Modeling the effects of future land use change on water quality under multiple scenarios: a case study of low-input agriculture with hay/pasture production

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Abstract

Understanding the relationship between land use and freshwater quality is necessary for effective water management. This study sought to evaluate the impacts of future land use change on water quality in an agriculture dominated watershed in South Dakota. Taking into account three cases (A1B, A2 and B1) of the FOREcasting SCEnarios (FORE-SCE) of Land use Change modeling framework, possible changes in surface runoff, sediment, nitrate and total phosphorus by the end of the 21st century were assessed relative to the baseline (National Land Cover Database 2011) comprising a multi-year period from the recent past (2006-2014). The Soil and Water Assessment Tool (SWAT) was used for simulating hydrology and water quality, where particular attention was given to land transformation from “high-input” to “low-input” agriculture. The analysis revealed that urban areas and low-input hay/pasture production would expand from conversion of forest, grassland and high-input cultivated cropland. While afforestation might also occur under certain future scenario assumptions such as B1, all the three scenarios are suggestive of complete grassland depletion by the first quarter of this century. Simulation results suggested that water quality would improve with expansion of hay/pasture production (as low-input agriculture), reducing surface runoff volume, sediment, nitrate and total phosphorus loads by 3-14% across all three future scenarios of land use change. This study provides an example on how physically-based land use projections can be ingested in SWAT modeling to identify possible environmental implications of future land use changes. Similar studies adopted on large scales would be useful to support holistic water resources management strategies and associated policy interventions.

Key Terms: FORE-SCE, grassland, hay production, land use, South Dakota, SWAT

INTRODUCTION

Under the prevailing global environmental stress, there is a need to develop holistic approaches for evaluating future sustainability of water resources, including the associated ecosystems. Studying climate change impacts on water resources has drawn increasing attention over the last decades (IPCC, 2014a,b), although land use change may be of equal importance. While climate change is relatively a slower process, change in land use can be drastic with considerable impacts on ecosystems and environmental processes (e.g. DeFries et al., 2004; El-Khoury et al., 2015; Fan and Shibata, 2015).

With the increasing population and associated food demand, conversion of natural grassland to agricultural areas has been the typical trend of land use change in the Midwest and Great Plains regions of United States after mid 20th century (e.g. Claassen et al., 2011; Ramankutty and Foley, 1999; Sleeter et al., 2013; Wright and Wimberly, 2013). In the Great Plains States (Kansas, Nebraska, North Dakota, and South Dakota), about 57% of grass/rangeland went under agricultural production between 1997 and 2007, an estimated total of more than 3200 km². Nationally, approximately 110,000 km² of grasslands were converted mostly to agricultural areas between 1950 and 1990 (Claassen et al., 2011). Research showed that replacement of native grassland by agricultural land is associated with many environmental consequences such as alteration of natural hydrological processes, acceleration of soil erosion, decrease in ecosystem carbon sequestration functions, and decrease in the variety and abundance of wildlife (e.g. Cox and Rundquist, 2013; Dahlgren et al., 2001; Schilling et al., 2008). Another trend of land use change in the Midwest has been the transitions of grassland or cultivated cropland to urban areas. With an average annual growth rate of 3.8% during 2000–2009, the population is growing rapidly in the Midwestern states (Eathington, 2010), leading to increased

urbanization and the trend seems to persist in future (DeFries et al., 2004; Drummond et al., 2012; Sleeter et al., 2013).

Trend of land use change within a region is predominantly dictated by demand-supply balance, economic conditions, governmental policy, conservation practices and emission developments. As an implication of recent public policy enactment such as the Energy Independence and Security Act (EISA, 2007), researchers indicated that increased agriculture in the Midwest and Great Plains in recent years is mostly governed by biofuel production (e.g. Drummond et al., 2012; Schilling et al., 2008; Sohl et al., 2012a, 2014). Corn-based ethanol and soybean-based biodiesel so far have been the two appreciated biofuels in United States (Hill et al., 2006). However, a thorough consideration of their life cycle tags them as a “high-input” agriculture, mostly due to their intense production process involving energy for machineries, irrigation and transportation, space for grain storage, nutrient input and the associated environmental impacts (Thomas et al., 2014). Deterioration of water quality by agricultural activities received considerable attention as a major contributor of nonpoint source pollution (Chaubey et al., 2010; Yu et al., 2004). Intensive agricultural practices have been identified to release substantial amounts of nutrients, especially nitrogen and phosphorus to receiving water bodies; sediment loss from top soils containing relatively large amounts of nutrients can decrease soil productivity (Gillingham and Thorrold, 2000; Monaghan et al., 2005; USEPA, 1990). Therefore, it becomes necessary to analyze possible future changes in agricultural land use practices and the expectancy of increase or decrease in their ability to affect the environment.

Besides row crops such as corn and soybean, livestock production is a major agricultural enterprise in the Great Plains and Midwestern states, pointing towards the importance of the forage industry in this region. As human population continues to grow, so is the increase in

demand for meat and dairy products. Since hay/pasture is widely used for livestock fodder as well as land conservation and reclamation, their production drives the market value of grassland (Claassen et al., 2011). Considering the demand to increase production of cellulosic material to meet the need of environment-friendly biofuel (e.g. switchgrass), and at the same time, the market value of livestock fodder, growing perennial grasses on the land unsuitable for food-based row crop production is recently being encouraged in the Midwest and Great Plains (McGuire and Rupp, 2013; Oak Ridge National Laboratory, 2011; CenUSA Bioenergy, 2013). Use of degraded and marginal lands in such cases of course does not compete with the demand of more traditional crops. However, it is also not impossible that a cropland used for growing corn or soybean might get used for perennial grasses depending on how the biofuel feedstock and livestock industry boom in near future. Therefore, even though a decline in grassland in Upper US Midwest and Great Plains logically presumes a resultant increased percentage of agricultural land, a closer look might reveal a paradigm change from “high-input” to “low-input” agriculture. The notion of low-input agriculture in this study refers to the low production input (e.g. fertilizer) and maintenance requirements for perennial grasses compared to that in traditional crop production (e.g. corn and soybean) which involve relatively intense use of off-farm resources (Graham et al., 1995; McLaughlin and Walsh, 1998; McLaughlin et al., 1999).

