Nitrate in Tile Drainage of the Semiarid Palouse Basin

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Topographically heterogeneous agricultural landscapes can complicate and accelerate agrochemical contamination of streams due to rapid transport of water and chemicals to poorly drained lower-landscape positions and shallow ground water. In the semiarid Palouse region, large parts of these landscapes have been tile drained to enhance crop yield. From 2000–2004 we monitored the discharge of a tile drain (TD) and a nearby profile of soil water for nitrate concentration ([NO$_3$]), electrical conductivity level (EC), and water content to develop a conceptual framework describing soil nitrate occurrence and loss via subsurface pathways. Tile-drain baseflow [NO$_3$] was consistently 4 mg N L$^{-1}$ and baseflow EC was 200 to 300 $\mu$S cm$^{-1}$. Each year sudden synoptic increases in TD discharge and [NO$_3$] occurred in early winter following ~150 mm of fall precipitation, which saturated the soil and mobilized high-[NO$_3$] soil water throughout the profile. The greatest TD [NO$_3$] (20–30 mg N L$^{-1}$) occurred approximately contemporaneous with greatest TD discharges. The EC decrease each year (to ~100 $\mu$S cm$^{-1}$) during high discharge, a dilution effect, lagged ~1 mo behind the first appearance of high [NO$_3$] and was consistent with advective transport of low-EC water from the shallow profile under saturated conditions. Water-budget considerations and temporal [NO$_3$] patterns suggest that these processes deliver water to the TD from both lower- and upper-slope positions, the latter via lateral flow during the high-flow season. Management practices that reduce the fall reservoir of soil nitrate might be effective in reducing N loading to streams and shallow ground water in this region.

Both worldwide and in North America, the rate of synthetic fertilizer N application to cultivated lands in the 1990s was two to three times that of biological N$_2$ fixation on those lands, and similar to total terrestrial biological N$_2$ fixation (Boryer et al., 2004; Vitousek et al., 1997). The environmental and economic consequences of this massive agricultural perturbation to the N cycle are of great interest and concern. In the United States, thousands of river systems are considered by the USEPA to be impaired, and many of these cases are attributable to nonpoint-source agricultural contamination. The 2004 national water quality report (USEPA, 2004a) cited agricultural fertilizer as the most widespread source of pollution of rivers, streams, lakes, reservoirs, and ponds, with N considered a leading pollutant. Nitrogen lost from agricultural fields in runoff or drainage can result in degraded surface water and ground water quality and potentially threaten aquatic ecosystems, as well as contribute to reduced yields and financial losses for farmers (Hatfield et al., 1999).

Artificial soil drainage through the use of subsurface tile drains is a common practice. In North America, these systems and their environmental effects have been extensively studied in poorly drained soils and relatively flat landscapes of the Midwest (e.g., Randall and Iragavarapu, 1995; Dinnes et al., 2004) and in the irrigated soils of the arid Southwest. Tile drains extract water from saturated soil at the scale of the field, thereby providing a measure of spatial integration of the processes causing chemical loading of effluents (Kladivko et al., 1991; Randall et al., 1997; David et al., 1997) and thus may be useful for both monitoring and understanding nutrient and agro-ecological delivery to surface water bodies adjoining fields (Kladivko et al., 1991; Buhler et al., 1993; Randall et al., 1997; Stamm et al., 1998; Cambardella et al., 1999; Zehe and Fluhler, 2001).

Several studies have shown sustained nitrate concentrations ([NO$_3$]) in tile-drain waters exceeding 10 mg N L$^{-1}$, even at low fertilizer application rates (Logan et al., 1980; Randall and Iragavarapu, 1995; Randall et al., 1997; Cambardella et al., 1999; Jaynes et al., 2001). In several regions with extensive subsurface drained cropland, tile drains have been identified as the primary conduits for nitrate loading to surface waters (David et al., 1997; Jaynes et al., 1999). Particularly large losses of fall-applied fertilizer have been shown to occur...
in tile drained areas of the Midwest where anhydrous ammonia is used in corn (Zea mays L.)–soybean [Glycine max (L.) Merr.] rotations (Randall et al., 2003; Randall and Vetsch, 2005).

