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Erosion and sediment delivery in southern Iowa watersheds: Implications for conservation planning

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Abstract: Soil sediment export from agricultural watersheds is a major environmental concern and is a primary contributor to nonpoint source sediments in streams. The purpose of this study was to quantify total suspended solid (TSS) export and current sediment delivery ratios (SDRs) in four southern Iowa watersheds and evaluate how existing and potential best management practices (BMPs) have affected SDRs. Our study updated estimates of SDRs that were previously developed using mid-20th century data and largely unknown methods. We estimated TSS export using continuous turbidity measurements and total phosphorus (P) data and measured discharge data to calculate TSS loads. By comparing annual TSS export to watershed-scale soil erosion estimated with the Revised Universal Soil Loss Equation (RUSLE) model, we calculated annual SDRs for the study watersheds and found that current SDRs were significantly lower than previous estimates. This new analysis provides an exceptional story of conservation progress in our study watersheds over the past four decades. Further, they are likely a worst-case scenario for sheet and rill eroded sediment export since TSS export does not distinguish among other sediment sources, such as stream bank and gully erosion. Based on the extent of BMP implementation in the watersheds and the potential for future BMPs determined using the Agricultural Conservation Planning Framework (ACPF) toolbox, we found that there is only limited potential for further reducing TSS export using additional in-field practices. Hence, we believe that further work toward reducing TSS export in these Iowa watersheds should be shifted to reducing contributions from other TSS sources including from streambed, bank, and gully erosion.

Key words: best management practices—sediment delivery ratios—soil conservation—soil erosion

Soil sediment export from agriculturally dominated watersheds is a global crisis that has vast-reaching and severe environmental and economic impacts (Brown and Wolf 1984; Myers et al. 1984; Siddiqui 1998). Multiple sources are known to contribute to sediment export from these watersheds, including in-field sheet and rill erosion (Johnson and Moldenhauer 1970), ephemeral gully erosion (Gomez et al. 2003), and stream bank erosion (Beck et al. 2018; Odgaard 1984). In the late 20th century, Pimentel et al. (1995) estimated that the effects of soil erosion alone produce societal costs exceeding US$40 billion annually. However, these costs are difficult to attribute to individual sources since the effects of soil erosion are not temporally or spatially consistent (Cruse et al. 2006) and relatively little research has been completed that accurately determined sources of sediment transport (Waters 1995). Still, new research over the last two decades is working to solve the mysteries of sediment sourcing by using new methods in tracking like rare earth element tracking (Kimoto et al. 2006), trace tracking (Collins et al. 2013), and biomarker tracking (Cooke et al. 2008). However, these methods are not refined and most are designed for very targeted, small-scale environments (Collins et al. 2013), rather than tracking landscape processes.

In Iowa, United States, agricultural watersheds are major contributors of nonpoint source sediments in streams (Schilling et al. 2011). It has been estimated that 93% of Iowa streams are primarily impaired by agricultural related sediment losses (Iowa Department of Natural Resources 2018b). Suspended sediments are a primary source of nitrogen (N) and phosphorus (P) to rivers, which contributes to eutrophication in streams (Carpenter et al. 1998) and development of seasonal hypoxic conditions in the Gulf of Mexico (Turner et al. 2007). Suspended sediments are also known to contribute to total carbon (C) export (Jones and Schilling 2013). Suspended sediments further negatively impact stream ecosystems by interfering with the growth and propagation of aquatic life (Newcombe and Jensen 1996; Newcombe and MacDonald 1991). Many states in the US Midwest including Ohio (Ohio Environmental Protection Agency 2013), Illinois (Illinois Environmental Protection Agency and Illinois Department of Agriculture 2014), Minnesota (State of Minnesota 2014), and Iowa (Iowa Department of Agriculture and Land Stewardship et al. 2013) have developed nutrient reduction strategies that aim to reduce the export of nutrients and sediments from agricultural watersheds to the Mississippi River (Christianson et al. 2018). However, accurately quantifying suspended sediment export from watersheds is difficult because the process relies largely on erosion and sedimentation modeling, which may be highly inaccurate depending on the data that feed the models (Parsons et al. 2004).

The sediment delivery ratio (SDR) can be used to quantify sediment export from a watershed. SDRs make comparisons of the estimated mass of soil eroded in a watershed to the estimated mass of total suspended solids (TSS) exported in streams. SDRs can be calculated for ephemeral soil erosion (USDA NRCS 1998), sheet/rill erosion (Johnson and Moldenhauer 1970), and stream bank erosion (Odgaard 1984). However, it is difficult to differentiate among TSS sources in watershed studies (Beck et al. 2018).

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Hence, this study focuses entirely on the TSS fraction of sediment transport. It is also important to note that not all sheet and rill eroded sediments are found in streams as suspended sediments. Postsettlement flood plain alluvium dominates the surfaces of Iowa's flood plains and is a measurable component of stream bank erosion contributions. Streambed load also contributes to total watershed-scale SDRs, but accurately sampling and quantifying bedload contributions is difficult (Bhowmik et al. 1986).

For watersheds in Iowa, the USDA published a report on erosion and sediment delivery, which estimated sediment sources by using a nomograph approach based on watershed area (USDA NRCS 1998). The nomograph method utilized mid-20th century erosion modeling and stream data (USDA NRCS 1998). While this report has been the standard for the last two decades, new studies have sought to update SDRs based on current soil erosion and TSS export conditions. While current estimates do not include sediments transported as bedload, Streeter et al. (2018) estimated TSS export from a small Iowa watershed using turbidity surrogates, which were created utilizing existing stream data (Iowa Department of Natural Resources 2019; Iowa Flood Center 2019a, 2019b), and compared TSS export to sediment erosion occurring in the watershed. Results from that study concluded that the SDR for the study watershed was 3.7%, which was less than one-third of the predicted value by the 1990s USDA report. Lower SDRs in Iowa watersheds are consistent with increased implementation of best management practices (BMPs) that have reduced sediment erosion and export over the last several decades (Jones and Schilling 2011).

Despite the positive effects of BMPs to reduce sediment export, actual quantification of their impacts on SDRs has not been done. Rundhaug et al. (2018) made comparisons of currently implemented BMPs to potential BMP locations in subbasins of the English River watershed in southern Iowa and determined that there was still the potential for additional BMPs in the watershed. However, they did not analyze the impacts of the existing or future BMPs on sediment export or SDRs. This may be possible by utilizing new geographic information systems (GIS)–based methods, such as the Agricultural Conservation Planning Framework (ACPF) toolbox (USDA 2019b), which has been developed to identify potential locations of future BMPs, and considering the effects of those BMPs on soil erosion and SDRs.

The primary goal of this study was to quantify current SDRs in four southern Iowa watersheds that vary in size and evaluate how existing and potential BMPs affect SDRs in the same landscape region. Our specific objectives were to (1) develop a current estimate of soil erosion for the English River and three nested Rapid Creek watersheds utilizing the Revised Universal Soil Loss Equation (RUSLE) model (Renard et al. 1997); (2) estimate TSS export from the four watersheds using surrogate measures and discharge rating curves; (3) calculate current SDRs for each study site watershed; and (4) evaluate the effects of current and future-adopted BMPs on SDRs and analyze the potential for future reductions in sediment export.