Understanding the possible future implications of growing perennial grasses on water quality would help develop land management practices compatibility with conventional farming practices, to foster biofuel feedstock and livestock, while still maintaining production of food-based crops (McGuire and Rupp, 2013; CenUSA Bioenergy, 2013). Many of the previous studies analyzing the impacts of land use change on hydrology/water quality in agriculture dominated watersheds have been conducted with “what if” land use scenarios (e.g. Neupane and Kumar,

2015; Wu and Liu, 2012; Yang et al., 2014), constructed by adjusting percentage changes of specific land classes. While this approach provides useful insight into how land transformation affects hydrology and water quality, the use of physically-derived scenarios in land use change analysis, not only from one class to another (e.g. forest to agriculture) but also within individual land use classes, is necessary to identify the critical source areas of environmental pollution and their spatio-temporal pattern of changes in future times. Specifically, there is a need to evaluate water quality impacts of converting annual crops (“high-input”) to perennial crops (“low-input”) within the agricultural land use class. Many of the previous studies slanted towards assessing the impacts/advantages of low-input agriculture, solely focused on second-generation biofuels (e.g. Demissie et al., 2012; Ng et al., 2010; Wu and Liu, 2012; Wu et al., 2012). Similar research on hay/pasture production is needed for a comprehensive understanding of the environmental benefits of low-input agriculture.

The general purpose of this study was to provide, through a case study, an elaborated view on the implications of increased hay/pasture production on water quality. The specific objective was to quantify the impacts of such land use changes on water quality indicators in an agriculture dominated watershed in the Upper Midwest of United States through direct ingestion of spatially explicit future land use projections into the Soil and Water Assessment Tool (SWAT). The water quality indicators considered in this study include sediment, total phosphorus and nitrate loadings. To help correlate water quality impacts with future land use under three plausible scenarios, an analytical comparison on the current and projected dynamics of land use change in the study watershed is also provided. Selection of water quality indicators was constrained by the availability of field-measurements essential to calibrate the SWAT model. South Dakota was chosen as the region of study since it is one of the states with highest

grassland conversion rates in the Great Plains and Midwestern United States (Singh, 2013; Wright and Wimberly, 2013).

2. STUDY AREA

This study was conducted in Skunk Creek watershed, located in southeast South Dakota with a drainage area of a 1605 km² (Figure 1). The annual average rainfall and snowfall in the watershed between 2005 and 2014 were 580 and 86 mm respectively, and the maximum annual stream discharge recorded at the watershed outlet was 215 m³/s. According to the National Land Cover Database (NLCD 2011), this watershed is primarily dominated by agricultural land, constituting 65% of the watershed area. About 6.5% of the watershed is urbanized, covering a large portion of Sioux Falls which is the largest urban area in the state. A comparison of NLCD 1992 and NLCD 2011 revealed more than 3% depletion of grassland and 5% increase in urban areas during the 20 year time period, while total agricultural area remained nearly the same (Paul et al., 2016). Based on the information extracted from State Soil Geographic (STATSGO) data, the dominant hydrologic soil group in the Skunk Creek watershed is "B", with more than 60% silt content in the top 100 cm of the soil profile. Major cultivated crops in the watershed are corn and soybean, followed by hay and wheat. The watershed has multiple poultry and dairy cattle production facilities with more than 200 animal feeding operations; 66% of the livestock being cattle (East Dakota Water Development District, 2004). Historically, water quality problems in the watershed are caused by *E. coli* or fecal coliform bacteria and Total Suspended Solids (East Dakota Water Development District, 2004; RESPEC, 2013).

3. METHODOLOGY

3.1 FORE-SCE of land use change

The FORE-SCE model was developed by the United States Geological Survey (USGS) Earth Resources Observation and Science (EROS) Center to provide spatially explicit projections of plausible future land use change (Sohl and Sayler, 2008; Sohl et al., 2007). FORE-SCE uses a ‘demand’ and a ‘spatial allocation’ module to project annual raster-based future land use maps. Historical land use data as well as long-term trends of change are combined with equally plausible demographic, socio-economic, technological and emission story lines (Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios, IPCC-SRES; IPCC-SRES (2000); Strengers et al., 2004; Van Vuuren and O’Neill, 2006) to construct scenarios with future proportions of land use changes over 84 ecoregions across United States. Guided by these ecoregion-based change proportions, the ‘spatial allocation’ component of the FORE-SCE model produces spatially explicit maps for each year, putting individual land use patches on the landscape until ‘demand’ for a scenario is met for a given year. Patch placements are dictated by the existing land use class and their change trends. Therefore, these land use data should not be considered as simple extrapolated projections of current land use types based on their historical trends within a domain of interest. Further details about the model and scenario characteristics can be found in Sleeter et al. (2012) and Sohl et al. (2012b).

In this study, FORE-SCE projected land use from 2012 to 2091, each for three IPCC-SRES scenarios (A1B, A2 and B1) were extracted from <http://landcover-modeling.cr.usgs.gov/> in raster format. The FORE-SCE raster datasets originally have a spatial resolution of 250 m with 17 distinct land classes. In order to maintain consistency, FORE-SCE data were re-sampled to 30 m and reclassified to match the land use class definition used in the NLCD 2011.

