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Quantifying Potential Long-term Changes in Erosion, Discharge, and Total Suspended Solids Resulting from Agricultural Land Use Change in South Dakota

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QUANTIFYING POTENTIAL LONG-TERM CHANGES IN EROSION, DISCHARGE, AND TOTAL SUSPENDED SOLIDS RESULTING FROM AGRICULTURAL LAND USE CHANGE IN SOUTH DAKOTA

BY

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This dissertation is approved as a creditable and independent investigation by a candidate for the Doctor of Philosophy in Biological Sciences degree and is acceptable for meeting the dissertation requirements for this degree. Acceptance of this does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

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This dissertation is dedicated to Diana Menendez.

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ABSTRACT

QUANTIFYING POTENTIAL LONG-TERM CHANGES IN EROSION, DISCHARGE, AND TOTAL SUSPENDED SOLIDS RESULTING FROM AGRICULTURAL LAND USE CHANGE IN SOUTH DAKOTA

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South Dakota is a mosaic of grasslands, wetlands, and cropland. A continuing shift from grassland to cropland has occurred over the past decade and is expected for the next 50 years. Rate of future conversion may vary greatly in response to regulatory, economic, and social factors. Concern has risen over environmental consequences associated with land conversion, which include but are not limited to changes in rill and sheet erosion rates from cultivated soils, stream and river discharge, and water quality. Quantifying future changes for these three externalities is important to understand the possible long-term consequences of complex grassland conversion decisions such as soil loss, flooding or drought, and diminished water quality. Systems Thinking and System Dynamics (SD) methodology was used to model complex land use and soil-related factors over time. The SD model replicated historic annual erosion rates (metric-tons/ha), discharge [million cubic meters (MCM)], and average total suspended solids (TSS; mg/L) from 1947 to 2012 with relative accuracy and precision in four South Dakota water-catchments, which included the Big Sioux, James, Bad, and Belle Fourche rivers. The SD model was utilized to forecast future annual and cumulative erosion [million metric-tons (Mt)], discharge (MCM), and TSS (mg/L) change under different potential future grassland conversion rates and conservation and conventional tillage from 2012 to 2062. Forecasted environmental externalities increased for policy scenarios that promoted grassland conversion but decreased for scenarios that limited grassland conversion to cropland or promoted grassland restoration. Policy implementation is likely to have the same general impact toward the reduction or increase of erosion, discharge, and TSS as cumulative estimates were $70 - 77\%$, $< 1 - 10\%$, and $70 - 76\%$ greater for the worst-case scenario compared to the best-case scenario estimates, respectively. However, externality change was greater in western verses eastern water-catchments. Results may provide producers, policymakers, and other stakeholders more specific quantitative estimates to assess the future impact of grassland conversion decisions. Additionally, comparisons between these estimates provide support that addressing grassland conversion issues and cultivation practices are important in order to preserve and conserve soil and water resources.

CHAPTER 1. INTRODUCTION

Since the 1900's, evolving farming technology (Dimitri, Effland, & Conklin, 2005) and ever-increasing grain demands (Clay et al., 2014) have accelerated the expansion of land conversion from grassland to cropland in the Midwestern U.S., and the rates of this type of land conversion have specifically increased in the past decade (Claassen, 2011; Clay et al., 2014). Wimberly and Wright (2013) found that rates of conversion from grassland to cropland in the Midwest between 2006 and 2011 (1.0-5.4% annually) were comparable to the deforestation rates in Brazil, Malaysia, and Indonesia (Lepers et al., 2005; Hansen et al., 2008). Worldwide, grassland conversion has been linked to increases in soil erosion rates, changes in hydrologic patterns, and decreased water quality (Bielders, Ramelot, & Persoons, 2003; Helmers et al., 2012).

One of the most noted consequences of grassland losses across the globe is an increase in soil erosion (Lal, 2004; Pimentel, 2000). Approximately 75 billion tons of topsoil are lost each year from global agriculture production, and roughly 6.9 billion tons of soil (9.2% of worldwide erosion estimates) are lost each year in the United States alone (Pimentel, 2000). Soil erosion may result from wind or water activity. Cultivated soil has less cover (e.g., plants and litter) and is more susceptible to wind energy, which increases the amount of soil particles that are dislodged and transported (i.e., creep, saltation, and suspension), sometimes over thousands of miles (Pimentel and Kounang, 1998; Zhang, Zhang, Chang, Wang, & Liu, 2017). One example of wind erosion is the U.S. Dust Bowl Era with an estimated loss of 14 billion metric tons of topsoil between 1932 and 1939 (Bolles, Forman, & Sweeney, 2017). Erosion by water can be sheet or rill erosion or both and occurs at the highest rates during intense rainfall events (Larson, Lindstrom, & Schumacher, 1997). Sheet erosion is a uniform removal of soil in thin layers and rill erosion is water concentration in streamlets or head cuts (Horton, 1945). Both sheet and rill erosion may lead to reduced nutrient uptake by plants, decreased rooting depth, diminished water-holding capacity of soils, and increased runoff over time (O'geen & Schwankl, 2006).

Similar to erosion, hydrologic processes are impacted by grassland conversion to cropland. Lower soil permeability in cropland has been shown to reduce water infiltration by five times than that of grassland (Bharati, Lee, Isenhart, & Schultz, 2002; Gerla, 2007). Diminished plant water uptake (transpiration) and soil infiltration alters surface runoff, evapotranspiration rates, and baseflows of lotic systems within the watershed (Foley et al., 2005). Changes in hydrological processes may also reduce groundwater storage as accelerated runoff reduces subsurface water infiltration (Foley et al., 2005; Rosegrant, Cai & Cline, 2002). Consequently, stream and river flow regimes change from historic patterns and discharge typically increases as natural vegetation in riparian zones is cleared for anthropogenic use (Costa, Botta, & Cordille, 2003; Polyakov, Nichols, & Nearing, 2016).

Increased erosion coupled with hydrologic changes may lead to increased transport of sediment (sand, silt, and clay particles) by overland flow into streams and rivers, which then end up either suspended or deposited in waterways (Langendoen, Simon, Klimetz, Bankhead, & Ursic, 2012; Morrissey, Rizzo, Ross, & Alves, 2011; Santos, Andrade, Medeiros, Guerreiro, & Palácio et al., 2017; Stryker, Wemple, & Bomblies, 2017). Sedimentation is a naturally occurring event in stream and river morphological processes (Leopold, Wolman, & Miller, 1964) and is most influenced by flow velocity, whereby larger sediments are transported at greater rates under higher velocities and settle out of the water column at lower velocities (Waters, 1995). Grasslands converted for agricultural use can lead to alterations of field surface slopes and stream gradients, making field surfaces and stream gradients more susceptible to erosion by water, which further induces deposition of sediment in waterways (Lowdermilk, 1953; Trimble, 2008). Over time, sediment transportation and deposition may increase the amount of total suspended solids (TSS) in the water column, which reduces water quality. Excessive sedimentation may lead to additional environmental consequences that may cascade to further impacts. For example, sedimentation may decrease light penetration in water bodies (Irving & Connell, 2002), which changes aquatic plant communities (Mahaney, Wardrop, and Brooks, 2005) and alters nutrient cycling processes (Irving & Connell, 2002), which, in turn, may alter animal communities in those systems (Bartelet, 2016;). Anthropogenically induced sedimentation in waterways may also have other consequences to society, including decreases in storage capacity of reservoirs, rivers, and streams and increases in flooding frequency and intensity (Cakula, Ferreira, & Panagopoulos, 2012; Santos et al., 2017).

Presently, South Dakota is one of the states in the U.S. where grassland-to-row crop conversion rates are the highest (Claassen, 2011; Clay et al., 2014; Wright and Wimberly, 2013). South Dakota is roughly bisected longitudinally by the Missouri River (Figure 1), and precipitation, geology, topography, and consequently, land use differ between the eastern and western portions of the state. Eastern South Dakota is primarily within the Prairie Pothole Region (PPR) and receives an annual average of 50 – 60 cm of precipitation (Hubbell, Stevens, Skinner, & Beverage, 1987). The PPR was created during Cenozoic period when expanding and receding glaciers deposited sediments and formed kettles (i.e., potholes) throughout the region (Samson & Knopf, 1994; see http://www.sdgs.usd.edu/geologyofsd /geosd.html for map). Historically, this area was used for grazing livestock, but now all but 2,220,925 hectares of the once native prairie has been converted to cultivated land (*Zea mays*, *Glycine max*, and *Tricticum aestivum*; (Bauman, Carlson, & Butler, 2016; Samson & Knopf, 1994). Western South Dakota is relatively drier and receives 30 − 40 cm of precipitation annually (Hubbell et al., 1987; Pieper, 2005). The geology of this region is composed of older Mesozoic sediments, including eroded clay, shale, and sandstone (see http://www.sdgs.usd.edu/ Geologyofsd/geosd.html for map). The landscape is composed of rolling hills, eroded stream valleys, and the Black Hills, and most of the land use is primarily for rangeland (USDA, 2006). Thus, South Dakota is unique in soil, topography, and climate and provides an opportunity to study how various soil types and watersheds respond to such change.

Changes in land use in both western and eastern South Dakota may be contributing to externalities related to erosion, hydrologic regimes (discharge), and water quality (namely, TSS) as other areas of the globe that have experienced similar land conversion. Externalities are defined as the consequence of one activity (in this case, grassland conversion) to a group that was not involved in the original process, such as extreme runoff (e.g., downstream residents who may experience increased flooding, decreased reservoir storage, or poorer water quality; Buchanan & Stubblebline, 1962; Lafont, 2008). Recent work in the Northern Great Plains (NGP) indicates that there is some concern of soil and water externalities associated with grassland conversion to cropland (Turner et al., 2016, 2017). Turner et al. (2016, 2017) modeled various policy, cultural, and economic scenarios that influence cropland expansion rates in the NGP. With each of these scenarios, future forecasts indicate that soil externalities may improve, stay the same, or worsen. These potential externalities were previously quantified by Turner et al. (2016, 2017) using a dimensionless index called Soil Environmental Risk (SER), but uncertainties exist as to how the externalities captured in this index will be realized on the landscape, particularly in regard to erosion rates, hydrological changes, and water quality (TSS).

Combining forecast grassland conversion scenarios to model future erosion, water quantity, and water quality externalities is a complex process. Turner et al. (2016, 2017) used a Systems Thinking and System Dynamics approach to model grassland-to-row crop conversion in the NGP and the associated SER consequences of various scenarios that may result from such conversion rates. Thus, I am using the same approach to specifically quantify externalities associated with SER. Meadows (2008) defines a system as "a set of things—people, cells, molecules, or whatever—interconnected in such a way that they produce their own pattern of behavior over time." Systems Thinking has been used to investigate complex problems (Sterman, 2000) and allows exploration of the underlying structure of a system (e.g., grassland conversion); System Dynamics then builds a model describing how the structure of a problem creates patterns of behavior over time (e.g., historic erosion rates). Systems Thinking and System Dynamics differ from the traditional scientific method in that the standard methods tend to be more linear (Figure 2) and often do not account for feedback within a system (Figure 3). Feedback can be described as symptoms, actions, and solutions that are not isolated in a linear fashion but rather exist in cause-and-effect relationships within systems, forming links known as feedback loops (Senge, 1990). Additionally, Systems Thinking and System Dynamics has the ability to integrate large amounts of data and diverse information and provides a quick and user-friendly interface to experiment with alternative scenarios (policy testing) and generate forecasts (Ahmad & Simonovic, 2004; Balali & Viaggi, 2015; Forrester, 1961; Rodrigues & Bowers, 1996; Sterman, 2000). Thus, Systems Thinking and Systems Dynamics offers an opportunity to specifically quantify and predict externalities related to grassland-to-row crop conversion by combining various factors that contribute to land cultivation decisions over time with specific models that quantify soil erosion rates, hydrological changes, and associated changes in TSS.

Potential externalities from grassland conversion are likely to vary across South Dakota primarily from the differences in soil, topography, and climate between the eastern and western portions of the state as well as differences in land use decisions (e.g., farming versus ranching). Therefore, four unique South Dakota water-catchments the Big Sioux, James, Bad, and Belle Fourche rivers were selected as the study area(s) to represent differences in soil, topography, climate, and land use throughout the state. Specific quantification of estimated changes of my three selected externalities in each of the four water-catchments identified aides in further evaluating the risk of accelerated grassland conversion now and into the future, especially as economics, policies, and culture continue to change. In order to address these potential environmental externalities, I will be addressing two specific focusing questions:

1.What are the possible changes to erosion rates, hydrologic regimes, and water quality (as indicated by TSS) that may occur in South Dakota lotic systems as a result of conversion from grassland to cropland agriculture in the future?

2.What effects might changes in policy and tillage type have on reducing or exacerbating those changes in erosion rates, hydrologic regimes, and water quality in South Dakota lotic systems in the future?

The focusing questions address potential erosion, hydrologic, and water quality externalities that are indicative of rapid changes in grassland conversion to agriculture production. Unknown risk of future changes in erosion, hydrologic regimes and water quality externalities merits the investigation of future environmental impacts of grassland conversion scenarios within the study area (Figure 1). Therefore, the specific objectives this study were to: 1) provide a detailed quantitative evaluation of the potential environmental consequences of land use change in South Dakota, specifically forecasting soil erosion, water quantity and water quality changes for the next 50 years (2012-2062); 2) evaluate the potential impacts that various policy and tillage decisions may have on the three externalities. The results of this study could provide more specific information to guide policy and tillage decisions or inform stakeholders about the potential environmental consequences of grassland-to-row crop agriculture in South Dakota and the NGP.

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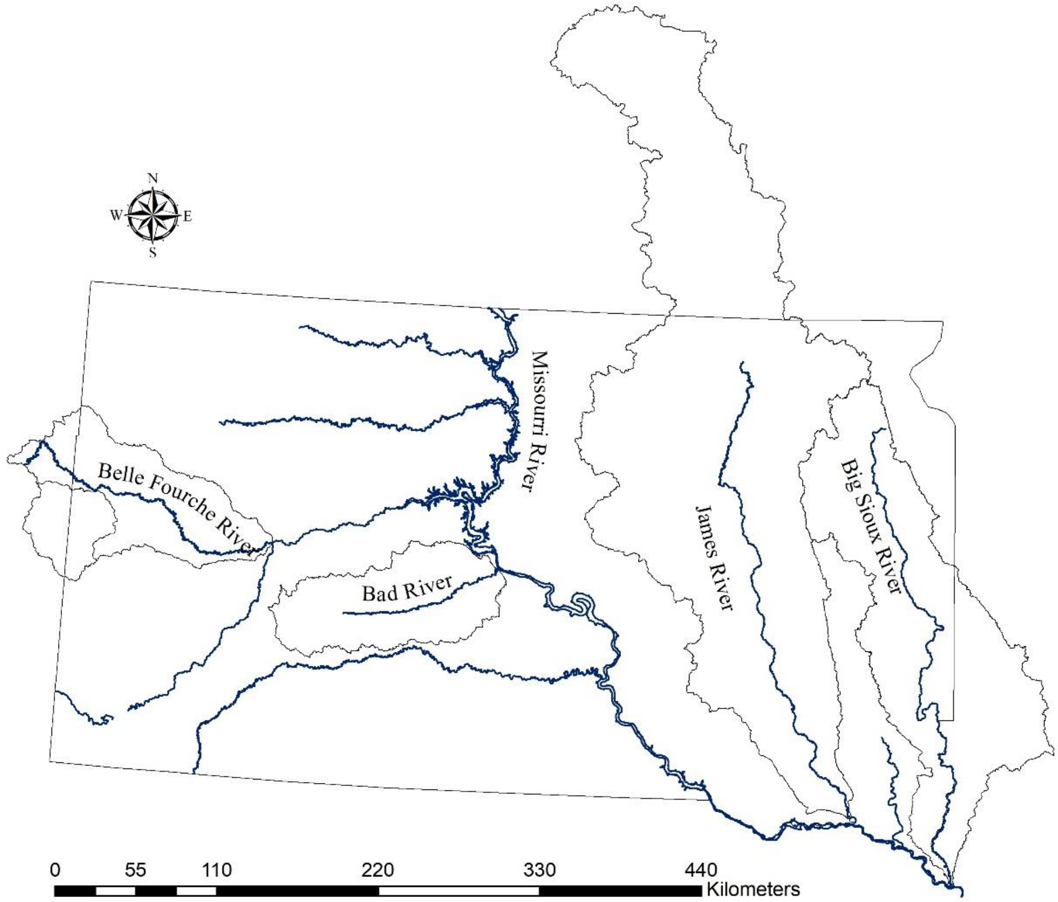

Figure 1. Map of the state of South Dakota, USA, with the four watersheds included in this study Big Sioux River (22,910 km^2), James River (54,742 km^2), Bad River (8,225 km^2), and Belle Fourche River (11,129 km^2).

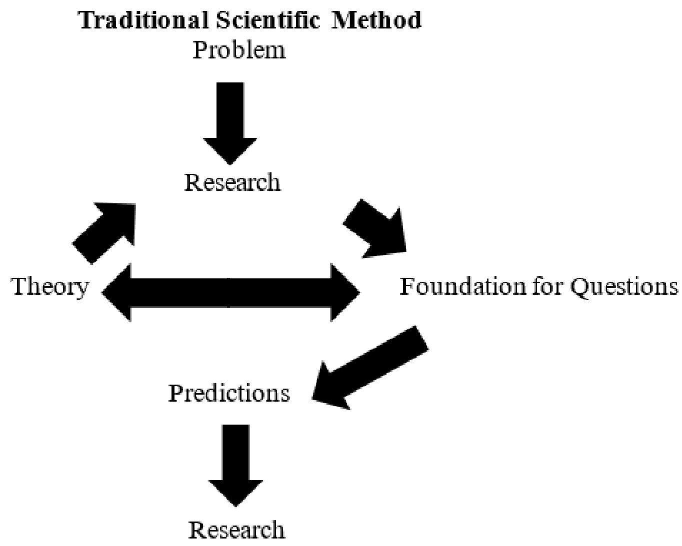

Figure 2. General outline of the scientific method (modified from Ford, 2000; Garton Ratti, & Giudice, 2005).

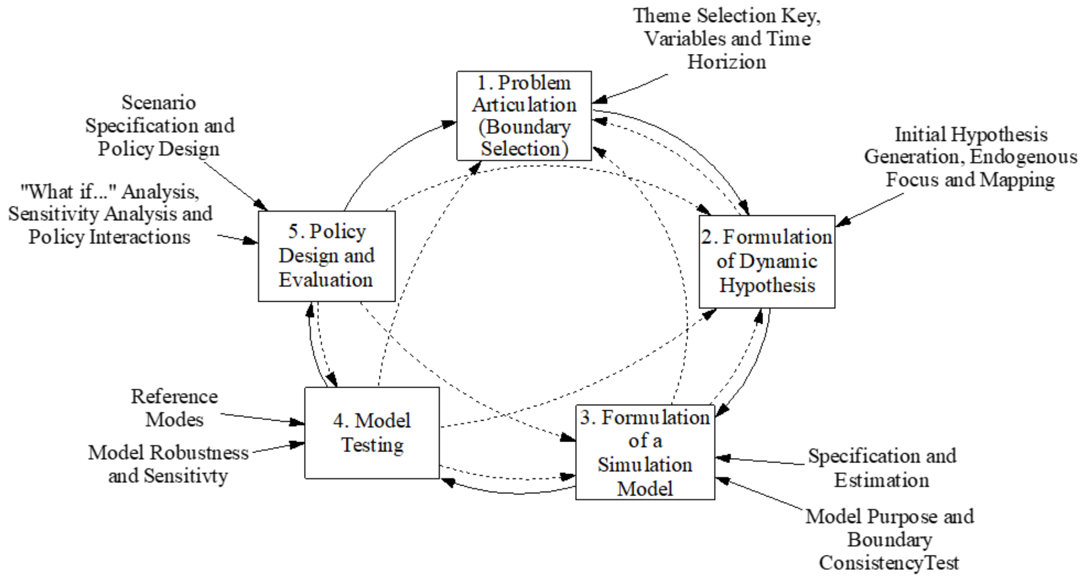

Figure 3. Summary of the iterative SD modeling process (steps 1-5 connected by solid lines). The point-ins describe activities performed at that step. Results in any one-step may yield insights that lead to revisions of earlier ones (dashed lines; adapted from Turner, 2016; and from Sterman, 2000).

CHAPTER 2. A SPATIAL LANDSCAPE SCALE APPROACH FOR ESTIMATING EROSION, WATER QUANTITY, AND QUALITY IN RESPONSE TO SOUTH DAKOTA GRASSLAND CONVERSION

INTRODUCTION

Accelerated land conversion to cultivated landscapes is being driven, in part, by increased demands for agricultural commodities as a result of an increasing global population (de Ruiter et al., 2017; Haberl, 2015). Expansion of land conversion has diminished grasslands worldwide, and rates of grassland conversion to row-crop agriculture have accelerated within the Northern Great Plains (NPG) region of the United States and Canada during the past decade (Foley et al., 2011; Ramankutty, Evan, Monfreda, & Foley, 2008; Ramankutty & Foley, 1999; Wimberly et al., 2017). Grassland conversion rates vary by province, state, or region within the NPG (Claassen, 2011; Wimberly & Wright, 2013). Conversion may lead to some environmental consequences such as changes in soil erosion rates, altered hydrologic flow in streams and rivers, and impairments to water quality, and such consequences may be more severe in grasslands that are typically considered to be less suitable for row crop agriculture (Claassen, 2011; Foley et al., 2005; Helmers et al., 2012; Lowdermilk, 1953; Wimberly et al., 2017).

One of the most studied consequences of grassland losses across the globe is an increase in soil erosion (Lal, 2004; Pimentel, 2000). Approximately 75 billion tons of topsoil are lost each year from global agriculture production, and roughly 6.9 billion tons of soil (9.2% of worldwide erosion estimates) are lost each year in the United States alone (Pimentel, 2000). Grasslands that have been tilled for the purposes of row crop agriculture have been found to increase wind and water erosion rates (Pimentel et al., 1995). An estimated 14 billion metric tons of topsoil were lost due to wind erosion between 1932 and 1939 during the Dust Bowl in the U.S. Great Plains, an environmental disaster strongly related to rapid rates of grassland-to-row crop conversion (Bolles, Forman, & Sweeney, 2017; Hansen & Libecap, 2004; Joel, 1937). Similarly, Lindstrom, Schumacher, Cogo, and Blecha (1998) estimated an increase in water erosion from 0.0 ton/ha to 6.7−18.2 tons/ha on recently converted grassland plots located within the NPG near White, South Dakota, subjected to simulated rainfall. Further, SooHoo, Wang, & Li (2017) found that erosion potential increased by 4% to >33 ton/ha/yr in response to a 15% increase in grassland conversion to row crop agriculture in the Missouri River Basin, which includes a large area of the NGP.

Land use changes worldwide, including, but not limited to grassland conversion, have also been shown to change hydrologic patterns and increase runoff rates by 6.8%, potentially altering the frequency and intensity of flood and drought events (Sterling, Ducharne, & Polcher, 2013). The conversion of forests and grasslands attributed to significant flooding of the Yangtze River in China in the late 1980s (Wenming, Landell-Mills, Jinlong, Jintao, & Can, 2002; Qiu, Yin, Tian, & Geng, 2011). Grassland conversion to cropland has altered hydrologic function (i.e., evapotranspiration, streamflow variation, and runoff) within the U.S. Great Plains (Dale et al., 2015; Gao, Sheshukov, Yen, Kastens, & Peterson, 2017; Krueger, Yimam, & Ochsner, 2017). Additionally, conversion of perennial grassland to cropland in the U.S. Midwestern Corn Belt has increased surface water runoff from an average 84 mm (1995) to 91 mm (2004, 10%); this relationship has inversely impacted evapotranspiration (10% decrease) throughout the 20th Century (Schilling, Jha, Zhang, Gassman, & Wolter, 2008).

Additionally, Lindstrom (1988) estimated that water runoff increased from 0 to 66% when precipitation was simulated on grasslands converted to cropland by moldboard plow within the NGP.

Changes in erosion rates and hydrology influence water quality of lakes, streams, wetlands, and aquifers within water catchments, particularly in the amount of suspended solids captured in streams and rivers (Foley et al., 2005; Lowdermilk, 1953; Strauch, Lima, Volk, Lorz & Makeschin, 2013). Agricultural expansion and conversion of native vegetation that occurred between 1963 and 2013 have been linked to increased levels of total suspended solids (TSS) in Brazil’s Pipiripau River Basin from 0 to 400 ton/day (Strauch et al., 2013). Within the U.S. Great Plains, TSS levels have been a concern in some areas where water-catchments that were formerly grasslands have changed to ones that are cropland dominated. For example, in the North Fork Ninnescah River and Cheney Reservoir of south-central Kansas, observed TSS levels more than doubled (250 mg/L) from the targeted TSS level standard (100 mg/L) between 1997 and 2003, partly due to precipitation driven surface water runoff from agricultural lands (Christensen, Graham, Milligan, Pope, & Ziegler, 2006).

Between 2006 and 2011, high rates of grassland-to-row crop conversion have been reported throughout the NGP, and South Dakota reported the highest rate of grassland-to-row crop conversion (1.0-5.4% annually) of any U.S. state within the NPG (Claassen, 2011; Clay et al., 2014; Wright & Wimberly, 2013). Several studies have noted changes in erosion rates, hydrology, and water quality in various water-catchments across the state, potentially due to these land use changes. Sishodia (2010) found that South Dakota soil water erosion rates increased from 0.9 to 28.7 ton/ha during peaks in grassland conversion rates as grasslands previously enrolled in the Conservation Reserve Program (CRP; a federal program designed to keep highly erodible soils out of production for 10-15 years) were converted to row-crop production. Cropland expansion in the Big Sioux River water-catchment has been linked to increases in mean annual surface runoff ($2 - 4\%$; Neupane & Kumar, 2015). Furthermore, estimations of annual sediment load have been shown to worsen by at least 7% compared to historical levels, following grassland conversion to cropland in eastern South Dakota that occurred between 1994 and 2014 (Hong, 2017). Environmental consequences related to grassland conversion to row-crop agriculture across South Dakota present real issues that need to be understood as land use decisions continue to be made.

Previous research indicates that understanding and estimating the environmental consequences of grassland conversion is complex and challenging to evaluate with certainty (Kibria, Ahiablame, Hay, & Djira, 2016; Strauch et al., 2013; Paul, Rajib, & Ahiablame, 2017). However, one approach that is well suited to handle the complexity of this issue is Systems Thinking (ST) and System Dynamics (SD; Forrester, 1961, 1990; Meadows, 2008; Sterman, 2000). Systems Thinking is an approach to understand complex systems. Meadows (2008) defines a system as “a set of things—people, cells, molecules, or whatever—interconnected in such a way that they produce their own pattern of behavior over time.” System Dynamics is the approach to model such complex problems within a system by accounting for complex dynamic feedback between variables over time and capturing the important drivers of a system’s behavior (Sterman, 2000).

Systems Thinking and SD have been previously applied to complex erosion, hydrologic, and water quality problems around the world. Erosion has been modeled using ST and SD in both Taiwan's Keelung River Basin (Yeh, Wang, & Yu, 2006) and Portugal's Alqueva Dam/reservoir water-catchment (Cakula, Ferreira, & Panagopoulos, 2012) to evaluate changing land use for urban development and agriculture over long periods of time. Hydrologic ST and SD models include the assessment of Iran's water limited Zayandeh-Rud River Basin (Madani & Marino, 2009), the modeling of snowmelt and flood management in Canada's Red River Basin (Ahmad & Simonovic, 2004), and the management of Idaho's water-dependent agricultural and energy systems within the Snake River Basin (Jeffers, 2013). Additionally, ST and SD have been used in the assessment of rural community shifts, irrigation management, and climate change in the headwater stream irrigation networks of New Mexico (Fernald et al. 2012; Turner et al. 2016a). Water quality issues have also been modeled using ST and SD to evaluate how changes in TSS may influence the Philippines' fragile coral reef and aquatic ecosystems (Bartelet, 2016). Additionally, SD models have been used to assess changes in sediment loading levels resulting from land use alterations and agricultural production in Taiwan (Yeh et al., 2006) and Portugal (Cakula et al., 2012).

Recent research in the NPG employed ST and SD to evaluate the potential consequences of accelerated grassland conversion and indicated that continued grassland conversion might increase risks to the environment (Turner et al., 2016b & 2017). Turner et al. (2016b) developed a soil environmental risk (SER) index which indicated that soil externalities, such as erosion or flooding severity, in the past, present, and future were related to various policy, economic, and social scenarios that altered the total number of grassland acres in production. In short, continued cropland expansion was found to increase SER values while decreased grassland conversion would reduce potential environmental risk (Turner et al., 2017). Although SER was a dimensionless index that was not able to measure specific soil and hydrologic responses unique to specific watershed (Turner et al., 2016b, 2017), SER estimates have corresponded to noteworthy erosion events and hydrologic regime changes where land use conversion has shown to be statistically significant (Turner et al., 2018).

The goal of this study is to quantify erosion, hydrologic, and TSS changes as a result of grassland-to-row crop conversion in four South Dakota water catchments using ST and SD. In order to build confidence in the resulting sub-models (i.e., three specific models that are within the SD model), each sub-model required calibration, rigorous testing, and evaluation. Here, I describe the construction process of the erosion, hydrologic, and TSS sub-models, the results of the calibration procedures and model tests, and compare predicted model results with historical data.

METHODS

Study Area

Four water catchments – the Bad, Belle Fourche, Big Sioux, and James rivers – were selected within the boundaries of South Dakota, USA (Figure 4). South Dakota is roughly bisected longitudinally by the Missouri River, and precipitation, geology, topography, and, consequently, land use differ between the eastern and western portions of the state. The Big Sioux and James River water catchments are located in the eastern half of the state. Eastern South Dakota is primarily within the Prairie Pothole Region

(PPR) and receives an annual average of 50 – 60 cm of precipitation (Hubbell, Stevens, Skinner, & Beverage, 1987). The topography of the PPR was influenced by the expansion and recession of glaciers that deposited sediments and formed kettles (i.e., potholes) during the Cenozoic period (Samson & Knopf, 1994; see http://www.sdgs.usd.edu/geologyofsd /geosd.html for map). Historically, this area was used for grazing livestock; now all but 24% (2,220,925 ha) of the once native prairie has been converted to cultivated land (*Zea mays*, *Glycine max*, and *Tricticum aestivum*; Bauman, Carlson, & Butler, 2016; Samson & Knopf, 1994).

The Bad and Belle Fourche River water catchments are located within the western half of South Dakota. Western South Dakota is relatively drier than the eastern part of the state, receiving 30 – 40 cm of precipitation annually (Hubbell et al., 1987; Pieper, 2005). The geology of this region is composed of older Mesozoic soils, including eroded clay, shale, and sandstone (see http://www.sdgs.usd.edu/Geologyofsd/geosd.html for map). The landscape is composed of rolling hills, eroded stream valleys, and the Black Hills, and most of the land use is primarily for rangeland (Gries, 1996; Sayler, 2014). Higgins et al., (2002) reported a 1.4 million hectare (14%) loss of rangeland to cropland in western South Dakota from 1977 – 1997.

The four water-catchments selected for this study thus differ in geology and land use (Figure 4). The Big Sioux and James River water-catchments are predominantly composed of the Mollisol soil order and characterized by multi-year cycles of wetter and drier periods (Dozark, 2010; Miller & Gardner, 2001). Elevation ranges from 284 to 663 m in the Big Sioux River (Neupane & Kumar, 2015) and from 305 to 625 m in the James River. The topography of both water-catchments consists mainly of plains and gentle rolling hills (USDA, 2006) and land use is predominantly row-crop agriculture (Dozark, 2010; Fayyadh, 2011).

Climate in the Bad and Belle Fourche river water-catchments is semi-arid with periods of reoccurring drought. The main soil order of the Bad River water-catchment is Entisol, though concentrations of Inceptisols, Mollisols, and Vertisols are present (Miller, 2014). Soil orders of the Belle Fourche River water-catchment include Entisols, Alfisols, Vertisols, and Inceptisols, and small amounts of Mollisols (USDA, 2006). Each western water-catchment includes unique geological features. The Bad River water-catchment includes the Badlands formation comprised of rock formations, steep canyons, and spires, and the Belle Fourche water-catchment includes the Black Hills comprised of high plateaus and very steep drainageways on peaks and ridges (USDA, 2006). Elevation varies between 430 and 990 m within the Bad River water-catchment and 1,000 and 2,208 m within the Belle Fourche River (USDA, 2006). Approximately 83% of the Bad River is used for livestock grazing, while cropland composes 14% of the total area (Paul et al., 2017). Land use in the Belle Fourche River includes timber harvest (Ball & Schaefer, 2000), livestock grazing, silvopasture (66%, Garret, Rietveld, & Fisher, 2009), and alfalfa (*Medicago sativa)* and small grain crop production (4%, USDA, 2012).

