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Assessment of the Iowa River's South Fork watershed: Part 1. Water quality

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Abstract: Iowa's South Fork watershed is dominated by corn (*Zea mays* L.) and soybean [*Glycine max* L. (Merr.)] rotations, and animal feeding operations are common. Artificial subsurface (tile) drainage is extensive; hydric soils cover 54% of the watershed. During spring and early summer, NO_3-N concentrations in tile and stream discharge often exceed 20 mg L^{-1}. Total N loads during 2002 to 2005 ranged from 16 to 26 kg NO_3-N ha^{-1} y^{-1} (14 to 23 lb ac^{-1} yr^{-1}). Nitrate concentrations increased linearly with log baseflow, effectively a surrogate measure of tile discharge. Phosphorus loads were only 0.4 to 0.7 kg P ha^{-1} y^{-1} (0.4 to 0.6 lb ac^{-1} yr^{-1}), but concentrations commonly exceeded 0.1 mg L^{-1}, a eutrophication-risk threshold. Mean *E. coli* populations in the stream exceeded 500 cells 100 ml^{-1} during summer. Statistical comparison of actual nitrate records with independent records generated using regression equations provided modeling efficiencies of 0.91 or less, suggesting performance targets for watershed model validation. Tile drainage is more important in transport of nitrate and dissolved phosphorus than *E. coli*. Variations in nitrate, phosphorus, and *E. coli* are uniquely timed, highlighting the complexity of integrated water quality assessments.

Key words: bacteria—Conservation Effects Assessment Project (CEAP)—conservation practices—*E. coli*—manure—nitrate—phosphorus—subsurface drainage

A number of water quality studies in the Midwestern Corn Belt have investigated the transport of nitrogen, phosphorus, pesticides, and fecal bacteria at a variety of scales, and improved our understanding of factors contributing to the entry of these contaminants into water resources (e.g., Blanchard and Lerch 2000; Klatt et al. 2003; Tomer et al. 2003; Royer et al. 2006). Likewise, many previous studies have led to an improved understanding of water quality effects of conservation practices at the scale of small plots or catchments. Watershed scale assessments of the effectiveness of conservation practices in protecting water quality capture a spectrum of landscapes, farming practices, crops and weather patterns that contribute to the observed water quality profile. The Conservation Effects Assessment Project (CEAP) has included benchmark watershed assessment studies to provide validation of modeling efforts and an improved understanding of conservation practice impacts in watersheds (Mausbach and Dedrick 2004). This and our companion paper (Tomer et al. 2008) provide an assessment of water quality, agricultural systems, and conservation practices in one CEAP benchmark watershed, the South Fork of the Iowa River. This watershed is in a region dominated by agricultural land that is intensively managed for livestock and row-crop production. Concentration of livestock production facilities in areas like the South Fork has led to greater concerns about agricultural impacts on the environment, and there is a need to understand how conservation systems can attenuate environmental impacts in these areas. This was a key issue that led to the South Fork's inclusion as a CEAP benchmark watershed.

Our objectives are to document the South Fork watershed's hydrology and to characterize the amounts and timing of nutrients (NO_3-N, phosphorus) and *E. coli* discharged from its streams and the relation of these discharges to agricultural practices. These water quality endpoints were chosen due to concerns over nutrient exports to local recreational waters and ultimately the Mississippi River. *E. coli* and nutrients are frequently cited as a water-quality impairment by the US Environmental Protection Agency (2008), and the effect of concentrated swine production on *E. coli* in streams have not been widely reported. A second objective was, for several stream gauging sites, to compare records of measured nitrate concentrations with statistically generated records developed using independent sets of nitrate measurements and their statistical relationships with hydrologic flows. Through these comparisons, we intend to show how replicated monitoring records can provide performance benchmarks useful in evaluating watershed model validation statistics.

Setting. The South Fork of the Iowa River covers about 78,000 ha (193,000 ac), including the tributaries of Tipton and Beaver Creeks (figure 1). The watershed dominantly lies in Hardin and Hamilton Counties, with small areas in Franklin and Wright Counties. It is representative of the Des Moines Lobe, the dominant landform region of north-central Iowa (Prior 1991). The terrain is young (about 10,000 years since the last glacial retreat), and therefore natural stream incision and development of alluvial valleys are generally limited to the lower (southeastern) third of the watershed. The upper parts of the watershed are occupied by till plains with many internally drained "prairie potholes" and several terminal moraines that cross through the watershed.

On this poorly dissected landscape, soil wetness is a major concern for land management and agricultural production. Hydric soils occupy about 54% of the watershed (Tomer and James 2004). Artificial drainage was installed to allow agricultural production, beginning more than 100 years ago. Subsurface tile drainage and dug ditches have greatly decreased water storage and travel time of water discharged from the watershed. Using a tile drainage routine within the Soil and Water Assessment Tool (SWAT) model, Green et al. (2006) estimated that 71% of total discharge from the watershed between 1996 and 2004 was tile flow, and that groundwater flow only comprised 6% of that total. The dominance of corn and soybean rotations on the landscape is facili-

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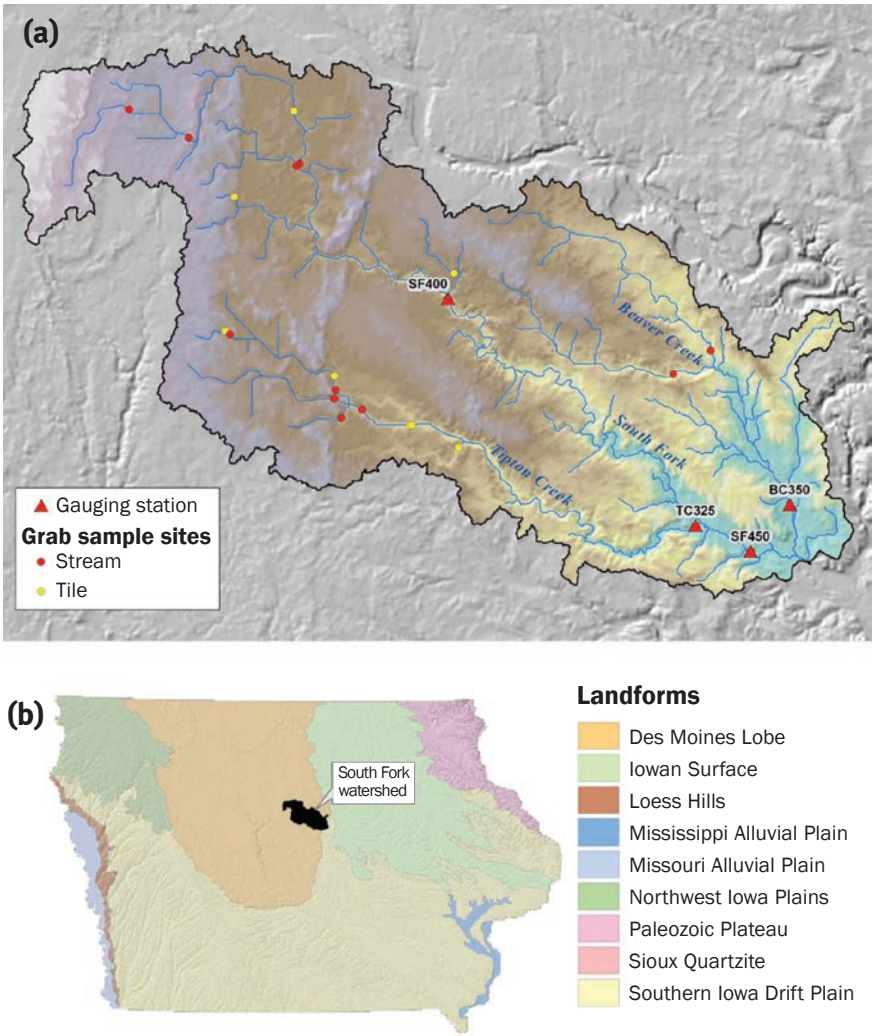
Figure 1

