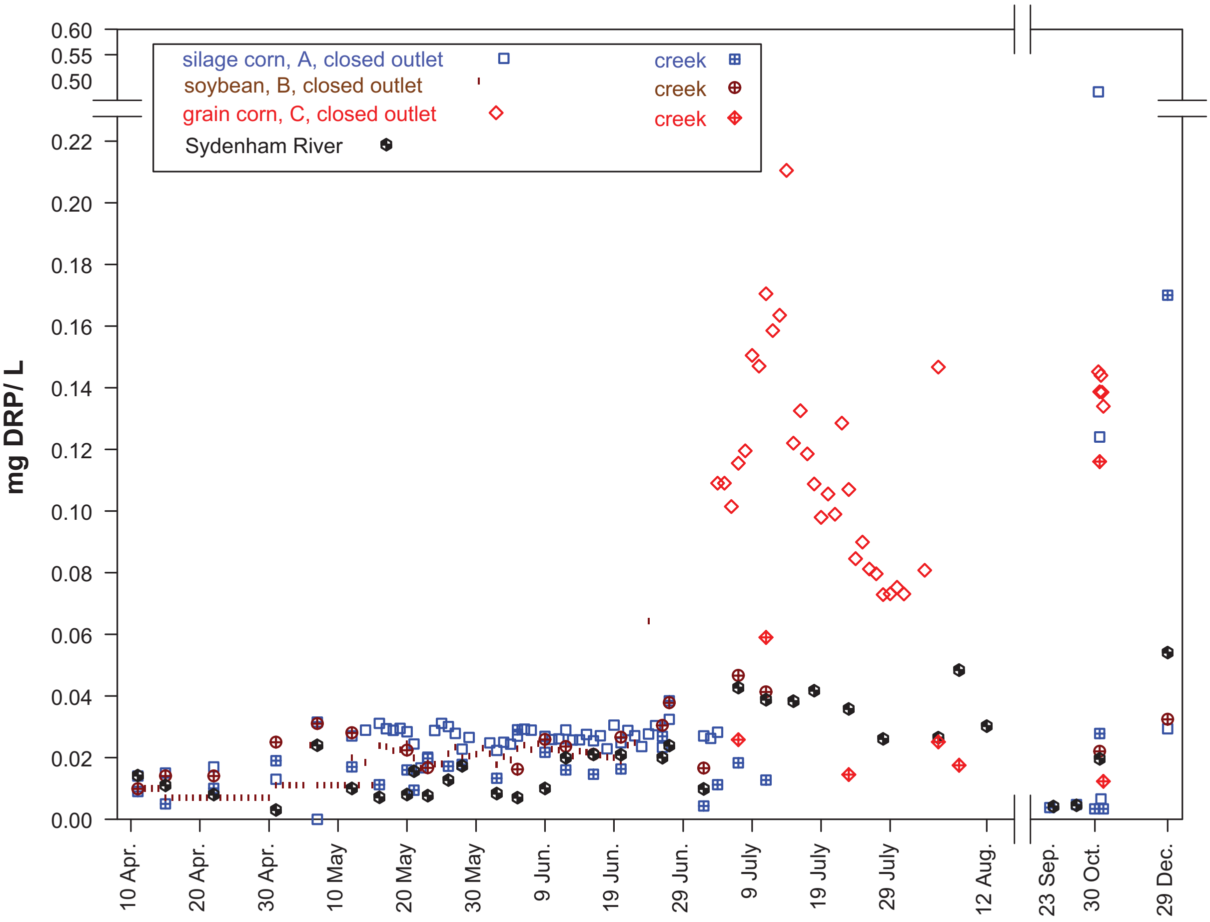

Figure 9. Daily average dissolved reactive P (DRP) concentration in outflows from *closed* drainage system Outlets A (silage corn), B (soybeans), and C (grain corn), adjacent surface waters and the Sydenham River in 2003.

concentrations from Outlet B, which was cropped with soybeans and did not flow in fall, occurred in spring and early summer (Fig. 13). Concentrations of NO_3-N were greater from *open* (D, F) than *closed* (E, G) drains (Figs. 2, 14), and were particularly low from Outlet E.

Point sources

The Strathroy plant has a tertiary water treatment process, with aeration, sand filter and UV treatment. Tertiary water treatment processes are also used at Wyoming and Petrolia, with extended aeration, sand filters and UV treatment. Extended aeration, P removal and disinfection water treatment processes are used at both the Alvinston and Dresden sites. Wallaceburg has 2-stage anaerobic digestion, belt press dewatering, and UV treatment. Oil Springs consists of a lagoon with batch P removal and has a 23 day discharge. Watford has a lagoon and 13 day discharge. Flows and concentrations of suspended solids and most P and N constituents were available for calculating annual loads from the STP effluent to the watershed (Fig. 15). Total load from all STP to the watershed was 44, 532 kg solids/yr, 3377 kg Tot-P/yr and 1342 kg NH_4-N/yr. Contribution of NO_3-N was much larger than that of NH_4-N or total Kjeldahl N which is comprised of organic- + NH_4-N (Fig. 15). NO_3-N data were not available from Strathroy and Petrolia. Assuming NO_3-N load contributed by Strathroy was the same as Wallaceburg, and from Petrolia the same as Wyoming (based on similarity of flows and concentrations of other constituents), total NO_3-N load from all STP was approximately 116, 900 kg/yr. Contribution from tile to the watershed, which was estimated from the seasonal average of the five monitored *closed* systems of 5.1 kg sediment/ha, 5 g DRP/ha, 3 g NH_4-N/ha and 3.8 kg NO_3-N/ha, over the total area tiled amounted to 833, 850 kg sediment, 818 kg DRP, 491 kg NH_4-N and 621, 300 kg NO_3-N.