3.2 SWAT model

SWAT (Neitsch et al., 2011; Arnold et al., 2012) is a physically-based, semi-distributed watershed model widely used to assess the impact of land management practices and climatic change on long-term water, sediment and pollutant yields. The model has been widely used for hydrologic and water quality simulations in many geographic regions (e.g. Heuvelmans et al., 2004; White and Chaubey, 2005; Abbaspour et al., 2007; Gassman et al., 2007; Jha et al., 2007). SWAT delineates a watershed with sub-watersheds, which can be further subdivided into Hydrologic Response Units (HRUs) (Neitsch et al., 2011; Arnold et al., 2010; Arnold et al., 2012). Each HRU is a homogeneous area of aggregated land use, soil, and slope, representing the smallest modeling unit. SWAT is undergoing continuous development in its geospatial structure, with a view to represent physical characteristics of a landscape as realistically as possible (e.g. Arnold et al., 2010; Bosch et al., 2010; Bonumá et al., 2014; Rathjens et al., 2015; Sun et al., 2015).

3.3 SWAT input data

In order to analyze the impacts of future land use change on water quality in the Skunk Creek watershed, a ‘baseline’ SWAT model was created for a 10-year time period (2005-2014). Creation of a SWAT model in the GIS interface (ArcSWAT) requires topography, soil texture, land use and climate data. For the baseline model, these data were obtained from the following sources: 30 m digital elevation model (DEM) from USGS National Elevation Dataset (USGS-NED, 2015); 30 m land use data from the National Land Cover Database (NLCD) for the year 2011 (USGS-NLCD, 2015); and 1:250,000 scale State Soil Geographic (STATSGO) data

included within SWAT database. Total daily precipitation, minimum and maximum daily temperature data for the study time period were obtained from the National Climatic Data Center for the stations that fall within or adjacent to the watershed boundary (Figure 1). All other related climatic components (e.g. solar radiation and relative humidity) were developed through the internal weather generator within ArcSWAT. Penman-Monteith equation was selected for computing potential evapotranspiration (PET). The study watershed was first divided into 81 sub-basins based on the topographic analysis of flow direction and accumulation; then all sub-basins were further discretized into HRUs using a 3% aggregation threshold for land use, soil and slope. A 3% HRU land aggregation threshold was used in this study to reduce the simulation time; a smaller (or zero) threshold value leads to higher number of HRUs, therefore, requiring greater computational demand. Curve Number and Variable Storage methods (Neitsch et al., 2011) were selected for surface runoff generation and channel routing simulation, respectively.

For agricultural land portions of the watershed, generic management operations were incorporated into the model at HRU level based on the best available information (Table 1). Corn and soybean, the two predominant crops in the watershed, were rotated on annual basis with no pre-harvest tillage. Timing for plantation, harvest and kill operations were set on the basis of personal communication with growers within the watershed area, and cross-checked with crop and state-specific database from the National Agricultural Statistics Service (NASS, 2010). Type, rate and number of chemical applications (fertilizer and herbicide) per crop per year were determined by taking the multi-year average survey results published by NASS Agricultural Chemical Use Program (NASS, 2012; 2014). Considering the large area of the watershed (1605 km²), it is very much likely that chemical type and application timing in respective fields can be quite subjective; hence, only the most common practice was chosen for inclusion in this study. In

cases where multiple applications were involved within the same growth cycle, timings are adjusted based on the relevant literature (Alley et al., 2009; Clay et al., 2009; Franzen, 2013; Moechnig et al., 2013a,b; Reitsma et al., 2008). For example, nitrogen application for corn followed a ‘split-dress’ approach where majority portion was applied during the very early growth stage (within 10 days of planting) and the rest within the first six weeks. Similarly, interval of successive herbicide applications before and after soybean plantation was determined based on the crop growth stages. Cattle grazing, manure application and tile drainage were not included in modeling.

3.4 Calibration and validation of SWAT

A logical order of parameter optimization was followed while calibrating the model (e.g. Abbaspour et al., 2007; White and Chaubey, 2005). The model was first calibrated for hydrology (streamflow) because of its ability to influence and control surface and subsurface nutrient transport processes. Measurement uncertainty was assumed negligible in the hydrologic data since the observed streamflow was obtained from gauge readings, whereas sediment and nutrient loads were basically regression-based estimates from the Load Estimator (LOADEST) model (Runkel et al., 2004). Hydrologic calibrations of the model were followed by sediment, then total phosphorus and finally nitrate calibration. A 9-year period (2005-2013) was selected for model calibration, where 2005 was used as the model warm-up period to initialize and train the model until reasonable values of model parameters were obtained. The calibrated model, being set with the best estimates of parameters, was validated with the 2014 data. A relatively longer period of calibration was preferred in this case in order to incorporate the influence of both dry and wet flow conditions of the watershed. This was important for water quality simulation due to the lack

of sufficient observed data. Any equifinality induced from the choice of calibration and validation phase may not be critical when the forward simulations (based on the calibrated model) are reported relative to a baseline reference period.

Daily streamflow data for model evaluation were obtained from the USGS gauge station (USGS 06481500) at the watershed outlet as shown in Figure 1. Measurements for sediment, total phosphorus and nitrate concentrations for the simulation period (2005-2014) were extracted from STORET/WQX data repository framework of the US Environmental Protection Agency (<http://www3.epa.gov/storet/>); these water quality measurements correspond to the same location as for the USGS streamflow observation. These data were a compilation of grab samples, not being collected on a regular basis. Therefore, continuous average monthly loads at the watershed outlet were estimated for the whole period of simulation (2005-2014) by ingesting the STORET concentration values into LOADEST (<http://water.usgs.gov/software/loadest/>). The resulting monthly time-series estimates were then used in place of measured data while conducting SWAT calibration and validation for water quality. While regression models in LOADEST aid conceptual understanding of the controls on watersheds' solute export, they can generate poor flux estimates due to inadequacy of regression formulas to capture entirely the relationship between constituent concentrations and available predictors such as discharge (Appling et al., 2015). Regardless, in case of discontinuous water quality data such as Skunk Creek, the use of LOADEST outputs has been a common practice (e.g. Hutchinson and Christiansen, 2013; White and Chaubey, 2005).