(a) Map of the Iowa River's South Fork watershed showing topography, streams, grab-sampling sites (stream sites and tile drainage discharge points), and stream gauge locations. (b) State map shows the watershed's location and Iowa's landform regions (based on Prior 1991).



tated by tile drainage; 80% of the watershed may be tile drained (Green et al. 2006). The soils are highly productive, with the Clarion-Nicollet-Webster soil association being dominant, forming a sequence (respectively) of moderately well drained Typic Hapludolls, somewhat poorly drained Aquic Hapludolls, and poorly drained Typic Endoaquolls (National Cooperative Soil Survey 1985; Soil Survey Staff 2003). The potholes are occupied by very poorly drained Okoboji soils (Cumulic Vertic Endoaquolls), often with calcareous and poorly drained Harps soils (Typic Calciaquolls) on their margins. Most potholes are cropped and have surface inlets to route surface water that ponds in them to the subsurface drainage network. This pathway, however, is poorly documented and remains a subject for future research.

In the past 35 years, agricultural land use has shifted from typically small, integrated crop-and-livestock farms toward larger operations focused on either livestock or row-crop production. Corn and soybean rotations now occupy about 85% of the land area. There are approximately 100 concentrated animal feeding operations in the watershed, with most producing swine. Two major subbasins, Tipton Creek and the upper South Fork (figure 1), contain most of these operations (see Tomer et al. 2008). Changes in agricultural systems continue to take place, as a facility for corn-based ethanol production has recently been constructed within only a few kilometers of the watershed's northeast boundary.

Materials and Methods

Hydrologic Discharge Data. In 1995, the US Geological Survey established a gauging station near New Providence, Iowa (SF450) (figure 1), as part of the Eastern Iowa Basins National Water Quality Assessment program (Becher et al. 2001). We expanded hydrologic and water-quality monitoring during 2000 and 2001 by establishing three additional gauging stations: one each for Tipton Creek (TC325) and Beaver Creek (BC350) and one at an upstream site along the South Fork's main stem (SF400). These stations were instrumented to measure stream stage height using bubbler-type water level recorder systems comprised of Waterlog H355 bubblers (Design Analysis Inc.) and Paroscientific PS2 pressure transducers. Periodic measurements of cross-sectional depths and flow velocities under varying flow conditions were used to establish and maintain rating curves defining the relationship between stage height and discharge at each station. These cross-sectional measurements were repeated after major runoff events to identify any changes in the streambed that could influence the rating curve. Hydrologic data were processed using the Water Information System Kisters (WISKI) hydrologic database software (Kisters 2007; Schlaeger et al. 2005; Malinsky et al. 2002), which includes customized software that automatically accounts for changes in rating curves and interpolates missing or aberrant data using methods that conform to US Geological Survey protocols for processing hydrologic data. Daily rainfall data were also acquired at three of the gauging stations (not SF450), using tipping-bucket type rain gauges. The discharge records were separated into runoff and baseflow components based on the filter method of Arnold and Allen (1999). Herein, discharge refers to total stream flow, runoff refers to that portion of discharge originating as overland flow, and baseflow refers to that portion of discharge originating from subsurface flow pathways including subsurface tile flow. All hydrologic flows were converted to depth equivalent (discharge volume divided by drainage area) to allow direct comparison between gauge stations.

Water Quality Sampling and Analyses. Water samples were manually collected from 23 locations (figure 1), including 11

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stream sites, eight tile-main discharge outlets and the four gauging stations. Discharge from tile mains comes from groups of fields organized into "drainage districts." Note discharge from these tile mains is comprised of subsurface tile flow (baseflow) and overland flow (runoff) entering the subsurface drainage network via surface inlets. Sampling frequency was weekly or biweekly during the growing season, and monthly during winter; only the gauging stations were sampled if frozen conditions required an ice auger for sample collection. Stream locations were sampled beginning in 2002, with tile locations in the upper South Fork and Tipton Creek basins added to the sampling program in July 2003. No accessible tile outlets in Beaver Creek were located. The manually collected (grab) samples were analyzed for NO_3-N and total P. For dissolved P, a subsample was passed through a 0.45 μm filter at the time of sample collection in the field. In addition, at each of the four gauging stations (figure 1), automated carousel-type samplers (Isco model 6712) were used to collect daily composite samples, composed of four subsamples collected at six-hour intervals, to represent an integral sample for each day. These daily samples were analyzed for nitrate and total P. The automated samplers were only operational when risk of freeze damage was minimal, averaging 220 d y^{-1}.

Sample concentrations of nitrogen as nitrate plus nitrite (herein denoted NO_3-N) were determined using a Lachat autoanalyzer employing Cd reduction (Wood et al. 1967) with detection limit of 0.3 mg L^{-1} (ppm). Data quality was assured by quality assurance/quality control procedures that included the use of (empty bottle) field blanks, a 1/15-frequency of calibration standards within the sample queue, and analytical sample duplicates. These quality assurance/quality control samples comprised 25% of the analytical workload. In their National Water Quality Assessment study, the US Geological Survey found that nitrate comprised about 90% of the total N load from eastern Iowa rivers (Becher et al. 2001). Therefore, other forms of N in surface water were not measured. Dissolved P concentrations were determined using flow injection analysis (US Environmental Protection Agency method 365, see O'Dell 1993) with a detection limit of 0.01 mg L^{-1}. Total P on unfiltered samples was analyzed by the same method but were acid-persulfate digested prior to analysis; the digest procedure increased the detection limit to 0.02 mg L^{-1}.

E. coli was measured in stream-water grab samples using 4-methyl-umbelliferyl-glucuronide in a defined medium within a modified most-probable-number (mpn) format (Colisure and Quantitray methods, see Idexx 2007). Grab samples were obtained as described previously and refrigerated in the field. In the laboratory, 100 ml (3.4 fl. oz) of sample was added to the Colisure media and dispensed into the Quantitrays. The Quantitray allows enumeration of populations between 1 and 2,419 cells (100 ml)$^{-1}$, (100 ml = 3.38 fl oz), so samples were diluted with deionized water when populations larger than 2,419 were expected. Analyses were initiated within 8 h after sample collection. Sealed Quantitrays were incubated for 24 h at 35°C (95°F), and *E. coli*-positive cells were visualized by the fluorescence of 4-methyl umbelliferone under ultraviolet light, indicating β-glucuronidase activity. The mpn values (cells 100 ml^{-1}) were calculated using the manufacturer's method (Idexx 2007).

A total of 240 *E. coli* samples were obtained from stream grab sampling sites (figure 1) Beaver Creek plus 428 from South Fork and 419 from Tipton Creek. An additional 230 samples were obtained from tile sites in Tipton Creek and the South Fork. Populations of *E. coli* were not normally distributed, but $\log_{10}$ transformation (geometric means) resulted in near-normal distributions. Differences due to season, watershed, and sites within watersheds were tested using analysis of variance (Proc, GLM, SAS) or the nonparametric Wilcoxon rank sum and Kruskall-Wallis tests (Proc NPARM, SAS) on the transformed data.

