ASSESSING THE IMPACTS OF LAND-USE AND CLIMATE CHANGE FOR WATER RESOURCE MANAGEMENT

BY

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Abstract

Sustainable management of water resources is a challenging interdisciplinary problem requiring the integration of fields such as hydrology, ecology, sociology, and public policy. In the past decade, there has been a great effort to understand how issues such as climate change and land-use change for biofuel feedstock production will affect water resources. This dissertation assesses the impacts of climate change and land-use change for water resource management in Kansas using an interdisciplinary approach and tools such as the Soil and Water Assessment Tool (SWAT), social surveys, and geospatial analysis. The SWAT model is used to simulate corn and grain sorghum biofuel-based land-use scenarios to assess water quality impacts and sustainability indicators in the Perry Lake and the Kanopolis Lake watersheds in Kansas. Modeling results suggest that corn scenarios produced significantly greater water quality impacts than grain sorghum scenarios, but that corn had a much higher crop yield, particularly in the Perry Lake watershed, and thus can provide more ethanol production potential per land, water, and nutrient input, which are efficiency metrics often used in agricultural studies. Overall, grain sorghum may be a more sustainable feedstock crop in drier climates and corn may be more sustainable in wetter climates. The sustainability measures utilized in this study allow for comparison between crops and between watersheds, yet they are typically not included in the current biofuel-based land-use analyses. This study shows the potential of integrating water quality analysis with sustainability indicators to develop a richer assessment of the trade-offs and benefits of landscape change for biofuel feedstock development.

The impact of climate change was assessed in three ways: first, with a review of the potential climate change impacts for reservoirs and a discussion of the potential in-lake and watershed management strategies for mitigation; second, with a social survey that explores perceptions of Kansas water managers towards climate change and planning for climate impacts; and third, with a study of the influence of reservoir management on greenhouse gas emissions from a tributary of the Three Gorges Reservoir in China.

The review of climate change impacts for reservoirs found that the sustainability of reservoir services will be threatened by climate change, but that there are a variety of management tools that may be able to mitigate impacts. The social survey demonstrated that anthropogenic climate change is a contentious issue within the state of Kansas, but that water managers believe it is important to consider future climate change in their planning efforts. Survey results, along with a review of key Kansas water management plans, suggest that Kansas water managers are indeed responsive to climate variability and are starting to integrate climate variability into planning efforts. The study of reservoir greenhouse gas emissions suggest that both CO_2 and CH_4 fluxes were influenced by reservoir water level and exhibited distinct patterns that correspond to the reservoir operation cycle. Over 90% of CO_2 effluxes occurred during the high water period, whereas the 58% of CH_4 effluxes occurred during the low water period. Results suggest that reservoir operations altered the hydraulic retention time, which along with water temperature, controlled the synthesis and decomposition of carbon in the backwater system.

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Chapter 1 – Introduction

1.1 General Background

From water supply shortages to diminishing water quality, sustainable management of water resources is one of the great challenges of this century. The challenge is made even greater by the interdisciplinary nature of most water resource problems. Water resource management at its broadest level requires integration of three major systems: the human system (i.e. water-related organizations, engineering projects, water use sectors, society), the physical system (i.e. hydrologic cycle, geomorphology), and the ecological and biogeochemical systems (i.e. aquatic organisms, nutrient cycles, biodiversity) (Pahl-Wostl 2007; Wagener et al. 2010). Therefore, it is necessary to integrate such fields as hydrology, ecology, sociology, and public policy to tackle the most challenging water resource problems.

In the past decade additional stressors, such as climate change, land-use change, and aging infrastructures have further challenged water resource managers. Available evidence shows that climate change may lead to increased occurrence and magnification of drought, alteration of geographic and temporal precipitation patterns, and intensification of precipitation events (Pachauri 2007; Seneviratne et al. 2012). Studies also show that changing landscape patterns due to urban growth and fluctuating agricultural land-use may further alter local and regional hydrology and water quality (Johnson and Host 2010).

In order to advance water resource research within the large topics of land-use change and climate change, a comprehensive set of tools are necessary. Such tools may include a broad monitoring network of hydrologic data, geospatial datasets, ecohydrological models for simulation testing, geospatial analysis, and social science methods to integrate findings back into management.

1.2 Ecohydrological Modeling for Sustainability Studies

Ecohydrological models are useful tools for studying the environmental impacts of land-use development and management scenarios. Popular tools include the Hydrologic Simulation Program Fortran (HSPF), the Environmental Policy Integrated Climate (EPIC) model, and the Soil and Water Assessment Tool (SWAT) (Tong et al. 2012; Secchi et al. 2011; Love and Nejadhashemi 2011). These models include hydrological, biogeochemical, and vegetation components that are coupled together to effectively simulate ecological and hydrological processes within a watershed (Krysanova and Arnold 2008). Such models usually require topographical, land-use, soil, and climate data sources as inputs for a combination of process and empirically-based mathematical equations.

In particular, SWAT has been shown to be an effective model for analyzing water quality and hydrologic impacts of agricultural management scenarios (Douglas-Mankin et al. 2010). SWAT has been used to study the water quality impacts of increased biofuel feedstock production in Michigan, North and South Dakota, the Arkansas-White-Red River Basin, and the Upper Mississippi River watershed (Kling et al. 2010; Secchi, Gassman, et al. 2011; Wu et al. 2012; Love and Nejadhashemi 2011; Jager et al. 2014). However, water quality impacts are often studied separately from other sustainability indicators, such as nutrient-, land-, and water-use efficiency, and biofuel production potential. Combining water quality impacts with sustainability indicators provides a more comprehensive examination of the costs and benefits of increasing biofuel feedstock production in a particular watershed.

1.3 Climate Change and Water Management

Climate change will pose many challenges for water resources, both in the management of artificial structures, such as dams and levees, as well as the management of natural water bodies for adequate water supply for municipal, agricultural, industrial, and ecological uses (Milly 2008; Bekele and Knapp 2010; Brekke 2010). While the impacts of climate change on reservoirs are often studied from a water supply perspective, see (Park and Kim 2014; Li et al. 2010; Raje and Mujumdar 2010; Alvarez et al. 2014; Georgakakos et al. 2012) reservoir water quality management issues are infrequently considered in the context of climate change (Zhou and Guo 2013). Currently, loss of storage capacity due to sedimentation, water quality degradation, and toxins from blue-green algal blooms are issues that threaten reservoir sustainability. Climate change is hypothesized to exacerbate these problems by increasing sediment and nutrient export from the surrounding watersheds, changing flow regimes, and increasing summer water temperatures.

In order to plan for climate change impacts, tools need to be available for water and natural resource managers to integrate predictive climate information into water resource planning and management. Tools are being developed in the academic environment, but it is often challenging to put these tools to use in practice. In order to improve integration of climate science into water resource management, information needs to be available to water resource planners. Publishing in journals specifically geared towards water resource managers is one potential method for starting a conversation. In addition, reaching out to managers to gain an understanding of information needs and issues of concern can help pave a joint path towards water management plans that integrate the uncertainty of climate change.

1.4 Study Goals and Objectives

This dissertation explores issues related to water quality and water resource management from multiple perspectives. The overall research is divided into three major components, which are organized into five chapters in this dissertation.

The first component of this dissertation includes both the development (Chapter 2) and implementation (Chapter 3) of SWAT models to study impacts of land-use change in Kansas. Chapter 2 includes the SWAT model development and calibration for the Perry Lake and the Kanopolis Lake watersheds. The goal was to develop two models to capture the climatic and geographically distinct features of eastern and west-central Kansas to be used for further analyses. The specific objectives were to:

1) Develop a distributed parameter watershed model, SWAT, for two different watersheds within Kansas at various scales;
2) Calibrate models based on multiple parameters, such as crop yield, daily, monthly and annual streamflow, and sediment load. Then, accurately simulate current hydrologic conditions.

Chapter 3 uses the developed SWAT models to simulate biofuel-based land-use change and to access water quality and sustainability indicators of the various scenarios. The main goal was to access the environmental impacts of increasing corn and grain sorghum land-use, the two dominant biofuel crops in the state of Kansas. Specific objectives were to:

1) Develop land-use scenarios that explore both intensification and extensification of current agricultural land, focusing on land-use types that are likely to be converted to biofuel feedstocks;

2) To test scenarios in the SWAT model at various scales and calculate water quality and sustainability indicators for each scenario;
3) Evaluate the water quality and sustainability indicators to determine scenarios most favorable for biofuel development.

The second component of this dissertation focuses on climate change and water management, both with respect to reservoir management (Chapter 4) and the perspective of Kansas water managers towards integrating climate change into planning and management strategies (Chapter 5) . Chapter 4 provides an assessment of the impacts of climate change for reservoir systems, and a review of watershed and in-reservoir management strategies to mitigate impacts of climate change. In addition, tools and data sources that have been successfully used in climate change studies are presented.

Chapter 5 utilizes both literature review and a survey instrument to explore the integration of climate change into water resource management. The major objectives of this study were to:

1) Use science-policy integration literature to identify common barriers to integrating climate change into water management;
2) Develop and utilize a survey to gather opinions of Kansas water managers towards climate change and state-based water management;
3) Access the degree to which current Kansas water management plans and programs integrate climate change or variability;
4) Use science-policy integration literature to identify useful strategies for integrating climate change into water resource management.

The final component of this dissertation, chapter 6, examines the concept of climate change and water management from a different perspective by examining how management of a large reservoir in China can influence greenhouse gas emissions. This research reflects collaboration at Chongqing University that was possible due to a National Science Foundation East Asia and Pacific Institute Fellowship. Both water quality conditions and water management patterns were used to understand emissions of methane and carbon dioxide from the Pengxi Tributary of the Three Gorges Reservoir. Geospatial methods were used to estimate overall emissions for the tributary. Geospatial methods are increasingly used to analyze environmental data and are a fundamental tool used to improve management decisions.

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Chapter 2 – SWAT Model Development and Calibration for Perry Lake and Kanopolis Lake Watersheds

2.1 Introduction

Hydrologic models are critical for land-use planning, and determining hydrologic impacts of changes in climate or watershed practices. However, distributed hydrologic models need to be developed for each study watershed with careful attention to detail through parameterization and evaluation of model success through calibration and validation. The goal of model parameterization is to accurately characterize field conditions through the best available knowledge. The purpose of calibration is to modify uncertain model parameters, within a realistic range, to improve model performance while testing on a set of observed data. Validation then tests the calibrated model on a new set of observed data, without altering model parameters further. Calibration and validation is often performed at one location within a model, usually at the watershed outlet. However, multi-site calibration and validation is a more robust means of ensuring accurate representation of a spatially distributed watershed model (Zhang et al. 2008).

In this study, a multi-objective calibration framework is used to develop models for two watersheds in Kansas - the Perry Lake and Kanopolis Lake watersheds. The Soil and Water Assessment Tool (SWAT) was utilized for model development, as it is one of the most widely used watershed models for agricultural systems in North America and the World (Douglas-Mankin et al. 2010; Gassman et al. 2007). The goal was to develop watershed models that could be used for analysis of land-use change impacts in Kansas.

2.2 Methods

2.2.1 Study Sites

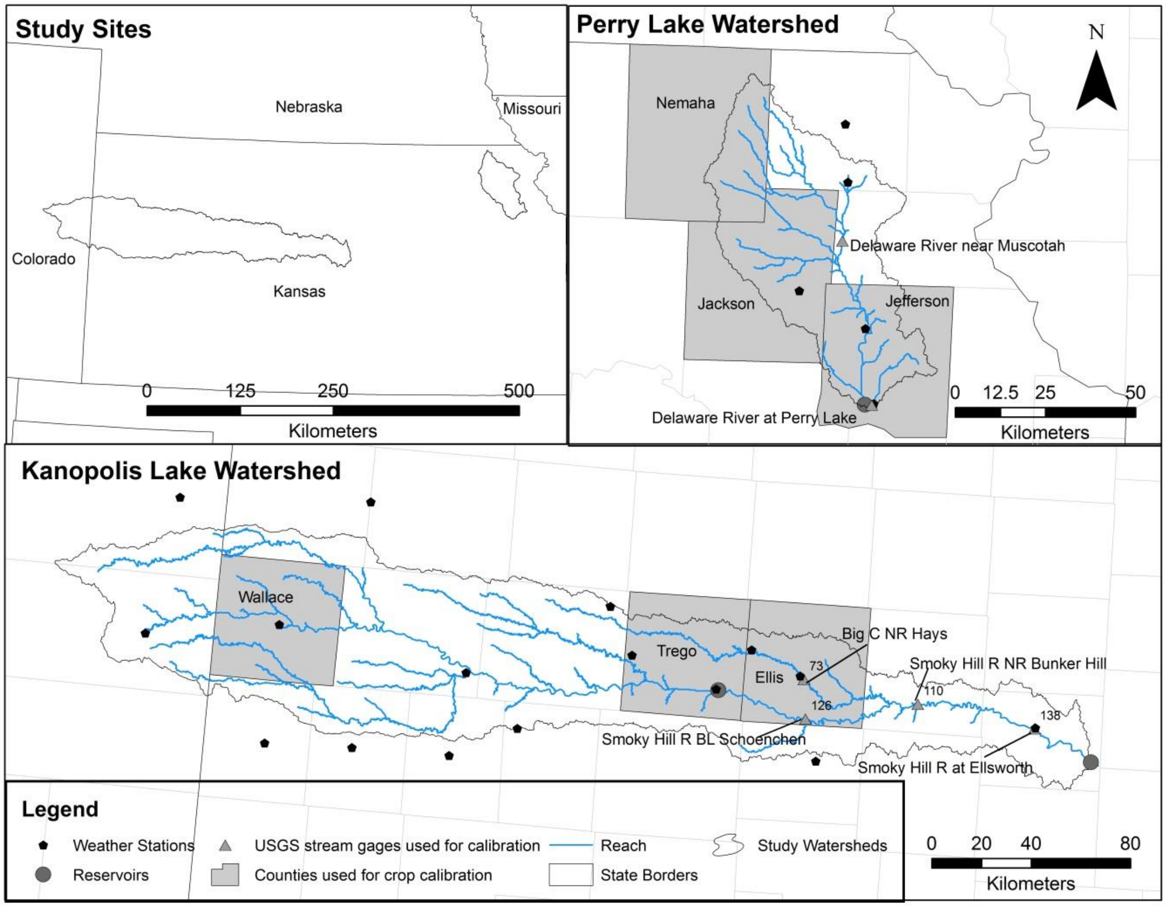

Figure 2-1: Location of study sites, Perry Lake watershed and Kanopolis Lake watershed, as well as weather stations and stream gages and counties used for calibration.

Perry Lake Watershed

The Perry lake watershed is a HUC-8 level watershed (10270103) located in northeastern Kansas within the Central Irregular Plains and the Western Corn Belt Plains Level III Ecoregions. The drainage area is approximately 2,924 km^2 and is utilized mostly for agricultural purposes with less than 0.05% total irrigated cropland. Hay (cool-season grassland) and rangeland (warm-season grassland) represent, respectively, 32% and 15% of the watershed, with corn and soybeans together representing 27% of the watershed. The mean annual precipitation ranges slightly from north to south with 890 mm at Horton, Kansas and 980 mm at Oskaloosa, Kansas. Most precipitation occurs during the April – September growing season (Juracek and Ziegler 2009). There are 7 major soil classes in the watershed, of which 23% have moderate infiltration rates (hydrologic soil group B), 30% have moderately high runoff potential (group C), and 47% have high runoff potential (group D). Approximately 17% of the Perry Lake watershed has a 0-2% slope, 39% is in the range of a 2-5% slope, and 44% has a slope greater than 5%.

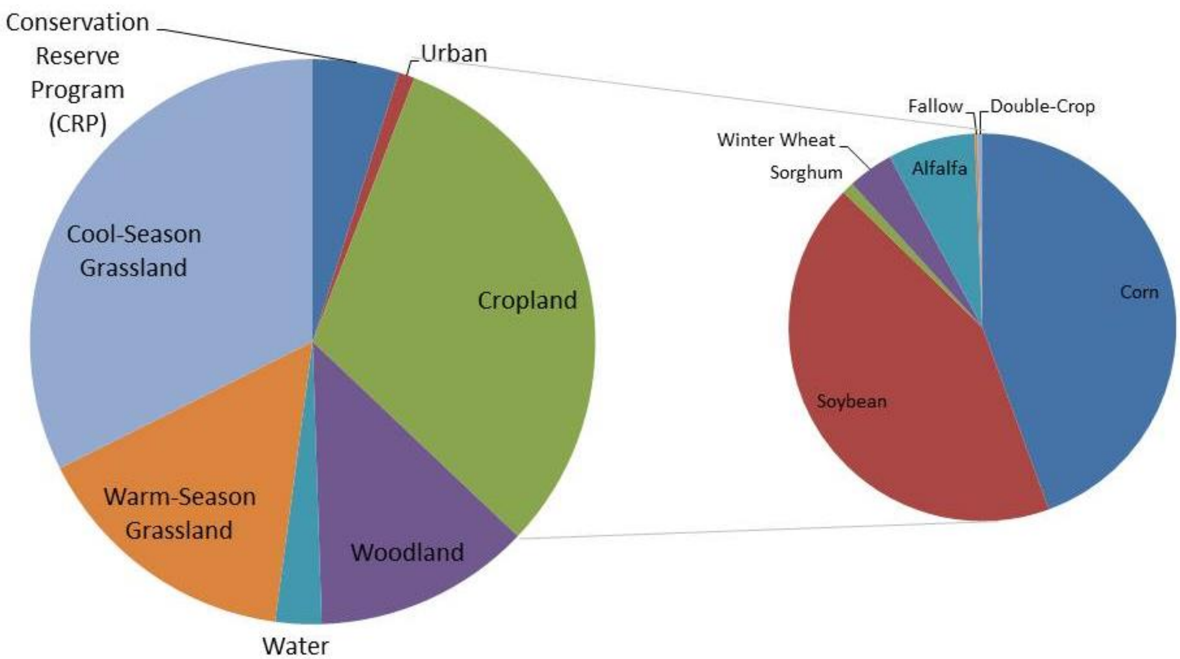

Figure 2-2: Perry Lake Watershed Land-use; major land-use types in left pie chart with cropland broken out into more specific types in the pie chart on the right.

The major water bodies in the watershed are the Delaware River, which drains into Perry Lake, a man-made reservoir operated by the Army Corp of Engineers, which then releases water which flows into the Kanas River. Perry Lake was opened in 1969 for the purposes of flood control, water supply, recreation, navigation, and wildlife management. Approximately 676,000 people visit the lake every year for recreation purposes, which contributes about $15.8 million in direct spending annually. Annual water supply benefits from the lake are estimated to be around $24.8 million when considering reservoir construction and mitigation costs (CDM Federal Programs Corporation 2011).

Kanopolis Lake Watershed

The Kanopolis Lake watershed is located in central to west-central Kansas and reaches across the state into the east-central portion of Colorado. It is located within the Central Great Plains and the Western High Plains ecoregions and includes HUC-8 subbasins 10260001-10260007. The watershed area is about 20,291 km^2. The predominant land-use types are rangeland (warm-season grassland; 40%), followed by winter wheat (29%); urban land-use represents less than 1% of the watershed. While irrigated cropland is more common in the Kanopolis watershed than the Perry watershed, it still represents a small portion of the overall watershed (4.4%). Precipitation varies greatly across the watershed with a long-term mean annual precipitation of 711 mm at Ellsworth, Kansas in the eastern portion of the watershed and only 483 mm at Sharon Springs, Kansas in the western portion of the watershed (Juracek 2011). The Kanopolis Lake watershed consists of 55 soil classes, and 89% have moderate infiltration (hydrologic group B). The narrow, elongated Kanopolis watershed has very little relief; 54% of the watershed has a 0-2% slope, 29% has a 2-5% slope, and only 17% of the watershed has a slope greater than 5%.

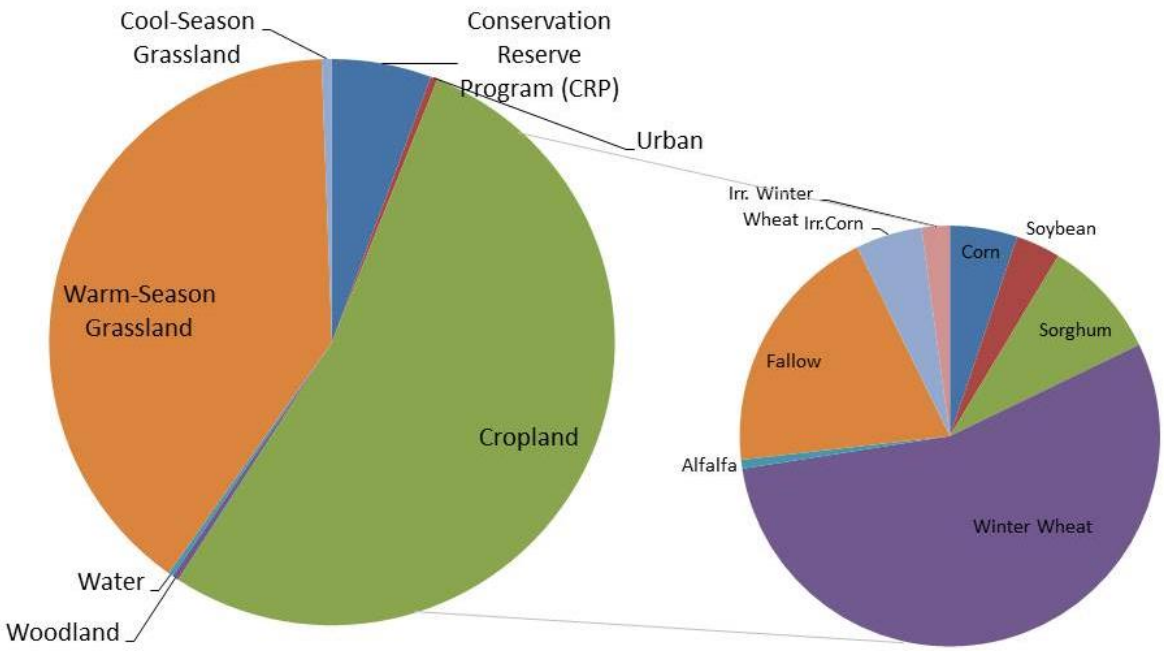

Figure 2-3: Kanopolis lake watershed land-use; major land-use types in the left pie chart with cropland broken out into more specific types in the pie chart on the right. (Irr.: irrigated)

The watershed drains the Smoky Hill River into Cedar Bluff Lake, located in the central portion of the watershed and then ultimately into Kanopolis Lake in Ellsworth county at the outlet of the watershed. Both reservoirs are operated and maintained by the Army Corps of Engineers. Cedar Bluff Lake was finished in 1950 with the purpose of providing irrigation, flood control, and water supply (KWO 2011b). However, low flows into the reservoir have limited the possible uses and the irrigation district was dissolved. Kanopolis Lake was finished in 1948 with the purposes of flood control, irrigation, recreation, fish and wildlife, downstream low flow augmentation, and water supply (KWO 2011a). Groundwater is also an important water source in this watershed. Alluvial and groundwater pumping associated with public water supply and agricultural practices create complex surface and groundwater interactions (Sophocleous and Wilson 2000).

Differences in Study Sites

The Perry Lake watershed and the Kanopolis Lake watershed vary in several key ways. The Kanopolis Lake watershed is an order of magnitude larger than the Perry Lake watershed and both watersheds are located in completely different ecoregions. The mean annual precipitation in the Perry Lake watershed is almost twice that of locations in the western portion of the Kanopolis Lake watershed. Consequently, water intensive land-use types are more common in the Perry Lake watershed, such as hay, corn, and soybeans. In addition, the Perry watershed has less than 1% of cropland with irrigation, while about 8% of cropland in the Kanopolis Lake watershed is irrigated. The Kanopolis Lake watershed has very little relief and most soil types have moderate infiltration, in addition groundwater-surface water interactions are common in the watershed (Sophocleous and Wilson 2000). Whereas in the Perry Lake watershed the dominant slope class is greater than 5% and the highest percentage of soils have high runoff potential with little to no groundwater interaction.

The two study watersheds are considerably different, which allows for interesting comparisons. First, the difference in location and mean annual precipitation allow for comparisons across the Kansas longitudinal climate gradient. The differences in climate are also represented in the dominant land-use types in each watershed. Comparing SWAT performance in the two watersheds can provide helpful information on model capabilities in simulating yield and streamflow in areas with low annual average precipitation and high degrees of groundwater interaction.

Table 2-1: Comparison of watershed characteristics in both the Perry Lake and Kanopolis Lake watersheds

Characteristic	**Perry Lake Watershed**	**Kanopolis Lake Watershed**
Size	2,924 km2	20,291 km2
Ecoregion	Central Irregular Plains and the Western Corn Belt Plains	Central Great Plains and the Western High Plains
Dominant Land-use	Hay	Rangeland
Dominant Crop Type	Corn-Soybean Rotation	Winter Wheat
Mean Annual Precipitation	890 – 980 mm	483 – 711mm
Percent of land irrigated	<0.05% (0.17% of cropland)	4.4% (8.3% of cropland)
Dominant Slope Class	>5%	0-2%
Dominant Soil Group	Group D (High runoff potential)	Group B (Moderate infiltration)

2.2.2 SWAT Model Description

SWAT is a continuous-time, spatially distributed simulator of the hydrologic cycle and agricultural pollutant processes and transport. Major model input components include climate conditions, soil properties, topography, plant growth, and land management. Model outputs include subbasin flow and loads of nutrients, sediment, pesticides, bacteria, and pathogens (Gassman et al. 2007; Ficklin et al. 2009). SWAT automatically distributes the main watershed into subwatersheds or subbasins, based on the placement of watershed outlets. Subbasins are then further divided into hydrologic response units (HRUs), which are characterized as units of homogeneous soil properties, land-use and slope (Ficklin et al. 2009; Gassman et al. 2007). In this study the SWAT version 2010-beta was used (available at: swat.tamu.edu).

SWAT is utilized for a full range of basin sizes – from small watersheds to large river basins. Several studies have examined the influence of scale and watershed subdivision on SWAT model calibration and sensitivity (Jha et al. 2004; Heathman and Larose 2007; Thampi et al. 2010). In general, larger watersheds tend to have greater uncertainty in modeling results, and smaller watersheds tend to generate predictions with greater accuracy (Thampi et al. 2010; Heathman and Larose 2007). However, larger watersheds such as the Upper Mississippi River Basin, have also been modeled successfully with evaluation statistics demonstrating reasonable accuracy (Jha et al. 2006; Srinivasan et al. 2010). SWAT has been used worldwide for a variety of environmental, hydrologic, and agricultural applications; for example: climate change sensitivity analysis, field-level targeting of agricultural best management practices, impact of sediment control structures, and water quality impacts of switchgrass production (Douglas-Mankin et al. 2010; Gassman et al. 2007; Jha et al. 2007; Mishra et al. 2007; Romanowicz et al. 2005; Srinivasan et al. 2010). SWAT has also been used in many Kansas studies from the field-level to the watershed scale (Daggupati et al. 2011; Nelson et al. 2006; Sheshukov et al. 2012; Sheshukov et al. 2011).

Sediment loads are estimated as a function of erosion from the landscape, as well as channel erosion or deposition. Within the landscape component, erosion and sediment yield are estimated for each HRU using the Modified Universal Soil Loss Equation (MUSLE) (Williams 1975). Sediment transport is modeled considering both deposition and degradation, which are estimated as a function of stream power, exposure of channel sides, and composition of channel banks and bed sediment. Degradation is estimated by the Simplified Bagnold Equation, where the maximum amount of sediment that can be transported is a function of peak channel velocity (Neitsch et al. 2011).

Nutrient cycles in SWAT are similar to those of the Erosion-Productivity Impact Calculator or EPIC model (Williams 1990). Nutrient inputs to the system include natural sources, such as wet and dry atmospheric deposition, nitrogen fixation, and organic matter mineralization, as well as anthropogenic inputs, such as fertilizer application, crop residue, animal waste, and wastewater discharges. Biochemical transformations of both nitrogen and phosphorus through mineralization, decomposition, and immobilization are all estimated with in the model. Sorption of inorganic P is also considered, by assuming a rapid equilibrium between solution P and the active mineral pool, then a slow equilibrium between the active and stable mineral pools. Nitrogen losses are simulated through plant uptake, denitrification, ammonia volatilization, and leaching of nitrate in surface runoff and lateral flow. Phosphorus losses are simulated through plant uptake, erosion, and run off (Neitsch et al. 2011). In-stream nutrient processes are simulated using equations from the model, QUAL2E (Brown and Barnwell 1987). All of the land-based and in-stream nutrient processing was estimated in SWAT using default values that rely on watershed specific inputs such as soil properties, land-use, and user-defined anthropogenic nutrient inputs (i.e. fertilizer applications and wastewater discharge). Watershed inputs will be defined further in the following section, model parameterization.

2.2.3 Model Parameterization

Land-use, Soil and Slope

To delineate hydrologic response units (HRUs), information on land-use, soil, and slope are necessary. For both watersheds, the 2005 Kansas Level IV Land Cover Patterns map was used to parameterize land-use within the model (Martinko et al. 2010) (available at http://kars.ku.edu/). The Kansas Level IV map was developed using multi-seasonal Landsat Thematic Mapper imagery from the 2004 and 2005 growing season to map both cool- and warm-season grasses, and MODIS NDVI time-series imagery from 2005 was used to map cropland. Irrigation status of the main crop types was also determined using the MODIS NDVI time-series (Martinko et al. 2010). New SWAT land-use subclasses were created to represent the irrigated crop types, such as irrigated corn (IRCN), soy (IRSB), winter wheat (IRWW), sorghum (IRSG), and alfalfa (IRAL). These subclasses were used to delineate irrigated HRUs, and then to apply appropriate irrigation management routines. Soil classes were represented by the STATSGO database provided within the SWAT model, and slope was determined using a 30-meter digital elevation model (DEM) (USDA 1997; Gesch et al. 2002). Both STATSGO and DEM databases were downloaded from the USDA Geospatial Data Gateway (gdg.sc.egov.usda.gov). Three slope classes were used to delineate HRUs: 0-2%, 2-5%, >5%. Overlapping land-use, soil and slope resulted in a total of 3,839 unique HRUs in the Perry Lake watershed and 14,353 HRUs in the Kanopolis Lake watershed.

Fertilizer and Management Practices

Dominant crop rotations in each watershed were determined using the USDA Cropland Data Layer (CDL) from 2006, 2007 and 2008 (data downloaded from: http://datagateway.nrcs.usda.gov/). In the Perry Lake watershed, dominant crop rotations were continuous non-irrigated corn, continuous non-irrigated soy, and non-irrigated corn – soybean rotation. In the Kanopolis Lake watershed, dominant crop rotations were non-irrigated corn – winter wheat, non-irrigated sorghum – winter wheat, continuous winter wheat, continuous irrigated sorghum and continuous irrigated corn. For each crop rotation, corresponding management practices related to fertilizer application rates, tillage, and planting/harvesting dates were developed based on guidelines provided through personal communication with Dr. Nathan Nelson or Kansas State Extension materials. Dr. Nelson is an Associate Professor in the

Department of Agronomy at Kansas State University and specializes in soil fertility and nutrient management. He has experience conducting agricultural research at the field-, lab-, and small plot-scale, in addition to watershed modeling with SWAT.

Applications of nitrogen (lbs) and P_2O_5 (lbs) per bushel were estimated using recommended rates per acre corresponding to a bushel yield goal, assuming an average soil organic matter content of 2.5% (Leikam, Lamond, and Mengel 2003). Then, National Agricultural Statistics Service (NASS) county-level yield averages from 2005 – 2010 (+10% to account for fertilizer losses or undershooting yield goals) were calculated and then used with application rates per bushel to determine nutrient application rates per acre for both irrigated and non-irrigated corn, soybeans, sorghum, and winter wheat. Within SWAT, 30% of fertilizer was applied to the top 10 mm of soil; the remaining fertilizer was applied below the surface at the time of planting. For winter wheat, however, only one third of the nitrogen was applied at planting, and the remaining application was scheduled for the January following planting (see Appendix A). Auto-irrigation management was applied at an efficiency of 0.7 to land-use classes identified as irrigated cropland when the plant stress was around 0.9 (on a 0-1 scale). A calendar-based management scheme was developed, which included user-defined dates for tillage, planting, fertilizer application and harvesting (specific dates for each watershed can be found in Appendix A). Potential evapotranspiration was calculated using the Penman/Monteith equation and the SWAT weather generator was used to generate required inputs such as solar radiation, relative humidity, and wind speed (Neitsch et al. 2011).

Climate

Daily precipitation and minimum and maximum temperature data were obtained from the National Climatic Data Center (NCDC) Global Historical Climatology Network (GHCN) for the period from 1975 to 2011 (Data available at: http://www.ncdc.noaa.gov/cdo-web/). Records were screened for completeness and those with extended periods without data were excluded. An Excel-based macro was used to expedite the processing and formatting of the GHCN precipitation and temperature time series for SWAT compatibility. The weather generator within SWAT was used to generate any missing values to complete the time series, as well as relative humidity, solar radiation and wind speed for the entire time series, which are required for the Penman/Monteith equation.

Point Source Inputs

Data on nutrient loads and discharges from municipal and industrial point sources were compiled from the Kansas Department of Health and Environment (KDHE) data records (compiled from a request to the office), the EPA Clean Water Act DMR Pollutant Loading Online Tool, and the EPA Clean Watersheds Needs Survey online database (EPA 2012b, 2012a). Only records with nitrogen (ammonia, nitrate, nitrite, organic nitrogen, kjeldahl, or total nitrogen), phosphorus (phosphate or total phosphorus), or sediment (solids) data were compiled. Average nutrient and solid concentrations were either calculated from the KDHE reported data, or from the EPA Clean Water Act Pollutant Loading Online Tool. Estimated annual flows were reported in the EPA Clean Watersheds online database. Average concentrations and estimated annual flows were used to calculate long-term pollutant loads for nitrate, nitrite, organic nitrogen, total phosphorus and solids. Pollutant loads were aggregated at the subbasin level within each watershed. In addition, many wastewater treatment plants (WWTP) did not report phosphorus concentrations, as it is not required for many NPDES permits. In order to estimate an approximate phosphorus loading for WWTPs, an N:P ratio of 6 was used to estimate phosphorus loads from the available nitrogen data. This N:P ratio was based on calculated values from the Lawrence Wastewater Treatment Plant data (Sturm et al. 2012).

Reservoir Parameters and Outflow

Reservoir structural information was obtained from the Bureau of Reclamation (Bureau of Reclamation 2012) and daily outflow records from 1975 to 2012 were obtained from the U.S. Army Corps of Engineers (USACE) through an open records request. Structural information includes volume and surface area of the reservoir at the emergency spillway (maximum operating conditions) and principal spillway (normal operating conditions). In addition, the initial sediment concentration and the equilibrium sediment concentration in the reservoir were required. Reservoir water quality and suspended sediment data was determined from KDHE lake monitoring records representing the period from 1975-2007 (obtained from Ed Carney at KDHE).

2.2.4 Calibration and Validation

During model calibration, parameters are adjusted within an acceptable range to determine a set of parameters that achieve best performance between observed and simulated values. Perry and Kanopolis watersheds were calibrated for streamflow and crop yield, and Perry was calibrated for stream sediment load as well. Flow was calibrated first, followed by crop yield, and then sediment. Ten flow parameters, four crop parameters and six sediment parameters were selected for calibration and are ranges tested for each parameter are listed in

Table 2-3 and Table 2-4 for Perry Lake watershed and Table 2-5 for Kanopolis Lake watershed (Sinnathamby 2014). Three quantitative statistics were used to evaluate performance: Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and the ratio of the root mean square error to the standard deviation of measured data (RSR). NSE is a dimensionless model evaluation statistic that determines the noise to information ratio by comparing the magnitude of residual variance to the measured data variance. It also demonstrates how well the plot of observed vs. simulated values match the 1:1 trendline, with a value of 1 being the optimal value. NSE ranges to -∞, but anything <0.0 is typically unacceptable, as it indicates that the mean observed value is a better predictor for each observation than the simulated value. For calibration and validation with SWAT, NSE values >0.5 are considered satisfactory at the monthly time scale (Moriasi et al. 2007).