Although the occurrence of N in agricultural settings has been extensively documented in surface waters and subsurface drainage, the transport processes and flow pathways involved at field scales are complex and seem to vary both temporally and from place to place. For example, Jaynes et al. (1999) observed seasonal changes in [NO$_3$] in surface waters that were attributed to changes in discharge rates. When flow increased, stream [NO$_3$] were observed to decrease, exhibiting a dilution effect. This dilution effect was also observed in tile-drain effluent and attributed to snowmelt (Patni et al., 1996). However, David et al. (1997) observed [NO$_3$] increasing with increasing tile drain flow and Schilling and Lutz (2004) found that [NO$_3$] were linearly related to stream flow at daily, monthly, seasonal, and even annual time scales. Tomer et al. (2003) observed that nitrate was not diluted by larger flows, both in tile-drained subbasins (500 and 900 ha) and at the outlet of a 5000-ha agricultural watershed. These differences may be due to management, geologic, pedologic, and hydrologic differences among the study areas, as well as observational technique and scaling differences among the studies themselves.

Fertilization timing and rates can influence [NO$_3$] in both stream and tile drain waters. However, the effects seem to vary for complex, interrelated reasons. Cambardella et al. (1999) reported that temporal patterns of [NO$_3$] in tile drainage waters were not directly related to the rate or timing of fertilizer application. Conversely, Jaynes et al. (2001) observed peak tile drainage [NO$_3$] subsequent to fertilizer application and the maximum drainage discharge period. It should also be recognized that nitrate observed in stream and tile drainage waters may not be from the most recent input of nutrients, whether collected under baseflow or runoff conditions (Blanchard and Lerch, 2000). The observations are generally complicated by the biological lability of N in soils, with high rates of soil N mineralization, denitrification, and large NO$_3$–N buildups over dormant periods (David et al., 1997; Cambardella et al., 1999; Kladiokvo et al., 1999; Fenelon and Moore, 1998).

In the semiarid Palouse region of Washington State, soft white winter wheat (Triticum aestivum L.) is the principal agricultural crop alternating with spring grains (especially barley, Hordeum vulgare L.) and dry legumes (especially lentil, Lens culinaris L.). Winter wheat is planted in the fall after the soil is prepared and fertilized; typical fertilizer application rates range from 150 to 200 kg N ha$^{-1}$. Precipitation falls dominantly from late fall to spring. Thus, heavy fertilization before fall and winter rains and subsequent subsurface movement of water may cause significant loading of soil water, shallow ground water, and streams with dissolved agrochemicals. In addition, tile drains are common in lower topographic positions of fields, where restricted infiltration and lateral water flow from upslope can pond water on the soil surface (Mallawatantri et al., 1996). These tile-drained areas may provide important pathways for water and chemicals to move from fields to streams (e.g., Geyer et al., 1992). Recent research focused on the subsurface hydrology of high-relief landscapes on the Palouse has demonstrated the complexity of flow processes in the loess deposits beneath hillslopes (O’Brien et al., 1996; O’Geen et al., 2003) and the importance of lateral flow in delivery of subsurface water to topographic concavities and lower slope positions (Mallawatantri et al., 1996; Brooks et al., 2004, 2006). The hydrology and associated solute transport processes beneath lower-landscape fields, which abut and deliver sediment and water to ditches and streams, has received little attention. In all, little is known about the mechanisms and pathways of chemical loading, specifically nitrate, from fields to surface waters in this area.

The goal of this research was to study the relationship between hydrologic processes, particularly the temporal variation of flowpath activity, and nitrate losses from fields to streams in the semi-arid climate of the Palouse region of southeastern Washington. Of particular interest was how the soil N reservoir, the timing of fertilizer application, seasonal precipitation and runoff patterns, and topography interact to contribute to nitrate loss via subsurface pathways. The objectives were to (i) perform multi-year monitoring of a tile drain and a nearby soil profile for soil water content, tile drain discharge, and soil water and tile drain water nitrate and electrical conductivity levels; and (ii) develop a conceptual framework for soil nitrate occurrence and movement to the tile drain, based on the dynamics observed in these long-term records.

