Climate change impacts on streamflow, water quality, and best management practices for the shell and logan creek watersheds in Nebraska

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Abstract: Improvements in the management of water, sediment, and nutrients under future climatic conditions are needed to ensure increased crop and livestock production to meet greater global needs and the future availability of water for competing demands and protection against adverse water quality impairments. This study determined the impacts of future climate change scenarios on streamflow, water quality, and best management practices (BMPs) for two watersheds in Nebraska, USA. The Soil and Water Assessment Tool (SWAT) was employed to simulate streamflow, sediment, total nitrogen (N) and total phosphorus (P) from the Shell Creek Watershed near Columbus, Nebraska and the Logan Creek Watershed near Sioux City, Iowa. Available streamflow and water quality records for the two watersheds were used to calibrate model parameters that govern streamflow, sediment, and nutrient responses in SWAT. For each watershed, precipitation, air temperature, and CO_2 concentrations were input to SWAT for four climatic conditions: a baseline condition for the 1980 to 2000 period and the SRES A2, A1B, and B1 climate scenarios for a future period from 2040 to 2059. Findings from this study suggest that under the three future climate change scenarios, sediment losses are expected to be about 1.2 to 1.5 times greater than the baseline condition for Shell Creek and 2 to 2.5 times greater for Logan Creek; total N losses are expected to be about 1.2 to 1.4 times greater for Shell Creek and 1.7 to 1.9 times greater for Logan Creek. Relative to the baseline, total P losses under the future climate scenarios are projected to be about the same for Shell Creek and 1.5 to 1.7 times greater for Logan Creek. Findings from this study also suggest that future projected increases in both precipitation and CO_2 concentration account for net increases in streamflow, but in different ways on each watershed.

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1 Introduction

The impacts of climate changes on water resources during the past few decades have caused considerable concern throughout the United States. These changes have resulted in streamflow frequency, peak discharge, flow volume, and baseflow shifts in addition to changes in sediment, nutrient, pesticide, and other pollutant loadings. Changes occurring in water resources as a result of climate change include both subtle effects, such as gradual increases or decreases in annual streamflow and seasonal shifts in flow frequency, and the increasing occurrence of dramatic events, such as floods and droughts. In the Upper Mid-West portion of the country, Tomer and Schilling[1] report that not only have increases in precipitation led to subsequent increases in stream

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discharge, but decreasing evaporative demand may also be driving increases in streamflow. Changes in climate, due to increases in greenhouse gas concentrations coupled with changes in air temperature, precipitation, relative humidity and other climate variables are projected in the coming decades to have profound impacts on stream systems across the nation, affecting channel morphology, aquatic life and biodiversity, regional water supplies, and the quality of drinking water[2-4]. The effects of climate on the nation's water storage capabilities and hydrologic functions will have significant implications for water management and planning as variability in natural processes increases[5].

Within the U. S. Heartland Region of Iowa, Nebraska, Kansas and Missouri, climate change has led to increases in average temperatures, with the largest increases occurring in the winter months. Relatively cold days are becoming less frequent and relatively hot days more frequent. In this region of the United States, temperatures are expected to continue to increase over this century, with larger changes expected under scenarios of higher heat-trapping emissions as compared to lower ones[6]. During this century, northern areas of the Heartland are expected to experience increasingly wetter winters and springs. Projected changes also include more frequent extreme events such as droughts, heat waves, and heavy rainfall. Several previous studies have been conducted to assess the impact of future climate change on the hydrology of the Upper Mississippi River and Missouri River Basins,[7-10] both of which cover portions of the Heartland Region. However, studies focused specifically on evaluating future climate change impacts on both streamflow and water quality in the region are very limited, with the recent work of Woznicki et al.[11,12] for the Tuttle Creek Watershed in Nebraska and Kansas being one of the few examples. Although impacts to streamflow frequency, peak discharge, flow volume, and baseflow in addition to changes in sediment, nutrient, pesticide, and other pollutant loadings are expected to occur throughout the region, these projected impacts are not well documented over a range of spatial and temporal scales. A host of factors, including non-linear relationships, multiple causation, lag effects, and lack of mechanistic understanding, complicate our understanding of the cause and effect relationships between climate change and hydrologic/water quality response. Moreover, distinguishing the effects of land use changes from concurrent climate variability poses a particular challenge[13,14].

Many uncertainties exist today in the assessment of possible impacts of future changes in climate on hydrologic and water quality responses at watershed or basin scales. This is because water availability is highly variable and not well understood on a case-by-case basis for individual watersheds under future climate scenarios[2,15,16]. In some cases, substantial changes in the magnitude and frequency of storm events will have profound effects on the detachment and transport of pollutants from the landscape, thus impacting downstream receiving waters such as streams and lakes. Considerable uncertainty also exists regarding the effectiveness of Best Management Practice (BMP) implementation on pollutant load reduction under anticipated future changes in climate. As climate changes, the magnitude of nonpoint source (NPS) pollutants may be more extreme within a watershed and current BMPs may not be appropriate to treat these conditions[17].

World population is expected to increase by 40 percent by the year 2050, causing the demand for food to nearly double. Improvements in the management of water, sediment, and nutrients under future changes in climatic conditions are needed to ensure increased crop and livestock production to meet greater global needs and to ensure the future availability of water for competing demands and protection against adverse water quality impairments. To assess future climate change impacts on streamflow and water quality in the Heartland Region would represent an important step forward for strategic watershed planning and future environmental protection. Such an assessment would not only be helpful in providing a better understanding of how changes in future water resources will affect regional livelihood, the environment, wildlife, and human health, but would also be helpful in determining how future climate change will impact the effectiveness of best management practice implementation on water quantity and water quality at watershed scales. Recent advances in computing capability and Geographical Information Systems (GIS) have led to the development of sophisticated watershed-scale models that incorporate climatic, soils, topographic, and land use characteristics and are capable of addressing a host of issues related to water resources and water quality. Many of the models developed consist of elaborate algorithms that describe erosion and sedimentation, nutrient cycling, and pesticide fate and transport. They can therefore simulate the movement and transformation of a number of water quality constituents from point, non-point, and channel sources within a watershed at various spatial and temporal scales. They are also capable of estimating the impacts of climate changes on water quantity and water quality, and can evaluate pollutant loading reductions due to BMP implementation.

The Watershed Analysis Risk Management Framework (WARMF) model,[18,19] Hydrologic Simulation Program-Fortran (HSPF) model,[20-21] and the Soil and Water Assessment Tool (SWAT) model[22,23] represent physically-based, continuous-simulation hydrologic models that are capable of quantifying hydrologic and water quality responses in large, complex watersheds. For agricultural watersheds, SWAT is especially well suited for assessing the impact of future climate change scenarios on streamflow and water quality constituents as well as accurately evaluating BMPs to assess pollutant load reductions, as demonstrated in several previous studies[7,24-30]. Although SWAT is generally applied to large river basins, it has also been validated at both river basin and small watershed scales in terms of annual water and sediment yield[23].

To better understand the impacts of future climate scenarios on streamflow and water quality losses at the watershed scale, SWAT was employed to conduct an investigation at two locations in Nebraska (Figure 1). SWAT was used to simulate the streamflow, sediment, total nitrogen (N) and total phosphorus (P) response from the Shell Creek Watershed near Columbus, NE and the Logan Creek Watershed near Sioux City, IA. The first objective of the study was to compare the streamflow and pollutant response from the two watersheds under existing climatic conditions as well as three future climate scenarios. The second objective was to determine how future climate change scenarios will impact the effectiveness of a suite of best management practices on streamflow and water quality constituents at the two respective locations.

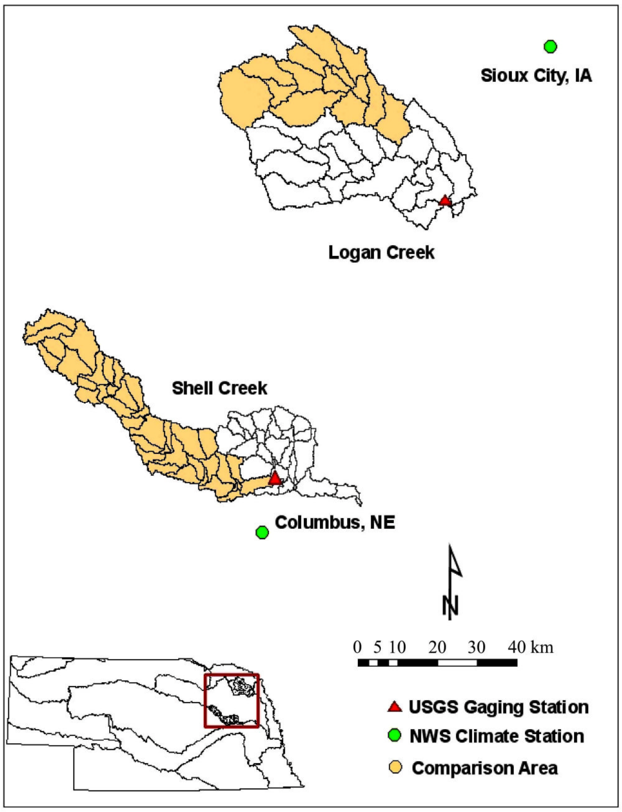


Figure 1 Location of the Columbus, NE and Sioux City, IA climate stations and the Shell and Logan Creek Watersheds

2 Materials and methods

2.1 Shell Creek Watershed

The Shell Creek Watershed is located just north of Columbus, Nebraska and drains an area of 1214 km^2. It is a tributary of the Platte River, one of the major rivers in Nebraska, and is located within the Lower Platte North Natural Resources District. The watershed, inhabited by nearly 1700 landowner/operators, is primarily an agricultural area that includes steep-sloped pastures, rolling pivot-irrigated hills, and gravity-irrigated flood plains. Average annual precipitation and runoff for the watershed are about 735 mm/year and 50 mm/year, respectively. Erosion and sedimentation, nitrogen, and phosphorus are major water quality issues, as well as degradation from other non-point sources and loss of aquatic and wildlife habitat. Extensive cultivation of corn, soybeans, and other crops contributes to substantial pollutant losses from the landscape. The presence of cattle and swine feedlot operations within the Shell Creek drainage also contributes to pollutant loadings. Land cover types on the watershed include corn (48%), soybean (28%), range (19%), alfalfa (3%), and misc. (2%). Most soils on the watershed are deep, silty loams and silty clay loams; soil series include the Nora (49%), Hobbs (27%), Belfore (19%), Moody (4%), and Gibbon (1%).

