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# Unrestricted cattle access to streams and water quality in till landscape of the Midwest

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ABSTRACT

Unrestricted cattle access to streams in traditionally pastoral regions has been linked to increased concentrations of bacteria, suspended sediments and associated contaminants in streams. However, there is a dearth of data available regarding the impact of cattle access to streams in poorly drained landscapes of the Midwest. In this study, we investigate changes in water quality on a 1005 m long stream section impacted by cattle grazing on the upper 130 m. We monitor discharge, water quality [nitrate, ammonium, total Kjeldahl nitrogen (TKN), total phosphorus (TP), total suspended sediments (TSS), turbidity, *Escherichia coli*] and chloride, atrazine, silica and major cation concentrations over a 12-month period. Cattle access to the stream does not significantly affect nitrate concentration but leads to large increases in TKN (fourfold increase), TP (fivefold increase), ammonium (fourfold increase), TSS (11-fold increase), turbidity (13-fold increase) and *E. coli* (36-fold increase) in the summer/fall period. In particular, *E. coli* concentration in the stream in the summer/fall period exceeds 235 colony forming unit (CFU)/100 ml 64% of the time upstream from the section impacted by cattle, but exceeds this same threshold 89% of the time immediately downstream. Despite the negative impact of cattle access to the stream on water quality, data indicate that dilution, in-stream processes, and natural stream geometry downstream from the impacted section help mitigate this pollution. We expect that this study will be an incentive for policy makers to promote stream rehabilitation and develop more stringent guidelines limiting cattle access to streams in many Midwestern states and other regions with poorly drained soils where the impact of cattle access to streams on water quality is often ignored.



## 1. Introduction

Land-use can strongly impact water quality with respect to suspended sediments, phosphorus, nitrogen and other nutrients/contaminants (Sharpley et al., 1992; Cooke and Prepas, 1998; Kuhnle et al., 2000; Vanni et al., 2001). For instance, high nitrate concentration in streams has been linked to agricultural land-use and the widespread application of fertilizers and manure to agricultural fields (Hill, 1978; Rock and Mayer, 2006; Royer et al., 2006; Tarkalson et al., 2006). Stream bank deterioration has also been linked to high phosphorus sediment losses and poor overall water quality (Sekely et al., 2002). In particular, streams impacted by cattle grazing have been shown to exhibit poorer water quality than streams

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where cattle access is restricted (Nagels et al., 2002; Line, 2003; Muenz et al., 2006). In a study of two buffered and three unbuffered streams impacted by cattle grazing, Muenz et al. (2006) indicate higher levels of nitrate, suspended solids and fecal coliforms in unbuffered streams than buffered streams. McKergrow et al. (2003) studied the impact of preventing cattle access to the stream along a 1.7 km long riparian zone on sediment exports in a catchment in Australia, and showed that stream fencing and riparian zone conservation led to a 90% reduction in sediment exports at the watershed scale. Collins (2004) also reported high levels of fecal contamination in wetlands affected by pastoral land-use. Contamination was attributed to both the transport of fecal coliforms to streams during precipitation events and direct excretion of fecal material in the wetland by cattle.

Literature clearly illustrates that unrestricted cattle access to streams has a negative impact on water quality. Nevertheless, most studies on the impact of cattle intrusion in or near streams take place in predominantly pastoral regions such as New Zealand or Australia (Nagels et al., 2002; McKergrow et al., 2003; Collins, 2004) and some areas of the U.S. where cattle operations are common such as Pennsylvania (Guiluano and Homyack, 2004), Virginia (Sheffield et al., 1997; Inamdar et al., 2002), Vermont and New York (Meals, 2001). There is a dearth of data on the impact of cattle access to streams in regions dominated by row crop agriculture like the U.S. Midwest. Unrestricted cattle access to streams nevertheless happens in these regions where poorly drained soils dominate since row crop agriculture is not the only regional land-use practice. For instance, 37% of agricultural products sold out of the state of Indiana are livestock products, including products derived from cattle operations (Farmland Information Center, 2006). Understanding the impact of unrestricted cattle access to streams on water quality in the Midwest is therefore necessary to better manage water quality in this region of the country that has been identified as a major source of contaminants/nutrients to the Gulf of Mexico (Goolsby, 2000; Royer et al., 2006). Gathering empirical evidence of the impact of cattle access to streams on water quality in poorly drained soils of the Midwest is also critical in order to promote the development of stronger environmental laws for stream protection in many Midwestern states, as well as to better characterize the impact of cattle access to streams in other landscapes with poorly drained soils.

In this study, we monitor changes in water quality with respect to nitrate, ammonium, total Kjeldahl nitrogen (TKN), total phosphorus (TP), total suspended sediments (TSS), turbidity, atrazine and *Escherichia coli* on a stream section impacted by cattle grazing. Water quality is monitored immediately upstream and downstream from the impacted stream section, as well as 875 m further in the downstream direction. Atrazine is typically not associated with pastoral land-use but widely used on corn in the Midwest (Kladivko et al., 1999). In this study, atrazine is monitored to allow for the differentiation between row crop derived nutrients and cattle derived nutrients in the spring, during which most nutrient losses from agricultural land occur (Royer et al., 2006). *E. coli* concentration has been used as an indicator of fecal contamination in many studies and is therefore used as such in this study (Nagels et al., 2002; Collins and Rutherford, 2004). Chloride, silica, calcium, magnesium, and sodium concentrations are measured in the stream in order to identify potential dilution effects in the downstream direction. The impact of cattle access to the stream on nitrate, ammonium, TKN, TP, TSS, turbidity and *E. coli* concentrations in the stream is quantified by season and streamflow conditions. The implications of our findings for watershed management are discussed.