Equation 1. $$NSE = 1 - \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)^2}{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{mean}\right)^2}\right]$$

PBIAS is an error index statistic that demonstrates the average tendency of simulated data to be larger or smaller than observed measurements. Positive PBIAS values indicate model underestimation and negative PBIAS values indicate overestimation. The optimal PBIAS value is 0.0. At the monthly time step, a PBIAS less than 25% for streamflow after calibration and a PBIAS less than 55% for sediment predictions after calibration are considered satisfactory (Moriasi et al. 2007).

Equation 2. $$PBIAS = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right) \times 100}{\sum_{i=1}^{n} Y_i^{obs}}\right]$$

RSR standardizes a commonly used error index statistic, the root mean square error (RMSE), by dividing by the standard deviation of measured data. RSR varies from 0.0 to large positive values; 0.0 is the optimal value indicating zero RMSE or perfect model simulation. For SWAT modeling at the monthly time step, RSR values greater than 0.70 are considered unsatisfactory (Moriasi et al. 2007). Performance ratings of the three recommended statics, NSE, PBIAS, and RSR, were determined by a group of hydrologic modelers, who compiled ratings for calibration and validation from many studies (Moriasi et al. 2007). A summary of these performance ratings at the monthly time step are provided in Table 2. In addition, hydrographs (Figures 4 – 15) are also used to visually analyze model performance.

Equation 3. $$RSR = \frac{RMSE}{STDEV_{obs}} = \left[\frac{\sqrt{\sum_{i=1}^{n} \left(Y_i^{obs} - Y_i^{sim} \right)^2}}{\sqrt{\sum_{i=1}^{n} \left(Y_i^{obs} - Y_i^{mean} \right)^2}} \right]$$

Table 2-2: Performance ratings for the SWAT model as determined by Moriasi et al. for recommended statistics on the monthly time step

Performance Rating	NSE	RSR	PBIAS - Streamflow	PBIAS - Sediment
Very Good	0.75 – 1.0	0.0 – 0.50	< ±10	< ±15
Good	0.65 – 0.75	0.50 – 0.60	±10 - ±15	±15 - ±30
Satisfactory	0.50 – 0.65	0.60 – 0.70	±15 - ±25	±30 - ±55
Unsatisfactory	< 0.50	> 0.70	> ±25	> ±55

Flow was calibrated and validated using United States Geological Survey (USGS) streamflow gauges at two locations in Perry Lake watershed (Delaware R NR Muscotah [USGS 06890100] and Delaware R at Perry Dam [USGS 06890900]), and at 4 locations in Kanopolis Lake watershed (Big Creek NR Hays [USGS 06863500], Smoky Hill R BL Schoenchen [USGS 06862850], Smoky Hill R NR Bunker Hill [USGS 06864050], and Smoky Hill R Ellsworth

[USGS 06864500]) (United States Geological Survey 2014b). Discharge data from 1978-1996 was used for calibration at all stations except Smoky Hill R BL Schoenchen, which only had data beginning from 01-10-1981. Data from 1997-2011 was used for validation at all stations. A three-year period from 1975-1977 was used to warm up the model.

For sediment calibration in the Perry Lake watershed, daily computed data for the Delaware R NR Muscotah stream gauge location was provided by the USGS through the USGS Kansas Real-Time Water Quality online database (United States Geological Survey 2014a). Daily predicted data was developed by the USGS using suspended sediment samples (n=181) and daily streamflow measurements that were collected between the years 2000 - 2002 to develop a regression equation to predict suspended sediment values based on streamflow measurements ($\log_{10}SSC = 1.270 + 0.257\log_{10}Q + 0.116(\log_{10}Q)^2$; $r^2 = 0.68$ and mean square error = 0.260 in log units; SSC: suspended sediment concentration in mg/L and Q: discharge in cubic feet per second) (Putnam and Pope 2003). The USGS-developed database is the most continuous record of suspended sediment data and provides the closest estimate of suspended sediment concentrations within the watershed.

Table 2-3: Perry Lake watershed streamflow and crop calibration parameters (Adapted from Sinnathamby 2014)

Parameters	Definition	Default value	Tested range		Magnitude of tested value	Final value
Flow						
ICN	Daily curve number calculation method	Antecedent soil moisture condition			0 or 1	Plant evapotranspiration
CNCOEF	Plant ET CN Coefficient	1	0.5	-1.5	±0.1	1.3
CN2.mgt	SCS runoff curve number for moisture condition 2	35-98	-15% ×CN2[1]	15% ×CN2[1]	±1%	-5% ×CN2 for all sub-watersheds above Perry at Delaware +12% ×CN2 for all sub-watersheds below Perry at Delaware
ESCO.hru	Plant evaporation compensation factor	0.95	0	1	±0.05	0.6 all HRU
SURLAG.bsn	Surface runoff lag coefficient	4	0	10	±1	1
ALPHA_BF	Baseflow alpha factor (days)	0.048	0	1	±0.001	0.10 all HRU
GW_DELAY	Groundwater delay (days)	31	0	500	±5	0 all HRU
RCHRG_DP	Aquifer fraction coef.	0.05	0	0.5	±0.01	0
CANMX	Maximum canopy storage	0	---	---	---	Agriculture 3 Forests 8 Urban 1.5
Crop						
BIO_E	Biomass-energy ratio *Corn* *Soybean*	 39 25	 40 28	 25 20	 ±1 ±1	 35 20
HVSTI	Harvest index *Corn* *Soybean*	 0.50 0.31	 0.6 0.3	 0.2 0.2	 ±0.01 ±0.01	 0.46 0.31
WYHI	Lower limit of harvest index *Corn* *Soybean*	 0.3 0.01	 0.35 0.3	 0.4 0.01	 ±0.01 ±0.01	 0.35 0.20
LAI	Leaf area index *Corn* *Soybean*	 5 3	 4 2	 6 5	 ±0.5 ±0.5	 5 2

Table 2-4: Perry Lake watershed sediment load calibration parameters

Sediment	Definition	Default value	Tested range		Magnitude of tested value	Final value
SPCON	Linear parameter for channel sediment routing	0.0001	0.005	0.01	±0.0001	0.008
SPEXP	Exponent parameter for channel sediment routing	1.0	1	1.75	±0.001	1.43
CH_COV1	Channel erodibility factor	0 /1*	0.50	1.5	±0.001	0.62
CH_COV2	Channel erodibility factor	0/1*	0.50	1.5	±0.001	0.62
USLE_P	USLE support practice factor	0	0.5	1	+0.001	0.86
CH_N(2)	Manning's "n" value	0.014	0.02	0.06	±0.001	0.05

Non-irrigated corn and soybean crop yields from Jackson and Brown counties were used to calibrate corn and soybean parameters in Perry Lake watershed, and non-irrigated grain sorghum and winter wheat crop yields from Wallace and Trego counties were used to calibrate grain sorghum and winter wheat parameters in Kanopolis Lake watershed, as these are the dominant crop types in the respective watersheds. Calibration was performed with data from 1996-2009 from the National Agricultural Statistics Service (NASS). For validation purposes NASS-reported Nemaha county corn and soybean crop yields were used to test for the accuracy of yield simulation in Perry Lake watershed, and NASS-reported Ellis county winter wheat and grain sorghum crop yields were used to test for the accuracy of yield simulation in for Kanopolis watershed (Sinnathamby 2014). The validation period was also 1996 – 2009. Different counties were used for validation purposes to determine if the calibrated model could accurately estimate yield in a nearby county within the respective watershed.

Table 2-5: Kanopolis Lake watershed flow and crop calibration parameters (Adapted from Sinnathamby 2014)

Parameters	Definition	Default value	Tested range		Magnitude of tested value	Final value
Flow						
CN2.mgt	SCS runoff curve number for moisture condition 2	35-98	-20% ×CN2[1]	20% ×CN2[1]	±1%	-20% ×CN2 for all sub-watersheds
ESCO.hru	Soil evaporation compensation factor	0.95	0	1	±0.05	0.5 for all HRUs above Big C NR Hays 0.8 all other HRUs
EPCO	Plant evaporation compensation factor	1	0	1	±0.01	0.01 all HRU
SURLAG.bsn	Surface runoff lag coefficient	4	0	10	±1	2
ALPHA_BF.gw	Baseflow alpha factor (days)	0.048	0	1	±0.001	0.001 all HRU
GW_DELAY.gw	Groundwater delay (days)		0		±5	0 all HRU
RCHRG_DP	Aquifer fraction coefficient	0.05	0	0.5	±0.01	0.75 all HRU
CANMX	Maximum canopy storage	0				Agriculture 3 Forests 8 Urban 1.5
Crop						
BIO_E	Biomass-energy ratio *Winter Wheat* *Grain Sorghum*	30 33.5	20 30	30 37	±1 ±1	30 36.5
HVSTI	Harvest index *Winter Wheat* *Grain Sorghum*	0.4 0.45	0.3 0.4	0.4 0.46	±0.01 ±0.01	0.41 0.46
WYHI	Lower limit of harvest index *Winter Wheat* *Grain Sorghum*	0.3 0.25	0.2 0.25	0.3 0.36	±0.01 ±0.01	0.3 0.4
LAI	Leaf area index *Winter Wheat* *Grain Sorghum*	4 3	4 3	5 5	±0.5 ±0.5	4 5

2.3 Calibration and Validation Results

2.3.1 Streamflow Calibration

Perry Lake Watershed

SWAT-predicted annual average daily streamflow values match the observed time series at both USGS stream gages: Delaware River near Muscotah and Delaware River at Perry Lake. Annual statistics for the Muscotah location are very good, with an NSE value of 0.80 for the calibration period (1978-1996) and 0.79 for the validation period (1997-2011). Annual statistics for the Perry Lake location are excellent with an NSE value of 0.99 for both the calibration and validation periods. Monthly and daily SWAT-predicted average streamflow values have a very good match at the Delaware River near Muscotah location, with NSE values between 0.65 - 0.84, and also at the Delaware River at Perry Lake location, with NSE values between 0.87 – 0.99. Calibration and validation statistics in Table 4 show that the SWAT model estimated streamflow with a high degree of accuracy before calibration (default values in table), but that calibration improved estimates at the monthly and daily time scales. Figures 4 – 7 demonstrate a time series of observed and SWAT-predicted annual and monthly streamflow, which further validate the model's ability to represent streamflow processes in the Perry Lake watershed. A more detailed study of model results indicate that simulated surface flow and baseflow were also well predicted (Sinnathamby 2014).

Table 2-6: Streamflow calibrated statistics for Perry watershed at two locations before and after calibration, respectively “default” and “final”.

	NSE		**PBIAS (%)**		**RSR**	
	Default	**Final**	**Default**	**Final**	**Default**	**Final**
Delaware River near Muscota						
Annual Calibration (1978-1996)	0.82	0.80	14.41	-3.75	0.43	0.19
Monthly Calibration	0.80	0.84	12.97	-15.66	0.44	0.26
Daily Calibration	0.37	0.65	-12.85	5.68	0.79	0.59
Annual Validation (1997-2011)	0.51	0.79	40.05	7.46	0.70	0.46
Monthly Validation	0.73	0.84	40.05	7.46	0.51	0.40
Daily Validation	0.15	0.74	25.84	0.13	0.92	0.51
Delaware River at Perry Lake						
Annual Calibration (1978-1996)	0.99	0.99	2.45	1.13	0.11	0.08
Monthly Calibration	0.98	0.98	5.7	4.70	0.13	0.09
Daily Calibration	0.91	0.90	5.04	1.44	0.31	0.32
Annual Validation (1997-2011)	0.99	0.99	7.22	1.14	0.10	0.10
Monthly Validation	0.99	0.99	4.73	3.96	0.10	0.10
Daily Validation	0.87	0.87	0.75	-1.75	0.36	0.36

Legend: NSE: Nash Sutcliffe Efficiency, PBIAS: Percent Bias, RSR: Ratio of the root mean square error to the standard deviation of measured data.

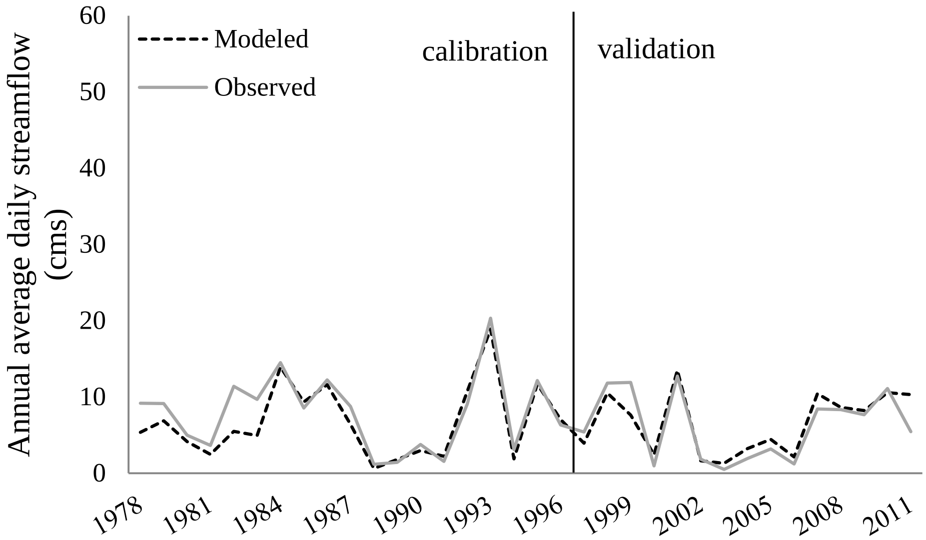

Figure 2-4: Calibrated and observed annual average daily streamflow at Delaware River near Muscotah for both calibration (1978-1996) and validation (1997-2011) time periods

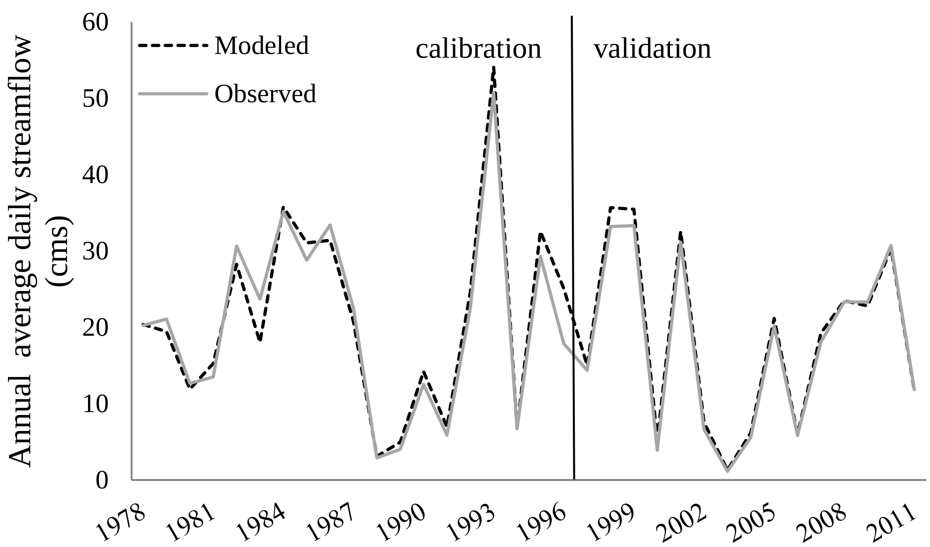

Figure 2-5: Calibrated and observed annual average daily streamflow at Delaware River at Perry Lake for both calibration (1978-1996) and validation (1997-2011) time periods

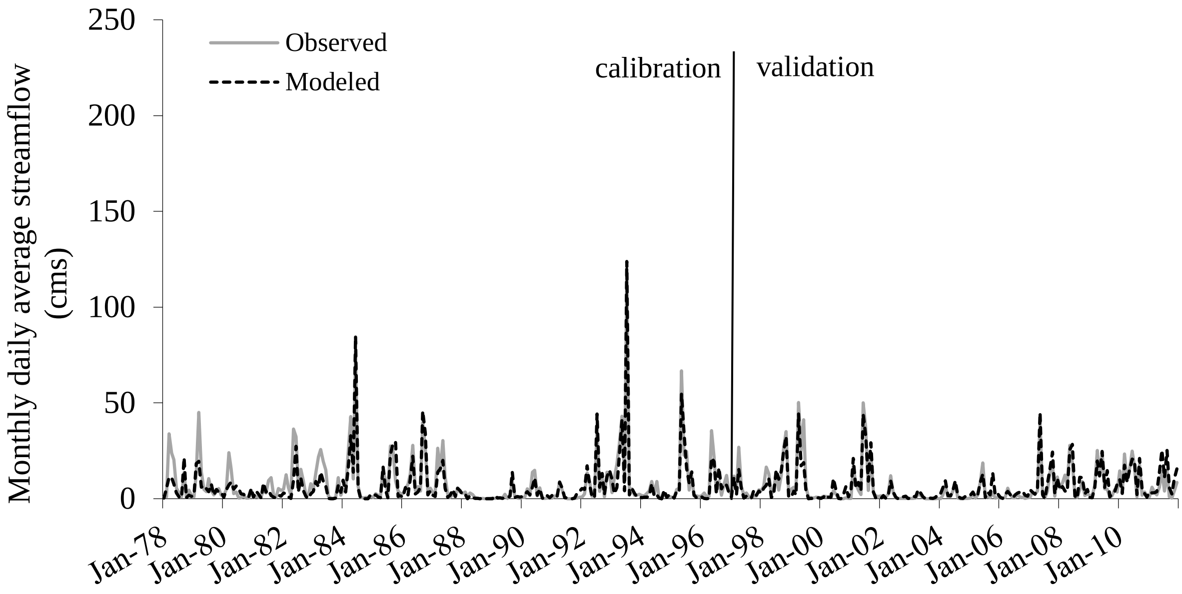

Figure 2-6: Calibrated and observed monthly average daily streamflow at Delaware River near Muscota for both calibration (1978-1996) and validation (1997-2011) time periods

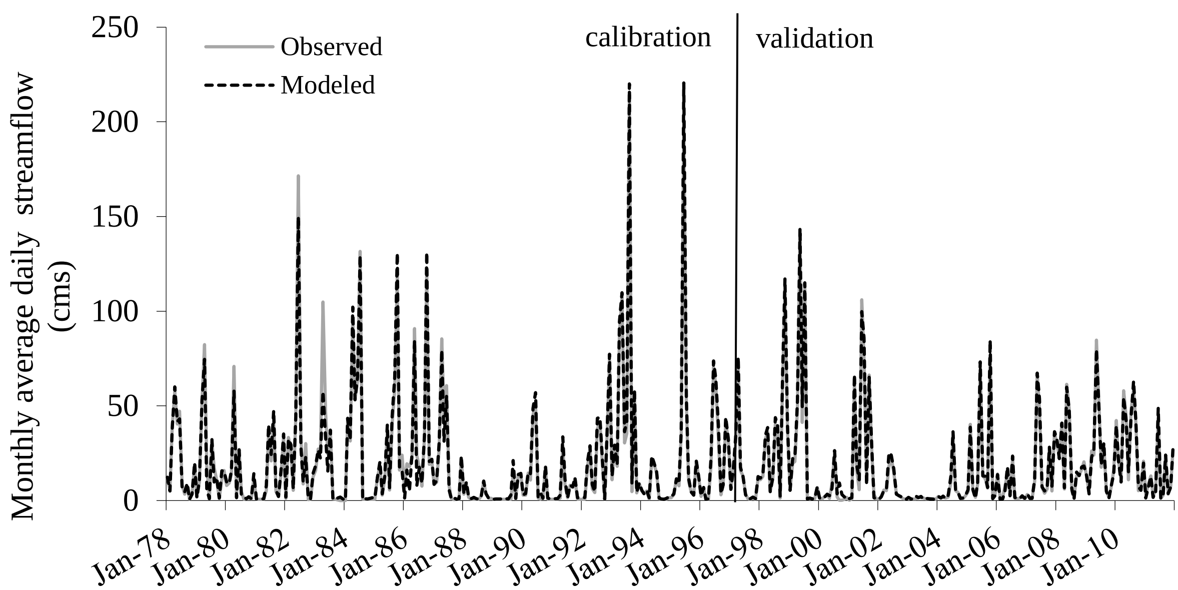

Figure 2-7: Calibrated and observed monthly average daily streamflow at Delaware River at Perry Lake for both calibration (1978-1996) and validation (1997-2011) time periods

Kanopolis Lake Watershed

Annual and monthly average daily predicted streamflow at the four evaluated locations in Kanopolis Lake watershed performed fairly well after calibration. Annual NSE statistics ranged from 0.49 – 0.85, and annual validation NSE statistics ranged from 0.31 – 0.66. Monthly NSE statistics for the calibration period ranged from 0.65 – 0.87; however, values were much lower for the validation period, ranging from 0.28 - 0.36. Daily calibration and validation statistics show that the model performed poorly at the daily scale with most NSE values less than 0. Figures 8 – 11 show annual observed and SWAT-estimated average streamflow. These figures show that SWAT does not predict well in years of high flow and low flow; however, it seems to simulate well in years of average flow. It is also clear in Figures 12 – 14 that SWAT underestimates average monthly flow especially in high flow months.

Calibration was more successful in the Perry Lake watershed model than in the Kanopolis Lake model. The Perry Lake watershed is the smaller of the two, receives more average annual precipitation and has very little groundwater – surface water interaction. However, the Kanopolis Lake watershed receives very little annual precipitation, which typically occurs in a few events, and potential evaporation exceeds available moisture, leading to a soil moisture deficit. Additionally, groundwater pumping for irrigation is more common in the Kanopolis watershed, and over time, groundwater pumping can cause streamflow magnitudes to decline (Sophocleous 1998). SWAT has a groundwater component, but it is not spatially distributed and therefore does not consider the spatial variability in hydraulic conductivity and recharge rates (Kim et al. 2008). Therefore, the ground water – surface water interactions are not well represented in the SWAT model, which is most likely why the Kanopolis Lake watershed model did not perform as well as the Perry Lake watershed model.

Table 2-7: Streamflow calibration statistics for Kanopolis Lake watershed at four locations

	NSE		PBIAS (%)		RSR	
	Default	**Final**	**Default**	**Final**	**Default**	**Final**
Big C NR Hays						
Annual Calibration (1978-1996)	-11.71	0.85	330.83	-27.20	3.56	0.39
Monthly Calibration	-3.98	0.87	330.83	-27.19	2.23	0.36
Daily Calibration	-15.57	-0.62	337.20	-27.90	4.00	1.79
Annual Validation (1997-2011)	.37.23	0.31	81.26	-10.43	6.18	0.83
Monthly Validation	-20.50	-0.17	414.06	-14.07	4.63	1.08
Daily Validation	-53.91	-8.25	317.48	0.10	7.4	3.04
Smoky Hill R BL Schoenchen						
Annual Calibration (1981-1996)	-8.44	0.83	445.04	-2.97	3.07	0.42
Monthly Calibration	-2.76	0.69	401.76	-0.91	1.94	0.56
Daily Calibration	-3.64	0.30	400.69	-2.90	2.15	0.83
Annual Validation (1997-2011)	-16.72	0.66	375.34	-0.38	4.21	0.58
Monthly Validation	-7.76	0.28	375.00	-1.26	2.96	0.85
Daily Validation	-0.19	-0.08	-72.87	-97.17	1.09	1.03
Smoky Hill R NR Bunker Hill						
Annual Calibration (1978-1996)	-3.44	0.60	180.38	-50.62	2.10	0.63
Monthly Calibration	-0.61	0.71	179.26	-50.40	1.27	0.54
Daily Calibration	-3.73	0.03	66.07	-44.45	2.17	0.8
Annual Validation (1997-2010)	20.85	0.36	283.37	-17.35	4.67	0.80
Monthly Validation	-7.69	0.32	275.00	-18.70	2.95	0.82
Daily Validation	-17.80	-1.57	63.38	-40.60	4.33	1.60
Smoky Hill R at Ellsworth						
Annual Calibration (1978-1996)	-1.60	0.49	141.19	-53.80	1.61	0.71
Monthly Calibration	-0.10	0.65	142.02	-51.19	1.05	0.59
Daily Calibration	-3.12	-0.15	152.59	-51.13	2.03	1.04
Annual Validation (1997-2010)	-6.68	0.34	-18.69	-15.75	4.44	0.81
Monthly Validation	-6.68	0.14	248.00	-21.00	2.77	0.92
Daily Validation	-6.07	-0.62	213.80	-57.34	2.66	1.27

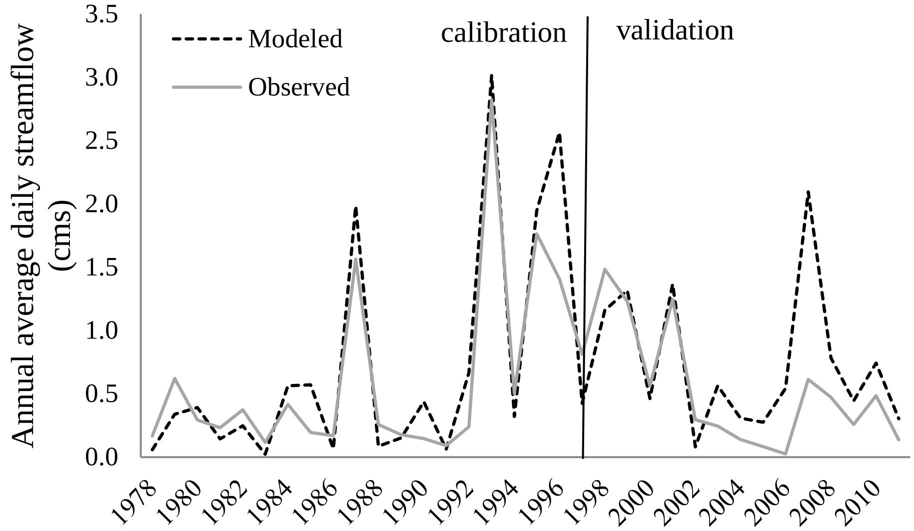

Figure 2-8: Calibrated and observed annual average daily streamflow at Big C NR Hays for both calibration (1978-1996) and validation (1997-2011) time periods

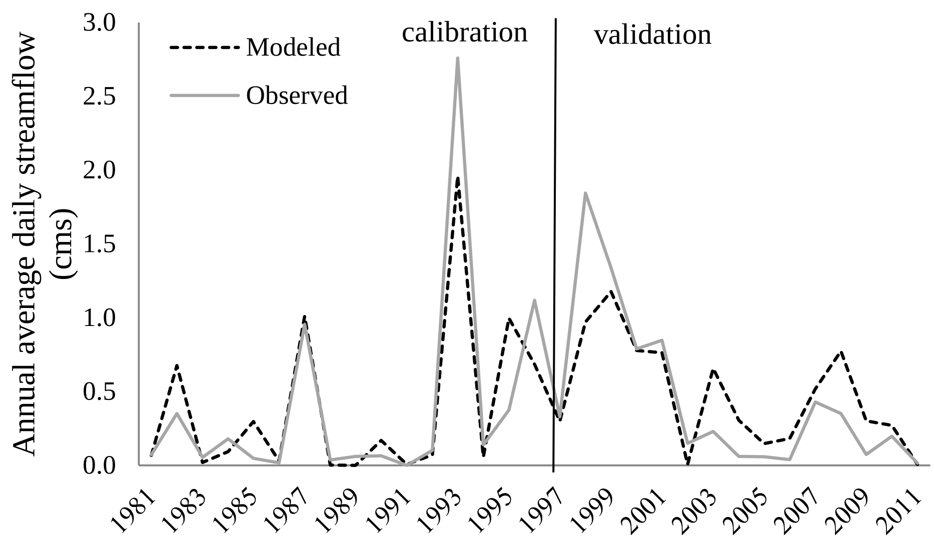

Figure 2-9: Calibrated and observed annual average daily streamflow at Smoky Hill R BL Schoenchen for both calibration (1981-1996) and validation (1997-2011) time periods

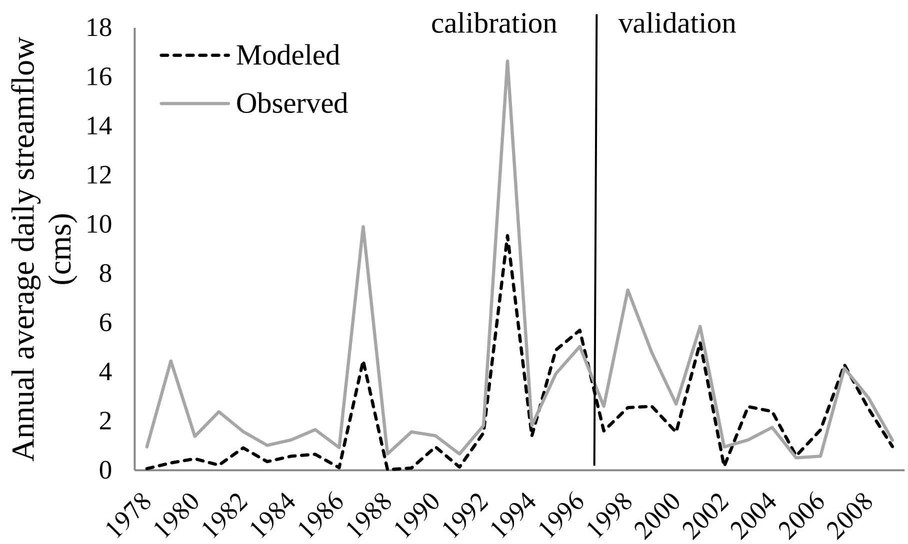

Figure 2-10: Calibrated and observed annual average daily streamflow at Smoky Hill R NR Bunker Hill for both calibration (1978-1996) and validation (1997-2010) time periods

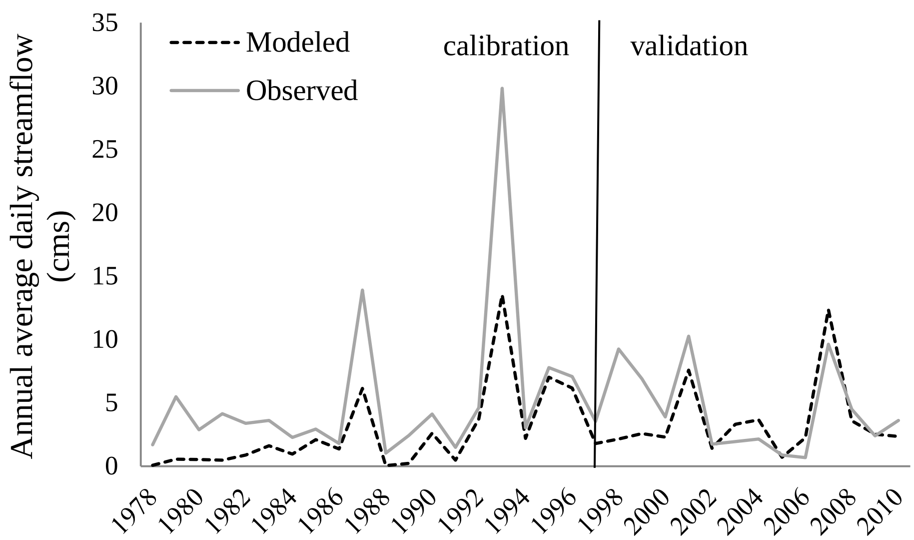

Figure 2-11: Calibrated and observed annual average daily streamflow at Smoky Hill R at Ellsworth for both calibration (1978-1996) and validation (1997-2010) time periods

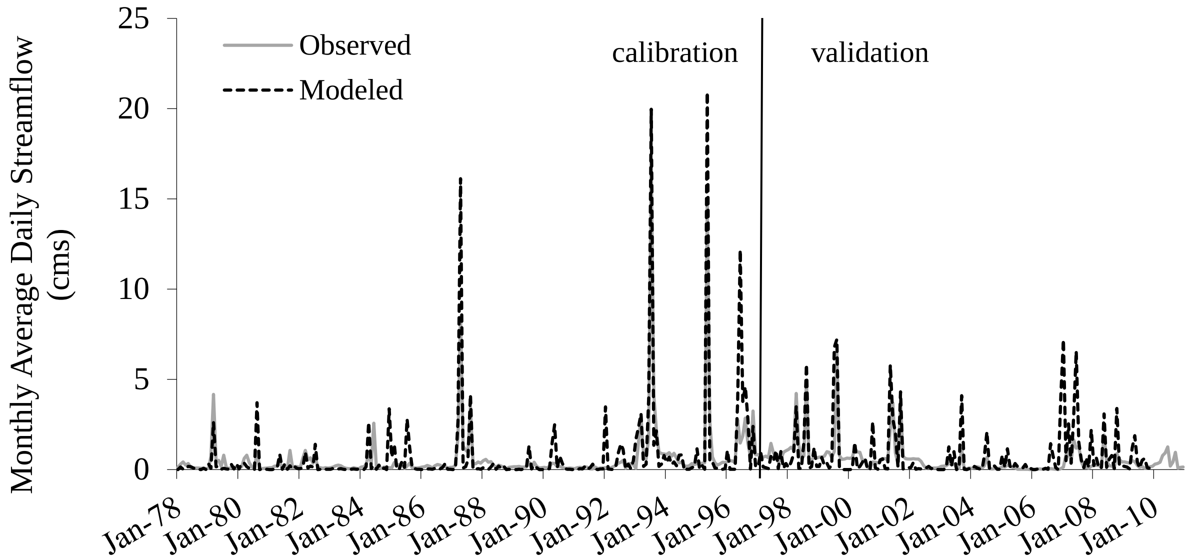

Figure 2-12: Calibrated and observed monthly average daily streamflow at Big C NR Hays for both calibration (1978-1996) and validation (1997-2010) time periods

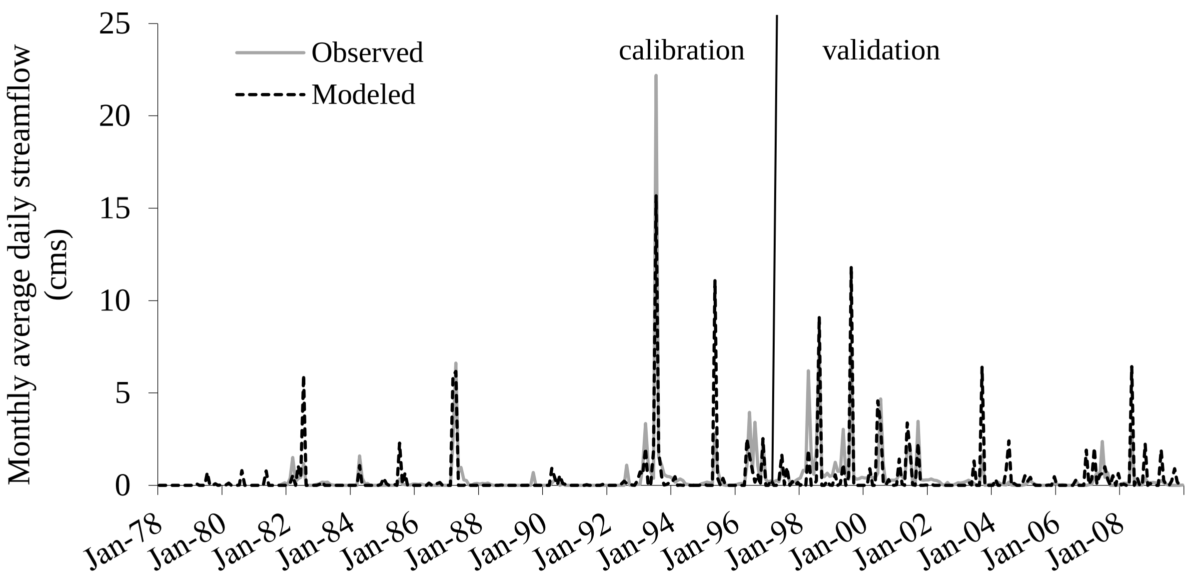

Figure 2-13: Calibrated and observed monthly average daily streamflow at Smoky Hill R BL Schoenchen for both calibration (1979-1996) and validation (1997-2009) time periods

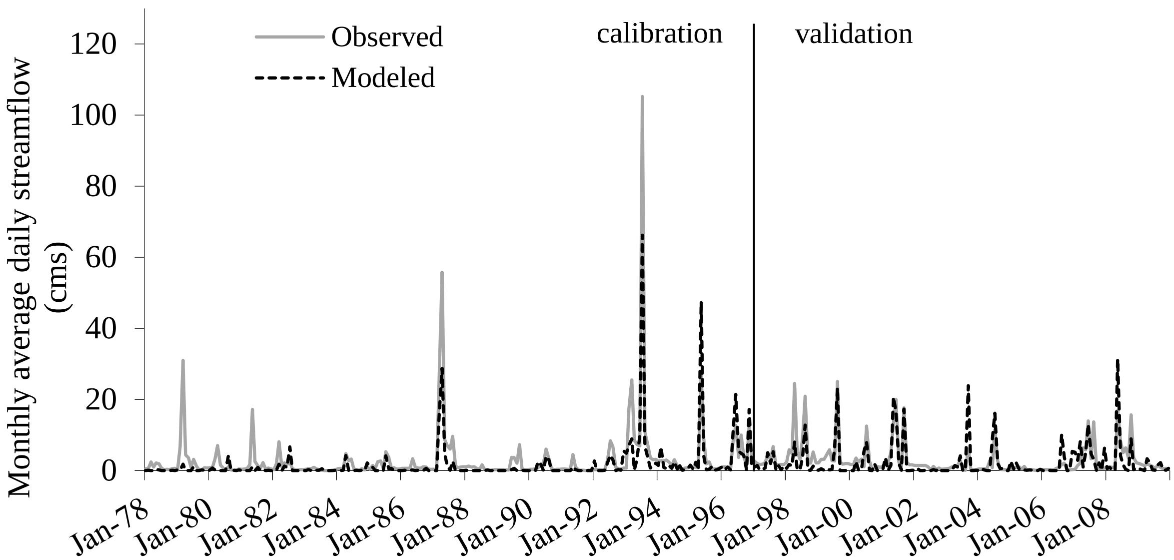

Figure 2-14: Calibrated and observed monthly average daily streamflow at Smoky Hill R NR Bunker Hill for both calibration (1978-1996) and validation (1997-2010) time periods

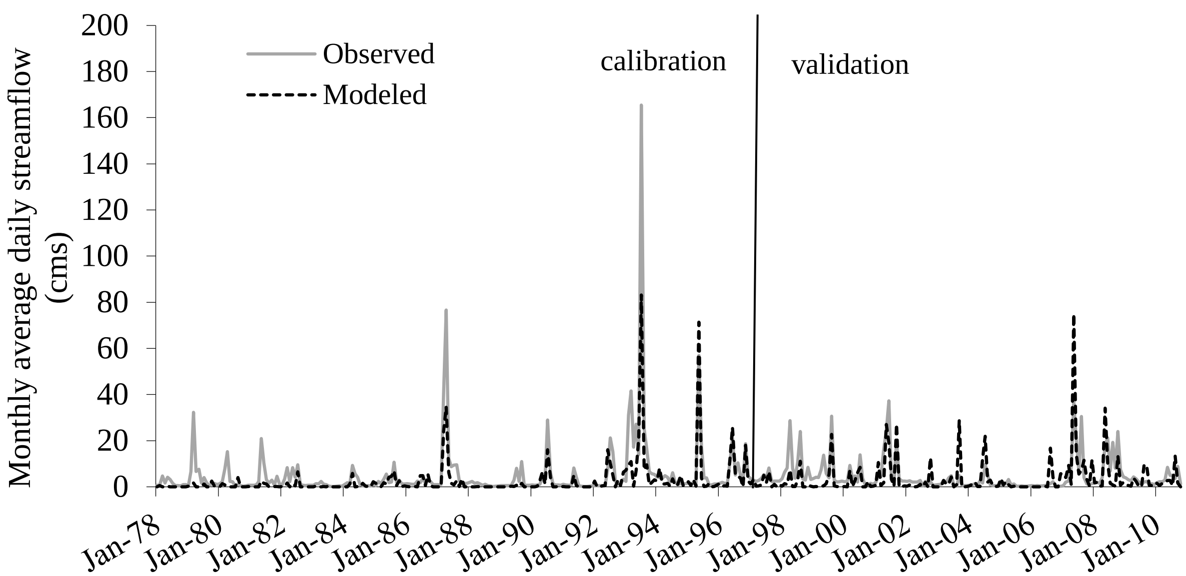

Figure 2-15: Calibrated and observed monthly average daily streamflow at Smoky Hill R at Ellsworth for both calibration (1978-1996) and validation (1997-2010) time periods.