Materials and Methods

Study Area

Research was conducted in the semiarid Missouri Flat Creek (MFC) watershed, located in the Palouse River Basin, Washington State (Fig. 1). Mean annual high and low air temperatures are 27°C in the summer and −7°C in the winter (Geyer et al., 1992). The water year begins on 1 October, at the end of the growing and cropping seasons. Precipitation is strongly seasonal with fall rains giving way to heaviest precipitation (rain and snow) between January and April. July through September is the driest period. The 38-yr average annual precipitation rate near the field site through

![Fig. 1. Topographic map (after Goodwin, 2006) of the Cook Agronomy Farm (CAF), showing the tile-drained study catchment, approximately 12 ha in area. Elevations are masl, contour interval is 2.5 m. NE-SW–running crop–rotation sections are delineated by light double lines. Drain line position is approximate. Bulls-eye symbol indicates location of drain outlet with nearby time domain reflectometry and soil water sampling instrumentation, installed via a backhoe trench (see text).](image-url)
the 1980s was 560 mm (Geyer et al., 1992). Precipitation during the four studied water years was 385, 480, 411, and 442 mm for 2000–2001, 2001–2002, 2002–2003, and 2003–2004, respectively. This was a relatively dry interval of years, and “open” winters with little snow cover characterized our study period. Although it was not addressed in this work, visual observation indicated relatively little overland runoff in the study area during this time, and there was no evidence of saturation-excess runoff (Dunne and Black, 1970) from the study catchment in particular. The greatest precipitation storage in snow, 10 to 15 mm moisture equivalent, occurred through January 2001. Recorded snow cover did not exceed 4 mm moisture equivalent in 2002, 2003, and 2004, except for 2 d in January 2004. In each water year, snowmelt had occurred and snow moisture storage was reduced to zero at the onset of wintertime high flows in MFC streams.

Basalts of the Columbia River Group dominate the regional bedrock geology. Quaternary deflation deposits of clayey silt loess have been “draped” over the bedrock, and preexisting topography has been accentuated by vegetation and erosion patterns developed during depositionally quiescent periods (McDonald and Busacca, 1992) to produce characteristic undulations with up to 50 m of local relief in the study area. At the study site, depth to bedrock is unknown but relatively unimportant to the surface hydrology because of the vertically impeding hydraulic characteristics of the soils and paleosols, as discussed below. These soils are dominantly silt loam Mollisols mapped as a part of the Palouse–Thatauna Association soil series (USDA, 1978). The lower-landscape portions of many agricultural fields are artificially tile-drained to increase productive acreage. The drains conduct water to ditches tributary to the MFC, and thus serve as a conduit for field infiltration to reach surface water. In these lower-slope positions, local infiltration is augmented by lateral inflows of soil water from adjacent slopes above restrictive argillic horizons (Brooks et al., 2006, 2004; O’Geen et al., 2003; Mallawatantri et al., 1996; O’Brien et al., 1996). These studies underline the difficulty of predicting the extent of upper-slope contributions of water to lower-slope fields due to the temporal and spatial variability of hydraulic conductivity.

Our research was conducted at the Washington State University Cook Agronomy Farm (CAF) (46°46′44″N, 117°05′19″W). The CAF is a research facility managed by the USDA using no-till conservation farming practices. Our work was focused on a tile-drained area of the CAF. The tile drain (TD) outlet is located at the southwest corner of the farm, intercepting and discharging water collected from beneath an approximately 12-ha area (Fig. 1). The trench was carefully backfilled after installation of instrumentation, and backfill was subsequently added to compensate for settlement so it could be cultivated as part of the field. Two types of suction samplers were used: one employed conventional high-flow, 0.1-MPa ceramic cups and plastic collection chambers, and the other used stainless-steel 0.1-MPa cups and chambers with the same physical configuration. All samplers were placed in augered holes angled approximately 20° downward from horizontal. Ceramic-cup samplers were replicated three times in each of three (23–30, 40–46, and 84–90 cm) depth intervals. These intervals will be referred to as “shallow,” “intermediate,” and “deep,” respectively. At each depth the replicates were installed approximately 2 m horizontally apart. Ceramic-cup samplers at the shallow depth yielded little water over the course of the study. Stainless-steel samplers were replicated three times, located approximately 50 cm laterally from the ceramic replicates, at the shallow and intermediate depths. Comparison of synoptic results for the two sampler types showed 30% lower [NO₃] in the ceramic-cup samplers than in the stainless samplers, and no significant pattern of difference for EC levels. Differences between sampler types were small relative to spatial and temporal variability, so results from both types were averaged.