2.2 Logan Creek Watershed

The Logan Creek Watershed is located southwest of Sioux City, Iowa and drains an area of 1990 km^2 in northeast Nebraska. It is also a tributary of the Platte River and is located within the Lower Elkhorn Natural Resources District. Average annual precipitation and runoff for the watershed are about 660 mm/year and 65 mm/year, respectively. Like the Shell Creek Watershed, erosion and sedimentation, nitrogen, and phosphorus are notable water quality issues. Animal feedlot operations within the drainage also contribute to pollutant loadings. Land cover types on the watershed include corn (45%), soybean (39%), range (14%), and alfalfa (2%). Most soils on the watershed are deep, silty loams and silty clay loams; soil series include the Nora (54%), Moody (24%), and Kennebec (22%).

2.3 SWAT model

SWAT was originally developed by the U.S. Department of Agriculture (USDA)-Agricultural Research Service (ARS) to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large ungaged basins[22,31,32]. Model simulations performed in SWAT are usually computed on a daily time step. For this study the USDA-Natural Resource Conservation Service (NRCS) runoff curve number (CN2) method was used to estimate surface runoff from daily precipitation[33] and evapotranspiration was computed using the Penman-Monteith[34] method. Model documentation is well formulated for SWAT, with considerable detail that is provided regarding model structure, algorithms, data input, and viewing of test results. SWAT version 2009 was used for this study, which is described in detail in the theoretical documentation manual[35].

SWAT is a distributed parameter model that partitions a watershed into a number of sub-basins. Each sub-basin delineated within the model is simulated as a homogeneous area in terms of climatic conditions, but with additional subdivisions within each sub-basin to represent various soils and land use types. Each of these subdivisions is referred to as a Hydrologic Response Unit (HRU) and is assumed to be spatially uniform in terms of soils, land use, topographic and climatic data.

On the landscape, erosion and sediment yield are estimated for each HRU in SWAT using the Modified Universal Soil Loss Equation (MUSLE)[36], an enhancement of the original USLE equation[37]. SWAT comprehensively models transfers and internal cycling of the major forms of nitrogen and phosphorus. The model monitors two pools of inorganic and three pools of organic forms of nitrogen as well as three pools of inorganic and three pools of organic forms of phosphorus. SWAT also incorporates in-stream nutrient dynamics using kinetic routines from the in-stream water quality model referred to as QUAL2E[38].

2.4 Watershed delineation, targeting of BMPs, and response comparison

Elevation, land use, and soil characteristics was obtained from GIS data layers for the Shell and Logan Creek Watersheds. The elevation layer was developed from the USGS National Elevation Dataset (NED)[39] at a 30 m resolution. The land use layer was obtained from the 2001 National Land Cover Dataset (NLCD)[40] at a 30 m resolution; the soil layer was obtained from the USDA-NRCS STATSGO database[41]. The ArcSWAT 2.3.4 interface[42] was used to delineate the Shell Creek Watershed into 70 subbasins and 3422 hydrologic response units (HRUs); the Logan Creek Watershed was delineated into 35 subbasins and 1235 HRUs. Crop management schedules, commercial fertilizer application rates, and manure obtained from swine feeding operations were input into the model for corn and soybeans based on professional judgment and estimate by USDA-NRCS personnel (Table 1). For HRUs delineated as irrigated corn and soybean, the auto irrigation scheme in SWAT was used to periodically irrigate crops during the growing season. A deep aquifer with an unlimited supply of water was assumed to be the source of irrigation for crops grown on each of the watersheds.

Table 1 Conventional tillage operations schedule for soybean and corn*

Crop	Date	Operation	Application rate (kg/ha)
Soybean	April 10th	tandem disk tillage	
	May 1st	pesticide application	1
	May 10th	plant	
	September 20th	harvest and kill	
	October 15th	swine manure application	50
	November 15th	phosphorus application	15
Corn	April 10th	tandem disk tillage	
	April 28th	plant	
	May 1st	pesticide application	1
	October 18th	harvest and kill	
	October 25th	swine manure application	50
	November 1st	anhydrous ammonia	90
	November 15th	phosphorus application	15

Note: *Operations schedule based on personnel communication with Nebraska NRCS personnel, Dec. 2009.

A targeting approach was employed in this study to evaluate the impact of BMP implementation on the reduction of sediment, total N, and total P constituent loadings for the two watersheds. Targeting criteria were specified a priori based on individual HRU slope steepness and USLE soil erodibility K factors. Within each watershed, BMPs were placed on all cropland HRUs with slope steepness $\geq$6% and USLE soil erodibility K factor $\geq$0.32. Based on these criteria, BMPs were placed on about 29% and 24% of the drainage areas for the Shell and Logan Creek Watersheds, respectively. Percent changes in streamflow, sediment, total N, and total P as a result of BMP implementation simulated by SWAT were compared at reach 63 in the Shell Creek Watershed and reach 15 in the Logan Creek Watershed. Five types of BMPs were implemented in SWAT to assess load reductions. These BMP types were arbitrarily chosen and do not necessarily reflect possible options that might be chosen by local producers at either location. BMP types included 1) conversion of crops to switchgrass, 2) conversion of crops to continuous pasture, 3) terraces 4) an 11 meter buffer strip, and 5) no-till. The assumption was made that cropland converted to pasture would be void of any management practices, while land converted to switchgrass would be planted and harvested each year. It was further assumed that all buffer strips and terraces that were implemented as BMPs were assumed to be fully functional and continuously maintained. A description of the BMP scenarios used in this study is presented in Table 2.

Table 2 A description of the best management practice scenarios for the Shell and Logan Creek Watersheds

Scenario	Method of simulation practice
Pasture	Changed crop type to pasture and reduced curve number in .mtg file
Switchgrass	Changed crop type to switchgrass and reduced curve number in .mtg file
Terraces	Reduced curve number and USLE P factor in .mtg file; reduced slope length in .hru file
No till	Reduced curve number and changed tillage code in .mtg file; reduced USLE C factor in crop code
11 meter buffer	Set width of field filter strip length to 11 m in .mtg file (FILTERW parameter)*

Note: *Although not employed in this study, an improved method for simulating filter strip impacts is available in SWAT2009[61].

The distributed approach to modeling in SWAT allows simulation results to be evaluated for every subbasin and reach delineated within a given project. To facilitate the comparison of hydrologic/water quality response and BMP impacts between the Shell and Logan Creek watersheds, output variables were evaluated at locations within each watershed such that the respective contributing drainage areas were nearly the same. Reach 63, with a drainage area of 781 km^2, was selected for the Shell Creek Watershed, while reach 15, with a drainage area of 785 km^2, was selected for the Logan Creek Watershed (Figure 1). Selected variables for the two drainage areas are compared in Table 3.

2.5 Observed data for model calibration

Observed climatic, streamflow, and water quality records were used to calibrate parameters that govern hydrologic and water quality processes in SWAT. Precipitation and air temperature data were obtained from the National Climate Data Center[43] climate stations at Columbus, NE and Sioux City, IA for the Shell and Logan Creek Watersheds, respectively. Streamflow and

Table 3 Comparison of selected watershed characteristics for the Shell Creek and Logan Creek Watersheds

Category		Shell creek		Logan creek
Drainage area/km^2		780.9		784.7
Change in elevation/m		199		159
Length of main channel/m		86.3		52.0
% of watershed as BMP		29.4		23.9
Land cover type/%	Alfalfa	3.0%	Alfalfa	1.9%
	Corn	23.5%	Corn	32.0%
	Irrigated corn	25.0%	Irrigated corn	15.0%
	Soybean	13.6%	Soybean	25.7%
	Irrigated soybean	16.0%	Irrigated soybean	11.0%
	Forest	1.1%	Forest	0.0%
	Range	16.8%	Range	14.2%
	Misc	1.0%	Misc	0.3%
Soil type/%	Belfore	9.6%		
	Hobbs	17.4%	Kennebec	16.6%
	Moody	2.7%	Moody	24.7%
	Nora	70.4%	Nora	58.7%

water quality data[44] were obtained for USGS gaging station 0679550, referred to as Shell Creek near Columbus, NE and for USGS gaging station 06799450, referred to as Logan Creek at Pender, NE. Data of a three year period from 1992 to 1994 were used to calibrate parameters governing hydrologic, sediment, nitrogen, and phosphorus response for the Shell Creek Watershed. Based on long term precipitation data, this period of record is about 16% wetter than average and was selected for calibration because it is the most sampled period from the available water quality record. Data of a five year period from 1971 to 1975 were used to calibrate streamflow, sediment, and nutrients for Logan Creek. This period of record is about 2% dryer than average and was selected for calibration because of the range of streamflow data and the availability of water quality records. Measured streamflow data for the period of record from 1998 to 2000 on Shell Creek and from 1984 to 1986 at Logan Creek were selected for streamflow validation. Parameters governing the streamflow response in the model were initially calibrated using the automated calibration procedure within the SWAT model framework. Manual adjustments were then made to fine tune the hydrologic calibration at the monthly time scale. SWAT was calibrated first on Shell Creek, and model parameters governing snow accumulation and melt on that watershed were assumed to be valid on Logan Creek. Parameters governing sediment, nitrogen, and phosphorus were sequentially calibrated on a monthly basis. Default and calibrated parameter values for the two watersheds are presented in Table 4.

