Source-Pathway Separation of Multiple Contaminants during a Rainfall-Runoff Event in an Artificially Drained Agricultural Watershed

M. D. Tomer* USDA–ARS
C. G. Wilson University of Iowa
T. B. Moorman and K. J. Cole USDA–ARS
D. Heer and T. M. Isenhart Iowa State University

A watershed’s water quality is influenced by contaminant-transport pathways unique to each landscape. Accurate information on contaminant-pathways could provide a basis for mitigation through well-targeted approaches. This study determined dynamics of nitrate-N, total P, *Escherichia coli*, and sediment during a runoff event in Tipton Creek, Iowa. The watershed, under crop and livestock production, has extensive tile drainage discharging through an alluvial valley. A September 2006 storm yielded 5.9 mm of discharge during the ensuing 7 d, which was monitored at the outlet (19,850 ha), two tile-drainage outfalls (total 1856 ha), and a runoff flume (11 ha) within the sloped valley. Hydrograph separations indicated 13% of tile discharge was from surface intakes. Tile and outlet nitrate-N loads were similar, verifying subsurface tiles dominate nitrate delivery. On a unit-area basis, tile total P and *E. coli* loads, respectively, were about half and 30% of the outlet’s; their rapid, synchronous timing showed surface intakes are an important pathway for both contaminants. Flume results indicated field runoff was a significant source of total P and *E. coli* loads, but not the dominant one. At the outlet, sediment, P, and *E. coli* were reasonably synchronous. Radionuclide activities of *Be* and *Pb* in suspended sediments showed sheet-and-rill erosion sourced only 22% of sediment contributions; therefore, channel sources dominated and were an important source of P and *E. coli*. The contaminants followed unique pathways, necessitating separate mitigation strategies. To comprehensively address water quality, erosion-control and nitrogen-management practices currently encouraged could be complemented by buffering surface intakes and stabilizing stream banks.

Water quality problems continue to vex agricultural producers in the midwestern United States. Studies addressing water quality tend to focus on single contaminants. Yet decisions regarding agricultural water quality management are often seen to involve tradeoffs among contaminants (Claassen et al., 2001). Integrated assessments are needed to better inform tradeoffs, and identify mitigation strategies that can address water quality comprehensively.

Nitrate-nitrogen, phosphorus, *Escherichia coli*, and sediment are among the most pervasive agricultural contaminants in the Midwest; an extensive research literature provides detailed analyses of these contaminants. In the Midwest, artificial subsurface (tile) drainage has modified the hydrology of many agricultural watersheds, and provides a major pathway for nutrients (Royer et al., 2006). Tiles contribute significant loads of NO$_3$–N to the Mississippi River, which have contributed to Gulf of Mexico hypoxia (Burkart and James, 1999). Phosphorus is also a concern for hypoxia, at least on a seasonal basis (EPA Science Advisory Board, 2008), and leads to eutrophication of freshwaters, particularly lakes and reservoirs (Klatt et al., 2003). Integrated assessments in glaciated areas of the Midwest have addressed N and P transport from tile-drained lands, and emphasized disproportionate loads occurring during high flow conditions (Royer et al., 2006), and the role of depressional (pothole) hydrology (Smith et al., 2008). Surface intakes are often installed in potholes to facilitate drainage of surface water, but provide an open conduit for surface water to enter the tile system.

Few watershed-scale studies have simultaneously addressed nutrients with other contaminants. The fecal-contamination indicator *E. coli* frequently impairs the use of water bodies in the United States, including at least one large Iowa watershed with significant tile drainage (Schilling et al., 2009). Ball Coelho et al. (2007) showed nutrients and *E. coli* were transported to tile drains within days following liquid manure injection in an...
Ontario study, particularly when applications exceeded crop N requirements. This indicates that preferential (macropore) flows are important in tile-drained soils, a phenomenon also implicated in delivering pesticides to tile discharge (Stone and Wilson, 2006).

Sediment loads have impacted society and agriculture in the United States since settlement (Montgomery, 2007), with ongoing impacts on aquatic habitat (Newcombe and Jensen, 1996) and navigation (Vörösmarty et al., 1997). Documentation of sediment sources in midwestern watersheds usually implicates soil erosion and stream banks as dominant sediment sources (Knox, 2006; Trimble, 1999). There are few reports of sediment loads discharged from tiles; among them two European studies indicate tile sediment loads can be significant (Russell et al., 2001; Uusitalo et al., 2000).

This study was conducted to simultaneously evaluate the timing and sources of these key contaminants discharged at three spatial scales within a tile-drained agricultural watershed in Iowa during a single rainfall-runoff event. The objective of this research was to determine if timing and loads of multiple contaminants at nested watershed scales could be interpreted to indicate dominant delivery pathways for each contaminant. Understanding differences in delivery mechanisms among the contaminants should in turn suggest how water quality management practices in the watershed could address each contaminant specifically. The research approach included chemical separations of water sources and radionuclide separations of sediment sources. Both were based on a simple mixing model approach, a technique that has been applied in evaluating transport of nutrients and pesticides from first-order agricultural watersheds (Fyer et al., 2001) and in tile-drainage discharge (Schilling and Helmers, 2008; Stone and Wilson, 2006). Sediment source separation based on 7Be;10Pb was described by Masitoff et al. (2005) and Wilson et al. (2008).

Materials and Methods

This research was conducted in Tipton Creek, Iowa, a tributary to the South Fork of the Iowa River. Research in this watershed has focused on assessing water quality and existing conservation practices (Tomer et al., 2008a, 2008b) and analyses to target new conservation practices (Tomer et al., 2003). The watershed lies on the eastern margin of the Des Moines Lobe ecoregion, a recently glaciated landscape region typified by limited stream development and poor natural drainage (Griffith et al., 1994). The Tipton Creek watershed is dominated by intensive crop and livestock production and has extensive artificial drainage. This research is focused on a single rainfall-runoff event in Tipton Creek that occurred during September 2006, but is also informed by a longer term record from this watershed. Therefore, the following describes monitoring data that are part of an ongoing effort to monitor discharge and nutrient concentrations, and data on bacteria populations and sediment sources that were collected for this individual event.