Calibration was performed in SWATShare (<https://mygeohub.org/groups/water-hub/swatshare>; Rajib et al., 2016a), which is a cyber-infrastructure (CI) for high performance simulation as well as sharing and visualization of SWAT models. The current version of

SWATShare uses the Parameter Solution (ParaSol) algorithm to perform Latin Hypercube Sampling and subsequent parameter optimization. SWAT parameters and their initial value ranges (see Tables 2 and 3) were selected based on the review of existing literature on adjacent Midwestern agricultural watersheds (e.g. Folle, 2010; Hutchinson and Christiansen, 2013; Jha et al., 2007; Neupane and Kumar, 2015) and suggestions from model developers (Abbaspour et al., 2007; Neitsch et al., 2011). During the first phase of calibration against observed streamflow data, 19 parameters involving surface, subsurface and channel hydrologic responses were optimized. In subsequent steps, parameters related to sediment, total phosphorus and nitrate (Table 3) were sequentially put into the optimization scheme to re-calibrate the partially calibrated model from the previous step. In total, 38 parameters were optimized to get the calibrated SWAT outputs on hydrology and water quality. Nash-Sutcliffe Efficiency (NSE) between simulated and reference data was used as the objective function, which was maximized to reach the best parameter estimates. Coefficient of Determination (R^2) and Percent Bias (PBIAS, Sorooshian et al., 1993) were also calculated to evaluate the goodness of fit between the observation/reference estimate and the best simulation having maximum objective function value (NSE in this case).

Average annual calibrated outputs, based on the NLCD 2011 land use over the 9-year period of recent past (2006-2014, excluding the warm-up year 2005), were taken as the reference condition (i.e. baseline) for this study.

3.5 Modeling the effects of future land use change scenarios in SWAT

Possible future dynamics and interchanges among different land use classes were first analyzed by comparing the three scenarios (A1B, A2 and B1) of FORE-SCE projections from

2012 to 2091. In order to represent the most extreme changes in water quality based on this preliminary analysis, land use scenarios for 2091 were selected for inclusion in SWAT modeling. Three separate SWAT configurations were created by replacing NLCD 2011 land use data in the calibrated baseline model (2006-2014) with the three 2091 land use scenarios (i.e. 2091-A1B, 2091-A2, and 2091-B1 scenarios), while the climate data was kept unaltered, with NLCD 2011 being the baseline. Such setups helped to isolate the contribution of land use change from climate effects. In other words, the difference between the simulated outputs from one of these three configurations and the baseline model would denote the magnitude of water quality changes caused by land use change if that particular scenario takes place in the future. All the three configurations together would produce an ensemble change extent.

It is important to clarify how the notion of low-input agriculture was implemented in the modeling for this study. Cropland in NLCD definition (http://www.mrlc.gov/nlcd11_leg.php) as well as in SWAT database specifically refers to the annual row crops such as corn and soybean. Another key point regarding the NLCD and SWAT land use definition protocol is that range/grassland, hay/pasture are all characterized as herbaceous vegetation, but the later (i.e. hay/pasture) typically refers to grasses planted for livestock fodder on a perennial cycle. However, many studies have modeled the impacts of biofuel production by assigning specific grass types (such as switchgrass and miscanthus) and associated management/nutrient operations to the entire hay/pasture land use patches (e.g. Demissie et al., 2012; Ng et al., 2010; Wu and Liu, 2012). While creating SWAT models for this study, no specific biofuel grass type was defined. Thus, it was implicit that the hay/pasture land use in all the FORE-SCE based SWAT configurations as well as in the baseline model represents perennial grass production areas.

4. RESULTS AND DISCUSSION

In this section, a general assessment of projected land use dynamics in the Skunk Creek watershed under three different scenarios is first presented relative to historical land use database. Next, performance of the calibrated SWAT model is evaluated during observed streamflow and water quality data at the watershed outlet. Finally, effects of land use change on water quality are evaluated by comparing model simulated outputs between the baseline and a future period by the end of 21st century. Overall, three levels of assessments are provided including the watershed-average percent changes in water quality indicators, their spatial patterns, as well as cases of water quality changes in specific sub-basins.

4.1 Future land use projections

The FORE-SCE land use projections for the year 2091 under one of the IPCC SRES scenarios (e.g. A2) is compared with NLCD 2011 in Figure 2. A visual comparison indicates that Skunk Creek will continue to be an agriculture-dominated watershed in future. However, it is likely that the watershed will undergo depletion of its natural grasslands, accommodating hay/pasture and the increasing trend of urbanization will also co-exist. Figure 2 also highlights the basis of FORE-SCE's land use allocation. For example, in the west-central portion of the watershed, it appears that the new extents of hay/pasture as projected by FORE-SCE are not randomly assigned; rather they are aggregated by replacing the grassland at the vicinity of the existing hay/pasture production areas. As stated earlier that grasslands in Skunk Creek depleted by 3% only in 20 years since 1992 and comprised only 6% of the study area in 2011 (Section 2), hence, it is not unlikely for the remaining grasslands to disappear by the end of 21st century (Figure 2). While extension of urban areas is possible all over the watershed, Figure 2 shows intensified urbanization around south-east which is historically a prominent growth center over

the years (city of Sioux Falls). Based on recent population movement in the state, Sioux Falls is growing quite rapidly (from 154,000 people in 2010 to 171,000 people in 2015 (US Census Bureau, 2016), and this increasing trend in population will likely to continue. FORE-SCE's predictions, as seen from Figure 2, are reasonable in capturing this trend. FORE-SCE, being informed of the past details of local land use trends and policy changes, the projections derived from therein should not be considered as simple "speculations".