Linear regressions between grab-sample nitrate concentrations and the natural log of baseflow were determined for each of the four gauging stations. Baseflow from this watershed is dominated by tile drainage discharge (Green et al. 2006). However, dates with baseflow <0.02 mm d^{-1} (0.0008 in day^{-1}) (unit-area basis) were deleted to perform the regressions because nondetectable NO_3-N concentrations were predominant at these low flows, and their inclusion created a bimodal distribution. There were two purposes for these regressions, the first being comparison of the nitrate-baseflow relationships among the four gauging stations, which could indicate if the hydrologic drivers of N loading varied among the South Fork's major subbasins. However, autocorrelation (i.e., the dependence of one observation on observations neighboring in time or space) can bias the standard errors of regression coefficients. We checked the grab-sample nitrate data for autocorrelation and found that it was diminished ($p < 0.1$) at sampling lags of 45 days. This corresponded to approximately every fourth of the grab-sample dates. Therefore, we repeated the regressions on four subgroups of samples corresponding to every fourth sampling date, which provided about thirty sampling dates per lag (subgroup). Averages of regression coefficient values among these four groups were within 3% of the values obtained by regression on the bulk data set. But standard errors for those coefficients were approximately doubled when regressions were based on non-autocorrelated (lagged) input data. These larger uncertainty estimates were used to compare regression results among sites.

The second purpose of these regressions was to construct an independent, statistically simulated record of daily NO_3-N concentrations and compare it to the measured daily record. This could provide a benchmark comparison to evaluate SWAT model simulations of NO_3-N transport in future studies. To begin, we compared NO_3-N concentrations in paired samples taken by grab and automated methods on the same date. Then daily concentrations of NO_3-N were estimated based on the linear regressions with natural log (baseflow). Baseflow showed higher correlation with NO_3-N concentration than total discharge, based on exploratory analysis. Schilling and Lutz (2004) showed NO_3-N concentrations were related to baseflow in Iowa's Raccoon River. The resulting regression-simulated NO_3-N records were validated against observed daily data. Comparison statistics were root mean square error, r^2, and the Nash-Sutcliffe modeling efficiency statistic (Nash and Sutcliffe 1970). Loads of NO_3-N were also calculated by multiplying daily discharge volumes by daily-composite-sample concentrations and summing for each year.

Concentrations of NO_3-N, total P, and dissolved P from grab samples were stratified by season and water source (tile and stream waters) and compared among watersheds. We compared only detectable concentrations for NO_3-N, but the nondetectable NO_3-N concentrations account for only 4% of the samples. Analysis of variance was con-

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Table 1

Average annual discharge (Q), runoff and baseflow amounts, and flow-weighted NO_3-N concentrations and loads at four stream gauging stations from 2002 through 2005.

Station	Drainage area (ha)	Precipitation (mm y^{-1})	Q (mm y^{-1})	Runoff (mm y^{-1})	Baseflow (mm y^{-1})	Baseflow fraction	NO_3-N (mg L^{-1})	NO_3-N load (kg ha^{-1} y^{-1})
TC325	19,850	655.3	148.6	50.5	98.1	0.66	20.4	24.1
SF400	25,600	697.3	201.4	74.1	127.4	0.63	15.7	25.8
SF450	58,050	—	167.8	59.4	108.4	0.65	18.2	21.6
BC350	18,200	525.6	167.9	62.4	105.5	0.63	14.2	18.5

Notes: Nitrate data are based on daily composite samples, collected 220 days per year. The total watershed area of 78,000 ha, measured from the Beaver Creek–South Fork confluence, includes 76,250 ha that are upstream from gauge stations and 1,750 ha below the gauge stations.

ducted to remove variance due to sampling date prior to testing for differences among watersheds (type 3 sum of squares). If the watershed effect was significant ($p < 0.05$), then watershed means were compared using Duncan's Multiple Range Test.

Log transforms were made on total P concentrations, and a value of 0.01 mg L^{-1} was assumed for the nondetectable concentrations, which is half of the minimum detectable level. Geometric means were calculated, but the standard deviations cannot be back-transformed without bias (Gilbert 1987) so these were not reported. Annual mean concentrations for the three watersheds were calculated as the grand mean of seasonal geometric means which effectively corrects for the disproportionately lower numbers of samples in autumn and winter.