Monitoring of Discharge and Water Quality Constituents

A stream gauge near the outlet of Tipton Creek, about 0.5 km upstream of its confluence with the South Fork of the Iowa River, was established in 2000. Gauging of stream discharge at this site (TC325; see Fig. 1), was described by Tomer et al. (2008a). Briefly, water stage was continuously recorded and a stage-discharge rating curve used to calculate discharge; the rating curve was adjusted whenever major runoff events altered the streambed cross section. The rating curve data also provide a relationship between discharge ($Q$) and velocity ($V$) used to estimate travel times along the stream course in this study. A carousel-type sampler (ISCO Model 6712, Teledyne Isco, Lincoln, NE) was used to collect discrete samples during runoff-hydrograph events. The timing of this sampling was triggered by changes in water stage. Sampling intervals were programmed to be as short as a half hour, but increased depending on the rate of change of stage and most sampling intervals were 3 h or more. This event sampling was in addition to daily samples, comprised of four subsamples collected at 6-h intervals and composited in a single bottle each day (Tomer et al., 2008a); these daily samples were analyzed for $\text{NO}_3$–$\text{N}$ concentration as detailed below.

Three additional gauging stations were installed upstream of the outlet (TC325). One of these measured and sampled surface runoff from a single agricultural field 10.6 ha in area (TC101; see Fig. 1), and was located in the lower part of the watershed and within about 500 m of the stream. The station included a 0.9-m H-type flume, with water stage recorded at 5-min intervals during runoff events. A similar carousel sampler collected one sample of surface runoff for every 7-m$^2$ interval of discharge. This flume station was placed along a grass waterway below a field dominated (58%) by a highly erodible Clarion (Typic Hapludoll) soil map unit with 5 to 9% slopes (National Cooperative Soil Survey, 1985; Soil Survey Staff, 2003). The field was managed in corn ($\text{Zea mays}$ L.) and soybean ($\text{Glycine max}$ L. (Merr.)) production and received swine manure applications following soybean harvest and before corn planting. Conservation practices in this field include two grassed waterways, minimum tillage in the cropped field, and a small area of steep (9–18%) slopes (about 0.5 ha) under perennial grass cover and enrolled in USDA’s Conservation Reserve Program. The field was in soybean production in 2006 and had lost received manure during spring 2005, before planting of corn that year. This field typifies small agricultural fields near the stream in the lower part of the watershed, which often include multiple conservation practices (Tomer et al., 2008b) to mitigate the risks of erosion and loss of contaminants carried with runoff.

The other two gauging–sampling stations were at adjacent outfalls of subsurface-drainage subbasins that were 1700 and 156 ha in area (TC240 and TC242; see Fig. 1), and located approximately 31 km upstream from TC325 (stream-course distance). These drainage district mains were clay tile pipes installed around 1910; TC242 was 0.91-m diam. and TC240 was 0.38-m diam. To measure pipe discharge, stage and velocity were measured using an open channel meter (Marsh-McBirney Flo-Tote 3, Hach Company, Frederick, MD). Water quality samples were collected based on change in stage during events, with discrete samples collected at a minimal interval of 0.5 h but a maximum interval of 24 h under steady flow conditions. Discharge gauging and water quality sampling for TC240 and TC242 was initiated in July 2005. The drainage area for
these tile gauge stations is dominated by the Clarion–Nicollet–Webster soil association, respectively comprised of Typic Hapludolls, Aquic Hapludolls, Typic Endoaquolls (National Cooperative Soil Survey, 1985; Soil Survey Staff, 2003). The landscape includes glacial “potholes” occupied by very poorly drained Okoboji soils (Cumulic Vertic Endoaquolls), with Harps soils (Typic Calciaquolls) on their margins. The drainage basin above TC242 is dominated by a glacial-till plain dominated by poorly drained soils, and, based on county soil survey data (National Cooperative Soil Survey, 1985), is 25% occupied by these pothole soils (compared with 17% of the watershed and 21% of its tile-drained lands). Most potholes are cropped; surface intakes in potholes and roadside ditches route surface water directly to the subsurface drainage network. These two tile outfalls drain 9% of the watershed area, or about 12% of its tile-drained lands.

Water samples collected during hydrograph events at all four gauge sites were analyzed for NO$_3$–N and total P. The NO$_3$–N concentrations were determined using a Lachat (Lachat Instruments, Loveland, CO) autoanalyzer with Cd reduction (Wood et al., 1967) at a detection limit of 0.3 mg L$^{-1}$. Total P concentrations were determined by acid-persulfate digestion and flow injection analysis using USEPA Method 365 (O’Dell, 1993).

This paper is focused on a rainfall-runoff event following a storm that occurred on 10 to 11 Sept. 2006, and two additional sets of analyses were conducted for this event. First, water samples collected from all four stations were also analyzed for $E. coli$ populations using 4-methyl-umbelliferyl-glucuronide and a modified most-probable-number (mpn) format known as Colisure and Quanti-Tray panel methods (Idexx, 2007). Carousel-collected samples were retrieved once or twice daily, and incubations were initiated on return to the laboratory, on the same day. In the laboratory, a 100-mL subsample was added to the Colisure media and dispensed into a Quanti-Tray/2000 panel. Undiluted samples could be enumerated between 1 and 2419 cells 100 mL$^{-1}$; most samples were diluted with deionized water to enable population counts >2419 to be counted. However, the need for and amount of dilution had to be anticipated because second incubations could require the maximum recommended sample-hold time to be exceeded. Following incubation for 24 h at 35°C, $E. coli$–positive cells in the sealed Quanti-Tray panels were visualized by ultraviolet fluorescence of 4-methyl umbelliferone, which indicates $\beta$-glucuronidase activity.
mpn values (cells 100 mL−1) were calculated using the manufacturer’s method (Idexx, 2007). For this September 2006 event, the number of E. coli samples analyzed ranged from 10 to 14 among the four stations, and the number of nutrient analyses varied from 12 to 14 per station.

The final set of analyses was on an additional set of water samples collected only at the outlet (TC325), to determine sediment loads and sediment sources using activities of 7Be and 210Pb isotopes. Methods are described in detail by Wilson et al. (2008), where, in a multiswatershed comparison, results detailed here were briefly summarized. The two naturally occurring nuclides are delivered to soils through atmospheric deposition, but the difference in half-lives (53.3 d for 7Be, 22.3 yr for 210Pb), together with processes of erosion and deposition, result in surface-soil-derived sediments having larger 7Be activity than those derived from channel and bank sources. During summer 2006, surface soils (0- to 20-mm depth) were sampled from five fields near the stream in the lower part of the watershed, including the field above TC101, and stream banks were cored at five locations. These samples were used to obtain the source signatures, which provided end member coordinates for a mixing model analysis. Precipitation (dry and wet) was also collected during the intervening period at the five fields to determine the atmospheric radionuclide deposition between sampling and the monitored runoff event.