Soil water samples were collected periodically, typically monthly, during winter and spring. Collection methods were consistent with USEPA guidelines (Barcelona et al., 1985; USEPA, 2004b) with details as follows: vacuum of 0.05 MPa was placed on suction samplers 1 d in advance of sample collection. Extracted soil water was then drawn from the sampler chamber directly into acid-washed, 250-mL plastic Nalgene bottles in line with a vacuum flask. All parts of the apparatus that contacted the sample were rinsed three times with nano-
pure water between each sample to avoid contamination (Simmons, 2003).

Soil water content reflectometers (model CS615, Campbell Scientific, Logan, UT) were installed at 25-, 40-, 55-, 70-, 85-, and 98-cm depths by insertion of probe rods directly into the trench wall. Factory calibration was adjusted to the measured water content range of soil samples at the CAF. Due to data acquisition problems, reliable water content measurements were not recorded until fall 2002. Drainage flow rate (discharge) from the TD was measured biweekly and more frequently during storm events using a manually recording tipping bucket or bucket and stopwatch. Tile drain samples were collected at approximately 2-wk intervals and more frequently during selected storm events. Samples were grabbed in 250-mL plastic acid-washed Nalgene bottles.

Sample aliquots for nitrate analysis were filtered in the laboratory with rinsed 0.45-μm Whatman cellulose nitrate membranes and placed into prewashed scintillation vials with foil-lined plastic caps, then frozen until analysis at the WSU USDA-ARS laboratory by continuous flow analyzer (model RFA 300, Alpkem/OI Analytical, College Station, TX). Electrical conductivity was measured in the laboratory using a temperature compensated system (model 115, Orion/ThermoFisher Scientific, Waltham, MA) and manual two-point calibration. In uncontaminated waters, electrical conductivity is a good proxy for ionic load due to water-mineral dissolution reaction progress (Freeze and Cherry, 1979). In N-contaminated waters, the nitrate anion and its counterion can affect and even dominate conductivity measurements (Smith and Doran, 1996) depending on their relative contribution to the total ionic load. To correct for this contamination effect, and to estimate the component of the measured conductivity value attributable to water-mineral reaction progress in our water samples, we experimentally determined and subtracted the effect of nitrate and its counterion on each sample’s conductivity measurement (Simmons, 2003).

The conductivities of prepared Ca(NO$_3$)$_2$ and NaNO$_3$ solutions were measured and regressed against their known [NO$_3$]. Slopes of the regressions, each with an $r^2$ of 0.99, were indistinguishable. The NaNO$_3$ regression was then used with the measured [NO$_3$] for each sample to generate a correction for each sample, and this correction was subtracted from the sample’s measured conductivity value to obtain nitrate-corrected electrical conductivity (EC). As a check on the success of the correction, cationic charge was measured on a subset of the samples (Wannamaker, 2005) and equivalents of nitrate counterion were subtracted from this quantity to yield corrected cationic charge. Linear regression of EC vs. corrected cationic charge yielded $r^2$ of 0.79, showing that EC satisfactorily reflects non-nitrate ionic load.

Combined analytical and sampling errors are estimated at less than 10% for [NO$_3$] and 10 to 15% for EC. Soil–water sampler results are reported below as averages for a given depth and sampling date, and spatial variability among replicates was generally larger than analytical and sampling errors. In general, temporal variability overwhelmed both spatial variability and method error.