Materials and Methods

Study Area. The English River, Rapid Creek and two subbasins within Rapid Creek were evaluated in this study (figure 1). The English River drains a watershed of 1,500 km^2 extending from its headwaters in Poweshiek County to discharge into the Iowa River in Washington County. A US Geological Survey (USGS) gauging station exists near Kalona in Washington County, Iowa (figure 1). Nearly 18 hydrologic unit code (HUC) 12 watersheds lie upstream of the English River gauging station near Kalona. The 88 km^2 HUC 12 Rapid Creek watershed in Johnson County flows into the Iowa River near Iowa City. Two subbasins in Rapid Creek were selected in order to analyze effects of scale on our results. The upstream (US) subbasin consists of the headwaters of the watershed and is approximately 8 km^2, whereas the downstream (DS) subbasin lies adjacent to the south of the US subbasin and is approximately 16 km^2 (figure 1). Land use in all of the study watersheds was predominantly row crop agriculture consisting of corn (*Zea mays* L.) and soybeans (*Glycine max* L.) (table 1). Mean annual precipitation for the study period (2017 to 2018) was 884 and 853 mm for the English River and Rapid Creek, respectively (table 1) (Iowa State University Department of Agronomy 2019a). Mean annual air temperature was 10°C (Iowa State University Department of Agronomy 2019b).

Land surface topography varies significantly across the study area. The study watersheds are situated within the Southern Iowa Drift Plain landform region of Iowa (Iowa Department of Natural Resources 2018a). This region is dominated by rolling hills of pre-Illinoian glacial till capped by Wisconsin loess and well-developed floodplains consisting of silty and sandy alluvium. The relic till plain has been further reshaped by frequent precipitation and overland sediment movement enhanced by postsettlement agricultural tillage (Cruse et al. 2006). The majority of the soils in our study were formed in loess, alluvium, and reworked glacial deposits. Soils formed in loess may typically be found in stable upland environments and have well developed, organic rich, surface horizons. Soils formed in glacial till–derived sediments on steeper slopes tend to have less well developed surface horizons and are lower in soil organic matter (Ritchie et al. 2007). Overall, the study area watersheds are located in a landform region that, even before commercial agriculture, has been defined by soil erosion and sedimentation.

RUSLE. In Iowa, a statewide RUSLE grid (10 m cell size) is available from the Iowa GeoData Portal (Iowa Department of Natural Resources 2018a). This model, which is driven primarily by precipitation, agricultural tillage, and conservation practices, provides an estimation of average annual soil erosion in Mg ha^{-1}. The components of the RUSLE model are the rainfall/runoff erosivity factor (R), soil erodibility (K), slope length and steepness factors (LS), cropping/management factor (C), and a conservation practice factor (P). The statewide RUSLE grid was created from a 10 m resolution land cover grid that provided the crop/tillage information (C factor), a 10 m Soil Survey Geographic database (SSURGO) data grid that provided the erodibility information (K factor), and rainfall (R) and slope length and steepness (LS) factors (using the 10 m SSURGO R and LS values), which were obtained from tables in "Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)" (Renard et al. 1997). For this study, we utilized the existing K and LS factors from the statewide RUSLE model. We then updated the R factor based on 2017 and 2018 precipitation data (US Environmental Protection Agency 2018). C factors were updated utilizing the newly created ACPF field boundary data sets (USDA 2019a). P factors were assigned by rasterizing recently completed maps of existing conservation practices (Iowa State University 2019) and assigning P values based on established values for each practice (USDA NRCS 2002). Watershed-scale soil loss estimates were developed by summing each individual raster's soil loss estimate and dividing by the total watershed area.

Sediment Delivery. Our study compared estimated annual soil erosion to estimated annual soil export from a watershed to determine the effectiveness of BMP implementation on soil trapping. The mass of soil exported to the stream in relation to the estimated mass of soil erosion in the watershed is the SDR. At the outlet of each watershed and subbasin, continuous stream stage and water quality data were collected using sensor technology from April through November in 2017 and April through October in 2018. In Iowa, during the winter months when temperatures are generally below freezing, the sensors are removed from the streams. Previous research in this region determined that less than 3% of stream sediment load is transported during the frozen months (Jones and Schilling 2011; Schilling 2000). We further used the Iowa Daily Erosion Project (Iowa State University Department of Agronomy 2019a) to determine that all but 3% of estimated soil erosion occurred from April through November in 2017 and April through October in 2018. Hence, we determined that the periods in 2017 and 2018 with stream sensors installed adequately captured the total sediment export from each watershed and subbasin.

Stream stage data were collected every 15 minutes using solar powered bridge sensors (Iowa Flood Center 2019b) and used to estimate stream discharge (figure 2). The bridge sensor utilizes a sonar signal to measure the distance between the sensor and the stream. The stream data are then made publicly available when they are transmitted via cell modem to the Iowa Flood Information System (Iowa Flood Center 2019a). Light Detection And Ranging (LiDAR) elevation data are available in a 1 m grid size from the Iowa GeoData Portal (Iowa Department of Natural Resources 2018a). We utilized that LiDAR data to create channel cross-sections at each bridge sensor location. The 1 m LiDAR grid had a horizontal accuracy with root mean square (RMS) error of 15 cm at a 95% confidence interval. Vertical accuracy was 18.5 cm RMS at a 95% confidence

Figure 1
Location map of English River and Rapid Creek watersheds in Iowa, United States, with exploded county view showing stream gauge sites and major rivers.