Fourteen suspended sediment samples were manually collected during the September 2006 event from the bridge at TC325 using a 20-L bucket. Sediment concentrations (g m−3) were measured after settling and drying the samples; the fine sediment was then separated by flocculation to determine nuclide activities using γ-ray spectroscopy, as described by Wilson et al. (2008). The analysis given here was supplemented by turbidity data from TC325 and therefore differs slightly from that presented by Wilson et al. (2008). Turbidity was measured using a Hydrolab Model DS4a multiparameter sonde (Hach Company, Loveland, CO) with a self-cleaning turbidity sensor. Comparing measured sediment concentrations to the turbidity readings (nephelometric turbidity units) showed a linear relationship with an R2 of 0.98. Therefore, the turbidity data were used to estimate the suspended sediment concentration (g m−3) throughout the hydrograph event. The radionuclide activities and end-member mixing model gave estimates of the fraction of sediment derived from surface soils, as opposed to channel (bank, bed, and any gully) sources. A semilog decay function was fit to a plot of time (T, hr after initiation of runoff) vs. percent surface-soil-derived sediment (S) with an R2 of 0.80:

\[ S = 0.70 T^{0.33} \]  \[ 1 \]

This function was used to estimate the proportion of sediment from surface and channel sources throughout the event.

**September 2006 Rainfall-Runoff Event**

Rainfall amounts were recorded using two tipping bucket rain gauges, which were located on a hilltop northeast of the field flume (TC101), and about 1 km north of the tile gauges. Rainfall began about 0300 h on 10 Sept. 2006 and continued for approximately 26 h (Fig. 2). The rain gauges indicated a total of 116 mm of rainfall was received near the field flume and 73 mm near the tile gauges. The greatest precipitation intensities occurred during the early-morning hours of 11 September. Radar rainfall data were obtained for these dates, which were calibrated using a network of rain gauges in Iowa to estimate daily rainfall totals as part of the Iowa Daily Erosion Project (Cruse et al., 2006). Although the radar-based rainfall estimates are about 25% below the rain gauge data, the radar data indicate that rainfall was relatively uniform over the upper watershed, except for increased amounts in about the lower (easternmost) 10% of the watershed (Fig. 2); but much of this area is under forest and pasture cover that would limit runoff compared with cropland. A separate analysis (D. James and M. Tomer, unpublished data, 2005) indicated the radar-based estimates of rainfall compare well with monthly rainfall amounts measured in this watershed, but not with daily amounts.

The runoff event resulting from this storm was plotted from the beginning of the hydrograph response through 16 Sept. 2006 (15 September at the tile gauges). Discharge was essentially stabilized at that time and additional rainfall on 16 September resulted in increases in discharge (observed early on 17 September at TC325). Discharge data at TC325, TC242, and TC240 were aggregated to 0.5-h averages, and at TC101

![Radar-estimated Rainfall (mm)](image)

**Fig. 2.** Rainfall amounts received on 10 to 11 Sept. 2006, with a map based on radar rainfall data showing relative distribution of rainfall across the Tipton Creek, Iowa, watershed, estimated based on methods described by Cruse et al. (2006).
to 300-s (5-min) averages. At each station, recorded sample times for each nutrient concentration and E. coli count datum were tabulated to match the time interval in which they fell, and linear interpolation was used to estimate concentrations—count values for intervening time periods. Contaminant loads and loading rates (expressed on cumulative-time and unit-time bases, respectively) were calculated by multiplying discharge by contaminant concentration and dividing by drainage area.

Hydrograph Separations

A common characteristic observed during hydrographs in this watershed is that total P concentrations increase during the rising limb of the hydrograph, then decline during hydrograph recession. Nitrate-N concentrations behave in opposite fashion, decreasing during the rising limb of the hydrograph then increasing during its recession. This difference results from the difference in timing of the dominant pathways delivering these two nutrients: NO$_3^-$-N via subsurface drainage, and total P via surface runoff. The dynamic lends itself to a chemical separation analysis, particularly for the tile outfalls where discharge originates from only overland flows entering surface intakes and subsurface tile drainage pathways. Some soil preferential (macropore) flow could be included in surface-intake flows, and this analysis lumps the two pathways to some extent. However, water that accumulates in potholes and enters intakes is also most likely to contribute to soil-macropore flows. This is because the Okoboji soils found in depressions have the greatest vertic (cracking) properties within this landscape, while better drained soils are loam-textured and less prone to cracking.

An approach to separate tile outfall hydrographs into tile drainage and surface intake components was developed as follows. This is essentially a dual mixing model for NO$_3^-$-N and total P. For any given time interval during the event, the mass balance of P is expressed as

$$P_iQ_i = P_iQ_i + P_iQ_i$$  \[2\]

where $Q$ is discharge, $P$ is total phosphorus concentration, and $e$, $t$, and $i$ subscripts refer to event (total observed) flow, and tile and intake components of flow, respectively. Similarly for NO$_3^-$-N:

$$N_iQ_i = N_iQ_i + N_iQ_i$$  \[3\]

where $N$ refers to NO$_3^-$-N concentration. These equations can be simplified because

$$Q = Q_i + Q_e$$  \[4\]

Dividing Eq. [4] by $Q_e$, the result allows substituting [1 – $Q_i/Q_e$] for [$Q/Q_i$] in Eq. [2–3], which for Eq. [3] is rearranged to:

$$Q = Q_i[(N_i - N)/N_i - N]$$  \[5\]

An estimate of $N_i$ can be obtained from the NO$_3^-$-N concentrations at the beginning and end of the event when discharge is stable and $Q_i$ is zero. Therefore, this equation has two unknowns, $Q_i$ and $N_i$. Here, a range of plausible values for $N_i$ were assumed and then Eq. [5] was solved for $Q_i$. Concentrations of NO$_3^-$-N in surface runoff from tile-drained soils in this region are known to be small; Zhao et al. (2001) reported 1.0 mg NO$_3^-$-N L$^{-1}$ was seldom exceeded in a Minnesota study on similar soils, while Cambardella et al. (1999) reported an average of 1.8 mg NO$_3^-$-N L$^{-1}$ in water standing in glacial potholes during a 4-yr study on the Iowa’s Des Moines Lobe. Recently from Ontario, Drury et al. (2009) reported nitrate in surface runoff from tile-drained plots (those without gated drainage control) usually varied between 1.5 and 3.5 mg NO$_3^-$-N L$^{-1}$, but these plots had tiles placed at 0.6-m depth which would tend to increase NO$_3^-$-N concentration compared with tiles installed at 0.9- to 1.2-m depth, which is typical in this Iowa landscape. The September timing of this event meant soil nitrate was well distributed in the profile, and nutrient uptake into the standing crops was culminating. Therefore, diffusion of nitrate from soil into runoff water should be minimal at this time of year. Noting that NO$_3^-$-N concentrations in surface runoff from TC101 were at or below the detection limit during this event, we assumed a value of 0.5 mg NO$_3^-$-N L$^{-1}$ in intake water, but also evaluated the effect of intake water having NO$_3^-$-N concentrations between 0.0 and 3.5 mg NO$_3^-$-N L$^{-1}$.