## 2. Study site

The stream section studied is a 1005 m long stream section of second-order Fishback Creek, located in Eagle Creek Watershed (ECW) in the Tipton Till Plain near Indianapolis, Indiana (Fig. 1). Indiana has a temperate continental and humid climate. The average annual temperature for central Indiana is 11.7 °C with an average January temperature for Eagle Creek watershed of −3.0 °C and an average July temperature of 23.7 °C. The long-term average annual precipitation (1971–2000) in the watershed is 105 cm and was obtained from NOAA (NOAA, 2005), while daily precipitation during the study period was measured at Eagle Creek Airpark (Fig. 1). Soils in ECW are characterized by a 30 cm thick A horizon and an E horizon, unless the soil has been ploughed extensively, and generally belong to the Crosby-Treaty-Miami association. The B horizon is typically higher in clay than overlying and underlying horizons (Hall, 1999). Most central Indiana soils are poorly drained and require artificial drainage in the nearly flat till plain.

Stream monitoring stations have been installed at three locations: one upstream from the section of the stream impacted by cattle grazing (station S1), one immediately downstream (station S2), and one 875 m downstream from station S2 (station S3) (Fig. 1). For this study, land-use was determined using 2003 National Resources Conservation Service 1 $m^2$ aerial photography. Contributing areas to each stream monitoring station were determined using ArcGIS surface hydrology tools and 30 m digital elevation model (DEM) data provided by the U.S. Geological Survey (USGS). The upstream contributing area to station S1 is 13.7 $km^2$, and land-use is dominated by row crop agriculture with 87% of land used for agriculture (mainly corn–soy rotation), 3.5% forest and 3.2% low-density residential land-use. The remaining 6.3% of land is mainly composed of herbaceous vegetation. The stream section between S1 and S2 (130 m) is bordered by a pasture in which 25 cows have free access to the stream all year long. The movement of cows in the 4-ha pasture and in and out of the stream was not directly monitored, so the frequency at which cows visited the stream was not determined; however, field observation indicates that cows are rarely near the stream during winter/spring months, whereas they are commonly observed near or in the stream in summer months. On the section impacted by cattle, near stream vegetation is lacking and stream bank erosion owing to cattle activity near the stream is obvious. No buffer of any type is therefore present between the pasture and the stream on the section impacted by cattle. Stream contamination from cattle activity on this section is therefore the result of direct excretion of fecal material in or near the stream by the cows, as well as the transport of contaminants from the pasture to the stream, especially during storm events. An intermittent

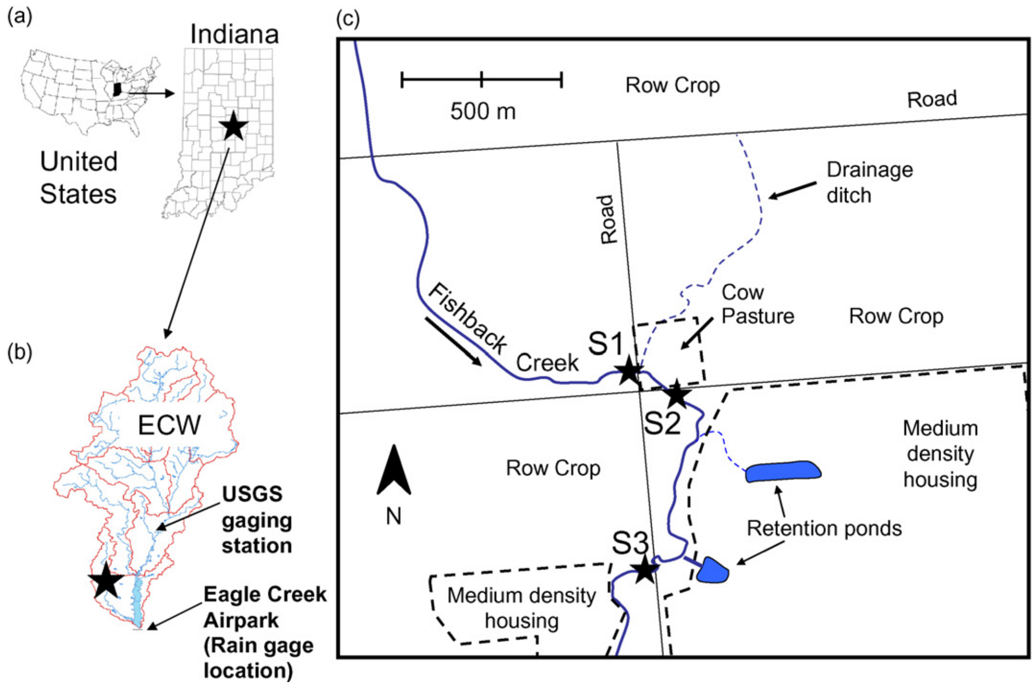


**Fig. 1 – Experimental site description. (a) State of Indiana; (b) Eagle Creek Watershed (ECW) and location of experimental site in ECW; (c) Land-use and location of stream monitoring stations S1, S2 and S3 on Fishback Creek.**