2.3.2 Crop Calibration

SWAT successfully modeled corn yield for the three selected counties in the Perry Lake watershed. The best results were achieved in Jefferson county, with an NSE value equal to 0.83 and bias less than 1%. The validation county, Nemaha, also had very good results with an NSE value equal to 0.64 and a bias of 1.35%. SWAT did not do as well simulating soybean yield; the best results were in Jackson county with an NSE value equal to 0.55 and bias near -2%. However, in Nemaha county, the NSE value was negative and bias was near -11%. Overall, SWAT simulated corn and soybean yields with acceptable accuracy (NSE > 0.5 in most cases).

Table 2-8: Perry Lake watershed crop calibration statistics

County	N	Crop yield (t/ha)		NSE		PBIAS (%)		RSR	
		Reported	Modeled	Default	Final	Default	Final	Default	Final
Corn									
Jackson	13	5.49	5.56	-13.32	0.55	78.65	1.83	3.80	0.67
Jefferson	10	5.65	5.69	-9.18	0.83	90.33	0.83	3.05	0.42
Nemaha	13	5.54	5.61	2.48	0.64	62.41	1.35	1.87	0.60
Soybean									
Jackson	6	2.13	2.08	-3.46	0.55	30.94	-2.09	2.11	0.67
Jefferson	10	2.07	1.87	-1.11	0.37	27.68	-9.85	1.45	0.80
Nemaha	6	2.29	1.90	0.38	-0.57	3.50	-11.24	0.76	0.99

Legend: N: the number of years of observation for each county, NSE: Nash Sutcliffe Efficiency, PBIAS: Percent Bias, RSR: Ratio of the root mean square error to the standard deviation of measured data.

SWAT did not simulate winter wheat nor grain sorghum yields with a high degree of accuracy in the Kanopolis Lake watershed (most NSE < 0). NSE values for winter wheat yield simulations were negative in all three counties studied. However, bias was less than 5% in all cases. Grain sorghum was successfully modeled in Trego county (NSE=0.51), but SWAT did not achieve good results in Wallace (NSE= -0.41) and Ellis (NSE = 0.1) counties. However, in most cases SWAT predicted winter wheat and grain sorghum yields within 1 – 16% bias. It was challenging to predict yield accurately in the Kanopolis Lake watershed due to the dry climate with high rates of evapotranspiration.

Table 2-9: Kanopolis Lake watershed crop calibration statistics

County	N	Crop yield (t/ha)		NSE		PBIAS (%)		RSR	
		Reported	**Modeled**	**Default**	**Final**	**Default**	**Final**	**Default**	**Final**
Winter Wheat									
Wallace	13	1.73	1.44	-5.12	-0.53	38.28	-1.91	2.47	1.24
Trego	13	1.90	1.86	-17.60	-0.07	87.78	3.70	4.31	1.03
Ellis	13	2.00	2.05	-9.15	-0.67	40.18	4.56	3.19	1.29
Grain Sorghum									
Wallace	12	2.58	1.85	-0.52	-0.40	-36.68	-28.27	1.81	0.85
Trego	12	3.50	2.95	-1.51	0.51	-18.0	-15.51	2.10	1.43
Ellis	12	3.74	3.93	-0.03	0.1	9.4	5.04	1.99	1.05

Legend: N: the number of years of observation for each county, NSE: Nash Sutcliffe Efficiency, PBIAS: Percent Bias, RSR: Ratio of the root mean square error to the standard deviation of measured data.

2.3.3 Sediment Calibration

SWAT was successful simulating sediment in the Perry Lake watershed. The NSE was equal to 0.92 during annual calibration and 0.83 during monthly calibration. Two years were used for validation to allow for more calibration data to find optimal parameters, and of those, 2011 demonstrated poorly predicted streamflow. Therefore, sediment was also not predicted well in 2011. The NSE for monthly values in 2010 was good (NSE=0.62), but the overall 2010-2011 monthly NSE was negative (-0.19). Percent bias was -4.2% for annual calibration and -20% for monthly calibration, which are both satisfactory results. Overall, SWAT did well predicting annual sediment loads, but was not always successful matching the load during peak events or low flow periods (see Figure 2-16.) For example, in the two months with peak sediment load SWAT under predicted total load by 1 – 1.5 million tons.

Table 2-10: Sediment calibration statistics at Delaware River near Muscotah in the Perry Lake watershed

	NSE		**PBIAS (%)**		**r^2**	
	After parameterization	**Final**	**After parameterization**	**Final**	**After parameterization**	**Final**
Delaware River near Muscotah						
Annual Calibration (1999-2008)	0.30	0.92	121	-4.21	0.91	0.94
Monthly Calibration	0.46	0.83	55	-20.0	0.88	0.85
Monthly Validation 2010 (2011)	0.36	0.62 (-0.19)	25	43.5	0.41	0.76 (0.59)

Legend: NSE: Nash Sutcliffe Efficiency, PBIAS: Percent Bias, r^2: Coefficient of determination

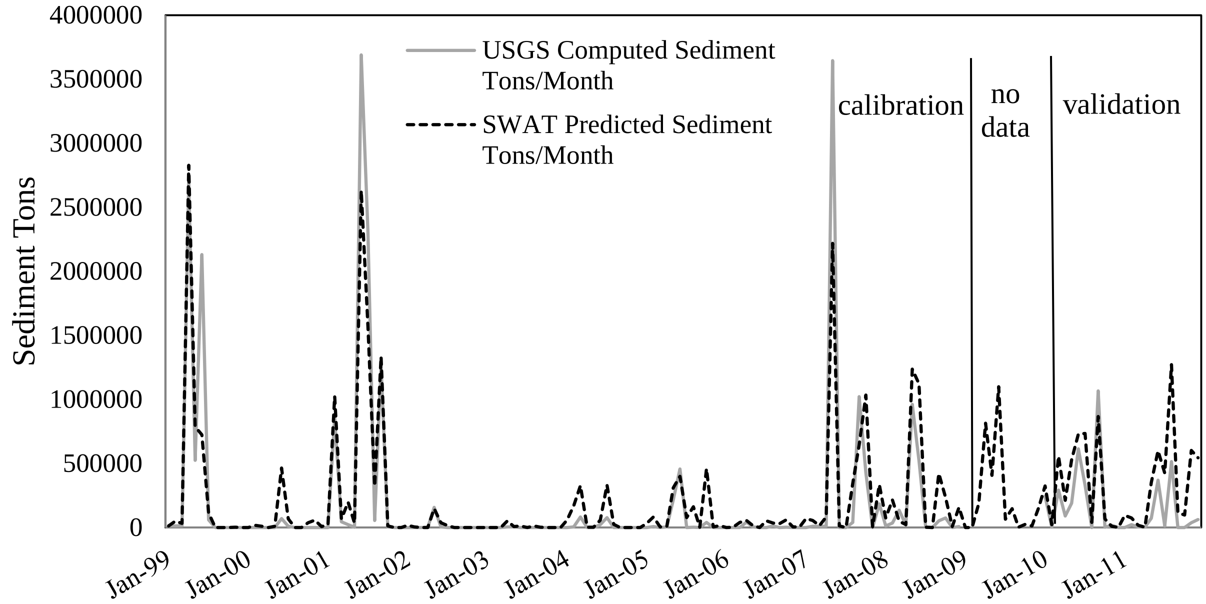

Figure 2-16: USGS-computed and SWAT-predicted calibrated suspended sediment load at the monthly time scale from January 1999 – July 2011 at the Delaware River near Muscotah station in the Perry Lake watershed.

2.4 Conclusion

SWAT was able to successfully simulate streamflow and dominant crop yields in two Kansas watersheds with different climate conditions. Additionally, sediment load was accurately predicted in the Perry Lake watershed over a wide range of hydrologic conditions. The SWAT model simulated corn and soybean yields with the greatest accuracy, but was least accurate at predicting winter wheat yields. Of the two different watersheds, SWAT was less accurate predicting hydrologic conditions in the Kanopolis watershed with a high level of groundwater-surface water interaction. However, SWAT predicted streamflow and sediment loads in the Perry Lake watershed with a high degree of accuracy. Model weaknesses in winter wheat yield simulation and groundwater-surface water interactions are related to model development and cannot be further improved through calibration of model parameters. Others have had success integrating physically-based, distributed groundwater models, such as MODFLOW, with SWAT for improved groundwater simulation (Kim et al. 2008); however, this was not part of the scope of this project. The overall performance was within the recommended metrics established by Moriasi et al., which provides the established criteria for evaluating model accuracy in watershed simulations, and also within the range of previously published studies (Moriasi et al. 2007; Douglas-Mankin et al. 2010). Overall, careful parameterization and calibration ensured two well-tuned models that can be used for a variety of agricultural and hydrological simulations in both eastern and western Kansas.

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Chapter 3 - Environmental Sustainability of Biofuel-Based Land-Use Change in Kansas

Abstract

The growth in ethanol production has sparked interest in potential land-use change and the associated environmental impacts that may occur in order to accommodate the increasing demand for grain feedstocks. In this study, water quality and sustainability indicators are used to evaluate the impacts of land-use change to increase corn and grain sorghum acreage for biofuel production in two Kansas watersheds: the Perry Lake watershed and the Kanopolis Lake watershed. Water quality indicators include total nitrogen, total phosphorus, and sediment loads per converted land acreage, and sustainability indicators include land-use, water use, and nutrient use efficiencies. Hay, CRP, and winter wheat were selected as targeted land-uses for conversion to biofuel feedstocks. The Soil and Water Assessment Tool was used to evaluate 12 different scenarios, each at 10 land-use change increments, for a total of 120 scenarios. Model simulations demonstrate that increased corn production generates significantly greater water quality impacts than increased grain sorghum production. Extensification of corn or grain sorghum cropland to hay or CRP land-uses resulted in the highest water quality impacts. Intensification of winter wheat cropland to either corn or grain sorghum produced the lowest water quality impacts. Corn had a higher yield potential per km^2 in the Perry Lake watershed resulting in better land, nutrient and water use efficiencies in comparison to grain sorghum. However, grain sorghum sustainability indicators increased in Western Kansas where annual average precipitation is lower. This study demonstrates that in dry climates grain sorghum is a more environmentally sustainable feedstock than corn.

3.1 Introduction

As of 2013, there were nearly 200 operating biorefineries in the United States, producing an estimated 13.3 billion gallons of ethanol per year (Renewable Fuels Association 2014). According to data from the US Department of Energy and the Renewable Fuels Association, ethanol production in the United States doubled within six years of passage of the 2007 Energy Independence and Security Act. This increase in ethanol production has displaced a volume of gasoline equivalent to the amount of crude oil imported annually from Venezuela and Iraq (Renewable Fuels Association 2014). The growth in ethanol production has sparked interest in potential land-use change (LUC) and the associated environmental impacts that may occur in order to accommodate the increasing demand for grain.

Many studies have focused on environmental changes in the Upper Midwest states, such as Iowa, Illinois, and Indiana, where the biofuel market is the strongest (Secchi, Gassman, et al. 2011; Secchi, Kurkalova, et al. 2011). These studies by Secchi et al. highlight the potential increase in sediment and nutrient non-point source pollution that may occur due to rising corn prices. Other studies have examined the impacts of corn and switchgrass production in North and South Dakota (Wu et al. 2012), corn stover removal in Indiana (Cibin et al. 2011), an array of biofuel feedstock rotations in Michigan (Love and Nejadhashemi 2011), and advanced cellulosic feedstock production in the Arkansas-White-Red river drainage basin (Jager et al. 2014). In general, these studies show that increased row-crop production for biofuel feedstocks results in increased non-point source nutrient pollution, but that replacing row-crop land-use with perennial feedstocks for cellulosic biofuel production shows the potential for improved water quality conditions. Overall, corn and switchgrass production have both been extensively studied in relation to water quality impacts. However, the water quality impacts of grain sorghum, a relevant biofuel feedstock in Kansas and other Great Plains states, have not been fully evaluated.

The studies available demonstrate that it is critical to evaluate the environmental impacts from biofuel feedstock production through location-dependent scenario analysis (Jager et al. 2014). However, a common set of sustainability indicators is necessary to measure and compare the impacts of biofuels on greenhouse gas emissions, soil fertility, water and air quality, biodiversity, and the global food system (Tilman et al. 2009; Hecht et al. 2009; National Research Council 2011). The indicators should be broadly applicable and allow for comparison across feedstocks and locations. Extensive research has been conducted on yield potentials, biomass to biofuel conversion factors, energy use, cost estimates, and water demands of specific crop types in order to compare which feedstocks may be the most energy, water, and cost efficient (Gelfand et al. 2010; Dominguez-Faus et al. 2009; Adler et al. 2006; Khanna 2008; Johnston et al. 2009). Further, a range of biofuel feedstocks have been evaluated using sustainability indicators based on average literature-reported values (de Vries et al. 2010; Scharlemann and Laurance 2008; Börjesson and Tufvesson 2011). Broad feedstock studies are critical for evaluating the biofuel market as a whole and for providing over-arching policy guidance and recommendations. However, decisions are often made on the state or local scale. Therefore it is critical to understand how sustainability indicators vary on a smaller scale relevant to local decisions.

In this study, environmental indicators are used to evaluate the impacts of LUC to increase corn and grain sorghum acreage for biofuel production in Kansas. Kansas is located in the periphery of the corn belt and the dominant region of US ethanol production, but is still ranked 9th in the US for total ethanol production (504 million gallons/year) and 7th for the total number of biorefineries (12 + 2 under construction) (Renewable Fuels Association 2014). Whereas some areas within the Corn Belt may soon be saturated with respect to ethanol biorefineries and available corn grain, Kansas remains an area with potential for expansion of ethanol production – especially with grain sorghum as a feedstock (Wang et al. 2008).

Ethanol production in Kansas uses predominantly corn and grain sorghum, with an often 50-50 mixture at biorefineries, but some have reported mixtures with up to 80% sorghum (Jessen 2010). Grain sorghum ethanol was approved by the United States Environmental Protection Agency (USEPA) as a renewable fuel under the Renewable Fuel Standards guidelines with 20% greenhouse gas (GHG) reduction if the biorefinery uses natural gas and a 50% GHG reduction, or advanced biofuel, if biogas is used. Approximately 30% of grain sorghum produced in the US is utilized for ethanol production, contributing about two percent to the total domestic ethanol production (Cai et al. 2013). Ethanol yield from sorghum grain is comparable to corn grain; however, a great deal of research has been done on maximizing corn conversion efficiency to ethanol, and therefore, currently corn has a higher conversion rate (Wang et al. 2008; Beach et al. 2010).

Grain sorghum is a drought tolerant C-4 grass, and typically does well in dry areas without irrigation. As such, sorghum is more water efficient than many other biofuel feedstocks. One study reported that forage sorghum (i.e. grain sorghum) produced biomass yields similar to corn using 33% less water (Rooney et al. 2007). With much of the environmental debate surrounding ethanol production centered on sustainable water use (Dominguez-Faus et al. 2009), grain sorghum could be a possible substitute for thirsty corn crops that require irrigation. However, to our knowledge, there are few studies that examine the water quality impacts of increased grain sorghum production (Love and Nejadhashemi 2011). Kansas is an ideal location to study the impacts of grain sorghum as the state is responsible for 42% of the national grain sorghum production, making it the nation's largest producer (Kansas Department of Agriculture 2014).

In this study, water quality and sustainability indicators were evaluated for land-use scenarios with increasing grain sorghum and corn production in two watersheds: the Perry Lake watershed in northeastern Kansas and the Kanopolis Lake watershed in central Kansas. In each watershed, six scenarios were examined: four with intensification of agricultural land (winter wheat to corn, winter wheat to grain sorghum, hay to corn, and hay to grain sorghum), and two with extensification of agricultural land (Conservation Reserve Program (CRP) land to corn, CRP land to grain sorghum). Winter wheat land-use was selected for analysis because it is not utilized as a biofuel crop, yet is a dominant crop in the Kansas landscape. Hay and CRP have been indicated as targeted land-uses for conversion to biofuel feedstock crops, such as corn and soybean, in the Western Corn Belt due to projected higher returns from rising commodity prices (Wright and Wimberly 2013). It is also expected that many land owners may not re-enroll CRP land due to high crop prices in recent years (Secchi, Kurkalova, et al. 2011; Hellerstein and Malcolm 2011). There is a total of 59 km^2 CRP land-use in the Perry Lake watershed and 919 km^2 in the Kanopolis Lake watershed that are set to expire between 2015 - 2025 (determined by geospatial analysis; see Figure 3-1 (USDA 2014). A recent study demonstrated that intensification and extensification of corn production is occurring in Kansas and that these phenomena are negatively related to the distance to the nearest ethanol refinery (Brown et al. 2014). However, no studies have yet analyzed the potential water quality impacts, or the possible ethanol production resulting from such LUC patterns. The goals of this study are two-fold: 1) to evaluate the water quality impacts of biofuel-based land-use scenarios in Kansas, by simulating intensification and extensification of corn and grain sorghum land-use in two watersheds; and 2) to compare the environmental sustainability indicators and water quality impacts of corn and grain sorghum in two watersheds with considerably different climate.

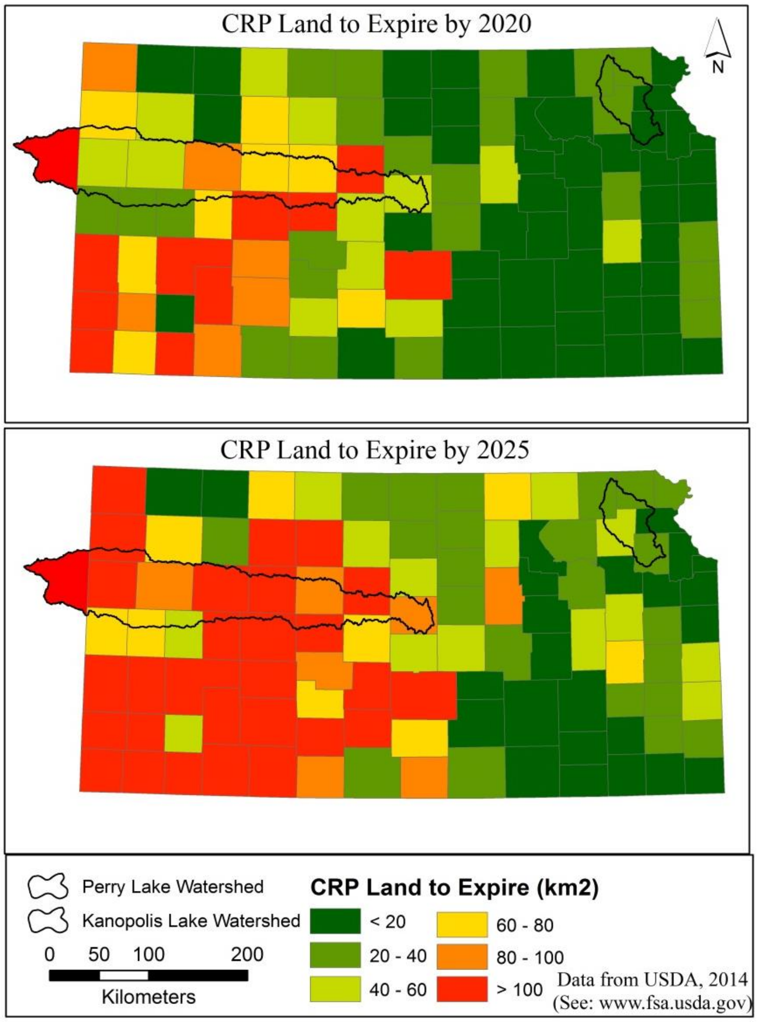

Figure 3-1: Total CRP land set to expire in Kansas by 2020 and 2025; based on data from (USDA 2014)

3.2 Methods

3.2.1 Study Sites

Two watersheds were selected for studying the impacts of LUC scenarios in Kansas, the Perry Lake and the Kanopolis Lake watersheds. Both watersheds drain into regionally important reservoirs and consist of primarily agricultural land-use, but they are located in very different regions within the state (see Figure 3-2).

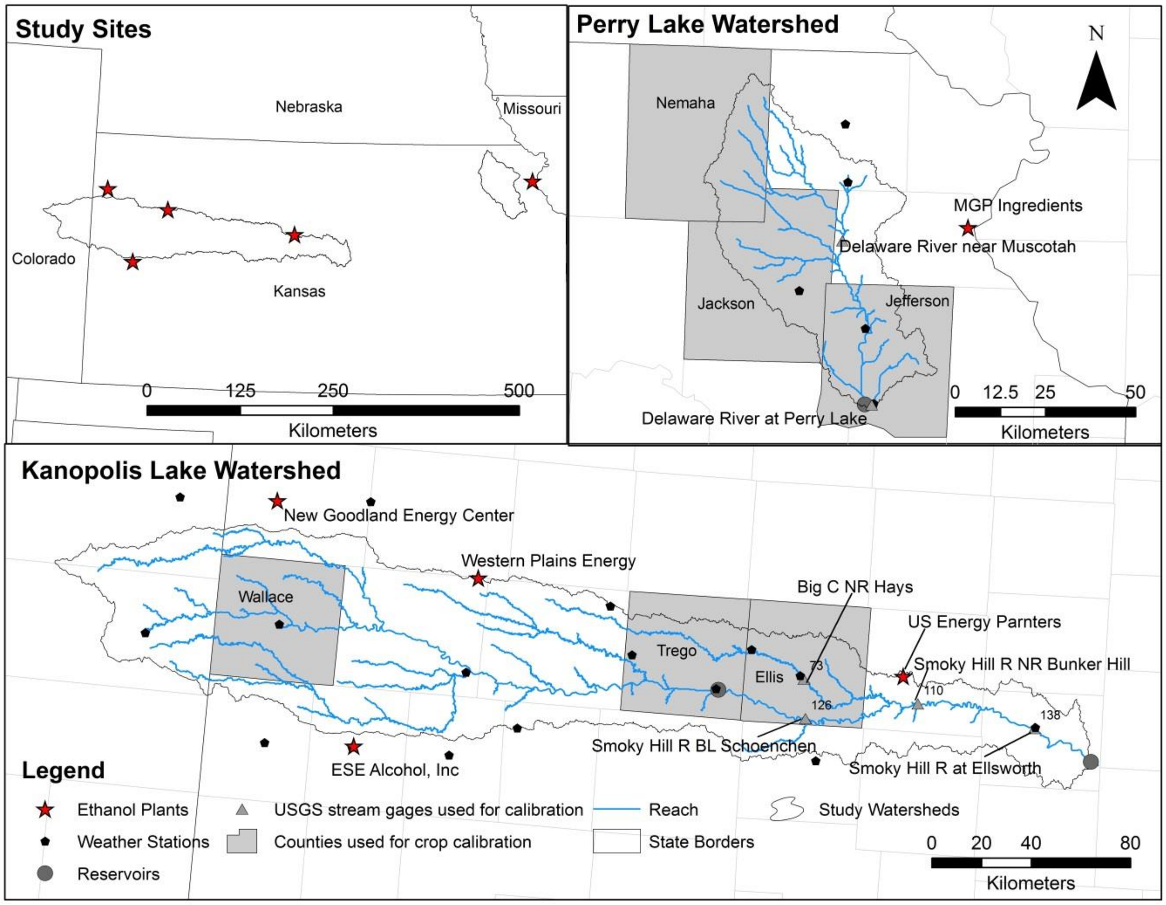

Figure 3-2: Location of study sites, Perry Lake watershed and Kanopolis Lake watershed, ethanol plants near study sites, as well as weather stations, stream gages, and counties used for calibration.

The two study watersheds also vary greatly in size, topography, average annual precipitation, dominant land-use types, dominant soil type, and groundwater interactions (Table 2-1). Particularly the difference in average annual precipitation drives many of the major land-use differences between the watersheds and is also critical when examining the potential for increasing production of corn and grain sorghum for biofuel production.

The Perry lake watershed is located in northeastern Kansas within the Central Irregular Plains and the Western Corn Belt Plains Level III Ecoregions. The drainage area is approximately 2,924 km^2 and is utilized mostly for agricultural purposes with very little irrigated crop land. Hay (cool-season grassland) and rangeland (warm-season grassland) represent, respectively, 32% and 15% of the watershed, with corn and soybeans together representing 27% of the watershed. Annual average precipitation ranges from 890-980 mm (Juracek and Ziegler 2009). The major water bodies in the watershed are the Delaware River, which drains into Perry Lake, a man-made reservoir operated by the Army Corp of Engineers, which then releases water into the Kanas River. There is one ethanol plant, MGP Ingredients, with 25 million gallons per year (MGY) capacity, located near the Perry Lake watershed.

The Kanopolis Lake watershed is located in central to west-central Kansas and reaches across the state into the east-central portion of Colorado. It is located within the Central Great Plains and the Western High Plains ecoregions has a watershed area of about 20,291 km^2. The predominant land-use types are rangeland (warm-season grassland; 40%), followed by winter wheat (29%). While irrigated cropland is more common in the Kanopolis watershed, it still represents a small portion of the overall watershed (4.4%). Annual average precipitation varies greatly across the long watershed with 711 mm at the watershed outlet and 483 mm in the western portion of the watershed (Juracek 2011). The watershed drains the Smoky Hill River into Cedar Bluff Lake, located in the central portion of the watershed and then ultimately into Kanopolis Lake in Ellsworth county at the outlet of the watershed. Both reservoirs are operated and maintained by the Army Corps of Engineers. There are three existing ethanol plants within or near the Kanopolis watershed: Western Plains Energy (41 MGY), US Energy Partners (55 MGY), and E.S.E. Alcohol, Inc (2 MGY). There is also an ethanol plant in construction nearby, New Goodland Energy Center (40 MGY), for a combined capacity of 138 MGY.

3.2.2 The SWAT Model

SWAT is a continuous-time, spatially distributed simulator of the hydrologic cycle and agricultural pollutants. Major model input components include climate conditions, soil properties, topography, plant growth, and land management. Model outputs include subbasin flow and loads of nutrients, sediment, pesticides, bacteria, and pathogens (Gassman et al. 2007; Ficklin et al. 2009). A more detailed description of the model, as well as sediment and nutrient processing can be found in Chapter 3. The LUC scenarios were applied through the land use updater (.LUC) within ArcSWAT version 2010-beta (Pai and Saraswat 2011).

3.2.3 Model Development and Calibration

A detailed description of model development and parameterization of land-use, soil, slope, climate, management practices, point source inputs, and reservoir outflow is presented in Chapter 3. Also in Chapter 3, a detailed description of the calibration procedure is given, and both calibration and validation statistics are presented for streamflow at two locations in Perry watershed and four locations in Kanopolis watershed, crop yield in three counties in each watershed, and sediment load at one location in Perry watershed. A brief summary of calibration and validation statistics for annual streamflow, crop yield, and annual sediment yield are provided in Table 3-1 and Table 3-2 at both default (pre-calibration) and final (post-calibration) stages. Performance evaluation statistics include the Nash-Sutcliffe Efficiency (NSE) and percent bias (PBIAS), which are both recommended statistics for evaluating the performance of hydrologic models (Moriasi et al. 2007).

In general, streamflow was simulated very well within the Perry Lake watershed with annual NSE values greater than 0.75 and PBIAS less than ± 10, indicating a very good performance rating by Moriasi et al. standards. Sediment calibration in Perry Lake watershed also performed well with a NSE monthly calibration value of 0.83. Streamflow in the Kanopolis Lake watershed was simulated best at the Hays and Schoenchen gage locations, with NSE values greater than 0.80 for the calibration period in both locations. The Schoenchen location also had a satisfactory validation NSE value (0.66) and very low bias.

Table 3-1 Annual streamflow calibration statistics for Perry Lake and Kanopolis Lake watersheds and monthly sediment calibration statistics for Perry Lake watershed

	NSE		**PBIAS (%)**	
	Default	**Final**	**Default**	**Final**
Perry Lake Watershed - Delaware River near Muscota - Streamflow				
Annual Calibration (1978-1996)	0.82	0.80	14.41	-3.75
Annual Validation (1997-2011)	0.51	0.79	40.05	7.46
Perry Lake Watershed - Delaware River near Muscota - Sediment				
Monthly Calibration	0.46	0.83	55	-20.0
Monthly Validation 2010 (2011)	0.36	0.62 (-0.19)	25	43.5
Perry Lake Watershed - Delaware River at Perry Lake - Streamflow				
Annual Calibration (1978-1996)	0.99	0.99	2.45	1.13
Annual Validation (1997-2011)	0.99	0.99	7.22	1.14
Kanopolis Lake Watershed - Big C NR Hays - Streamflow				
Annual Calibration (1978-1996)	-11.71	0.85	330.83	-27.20
Annual Validation (1997-2011)	.37.23	0.31	81.26	-10.43
Kanopolis Lake Watershed - Smoky Hill R BL Schoenchen - Streamflow				
Annual Calibration (1981-1996)	-8.44	0.83	445.04	-2.97
Annual Validation (1997-2011)	-16.72	0.66	375.34	-0.38
Kanopolis Lake Watershed - Smoky Hill R NR Bunker Hill - Streamflow				
Annual Calibration (1978-1996)	-3.44	0.60	180.38	-50.62
Annual Validation (1997-2010)	20.85	0.36	283.37	-17.35
Kanopolis Lake Watershed - Smoky Hill R at Ellsworth - Streamflow				
Annual Calibration (1978-1996)	-1.60	0.49	141.19	-53.80
Annual Validation (1997-2010)	-6.68	0.34	-18.69	-15.75

Crop calibration (see Table 3-2) demonstrated that SWAT satisfactorily estimates corn yield in the Perry Lake watershed. SWAT did not have good performance ratings for soybean yield; however, the best results were in Jackson county with an NSE value equal to 0.55 and bias near -2%. SWAT did not simulate winter wheat nor grain sorghum yields with a high degree of accuracy in the Kanopolis Lake watershed (most NSE < 0). NSE values for winter wheat yield simulations were negative in all three counties studied. However, bias was less than 5% in all cases.