Table 4 A listing of default and calibrated parameter values in SWAT for the Shell Creek and Logan Creek Watersheds

Category	Parameter	Description	Default Value	Calibrated value for shell creek near columbus, NE	Calibrated value for logan creek near Sioux City, IA
Basin	SURLAG	Surface runoff lag time	4	3.01	1.09
Snow	SMFMX	Melt factor for snow on June 21	4.5	4.35	4.35
	SMFMN	Melt factor for snow on Dec. 21	4.5	7.07	7.07
	SFTMP	Snowfall temperature	1	-2.29	-2.29
	SMTMP	Snowmelt base temperature	0.5	0.224	0.224
	TIMP	Snow pack temperature lag factor	1	0.381	0.381
Channel	CH_K2	channel hydraulic conductivity	0	122.0	62.8
	CH_N	Manning's n for channel reaches	0.025	0.03	0.041
Surface	ESCO	Soil evaporation compensation factor	0.95	0.80	1.00
	SOL_AWC	Available soil water capacity	0%	0%	-10%
Subsurface	ALPHA_BF	Baseflow recession constant	0.048	0.368	0.415
	GWQMN	Minimum threshold depth for return flow	0	0	0
	GW_REVAP	Ground water "revap" coefficient	0.02	0.021	0.024
	REVAPMN	Minimum threshold depth for "revap"	1	352	352
	RCHRG_DP	Deep aquifer percolation fraction	0.05	0.035	0.001
	GW_DELAY	Ground water delay	31	92	92

Category	Parameter	Description	Default Value	Calibrated value for shell creek near columbus, NE	Calibrated value for logan creek near Sioux City, IA
Sediment	SPCON	Coefficient for channel sediment transport	0.0001	0.020	0.016
	SPEXP	Exponent for channel sediment transport	1	2	2
	CH_EROD	Channel erodibility factor	0	0.22	0.22
	CH_COV	Channel vegetative cover factor	0	0.45	0.3
Nutrients	CMN	Rate factor for humus mineralization	0.0003	0.0003	0.0003
	N_UPDIS	Nitrogen uptake distribution factor	20	20	20
	P_UPDIS	Phosphorus uptake distribution factor	20	20	20
	NPERCO	Nitrogen percolation coefficient	0.2	0.01	0.01
	PPERCO	Phosphorus percolation coefficient	10	0.01	1
	PHOSKD	Phosphorus soil partitioning coefficient	175	175	175
	PSP	Phosphorus sorption coefficient	0.4	0.8	0.4
	RSDCO	Residue decomposition coefficient	0.05	0.05	0.05

2.6 Climate input data

Many types of uncertainty relate to the use of climate change models that are used to provide climatic input data, such as temperature and precipitation, for streamflow simulation models such as SWAT. Shrestha et al.[45] employed SWAT to assess the impact of climate change scenarios on streamflow response for the Lake Winnipeg Watershed in Central Canada. They reported substantial variability in mean annual precipitation that was input to the model for three regional climate models (RCMs) used in their study. The three RCM datasets used in the scenario simulations exhibited different spatial and temporal variability, which led to significant differences in the runoff simulations for two catchments. Shrestha et al.[45] reported that such uncertainties in modeling future hydrologic regimes using single RCM forcings reinforce the need to use an ensemble approach that relies on multiple RCMs, and provides a range of possible future changes. In a similar study, Zhang et al.[46] used SWAT to perform an uncertainty assessment of climate change impacts on the hydrology of small prairie watersheds in southern-central Saskatchewan, Canada. The two RCMs employed in their study showed significant discrepancies in simulating both the magnitude and timing of precipitation for future climatic conditions. They further reported that uncertainties in integrated downscaling were primarily derived from the choice of RCM, and were amplified through the incorporation of different weather generators.

The baseline climatic condition was obtained from National Weather Service observed data at Columbus, NE and Sioux City, IA for the 1980 to 2000 period of record. The future climate change scenarios were obtained from the World Climate Research Program's (WCRP) Coupled Model Intercomparison Project phase 3 (CMIP3) multi-model dataset, which were also used in the IPCC AR4[47]. The monthly temperature and precipitation data were downscaled as described by Maurer et al.[48] using the bias-correction/spatial downscaling method to 0.125° grids (approximately 10 km). The statistically-downscaled present-day control simulations and future climate change projections from 16 fully coupled climate models covering the contiguous United States were employed for the period from 1950 to 2099. These 16 climate models, a brief indication of their origin (with only the first institute shown in the case of multiple institutions), and the number of realizations available for each climate change scenarios are presented in Table 5. These climate models were chosen because each one has been run for the three Special Report on Emissions Scenarios (SRES), A2, A1B, and B1 that were employed in this study[48,49]. The statistical downscaled monthly temperature and precipitation from models with multiple realizations (model runs) were first averaged, and then the ensemble of the 16 models were computed by equal weighting of the 16 models. The A2, A1B, and B1 climatic conditions represent a range of future economic and energy demand scenarios. The A2 climate scenario represents a world with a continuously increasing global population, nations that are self-reliant in terms of development, and technological changes and improvements that are relatively fragmented in comparison to other SRES scenarios. The B1 climate scenario represents a world with a population that reaches 9 billion in 2050 and then gradually declines; world-wide economic development is more integrated and ecologically friendly. The B1 scenario represents a world that emphasizes the implementation of clean and resource efficient technologies and global solutions to societal and environmental stability. The A1B climate scenario represents a future world with rapid economic growth and reliance upon multiple energy sources[50].

Table 5 A listing of the climate models used in this study, a brief indication of their origin, and the number of realizations available for each climate change scenarios

Model name	Origin	Scenario		
		A2	A1b	B1
bccr_bcm2_0	Bjerknes Centre Clim. Res., Bergen, Norway	1	1	1
cccma_cgcm3_1	Canadian Centre, Victoria, B.C., Canada	5	5	5
cnrm_cm3	Meteo-France, Toulouse, France	1	1	1
csiro_mk3_0	CSIRO Atmos. Res., Melbourne, Australia	1	1	1
gfdl_cm2_0	Geophys. Fluid Dyn. Lab, Princeton, NJ	1	1	1
gfdl_cm2_1	Geophys. Fluid Dyn. Lab, Princeton, NJ	1	1	1
giss_model_e_r	NASA/Goddard Inst. Space Studies, NY	1	2	1
inmcm3_0	Inst. Num. Mathematics, Moscow, Russia	1	1	1
ipsl_cm4	Inst. Pierre Simon Laplace, Paris, France	1	1	1
miroc3_2_medres	Center Climate Sys. Res., Tokyo, Japan	3	3	3
miub_echo_g	German Meteor. Inst. U. Bonn, Bonn, Germany	3	3	3
mpi_echam5	German Max Planck Inst. Meteor., Hamburg, Germany	3	3	3
mri_cgcm2_3_2a	Meteor. Res. Inst., Tsukuba, Ibaraki, Japan	5	5	5
ncar_pcm	Nat. Center Atmos. Res., Boulder, CO	4	4	2
ncar_ccsm3_0	Nat. Center Atmos. Res., Boulder, CO	4	6	7
ukmo_hadCM3	UK Met Office, Exeter, Devon, UK	1	1	1

Downscaling of temperature and precipitation for this study was performed as follows. The statistically-downscaled temperature and precipitation during 2040-2059 on the grid point which is closest to Columbus (or Sioux City) were chosen first. The temperature and precipitation for the two sites from models with multiple realizations (model runs) were first averaged, and then the ensemble (average) of the 16-model projections was computed by equal weighting of the 16 models on each SRES scenario. The ensemble of the model projections was used because the ensemble of model outputs made by all the available climate models is often the best determinant for simulating mean global and regional climates[51-53]. Shrestha et al.[45] also reported that an ensemble of multiple climate models output is needed to assess the impact of climate scenarios on streamflow responses for a watershed in Central Canada using SWAT. Additionally, because the SWAT model uses daily input, the monthly outputs of future climate at Columbus (or Sioux City) were used in the stochastic weather generator, LARS-WG Version 5.0[54], to generate the weather variables at a daily timescale for both sites during 2040-2059. In this process, the LARS-WG first calculated the empirical and/or semi-empirical statistical mean and distributions of the observed daily weather conditions (e.g., precipitation and temperature) using the data of 1980-2000 in Columbus (or Sioux City). Then, the *monthly* future climate scenarios derived during 2040-2059 were used in LARS-WG to generate daily weather data for the future climate at Columbus (or Sioux City), following the same procedures as employed by Weiss et al.[55]. In order to describe the daily weather conditions for each SRES scenario, 52 years of daily meteorological data were generated, using an initial random seed for the weather generator and a two year warm-up period for model simulations. The 50 year simulation period was assumed to be representative of daily weather conditions expected in the future. Average annual generated versus observed baseline temperature data for the 1980-2000 period showed good agreement for both the Columbus and Sioux City stations. However, average annual generated precipitation data for the baseline period was on average 64 mm and 78 mm higher than the observed record for the Columbus and Sioux City stations, respectively. To better represent the baseline and projected future climate precipitation signals, the average annual generated precipitation files for the baseline, A2, A1B, and B1 scenarios were adjusted downward on a daily basis by these corresponding amounts for the two climate stations.