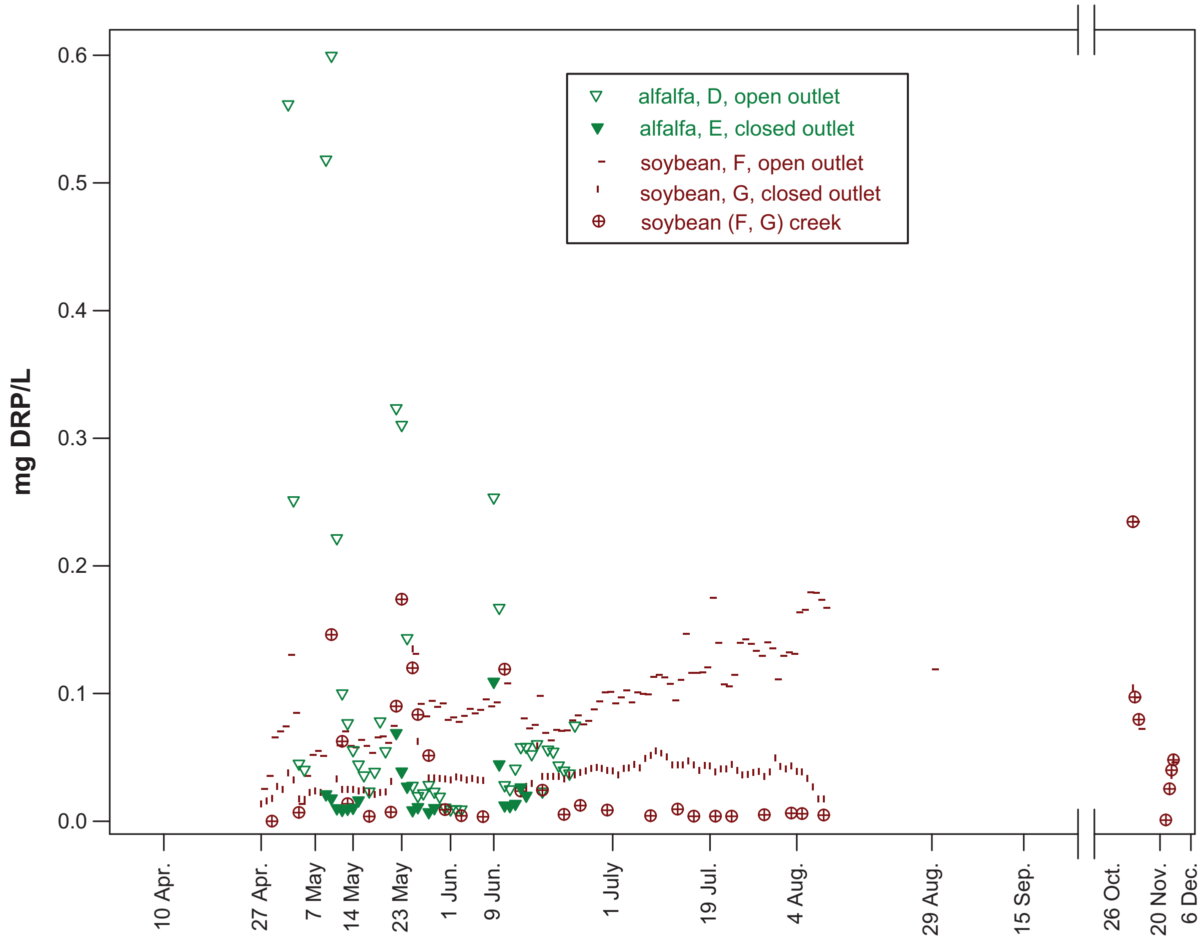

Figure 10. Daily average dissolved reactive P (DRP) concentration in outflows from *closed* and *open* drainage system Outlets D and E (alfalfa) and F and G (soybeans), and in the adjacent creek at the soybean field in 2004.

Discussion

The amount of precipitation required to trigger outlet flows depended on rain intensity, initial soil water content, presence of inlets, crop type and soil texture. The trend was a faster flow response with increased rain intensity, increased soil water content and presence of surface inlets; and slower or less response with alfalfa and coarse soil texture e.g. Outlet B flowed less frequently and drained a field of coarser texture than A. Contributions from drainage tile to base flows are variable and not well studied.[7] In an Illinois watershed where tile contribute 7% to base stream flow, drain flow responses were slow, about 94 h, with dry soil at the time of a storm, and peaked 43 h after stream flows. As these authors[7] speculated, we observed more rapid outlet flow responses when soil was wet, such as flow within 1 h when soil was near saturation, and where drainage systems had surface inlets, such as flow within 30 min, likely reflecting overland flow. Low flow from Outlet E was probably due to crop water uptake as topsoil was about 0.1 m^3/m^3 drier in the alfalfa than soybean field from mid-June through September 2004. Tile flow was similarly less from sod than corn in another Ontario study.[16]

Outlets that were monitored from late April or early May until November (A, D, E, F and G) had flow volumes equivalent to 7, 4, 1, 3 and 5% of rainfall, respectively. Year-round measures of water partitioning through tile in humid temperate climates range from 15 to 30% of precipitation,[26–28] greater than our within growing season measures, because much of the annual precipitation occurs over winter. For example, long-term 1971–2000 average precipitation from November–April for Petrolia is 437 mm, or 47% of total annual (Environment Canada weather station). We estimated that an additional 6 kg/ha of sediment was transported over winter through