### Results

#### Hydrology

The relationship between precipitation and runoff discharge for one 660-ha catchment containing the study site (Wannamaker, 2005) is shown in Fig. 2a. Comparison with discharge from the tile drain (Fig. 2b) and other flow data in the watershed (Wannamaker, 2005) shows that seasonal hydrographic responses exhibit similar patterns across a range of catchment scales up to 5000 ha. Suzuki (2005) reported total runoff discharges for the 2003–2004 water year as 110 to 120 mm or 25 to 28% of precipitation for the 660-ha catchment, and 150 mm or 34% of precipitation for the TD area. The similarity indicates that the 12-ha estimate of the tile-drained area is broadly consistent with local runoff fluxes, and suggests that this catchment may be hydrologically typical of this part of the MFC drainage. Given the small estimated rates of ground water recharge reported in previous studies (see above) and assuming that most TD-catchment runoff is captured and discharged by the tile line, evapotranspiration is equal to most of the difference between precipitation and TD discharge, and would amount to approximately two-thirds of precipitation in 2003–2004. These estimates represent the TD catchment as a whole.

The typical 3- to 4-mo lag between onset of fall precipitation and onset of flow is due to the absorption by the soil profile of approximately 150 mm of precipitation early in the water year (Fig. 2a). This quantity is in fact equal, within error, to the amount of water required to increase the soil water content of the upper 1 m of the profile, as measured at the tile drain outflow, from its state at the beginning of the water year to its saturated state (Fig. 2b; Wannamaker, 2005). After this large soil water storage capacity is recharged and saturation is attained, the first substantial increase in discharge is observed, typically between January and February of a given water year. After this initial discharge threshold, additional events are triggered by each successive precipitation event (Fig. 2a and 2b).

#### Nitrate

The [NO$_3$]$_-$ level in tile drainage at baseflow, in summer and fall, is approximately 4 mg N L$^{-1}$ (Fig. 2c). The [NO$_3$]$_-$ levels in drainage increase as soon as any substantial increase in the discharge has occurred (Fig. 2b and 2c); that is, at the temporal resolution of this study, the [NO$_3$]$_-$ response to the onset of winter flow is essentially instantaneous. The highest [NO$_3$]$_-$ occur with the highest flows (February–March). However, the concentrations also remain elevated, even after discharge has substantially decreased in the spring.

Soil water [NO$_3$]$_-$ profiles monitored near the TD outflow are variable and change from decreasing with depth to increasing with depth (Fig. 3). The magnitudes of [NO$_3$]$_-$ vary annually and seasonally, and appear to depend to some extent on fertilizer timing and application rate in the westernmost cropping strip (Fig. 1). In years when N fertilizer was applied in the fall, as in October 2000 and October 2001, [NO$_3$]$_-$ in soil waters were greater than when fertilizer was applied only in the spring as in April 2004 (Fig. 3). (The effect of the October 2002 fertilizer applica-
tion on soil–water \([\text{NO}_3^-]\) is not known because soil water was not sampled in the 2002–2003 water year.) These large variations among years contrast with the TD discharge water, in which \([\text{NO}_3^-]\) dynamics are relatively consistent from year to year and relatively unresponsive to fertilization timing in the westernmost cropping strip.

**Electrical Conductivity**

The temporal dynamics of EC in the TD discharge water are generally but not exactly opposite those of \([\text{NO}_3^-]\) (Fig. 2c). As has been reported in numerous watershed studies (e.g., Hooper et al., 1990; Peters et al., 1995), discharge water at high flow exhibits lowered EC relative to baseflow conditions. Careful inspection shows, however, that typically EC values do not decline to their lowest levels until after the first large TD hydrographic events and \([\text{NO}_3^-]\) “spikes” of the flow season (Fig. 2b and 2c).