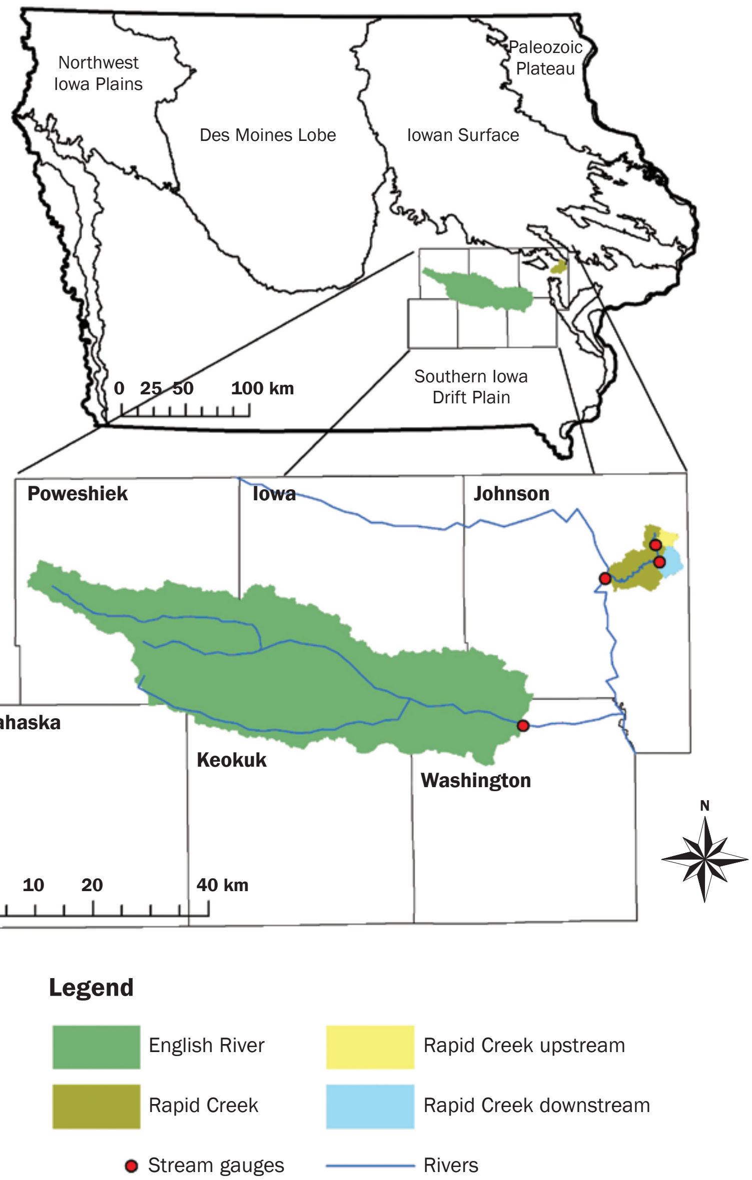
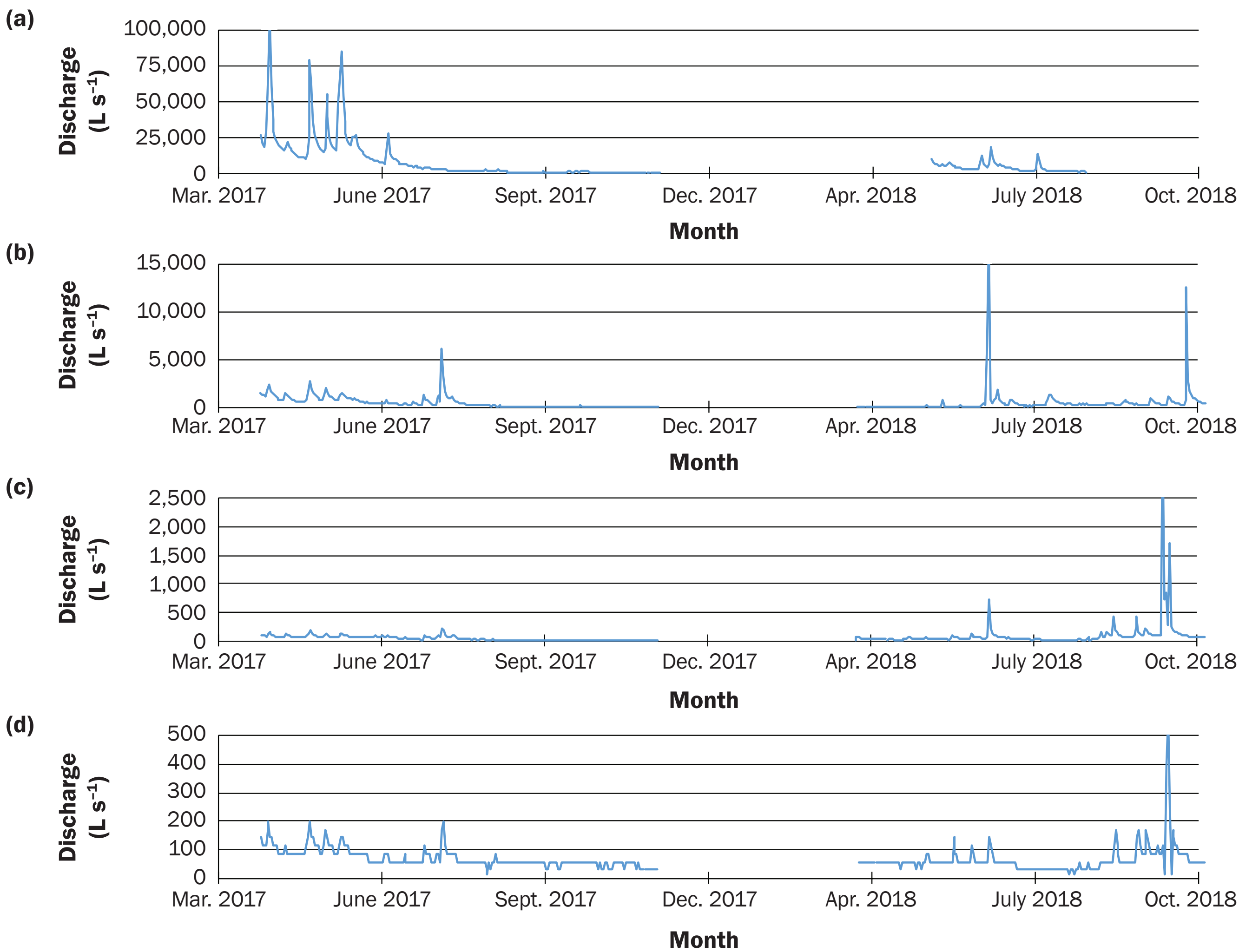


Table 1
Study watershed summaries.

Watershed	Watershed area (ha)	Agricultural area (%)	Mean LS value	Mean annual precipitation (mm)
English River	149,183	59	0.7	884
Rapid Creek	8,754	47	0.8	853
Rapid Creek upstream	777	87	0.6	853
Rapid Creek downstream	1,554	77	0.6	853

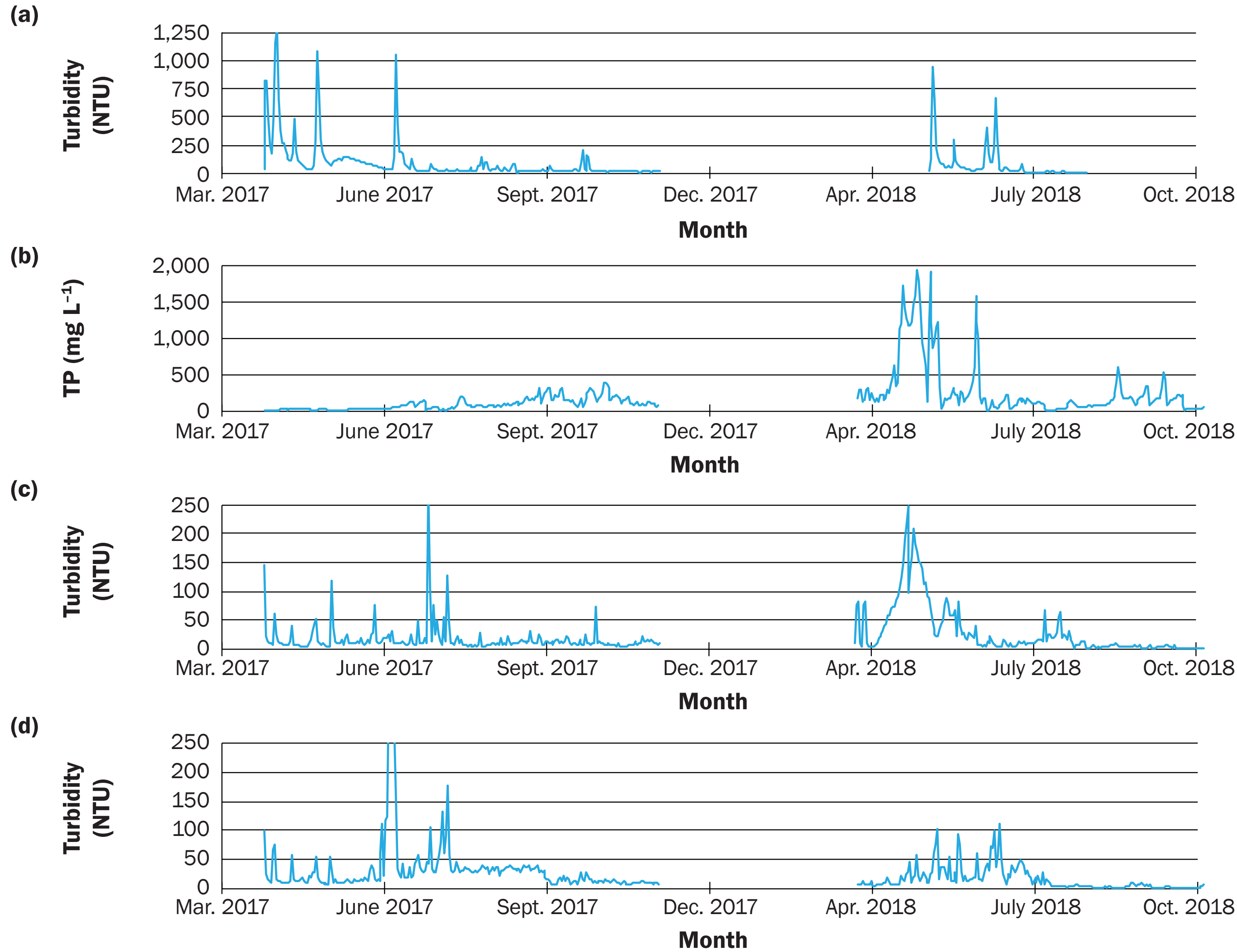
Figure 2
Stream discharge (L s⁻¹) for each watershed study area from April of 2017 through October of 2018: (a) English River, (b) Rapid Creek, (c) Rapid Creek upstream, and (d) Rapid Creek downstream.