A GIS analysis of NLCD 2011 and the FORE-SCE projections from 2012 to 2091 identifies the possible interchanges among different land use classes (Figure 3). The general pattern in Figure 3 is supportive of the probable land use change pattern deduced from Figure 2, showing that cropland, grassland and forest are most likely to be replaced by urban and hay/pasture areas. Reduced cropland as shown in Figure 3 does not contradict previous studies which reported increased agricultural areas in the Midwest and Great Plains states (e.g. Claassen et al., 2011; Wright and Wimberly, 2013). Since row crops and hay/pasture can be put under the broad classification of agricultural land use category, adding up 5-10% projected reduction in row crop and associated 35% increase in hay/pasture areas, ultimately results in 25-30% net increase of total agricultural land areas in the Skunk Creek watershed (Figure 3). Because water bodies and wetlands comprise only a small portion of the whole watershed area (less than 2% in NLCD 2011 and 1-4% in the future projections; not reported here), their effects on the hydrologic processes and associated water quality simulation can be considered insignificant.

Figure 3 reveals the relative influence of plausible scenarios (A1B, A2 and B1) on future land use trends. The estimated changes within the land use classes as shown in Figure 3 match well with the respective scenario driving storylines as described in IPCC-SRES (2000). For example, the A2 scenario, representing a very heterogeneous world with continuously increasing

population and slower economic/technological growth, leads to the highest extent of urbanization, grassland depletion and gradual deforestation (Figure 3). Being a sustainability-oriented scenario with a declining global population after the mid 21st century, intensive growth of clean technology and reduction in material intensity, B1 shows the least urbanization and grassland depletion, and 5-7% lesser cropland reduction than what was estimated in A1B and A2 scenarios. Least intensive emission in B1 can be attributed to a consistent 5-7% increase in forest within the study watershed throughout the century. Having the same population dynamics as in B1 scenario but with the use of both clean and fossil-intensive technology, A1B can be considered a relatively moderate scenario. Accordingly, A1B bears similar trends of future land use as in the case of A2, except with a slower degree of changes. Regardless of the scenario being employed in generating these future land use projections, the most noticeable feature of land use change is the 100% depletion of grasslands between years 2030 and 2050 (Figure 3). Moreover, the rate of change for all land use classes shows a very sharp increasing/decreasing trend during the first quarter of the century, with a possibility that the total agricultural land acreage might eventually reach a saturation limit.

As mentioned above, FORE-SCE land use projections for 2091 (Figure 4) was used in SWAT modeling for this study. Any one of the driving scenarios (A1B, A2 and B1) or its influence on land use projection does not have a greater probability over the others to actually occur in future (IPCC-SRES, 2000). Therefore, while modeling the plausible future impacts of land use change on water quality in Skunk Creek watershed, all these scenarios were considered equally possible.

4.2 Evaluation of SWAT performance

To ascertain that the calibrated model over the baseline period (2005-2014) is representative of the watersheds' hydrologic characteristics, simulated daily streamflow hydrographs were compared with the observed data at the watershed outlet (Figure 5(a)). SWAT simulations matched well with the observed, with some disagreements both during the high and low flow events. The goodness of fit scores (R^2 , NSE and PBIAS) are presented in Table 4. Daily streamflow simulation produced 22% bias even with reasonably high R^2 and NSE of nearly 0.6, indicating that the calibrated SWAT model for the Skunk Creek watershed slightly overestimated the total volume of flow at the watershed outlet. This can be attributed to uncertainties in precipitation forcing data (Fuka et al., 2014; Price et al., 2014) as well as the widely accepted issue of equifinality and sub-optimality based on the "streamflow-only" calibration scheme (Beven, 1993; Rajib et al., 2016b). Table 2 reports the optimized parameter values (best estimates) for streamflow simulation at daily time-scale.

The simulated monthly loads of sediment, total phosphorus and nitrate at the watershed outlet were compared with corresponding LOADEST-derived reference estimates (Figures 5b, c and d, respectively). Based on the evaluation criteria suggested by Moriasi et al. (2007), SWAT performance can be considered quite satisfactory with R^2 values close to 0.5 and NSE values in the range of 0.3-0.4 (except for total phosphorus) over the 10-year simulation period (Table 4). Despite the low NSE value (0.2) for total phosphorus simulation, the PBIAS value of 17.5% was within the range recommended by Moriasi et al. (2007). Similarly, PBIAS values for the entire simulation period for sediment and nitrate simulations (-4.5% and -3%, respectively) were found well below the maximum threshold reported by Moriasi et al. (2007), even with relatively high PBIAS during the validation period (Table 4). Table 3 reports the optimized parameter values (best estimates) for sediment, total phosphorus and nitrate simulation at monthly time-scale.

Overall, it appears that evaluation of model performance is influenced by the reference data (that is, "observed" data) being employed in model calibration. Considering the scope of uncertainties in processing the LOADEST water quality estimates, which are not actual observations, SWAT performance as denoted in the above description was deemed acceptable for assessing long-term environmental impacts defined in this project. Discussions in the following sections are based on the calibrated baseline SWAT model.

4.3 Land use change impact on water quality

SWAT simulations presented in this section reveal that the projected future land use change in the Skunk Creek watershed might produce significant improvement in water quality. Changes by the end of 21st century corresponding to the FORE-SCE land use in year 2091 show reduction in surface runoff volume, sediment, nitrate and total phosphorus loads regardless of the scenario (A1B, A2 or B1) being considered (Figure 6).