drainage ditch draining a field cropped with corn in 2005 drains into the stream between S1 and S2. Land-use between stations S2 and S3 (875 m stream length) is dominated by row crop agriculture on the right bank and medium density housing on the left bank. Two retention ponds collecting storm runoff from a residential area drain into the stream between S2 and S3 (Fig. 1). Stream geometry is typical of the region upstream from S1, with a deep artificially incised stream channel (approximately 3 m deep) from which vegetation is periodically removed. However, between S2 and S3, stream geometry progressively shifts to a wide (6–10 m wide) and shallow stream channel (0.5–1 m deep), with pools and riffles and a large forested riparian zone bordering the stream on both sides.

## 3. Methods

Stream discharge at station S1 was estimated based on mean daily discharge measurements made at the USGS Zionsville stream gaging station (Fig. 1) following the general equation:

$$Q_{S1} = \left[\frac{A_{S1}}{A_Z}\right] Q_Z \qquad (1)$$

where $Q_{S1}$ is the discharge at S1 ($m^3/s$), $Q_Z$ the discharge measured at the USGS Zionsville stream gaging station ($m^3/s$), $A_Z$ the contributing area upstream from Zionsville monitoring station ($km^2$) and $A_{S1}$ the contributing area upstream from stream monitoring station S1 ($km^2$) (USGS, 2005). This equation was used because discharge typically scales linearly or nearly linearly with contributing area (Dunne and Leopold, 1978; Pazzaglia et al., 1998). Instantaneous discharge was also measured in the field in 2005–2006 to check for accuracy of estimated discharge and to compare instantaneous discharge measurements between stations using a Doppler velocity meter (SONTEK Flow Tracker). However, due to equipment failure, instantaneous discharge was only measured for 15 of the 27 sampling dates. Experimental error on instantaneous discharge measurements using the Doppler velocity meter was determined in the field to be 25–30% (unpublished data). High flow or event flow at S1 ($Q_{75}$) is defined as the discharge exceeded 25% of the time and was estimated based on long-term discharge measurements at the USGS Zionsville stream gaging station.

Water samples were collected on a biweekly to monthly basis between March 2005 and March 2006 (27 sampling dates) (Fig. 2). A total of 10 sampling dates correspond to high flow conditions ($Q > Q_{75}$). Field blanks and triplicate analysis of selected samples were performed for quality control/quality assurance and samples were kept on ice after sampling until return to the laboratory. *E. coli* concentration was measured within a few hours of collection and samples for total phosphorus (TP) and total Kjeldahl nitrogen (TKN) analyses were collected in pre-acidified containers to maintain the pH < 2 until analysis. All other samples were filtered using disposable GF/F filters within 36 h of sampling and frozen until analysis. Nitrate/nitrite, ammonium, chloride and silica were measured according to standard colorimetric methods (Clesceri et al., 1998) using a photometric analyzer (Aquachem 20-EST Analytical). Calcium, magnesium and sodium were measured by ion chromatography using a Dionex DX500 Ion Chromatograph equipped with a CS15 analytical column and methasulfonic acid eluent. TKN was measured using the standard Kjeldahl method (EPA method 351.4). TP was determined using Environmental Protection Agency (EPA) standard method 4500 PE consisting of a strong acid and