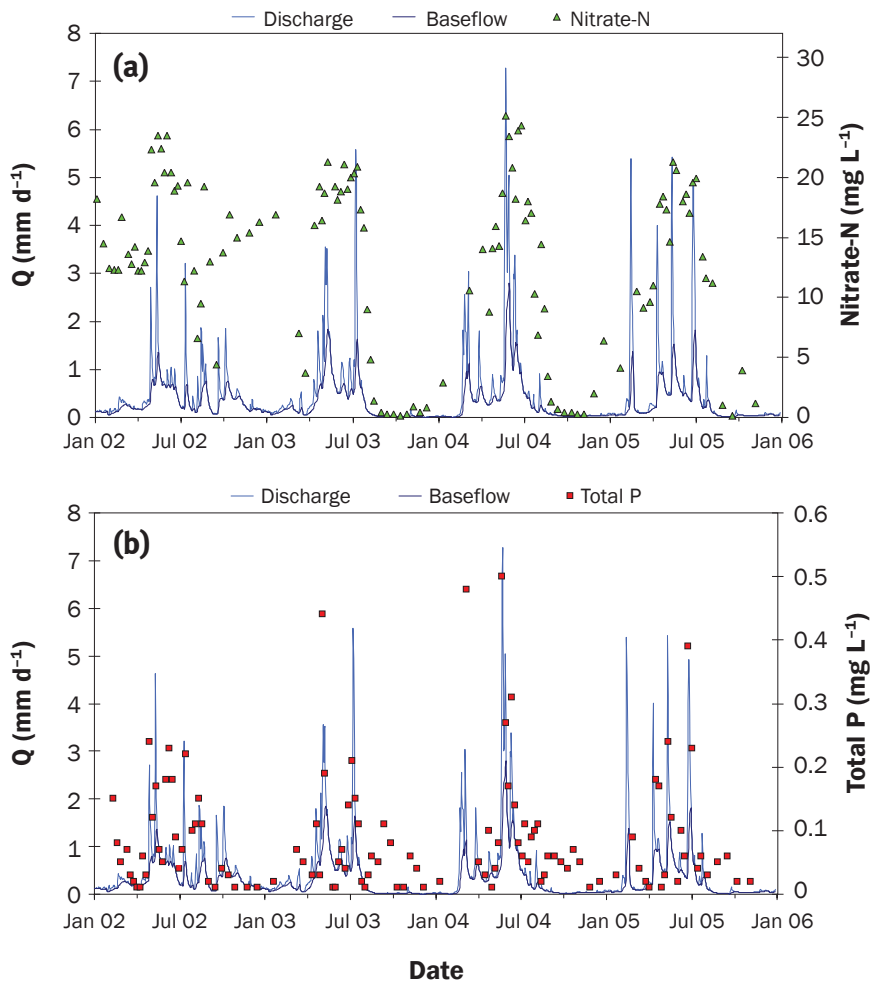
Results and Discussion

Water Quantity. Average stream discharge among the four gauged stations for 2002 to 2005 varied between 149 and 201 mm y^{-1} (5.87 and 7.91 in yr^{-1}) (table 1). This is a fairly wide range but does reflect differences in precipitation patterns that occurred. Several storm events caused increased discharge from the upper South Fork (SF400) watershed but little response from Tipton or Beaver Creeks. At Walnut Creek, a smaller (5,134 ha [12,681 ac]) watershed also located on the Des Moines Lobe region, Tomer et al. (2003) reported annual discharge was 29% of annual precipitation, when averaged across nine years (1992 to 2000). Data from the SF400 and BC350 gauge stations are consistent with this value. However, the TC325 gauge showed a smaller fraction of precipitation as discharge (23%) (see table 1), probably because precipitation at the gauge site underrepresented the watershed's precipitation. Also, the TC325 gauge is located about 1 km (0.62 mi) downstream from Tipton Creek's entrance into the South Fork's alluvial valley, hence the smaller discharge may partly result from stream water losses to ground water. Stream water loss to an alluvial aquifer was also documented at Walnut Creek, near its confluence with the Skunk River but below Walnut Creek's gauging station (Burkart et al. 1999).

Daily stream discharge (SF450) and water quality results from grab samples (figure 2)

Figure 2

Daily stream discharge shown with (a) NO_3-N and (b) total phosphorus concentrations measured in grab samples collected near the outlet of the Iowa River's South Fork (SF450, see figure 1).



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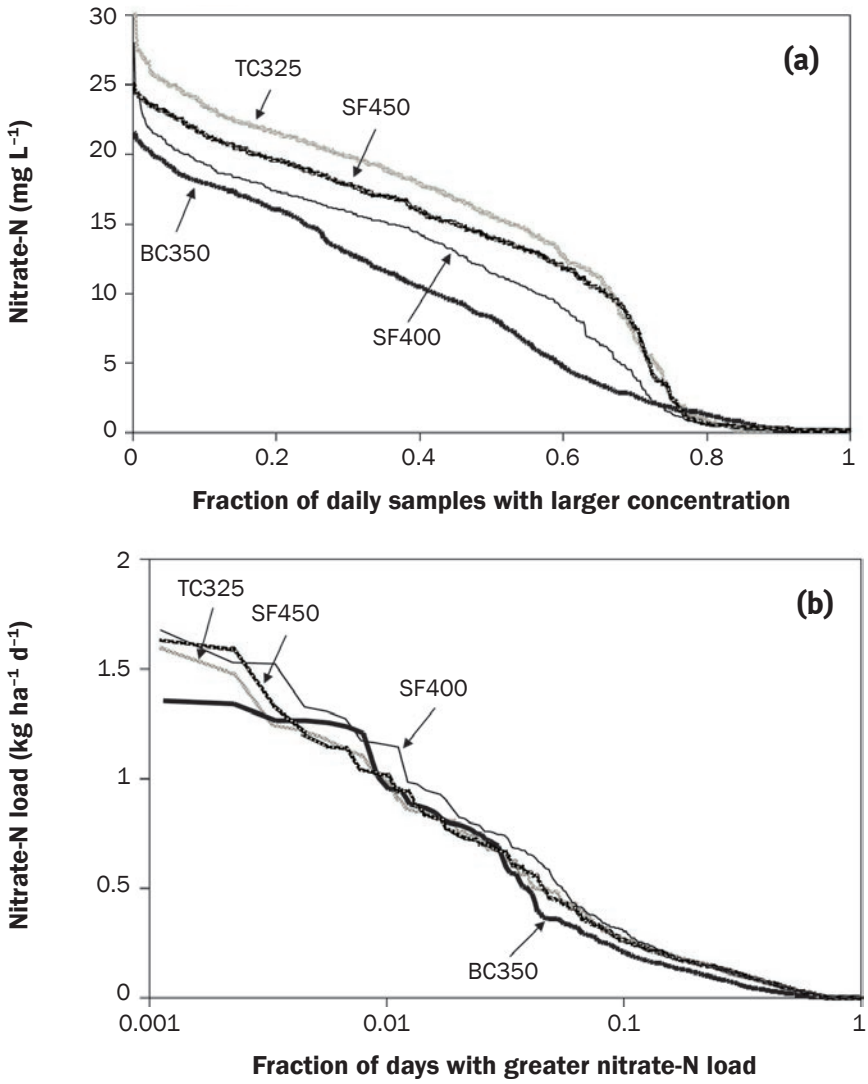
show distinct seasonality, with 70% of the discharge occurring during spring and early summer (i.e., March 22 to July 22). This generally follows timing of seasonal precipitation but is accentuated because little crop water use occurs during much of this time. About two thirds of the discharge is estimated to be baseflow by hydrograph separation. We regard this baseflow to be dominantly tile drainage, with direct ground water flow to the stream as a minor component. Kalita et al. (2006) reported tile drainage ranging from 129 to 200 mm yr^{-1} in central Illinois, while Tomer et al. (2003) reported between 100 and 200 mm of tile drainage in most years in the Walnut Creek watershed. This assumption is consistent with SWAT simulations of the South Fork watershed's hydrology, with the runoff component of our separations including perhaps that 5% to 10% of tile drainage most rapidly discharged to streams following rainfall events.

Nitrate-Nitrogen. Flow-weighted NO_3-N concentrations were greatest from Tipton Creek, which also had the smallest discharge. Consequently the upper South Fork and Tipton Creek basins had similar NO_3-N loads (table 1) (figure 3). The drainage area for SF450 includes the Tipton Creek (TC325) and upper South Fork (SF400) basins combined, plus 12,600 ha (31,000 ac) that includes much of the watershed's alluvial valley and pasture land, which decreases the area-weighted NO_3-N load passing this station (table 1). Beaver Creek had the least average NO_3-N concentration of the four stations (figure 3) and hence a NO_3-N load about 25% less than the other two gauging stations draining similar size areas. Differences during low and intermediate flows were responsible for this. At higher flows, NO_3-N loads exceeded 1 kg ha^{-1} d^{-1} (0.89 lb ac^{-1} day^{-1}) about 1% of the time at all four gauging stations (figure 3). The largest nitrate loads at SF400 and TC325 correspond to the drainage areas containing most of the confined livestock operations within the watershed.

Use of two sampling methods (grab and daily-automated) to obtain NO_3-N concentrations at the four gauging stations provided 359 paired measurements. Comparison of these paired data showed an r^2 of 0.94, root mean square error of 2.7 mg L^{-1}, and modeling efficiency of 0.88. Regression between the paired values showed the automated daily samples averaged about 8% less than the grab samples. Apparently, field-storage of daily samples between dates of sample retrieval occasionally decreased NO_3-N concentrations.

Figure 3

Frequency distributions of (a) NO_3-N concentrations and (b) NO_3-N loads based on daily automated sampling at four gauging stations in the Iowa River's South Fork watershed.