Agricultural operations related to annual row crops (corn/soybean) potentially impart a compaction effect and consequent reduction in the permeability of soil cover, which can be regarded as a contributing factor for surface runoff generation in agricultural landscapes (DeJong-Hughes et al., 2001; Wolkowski and Lowery, 2008). Using assumed future land use change scenarios in another agricultural watershed close to the Skunk Creek watershed, Neupane and Kumar (2015) reported 4% increase in surface runoff associated with 10% expansion of corn cultivation. As the cultivation of hay/pasture does not involve any intensive agricultural operation, reduced cropland and increased hay/pasture areas in the Skunk Creek might contribute in lowering the surface runoff volume (mmH₂O) up to 13% by the year 2091 (Figure 6). While 60-75% expansion of urban areas (Figure 3) should potentially augment surface runoff in limited

areas, the net surface runoff volume generated over the entire watershed would still be decreasing because of the much smaller proportion of urban land use (Figure 4).

Relatively denser roots of properly managed hay/pasture can control the erosion potential of the top soil (Boval and Dixon, 2012; Carlier et al., 2009; Speir, 2009). The impermeable surfaces in urban areas can also reduce (or fully eliminate) soil erosion. Wu and Liu (2012) found 4.5% reduction in sediment load for a watershed in Iowa when 10% of cornfield was converted to perennial biofuel pasture (switchgrass and miscanthus); virtually no change could be found when native grassland was replaced by switchgrass and miscanthus. This is indicative of the fact that both native grassland and hay/pasture crops ideally have the same resistance to soil erosion. Thus, the 6-14% reduction of sediment load (ton/ha) in the Skunk Creek watershed by 2091 (Figure 6) would mostly result from hay/pasture cultivation in the existing degraded and marginal croplands. Expansion of hay/pasture cultivation areas by depletion of grasslands may not contribute in this regard.

Growing perennial biofuel crops in the newly evolved hay/pasture areas would require fertilization (Wu et al., 2013, 2012). In contrast, hay/pasture production in the study watershed would involve minimal nutrient and management input compared to that being applied for row crops (corn/soybean) under present acreage (Table 1) or that might be required for biofuel. As the SWAT model for Skunk Creek watershed did not include any specific biofuel crop type to the hay/pasture HRU classes, it can be presumed that the increased proportions of hay/pasture, either converting from grasslands or occupying the currently degraded and marginal croplands, would be entirely used in livestock fodder production. The possible low-input hay/pasture production is likely to cause an overall reduction of nitrate (7.5-14%) and total phosphorus (2.5-9%) loads (kg/ha). This is comparable to the finding reported by Yang et al. (2014) which

showed substantial reduction of total nitrogen and total phosphorus load in three Canadian watersheds under hypothetical scenarios of marginal annual cropland converting into perennial hay/pasture. A similar study by Wu et al. (2013), applying FORE-SCE projections of land use change on a watershed in Iowa, showed nominal changes in nitrate load when no biofuel crop was specified on the increased hay/pasture extents; 3% increase in cropland in that particular watershed might have offset the possible improvement in water quality. However, the authors reported 2.5% increase in nitrate load by simulating biofuel crop (switchgrass) cultivation on the entire extents of newly added hay/pasture land use class. Similarly, Ng et al. (2010) showed that the potential of increased hay/pasture cultivation toward improving water quality essentially depends on several inter-related agricultural practices, including whether or not biofuel crops should be cultivated in the hay/pasture areas, the relative proportion of cropland and hay/pasture land use classes within a watershed, and the fertilizer application rate for respective biofuel crop types. Based on the above discussion, establishment of fodder production in the possible new hay/pasture areas is a viable option for "low-input" agriculture that can minimize environmental pollution and maximize economic benefits for the food and livestock industry in the Great Plains and Midwestern States.

Out of the three future scenarios employed in SWAT modeling for the Skunk Creek watershed, B1, because of its highest percentage reduction of high-input croplands (row crops), associated expansion of low-input hay/pasture areas and noticeable afforestation, should appear as an "environment-friendly" future scenario in terms of less nitrate and total phosphorus loads generated from the landscapes. Thus, SWAT simulations summarized in Figure 6 are quite reflective of the respective future scenario assumptions and storylines.

Figure 7 shows the trend of changes in surface runoff volume, sediment, nitrate and total

phosphorus loads with the reduction in cropland (row crops) areas, comprising of the modeled outputs from all the three scenarios. Figure 7 clearly indicates the direct influence of varying high-input agricultural acreage on the potential improvements in water quality. An important feature that is revealed from Figure 7 is the consistency of SWAT's water quality predictions in terms of the physical transport mechanism of phosphorus and nitrate and their relationships with surface runoff and sediment. Surface runoff being the principal driving factor to carry the eroded soil sediment from landscape to the stream, their linear relationship as shown in Figure 7 is quite obvious. Similarly, total phosphorus follows the same trend as sediment transport (Figure 7). Because of the high affinity for soil particles, phosphorus can adhere to the soil surface and become readily drivable by surface runoff. On the contrary, nitrate is highly mobile and is not adsorbed by soil particles (Neitsch et al., 2011). Thus, surface runoff sediment directly influences the transport processes of phosphorus but not of nitrate (Figure 7).

While the above discussions correspond to the watershed-scale changes, Figure 8 represents a sub-basin scale spatial evaluation. Changes calculated for the year 2091 are relative to the multi-year average values from the baseline SWAT simulations (2006-2014; NLCD 2011). Spatial maps for surface runoff, sediment, nitrate and total phosphorus only for the A2 scenario are shown here. Overall, the spatial patterns for the other two scenarios (A1B and B1) were found to be nearly the same, with differences in the magnitude of changes. A distinct upstream versus downstream gradient can be observed from Figure 8; the upstream portion of the watershed mostly indicates reduced surface runoff, sediment and nutrient loads, while these might increase in many of the downstream sub-basins.