The stochastically generated daily data during 2040-2059 were in turn used to drive the SWAT model for each of the three future climate scenarios. For model simulations performed in this study, the concentration of atmospheric carbon dioxide (CO_2) was assumed to be constant for each climatic condition. CO_2 input values to SWAT were assumed to be 330, 525, 525, and 475 ppm (1 ppm = μmol/mol) for the baseline, A2, A1B, and B1 scenarios, respectively.

3 Results

3.1 Model calibration and validation

Results of the model simulations during the calibration periods were evaluated based on the monthly values of percent bias (PBIAS) and Nash-Sutcliffe[56] coefficient of efficiency (NSE) (Table 6). Based on suggested guidelines by Moriasi et al.[57], simulated streamflow, sediment, total N, and total P for the Shell Creek Watershed were all considered very good at the monthly time scale. For the Logan Creek Watershed, simulation results for streamflow, sediment, and total P were considered very good, and total N was considered good. For the validation data sets, streamflow simulation results were considered very good for the Shell Creek Watershed and satisfactory for the Logan Creek Watershed (Table 6). Based on computed values of PBIAS, the average tendency of the simulated streamflows for the calibration data sets was within ±5% of the observed flows. Average tendencies of the simulated sediment, total N, and total P loads for the calibration data sets were within ±10%, ±15%, and ±20% of the observed loads, respectively. Computed values of NSE for the calibration periods suggest that in most cases, SWAT did a good job replicating monthly variations in the observed streamflow and water quality constituents. Comparison of the four monthly measured versus simulated output variables for the two watersheds is presented in Figure 2. Model simulations indicate that in general, SWAT performed better on Shell Creek than on Logan Creek, primarily because the Columbus, NE climate gage is located closer to Shell Creek than the Sioux City gage is to Logan Creek. For the most part, this in turn led to more accurate streamflow responses to precipitation for Shell Creek than for Logan Creek. Test results show that for the Shell Creek Watershed, SWAT underestimated the streamflow response from snowmelt during March of 1993, but overestimated the sediment and nutrient responses for that month. SWAT performed well in simulating the February 1971 hydrologic and water quality responses for the Logan Creek Watershed, but underestimated responses from storms during June of 1971 and overestimated them during July 1972. During the validation periods, SWAT performed well in simulating streamflow on Shell Creek, but overestimated flows on Logan Creek for July 1982 and May/June 1983 and underestimated them for April/May 1984 (Figure 3). Discrepancies between measured versus simulated responses were largely attributed to data deficiencies in the spatial representation of precipitation on the two respective watersheds.

Table 6 Monthly streamflow, sediment, total nitrogen, and total phosphorus percent bias and coefficient of efficiency statistics for the Shell Creek and Logan Creek Watersheds

Watershed name	Time Series	Streamflow		Sediment		Total N		Total P	
		PBIAS** /%	Streamflow NSE***	PBIAS /%	Sediment NSE	PBIAS /%	Total N NSE	PBIAS /%	Total P NSE
Shell Creek	1992-1994 C*	3.9	0.82	-9.5	0.90	-7.6	0.90	-17.7	0.78
Logan Creek	1971-1975 C	2.6	0.88	-8.6	0.84	13.9	0.71	7.9	0.94
Shell Creek	1998-2000 V	9.2	0.83						
Logan Creek	1984-1986 V	-22.7	0.58						

Note: * C = Calibration; V = Validation. ** PBIAS = Percent Bias. *** NSE = Nash Sutcliffe Coefficent of Efficiency.

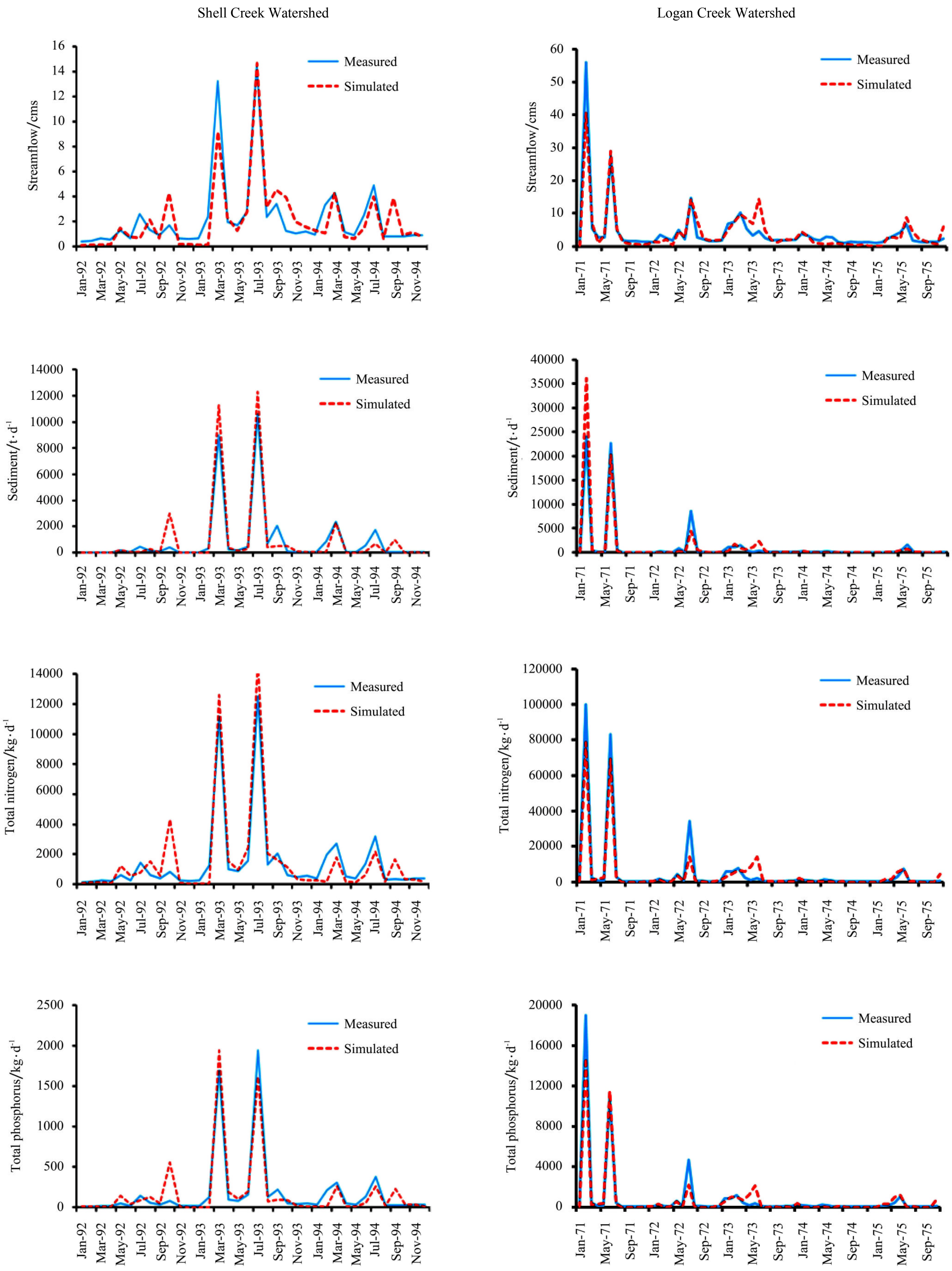


Figure 2 Comparison of measured versus simulated monthly streamflow, sediment, total nitrogen, and total phosphorus during calibration periods for the Shell and Logan Creek Watersheds