interval. The channel cross section invert was adjusted for the lowest elevation measured by the stage sensor. Manning's roughness coefficients were selected to represent the overbank and channel areas based on engineering judgement and reference materials (Chow 1959). Channel reach slope was estimated for each site based on LiDAR data. We then solved for water surface elevations based on a range of discrete discharges by utilizing the Hydrologic Engineering Center's River Analysis System (HEC-RAS) uniform flow hydraulic design function (US Army Corps of Engineers 2019). Discharge sensitivity was estimated based on low and high Manning's roughness values. The typical value scenario was 0.04 for the channel and 0.1 for the overbanks. The high value scenario was 0.05 for the channel and 0.15 for the overbanks, whereas the low value scenario was 0.03 for the channel and 0.05 for the overbanks. Overall, the estimated daily discharge sensitivity using the rating curve method was ±25% for the range of stream fluctuations measured during the study.

The two subbasins of Rapid Creek and the English River watershed were implemented with turbidity sensors in addition to the stream stage sensor. Turbidity was measured every 15 minutes at each sensor using Hach instrumentation (Hach Instrumentation, Loveland, Colorado). The data are publicly available through the Iowa Water Quality Information System (University of Iowa 2019). Turbidity data were downloaded for our study timeframe and compiled as daily averages for analysis (figure 3). We then correlated TSS to turbidity using monitoring data collected at Old Man's Creek located nearby. Old Man's Creek is located in the same landform region and is immediately adjacent to the English River watershed. Previous research has shown that the relation of turbidity to sediment-bound total P (TP) is very consistent across landform regions in Iowa (Schilling et al. 2017). Hence, we considered the relationship of TSS to turbidity developed for Old Man's Creek to be representative of the conditions in our study watersheds. Turbidity and TSS data were downloaded from the Iowa DNR ambient

Figure 3
Stream turbidity (NTU) for (a) English River, (c) Rapid Creek upstream, and (d) Rapid Creek downstream study areas from April of 2017 through October of 2018. (b) Stream TP (estimated by LOADEST) for the Rapid Creek watershed study area.



monitoring database (Iowa Department of Natural Resources 2019), and a simple linear regression was assigned to the data (R^2 = 0.92) with the following equation:

$$TSS = 2.2369 \times \text{turbidity} - 3.2859, \quad (1)$$

where TSS is in mg L^{-1} and turbidity is in NTU. We then used this equation to convert daily turbidity in our watershed streams to daily TSS. We further multiplied TSS by daily discharge to estimate daily TSS export (in kg) from each watershed and subbasin.

While each subbasin of the Rapid Creek watershed had turbidity sensors in place, the HUC 12 Rapid Creek watershed did not. Instead, TP was measured at the Rapid Creek bridge sensor location monthly from October of 2014 through June of 2017. We used the USGS software program LOADEST (Runkel et al. 2004) to estimate daily TP loads for Rapid Creek. LOADEST estimates daily TP loads using a rating curve approach by incorporating discrete samples and continuous streamflow data. We used the seven-parameter load prediction model:

$$\ln(L) = \beta0 + \beta1\ln(Q) + \beta2[\ln(Q)]2 + \beta3t + \beta4t2 + \beta5\sin(2\pi t) + \beta6\cos(2\pi t) + \varepsilon, \quad (2)$$

where L = CQ is the load or flux, C is concentration, Q is discharge, t is time in decimal years, $\beta0$, $\beta1$,..., $\beta6$ are regression coefficients, and ε is assumed to be an independent and normally distributed error with zero mean and constant variance. This approach still required continuous streamflow data, but continuous streamflow data are common in Iowa (US Geological Survey 2019) and may be a reasonable approach for future large-scale SDR estimations where turbidity data are not available. Still, estimation of nutrient loads in rivers is challenging and has the potential for a high degree of error and uncertainty (Hirsch 2014; Schilling et al. 2017). Indeed, Stenback et al. (2011) found that 15% of case sites over/underestimated TP levels by more than 25%. With those limitations in mind, we analyzed the relationship of TSS to TP for Old Man's Creek. TP and TSS data were downloaded from the Iowa

Department of Natural Resources ambient monitoring database (Iowa Department of Natural Resources 2019), and a simple linear regression was assigned to the data (R^2 = 0.99) with the following equation:

$$TSS = 2.0934 \times TP. \quad (3)$$

We then used this equation to convert daily TP estimates from LOADEST for Rapid Creek to daily TSS. We further multiplied TSS by daily discharge to estimate daily TSS export (in kg) for the watershed.

Conservation Practices. Current inventory of existing BMPs in the four study watersheds were located as part of the Iowa BMP Mapping Project (Iowa State University 2019). The Iowa BMP Mapping Project identified and mapped existing contour buffer strips, contour terraces, and water and sediment control basins (WASCOBs) in each of our study site watersheds in 2018 using a combination of LiDAR elevation data and color infrared photographs. The BMP mapping project provided a baseline for current progress and future development of conservation practices.

The ACPF toolbox is a series of GIS tools that analyze high-resolution soil, terrain, and land use data in order to generate maps that identify ideal locations for many conservation practices in agricultural fields and riparian zones (USDA 2019b). Several of the conservation practices that the ACPF identifies, namely contour buffer strips, contour terraces, and WASCOBs, directly influence SDRs by retaining or "trapping" dislodged sediments behind them, thereby serving to reduce sediment export from the watershed. The ACPF toolbox was completed for each of our study site watersheds for the period between 2017 and 2018.

By utilizing the 2018 BMP mapping results, we were able to modify the P factor in our RUSLE model to account for current conservation practices. This was accomplished by utilizing a GIS tool developed by the Iowa DNR (Asell 2019), which determined the catchment area that drained to each existing conservation practice (figure 4). This tool intersected the location of each BMP with streamflow paths. Land area above each intersection point was then determined. We then created a raster grid for each conservation practice where each cell was assigned either "1" for no practice areas or the appropriate P value where practices existed (0.8 for terraces and contour buffer strips and 0.7 for WASCOBs [USDA NRCS 2002]). These new P value parameters were used when estimating current soil erosion with RUSLE (table 2). Finally, we compared the total area impacted by each conservation practice to the total watershed area in agricultural row crop production to determine a "percentage saturation" for each BMP (table 3).