Depth profiles of EC in soil water, collected near the TD outflow, show increasing values with depth (Fig. 4). The shapes of the several profiles are generally similar, and in contrast with the TD EC data, there is no clear temporal pattern. The depth trend is consistent with increase in water–mineral reaction progress with time, as soil water moves downward in the profile. The low-EC sides of the data envelope at deep and shallow depths exhibit values similar to TD discharge water at baseflow and high-flow conditions, respectively (Fig. 2c). This suggests that TD waters could be mixtures of time-varying proportions of deep and shallow soil waters flowing laterally and downward from a depth-variable profile (e.g., Bishop et al., 2004) as discussed further below.

**Discussion**

**Onset of Wintertime High-Nitrate Discharge**

The immediate appearance of high \([\text{NO}_3^-]\) with the onset of high wintertime tile-drain flows (Fig. 2b and 2c) is an arresting feature of our data. The time of accumulation of 150 mm of fall precipitation (Fig. 2a) is in fact an excellent predictor not only of the first high TD discharge rates, but also of the time of the first \([\text{NO}_3^-]\) “spike” of the season (Fig. 2c). Taken together these data suggest that at the onset of saturated wintertime soil water flow and the runoff season, nitrate is already present at substantial levels at the depth of the tile-drain perforations, and is thus immediately available for mobilization into drainage. The notion of a soil nitrate reservoir present following harvest, or residual soil nitrate, is documented in the literature on the Palouse (e.g., Fuentes et al., 2003; Schillinger et al., 2003) and is supported by fall 2002 prefertilization measurements of extracitable nitrate beneath the tile-drained field (Fig. 5). Conversion of the dry soil mass–basis \(\text{NO}_3^-\)–N values in Fig. 5 to areal basis, assuming average soil bulk density of approximately 1.4 g cm\(^{-3}\), yields an estimate of 80 to 100 kg NO\(_3^-\)–N ha\(^{-1}\) residual soil nitrate in the upper 1 m of the profile. If this nitrate were dissolved into otherwise nitrate-free soil water saturating the pore space, assuming bulk density as above and average saturated porosity of approximately 0.35 cm\(^3\) cm\(^{-3}\), the estimated \([\text{NO}_3^-]\) at 1-m tile-drain depth (Fig. 5) would be very similar to the peak \([\text{NO}_3^-]\) exhibited in winter–spring tile drainage (Fig. 2c). Tomer and Burkart (2003) reported that in long-term fertilized agricultural settings it may be common for nitrate to accumulate becoming a reservoir subject to movement with local hydrology.

An alternative explanation for the rapid appearance of nitrate could, in principle, be rapid vertical transport in percolating soil water from the fertilization zone (upper 15 cm). Neglecting components of lateral soil–water flow above the tile drain and upslope, a reasonable estimate of advective travel time from the upper 15 cm to the drain perforations at 1-m depth, assuming saturated soil hydraulic conductivity measured at a field 2 km north (Johnson, 1991) and porous-media flow under a conservatively large unit gradient, is approximately 30 d. This is clearly much too long for the practically instantaneous appearance of high tile-drain nitrate levels once saturated.

Fig. 2. Watershed and study site data for the 4-yr study period. (a) Event and cumulative daily precipitation by water year (October–September). Vertical arrows indicate time of cumulative of 150 mm of fall precipitation (pptn). Co-incident onset of high-flow season is evident in the discharge hydrograph for a 660-ha catchment (grayscale) containing the study site. (b) Tile drain (TD) discharge rate and 48-h mean soil volumetric water content at selected depths; (c) Tile drain nitrate concentration and nitrate-corrected electrical conductivity (EC), with solid vertical bars showing times of 150-mm precipitation accumulation from panel a. Subsequent vertical bars denote passage of 30 days time.
flow begins; it strengthens the inference that at that moment, high-nitrate soil water must already be present at depth.