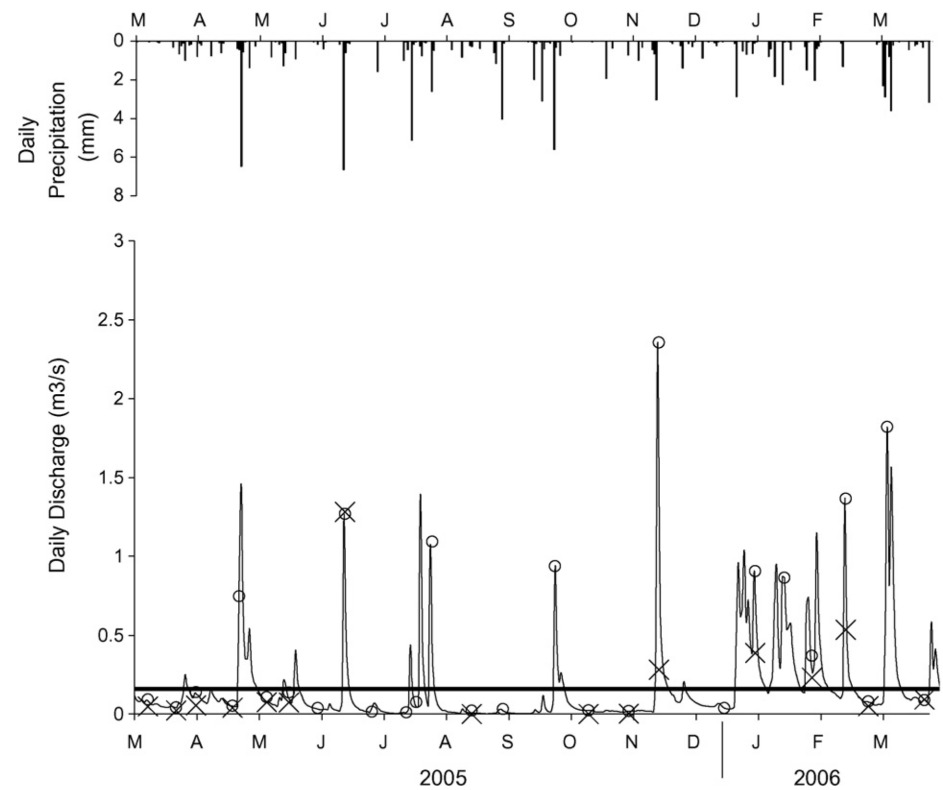


**Fig. 2 – Daily precipitation (mm) and average daily discharge at stream monitoring station S1 ($m^3$/s) between March 2005 and March 2006. Clear dots indicate sampling dates for all stations and average daily discharge at the time of sampling at S1. × indicates the measured discharge at S1 at the time of sampling. The horizontal solid line indicates the 75th discharge percentile (high flow) at stream monitoring station S1.**

persulfate digestion analyzed colorimetrically using the ascorbic acid–molybdate blue method. *E. coli* concentration [most probable number (MPN) of colony forming units (CFU) per 100 ml] was measured, according to the *E. coli* Test, with EC-MUG Medium and read using a fluorometer (long-wavelength UV) (standard method SM9221-F). The most probable number of colony forming units is used as an estimate of CFU. *E. coli* concentration is therefore reported as CFU/100 ml hereafter. Atrazine levels were determined by immunoassay using the Beacon Analytical Immunoassay. Turbidity was measured using a turbidimeter (Hach 2100N) and total suspended sediment concentration (TSS) was determined by weighing oven dried (65 °C) sediments collected on pre-washed Whatman GF/F Fiber Glass filters (0.7 μm pore size). Student's *t* tests were used to determine significant differences in water quality between stations. Throughout this manuscript, seasons are based on the calendar year. The summer/fall period therefore starts on June 21 and terminates on December 20.

## 4. Results

### 4.1. Stream hydrology

Fig. 2 shows daily precipitation (mm), daily discharge at station S1 ($m^3$/s), the date at which samples were taken for all stations between March 2005 and March 2006 and the measured discharge at S1 for each sampling date. Total precipitation between 1 April 2005 and 31 March 2006 was 1000 mm or 4.8% drier than the 30-year normal. The average daily discharge at S1 over the duration of the study was 0.17 $m^3$/s with a maximum discharge of 2.35 $m^3$/s and a minimum discharge of 0.003 $m^3$/s. Average discharge during the summer/fall period was 0.087 and 0.27 $m^3$/s during the winter/spring period. The 75th discharge percentile (high flow) at station S1 was calculated to be 0.16 $m^3$/s and is shown on Fig. 2.

Instantaneous discharge measurements (Fig. 3) at S1, S2 and S3 indicate that except during the summer, there are generally no significant differences (variation within 30% measurement error) in discharge between stations. At low flow in the summer, discharge is nevertheless typically higher at S3 than at S2 or S1. Despite the lack of significant differences between stations, the average measured discharge at S3 is 28% higher ($n = 13$) than average measured discharge at S1 during the winter/spring period and 107% higher ($n = 2$) in the summer/fall period for an increase in contributing area of 8.9% between S1 and S3.

Chloride, calcium, magnesium, sodium, and silica concentrations for the duration of the study (including events), events only, the summer/fall period (including events), summer/fall events only, the winter/spring period (including events) and winter/spring events only are shown in Table 1. *T*-tests show that over a 12-month period, average concentra-

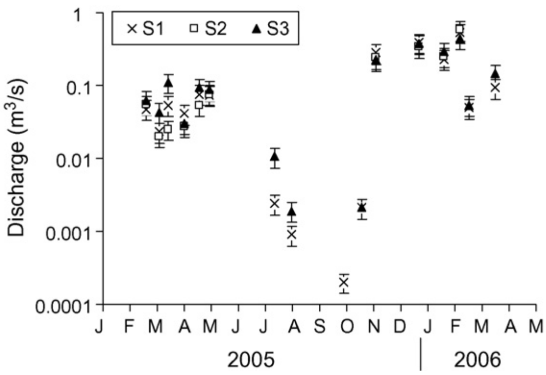


**Fig. 3 – Instantaneous discharge measurements made in the field between March 2005 and March 2006 at stream monitoring stations S1, S2 and S3. Error bars correspond to a 30% measurement error as determined in the field.**

tions do not change significantly between stations ($p < 0.001$). Similarly, no significant variations ($p < 0.001$) in chloride, calcium, magnesium, sodium and silica concentrations are observed between stations during the winter/spring period, including winter/spring events. During the summer/fall period, chloride, magnesium, sodium and silica are also similar ($p < 0.001$) at S1 and S2. Nevertheless, chloride, magnesium, sodium and silica concentrations decrease by 26.7, 17.3, 39.0 and 41.9%, respectively, between S1/S2 and S3 during the summer/fall period. However, this trend is not observed for Ca, as no significant changes ($p < 0.001$) in Ca concentrations are observed between the three stations in summer/fall.