Development of the System Dynamics Model

Complex land use changes and water-catchment characteristics were used to model associated erosion, hydrologic, and TSS systems by following SD methodology, which includes dynamic hypothesis formulation, model formulation, model calibration, and model testing (Sterman, 2000), each phase of which is described in the sections that follow.

Dynamic Hypothesis Formulation

The core structure of the dynamic hypothesis (DH) is created by linking key variables via the feedback processes that create and perpetuate the problem at hand. System Dynamics variables are categorized into two types: endogenous and exogenous (Table 1). Endogenous variables are those embedded within the feedback loops of the system, and exogenous variables are components whose values are not directly affected by the system (Albin & Forrester, 1997). For example, cattle nutrient requirements is considered an endogenous variable because body temperature may be influenced by changes in air temperature which alters energy requirements to maintain body temperature. Conversely, air temperature is considered an exogenous variable as air temperature was not influenced by any other model variable, such as nutrient or energy requirements. Dominant feedback loops (i.e., feedback relationships) were identified and interconnected with key equations within each of the three sub-models (i.e., erosion, hydrology, and TSS models) as they represented the over-arching drivers of change over time. Feedback loop structures may reinforce (increase; positive feedback) a behavior over time or balance (limit; negative feedback) a behavior over time within a model (Meadows, 2008). For example, animal populations can increase when food, water, and habitat are available (reinforcing) but a shortage of those resources will create a limiting action on population growth rates (balancing). Key dynamic feedback relationships between each sub-model within the DH highlight the shared endogenous variables that are responsible for change in the three sub-models and where sub-models can be joined to estimate changes in all three externalities (Figure 7; see Appendix A for the Dynamic Hypothesis statement).

Formulation of the Sub-Models

Erosion, hydrologic, and TSS systems and associated feedback were simplified into three separate sub-models in Vensim™ through the use of key equations from existing models (Gassman, Reyes, Green, & Arnold, 2007; Vanoni, 2006; Wischmeier & Smith, 1978). Erosion rates (metric-tons/ha/yr) were estimated using the Revised Universal Soil Loss Equation 2 [RUSLE2 and earlier versions (e.g., USLE; Wischmeier & Smith, 1978)]. Soil erosion by water, specifically aggregate rill and sheet erosion estimates from the RUSLE2, should not be confused with sediment deposition into a stream or river (Foster et al., 2002). Rather, these estimates are limited to soil erosion and movement from one landscape position to another, such as soil movement from a hilltop to a toe-slope position. Annual rill and sheet erosion (A) were estimated as:

$$A = R * K * LS * C * P$$

where R = rainfall erosivity factor; K = soil erodibility factor; LS = slope, length and steepness factor; C = vegetation cover factor; and P = conservation practice factor. Each factor, except for the vegetation cover factor, were endogenously incorporated into the erosion model. Typically, each RUSLE2 factor contains complex equations within themselves, but factors were simplified using established parameter values found in the literature (Franzmeier, Yahner, Steinhardt, & Schulze, 1986; Wischmeier & Smith, 1978). Specific parameter values for R, K, LS, and P were dynamically altered within the erosion sub-model according to land use change, soil type, topography, and climate (Cakula et al., 2012). The vegetation cover factor is a dimensionless index (range = 0.00 − 0.32) which attributes lower values to conservation tillage and higher values to conventional tillage (Wischmeier & Smith, 1978). This index was parametrized in the model using historic tillage trends for South Dakota from 1982 to 2012 (Miller, 2014; USDA, 2017). Erosion model equations were driven by data from several exogenous variables including daily rain, snowfall, and minimum and maximum temperature. All weather and climate data were obtained from the National Oceanic and Atmospheric Administration (NOAA; https://www.ncdc.noaa.gov/data-access, Table 2). Soil data [infiltration, Land Capability Classes (LCC), topography, and texture] were obtained from the National Resource Conservation Service (NRCS) Spatial Gateway (https://datagateway.nrcs.usda.gov). Annual land use data were obtained from the United States Geological Survey-Earth Resource Observation and Science (USGS-EROS) Center to represent land use type (ha/yr) in the Big Sioux, James, Bad, and Belle Fourche water-catchments from 1947 to 2012 (https://landcovermodeling.cr.usgs.gov/projects-.php). Specific crop production trends for corn (*Zea maize*) soybean (*Glycine max*), and spring/winter wheat (*Triticum aestivum* were provided at the county level from 1947 to 2012 by the U.S. Agriculture Census (https://www.agcensus.usda.gov/Publications-/2012/Full_Report/Census_by_State) and overlaid across each water-catchment (Figure 5, see Appendix B for additional information).

Similar relationships inform both the erosion and hydrologic models; thus, some data was shared between the two models. Surface runoff (m^3/day) was calculated through the use of a landscape-scale water balance equation similar to the water balance equation used by the Soil and Water Analysis Tool (SWAT; Chow, Maidment, & Mays, 1988; Gassman et al., 2007), which uses a combination of key hydrologic equations. Daily water runoff (Wr) was parameterized as:

$$\mathrm{Wr} = \mathrm{I} - (P_i + \mathrm{I}_w + S + ET) * RC$$

where, I = inflow, P_i = precipitation interception, I_w = water infiltration into the soil, S = seepage, ET = evapotranspiration, and RC = runoff coefficient. Inflow is the total amount of rain and snow entering each water-catchment each day (cm/day). Precipitation interception rate is defined as the percent of precipitation per daily precipitation event (cm/day) that does not reach the soil. This interception rate varies based on precipitation intensity, plant type, and plant growth stage; thus, the rate was calculated for each crop type as well as all grasses, in general, using seasonal leaf interception and plant growth stage relationships (Couturier & Ripley, 1973; Kang, Wang, & Liu, 2005; Ma, Gale, Ma, Wu, Li, & Wang, 2013; Ostrem et al., 2016). Plant growth (kg/ha) was calculated based on biomass and growth stage relationships which were regulated by daily plant available soil moisture (range = -1500 to -33 kPa) and temperature requirements (i.e., growing degree days) for plant growth and development (Miller & Gardner, 2001). Growing degree days (GDD) was calculated as:

$$GDD = \frac{\text{Maximum daily temperature } + \text{ Minimum daily temperature}}{2} - Base\ °C$$

where Base °C is the specific base temperature required for corn and soybeans (10°C) and wheat and grass (0 °C) germination (McMaster & Wilhelm, 1997).

Once precipitation interception components were parameterized to account for each crop type and grass and effective rain and snow (i.e., snow water equivalent; NRCS, 2017) that reached the soil surface, the precipitation was then infiltrated into the soil. Water infiltration is the rate (m^3/day) that water was absorbed into the soil profile, determined by daily soil water holding capacity of each soil type and its soil organic matter content. Effective precipitation infiltration was halted if the average daily temperature was below 0°C. In this case, the model assumed that the resulting water was stored above ground as either frozen water or snowpack. Once the temperature condition for infiltration was met (temperature ≥ 0°C), infiltration occurred (m^3/day) unless the first 0.3 m of soil was at field capacity. Field capacity limits (m^3/day) were altered by changes in percent soil organic matter. Soil organic matter levels were altered in relation to tillage type (i.e., conventional or conservation) and land use as crop production typically decreases soil organic matter while grasslands maintain or increase soil organic matter (Rhoton, 2000; Schipper et al., 2017). In general, a 1% increase in soil organic matter can increase soil water holding capacity by 60,567 L/ha (Overstreet & Dejong-Huges, 2009; Sullivan, 2000). Therefore, volumetric soil water holding capacity altered the daily infiltration rate needed to reach field capacity, which increased water infiltration rates and decreased surface runoff.

After water infiltration occurred, the infiltrated water was then converted to groundwater. The amount of total groundwater was influenced by groundwater seepage [percent daily loss of infiltrated water (m^3/day)] and evapotranspiration (mm/day). The Hargreaves method was used to calculate daily evapotranspiration:

$$ET_0 = K_{ET} \times RA \times TD^{0.50} (T + 17.8)$$

where ET_0 = evapotranspiration; K_{ET} = crop evapotranspiration coefficient; RA = extraterrestrial radiation; TD = mean temperature (℃); T = temperature (℃). Evapotranspiration rates were adjusted based on available groundwater and was calculated as:

$$E(s) = \begin{cases} E_w \dfrac{S - S_h}{S_w - S_h}, & S_h < S \leq S_w, \\ E_w + (E_{max} - E_w) \dfrac{S - S_w}{S^* - S_w}, & S_h < S \leq S^*, \\ E_{max} & S^* < S \leq 1, \end{cases}$$

where E = evapotranspiration; E_{max} = maximum evapotranspiration rate; E_w = evaporation rate at wilting point; S = percent soil water; S* = plant water stress level; S_w = wilting point; S_h = hydroscopic point. Soil water and evapotranspiration decreased until only evaporation was possible because water at the hydroscopic level is unavailable for plant transpiration to occur (Laio, Porporato, Ridolfi, & Rodriguez-Iturbe, 2001). Soil texture (e.g., clay, loam, silt, and sand) determined the potential water availability which controlled plant water stress level, wilting point, and hydroscopic point for evapotranspiration rates throughout the growing season (Cosby, Hornberger, Clapp, & Ginn, 1984; Dingman, 1994; Lai & Katul, 2000).

After initial water losses, excess water inflow was multiplied by a runoff coefficient to calculate daily runoff for cropland, grassland, and all other land within each of the four water-catchments from 1947 to 2012. Runoff coefficient values were calculated by averaging the Rational Method index; a simplistic method commonly used to estimate surface water runoff based on relief (i.e., topography, soil infiltration, vegetative cover, and surface storage (Thompson, 2006). Index values were parameterized from LCC characteristics within each hydrologic unit code 10 (HUC) water-catchment and for each land use type. The runoff coefficient was altered annually based on changes in ha of cropland and grassland within each LCC.

Output of the erosion and hydrologic sub-models were then used as input for the TSS model (Figure 6). Estimates of TSS were dependent upon eroded soil transport and deposition into waterways and stream or river flow volume. Total annual rill and sheet erosion from the erosion sub-model were used to calculate sediment deposition (metric-tons/ha) into a stream or river. Sediment deposition was computed using the Vanoni (2006) power function that expresses the amount of annual rill and sheet erosion that reached ephemeral streams, gullies, or rivers. The sediment delivery ratio (SDR; metric-tons/water-catchment/yr) into streams and rivers in each water-catchment was estimated as:

$$\text{SDR} = \Sigma\, HUC10_{i=n}(0.42\, A^{(-0.125)} \times \text{Annual erosion})$$

where $HUC10_i$ is each sub-water-catchment that comprise the entire water-catchment, A is the area (km^2) of each HUC10 water-catchment, and annual erosion is the total annual erosion of each HUC10. Annual sediment deposition altered sediment bedload and TSS levels in each river system either through sediment suspension or settlement. Total suspended solids were altered with streamflow velocity (Vanoni, 2006) and were adjusted in the model using a table function in Vensim™. The table function was parameterized by a TSS and streamflow velocity curve for each water-catchment, which dynamically improved all simulated annual TSS values (Figure 8). A TSS correction value was developed to improve simulated TSS settling rates for each water-catchment by obtaining a ratio from the observed and predicted mean TSS values across all years and was calculated as:

$$Correction\, Value = \frac{\bar{X}}{\bar{Y}}$$

where $\bar{Y}$ is the mean of simulated TSS values and $\bar{X}$ is the mean of observed TSS values across all years. This ratio was used as a constant TSS correction value for each-catchment across all years to avoid overfitting annual TSS values (i.e., avoid biasing annual TSS simulation values).

Model Calibration

Several tools, procedures, and tests were used to calibrate and test each model in order to improve the model's reliability to simulate reality (Table 3; Homer, 1996; Sterman, 2000). Each model was calibrated using both automatic and hand calibration to obtain a fit of simulated estimates to historical reference modes (i.e., observed data). Automatic calibration obtains the best fit for model variables and facilitates sensitivity testing (Oliva, 2003). However, failure to specify the correct variables and their associated values may obtain an optimal numerical fit but be difficult to intuitively interpret and methodologically wrong. For example, it is possible to set automatic calibration to adjust the runoff coefficient beyond 100%, which may produce an optimal fit for discharge but is physically impossible. Thus, it is important to be aware of constants used in automatic calibration that may provide a numerical fit but be intuitively wrong when selecting constants for automatic calibration. Hand calibration minimizes obvious errors in model parameters that are more easily overlooked in automatic calibration and is accomplished by adjusting variable values one-by-one to obtain an optimal fit, which requires much more time than automatic calibration. Automatic calibration was used in the hydrologic model to adjust parameters and minimize calibration time since discharge was measured daily for 66 years (1947-2012 or 24,107 days). This procedure was completed using the optimization function in Vensim DSS™. Hand calibration was used to adjust the RUSLE2 factors and sediment parameter values, within their established ranges to match historical erosion (1982-2012) and TSS values

(1964-2012). Both automatic and hand calibration were used for each of the four water-catchments before SD statistical testing.

Model Testing

After rigorous iterative calibration of each sub-model, the simulated erosion, hydrologic, and TSS estimates were compared to the observed data using SD statistical calibration tests. Long-term reference mode data were obtained for each model and water-catchment. Cropland and grassland erosion rates (metric-ton/ha/yr) were used as reference modes for the erosion model and were collected from the United States Department of Agriculture-National Resource Conservation Service 2012 National Resource Inventory (USDA-NRCS, 2012 NRI by request, see https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/nra/nri/) from 1982-2012 at five-year intervals. The hydrologic model was tested against the hydrologic discharge reference modes [total annual Million Cubic Meters (MCM)] for the Big Sioux, James, Bad, and Belle Fourche rivers. Hydrologic discharge reference mode data were obtained from the following sources: Big Sioux (USGS 06485500 at Akron, Iowa), James (USGS 06478500 near Scotland, South Dakota), Bad (USGS 06441500 near Fort Pierre, South Dakota) and Belle Fourche River (USGS 06438000 near Elm Springs, South Dakota) from 1947 − 2012. Simulated TSS were compared to TSS reference mode data for each of the four water-catchments. Total suspended solids reference mode data were obtained (1967 − 2012) from previously mentioned USGS stations [mg/L; Suspended sediment concentration (parameter code: P80154), Field/Lab Water Quality Samples, see https://waterwatch.usgs.gov/?m=real&r=sd] and from the Eastern Dakota Water Development District (see http://www.eastdakota.org/).

Statistical calibration tests compared reference modes and simulated estimates using Model Evaluation Software™ (see Tedeschi, 2006 for details on mathematical equations) and included three measurements of accuracy. Bias correction factor (C_b; Lin, 1989) calculated as:

$$C_b = \frac{2}{V + \frac{1}{V} + \mu^2}$$

where, *V* is the variance and μ is the mean of the population or sample, indicating how far the regression line deviates from the slope of unity (45°). Mean bias (MB; Cochran & Cox, 1957) computed as:

$$MB = \frac{\sum_{i=1}^{n}(Y_i - f(X_1, \ldots, X_p)_i)}{n}$$

where Y_i = ith observed value and $f(X_1, \ldots, X_p)_i$ = ith model-predicted values and n = sample size, which indicates the mean difference between observed and predicted values. The root mean square error of prediction (RMSEP; Bibby & Toutenburg, 1977) was calculated as:

$$RMSEP = \sqrt{\frac{\sum_{i=1}^{n}(Y_i - f(X_1, \ldots, X_p)_i)^2}{n}}$$

where Y_i = ith observed values, $f(X_1, \ldots, X_p)_i$ = ith model-predicted values, and n = sample size, which indicates the root difference between observed and model-predicted values. (Mitchell & Sheehy, 1997). Three measurements of precision were used to evaluate the model. Coefficient of determination (R^2; Kvålseth, 1985) calculated as:

$$R^2 = \left(\frac{n(\sum yx) - \sum y \sum x}{\sqrt{n\sum y^2 - (\sum y)^2}\sqrt{n\sum x^2 - (\sum x)^2}}\right)^2$$

where, y = observed values, x = predicted values, and n = sample size, which measures the proportion of variance between observed and predicted values. Modeling efficiency (MEF; Loague & Green, 1991) was calculated as:

$$MEF = \frac{\left(\sum_{i=1}^{n}(Y_i - \bar{Y})^2 - \sum_{i=1}^{n}(Y_i - \mathrm{f}(X_1, \dots, X_p)_i)^2\right)}{\sum_{i=1}^{n}(Y_i - \bar{Y})^2}$$

$$= 1 - \frac{\sum_{i=1}^{n}(Y_i - \mathrm{f}(X_1, \dots, X_p)_i)^2}{\sum_{i=1}^{n}(Y - \bar{Y})^2}$$

where, Y_i = ith observed value, $\bar{Y}$ = mean of observed values, $\mathrm{f}\left(X_1, \dots, X_p\right)_i$ = ith model-predicted values, and n = sample size, which is the proportion of variation explain from the line of predicted values rather than the fitted line. The concordance correlation coefficient [CCC ($\widehat{\rho}_C$); Lin, 1989] calculated as:

$$\widehat{\rho}_C = \frac{2 \times S_{\mathrm{f}}(X_1, \dots, X_p\,)Y}{S_Y^2 + S_{\mathrm{f}(X_1,\dots,X_p)}^2 + (\bar{Y} - \bar{\mathrm{f}}(X_1\,, \dots, X_p\,))^2}$$

where, Y_i = ith observed value, $\bar{Y}$ = mean of observed values, $\mathrm{f}\left(X_1, \dots, X_p\right)_i$ = ith model-predicted values, S_Y = standard deviation for Y, $S_{\mathrm{f}(X_1,\dots,X_p)}^2$ = standard deviation for predicted values, $\bar{\mathrm{f}}\left(X_1\,, \dots, X_p\,\right)$ = the mean of predicted values, and n = sample size, which indicates the reproducibility of two variables (i.e., a measurement of both accuracy and precision over a given time). The RMSEP values were then decomposed ($RMSEP_d$) to screen for systemic errors using Thiel's inequality statistics calculated as:

$$RMSEP_d = (\bar{f}\,(X_1, \dots, X_P) - \bar{Y})^2 + (s_{f(X_1,\dots,X_p)} - r \times s_Y)^2 + (1 - r^2) \times S_Y^2$$

where, $\bar{Y}$ = mean of observed values, $\bar{f}(X_1, \ldots, X_p)$ = mean of predicted values, S_Y = the standard deviation for observed values, $s_{f(X_1,\ldots,X_p)}$ = the standard deviation for predicted values, and r = Pearson's correlation coefficient, and r^2 = coefficient of determination. The decomposition of RMSEP simply calculates the proportion of mean, variance, and covariance that when added together equal the total RMSEP (i.e., equal to 1, referred to as unequal mean, variance, and co-variance) and is an indication of SD structural adequacy (Oliva, 1995; Sterman, 1984; Tedeschi, 2006; Thiel, 1961).

RESULTS

Erosion Modeling

The erosion model accurately replicated past behavior of erosion rates on croplands within each of the four water-catchments as indicated by C_b, MB, and RMSEP (Table 4). Coefficient of determination and CCC measurements of cropland erosion indicated high precision for the Big Sioux and James rivers but lower precision for the Bad and Belle Fourche rivers. Modeling efficiency results for cropland erosion indicated a lack of precision in each water-catchment, except for the Big Sioux, likely due to small sample size (observed years = 5 − 7 for erosion; Table 5). Measurements of cropland erosion RMSEP decompositions indicated no systematic errors from percentages of unequal mean, unequal variance and unequal covariance, except for the James River (Table 6). However, errors in the James River were considered model "noise" (i.e., non-systematic) as overall observed and predicted cropland erosion means followed similar long-term trends across years (Figure 9); therefore, these errors were disregarded (Sterman, 2000). Grassland erosion measurements of accuracy and precision were relatively high for each water-catchment (Tables 4 and 5). Measurements of grassland erosion RMSEP decompositions percentages of unequal mean, unequal variance, and unequal co-variance indicated that the Big Sioux River was structurally sound, but not for the James, Bad, and Belle Fourche rivers (Table 6). Root mean square error of prediction decomposition errors for James, Bad, and Belle Fourche rivers grassland erosion were found to be unsystematic errors (i.e., model noise) as mean observed and predicted values were similar across years. (Figure 10; Sterman, 2000; see Appendix C for additional model results).

Hydrology Modeling

Measurements of accuracy from the Cb and MB indicated that each water-catchment accurately replicated reference mode data, but not from RMSEP values (Table 7). Measurements of precision indicated that the Big Sioux River and James River were more precise than the Bad and Belle Fourche rivers (Table 8); however, the hydrologic model's level of precision for each catchment was considered adequate based on R^2 criteria (> 0.50; Moriasi et al., 2007; Santhi et al., 2001; Van Liew, Arnold, & Garbrecht, 2003). Measurements of RMSEP decompositions indicated that the structure of the models for the Big Sioux and Belle Fourche rivers were adequate, but the models for the James and Bad rivers were less structurally sound (Table 9). Structural model errors were evaluated and were determined to be model "noise" and non-systematic; thus, the model's purpose to replicate historical reference modes was accomplished, and the model was able to capture long-term high and low discharge extremes (Figure 11; see Appendix C for additional model results).

Total Suspended Solids Modeling

Measurements of accuracy (MB and C_b) indicated that all water-catchments were moderately accurate, due to larger RMSEP values in each of the four water-catchments (Table 10). Estimates of TSS indicated low precision for each water-catchment (Table 11). Moreover, estimates of TSS, RMSEP decompositions for each water-catchment indicated a high degree of model structure (i.e., structurally sound model) in the TSS model (Figure 12; Table 12; see Appendix C for additional model results).

DISCUSSION

The novel approach of SD for complex erosion, hydrologic, and TSS systems is useful for our purpose of estimating structural and behavioral changes over time. Each SD sub-model utilized the best available time series data and methodology to simulate real-world behaviors as compared to historically observed data from four unique water-catchments in South Dakota. Simulated cropland and grassland erosion rates matched the observed data with accuracy and precision across all years and in each water-catchment. Further, replication of long-term erosion patterns indicated that land use change and erosion dynamics adequately captured structural and behavioral changes over time as land use, topography, climate, and soils differ in each study area.

Hydrologic sub-model simulated estimates followed observed discharge trends across years, despite some larger differences between observed and predicted values during high precipitation years. Differences between observed and predicted annual discharge volumes were mostly caused by additional streamflow dynamics during flooding. Extreme flooding events (10 − 500-year events) have been shown to cause additional changes in hydrologic processes from increased upstream flow and river inundation dynamics (Niehus, 1996), and these processes may not have been completely captured in the model. Long-term hydrologic historical trends of increased discharge magnitude, number of extreme events, and low water years were adequately captured in each water-catchment from 1947 – 2012. Historical trends of increased river discharge over time have been mainly attributed to changes in climate, especially in eastern South Dakota (Kirbia et al., 2016).

Simulations of TSS were highly variable but followed general long-term trends in each water-catchment. In all water-catchments, simulated TSS values improved after 1982 (start of erosion calibration), indicating that erosion calibration made a significant difference in simulated TSS values. Observed data (n = 12 years for TSS) were more limited for the eastern Big Sioux River water-catchment than the other water-catchments, which may have attributed to increased differences for long-term trends when compared to the other three water-catchments. The western Bad River water-catchment is characterized by its highly erodible soils and high levels of TSS, which were successfully captured in relation to observed TSS values. In general, both eastern and western water-catchments followed observed trends of decreased TSS from 1967 to 2012, most likely from the adoption of no-tillage cultivation (Fayyah, 2011, Hong, 2017, Stoltenberg & Rutz et al., 2013, Smart et al., 2015).

The results of my sub-models follow similar international and domestic environmental land use change models. For example, Bakker et al. (2008), estimated that erosion increased from 9 metric-tons/ha/yr in 1995 to 16 metric-tons/ha/yr in 2001 due to cropland expansion onto grassland with erodible soils in Hageland, Belgium, using the RUSLE (Bakker et al., 2008). Results from my study showed that grassland reestablishment in South Dakota decreased from a maximum of 11.0 to minimum 0.5 metric-tons/ha/yr from 1982 to 2012 as lands with erodible soils were taken out of cropland production and put into CRP. These findings support the use of RUSLE components that effectively linked changes in erosion to alteration of cropland and grassland in both Belgium and South Dakota. Changes in erosion rates from grassland conversion have been documented in the NPG. Clay et al. (2014) reported an overall decrease in cropland and grassland erosion from 7.2 metric-ton/ha/yr in 1982 to 4.8 metric-ton/ha/yr in 2007 for South Dakota, North Dakota, and Nebraska. Declines in erosion rates were attributed to improvements and use of conservation tillage (i.e., no-till; Clay et al., 2014), although 2012 NRI erosion rates ≤ 6.77 metric-tons/ha/yr for these states were greater than those in 2007 (NRCS, 2012). The results of erosion modeling in my study did not cover the entire state, like those reported by Clay et al., (2014), but each water-catchment followed statewide erosion trends, which indicated that simulated grassland and cropland erosion rates from 1982 – 2012 were reasonable; that is to say the model parameters and structure were adequate for erosion systems.

Hydrologic changes (runoff, streamflow, evapotranspiration) were similar amongst the SD hydrologic model and other models that evaluated long-term land-use change. Hydrologic responses to land use change have been evaluated across the globe using the soil and water analysis tool (SWAT). For example, SWAT was used in Ethiopia, where runoff increased from 159 –167 mm/yr and discharge decreased from 538 – 467 mm/yr in relation to a 20% increase of cropland and a 4% loss of grassland from 1973 to 2010 (Woldesenbet, 2017) and SD model results demonstrated a general increase in discharge in all four water-catchments between 1947 and 2012. In the

Midwestern U.S., Xu, Scanlon, Schilling, and Sun (2013), reported that baseflow increased by approximately 25% from land use change including grassland in 55 midwestern water-catchments from 1930 to 2010. Further, Hong (2017) indicated that SWAT runoff estimates in eastern South Dakota increased or decreased as much as 7% under cropland expansion or grassland reestablishment, respectively in the NPG. In South Dakota, Neupane and Kumar (2015) evaluated hydrologic changes from cropland expansion between 1980 and 2013 using SWAT and indicated that surface water runoff increased between 2 – 4% annually as cropland hectares increased within the Big Sioux River. Kibria et al. (2016) reported that only two of 18 western-catchments (Castle Creek near Deerfield Reservoir and Hill City, South Dakota, and Rhoads Fork near Rochford, South Dakota) were responsive to rapid decreases in grassland for two specific years, 1951 (7 – 17% decrease) and 2011 (2 – 7% decrease); streamflow in the remaining catchments were responsive only to changes in climate. Overall, estimates from the SD model and other studies indicated that hydrological response to grassland conversion are limited and that climate is a factor that contributes to changes in discharge within South Dakota.

Similar to the erosion and hydrologic sub-models, the TSS model performance was comparable to other water quality models, which indicate that modeling TSS concentration flux is difficult and highly variable (Meybeck, Laroche, Dürr, & Syvitski, 2003; Kettner, Gomez, & Syvitski, 2007). For example, Strauch et al. (2013) estimated Brazilian sediment deposition (i.e., directly tied to TSS) change from forest and grassland conversion to cropland but was unsuccessful, reporting that calibrated sediment deposition values were excessively large due to the lack of daily observed sediment values and incomplete climate data. Despite limitations, the Brazilian model was determined useful as it provided a percent change from baseline sediment deposition with various water quality mitigations scenarios (reductions in sediment deposition = 40%). Likewise, the TSS model in my study is useful as simulated TSS data were able to indicate a departure from the South Dakota Department of Environmental and Natural Resources (SD-DENR) environmental standards of average daily TSS 158 mg/L (see https://denr.sd.gov/des/sw/-swqstandards.aspx). Within the Great Plains, North Fork Ninnescah River and Cheney Reservoir of south-central Kansas, Christensen et al., 2006 reported that during large rain events surface runoff from agricultural dominated lands increased TSS (>550 mg/L), which exceeded state standards (> 100 mg/L) in 2001 and 2002. Similar TSS patterns were shown in my model which indicated increased TSS during high runoff years. More recently, Fayyadh (2011) analyzed TSS and river flow relationships from 1975 to 2008 for the James River near Columbia, South Dakota, and found that TSS ranged from 3 − 166 mg/L (mean = 44 mg/L) during dry years and 6 −135 mg/L (mean = 26 mg/L) during wet years. Mean TSS calculated from the SD TSS model near Scotland, South Dakota, (484 km south of Columbia, South Dakota) was 106 mg/L (range = 56 − 341 mg/L). Thus, complex TSS flux estimates would likely improve with additional data, but TSS sub-model’s calibration meet the current purpose to indicate change in this environmental externality relative to grassland conversion to cropland in South Dakota.

The calibration process for each model presented limitations, but despite these limitations, the model was still quantitatively and structurally (i.e., correct parameters) sound to replicate historical reference modes of erosion, hydrologic discharge, and TSS.

The model calibration process was limited by the amount and quality of available land use, climate, and TSS data; put simply low-quality data into the model equals low-quality estimates out of the model. Land use data that informed each model were estimated data from 1947 – 2012 and lacked some degree of spatial accuracy (Sohl et al., 2016); therefore, any errors in the land use data limited the calibration process of each sub-model. For example, errors in spatially explicit annual land use could amplify errors in hydrologic and TSS responses over time as annual hydrologic responses vary depending on land use hydrologic characteristics. Availability of climate data at the appropriate spatial scale also limited the calibration of each sub-model in my study in some respect. Regionalized climate data from 1947 to 2012 may have had disparities between actual climate patterns in local sub-water-catchments which may have biased the calibration process. Total suspended solids values are challenging to simulate due to limited observed data and being variable in nature (Christensen et al., 2006; Strauch et al., 2013; Vanoni, 2006). Despite these limitations, the TSS model results captured long-term TSS trends, which is useful to indicate departure from environmental standards in each of the four water-catchments. Further, each model is useful as core dynamics that drive each system were incorporated and provide a basis for understanding how erosion, hydrologic and TSS systems respond to grassland conversion in each of the four water-catchments over time.

Overall, the aggregate System Dynamics model (combination of the erosion, hydrologic, and TSS sub-models) used to model environmental externalities in these four South Dakota water-catchments provides a reliable method to explain past and present trends in soil erosion, hydrologic regimes, and TSS variability. Thus, this model may be used to reasonably estimate the same externalities into the future under various land use change scenarios. The estimates may help identify high-leverage and long-term solutions to mitigate potential environmental risk. Furthermore, the model could be used to provide information to producers and other stakeholders (e.g., policy makers) by allowing them to experiment with grassland conversion scenarios and mitigation strategies and make proactive management decisions using the best information available.

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Table 1. Definitions of key endogenous and exogenous variables used in the System Dynamics model (Chow et al., 1988; Turner et al., 2016).

Variable type	**Variable name**	**Definition and unit**
Endogenous	Farmland	Total land in crop production (ha/yr; Turner et al., 2016).
	Grassland	Total land used for hay, pasture, or fallow (ha/yr; Turner et al., 2016).
	Total plant biomass	Total alive and dead (above and below ground) plant material throughout the growing season (kg/ha).
	Surface water runoff	Volume (m^3/s).
	Aggregate sheet and rill erosion	Detached soil particles (metric-tons/ha/yr).
	Soil organic matter	Percent organic matter in the soil profile (% in topsoil layer).
	Best management practices (BMP)	Tillage type: Conservation or conventional tillage (dimensionless)
	Total suspended solids	Total soil particles suspended in a stream or river (mg/L).
	Soil infiltration rate	Daily rate of water movement into the ground (m^3/day).
Exogenous	Projected land use	Farmland estimates for each Land Capability Class (LCC; ha; Turner et al., 2016).
	Climate	Precipitation (cm/day), temperature (°C) and snow (cm/day).
	Crop diversity and distribution	Distribution of corn, soybeans, winter wheat, and spring wheat planting (ha) based on USDA Agriculture Census data from 1945 – 2012.
	Land capability classes 1-8	USDA-NRCS land suitability rating for agricultural production (dimensionless).
	Slope length and steepness factor	Hydrologic factor dependent on average slope length and steepness characteristic of each basin (dimensionless).

Table 2. Climate data source description. Weather station name, state, global historical climatology network daily documentation (GHCND), latitude and longitude (Lat/Long), water-catchment, years of available data ("avail."), and percent coverage (C) of data with reported years.