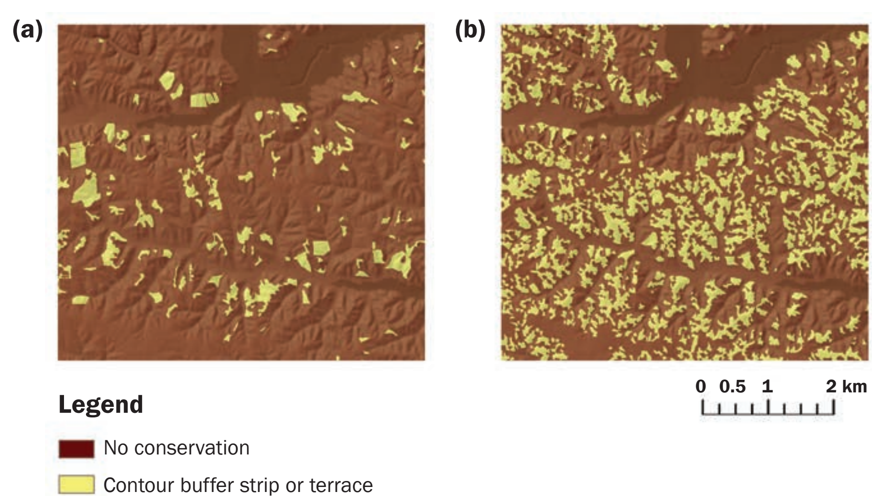
Similar to the methods used for determining the land area impacted by existing BMPs, we again used the Iowa DNR's GIS tool (Asell 2019) to determine the land area that would be impacted by conservation practices if all of the practices suggested by the ACPF tool were implemented. We further calculated new P values for watershed areas and ran the RUSLE model again to provide a best case estimate of soil erosion if all of these practices were implemented. In doing so, we were able to make comparisons of soil erosion estimates (table 2) and percentage saturation of conservation practices (table 3) for current BMP mapping and potential ACPF conservation scenarios.

Results and Discussion

English River. The English River watershed land area was comprised by a majority (59%) in row crop agriculture (table 1). This accounts for over 88,000 ha of the watershed. BMPs, namely contour buffer strips, contour terraces, and WASCOBs, are widespread in the watershed. A total of 19,119 existing BMPs were identified, which impacted 9% of the total watershed area and more specifically, 16% of the agricultural row crop areas (table 3). When the effects of existing BMPs are included in the RUSLE calculations, average estimated soil loss for the entire watershed in both 2017 and 2018 was approximately 8.0 Mg ha^{-1} y^{-1}, which is less than the defined "tolerable" value of 11.2 Mg ha^{-1} y^{-1} (Schertz 1983). However, if we assume that no soil loss occurred in non-row crop areas (total soil loss divided by land area in row crop production only), the average estimated soil loss for the row crop areas could be as high as 13.6 Mg ha^{-1} y^{-1} (table 2).

Stream discharge monitored in the English River in 2017 showed several large events, with fewer large events observed in 2018 (figure 2). Coincident with discharge, turbidity also fluctuated during the two years (figure 3). Based on the turbidity-TSS regression developed from Old Mans Creek, approximately 95,000 Mg of TSS were estimated to be exported from the watershed or 0.64 Mg ha^{-1} (table 4). If we assume these sediments to be entirely derived from the row crop portion of the watershed rather than the watershed as a whole, TSS export attributed to those areas

Figure 4
Example of area saturated by contour buffer strip or terrace in English River watershed as identified by (a) best management practice (BMP) mapping and (b) potential saturation of the same conservation practices as identified by the Agricultural Conservation Planning Framework (ACPF) toolbox.



could be as high as 1.08 Mg ha^{-1}. The SDRs for the English River watershed were 8.0% for the entire watershed and 13.5% if the TSS is apportioned to the row crop land. Although the monitoring periods were not complete calendar years, the 2017 and 2018 monitoring season accounted for over 99% of the precipitation measured for the entire two-year period. Therefore, we can be confident that the turbidity-derived TSS data were a reasonable estimate of total suspended sediment exported from the English River.

The potential for further reductions in soil erosion was evaluated by considering results from running the ACPF toolbox. The ACPF toolbox identified a total of 67,415 potential practices, which could potentially impact 65% of the agricultural land area or 38% of the total watershed. The ACPF toolbox indicated the potential for a three-fold increase in the number of conservation practices compared to current BMPs, which could potentially impact four times the land area. However, when the ACPF-located practices were input into the RUSLE model, estimated soil loss was only reduced to 7.3 Mg ha^{-1} y^{-1} for the entire watershed and 12.4 Mg ha^{-1} y^{-1} when considering only the agricultural land area (table 2). Hence, potential practices reduced soil erosion by 9% compared to the 300% to 400% increase in the number of practices.

Rapid Creek. The approach used to determine the estimated soil erosion over the study period for the Rapid Creek watershed was similar to that used for the English River watershed. Similarly, we found that nearly all of the precipitation (98%) occurred during the months when stream sensors were installed and water quality data were collected. Less watershed land area was devoted to row crop production in Rapid Creek (47% or approximately 4,100 ha) compared to the English River (table 1). Likewise, fewer BMPs were mapped in Rapid Creek

Table 2
Results from Revised Universal Soil Loss Equation (RUSLE) soil erosion modeling for current watershed scenarios (with current best management practices [BMPs]) and scenarios including the incorporation of Agricultural Conservation Planning Framework (ACPF) toolbox suggested conservation. Full watershed and row crop area only (ag.) results are shown.

	Current BMPs (Mg ha^{-1} y^{-1})			Current BMPs ag. only (Mg ha^{-1} y^{-1})	ACPF practices (Mg ha^{-1} y^{-1})			ACPF ag. only (Mg ha^{-1} y^{-1})
Watershed	Range	Mean	Std. Dev.	Mean	Range	Mean	Std. Dev.	Mean
English River	0 to 180.0	8.0	8.2	13.6	0 to 180.5	7.3	7.5	12.4
Rapid Creek	0 to 78.5	7.6	8.5	16.2	0 to 112.7	7.0	8.1	14.9
Rapid Creek upstream	0 to 44.4	9.3	6.3	10.7	0 to 45.5	8.0	6.0	9.2
Rapid Creek downstream	0 to 74.4	8.7	6.4	11.3	0 to 76.2	7.4	5.8	9.6

Table 3
Best management practice (BMP) mapping and Agricultural Conservation Planning Framework (ACPF) number (#) of practices and impacted watershed area as percentage of row crop area (ag.) and percentage of watershed area.

Watershed	BMP # of practices	BMP % of ag. area	BMP % of watershed area	ACPF # of practices	ACPF % of ag. area	ACPF % of watershed area
English River	19,119	16	9	67,415	65	38
Rapid Creek	163	5	2	1,822	66	31
Rapid Creek upstream	3	1	1	295	83	73
Rapid Creek downstream	55	10	8	637	86	66

Table 4
Summary of sediment export for the study watersheds and row crop area only (Ag.) including total suspended solids (TSS) and sediment delivery ratio (SDR).

Watershed	Overall TSS (Mg)	Watershed TSS (Mg ha^{-1})	Watershed SDR (%)	Ag. TSS (Mg ha^{-1})	Ag. SDR (%)
English River	95,367	0.64	8.0	1.08	13.5
Rapid Creek	1,608	0.18	2.4	0.36	5.1
Rapid Creek upstream	82	0.11	1.1	0.12	1.3
Rapid Creek downstream	135	0.08	1.0	0.11	1.3

than the English River, and they impacted 5% of the agricultural area and 2% of the total watershed area (table 3). Annual soil erosion for the 2017 to 2018 study period was estimated to be 7.6 Mg ha^{-1} y^{-1} for the entire watershed and 16.2 Mg ha^{-1} y^{-1} for the row crop land area (table 2). Similar to the results from the English River, these two values for estimated annual soil loss were below and above the defined tolerable value of 11.2 Mg ha^{-1} y^{-1} (Schertz 1983).