Nitrate concentrations in water delivered by subsurface tile drains (N) in this watershed is clearly >3.5 mg NO$_3^-$-N L$^{-1}$. The success of the hydrograph separation relies on this difference, but any change in the NO$_3^-$-N concentration in tile waters during the event must be accounted for to ensure the separation is accurate. Commonly, during the growing season, NO$_3^-$-N concentrations in tile discharge at the end of a runoff hydrograph have been greater than antecedent concentrations. Such increases can be attributed to mineralized N that builds up in soils during dry periods (Cambardella et al., 1999), which with time can increase the amount of NO$_3^-$-N that is subject to leaching through soils to shallow groundwater and tiles. To conduct the separation, the timing of this increase in tile NO$_3^-$-N concentration (i.e., increase in $N$) during the event must be estimated. Several timings were considered using trial and error, and a linear increase from the pre-event to the maximum postevent concentration was assumed to occur between the initiation of runoff to the time of peak discharge. Extending the increase to significantly after the peak resulted in mass imbalance during recession, and the separation results were relatively insensitive to other plausible variations in timing of the increase. Therefore, variation in intake nitrate ($N$) was used to provide uncertainty estimates for the separations. Calculations based on these assumptions provided hydrograph separations for both tiles (TC240 and TC242). The fraction of total P loads associated with surface intake and tile flows was quantified for the tile gauge sites, based on Eq. [2] and [4].

At the watershed outlet (TC325), groundwater is a source of stream discharge that must also be considered in conducting the separations. For this event, antecedent discharge at TC325 was 0.011 mm d$^{-1}$, and NO$_3^-$-N concentrations were <0.3 mg L$^{-1}$. Tomer et al. (2008a) reported nondetectable NO$_3^-$-N concentrations were modal when discharge was <0.02 mm d$^{-1}$ in this watershed, and interpreted this to mean groundwater discharge dominated at $Q < 0.02$ mm d$^{-1}$. Unit-area antecedent discharge (i.e., on 9 Sept. 2006) at TC325 exceeded that observed at TC240 and TC242, supporting the notion that groundwater flow dominated at the outlet at the beginning
of the event. Peak event discharge at TC325 was >200 times the pre-event discharge, and occurred about 35 h after runoff initiated. Groundwater flows to the stream would not significantly increase from pre-event discharge until recession of water stage enabled groundwater return flows to occur from saturated stream banks. Therefore, increased groundwater flows at TC325 could only occur once the hydrograph began its recession. The maximum groundwater (recessional) flow was estimated as the difference in discharge between the tiles and TC325 at the end of the hydrograph; a linear increase from pre-event to this maximum groundwater discharge rate was assumed to occur during the first 30 h of recession; that is, until the inflection of the recessional curve occurred based on visual interpretation. This groundwater discharge was subtracted from the observed hydrograph at TC325 and then the separation was then conducted as described for the tile outfalls. This approach could be checked based on NO₃–N dilution in the stream; tile outfall NO₃–N concentrations, if diluted by the proportion of estimated groundwater flow, were similar to those (within 1.5 mg NO₃–N L⁻¹) observed during late recession at TC325. Biological losses in the stream could affect this check but would be limited because the travel time from TC240 to TC325 was estimated to be only 17 h for discharge of 1.0 m³ s⁻¹, based on the discharge at the end of 16 September and the discharge-velocity curve at TC325. Note groundwater discharge is of hydrological interest, but in this watershed is not a key pathway for the contaminants considered here. At TC325, values of $N_i$, both initial and recessional nitrate concentrations, were based on those observed at TC240 and TC242, and the sensitivity of the separation to variation in $N_i$ values was determined.

Results and Discussion

A summary of a 3-yr record at Tipton Creek provides context to the September 2006 event. During 2005 to 2007, annual precipitation averaged 838 mm near TC101, and 817 mm near the tile outfalls. Rainfall during 2006 was near the average; however, the event of 10 to 11 Sept. 2006 was preceded by dry summer weather, with only 68 mm of rainfall during the prior month.

During these 3 yr of monitoring, there were 13 rainfall-runoff hydrograph events during which peak total P concentrations exceeded 1 mg P L⁻¹ at TC240 and/or TC242, and 11 hydrograph events that were associated with peak total P concentrations >1 mg P L⁻¹ at TC325 (Fig. 3). Large P concentrations (i.e., >1 mg P L⁻¹) did not occur during all events at all three stations, but this does confirm that the magnitude of P concentrations observed during the September 2006 event (indicated in Fig. 3) was not unusual. Large nitrate concentrations were also common during this 3-yr period. Concentrations exceeded 10 mg NO₃–N L⁻¹ in 95% of the event samples collected at the two tiles, and in 72% of the samples at TC325. Concentrations ranged up to 34 mg NO₃–N L⁻¹ in the tiles, with median concentrations of 24 mg NO₃–N L⁻¹ at TC240 and 21 mg NO₃–N L⁻¹ at TC242. At TC325 the maximum observed concentration was 28.6 mg NO₃–N L⁻¹, with a median of 12.7 mg NO₃–N L⁻¹.

Total discharge during 3 yr of monitoring was greater at TC325 than observed from the tile-drained basins (Table 1). On a unit-area basis, the larger tile basin (TC240) underrepresented the discharge from the watershed by about one-third, but the difference between TC242 and TC325 was only about 10%. Tile systems obviously do not capture all surface and subsurface flows and this appears particularly true of the larger basin. Despite the smaller hydrologic discharge, NO₃–N loads discharged from the tile basins exceeded that observed

![Fig. 3. Three-year record of total P concentrations observed during rainfall-runoff events at two tile outfalls (TC240 and TC242) and at the outlet (TC325), Tipton Creek, Iowa. Arrows indicate September 2006 event.](image)

Table 1. Comparison of hydrologic discharge, and loads of NO₃–N and total P from Tipton Creek, Iowa (TC325), and from two tile-drained subbasins (TC240 and TC242).