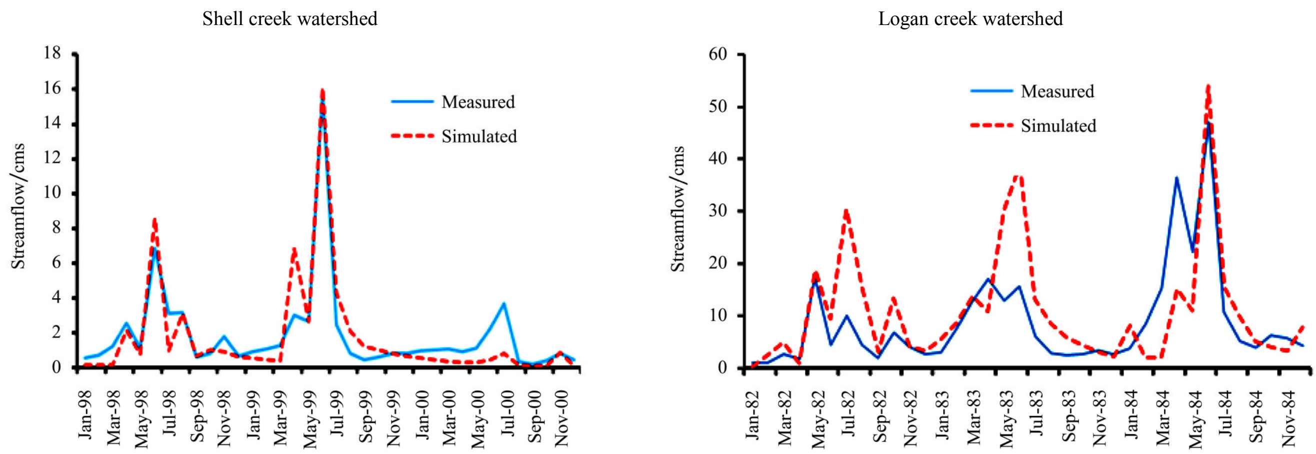
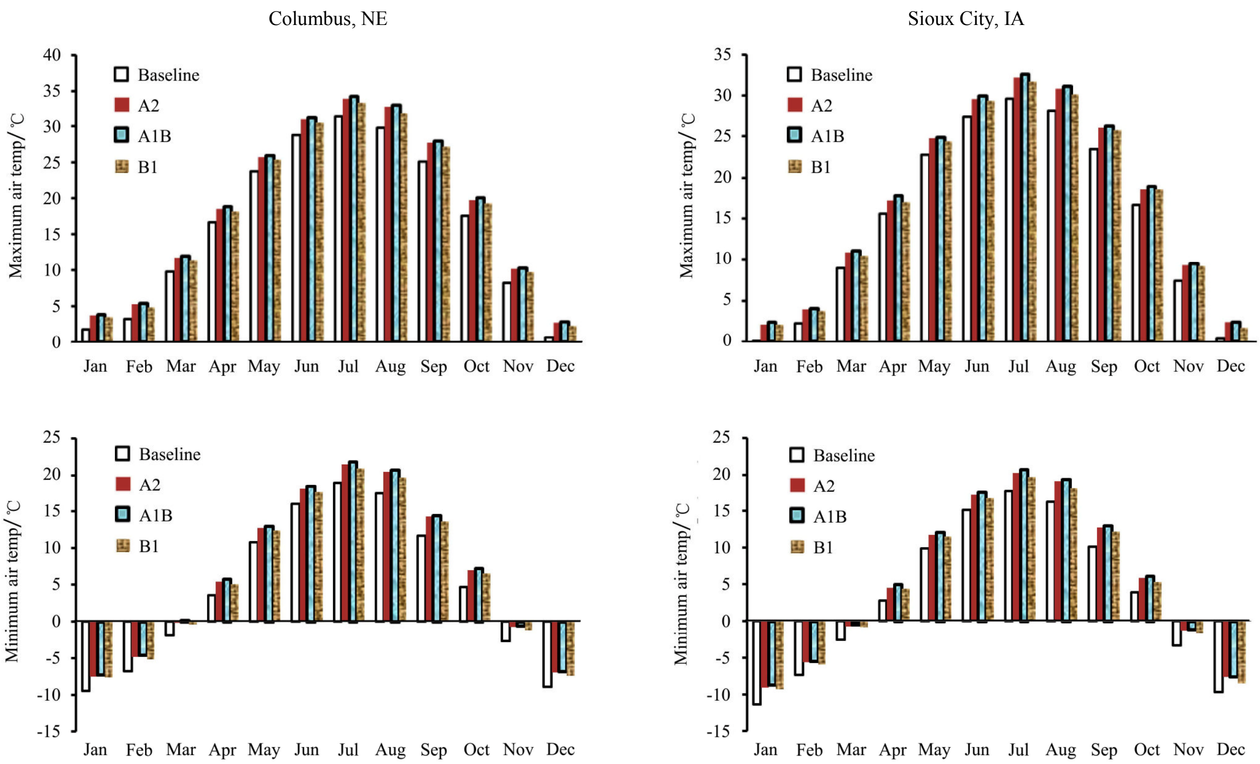


Figure 3 Comparison of measured versus simulated monthly streamflow during validation periods for the Shell and Logan Creek Watersheds

3.2 Comparison of watershed response for existing and future climate scenarios

3.2.1 Climate

Average annual maximum and minimum air temperatures for baseline, A2, A1B, and B1 climate scenarios for the Shell Creek Watershed were 16.5 and 4.5, 18.5 and 6.7, 18.8 and 6.9, and 18.2 and 6.2℃. Average annual maximum and minimum air temperature for baseline, A2, A1B, and B1 climate scenarios at the Logan Creek Watershed were 15.3 and 3.5, 17.3 and 5.6, 17.6 and 5.9, and 17.1 and 5.2℃. For both watersheds, average annual maximum air temperatures are projected to increase about 2.0, 2.3, and 1.8℃, respectively for the A2, A1B, and B1 future climate scenarios relative to the corresponding baseline temperatures; minimum air temperatures are expected to increase about 2.2, 2.4, and 1.7℃, respectively. Relative to the baseline condition, average annual snowfall simulated by SWAT is projected to decrease 27%, 22%, and 22% for the A2, A1B, and B1 climate change scenarios for Shell Creek and 22%, 22%, and 6% for Logan Creek, respectively. Average monthly maximum and minimum air temperatures at the two sites for each climatic condition are presented in Figure 4.



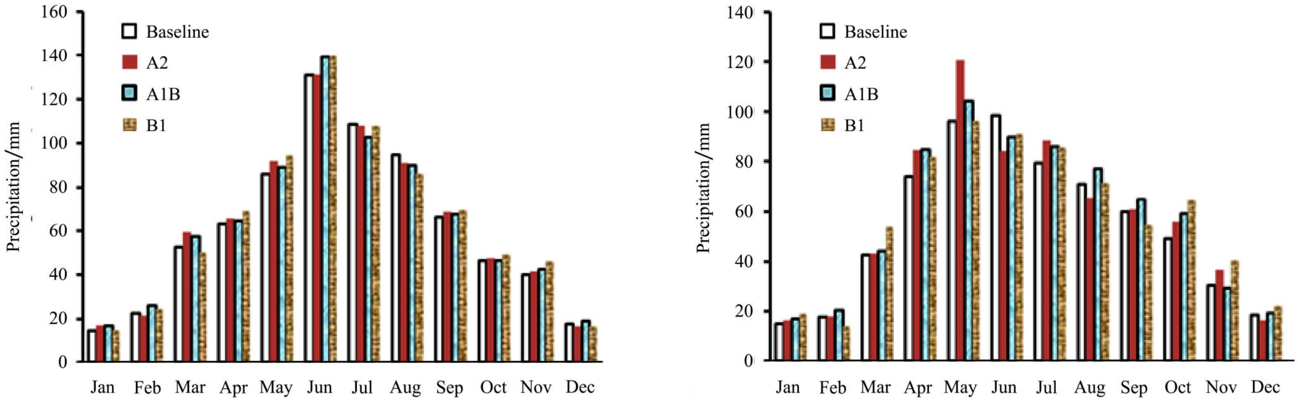


Figure 4 Monthly variations in maximum and minimum air temperature at the Columbus, NE and Sioux City, IA climate stations for the baseline and three future climate scenarios

Average annual precipitation for the baseline, A2, A1B, and B1 climate scenarios for Shell Creek were 743, 760, 762, and 766 mm, respectively. This represents increases in precipitation of 2.2%, 2.3%, and 3.1% for the A2, A1B, and B1 future climatic conditions. Although only representing small amounts in terms of the total annual precipitation, the largest monthly percentage increases in precipitation relative to the baseline condition were 12.9% (Mar), 15.9% (Feb), and 15.1% (Nov) for the A2, A1B, and B1 scenarios, respectively; the largest percentage decreases were 6.1% (Feb), 5.7% (Jul), and 9.2% (Aug). Modest increases in seasonal precipitation of 7.7%, 5.0% and 6.0% were exhibited for the spring months of March to May for the A2, A1B, and B1 scenarios relative to the baseline. Similar increases of 3.2%, 2.5%, and 7.3% were also exhibited for the fall months of September to November for the A2, A1B, and B1 scenarios. The projected changes in precipitation for the summer months of June to August were nearly negligible for the three future climate scenarios.

Average annual precipitation for the baseline, A2, A1B, and B1 climate scenarios for Logan Creek was 652, 690, 697, and 694 mm, respectively. For the A2, A1B, and B1 future climatic scenarios, this represents increases in precipitation of 5.8%, 6.9%, and 6.4% relative to the baseline condition. Percent increases in precipitation for the future climate change scenarios were therefore more pronounced than those for Shell Creek, thus reflecting projected spatial variability in the precipitation signal between the two sites. The largest monthly percentage increases in precipitation relative to the baseline condition were 25.0% (May), 20.0% (Oct), and 32.8% (Nov) for the A2, A1B, and B1 scenarios, respectively. Substantial increases in seasonal precipitation of 16.6%, 9.6%, and 8.9% were exhibited for the spring months of March to May for the A2, A1B, and B1 scenarios relative to the baseline. Similar increases of 9.6%, 9.9%, and 14.1%, respectively, were exhibited for the fall months of September to November. Seasonal changes in precipitation for the summer months of June to August are projected to be -4.3%, 1.8%, and -0.4% for the A2, A1B, and B1 scenarios, respectively.