Station Name	State	GHCND	Lat/Long	Water-catchment	Years avail.	C (%)
Akron	IA	USC00130088	42.8258, -96.5514	Big Sioux River	1900 – 2017	63%
Luverne	MN	USC00214937	43.6658, -96.2022	Big Sioux River	1893 – 2017	52%
Brookings 2 Northeast	SD	USC00391076	44.3252, -96.7686	Big Sioux River	1893 – 2017	99%
Watertown Regional Airport	SD	USW00014946	44.9047, -97.1494	Big Sioux River	1893 – 2017	97%
Canton	SD	USC00391392	43.3112, -96.5877	Big Sioux River	1896 – 2017	92%
Jamestown Municipal Airport	ND	USW00014919	46.9258, -98.6691	James River	1948 – 2017	100%
Fullerton 1 East-Southeast	ND	USC00323287	46.158, -98.4	James River	1898 – 2017	99%
Aberdeen Regional Airport	SD	USW00014929	45.4433, -98.413	James River	1893 – 2017	99%
Huron Regional Airport	SD	USW00014936	44.3981, -98.2231	James River	1881 – 2017	100%
Alexandria	SD	USC00390128	43.6513, -97.7847	James River	1893 - 2017	97%
Pierre Regional Airport	SD	USW00024025	44.3813, -100.2855	Bad River	1983 – 2017	92%
Cottonwood 2 East	SD	USC00391972	43.9611, -101.8605	Bad River	1909 – 2017	98%
Newell	SD	USC00396054	44.7158, -103.4275	Belle Fouche River	1920 – 2017	99%
Lead	SD	USC00394834	44.3544, -103.7431	Belle Fouche River	1909 – 2017	100%

Table 3. Tests for assessment of dynamic models adapted from Sterman (2000).

Test	**Purpose**	**Procedures and tools**
Boundary Adequacy	Are the significant ideas for addressing the problem endogenous to the model? Does the behavior of the model change when boundary assumptions are altered? Do the policy recommendations alter when the model boundary is enlarged?	Use model or sub-model diagrams, causal diagrams, stock and flow conceptual maps, and review model equations. Utilize interviews, workshops to gain expert opinion, historic materials, review of literature, direct inspection or involvement in system processes. Adapt model to include likely additional structure, make constants and exogenous variables endogenous, afterward repeat sensitivity and policy analysis.
Structure Assessment	Is the model structure consistent with appropriate descriptive information of the system? Is the aggregation level appropriate? Are basic physical laws followed by the model? Is the stakeholder behavior captured in the system by the decision rules?	Application of policy structure diagrams, causal diagrams, stock and flow maps, and direct review of model equations. Utilization of interviews, workshops to gain expert opinion, historic materials, direct review or involvement in system processes. Perform partial model tests of the decision rules to evaluate rationale. Design separate sub-models and contrast behavior to aggregate model formulations. Disconnect structures of interest, then redo sensitivity and policy analysis.
Dimensional Consistency	Do model parameters have dimensional consistency and actual meaning?	Analyze dimensional consistency with model program. Review model equations for suspicious parameters.

Table 3. Tests for assessment of dynamic models (continued).

Assessment of parameters	Is quantitative information of the system captured by the parameter values? Are parameters representative of actual real-world variables?	Utilize statistical methods to estimate parameters and partial model tests to calibrate subsystems. Apply subjective methods based on consultations, expert judgment, focus groups, historic materials and experience. Disaggregate and evaluate sub-models.
Extreme Conditions	Is each equation intuitively correct relative to extreme values changes? Does the model respond reasonably to extreme policies, perturbations, and parameters?	Review equations. Evaluate response to extreme values of each input(s). Subject model to large perturbations and varying extreme conditions.
Integration	Does selection of time step or numerical integration method indicate sensitivity in results?	Alter the timestep (e.g., half). Utilize various integration methods and test for behavioral changes.
Behavior Reproduction	Is the systems behavior of interest replicated? Does it endogenously create the symptoms of the problem inspiring the study? Does the model create the various changes in behavior?	Calculate statistical measures of agreement between model and data. Contrast model results and data qualitatively: modes of behavior, shape of variables, asymmetries, relative amplitudes and phasing and unusual events. Evaluate response of model to test inputs, shocks, and noise.

Table 3. Tests for assessment of dynamic models (continued).

Behavior Anomaly	When assumptions of the model are altered or removed do results indicate sensitivity?	Zero-out key effects (i.e., loop knockout analysis).
Family Member	Is the model able to generate the observed behavior in other cases of the same system?	Calibrate the model to the broadest conceivable range of associated systems.
Surprise Behavior	Is unobserved or unrecognized behavior generated by the model? Under novel conditions does the model successfully anticipate the systems response?	Record simulation results and use model to simulate possible future behavior of system. Clarify inconsistencies between model behavior and current understanding of the real-world system. Document existing mental models of participant and client before the beginning of the modeling work.
Sensitivity Analysis	Do the quantitative values change significantly? Do the modes of behavior created by the model change drastically? Do the policy implications alter drastically?	Conduct univariate and multivariate sensitivity analysis. Use other analytical methods. Perform model boundary and aggregation tests. Apply optimization tools to find the optimum model parameters and policies. Utilize optimization tools to identify parameter combinations that produce improbable outcomes or reverse policy results.
System Improvement	Did the modeling procedure improve the system?	Develop means to evaluate model impact on mental models, behavior, and outcomes prior to the study. Conduct before and after intervention evaluation using controlled experiments (e.g., treatment, control groups, and random assignment).

Table 4. Statistical measurements of accuracy for the erosion model. Values for correction bias (C_b) closer to 1, mean bias closer to 0, and root mean square error of prediction (RMSEP) closer to 0 indicate higher levels of accuracy. Mean bias and RMSEP are reported as percentages of observed values.

Land Use	**Water-catchment**	**n**	**Observed Mean**	**Predicted Mean**	**C_b**	**Mean Bias**	**RMSEP**
Cropland	Big Sioux River (HUC6)	7	6.39	6.11	0.98	-4.54%	11.65%
Cropland	James River (HUC6)	7	1.0	0.92	0.80	8.76%	15.25%
Cropland	Bad River (HUC 8)	7	2.13	2.43	0.81	-14.09%	32.93%
Cropland	Belle Fourche River (HUC 8)	5	0.45	0.46	0.90	-3.06%	28.54%
Grassland	Big Sioux River (HUC6)	7	1.26	1.26	0.99	-0.07%	1.62%
Grassland	James River (HUC6)	7	0.23	0.22	0.96	3.02%	13.50%
Grassland	Bad River (HUC 8)	7	0.55	0.59	0.80	-8.07%	40.70%
Grassland	Belle Fourche River (HUC 8)	5	0.24	0.23	0.97	-7.0%	13.53%

Table 5. Statistical measurements of precision for the erosion model. Values for coefficient of determination (R^2), modeling efficiency (MEF), and correlation concordance coefficient (CCC) closer to one indicate higher precision.

Land Use	Water-catchment	n	R^2	MEF	CCC
Cropland	Big Sioux River (HUC6)	7	0.89	0.83	0.93
Cropland	James River (HUC6)	7	0.85	-0.17	0.73
Cropland	Bad River (HUC 8)	7	0.28	-1.33	0.43
Cropland	Belle Fourche River (HUC 8)	5	0.20	0.15	0.40
Grassland	Big Sioux River (HUC6)	7	0.99	0.99	0.99
Grassland	James River (HUC6)	7	0.90	0.86	0.91
Grassland	Bad River (HUC 8)	7	0.98	0.73	0.78
Grassland	Belle Fourche River (HUC 8)	5	0.92	0.89	0.94

Table 6. Root mean square error of prediction (RMSEP) decomposition for the erosion model. Percentages represent portion of total error derived from unequal mean, variance, or covariance.

Land Use	**Water-catchment**	**Unequal Mean**	**Unequal Variation**	**Unequal Covariation**
Cropland	Big Sioux River (HUC6)	15.18%	11.29%	73.52%
Cropland	James River (HUC6)	33.02%	44.1%	22.88%
Cropland	Bad River (HUC 8)	18.31%	16.66%	65.02%
Cropland	Belle Fourche River (HUC 8)	1.15%	15.38%	83.47%
Grassland	Big Sioux River (HUC6)	0.21%	0.81%	98.98%
Grassland	James River (HUC6)	5.0%	42.1%	52.89%
Grassland	Bad River (HUC 8)	3.94%	92.70%	3.37%
Grassland	Belle Fourche River (HUC 8)	26.67%	14.35%	58.98%

Table 7. Statistical measurements of accuracy for the hydrologic model. Values for correction bias (C_b) closer to 1, mean bias closer to 0, and root mean square error of prediction (RMSEP) closer to 0 indicate higher levels of accuracy. Mean bias and RMSEP are reported as percentages of observed values.

Water-catchment	n	Observed Mean	Predicted Mean	C_b	Mean Bias	RMSEP
Big Sioux River (HUC6)	66	1376	1338	0.99	2.7%	61.98%
James River (HUC6)	63	1651	1097	0.85	-37.49%	75.18%
Bad River (HUC 8)	66	162	345	0.56	-112.11%	139.22%
Belle Fourche River (HUC 8)	66	338	279	0.93	-21.48%	71.91%

Table 8. Statistical measurements of precision for the hydrologic model. Values for coefficient of determination (R^2), modeling efficiency (MEF), and correlation concordance coefficient (CCC) closer to one indicate high precision.

Water-catchment	**n**	**R^2**	**MEF**	**CCC**
Big Sioux River (HUC6)	66	0.55	0.49	0.74
James River (HUC6)	63	0.79	0.71	0.82
Bad River (HUC 8)	66	0.48	-0.46	0.39
Belle Fourche River (HUC 8)	66	0.47	0.41	0.64

Table 9. Root mean square error of prediction (RMSEP) prediction decomposition for the hydrologic model. Percentages represent portion of total error derived from unequal mean, variance, or covariance.

Water-catchment	**Unequal Mean**	**Unequal variation**	**Unequal covariation**
Big Sioux River (HUC6)	0.19%	<0.01%	99.81%
James River (HUC6)	24.87%	0.40%	74.73%
Bad River (HUC 8)	64.85%	7.24%	27.91%
Belle Fourche River (HUC 8)	8.92%	8.75%	82.33%

Table 10. Statistical measurements of accuracy for the total suspended solids model. Values for correction bias (C_b) closer to 1, mean bias closer to 0, and root mean square error of prediction (RMSEP) closer to 0 indicate higher levels of accuracy. Mean bias and RMSEP are reported as percentages of observed values.

Water-catchment	n	Observed Mean	Predicted Mean	C_b	Mean Bias	RMSEP
Big Sioux River (HUC6)	12	129.91	93.62	0.76	-38.76%	87.53%
James River (HUC6)	18	263.83	96.50	0.19	7.59%	49.29%
Bad River (HUC 8)	31	163.83	197.76	0.87	17.15%	63.62%
Belle Fourche River (HUC 8)	14	112.43	88.85	0.85	20.97%	55.32%

Table 11. Statistical measurements of precision for the total suspended solids model. Values for coefficient of determination (R^2), modeling efficiency (MEF), and correlation concordance coefficient (CCC) closer to one indicate high precision.

Water-catchment	**n**	**R^2**	**MEF**	**CCC**
Big Sioux River (HUC6)	12	0.05	-0.37	0.17
James River (HUC6)	18	0.04	-0.68	0.04
Bad River (HUC 8)	31	0.19	0.07	0.38
Belle Fourche River (HUC 8)	14	0.10	-0.22	0.27

Table 12. Root mean square error of prediction (RMSEP) decomposition in the total suspended solids model. Percentages represent portion of total error derived from unequal mean, variance, or covariance.

Water-catchment	**Unequal Mean**	**Unequal Variation**	**Unequal Covariation**
Big Sioux River (HUC6)	19.61%	10.18%	70.21%
James River (HUC6)	2.37%	16.88%	80.75%
Bad River (HUC 8)	7.27%	13.13%	79.61%
Belle Fourche River (HUC 8)	14.37%	7.07%	78.56%

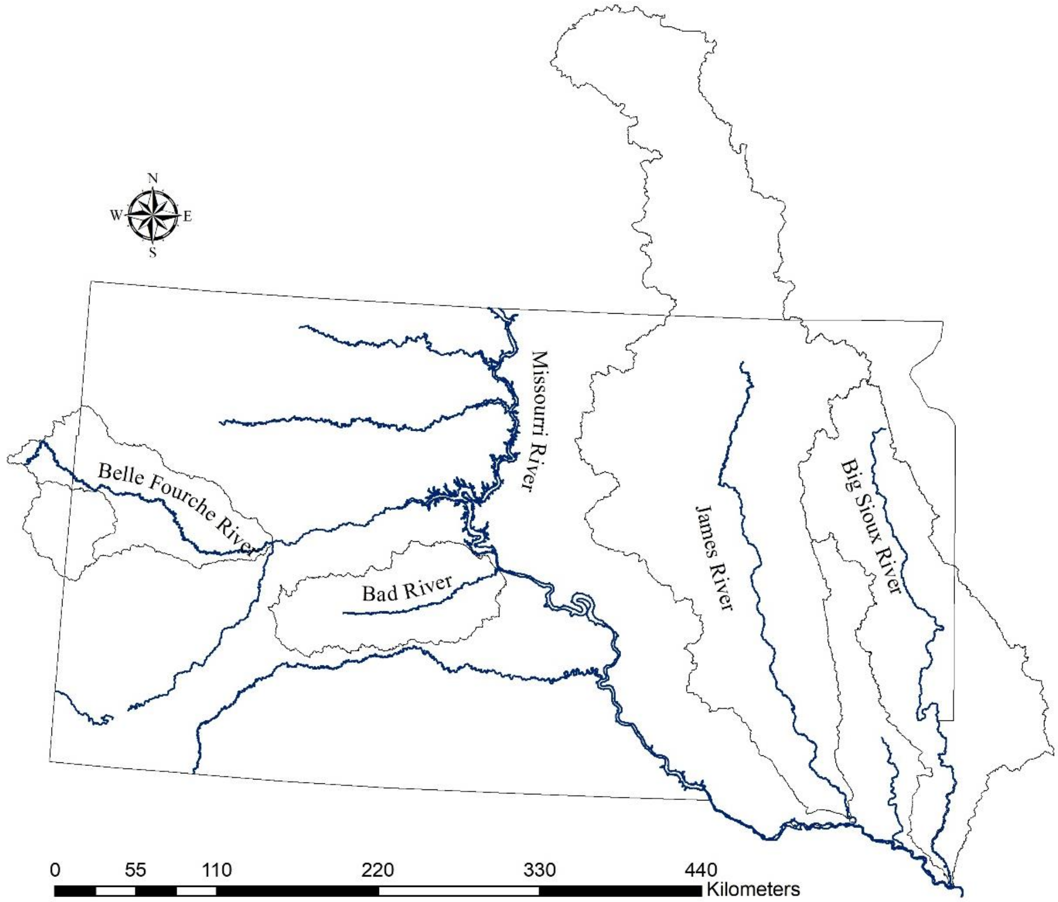

Figure 4. Map of the state of South Dakota, USA, with the four water-catchments included in this study: Big Sioux (area = 22,910 km^2), James (area = 54,742 km^2), Bad (area = 8,225 km^2), and Belle Fourche (area = 11,129 km^2) rivers.

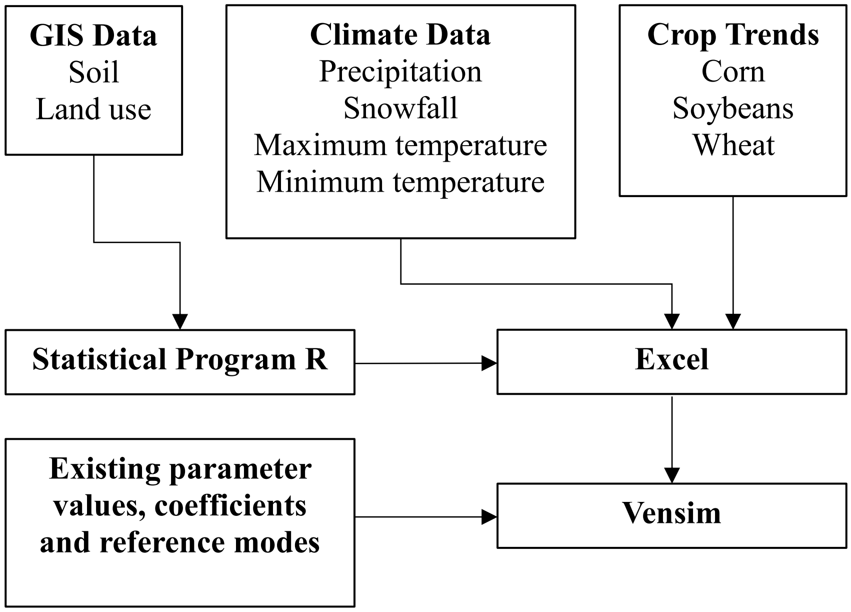

Figure 5. Collection and importation of model data into the SD modeling program Vensim DSS™ (Access™, ArcGIS 10.3.1™, Excel™, Program R™).

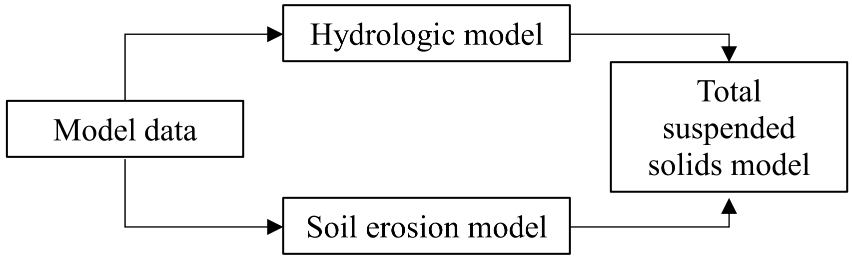

Figure 6. Overview of the process for integrating data into the soil erosion and hydrology models and then into the total suspended solids model.

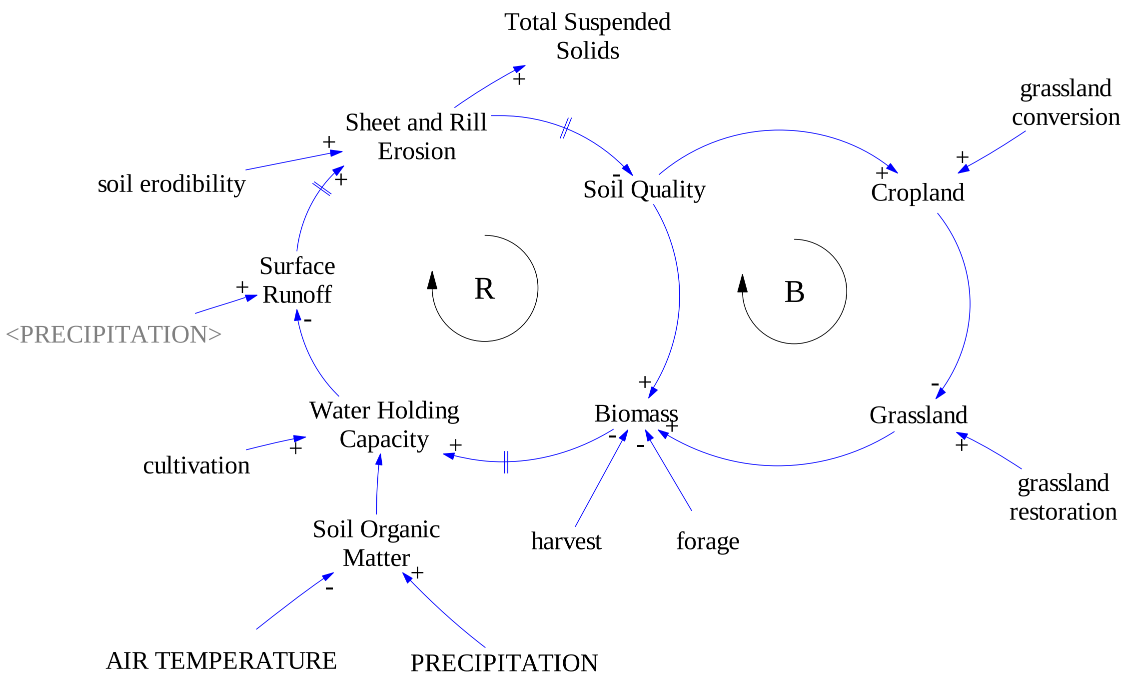

Figure 7. Dynamic Hypothesis conceptual diagram of dominant feedback loops within and between the erosion, hydrologic, and TSS model's components (for details on variables, see Menendez et al., 2017).

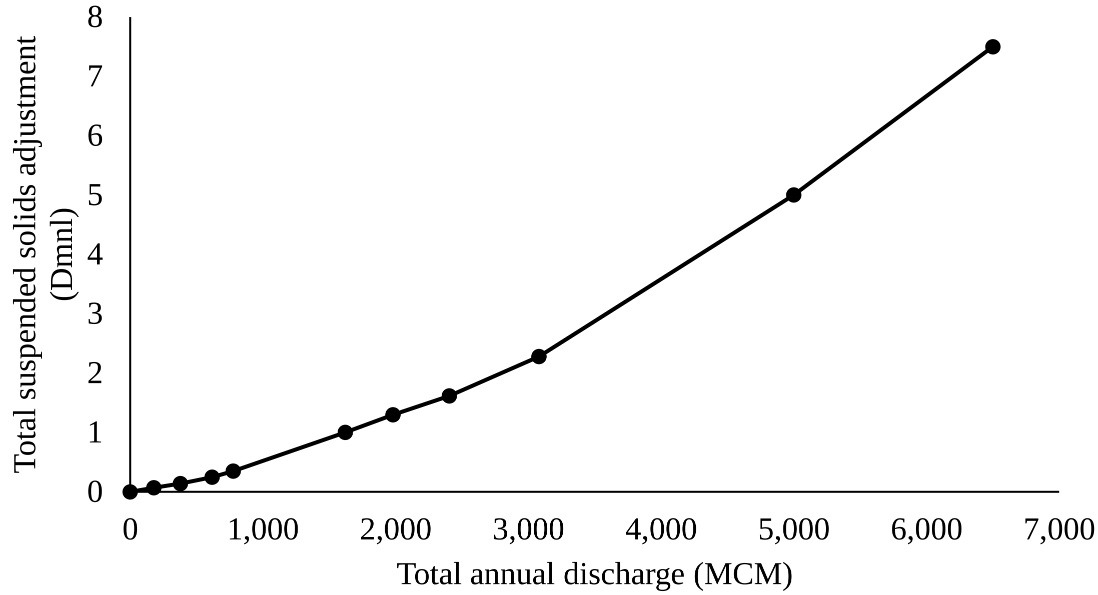

Figure 8. Example of total suspended solids lookup adjustment variable applied to the Big Sioux River water-catchment from 1947-2012 [input = annual discharge (million cubic meters) and output = dimensionless].

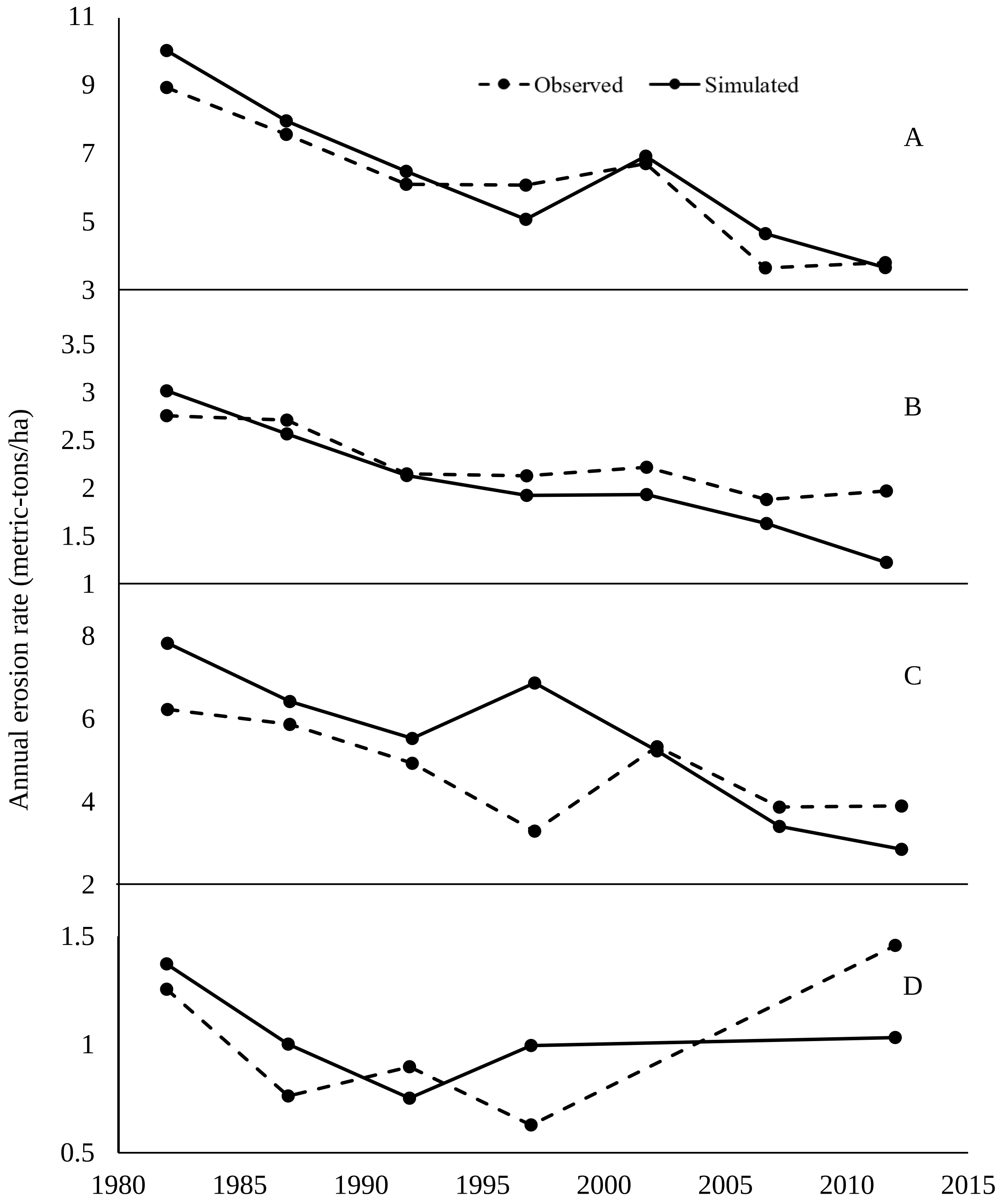

Figure 9. Observed and simulated rill and sheet erosion (metric-tons/ha/yr) from cropland in the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) rivers from 1982 to 2012.

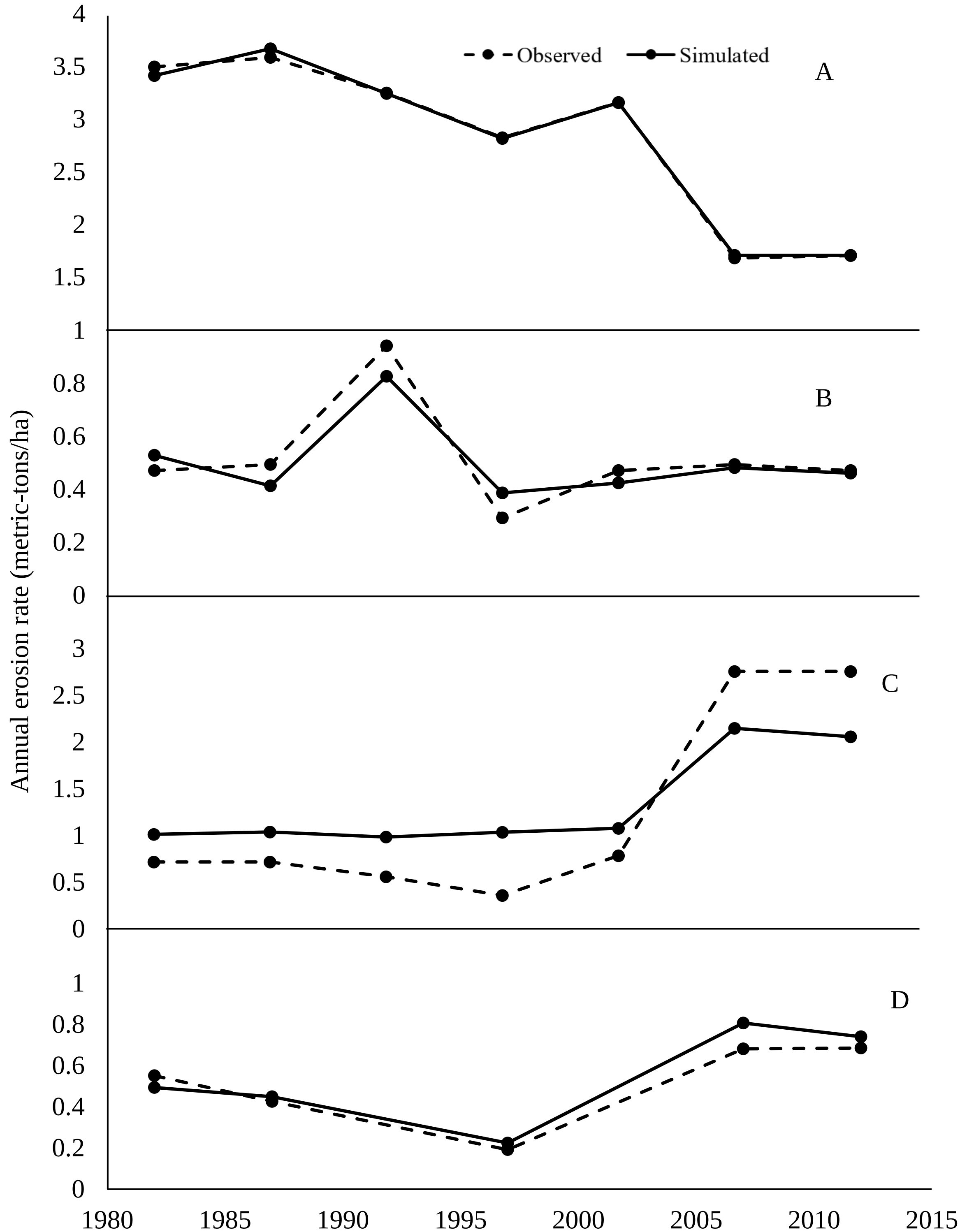

Figure 10. Observed and simulated rill and sheet erosion (metric-tons/ha/yr) from grassland in the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) rivers from 1982 to 2012.

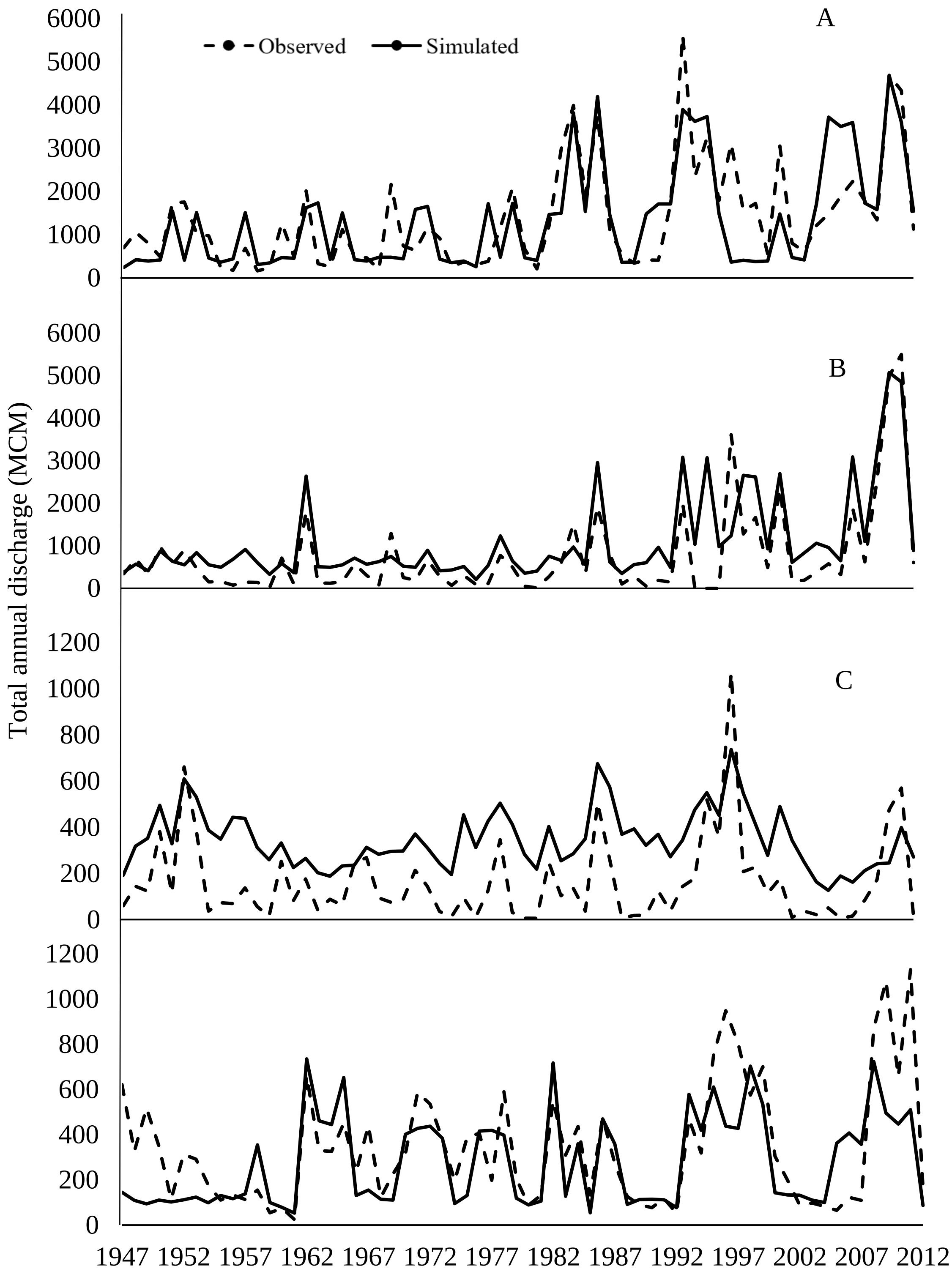

Figure 11. Observed and simulated total annual discharge [Million Cubic Meters (MCM)] for the Big Sioux River (A), James River (B), Bad River (C), and Belle Fourche River (D) from 1947-2012.