### 4.2. *Water quality*

Variations in turbidity and concentrations of nitrate, ammonium, TKN, TP, TSS, *E. coli* and atrazine at stations S1, S2, and S3 between March 2005 and March 2006 are presented in Fig. 4. Table 2 summarizes average values for each of these variables by season. Average nitrate concentrations between S1 and S2 over a 12-month period increase by 29.6%; however, nitrate is only significantly higher ($p < 0.001$) in S1 than S2 between March and June. Ammonium concentrations are generally higher in S2 than S1, especially during the summer/fall period. Overall, over a 12-month period, ammonium concentrations increase by 64.7% between S1 and S2. TKN and TP are also significantly higher in S2 than S1 ($p < 0.001$) with average increases in concentration between S1 and S2 of 175 and 192%, respectively. As shown on Fig. 4, TSS and turbidity are also much higher in S2 than S1, especially between April and November, with overall increases in concentration over a 12-month period between S1 and S2 of 300 and 283%, respectively. Differences in *E. coli* concentrations between S1 and S2 are mainly seasonal with *E. coli* increases between S1 and S2 occurring mainly during the summer/fall period. Over a 12-month period, *E. coli* concentrations increase, on average, by 2380% between S1 and S2 (from 1296 to 32,188 CFU/100 ml). Average Atrazine concentrations are 41.1% higher in S2 than S1 over a 12-month period. Overall, data indicate that turbidity and nitrate, ammonium, TKN, TP, TSS, *E. coli* and atrazine

**Table 1 – Average concentrations for chloride ($Cl^-$), calcium ($Ca^{2+}$), magnesium ($Mg^{2+}$), sodium ($Na^+$) and silica ($Si^+$) for a 12-month period (including events), events only, the winter/spring period (including events), winter/spring events only, the summer/fall period (including events) and summer/fall events only for stream monitoring stations S1, S2 and S3 between March 2005 and March 2006**

| Time Period | $Cl^-$ (mg/l) | | | $Ca^{2+}$ (mg/l) | | | $Mg^{2+}$ (mg/l) | | | $Na^+$ (mg/l) | | | $Si^+$ (mg/l) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | S1 | S2 | S3 | S1 | S2 | S3 | S1 | S2 | S3 | S1 | S2 | S3 | S1 | S2 | S3 |
| 12 months ($n = 27$) | 36.10 | 34.94 | 32.29 | 38.10 | 38.04 | 39.45 | 22.19 | 22.25 | 20.23 | 12.55 | 11.71 | 9.14 | 2.94 | 3.30 | 2.41 |
| Events ($n = 10$) | 26.76 | 28.25 | 25.38 | 37.17 | 36.08 | 36.83 | 16.96 | 16.99 | 16.09 | 7.51 | 6.75 | 6.12 | 4.33 | 4.53 | 4.29 |
| Winter/spring ($n = 16$) | 33.90 | 34.04 | 34.45 | 43.58 | 42.81 | 44.67 | 21.70 | 21.69 | 21.04 | 8.67 | 8.21 | 7.88 | 2.43 | 2.83 | 2.57 |
| Winter/spring events ($n = 7$) | 23.43 | 26.96 | 25.61 | 41.83 | 39.07 | 41.93 | 16.59 | 16.12 | 16.31 | 6.38 | 5.12 | 5.21 | 4.38 | 4.52 | 4.91 |
| Summer/fall ($n = 11$) | **39.52** | **36.34** | **28.96** | **29.64** | **30.66** | **31.38** | **22.95** | **23.11** | **18.98** | **18.19** | **17.12** | **11.09** | **3.72** | **4.02** | **2.16** |
| Summer/fall events ($n = 3$) | 33.41 | 30.84 | 24.92 | 27.86 | 30.10 | 26.63 | 17.71 | 18.73 | 15.64 | 9.39 | 10.01 | 7.94 | 4.21 | 4.56 | 3.06 |

Bold text indicates the period when the largest changes for each parameter (%) are observed between S1/S2 and S3.