As mentioned earlier, FORE-SCE model does not only project the amount of change in different land use categories, but also applies a 'spatial allocation' module in its algorithm which

determines where in the landscape the new land use patches will be placed. In case of a condition when cropland might expand, FORE-SCE raster data would show which cells are the new cropland areas, and as such specific crop types/managements can be assigned only to the corresponding SWAT HRUs. Declining cropland as shown in Figures 3 and 4 is the "watershed-average" trend projected by FORE-SCE. Hence, these figures do not indicate depletion of cropland all over the watershed. FORE-SCE land use projections are spatially explicit, derived from the historical change pattern as well as current policy, socio-economic/demographic trends and future emission scenarios. From these perspectives, assuming an increased cropland scenario for the whole watershed would be contradictory with the "physically-based" approach established in this study. To further ascertain correlation of the water quality indicators with specific land use change cases, four sub-basins were analyzed individually as examples (Table 5). Sub-basin A (Figure 8 and Table 5) represents the upstream part of the watershed where hay/pasture areas might increase by 10 km² based on A2-projection; accordingly, cropland and grassland might decrease by 6 km² and 4 km² respectively. Based on SWAT simulations, such changes would reduce surface runoff by 16 mm; sediment and nutrient loads would also decrease. Sub-basin B exemplifies the opposite scenario of sub-basin A where both grassland and hay/pasture areas might get converted into croplands. 10 km² increase in cropland acreage would produce more surface runoff (25 mm) in B with associated increase in sediment (28 ton/ha) and nutrient loads (23 kg/ha for nitrate and 17 kg/ha for total phosphorus). The opposite cases represented by sub-basins A and B support the findings shown in Figure 7 and validate the point that increasing the extents of low-input agriculture would reduce surface runoff, which will likely improve water quality. Sub-basin C and C' represent a different possible case where cropland, grassland, hay/pasture all would be converted into urban areas. This is a distinctive

example where even a substantial increase in surface runoff (57 mm for C) would not degrade water quality. Because of the relatively small urban areas compared to agricultural areas in this watershed, non-point source pollution from urban settings will likely not exceed pollution from agricultural lands when assessing the effects of land use change at the watershed outlet.

The discussion presented above is useful in understanding the possible spatio-temporal dynamics of future land use change in agricultural contexts and their implications for environmental quality.

5. CONCLUSIONS

Analysis of physically-based FORE-SCE land use projections indicated that grassland conversion will continue to be a typical trend of land use change in the watershed. Hay/pasture production, together with urban areas, would expand at the detriment of cultivated cropland, grassland and forest. Being a relatively environment-friendly scenario, B1, would cause afforestation, giving way to less grassland depletion and urbanization. The estimated rates of change for all land use classes, regardless of the scenario, suggested an abrupt change in all land uses during the first quarter of the century, when entirety of grassland in the watershed will likely be completely depleted and moved to other land uses. The rate of land use change would be slower for the remainder of the century.

While land use change is generally linked to adverse environmental impacts, the simulation results showed that the future land use change in Skunk Creek watershed may lead to water quality improvement with reduction in surface runoff, sediment, nitrate and total phosphorus loadings for all three scenarios examined. This improvement in water quality is the reflection of the nature of land use conversion and associated activities, which are characterized

in this case by gain in low-input agriculture with hay/pasture production. This study provides land use planners a view to anticipate the effects of any proposed land transformation on water quality. Integrating hay/pasture (i.e. perennial grass) production into existing farming practices may be a sustainable approach worth considering in agricultural land use planning.

The major theme of this study was to establish the use physically-derived future land use projections as a more authentic approach for hydrology and water quality modeling. However, FORE-SCE database is basically model-generated, and as such it has its own sources of limitations. When clipping for a small watershed from a large-scale database such as FORE-SCE, it would be useful to analyze potential drivers of land use change at the scale at which they are analyzed, to assess the degree at which large-scale drivers used at national scale as well as the local drivers are relevant for the specific domain of interest. The hydrology-water quality modeling results presented in this study are influenced by agricultural management practices, including those related to type, rate and timing of nutrient input, as well as the LOADEST-derived estimates of water quality indicators used for model calibration. Specific management practices such as cattle grazing and manure application were not included in the SWAT model due to the lack of reliable field data. Since the model configurations are designed in a way that the future impacts are analyzed with respect to a current baseline and the results were shown as percent changes, modeling uncertainties may not have critically affected the overall perspective being presented. Overall, this study imparts insightful information that provides a base for additional research to foster understanding of the potential influence of future land use change scenarios and their trajectories on water quality in the state and the Upper Midwest region as a whole.

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Table 1. Agriculture management operations as used for water quality modeling in Skunk Creek watershed, South Dakota

Corn				
Tillage	No till			
Planting	May 15			
Chemical Input	Type	No. of Application(s)	Timing	Rate/Crop Year (kg/ha)
N-Fertilization	Anhydrous Ammonia	1.3	May 23 (1.0) July 1 (0.3)	85
P-Fertilization	Monoammonium Phosphate	1.0	May 25	40
Herbicide	Atrazine	1.0	May 2	0.7
Harvest and Kill	October 10			
Soybean				
Tillage	No till			
Planting	May 20			
ChemicalInput	Type	No. of Application(s)	Timing	Rate/Crop Year (kg/ha)
P-Fertilization	Monoammonium Phosphate	1.0	May 10	40
Herbicide	Glyphosphate	1.4	May 15 (0.8) June 15 (0.3) July 5 (0.3)	1.04
Harvest and Kill	October 5			