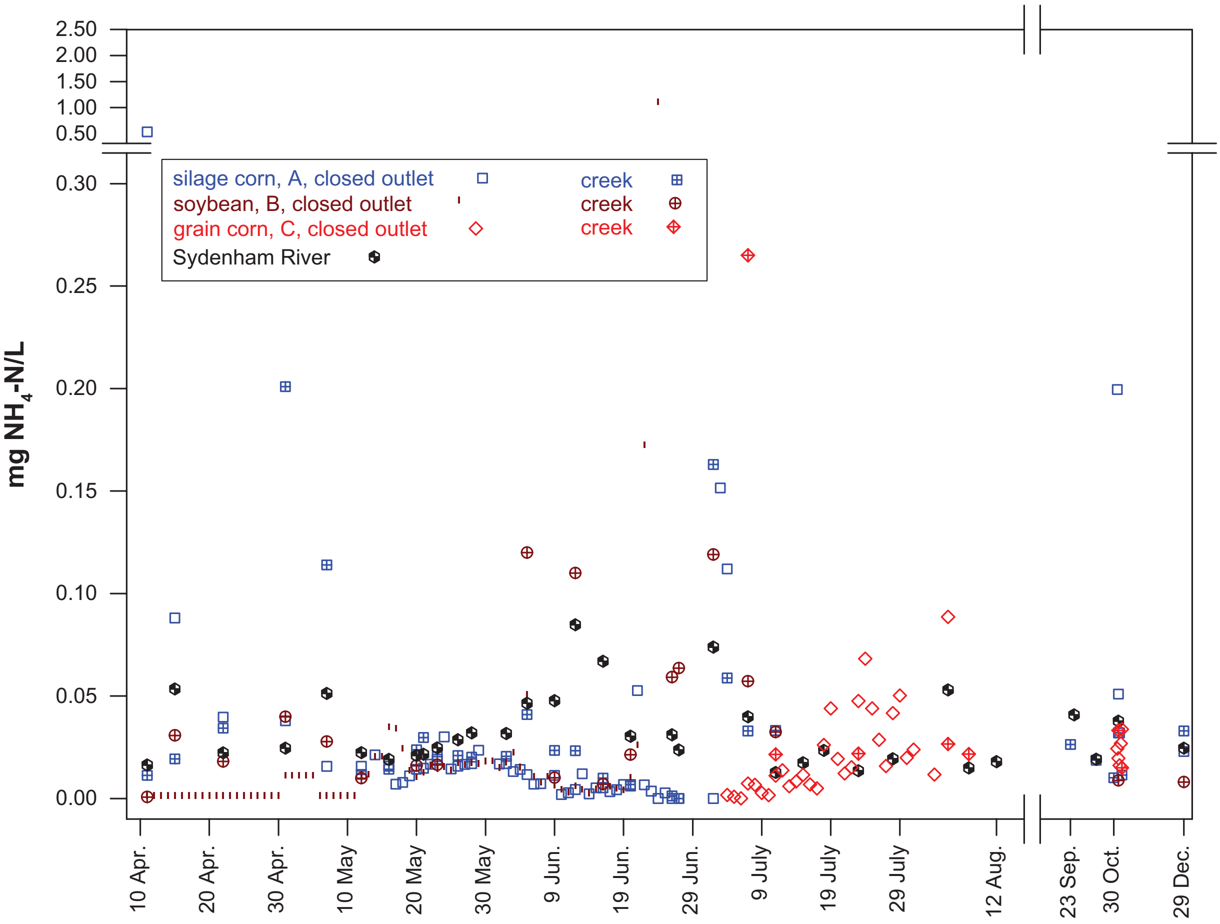

Figure 11. Daily average NH_4-N concentration in outflows from *closed* drainage system Outlets A (silage corn), B (soybeans), and C (grain corn), adjacent surface waters and the Sydenham River in 2003.

Outlet A, comprising 40% of total annual load. In a Minnesota study, a similar proportion, about 40%, of total annual sediment load moved during snowmelts, with the remainder transferred with rainfall. Total annual movement through systems with hickenbottom inlets in that study averaged 46 kg solids/ha/yr over a 3-yr period, but with wide temporal variation ranging from 1 to 125 kg/ha/yr.[29] Snowmelt also predominated annual subsurface and surface sediment movement in a Quebec study.[30]

To protect aquatic life, 46 mg suspended sediment/L is a proposed threshold for turbidity,[31] and Canadian water quality guidelines advise a maximum suspended sediment concentration increase from background of 25 mg/L at times of high flow, when background levels are between 25 and 250 mg/L.[32] Fluctuations of this magnitude occurred in our study during storm events in creek (Figs. 7, 8) and tile outflow (Figs. 3, 5, 6) water, particularly after dry periods. Elevated tile water sediment concentrations at the beginning of flow events followed by a rapid decrease to relatively constant levels was also observed from a silty clay/silt loam field in Indiana.[33] Movement of sediment into tiles is attributed to preferential flow through undisturbed macropores[15] and swelling and shrinking of clays.[34] The field drained by Outlet C had greatest flow-weighted sediment concentration for *closed* systems (Fig. 2) and greatest topsoil clay content (Table 1). Well-developed cracks were observed in this field in August, and subsequently (fall 2003) high outflow sediment concentrations (Fig. 7), particularly on 2 Nov. after a dry period of 89 d with no flow (Fig. 3). The trend of greater sediment concentrations in water from *open* than *closed* drainage systems (Figs. 2, 8) was previously observed in grab samples collected from 26 outlets in the Sydenham watershed which averaged over multiple dates 200 mg/L from *open* vs. 50 mg/L from *closed* systems.[35] There was no apparent effect of tillage practice on sediment movement through tile in our study.

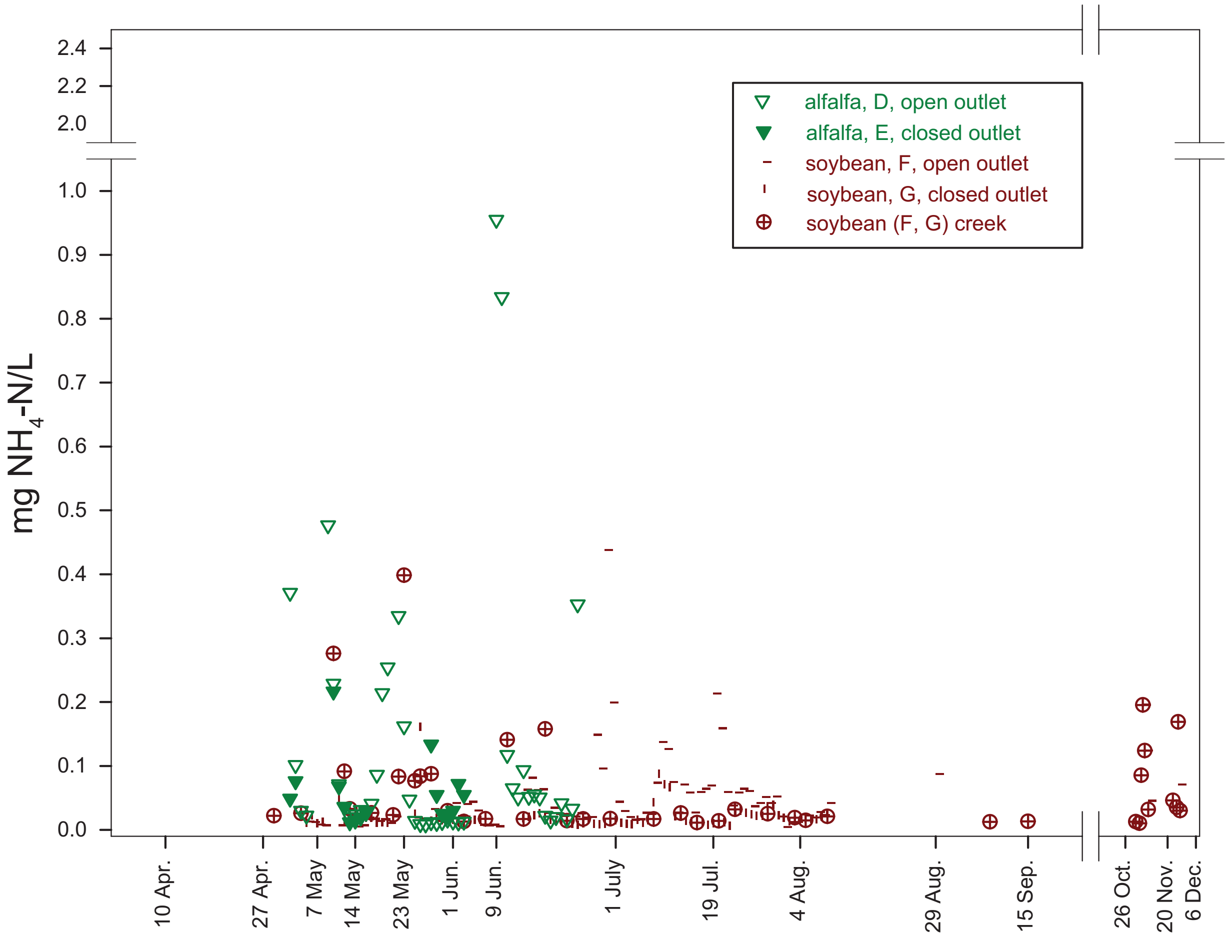

Figure 12. Daily average NH_4-N concentration in outflows from *closed* and *open* drainage system Outlets D and E (alfalfa) and F and G (soybeans), and in the adjacent creek at the soybean field in 2004.