Table 3-2: Yield calibration statistics for Perry Lake and Kanopolis Lake watersheds

County	N	Crop yield (t/ha)		NSE		PBIAS (%)	
		Reported	Modeled	Default	Final	Default	Final
Perry Lake Watershed – Corn Yield							
Jackson	13	5.49	5.56	-13.32	0.55	78.65	1.83
Jefferson	10	5.65	5.69	-9.18	0.83	90.33	0.83
Nemaha	13	5.54	5.61	2.48	0.64	62.41	1.35
Perry Lake Watershed – Soybean Yield							
Jackson	6	2.13	2.08	-3.46	0.55	30.94	-2.09
Jefferson	10	2.07	1.87	-1.11	0.37	27.68	-9.85
Nemaha	6	2.29	1.90	0.38	-0.57	3.50	-11.24
Kanopolis Lake Watershed - Winter Wheat Yield							
Wallace	13	1.73	1.44	-5.12	-0.53	38.28	-1.91
Trego	13	1.90	1.86	-17.60	-0.07	87.78	3.70
Ellis	13	2.00	2.05	-9.15	-0.67	40.18	4.56
Kanopolis Lake Watershed - Grain Sorghum Yield							
Wallace	12	2.58	1.85	-0.52	-0.40	-36.68	-28.27
Trego	12	3.50	2.95	-1.51	0.51	-18.0	-15.51
Ellis	12	3.74	3.93	-0.03	0.1	9.4	5.04

3.2.4 Land-use Scenarios

Land-use scenarios were developed for the Perry and Kanopolis watersheds to consider both expansion of cropland into non-cultivated land (i.e. extensification), as well as intensification of production on agricultural land. There were a total of 6 base scenarios per watershed: 2 representing extensification and 4 representing intensification. In the extensification scenarios, corn and grain sorghum replaced conservation reserve program (CRP) land-use. In the intensification scenarios corn and grain sorghum-based rotations replaced either winter wheat or hay land-uses. In the Perry Lake watershed, all corn land-use is represented by a corn-soy rotation, as this is the realistic land management practice in this watershed. In the Kanopolis Lake watershed, all corn land-use is represented by a corn-winter wheat rotation, and all grain sorghum land-use is represented by a grain sorghum-winter wheat rotation, as these are both the dominant rotations in this watershed.

Table 3-3: Study design matrix demonstrating the range of biofuel feedstock land-use transitions studied in each watershed; each range was broken into 10 simulations to study how impacts vary within the range.

	Perry Lake Watershed		**Kanopolis Lake Watershed**	
Original Land-use	**Grain Sorghum**	**Corn**	**Grain Sorghum**	**Corn**
Winter Wheat	2.4 – 24 km^2	1.7 - 17 km^2	82 – 820 km^2	31 – 310 km^2
Hay	13 – 127 km^2	9.3 - 93 km^2	10 – 96 km^2	3.5 – 35 km^2
CRP	5.4 – 54 km^2	3.6 – 36 km^2	39 – 394 km^2	14 – 140 km^2

Each of the 12 total scenarios was simulated at 10 different land-use percentages (resulting in 120 different simulations). The goal was to vary the land-use percentage over a reasonable range and show results at 2, 5, and 10% increments. However, in many cases the model was unable to reach these target percentages due to constraints in the land-use updater (.LUC) tool within ArcSWAT. The .LUC tool is only able to convert an HRU from its original land-use type to a land-use type that is located in an adjacent HRU. Therefore, often the model did not reach the targeted LUC percentage. For example, in the Kanopolis Lake watershed the goal was to simulated 2 – 20% winter wheat to grain sorghum LUC at 2% intervals. However, the model was only able to change 1.2 – 12% of winter wheat land-use at approximately 1 – 1.2% intervals. The targeted LUC percentages, as well as the actual LUC percentages, are provided in Appendix C – Land-use change scenarios

In the Perry Lake watershed the hay scenarios resulted in the greatest overall LUC by area, as hay was the dominant land-use. Conversely, winter wheat scenarios resulted in the lowest overall LUC, as winter wheat represents less than 1% of overall watershed land-use. In the Kanopolis Lake watershed winter wheat represents 19% of total watershed land-use; therefore, winter wheat scenarios resulted in the greatest overall LUC by area. Conversely, hay scenarios resulted in the lowest overall LUC, as hay represents less than 1% of overall watershed land-use. In order to account for these differences in land-use, all water quality indicators were examined per km^2 land changed.

3.2.5 Water Quality Indicators

Total phosphorus, total nitrogen, and sediment loads were used as indicators of water quality impacts from LUC scenarios. Total phosphorus (TP) loads were calculated by summing organic phosphorus (ORGP) and mineral phosphorus (MINP) for each year from 2000 – 2011 and then averaging the annual values for each scenario. Total nitrogen (TN) loads were calculated by summing organic nitrogen (ORGN), nitrate (NO3), ammonium (NH4), and nitrite (NO2) outputs or each year from 2000 – 2011 and then averaging the annual values for each scenario. The loads were analyzed at the subbasin outlet(s) closest to the inlet of the reservoir in each watershed. For the Perry Lake watershed, the outlets for reaches 33-37 were analyzed, as each of the five outlets ended at one of the major branches of the reservoir. Loads from subbasins 33-37 were then summed to get a total load for the reservoir. In the Kanopolis Lake watershed the outlet for reach 144 was analyzed, as this was the only outlet prior to the reservoir inlet. A baseline load was determined by calculating average TP, TN, and sediment loads from the 2000 – 2011 period before any land-use modifications. Then, baseline values were subtracted from all TP, TN, and sediment loads from the land-use scenarios to determine the difference from baseline conditions. In order to compare water quality outcomes of all scenarios, regardless of the total amount of land changed, 2000 – 2011 average loads of sediment, TN, and TP were divided by the land-area changed in each model iteration. Ratios of sediment-tons/km^2, TN-kg/km^2, and TP-kg/km^2 were tested for significant differences between corn and grain sorghum LUC scenarios using a Student's T-Test. The average water quality ratios for the 10 iterations of each scenario can be found in Table 3-4.

3.2.6 Sustainability Indicators

The following sustainability indicators were used to account for nutrient, land, and water resources used to grow biofuel feedstocks and to produce metrics that can be compared across crop types and watersheds. Water resource use is accounted for by the water use efficiency (WUE; kg/m^3) indicator, which is calculated by taking the ratio of yield (Y; g/m^2) to crop evapotranspiration (ET; mm) (Tolk and Howell 2003). In this study, only non-irrigated crops were considered, so irrigated water use was irrelevant. As such, the WUE values varied mostly with the weather conditions of the two watersheds that dictated crop evapotranspiration. The average and standard deviation of WUE values are reported for both corn and grain sorghum for each watershed, as the values did not vary between scenarios. Nutrient resource use is represented by the nutrient use efficiency (NUE) indicator for nitrogen (NUE-N) and for phosphorus (NUE-P), which are calculated by dividing the grain yield (kg) by the amount of nitrogen (N; kg) and phosphorus (P; kg) applied as fertilizer (Good et al. 2004). Nitrogen and phosphorus application rates were determined based on recommended rates per acre for a specific bushel yield goal using county average yields from 2005 – 2011 from the National Agricultural Statistics Service (Leikam et al. 2003). Spatial land-use impact, or land-use efficiency (LUE; km^2/L) is accounted for by the ratio of land area changed (km^2) to the liters of ethanol produced. An estimate of ethanol production was calculated using values from the Forest and Agricultural Sector Optimization Model with Greenhouse Gases (FASOMGHG). The FASOMGHG model estimates an ethanol yield of 2.71 gallons per bushel of corn (using a dry milling process) and 2.38 gallons per bushel of grain sorghum (Beach et al. 2010). In all cases, indicators were calculated using average values for the entire watershed from 2000-2011. Other factors necessary to grow, harvest, or process the biofuel crop are not included in this analysis.

3.3 Results

3.3.1 Perry Lake Watershed – Water Quality Indicators

Figure 3-3A-D show the water quality impacts of replacing winter wheat with grain sorghum and corn with the Perry Lake watershed. When winter wheat was replaced by grain sorghum or corn there was an increase of less than 0.4% for sediment and less than 0.75% for TP loads, when compared to baseline values. The sediment load increased from 1,600 to 17,000 tons above baseline with a respective additional 2.4 km^2 to 24 km^2 grain sorghum, which corresponds to a 0.03% to 0.34% increase from the baseline sediment output. With an additional 1.7 km^2 to 17 km^2 corn, the sediment load increased by 994 to 20,000 tons, respectively, a 0.02% to 0.39% increase from baseline sediment values. While all changes are less than 0.4%, there are still significantly different from the baseline in all scenario iterations with a p-value < 0.05. Also, the corn land-use scenarios produced higher sediment loads per land area compared to the grain sorghum land-use scenarios (see Table 3-4) the difference is significant with a p-value = 0.001.

With an increase in grain sorghum land-use from 2.4 to 24 km^2, TP loads increased from 1,412 to 9,788 kg, which corresponds to a 0.11 to 0.73% increase from baseline values. Results are significantly different from the baseline at a LUC above 9.7 km^2, with p-values < 0.05., With an increase in corn land-use from 1.7 – 17 km^2, TP loads increased from 2,500 to 7,400 kg, which corresponds to a 0.18 to 0.55% increase from baseline values. However, only the LUC scenarios with increases of 3.4 -8.5 km^2 were statistically different from the baseline with p-values < 0.05. However, there is no significant difference in TP output between corn and grain sorghum scenarios.

Nitrogen loads, however, did not behave similarly to phosphorus or sediment loads. TN increased when additional grain sorghum was planted, but decreased when additional corn was planted (See Figure 3-3B-D). TN increased from 4,500 kg to 14,000 kg with an increase of 2.4 km^2 to 24 km^2 grain sorghum, respectively. These increases correspond to a 0.06% to 0.20% increase from baseline values. When an additional 1.7 km^2 corn was planted, TN loads initially increased by 9,200 kg (0.13% change from baseline), but then began to decrease and bottomed out at a decrease in 41,000 kg TN (-0.58% change from baseline) when corn acreage increased by 17 km^2. The TN load decreased mainly due to a reduction in nitrate export due to the corn-soy rotation that replaced winter wheat. The alternating years of soy production and associated nitrogen fixation resulted in alternating years without chemical nitrogen fertilizer added. In addition, corn grown in rotation with soybeans has been shown to have overall reduced nitrate runoff compared to continuous corn (Drinkwater et al. 1998). TN load per km^2 is significantly higher in the continuous grain sorghum scenario than in the corn-soy scenarios (Table 3-4; p=0.03). However, the TN loads from both the corn and grain sorghum scenario results are not significantly different from baseline TN loads (see Appendix E. Statistical significant of water quality changes from land-use change scenarios).

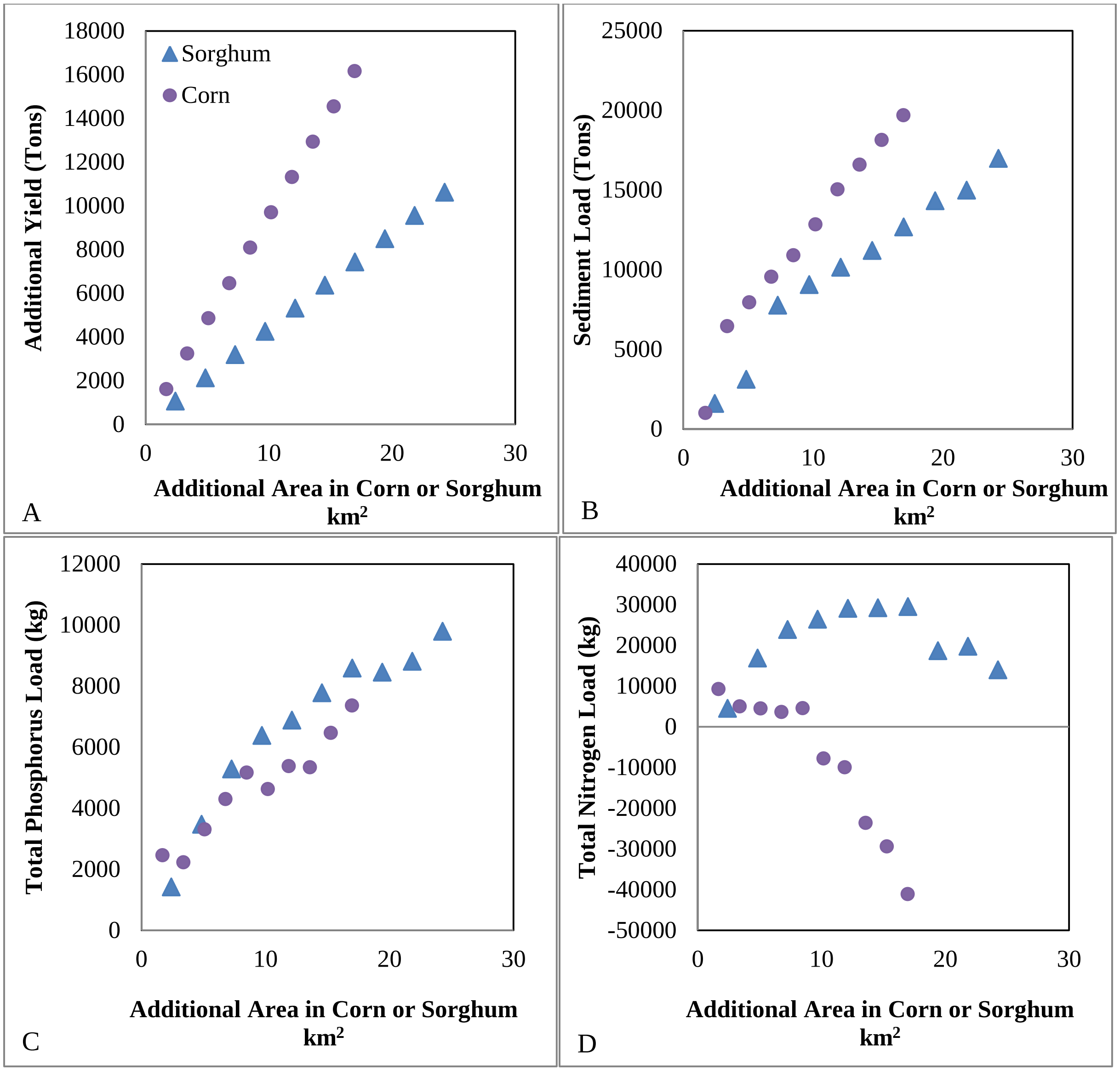

Figure 3-3A-D: Results from land-use scenarios replacing winter wheat for grain sorghum (blue triangle) or corn (purple circle). Figure A (top left) shows the additional yield in grain sorghum and corn resulting from replacing winter wheat land-use. Figures B-D demonstrate the change in sediment yield (B; top right), TP load (C; bottom left), and TN load (D; bottom right) at the Perry reservoir inlets due to the LUC. All values are shown as the difference from the baseline model simulations.

When grain sorghum and corn replaced hay land-use, there was an overall increase in sediment, TP, and TN loads. The sediment load increased from 22,000 to 200,000 tons, 0.44 to 3.78% higher than baseline values, with respective increases in grain sorghum from 13 to 127 km^2. With increases in corn land-use from 9.3 to 93 km^2, there was a respective increase in sediment load from 34,000 to 280,000 tons, corresponding to an increase of 0.68 to 5.7% above baseline values.

With respective increases in grain sorghum land-use from 13 to 127 km^2, TP increased from 27,000 to 260,000 kg, a 2.0 to 20% increase from baseline values, and TN increased from 110,000 to 1,000,000 kg, a 1.6 to 15% increase from baseline values. With increases in corn land-use from 9.3 to 93 km^2, there was a respective increase in TP values from 37,000 to 350,000 kg, corresponding to an increase of 2.8 to 26% from baseline values. TN loads increased from 140,000 to 1,200,000 kg, corresponding to increases of 1.9 to 18% from baseline values, respectively. Of the two feedstocks, corn scenarios produced greater sediment, TN, and TP loads per km^2 than grain sorghum scenarios when compared using a T-test (see Table 3-4); the differences are all significant with a p-value < 0.001. Also, all water quality load increases from both corn and grain sorghum scenarios were significantly different from the baseline with p-values ≤ 0.001.

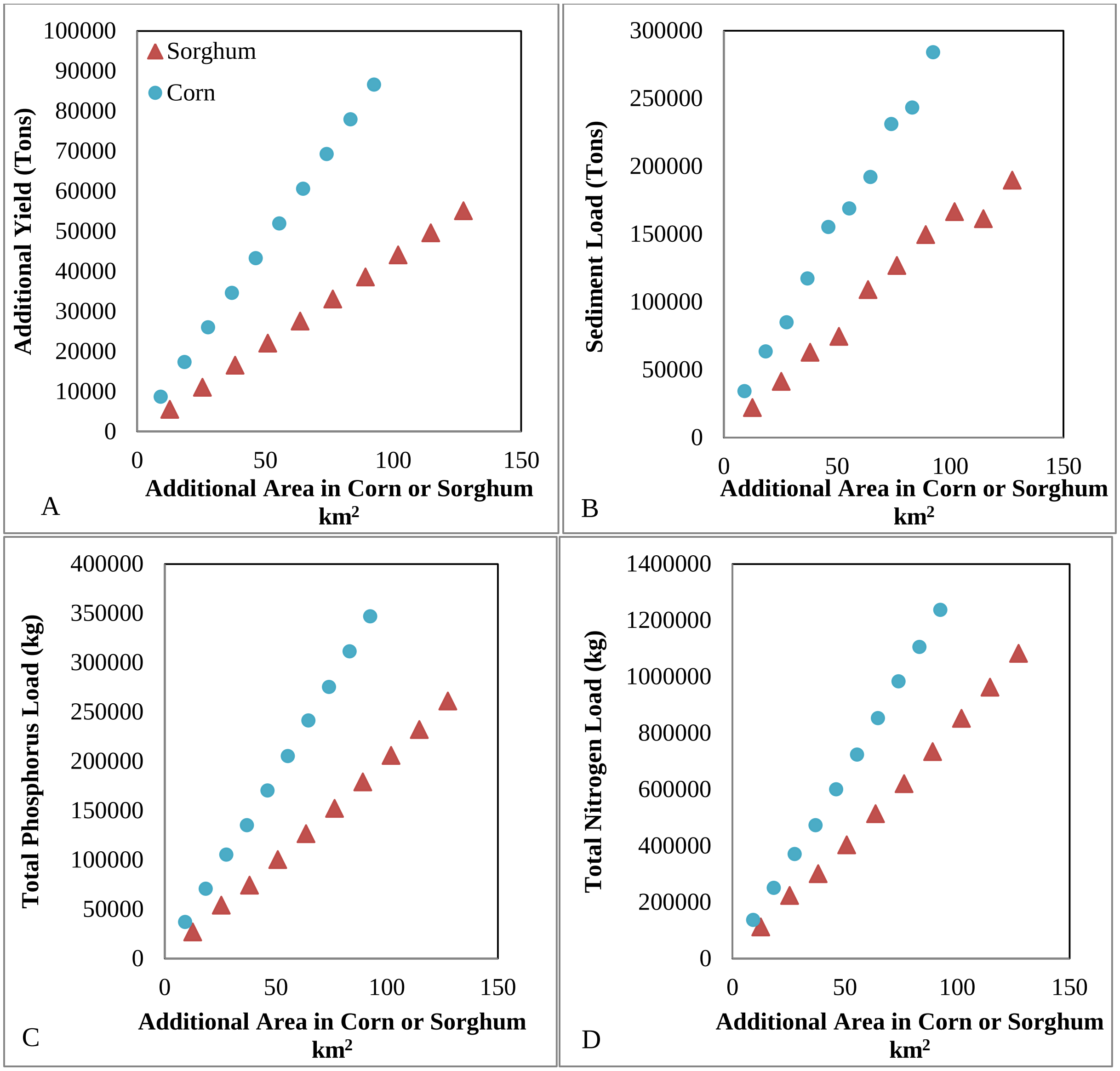

Figure 3-4A-D: Results from land-use scenarios replacing hay for grain sorghum (red triangle) or corn (blue circle). Figure A (top left) shows the additional yield in grain sorghum and corn resulting from replacing hay land-use. Figures B-D demonstrate the change in sediment yield (B; top right), TP load (C; bottom left), and TN load (D; bottom right) at the Perry reservoir inlets due to the LUC. All values are shown as the difference from the baseline model simulations.

When grain sorghum and corn replaced CRP land-use, there was also an overall increase in sediment, TP, and TN loads (Figure 3-5A-D). The sediment load increased from 2,700 to

30,000 tons, 0.05 to 0.60% above baseline values, with respective increases in grain sorghum from 5.4 to 54 km^2 above baseline values. With increases in corn land-use from 3.6 to 36 km^2 there was a respective increase in sediment load from 2,300 to 36,000 tons, corresponding to an increase of 0.05 to 0.72% from baseline values. All increases are significantly different from the baseline at p-values < 0.005.

With respective increases in grain sorghum land-use from 5.4 to 54 km^2, TP increased from 8400 to 85,000 kg, a 0.63 to 6.4% increase above baseline values. TN increased from 37,000 to 350,000 kg, which corresponds to a 0.52 to 5.0% increase from baseline values. With increases in corn land-use from 3.6 to 36 km^2, there was a respective increase in TP values from 11,000 to 98,000 kg, a 0.84 to 7.4% increase above baseline values. TN loads increased from 43,000 to 350,000 kg, corresponding to increases of 0.60 to 4.9% from baseline values, respectively. Again, the corn scenarios produced greater sediment, TN, and TP loads per km^2 than the grain sorghum scenarios; the differences are significant with a p-value < 0.001. Also, all TN and TP load increases from both corn and grain sorghum scenarios were significantly different from the baseline with p-values ≤ 0.001.

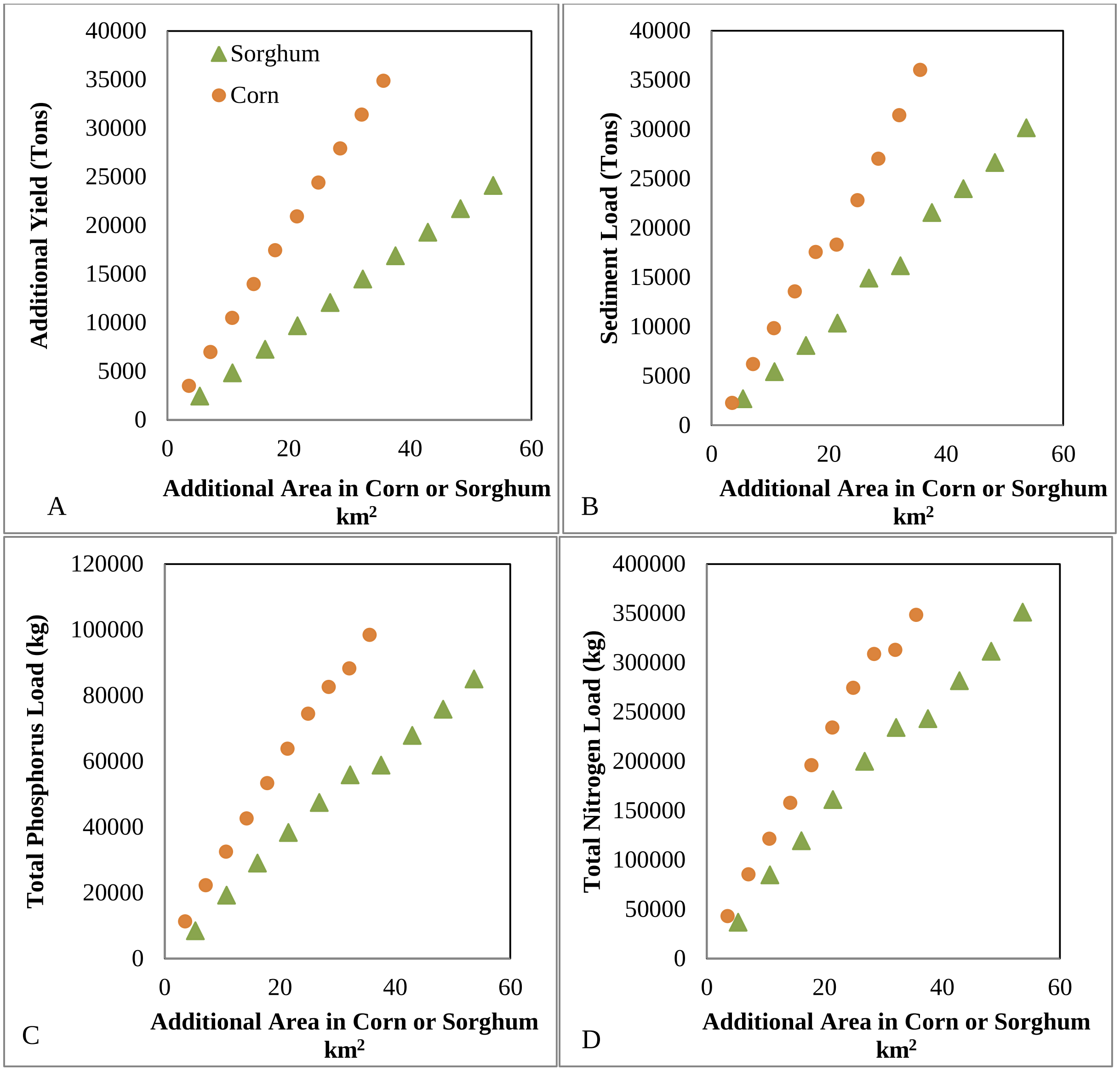

Figure 3-5A-D: Results from land-use scenarios replacing grain sorghum (green triangle) or corn (orange circle) for CRP land. Figure A (top left) shows the additional yield in grain sorghum and corn resulting from replacing CRP land-use. Figure B-D demonstrate the change in sediment yield (B; top right), TP load (C; bottom left), and TN load (D; bottom right) at the Perry reservoir inlets due to the LUC. All values are shown as the difference from the baseline model simulations.

Table 3-4: Average annual water quality indicators for land-use scenarios in Perry Lake watershed calculated using the 10 iterations of each scenario. Ratios reflect an increase in sediment, TN, or TP export per area of land changed relative to the baseline model.

	Water Quality Indicators Average ± Standard Deviation		
	Sediment /Area ($Tons/km^2$)	**TN /Area (kg/km^2)**	**TP /Area (kg/ km^2)**
Winter Wheat to Grain Sorghum	776 ± 134	1,984 ± 992	552 ± 120
Winter Wheat to Corn-Soy	1,282 ± 331	114 ± 2,280	614 ± 311
Hay to Grain Sorghum	1,602 ± 110	8,282 ± 321	2,017 ± 57
Hay to Corn-Soy	3,179 ± 237	13,333 ± 535	3,752 ± 97
CRP to Grain Sorghum	528 ± 34	7020 ± 522	1671 ± 108
CRP to Corn-Soy	907 ± 107	10970 ± 755	2967 ± 132

3.3.2 Perry Lake Watershed – Sustainability Indicators

Sustainability indicators for both grain sorghum and corn in Perry Lake watershed are shown in Table 3-5. The SWAT model estimated average NUE-N and NUE-P values of 36 and 137 for grain sorghum. In other words, for every 1 kg-N yielded 36 kg grain sorghum grain. Equivalently, an average of 27 kg-N and 7.2 kg-P are required to produce one metric ton of grain sorghum. For corn production average NUE-N and NUE-P values of 56 and 212 were estimated. Equally, about 17.5 kg nitrogen and 4.7 kg phosphorus are needed to produce one ton of corn grain. Therefore, corn has a higher NUE both in terms of nitrogen and phosphorus in the Perry Lake watershed. LUE-yield, or grain yield per area, is also more than twice that of grain sorghum: 934 to 970 $tons/km^2$ for corn compared to 433 to 452 $tons/km^2$ for grain sorghum. Consequently, LUE-ethanol, or ethanol yield per km^2, is also more than double for corn (377,000 to 392,000 L/km^2) compared to grain sorghum (153,000 to 160,000 L/km^2). Average water use efficiency in the Perry Lake watershed was estimated to be 0.66 ± 0.22 kg/m^3 for grain sorghum and 1.49 ± 0.37 kg/m^3 for corn. Corn achieves about twice as much grain yield per m^3 water consumed in the Perry Lake watershed.

Table 3-5: Sustainability indicators for nutrient and land use resource requirements per ton of grain and per liter ethanol in Perry Lake watershed

Starting Land-use	Ending Land-use	NUE-N	NUE-P	LUE -Yield (tons/km^2)	LUE -Ethanol (1000 L/km^2)
Winter Wheat	Grain Sorghum	36	136	446 ± 31	158 ± 11
Hay	Grain Sorghum	36	135	433 ± 4	153 ± 2
CRP	Grain Sorghum	37	140	452 ± 12	160 ± 4
Winter Wheat	Corn	56	211	954 ± 69	385 ± 28
Hay	Corn	55	208	934 ± 13	377 ± 5
CRP	Corn	58	218	970 ± 38	392 ± 15

3.3.4 Kanopolis Lake Watershed – Water Quality Indicators

When winter wheat was replaced by grain sorghum or corn, there was an increase in predicted TN loads. The predicted TP load decreased with additional grain sorghum land-use and increased with additional corn land-use (See Figure 3-6C). However, both TN and TP load changes are not statistically different from baseline scenarios in both grain sorghum and corn scenarios (Appendix E. Statistical significant of water quality changes from land-use change scenarios). The sediment load appeared to increase in both scenarios, but it is not clear if this is a reliable result. In general, the Kanopolis Lake SWAT model did not produce reliable or consistent sediment predictions. Sediment loads unexplainably fluctuate up and down as corn or grain sorghum land-use increased. There is little available data to evaluate the SWAT-predicted sediment performance in the Kanopolis Lake watershed; therefore, there is a great deal of uncertainty in sediment predictions. Overall, scenario predicted sediment loads from all models are not statistically significant when compared to baseline results.

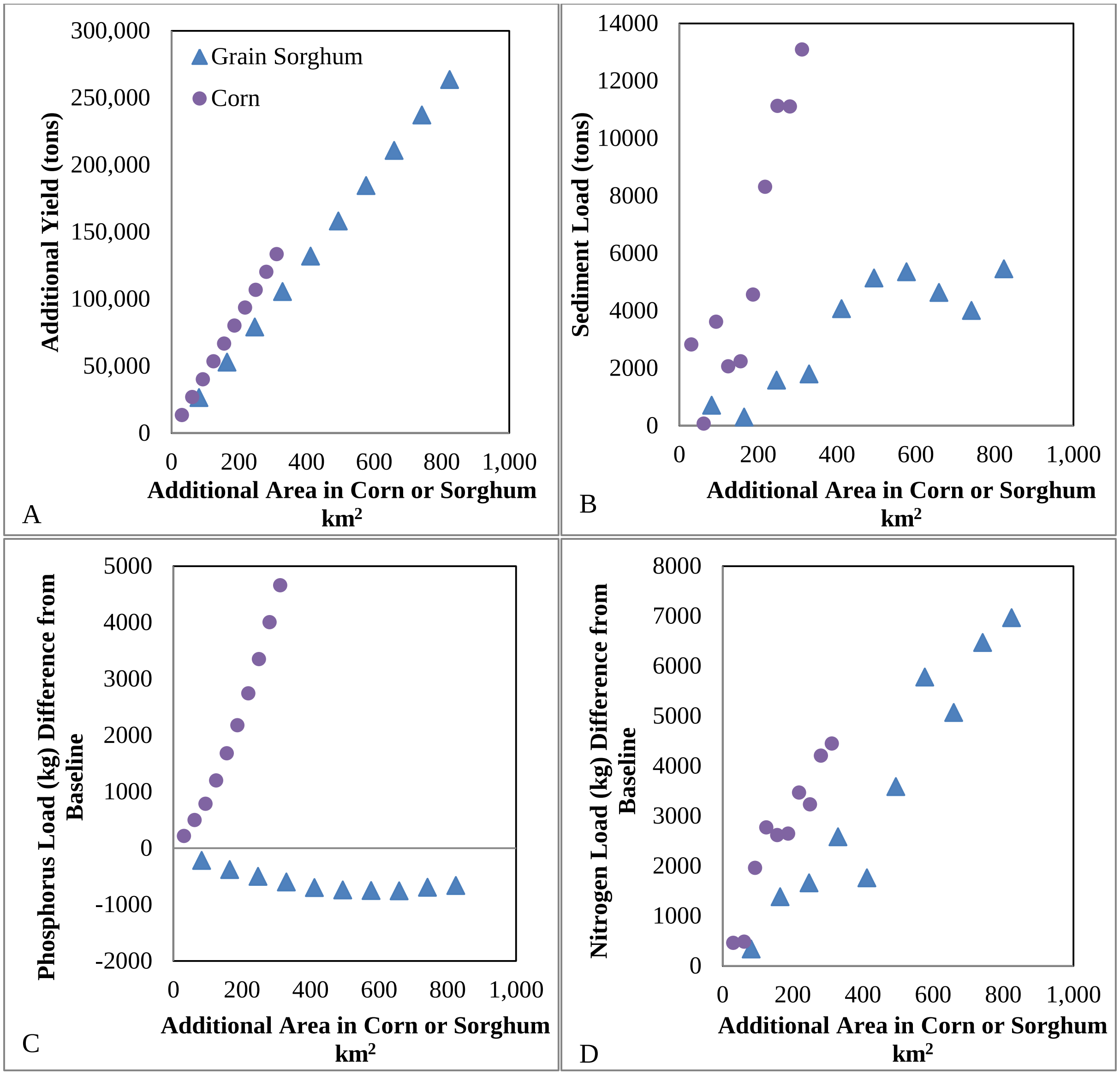

Figure 3-6A-D: Results from land-use scenarios replacing winter wheat for grain sorghum (blue triangle) or corn (purple circle). Figure A (top left) shows the additional yield in grain sorghum and corn resulting from replacing winter wheat land-use. Figures B-D demonstrate the change in sediment yield (B; top right), TP load (C; bottom left), and TN load (D; bottom right) at the Perry reservoir inlets due to the LUC. All values are shown as the difference from the baseline model simulations.

When hay land-use was replaced by grain sorghum and corn-based rotations, there was an overall increase in TP and TN loads. With respective increases in grain sorghum land-use from 10 - 96 km^2, TP increased from 370 to 3,800 kg corresponding to a 0.19 to 2.0% increase from baseline values, and TN increased from 1,600 to 7,400 kg, which corresponds to 0.42 to 2.0% increases from baseline. The TP increases are statistically different from the baseline with a p-value = 0.01, but the TN increases are not statistically significant. As corn land-use increased from 3.5 to 35 km^2, there was a respective increase in TP values from 404 to 4,400 kg, corresponding to an increase of 0.21 to 2.3% from baseline values. Similarly, TN loads increased from 780 to 6,900 kg, corresponding to increases of 0.21 to 1.8% from baseline values. Both TN and TP increases are statistically significant with p-values < 0.01. Also, the corn scenarios produced elevated loads per area of nitrogen and phosphorus, compared to grain sorghum scenarios, and these differences were significant with a p-value < 0.001.