Stream discharge in Rapid Creek showed fewer large stream events compared to the English River during the study period (figure 2). Although we didn't have access to turbidity data for the Rapid Creek watershed, we utilized monthly TP data and the USGS software program LOADEST (Runkel et al. 2004) to estimate daily TP loads in Rapid Creek (figure 3). Then, using the correlation developed for TP and TSS in the Old Man's Creek watershed, we estimated that over 1,600 Mg of TSS were exported from the watershed during the study period (table 4). This equates to 0.18 Mg ha^{-1} for the entire watershed or 0.36 Mg ha^{-1} if we exclude nonagricultural areas. Over the study period, SDRs for Rapid Creek were 2.4% for the watershed and 5.1% for row crop area only (table 4).

Since less land area in Rapid Creek was in row crops, the overall estimated soil loss for the watershed was slightly lower than for the English River (7.6 compared to 8.0 Mg ha^{-1} y^{-1}). However, when the soil erosion estimates were attributed to only row crop areas, soil erosion was higher in Rapid Creek (16.2 Mg ha^{-1} y^{-1}) compared to the English River (13.6 Mg ha^{-1} y^{-1}), due mainly to fewer BMPs observed in Rapid Creek. Interestingly, the ACPF toolbox identified 1,822 potential BMP locations in Rapid Creek watershed, which could potentially impact 66% of the row crop areas. This percentage was similar to the area impacted by potential BMP locations in the English River (65%) (table 3). Still, the percentage area impacted by ACPF BMPs for the entire Rapid Creek watershed was less than that for the English River (31% compared to 38%). However, existing BMPs impacted a very small area of Rapid Creek compared to their potential impact (31%).

Rapid Creek Subbasins. Land use in two headwater subbasins of Rapid Creek were predominantly in row crops, comprising 87% of the US subbasin and 77% of the DS subbasin (table 1). BMP mapping identified only three practices in the US subbasin, impacting 1% of the land area, whereas 55 BMPs were identified in the DS basin, impacting 10% of the land area. Incorporating the current BMPs in the RUSLE model, we estimated soil loss to be 9.3 and 8.7 Mg ha^{-1} y^{-1} for US and DS subbasins, respectively (table 2). These values increased to 10.7 and 11.3 Mg ha^{-1} y^{-1} for US and DS subbasins, respectively, when applied to just row crop areas.

Like discharge in the larger watershed, several large discharge events were identified in the gaging records, although several events observed in the US subbasin in 2018 were not identified in the DS subbasin (figure 2). Based on turbidity data available for the subbasins (figure 3), we estimated that 82 Mg of TSS were exported from the US subbasin during 2017 and 2018, while 135 Mg of TSS were exported from the DS subbasin (table 4). For their entire area, this equates to 0.11 and 0.08 Mg ha^{-1} for the US and DS sites, respectively, but were 0.12 and 0.11 Mg ha^{-1} when applied to just the row crop areas. The soil erosion estimates were higher in these subbasins compared to Rapid Creek or English River watersheds. However, the SDRs for these subbasins were lower, 1.1% and 1.0% for the entire land areas within the US and DS subbasins, respectively, whereas the SDR for the row crop area was 1.3% at both locations (table 4). Low SDRs were mainly due to a decrease in overall TSS export compared to the larger watersheds rather than a decrease in the estimated soil erosion.

Although more BMPs were mapped in the DS subbasin, both watersheds have potential for future BMP implementation. The ACPF toolbox identified 295 and 637 sites for potential practices in the US and DS subbasins, respectively. If these were implemented, the impacted watershed area would increase to 73% and 66% for the US and DS locations, respectively, while impacted row crop area would increase to 83% and 86% for the same locations. Comparing the existing BMPs to the potential BMPs suggests that these smaller basins have the largest potential for increases in BMP saturation since they are dominated by row crops and losses could be reduced with relatively few BMPs.

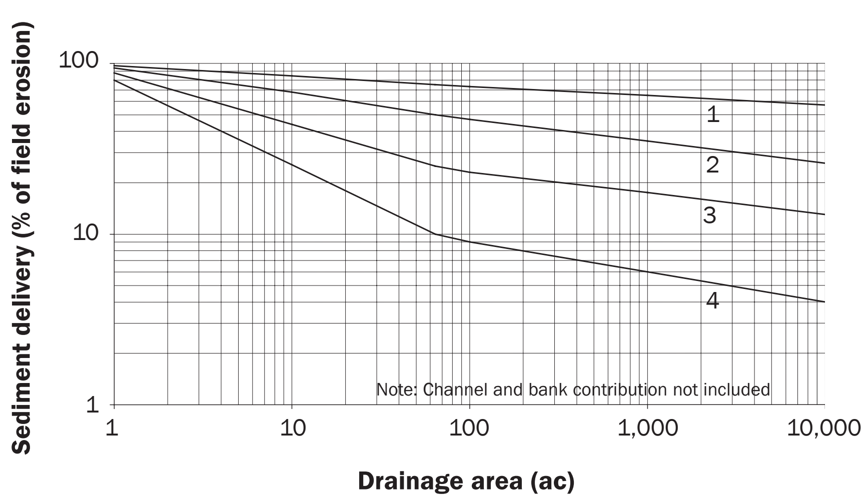
Soil Erosion. Sustainable soil loss has been a policy defined term for decades. The value for sustainable soil loss, which is to say, soil loss rates that are equal to soil development rates, is traditionally considered to be 11.2 Mg ha^{-1} y^{-1} (Schertz 1983). However, this value is likely much higher than actual rates of soil development. Soil development has been found to be closer to 1 Mg ha^{-1} y^{-1} (DeWitt 1981) and as low as 0.5 Mg ha^{-1} y^{-1} (Alexander 1988). For our study, we modeled two years of soil erosion using RUSLE for four Iowa watersheds using 2017 and 2018 rainfall data as well as current land use type estimates and current P values derived from BMP mapping. The overall average annual soil loss was estimated to be 7.6 to 9.3 Mg ha^{-1} y^{-1} for the entire watershed areas of the English River, Rapid Creek, and two Rapid Creek subbasins, and 10.7 to 16.2 Mg ha^{-1} y^{-1} if the soil erosion were applied to just the row crop acres (table 2). Differences in soil erosion rates among the watershed areas are due, in part, to differences in the amount of land under row crop cultivation, as the English River had 59% of the total area in agricultural production, the Rapid Creek watershed had 47%, and the two subbasins of Rapid Creek had the highest percentage (77% to 87%; table 1). It is interesting to note that soil erosion rates increased with row crop percentage when rates were averaged across the watershed scale, but when the rates were isolated to just the row crop acres in the watershed, soil erosion per cropped area was less in the intensely cropped Rapid Creek subbasins compared to the larger watersheds. This was due to smaller length slope (LS) factors in the headwater basins (0.6) compared to Rapid Creek overall (0.8) and English River watershed (0.7) (table 1). Overall, despite similar topographic, hydrologic, and soil conditions among the watersheds located in the same landform region, soil erosion rates differ because of where row cropping occurs. Cropping fewer acres in a watershed but on steeper slopes will generate more soil erosion per crop area compared to greater cropping intensity on gentler slopes. Further, agricultural management decisions involving soil tillage have large impacts on the RUSLE model by altering the C factor. The range of soil erosion rates in our study watersheds provides valuable insight into the current status of row crop management and helps to determine the watershed's potential for future reductions in sediment export.