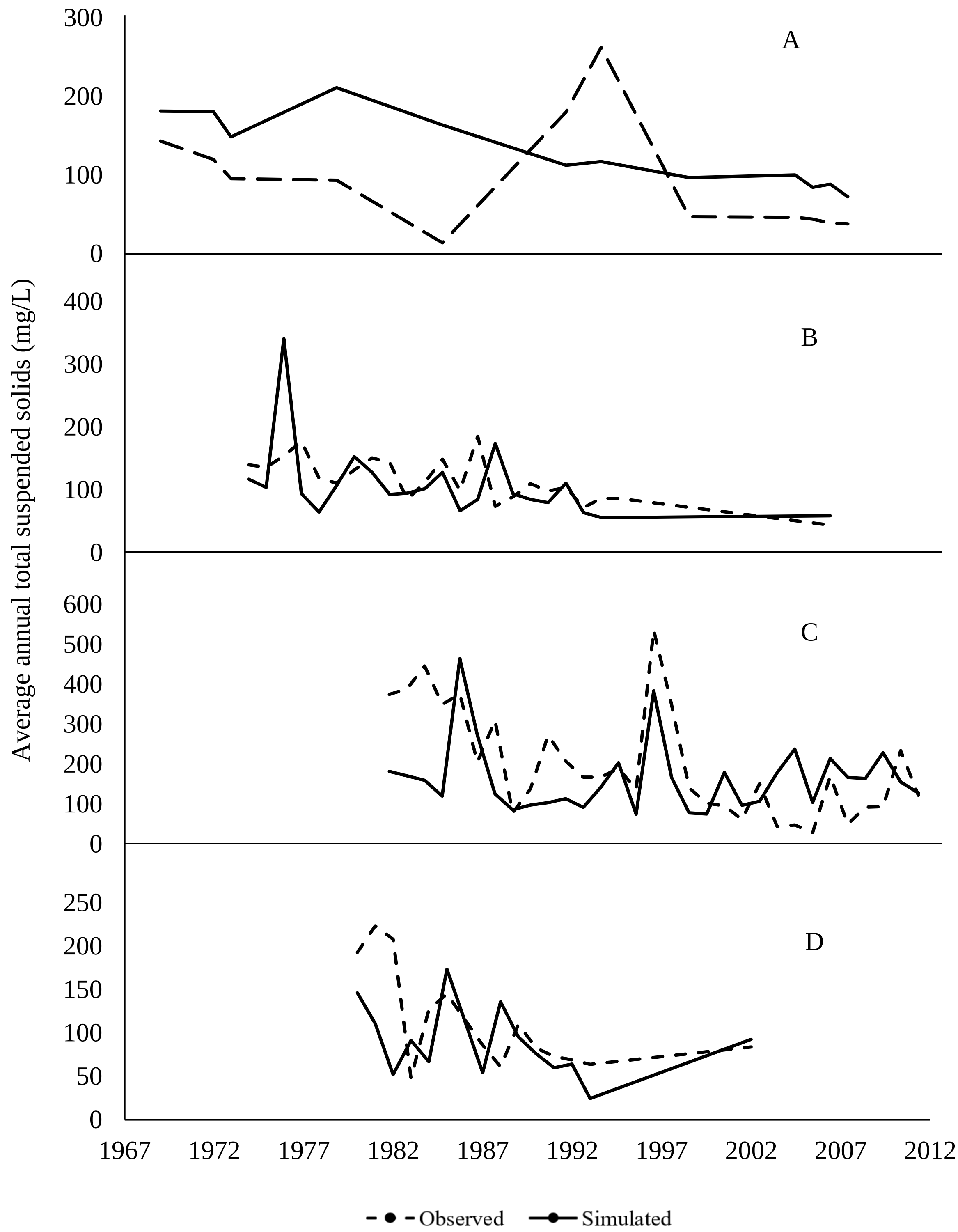

Figure 12. Observed and simulated average annual total suspended solids (mg/L) for the Big Sioux River (A), James River (B), Bad River (C), and Belle Fourche River (D). Years vary with available reference mode data for each water-catchment.

CHAPTER 3. ESTIMATING FUTURE CONSEQUENCES OF GRASSLAND CONVERSION IN FOUR SOUTH DAKOTA WATER-CATCHMENTS

INTRODUCTION

Increasing rates of grassland conversion to row-crop agriculture around the world have raised concerns of environmental consequences that may threaten the structure and function of ecosystems (Borrelli et al., 2017; Foley et al., 2005; Sterling, Ducharne, & Polcher, 2013; Turner et al., 2018). Between 2001 and 2012, an estimated 90% of new cropland (0.76 million km^2) around the world resulted from the conversion of grassland (Borrelli et al., 2017). Environmental consequences of such land use change may include, but are not limited to, changes in soil erosion rates, altered hydrologic regimes, and reductions in water quality (Foley et al., 2005; Koch et al., 2013; Sterling et al., 2013; Turner et al., 2018). Estimated erosion from cropland alone contributed to 80% of total annual erosion from 2001 to 2012 around the world (i.e., 0.69 billion metric-tons/yr of 0.86 billion metric-tons/yr; Borrelli et al., 2017). Changes in soil erosion alter the amount of topsoil retained or lost within a landscape which may alter biotic community interactions and resource availability within ecosystems (e.g., nutrient cycling processes; Matson, Parton, Power, & Swift, 1997) and agricultural systems (Pimentel, 2000). Similar to erosion rates, estimations of surface water runoff rates have also increased (6.8%) across the world, at least in part in response to land use change (including grassland conversion) between 1950 and 2000 (Sterling et al., 2013). Changes in runoff may alter the frequency and magnitude of flood and drought events within waterways (Bielders, Ramelot, & Persoons, 2003).

Erosion and surface water-runoff generated from grassland conversion may impact water quality as some portion of eroded soil may be transported by surface water runoff into waterways (Foley, 2005; Ojima, Galvin, & Turner, 1994). For example, in the Philippines, Alibuyog et al. (2009) reported that sediment yield into the Manupali River was estimated to increase by 200 − 273% as a result of increased soil erosion and surface water runoff resulting from the conversion of 50% of pasture area and grassland to cultivated cropland. Increased sediment yield is directly tied to increased total suspended solids (TSS, Meybeck, Laroche, Dürr, & Syvitski, 2003). Higher levels of TSS in waterways has been linked to altered aquatic plant (Mahaney, Wardrop, and Brooks, 2005) and animal (Bartelet, 2016) communities, nutrient cycling processes (Irving & Connell, 2002), and other social consequences such as decreased storage capacity of reservoirs, streams, and rivers (Cakula, Ferreira, & Panagopoulos, 2012; Santos, Andrade, Medeiros, Guerreiro, & Palácio et al., 2017). Thus, erosion and hydrological changes resulting from grassland conversion may drive other changes within waterways, including water quality.

The possibility exists that the environmental consequences of grassland conversion may continue to worsen in the future as conversion rates are likely to continue to escalate in response to human population growth and increased demand for agricultural commodities (e.g., grain, biofuel, livestock, and textiles; de Ruiter et al., 2017; Haberl, 2015; Pelletier & Tyedmers, 2010). In addition, the majority of the most productive grasslands suitable for cultivation has already been converted, and remaining grasslands that are less suitable for row-crop production are being encroached at higher rates each year (Carbutt, Henwood, & Gilfedder, 2017; Wimberly & Wright, 2013).

Cultivation of less suitable land may lead to even more severe changes to erosion rates, hydrologic regimes shifts, and water quality degradation (Borrelli et al., 2017; Foley et al., 2005; Meadows, Randers, & Meadows, 2004).

Regulations (e.g., laws, ordinances) established at local, state, and national levels have been used to mitigate environmental consequences related to the use of soil and water resources (Claassen, 2011; Samson, Knopf, & Ostlie, 2004). Typically, regulatory policy is designed to mitigate environmental consequences through enforcing/regulating direct changes in management practices of soil and water resources. One example of a regulatory policy that was implemented to directly reduce erosion is the European Common Agriculture Policy. Under this policy, annual erosion rates were reduced by 9.5% over the past decade through the reduction of agricultural expansion on erodible lands and implementation of support practices such as conservation tillage across the continent (Panagos et al., 2015). Water resources have also been directly impacted by regulatory policy to address diminished water quality from grassland conversion to cropland in Brazil (Strauch, Lima, Volk, Lorz, & Makeschin, 2013). Brazil's National Water Agency initiated research through the Water Producer Program and found a potential 40% reduction in sediment loading to the Pipiripau River if agricultural best management practices (BMPs) such as sediment collection ponds, terraces, and multi-diverse crop rotations are implemented (Strauch et al., 2013). Thus, water quality in the Pipiripau River could be improved through the implementation of BMP's as a direct result of regulatory water policy. Overall, regulatory policy appears to be effective for the mitigation of environmental consequences related to soil and water resources as direct action is taken to address specific environmental issues.

Unlike the direct effects of regulatory policy, changes in economic incentives and societal structures may indirectly affect soil and water related externalities (Keeney & Hertel, 2009). Madani and Marino (2009) evaluated water use and availability within Iran's water limited Zayandeh-Rud River Basin and estimated that water consumption increased from 550 to 1100 million cubic meters (MCM) and then plateaued from 2006 to 2025 as an indirect consequence of social policy in which additional water was obtained from other regions to satisfy, in part, increasing public water demands (i.e., luxury demands). Water availability was eventually limited as continually meeting higher water demands altered public perception that water was readily available which caused unsustainable growth in water consumption per capita. Additionally, indirect effects of social and economic policy have been reported in Vietnam, where freshwater availability was estimated to decrease from 19 MCM to <15 MCM during the dry season, when the population increased by 1% and industrial water use increased by 2.5% (i.e., social & economic policies; Phan, Smart, Sahin, Capon, & Hadwen, 2018). Thus, indirect consequences of social and economic policy have the potential to alter soil and water resources and may be difficult to identify since environmental consequences may not be directly linked to social or economic policy change.

Predicting the outcomes of regulatory, economic, and social changes over the long-term may be difficult given the complexity of the system in which they occur. Meadows (2008) defines a system as "a set of things—people, cells, molecules, or whatever—interconnected in such a way that they produce their own pattern of behavior over time." Systems produce outcomes over time, and these outcomes are influenced by a set of decision rules, strategies, and structure that are referred to as "*policies*" (Sterman, 2000). The SD definition of a "policy" differs from the traditional use of the term. Traditional uses of the term "policy" refers to the proposal or adoption of a specific action, whereas the use of the term "policy" in SD refers to the evaluation of an action or other changes to model inputs and their influence on the outcome(s) over time (Sterman, 2000). Additionally, all SD policies result in direct effects on outcomes in a system compared to the traditional indirect and direct influences from policies on outcomes (Sterman, 2000). Direct effects of SD policies may reveal unintended (i.e., unexpected) outcomes or consequences. Because SD policy changes in one system often have unintended impacts on other model inputs, finding an optimal long-term solution that satisfies all stakeholders and their respective concerns may be difficult (Turner et al., 2016a). Predicting long-term impacts of several policies over the same time period and comparing the outcomes of those policies may inform the decision-making process by providing information on the possible intended and unintended outcomes of various proposed solutions (Barlas, 2007; Horschig, Adams, Gawel, Thrän, 2017; Phan et al., 2018; Turner et al., 2013). Therefore, SD policy evaluation techniques may be useful to assess environmental consequences of grassland conversion to cropland resulting from potential changes in regulatory, economic, and social policies, especially in areas with large amounts of grassland where changes may be more notable (Borrelli et al., 2017; Carbutt et al., 2017; Foley et al., 2005).

The North American Great Plains (NAGP) is one of the world's largest major grasslands (Samson et al., 2004). This ecosystem has experienced high conversion rates of native grassland to cropland since the 1920s, in part due to improved farming technology that increased cultivation efficiency as well as federal agricultural policy that directly and indirectly promoted cropland expansion (Barnes, 1993; Samson et al., 2004). The Dust Bowl that occurred within the U.S. Great Plains during the 1930s is an oft-cited example of a soil-related environmental consequence of grassland conversion. For a decade, severe wind erosion led to an estimated 14 billion tons of soil loss (Bolles, Forman, & Sweeny, 2017). More recently, land conversion in the northern portion of the U.S. Great Plains has been of concern as conversion rates in this area were among the highest reported in the U.S. from 2006 to 2011 (Claassen, 2011; Clay et al., 2014; Wimberly & Wright, 2013). Accelerated conversion in the U.S. Great Plains has been attributed to the U.S. Energy Policy Act (2005) which incentivized crop production to meet biofuel demands for the Renewable Fuel Standard (RFS), a policy designed to address climate change concerns and reduce dependence on foreign oil (see Figure 13; McPhail, Westcott, & Lutman, 2011). The RFS indirectly increased grassland conversion to cropland as a result of increased commodity prices and these incentives cascaded into other factors that have been linked to grassland conversion within the Northern Great Plains (NGP; Claassen. 2011; Keeney & Hertel, 2009; Lark, Salmon, & Gibbs, 2015**;** Wimberly & Wright, 2013). For example, Soohoo et al. (2017) reported that erosion potential might increase by 4% in response to a 15% increase in grassland conversion for renewable biofuel crop production within the Missouri River Basin, USA.

Complex social components may also influence environmental consequences within the NGP. The average age of a farm owner is 65 years and fewer younger individuals are remaining within rural communities (USDA, 2012); thus, current land in working farms or ranches may be at risk of not being passed on to future generations but rather leased or sold to larger operations (Claassen, 2011; Pfrimmer, Gigliotti, Stafford, Schumann, & Bertrand, 2017; Sweikert., 2017; Turner et al., 2014). Removal of family ownership to large non-family based operations has been linked to increased grassland conversion to cropland cultivation and potential increases of environmental externalities (Turner et al., 2014, 2017).

Soil-related externalities in the NPG have been partly mitigated through regulatory federal government-sponsored conservation programs such as the Conservation Reserve Program (CRP; part of the U.S. Farm Bill), which provides landowners an option to reestablish grasslands from cropland for 10 to 15 years (Glaser, 1986). However, regulatory programs such as CRP may depend on continued fiscal support from the U.S Farm Bill and may also be altered by the previously mentioned economic incentives and social dynamics which influence decisions related to grassland conversion (Claassen, 2011; Pfrimmer et al., 2017; Sweikert., 2017; Turner et al., 2014, 2017).

Complex regulatory, economic, and social dynamics have been used to estimate grassland conversion under various scenarios within the NGP (Turner et al., 2016b, 2017). Turner et al. (2016b) found that soil externalities might improve, stay the same, or worsen, depending on the policy. For example, Turner et al., (2017) estimated that if the CRP were eliminated in the NGP, soil externalities would reach levels comparable to those estimated during the Dust Bowl era by 2062. Potential soil externality changes across all of the scenarios were classified by a dimensionless index called "Soil Environmental Risk": (SER). However, uncertainties exist as to how the externalities indexed by SER will be realized on the landscape, specifically in regards to changes in erosion rates, hydrological regimes (discharge), and water quality (namely, TSS).

Future changes in SER and specific externalities therein may be of greatest concern in South Dakota as that state has reported the highest rates of grassland-to-row crop conversion (1.0-5.4% annually between 2006 to 2011) compared to any other northern U.S. Great Plains state (Claassen, 2011; Clay et al., 2014; Wright & Wimberly, 2013). Therefore, I forecasted potential changes in annual and cumulative erosion, hydrologic discharge, and TSS in four South Dakota water-catchments from 2012 to 2062 (50 years) under various grassland conversion scenarios. Grassland conversion rates were estimated under various policy options using Systems Thinking (ST) and System Dynamics (SD) approaches. Such information could be useful to decision-makers in evaluating the potential long-term intended and unintended consequences resulting from policy changes in the present and into the future.

METHODS

Study Area

South Dakota is roughly bisected longitudinally by the Missouri River (Figure 14). Climate, soil type, topography, and land use differ between the eastern and western halves of the state. The eastern half of the state has a variable climate characterized by multi-year cycles of wetter and drier periods and receives an average of 50 – 60 cm of precipitation annually (Dozark, 2010; Hubbell, Stevens, Skinner, & Beverage, 1987). The dominant soil order is Mollisol (Miller & Gardner, 2001), and the topography is characterized predominately by plains and rolling hills (USDA, 2006). Land is primarily used for row-crop agriculture production, including corn (*Zea mays*), soybeans (*Glycine max*), and wheat (*Triticum aestivum*; USDA, 2012; see Chapter 2). The western half of the state has a semi-arid climate; annual precipitation levels average between 30 − 40 cm, and the region is frequently subject to drought conditions (Hubbell et al., 1987; Pieper, 2005). The dominant soil orders include Inceptisols, Mollisols, Vertisols, and Alfisols (Miller, 2014), and the topography includes rolling hills, eroded stream valleys, and the Black Hills (USDA, 2006). Land is used primarily for livestock grazing and crop production is mostly wheat, small grains, and alfalfa (*Medicago sativa*; USDA, 2012).

I selected two eastern water-catchments and two western water-catchments as soil and water externalities from grassland conversion are likely to vary between the eastern and western portions of the state due to the aforementioned differences in topography, elevation, soil, climate, and land use. Eastern water-catchments included the Big Sioux and the James rivers, and western water-catchments included the Bad and Belle Fourche rivers (Figure 14). Elevation varies between 284 to 663 m and 305 to 625 m in the Big Sioux and James river water-catchments, respectively (Neupane & Kumar, 2015; USDA, 2009). Croplands comprise 61% and 52% and grasslands comprise 27% and 43% of each water-catchment (Ahiablame, Sinha, Paul, Ji, & Rajib, 2017; Neupane & Kumar, 2015). The elevation of the Bad River water-catchment ranges between 430 and 990 m, and the elevation of the Belle Fourche River water-catchment ranges between 1,000 and 2,208 m (USDA, 2006). Approximately 83% and 14% of the entire Bad River water-catchment is grassland and cropland, respectively, and approximately 66% and 4% of the Belle Fourche River water-catchment is grassland and cropland. (Paul, Rajib, & Ahiablame 2017, see Chapter 2).

System Dynamics Model

An SD model was used to explain past changes in erosion, hydrologic regimes, and TSS as they related to land use (grasslands vs. row crops; see Chapter 2). The model showed relatively high accuracy and precision and, thus, was used in this study to predict future changes for each of these externalities under various policy scenarios. The SD model utilized climate (i.e., precipitation and air temperature), soil type, land capability classifications (LCC), land use area, and crop production data that were appropriate for the size of the study area(s) and timeline of the model (1947 to 2012 Table 13, Figure 14). Land capability classifications provide descriptions of land suitability for agricultural production and range from LCC1 to LCC8, where LCC1 indicates prime farmland and LCC8 indicates lands unable to support agricultural production (Klingebiel & Montgomery, 1961). Within the SD model, three unique sub-models (i.e., erosion, hydrological discharge, and TSS) were linked to reflect the relationships between the externalities; these sub-models and their linkages were evaluated to estimate each specific externality under the various scenarios. Thus, each sub-model utilized the same input data (see Chapter 2).

Eight unique scenarios of regulatory, economic, or social changes were selected to simulate potential changes in erosion, discharge, and TSS (see Table 14 for a full description of each scenario). These scenarios were chosen to provide a range of potential changes in externalities and reveal unintended or delayed consequences of plausible shifts in future land use decisions; such decisions may be influenced by agricultural economics, federal environmental regulations, local farm and ranch culture and rural community dynamics (see Turner et al., 2017). Each scenario was associated with an annual percent change of total cropland and grassland (ha/yr) within each USDA LCC. Simulations were conducted between the years 2012 and 2062, which provided 50 years of quantitative estimates for each externality.

Externality changes within the eight scenarios were further evaluated under two different crop cultivation practices: conservation tillage and conventional tillage. Alteration of soil structure and cover was assumed to be minimized under conservation tillage and increased under conventional tillage. Each tillage practice was obtained from a previously established Revised Universal Soil Loss Equation cover and management (i.e., tillage) factor index, where 0.05 is the value for no-tillage a conservation tillage practice and 0.16 is the value for chisel-tillage a conventional tillage practice (Foster et al., 2002; Franzmeier, Yahner, Steinhardt, & Schulze, 1986; see Chapter 2). The inclusion of these values within the SD model altered erosion and water infiltration rates in the historic estimates (see Chapter 2). The inclusion of each tillage practice within the scenarios assumes that the practice will occur at a constant rate over time (i.e. fixed). Thus, the effect of tillage type on each of the externalities was evaluated under this assumption for each scenario. The simulation of each policy scenario and tillage practice combination resulted in annual estimates of erosion rates [million metric-tons (Mt/yr)], discharge [million cubic meters (MCM/yr)], and average TSS (mg/L/yr) for each water-catchment from 2013 to 2062. Differences in annual rates were compared among the various scenarios through visual analysis. Annual estimates for each externality were added together across the entire simulation period (2013 to 2062) to estimate cumulative values for erosion, discharge, and TSS over this 50-year period. Cumulative values for each scenario and tillage combination were evaluated to identify the best-case (lowest) and worst-case (highest) scenario for each externality. Additionally, each scenario and tillage practice combination was compared to the base-case by calculating the percent differences in cumulative erosion (Mt), discharge (MCM) and TSS (mg/L) for each scenario from the base case scenario.

RESULTS

Overall, higher annual and cumulative erosion estimates were noted under conventional tillage practices compared to those under conservation tillage practices for each of the eight policy scenarios (Figures 15 – 17). Removal of all grassland and the elimination of CRP were the only two scenarios to increase annual erosion above the base-case in all four water-catchments (Figures 15 and 16). The inclusion of tillage practices in the scenarios altered the overall patterns of annual erosion over time. Annual erosion substantially increased throughout the forecast and then plateaued under conventional tillage. However, under conservation tillage, annual erosion tracked closely to the base-case initially but then substantially increased above the base-case until the end of the forecast. Annual erosion estimates were lower than the base-case for the scenarios that included doubled farmland costs, integrated livestock, reinvigorated youth, and doubled conservation compliance (Figures 15 and 16), but patterns of annual erosion were influenced by tillage practices. Annual erosion estimates initially increased at a similar rate as the base-case under conventional tillage but then gradually decreased for the remainder of the forecast. Conversely, under conservation tillage, estimates of annual erosion were initially lower than the base-case but then plateaued and remained low throughout the simulation. Cumulative erosion estimates were lower than the base-case for doubled farmland costs, integrated livestock, reinvigorated youth, and doubled conservation compliance under both tillage types and in each water-catchment (Figure 17, Tables 15 and 16). However, cumulative erosion was above the base-case for removal of all grassland and the elimination of CRP under both tillage types and in each water-catchment (Figure 17, Tables 15 and 16). Overall, cumulative erosion estimates were 70 – 77% higher when all grasslands were removed and conventional tillage was in practice (i.e., the worst-case scenario) compared to cumulative erosion estimated when decreased livestock costs were coupled with conservation tillage (i.e., the best-case scenario; Figure 17 and Appendix D).

Similar to erosion, annual and cumulative hydrologic discharge estimates were higher under conventional tillage compared to conservation tillage (Figures 18 − 20). However, patterns of annual discharge and cumulative totals varied by scenario, tillage type, and between each study area (Figures 18 − 20, Tables 15 and 16). Under both tillage types, annual discharge patterns in the Big Sioux River tracked slightly under the base-case scenario until 2034 and then rose and remained marginally higher than the base-case throughout the forecast for the five scenarios that promoted grassland conservation or restoration (i.e., livestock costs were decreased, conservation compliance doubled, youth reinvigorated, livestock integrated, and land cost were doubled; Figures 18A and 19A). Conversely, Big Sioux River patterns of annual discharge resulting from the removal of CRP or all grassland were above the base-case until 2034 and then decreased slightly below the base-case throughout the forecast under both tillage types (Figures 18A and 19A). Cumulative discharge estimates for the elimination of CRP and removal of all grassland were also below the base-case for both tillage types (Figure 20A and Table 15). However, cumulative discharge estimates were greater than the base-case when livestock costs were decreased, conservation compliance doubled, youth reinvigorated, livestock integrated, and land cost were doubled under both tillage types (Figure 20A and Table

15). Overall, Big Sioux River cumulative discharge estimates were 4% higher for decreased livestock costs with conventional tillage (i.e., worst case) compared to the removal of all grasslands coupled with conservation tillage (i.e., the best-case scenario; Figure 20A and Appendix D).

In the James River, the elimination of CRP and removal of all grassland were the only two scenarios that resulted in annual discharge estimates that were greater than the base-case under both conservation and conventional tillage (Figures 18B and 19B). The pattern of annual discharge was slightly greater than the base-case throughout the forecast for these two scenarios and under both tillage types. James River annual discharge estimated under reduction of livestock costs, doubled conservation compliance, reinvigorated youth, integrated livestock, or doubled land costs tracked closely but slightly below the base-case under both conservation and conventional tillage (Figures 18B and 19B). Additionally, cumulative discharge for each of these five scenarios was below the base-case (Figure 20B, Table 15). However, cumulative discharge was greater than the base-case when all grassland was removed and CRP eliminated under both tillage types (Figure 20B and Table 15). Overall, cumulative discharge was 7% higher from the removal of all grasslands under conventional tillage (i.e., the worst-case scenario) when contrasted with decreased livestock costs under conservation tillage (i.e., the lowest and best-case scenario; Figure 20B and Appendix D).

In the Bad River, the pattern of annual discharge was slightly above the base-case for each scenario throughout the forecast when coupled with conservation tillage (Figure 18C). However, under conventional tillage, the pattern of annual discharge was slightly above the base-case when CRP was eliminated, land cost doubled, or all grassland removed throughout the entire forecast (Figure 19C). Conversely, Bad River annual discharge tracked slightly beneath the base-case throughout the entire forecast when the scenarios for reinvigorated youth, doubled conservation compliance, decreased livestock costs, or integrated livestock were coupled with conventional tillage (Figures 19C). Under conservation tillage, cumulative discharge was greater than the base-case for each scenario (Figure 20C and Table 16). However, under conventional tillage cumulative discharge was above the base-case when CRP was eliminated, land cost doubled, or all grassland removed (Figure 20C and Table 16). Additionally, cumulative discharge was below the base-case for reinvigorated youth, doubled conservation compliance, decreased livestock costs, or integrated livestock under conventional tillage (Figure 20C and Table 16). Cumulative discharge estimates were < 1% higher from the removal of all grasslands under conventional tillage (i.e., the worst-case scenario) compared to the base-case under conservation tillage (i.e., the best-case scenario; Figure 20C and Appendix D).

Within the Belle Fourche River, the elimination of the CRP and removal of all grassland patterns of annual discharge estimates remained slightly higher than the base-case throughout the forecast with the use of conservation tillage (Figure 18D). However, the pattern of annual discharge tracked slightly under the base-case for scenarios that promoted greater grassland conservation or restoration (i.e., the reduction of livestock costs, doubled conservation compliance, reinvigorated youth, integrated livestock, or doubled land costs) when coupled with conservation tillage (Figure 18D). Under conventional tillage, the pattern of annual discharge estimates from the removal of all grassland was above the base-case, while estimates from the elimination of CRP was synchronous with the base-case throughout the forecast (Figure 18D). Additionally, the pattern of annual discharge tracked slightly below the base-case throughout the forecast when livestock costs were decreased, conservation compliance doubled, youth reinvigorated, livestock integrated, and land cost doubled coupled with conventional tillage (Figure 18D). Cumulative discharge was below the base-case under these five previously mentioned scenarios when coupled with conservation tillage (Figures 20D and Table 16). However, cumulative discharge was greater than the base-case for the elimination of CRP and the removal of all grassland under conservation tillage (Figure 20D and Table 16). Under conventional tillage, cumulative discharge was greater than the base-case when all grassland was removed and about equal to the base-case with the elimination of CRP (Figure 20D and Table 16). Conversely, cumulative discharge was below the base-case when land cost doubled, livestock integrated, youth reinvigorated, conservation compliance doubled, and livestock production cost decreased under conventional tillage (Figure 20D, Table 16). Cumulative discharge estimates were 10% higher from the removal of all grassland coupled with conventional tillage (i.e., the worst-case scenario) when compared to decreased livestock cost under conservation tillage (i.e., the best-case scenario; Figure 20D and Appendix D).

Estimates of TSS were directly influenced by both erosion (i.e., more influence) and discharge (i.e., less influence) estimates, and thus annual patterns of TSS under various scenarios and tillage practices followed similar patterns as those reported for erosion and discharge above (Figures 21 – 23). Elimination of CRP and removal of all grassland were the only two scenarios to increase annual TSS above the base-case in all four water-catchments under both tillage types (Figures 21 – 23). The inclusion of tillage practices with these two scenarios altered annual TSS patterns. Annual TSS estimates steadily increased throughout the forecast and then plateaued under conventional tillage. Under conservation tillage, the same two scenarios as mentioned above tracked closely to the base-case but then gradually increased above the base-case until the end of the forecast.

Annual TSS estimates were smaller than the base-case for the scenarios that promoted grassland conservation and restoration (i.e., doubled farmland costs, integrated livestock, reinvigorated youth, and doubled conservation compliance; Figures 21 – 23), but patterns were influenced by tillage practices. Under conservation tillage, estimates of annual TSS for these four scenarios remained slightly lower than the base-case throughout the simulation. Conversely, under conventional tillage, annual TSS estimates for these four scenarios initially increased at a similar rate as the base-case but then gradually decreased for the remainder of the forecast. Cumulative TSS was greater than the base-case for the elimination of CRP and removal of all grassland under both tillage types (Figure 23, Tables 15 and 16). However, cumulative TSS was smaller than the base-case for doubled farmland costs, integrated livestock, reinvigorated youth, and doubled conservation compliance under both tillage types (Figure 23, Tables 15 and 16). Overall, cumulative TSS was 70 – 76% higher when all grassland was removed and conventional tillage was in practice (i.e., the worst-case scenario) compared to TSS from decreased livestock costs coupled with conservation tillage (i.e., the best-case scenario; Figure 23 and Appendix D).

DISCUSSION

To my knowledge, this is the first study to predict annual and cumulative erosion, discharge, and total suspended solids in the Northern Great Plains (NGP) under current conditions and potential policy changes over a 50-year period. Evaluation of grassland conversion scenarios indicated that erosion, discharge, and water quality may indeed be influenced by regulatory, economic, and social policy changes. In general, expected responses were captured for each environmental externality, tillage practice, and policy type throughout the eastern and western South Dakota water-catchments.