The P limit to prevent eutrophication of surface water of 0.035 mg P/L[36] was exceeded in STP effluent from all plants except Oil Springs, which uses batch removal of P in the treatment process (Fig. 15), and in drainage water from five of seven monitored outlets (Fig. 2). Seasonal flow-weighted DRP concentrations from the *closed* drainage systems (Fig. 2) were: comparable to annual flow-weighted tile water DRP concentrations in an Illinois watershed which ranged 0.08 to 0.20 mg/L[4]; and less than flow-weighted Tot-P concentrations in water draining through surface inlets in Minnesota when flow was rainfall-induced (0.7–6.5 mg/L), but not snowmelt-induced (0.2–2.9 mg/L).[29] Four of the five outlets from *closed* systems had flow-weighted concentrations that were less than DRP concentrations in surface runoff collected from agricultural watersheds in Ontario of 0.098 to 0.69 mg/L[37] and 0.1 mg/L,[16] or Quebec of 0.06 to 0.42 mg/L.[30]

Similar to sediment, P transport through tile in the first flush of drainage water following dry weather has been attributed to the washing of fine material from the sides of cracks, fissures and earthworm burrows.[38] In a silty clay loam pasture about 17% of Tot-P loss through tile was associated with particles.[39] Greater flow-weighted DRP concentrations in water from Outlet C than from the other *closed* systems (Fig. 2) may have been related to soil P. Available P in topsoil of fields drained by Outlets C and F/G was greater than in the other fields (Table 1). Soil test P values were less than the threshold of 60 mg/kg available soil P, above which it is proposed that P moves to tile and concentrations will exceed 0.3 mg Tot-P/L.[11]

Seasonal P loads through the *closed* drainage systems (Fig. 2) were less than annual loads reported from: cracking clay soil in the UK of 0.1 to 0.6 kg Tot-P/ha/yr;[40] intensively managed pastures with fertilizer P and manure additions in Scotland of up to 5 kg

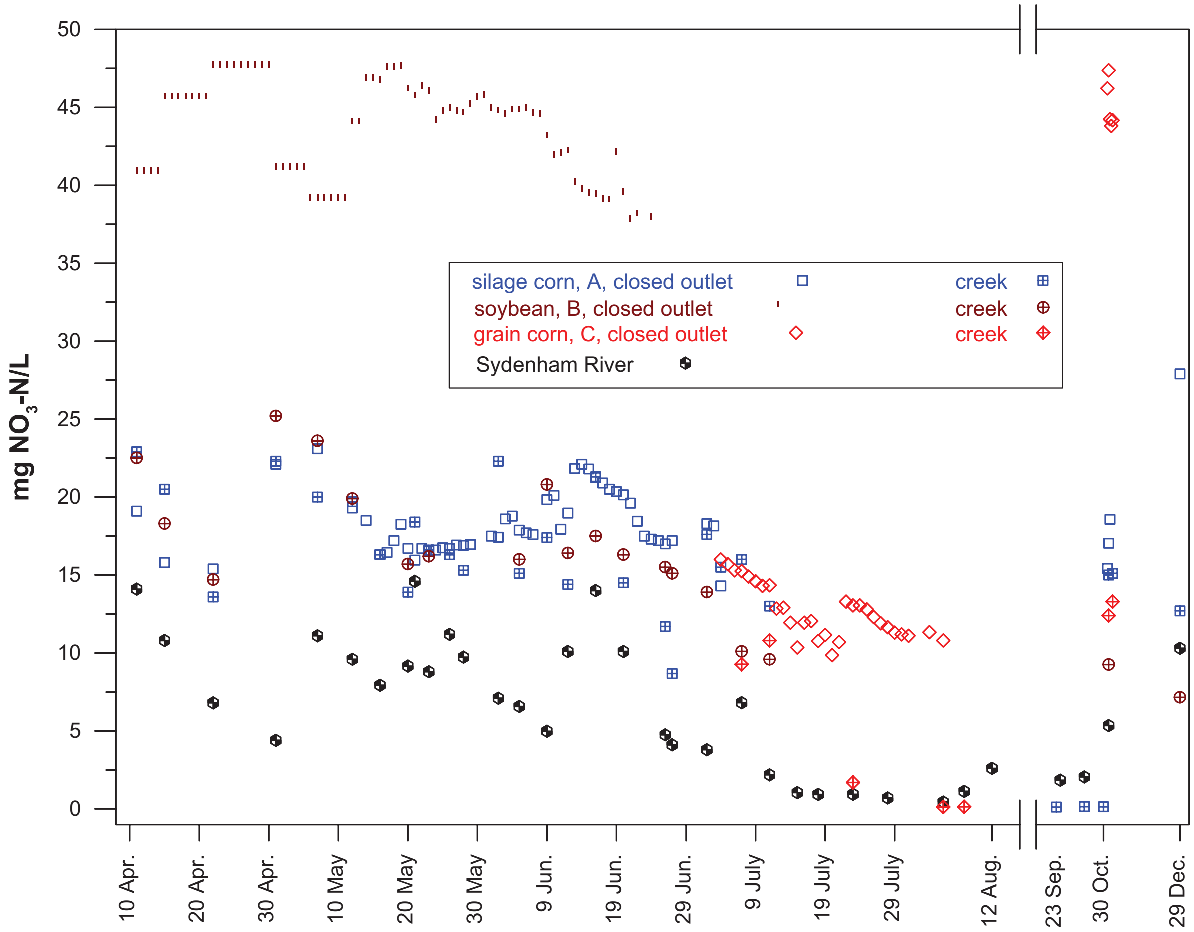

Figure 13. Daily average NO_3-N concentration in outflows from *closed* drainage system Outlets A (silage corn), B (soybeans), and C (grain corn), adjacent surface waters and the Sydenham River in 2003.