<table>
<thead>
<tr>
<th>Gauge station</th>
<th>Drainage area (ha)</th>
<th>Mean discharge (mm yr⁻¹)</th>
<th>NO₃–N load (kg ha⁻¹ yr⁻¹)</th>
<th>Total P load (g ha⁻¹ yr⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TC325</td>
<td>19,850</td>
<td>280</td>
<td>31.1</td>
<td>581</td>
</tr>
<tr>
<td>TC240</td>
<td>1,700</td>
<td>187</td>
<td>33.1</td>
<td>170</td>
</tr>
<tr>
<td>TC242</td>
<td>156</td>
<td>252</td>
<td>38.2</td>
<td>195</td>
</tr>
</tbody>
</table>
at TC325 on a unit-area basis during 2005 to 2007. The tile basins also discharged a substantial amount of total P, but only about one-third the unit-area load observed at TC325 (Table 1).

September 2006 Hydrograph Event

Rainfall began during the early morning of 10 Sept. 2006 (Fig. 2), and an increase in stream stage at TC325, in response to direct rainfall into the stream channel, was quickly evident (Fig. 4). Thereafter, the TC325 hydrograph indicated surface runoff contributions began about 1600 h on 10 September, peaked on 11 September around 0800 h, and then decreased until about 1600 h. At this time, increased discharge from tile-drained areas upstream began to dominate at TC325, and peaked at about 0600 h on 12 September, with hydrograph recession thereafter (Fig. 4). The peak in surface runoff observed at TC325 lagged the hydrograph peak at TC101 by about 4.5 h, and the second peak at TC325, attributed to tile discharge, lagged the peak discharge observed at TC240 by 24 h (Fig. 4). Based on discharge-velocity data, that water actually discharged during hydrograph peak at the tile outfalls could have arrived at TC325 within about 15 h, and then contributed to the leading edge of the second rising limb observed at TC325. The bulk of tile water discharged during the hydrograph peak at TC325 would have originated from the watershed’s upper reaches, and would have passed the monitored tile outfalls at least 12 h before reaching TC325. These time estimates are minimum values because they assume uniform channel velocity, but in this watershed, channel slopes are least in the upper reaches, which are comprised of dug ditches. The lag between surface runoff and tile discharge within the hydrograph at TC325 (Fig. 4) was not only due to the greater distance the upstream tile discharge had to travel in the channel, but also the landscape contrast between steeper slopes in the lower watershed and flatter tile-drained lands in the upper watershed (Tomer et al., 2008b). Greater rainfall in the lower watershed (Fig. 2) may have accentuated the lag seen in this hydrograph, but lagged timing of surface runoff and tile discharge is typical of hydrographs in this watershed (data not shown).

Discharge from tile gauges for this event (Fig. 5) showed a more typical hydrograph form (particularly for TC240) reflecting a reasonably uniform rainfall intensity during the event in these subsbasins and essentially uniform terrain. Discharge from the field gauge at TC101 (Fig. 6) is more sporadic, but does reflect greater rainfall intensity late in the event at that location (Fig. 2).

Event Water Quality

Water quality results for all four gauges show the greatest concentrations of total P and populations of E. coli occurred early in the hydrograph event (Fig. 5–7). At TC240, the E. coli in three early samples saturated the Quanti-Trap panels despite a 20-fold dilution, and therefore represented minimum populations of 48,384 cells 100 mL⁻¹. Saturated counts also occurred with two samples collecting during the rising limb of the hydrograph at TC325, with the same 20-fold dilution for one sample and a 1000-fold dilution for another taken at 1820 h on 11 September that indicated a minimum population of
2,419,200 cells 100 mL$^{-1}$. Total loads of *E. coli* were calculated using these minimum numbers and, therefore, are conservative for TC240 and TC325. The maximum *E. coli* count at TC242 was 16,343 cells 100 mL$^{-1}$, and at TC101 was 86,645 cells 100 mL$^{-1}$. During hydrograph recession, *E. coli* populations decreased to 333 cells 100 mL$^{-1}$ at TC240 and 79 cells 100 mL$^{-1}$ at TC242, while the final sample of surface runoff collected at TC101 was 2605 cells 100 mL$^{-1}$. Yet, at TC325, populations only decreased to 6690 cells 100 mL$^{-1}$ by the time the hydrograph receded to about one-third of peak discharge on the morning of 14 Sept. 2006.

Pre-event total P concentrations in tile discharge (Fig. 5–7) were 0.2 mg P L$^{-1}$ at TC240 and 0.09 mg P L$^{-1}$ at TC242, but were smaller during late recession; that is, 0.08 mg P L$^{-1}$ at TC240 and 0.06 mg P L$^{-1}$ at TC242. At TC325, pre-event total P was 0.1 mg P L$^{-1}$, but was 0.2 mg P L$^{-1}$ during late recession; this increase can be attributed to greater suspended sediment. During the event, maximum total P concentrations were 3.74 mg P L$^{-1}$ at TC240, 2.74 mg P L$^{-1}$ at TC242, 2.72 mg P L$^{-1}$ at TC325, and 1.44 mg P L$^{-1}$ in surface runoff at TC101 (Fig. 5–7). That the greatest total P concentrations were observed in tile discharge rather than at the flume may be surprising, but the tile discharge included surface intake flows from pothole soils that may contain excess P due to a history of repeated manure applications. Specific information about this was not available, but Karlen et al. (2008) documented greater P concentrations in manure-amended soils across the South Fork watershed. Tomer et al. (2008b) estimated that most fields in these tile-drained subbasins would be expected to receive manure, based on the number, sizes, and proximity of livestock facilities in the watershed. At TC101, manure application had not occurred in the past year at the time of this hydrograph event.