Advective transport is nonetheless an important process. A second set of vertical bars is plotted on Fig. 2c to show the passage of 30 d following coincident profile wetup, onset of saturated flow, and high-nitrate release by the tile drain. The data show that this is typically about the time at which the lowest EC values are exhibited; that is, EC "troughs" lag behind the first nitrate "peaks" by a reasonable estimate of vertical advective travel time through the profile. The simplest interpretation of this lag is that on wetup, high-[NO3] water flows from the deep profile into the tile drain; about a month later, high-[NO3], low-EC water arrives at the tile drain by advective transport in saturated flow from shallow soil. The horizontal components of such flows might be large, given the dramatic increases of lateral hydraulic conductivity with increases in saturation, which have been documented in soils of the CAF and adjoining farms (Mallawatantiri et al., 1996; Brooks et al., 2006). In the 2003 wet season the EC trough lagged the first [NO3] peak by a considerably greater time, probably because runoff and tile drainage in particular were later and smaller than usual that year (Fig. 2b), such that vertical flow and transport in the profile would also have been slower than usual. Smaller deviations from the 30-d lag in other years, due to such processes, are not surprising. The simple advective transport model is also consistent with the dynamics of soil–water [NO3] responses following fertilization, particularly in the shallow and middle samplers, during water years 2001 and 2002 (Fig. 3). Thus, wintertime flow evidently does advectively redistribute dissolved nitrate downward in the profile, and this would sustain high-[NO3] tile discharge through the high-flow season (Fig. 2). Certainly unsaturated advective transport must also move solutes downward during the wetup period; our data suggest that this may be secondary in importance to subsequent transport under saturated conditions.

The data also suggest the possibility that macropore flow occurs and contributes to redistribution (e.g., Ersahin et al., 2002; Zehe and Flühler, 2001; Lennartz et al., 1999). Shallow and intermediate soil water [NO3] exhibit "spikiness" (Fig. 3) and the same is true of EC values in tile drainage, particularly early in the period of saturated flow each year (Fig. 2c). However, rapid downward movement of shallow soil water via macropore flow cannot be the main reason for the consistently coincident onset of high [NO3] and high flow, because that process would coincidentally depress EC to sustained low levels, and we do not in fact observe this depression until about a month later.

**Seasonal Dynamics of Nitrate in Tile Drainage**

We propose a conceptual framework, explaining TD hydrologic and [NO3] dynamics, which divides the seasonal cycle into three principal phases as shown in Fig. 6. This division is similar to that used by Fuentes et al. (2003) in their study of seasonal soil water and N changes in eastern Washington. In Phase 1, from crop senescence (approximately 1 August) through fall, baseflow drainage exhibits constant low nitrate and high EC levels attributable to a lateral/deeper-flow source of shallow ground water from beneath...
adjacent slopes. This is consistent with the greater water contents observed at the base of the profile during late summer. Residual soil nitrate is held under tension at shallower depths and this pool is augmented by ongoing mineralization and nitrification of soil N, and any fall application of fertilizer. The production of nitrate in soil, asynchronous with its uptake by crops, has been previously implicated as a key process supplying nitrate to tile drains (Cambardella et al., 1999). During this period fall precipitation gradually recharges soil moisture from above.

At the beginning of Phase 2 (Fig. 6), saturation develops and pore imbibition mobilizes nitrate at the tile drain perforations and throughout the profile. Saturated flow redistributes soil- and fall fertilizer-derived nitrate downward and drives tile drainage, which exhibits sustained high [NO₃] and peaking flows. The TD water is first derived from the bottom of the profile (high EC), then is mixed with contributions from shallower depths (lower EC), and finally is mainly derived from the bottom of the profile again (recovering high EC).

In Phase 3, baseflow conditions are reestablished as precipitation declines and crop growth extracts water and nitrate from the profile. Rates of mineralization and nitrification of soil N increase as the temperature increases, and that fraction not taken up by the crop contributes to replenishment of the residual soil nitrate reservoir for Phase 1 of the following water year.