Tot-P/ha/yr;[39] and clay soil in Ontario of 1 kg/ha/yr with tile 0.6 m deep and 0.4 kg/ha/yr with tile 1 m deep.[16] Loading of DRP through the *open* systems (Fig. 2) was comparable to annual loads in Minnesota lacustrine landscapes of 0.12 kg particulate P/ha/yr from both *closed* drains and systems with hickenbottom inlets.[29] The similar P transport with or without inlets was attributed to ponding at the hickenbottom, which allowed entrained particles to settle.[29] Type of inlet structure was confounded with crop in our study with catch basins in the alfalfa field and hickenbottoms in the soybean field, and so the effect of inlet design on sediment and nutrient loading remains an area for future study.

Aquatic surface water quality standards for NH_4-N in Canada of 0.019 mg unionized NH_3/L or 3.4 mg NH_4-N/L at pH 7 and 25 °C[3] or in the US of 2.5 mg NH_4-N/L[41] were not exceeded in STP effluent or tile outflows. The low NH_4-N concentrations observed are typical of tile water, except when preferential flow occurs during manure application.[8] We observed greater NH_4-N concentrations in tile water draining legume (soybean, alfalfa) than corn crops.

Guidelines for aquatic life of 2.9 mg NO_3-N/L[42] were exceeded in STP effluent except from lagoons at Watford and Oil Springs, all tile outflows except Outlet E (Figs. 2, 15), and frequently in creek and river water (Figs. 13, 14). Elevated drainage water NO_3-N concentrations from the 2003 soybean field (Outlet B) was not an inherent site characteristic, since water collected from Outlet B the previous spring when the field was planted with corn contained only 7 mg NO_3-N/L on 14 May and 6 mg NO_3-N/L on both 24 May and 3 June 2002.[35] NO_3-N concentrations in water draining from soybean fields can be a function of the N fertilizer rate applied to the previous crop. In Pennsylvania, leachate NO_3-N concentrations in soil water 1.2 m below soybean ranged from 19 to 36 mg/L following corn that received 200 kg fertilizer N/ha, but only 3 to 11 mg/L when the preceding

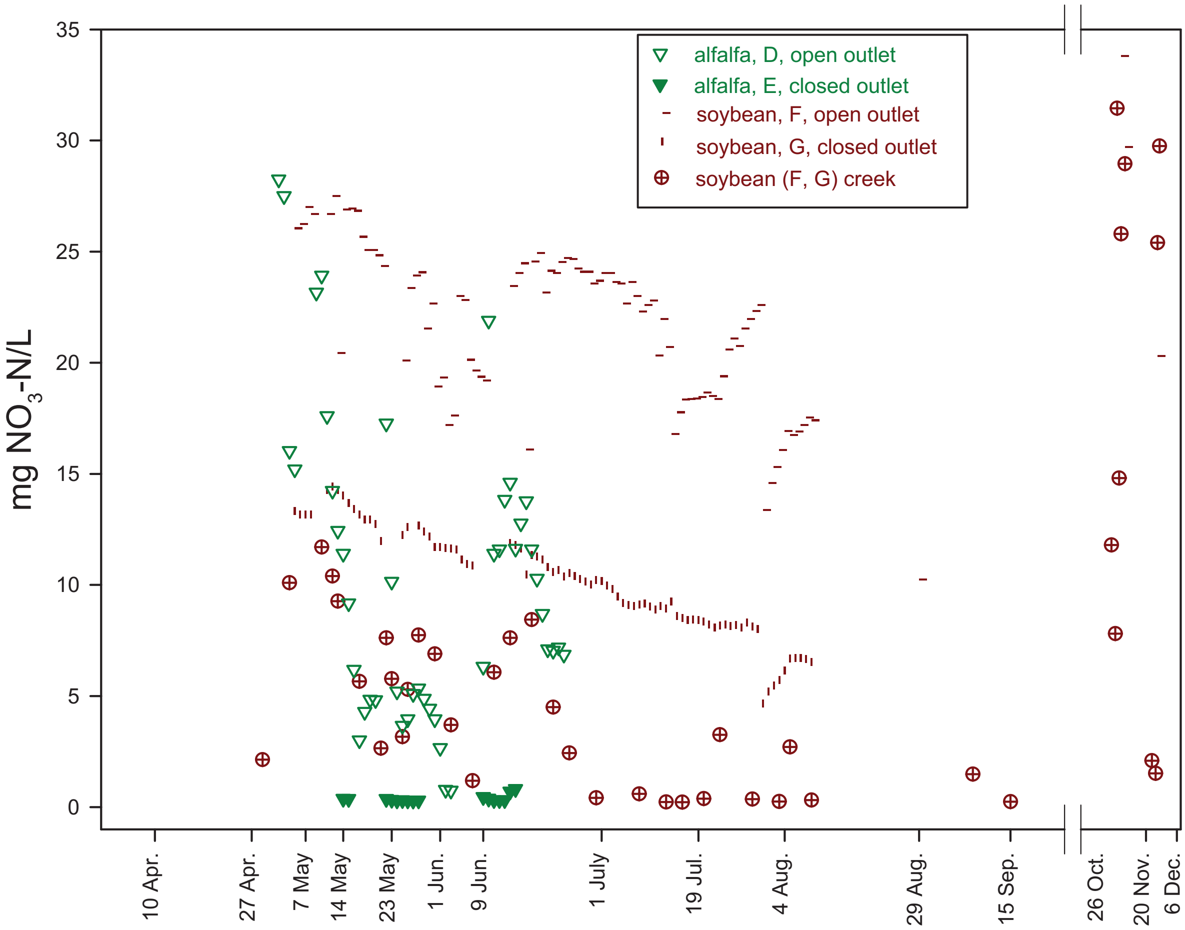

Figure 14. Daily average NO_3-N concentration in outflows from *closed* and *open* drainage system Outlets D and E (alfalfa) and F and G (soybeans), and in the adjacent creek at the soybean field in 2004.

corn was fertilized with 0 to 100 kg N/ha.[9] Increased surface water NO_3-N concentrations in fall as noted in the Sydenham River and creeks receiving Outlets C and F/G (Figs. 13, 14), may be related to less removal by denitrification as stream water temperature cools[43] and/or greater inputs from tiles when N uptake by annual crops ceases, as illustrated by the increased concentrations from Outlets C and F.