Overall, the System Dynamics (SD) model forecasts for both annual and cumulative erosion increased over time under scenarios that reduced grassland and decreased under scenarios that conserved or restored grassland. Other studies have reported similar changes in erosion from grassland conversion to cropland. Bakker et al. (2008), estimated that erosion increased from 9 metric-tons/ha/yr in 1995 to 16 metric-tons/ha/yr in 2001 due to cropland expansion onto grassland with erodible soils in Hageland, Belgium. Within South Dakota, Sishodia (2010) found that annual erosion rates increased from 0.9 to 28.7 metric-ton/ha/yr when CRP grassland was converted to cropland in 2009. Further, erosion results from my study worsened under conventional tillage and improved under conservation tillage for all scenarios and in each study area and supported previous research that tillage practices may further exacerbate or ameliorate erosion. Miller (2014) estimated that mean erosion rates increased from 1 metric-tons/ha/yr under conservation tillage to 10 metric-tons/ha/yr within South Dakota’s Bad River and Big Sioux River water-catchments in 2014. Clay et al. (2014) reported that erosion decreased from 7.2 metric-ton/ha/yr in 1982 to 4.8 metric-ton/ha/yr in 2007 for South Dakota, North Dakota, and Nebraska (NRCS, 2007) through grassland conservation under CRP coupled with the use of conservation tillage practices on croplands. During this time (i.e., 1987 to 2007), CRP increased by 372% (146,011 to

543,614 hectares) throughout South Dakota and no-tillage had a 41% average adoption rate from 2004 to 2010 in the eastern half of the state (i.e., spatially ununiform; Clay et al., 2012, 2014). In general, grasslands play a vital role in the prevention and mitigation of soil loss by erosion. Erosion is diminished within grassland areas through the addition of ground cover which reduces the force of kinetic energy from precipitation that causes soil displacement (i.e., rainfall erosivity; Foster et al., 2002). Further, fibrous roots of grasses hold soil in place which also reduces soil erosion and transport from surface water runoff (Kort, Collins, & Ditsch, 1998). In respect to cultivation, no-tillage reduces erosion by improving ground cover (Nouwakpo, Song, & Gonzalez, 2018) and decreasing soil disturbance, preserving water-stable soil aggregates that are more resistant to erosion than soil aggregates under conventional tillage (Beare, Hendrix, Cabrera, & Coleman, 1994). Therefore, understanding grassland conversion to cropland and tillage impacts on erosion is important for future soil conservation efforts as land use change is anticipated to continue.

Discharge estimates did not appear as responsive to grassland area changes compared to annual erosion estimates; I found that discharge volumes may only slightly increase as grassland conversion to cropland increases. My results were somewhat surprising as general increases in discharge following grassland conversion to cropland have been reported in other areas. For example, runoff increased from 159 – 167 mm/yr (6.2%) and discharge decreased from 538 – 467 mm/yr (-13%) following a 20% expansion of cropland and a 4% reduction in grassland from 1973 to 2010 in Ethiopia (Woldesenbet, Elagib, Ribbe, & Heinrich, 2017). However, hydrologic studies within South Dakota water-catchments have shown less of a response of discharge in relation to land use change. Kibria, Ahiablame, Hay, and Djira (2016), found streamflow changes in only two of 18 South Dakota water-catchments that underwent grassland conversion from 1951 to 2013. Neupane and Kumar (2015) estimated that surface water runoff increased between 2 – 4% annually from a 2 – 10% expansion of cropland hectares within the Big Sioux River from 1980 to 2013. Interestingly, Hong (2017) estimated that streamflow increased by 7% from increased baseflow rather than surface water runoff when grasslands were expanded within the Big Sioux River from 1996 to 2014. Similarly, increased discharge was estimated for scenarios that increased grassland from the SD model within the Big Sioux River and by other studies in areas outside of South Dakota (Qiu, Yin, Jian, & Geng, 2011). Within the James River, discharge was projected to increase by only 6 – 8% as a result of crop, grass, water, and urban land use change by 2055 (Ahiablame et al., 2017). Surface water runoff was estimated to increase by 5 – 6% between 1981 and 2014 as a result of a 1.4% decrease in grassland within the Bad River water-catchment (Paul et al., 2017). Changes in discharge may be more sensitive to climatic factors such as precipitation intensity, duration, frequency, and seasonality (e.g., wetter falls and springs) compared to grassland conversion to cropland (Ahiablame et al., 2017; Kibria et al., 2016). The SD model I developed for my study included some climatic factors, but it may not have captured the variability of climate in the future. Several studies note that climate within South Dakota will have warmer air temperatures and experience more frequent and intense precipitation events over the next 50 years (EPA, 2016; Meehl et al., 2017; Pierce, Cayan, Maurer, Abatzoglou, & Hegewisch, 2015; Pierce, Cayan, & Thrasher, 2014) Thus, my estimated projections of change may be conservative relative to what is possible and future evaluation of discharge from grassland conversion to cropland may need to include anticipated climate change projections to detect more substantial changes in annual discharge over time.

Alteration of TSS estimates appeared to be synchronous with changes in erosion rates and less related to changes in discharge; these results agree with other water quality studies (Alibuyog et al., 2009; Hong, 2017; Lentsch, 2011). To my knowledge, only one study has reported the impact of grassland conversion to row-crop agriculture on TSS concentrations (Weller, Jordan, Correll, & Liu, 2003). Weller et al. (2003) found that TSS concentration increased from 124 to 229 mg/L as a result of doubled cropland and decreased grassland in Maryland's Patuxent watershed. Similarly, observed TSS decreased from > 158 mg/L in 2004 to < 29 mg/L in 2007 within the Belle Fourche River when the best management practice (BMP) of grassland riparian strips was utilized to improve water quality (EPA, 2011). Preventing eroded soil from reaching waterways is important as this sediment yield influences TSS levels (Meybeck et al., 2013; Walling, 1999). Upon entering a waterway, sediments are either suspended in the water column or settle-out to the bottom, which becomes sediment bedload (Vanoni, 2006). Increased discharge flow velocity may cause bedload to be resuspended in the water column through saltation (i.e., sheer-stress; Fernandez-Luque, & Van Beek, 1976). Total discharge volume may decrease or increase TSS levels through dilution from large volumes of water during wet years (e.g., flood conditions) or low volumes during dry years (e.g., drought; Fayyadh, 2011). Thus, TSS is a direct consequence of altered erosion and discharge from grassland conversion and may continue to change in relation to future policy decisions regarding land use in South Dakota.

Land use decisions have been linked to the direct and indirect influence of regulatory, economic, and social policies (Claassen; 2011; Dimitri, Effland, & Conklin, 2005; Keeney & Hertel, 2009; Rosegrant, Cai, & Cline, 2002). Within the SD model, changes to annual grassland and cropland area used to drive each scenario were directly altered by regulatory policies (i.e., elimination of CRP and doubled conservation compliance) or were indirectly changed by economic (i.e., doubled land cost and decreased livestock production costs) and social policies (i.e., integrated livestock and reinvigorated youth). Other studies have documented similar direct and indirect policy influences on land use decisions. For example, the European Union (EU) Common Agriculture Policy (CAP) has addressed the impact of climate change on land use where regulatory policy directly maintained current agricultural land to support local food production, despite climate changes (Olesen & Bindi, 2002). Additionally, EU CAP has been used to indirectly decrease land use change by maintaining viable rural societies and their cultural heritage through subsidies to maintain agricultural profitability despite changes in market structures or technology which, in turn, prevents land abandonment and desertification (Olesen & Bindi, 2002; Rounsevell, Ewert, Reginster, Leemans, & Carter, 2005). Within the U.S., the Farm Security and Rural Investment Act (2002) had an indirect impact on land use decisions through counter-cyclical payments that only provided financial assistance if commodity prices fell below expected market values; thus, this safety net may have indirectly encouraged land expansion as it mitigated the risk of market commodity prices (Westcott, Young, & Price, 2002). The U.S. Energy Policy Act (2005) also indirectly increased cropland expansion onto grassland to meet biofuel demands for the Renewable Fuel Standard (RFS) that increased crop commodity prices (McPhail et al., 2011). Therefore, consideration of the role of policy is important as SD model results and other studies indicated that policies directly and indirectly influenced annual grassland conversion or restoration rates that in turn directly altered environmental consequences over time (Foley et al., 2005; Wang, Lin, Glendinning, & Xu, 2018).

Overall, specific environmental consequences of annual and cumulative erosion, discharge, and TSS were influenced in similar ways by policy scenarios and across water-catchments with only one exception (see model results of discharge within the Big Sioux River). Similar policy effects throughout the four water-catchments imply that policies may be implemented throughout the state and achieve the same general results of worsening or mitigating each environmental consequence over a 50-year period. However, changes in environmental consequences differed between western (Bad and Belle Fourche rivers) and eastern (Big Sioux and James rivers) South Dakota water-catchments in response to policy scenarios. Large differences in scale exist between eastern (area = 23,000 − 53,000 km^2) and western water-catchments (area = 8,000 − 11,000 km^2) but cumulative percent changes above and below the base-case revealed differences in externalities between eastern and western water-catchments. Results indicated that eastern water-catchments may be less sensitive to grassland conversion to cropland or grassland conservation and restoration compared to western water-catchments for each externality. Additionally, eastern water-catchments had less annual variability compared to western water-catchments, which indicated that western water-catchments may be more susceptible to annual changes of environmental consequences from grassland conversion-to-row crop agriculture. These implications concur with factors known to increase potential environmental consequences from grassland conversion such as erodible soils (e.g., clay; Foster et al., 2002) and steeper slopes that accelerate surface water runoff (Blanco & Lal, 2010). Western South Dakota water-catchments are characterized by these previously mentioned factors compared to the less erodible soils and gentler slopes within the eastern water-catchments (see study area description in the Methods section above). Consequently, policy scenarios may have greater magnitudes of increased or decreased environmental consequences in western water-catchments compared to eastern water-catchments in South Dakota. Environmental consequences in other areas may respond in a similar fashion to policies that impact grassland conversion as most policy changes are set at the federal level and affect entire states or regions within the U.S. (Claassen, 2011; Glaser, 1986).

The System Dynamics (SD) model in this study was able to incorporate complex regulatory, economic, and social factors in order to forecast changes in environmental consequences over time. This model provides a robust and powerful tool to forecast erosion, discharge, and TSS under various landscape-scale scenarios. Forecasts indicated that there may be concerns regarding the consequences of future land use change as grassland conversion was estimated to increase each potential environmental externality. Additionally, direct and indirect effects of policies on grassland conversion to cropland or grassland conservation and restoration may have strong influences on future environmental consequences from land use change. System Dynamics model forecasts indicated that each policy scenario had the same general effect on each environmental externality, but policy changes may be of more concern in the western water-catchments compared to the eastern water-catchments over time. Information presented here may provide producers, policymakers, and other stakeholders more specific quantitative estimates to assess the future impact of grassland conversion. Additionally, comparisons between these estimates provide support that addressing grassland conversion issues and cultivation practices are important in order to preserve and conserve soil and water resources.

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Table 13. Data sources used in the erosion, hydrologic, and total suspended solids sub-models. For additional information see the Methods section in Chapter 2: Methods.

Data type	**Data source**
Climate	National Oceanic and Atmospheric Administration (NOAA; https://www.ncdc.noaa.gov/data-access)
Soil characteristics and land capability classifications	National Resource Conservation Service (NRCS) Spatial Gateway (https://datagateway.nrcs.usda.gov)
Annual land use	United States Geological Survey-Earth Resource Observation and Science (USGS-EROS; https://landcovermodeling.cr.usgs.gov/projects-.php)
Crop Production	U.S. Agriculture Census (https://www.agcensus.usda.gov/Publications-/2012/Full_Report/Census_by_State)

Table 14. Name, scenario category, and description of regulatory, economic, and social policy scenarios from Turner et al. (2017).

Scenario name	Scenario category	Scenario description
Base-case	Baseline	Estimated future rates of grassland conversion (2013-2062) to create a "status quo" in which to compare other scenarios.
Livestock Integration	Social	Altered grassland conservation rates with livestock integration which involves the recoupling of cattle ranching and farming production.
Reinvigorated Youth	Social	Targeted the youth-demographics of farmers, ranchers, and landowners within agricultural communities to play a stronger role in land use decisions.
CRP 0%	Regulatory	The Conservation Reserve Program enrollment was set to zero from 2012 to 2062, which decreased cropland.
Cons. Comp. X2	Regulatory	Conservation compliance was doubled which increased grassland by 32% for every 1% increase in cropland.
Land cost X2	Economic	Total farmland was reduced as average cropland cost was doubled slightly lowering cropland.
Livestock costs X0.75	Economic	Livestock production costs were reduced by 25%, which increased grassland.
Grassland 0%	Environmental	A scenario was added to those developed by Turner et al. (2017), which decreased grassland by 10% each year from 2012-2062 until very few hectares of grassland remained in each water-catchment. The purpose of this scenario was to capture the upper extreme of environmental externalities.

Table 15. Changes in cumulative erosion [million metric-tons (Mt)], discharge [million cubic meters (MCM)], and total suspended solids (mg/L; TSS) from 2013 to 2062 for each scenario (reported as a percentage) compared to the "Base-case" scenario (reported as a whole number) for two eastern South Dakota water-catchments (Big Sioux and James rivers). "Conservation tillage and conventional tillage are denoted by "cons." and "conv.", respectively, for each metric. See Table 14 for a full description of scenario names.

Water-catchment	Scenario	Erosion (cons.)	Erosion (conv.)	Discharge (cons.)	Discharge (conv.)	TSS (cons.)	TSS (conv.)
Big Sioux River	**Base-case**	**334**	**397**	**68,391**	**69,387**	**1,304**	**3,972**
	Livestock Integration	-3	-5	1	3	-2	-2
	Reinvigorated Youth	-5	-7	4	3	< 1	-5
	CRP 0%	1	2	< 1	-1	1	< 1
	Cons. Comp. X2	-5	-8	4	3	-1	-6
	Land cost X2	-1	-2	1	< 1	< 1	-3
	Livestock costs X0.75	-8	-12	4	3	-4	-10
	Grassland 0%	10	16	< 1	-2	12	16
James River	**Base-case**	**365**	**968.5**	**64,087**	**64,341**	**4,401**	**11,791**
	Livestock Integration	-2	-2	< 1	< 1	-1	-2
	Reinvigorated Youth	-2	-3	< 1	< 1	-2	-2
	CRP 0%	1	1	< 1	< 1	1	1
	Cons. Comp. X2	-2	-3	< 1	< 1	-2	-3
	Land cost X2	-1	-1	< 1	< 1	-1	-1
	Livestock costs X0.75	-3	-4	-2	< 1	-3	-4
	Grassland 0%	17	24	5	5	15	21

Table 16. Changes in cumulative erosion [million metric-tons (Mt)], discharge [million cubic meters (MCM)], and total suspended solids (mg/L; TSS) from 2013 to 2062 for each scenario (reported as a percentage) compared to the “Base-case” scenario (reported as a whole number) for two western South Dakota water-catchments (Bad and Belle Fourche rivers). “Conservation tillage and conventional tillage are denoted by “cons.” and “conv.”, respectively, for each metric. See Table 14 for a full description of scenario names.

Water-catchment	**Scenario**	**Erosion (cons.)**	**Erosion (conv.)**	**Discharge (cons.)**	**Discharge (conv.)**	**TSS (cons.)**	**TSS (conv.)**
Bad River	**Base-case**	**125**	**261**	**26,936**	**26,977**	**13,524**	**27,784**
	Livestock Integration	-6	-19	< 1	< 1	-8	-3
	Reinvigorated Youth	-7	-24	< 1	< 1	-10	-25
	CRP 0%	5	6	< 1	< 1	2	6
	Cons. Comp. X2	-10	-30	< 1	< 1	-13	-31
	Land cost X2	-1	-8	< 1	< 1	-4	-8
	Livestock costs X0.75	-14	-39	< 1	< 1	-17	-40
	Grassland 0%	35	65	< 1	< 1	32	66
Belle Fourche River	**Base-case**	**30**	**50**	**12,795**	**12,996**	**2,623**	**4,437**
	Livestock Integration	-8	-22	-1	-2	-7	-22
	Reinvigorated Youth	-11	-25	-2	-4	-10	-25
	CRP 0%	2	7	1	< 1	1	5
	Cons. Comp. X2	-15	-35	-4	-4	-15	-35
	Land cost X2	-3	-9	< 1	-2	-3	-8
	Livestock costs X0.75	-16	-38	-6	-7	-17	-40
	Grassland 0%	49	125	5	3	39	106

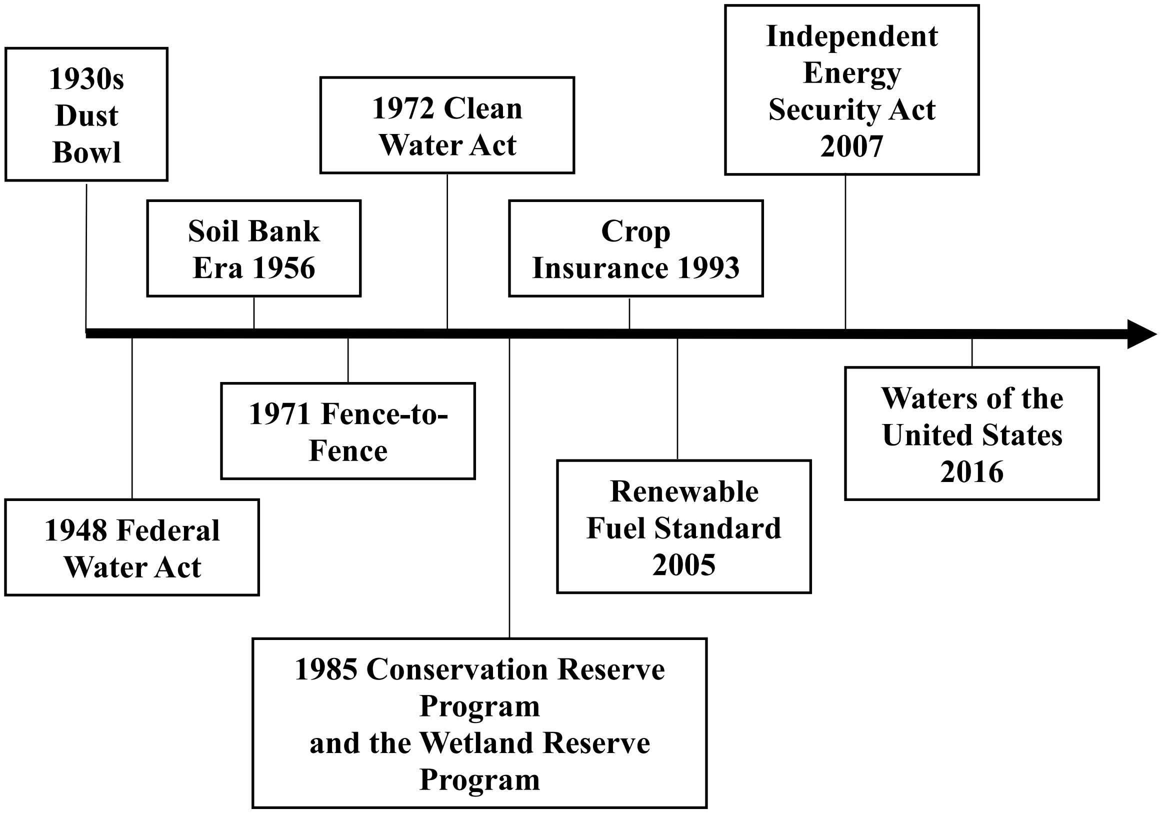

Figure 13. Major United States federal policies, historical conservation landmarks, and programs related to soil, water, and land use. Note that “fence-to-fence” was not a federal policy but rather a political agenda to maximize crop production by increasing cropland area (see Turner et al., 2014).

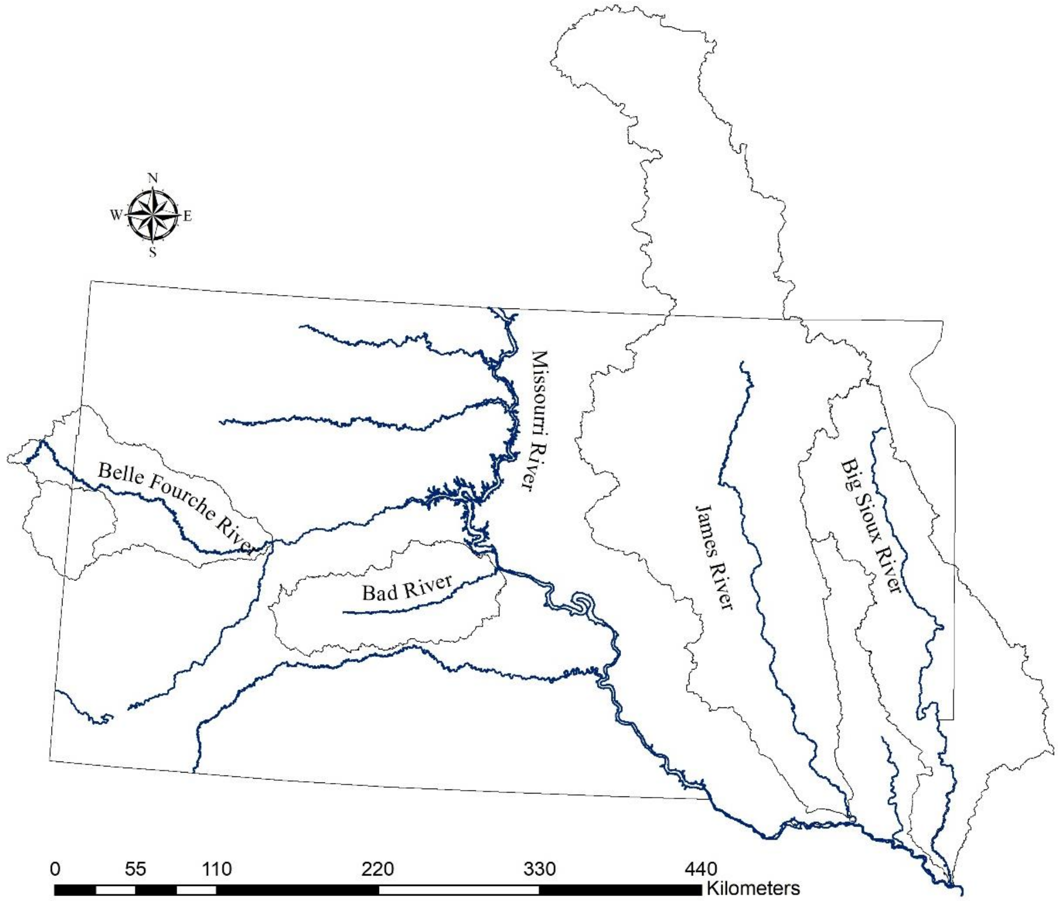

Figure 14. Map of the state of South Dakota, USA, including the four water-catchments of this study: Big Sioux (area = 22,910 km^2), James (area = 54,742 km^2), Bad (area = 8,225 km^2), and Belle Fourche (area = 11,129 km^2) rivers.

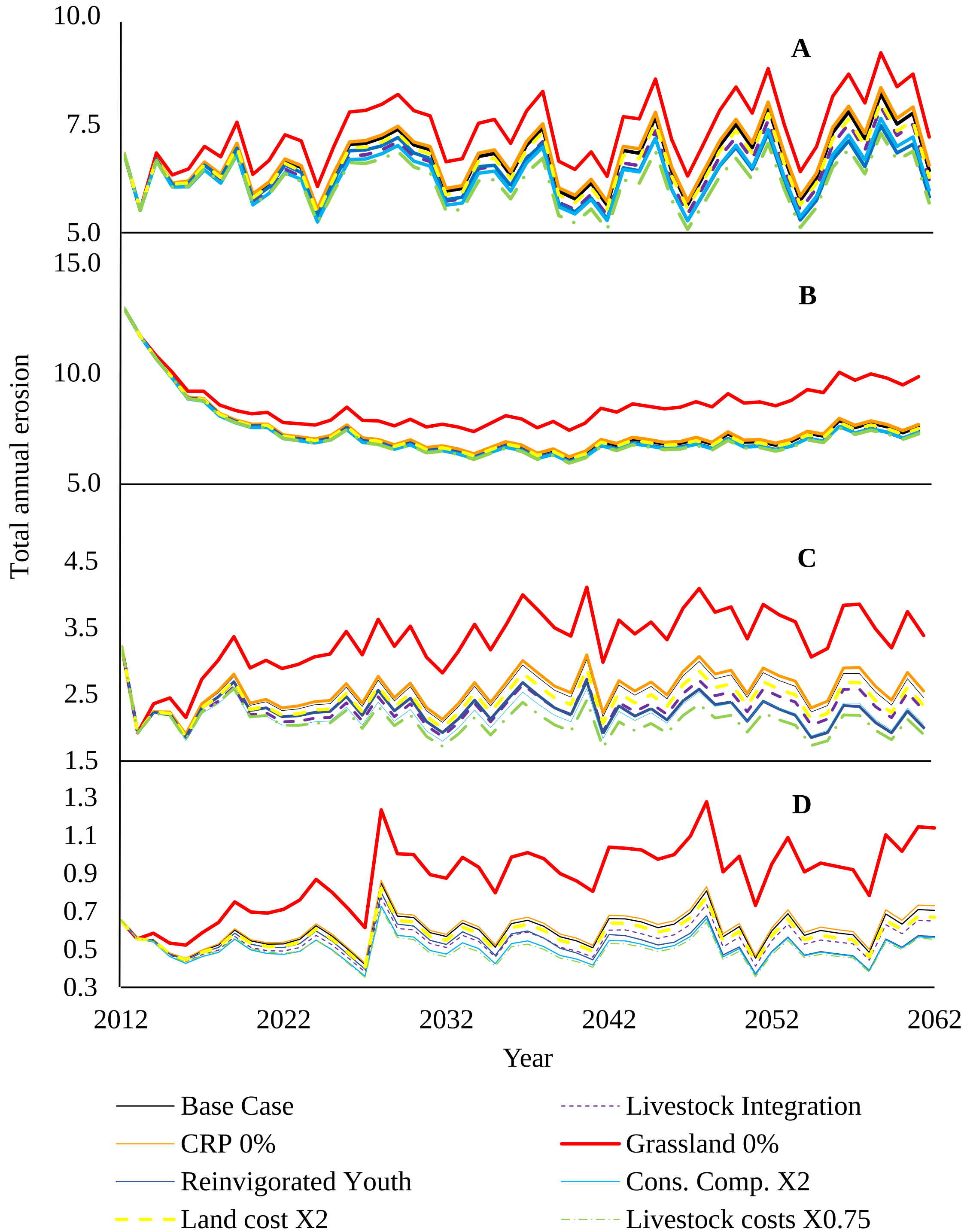

Figure 15. Annual erosion estimates [million metric-tons (Mt)] between 2012 and 2062 for the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments for the eight policy scenarios modeled under conservation tillage practices. Scenario names are described in Table 14.

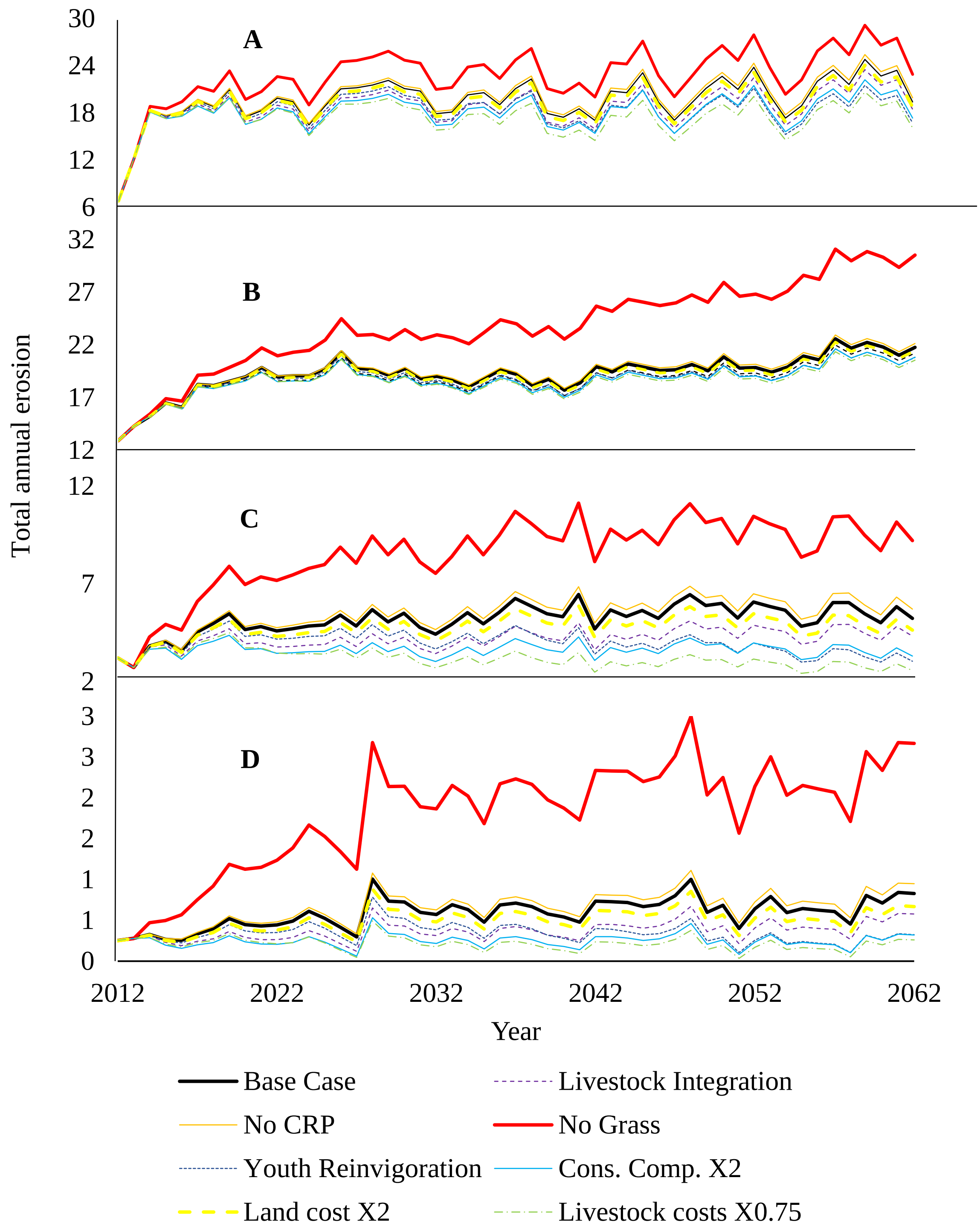

Figure 16. Annual erosion estimates [million metric-tons (Mt)] between 2012 and 2062 for the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments for the eight policy scenarios modeled under conventional tillage practices. Scenario names are described in Table 14.

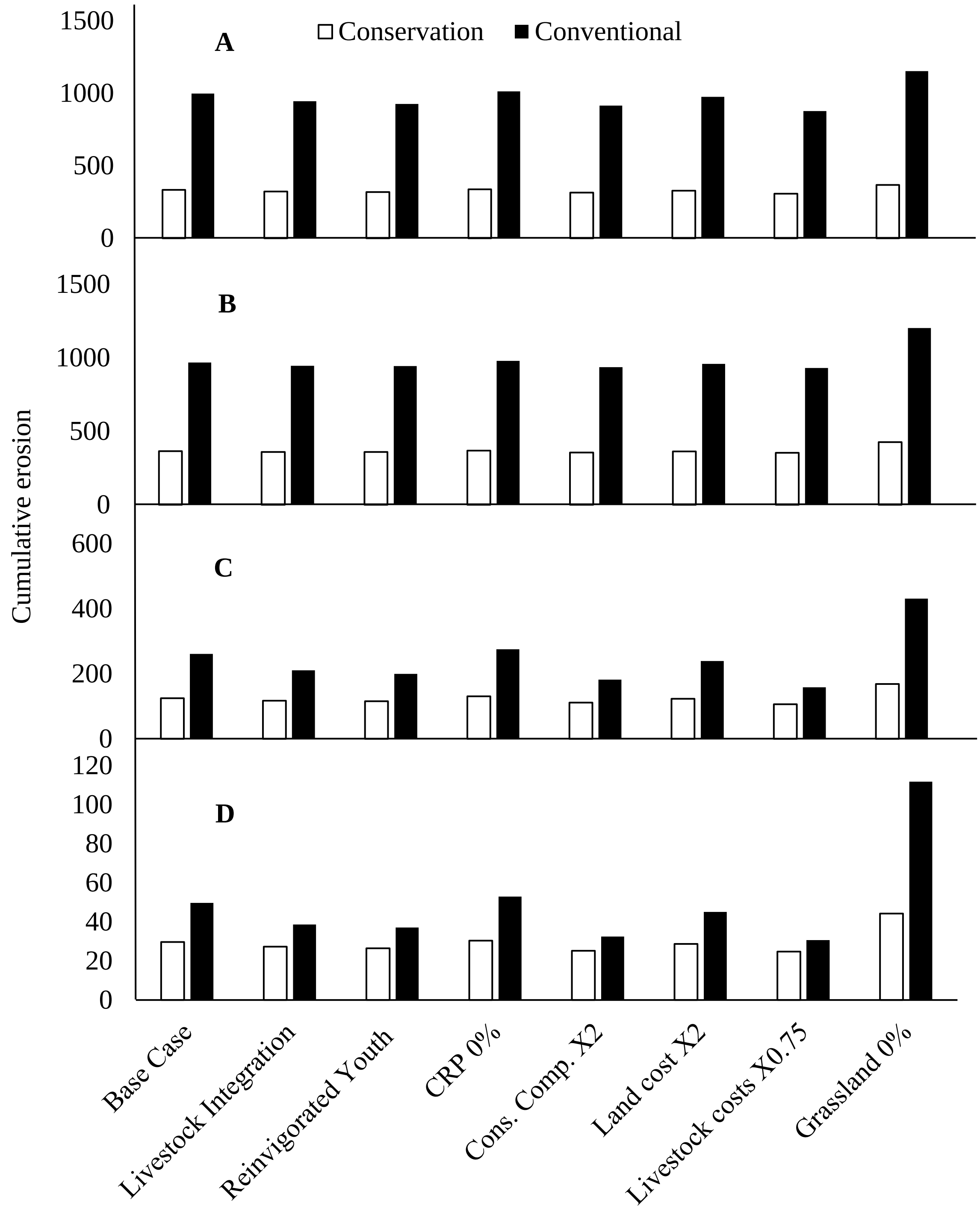

Figure 17. Total cumulative erosion [million metric-tons (Mt)] estimated from 2013 to 2062 for each of the eight scenarios (see Table 14 for scenario names) under both conservation and conventional tillage for Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments.