Nitrate-N concentrations in stream waters were strongly dependent on baseflow (table 2) (figure 4), particularly at SF450 and BC350. Regression equations between grab-sample NO_3-N concentrations and baseflow (in essence a surrogate for tile discharge) are similar among the four stations, although the regression slopes at the SF400 and BC350 sites are different at $p = 0.10$. The equations show slopes similar to those found by Schilling and Lutz (2004) for Iowa's Raccoon River. Validation of these regression equations was conducted by comparing regression-estimated daily records to measured daily NO_3-N concentrations obtained through automated sampling. That is, we estimated the daily NO_3-N record using the equations given in table 2, on dates when baseflow exceeded 0.02 mm d^{-1} (0.0008 in day^{-1}) and compared it with the results of the daily composite samples. The predicted records successfully captured the dynamics of the daily record, but not equally well at all four stations (table 2). At the SF450 station, which had the best results (table 2) (figure 5), the standard error of the predicted versus observed was 2.22 mg L^{-1}, with an r^2

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Table 2
Regression of grab sample NO_3-N concentration and baseflow (>0.02 mm d^{-1}).

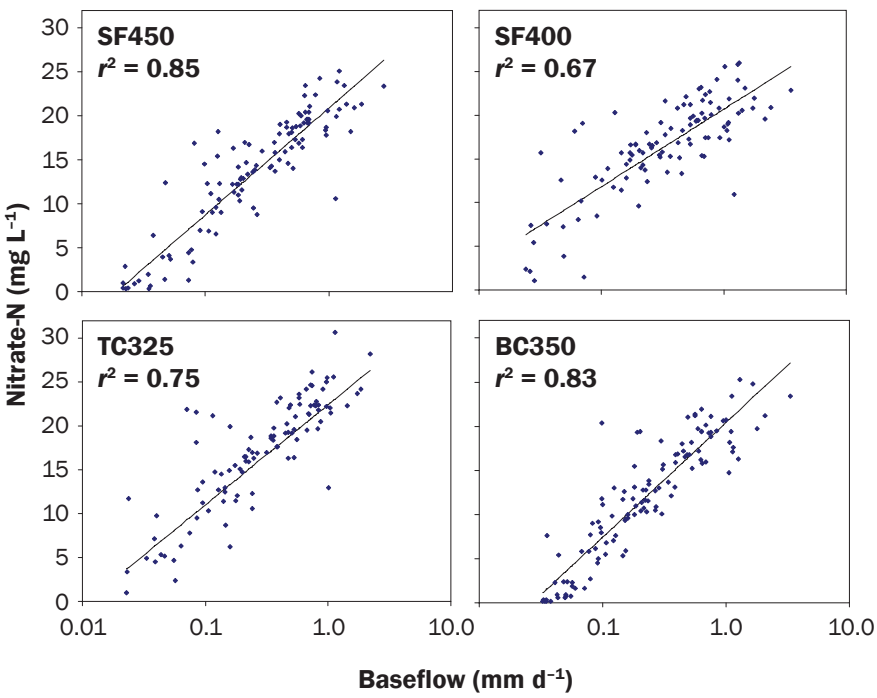
Station	Equation [NO_3-N = a + b ln(BF)]	se*	se†	RMSE	r^2	Comparison with daily samples r^2	RMSE	E
TC325	NO_3-N = 23.43 + 5.08 ln (BF)	1.01	0.61	3.21	0.75	0.89	3.0	0.82
SF400	NO_3-N = 20.71 + 3.85 ln (BF)	0.89	0.56	3.46	0.67	0.75	4.9	0.24
SF450	NO_3-N = 20.89 + 5.28 ln (BF)	0.86	0.48	2.82	0.85	0.92	2.2	0.91
BC350	NO_3-N = 20.44 + 5.65 ln (BF)	0.90	0.48	2.90	0.83	0.87	3.7	0.69

Notes: Standard errors of the coefficients were estimated based on noncorrelated subsets of the data. The comparison with daily samples reports validation statistics between daily NO_3-N records based on regression estimates with independent measured data from automated samplers.
BF = baseflow; se = standard error of the estimate; RSME = residual mean square error; E = modeling efficiency.

* Regression equation intercept.

† Regression equation slope.

Figure 4
Relationships of NO_3-N concentrations with baseflow (>0.02 mm) at four stream gauges in Iowa River's South Fork watershed.



Note: Station locations are given in figure 1, and best-fit equations are given in table 1.

of 0.92 (figure 5b), and the Nash-Sutcliffe model efficiency for the estimates was 0.91, based on dates with baseflow >0.02 mm day^{-1}. The weakest comparison occurred for SF400. This type of information could be applied to evaluating watershed-model simulations of nitrate losses from this watershed, suggesting the level of modeling accuracy at which simulated data essentially provide a replicate realization of the measured nitrate losses. We can also evaluate how model-validation statistics shift as comparisons shift from directly comparing two sampling methods, to comparing a statistically interpolated record with an independent record of measurements. Here, use of a statistical model to interpolate the complete daily nitrate record had minimal impact on the modeling statistics at two of the four stations. Certainly, watershed simulation models cannot logically be expected to outperform a replicate sampling record in terms of validation statistics. We will continue to pursue this idea as an avenue of research.

The baseflow cutoff of 0.02 mm d^{-1} (0.0008 in day^{-1}), below which nondetectable NO_3-N concentrations (<0.3 mg L^{-1}) were modal, may approximate an average discharge rate of low-nitrate groundwater to the stream. If so, then the annual groundwater discharge would be 7.3 mm (0.29 in), which is close to the 9.4 mm y^{-1} (0.37 in yr^{-1}) that Green et al. (2006) estimated to be the groundwater component of stream discharge at SF450 for their validated SWAT model for 2001 to 2004. Nondetectable concentrations of NO_3-N at low flow are associated with minimal tile drain discharge and may be attributable to reducing conditions at depth in aquitards contributing groundwater to stream discharge (Rodvang and Simpkins 2001), and/or biological processing of nitrate in the stream during low flow (Royer et al. 2004; Sobczak et al. 2003).

We stratified the NO_3-N concentration data from all sample sites (figure 1) data by source (stream versus tile waters), season (between equinox and solstice dates), and watershed. Nondetectable concentrations were only observed in stream water during late summer and autumn under low flow conditions but never in tile flows (table 3). The largest concentrations occurred during spring and summer. Tipton Creek showed higher concentrations than the other two subbasins during at least three seasons, and Beaver Creek tended to have the smallest concentrations, which was consistent with the daily data collected at the gauge stations (table 1). Mean concentrations were typically larger in tile waters than in stream waters, supporting the idea that they are the dominant source of NO_3-N contributed to streams.

Phosphorus. Estimates of annual total P loads at the four gauged sites were 0.4 to 0.7 kg ha^{-1} y^{-1} (0.3 to 0.6 lb ac^{-1} yr^{-1}) (data not

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shown). This magnitude of loss is similar to that reported for other Midwestern watersheds (Royer et al. 2004; Algoazany et al. 2007). These losses are of little agronomic importance compared to the ecological effects of P in local waters and reservoirs (Dodds and Welch 2000). Therefore, our subsequent analyses examine phosphorus concentrations rather than loads. Total phosphorus concentrations show a greater response to storm events than nitrate concentrations (figure 2). Total P concentrations were skewed, and autocorrelation was weak. Both of these results are consistent with high P concentrations resulting from transport in runoff rather than baseflow. After evaluating the data from the individual gauging stations, the pooled data show a weak relationship with discharge below 0.9 mm d^{-1} (0.04 in day^{-1}) but are more responsive to higher discharge levels (figure 6).