At both tiles and at TC325, NO$_3$–N increased from pre-event to late recession. The increase at TC325 was from nondetectable (<0.3 mg NO$_3$–N L$^{-1}$) to a maximum of 19.3 mg NO$_3$–N L$^{-1}$ during recession (Fig. 7). At the tiles, NO$_3$–N concentrations decreased at first due to tile intake (surface runoff) contributions, then increased to exceed pre-event concentrations (Fig. 5). At TC240, pre-event NO$_3$–N concentration was 14.5 mg NO$_3$–N L$^{-1}$, which decreased to 9.4 mg NO$_3$–N L$^{-1}$, then increased to a maximum of 23.2 mg NO$_3$–N L$^{-1}$ during recession. At the smaller tile TC242, pre-event NO$_3$–N concentration was 14.6 mg NO$_3$–N L$^{-1}$, which decreased to 1.9 mg NO$_3$–N L$^{-1}$, then increased to 18.4 mg NO$_3$–N L$^{-1}$ during recession.

### Hydrograph Separations

Hydrograph separations provided estimates of the proportion of tile discharge originating from surface intakes and the proportion of total stream discharge originating from overland flows. Results indicated that 12.7% of the discharge from tile outfalls originated from surface intakes, and that 21.2% of the discharge from the watershed outlet originated from overland flows (Fig. 8 and Table 2). These values assume that overland flows contain 0.5 mg NO$_3$–N L$^{-1}$. Varying this assumed value between 0 and 3.5 mg NO$_3$–N L$^{-1}$ indicated between 12 and 15% of the tile outfall may have originated from surface intakes, and that between 20.9 and 24.8% of total watershed discharge may have been overland flow contributions. Similar separations of tile-outfall hydrographs from a small watershed in central Iowa yielded estimates that tile intakes contributed about 17% of storm discharge (Schilling and Helmers, 2008).

The hydrograph separation for TC325 assumed that initial and recessional NO$_3$–N concentrations at the gauged tiles represented the end-member concentrations for the watershed. The initial NO$_3$–N concentrations at TC240 and TC242 were nearly identical (14.5 and 14.6 mg NO$_3$–N L$^{-1}$, respectively).
During late hydrograph recession, the area-weighted average of TC240 and TC242 was 22.5 mg NO$_3$–N L$^{-1}$. The hydrograph separation at TC325 was insensitive to the initial concentration; varying ±2 mg NO$_3$–N L$^{-1}$ influenced the separation results by 0.02 mm. The separation results were more sensitive to variation in the recessional tile concentration; a change of ±0.5 mg NO$_3$–N L$^{-1}$ in concentration altered the separation results by about 1% (0.06 mm). Varying the recessional concentration also determines the duration of runoff by an amount that depends on the minor changes in observed NO$_3$–N concentration during recession. Using 22.5 mg NO$_3$–N L$^{-1}$ for this recessional tile-end-member concentration, surface runoff was estimated to cease near 0600 h on 14 Sept. 2006, which was about 22 h after surface runoff (i.e., inlet flows) ceased at TC240 and was <5% of discharge at TC242. This travel time is deemed reasonable given discharge-velocity data at TC325 and the position of the gauged tiles in the watershed (Fig. 1). The estimated duration of runoff at TC325 decreased by 4 h if the recessional tile concentration was decreased to 22 mg NO$_3$–N L$^{-1}$, and increased by 3 h if it was increased to 23 NO$_3$–N L$^{-1}$. Larger changes led to runoff cessation times that did not appear realistic based on graphical hydrograph interpretation and travel time information.

On a unit-area basis, cumulative event discharge at TC325 exceeded that from the tiles by about 1.8 mm (Table 2); the difference results mainly from groundwater contributions at the watershed scale, and heavier rains and surface runoff in the lower part of the watershed. An estimate of groundwater discharge late in the event at TC325 was determined from the difference in discharge rates between tile subbasin and watershed scales at the end of the hydrograph, assuming a 17-h time lag between the tile and stream gauges, which was calculated based on the discharge rate at the end of the hydrograph and discharge-velocity data. The hydrograph-estimated groundwater contribution amounted to about 10% of the total event discharge (Fig. 8 and Table 2), and comprised 22% of the cumulative discharge during the final 3 d of the event. A second estimate of groundwater contribution, based on dilution of maximum NO$_3$–N concentration observed at the tile outfalls (weighted average of 22.5 mg NO$_3$–N L$^{-1}$) compared with that observed at TC325 (16.8 mg NO$_3$–N L$^{-1}$), suggests groundwater flow late in the event at TC325 was 25% of $Q$, assuming that measured tile concentrations were representative of the watershed and that groundwater flow was fully denitrified. A watershed modeling (SWAT) study of the South Fork watershed estimated groundwater flows comprised about 6% of annual discharge (Green et al., 2006).

Tile drainage contributions to flow at TC325 comprised an estimated 68.6% of the total discharge, which is in close agreement with a SWAT model estimate of 71% of annual average discharge originating from tiles (Green et al., 2006). Hydrograph-separation estimates of unit-area tile flow contributions from TC240 and TC242 were less than the estimate at TC325 by about 12% (Table 2). However, the watershed is not all tile drained and the actual discrepancy is therefore >12%. If 85% of the watershed is tile drained, then estimated tile discharge at TC240 (per unit area) is 21% less than the watershed average. While groundwater discharge may have been underestimated, these tile basins appear to underrepresent tile drainage from the watershed as a whole (Table 1), due to a combination of factors that include the size, flatness,
and drainage inefficiency of the TC240 basin. Also, part of the TC242 basin is sloping and probably not tile drained.

Cumulative surface runoff from TC101 was 5.28 mm (Table 2), while surface runoff from the watershed was estimated to be 1.26 mm based on hydrograph separation at TC325 (Table 2). If tile intake contributions at TC240 and TC242 represent those from the watershed’s tile-drained lands and 85% of the watershed is tile drained, then surface runoff from the non-tile-drained lands in the watershed during this event would have averaged 4.24 mm. Therefore, the TC101 runoff response during this event appears to exceed that of the watershed’s non-tiled lands (by about 20%), which would be expected given the rainfall distribution (Fig. 2). Cropped fields similar to the TC101 site (i.e., with >50% highly erodible soils and within 1 km of the stream) occupy about 10% of the Tipton Creek watershed. These comparisons show the hydrologic balance is reasonable (within about 20%) given the difference in scales, and provide context for evaluating contaminant loads from the different gauge sites.