As a result of low flows and concentrations, the *closed* drainage system cropped with alfalfa (E) contributed the least N load of all outlets, and less sediment and P load than all outlets except C which was only monitored from summer to fall (Fig. 2). Low NO_3-N concentration (<5 mg/L) is typical of tile water draining alfalfa,[12] and less N load through tile draining alfalfa than corn was likewise observed in Ohio (<1 kg N/ha vs. 41 kg/ha).[44] A trend of less sediment and nutrient movement, particularly N, from tiles draining pasture, forage and winter cereals than from systems draining corn or soybeans was noted in grab samples collected previously from the Sydenham watershed, and was in part due to less flow.[35] Due to extensive root development and quasi-continuous water and nutrient uptake, forage can also reduce overland sediment movement dramatically, by up to 760-fold, relative to row crops such as corn.[45] Overland nutrient movement from catchments with pastures and cereals in Norway of 14 to 35 kg total N/ha/yr and 0.2 to 2.3 kg Tot-P/ha/yr was less than from catchments in vegetable producing areas of 55 to 112 kg total N/ha/yr and 1.0 to 18.9 kg Tot-P/ha/yr.[46]

In the absence of artificial drainage, overland flow and potential sediment and nutrient movement in runoff waters increases. Decreased sediment concentration over time in Ohio rivers despite increased storm intensity has been attributed in part to tile drain installation,[47] but few studies however, have quantified the net effects of drainage on sediment and nutrient loading. In a grassland, Tot-P transport was 30% less with than without tile and the authors speculated that the

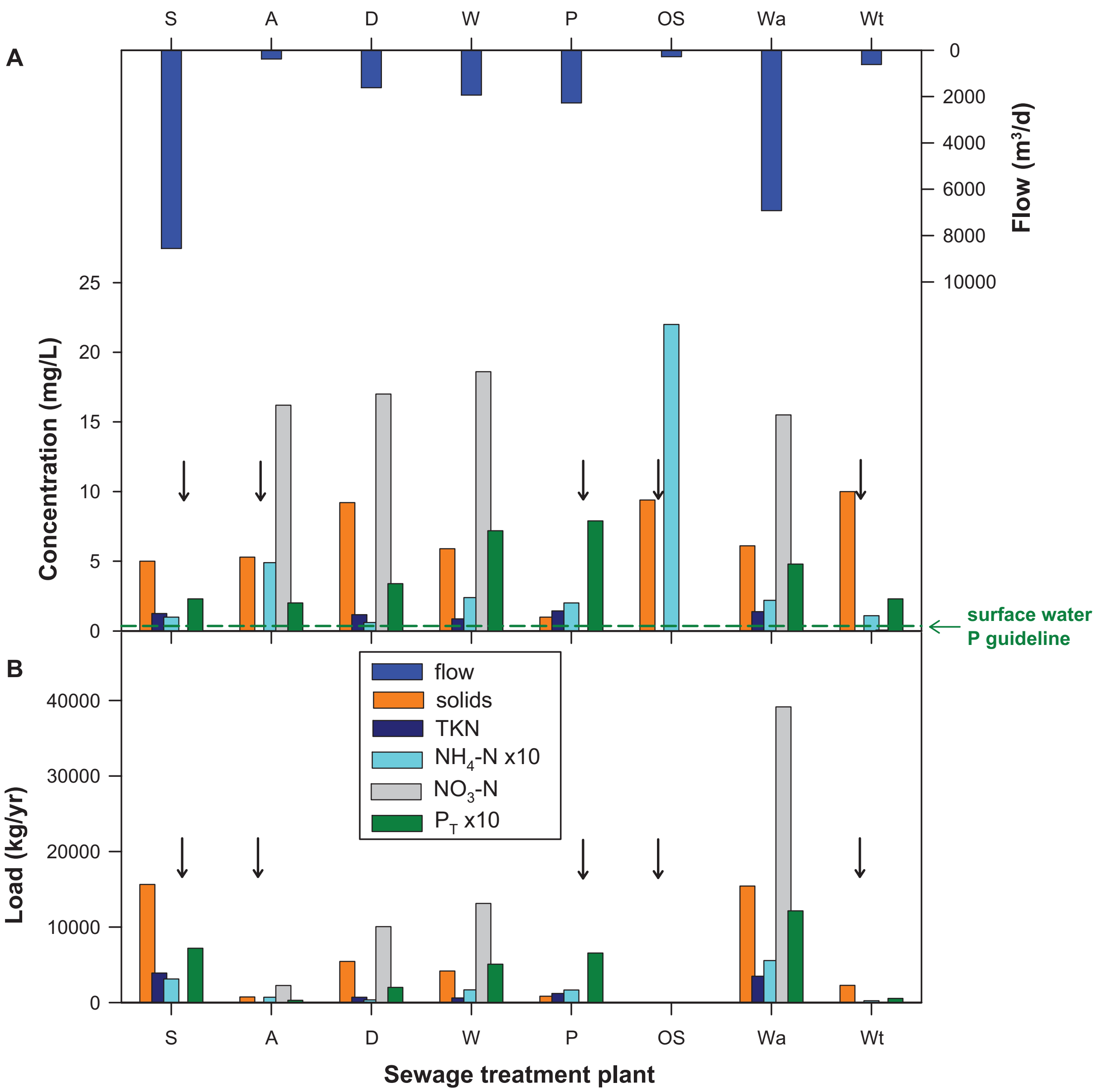

Figure 15. Effluent flows and annual average concentrations **A**) and annual loading **B**) to the Sydenham watershed of suspended solids, N, and Tot-P from eight sewage treatment plants.
Arrows indicate constituents that were not monitored at the specified treatment plant.
Abbreviations: S, Strathroy; A, Alvinston; D, Dresden; W, Wyoming; P, Petrolia; OS, Oil Springs; Wa, Wallaceburg; Wt, Watford; TKN, total Kjeldahl N.

reduction in P transport due to tiling would be greater from cultivated than from grass systems.[38] Subsurface drainage combined with a vegetative filter strip in Illinois reduced the quantity of runoff water as well as concentrations of P in surface and subsurface outflows, but sometimes increased transport of NO_3-N in subsurface flow.[19]

Soil erosion in Ontario can be up to 20, 000 kg/ha[48] but processes such as deposition or sediment delivery determine how much enters surface waters. Overland sediment delivery to surface water ranged: 60 to 1300 kg/ha from the tile-shed areas in our study (estimates); 31 to 3000 kg/ha/yr from eight catchments in Norway;[46] and 400 to 900 kg/ha from an Essex County, Ontario watershed.[16,49] The proportion of sediment that moved through tile during the growing season was 1 to 5% of the estimated annual overland delivery, and 9% when the additional estimate of sediment movement over winter through Outlet A was included. Proportion of total (tile + overland) sediment load that moved through tile following rainfall simulation on clay loam in Minnesota of 2% under conventional and 15% under ridge till was similar in magnitude.[15]