There were differences between subwatersheds in average annual concentration of grab sample total P; Tipton Creek had a significantly ($p < 0.10$) lower mean concentration of total P (0.057 mg L^{-1}) than did the South Fork (0.062 mg L^{-1}) and Beaver Creek (0.068 mg L^{-1}). Seasonal geometric means of total P in grab samples show lower concentrations in spring and autumn than winter and summer (table 4). The low stream concentrations of P in spring relative to summer are unexpected given that surface runoff, typically the dominant pathway for P loss, is greatest in this season. The majority of the P in tile water is dissolved inorganic (ortho) P, regardless of watershed or season. In South Fork and Tipton Creek, only 25% and 27%, respectively, of the tile samples had a dissolved P/total P ratio of less than 0.9. In contrast, stream water during the 2002 to 2005 period showed average dissolved P/total P ratios of 0.55 in Beaver Creek and 0.68 in Tipton Creek and the South Fork. Dissolved P accounts for greater fractions of total P in streams during the fall and winter. The greater relative amounts of particulate P in spring and summer (lower dissolved P/total P ratios) occur despite the contribution of dissolved P in tile discharge. Presumably this reflects the influx of sediment P from runoff or from stream channels.

Water samples obtained with grab and automated sampling methods had differing P concentrations (figure 6b). Sampling tubes for the automated samplers were set 0.1 to 0.2 m (4 to 8 in) above the stream bottom and pointed downstream to allow sampling during low flow conditions, without disturbing streambed sediments during line purging that is a necessary part of the auto-sampler's program. Grab samples were typically collected at or near the water surface. This difference in sampling depth (typical stream depths were 0.2 to 1.2 m [4 ft]) caused distributions of P concentrations to differ, with larger concentrations produced from the automated samplers. The largest differences occurred within the 0.1 to 1.0 mm d^{-1} (0.004 to 0.04 in day^{-1}) range of flows, but larger concentrations in grab than daily samples on a common date and site almost never occurred. This effect of sampling method still allows for relative comparisons among sampling stations but may be important to interpretation of ecological impairments because concentrations are compared to reference values. For instance, 32% of the grab samples had concentrations exceeding 0.1 mg L^{-1} while 82% of the automatic samples concentrations exceed 0.1 mg L^{-1}. Correll

Figure 5

Validation of NO_3-N prediction by a base flow regression model at SF450 (given in table 2) using results from daily composite samples collected by an automated sampler on dates when baseflow exceeded 0.02 mm d^{-1} (797 dates): (a) time-series comparison; (b) regression-model predictions, based on grab-sample data (table 1), predicted daily composite sample concentrations with an RMSE of 2.22 mg L^{-1}

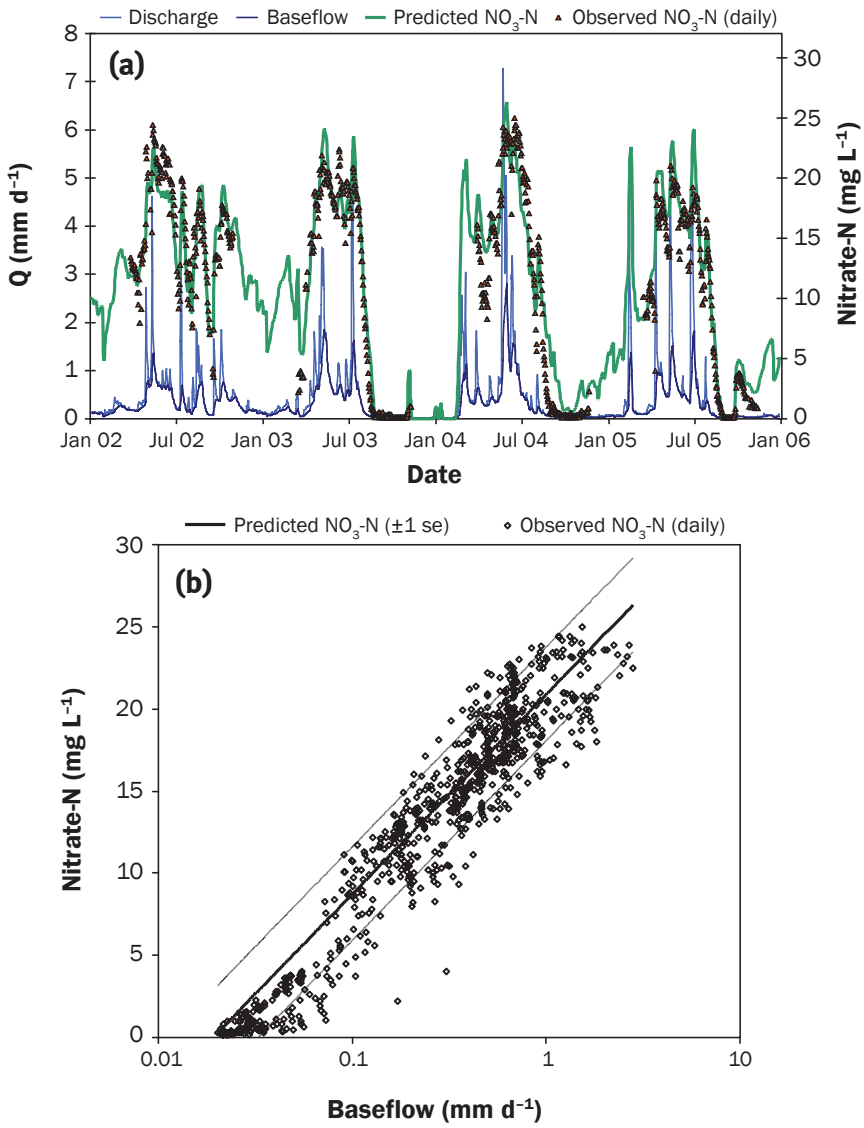

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Table 3
Seasonal effects on NO_3-N concentrations in stream and tile-discharged waters.

Water source	Season	Watershed	Mean NO_3-N (mg L^{-1})	sd	*n*	Nondetects
Stream	Spring	Tipton	23.03a	4.80	278	0
		South Fork	17.94c	4.22	263	0
		Beaver	18.61b	4.17	148	0
	Summer	Tipton	15.67a	8.21	209	9
		South Fork	13.15b	6.27	212	9
		Beaver	12.91b	6.72	120	5
	Autumn	Tipton	9.46a	8.34	80	17
		South Fork	8.30ab	6.75	78	21
		Beaver	7.07b	5.82	48	7
	Winter	Tipton	13.86a	7.15	40	0
		South Fork	11.12b	5.25	53	0
		Beaver	10.38b	4.27	28	0
Tiles	Spring	Tipton	22.08b	3.61	88	0
		South Fork	23.63a	6.03	54	0
	Summer	Tipton	21.75	5.42	88	0
		South Fork	23.12	5.51	63	0
	Autumn	Tipton	12.72b	5.97	35	0
		South Fork	16.07a	4.97	22	0
	Winter	Tipton	14.32	3.53	9	0
		South Fork	15.96	4.10	5	0

Notes: Mean and standard deviations (sd) exclude nondetectable concentrations (<0.3 mg L^{-1}). Significance groupings (a, b, c), based on t-tests, are given where ANOVA indicated a significant watershed effect after variation among sampling dates was removed (type 3 sum of squares).

(1998) suggested that a total P concentration of 0.1 mg L^{-1} would lead to eutrophication in fresh water bodies, while Dodds and Welch (2000) suggested a 0.06 mg L^{-1} threshhold.