**Contaminant Loads and Sources: Nutrients and E. coli**

Cumulative NO$_3$–N loads were similar between the tile outfalls and TC325; that is, the NO$_3$–N load from the gauged subbasins was 89% of that observed at the outlet, on a unit-area basis (Fig. 9a and Table 3). This confirms the dominant role that tile drainage plays in delivering NO$_3$–N loads discharged from this watershed. The similarity in the shape of the cumulative-load curves with time (Fig. 9a) between the tiles and the stream outlet further supports tile drainage as the dominant pathway for NO$_3$–N. The difference in area-weighted NO$_3$–N loads between TC325 and the tiled subbasins probably result from TC240 and TC242 underrepresenting the watershed’s bulk tile discharge. The unit-area load from the TC242 subbasin was less than that from the TC240 subbasin, where unit-area loads were 95% of that observed at TC325.

Cumulative loads of total P in contrast, showed how multiple sources were involved in delivering P loads from this watershed (Fig. 9b and Table 3). Tile gauges (TC240 and TC242) generated about half the total P load observed at the watershed scale, on a unit-area basis. This may underrepresent the watershed-scale contribution of P from tile-drained lands given the differences in discharge, but if 85% of the watershed is tile drained and these basins are representative, then 42% of the P load at TC325 for this event originated from tiles. The magnitude of the tile P load was unexpected, particularly as tile samples had good clarity even at the beginning of events when P concentrations were high. Based on observation, water entering tile intakes is ponded and any sediment load it carried as overland flow has become deposited in the glacial depressions and roadside ditches where the intakes were installed. These intakes delivered at least 75% of the P load discharged from the tiles, based on hydrograph separation results (Table 3). The actual fraction of this event’s tile P load that was dissolved is not known, but Tomer et al. (2008a) found dissolved P to total P ratios were at least 0.90 in 73% of tile-water samples manually collected from Tipton Creek during 2002 through 2005.

The surface flume (TC101) also discharged a significant P load during this September 2006 event (Fig. 9b and Table 3). However, if the P load from the surface flume scales up to represent 10 to 15% of the watershed, then these near-stream sources of overland flow contributed about one-quarter to one-third of the total P load, which was less than that estimated from tiles. The shape of the curves for P load accumulation (Fig. 9b) showed the field and tile discharge were dominated by rapid pulse of P, with the difference in slope of P accumulation reflecting differences in hillslope gradients on the landscape. The lag between these pathways was apparent on the total P curve for TC325 (Fig. 9b). Bank sediments also played an important role in P transport and this will be further considered with sediment-source information below. Hydrograph separation of P loads at TC325 was not attempted at the outlet due to bank sediments comprising a third source of P discharged at the watershed scale; that is, Eq. [1–4] are only valid for a two-source separation.

Loads of E. coli were substantial, also with multiple sources contributing (Fig. 9c and Table 3). Tile-discharge loads of E. coli...
were estimated at 30% that observed at TC325 on a unit-area basis. Loads of \textit{E. coli} through the TC101 were also considerable, but did not exceed the unit-area load at TC325 even using minimum counts from saturated media. Results reflect the ubiquity of this indicator of fecal contamination, indicating agricultural applications of livestock manure to cropland, which occur before planting in spring or following harvest in fall, could not be its sole source during this event, which occurred under full crop cover. Tomer et al. (2008a) reported \textit{E. coli} populations in South Fork waters were least in tile discharge, and increased in stream water during times of surface runoff contributions. However, significant seasonal differences in mean populations among watersheds (i.e., South Fork, and Tipton and Beaver tributaries) did not follow expectations based on the number of livestock facilities in the watersheds. Cattle grazing, wildlife, and humans are other sources that may be important. In particular, runoff from riparian pastures can more directly contribute to \textit{E. coli} loads than upland manure applications, which dominantly are incorporated or injected into the soil. Long-term survival of \textit{E. coli} in streambed sediments also appears to contribute to \textit{E. coli} loads, based on two lines of evidence. First, during hydrograph recession when surface runoff has attenuated but some sediment from channel sources remained suspended, \textit{E. coli} populations remained greater than pre-event populations. Second, and as further discussed below, the accrual of \textit{E. coli} load (Fig. 9c) and changes in population (Fig. 7) during this event followed the timing of sediment concentrations more closely than they did hydrologic fluxes. Tomer et al. (2008a) reported viable populations in stream sediment, up to 1181 cells g$^{-1}$ of sediment.

\textbf{Sediment Loads and Sources}

Changes in turbidity during this runoff event showed two distinct pulses of increased sediment associated with surface runoff from the lower watershed and then with tile contributions from the upper watershed (Fig. 10). The second, tile-sourced pulse of water clearly picked up bed and lower bank sediments as discharge began to increase, but sediment concentrations attenuated before the hydrograph peak, by about 6 h. This may seem unusual, but peak discharge during this event was only about half the channel capacity, with bank full discharge estimated to be 11.6 m$^3$ s$^{-1}$. Therefore, the discharge had limited capacity to scour banks, and stream banks did not become saturated to the extent they would during large events. There were consequently minimal sediment contributions resulting from collapse of banks during hydrograph recession, an often-important process described elsewhere (Rinaldi and Casagli, 1999; Fox et al., 2007). The largest sediment concentrations occurred during the initial discharge of surface runoff in the event, when upland contributions were also greatest. The

\begin{table}[h]
\centering
\begin{tabular}{|c|c|c|c|c|}
\hline
\textbf{Gauge site} & \textbf{Description} & \textbf{NO$_3$-N} & \textbf{Total P} & \textbf{E. coli} & \textbf{Sediment} \\
& & $g$ ha$^{-1}$ & $g$ ha$^{-1}$ & $10^9$ ha$^{-1}$ & Mg ha$^{-1}$ \\
\hline
\text{TC325} & Watershed outlet & 940 & 27.3 & 12.50 & 0.016 & 0.004" \\
& Overland sources & 4 & X† & X & X \\
& Subsurface (tile + groundwater) & 936 & X & X & X \\
\text{TC240} & Tile outfall & 890 & 13.5 & 3.98 & X \\
& Surface intakes & 3 & 10.2 & X & X \\
& Subsurface tiles & 887 & 3.3 & X & X \\
\text{TC242} & Tile outfall & 276 & 7.6 & 0.45 & X \\
& Surface intakes & 3 & 6.7 & X & X \\
& Subsurface tiles & 273 & 0.9 & X & X \\
\text{TC240 + 242} & Area-weighted average & 839 & 13.0 & 3.70 & X \\
\text{TC101} & Surface flume & 11§ & 66.4 & 8.33 & X \\
\hline
\end{tabular}
\caption{Cumulative loads of four contaminants observed at four gauging stations during the September 2006 rainfall-runoff event, Tipton Creek, Iowa, expressed on a unit-area basis. Nutrients and sediment loads are apportioned to major source-pathways for TC325, TC240, and TC242, based on results of source-pathway separation methods discussed in the text.}
\end{table}

† X, not determined.
‡ The remaining sediment load is attributed to channel sources.
§ Includes nondetectable concentrations in multiple samples, assumed to be 0.15 mg NO$_3$-N L$^{-1}$
upland (surface-soil) sources of sediment attenuated quickly as rainfall ceased and surface runoff diminished. As increased tile discharge was routed toward TC325, it entrained greater amounts of sediment sourced from the bed and lower banks (Fig. 10). Suspended sediment collected before the hydrograph peak averaged 31% from upland sources, whereas samples collected during recession averaged only 17% upland sources.