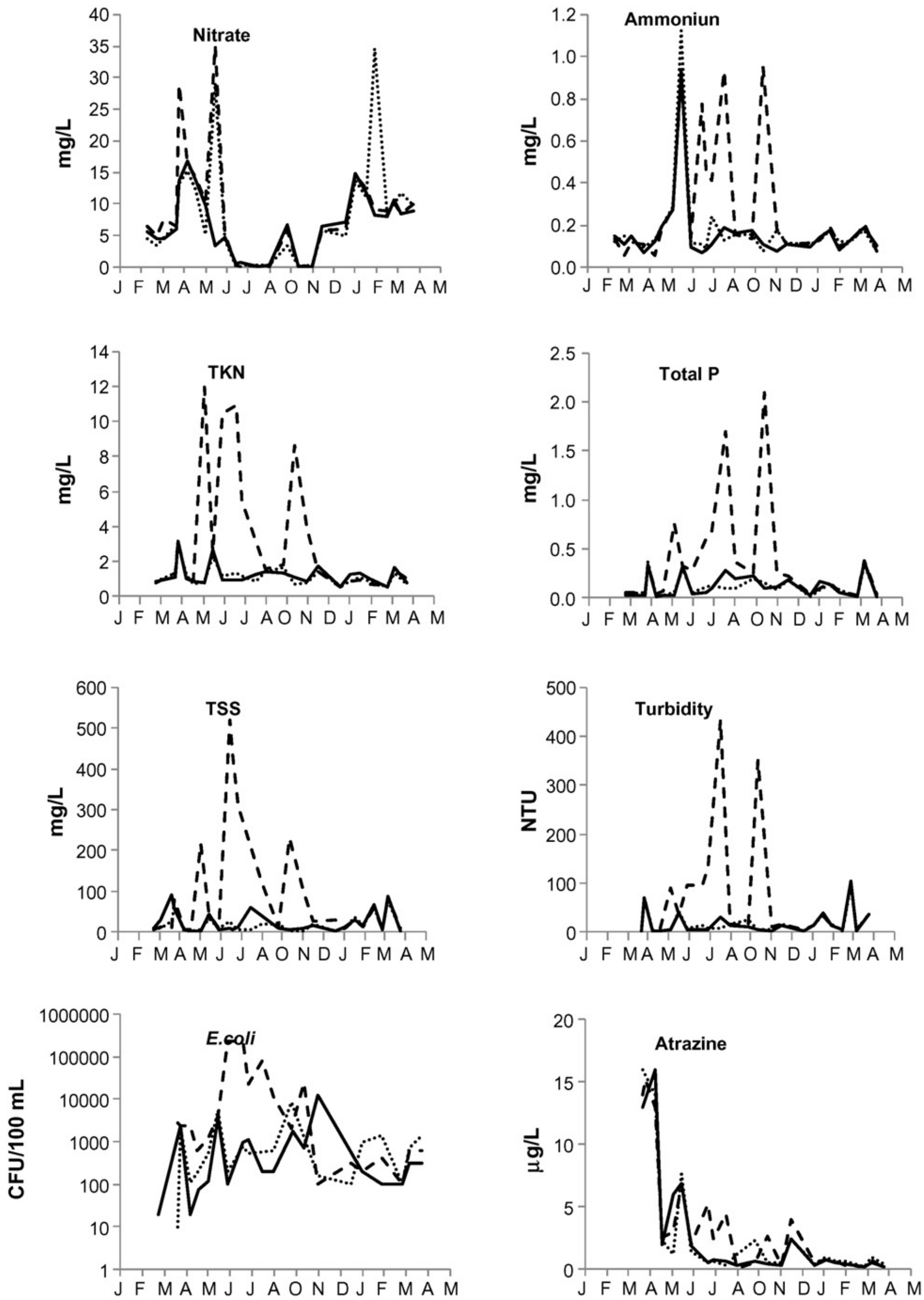


**Fig. 4 – Nitrate, ammonium, total Kjeldahl nitrogen (TKN), total phosphorus (Total P), total suspended sediment (TSS), turbidity, *Escherichia coli* and Atrazine concentrations between March 2005 and March 2006 at stream monitoring stations S1 (solid line), S2 (dashed line) and S3 (dotted line).**

concentrations generally increase between S1 and S2, and that differences between these two stations are larger in the spring for nitrate and in the summer/fall period for ammonium, TKN, TP, TSS, turbidity and *E. coli*.

With the exception of one date in February where nitrate concentration in S3 is much higher than at S2 or S1, nitrate, ammonium, TKN, TP, TSS, turbidity, *E. coli* and atrazine concentrations/levels at S3 fall back to levels similar ($p < 0.001$) to or lower than those observed at S1 during the same period (Table 2 and Fig. 4).

## 5. Discussion

### 5.1. Stream hydrology

The analysis of discharge distribution throughout the year at stream monitoring station S1 indicates that discharge in the stream is approximately three times higher in the winter/spring period than in the summer/fall period. In addition, instantaneous discharge measurements at S1 and S3 indicate that discharge increases on average by 28% between S1 and S3

**Table 2 – Average concentrations for nitrate-N ($NO_3^-$), ammonium-N ($NH_4^+$), total Kjeldahl nitrogen (TKN), total phosphorus (TP), total suspended sediment (TSS), *E. coli* and atrazine, and average turbidity levels for a 12-month period (including events), events only, the winter/spring period (including events), winter/spring events only, the summer/fall period (including events) and summer/fall events only for stream monitoring stations S1, S2 and S3 between March 2005 and March 2006**