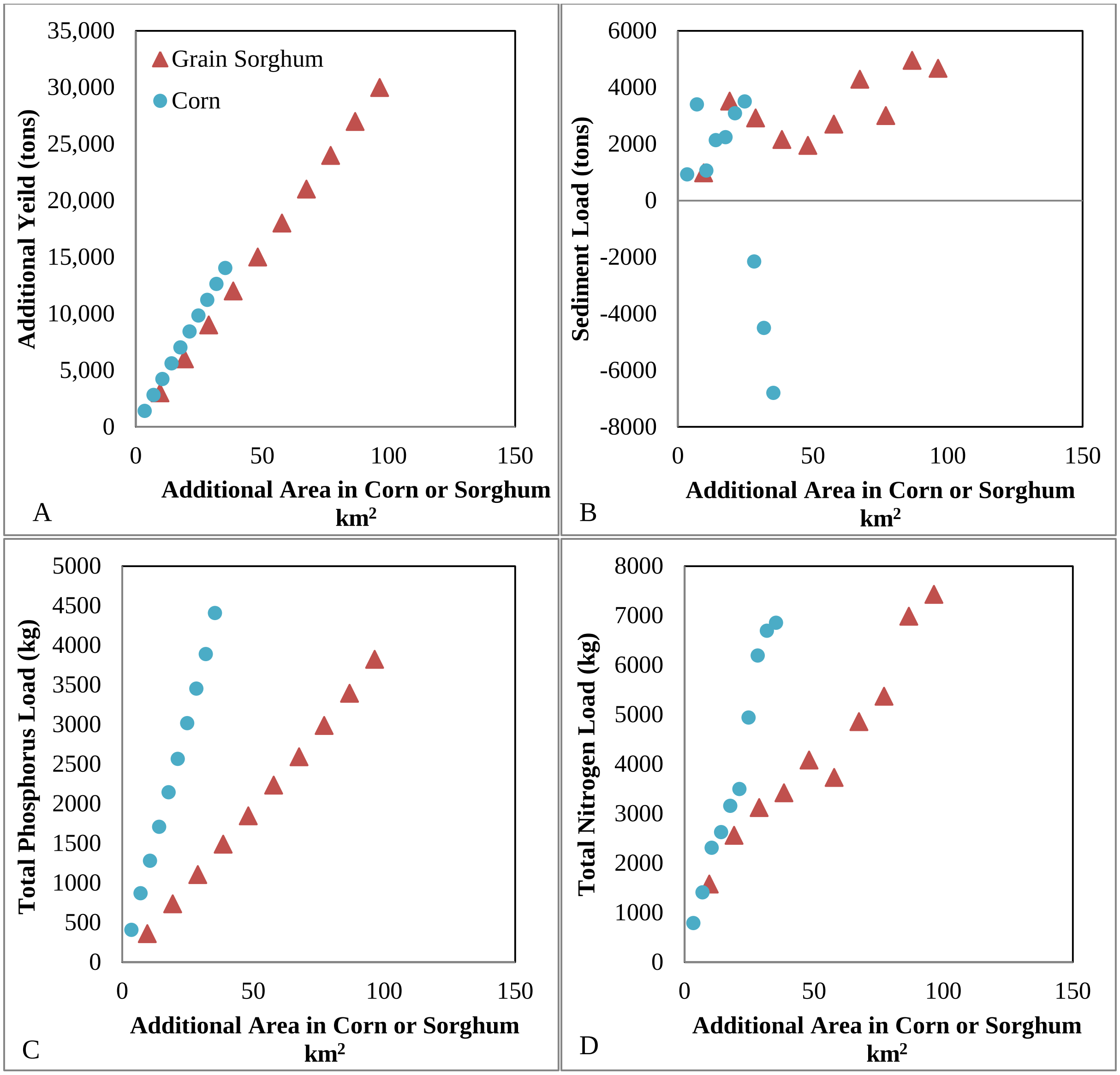

Figure 3-7A-D: Results from land-use scenarios replacing hay for grain sorghum (red triangle) or corn (blue circle). Figure A (top left) shows the additional yield in grain sorghum and corn resulting from replacing hay land-use. Figures B-D demonstrate the change in sediment yield (B; top right), TP load (C; bottom left), and TN load (D; bottom right) at the Perry reservoir inlets due to the LUC. All values are shown as differences from the baseline model simulation.

When grain sorghum and corn replaced CRP land-use, there was also an overall increase in TP and TN loads. With an additional 39 to 394 km^2 grain sorghum land-use, the TP load increased from 2,000 to 21,000 kg, a 1.1 to 11% increase from baseline values, and the TN load increased from 5,900 to 52,000 kg, a 1.6 to 14% increase from baseline values. TN load increases were not statistically different from baseline loads, but TP loads were statistically different with a p-value = 0.05. With an additional 14 to 140 km^2 corn land-use, the TP load increased from 2,200 to 24,000, a 1.2 to 13% increase from baseline values; the TN load increased from 5,600 to 45,000 kg, a 1.5 to 12% increase from baseline values. Both TN and TP increases were statistically different from the baseline with a p-value < 0.01. The corn scenarios had elevated loads of nitrogen and phosphorus compared to grain sorghum scenarios, determined with statistical significance of a p-value < 0.001.

Table 3-6: Average annual water quality indicators for land-use scenarios in Kanopolis Lake watershed calculated using the 10 iterations of each scenario. Ratios reflect a change in sediment, TN, or TP export per area of land changed relative to the baseline model.

Kanopolis Lake Watershed	**Average Water Quality Indicators per Scenario**		
Land-use Scenarios	**Sediment /Area (Tons/km^2)**	**TN /Area (kg/km^2)**	**TP /Area (kg/ km^2)**
Winter Wheat to Grain Sorghum	7.07 ± 2.6	7.33 ± 1.89	-1.64 ± 0.62
Winter Wheat to Corn	34.9 ± 24.3	15.4 ± 4.05	11.0 ± 2.79
Hay to Grain Sorghum	73.3 ± 44	93.9 ± 31.5	38.4 ± 0.699
Hay to Corn	98.9 ± 197	198 ± 18.8	121 ± 2.66
CRP to Grain Sorghum	1.58 ± 20.3	137 ± 8.81	52.8 ± 1.04
CRP to Corn	17.2 ± 27.5	329 ± 26.3	164 ± 5.13

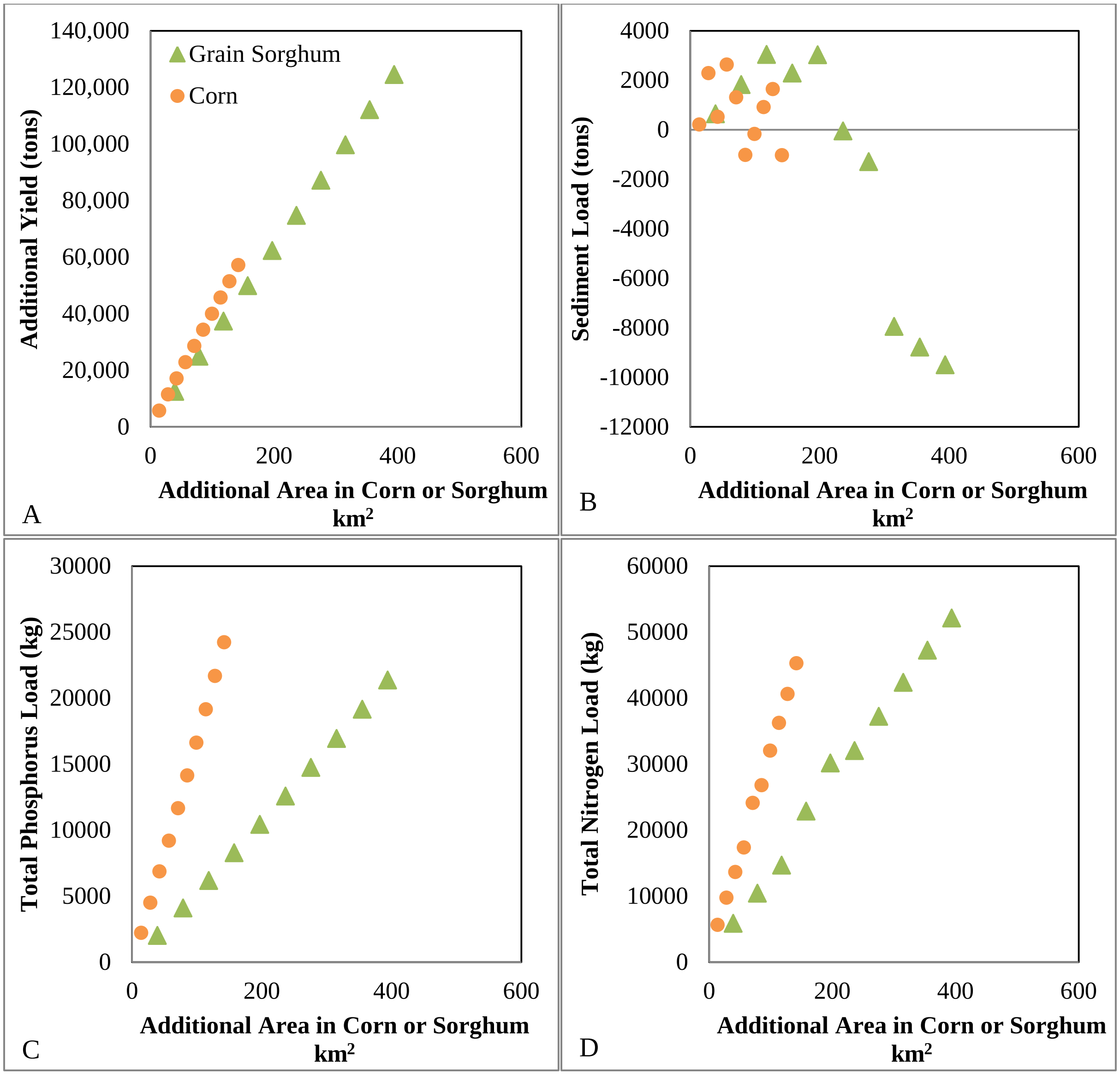

Figure 3-8A-D: Results from land-use scenarios replacing CRP land with grain sorghum (green triangle) or corn (orange circle). Figure A (top left) shows the additional yield in grain sorghum and corn resulting from replacing CRP land-use. Figure B-D demonstrate the change in sediment yield (B; top right), TP load (C; bottom left), and TN load (D; bottom right) at the Perry reservoir inlets due to the LUC. All values are shown as the difference from the baseline model simulations.

3.3.5 Kanopolis Lake Watershed – Sustainability Indicators

For grain sorghum production the SWAT model estimated average NUE-N and NUE-P values of 54 and 197, respectively. For example, this means that for every 1 kg-N, 54 kg grain sorghum grain was produced. Equivalently, an average of 19 kg-N and 5 kg-P are required to produce one metric ton of grain sorghum. For corn production average NUE-N and NUE-P values were estimated to be 43 and 161, respectively. In other words, about 23 kg nitrogen and 6 kg phosphorus are needed to produce one ton of corn grain in the Kanopolis Lake watershed. In Kanopolis Lake watershed the NUE of grain sorghum is higher than corn, whereas in Perry Lake watershed the relationship was reversed. The average LUE-yield is slightly lower for grain sorghum (310 to 316 tons/km^2) than corn (391 to 428 tons/km^2). Consequently, the LUE-ethanol is also lower for grain sorghum (110,000 to 113,000 L/km^2) than for corn (158,000 to 173,000 L/km^2) (see Table 3-5). Average water use efficiency (± standard deviation) for grain sorghum and corn in the Kanopolis Lake watershed was 0.60 ± 0.18 kg/m^3 and 0.88 ± 0.30 kg/m^3, respectively. Corn, therefore, has slightly higher water use efficiency, which is statistically significant with a p-value < 0.000 (determined with a T-Test).

Table 3-7: Sustainability indicators for nutrient and land use resource requirements per ton of grain and per liter ethanol in Kanopolis Lake watershed

Starting Land-use	**Ending Land-use**	**NUE-N**	**NUE-P**	**LUE-Yield (tons/km^2)**	**LUE-Ethanol (1000L/km^2)**
Winter Wheat	Grain Sorghum	54	200	320 ± 1	113 ± 0.2
Hay	Grain Sorghum	53	194	310 ± 4	110 ± 1
CRP	Grain Sorghum	54	197	316 ± 1	112 ± 0.4
Winter Wheat	Corn	44	165	428 ± 2	173 ± 1
Hay	Corn	42	157	391 ± 16	158 ± 6
CRP	Corn	43	161	403 ± 3	163 ± 1

3.4 Discussion

3.4.1 Corn vs. Grain Sorghum

LUC was simulated in two watersheds in Kansas with different dominant land-use, climate, and size. LUC simulations focused on increasing the area of two major biofuel crops, grain sorghum and corn, by replacing winter wheat, hay, and CRP land. For most scenarios, LUC with continuous corn (or corn-soy rotation in the Perry Lake watershed) produced significantly higher loads of TN and TP per land area as compared to continuous grain sorghum scenarios ($p < 0.05$ in all scenarios; except winter wheat TP in the Perry watershed). These results vary slightly from the only other published study comparing corn and grain sorghum LUC (Love and Nejadhashemi 2011). In the Love and Nejadhashemi study, continuous production of grain sorghum resulted in higher median sediment and TP loads, but lower TN loads in comparison to continuous corn and corn-soy LUC in a Michigan watershed.

Table 3-8: Overview of relationships between LUC scenarios with two biofuel feedstock crops and sustainability indicators (nutrient use, land-use, and water use efficiency) and water quality impacts (sediment, TN, and TP) in the two study watersheds. Green/red colors represent better/worse relationships, respectively, and the direction of the arrow represents the direction of the relationship, N/A stands for not available.

		Sustainability Indicators			Water Quality Impacts		
		NUE	LUE	WUE	Sediment	TN	TP
Perry Lake Watershed (High Precipitation)	Corn	$\color{green}\blacktriangle$	$\color{green}\blacktriangle$	$\color{green}\blacktriangle$	$\color{red}\blacktriangle$	$\color{red}\blacktriangle$	$\color{red}\blacktriangle$
	Grain Sorghum	$\color{red}\blacktriangledown$	$\color{red}\blacktriangledown$	$\color{red}\blacktriangledown$	$\color{green}\blacktriangledown$	$\color{green}\blacktriangledown$	$\color{green}\blacktriangledown$
Kanopolis Lake Watershed (Low Precipitation)	Corn	$\color{red}\blacktriangledown$	$\color{green}\blacktriangle$	$\color{green}\blacktriangle$	N/A	$\color{red}\blacktriangle$	$\color{red}\blacktriangle$
	Grain Sorghum	$\color{green}\blacktriangle$	$\color{red}\blacktriangledown$	$\color{red}\blacktriangledown$	N/A	$\color{green}\blacktriangledown$	$\color{green}\blacktriangledown$

In the Perry Lake watershed, corn's higher water quality impacts are offset by the increased yield per land area; corn yield per km^2 is twice as high as grain sorghum yield (see Table 3-5). The higher yield and higher conversion rate from grain to ethanol make it possible to produce more than twice the ethanol with corn than grain sorghum with the same amount of land in the Perry Lake watershed (see Appendix D. Ethanol production potential from land-use scenarios . Therefore, it could be possible to strategically convert land to corn in order to limit the water quality impacts. However, the same situation does not apply in the Kanopolis Lake watershed where non-irrigated corn yield is only slightly higher than grain sorghum yield (see Table 3-7). In this case, higher corn yields do not offset the higher water quality impacts.

The nutrient requirements for grain sorghum and corn per ton grain also differ between the two watersheds. Grain sorghum has a higher input of N and P per ton grain, or lower NUE, than corn in the Perry Lake watershed. This relationship is reversed in the Kanopolis Lake watershed, where grain sorghum requires slightly less N and P per ton grain than corn (i.e. higher NUE). This, again, is related to the difference in yields between the two watersheds. The Perry Lake watershed is located at the edge of the Corn Belt ecoregion and, on average, receives a sufficient amount of rainfall to grow non-irrigated corn with high yields. The Kanopolis Lake watershed is located in the central to west-central portion of Kansas, where average rainfall is less and decreases westward. Therefore, the corn yield per area is much higher in the Perry watershed than the Kanopolis watershed.

With respect to water use, corn has higher water use efficiency (WUE) than grain sorghum in both watersheds; however, the difference between the WUE of crops is much higher in the Perry watershed than the Kanopolis watershed. The WUE results were somewhat unexpected as grain sorghum is considered a drought-tolerant plant and has been shown to produce greater biomass yield per water use, when compared to corn (Rooney et al. 2007). However, this study is focusing on grain yield and not overall biomass. Other studies indeed show that while corn has a higher max and threshold ET compared to grain sorghum, corn also has a higher yield to ET relationship (i.e. WUE) (Stone and Schlegel 2006). It is important to note that these crops were all simulated without irrigation.

Overall, this study suggests that corn will most likely be favored over grain sorghum as a biofuel feedstock in the Perry Lake watershed, and similar watersheds, due to the higher yield potential and suitable climate to produce high corn yields without irrigation. In the Kanopolis Lake watershed, grain sorghum is the favored biofuel feedstock due to the similar yield potentials between corn and grain sorghum, but the lower water quality impacts of grain sorghum production.

3.4.2 Extensification vs. Intensification

The water quality impacts differ between extensification of the landscape (i.e converting CRP land to cropland) and, intensification of current agricultural land (i.e. replacing winter wheat and hay with biofuel feedstock crops). In the Perry Lake watershed, the hay LUC scenarios produced the highest sediment, TN, and TP outputs per area. In the Kanopolis watershed, the CRP LUC scenarios produced the highest TN and TP outputs per area. In both cases the differences are statistically significant ($p < 0.001$) as determined through a one-way ANOVA. Overall, changing winter wheat land-cover to either corn or grain sorghum produced the lowest water quality outputs per km^2. In fact, TN and TP loads from winter wheat scenarios were not statistically different from baseline results. These results suggest that converting current cropland to a more intensive crop, such as corn or grain sorghum, may cause less water quality impacts than converting less intensive agricultural land-uses, such as hay or pasture. Other studies have also confirmed that conversion of current row crop land to biofuel feedstock provides the lowest water quality changes (as opposed to converting non-row crop land), and in some cases can result in a decrease in water quality outputs compared to the current baseline (Love and Nejadhashemi 2011). For example, the Love and Nejadhashemi study found that a sorghum-soybean rotation had the potential to reduce nitrogen loads when grown on current row crop land.

Returning CRP land-use into production would also have greater environmental impacts beyond what can be analyzed in this study. CRP land sequesters carbon, maintains marginal land, provides habitat for birds and grassland species, and supports re-emerging grassland ecosystems (Wright and Wimberly 2013). Tallgrass and mixed-grass prairies have been converted to agriculture at extremely high rates in the past, and remaining areas are critical for ecosystem conservation (Samson and Knopf 1994). Interviews of Kansas farmers indicate that many farmers support the CRP program and see their participation as important for being a good steward to the land. However, some farmers indicated that they have already converted CRP land to cropland at the end of contracts, or have expressed interest in doing so (Brown et al. 2014; Gray and Gibson 2013). Farmers cite income potential of grain production and land scarcity as reasons for converting CRP to cultivated land. Therefore, it is challenging to predict the amount of CRP land that may be converted back to agriculture, as personal values and economic factors both play a large role in land-use decision-making.

Intensification scenarios also pose additional problems that are not quantified in this study, such as the direct and indirect effects of replacing food-related crops with crops dedicated to the biofuel market. Kansas is consistently the number one or two producer of winter wheat in the United States, representing about 14% of the market (Kansas Department of Agriculture

2014). Replacing winter wheat with biofuel feedstocks could interfere with the commodity market for wheat, causing a rise in food prices. Similarly, hay production is used to support the cattle industry, especially in Eastern Kansas. Therefore, substituting either of these crops at a large scale may have consequences for agricultural production for human consumption. The food vs. fuel issue is central to the biofuel debate, including the concern for rising food prices as food crops are diverted to fuel production (Cassman and Liska 2007; Tilman et al. 2009). Specifically, there have been concerns about increasing greenhouse gas emissions, biodiversity loss, and water quality degradation due to cropland expansion for food crops if biofuel feedstocks are grown on currently utilized fertile land (Searchinger et al. 2008; Wright and Wimberly 2013).

3.4.3 Water Quality Costs

Water quality indicators show that increasing production of ethanol feedstocks in the Kanopolis Lake and Perry Lake watersheds will cause increased sediment, nitrogen, and phosphorus inputs into the respective reservoir systems. Based on bathymetric surveys, the Kanopolis reservoir has already lost about 36% of the multipurpose pool water-storage capacity due to sedimentation, and the Perry reservoir has lost 19% (Juracek 2015). The sediment trap efficiency for Kanopolis Lake is estimated to be 95%; therefore, most incoming sediment will remain in the reservoir causing further storage loss (Juracek 2011). Sediment removal is costly and can cause further environmental damage due to the invasive nature of the process and disposal of removed sediment (deNoyelles and Jakubauskas 2008). Reducing the current sediment load by 72% (about 736,000 tons/year) to Perry Lake through a combination of cropland best management practices (BMPs) is estimated to cost about $500,000 over the next 30 years (or $212,000 with use of cost sharing programs) (Bosworth 2011).

In addition, all uses in both Perry Lake and Kanopolis Lake are impaired by eutrophication caused by nonpoint source nutrient pollution, and both lakes are under total maximum daily load (TMDL) plans in order to improve water quality conditions (Kansas Department of Health and Environment 2012, 2004). The nitrogen load needs to be reduced by 70%, or a total of 388,000 kg/year, and the phosphorus load also needs to be reduced by 71%, or 80,000 kg/year, in order to meet the Perry Lake TMDLs. Targeted BMPs to reach water quality goals are primarily focused on reducing non-point source inputs from cropland, livestock, and streambank erosion (Bosworth 2011). In the Kanopolis Lake watershed, non-point sources need to be reduced by 52% for TN (131,000 kg/year) and 48% for TP (17,000 kg/year) in order to meet TMDL targets (Minson et al. 2011). The total estimated cost of cropland BMPs in the Kanopolis Lake watershed is estimated to be $9.5 million over the next 30 years, with landowner investment representing 30% of the total cost after cost share programs. It is clear that water quality improvement is necessary in these two study watersheds, and unfortunately it will not be achieved without substantial investment from both landowners and government agencies. Therefore, increased development of biofuel feedstocks that would further degrade watershed water quality should be carefully considered.

One important consideration is that BMPs were not modeled in this study. It is possible to model BMPs within SWAT, but this requires the modification of many different parameters such as the curve number, the USLE P-factor, and channel routing variables (Tuppad et al. 2010). These parameters were all included in the parameter set used to calibrate streamflow and sediment load. Altering parameters further without any knowledge of BMP location would further increase model uncertainty. As scenario results were compared to a baseline scenario the omission of BMPs should not change the trend of the results discussed here; if BMPs were to be modeled, they should impact the baseline and scenario results similarly. However, in light of future research, there is potential for biofuel feedstock land-use development to coincide with BMP development, and this could offset some of the negative water quality impacts. Future research should focus on coupling LUC scenarios with BMP development, which would require a different LUC allocation approach other than the one used in this study.

3.5 Conclusion

It is impossible to know for certain how land-use patterns will respond to future grain and fuel prices, land scarcity, government regulations, and farmers' decisions. The literature suggests that both intensification and extensification of agricultural land for biofuel feedstock development is highly possible, and in fact already occurring in Kansas (Brown et al. 2014). However, until now, there was not a study that explored the impacts of such LUC in Kansas. This study contributes to the discussion of environmental impacts of biofuel-based LUC in Kansas.

In this study, the SWAT model was used to simulate grain sorghum and corn production on current winter wheat, hay, and CRP land-uses in two Kansas watersheds. The overall results indicate that replacing hay and CRP land with grain sorghum or corn will cause, on average, an increase of 7,020 – 13,333 kg-TN/km^2 and 1,602 – 3,179 kg-TP/km^2 in the Perry Lake watershed, and an increase of 94 – 329 kg-TN/km^2 and 38 -164 kg-TP/km^2 in the Kanopolis Lake watershed (in all cases increases are compared to the baseline scenario). Replacing winter wheat land-use with grain sorghum and corn produced smaller increases in non-point source nutrient and sediment pollution, which are not statistically different from baseline values. For example, from winter wheat scenarios there was an average increase in 114 – 1,984 kg-TN/km^2 and 552 – 614 kg-TP/km^2 in the Perry Lake watershed. These results suggest that intensification of current row-crop agricultural land may be a more environmentally sustainable option for increasing biofuel feedstock production that converting hay or CRP land.

In addition, this study evaluates sustainability measures for grain sorghum and corn scenarios in two watersheds. Corn had a higher land, nutrient, and water use efficiency in the Perry Lake watershed as corn yield per km^2 is twice as high as grain sorghum yield. The higher yield and higher conversion rate from grain to ethanol, makes it possible to produce more than twice the ethanol with corn than grain sorghum with the same amount of land in the Perry Lake watershed. However, in the Kanopolis Lake watershed the land-use efficiency was similar between corn and grain sorghum. Also, grain sorghum had a higher nutrient use efficiency and corn had a higher water use efficiency; however, the differences were small in both cases. These results suggest that grain sorghum may be a more sustainable feedstock crop in drier climates and corn may be more sustainable in wetter climates. Sustainability measures allow comparison between crops and between watersheds, yet they are typically not included in the current biofuel-based land-use analyses in the literature. This study integrates water quality analysis with sustainability indicators to develop a richer assessment of the trade-offs and benefits of landscape change for biofuel feedstock development.

The land-use simulations explored in this study can help aid decision-making by providing guidance on expected yield from feedstocks in varying geographic locations, as well as potential environmental degradation that may occur from enhanced feedstock development. Simulations need to occur at various scales, from regional to local, in order to aid decision-making from the federal to state levels. This study provides a Kansas perspective and may be helpful in considering environmental impacts of biofuel development in other Great Plains ecoregions as well.

3.6 References

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Chapter 4 – Impacts of climate change on reservoir services and strategies for management

Abstract

Reservoirs are critical resources for economic growth, and provide numerable social and ecological services. Yet, climate change may drastically alter reservoir systems, requiring adaptive management techniques. Currently, loss of storage capacity due to sedimentation, water quality degradation, and toxins from blue-green algal blooms are issues that threaten reservoir sustainability. Climate change is hypothesized to exacerbate these problems by increasing sediment and nutrient export from the surrounding watersheds, changing flow regimes, and increasing summer water temperatures. This study adds to the reservoir management literature by providing a synthesis of the disparate literature on potential impacts of climate change to reservoir services, and provides a review of both watershed and in-reservoir management strategies to mitigate the impacts of climate change. In addition, this study can serve as a resource for managers that seek to study the impacts of climate change on a particular system by providing a compilation of tools and data sources that have been successfully used to study the impacts of climate change on reservoir systems.

4.1 Introduction

Reservoirs are critical economic infrastructures, reflecting billions of dollars invested around the world. Many large reservoirs in the United States have been built for hydroelectric power, water supply, or flood control purposes. Yet despite the initial prescribed reservoir purpose, reservoirs are perceived as multi-use infrastructures with the ability to provide several essential services to regional populations. For example, more than 27,000 reservoirs listed in the National Inventory of Dams (NID) have a primary purpose of recreation, which is 32% of total listed reservoirs (USACE, 2013). Many others have purposes such as irrigation, debris control, navigation, fire protection, and fish and wildlife support. Approximately 50% of all dams listed in the NID were built between 1950 and 1979, with only 10% completed in the last two decades (USACE, 2013). While some dams are nearing the end of their prescribed design life, there are economic, social, and environmental incentives to use reservoir and watershed management to ensure continued utility of these existing investments.

Managers are already faced with serious issues, such as declining water levels in some Western US reservoirs and sedimentation and algal blooms in Midwest US reservoirs, both challenging the long-term sustainability of multiple reservoir uses. However, climate change is expected to amplify water shortages, erosion (Nearing et al. 2004), and the frequency of algal blooms (Paerl and Huisman 2009), which will create additional complications for reservoir management. The uncertainty of the intensity and duration of future droughts, as well as extreme precipitation events, are both of concern and challenging to planning efforts.

The impacts of climate change for natural lakes are studied quite often (for example, see Mortsch and Quinn 1996, Blenckner 2005, Pham et al. 2008, Adrian et al. 2009, and Schindler 2009). However, reservoirs differ from lakes in several critical ways. First, reservoirs typically have larger watersheds compared to lakes, which means there is generally higher water, phosphorus, and nitrogen loads into reservoirs (Kennedy 2005). Second, surface area is also generally greater for reservoirs, which increases evaporative potential. Finally, reservoir drawdown zones can be much greater than those of lakes, which can have an effect on erosion and ecological processes in the littoral regions of reservoirs (Furey et al. 2004). While the impacts of climate change on reservoirs are often studied from a water supply perspective, see: (Park and Kim 2014, Li et al. 2010a, Raje and Mujumdar 2010, Alvarez et al. 2014, Georgakakos et al. 2012), reservoir water quality management issues are infrequently considered in the context of climate change (Zhou and Guo 2013).

The goal of this study is to synthesize the available literature and to review data sources and tools that can be used to understand the possible impacts of climate change specifically to reservoir systems. While specific management decisions for a given system will be case-specific, and it is impossible to make over-arching recommendations for all reservoirs, this review can serve as an inventory of possible management solutions, tools, and data sources that may be useful to develop climate adaption strategies for reservoir systems. Kansas is used as a case study to discuss particular impacts and management efforts; however, the management strategies and tools are broadly applicable.

4.2 Reservoir-related services

The concept of reservoir services provided is a useful framework to evaluate the current benefits derived from reservoirs, especially as water quality metrics that are typically used in reservoir management may not be relevant to the public. For example, metrics such as total phosphorus, total nitrogen, and chlorophyll-*a* concentrations are typically used to assess water quality conditions. Yet the public is more interested to know if they can swim, fish, or boat safely on the water (Keeler et al. 2012). The framework of reservoir services can capture the link between biological and physical measurements and the economic and social importance of the water body. Reservoir services include: hydropower, flood control, recreation, nutrient attenuation, aquatic ecosystem support, and water supply for municipal, industrial, and agricultural uses.

For example, the lack of natural lakes in Kansas amplifies the value of reservoirs within the state. There are few alternative natural water bodies for outdoor recreation, boating, sailing, fishing, and swimming, besides the limited use of rivers and streams. The annual value of Kansas reservoirs for recreation has been estimated around $15 million for Perry Lake, $17 million for Milford Lake and $12 million for Tuttle Creek Lake (in 2009 dollars) (CDM Federal Programs Corporation 2011). Additionally, there are 93 reservoirs within the state that serve as water supply and approximately 60% of the state's population receives drinking water from these reservoirs (deNoyelles and Jakubauskas 2008). For Perry, Milford, and Tuttle Creek reservoirs, the value of water supply is estimated at approximately $294 million when including the avoided costs of constructing new reservoirs and estimated mitigation costs for maintaining water supply (CDM Federal Programs Corporation 2011). Valuing reservoir services has provided useful in other case studies: such as, reregulating flow releases from the Glen Canyon Dam on the Colorado River, and determining the recreational benefits of maintaining higher water levels on dams operated by the Tennessee Valley Authority (Loomis 2000).

While all reservoir services are important, some are irreplaceable. Surface water storage is an essential service. In many parts of the US groundwater sources are declining and reservoirs are relied on to provide freshwater to support growing populations and agricultural production.

Reservoir-derived ecosystem services such as water supply, water quality, recreation, nutrient attenuation, habitat and navigation will all be affected by climate change (Table 1).

Table 4-1: Reservoir services and possible impacts due to climate change

Reservoir-Derived Ecosystem Services	**Possible Effects due to Climate Change**
Flood Control	Overwhelm flood control capacity
Water Supply: Municipal	Sedimentation diminishes water supply capacity; uncertainty in drought adds water supply stress; increased nutrient loading will increase eutrophication and algal blooms creating taste and odor events
Water Supply: Industrial	Sedimentation diminishes water supply capacity; uncertainty in drought adds water supply stress; decrease in water quality may require additional treatment before water use
Water Supply: Agriculture	Sedimentation diminishes water supply capacity; uncertainty in drought adds water supply stress; potential for increased salinity
Power Generation	Decreased inflow may bring water levels below turbines and decrease power generating potential
Recreation	Decreased water levels may prevent many aquatic recreation activities; poor water quality and algal blooms limit recreation availability
Habitat for Aquatic Organisms	Poor water quality and turbid waters may stress some aquatic species and allow invasive species to take over, or may cause loss of threatened or endangered species
Nutrient Attenuation	Stressed water bodies will not be able to effectively attenuate nutrients and may export additional nutrients downstream
Navigation (releases to rivers)	Water supply stress may limit the amount of water available for navigation releases

4.3 Possible impacts of climate change on reservoir services

The scientific community has unequivocally demonstrated that the earth is warming due to increased concentrations of greenhouse gases in the atmosphere, which could lead to increased occurrence and magnification of drought, alteration of geographic and temporal precipitation patterns, and intensification of precipitation events (Pachauri and Reisinger 2007, Seneviratne et al. 2012). While it is challenging to reliably predict impacts on a local scale, evidence suggests that climate change will have serious consequences for water management systems both due to increased vulnerability to drought and flooding (Handmer et al. 2012). The scientific research community is currently evaluating the potential global and local impacts of climate change scenarios on water resources and water quality through modeling and empirical studies (Firth and Fisher 1992, Schindler 1997, Whitehead et al. 2009, Brekke 2010).

4.3.1 Climate impacts on erosion and reservoir sedimentation

Reservoirs in agricultural watersheds typically have problems with excess sedimentation. It is costly to dredge and remove annual sediment inflow, and challenging to dispose of dredged material without causing further environmental degradation (CDM Federal Programs Corporation 2011). Major sediment sources come from cropland and grazing lands within the watershed, but also from eroding streambeds and streambanks (Devlin and Barnes 2008, Juracek and Ziegler 2009, Juracek 2011). On land, soil is mobilized by three types of erosion: sheet, rill, and gully erosion, which are all expected to increase with climate change. Soil erosivity for the US as a whole is expected to increase anywhere from 17-58%, with a great deal of variability between regions (Nearing 2001). While the sensitivity of runoff and soil loss to precipitation change is also expected to increase, with an expected 1.7% change in erosion for each 1% change in precipitation (Nearing et al. 2004). Simulations in Midwestern watersheds show that a later planting date for crops such as corn and soybeans, in combination with climate change, can have a significant impact on the severity of erosion, as cropland will be uncovered during April and May storms (O'Neal et al. 2005).

Streambank erosion is also a significant problem. A study of Perry Lake in Kansas demonstrated that stream bank erosion above the reservoir was more important than surface soils in the overall amount of transported sediment (Juracek and Ziegler 2009). Stream segments below reservoirs also experience erosion: when the majority of the sediment load is deposited in the reservoir, outflow has very low total suspended solids and has the capacity to erode sediment from the channel and streambed to reach equilibrium sediment load (Juracek 2011). High flow events, which are predicted to increase with climate change, would also most likely increase the amount of streambank and bed erosion; as flow velocity and turbulence increases, the sediment load capacity would also increase allowing for additional bank erosion.

4.3.2 Increased eutrophication potential

Both increased temperature and altered precipitation regimes could increase the export of nitrogen, phosphorus, and carbon from surrounding watersheds into reservoirs. Increased temperature has the potential to increase nitrification in soils and N-availability, as well as mineralization and release of phosphorus and carbon from soil organic matter (Delpla et al. 2009). After a period of drought, there would be high concentrations of mineralized nutrients in the soil and a precipitation event would wash that material into the nearest waterbody.

In contrast, periods of minimal precipitation and low inflow will increase lake residence time. Reservoirs may become fairly stagnant with little outflow in order to preserve water levels and storage. Increased lake residence time will increase the risk for elevated nutrient concentrations and algal blooms (Delpla et al. 2009). For example, Olds et al. found that water quality parameters varied significantly between drought and normal conditions for Harlan County Reservoir in Nebraska. Chl a and turbidity were significantly higher, and dissolved oxygen was significantly lower, during drought conditions, even though water temperature remained constant (Olds et al. 2011). However, a rise in water temperature can also create a decrease in dissolved oxygen (DO) solubility. If increased water temperatures coincide with high concentrations of nutrients and algae in the water column, DO could reach very low levels in bottom waters, putting aquatic life at risk (Whitehead et al. 2009). Additionally, low DO concentrations lead to a reducing environment in which iron hydroxides dissolve releasing additional phosphorus that was bound to iron-rich sediments, creating an internal P-load which further exacerbates eutrophication. Internal P-load can also be amplified by suspension of benthic sediments due to lower water levels and increased turbidity from drought conditions (Dzialowski et al. 2008, Wildman Jr and Hering 2011). A conceptual diagram showcasing the relationships between increasing precipitation and temperature and eutrophication is shown in Figure 1.