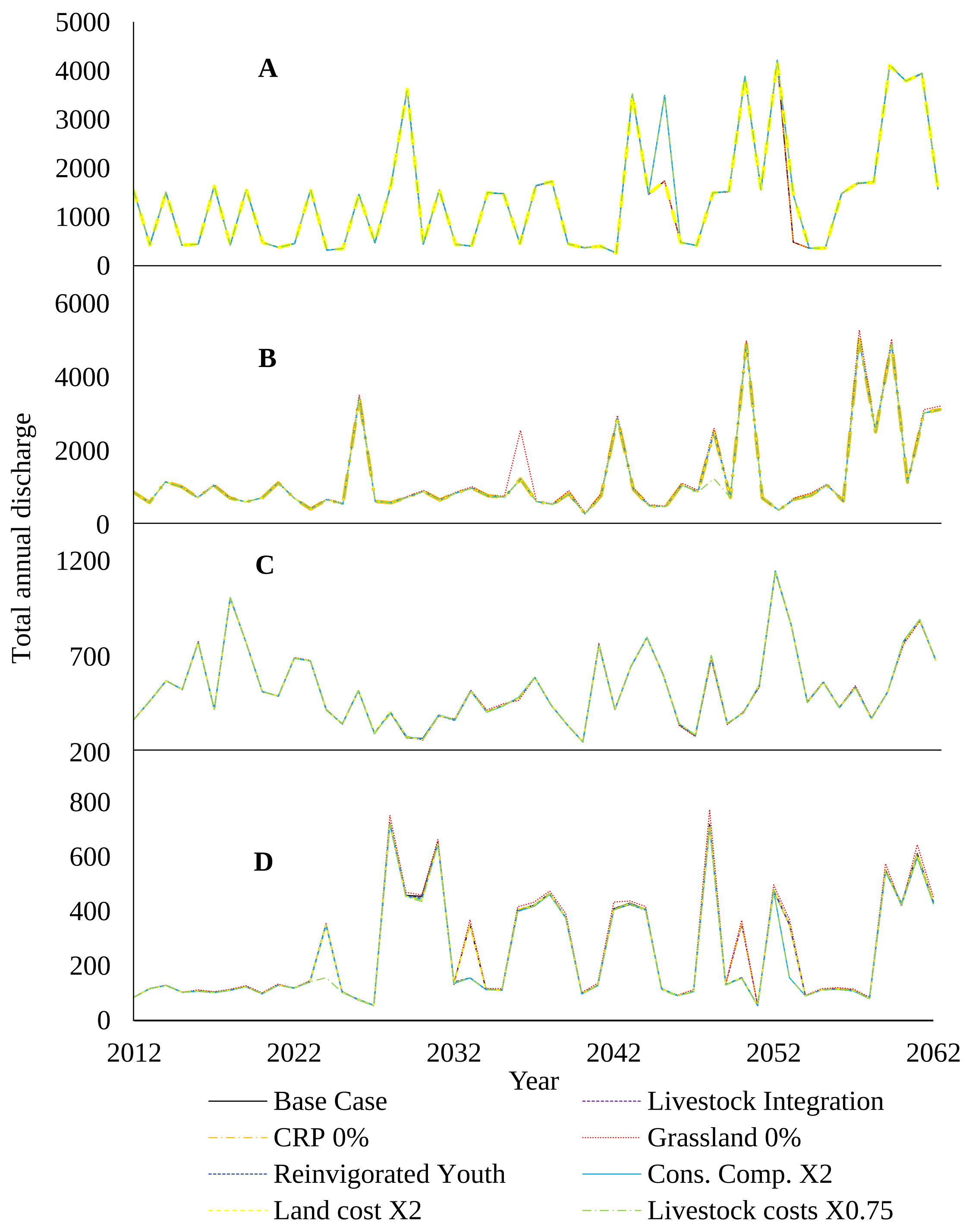

Figure 18. Annual discharge estimates [million cubic meters (MCM)] between 2012 and 2062 for the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments for the eight policy scenarios modeled under conservation tillage practices. Scenario names are described in Table 14.

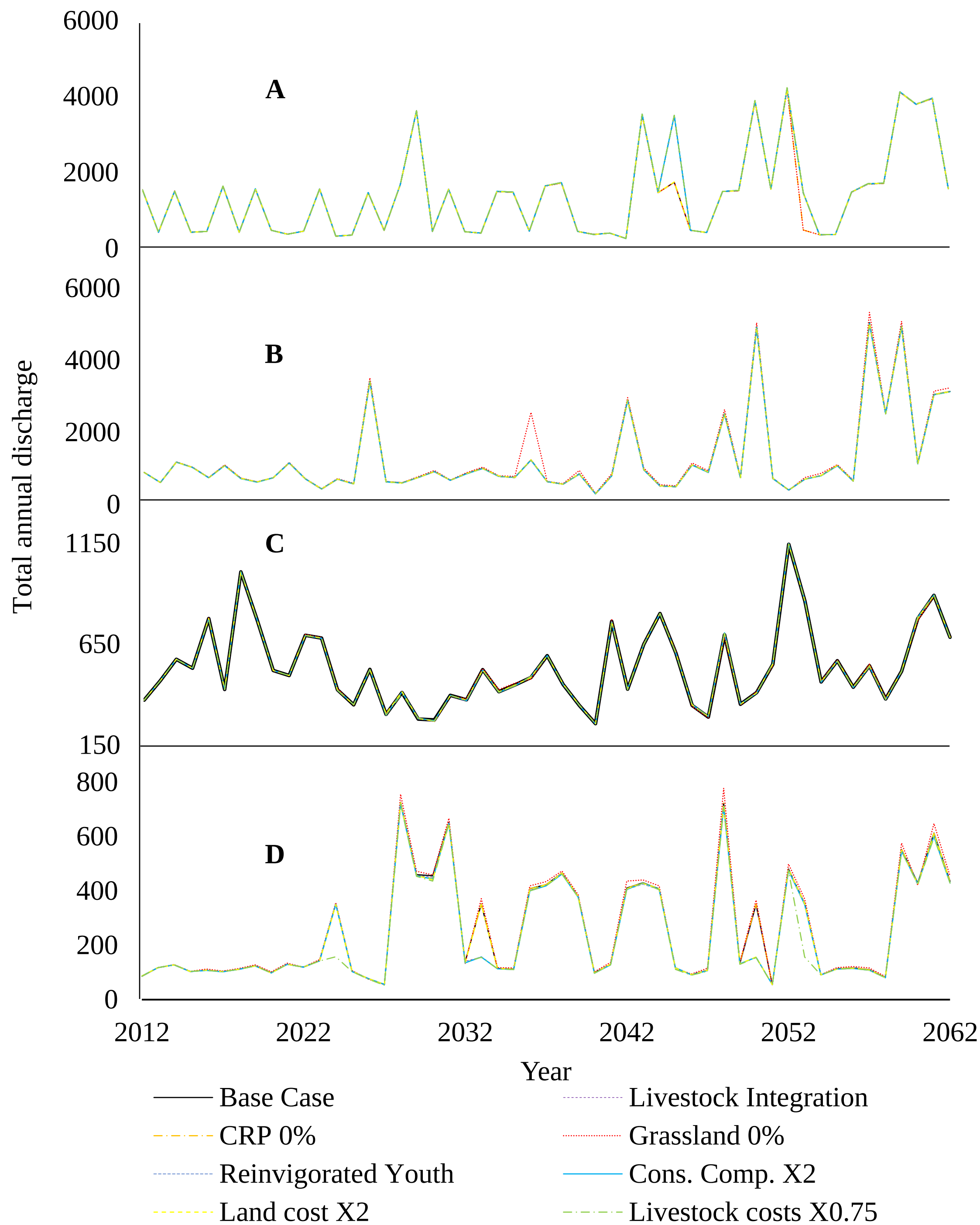

Figure 19. Annual discharge estimates [million cubic meters (MCM)] between 2012 and 2062 for the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments for the eight policy scenarios modeled under conventional tillage practices. Scenario names are described in Table 14.

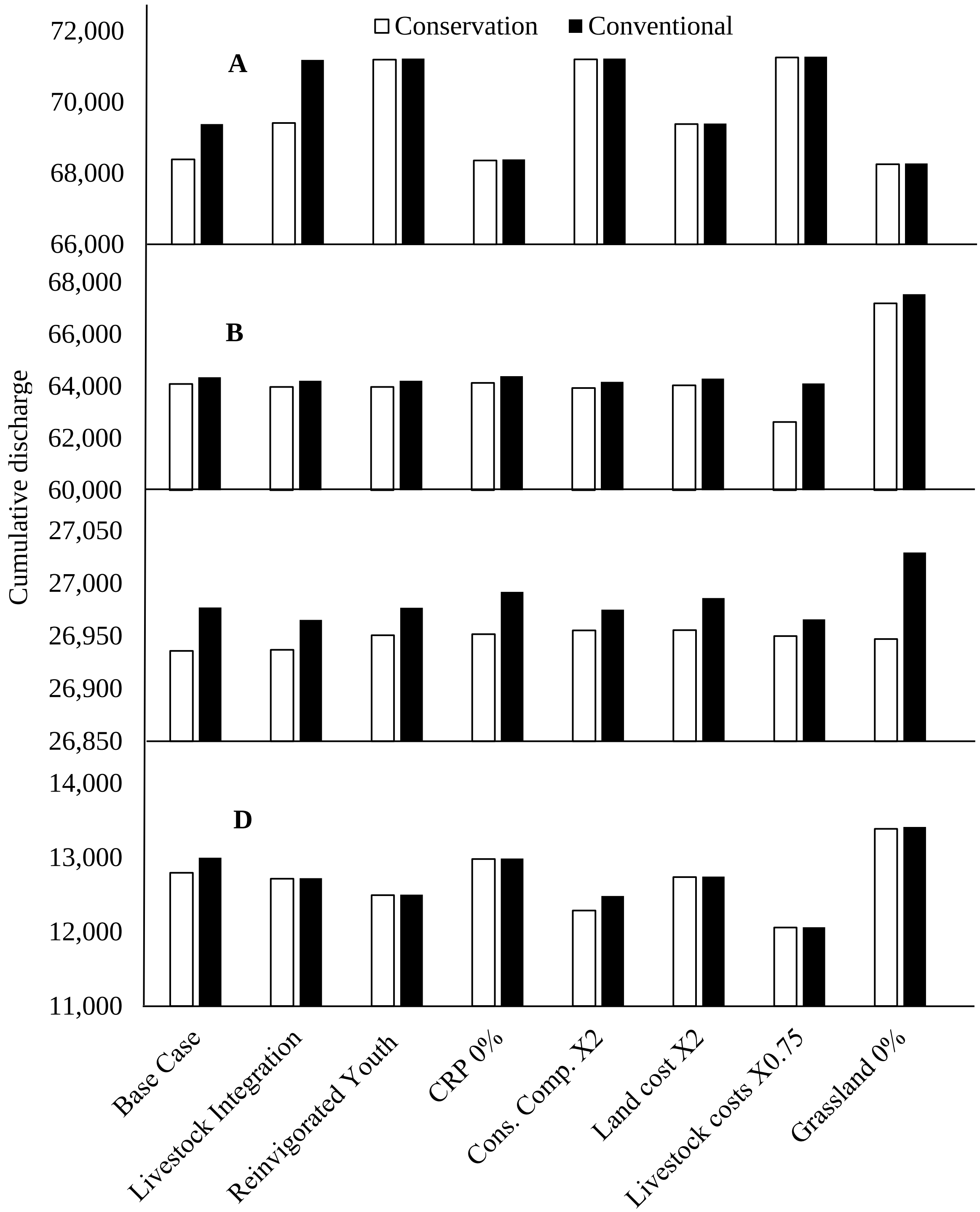

Figure 20. Total cumulative discharge [million cubic meters (MCM)] estimated from 2013 to 2062 for each of the eight scenarios (see Table 14 for scenario names) under both conservation and conventional tillage for Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments.

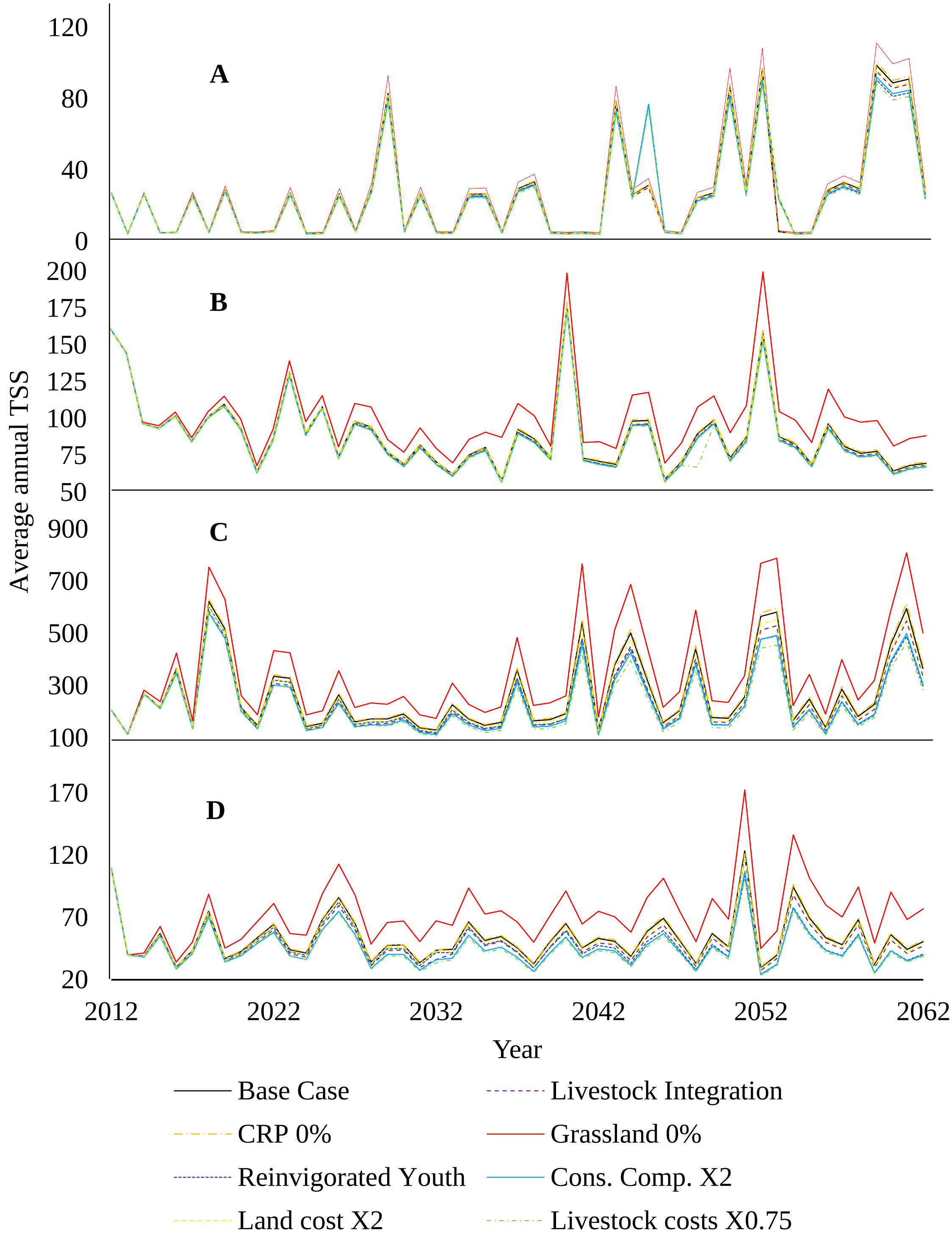

Figure 21. Average annual total suspended solids (TSS; mg/L) between 2012 and 2062 for the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments for the eight policy scenarios modeled under conservation tillage practices. Scenario names are described in Table 14.

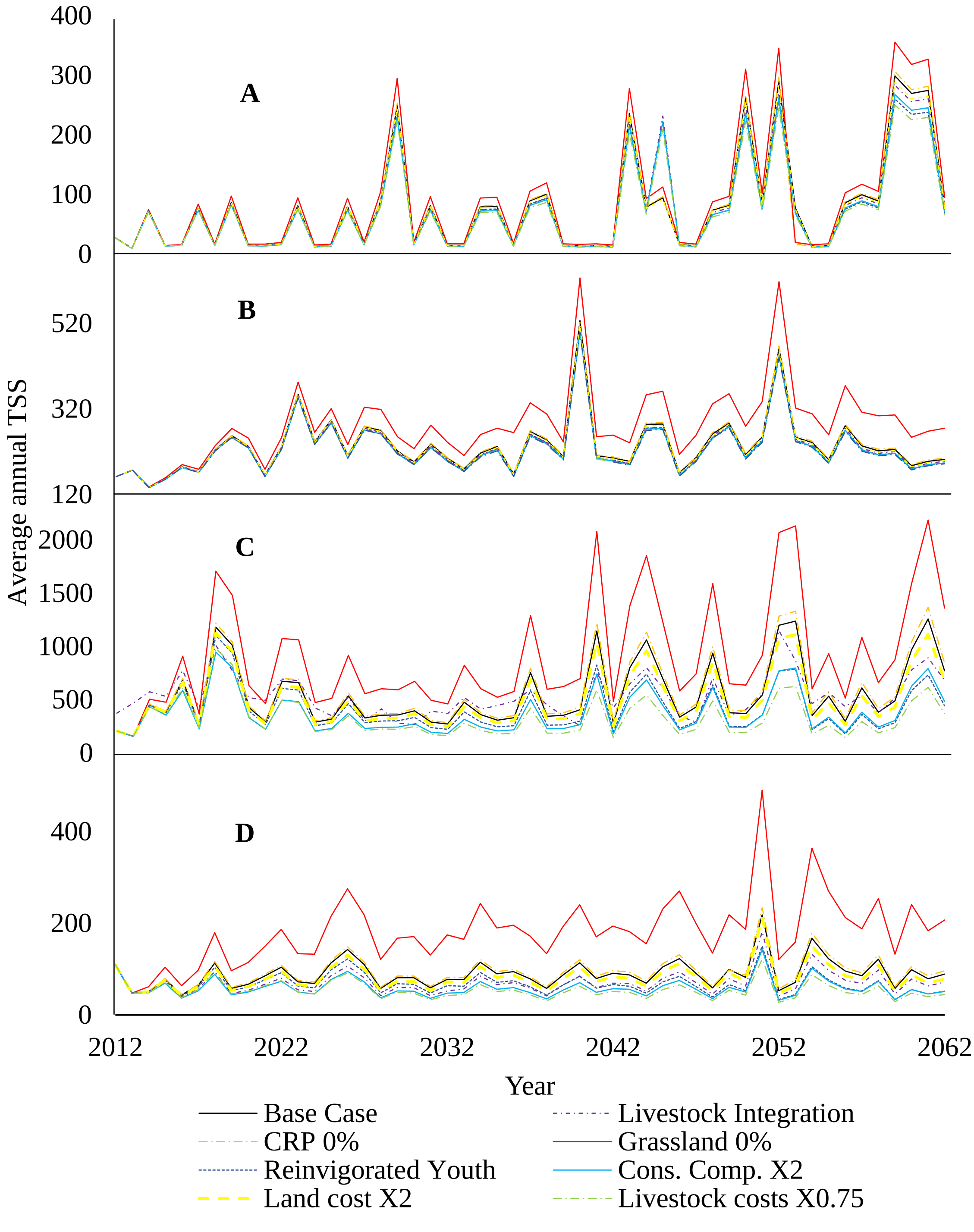

Figure 22. Average annual total suspended solids (mg/L) between 2012 and 2062 for the Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments for the eight policy scenarios modeled under conventional tillage practices. Scenario names are described in Table 14.

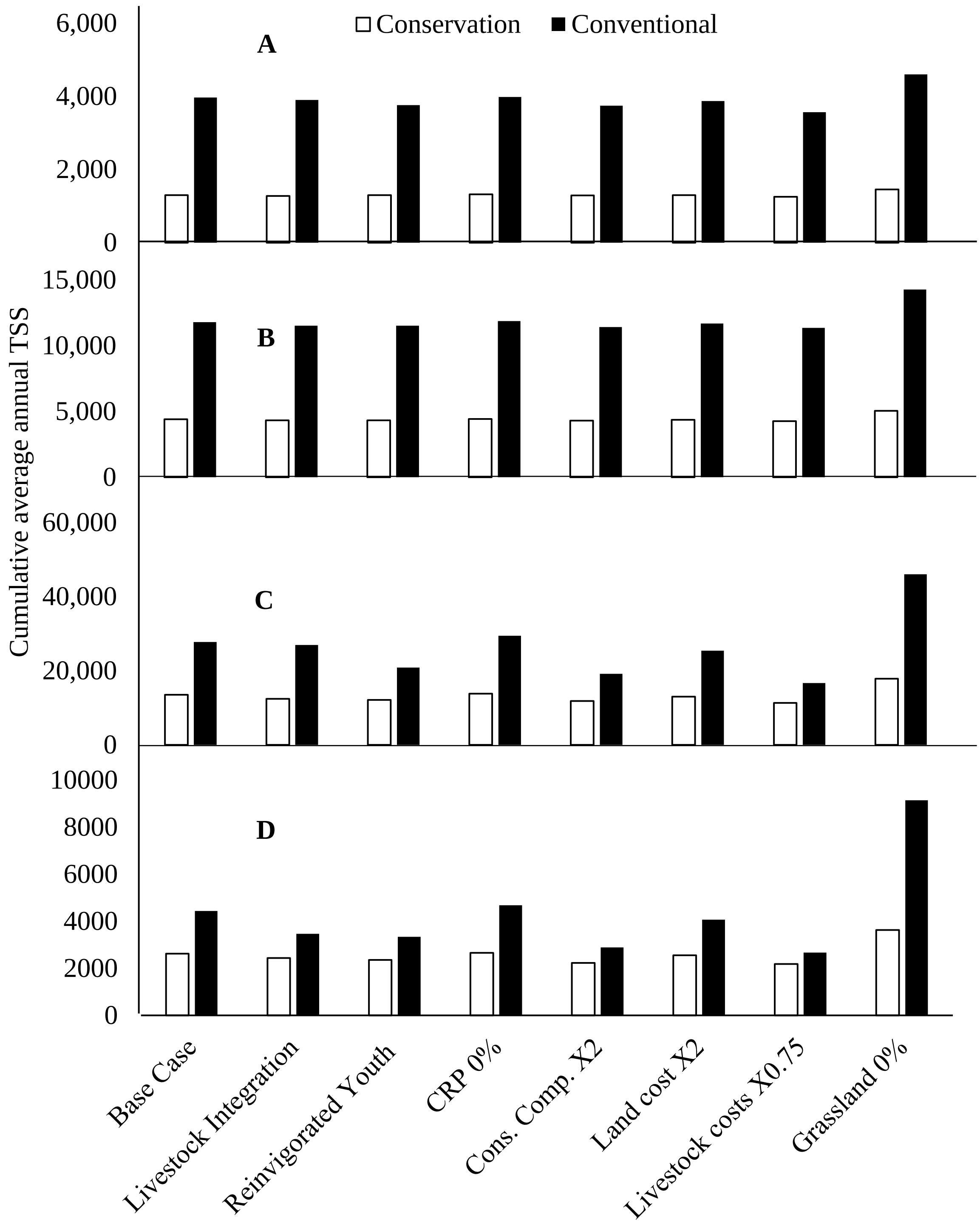

Figure 23. Cumulative annual average total suspended solids (mg/L) estimated from 2013 to 2062 for each of the eight scenarios (see Table 14 for scenario names) under both conservation and conventional tillage for Big Sioux (A), James (B), Bad (C), and Belle Fourche (D) water-catchments.

CHAPTER 4. DISCUSSION

This study is the first to my knowledge to estimate the potential environmental risk from grassland conversion-to-row crop agriculture under various regulatory, economic, and social scenarios. Concern of potential environmental risk in the U.S. Great Plains (including South Dakota) was initially reported by Turner et al. (2017) under the same scenarios that were included in my study; this risk was calculated as a dimensionless index [Soil Environmental Risk (SER)]. Overall, Turner et al. (2017) found that soil-related externalities from grassland conversion to cropland may worsen, stay the same, or improve over the next 50 years, depending on the policy under consideration. However, the SER index did not specify what type of soil-related externalities may occur nor specifically quantify those particular risks. In order to quantify unknown SER associated with each scenario, I developed a System Dynamics model that was able to evaluate three specific soil-related externalities [i.e., erosion, discharge, and total suspended solids (TSS)] commonly associated with grassland conversion to cropland (Foley et al., 2005; Sterling, Ducharne, & Polcher, 2013; Turner et al., 2018). Historic environmental externalities were replicated with relative accuracy and precision using the System Dynamics model from 1947 to 2012 (See Chapter 2: Results). Thus, the System Dynamics model forecasts addressed previously unknown SER by providing annual and cumulative estimates for erosion, discharge, and TSS externalities associated with future grassland conversion to cropland in four South Dakota water-catchments (i.e., Big Sioux, James, Bad, and Belle Fourche rivers).

Overall, estimates from this study indicated that soil related externalities are influenced by the implementation of policies that alter conversion of grassland to cropland, grassland conservation, or grassland restoration. In general, erosion, discharge, and TSS increased or decreased in a similar fashion throughout each study area from each policy scenario and tillage practice combination. However, environmental impacts from each policy were more significant in western South Dakota compared to the eastern side of the state, which indicated that areas with similar soils and topography may be more prone to environmental consequences from policies that alter grassland conversion. Similar studies have attributed changes in erosion, discharge, and water quality with grassland conversion to cropland, especially on landscapes with highly erodible soils and steep slopes (Ahiablame, Sinha, Paul, Ji, & Rajib, 2017; Clay et al., 2014; EPA, 2011; Qiu, Yin, Jian, and Geng, 2011; Sishodia, 2010, Sterling, Ducharne, & Polcher, 2013; Weller, Jordan, Correll, & Liu, 2003). Furthermore, changes in externalities presented in this study may also occur in areas with remaining grassland outside of South Dakota since most regulatory, economic, and social policies that directly or indirectly impact grassland conversion to cropland are set at a federal level (Claassen, 2011; McPhail, 2011; Pfrimmer, Gigliotti, Stafford, Schumann, & Bertrand, 2017; Turner et al., 2014). Therefore, understanding the impacts of current and future policies on environmental externalities is important as grassland conversion decisions may continue to influence soil and water resources (Foley et al., 2005; Koch et al., 2013; Rosegrant, Cai, & Cline, 2002)

Model forecasts provided useful information in evaluating scenarios, but limitations in the forecasts existed when potential changes in erosion, discharge, and TSS estimates from grassland conversion to cropland were captured. For example, integration of livestock was not as effective in reducing environmental consequences as expected, despite that livestock integration into cropland has been linked to decreased environmental externalities (Faust et al., 2018; Liebig, Tanaka, Kronberg, Scholljegerdes, & Karn, 2011; Russelle, Entz, & Franzluebbers, 2007; Turner et al., 2017). Additionally, Turner et al. (2017) estimated SER would decline from livestock integration into cropland. Each scenario altered the amount of annual cropland and grassland area within the System Dynamics model, which, in turn, drove the erosion, hydrologic, and TSS sub-models that generated estimates. However, System Dynamics model forecasts from livestock integration did not substantially reduce externalities as I was unable to account for livestock impacts on soil erodibility and surface water runoff. Reduced erosion and surface water runoff have been linked to increased ground cover from manure or trampled plant litter which may also cause increased water infiltration and soil water holding capacity from accumulated soil organic matter (Faust et al., 2018; Overstreet & DeJong-Huges, 2009; Tanaka, Kronberg, Scholljegerdes, & Karn, 2011). Water quality may also be improved as a result of decreased sediment deposition into waterways due to reduced erosion and surface water runoff from livestock integration (TSS; Faust et al., 2018). Future studies could include livestock integration to further evaluate the long-term impacts of this practice on environmental externalities.

Other factors may contribute to the current System Dynamics evaluation of erosion, discharge, and TSS from plausible grassland conversion scenarios. Climate change is expected to alter the intensity, frequency, and magnitude of precipitation events and minimum and maximum air temperature (EPA, 2017). Altered precipitation and temperature climate factors have been linked to increased erosion (Pruski & Nearing, 2002), altered hydrologic discharge (Ahiablame et al., 2017) and diminished water quality (Whitehead, Wilby, Battarbee, Kernan, & Wade, 2009) using regionalized climate projections from the Intergovernmental Panel on Climate Change (IPCC; see http://www.ipcc.ch/; Meehl et al., 2017). Additionally, climate change in South Dakota is expected to increase air temperature and the frequency, intensity, and magnitude of precipitation events (EPA, 2016, 2017; Pierce, Cayan, Maurer, Abatzoglou, & Hegewisch, 2015; Pierce, Cayan, & Thrasher, 2014). Therefore, current System Dynamics model forecasts may capture more considerable extremes in change of environmental externalities from grassland conversion to cropland through the incorporation of anticipated future climate projections for South Dakota.

Current estimates of environmental externalities may also be influenced by the incorporation of future changes in crop commodity factors. The spatial distribution of corn (*Zea mays*), soybeans (*Glycine max*), and wheat (*Triticum aestivum*) may vary across the landscape in response to future crop commodity demands (Roesch-McNally, Arbuckle, & Tyndall, 2018). Wang (2018) evaluated wheat production and reported that wheat acres have decreased by 32% from 2015 to 2018 within South Dakota (USDA-NASS, 2018), which may be a result of increased corn and soybean prices and decreased wheat profitability (Schnitkey, 2017). Landscape-scale shifts in corn, soybeans, and wheat have been linked to changes in erosion, discharge, and water quality **(**Hong, 2017; Neupane & Kumar, 2015; Rounsevell et al., 2005). Corn, soybeans, and wheat (i.e., spring and winter wheat) are linked to changes in environmental externalities because they differ in growing season time and duration, ground cover, and evapotranspiration (i.e., plant water requirements; see Chapter 2: Methods; Couturier & Ripley, 1973; Foster et al., 2002; Gassman, Reyes, Green, & Arnold, 2007; Kang, Wang, & Liu, 2005; Ma, Gale, Ma, Wu, Li, & Wang, 2013; Ostrem, Trooien, & Hay, 2016). Therefore, current evaluation of changes in environmental externalities may be improved through the coupling of grassland conversion scenarios with expected future changes in corn, soybeans, and wheat production in South Dakota.

Unlike livestock integration, climate, and crop-type model factors, soil erosion by wind was not evaluated in the System Dynamics model. Historically, grassland conversion to cropland coupled with drought caused an estimated 14 billion metric-tons of topsoil loss from wind erosion during the 1930s Dust Bowl in the U.S. Great Plains (Bolles, Forman, & Sweeny, 2017). Recently, concern of increased wind erosion has again risen from accelerated grassland conversion to cropland within the U.S. Northern Great Plains (NRCS, 2012; Wienhold, Vigil, Hendrickson, & Derner, 2018). Additionally, potential soil loss from wind erosion has been linked to cultivation within South Dakota (Miller, 2014). Wind erosion was purposely excluded from the System Dynamics model because it is not driven by precipitation and required a unique spatial resolution and spatial components (e.g., tree barriers; Wagner, 2013) which made it less related to rill and sheet erosion, discharge, and TSS externalities that shared a common model structure for precipitation and spatial components. Future efforts to incorporate wind erosion dynamics may potentially improve estimations of annual and cumulative erosion in model forecasts.