3.2.2 Water budget

Average annual water budgets for the two test watersheds under the baseline and three future climate change scenarios are presented in Table 7. Hydrologic inputs of precipitation and irrigation water are balanced against abstractions consisting of surface and subsurface flow and evapotranspiration (ET). On a percentage basis, simulation results show very small changes in ET for any of the future climate change scenarios in comparison to the existing baseline condition. For the Shell Creek Watershed, larger percentage increases are expected to occur for subsurface flow in comparison to surface flow under future climatic conditions; just the opposite is true for Logan Creek Watershed. For both watersheds, notable decreases in water for irrigation are anticipated under future climatic conditions: percentage decreases in irrigation range from 28% to 35% for Shell Creek and 42% to 47% for Logan Creek. Smaller irrigation amounts expected under future climate scenarios reflect the impact of elevated carbon dioxide levels that lead to increased plant productivity and decreased crop water requirements.

Table 7 Average annual water budget for the Shell Creek and Logan Creek Watersheds under the baseline and three future climate change scenarios

Watershed	Climate Scenario	Precip. /mm	% Change in precip. from baseline	Irrigation from deep aquifer /mm	% Change in irrig. from baseline	Surface runoff /mm	% Change in surface runoff from baseline	Subsurface runoff /mm	% Change in subsurface runoff from baseline	ET /mm	% Change in ET from baseline
Shell	Baseline	743		46		48		7		734	
	A2	760	2%	30	-35%	62	29%	16	129%	712	-3%
	A1B	762	2%	32	-30%	61	27%	11	57%	722	-2%
	B1	766	3%	33	-28%	57	19%	10	43%	732	0%
Logan	Baseline	652		19		38		25		608	
	A2	690	6%	11	-42%	63	66%	36	44%	602	-1%
	A1B	697	7%	10	-47%	63	65%	34	26%	610	0%
	B1	694	6%	11	-42%	55	45%	38	49%	612	1%

3.2.3 Streamflow

As noted in Table 8, average annual stream discharge for the baseline, A2, A1B, and B1 climate change scenarios was 1.35, 1.90, 1.74, and 1.62 cms for the Shell Creek Watershed and 1.59, 2.41, 2.41, and 2.31 cms for the Logan Creek Watershed, respectively. The percent change in discharge from Shell Creek for the A2, A1B, and B1 scenarios relative to the baseline condition is projected to be 41%, 29%, and 20%, while that from Logan Creek is 52%, 52%, and 45%, respectively. For the A2, A1B, and B1 climate change scenarios, the actual projected increase in discharge for the Shell Creek Watershed is 0.55, 0.39, and 0.27 cms and 0.82, 0.82, and 0.72 cms, respectively, for the Logan Creek Watershed.

Table 8 Average annual streamflow, sediment, total nitrogen, and total phosphorus for the Shell Creek and Logan Creek Watersheds under the baseline and three future climate change scenarios

Watershed	Climate Scenario	Runoff /cms	% Change in streamflow from baseline	Sediment yield /t • d-1	% Change in sediment from baseline	Total nitrogen /kg • d^{-1}	% Change in total N from baseline	Total phosphorus /kg • d^{-1}	% Change in total P from baseline
Shell	Baseline	1.35		359		1250		112	
	A2	1.90	41%	535	49%	1770	42%	119	6%
	A1B	1.74	29%	514	43%	1680	26%	128	14%
	B1	1.62	20%	440	23%	1520	22%	109	-3%
Logan	Baseline	1.59		183		1020		149	
	A2	2.41	52%	351	92%	1910	87%	259	74%
	A1B	2.41	52%	451	146%	1920	88%	259	74%
	B1	2.31	45%	362	98%	1790	75%	223	50%

Average monthly variations in streamflow, sediment, total nitrogen, and total phosphorus from the two watersheds under the four climatic conditions are presented in Figure 5. Although not readily apparent from the figure, the peak discharge months from May to July account for about 45%, 46%, 49%, and 52% of the total annual streamflow for the baseline, A2, A1B, and B1 climate scenarios on Shell Creek and about 41%, 47%, 40%, and 40% of the total for the four scenarios on Logan Creek, respectively. With a few exceptions, increases in streamflow are projected for each month for both watersheds. For the Shell Creek Watershed, the largest projected monthly increases in streamflow will occur in June (1.40, 1.48, 1.64 cms) for the A2, A1B, and B1 scenarios, respectively, and in May (2.87, 2.11, and 1.65 cms) for the Logan Creek Watershed (Figure 5).

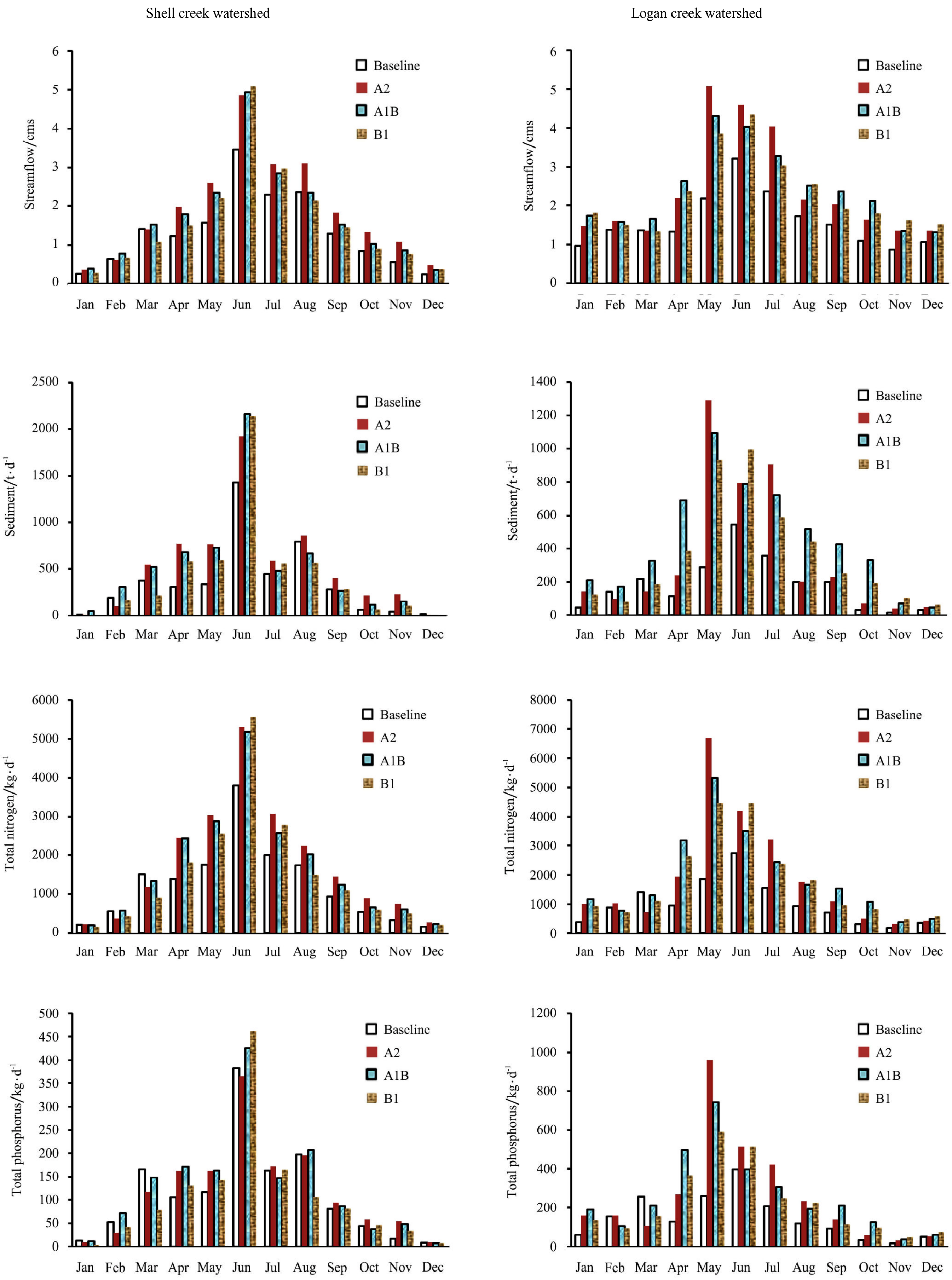


Figure 5 Average monthly variations in streamflow, sediment, total nitrogen, and total phosphorus losses from the Shell and Logan Creek Watersheds for the baseline and three future climate scenarios

3.2.4 Sediment

The Shell Creek Watershed average annual sediment yield for the baseline, A1, A1B, and B1 climatic conditions are 359, 535, 514, and 440 tons/day, respectively, compared to 183, 351, 451, and 362 tons/day for the Logan Creek Watershed (Table 7). Relative to the baseline condition, the projected annual percentage increases in sediment under the A2, A1B, and B1 future climate change scenarios are 49%, 43%, and 23% for Shell and 92%, 146%, and 98% for Logan, respectively (Table 8). For both watersheds, monthly variations in sediment follow similar monthly patterns for simulated streamflow (Figure 5). Model predictions suggest that the peak months from May to July account for about 51%, 51%, 55%, and 62% of the annual sediment load for Shell Creek and 54%, 71%, 48%, and 58% for Logan Creek under the baseline, A2, A1B, and B1 climatic conditions, respectively. Relative to the baseline, the greatest monthly net increases in sediment for the Shell Creek Watershed are projected to occur in June (493, 732, and 709 tons/day) for the A2, A1B, and B1 scenarios, respectively, and in May (1000, 808, and 644 tons/day) for the Logan Creek Watershed.