Associations of P with point sources, and N and solids with non-point sources have been observed in other watersheds. Fine sediment loading to rivers in the UK was primarily from diffuse sources rather than from sewage or industrial effluents.[1] In an Oregon watershed, higher N concentrations correlated primarily with agricultural land use, and elevated P concentrations with urban land use, although P trends were attributed mainly to factors such as storm drains and impermeable surfaces[50] rather than STP. In streams of the Upper Midwestern US, NO_3-N concentrations correlated positively to stream flow, upstream areas planted with corn, and upstream N-fertilizer application rates; and weakly with population density.[51] In an Illinois agricultural watershed, from a town with a population of 2500, comparable to that of Wyoming and Petrolia, STP effluent P contribution of 450 kg DRP/yr was also comparable (Fig. 15), and did not add greatly to the annual P load in the river, but impacted water quality during low discharge periods in summer and fall.[4]

Conclusions

Sediment movement through tile increased with field clay content and the presence of surface inlets. P concentrations in tile drainage water increased with field soil P content and presence of surface inlets. Quantitatively, tiles were a less important source of solids than overland movement, and P loads from tiles were small relative to that from STP. STP are a major source of P and NH_4-N load to the Sydenham watershed, whereas drainage tile contribute greater quantities of NO_3-N and solids than STP. Our results suggest that targeting: subsurface drainage pathways for NO_3-N (e.g. optimize N application rate, winter cover crop);[52,53] overland pathways for sediment (e.g. minimum tillage, filter strips); and STP for P (e.g. batch removal); would be effective strategies for reduction of loading to surface waters in tile drained agricultural watersheds. Forage demonstrated low risk for subsurface sediment and nutrient movement in this study. Coupled with effective control of overland movement documented in other studies, this cropping system is therefore well-suited to areas with high risk of both surface and subsurface movement. Simultaneous quantification of sediment and nutrient transport overland and through drainage systems during winter is needed to better understand partitioning to surface waters in humid temperate regions.

Disclosures

This manuscript has been read and approved by all authors. This paper is unique and not under consideration by any other publication and has not been published elsewhere. The authors report no conflicts of interest.

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Abbreviations

DRP, dissolved reactive phosphorus; STP, sewage treatment plant; θv, soil volumetric water content; Tot-P, total phosphorus.

References

1. Mainstone CP, Dils RM, Withers PJA. Controlling sediment and phosphorus transfer to receiving waters—a strategic management perspective for England and Wales. *Journal of Hydrology*. 2008;350(3–4):131–43.
2. Staton SK, Dextrase A, Metcalfe-Smith JL, et al. Status and trends of Ontario's Sydenham River ecosystem in relation to aquatic species at risk. *Environmental Monitoring and Assessment*. 2003;88:283–310.
3. Canadian Council of Ministers of the Environment. Canadian water quality guidelines for the protection of aquatic life. Ammonia. 2000, revised 2009. p. 8. In:Canadian environmental quality guidelines, Winnipeg. Available at: http://ceqg-rcqe.ccme.ca. Accessed July 2009.
4. Gentry LE, David MB, Royer TV, Mitchell CA, Starks KM. Phosphorus transport pathways to streams in tile-drained agricultural watersheds. *Journal of Environmental Quality*. 2007;36:408–15.
5. Stone M, Krishnapan BG. The effect of irrigation on tile sediment transport in headwater stream. *Water Resources*. 2002;36:3439–48.
6. Madramootoo CA, Johnston WR, Ayars JE, Evans RO, Rausey NR. Agricultural drainage management, quality and disposal issues in North America. *Irrigation and Drainage*. 2007;56:S35–45.
7. Goswami D, Kalita PK, Cooke RA, Hirschi, MC. Estimation and analysis of baseflow in drainage channels in two tile-drained watersheds in Illinois. *Transactions of the American Society of Agricultural and Biological Engineers*. 2008;54(4):1201–13.