While our estimated values for soil loss may appear high, the eastern Iowa values are comparable or lower than many agriculturally managed regions throughout the United States and the world. In 2005, researchers in China estimated that only 60% of similarly sized watersheds had RUSLE estimated soil loss at or below 10 Mg ha^{-1} y^{-1} (Chen et al. 2011). Some watershed areas in China experienced erosion rate estimates of up to 150 Mg ha^{-1} y^{-1}. Another study in a smaller watershed in China reported soil loss estimates via RUSLE to be between 26 and 52 Mg ha^{-1} y^{-1}, but noted that erosion could be greatly reduced by the incorporation of conservation practices (Shi et al. 2004). However, when comparing the average soil loss for our study areas to the average soil loss for all of the row crop agricultural land in Iowa, our study watersheds were much higher (~10 compared to 4.3 Mg ha^{-1} y^{-1} for our study watersheds and the entire state, respectively [Iowa Department of Natural Resources 2018a]). These data show that while the English River and Rapid Creek watersheds may be performing well in terms of soil loss estimates compared to other regions of the world, there is still much work to be done in order to build the resiliency of the soil within these watersheds.

Sediment Delivery. In 1998 the USDA published a report on "Erosion and Sediment Delivery," which provided an equation-based approach to estimate SDRs in Iowa (USDA NRCS 1998). The SDR equations for sheet and rill erosion were based entirely on a nomograph whose parameters primarily included watershed size, shape, and drainage network path (figure 5). Little is known about the specific methodology that led to the equation, but it appears to have been derived from soil erosion and sedimentation data collected in the 1960s by the Soil Conservation Service in Iowa (Johnson and Moldenhauer 1970). Despite the lack of provenance, the graph continues to be used extensively in Iowa watershed studies to estimate nutrient load reductions by agricultural practices (Mallarino et al. 2002; Vadas et al. 2009). Using this graph, the four study watersheds considered herein would have been expected to have SDRs ranging from ~25% (English River and Rapid Creek) to ~30% (Rapid Creek subbasins) (figure 5). However, in this study, we estimated TSS export using continuous turbidity measurements (English River and Rapid Creek subbasins) and TP data (Rapid Creek) calibrated to Old Mans Creek and compared estimated TSS loads to watershed-scale RUSLE soil erosion to calculate an annual SDR for the basins. Based on these methods, we estimated the SDRs to be 8.0%, 2.4%, 1.1%, and 1.0% for the English River, Rapid Creek, Rapid Creek US, and Rapid Creek DS watersheds, respectively (figure 6), values that are approximately 1/30th to 1/3rd of the USDA estimates. Although the accuracy of the original graphs were considered by Johnson and Moldenhauer (1970) to be of low accuracy (actual values ranged from 30% to 300% of those reported), our current data present an exceptional story of conservation progress in our study watersheds over the past four decades showing SDRs far below the range of error described by Johnson and Moldenhauer (1970). SDRs were substantially lower for the Rapid Creek watershed (including the subbasins) than the English River. With less row crop in the English and Rapid Creek watersheds, we expected the SDR for the English River and the entire Rapid Creek watershed to be the same or less than the subbasins. This would have been consistent with the 1998 graph (figure 5), which suggested that larger drainage areas have lower SDRs (Johnson and Moldenhauer 1970; USDA NRCS 1998). We hypothesize that the higher SDR found in the English River compared to the smaller subbasins may be due to a greater fraction of TSS loads originating from concentrated flow, stream banks, and bed load rather than from agricultural fields. Beck et al. (2018) estimated that 4% to 44% of stream sediment loads were derived from stream bank erosion in the same landform region of Iowa. Further, ephemeral gully erosion was found by Poesen et al. (1996) to represent 44% of total sediment production in small cultivated watersheds in Spain and Portugal. Hence, we consider the SDRs estimated in this study to be a worst-case scenario because we assigned all the TSS export to sheet and rill erosion when as much as 50% or more of the TSS load may be due to bed, bank and gully erosion (Schilling et al. 2011). Still, we also recognize that not all sheet and rill eroded sediments are found in streams as suspended sediments. Many of Iowa's stream banks are composed of upland sediments deposited in the flood plain as postsettlement alluvium and should in some way be considered as part of the delivery equation. Likewise, bed load measurements should be considered in the watershed-scale SDR, but they are rarely made in the US Midwest and there are no standard procedures and equipment to sample bed load accurately for different types of streams (Bhowmik et al. 1986). Nakato (1981) estimated that the bed load of a tributary streams to the Mississippi River ranged from 6% to 26% and averaged approximately 11% of the total suspended load, whereas bed load ranged from 1% to 2% of the total yearly suspended load in the Kankakee River in Illinois (Bhowmik et al. 1980). Although we focus exclusively on suspended sediment

Figure 5

1970s nomograph showing estimated sediment delivery ratios for landform regions. Note that the nomograph is displayed in acres rather than hectares. Landform regions are (1) Loess Hills, (2) Southern Iowa Drift Plain, (3) Iowan Surface, and (4) Des Moines Lobe.



in this study, we cannot rule out the contribution of bed load to sediment delivery and believe it to be a measurable, but small component of sediment export from Iowa watersheds. If we consider that some upland eroded sediments are being transported in the river as bed load, our SDR estimates may increase to a degree. However, if we assume that the fraction of sheet and rill erosion being transported as suspended sediments compared to the fraction being transported as bedload remains consistent over time, our methods will provide a baseline for future changes in SDR rates.

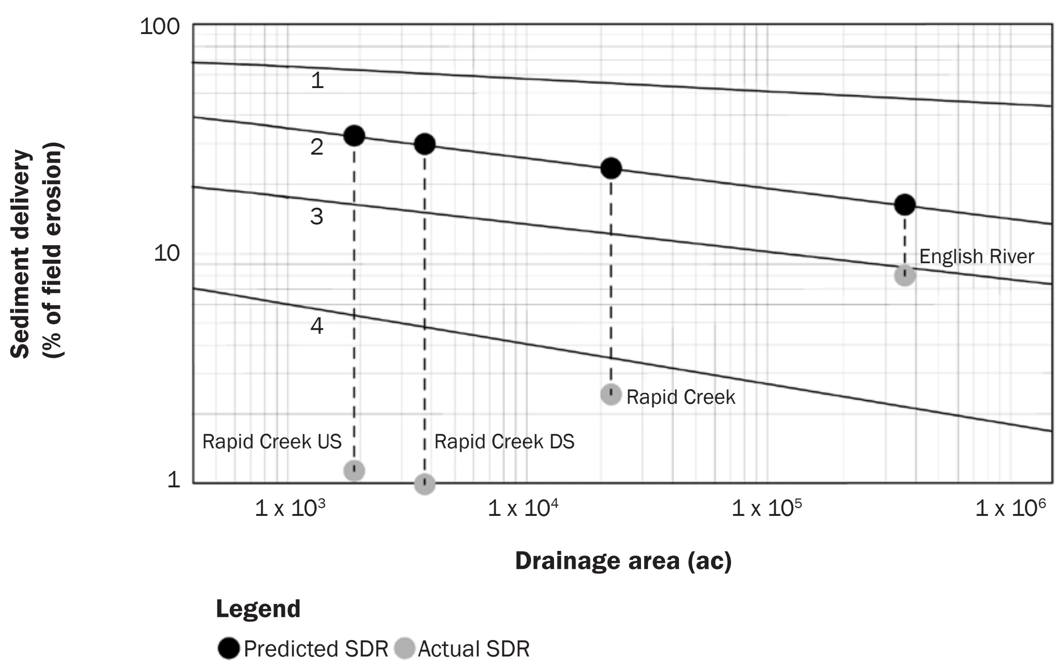
Conservation Practices. The final objective in our project was to evaluate the effects of current and future conservation practices on TSS export and SDRs. A recent study that utilized the same Rapid Creek US subbasin estimated the SDR to be 3.7% in 2015 (Streeter et al. 2018). In this study we found the SDR to be even lower based on 2017 to 2018 data (1.1%; table 4). Although Streeter et al. (2018) did not account for the implementation of BMPs that reduce soil erosion estimates, the Rapid Creek US subbasin had minimal BMPs in place that impacted only 1% of the agricultural area. The Rapid Creek DS subbasin had several more BMPs, but agricultural area impacted was still low (10%) (table 3). The major reason for lower SDRs in the Rapid Creek US subbasin in 2017 to 2018 was substantially less TSS export measured in the stream. In 2015, 193 Mg of TSS were exported from the US subbasin, whereas in 2017 and 2018 only 82 Mg were exported in total (approximately 40 Mg y^{-1}). This suggests that caution should be used in extrapolating TSS export from a watershed based on a single year of data. In this case, the multiple years of TSS data from a single watershed all indicated a low SDR for the Rapid Creek US subbasin (1.5% to 3.7%) even though the exact SDR was unknown with certainty.