Application of SD methodology provided a robust and comprehensive tool to evaluate plausible regulatory, economic, and social policies that may influence land use change scenarios and their interaction with tillage practices to estimate erosion, discharge, and TSS. Forecasts indicated that there may be concern of exacerbated environmental consequences from grassland conversion, but these consequences may be mitigated through grassland restoration. Implementation of policy scenarios appears to be an effective means of altering grassland conversion rates and associated environmental consequences. Additionally, continuation of current policies or incorporation of new policies may have similar effects on environmental consequences in other areas that are subject to current or future changes in grassland conversion rates. Thus, my results may provide information for producers, policymakers, and other stakeholders to address complex grassland conversion decisions and potential environmental consequences with a long-term view in order to conserve and preserve soil and water resources in South Dakota.

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APPENDIX

Appendix A: Dynamic Hypothesis statement.

A DH statement was developed which described the central endogenous variables that influence the model's structure. Endogenous variables are used to simplify problem articulation, which leads to a concise and easily communicated statement reflective of the DH. The following statement is my endogenous articulation of the hypothesized structure:

Land use change from grassland to row crop agriculture has cascading effects within the plant-soil-water continuum at the field level, including: plant cover, rooting structure, plant residue, soil aggregate stability and soil permeability. The cumulative effect of these changes influence surface water hydrologic patterns and soil erodibility, impacting soil quality, which subsequently alters natural (baseline) total suspended solids in streams and rivers. Unforeseen consequences from soil loss (aggregate sheet and rill erosion; metric-tons/ha/yr), hydrologic changes [too much or too little; million cubic *meters (MCM)/yr] and impaired water quality (TSS; mg/L) may reduce the functionality of ecological goods and services. Impairment of these resources may limit hectares of land available for production in the form of mandates to mitigate environmental externalities, for example, removal of land in production (e.g., CRP) or that degradation has made vulnerable land unsuitable for agricultural production.*

Appendix B: Description of model data water-catchment delineation.

Water-catchment characteristics (i.e., climate, soil, spatial land use, and crop type) were delineated at a hydrologic unit code (HUC) 6 (Big Sioux and James rivers) and HUC8 (Bad and Belle Fourche rivers) and then delineated to smaller HUC10 boundaries within each of the four water-catchments. Water-catchment delineation to HUC10 provided greater data resolution within each water-catchment. Water-catchment data was then integrated into Vensim™ using subscripting (Vensim, 2007). Subscripts allowed for multiple uses of the same model structure to represent HUC10 water-catchments within each of the four study areas. For example, the Big Sioux River HUC6 contains 53 unique HUC10 water-catchments, and each HUC10 was integrated into the model using a unique subscript reference for each of the 53 HUC10s data (see Tables B1 – 4 for subscript information for each water-catchment). Subscripted water-catchments (HUC10s) were then aggregated to represent the entire Big Sioux River (HUC6) water-catchment (Figure B-1).

Table B-1. Subscripted hydrologic unit code (HUC10) for the Big Sioux River water-catchment including Vensim subscript identification (ID), water-catchment name, area (ha), state, and the United States Geological Survey (USGS) identification.

HUC subscript ID #	**HUC 10 water-catchment**	**Area (ha)**	**State(s)**	**USGS HUC ID**
1	Big Ditch-Big Sioux River	72,566	IA, SD	1017020325
2	Skunk Creek	46,904	SD	1017020311
3	Broken Kettle Creek	25,587	IA	1017020324
4	Deer Creek-Medary Creek	17,313	MN, SD	1017020209
5	Rock River	42,124	IA	1017020408
6	North Deer Creek	32,426	SD	1017020207
7	Split Rock Creek	47,042	MN, SD	1017020316

Table B-1. Subscripted hydrologic unit code (HUC-10) for the Big Sioux River water-catchment (continued).

HUC subscript ID #	HUC 10 water-catchment	Area (ha)	State(s)	USGS HUC ID
8	Grass Lake	77,997	SD	1017020104
9	Willow Creek	30,251	SD	1017020107
10	Riverview Cemetery-Big Sioux River	22,051	SD	1017020302
11	Sixmile Creek	27,903	IA	1017020320
12	Dry Lake Number One	84,270	SD	1017020201
13	Mud Creek-Rock River	35,889	IA	1017020404
14	Hidewood Creek	42,976	SD	1017020204
15	Champepadan Creek-Rock River	63,970	IA, MN	1017020403
16	Beaver Creek	32,936	SD	1017020318
17	Beaver Creek-Split Rock Creek	41,444	MN	1017020315
18	Brookfield Creek-Big Sioux River	43,108	SD	1017020306
19	Sioux Falls Diversion Channel-Big Sioux River	28,954	SD	1017020312
20	Lake Kampeska	48,126	SD	1017020105
21	Lake Marsh	59,711	SD	1017020202
22	Otter Creek-Little Rock River	54,512	IA	1017020405
23	Oakwood Lakes	22,321	SD	1017020205
24	Little Rock River	66,459	IA, MN	1017020406
25	Medary Creek	34,729	MN, SD	1017020210
26	Kanaranzi Creek	53,018	IA, MN	1017020402
27	West Branch Skunk Creek	19,662	SD	1017020309
28	Sixmile Creek	27,903	IA	1017020320
29	Waubay Lakes	75,692	SD	1017020102
30	Stray Horse Creek	21,372	SD	1017020108
31	Ninemile Creek-Big Sioux River	50,510	IA, MN, SD	1017020317
32	Bitter Lake	30,004	SD	1017020103
33	Colton Creek-Skunk Creek	36,524	SD	1017020310
34	City of Watertown-Big Sioux River	56,962	SD	1017020109
35	Squaw Creek	14,929	SD	1017020304
36	Headwaters Rock River	84,072	MN	1017020401
37	West Pipestone Creek	22,515	SD	1017020314
38	Green Creek-Big Sioux River	29,454	IA	1017020321

Table B-1. Subscripted hydrologic unit code (HUC-10) for the Big Sioux River water-catchment (continued).

HUC subscript ID #	**HUC 10 water-catchment**	**Area (ha)**	**State(s)**	**USGS HUC ID**
39	Headwaters Skunk Creek	32,726	SD	1017020307
40	Lakes Inlet-Big Sioux River	38,321	SD	1017020106
41	Battle Creek	65,231	SD	1017020208
42	Flandreau Creek	30,185	MN	1017020303
43	Buffalo Creek	25,479	SD	1017020308
44	Headwaters Big Sioux River	44,231	SD	1017020101
45	Indian Creek	16,137	IA	1017020322
46	Upper Big Sioux River	86,861	SD	1017020211
47	Pipestone Creek	57,255	MN	1017020313
48	Brule Creek	55,442	SD	1017020323
49	Pattee Creek-Big Sioux River	64,087	IA, SD	1017020319
50	Tom Creek-Rock River	35,008	IA	1017020407
51	Bachelor Creek	25,585	SD	1017020305
52	Lake Poinsett	82,481	SD	1017020203
53	Spring Creek	16,654	MN, SD	1017020301

Table B-2. Subscripted hydrologic unit code (HUC10) for the James River water-catchment including Vensim subscript identification (ID), water-catchment name, area (ha), state, and the United States Geological Survey (USGS) identification.

HUC subscript ID #	**HUC 10 water-catchment**	**Area (ha)**	**State(s)**	**USGS HUC ID**
1	Newport/Weston Ditch	40,348	SD	1016000317
2	Dry Branch	42,242	ND	1016000406
3	West Branch Firesteel Creek	36,065	SD	1016001108
4	Antelope Creek	52,592	SD	1016000501
5	Pleasant Lake	61,095	SD	1016001107
6	Stevens Slough	36,964	ND	1016000312
7	Upper Turtle Creek	85,398	SD	1016000901
8	Lower North Fork Snake Creek	26,486	SD	1016000704
9	Lower Mud Creek	35,470	SD	1016000504
10	Lower Pipestem Creek	88,773	ND	1016000205
11	Headwaters Pipestem Creek	65,987	ND	1016000201
12	Jamestown Reservoir	92,039	ND	1016000106
13	Lower Turtle Creek	31,644	SD	1016000907
14	Rocky Run	61,797	ND	1016000103
15	Moccasin Creek-James River	46,872	SD	1016000321
16	Lonetree Creek	28,397	SD	1016001116
17	South Fork Maple River	25,938	ND	1016000403
18	Middle Pipestem Creek	39,495	ND	1016000204
19	Crow Creek Drainage Ditch	86,941	ND, SD	1016000318
20	Timber Creek	92,773	SD	1016000603
21	Upper Pipestem Creek	32,322	ND	1016000203
22	City of Jamestown	35,294	ND	1016000303
23	Beaver Creek-Upper James River	70,229	ND	1016000305
24	Redstone Creek	78,671	SD	1016000612
25	North Wolf Creek	55,409	SD	1016000904
26	Long Lake	20,023	SD	1016001101
27	Upper Bear Creek	60,663	ND	1016000310
28	Dry Run	29,296	ND	1016000313

Table B-2. Subscripted hydrologic unit code (HUC10) for the James River water-catchment (continued).

HUC subscript ID #	HUC 10 water-catchment	Area (ha)	State(s)	USGS HUC ID
29	Willow Creek	49,166	SD	1016000407
30	Melby Hills	48,606	ND	1016000105
31	School Section Lakes	19,374	SD	1016001105
32	Dry Coulee	35,684	ND	1016000306
33	Firesteel Creek	81,604	SD	1016001109
34	Bone Hill Creek	54,725	ND	1016000307
35	Sand Creek	103,404	SD	1016000613
36	Silver Lake	80,693	ND	1016000101
37	Foster Creek	62,692	SD	1016000606
38	Lower South Fork Snake Creek	67,606	SD	1016000805
39	Foot Creek	50,779	SD	1016000319
40	Lower Wolf Creek	32,152	SD	1016000905
41	Beaver Creek	37,563	SD	1016001119
42	Town of Freedonia	74,692	ND	1016000401
43	Sevenmile Coulee	49,320	ND	1016000301
44	Upper Wolf Creek	86,608	SD	1016000902
45	Firesteel Creek-James River	59,187	SD	1016001114
46	Foster Creek-James River	104,509	SD	1016000610
47	Lower Preachers Run-Scatterwood Lakes	54,169	SD	1016000803
48	Little Pipestem Creek	50,672	ND	1016000202
49	Dawson Creek	18,128	SD	1016001117
50	Timber Creek-James River	45,832	SD	1016000604
51	Elm Lake	72,406	ND, SD	1016000405
52	Beaver Creek-James River	42,109	SD	1016001120
53	Streaman Coulee	44,679	ND	1016000302
54	Pierpont Lake	54,471	SD	1016000502
55	Pearl Creek	74,430	SD	1016000611
56	Upper-North Fork Snake Creek	51,686	SD	1016000701
57	Lost Creek	28,417	SD	1016000903
58	Jim Creek-James River	33,131	SD	1016001103
59	Buffalo Creek	50,323	ND	1016000304
60	Upper South Fork Snake Creek	89,877	SD	1016000804
61	Cain Creek	98,494	SD	1016000609

Table B-2. Subscripted hydrologic unit code (HUC10) for the James River water-catchment (continued).

HUC subscript ID #	**HUC 10 water-catchment**	**Area (ha)**	**State(s)**	**USGS HUC ID**
62	Columbia Road Reservoir-James River	114,669	ND, SD	1016000314
63	Twelvemile Creek	71,755	SD	1016001112
64	Wolf Creek	103,466	SD	1016001115
65	Rock Creek	72,411	SD	1016001106
66	Dry Run	55,341	SD	1016000601
67	Redstone Creek-James River	49,061	SD	1016000614
68	Lower Snake Creek	92,578	SD	1016000806
69	Upper Snake Creek	93,233	SD	1016000801
70	Shue Creek	44,131	SD	1016000607
71	Enemy Creek	46,314	SD	1016001110
72	Kelly Creek	59,812	ND	1016000104
73	Moccasin Creek	49,676	SD	1016000320
74	Cresbard Lake	38,556	SD	1016000703
75	Medicine Creek	69,085	SD	1016000906
76	Dry Creek	33,865	SD	1016001113
77	Crow Creek	51,455	ND, SD	1016000316
78	Maple Creek	57,265	ND	1016000402
79	Sweetwater Lake	67,144	SD	1016000602
80	Broadland Creek	42,733	SD	1016000608
81	Pierre Creek	24,231	SD	1016001111
82	Upper Preachers Run	43,463	SD	1016000802
83	Lower Elm River	27,556	SD	1016000408
84	Northern Coteau Lakes-Upper James River	75,097	SD	1016000315
85	Maple River	51,776	ND	1016000404
86	Jim Creek	26,456	SD	1016001102
87	Wolf Creek-James River	69,907	SD	1016001118
88	Crandon Creek	41,168	SD	1016000605
90	Dry Run-James River	57,777	SD	1016001104
91	Upper Mud Creek	74,558	SD	1016000503
92	Hamak Lake	131,314	SD	1016000702
93	Twin Lakes	44,320	ND	1016000308
94	Lower Bear Creek	39,526	ND	1016000311
95	Big Slough	96,697	ND	1016000102

Table B-3. Subscripted hydrologic unit code (HUC10) for the Bad River water-catchment including Vensim subscript identification (ID), water-catchment name, area (ha), state, and the United States Geological Survey (USGS) identification.

HUC subscript ID #	**HUC 10 water-catchment**	**Area (ha)**	**State(s)**	**USGS HUC ID**
1	Whitewater Creek	50,740	SD	1014010211
2	Cottonwood Creek	55,538	SD	1014010211
3	South Fork Bad River	34,899	SD	1014010211
4	North Fork Bad River	48,791	SD	1014010211
5	Little Prairie Dog Creek-Bad River	59,152	SD	1014010211
6	Dry Creek	42,924	SD	1014010211
7	White Clay Creek	36,332	SD	1014010211
8	Big Prairie Dog Creek-Bad River	44,747	SD	1014010211
9	Frozen Man Creek	29,568	SD	1014010211
10	Plum Creek	47,159	SD	1014010211
11	Brave Bull Creek	33,774	SD	1014010211
12	White Willow Creek	34,456	SD	1014010211
13	Grindstone Creek-Bad River	43,515	SD	1014010211
14	Indian Creek	25,690	SD	1014010211
15	Buzzard Creek-Bad River	33,652	SD	1014010211
16	Mitchell Creek	41,954	SD	1014010211
17	Lance Creek	27,314	SD	1014010211
18	War Creek	32,969	SD	1014010211
19	Willow Creek	26,702	SD	1014010211
20	Outlet Bad River	72,714	SD	1014010211

Table B-4. Subscripted hydrologic unit code (HUC10) for the Belle Fourche River water-catchment including Vensim subscript identification (ID), water-catchment name, area (ha), state, and the United States Geological Survey (USGS) identification.

HUC subscript ID #	**HUC 10 water-catchment**	**Area (ha)**	**State(s)**	**USGS HUC ID**
1	Upper Belle Fourche River	97,644	SD, WY	1012020201
2	Sand Creek	77,717	SD, WY	1012020301
3	Lower Redwater Creek	71,556	SD, WY	1012020304
4	Upper Redwater Creek	68,244	SD, WY	1012020303
5	Horse Creek	41,732	SD	1012020204
6	Willow Creek	48,587	SD	1012020206
7	Owl Creek	61,025	SD, MT	1012020202
8	West Elm Creek	39,612	SD	1012020210
9	Middle Belle Fourche River	101,782	SD, WY	1012020205
10	Indian Creek	93,327	SD, MT	1012020203
11	Spearfish Creek	54,624	SD	1012020302
12	Elm Creek	66,719	SD	1012020212
13	East Elm Creek	20,388	SD	1012020211
14	Bull Creek-Belle Fourche River	54,730	SD	1012020213
15	Alkali Creek	49,131	SD	1012020209
16	East Killdeer Creek-Belle Fourche River	43,811	SD	1012020214
17	Bear Butte Creek	57,668	SD	1012020207
18	Fourmile Creek-Belle Fourche River	64,663	SD	1012020208

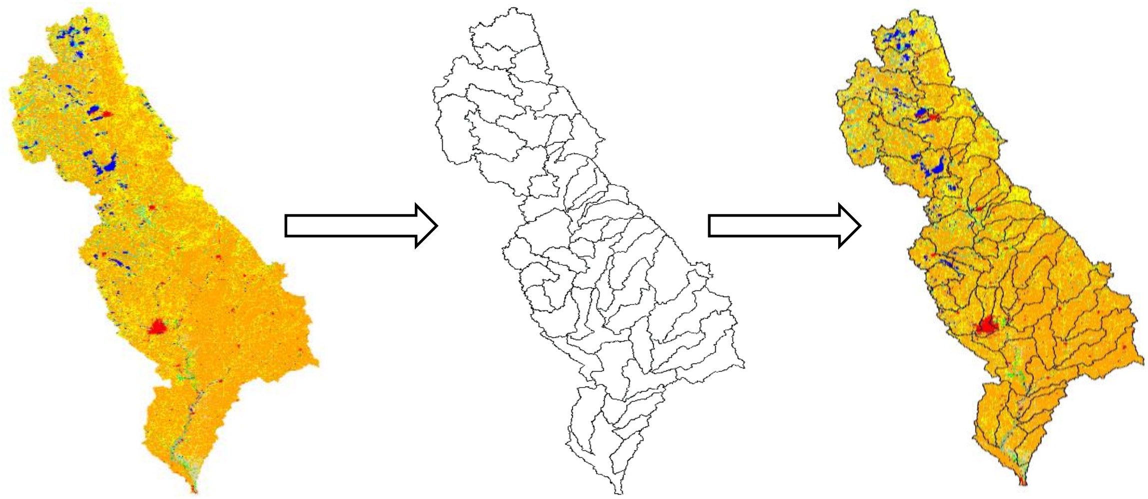

Figure B-1. Example of the Big Sioux River land use data collected at a HUC6 level and then delineated by 53 unique HUC10s which were integrated using subscripts (53 unique models) and aggregated into a HUC6 model (aggregate of the 53 HUC10s).

Appendix C: Additional model tests and results.

Supplemental model tests were performed in addition to the statistical tests, which were a sensitivity analysis and an extreme conditions test. A sensitivity analysis of erosion, discharge, and TSS was conducted, where rain and snow were multiplied by a range from 0 – 10 of randomly generated constants with the Latin Hyper Cube method, assuming a normal distribution, which were used to perform 200 simulations in Vensim™. Results indicated that erosion [million metric-tons(Mt/yr)], discharge [million cubic meters (MCM/yr)], and average annual TSS (mg/L) were sensitive to changes in precipitation values (see Figures C1 – 3 for Belle Fourche River example). The extreme conditions test consisted of three simulations that adjusted rain and snow by multiples of 0, 1, and 10. Results of the extreme conditions test indicated that the model did not produce integration errors when pushed to the extremes; for example, the model did not produce negative values of any metric nor did the model fail to perform calculations. Moreover, behaviors for erosion and discharge were as expected which increased and decreased as precipitation was changed from 10 to 0. Total suspended solids also displayed the same behavior as erosion and discharge, except when precipitation was increased times 10, which decreased TSS as increased discharge levels diluted TSS concentration (see Figures C4 – 6 for Belle Fourche River example).

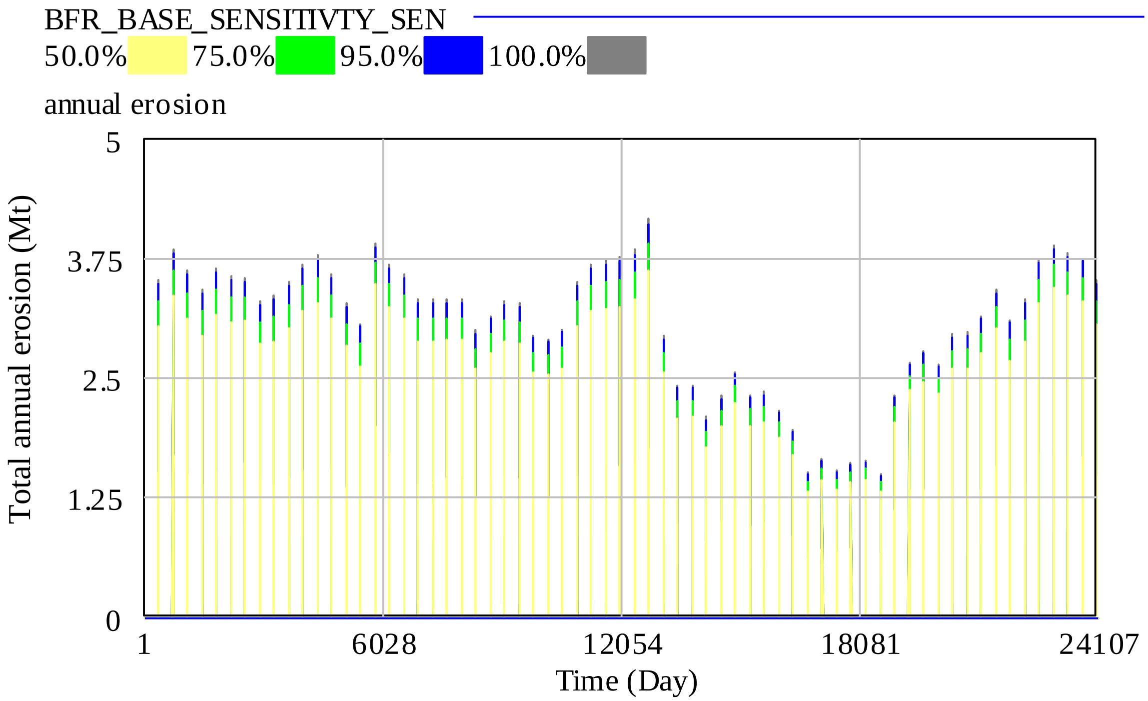

Figure C-1. Belle Fourche River sensitivity analysis of annual erosion [million metric-tons (Mt)/yr] where rain and snow were multiplied by a range from 0 − 10 (constant) from 1947 (i.e., 0) to 2012 (i.e., 24107).

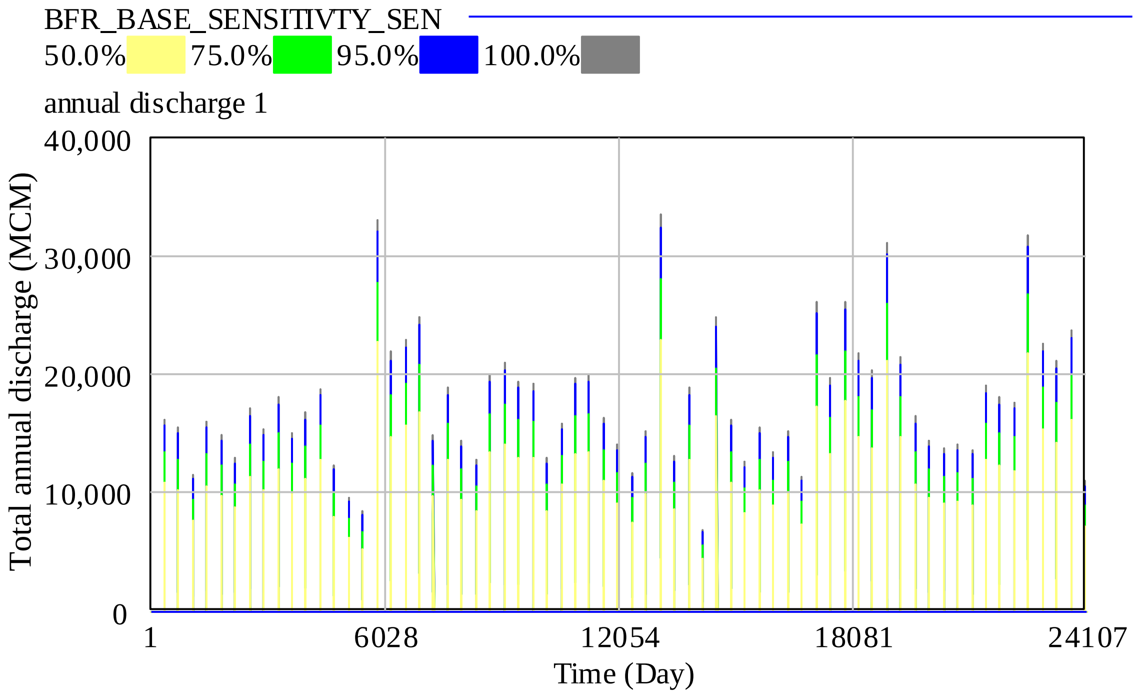

Figure C-2. Belle Fourche River sensitivity analysis of annual discharge [million cubic meters (MCM)/yr] where rain and snow were multiplied by a range from 0 − 10 (constant) from 1947 (i.e., 0) to 2012 (i.e., 24107).

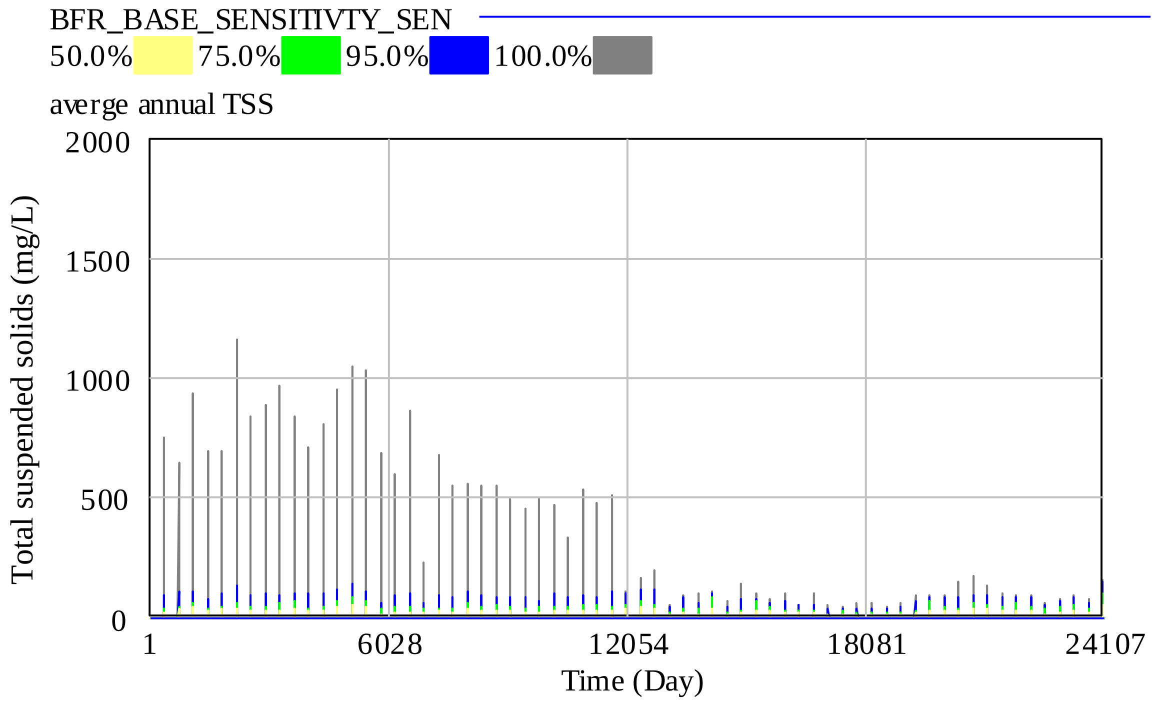

Figure C-3. Belle Fourche River sensitivity analysis of annual average total suspended solids (mg/L/yr) where rain and snow were multiplied by a range from 0 – 10 (constant) from 1947 (i.e., 0) to 2012 (i.e., 24107).

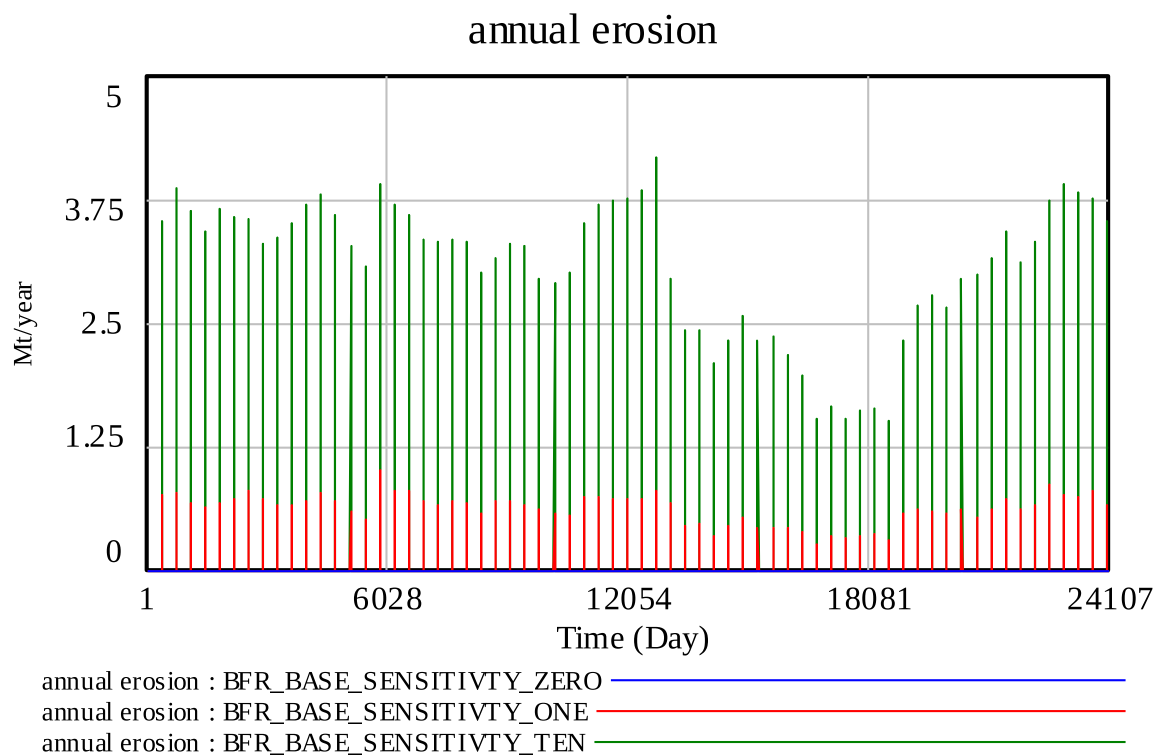

Figure C-4. Total annual erosion [million metric-tons (Mt)/yr] extreme conditions tests where rain and snow were multiplied by zero (blue line), one (red line) and 10 (green line) from 1947 (i.e., 0) to 2012 (i.e., 24107).

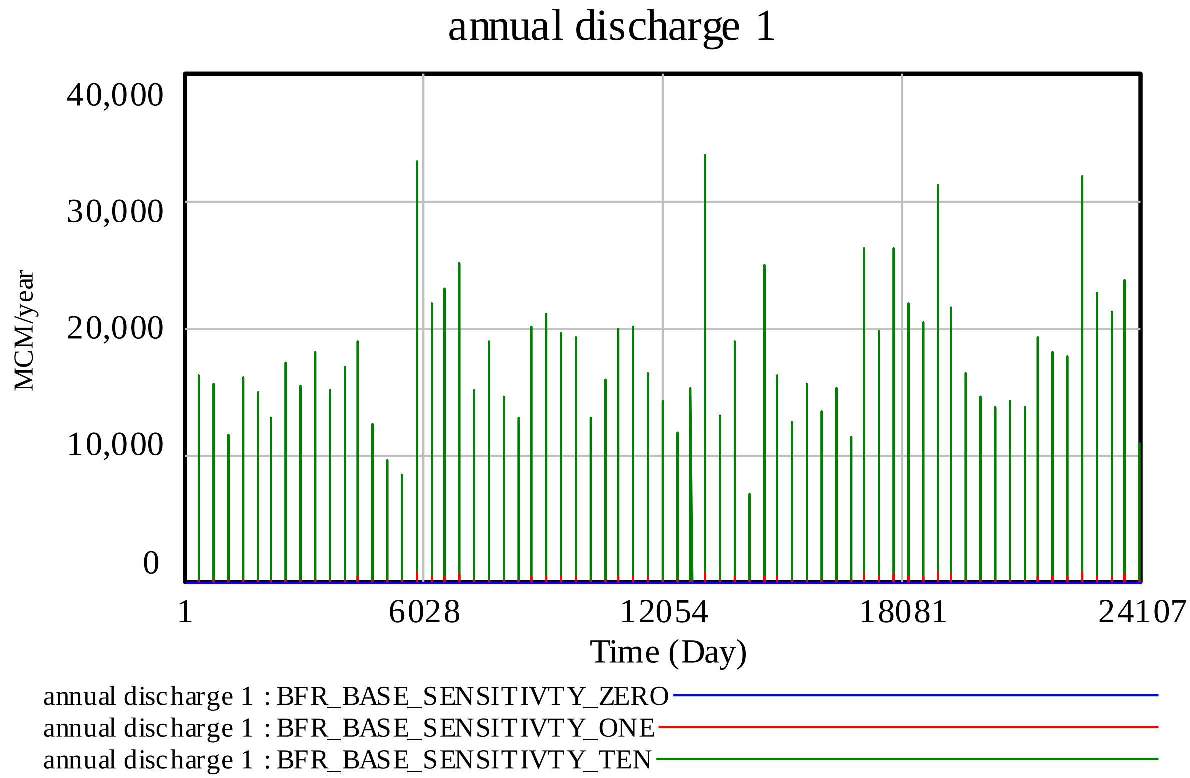

Figure C-5. Total annual discharge [million cubic meters (MCM)/yr] extreme conditions tests where rain and snow were multiplied by zero (blue line), one (red line) and 10 (green line) from 1947 (i.e., 0) to 2012 (i.e., 24107).