3.2.5 Total Nitrogen

Average annual total nitrogen yield for the Shell Creek Watershed under the baseline, A1, A1B, and B1 climatic conditions are 1250, 1770, 1680, and 1520 kg/day, respectively, compared to 1020, 1910, 1920, and 1790 kg/day for the Logan Creek Watershed (Table 8). Relative to the baseline condition, the projected annual percentage increases in total N under the A2, A1B, and B1 future climate change scenarios are 44%, 34%, and 22% for Shell and 87%, 88%, and 75% for Logan, respectively. In terms of actual annual changes relative to the baseline, the A2, A1B,and B1 increases are 520, 430, and 270 kg/day and 890, 900, and 770 kg/day for the Shell and Logan Creek Watersheds, respectively. Based on model simulations, decreases in total nitrogen of 21%, 4%, and 33% for the A2, A1B, and B1 scenarios, respectively, are projected to occur during the winter months for the Shell Creek Watershed. Relative to the baseline climate scenario, the largest A2, A1B, and B1 net increases in total N for Shell Creek will occur in June (1510, 1400, and 1770 kg/day) and in May (4840, 3480, and 2610 tons/day) for the Logan Creek Watershed.

3.2.6 Total phosphorus

Average annual total phosphorous loss for the baseline, A1, A1B, and B1 climatic conditions are 112, 119, 128, and 109 kg/day, respectively, for the Shell Creek Watershed. This compares to 149, 259, 259, and 223 kg/day for the Logan Creek Watershed. Relative to the baseline condition, the projected annual percentage changes in total P under the A2, A1B, and B1 future climate change scenarios are 6%, 14%, and -3% for Shell and 74%, 74%, and 45% for Logan, respectively (Table 8). Model predictions suggest that the spring months from April to June account for about 44%, 48%, 49%, and 56% of the annual total P loss for Shell Creek and 44%, 56%, 53%, and 55% for Logan Creek under the baseline, A2, A1B, and B1 climatic conditions, respectively. Moderate decreases in total P are projected to occur during the winter months for the Shell Creek Watershed under the A2 and B1 future climate change scenarios. For the Logan Creek Watershed, moderate decreases in monthly total P for the future climate scenarios are anticipated during the winter months: February (59%) for the A2 and February/March (22% and 39%) for the A1B and B1, respectively (Figure 5). Relative to the baseline, the largest monthly net increases in total P for the Shell Creek Watershed are projected to occur in April (57 and 68 kg/day) for the A2 and A1B and in June (80 kg/day) for the B1 scenarios, respectively. For the Logan Creek Watershed, the largest monthly net increases in total P are projected to be 703, 486, and 330 kg/day for the A2, A1B, and B1 scenarios during the month of May.

3.3 Comparison of watershed response with best management practice implementation

Comparisons of changes in streamflow and constituent loadings among the BMPs are presented in Figure 6. Although not shown in the figure, test results show that with a few exceptions, the percent change in implementing a particular BMP on the four output variables did not vary appreciably between the baseline and any of the future climate scenarios. Model simulations indicate that the terrace and no-till BMPs had minimal impact on changes in average annual streamflow on either watershed; in general, these two BMPs led to small decreases in streamflow. No changes in streamflow were noted for either watershed with the

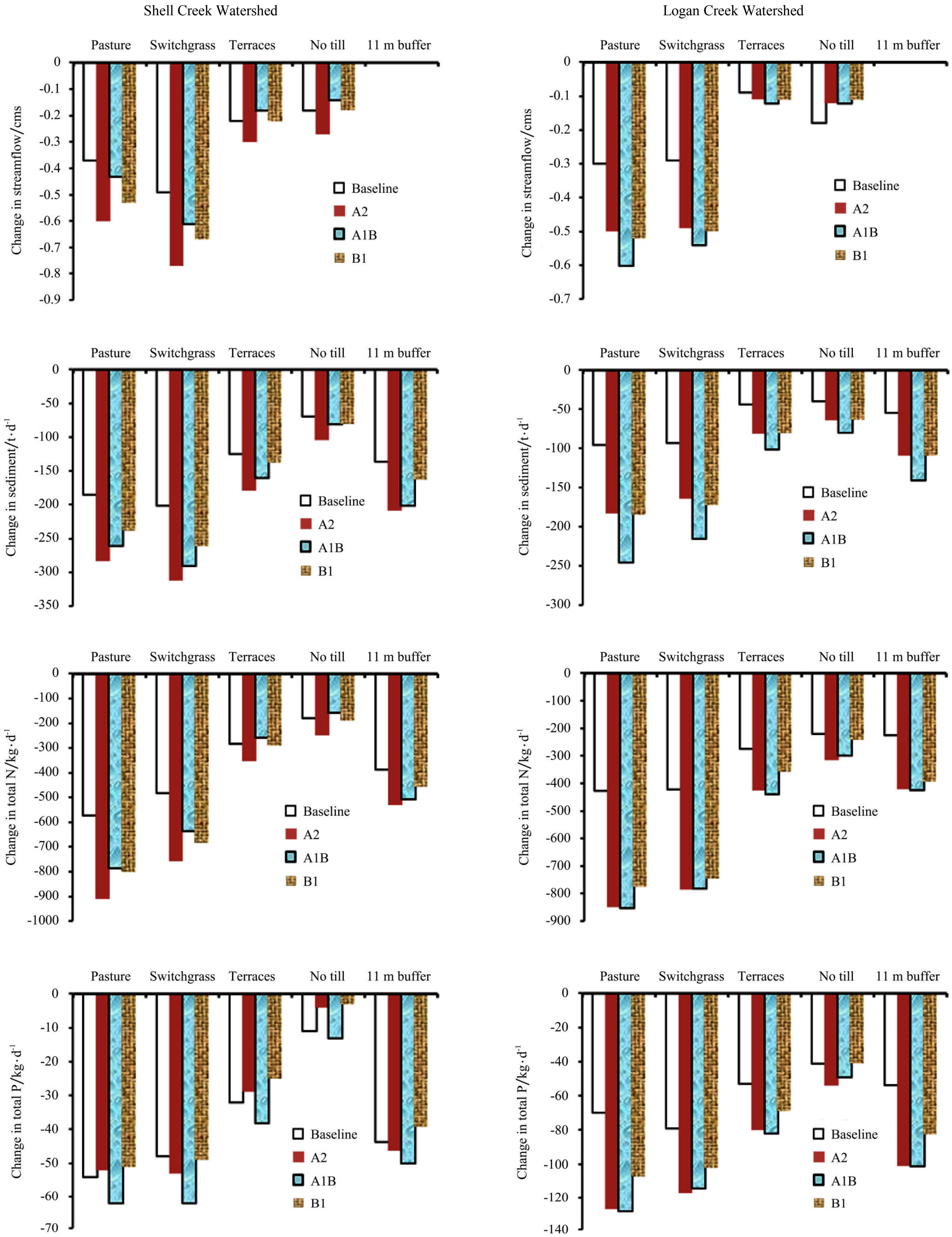


Figure 6 Impacts of BMP implementation on changes in streamflow, sediment, total nitrogen, and total phosphorus for the Shell and Logan Creek Watersheds under the baseline and three future climate scenarios

implementation of the 11 meter buffer BMP. For both watersheds, the conversion of existing corn and soybean cropland to either pasture or switchgrass is expected to result in moderate decreases in streamflow for each of the four scenarios. For the Shell Creek Watershed, for example, decreases in streamflow of 0.37%, 0.61%, 0.44%, and 0.53% with the implementation of the pasture BMP are expected for the baseline, A2, A1B, and B1 scenarios. Even more pronounced decreases on that watershed are expected for the switchgrass BMP. Very similar reductions in streamflow for all four scenarios are anticipated for the pasture and switchgrass BMPs on Logan Creek.

For both watersheds, conversion of existing corn and soybean cropland to pasture or switchgrass had the most pronounced effect among the five BMPs on decreasing sediment losses. For the pasture BMP, these average annual reductions due to BMP implementation were 185, 283, 260, and 238 tons/day for the Shell Creek Watershed and 96, 183, 245, and 184 tons/day for the Logan Creek Watershed under the baseline, A2, A1B, and B1 scenarios, respectively (Figure 6). In general, the 11 meter buffer BMP performed somewhat better than the terrace or no till BMPs in reducing sediment losses on both Shell Creek and Logan Creek.

Simulation results show that the conversion of cropland to pasture or switchgrass leads to marked decreases in total nitrogen losses for all four climate scenarios on both the Shell Creek and Logan Creek Watersheds; both of these treatments lead to similar responses on each of the watersheds. For switchgrass, the projected annual reductions in total N were 484, 759, 635, and 684 kg/day for Shell Creek under the baseline, A2, A1B, and B1 climate scenarios, while the respective projected annual reductions in total N were 422, 786, 779, and 746 kg/day for Logan Creek. Among the other three BMPs, the 11 meter buffer performed somewhat better than terrace or no till. SWAT simulations suggest that for Logan Creek, the conversion of existing corn and soybean to switchgrass brought about average annual reductions in total phosphorus losses that were nearly twice as great as those on Shell Creek under future climatic conditions. With the switchgrass BMP, total P reductions on Logan Creek were 127, 128, and 107 kg/day under the A2, A1B, and B1 scenarios; these compare to respective reductions of 52, 62, and 51 kg/day on Shell Creek. Similar to that which was reported for sediment and total N, pasture and switchgrass performed best on reducing total P losses on both watersheds, followed by the 11 meter buffer, terrace, and no-till treatments.