In addition, our study using turbidity-based TSS estimates provides valuable insight about how dynamic these transport processes can be and reiterates the necessity for real-time data collection. Lu et al. (2006) determined that the ratio of sediment residence time to rainfall duration was a primary contributor for controlling rates of sediment delivery. The USDA's report on erosion and sediment delivery does not account for rainfall duration, but rather focuses primarily on watershed shape, size, and drainage network length (USDA NRCS 1998). Our study methods indirectly accounted for rates and duration of rainfall by measuring real-time turbidity within the stream as well as stream stage. In doing so, we were able to account for actual changes in stream discharge and sediment export, which accounts for some of the differences in our SDR estimates compared to the USDA report.

Although the SDRs were lowest in the entire Rapid Creek watershed, the ACPF toolbox suggested that this watershed area has greater potential for future BMP implementation compared to the English River. The ACPF toolbox identified an additional 158 potential BMP locations for Rapid Creek, which would increase the impacted agricultural area from 5% to 66%. The area of potential BMP impact was even greater

Figure 6

1970s nomograph showing predicted sediment delivery ratios (SDR, black) and our new actual estimates (gray) for each study watershed. Note that the nomograph is displayed in acres rather than hectares. Landform regions are (1) Loess Hills, (2) Southern Iowa Drift Plain, (3) Iowan Surface, and (4) Des Moines Lobe.



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in the Rapid Creek subbasins where the ACPF toolbox indicated that BMPs could increase impact areas from 1% to 83% and from 10% to 86% for the US and DS subbasins, respectively (table 3). In contrast, in the English River watershed, we identified over 19,000 existing BMPs that work to reduce sediment loss. However, these existing BMPs only impacted 16% of the land area in agricultural production. Results from the ACPF toolbox suggested the addition of nearly 50,000 more practices that would bring the impact saturation up to 65% of the agricultural land area. A recent study in the same English River watershed sought to make similar comparisons of existing BMPs and potential BMPs identified with the ACPF toolbox (Rundhaug et al. 2018). In that study, Rundhaug et al. (2018) compared the current and potential implementation of WASCOBs near the headwaters of the English River watershed and found that 12.2% of the total land area had WASCOBs implemented compared to a potential total land area of 16.5%. In our study we did not explicitly differentiate among WASCOBs, contour buffer strips, and contour terraces, but when focusing on WASCOBs only for the entire English River watershed, we found the current BMP saturation of this practice (4%) to be less than the ACPF potential saturation (13%). The ratio of existing WASCOBs to potential locations was lower in our study because we evaluated the entire watershed and not just the headwater areas as done by Rundhaug et al. (2018), which was approximately 10 times the area evaluated.

Despite the significantly greater number of BMPs installed in the English River, which impacted 9% of the total watershed area compared to only 2% of the Rapid Creek watershed and 1% of the Rapid Creek US subbasin, the SDR estimation in the Rapid Creek watershed and its subbasins suggested TSS export levels that were approximately 25% of the English River. Although the SDRs were lowest in the Rapid Creek watershed, the ACPF toolbox identified this area to have the greatest potential for future BMP implementation over the English River.

Implications for Future Management. BMPs like contour buffer strips, WASCOBs, and contour terraces have many potential environmental benefits including improved surface water and groundwater quality (Bracmort et al. 2006; Gassman et al. 2010; Schilling 2000), increased soil trapping (Streeter and Schilling 2019), reduced soil erosion (Lovell and Sullivan 2006; Lowrance et al. 2002), and improved wildlife habitat (Fischer and Fischenich 2000). However, when considering their effects on TSS export as indicated by SDRs, increased BMP implementation may not be the most appropriate management practice. For the English River watershed, current average soil erosion was estimated to be 8.0 Mg ha^{-1} y^{-1}, current SDR was 8.0%, and agricultural land impacted by BMPs was 16%. The ACPF toolbox identified appropriate locations for nearly 50,000 additional BMPs in the watershed. If each of these BMPs were installed, the estimated soil erosion would decrease to 7.3 Mg ha^{-1} y^{-1}, SDR would remain largely unchanged (assuming a proportionate decrease in stream turbidity), and the agricultural land impacted by BMPs would increase to 65%. This expectation of no-change to SDR is due to the in-stream turbidity being heavily influenced by factors other than the soil erosion estimates modeled by RUSLE since a large (but unknown) fraction of TSS in streams is derived from concentrated flow and stream bank erosion (Prosser et al. 2000; Schilling 2000; Shields Jr et al. 1995). Neither of those types of erosion may be significantly reduced by the BMPs that we considered for this study. A similar result can be seen when comparing current BMP implementation in Rapid Creek compared to those suggested by the ACPF toolbox. In Rapid Creek, the current soil erosion was 7.6 Mg ha^{-1} y^{-1}, the current SDR was 2.4%, and the current agricultural land impacted by BMPs was 5%. If all of the ACPF toolbox practices were implemented, soil erosion would decrease to 7.0 Mg ha^{-1} y^{-1}, SDR would remain largely unchanged, and agricultural land impacted by BMPs would increase to 66%. Even though these structural BMPs trap significant amounts of in-field sediments (USDA NRCS 2002), increased implementation of in-field BMPs may not be the most appropriate management decision for significantly decreasing TSS in streams. Instead we must look more closely at stream bank vulnerability and stabilization as well as practices that work to reduce soil erosion due to concentrated flow.

Summary and Conclusions

The primary goal of this study was to quantify TSS export and current SDRs in four southern Iowa watersheds and evaluate how existing and potential BMPs have affected SDRs. Our study updated estimates of SDRs that were previously developed using mid-20th century data and largely unknown methods (USDA NRCS 1998). Herein we estimated TSS export using continuous turbidity measurements (English River, Rapid Creek subbasins) and TP data (Rapid Creek) and measured discharge data to calculate TSS loads. By comparing annual TSS export to watershed-scale soil erosion estimated with the RUSLE model, we calculated annual SDRs for the basins and found that current SDRs for the four basins were significantly lower than previous estimates. We estimated current SDRs to be 8.0%, 2.4%, 1.1%, and 1.0% for the English River, Rapid Creek, Rapid Creek US, and Rapid Creek DS watersheds, respectively. This new analysis provides an exceptional story of conservation progress in our study watersheds over the past four decades. Further, they are likely a worst-case scenario for sheet and rill eroded sediment export since TSS export does not distinguish among other sediment sources such as stream bank and gully erosion. Based on the extent of BMP implementation in the watersheds and the potential for future BMPs determined using the ACPF toolbox, we found that there is only limited potential for further reducing TSS export using additional in-field practices. Hence, we believe that further work toward reducing TSS export in these Iowa watersheds should be shifted to reducing contributions from other TSS sources including from streambed, bank, and gully erosion.

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