| | $NO_3^-$ (mg/l) | | | $NH_4^+$(mg/l) | | |
|---|---|---|---|---|---|---|
| | S1 | S2 | S3 | S1 | S2 | S3 |
| 12 months (*n* = 27) | 6.51 | 8.44 | 7.96 | 0.17 | 0.28 | 0.18 |
| Events (*n* = 10) | 8.20 | 12.91 | 13.20 | 0.24 | 0.27 | 0.28 |
| Winter/spring (*n* = 16) | 9.29 | 12.72 | 12.16 | 0.20 | 0.19 | 0.21 |
| Winter/spring events (*n* = 7) | **9.76** | **16.73** | **17.48** | 0.30 | 0.30 | 0.34 |
| Summer/fall (*n* = 11) | 2.46 | 2.21 | 1.85 | **0.12** | **0.40** | **0.13** |
| Summer/fall events (*n* = 3) | 4.55 | 4.00 | 3.22 | 0.13 | 0.22 | 0.17 |
| | TKN (mg/l) | | | TP (mg/l) | | |
| | S1 | S2 | S3 | S1 | S2 | S3 |
| 12 months (*n* = 27) | 1.20 | 3.30 | 1.17 | 0.13 | 0.38 | 0.12 |
| Events (*n* = 10) | 1.67 | 2.05 | 1.53 | 0.21 | 0.28 | 0.21 |
| Winter/spring (*n* = 16) | 1.05 | 1.72 | 0.98 | 0.12 | 0.18 | 0.12 |
| Winter/spring events (*n* = 7) | 1.59 | 1.39 | 1.35 | 0.23 | 0.22 | 0.22 |
| Summer/fall (*n* = 11) | **0.98** | **4.09** | **1.02** | **0.13** | **0.65** | **0.12** |
| Summer/fall events (*n* = 3) | 1.32 | 2.90 | 1.42 | 0.17 | 0.38 | 0.18 |
| | TSS (mg/l) | | | Turbidity (NTU) | | |
| | S1 | S2 | S3 | S1 | S2 | S3 |
| 12 months (*n* = 27) | 24 | 96 | 19 | 18 | 69 | 18 |
| Events (*n* = 10) | 34 | 68 | 35 | 34 | 48 | 35 |
| Winter/spring (*n* = 16) | 30 | 40 | 25 | 25 | 32 | 25 |
| Winter/spring events (*n* = 7) | 44 | 46 | 43 | 46 | 45 | 45 |
| Summer/fall (*n* = 11) | **16** | **179** | **12** | **9** | **117** | **10** |
| Summer/fall events (*n* = 3) | 11 | 117 | 16 | 10 | 55 | 16 |
| | *E. coli* (CFU/100 ml) | | | Atrazine (μg/l) | | |
| | S1 | S2 | S3 | S1 | S2 | S3 |
| 12 months (*n* = 27) | 1296 | 32,188 | 1308 | 2.58 | 3.64 | 2.64 |
| Events (*n* = 10) | 1349 | 4,799 | 2776 | 1.59 | 3.52 | 2.04 |
| Winter/spring (*n* = 16) | 606 | 1,396 | 1064 | 4.22 | 4.80 | 4.10 |
| Winter/spring events (*n* = 7) | 1324 | 1,532 | 2098 | **1.81** | **4.10** | **2.18** |
| Summer/fall (*n* = 11) | **1929** | **69,823** | **1348** | 0.78 | 2.25 | 1.03 |
| Summer/fall events (*n* = 3) | 1410 | 12,965 | 4470 | 1.22 | 2.35 | 1.80 |

Bold text indicates the period when the largest changes for each parameter (%) are observed between S1 and S2.

in the winter/spring period and by 107% in the summer/fall for an increase in contributing area of only 8.9%. The limited number of instantaneous discharge measurements at all stations (*n* = 15), especially in the summer (*n* = 2), does not allow for the precise quantification of the typical increase in discharge between S1 and S3. However, data suggest that discharge does increase by at least 20–30% between S1 and S3 in the winter/spring period and by more than 50% (likely 100%) between S1 and S3 during the summer/fall period. Considering that the contributing area to S3 is only 8.9% larger than at S1, this suggests a large addition of water to the stream between S1 and S3. In addition, 17–41% lower chloride, calcium, magnesium, sodium and silica concentrations at S3 than at S1/S2 during summer months also suggest that dilution by a source of water external to the stream containing low chloride, calcium, magnesium, sodium and silica concentrations occurs in the summer. As indicated in Fig. 1, there is a drainage ditch feeding into the stream between S1 and S2 and two residential retention ponds discharging into Fishback Creek between S2 and S3. Although the drainage ditch is dry during most of the summer, water level in the retention ponds collecting runoff from precipitation and lawn irrigation in the residential area is maintained. It is likely that discharge from the retention ponds in the summer contributes to the higher discharge observed at S3. Changes in water quality between S1, S2 and S3 are evaluated in the context of this dilution phenomenon.

### 5.2. Water quality analysis

Over a 12-month period, average values for turbidity and nitrate, ammonium, TKN, TP, TSS, *E. coli* and atrazine concentrations all increase by more than 30% (more than 2000% for *E. coli*) over the 130 m distance between S1 and S2 (Table 2), with especially high increases in turbidity and ammonium, TKN, TP, TSS, and *E. coli* concentrations between S1 and S2 during the summer/fall period (low flow period). This suggests that unrestricted cattle access to streams and the lack of adequate buffer between the pasture and the stream can lead to large increases in nutrient/contaminant concentrations in the stream over a very short distance (130 m). This result is consistent with previous studies documenting the impact of cattle access to streams on water quality (Nagels et al., 2002; Line, 2003; McKergrow et al., 2003; Muenz et al., 2006).