Integrated across the event, results showed approximately 78% of the sediment load was sourced from the channel, and 22% from surface soils (Fig. 11). This surface-soil contribution compares closely with an estimated 25% of P loads sourced from surface soils, if P loads discharged from TC101 represented the 10% of the watershed in cropped fields with erodible soils. Most P, sediment, and *E. coli* were discharged during the rising limb of this event hydrograph. Relative to sediment, P and *E. coli* were discharged more rapidly when one compares rates of accumulation (Fig. 9 and 10). This probably resulted from a “first flush” effect and delivery of these contaminants in surface runoff early in the event, which was observed at all gauge locations. An estimated 1612 g of P was discharged for every Mg of sediment during this event. The accrual of loads of both P and *E. coli* (Fig. 9) follow that of sediment in one important respect: that the apparent inflection (where the rate of accrual of P and *E. coli* begins to diminish) occurred at the beginning (near 2400 h) of 12 September, exactly when sediment concentrations peaked but before the hydrograph peak by about 6 h. Note that for both contaminants, peak concentrations occurred before the hydrograph peak (Fig. 7), and this occurred with sediment as well (Fig. 10). This emphasizes the likelihood that channel sediment is a significant source of both contaminants. However, channel-derived sediment appeared to be more important in delivering *E. coli* than P. In contrast to P, accrual of the *E. coli* load becomes steepest only on arrival of the tile-derived flows, when sediment loads become >75% channel derived. Both saturated Quanti-Tray panel counts of *E. coli* at TC325 also occurred when tile flows, rather than runoff, began to dominate the hydrograph (i.e., the fifth and sixth sample points shown in Fig. 7 represent saturated count values).

### Interpreting Results for Contaminant-Specific Mitigation Strategies

In any watershed, each rainfall-runoff event is unique and influenced by storm dynamics and antecedent moisture, discharge, and vegetation. This particular event was a significant storm that triggered a somewhat muted hydrologic response due to dry antecedent conditions. Mature stands of corn and soybean crops were the dominant cover. Typically, corn is near physiological maturity and soybean is initiating senescence in early September in this watershed. Results of monitoring this event must be viewed in the context of these conditions. One key implication of these conditions is that recent tillage, manure applications, or other operations did not impact contaminant loads. Results from this event are reasonably consistent when compared with 3-yr annual averages for discharge and nutrients (Table 1). Therefore, on balance, results of hydrologic and water quality monitoring at several scales in this watershed indicate the dominant pathways each contaminant followed in this event.

This paper’s intent was to use the study results and interpretations to then make statements about the pathways that should be addressed in mitigating specific contaminants, which are put forth as follows.

#### Nitrate-Nitrogen

Tile drainage is the dominant pathway. Mitigation practices include improved agricultural N management, nutrient removal wetlands, and other biological filters that can achieve full denitrification of NO$_3$–N to N$_2$.

#### Total Phosphorus

Surface intakes in tile drained areas are an underappreciated source of P, and calculated to be the most significant transport pathway for
P in this event. Manure application histories probably help determine the variation in P contributions via this pathway among individual fields. Channel-source sediments and surface runoff also play important roles as sources of P. Buffered surface intakes in tile-drained areas, optimized erosion control on steeper cropped fields that are near the streams, and vegetation management to stabilize stream banks, if combined, should comprise an effective strategy to reduce P loads. Surface intakes could be buffered using vegetated filter strips, and/or include filter media that can adsorb P (e.g., water treatment residuals) as part of the design of the intake.

E. coli

Reducing bacterial loads could in part be accomplished through the same practices that would reduce P loads from tile intakes and surface runoff. However, direct fecal sources (riparian pasture management) should be considered for this contaminant to be addressed comprehensively.

Sediment

Channel sources dominate, and could be most viably addressed through conservation easements (e.g., riparian buffers) or rotational grazing to increase rooting of riparian vegetation and thereby enhance bank stability. This would be most beneficial in the lower riparian valley. (This interpretation is aided by interpreting bank conditions shown by video taken along the stream from a helicopter; see Simon and Klimetz, 2008). Other practices that can reduce stream power, such as reestablishing meanders along straightened reaches (Yan et al., 2010), could also be considered.

Results highlight the importance of adding surface-intake buffers and improved riparian management to the suite of conservation measures currently being encouraged in the watershed. Other than improved nutrient management to limit NO₃–N losses, practices to reduce these contaminants would involve a small proportion of this watershed's land area, yet would address these contaminants comprehensively, if supported by continued monitoring to assess and improve effectiveness.

Conclusions

This study showed that results from comprehensive monitoring of a single rainfall-runoff event, when viewed in context, can inform water quality planning in an agricultural watershed. Monitoring of multiple contaminants at multiple scales, when combined with hydrograph separations and sediment source tracking supported what was already understood about NO₃–N loads in this watershed, but also revealed new information about total P, E. coli, and sediment sources that helps identify key pathways of several specific contaminants. In this watershed, while sediment is dominantly sourced from stream channels and NO₃ is dominantly sourced by tile drainage, multiple sources of total P and E. coli need to be addressed to mitigate these two contaminants comprehensively. The role of surface intakes in delivering these two contaminants in tile-drained watersheds is generally not well recognized. During this event, hydrograph separations indicated tile intakes delivered about half the total P load and a third of the E. coli load observed at the watershed outlet, when scaled on a unit-area basis. Short-term, intensive, and multiscale monitoring of rainfall-runoff events could be used more frequently to assess and quantify important source-pathways of multiple agricultural contaminants. In this study, management interpretations were aided by a 3-yr monitoring record, aerial video of the stream channel, and a hydrologically calibrated watershed model.

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