The foregoing framework is subject to testing in ongoing work. It must be considered as averaging the time variation of relative water and solute contributions from lower and upper slope positions in the TD catchment. Infiltration on the lower slope would logically drive drainage at the beginning of saturated soil–water flow and high TD discharge rates. Soil water from nearby upper slopes may arrive rapidly, however, due to large (and difficult to quantify) lateral saturated soil hydraulic conductivities and transport rates above restrictive argillic horizons (Mallawatantri et al., 1996; Brooks et al., 2006). Substantial contributions from upper slopes over the course of the flow season are implied by the available estimate of the TD runoff flux (i.e., approximately one-third of precipitation on 12 ha). On the one hand, given small loss to deep percolation, evapotranspiration of the other two-thirds of precipitation—about 300 mm—is reasonable (e.g., Fuentes et al., 2003); on the other hand, the 3 to 5 ha of gently sloping ground at the western edge of the catchment could physically account for tile drainage, but only if all precipitation falling on this area were intercepted and discharged by the TD with no evapotranspiration. Year-round baseflow discharge from the tile drain, and late-summer wetting of the bottom of the soil profile implied by monitoring near the tile-drain outflow, also strongly imply hydraulic connection of the lower-slope area to adjacent upper slopes. Moreover, the year-to-year stability of the [NO₃] patterns in tile drainage, compared with soil–water patterns monitored on the lower slope (Fig. 3), is consistent with a catchment area that averages the year-to-year fluctuations of soil conditions across a range of slope positions. More research into these relationships is needed, both to understand the processes and the natural variation of the system’s response. It is possible for example that an interval of years with colder and wetter winters might increase the importance of snow storage, soil freezing, and overland flow (Brooks et al., 2006), which had little evident effect on our observations.

**Conclusions and Implications**

Seasonal precipitation variations are a primary control of N loss from soils to streams in our region. Late-fall and wintertime recharge of soil profiles, dried down by crops during the preceding growing season, is required to mobilize nitrate and generate sustained winter–spring streamflow. Our data and interpretations suggest that fall fertilization and spring–summer soil N mineralization and nitrification generate a reservoir of nitrate more or less throughout the profile, such that the whole system is “primed” to release large dissolved nitrate mass discharges as soon as the winter flow season commences.

According to our reasoning, controlling hydrologic losses of N would involve reducing the residual and fertilizer-application components of the Phase 1 nitrate pool, which are vulnerable to transport during Phase 2 (Fig. 6). Altering the fall–spring distribution of fertilization should affect N loss, but as long as total applications exceed crop uptake on average, a Phase 1 pool will exist and be vulnerable to transport loss during the cold, high-flow portion of the Palouse water year. Suzuki (2005) measured 100 mm of cumulative runoff discharge from the 660-ha and TD catchments (Fig. 2a and 2b, respectively) from November through April of 2003–2004, which was the cold season of high flows and high nitrate concentrations in runoff in that water year. Assuming a rough mean [NO₃] of 15 mg N L⁻¹ during that period (TD and 660-ha data shown in Fig. 7), the cumulative N loss in 2003–2004 was about 15 kg N ha⁻¹ in both catchments. Studies of N mass discharges in Palouse streams are ongoing; good estimates of longer-term averages of hydrologic loss, over a greater range of catchment scales, are not yet available. Clearly, for example, the N losses in the two catchments would have been substantially smaller.
in the low-runoff year 2000–2001 (Fig. 2 and 7). Nonetheless, the foregoing rough calculation shows that hydrologic loss of only a small fraction of winter wheat crop uptake and typical annual fertilizer application—estimated at 130 to 170 (Koenig, 2005) and 150 to 200 kg N ha$^{-1}$, respectively—is enough to cause substantial nitrate contamination in surface waters. It is well known that cover crops can take up N and limit its hydrologic losses (Dinnes et al., 2004). Long-term amelioration of the N contamination problem in our region may require the use of perennial crops with deep root systems that cultivate relatively “tight” subsurface N cycles. This is under investigation in ongoing research.

The synoptic TD and 660-ha [NO$_3$]$_{-}$ records (Fig. 7) are highly similar for the duration of our study. This similarity is particularly striking, given an areal scaling difference of almost two orders of magnitude and that the larger catchment contains fields with and without tile drains. Thus, the ideas presented here may apply more broadly to subsurface flow pathways from cultivated fields to streams on the Palouse, whether that flow occurs as tile drainage or as distributed seepage. We are presently investigating the applicability of our ideas to larger watershed scales and to the broader Palouse physiographic region.

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