Escherichia coli. The South Fork, Tipton Creek, and Beaver Creek all show substantial populations of *E. coli* (table 5) (figure 7). Significant differences ($p < 0.05$) in geometric mean *E. coli* populations were found to be attributable to season and watershed by both ANOVA and nonparametric Wilcoxon rank sum–Kruskal Wallis tests. All watersheds had the greatest stream water populations in summer (June 21 to September 21) and the least in winter (December 21 to March 19). The data in table 5 and figure 7 include nondiluted samples where the culture tray was saturated resulting in a mpn of >2,419 cells 100 ml^{-1} (3.4 fl oz). These data were included in the analysis, but they underestimate the population in those samples to an unknown extent. Removing these data has the effect of underestimating *E. coli* mean populations because mpn estimates of >2,419 exceed the seasonal and annual means. The seasonal trends shown in table 5 are based on the cumulative data set from July 2001 through November 2005. Similar trends are seen in data from individual years (data not shown). The geometric means in table 5 can be compared to the class A2 water standard for *E. coli* of 630 cells 100 ml^{-1} (Iowa DNR 2007). Tile drainage water *E. coli* populations were substantially lower than those in stream water, averaging 13 *E. coli* 100 ml^{-1} in two south Fork tiles and 30 *E. coli* 100 ml^{-1} in the Tipton Creek tile waters.

Tipton Creek has a greater percentage of stream water samples with populations of 100 cells 100 ml^{-1} (3.4 fl oz) or less and fewer samples with populations exceeding 1,000 cells 100 ml^{-1} (figure 7). This is consistent with the smaller mean *E coli* populations in Tipton Creek. The opposite trend was found for Beaver Creek, which had greater frequency of samples exceeding 1,000 cells 100 ml^{-1}. The maximum single sample concentration of *E. coli* in Beaver Creek was 15,523 cells 100 ml^{-1}, 6,486 in South Fork, and 24,154 in Tipton Creek. In contrast, the tile discharge waters (Tipton and South Fork combined) had 73 % of samples below 100 *E. coli* 100 ml^{-1} and only 6% of samples exceeding 1,000 *E. coli* 100 ml^{-1}.

Stream water populations of *E. coli* are dynamic and dependent upon inputs, survival, and transport (Jamieson et al. 2004). The following regression equation explains about half of the variation ($r^2 = 0.497$, $p < 0.01$) in *E. coli* populations (cells 100 ml^{-1}) at the stations where stream flow data were measured:

$$\text{Log}_{10}(E.\ coli) = (0.052\ \text{AT}) + (0.059\ \text{RQ}) + 1.767, \quad (1)$$

where AT is air temperature (°C) and RQ is daily average discharge ($m^3\ s^{-1}$) attributed to runoff obtained from hydrograph separations. Air temperature is a surrogate for water temperature and describes the seasonal effects on *E. coli*. Runoff discharge likely reflects overland flow contributions and possible suspension of stream bed sediment into the water column at greater discharge, as viable *E. coli* populations exist in the bed sediment. A preliminary sampling of stream sediments (surface 2 cm [0.8 in]) showed *E. coli* to be present in populations from 29 to 1,181 cells g^{-1} of sediment. These populations may also serve as a source during periods of low stream flow, but this has not been reported.

We estimated swine populations in the different subwatersheds to be 75,379 for Beaver Creek, 301,628 for South Fork (excluding Tipton Creek), and 224,186 for Tipton Creek (Tomer et al. 2008). These resulted in swine densities of 4.14, 7.9, and 11.29 swine ha^{-1} (1.7, 3.2, and 4.6 swine ac^{-1}) for Beaver Creek, South Fork, and Tipton Creek, respectively. Using similar geographic information system–based methods, we estimate that 66% of the cropped land would receive manure. If swine manure applications were the principal source of *E. coli*, then lower populations of *E. coli* would be expected in Beaver Creek than in the other subwatersheds. Furthermore, greater stream water

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E. coli populations would be expected in spring and fall when swine manure is applied rather than in summer when the greatest populations are found. Other potential sources of *E. coli* include cattle, humans, and wildlife, and are apparently important as well. Based on observation, riparian areas have the greater concentrations of wildlife and grazing cattle compared to upland areas. We intend to undertake bacterial source tracking in future research in this watershed.

Summary and Conclusions

Water quality in the South Fork of the Iowa River and its tributaries shows several features that appear to be common to tile-drained agricultural landscapes of the midwestern Corn Belt. Nitrate concentrations in stream water display peaks in spring and summer that correspond to the times when tile drainage is most important. Quantifying tile drainage in watersheds of this size is not yet feasible due to cost and accessibility, but the extent of tile drainage and agriculture strongly indicate that the NO_3-N exported is agricultural in origin. Nitrate export from these streams was greater than 20 kg ha^{-1} y^{-1} (18 lb ac^{-1} yr^{-1}) during 2002 to 2005 and is likely related to the formation and leaching of nitrate prior to the onset of crop demand for N in June (Dinnes et al. 2002). These losses of NO_3-N likely occur through the tile drainage system which means that conservation practices such as riparian buffers are not likely to have substantial impact. Observed NO_3-N concentrations were strongly related to baseflow, essentially a surrogate expression of subsurface tile flow contributions to discharge. By monitoring of NO_3-N concentrations using both grab and automated methods, we can statistically compare replicate records of water quality to suggest benchmark targets for modeling efforts in the watershed. However, P concentrations obtained from the two methods did not compare well, probably due to differences in sampling depth in the water column. Relationships of P concentration to hydrologic discharge were observed but weaker for P than for nitrate.

Total P and *E. coli* also showed seasonal trends, but these peak concentrations occurred later than those observed with NO_3-N. Both of these contaminants are transported in runoff. Subsurface tile drainage contributes primarily dissolved P, although surface inlets allow some overland flow (runoff) to enter the subsurface drainage network. Variations in the ratio of dissolved P/total P suggest that runoff is important to total P transport in summer months. The major sources of P are swine manure and fertilizer applied to soil, and sediments from stream channels, but estimating the relative magnitude of these sources is not currently possible. Riparian buffers and grassed waterways are likely to be effective in mitigating P losses in runoff from crop land (Dillaha et al. 1989; Lee et al. 2003) but will not mitigate P losses in tile drainage. Concentrations of total P are strongly affected by sampling method, but regardless of the method used, impacts on stream ecology are indicated, given that concentrations exceeding 0.1 mg L^{-1} are commonly observed by both methods.

The *E. coli* data show that these waterways are at least seasonally impaired for recreational use. The sources of *E. coli* include animal manures, wildlife and human inputs. A combination of temperature and run-

Figure 6

(a) Relationship of total phosphorus concentrations measured in grab samples taken at the four gauging stations and daily discharge rate. (b) Phosphorus concentrations increased during high flows, but results were strongly influenced by sampling method.