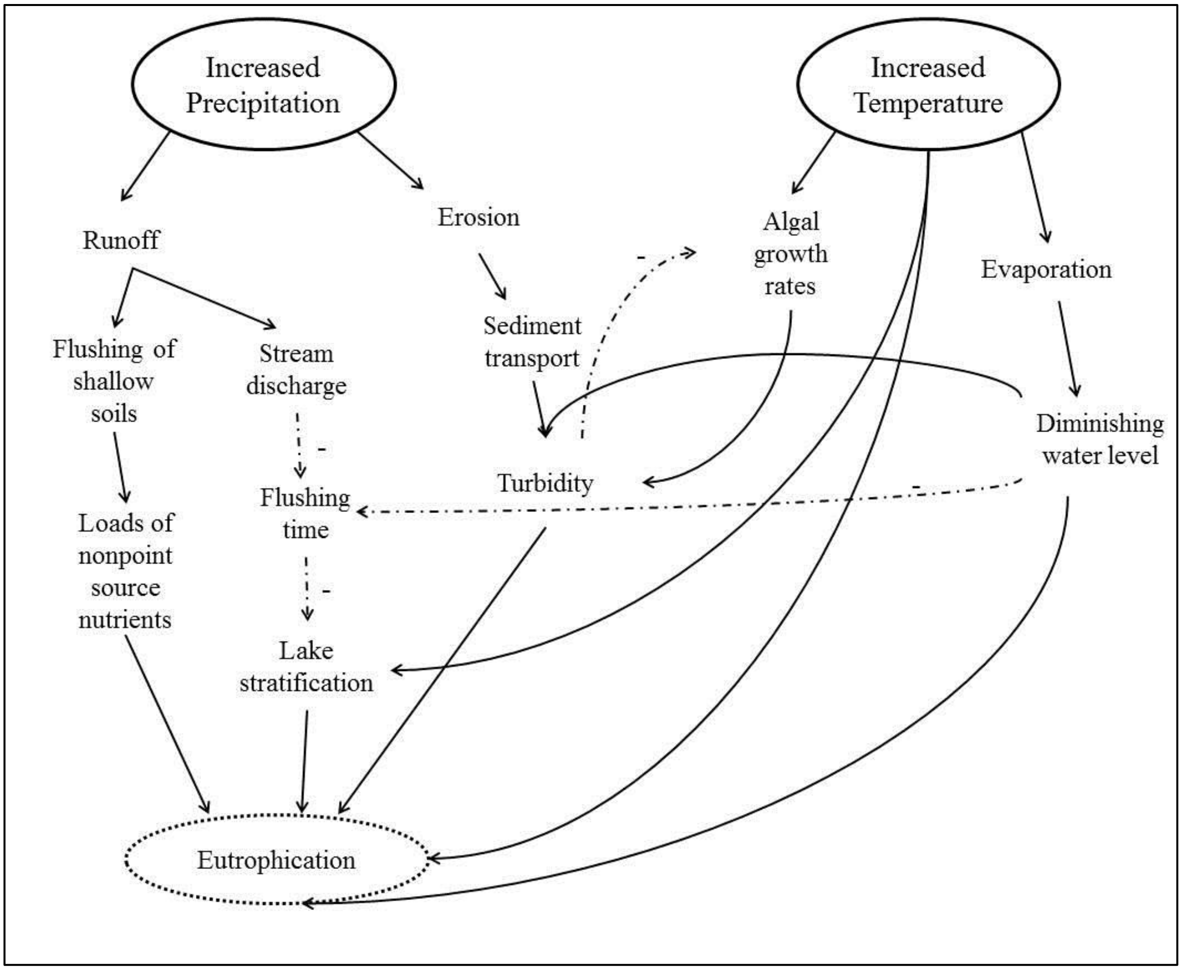

Figure 4-1: Positive and negative feedback loops between climate factors, watershed processes, and eutrophication. Negative feedback is indicated with a dotted line and a negative sign (-).

It is hypothesized that climate change will be a major factor in the rising occurrence of algal blooms globally through increases in water temperature, water residence time, vertical stratification, and nutrient pulses from extreme events (Paerl and Huisman 2009). In a warmer climate, cyanobacteria may have a competitive advantage for resources, as cyanobacteria often have optimal growth rates at high temperatures (i.e. above 25°C) (Johnk et al. 2008, Paerl and Huisman 2009). Blue-green algae, or cyanobacteria, can release toxins that are harmful for both humans and animals when ingested or in skin contact (Codd 2000, Codd et al. 2005). Whether or not toxins are produced, algal blooms create taste and odor problems for municipal water suppliers. Taste and odor events can be treated with powdered activated carbon, but conventional treatment methods do not seem to remove all toxins or taste and odor compounds (Jung et al. 2004). In some cases, it may be necessary to invest in expensive water treatment systems that rely on ozone or advanced oxidation treatment technologies (Srinivasan and Sorial 2011). Treatment is further complicated because it is challenging to predict the occurrence of algal blooms, as many variables, such as nitrogen and phosphorus concentrations, temperature, stratification, and the presence and/or dominance of other competing species, may be critical factors (Smith 1983, Dzialowski et al. 2011).

4.3.3 Increased likelihood of drought-related impacts

The importance of adaptive reservoir management is brought into focus during times of drought. The Colorado River Basin developed a plan for drought and reservoir management due to a prolonged period of drought that began in 2000. The plan was known as the "Interim Guidelines" and ushered forth a new strategy for water conservation and coordinated operation of Lakes Powell and Mead (Rajagopalan et al. 2009). In addition, drought conditions highlighted the need to prepare for future climate change, where diminishing water levels may be the norm (Barnett and Pierce 2008).

Similarly, in a much smaller system, drought conditions during 2012 and 2013 brought the issue of reservoir sustainability into focus for Kansas, as many Kansas reservoirs declined to critically low levels. The concern for water management initiated a 50-year water planning effort within the state. Drought often highlights reservoir vulnerabilities that often have been present all along. For example, declining reservoir inflow in Kansas has been persistent over time. Several reservoirs in Western Kansas, such as Cedar Bluff, Keith Sebelius, Webster, and Kirwin reservoirs, have experienced drastic reduction in inflow since construction periods: -88% since 1950, -56% since 1962, -77% since 1945, and -50% since 1953, respectively (percent change is annual inflow compared from first to last decades of record). Additionally, approximately 68% of Cedar Bluff's inflow, and 83% of Keith Sebelius' inflow volume is calculated to go to evaporation demand (Brikowski 2008). These statistics demonstrate that Cedar Bluff and Keith Sebelius are possibly unsustainable reservoirs that could fail during drought conditions.

Planning for drought conditions is extremely challenging, as a limited historical dataset is available to study recent drought periods and use as a proxy for future conditions. Using past records to determine a range of variability uses the assumption of stationarity. Stationarity is the idea that natural systems operate within an envelope of climate variability that can be inferred from historically measured streamflow and weather variations in a designated spatial area. Climate change predictions debunk the assumption of stationarity and several prominent hydrologists have boldly stated that "stationarity is dead" (Milly 2008, Wagener et al. 2010). Without the use of the methods derived from the assumption of stationarity, water resource managers are faced with complex problems and a set of aging tools that may not be suited to provide answers.

4.3.4 Increased watershed transport from extreme events

The assumption of stationarity has also become invalid when we consider extreme precipitation events that may occur in the future. An increase in greenhouse gases in the atmosphere will lead to higher temperatures, and thus increased evaporation, water-holding capacity and content in the atmosphere, which ultimately can lead to enhanced precipitation rates (Trenberth 1999). Modeling results confirm that the proportion of rainfall from heavy events will likely increase for most areas of the globe in the 21st century (Seneviratne et al. 2012).

Currently, there are not any modeling studies that demonstrate the possible range of water quality impacts from an extreme event analysis. Yet analysis of past extreme events have shown that high intensity precipitation events are linked to increased turbidity and high sediment loads, which in turn are connected to high phosphorus export from the watershed (Murdoch et al. 2000, Stutter et al. 2008). Also, with longer dry periods in between storms, pollutants will build up in the watershed and result in high pollutant concentrations in storm runoff. Large pulses of nutrient- and sediment-rich storm water could create significant short term water quality changes that may exceed biologically relevant thresholds and have long term effects on the ecosystem balance (Murdoch et al. 2000). Extreme events would also likely increase both soil detachment and transport capacity of eroded material. There is empirical evidence that suggests that infrequent, yet large, rainfall and runoff events are already responsible for a greater proportion of overall erosion. In a 28-year study it was found that the five-largest events contributed 66% of the total erosion within Ohio watersheds (Edwards and Owens 1991). Also, in Kansas a study of sediment sources to large federal reservoirs in Kansas, Kanopolis Lake and Tuttle Creek Lake, indicated that large storms are responsible for the majority of transported sediment. For example, at the Ellsworth streamgage located upstream of Kanopolis Lake, in 2010 seven storms accounted for about 48% of the total discharge and 88% of the total suspended sediment load (Juracek 2011).

4.4 Management solutions for addressing the impacts of climate change on reservoir ecosystem services

4.4.1 Watershed management

Solutions to water quality problems are often found at the watershed scale. As watershed-focused management considers climate and landscape factors, it can be used to develop practices that are well-designed to accommodate future climate changes.

Land management

Land treatment and management strategies can vary greatly. Common vegetative treatments, or Best Management Practices (BMPs), for cropland include: cover cropping, crop rotations with grasses or legumes, crop residue application, mulching, and planting woody or grass species in critical areas (Vanoni 2006). Vegetative BMPs are commonly used in the agricultural sector as they also decrease high nutrient runoff, and sustain healthy cropland. Other more mechanical BMPs may include contour farming, no-till, or terraces. In addition, grassed waterways, vegetative buffers, sediment traps, and riparian forest buffers can be built to intercept sediment and nutrient-rich runoff from entering waterways (Vanoni 2006, Devlin and Barnes 2008). On grazing land increasing plant density and adequate cover, while also limiting grazing on erosive lands or critical runoff areas, can prevent excess sediment loss.

Land management approaches often require personal investment from land owners and, therefore, are challenging to implement without an economic incentive. BMPs must be targeted to the land that will provide the greatest reduction in sediment and/or nutrient export (Tuppad et al. 2010, Daggupati et al. 2011). For example, a study in Tuttle Creek Lake watershed in Kansas determined through model simulations that targeted BMP application cost-effectively prevented

260,893 tons of sediment transport into Tuttle Creek Lake per year, but that random BMP application was less cost-effective than reservoir dredging (Smith et al. 2013).

Structural measures can also be applied within the watershed to erosive streambanks and reservoir shorelines. Streambank restoration has demonstrated potential to reduce sediment load to reservoirs and may be necessary to increase stability for high flow events (USEPA 2008). The contribution of reservoir shorelines is often overlooked in studies of sediment sources. Yet, erosion of unprotected shorelines may occur during reservoir level fluctuations or by increased wave energy from storm events. Installation of breakwaters in strategic locations may stabilize reservoir shorelines and decrease sediment inflow, but can be costly, ranging from $200 - $855 per linear meter (Pape 2004, Severson et al. 2009).

Watershed structures

Upstream debris dams and sediment basins can help slow flow and trap sand and silt that may be transported down into the reservoir. These impoundments can be periodically dredged of material at a greater convenience and reduced cost compared to large reservoir dredging (Vanoni 2006). Wetlands are also effective at trapping sediment, retaining water during high flow periods, and attenuating nutrient loads. Strategically-placed constructed or restored wetlands in watershed headwaters or near reservoirs could possibly ameliorate the impacts of large precipitation events (Kadlec and Wallace 2008). Finally, off-channel storage could help store water, trap sediment and attenuate nutrients before water is transported to the reservoir.

Comprehensive watershed management: Case Study - Kansas WRAPS Program

Kansas has a highly developed watershed-based management program, the Watershed Restoration and Protection Strategy (WRAPS) Program that is based out of the Kansas

Department of Health and Environment (KDHE), and involves collaboration between several state agencies. WRAPS involves a planning and management framework that is based on local stakeholder involvement. Stakeholders are responsible for developing a watershed assessment, establishing goals and identifying necessary actions and costs, preparing a watershed plan, and securing resources needed to execute that plan (KDHE 2012). WRAPS groups are guided by KDHE staff and scientists working through Kansas State Extension Services.

The WRAPS program works primarily to establish BMPs where they are most needed in the watershed. Targeted areas are determined with watershed modeling and then stakeholder groups work to achieve cooperation from necessary landowners. This program seems to be an effective way to implement watershed management on a local scale. As stakeholders are intimately involved in planning and goal-setting, this program represents a semi bottom-up approach that can take into account the views and values of watershed residents. Success has already been demonstrated in several WRAPS watersheds where 303d listings (in relation to the Clean Water Act) have been lifted after conditions improved (USEPA 2009, USEPA 2012). For example, dissolved oxygen conditions improved in Toronto Reservoir after repairs to and installation of agricultural ponds, livestock fencing, and watering facility units. Land use changes, such as pasture and hay land planting, and critical area planting to reduce runoff, also were utilized in the Toronto Reservoir watershed.

Can these plans aid the management of highly variable flow and water quality conditions due to climate change? While climate change is not addressed in these plans, it is being considered by some of the researchers collaborating with the WRAPS program (Sheshukov et al. 2011). Additionally, these plans are a huge step forward with respect to local engagement in the problem of water quality degradation. The planning and action steps involved in the WRAPS program also would decrease the vulnerability of the participating watersheds to climate change.

4.4.2 In-lake management

In most cases, watershed management is preferred as a long-term solution to reservoir water quality and sedimentation issues; however, there is a place for in-lake management as well. There are a variety of different management techniques available with varying applicability.

In-lake sediment management

In many areas reservoirs are rapidly filling with sediment and action is needed to preserve water resource investments. Some techniques, such as inflow routing and density current venting, require some degree of thermal or density stratification in order to work correctly (Baker and deNoyelles 2008). Inflow routing and density current routing attempt to route turbid inflow water through the reservoir as a density current where it is then released downstream, preventing maximum sedimentation in the reservoir (Morris et al. 2006). Sluicing, or flood flushing, moves the sediment load through the reservoir during a high-flow event and requires a low water level during the flood season to maintain flow velocity. Sluicing takes advantage of the silt-carrying capacity of the floodwaters to flush these particles closer to the dam and then out of the reservoir. Sediment loaded water is then flushed out of the reservoir as the hydrograph rises, and gates can be closed to trap relatively low-sediment waters on the falling limb of the hydrograph (Durgunoglu and Singh 1993).

Once sediment has settled in the reservoir, it can be removed by hydraulic flushing, where sediment is carried by water through a low-level outlet. Reservoir levels must be low and reservoir inflow must be high for this method to be successful (Palmieri et al. 2001). Any of these techniques that flush sediment through the reservoir could be detrimental for downstream ecosystems depending on the amount of sediment released. Also, in some cases, managing water levels for flushing or routing may compromise the ability to retain adequate water storage, especially if the reservoir pool is not able to rise and hold seasonal inflow. As mentioned, these methods require some degree of density or thermal stratification, which may not be possible to obtain in reservoirs that typically have long fetches and are adequately wind-mixed.

More drastic solutions for removing sediment include hydraulic dredging and dry excavation. Dredging involves using a barge to loosen consolidated sediments and pump out this sediment-rich slurry. Dry excavation requires the reservoir to be drained for the sediment to be excavated and removed. Both options are expensive, require a large area for sediment disposal or storage, and can be damaging to the reservoir ecosystem.

In-lake nutrient management

While the first step in reservoir nutrient management is reducing nutrient loads into the waterbody through watershed management efforts, in some cases when a precipitation event occurs soon after fertilizer application, in-lake management might be desired to reduce the impacts of incoming nutrients. In-lake management includes selective withdrawal, aeration, change in lake level management plants, and altering hydraulic residence time (Baker and deNoyelles 2008). Multi-level selective withdrawal requires some degree of stratification, and therefore may not be a useful for all reservoirs, or those that do not have the proper outlet structure. If stratification occurs at the dam, water can be released from a layer that may contain undesired water quality conditions, such as an anoxic hypolimnion or a eutrophic epilimnion. Selective withdrawal forces mixing of the water column and has been shown to reduce hypolimnetic anoxia and algal blooms (Lehman 2014). Releasing anoxic waters could cause problems for downstream ecosystems, yet this can be mitigated through release mechanisms that oxygenate the water. Aeration can also be used in cases where the hypolimnion is anoxic. Aeration will help keep organic matter suspended in the water column and prevent it from settling into the hypolimnion where bacterial consumption will deplete oxygen (Beutel and Horne 1999). Aeration may also help correct eutrophication-related taste and odor problems.

Change in lake level management plans, or rule curves, is a way to alter hydraulic residence times (HRT) to either retain or flush flow through the reservoir. These plans allow seasonal changes in elevation that allow for flood control, water storage, hydropower generation, recreation and ecological needs. Rule curves vary greatly by reservoir depending on regional climate and dominant reservoir functions. If possible, plans could be altered in order to flush water high in nitrogen or phosphorus and aim for a strategic HRT that would maintain higher quality water throughout the summer. Studies would need to be done to determine the optimal HRT for each reservoir.

4.5 Climate adaptation in reservoir management

Many state and federal agencies are involved in the collection and analysis of reservoir information. Reservoir sustainability is an important issue that is not over-looked by resource managers. However, in the midst of many proactive studies and projects, climate remains the wild card. Past climate data is used as a proxy for possible future droughts and floods. Yet is this past data sufficient? With the concept of stationarity no longer valid, should there be a push to move to a new paradigm that includes the uncertainty of both gradual atmospheric warming and increased precipitation variability?

In an era of uncertainty reservoir managers will need to use flexible methods to adapt to a changing climate. Adaptive policies and strategies can be developed through simulation modeling. The most common approach is to combine a series of climate, hydrologic, and reservoir and/or ecological models. First, results from Global Climate Models (GCMs) are used to drive regional climate models or weather generators in order to generate more location-specific climate parameters. Next, the generated climate parameters are used in a calibrated hydrologic model that can generate streamflow and nutrient inputs into the reservoir system. Then the streamflow results are input into a reservoir optimization model to examine possible management strategies, or into an ecological or water quality model to predict algal biomass, nutrient concentrations, oxygen demand, and a variety of other parameters of concern. For example, such studies have been conducted for the Hirakud reservoir in India (Raje and Mujumdar 2010), the Chungju dam in South Korea (Park and Kim 2014), and for the Northern California water and power system (Georgakakos et al. 2012).

Such an approach is incredibly time and resource intensive, unless hydrologic and/or water quality models are already developed for a system, and also requires a great deal of data and technical expertise. Examples of the modeling tools and mathematical approaches, and data sources that have been used to conduct such analyses are presented in tables 2, 3, and 4, respectively. In some cases the use of future climate data may require collaboration between resource managers and climate scientists.

Reservoirs are built for multi-generational use and are often operated with a long-term approach; therefore modeling efforts that project 20-50 years into the future may be helpful for guiding long-term planning efforts. Such simulations can provide an estimation of the range and probabilities of impacts to local systems, which can be useful for a risk assessment framework, and for reservoir planning and management efforts.

Table 4-2: Mathematical approaches used to study impacts of climate change on reservoirs.

Approach	Description	Example of Application
System Dynamics	Describes relationship between the rate of change in system variables in relation to inputs by using first-order differential equation with a time lag	Li et al. 2010b, Chen and Wei 2014
Analog ESP	Develops relationship between streamflow and hydro-climate factors, which can then be extended to future climate	Yao and Georgakakos 2001
Statistical P-loss Model	Develops relationship between P loss from watershed and critical hydrologic and watershed variables and then can use projected flow to estimate P loss under climate change	Jeppesen et al. 2009
2D Reservoir Water Quality Mathematical Model	Links mathematical equations describing flow and transport, ecological interactions, and water-sediment interchange	Komatsu et al. 2007

Table 4-3: Tools used to study impacts of climate change on reservoirs and adaptive management techniques.

Tool	**Description**	**Developer**	**Example of Application**
Integrated Adaptive Optimization Model (IAOM)	Contains three modules: weather generator, hydrological simulator, and multipurpose reservoir optimization to develop optimal operating rule curves under climate change	Y. Zhou and S. Guo	Zhou and Guo 2013
Hydrologic Engineering Center – Reservoir System Simulation (HEC-ResSim)	Uses rule based approach to mimic decision making process	USACE	Park and Kim 2014
Dynamic Hydroclimatological Assessment Model (DYHAM)	Utilizes system dynamics theories and feedback causal loops to simulate dynamic processes within watershed and reservoir	SP Simonovic and LH Li	Li et al. 2010
Phytoplankton Responses to Environmental Change: PROTECH	Simulates the daily change in chl a concentration for up to 10 algal species in response to environmental variability in lakes and reservoirs	A. Elliott, C. Reynolds, T. Irish	Alex Elliott et al. 2005
NAM Rainfall Run-off Model	Deterministic, non-distributed hydrological model	DHI Inc.	Jeppesen et al. 2009
Soil and Water Assessment Tool (SWAT)	Simulates water quality and quantity of surface water and can test scenarios related to land use, land management practices, and climate change	USDA-ARS and Texas A&M	White et al. 2010a, White et al. 2010b
Hydrologic Simulation Program Fortran (HSPF)	Simulates hydrologic and water quality processes on land, in streams, and in well-mixed impoundments.	USGS	Göncü and Albek 2010
Agricultural Non-Point Source Pollution Model (AGNPS)	Evaluates effect of management decisions that may impact water, sediment, and chemical loadings within a watershed	USDA	Booty et al. 2005
Generalized Watershed Loading Function (GWLF)	Dynamically simulates variations in stream discharge and combines with sources of P to estimate P export	K.H. Reckhow	Pierson et al. 2010
BASINS-CAT: Better Assessment Science Integrating Point and Non-Point Sources Climate Assessment Tool	Integrates GIS, national water data, and watershed modeling tools (HSPF) with the added flexibility to incorporate climate change scenarios	USEPA	Taner et al. 2011

Table 4-4: Examples of commonly used data sources available to implement tools and approaches outlined in Table 3 and to study climate impacts on reservoirs

Data Network	**Description**	**Data Source**
Geospatial Data Gateway	Census, average climate, easements, elevation, geology, government units, hydrography, land use and land cover, soils, topography, and transportation data layers available for the US	United States Department of Agriculture (USDA) Natural Resources Conservation Service
Hydroclimatic Data Network (HCDN)	Subset of streamflow stations that can be used to study climate fluctuations in the US	United States Geological Survey (USGS)
US Cooperative Observer Network (COOP)	Over 8000 climate stations operating from as early as 1886, providing daily min and maximum temperature and precipitation data	National Climatic Data Center (NCDC)
Global Historical Climate Network	Daily climate observations from around the world, including min and maximum temperatures, total precipitation, snowfall, and depth of snow	NCDC
US Climate Reference Network (USCRN)	114 climate stations in the conterminous US, as well as 29 stations in Alaska and 2 in Hawaii to detect the national signal of climate change	National Oceanic and Atmospheric Administration (NOAA)
National Weather Service River Forecast Centers	High resolution gridded hydrologic state variables and flux datasets; both observed and forecasted river conditions and precipitation are available	NOAA
Next Generation Radar (NEXRAD) Level III	40+ data products derived from NEXRAD Level II data including precipitation estimates and storm relative velocity	NOAA

4.6 Conclusion

Reservoirs provide numerable services and represent large fiscal investments from previous generations. The sustainability of reservoir services is threatened by excessive sedimentation, algal blooms, and water supply shortages. Climate change could exacerbate these issues and further complicate management of reservoir systems. Watershed and in-reservoir management techniques are available, yet climate adaptation may require thinking beyond current practices and employing simulation modeling to estimate nutrient and water loads, as well as future rates of sedimentation. There are a large variety of tools available, but the data requirements and technical expertise necessary are often limiting. Collaborations between reservoir managers and climate scientists may be necessary to develop simulation modeling platforms that can explore and virtually test adaptive management strategies in the context of altered climate patterns.

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Chapter 5 – Climate change and Kansas water management: perspectives and opportunities

Abstract

Climate change is a critical issue that has begun to shape water management and planning on the federal and state levels. This study focuses on the potential for Kansas water managers to integrate climate change into statewide water planning and management. A survey was employed to understand the personal perspectives of Kansas water managers towards climate change and its integration into state-based water planning. Respondents were targeted at three agencies: the Kansas Department of Health and Environment, the Kansas Water Office, and the Kansas Department of Agriculture – Division of Water Resources; 37 of 64 respondents finished the survey. The survey results, along with a review of key Kansas water management plans, suggest that Kansas water managers are indeed responsive to climate variability and are starting to integrate climate variability into planning efforts. To promote successful integration, helpful lessons from the climate science-policy literature are provided, such as a description of potential barriers and strategies useful for effective integration.

5.1 Introduction

The scientific community has unequivocally demonstrated that the earth is warming due to increased concentrations of greenhouse gases in the atmosphere (Pachauri 2007). Global circulation models have been extensively developed in the past decade and are widely used by the scientific community to evaluate possible future scenarios that may result from an alteration to global atmospheric chemistry (Gent et al. 2011). Yet, it still remains a challenge to integrate future climate simulations and forecasting into water resource planning, especially at the local level (Kiparsky et al. 2012; Rayner et al. 2005).

It is unclear how future climate change directly influences water management and policy decisions in Kansas. In past years Kansas has lagged behind other states in efforts to develop climate change adaptation strategies (Chou and Schroeder 2012). For example, Alaska, California, Colorado, Connecticut, Florida, Massachusetts, Maryland, Maine, Oregon and Pennsylvania are just a few examples of states that have created climate change adaptation and mitigation plans (Georgetown Law 2012). At the time of this study Kansas state agencies have not published any reports or documents that discuss climate adaptation strategies for state planning. Kansas is particularly vulnerable to the effects of climate change, since agriculture is one of the dominant economic sectors of the state. With respect to water management, the effects of climate change on precipitation trends, future water availability, and runoff events are relevant issues for the state.

After years of intense drought, Kansas water managers have begun to focus on climate variability with greater intensity and water management issues have gained greater attention within the realm of statewide policy and planning. For example, in recent years the Governor's Conference on the Future of Water in Kansas has become a widely attended event, offering an arena for interdisciplinary discussion of water issues with state scientists, politicians, academics, farmers, and interested citizens (see: http://www.kwo.org) . In addition, inter-agency efforts directed at long-term water planning have emerged as a means to improve management of state resources. Within this context, a study was developed to analyze the potential for Kansas water managers to integrate climate change into statewide water planning and management. In particular this study seeks to answer several questions:

1) What does the literature of science-policy integration suggest are the common barriers to integrating climate change into water management?
2) What are the personal perspectives of Kansas water managers on climate change and its integration into state-based water planning?
3) How do current Kansas water management plans and programs integrate climate change or variability?
4) What are useful strategies to promote integration of climate change into water resource management?

These questions are explored through a literature analysis, an examination of Kansas water management documents, and the results of a survey designed to gather the perspectives of water resource decision-makers and managers in Kansas. The survey focuses on the integration of predictive climate data into statewide water management. The ultimate goal is that this study can help illuminate the gaps between perceptions and action, and can be used to improve collaborative efforts between university researchers and water managers in Kansas on climate and water-related projects.

5.2 Barriers to Integrating Climate Science into Water Resource Management

There is a great need to integrate predictive climate information into water resource planning and management; however, such integration proves to be a challenge for both scientists and practitioners. With respect to the practitioner, previously identified barriers to integrating climate forecasts into resource management systems include issues such as: competition with other decision-making factors, institutional barriers, and perceived problems with the climate forecast product, such as lack of accuracy, reliability, or credibility, as well as timeliness and dissemination (Kirchhoff 2010; Dilling and Lemos 2011; Rayner et al. 2005). Methods of communication and information transfer can also create significant barriers to using climate science to make resource management decisions. These barriers are described in greater detail below.

Competition with other decision-making factors: There are many factors that go into natural resource decisions, and scientific research is just one of these many components. Scientists may be able to improve the usability of their work if they understand the context and process of decision making and can work to integrate their work within the context of other decision-making factors (Dilling and Lemos 2011).

Institutional barriers: Organizational culture and incentive structures in different institutions (e.g. academia, state agencies, non-governmental organizations) create differing motivations and may cause conflict when trying to conduct collaborative research or implement research findings (Buizer et al. 2010). For example, academic institutions may not reward applied research that is directed towards local use, rather than discoveries in theoretical research, or research with a national or international focus. In addition, inflexible institutional or organizational rules or a risk-adverse or routine-oriented organizational culture in the water resource management sector may encourage managers to rely on traditional planning methods in order to avoid public criticism for failures when using untested approaches (Dilling and Lemos 2011; Rayner, Lach, and Ingram 2005). Preference for decision-making that relies on established practices will impede the integration of untested climate forecasts or new modeling tools.

Perceived problems with climate forecast products: The perception that climate forecasts may be inaccurate, unreliable, or have a high degree of uncertainty is a major roadblock for many practitioners (Brenner 2011). This is an area that can be improved through increased and ongoing communication between scientists and practitioners. Additionally, problems such as timeliness and scale are consistently reported as a barrier to integrating climate information into local management. Information needs to be available at the appropriate scale at the time of the decision; however, often this is not the case. Practitioners may need to make a quick decision and producing scientific results within several days' notice is typically not possible. Researchers are increasingly downscaling climate results for local studies, which will cause the issue of geographic scale to become less relevant.

Process and communication barrier: Another barrier is the way in which scientists and practitioners may engage with one another. There is a historically prevalent perception that the link between science and practice is a "bridge", "pipeline", "superhighway", or other linear structure (Kasperson et al. 2011). The researchers may set the agenda and once the information is prepared, transport this information to potential users, what Dillings and Lemos describe as the "science push" (2011). The push could consist of peer-reviewed articles and conference presentations, whose audience will most likely not include the targeted "user" of this information. In addition, the push could be described as outreach, which more often is a requirement to receive federal funding. However, without careful design and implementation, outreach may not reach, or be understood by, the intended audience. In addition, without knowledge of the institutional setting, scientists may propose solutions or make recommendations that are infeasible or provide tools that are unusable by staff (Weichselgartner and Kasperson 2010).

On the other hand, the practitioners may set the research agenda, either through a call for proposals for funding, or by developing collaborative opportunities, therefore creating the “demand pull”, which then scientists will respond to with the requested information (Dilling and Lemos 2011). This approach has been promoted in the climate change community as a means to develop highly usable research (McNie 2007). Indeed if the “demand pull” creates a collaborative relationship between the researcher and practitioner, this may result in science that is usable for policy and practice. However, in this case the relationship is no longer uni-directional and instead is cyclical and evolving with time and experience (Kasperson and Berberian 2011; Kasperson et al. 2011).

5.3 Water Management in Kansas

Kansas is a predominately agricultural state, and an important contributor to the US and global food market. In 2010 Kansas ranked sixth in the nation for total agricultural export revenue and second for total cropland at over 28 million acres (USDA, 2011). Agriculture is the fuel that continually feeds the state’s economy, and it is a major activity that defines local culture and community in Kansas. Water is inexorably tied to agriculture - water resources are necessary for crop production, and water quality is often damaged as a result of agricultural activities (Carney 2009). Irrigation for crop production is the state’s largest water user, accounting for 80-85% of diverted water. Groundwater represents about 90% of water used in the state (Foley et al. 2014). Portions of western Kansas overly the Ogallala-High Plains aquifer and utilize groundwater, while surface water use from reservoirs and rivers is common in the eastern half of Kansas, as shown in Figure 5-1 (Sophocleous and Wilson 2000). Consequently, the two largest water supply issues in the state are groundwater depletion and sedimentation of reservoirs.

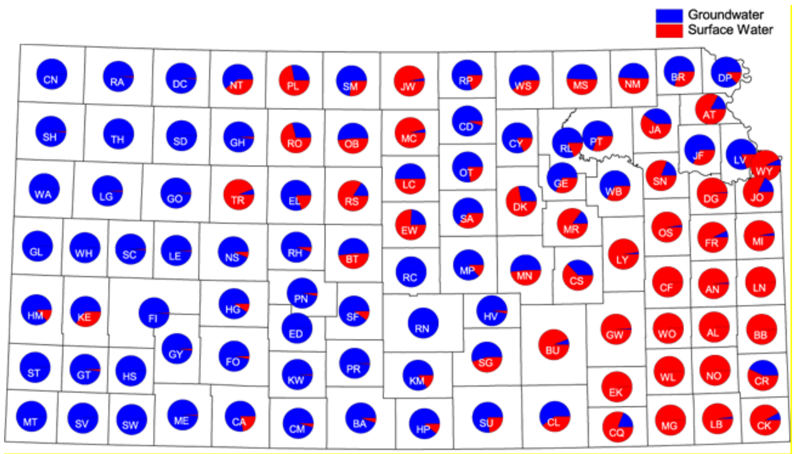

Figure 5-1: Proportion of groundwater and surface water rights in each county, data from 2000 (Sophocleous and Wilson 2000).

Kansas water management policy is guided by the Water Appropriate Act, which mandates that the state manage the system of water rights (Foley et al. 2014). A surface or ground water right allows for beneficial use of such resources. The date of the water right, not the type of use, determines the priority of the water right user (Peck 1994). The primary tool to plan for future water resource needs is the Kansas Water Plan, which is prepared every five years by the Kansas Water Office (KWO), in cooperation with many other local, state, and federal partners. More specifically, the KWO works in cooperation with the Kansas Water Authority (KWA), which provides policy guidance to the Governor and Legislature of the State of Kansas (Kansas Water Office 2013). Other state agencies are also critical for water management efforts in the state. The Kansas Department of Agriculture – Division of Water Resources (KDA-DWR) administers statutes related to dam and levee construction, the state's four interstate river compacts, the Kansas Water Appropriate Act, and the national flood insurance program in Kansas.

The Kansas Department of Health and Environment (KDHE) Bureau of Water is the agency responsible for administrating programs related to public water supplies, wastewater treatment, sewage disposal, and nonpoint source pollution. KDHE also assures compliance with the Clean Water Act and Safe Water Drinking Act. Within the Bureau of Water, there are several sub-sections, such as Watershed Management; Watershed Planning, Monitoring and Assessment; and Public Water Supply. The Watershed Management section develops and reviews strategies and local environmental protection plans intended to control nonpoint source pollution. The Watershed Planning, Monitoring and Assessment Section monitors water quality conditions in streams and publicly owned lakes and wetlands; identifies and prioritizes impaired streams, lakes and wetlands; establishes Total Maximum Daily Loads (TMDLs) for these waterbodies; and develops statewide surface water quality standards. These sections have complementary roles in water quality assessment and improvement in the state of Kansas. The Public Water Supply Section implements plans that regulate public water supply systems within the state. There are many other state and federal agencies involved in Kansas water management; however, the KWO, KDA-DWR, and the KDHE are the three main agencies that will be examined in this study.

Table 5-1: A summary of the main water management and planning roles of the three agencies considered in this study.

Agency	Summary of main water management and planning roles
Kansas Water Office (KWO)	Prepares the Kansas Water Plan; works in cooperation with the Kansas Water Authority (KWA) to provide water policy guidance to the Governor and Legislature of the State of Kansas; oversees stakeholder basin groups throughout the state.
Kansas Department of Agriculture – Division of Water Resources (KDA-DWR)	Administers the state's four interstate river compacts, the Kansas Water Appropriate Act (i.e. water rights), statutes related to dam and levee construction, and the national flood insurance program in Kansas.
Kansas Department of Health and Environment (KDHE) Bureau of Water	Responsible for administrating programs related to public water supplies, wastewater treatment, sewage disposal, and nonpoint source pollution; works to assure compliance with the Clean Water Act and Safe Water Drinking Act.

5.4 Study Approach

5.4.1 Survey

In November of 2012, an electronic survey was sent to pre-selected state employees who have some involvement in water management and water resource planning within the state of Kansas within the KWO, the KDHE, and the KDA-DWR. At the KWO 14 respondents in either technical or managerial positions were selected. At the KDHE 21 state employees were identified in three sub-sections of the Bureau of Water: Watershed Management; Watershed Planning, Monitoring and Assessment; and the Public Water Supply sections. At the KDA-DWR a total of 29 employees working within the water management services, interstate water issues, and basin management teams were targeted as possible respondents.