Large increases in stream nitrate concentrations between stations S1 and S2 are associated with high atrazine levels and typically occur during winter/spring events (Table 2 and Fig. 4). This suggests that these changes in nitrate concentrations are likely due to inputs of nitrate and atrazine from the intermittent drainage ditch discharging into Fishback Creek between S1 and S2, rather than unrestricted cattle access to the stream as atrazine is typically associated with row crop agriculture in the Midwest (Kladivko et al., 1999). Nitrate concentrations do not increase between S1 and S2 in the summer/fall period, which is the time period when cattle are often observed in or near the stream. This suggests that unrestricted cattle access to the stream has little to no impact on stream nitrate concentration. Although some studies have reported an increase in nitrate in streams not protected from cattle access (Muenz et al., 2006), most studies generally report changes in TKN, TP, *E. coli*, suspended sediment concentrations and turbidity in streams impacted by cattle grazing (Nagels et al., 2002; Line, 2003; McKergrow et al., 2003).

Data indicate that unrestricted cattle access to the stream during the summer has a dramatic effect on turbidity and ammonium, TKN, TP, TSS concentrations, and especially *E. coli* concentration in the stream. Indeed, *E. coli* concentration increases from an average of 1929 CFU/100 ml at S1 in the summer/fall period to 69,823 CFU/100 ml at S2. The U.S. Environmental Protection Agency (USEPA, 2006) identifies a threshold of 235 CFU/100 ml as the geometric mean of four to six measurements within a month as the maximum concentration of *E. coli* in water for full body contact. Although single values cannot be directly compared to the geometric mean of four to six measurements within a month, it is interesting to note that this threshold is exceeded 64 and 89% of the time in the summer/fall period at S1 and S2, respectively. Similarly, TSS, TKN, ammonium, TP concentrations and turbidity show, on average, large increases (more than threefold increases) on the 130 m long stream reach affected by cattle grazing (Table 2). This indicates that the effect of cattle on stream water quality is especially acute in the summer/fall period when flow is low and when cattle is more often observed in or near the stream.

However, despite the obvious deterioration of water quality at S2, especially during summer/fall months, this study also illustrates the resilience of the stream to disturbance as water quality at S3, 875 m downstream from S2, is similar to water quality at S1 (Fig. 4, Table 2). Hydrological data (discharge, cation and chloride data) suggest that some dilution is taking place in the stream between S2 and S3, likely due to the addition of residential pond water to the stream. However, even assuming very low nutrient/contaminant concentrations in pond water, this dilution, estimated to be around a 1:2 ratio in the summer, is not large enough to explain the changes in concentration between S2 and S3 during summer/fall months. Indeed, ammonium concentration during the summer/fall period is approximately three times smaller at S3 than at S2, TKN concentration is four times smaller, TP concentration is five times smaller, TSS concentration 15 times smaller, turbidity 17 times smaller and *E. coli* concentration 51 times smaller. These large decreases in contaminant concentrations between S2 and S3 during summer months suggest that processes other than a 1:2 dilution are affecting TKN, TP, TSS, turbidity, and *E. coli* concentrations in the stream.

In-stream processes were not investigated in this study; however, site description information indicates a progressive shift from an artificially incised stream channel upstream from S1 to a wide and shallow stream channel with well formed pools and riffles bordered by a large forested riparian area at S3. One possible reason for the improvement of water quality at low flow between S2 and S3 is that natural stream geometry may increase hyporheic exchange at low flow and the ability of the stream to process nutrients and trap sediments and associated contaminants such as TKN and TP. Although a detailed investigation of in-stream processes would be necessary to obtain a definitive answer, data suggest that the more natural stream channel geometry and the occurrence of forested riparian buffer immediately upstream from S3 likely contribute to the improvement of water quality in the stream, especially during low flow periods during which hyporheic processes generally have the largest impact on water quality.

## 6. Implications for watershed management and conclusions

By offering a direct quantification of the localized impact of unrestricted cattle access to streams on water quality, this study provides a means to assess the potential impact of widespread cattle access to streams on water quality at the watershed scale. Although most nutrients are exported in the spring (Royer et al., 2006), this study indicates that unrestricted cattle access to streams in regions traditionally dominated by row crop agriculture can have dramatic effects on water quality during the summer/fall period when flow is low. However, this study also illustrates that when restricted to a small area, the impact of cattle access to streams on water quality is limited. Results also suggest that natural stream geometry can positively impact water quality regarding TKN, TP, TSS, turbidity, and *E. coli*, especially when flow is low. Stream rehabilitation is therefore likely to be an efficient tool to improve water quality in traditionally agricultural regions of the Midwest and other regions with poorly drained soils and flat to gentle topography. This study will be an incentive for policy makers to promote stream rehabilitation and to develop more stringent guidelines limiting cattle access to streams in many Midwestern states and other regions dominated by row crop agriculture. By illustrating the potential impact of unrestricted cattle access to streams in regions with flat to gentle topography and poorly drained soils, a positive impact on the management of water in agriculture is expected from this study by providing policy makers and landscape managers with strong evidence of the negative impact of cattle access to streams in regions where this issue is rarely investigated.

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