8. Ball Coelho BR, Roy RC, Topp E, Lapen DR. Tile water quality following liquid swine manure application into standing corn. *Journal of Environmental Quality*. 2007;36:580–87.
9. Zhu Y, Fox RH. Corn-soybean rotation effects on nitrate leaching. *Agronomy Journal* 2003;95:1028–33.
10. Withers PJA, Ulen B, Stamm C, Bechmann M. Incidental phosphorus losses—are they significant and can they be predicted? *Journal of Plant Nutrition and Soil Science*. 2003;166:459–68.
11. Heckrath G, Brookes P, Poulton P, Goulding K. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. *Journal of Environmental Quality*. 1995;24:904–10.
12. Randall GW, Huggins DR, Russelle MP, Fuchs DJ, Nelson WW, Anderson JL. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. *Journal of Environmental Quality*. 1997;26:1240–47.
13. Gaynor JD, Findlay WI. Soil and phosphorus loss from conservation and conventional tillage in corn production. *Journal of Environmental Quality*. 1995;24:734–41.
14. Oygarden L, Kvaerner JK, Jenssen PD. Soil erosion via preferential flow to drainage systems in clay soils. *Geoderma*. 1997;76:65–86.
15. Zhao SL, Gupta SC, Huggins DR, Moncrief JF. Tillage and nutrient source effects on surface and subsurface water quality at corn planting. *Journal of Environmental Quality*. 2001;2:998–1008.
16. Culley JLB, Bolton EF, Bernyk V. Suspended solids and phosphorus loads from a clay soil:I. Plot studies. *Journal of Environmental Quality*. 1983;12:493–8.
17. Enright P, Jamieson A, Madramootoo C. Field sites for evaluating surface and subsurface phosphorus movement in the Pike river watershed. CSAE/SCGR Meeting, Guelph, ON. 2001. p. 10. Paper 01–111.
18. Barton DR. The use of percent model affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of southern Ontario, Canada. *Freshwater Biology*. 1996;36:397–410.
19. Bhattarai R, Kalita PK, Patel MK. Nutrient transport through a vegetative filter strip with subsurface drainage. *Journal of Environmental Management*. 2009;90:1868–76.
20. Sheldrick BH, Wang C. Particle Size Distribution. In:Carter MR, ed. Soil sampling and methods of analysis. Boca Raton, FL:Lewis Publishers; 1993:499–511.
21. Liao N. QuikChem Method 10-115-01-3-A. Determination of phosphorus by FIA colorimetry (in-line persulfate digestion method). Lachat Instruments, Milwaukee, WI. 2001. p. 18.
22. Diamond D. QuikChem Method 10-107-04-3-P. Determination of total nitrogen in waters by in-line digestion followed by flow injection analysis. Lachat Instruments, Milwaukee, WI. 2007. p. 19.
23. Liao N. QuikChem Method 10-107-06-1-J. Determination of ammonia (phenolate) by flow injection analysis colorimetry (low flow method). Lachat Instruments, Milwaukee, WI. 1999. p. 24.
24. Diamond D. QuikChem Method 10-115-01-1-A. Determination of orthophosphate in waters by flow injection analysis colorimetry. Lachat Instruments, Milwaukee, WI. 2000. p. 15.
25. Snell EA. A manual for regional targeting of agricultural soil erosion and sediment loading to streams. Working paper No. 36, Land Directorate, Environmental Conservation Service, Environment Canada, Ottawa, Ontario. 1984. p. 39.
26. Drury CF, Tan CS, Gaynor JD, Oloya TO, Welacky TW. Influence of controlled drainage-subirrigation on surface and tile drainage nitrate loss. *Journal of Environmental Quality*. 1996;25:317–24.
27. Goulding KWT, Poulton PR, Webster CP, Howe MT. Nitrate leaching from the Broadbalk wheat experiment, Rothamsted, UK, as influenced by fertilizer and manure inputs and the weather. *Soil Use and Management*. 2000;16(4):244–50.
28. Kladivko EJ, Frankenberger JR, Jaynes DB, Meek DW, Jenkinson BJ, Fausey NR. Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system. *Journal of Environmental Quality*. 2004;33:1803–13.
29. Ginting D, Moncrief JF, Gupta SC. Runoff, solids, and contaminant losses into surface tile inlets draining lacustrine depressions. *Journal of Environmental Quality*. 2000;29:551–60.
30. Jamieson A, Madramootoo CA, Enright P. Phosphorus losses in surface and subsurface runoff from a snowmelt event on an agricultural field in Quebec. *Canadian Biosystems Engineering*. 2003;45:1.1–1.7.
31. Vondracek B, Zimmerman JKH, Westra JV. Setting an effective TMDL: sediment loading and effects of suspended sediment on fish. *Journal of the American Water Resources Association*. 2003;39:1005–15.
32. Canadian Council of Ministers of the Environment. Canadian water quality guidelines for the protection of aquatic life. Total Particulate Matter. 2002. p. 13. In:Canadian environmental quality guidelines, Winnipeg. Available at:http://ceqg-rcqe.ccme.ca. Accessed July 2009.
33. Bottcher AB, Monke EJ, Huggis EF. Nutrient and sediment loadings from a subsurface drainage system. *Transactions of the American Society of Agricultural Engineers*. 1981;24:1221–6.
34. Skaggs RW, Nassehzadeh-Tabrizi A, Foster GR. Subsurface drainage effect on erosion. *Journal of Soil and Water Conservation*. 1982;37:167–72.
35. Staton S, Ball Coelho B. Sydenham River Recovery Strategy, Part 1:Riparian Inventory and Sediment/Nutrient Loading. Interdepartmental Recovery Fund Annual Report. 2002. p. 85.
36. Canadian Council of Ministers of the Environment. Canadian water quality guidelines for the protection of aquatic life. Phosphorus:Canadian guidance framework for the management of freshwater systems. 2004. p. 6. In: Canadian environmental quality guidelines, Winnipeg. Available at:http://ceqg-rcqe.ccme.ca. Accessed July 2009.
37. Dickinson WT, Kachanoski RG, Lobb DA, Smithers JA, Vyn TJ, Miller MH. (The Centre for Soil and Water Conservation, University of Guelph). Soil Erosion and Phosphorus in Runoff from Agricultural Cropland in Southwestern Ontario. A review of relevant literature prepared for Ecological Services for Planning Ltd. 1987. p. 73.
38. Simard RR, Beauchemin S, Haygarth PM. Potential for preferential pathways of phosphorus transport. *Journal of Environmental Quality*. 2000;29:97–105.
39. Hooda PS, Moynagh M, Svoboda IF, Edwards AC, Anderson HA, Sym G. Phosphorus loss in drainflow from intensively managed grassland soils. *Journal of Environmental Quality*. 1999;28:1235–42.
40. Addiscott TM, Brockie D, Catt JA, et al. Phosphate losses through field drains in a heavy cultivated soil. *Journal of Environmental Quality*. 2000;29:522–32.
41. Eghball B, Gilley JE. Phosphorus and nitrogen in runoff following beef cattle manure or compost application. *Journal of Environmental Quality*. 1999;28:1201–10.
42. Canadian Council of Ministers of the Environment. Canadian water quality guidelines for the protection of aquatic life. Nitrate ion. 2003. p. 7. In: Canadian environmental quality guidelines, Winnipeg. Available at: http://ceqg-rcqe.ccme.ca. Accessed July 2009.
43. van der Perk M. Soil and water contamination—from molecular to catchment scale. Leiden, The Netherlands:Taylor and Francis; 2006.
44. Logan TJ, Schwab GO. Nutrient and sediment characteristics of tile effluent in Ohio. *Journal of Soil and Water Conservation*. 1976;35:24–7.
45. Clark EA, Poincelot RP. The contribution of managed grasslands to sustainable agriculture in the Great Lakes Basin. Binghampton, NY:Food Products Press; 1996.
46. Bechmann ME, Deelstra J, Stalnacke P, Eggestad HO, Oygarden L, Pengerud A. Monitoring catchment scale agricultural pollution in Norway:policy instruments, implementation of mitigation methods and trends in nutrient and sediment losses. *Environmental Science and Policy*. 2008;11:102–14.
47. Richards RP, Baker DB, Crumrine JP, Kramer JW, Ewing DE, Merryfield BJ. Thirty-year trends in suspended sediment in seven Lake Erie tributaries. *Journal of Environmental Quality*. 2008;37:1894–908.
48. Wall GJ, Pringle EA, Sheard RW. Intercropping red clover with silage corn for soil erosion control. *Canadian Journal of Soil Science*. 1991;71:137–45.

49. Gaynor J, Bissonnette D. The effect of conservation tillage practices on the losses of phosphorus and herbicides in surface and subsurface drainage waters. SWEEP Report #60. 1992. p. 6. Available at:http://gis.lrs.uoguelph.ca/AgriEnvArchives/sweep/sublist.html#TED. Accessed July 2009.
50. Sonoda K, Yeakley JA, Walker CE. Near-stream landuse effects on stream-water nutrient distribution in an urbanizing watershed. *Journal of the American Water Resources Association*. 2001;37:1517–32.
51. Mueller DK, Ruddy BC, Battaglin WA. Logistic model of nitrate in streams of the Upper-Midwestern United States. *Journal of Environmental Quality*. 1997;26:1223–30.
52. Ball Coelho BR, Roy RC, Bruin AJ. Nitrogen recovery and partitioning with different rates and methods of sidedressed manure. *Soil Science Society of America Journal*. 2006;70:464–73.
53. Ball Coelho BR, Roy RC, Bruin AJ. Long-term effects of late-summer overseeding of winter rye on corn grain yield and nitrogen balance. *Canadian Journal of Plant Science*. 2005;85:543–54.