In total, 64 respondents were targeted, and 37 responses were received, for a 58% response rate. Response varied by agency, as indicated in Table 5-2. Respondents were sent a link to an online survey facilitated by SurveyMonkey (https://www.surveymonkey.com/). Employees at each separate agency were provided with unique links that could identify the agency, but not the individual respondent. Respondents were assured anonymity, and this was provided through SurveyMonkey's secure online system.

Table 5-2: Survey response rate by agency; includes the total number of possible respondents who received the survey, the number of survey responses received, and the calculated response rate

Agency	Number of possible respondents	Number of responses received	Response rate
KWO	14	13	93%
KDHE	21	9	43%
KDA-DWR	29	15	52%
Total	64	37	58%

Respondents were asked questions related to four categories: 1) global climate change occurrence and impacts; 2) global climate change and resource management and planning; 3) global climate change information: data use and availability; and 4) global climate change vulnerability and adaptation. More specifically, these questions explored respondents' personal opinions about the occurrence of global climate change and associated impacts, views towards the importance of integrating climate change into agency water resource management and planning, the time scale corresponding to the majority of individuals' work within the agencies, and the availability and usability of global climate change information, including climate predictions, climate change vulnerability, and adaptation assessments. A complete list of survey questions are found in Table 5-3. Survey results were analyzed and plotted in Excel. Some responses were coded with numbers in order to calculate numerical averages for groups and statistically compare results. For example, question 2 was coded as following: “Not concerned” as 1, “Slightly concerned” as 2, “Concerned” as 3, and “Very concerned” as 4. Question 3 was coded: “Not at all important” as 1, “A little important” as 2, “Reasonably important” as 3, “Very important” as 4, and “Extremely important” as 5.

Table 5-3 Survey questions asked to Kansas water managers

	Question	**Possible Responses:**
	Global Climate Change Occurrence and Impacts	
1	Which of the following statements best reflects your personal opinion about climate change:	• Climate change is occurring; it is caused by emissions of greenhouse gases and other human-based causes. • Climate change is occurring; it is part of a natural cycle with no human influence. • I do not believe the climate is changing.
2	For each of the categories below, select the degree to which you are personally concerned about the impacts of global climate change: - Impact to my local community - Impact to state resources - Impact to the environment - Impact to global society	• Not concerned • Slightly concerned • Concerned • Very Concerned
	Global Climate Change and Resource Management and Planning	
3	How important is future climate change for management and planning efforts at your agency?	• Not at all important • A little important • Reasonably important • Very important • Extremely important
4	Do you think that your agency should consider future climate change in agency planning and management programs that your agency operates?	• Yes • No • Not sure
5	On average, what percentage of your work is focused on management or planning of water resources in the following time scales? Please ensure that the total adds up to 100%.	Respondents filled in percentages for the following 3 categories: • Short term (0-5 years in the future) • Medium term (5-15 years in the future) • Long term (15+ years in the future)

6	If you are engaged in long-term planning, at which time scale is climate information most relevant for your long-term planning efforts? Please rank time periods according to relevance.	Respondents chose from 1 (lease relevant) to 6 (most relevant), or “I do not engage in long-term planning” for the following time periods: • 2012-2020 • 2021-2040 • 2041-2060 • 2061-2080 • 2081-2100 • The entire period 2012-2100
	Global Climate Change Information – Data Use and Availability	
7	Which of the following climate information sources would you use if you were seeking climate data and projections? Please mark all that apply.	• Global reports, such as those from the Intergovernmental Panel on Climate Change (IPCC) • A national climate agency, e.g. the National Climatic Data Center • A regional climate data center, e.g. the High Plains Regional Climate Center • A state climate data center, e.g. K-State Research and Extension State Climatologist • Climate scientists at local or regional universities • I would not be interested in this information
8	How would you rate the availability of information on climate predictions and tools to integrate future climate scenarios into Kansas or regional studies?	• Widely available and easy to access • Widely available but not very easy to access • Fairly available • Not available at all • Not sure how to access • I am not interested in this information
	Global Climate Change Vulnerability and Adaptation	
9	*Climate vulnerability can be defined as exposure and sensitivity to adverse consequences that would be caused by changes in climate, as well as natural hazards due to extreme climate conditions. Vulnerability assessments provide background for determining populations and resources that may be vulnerable to changes in climate.* How would you rate the availability of climate vulnerability assessments for Kansas and/or the region?	• Widely available and easy to access • Widely available but not very easy to access • Fairly available • Not available at all • Not sure how to access • I am not interested in this information

10	*Climate adaptation includes the management of risks due to climate change, including variability. Adaptation may include actions, programs, or policies that mitigate risks that could be caused by extreme climate events and long-term climate change.* How would you rate the availability of climate adaptation strategies and practices tailored for Kansas and/or the region?	• Widely available and easy to access • Widely available but not very easy to access • Fairly available • Not available at all • Not sure how to access • I am not interested in this information
11	What do you see as the roadblocks to integrate predictive climate science with Kansas water resource planning and management? Please choose all that apply.	• Insufficient data • Insufficient funding • Insufficient staff resources • Lack of agency or staff interest in climate change • Technical complexity • None of the above • Other (please specify)
12	Do you have any suggestions to improve the integration of climate science with Kansas water resource planning and management?	Open Ended

5.4.2 Kansas Water Management Documents

Beyond the employee perspective, an independent review of key documents was conducted for evidence that climate change is considered in Kansas water management. Several documents were examined, such as the Kansas Water Vision, a 50-year plan for water management in Kansas, as well as the KWA Report to the Governor from 2014, and the 2014 Kansas Integrated Water Quality Assessment by the KDHE (Foley et al. 2014; Kansas Department of Health and Environment 2014; Kansas Water Authority 2014). While there are many other reports and documents available, these documents represent the most recent and synthesized reports and plans for water resource management in the state. These documents were analyzed for any mention of climate variability, climate change, extreme weather conditions (drought or flooding), future conditions, hydrologic modeling, and the general planning approach. The Kansas Water Vision is a recent initiative heavily supported by the governor of Kansas and spearheaded by a team comprised of employees of the KWO, the KDA, and the KWA (Foley et al. 2014).

5.5 Results and Discussion

5.5.1 Survey Results

Climate Change Occurrence and Impacts

When asked their opinions about climate change, 65% of respondents reported that they believe climate change is occurring, and it is caused by emissions of greenhouse gases and other human-based causes. The remaining 35% reported that they believe climate change is occurring, and it is part of a natural cycle with no human influence. No respondents reported that they did not believe the climate is changing. These responses demonstrate that the dominant cause of climate change is still a contentious issue among natural resource planners in Kansas, which reflects a larger pattern that is seen in climate perceptions in the state. For example, a few studies have indicated that Kansans have little concern for climate change and/or do not believe that climate change could be caused by human activities (Hamilton and Keim 2009; Harrington 2010). Many of the environmental scientists in the state believe that climate change has no anthropogenic influence, and this could influence their willingness to use climate forecasts and model output that utilize anthropogenic forcing.

Most respondents indicated that they had some level of concern for the impacts of global climate change on several scales: the local community, state resources, the environment, and global society. The highest levels of concern were indicated for state resources and the environment. The majority of respondents were either "concerned" or "very concerned" about climate change impacts to state resources (75%) or to the environment (73%). There was also a majority of respondents who indicated that they were "concerned" or "very concerned" about climate change impacts to their local community (59%) and to global society (57%). Very few respondents had no concern for climate change impacts – only four indicated that they had "no concern" for impacts to their local community, and two had "no concern" for impacts to global society.

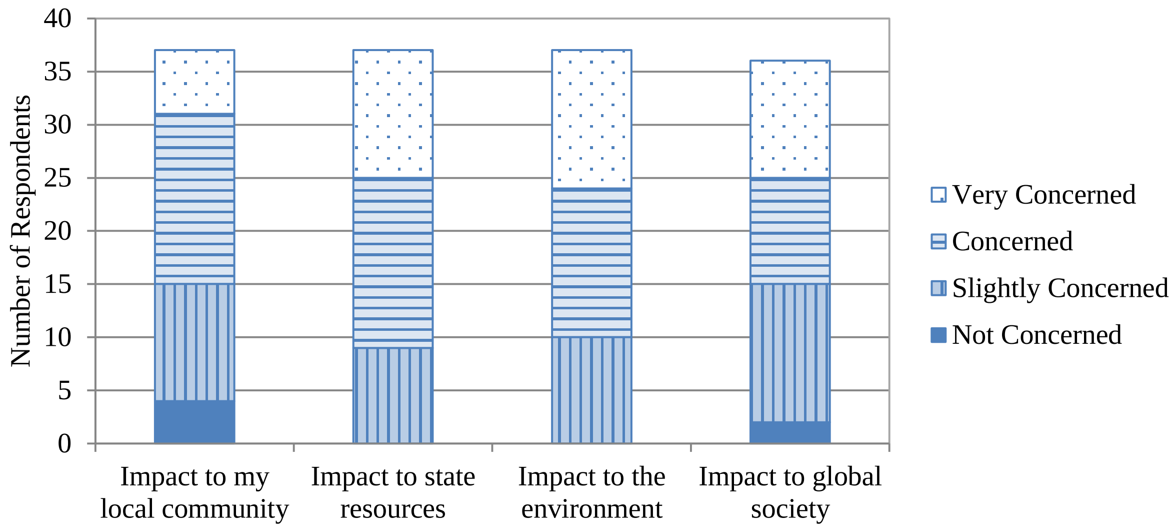

Figure 5-2: Responses to survey question 2: respondents' concern about the impacts of climate change to the local community, state resources, the environment, and to global society

There was a significant difference in the rating of concern for climate change impacts to local community, the environment, and global society between groups that either believed climate change was an entirely natural phenomenon with no human influence, and those that believed climate change is caused by greenhouse gas emissions and other human influences. The group that believed climate change has an anthropogenic component rated their concern for climate change impacts to the local community, the environment, and global society higher than those of the group that believe climate change is an entirely natural phenomenon with no human influence. However, there was no significant difference between the concerns for climate change impact to state resources. These results suggest that managers who believe that climate change is anthropogenic have greater concern for the impacts. Yet, it is not possible to disentangle if those that believe climate change is anthropogenic have greater concern overall for the environment or local and global societies.

Table 5-4: Average scores for survey question 2 broken up into two groups: those that believe climate change is an entirely natural phenomenon and those that believe climate change has an anthropogenic influence. The average scores of each group were compared by a two-way t-test with unequal variance and the corresponding p-value is reported.

	Average score for respondents who believe climate change is an entirely natural phenomenon with no human influence	Average score for respondents who believe climate change is caused by greenhouse gas emissions and other human influences	p-value
Impact to my local community	2.18	2.88	0.04
Impact to state resources	2.64	3.25	0.07
Impact to the environment	2.45	3.42	0.00
Impact to global society	2.09	3.17	0.00

Global Climate Change and Resource Management and Planning

Almost all respondents gave some degree of importance to climate change for management and planning efforts at their agencies. The highest percentage of respondents (38%) indicated that it is “very important” to incorporate future climate change for management and planning efforts at their agency, with the next highest response (27%) indicating that is was “reasonably important”. There was no statistical difference (p=0.89 according to two-tailed Student’s T-Test with unequal variance) in the average rating of importance between those who believe climate change is an entirely natural phenomenon and those that believe climate change is primarily human-caused. Therefore, regardless of what the respondents believe is causing climate change, they both view it as equally important to consider in planning efforts. Consequently, the responses from survey question 4 indicate that the majority of respondents (85%) believe that Kansas agencies should consider future climate change in planning and management programs that their agencies offer, with only 2.9% indicating that climate change should not be considered, and 12% unsure.

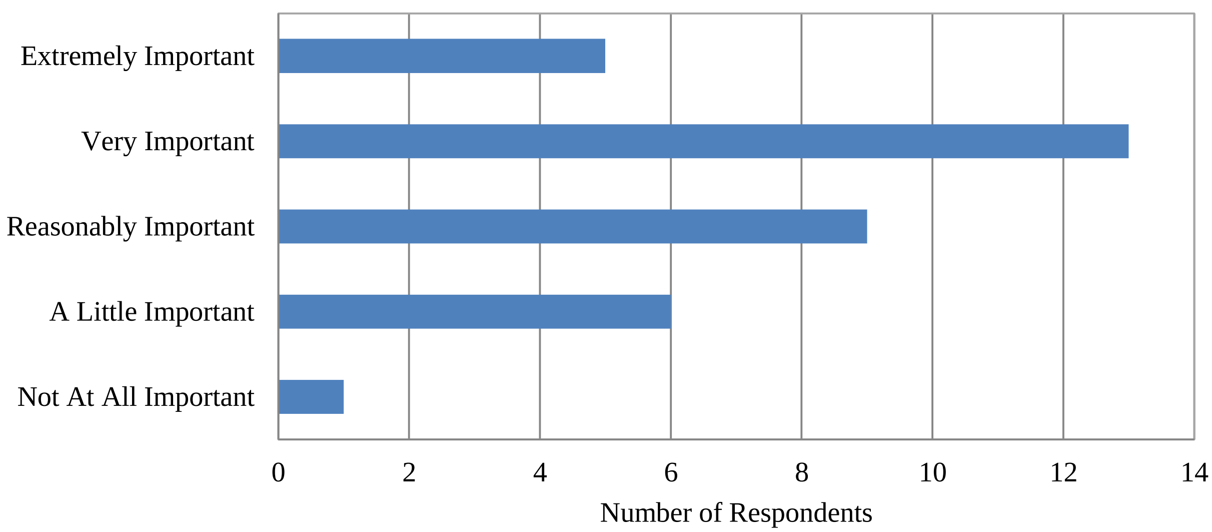

Figure 5-3: Respondents’ response to survey question 3: “How important is future climate change for management and planning efforts at your agency?”

To gain a perspective on the usefulness of climate model predictions for statewide water management efforts, respondents were asked what percentage of their work focused on management or planning of water resources in the short, medium, and long term. While there was some variation between agencies, 47% was the average percentage of work time spent on short term planning (0-5 years), 31% for medium term planning (5-15 years), and 25% for long term planning long term planning (+15 years) (results do not add to 100% as these are averages of indicated percentages from all individuals). It is important to consider how much time is spent on long term planning, as this is a helpful indicator to the degree of usefulness of long-range climate predictions in day-to-day water management activities. However, it is also clear that water managers spend almost half of their time on short-term issues, and therefore the timeliness of information is important (i.e. information provided as quickly as possible to be used in decision making).

In the same vein, respondents were asked to indicate which time scale climate information would be most useful for planning efforts, given 20 year periods starting from the present to 2020, and then continuing up to 2100. By understanding the time period that is perceived as most useful for planning efforts, climate scientists can focus on generating results within this time frame and provide more usable science for planning. For this question agency-specific responses diverged, with the KWO indicated that the 2041-2060 time period would be the most useful for planning, while KDHE and DWR both indicated that 2012-2020 would be the most useful time period. As the most useful time period varies between agencies, this would most likely need to be modified for each project in order to serve the needs of the particular agency or management concern. Current climate modeling efforts in Kansas have analyzed climate changes through 2100 using a composite of 21 global climate models (Brunsell et al. 2010). Most IPCC climate projections also extend to 2100 (Pachauri 2007). Published hydrologic modeling efforts of future climate in Kansas show snapshots at 2050 and 2100 (Sheshukov et al. 2011).

Global Climate Change Information – Data Use and Availability

Most respondents (28 of 37) reported that they use a state climate data center, such as the K-State Research and Extension State Climatologist (Accessible at: http://www.ksre.ksu.edu/wdl/), for climate data and projections. Regional climate data centers (e.g. the High Plains Regional Climate Center) and national climate data centers (e.g. the National Climatic Data Center) were also popular sources of climate information. Climate scientists at local or regional universities were reported to be a source of climate data for 18 respondents, and only 11 indicated that they use global reports such as those from the Intergovernmental Panel on Climate Change (IPCC). Based on these results, it seems that the best place to provide climate data and tools would be through a state or regionally-based center.

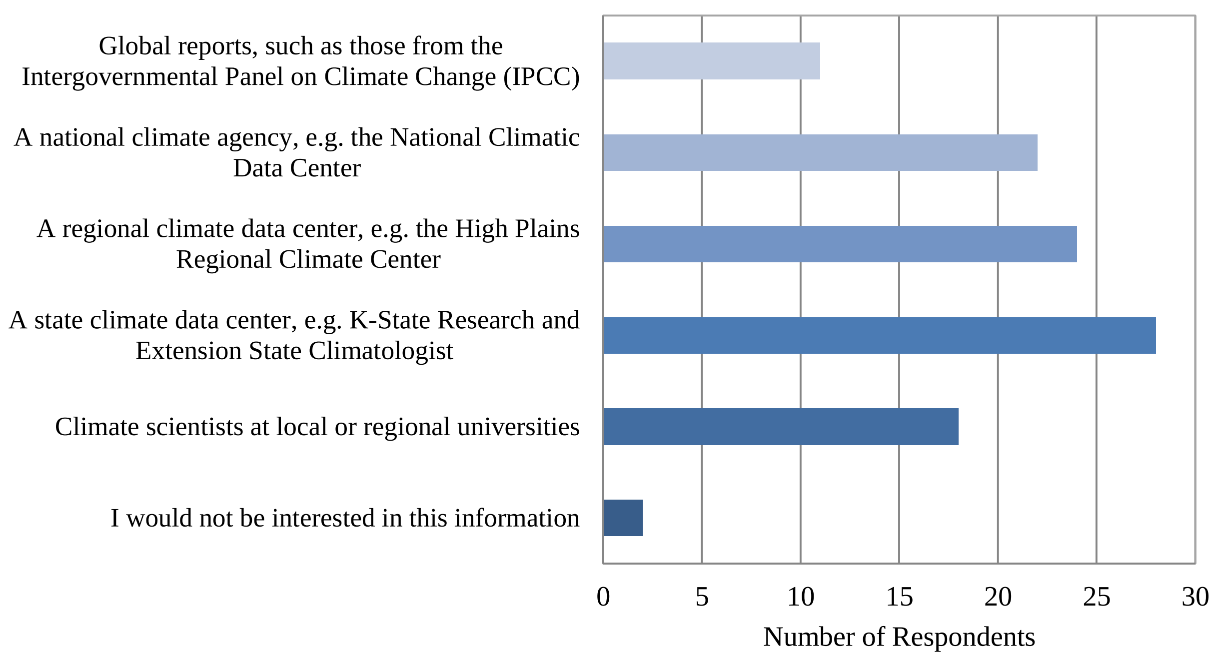

Figure 5-4: Responses to survey question 7: "Which of the following climate information sources would you use if you were seeking climate data and projections? Please mark all that apply."

Global Climate Change Vulnerability and Adaptation

Finally, respondents were asked to rate the availability of information on climate predictions and tools to integrate future climate scenarios into Kansas studies, climate vulnerability assessments for Kansas and/or the region, and climate adaptation strategies tailored for Kansas and the region. Climate predictions and tools were deemed the most available with higher ratings of "widely available" and "fairly available", yet there were also 10 respondents that indicated they were "not sure how to access the information", and three who believed this information was "not available at all". As there is already a focused source for local climate data through the K-State Research and Extension State Climatologist, and most respondents indicated that they would seek information there, perhaps climate predictions and tools could be made available through this data venue to increase access to natural resource managers and other interested parties in the state.

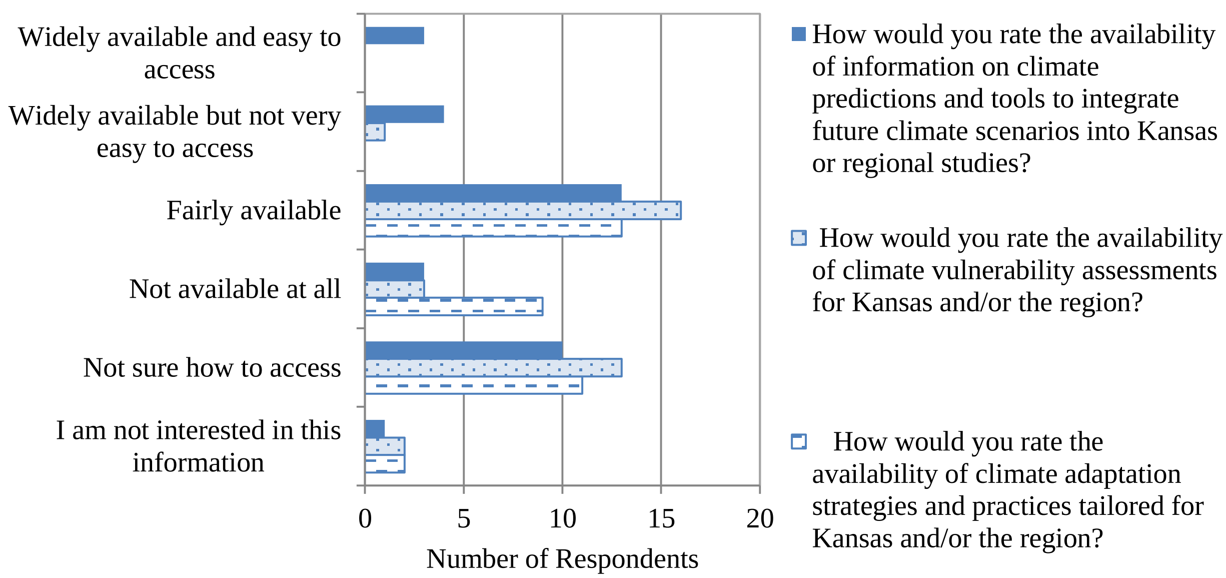

Figure 5-5: Responses to survey questions 8-10: views on the availability of climate predictions and tools (blue), Kansas or regional climate vulnerability assessments (light blue spotted), and

climate adaptation strategies/practices (white with dashes). Survey questions are indicated in the legend. Definitions for vulnerability and adaptation were provided in the survey and can be found in Table 5-3.

Climate vulnerability assessments and adaptation strategies were similarly rated, with approximately one third of the respondents rating the information as “fairly available” and the remaining two thirds either believed the information was “not available at all”, were “not sure how to access” the information, or were not interested. With about one third of respondents unsure how to access information, there is a demonstrated need for more outreach between researchers and managers. At minimum, as such information becomes available it should be provided electronically on a continually updated website, which is connected to other state-based climate and natural resource data. Moreover, tabletop exercises and two-way discussions on how to utilize information for state-based management would be useful for translating knowledge to action and sparking new ideas for future research.

Roadblocks to Climate Integration

The majority of respondents see “insufficient staff resources” and “insufficient funding” as the major barriers to integrating climate science into Kansas water resource planning and management. “Insufficient data” and “technical complexity” were also highly rated as impediments to working with climate change science. Approximately one third of respondents indicated that “lack of agency or staff interest in climate change” was also a roadblock. Interestingly, 12 of the 17 respondents that indicated that “insufficient data” is a roadblock, rated the availability of information on climate predictions and tools to integrate future climate scenarios into Kansas studies as “fairly available” (7/12) to “widely available” (5/12). The inconsistent response towards the perception of climate availability could be an indication that the data currently available is not compatible with agency planning tools. In a follow up discussion with the Kansas Water Office, agency staff mentioned that the available climate model results are spatially not applicable and that a finer scale is necessary to make work applicable to state-based water management.

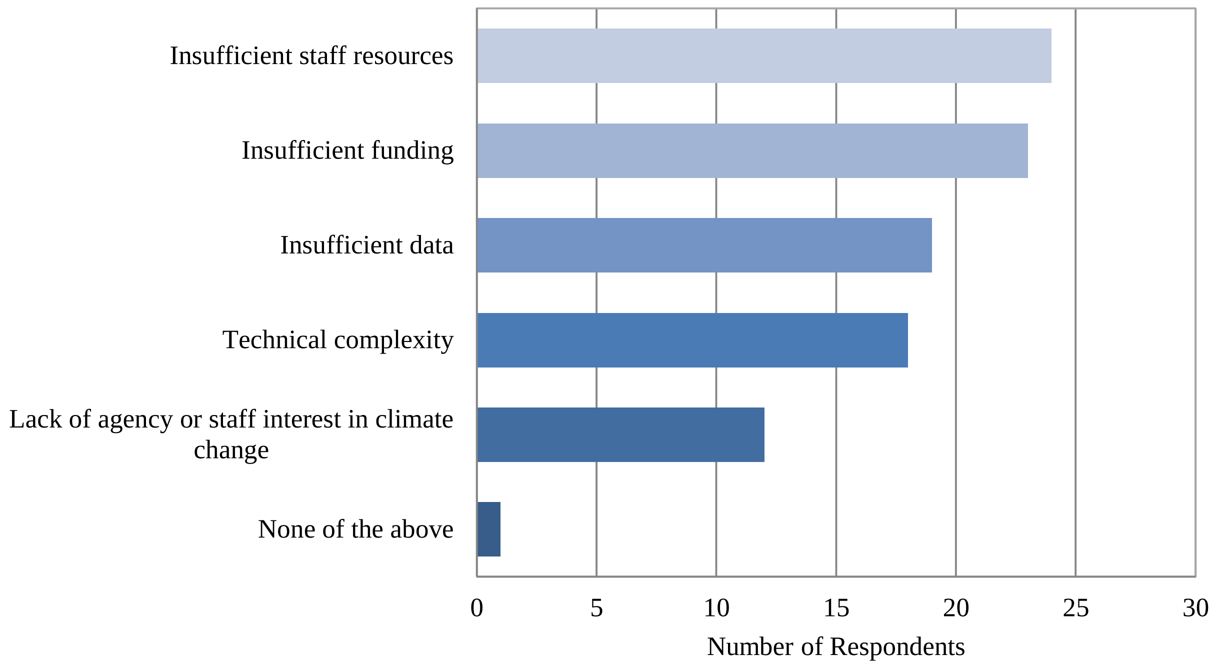

Figure 5-6: Responses to survey question 11: “What do you see as the roadblocks to integrate predictive climate science with Kansas water resource planning and management? Please choose all that apply.”

Survey participants indicated that the major roadblocks to integrating climate science into water planning are funding and staff resources. The issues of insufficient data and technical complexity will not be resolved without additional support for research and training, which requires more staff and financial resources. Therefore, it is challenging for resource managers to take up the issue of climate change within their organizations without some type of external incentive or support.

At the same time, university researchers are faced with the challenge to make their research meaningful and relevant to society. University researchers could play an invaluable role as collaborators and provide meaningful research for government agencies. While it is already common for agencies to fund contracts for specific research work, such as watershed modeling and field studies, university researchers should approach government agencies with ideas for projects, or calls for funding that may generate ideas for proposals. While some researchers may already be collaborating actively with state agencies, it was indicated by some staff at the KWO that further engagement is desired during brainstorming and proposal development. Dialogue between researchers and agencies early in project formation is critical for both tailoring the research to the agency needs, and providing a solid platform to develop the research proposal. This collaborative approach is called co-production of knowledge, and has been indicated as a means to increase the usability of climate data, but has been indicated by some to be an unsustainable method of engagement due to the high level of human, financial, and technical capital needed (Lemos, Kirchhoff, and Ramprasad 2012).

5.5.2 Is Climate Change Considered in Kansas Water Management?

Climate change is not specifically mentioned in the three analyzed planning documents (the Kansas Water Vision, the KWA Report to the Governor from 2014, and the 2014 Kansas Integrated Water Quality Assessment by the KDHE), but climate-related issues are drivers for many of the proposed plans in the Kansas Water Vision (abbreviated as the Vision). In fact, the creation of the Vision seems to have been driven by the ongoing drought that occurred between 2010-2013, as mentioned in the opening statement of the plan: “the multi-year drought has brought water issues to the forefront; we must plan for the future now (p.6) (Foley et al. 2014).” Drought forced water issues to become a priority for the public and the state government. The importance of weather and climate variability are also emphasized in the opening statement: “Due diligence in protecting water resources and adapting to future climate variability will be important to maintaining and improving quality of life and the state's economy (p.6) (Foley et al. 2014).”

The Vision proposes to conduct drought simulation exercises, to promote regional drought and water conservation planning, and to develop regional plans that acknowledge the significance of planning for state resiliency to the impacts of climate variability. Reservoir operations will be analyzed to protect systems from sediment influx from high-flow events, reservoir drought risk will be evaluated, and information will be developed to assess future reservoir operations and management changes. Model development is highlighted in several instances as a path to testing future scenarios and moving towards adaptive management. In addition, the document highlights the importance of annual interaction with university researchers regarding collaborative research that supports implementation of the Vision. Multi-disciplinary approaches are encouraged, and the plan hopes to develop research proposals that would be ready for incoming funding, as it becomes available (Foley et al. 2014).

The KDHE non-point source report also highlights impacts of extreme climate on Kansas water:

> "Kansas experienced major statewide droughts in 2001-2006 and again in 2011. In 2007, major floods in southeastern Kansas scoured many rivers and creeks and produced sustained high stream flows for much of the summer. The combined effects of these dramatic weather-related events exacerbated many of the water quality impairments documented in the past decade (p.8) (Kansas Department of Health and Environment 2014)."

The report highlights that the major causes of water quality impairment are municipal point sources and agriculture, yet there is some reflection on the role of climate in magnifying water quality impairments, especially in streams. It was also mentioned that temperature increases in water bodies have a role in blue green algae blooms, and in one case, temperature was the cause of a reported fish kill. However, in the KDHE report a very small portion of impaired lake acreage is primarily attributed to natural sources of impairment, including climate and weather-driven impacts (Kansas Department of Health and Environment 2014).

From a review of these key documents, it can be determined that climate variability and extremes both directly and indirectly influence Kansas water resource management and are recognized by state decision makers. While Kansas water managers do not use the phrase "climate change", this may be a reflection of the political climate of the state, which has not fully embraced the idea of anthropogenic climate change, rather than the views of the managers (House Bill no 2306; Associated Press 2014). From the survey results it is apparent that most water resource managers recognize that climate change is occurring and believe it should be used in statewide management efforts. The Vision makes clear that future planning will involve hydrologic modeling efforts, but it is not mentioned if these models will be used to examine future climate scenarios or forecasts.

5.5.3 A Way Forward: Elements Necessary for Successful Integration of Climate Science into Water Resource Management

Successful integration of climate science into management will require the development of usable science products or tools explicitly for decision makers. Usable science is a term used to describe the usefulness of science in the context of decision-making, providing a solution for a problem, or contributing to the design of policy (Dilling and Lemos 2011). There are several factors that have been identified as critical elements for developing usable science in the climate and water resource sectors, which can be divided into three types: product, process, and context (Kirchhoff 2010).

Product: Many basic factors relate to the scientific product provided; these include obvious, yet incredibly essential, elements such as accuracy, reliability, credibility, salience, and timeliness (Kirchhoff 2010). Salient information is responsive to regulatory and legal constraints, as well as ecological, spatial, temporal, and administrative scales (McNie 2007). Assessments should be tailored to the producer and the user of the information, while considering aspects such as availability of resources, flexibility, and the knowledge base of participants (Brenner 2011). It may go without saying, but information also needs to be accessible. Accessibility includes the ability to obtain forecasts or climate data, the ease of access of data formatting and representation, as well as the ability for users to comprehend and implement information (Dilling and Lemos 2011). With respect to Kansas, many specific scientific needs were explicitly mentioned in the Vision, along with expected timelines, which should provide a starting point for scientists wishing to contribute to statewide water resource management.

Process: Through evaluation of case studies, Dilling and Lemos found that usability increased when information producers were knowledgeable about the specific decision contexts for the science they were providing. In addition, it was important for the information users to value and understand the usability of the science for their own decision making (Dilling and Lemos 2011). Such mutual understanding is developed by collaboration and repeated interaction between information producers and users. Trust, bilateral communication, and co-production of knowledge are essential components of developing a successful collaboration and long-term relationship. Information can be more easily shared between organizations and institutions when time has been invested to develop a collective meaning and identity (i.e. co-production), and ensuring that information maintains the goals of the organization (Rayner et al. 2005). There is no shortcut to successful collaboration. Time must be invested, and a long-term perspective is necessary. This sentiment is also echoed in the Kansas Vision, which discusses the need for ongoing and continual interaction with Kansas scientists on issues of future water management. This provides an opportunity for scientists to be involved in co-production of usable science for local and regional water resource planning. In addition, collaboration provides the opportunity to emphasize issues such as future climate change and to develop locally-relevant adaptation and mitigation strategies.

Iterative approaches have also been shown to have success. Various iterative assessments can focus on a specific outcome, instead of attempting to cater to multiple users in one comprehensive report or analysis (Brenner 2011). Iterations can also leave room for flexible approaches and adaptation to successes and failures of past approaches. The Kansas Vision itself is being developed with an iterative approach; each phase includes a period of stakeholder outreach, followed by meetings and discussion drafts, then the plan is released for additional feedback. There is also a 5-year review process in place to continue to update the Vision and amend actions and goals, as necessary.

Context: Developing an accurate and timely product and investing in a trustworthy, long-term collaboration are the key components necessary for success. Yet, sometimes political, economic and social contexts can be the ultimate driving factors determining the success or failure of science-policy integration. Scientists need to consider how their work may be perceived from these various lenses, and also consider the local political and economic realities, especially when working in the climate change arena (Brenner 2011). The contentious nature of climate change in the state of Kanas sets the context for integrating climate change into water resource management. However, it seems that one can be successful at sidestepping this issue by focusing on the issue of climate variability and highlighting the damaging effects of past extreme events. In this way, climate change can be considered in water resource planning without explicitly mentioning it.

Key stakeholders and boundary organizations can play a central role in navigating complex political or social contexts and creating common ground for collaborating parties to work from. Stakeholders are the critical ingredient to effectively integrate information into local action. Stakeholders have access to, and experience with, practical, local knowledge to evaluate adaptation and mitigation efforts that consider climate change (Dessai and Hulme 2004). Therefore, integration of stakeholders early in climate vulnerability assessment is critical to creating usable science (Dilling and Lemos 2011). Stakeholders and scientists/researchers should work together to set the research agenda, which will continually be shaped by the “science push” or the pursuit of knowledge, as well as the “demand pull” or the search for a solution to a pressing problem (Dilling and Lemos 2011). Stakeholders were heavily involved in the development of the Kansas Vision. While, again, the plan is not focused on climate change adaptation, there are many planned actions and studies within the plan that will make water supplies more resilient to climate change. Stakeholder involvement was critical for ensuring that all issues were considered and that the planned efforts align with the most pressing needs.

Boundary organizations do not always refer to formal organizations, but can also indicate a one-time forum, or other arena fostering interdisciplinary cooperation. In general, boundary organizations connect and integrate professionals from various backgrounds and organizations, aid in communication and translation of information, and mediate between producers of information and the users (Feldman and Ingram 2009). Boundary organizations can fill the knowledge and cultural gaps between collaborating organizations by serving several key roles and duties. For example, boundary organizations connect information needs with sources, while also facilitating integration and communication of available knowledge. These organizations are also critical for facilitating dialogue between various parties and ensuring equitable partnerships. Moreover, due to their unique position as an integrator, boundary organizations participate in synthesizing and creating new knowledge through their interactions with various parties (Buizer et al. 2010). There is a great need for such organizations at the interface of climate change and water management, especially in Kansas. Water issues have been given greater attention through recurring events such as the Governor's Conference on the Future of Water in Kansas and through venues supported by a new group, KU Water Research, at the University of Kansas (http://water.ku.edu/). Perhaps in time the participants in these events will form the connections necessary to create a more permanent climate-water boundary organization in Kansas. Other universities have had success developing regional climate-water collaborations through boundary organizations and their experience may be helpful for considering the next step in Kansas.