Table 2. Parameters used in SWAT model calibration for streamflow

No.	Parameter ^a	Definition ^b	Initial range	Best estimate
1	v_ALPHA_BF	Baseflow recession constant (days)	0.01 - 1	0.30
2	v_CANMX	Maximum canopy storage (mm H ₂ O)	0.01 - 25	20.12
3	v_SURLAG	Surface runoff lag coefficient (days)	0.05 - 24	2.24
4	v_CH_K2	Main channel hydraulic conductivity (mm/hr)	5 - 100	40.25
5	v_CH_N2	Main channel Manning's <i>n</i>	0.01 - 0.15	0.08
6	r_CN2	Curve number (moisture condition II)	-20 - 20	0.11
7	v_EPCO	Plant uptake compensation factor	0 - 1	0.81
8	v_ESCO	Soil evaporation compensation factor	0 - 1	0.99
9	a_GW_DELAY	Groundwater delay (days)	-10 - 10	-0.79
10	v_GW_REVAP	Groundwater "revap" coefficient	0.01 - 0.2	0.02
11	v_GWQMN	Threshold depth for return flow (mm H ₂ O)	0.01 - 5000	3537
12	v_REVAPMN	Re-evaporation threshold (mm H ₂ O)	0.01 - 500	325
13	r_SOL_K	Soil saturated hydraulic conductivity (mm/hr)	-15 - 15	-14.73
14	r_SOL_AWC	Available soil water capacity (mm/mm)	-15 - 15	-0.08
15	v_SFTMP	Snowfall temperature (°C)	0 - 5	1.05
16	v_SMFMN	Melt factor for snow on December 21 (mm H ₂ O/°C-day)	0 - 10	5.19
17	v_SMFMX	Melt factor for snow on June 21 (mm H ₂ O/°C-day)	0 - 10	5.39
18	v_SMTMP	Snow melt base temperature (°C)	-2 - 5	-1.36
19	v_TIMP	Snow pack temperature lag factor	0 - 1	0.55

^aThe indices shown with the parameter names denote the type of change to be applied over the existing parameter value: 'v_' means the original value is to be replaced by a value from the range, 'a_' means a value from the range is added to the original value, 'r_' means the original value is multiplied by the adjustment factor (1+ given value within the range).

^b Source: Neitsch et al. (2011)

Table 3. Parameters used in SWAT model calibration for sediment, total phosphorus and nitrate

No.	Parameter ^a	Definition ^b	Initial range	Best estimate
Sediment				
1	v_SPEXP	Exponent factor for channel re-entrainment	1 - 2	1.05
2	v_SPCON	Maximum channel re-entrainment factor	0.0001 - 0.01	0.008
3	v_PRF	Peak rate adjustment factor in main channel	0.01 - 1	0.06
4	v_CH_COV2	Channel cover factor	0.01 - 0.5	0.08
5	v_USLE_P	Support practice factor	0.001 - 1	0.11
Total Phosphorus				
6	v_PSP	Phosphorus availability index	0.01 to 0.7	0.35
7	v_PHOSKD	Soil partitioning coefficient (m ³ /Mg)	100 to 200	158
8	v_PPERCO	Phosphorus percolation coefficient (m ³ /Mg)	10 to 17.5	12.54
9	v_ERORGP	Phosphorus enrichment ratio	0.001 to 5	0.11
10	v_BC4	Rate constant for mineralization of organic P to dissolved P in reach (day ⁻¹)	0.01 to 0.7	0.27
11	v_RS5	Organic P settling rate in reach (day ⁻¹)	0.001 to 0.1	0.04
Nitrate				
12	v_RCN	Nitrogen in rain (mg/L)	0.001 to 10	4.77
13	v_NPERCO	Nitrate percolation coefficient	0.01 to 1	0.03
14	v_ANION_EXCL	Fraction of porosity from which anions are excluded	0.01 to 1	0.39
15	v_BC1	Rate constant for biological oxidation of NH ₄ to NO ₂ in reach (day ⁻¹)	0.1 to 1	0.98
16	v_BC2	Rate constant for biological oxidation of NO ₂ to NO ₃ in reach (day ⁻¹)	0.2 to 2	1.39
17	v_BC3	Rate constant for hydrolysis of N to NH ₄ (day ⁻¹)	0.2 to 0.4	0.28
18	v_RS4	Organic N settling rate in reach (day ⁻¹)	0.001 to 0.1	0.05
19	v_ERORGN	Organic N enrichment ratio for sediment	0 to 5	0.89

^{a, b}Table 1 provides necessary explanations.

Table 4. Goodness-of-fit scores of SWAT model evaluation^a

	Statistics	Calibration	Validation	Entire study period
Streamflow	R ²	0.57	0.76	0.61
	NSE	0.57	0.51	0.58
	PBIAS	7.9	21.8	22.0
Sediment	R ²	0.45	0.91	0.47
	NSE	0.31	0.31	0.32
	PBIAS	-7.3	47	-4.50
Total Phosphorus	R ²	0.5	0.91	0.50
	NSE	0.26	0.21	0.20
	PBIAS	15.4	25.5	17.5
Nitrate	R ²	0.42	0.44	0.42
	NSE	0.41	0.38	0.42
	PBIAS	-5.6	43	-3

^aStatistics for streamflow is based on daily simulation, while statistics for water quality are in monthly time-scale.

Table 5. Changes in land use and water quality in specific sub-basins^{a, b}

Land use change (2091-2011, km ²)				
Sub-basin ^a	Cropland	Hay/Pasture	Range/Grassland	Urban
A	-6	+10 ^b	-4	Nil
B	+10 ^b	-5	-4.5	+0.5
C	-10	-3	-0.5	+13.5 ^b
C'	-2	-1.5	-2.5	+6 ^b
Change in runoff and water quality (2091-2011)				
Sub-basin ^a	Surface runoff (mm)	Sediment (ton/ha)	Nitrate (kg/ha)	Total Phosphorus (kg/ha)
A	-16	-26	-27	-23
B	+25	+28	+23	+17
C	+57	-51	-26	-23
C'	+14	-11	-3	-11

^aSee Figure 8 for sub-basin ID.^bMajor change in land use.

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