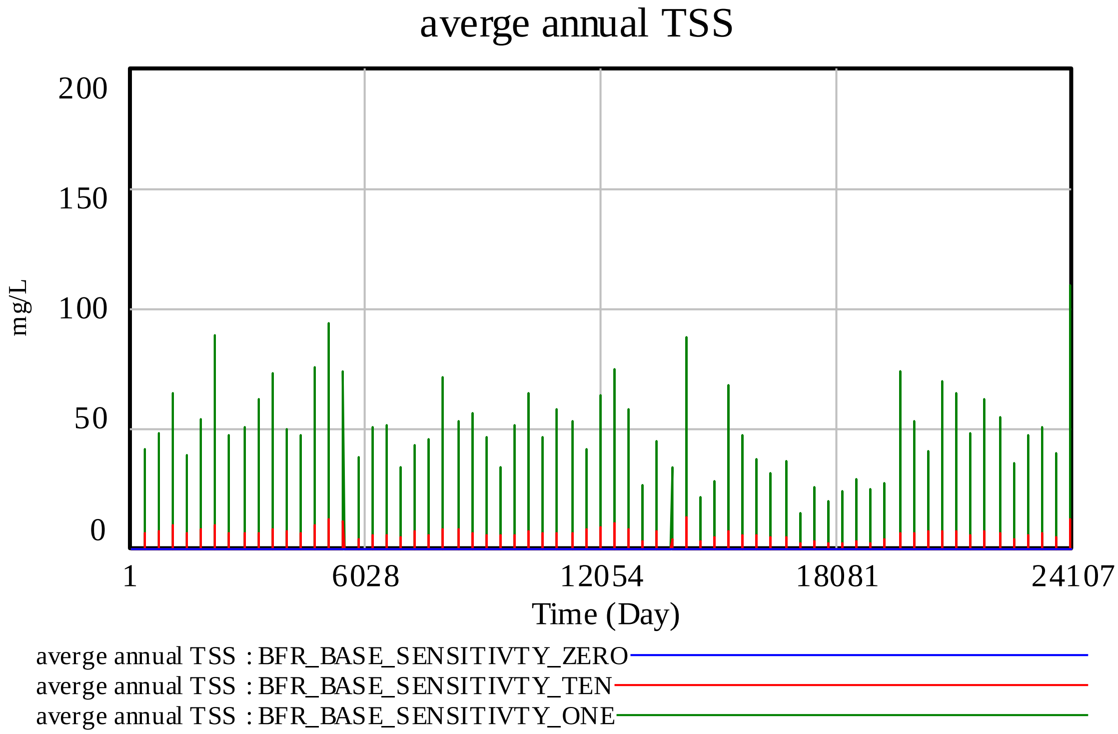

Figure C-6. Average annual total annual discharge (mg/L) extreme conditions tests where rain and snow were multiplied by zero (blue line), one (green line) and 10 (red line) from 1947 (i.e., 0) to 2012 (i.e., 24107).

Appendix D: Additional model forecast results for the Big Sioux, James, Bad, and Belle Fourche water-catchments.

Table D-1. Big Sioux River total erosion [megatons (million metric-tons)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative erosion (Total), and rank of cumulative erosion within each tillage type (1 = highest erosion estimates and 8 = lowest erosion estimates).

Tillage type	**Scenario**	**2013**	**2062**	**Minimum**	**Maximum**	**Mean**	**SD**	**Total**	**Rank**
Conservation	Base-case	5.5	6.4	5.5	8.2	6.7	0.7	333.7	3
	Livestock Integration	5.5	6.2	5.3	7.9	6.4	0.6	322.2	5
	Reinvigorated Youth	5.5	5.8	5.3	7.5	6.4	0.6	318.0	6
	CRP 0%	5.5	6.6	5.5	8.3	6.8	0.7	337.6	2
	Cons. Comp. X2	5.5	6.0	5.2	7.6	6.3	0.6	315.8	7
	Land cost X2	5.5	6.3	5.4	8.0	6.6	0.6	328.8	4
	Livestock costs X0.75	5.5	5.7	5.1	7.3	6.1	0.6	307.2	8
	Grassland 0%	5.5	7.2	5.5	9.1	7.4	0.8	367.9	1
Conventional	Base-case	4.9	7.7	4.9	9.9	7.9	0.9	397.2	3
	Livestock Integration	4.9	7.4	4.9	9.4	7.5	0.8	376.8	5
	Reinvigorated Youth	4.9	6.7	4.9	8.6	7.4	0.8	369.1	6
	CRP 0%	4.9	7.9	4.9	10.1	8.1	1.0	403.9	2
	Cons. Comp. X2	4.9	6.9	4.9	8.9	7.3	0.8	364.5	7
	Land cost X2	4.9	7.5	4.9	9.6	7.8	0.9	388.8	4
	Livestock costs X0.75	4.9	6.4	4.9	8.2	7.0	0.7	349.6	8
	Grassland 0%	4.9	9.1	4.9	11.6	9.2	1.2	459.4	1

Table D-2. Big Sioux River percent of calibrated (“cal.”) and forecasted (“fore.”) mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity.

Land use and tillage type	**Scenario**	**Erosion (cal.)**	**Erosion (fore.)**	**Discharge (cal.)**	**Discharge (fore.)**	**TSS (cal.)**	**TSS (fore.)**
Cropland under conservation tillage	Base-case	50	99	15	13	73	273
	Livestock Integration	50	100	17	15	74	287
	Reinvigorated Youth	51	104	15	13	73	268
	CRP 0%	50	99	15	13	70	232
	Cons. Comp. X2	50	101	20	17	70	231
	Land cost X2	50	100	20	17	70	230
	Livestock costs X0.75	51	104	17	15	74	281
	Grassland 0%	51	104	20	17	71	243
Cropland under conventional tillage	Base-case	61	38	17	15	19	24
	Livestock Integration	60	38	20	17	8	9
	Reinvigorated Youth	57	36	15	13	17	21
	CRP 0%	61	38	15	13	4	4
	Cons. Comp. X2	59	37	20	17	12	14
	Land cost X2	60	38	20	17	12	14
	Livestock costs X0.75	57	36	17	15	21	27
	Grassland 0%	57	36	20	17	17	21

Table D-3. Big Sioux River percent of calibrated ("cal.") and forecasted ("fore.") mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity (continued).

Land use and tillage type	**Scenario**	**Erosion (cal.)**	**Erosion (fore.)**	**Discharge (cal.)**	**Discharge (fore.)**	**TSS (cal.)**	**TSS (fore.)**
Grassland	Base-case	63	171	-	-	-	-
	Livestock Integration	62	162	-	-	-	-
	Reinvigorated Youth	60	150	-	-	-	-
	CRP 0%	63	172	-	-	-	-
	Cons. Comp. X2	61	154	-	-	-	-
	Land cost X2	63	169	-	-	-	-
	Livestock costs X0.75	59	147	-	-	-	-
	Grassland 0%	62	163	-	-	-	-

Table D-4. Big Sioux River total discharge [million cubic meters (MCM)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative discharge (Total), and rank of cumulative discharge within each tillage type (1 = highest discharge estimates and 8 = lowest discharge estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	429	1,577	266	4,211	1,368	1,163	68,391	6
	Livestock Integration	429	1,578	265	4,220	1,388	1,157	69,413	4
	Reinvigorated Youth	429	1,578	265	4,228	1,424	1,196	71,201	3
	CRP 0%	429	1,578	266	4,204	1,367	1,162	68,363	7
	Cons. Comp. X2	429	1,578	265	4,226	1,424	1,195	71,207	2
	Land cost X2	429	1,577	265	4,215	1,388	1,156	69,383	5
	Livestock costs X0.75	429	1,579	265	4,238	1,425	1,197	71,262	1
	Grassland 0%	429	1,582	267	4,183	1,365	1,159	68,254	8
Conventional	Base-case	430	1,578	266	4,212	1,388	1,156	69,387	6
	Livestock Integration	430	1,578	265	4,222	1,424	1,195	71,188	4
	Reinvigorated Youth	430	1,578	265	4,228	1,424	1,196	71,225	3
	CRP 0%	430	1,578	266	4,208	1,368	1,162	68,388	7
	Cons. Comp. X2	430	1,578	265	4,226	1,425	1,196	71,225	2
	Land cost X2	430	1,578	265	4,216	1,388	1,157	69,402	5
	Livestock costs X0.75	430	1,579	265	4,237	1,426	1,197	71,282	1
	Grassland 0%	430	1,582	267	4,183	1,366	1,159	68,280	8

Table D-5. Big Sioux River total suspended solids (TSS; mg/L) for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative TSS (Total), and rank of cumulative TSS within each tillage type (1 = highest TSS estimates and 8 = lowest TSS estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	4	26	4	99	26	28	1,304	3
	Livestock Integration	4	26	4	96	26	27	1,280	7
	Reinvigorated Youth	4	24	4	91	26	27	1,299	5
	CRP 0%	4	27	4	101	26	28	1,320	2
	Cons. Comp. X2	4	25	4	92	26	27	1,295	6
	Land cost X2	4	26	4	97	26	27	1,301	4
	Livestock costs X0.75	4	24	4	89	25	26	1,257	8
	Grassland 0%	4	30	4	111	29	32	1,455	1
Conventional	Base-case	10	80	10	299	79	84	3,972	3
	Livestock Integration	10	76	10	283	78	83	3,905	4
	Reinvigorated Youth	10	69	10	260	75	78	3,766	6
	CRP 0%	10	82	10	306	80	87	3,985	2
	Cons. Comp. X2	10	71	10	268	75	79	3,750	7
	Land cost X2	10	77	10	289	77	82	3,871	5
	Livestock costs X0.75	10	66	10	250	71	74	3,572	8
	Grassland 0%	10	95	10	356	92	101	4,601	1

Table D-6. James River total erosion [megatons (million metric-tons)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative erosion (Total), and rank of cumulative erosion within each tillage type (1 = highest erosion estimates and 8 = lowest erosion estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	11.7	7.6	6.1	11.7	7.3	1.1	365.0	3
	Livestock Integration	11.7	7.4	6.0	11.7	7.2	1.1	358.6	5
	Reinvigorated Youth	11.7	7.3	6.0	11.7	7.2	1.1	358.3	6
	CRP 0%	11.7	7.7	6.2	11.7	7.4	1.0	367.8	2
	Cons. Comp. X2	11.7	7.3	5.9	11.7	7.1	1.1	356.0	7
	Land cost X2	11.7	7.5	6.1	11.7	7.2	1.1	362.4	4
	Livestock costs X0.75	11.7	7.2	5.9	11.7	7.1	1.1	354.4	8
	Grassland 0%	11.7	9.8	7.3	11.7	8.5	0.9	426.9	1
Conventional	Base-case	14.3	21.8	14.3	22.6	19.4	1.6	968.5	3
	Livestock Integration	14.3	21.2	14.3	22.0	18.9	1.5	945.3	5
	Reinvigorated Youth	14.3	20.8	14.3	21.6	18.9	1.4	944.0	6
	CRP 0%	14.3	22.1	14.3	23.0	19.6	1.7	978.2	2
	Cons. Comp. X2	14.3	20.8	14.3	21.6	18.7	1.4	936.1	7
	Land cost X2	14.3	21.5	14.3	22.3	19.2	1.5	958.9	4
	Livestock costs X0.75	14.3	20.6	14.3	21.4	18.6	1.3	929.7	8
	Grassland 0%	14.3	30.5	14.3	31.1	24.1	3.9	1202.9	1

Table D-7. James River percent of calibrated (“cal.”) and forecasted (“fore.”) mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity.

Land use and tillage type	Scenario	Erosion (cal.)	Erosion (fore.)	Discharge (cal.)	Discharge (fore.)	TSS (cal.)	TSS (fore.)
Cropland under conservation tillage	Base-case	22	28	160	62	26	36
	Livestock Integration	23	29	160	62	28	38
	Reinvigorated Youth	23	30	160	62	28	39
	CRP 0%	21	26	161	62	26	34
	Cons. Comp. X2	23	30	160	62	28	40
	Land cost X2	22	29	160	62	27	37
	Livestock costs X0.75	23	30	154	61	30	42
	Grassland 0%	22	28	174	63	11	12
Cropland under conventional tillage	Base-case	151	60	162	62	114	53
	Livestock Integration	148	60	161	62	109	52
	Reinvigorated Youth	147	60	162	62	107	52
	CRP 0%	154	61	175	64	117	54
	Cons. Comp. X2	147	59	161	62	106	52
	Land cost X2	150	60	161	62	112	53
	Livestock costs X0.75	146	59	161	62	104	51
	Grassland 0%	150	60	161	62	175	64

Table D-8. James River percent of calibrated ("cal.") and forecasted ("fore.") mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity (continued).

Land use and tillage type	Scenario	Erosion (cal.)	Erosion (fore.)	Discharge (cal.)	Discharge (fore.)	TSS (cal.)	TSS (fore.)
Grassland	Base-case	29	40	-	-	-	-
	Livestock Integration	28	40	-	-	-	-
	Reinvigorated Youth	28	39	-	-	-	-
	CRP0%	29	41	-	-	-	-
	Cons. Comp. X2	28	39	-	-	-	-
	Land cost X2	29	40	-	-	-	-
	Livestock costs X0.75	28	38	-	-	-	-
	Grassland 0%	28	40	-	-	-	-

Table D-9. James River total discharge [million cubic meters (MCM)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative discharge (Total), and rank of cumulative discharge within each tillage type (1 = highest discharge estimates and 8 = lowest discharge estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	607	3,127	299	5,020	1,282	1,192	64,087	3
	Livestock Integration	608	3,127	297	5,010	1,279	1,191	63,975	6
	Reinvigorated Youth	607	3,123	298	5,006	1,280	1,190	63,975	5
	CRP 0%	608	3,129	298	5,023	1,283	1,193	64,127	2
	Cons. Comp. X2	608	3,124	297	5,001	1,279	1,190	63,926	7
	Land cost X2	608	3,126	297	5,016	1,281	1,192	64,030	4
	Livestock costs X0.75	607	3,122	297	4,991	1,252	1,175	62,624	8
	Grassland 0%	608	3,223	312	5,289	1,344	1,244	67,191	1
Conventional	Base-case	611	3,135	296	5,046	1,287	1,197	64,341	3
	Livestock Integration	612	3,133	297	5,034	1,284	1,195	64,213	5
	Reinvigorated Youth	611	3,131	296	5,026	1,284	1,194	64,206	6
	CRP 0%	612	3,138	298	5,049	1,288	1,198	64,391	2
	Cons. Comp. X2	612	3,132	295	5,024	1,283	1,193	64,165	7
	Land cost X2	612	3,133	297	5,041	1,286	1,196	64,295	4
	Livestock costs X0.75	611	3,126	296	5,017	1,282	1,192	64,106	8
	Grassland 0%	612	3,234	310	5,330	1,351	1,250	67,545	1

Table D-10. James River total suspended solids (TSS; mg/L) for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative TSS (Total), and rank of cumulative TSS within each tillage type (1 = highest TSS estimates and 8 = lowest TSS estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	145	70	58	177	88	24	4,401	3
	Livestock Integration	145	68	57	175	87	24	4,338	5
	Reinvigorated Youth	145	67	57	175	87	24	4,331	6
	CRP 0%	145	71	59	179	89	24	4,431	2
	Cons. Comp. X2	145	67	57	174	86	24	4,307	7
	Land cost X2	145	69	58	177	88	24	4,377	4
	Livestock costs X0.75	145	67	57	173	85	24	4,269	8
	Grassland 0%	145	89	68	200	101	26	5,046	1
Conventional	Base-case	177	202	137	526	236	68	11,791	3
	Livestock Integration	177	198	136	502	231	65	11,535	5
	Reinvigorated Youth	177	194	136	509	230	66	11,522	6
	CRP 0%	177	205	136	523	238	68	11,883	2
	Cons. Comp. X2	177	194	136	502	229	65	11,428	7
	Land cost X2	177	199	136	515	234	67	11,681	4
	Livestock costs X0.75	177	192	136	498	227	64	11,362	8
	Grassland 0%	177	275	138	626	285	89	14,271	1

Table D-11. Bad River total erosion [megatons (million metric-tons)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative erosion (Total), and rank of cumulative erosion within each tillage type (1 = highest erosion estimates and 8 = lowest erosion estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	1.9	2.4	1.9	3.0	2.5	0.3	124.7	3
	Livestock Integration	3.2	2.3	1.8	3.2	2.3	0.3	117.4	5
	Reinvigorated Youth	3.2	2.0	1.8	3.2	2.3	0.2	115.4	6
	CRP 0%	3.2	2.5	1.9	3.2	2.6	0.3	130.7	2
	Cons. Comp. X2	3.2	2.0	1.8	3.2	2.2	0.2	111.8	7
	Land cost X2	3.2	2.3	1.9	3.2	2.4	0.3	123.0	4
	Livestock costs X0.75	3.2	1.9	1.7	3.2	2.1	0.2	106.8	8
	Grassland 0%	3.2	3.4	1.9	4.1	3.3	0.5	168.4	1
Conventional	Base-case	2.7	5.2	2.7	6.5	5.2	0.7	261.1	3
	Livestock Integration	2.7	4.3	2.7	5.1	4.2	0.5	210.9	5
	Reinvigorated Youth	2.7	3.0	2.7	5.1	4.0	0.5	199.7	6
	CRP 0%	2.7	5.7	2.7	6.9	5.5	0.8	275.8	2
	Cons. Comp. X2	2.7	3.3	2.7	4.4	3.6	0.3	181.9	7
	Land cost X2	2.7	4.6	2.7	5.9	4.8	0.6	239.7	4
	Livestock costs X0.75	2.7	2.6	2.4	4.5	3.2	0.4	159.0	8
	Grassland 0%	2.7	9.2	2.7	11.1	8.6	1.8	431.4	1

Table D-12. Bad River percent of calibrated ("cal.") and forecasted ("fore.") mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity.

Land use and tillage type	Scenario	Erosion (cal.)	Erosion (fore.)	Discharge (cal.)	Discharge (fore.)	TSS (cal.)	TSS (fore.)
Cropland under conservation tillage	Base-case	8	8	260	72	28	22
	Livestock Integration	9	8	260	72	17	15
	Reinvigorated Youth	8	8	260	72	7	7
	CRP 0%	8	7	260	72	32	24
	Cons. Comp. X2	9	8	260	72	8	8
	Land cost X2	9	8	260	72	22	18
	Livestock costs X0.75	8	8	260	72	0	0
	Grassland 0%	1	1	260	72	73	42
Cropland under conventional tillage	Base-case	247	71	260	72	172	63
	Livestock Integration	249	71	260	72	144	59
	Reinvigorated Youth	247	71	261	72	69	41
	CRP 0%	246	71	261	72	193	66
	Cons. Comp. X2	249	71	260	72	74	43
	Land cost X2	249	71	260	72	143	59
	Livestock costs X0.75	248	71	260	72	36	26
	Grassland 0%	218	69	260	72	368	79

Table D-13. Bad River percent of calibrated ("cal.") and forecasted ("fore.") mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity (continued).

Land use and tillage type	**Scenario**	**Erosion (cal.)**	**Erosion (fore.)**	**Discharge (cal.)**	**Discharge (fore.)**	**TSS (cal.)**	**TSS (fore.)**
Grassland	Base-case	159	61	-	-	-	-
	Livestock Integration	168	63	-	-	-	-
	Reinvigorated Youth	179	64	-	-	-	-
	CRP 0%	151	60	-	-	-	-
	Cons. Comp. X2	177	64	-	-	-	-
	Land cost X2	162	62	-	-	-	-
	Livestock costs X0.75	182	65	-	-	-	-
	Grassland 0%	163	62	-	-	-	-

Table D-14. Bad River total discharge [million cubic meters (MCM)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative discharge (Total), and rank of cumulative discharge within each tillage type (1 = highest discharge estimates and 8 = lowest discharge estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	470	684	256	1,145	539	197	26,936	8
	Livestock Integration	470	684	255	1,148	539	198	26,937	7
	Reinvigorated Youth	470	683	255	1,148	539	198	26,951	4
	CRP 0%	470	684	256	1,148	539	197	26,952	3
	Cons. Comp. X2	470	684	255	1,149	539	198	26,955	2
	Land cost X2	470	684	256	1,145	539	197	26,956	1
	Livestock costs X0.75	470	683	255	1,147	539	198	26,950	5
	Grassland 0%	470	685	258	1,144	539	197	26,947	6
Conventional	Base-case	470	685	256	1,146	540	197	26,977	4
	Livestock Integration	470	684	255	1,149	539	198	26,965	8
	Reinvigorated Youth	470	684	255	1,148	540	198	26,977	5
	CRP 0%	470	685	256	1,149	540	197	26,992	2
	Cons. Comp. X2	470	684	255	1,149	540	198	26,975	6
	Land cost X2	470	684	256	1,146	540	197	26,986	3
	Livestock costs X0.75	470	684	255	1,147	539	198	26,966	7
	Grassland 0%	470	687	258	1,146	541	197	27,029	1

Table D-15. Bad River total suspended solids (TSS; mg/L) for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative TSS (Total), and rank of cumulative TSS within each tillage type (1 = highest TSS estimates and 8 = lowest TSS estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	114	366	114	622	270	145	13,524	3
	Livestock Integration	114	338	114	585	248	134	12,394	5
	Reinvigorated Youth	114	299	114	604	243	129	12,142	6
	CRP 0%	114	380	114	631	277	149	13,829	2
	Cons. Comp. X2	114	306	113	579	236	127	11,817	7
	Land cost X2	114	348	114	608	260	140	12,994	4
	Livestock costs X0.75	114	284	105	583	226	122	11,287	8
	Grassland 0%	114	502	114	809	356	194	17,822	1
Conventional	Base-case	157	773	157	1,259	556	304	27,784	3
	Livestock Integration	470	684	255	1,149	539	198	26,965	4
	Reinvigorated Youth	157	442	157	1,104	417	220	20,845	6
	CRP 0%	157	844	157	1,367	588	325	29,421	2
	Cons. Comp. X2	157	484	157	947	383	207	19,145	7
	Land cost X2	157	681	157	1,124	509	275	25,425	5
	Livestock costs X0.75	157	372	146	981	333	188	16,657	8
	Grassland 0%	157	1,360	157	2,188	921	521	46,036	1

Table D-16. Belle Fourche River total erosion [megatons (million metric-tons)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative erosion (Total), and rank of cumulative erosion within each tillage type (1 = highest erosion estimates and 8 = lowest erosion estimates).

Tillage type	**Scenario**	**2013**	**2062**	**Minimum**	**Maximum**	**Mean**	**SD**	**Total**	**Rank**
Conservation	Base-case	0.6	0.7	0.4	0.8	0.6	0.1	29.7	3
	Livestock Integration	0.6	0.6	0.4	0.8	0.5	0.1	27.3	5
	Reinvigorated Youth	0.6	0.6	0.4	0.8	0.5	0.1	26.5	6
	CRP 0%	0.6	0.7	0.4	0.9	0.6	0.1	30.4	2
	Cons. Comp. X2	0.6	0.6	0.4	0.7	0.5	0.1	25.2	7
	Land cost X2	0.6	0.7	0.4	0.8	0.6	0.1	28.7	4
	Livestock costs X0.75	0.6	0.6	0.4	0.7	0.5	0.1	24.8	8
	Grassland 0%	0.6	1.1	0.5	1.3	0.9	0.2	44.3	1
Conventional	Base-case	0.7	1.2	0.7	1.4	1.0	0.2	49.7	3
	Livestock Integration	0.7	1.0	0.5	1.1	0.8	0.1	38.6	5
	Reinvigorated Youth	0.7	0.7	0.5	1.2	0.7	0.1	37.1	6
	CRP 0%	0.7	1.3	0.7	1.5	1.1	0.2	53.0	2
	Cons. Comp. X2	0.7	0.7	0.5	0.9	0.7	0.1	32.5	7
	Land cost X2	0.7	1.1	0.6	1.3	0.9	0.1	45.1	4
	Livestock costs X0.75	0.7	0.7	0.4	0.9	0.6	0.1	30.7	8
	Grassland 0%	0.7	3.1	0.7	3.4	2.2	0.6	111.8	1

Table D-17. Belle Fourche River percent of calibrated ("cal.") and forecasted ("fore.") mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity.

Land use and tillage type	Scenario	Erosion (cal.)	Erosion (fore.)	Discharge (cal.)	Discharge (fore.)	TSS (cal.)	TSS (fore.)
Cropland under conservation tillage	Base-case	111	53	16	20	38	61
	Livestock Integration	111	53	17	20	42	72
	Reinvigorated Youth	111	53	18	23	49	94
	CRP 0%	524	84	15	18	38	60
	Cons. Comp. X2	112	53	20	25	49	95
	Land cost X2	111	53	17	20	40	66
	Livestock costs X0.75	112	53	21	27	51	103
	Grassland 0%	557	85	13	14	10	11
Cropland under conventional tillage	Base-case	577	85	15	18	10	9
	Livestock Integration	578	85	17	20	12	14
	Reinvigorated Youth	577	85	18	23	17	14
	CRP 0%	1,899	95	15	18	143	59
	Cons. Comp. X2	579	85	19	23	30	42
	Land cost X2	577	85	17	20	33	49
	Livestock costs X0.75	580	85	21	27	1	1
	Grassland 0%	938	90	12	14	42	72

Table D-18. Belle Fourche River percent of calibrated ("cal.") and forecasted ("fore.") mean bias for erosion, discharge, and total suspended solids (TSS). Results are reported for conservation and conventional tillage and grassland erosion rates, but not for discharge and TSS as there was no historical data for discharge or TSS specifically from grassland (see Chapter 2). If percent predicted mean bias is greater than percent calibrated it is an indication of sensitivity (continued).

Land use and tillage type	Scenario	Erosion (cal.)	Erosion (fore.)	Discharge (cal.)	Discharge (fore.)	TSS (cal.)	TSS (fore.)
Grassland	Base-case	56	36	-	-	-	-
	Livestock Integration	57	36	-	-	-	-
	Reinvigorated Youth	47	32	-	-	-	-
	CRP 0%	55	36	-	-	-	-
	Cons. Comp. X2	47	32	-	-	-	-
	Land cost X2	57	36	-	-	-	-
	Livestock costs X0.75	30	23	-	-	-	-
	Grassland 0%	38	27	-	-	-	-

Table D-19. Belle Fourche River total discharge [million cubic meters (MCM)] for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative discharge (Total), and rank of cumulative discharge within each tillage type (1 = highest discharge estimates and 8 = lowest discharge estimates).

Tillage type	Scenario	2013	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	118	435	55	726	256	196	12,795	3
	Livestock Integration	118	431	55	721	254	194	12,716	5
	Reinvigorated Youth	118	430	56	723	250	193	12,494	6
	CRP 0%	118	437	55	725	260	195	12,979	2
	Cons. Comp. X2	118	429	56	721	246	193	12,287	7
	Land cost X2	118	433	55	724	255	194	12,736	4
	Livestock costs X0.75	118	429	53	721	241	192	12,059	8
	Grassland 0%	118	453	54	774	268	204	13,386	1
Conventional	Base-case	118	435	55	726	260	196	12,996	2
	Livestock Integration	118	431	55	722	254	194	12,721	5
	Reinvigorated Youth	118	430	56	723	250	193	12,499	6
	CRP 0%	118	437	55	725	260	195	12,987	3
	Cons. Comp. X2	118	429	56	721	250	193	12,483	7
	Land cost X2	118	433	55	725	255	195	12,744	4
	Livestock costs X0.75	118	429	53	721	241	193	12,061	8
	Grassland 0%	118	454	53	778	268	204	13,410	1

Table D-20. Belle Fourche River total suspended solids (TSS; mg/L) for conservation tillage and conventional tillage scenarios from 2013 to 2062 including minimum, maximum, mean, standard deviation (SD), cumulative TSS (Total), and rank of cumulative TSS within each tillage type (1 = highest TSS estimates and 8 = lowest TSS estimates).

Tillage type	Scenario	2012	2062	Minimum	Maximum	Mean	SD	Total	Rank
Conservation	Base-case	40	51	30	124	52	17	2,623	3
	Livestock Integration	40	47	27	118	49	17	2,440	5
	Reinvigorated Youth	40	41	24	103	47	15	2,359	6
	CRP 0%	40	52	30	121	53	17	2,655	2
	Cons. Comp. X2	40	40	24	107	45	15	2,235	7
	Land cost X2	40	48	29	120	51	17	2,552	4
	Livestock costs X0.75	40	39	23	101	44	15	2,184	8
	Grassland 0%	40	77	34	173	73	25	3,633	1
Conventional	Base-case	48	89	44	218	89	31	4,437	3
	Livestock Integration	48	72	39	181	69	25	3,464	5
	Reinvigorated Youth	48	52	34	150	67	23	3,343	6
	CRP 0%	48	97	45	234	94	34	4,680	2
	Cons. Comp. X2	48	51	32	143	58	20	2,888	7
	Land cost X2	48	78	42	212	81	29	4,068	4
	Livestock costs X0.75	48	45	28	122	53	18	2,666	8
	Grassland 0%	48	207	48	490	183	75	9,143	1

Table D-21. National Resource Conservation Service general cropland erosion tolerance levels. Exceedance of maximum tolerance levels threatens cropland productivity (USDA, 2001).

Tolerance value	Tons/ac/yr	Metric-tons/ha/yr
1	1.0	2.2417
2	2.0	4.4834
3	3.0	6.72511
4	4.0	8.96681
5	5.0	11.2085

Table D-22. Erosion rates for the Big Sioux, James, Bad, and Belle Fourche water-catchments which includes land use, tillage (if land use is cropland), minimum and maximum erosion rates (metric-tons/ha/yr). An indication of exceedance above the maximum tolerable erosion rate (11.2085 metric-ton/ha/yr; see Table A-17) is denoted by "yes" or "no" and percent of estimates for all scenarios that exceeded tolerable erosions standards are reported.

Water-catchment	**Land use type and tillage**	**Minimum**	**Maximum**	**Exceedance: yes or no**	**Percent exceedance**
Big Sioux River	Cropland conservation	≥ 2.0	≤ 5.0	No	0
	Cropland conventional	≥ 10.0	≤ 15.0	Yes	80
	Grassland	≥ 1.0	≤ 1.7	-	-
James River	Cropland conservation	≥ 2.0	≤ 5.0	No	0
	Cropland conventional	≥ 5.0	≤ 8.0	No	0
	Grassland	≥ 0.4	≤ 0.5	-	-
Bad River	Cropland conservation	≥ 5.0	≤ 7.0	No	0
	Cropland conventional	≥ 9.0	≤ 22.0	Yes	98
	Grassland	≥ 1.3	≤ 2.0	-	-
Belle Fourche River	Cropland conservation	≥ 2.0	≤ 8.0	No	0
	Cropland conventional	≥ 4.0	≤ 26.0	Yes	19.5
	Grassland	≥ 0.4	≤ 0.9	-	-

Table D-23. Total suspended solids rates for the Big Sioux, James, Bad, and Belle Fourche water-catchments which includes tillage, minimum and maximum TSS rates (mg/L/yr). An indication of exceedance of the maximum tolerable TSS rate (158 mg/L/yr, see http://denr.sd.gov/dfta/wp/wqinfo.aspx) is denoted by "yes" or "no" and percent of estimates for all scenarios that exceeded tolerable TSS standards.

Water-catchment	**Tillage type**	**Minimum**	**Maximum**	**Exceedance: yes or no**	**Percent exceedance**
Big Sioux River	Conservation	≥ 4	≤ 101	No	0
	Conventional	≥ 250	≤ 356	Yes	100
James River	Conservation	≥ 57	≤ 200	Yes	≤ 1
	Conventional	≥ 136	≤ 626	Yes	96 − 98
Bad River	Conservation	≥ 105	≤ 809	Yes	56 −100
	Conventional	≥ 146	≤ 2,188	Yes	96 −100
Belle Fourche River	Conservation	≥ 23	≤ 173	Yes	≤ 2
	Conventional	≥ 28	≤ 490	Yes	0 − 66