4 Discussion

In spite of a number of similarities that exist between the Shell Creek and Logan Creek drainages that were selected for this study, noticeable differences are evident upon comparison of the hydrologic and water quality responses of the two watersheds. Although average annual precipitation for Shell Creek is about 15% greater than that for Logan Creek for the baseline climatic condition, average annual streamflow for Logan Creek is more than 1.5 times as great as that from Shell Creek. Even more pronounced differences in watershed response are noted at the monthly time scale under the baseline condition, especially for the late fall to spring months. During the month of May, for example, average monthly precipitation is about 95 mm on Shell Creek and 103 mm on Logan Creek, with similar antecedent precipitation amounts during the months of March and April for both watersheds. However, streamflow, sediment, total N, and total P are about 2.1, 2.0, 2.9, and 4.3 times greater during that month for Logan Creek than for Shell Creek, respectively.

Differences in streamflow simulation between the two watersheds may in large part be attributed to both the integrated effects of topographic, land cover, and soil differences and the values selected for model calibration of each watershed. To test the impact of model calibration between the two watersheds, model output from Shell Creek was compared to output from Logan Creek using the Shell Creek climate input data and calibrated parameter set. Cursory testing revealed that using the Shell Creek set of model parameters for Logan Creek resulted in streamflow, sediment, total N, and total P reductions of 35%, 63%, 60%, and 75%, respectively under the baseline scenario. Among the most sensitive parameters calibrated in this study was the soil evaporation compensation factor (ESCO). Change of this parameter alone from 1.00 (calibrated value for Logan Creek) to 0.80 (calibrated value for Shell Creek) led to reductions in streamflow, sediment, total N, and total P reductions of 34%, 67%, 53%, and 64%, respectively under the baseline scenario. These marked changes in streamflow and water quality constituents illustrate the need for considerable care when performing model calibrations.

Model simulations suggest that under the three climate change scenarios investigated in this study, the average annual impact on streamflow will be somewhat greater on Logan Creek than on Shell Creek. However, the impact on average annual sediment, total N, and total P will be much stronger on the former in comparison to the latter. Streamflow is expected to be about 1.2 to 1.4 times greater than the baseline condition for Shell Creek and about 1.5 times greater for Logan Creek. Under the three future climate change scenarios, sediment losses are expected to be about 1.2 to 1.5 times greater than the baseline condition for Shell Creek and 2 to 2.5 times greater for Logan Creek; total N losses are expected to be about 1.2 to 1.4 times greater for Shell Creek and 1.7 to 1.9 times greater for Logan Creek. Relative to the baseline, total P losses under the future climate scenarios are projected to be about the same for Shell Creek and 1.5 to 1.7 times greater for Logan Creek. SWAT simulations indicate that for the Shell Creek Watershed, the A2 climate change scenario had the greatest projected overall impact on the four output variables, followed respectively by the A1B, and B1 scenarios. For the Logan Creek Watershed, the A1B climate change scenario is expected to have the greatest overall impact on streamflow and water quality, followed by the A2 and B1 scenarios, respectively.

Only three variables were modified in this study to determine the impacts of future climate change scenarios on the four output variables modeled in SWAT. These variables included air temperature, precipitation, and the concentration of CO_2. In a comparison of average annual streamflow response, a 1.4 fold increase in discharge was noted for the A2 climate scenario relative to the baseline for Shell Creek and a 1.5 fold increase was noted for Logan Creek. However, only a 2.2% increase in average annual precipitation is projected for the Shell Creek Watershed for the A2 scenario relative to the baseline, compared to a 5.8% increase for the Logan Creek Watershed. To help explain the reason for this apparent discrepancy, cursory testing was employed to assess the impact of air temperature, precipitation, and CO_2 concentration on streamflow for the baseline and A2 climate scenarios. Model simulations were performed for varying each of the three input variables one at a time, and the results of the modeled output were then compared between the two watersheds. The results of this analysis showed that on an annual basis, increases in CO_2 projected for the A2, A1B, and B1 climate scenarios relative to the baseline accounted for 27%, 29%, and 31% of the net increase in streamflow for the Logan Creek Watershed, respectively, while increases in precipitation accounted for the remaining 73%, 71%, and 69%. Different results were obtained on the Shell Creek Watershed, where increases in CO_2 and precipitation under the A2 climate scenario accounted for about 74% and 26% of the net increase in streamflow, respectively. For the A1B and B1 climate scenarios on Shell Creek, increases in CO_2 accounted entirely for the net increase in streamflow, in spite of the projected increases in precipitation for those two scenarios. For both watersheds, increases in air temperature projected for the A2 relative to the baseline led to small decreases in streamflow. Though further study is warranted, these surprising results illustrate the important influence that higher CO_2 concentrations appear to have on reductions in evapotranspiration and consequent increases in streamflow for watershed systems.

In this study only a single targeting approach was employed to select the location of BMPs that were implemented on the landscape. Targeting criteria were specified a priori based on individual HRU slope steepness and USLE soil erodibility K factors within each watershed. The targeting approach used in this investigation did not necessarily reflect the greatest load reductions that could be expected to occur on either watershed. Other approaches for implementing the equivalent number of BMPs or HRU areas impacted by BMPs could lead to considerably different load reductions than those reported in this study.

Simulation results indicate that of the five BMPs tested in this study, the conversion of cropland to pasture or to switchgrass provided the greatest reductions in constituent loadings. It must be recognized, however, that the conversion of corn or soybeans to switchgrass or pasture may not represent a viable economic alternative in the future. For each type of BMP, overall efficacy was generally about the same on both watersheds for sediment and total N, while efficacy was greater on Logan Creek than on Shell Creek for total P. However, it must be recognized that under the future climate change scenarios, a much broader and/or more effective BMP strategy would need to be employed if future constituent loads were to be maintained at levels that are comparable to those simulated for the baseline condition. Based on the results of the targeting approach used in this study, model simulations suggest that the pasture, switchgrass, and 11 meter buffer BMPs implemented on Shell Creek and only the pasture and switchgrass BMPs implemented on Logan Creek would be suitable choices under the future climate scenarios for providing sufficient pollutant load reduction that is comparable to the loads simulated for the current day baseline condition. Although only a relatively straight forward targeting approach was undertaken in this study, the results of this comparison point to the daunting challenges that will exist in the future for developing and implementing watershed management plans that are effective in improving the quality of water in stream systems throughout the Heartland as well as the nation.

5 Conclusions

Findings from this investigation demonstrate that in spite of the close proximity and many similarities between the two study watersheds, considerable differences were noted in the hydrologic and water quality responses for both the present day and future climate change scenarios. This suggests that modeling investigations used to evaluate the impact of climate change on streamflow or water quality constituents are not only highly sensitive to calibration, but also to the spatial and temporal variations in the input data used to simulate future climate change scenarios. Considerable care must therefore be taken in model calibration and extending applications from one watershed to another, even on a regional basis.

Results of this study indicate that for the Shell Creek Watershed, the A2 climate change scenario is expected to have the greatest projected overall impact on the four output variables, followed respectively by the A1B, and B1 scenarios. For the Logan Creek Watershed, the A1B climate change scenario is expected to have the greatest overall impact on streamflow and water quality, followed by the A2 and B1 scenarios, respectively. Under the future climate change scenarios examined in this study, modest to moderate increases in streamflow, sediment, and nutrients are projected to occur on Shell Creek while substantial increases are expected for Logan Creek. With the wide array of climatic, soils, and land use conditions that exist in the U.S. Heartland, modeling studies similar to the one performed in this investigation need to be undertaken in a variety of watersheds throughout the region to assess the projected impacts of future climate change on streamflow and water quality.

Findings from this study suggest that future projected increases in both precipitation and CO_2 concentration account for net increases in streamflow and attendant pollutant loadings, but in different ways on each of the test watersheds. Although these findings are preliminary, they point to the need for a better understanding of how future changes in these and other climatic variables will impact components of the hydrologic cycle and the fate and transport of biological and chemical constituents throughout watershed systems.

A targeting approach employed in this study compared the impact of five BMPs on streamflow and water quality for each watershed. Results of this study indicate that for the most part, pollutant responses to BMP treatments were about the same on the two watersheds under existing or future climate change scenarios. Simulation results indicate that of the five BMPs tested in this investigation, the conversion of cropland to switchgrass and the conversion of cropland to pasture were the overall most effective BMPs while no-till was the least effective. These results are similar to those reported by Woznicki et al.[12] who employed SWAT to assess BMP impacts for the Tuttle Creek Lake Watershed in Nebraska and Kansas under future climate change scenarios. Findings of this study indicate that the switchgrass and pasture treatments under the future climate change scenarios would provide sufficient sediment, total N, and total P load reductions that are comparable to the respective loads simulated for the current day baseline condition. Findings from this investigation also accentuate the need to explore new methodologies for BMP placement. In recent years the development of sophisticated optimization searches has shown tremendous promise for identifying the cost effective placement of BMPs to reduce pollutant loadings in stream systems[58-60]. Given the likelihood of projected increases in pollutant loadings under future climate change scenarios, a need exists to determine how new methodologies and optimized searches can best be employed to address future water quality concerns.

Acknowledgements

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