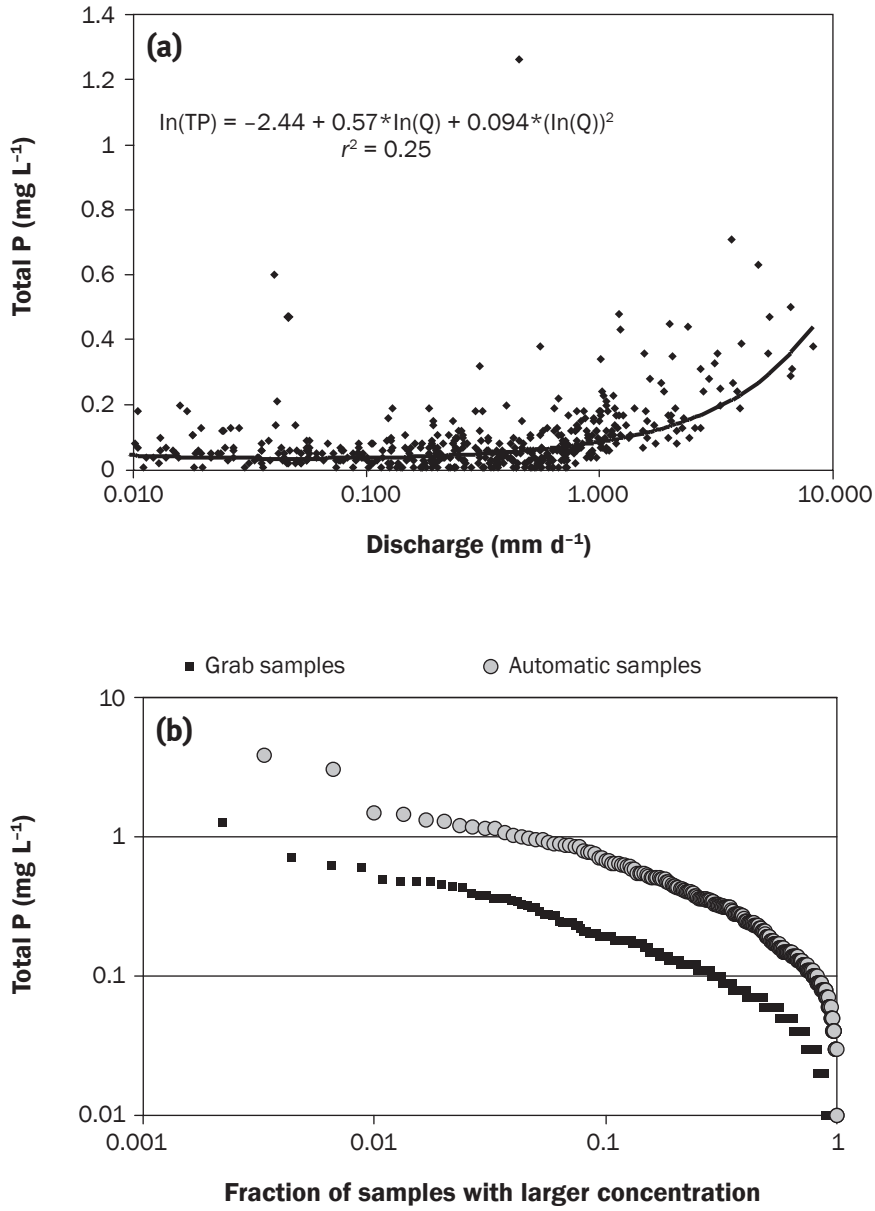

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Table 4
Seasonal effects on mean total phosphorus concentrations and ratios of dissolved phosphorus–to–total phosphorus in stream and tile-discharge waters.

Water source	Season	Watershed	Mean total P (mg L^{-1})	DP/TP ratio	Detects	Nondetects
Stream	Spring	Tipton	0.043	0.64	216	62
		South Fork	0.052	0.68	220	43
		Beaver	0.050	0.61	125	23
	Summer	Tipton	0.068	0.68	201	17
		South Fork	0.092	0.64	219	2
		Beaver	0.080	0.50	125	1
	Autumn	Tipton	0.041	0.82	69	26
		South Fork	0.040	0.82	79	20
		Beaver	0.054	0.66	51	4
	Winter	Tipton	0.085	1.00	34	4
		South Fork	0.080	0.71	46	2
		Beaver	0.086	0.42	25	0
Tiles	Spring	Tipton	0.056	1.00	87	1
		South Fork	0.048	1.00	54	0
	Summer	Tipton	0.060	0.85	74	12
		South Fork	0.040	0.90	59	4
	Autumn	Tipton	0.048	0.91	24	10
		South Fork	0.029	0.97	14	8
	Winter	Tipton	0.084	1.00	8	1
		South Fork	0.131	1.00	5	0

Notes: Mean total phosphorus values shown are back-transformed geometric means (grab samples) for total phosphorus and arithmetic means for the ratios. DP = dissolved phosphorus. TP = total phosphorus. Numbers of detects and nondetects are for the total phosphorus estimation of the mean. For nondetectable concentrations, a total P concentration of 0.1 was used in the calculations. For the DP/TP ratio, nondetects were omitted from the calculation.

Table 5
Seasonal and annual mean populations of *E. coli* in stream water from 2002 through 2005.

	E. coli (cells 100 ml^{-1})*		
Season	Beaver Creek	South Fork	Tipton Creek
Spring	232a	201a	104b
Summer	1047a	649b	500b
Autumn	208a	139a	87b
Winter	21a	19a	14a
Annual	182	136	90

* Geometric means were back-transformed after statistical analysis. The annual mean was calculated as a grand mean of equally weighted seasonal geometric means. Means on the same row followed by different letters are significantly different (p = 0.05).

off explained about 50% of the variation in *E. coli* in stream water, but the elevated concentrations of *E. coli* in late summer do not coincide with runoff from manured lands which would occur primarily in fall and to a lesser extent in spring. Similarly the distribution of *E. coli* among the three catchments was not related to estimated numbers of swine. While transport of *E. coli* from manured lands does occur, a better understanding of the different sources is needed to assess effectiveness of conservation practices. The unique dynamics of the three contaminants considered highlight the complexity of integrated water quality assessments in large watersheds.

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Disclaimer

Mention of trade names is for information only does not constitute an endorsement by USDA.

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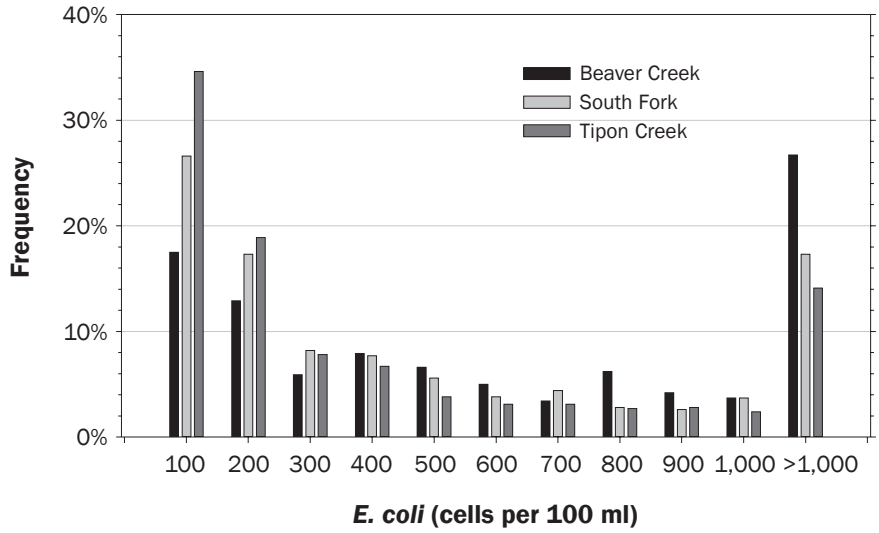
Figure 7

Frequency distribution of *E. coli* populations in the South Fork of the Iowa River and two principal tributaries for measurement dates 2002 through 2005.



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