Examples of Successful Climate-Water Collaborations

In 1995 the Climate Impacts Group (CIG) at the University of Washington was formed. Over the following years a successful collaboration was formed between water managers and researchers in CIG. CIG conducted interviews to determine how agencies might use climate information and then used this knowledge to direct the type of information developed, and to develop a plan for outreach and information dissemination. The CIG group determined that agencies did not have the technical and financial resources to develop their own hydrologic scenarios for climate change planning, but they needed more focused descriptions of potential impacts and climate information that could be easily integrated into their current operating models (Snover et al. 2003). The CIG group developed hydrologic scenarios using the "delta" method, which adjusts historical regional climate in each month by projected changes in monthly mean precipitation and temperature. This simple method allowed CIG to quickly integrate climate change into the agencies' planning framework and begin assessing climate vulnerability. In addition, the group has developed long-range streamflow forecasts for water management and methods for integrating climate information into water resource planning, operation, and management (Kirchhoff 2010).

Another example of a successful collaboration is the Climate Assessment for the Southwest (CLIMAS) group, which began in 1998. CLIMAS is located at the University of Arizona in Tucson and works to bring together natural and social scientists studying climate processes and impacts with decision makers and resource managers (Kirchhoff 2010). The CLIMAS group has developed seasonal forecasts for urban water managers and has helped to analyze the sensitivity of urban systems to drought. CLIMAS built long-term, interactive relationships with water managers, and this encouraged trust and the development of useable data, which ultimately determine the successful use of CLIMAS data in water resource planning and decision making (Kirchhoff et al. 2013). CIG and CLIMAS are both considered boundary organizations, yet they have a strong foothold in universities. The research produced by these groups is directed towards regional use in natural resource management, and there is direct feedback from information users, which helps to increase its usability.

5.6 Conclusions

Anthropogenic climate change is still a contentious issue within the state of Kansas. While Kansas water managers do not have a direct plan for climate change adaptation, there is interest from state water planners to include climate change in their future efforts. Through analysis of recent Kansas water planning documents, management and planning approaches call for a more thorough examination of impacts of climate extremes to Kansas water resources. The Kansas Water Vision has put forth an agenda for water resource management for the next several decades. This Vision will require institutional collaboration and research. Currently, the state government is spearheading this effort; however, a state-based boundary organization may be helpful for integrating the vision with the efforts of various Kansas water agencies and university researchers. Indeed, the Vision provides researchers with opportunities for future collaboration on projects of scientific interest and critical importance to the state of Kansas. Working together, researchers and state planners can produce knowledge that is credible, timely, salient, and can be directly applied to water management issues in the state.

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Chapter 6 – Emission characteristics of CO_2 and CH_4 in the Pengxi River during an annual cycle of storage operations of the Three Gorges Reservoir, China

Abstract

The emission of greenhouse gases (GHG) from freshwater reservoirs has received a great deal of attention in recent years. Features such as reservoir age, geographical distribution, and submerged soil type have been determined to have a great impact on reservoir GHG emissions; however, the effect of artificial water storage management has been largely overlooked. A field study was conducted from June 2010 to May 2011, an annual cycle of reservoir storage operations, to evaluate potential ecological processes and environmental conditions that regulate carbon dioxide (CO_2) and methane (CH_4) fluxes in the Pengxi River backwater area, a typical tributary of the Three Gorges Reservoir, China. Both CO_2 and CH_4 fluxes were influenced by water level and exhibited distinct patterns that correspond to the reservoir operation cycle. Over 90% of CO_2 efflux occurred during the high water period, whereas the 58% of CH_4 efflux occurred during the low water period. Our results suggest that reservoir operations altered the hydraulic retention time, which along with water temperature, controlled the synthesis and decomposition of carbon in the backwater system. In particular, CO_2 fluxes were highly influenced by algal growth, which at times caused an influx of CO_2 into the surface water. The overall CO_2 fluxes from the PBA were relatively higher than that of temperate reservoirs, and similar to subtropical and tropical reservoirs. However, the CH_4 fluxes were closer to the median values for temperate reservoirs globally.

6.1 Introduction

The majority of the world's lakes, rivers, reservoirs, estuaries and other inland waters are supersaturated with CO_2 and CH_4 (Cole et al. 1994; Cole et al. 2007). St. Louis et al. estimated that reservoir CO_2 fluxes are equivalent to 4% of total global anthropogenic emissions of CO_2, but that reservoir CH_4 fluxes are equal to approximately 20% of global anthropogenic CH_4 emissions (Louis et al. 2000). Although biogeochemical processes leading to greenhouse gas (GHG) production and emission in reservoirs are well identified and similar to those occurring in natural lakes (Goldenfum and Association 2010), in the past decades, various research has been carried out to test the hypothesis that damming rivers has a positive effect on GHG emissions in watersheds (Fearnside 2014, 1997, 1995; dos Santos et al. 2006; Ramos et al. 2009; Rosa et al. 2006; Rosa and Schaeffer 1995). It was widely accepted that factors affecting the carbon budget in reservoir systems include: the input of allochthonous organic carbon, the amount and type of organic carbon deposits in the flooded land, the reservoir age, and meteorological background (Abril et al. 2005; Barros et al. 2011a; Delmas et al. 2001; Hertwich 2013). Reservoir age, reservoir depth, and regional climate might be the key factors regulating gross emissions of GHG in reservoirs (Barros et al. 2011a; Hertwich 2013). However, current knowledge is not enough to quantify the net GHG effects of reservoir creation and impoundment (Teodoru, et al. 2011; Teodoru et al. 2012).

The Three Gorges Reservoir (TGR) is currently China's largest reservoir with a full capacity of 39.3km^3. Recent years witnessed a growing concern related to GHG emissions from the TGR. However, recent research suggests that gross GHG emissions were not as high as estimated by Chen et al. (2009) (Yang, et al. 2013a; Yang et al. 2013b; Zhao, et al. 2013; Chen et al. 2011a). The TGR is operated for various functions, such as flood control, navigation, and hydropower generation. Water level in the TGR decreases to 145m in May before the flood season, creating about 22 km^3 capacity for the incoming floods. At the end of October water level in the TGR increases to 175m for hydropower generation during the winter drought season. Previous studies indicated that reservoir operations created distinctive seasonal habitats that potentially regulate the carbon budget in the TGR (Li et al. 2014). Also, as a river-valley dammed reservoir, the running nature creates a longitudinal hydrodynamic gradient that potentially regulates carbon transport and storage (Straškraba et al. 1993; Thornton et al. 1990). Although previous studies reported the spatial patterns of surface GHG emissions along the mainstream Yangtze River of the TGR (Yang et al. 2013a; Yang et al. 2013b), the impact of reservoir operations on the surface GHG emissions in tributaries of the Yangtze were not well documented. In addition, there is need for accurate estimation of gross air-surface GHG emissions in this river-reservoir hybrid system. This requires the use of geospatial methods utilizing monthly data from limited sampling sites.

A one-year field survey was carried out in the Pengxi River, a typical tributary of Yangtze in the TGR, from June 2010 to May 2011. Monthly air-water CO_2 and CH_4 emissions along the river-reservoir longitudinal gradient were measured in an annual reservoir operational cycle. Environmental parameters such as temperature, dissolved oxygen (DO), carbon, nitrogen, and phosphorus concentrations in the water column, and hydrological characteristics were analyzed. This study examines the relationships between environmental parameters and monthly GHG fluxes to determine if reservoir operations regulated surface GHG emissions. In addition, gross CO_2 and CH_4 emissions are estimated in the backwater area of the Pengxi River using ArcGIS and a geospatial estimation approach.

6.2 Material and methods

6.2.1 Study sites

The Pengxi River is the largest tributary of the TGR and located at the mid-reach of the reservoir region, about 250 km upstream from the Three Gorges Dam (Figure 6-1). The Pengxi River watershed area is 5173 km^2, ranging from N31°00′ E107°56′ to N31°42′ E108°54′. Annual rainfall in the watershed is 1100-1500mm, and the annual discharge of the river is $118 m^3 \cdot s^{-1}$.

After impoundment of the TGR to the water level of 145 m, the Pengxi River forms a backwater area of approximately 60 km from the town of Yunyang, located at the confluence of the Yangtze River, to the upstream town of Yanglu. The backwater area of the Pengxi River from Yunyang to Yanglu has a water surface area of 31.5 km^2. However, during high water operation (up to 175m), the terminal backwater region extends to the upstream region of Kaixian, with a water surface area and length about 79.2 km^2 and 80km long, respectively. During the discharge period, the backwater area ends between the towns of Yanglu and Kaixian. During the operating cycle of 145m – 175m, the Pengxi Backwater Area can be divided into two distinctive parts: the fluctuating backwater area (FBA) and the perennial backwater area (PBA). Yanglu is located approximately at the boundary between the FBA and the PBA of the Pengxi River. The aquatic ecosystem in the FBA is riverine when the TGR decreases its water level during the flood season. In winter, during the high water level period, the FBA is re-inundated and becomes part of the reservoir. The PBA is part of the TGR regardless of water level fluctuations.

There were seven sampling spots used in this study along the 80 km backwater area; the locations of these spots are subsequently listed from upstream to downstream. (1) Wenquan (WQ, N 31°20′1.3″E 108°30′48.8″) is an upstream river location. WQ controls 24% of the watershed area in the Pengxi River Watershed. It is a river background sampling site, representing the carbon input from upstream into the reservoir. (2) Kaixian (KX, N 31°11′12.9″E 108°26′34.2″) and (3) Baijiaxi (BJX, N31°6′48.5″E108°32′56.5″) are in the FBA in the Pengxi River backwater area. (4)Yanglu (YL, N31°5′7.7″E108°33′47.6″), as indicated above, is the geophysical boundary between river reaches and the PBA area during the summer low water level period, but in the winter high water level period YL is part of the PBA. (5) Gaoyang (GY, N31°5′48.2″E108°40′20.1″), (6) Huangshi (HS, N31°00′29.4″E108°42′39.5″), and (7) Shuangjiang (SJ, N30°56′51.1″E108°41′37.5″) are the 3 sampling sites in the PBA, the permanently flooded region.

Table 6-1: Sampling locations and depths

		Water depth(m) (max, min)	Samping depth (m)		
			Temperature, pH, DO, pCO_2, Chl-a	Dissolved nutrients	Gas flux
Background	WQ	(1.5, 0.5)	0.5	0.5	Air-water interface
FBA	KX	(10.0, 1.0)	0.5	0.5	
	BJX	(10.0, 3.0)	0.5	0.5	
	YL	(20.0, 3.0)	0.5	0.5	
PBA	GY	(30.0, 10.0)	0.5, 1, 2, 3, 5, 8, 10	0.5	
	HS	(50.0, 20.0)	0.5, 1, 2, 3, 5, 8, 10	0.5	
	SJ	(60.0, 30.0)	0.5, 1, 2, 3, 5, 8, 10, 12, 15 ,20, 25, 30	0.5	

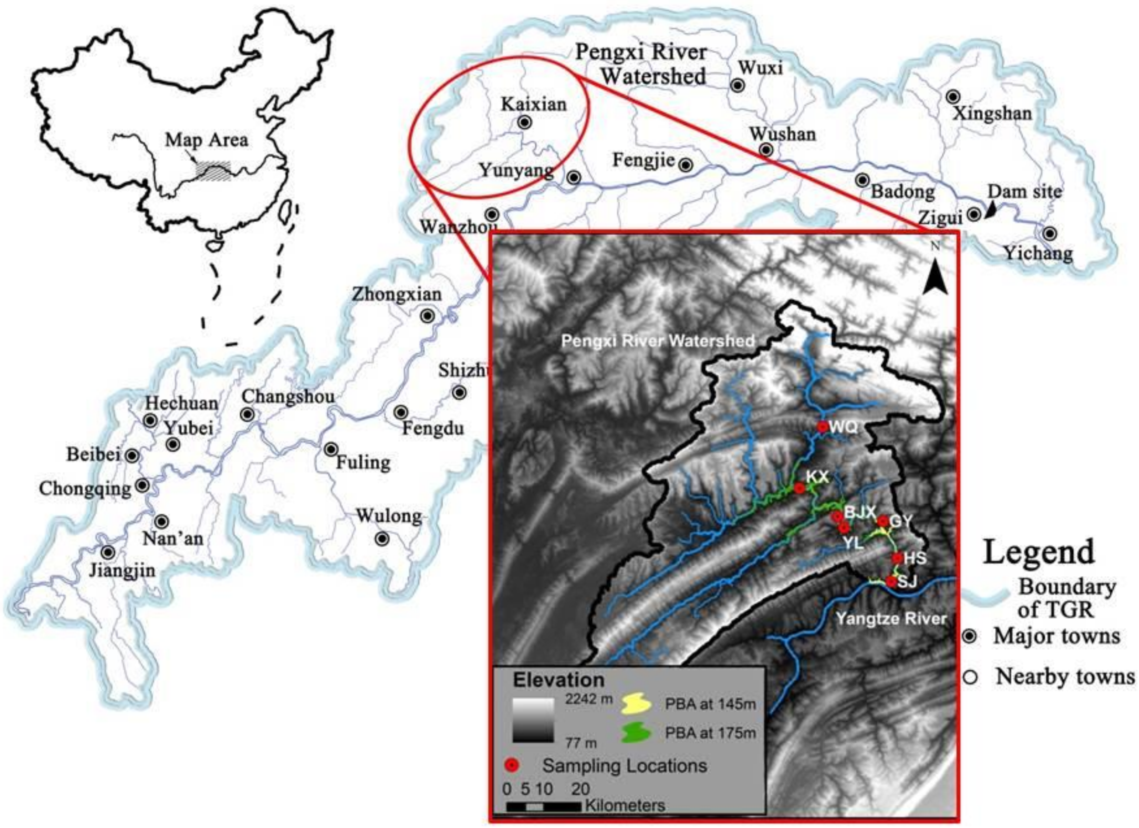

Figure 6-1: Map of the Three Gorges Reservoir and Pengxi River. Pengxi River (also named as Xiaojiang River) is located at the mid-reach of the Yangtze in the Three Gorges Reservoir, about 250km upstream from the Three Gorges Dam. There are 7 sampling spots along the 80km backwater area of the Pengxi River. From upstream to downstream they are: (1) Wenquan (WQ), an unaltered river location; (2) Kaixian (KX), the terminal backwater region at a high water level; (3) Baijiaxi (BJX), the terminal backwater area in the discharge period; (4) Yanglu (YL), the terminus of low water operation; (5-7) Gaoyang (GY), Huangshi (HS), and Shuangjiang (SJ), three permanent backwater regions.

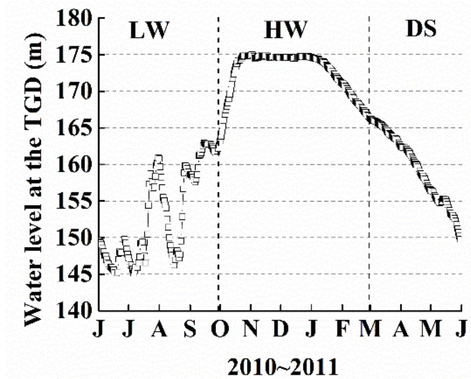

Figure 6-2: Water level variation from June 2010 to May 2011; the low water period (LW) is from June – September, the high water period (HW) is from October – February, and then the discharge period (DS) is from March – May. Data came from the website: www.cwic.com.cn.

6.2.2 Sampling

Monthly sampling events were carried out from June 2010 to May 2011. Sampling time was controlled between 9:30 AM to 4:30 PM for all six sampling spots. Water samples were collected from the main channel with 3L Kitahara's water sampler. pH and conductivity were measured in situ using a YSI® sonde (YSI 63), with a precision and estimated accuracy of ±0.01 pH and ±0.1 units, respectively. The pH sensor was calibrated with standard solutions (pH 4, 7, and 10) before sampling events. Temperature and dissolved oxygen (DO) were measured with a YSI® Pro ODO® probe, which has a precision of ±0.1℃, and ±0.01mg-DO L^{-1}. The probe was calibrated in water-saturated air prior to sampling. Three-minute continuous wind speed measurements were taken on site before and after each sampling event using a hand-held anemometer (Smart® AR-826, accuracy 3%, response 1s, operational range 0.3-45 m s^{-1}) at 2m above the water surface. The results from the two measurements were then averaged to determine the representative on-site instantaneous wind speed of the region.

6.2.3 Diffusive fluxes of CO_2 and CH_4

Diffusive CO_2 and CH_4 fluxes were measured directly with floating chambers from a small boat that was left to drift during measurements (Duchemin et al. 1999; Matthews et al. 2003). The floating chambers (0.14 m^2 and 14.2 L) were covered with heat-resistant material, Mylar paper and fitted with a stabilizing Styrofoam collar which served to maintain the upper, closed portion of the chamber about 4 cm above the water surface. During each sampling period, two chambers were simultaneously deployed on the surface water. Six gas samples of 50 mL were collected every 2 min over a 10 min period using 100-mL polypropylene syringes from each chamber. An equalizer pipe was fixed on the head to maintain the balance of gas pressure inside and outside the chamber.

Analyses of CO_2 and CH_4 concentrations in the gas samples were carried out in the laboratory within 24 hours. The gaschromatograph (Agilent 7820A) was equipped with a 0.25 mL sampling loop, a steel packed TDX-01 column, a flame ionization detector and a methane reformer. The diffusive fluxes were calculated using linear regression based on the concentration change as a function of time for the six samples. Acceptance of the results was based upon three criteria: (1) initial gas concentrations inside the chamber had to be ±10% of those measured in the atmosphere; (2) correlation coefficients (R^2) had to be >90% for CH_4 and CO_2 (Duchemin, Lucotte, and Canuel 1999; Soumis et al. 2004).

A previous validation experiment demonstrated that these static chambers allowed for an accurate estimate of GHG transfers across the air/water interface. Also, wind conditions were appropriate (i.e., < 3 m s^{-1}) for chamber deployment on most occasions. Accordingly, there was confidence in the accuracy of measurements.

6.2.4 Dissolved nutrients

The surface water at each site was sampled for chlorophyll a (chl-a), nitrogen, phosphorus, and inorganic and organic carbon concentrations. In the laboratory, samples were filtered through Whatman® GF/C membrane and chlorophyll pigments were extracted using 90% acetone solution for 36h and analyzed spectrophotometrically at 750, 665, 645 and 630nm. Water samples were filtered through pretreated (450°C for 4 h in Muffle furnace) Whatman® GF/F glass fiber membranes for soluble reactive phosphorus (SRP) and dissolved total nitrogen (DTN). All chemical analyses used visible or ultraviolet spectrophotometric methods (APHA 1995). Dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) concentrations were determined using a high temperature combustion method with a Shimadzu® TOC-V TOC analyzer (Shimadzu®, Japan). The dissolved CO_2 concentration, which is indicated as CO_2 partial pressure (pCO_2) was calculated by measured pH, water temperature, and DIC concentrations (Goldenfum and Association 2010).

6.2.5 Statistical analyses

According to previous hydrological, meteorological, water quality, and microbiological monitoring data, the TGR operation cycle was divided into three stages (Figure 6-2): low water operation (LW; June - September), high water operation (HW; October - February), and discharge period (DS; March - May). Datasets were divided into 3 respective sub-sets to elucidate the potential regulation of reservoir operations on carbon processing in the PBA. GHG fluxes and environmental variables were not normally distributed and therefore did not meet the criteria for Pearson's correlation coefficient analysis. Therefore, Spearman's Rank correlation analysis was used. Minitab 17.0 was used to conduct all statistical analyses.

6.2.6 Estimation of Gross CO_2 and CH_4 Emissions

Gross CO_2 and CH_4 emissions for the Pengxi Backwater Area were estimated using spatially – weighted monthly emissions. First, a 30-m digital elevation model (DEM) was used to determine the area inundated when the water level at the dam was at an elevation of 145m, 150m, 155m, 160m, 165m, 170m, and 175m. The inundated areas were digitized to create ArcGIS shapefiles and to calculate the area. Then, using ArcGIS tools the inundated areas were split into six smaller polygons that correspond to one of the six sampling locations within the Pengxi inundated area (WQ is excluded because it is upstream of the inundated area). To determine the splitting point the stream length and mid-point along the stream segment were calculated between each of the sampling locations. The mid-point between locations was used as the splitting point (see Figure 6-3 for a visual example). The areas of the six polygons corresponding to each sampling location were calculated at water elevations of 145m, 150m, 155m, 160m, 165m, 170m, and 175m. Daily records of water level at the TGR dam (see Figure 6-2) were then used to match the Pengxi water body shape area to the corresponding monthly flux measurements.

Monthly fluxes from each sampling location were multiplied by the corresponding surface area to determine gross emissions of both CO_2 and CH_4. For example, when water samples were collected in May the water level at the dam was approximately 155m. Therefore, May fluxes are multiplied by the segment areas determined at 155m elevation. However, in some months there was great variation in the water level at the dam between the several days when water sampling occurred. In these cases, the area most closely corresponding to the water level at the time of sampling was used. For example, in August locations SJ, HS, and GY were sampled when the water level at the dam was near 150m, but locations YL, BJX, and KX were sampled when the water level at the dam was near 155m. Therefore, in this month fluxes from SJ, HS, and GY were multiplied by the segment areas calculated at 150m and fluxes from YL, BJX, and KX were multiplied by the segment areas calculated at 155m. This method assumes that measured fluxes are representative of overall fluxes for each month and that they are also representative for the spatially proximate inundated areas near that sampling location.

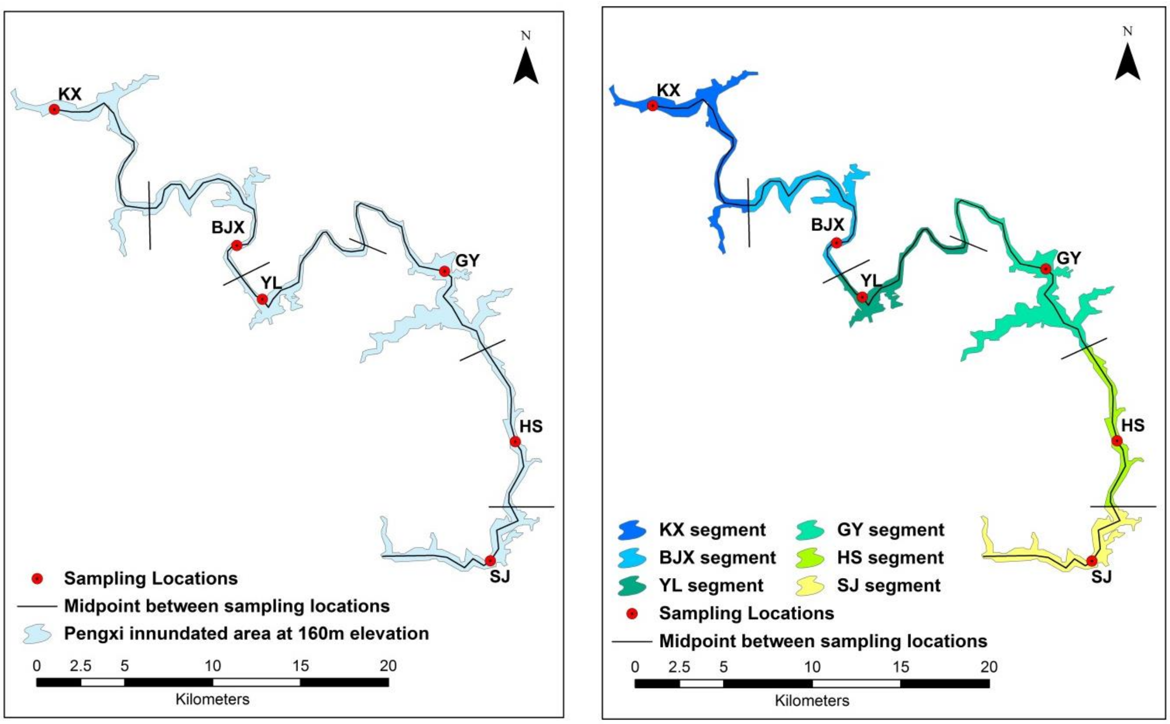

Figure 6-3: Left image depicts the inundated area of the Pengxi tributary at 160m elevation and the determined midpoints between the sampling locations. Midpoint locations were used to split the full area into six segments that correspond to sampling locations, as shown in the right image.

Table 6-2: Calculated area of each sampling segment corresponding to the water level elevation at the dam as it fluctuates throughout the year.

	Calculated area of sampling segment in m^2 at various water level elevations						
Sampling Station	145m	150m	155m	160m	165m	170m	175m
SJ	6218586	6617892	7317762	8623835	9213182	10122657	10102739
HS	3395344	3408474	3863330	4885559	4706218	5439536	5362384
GY	9024510	10109271	11992128	14793017	14971728	17695896	17326398
YL	1031645	3540217	4652533	6094607	5779014	7132688	7177900
BJX	1216506	3957613	5870446	8331569	9383433	11825563	12207834
KX	1072543	4900413	4616706	9554633	18532647	35226747	46867968

6.3 Results

6.3.1 Meteorological and aquatic chemistry conditions

The air temperature 2 m above the water surface varied from 6.5°C in January to 38.5°C in July, with an average annual temperature of 22.4±9.0°C; the total precipitation was about 1200 mm during 2010. Wind speed at 10m varied between 0 and 3.4 m s^{-1} and air humidity was between 45.2% and 95.3% during the study period. Mean surface water temperature deceased from 32.2 ± 3.7°C in August to 13.3 ± 0.84°C in January. All the sampling sites had similar seasonal variations in surface water temperature and surface water DO. However, the boxplots demonstrate a slight increasing gradient in DO from upstream to downstream. WQ had the lowest surface water temperature and DO among the sampling sites. The highest measured DO and pH occurred during at the SJ location in March 2011, with a DO value of 20.67 mg L^{-1} and pH of 9.47. The lowest values were observed at the BJX location in November 2010 at the beginning of the high water operation.

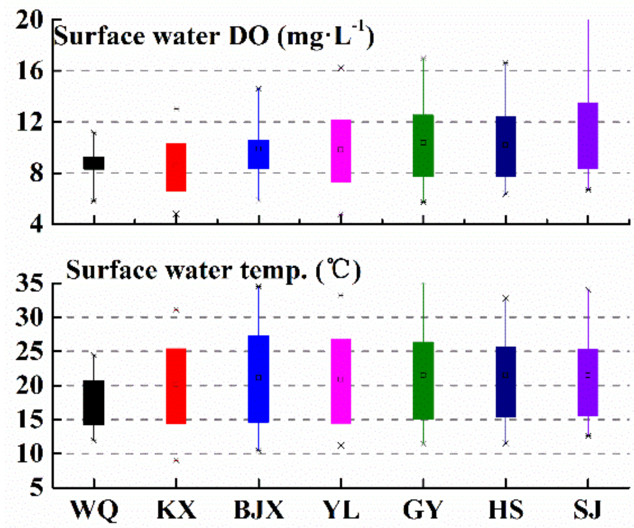

Figure 6-4: Boxplots of surface water temperature and DO; surface water temperature ranged from 9.0°C to 35.1°C, and surface DO ranged from 4.77 mg·L^{-1} to 20.67 mg·L^{-1} in all the sampling sites during the study.

Strong thermal stratification began in May and was maintained throughout the summer flood season (Figure 6-5). The estimated mixing layer in the water column was approximately around 8-10 m below the water surface. No thermal stratifications occurred in winter. Nevertheless, this monomictic system showed several instances of DO stratification and mixing events in the water column during an annual cycle. As shown in Figure 6-5, a significant DO stratification event initiated in late June, intensified in August, and diminished in mid-September. During this time, DO was super-saturated in the upper water column and a zone of anoxia formed in the lower water column. When the TGR water level increased, the DO re-mixed in the water column. Weak DO stratification also occurred in October and diminished in

November. Then in March and May strong DO stratification events occurred at both GY and SJ. However, these events did not persist and DO mixing was detected in April.

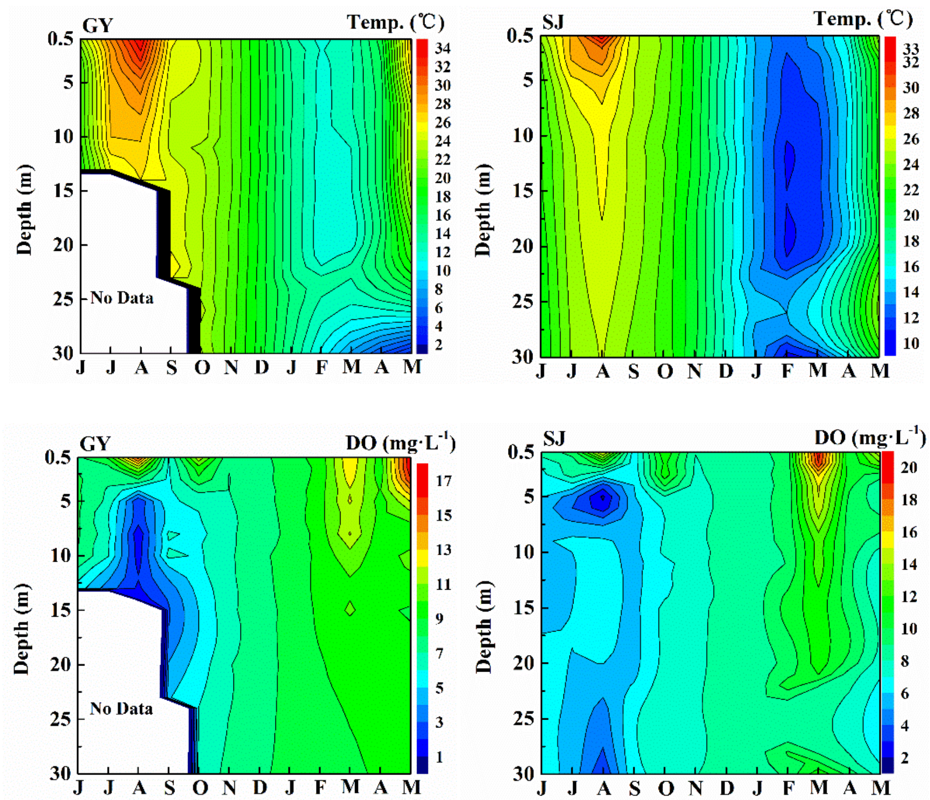

Figure 6-5: Vertical profiles of water temperature and DO in GY and SJ in PBA, Pengxi River.

6.3.2 pCO_2 and Chl-a

pCO2 at WQ varied between 1200 and 3400 ppm with an average of 2049.5±197.2 ppm and was frequently supersaturated and the highest among the sampling sites. Although the maximum pCO2 at WQ was detected in September, there were no significant differences among the LW, HW, and DS periods (ANOVA, sig. >0.05) at WQ. In both the FBA and PBA, surface water pCO2 values were the lowest in the DS period but the highest during the HW period. Low levels of surface water pCO2 were also detected in July and August in the LW period. Surface water pCO2 showed a general decrease from upstream to downstream.

With minimal seasonal variation, Chl-a measurements at WQ were also among the lowest in all sampling sites. The average value of Chl-a at WQ was 2.71 μg L-1, and ranged from 0.61 to 9.85μg L-1. Average Chl-a concentrations and monthly variation increased slightly downstream. Blooms of phytoplankton (Chl-a >10μg·L^{-1} and up to 290 μg·L^{-1}) occurred in both the FBA and the PBA in August and October of 2010, as well as in March and May of 2011. Generally, levels of Chl-a at the PBA sampling sites were significant higher than those in FBA. During the HW period Chl-a was at its lowest values for the year.

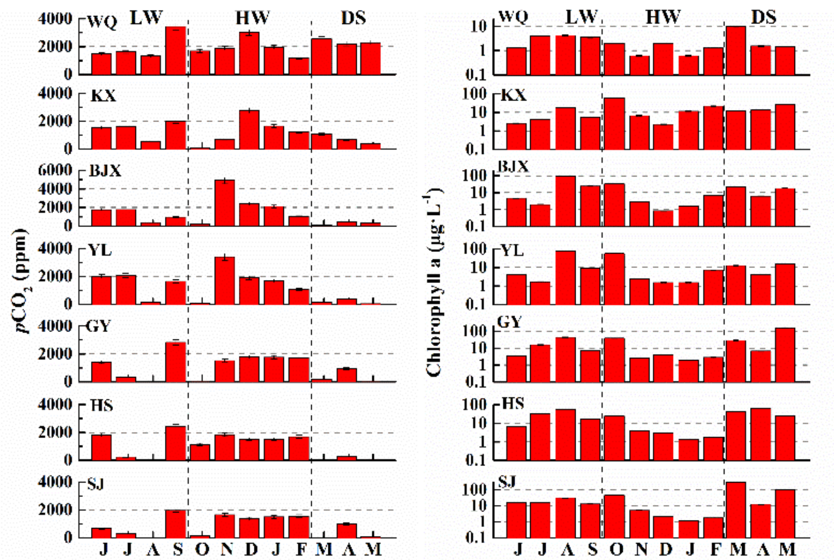

Figure 6-6: Monthly surface partial pressure of carbon dioxide (pCO2) and chlorophyll a (Chl-a) at each sampling site.

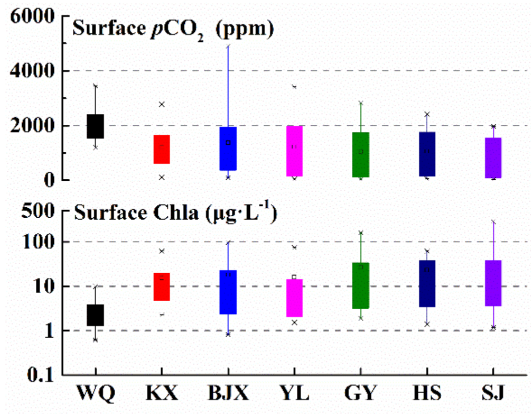

Figure 6-7: Box-plots of surface partial pressure of carbon dioxide (pCO2) and chlorophyll a (Chl-a) at each sampling site.

Vertical profiles of pCO_2 and Chl-a in PBA (at GY and SJ locations) displayed similar patterns to DO, which provides evidence for phytoplankton blooming events in the FBA and PBA. At GY and SJ, the first event of pCO_2 and Chl-a stratification initiated in late June at a water depth of approximately 5 m and extended to September. As the water level increased from September to October, surface water pCO_2 at GY and SJ experienced a steep rise and fall. At GY, surface water pCO_2 increased from 30.5±2.1 ppm in August to 2831.2±198.2 ppm in September. In October, pCO_2 was down to 46.8±3.3ppm, which is only 1.7% of surface water pCO_2 in September. However, the pCO_2 stratification in October disappeared in November. In March and May vertical profiles of pCO_2 in PBA indicated that there were also 2 stratification events. Surface water pCO_2 was at a low level (less than 500 ppm) during this time. Especially, there was a clear decrease in pCO_2 within the 30m water column from Feburary and March. Water column pCO_2 was below 1000ppm from March to May.

There were 4 events of Chl-a stratification throughout the year. Chl-a stratification occurred simultaneously with stratification of pCO_2 and DO. In GY, surface Chl-a was 44.2±

1.8 μg·L^{-1} in August and 38.9±1.6 μg·L^{-1} in October. Chl-a stratification occurred at a water depth of approximately 5m. In HW, Chl-a values were below 5 μg·L^{-1} until March. Surface Chl-a was 162.7±1.6 μg·L^{-1} in May which was the maximum of the year. In SJ, the maximum surface Chl-a was in March, which was 286.1±11.4 μg·L^{-1}. Surface Chl-a in May was 100.6±4.0μg·L^{-1}. Although phytoplankton blooms occurred, concentrations of Chl-a in August and October were much lower than those of March and May, which were 31.7±1.3μg·L^{-1} and 44.7±1.8 μg·L^{-1}, respectively. Nevertheless, 4 events of Chl-a stratification occurred at SJ at an approximate water depth of 5m.

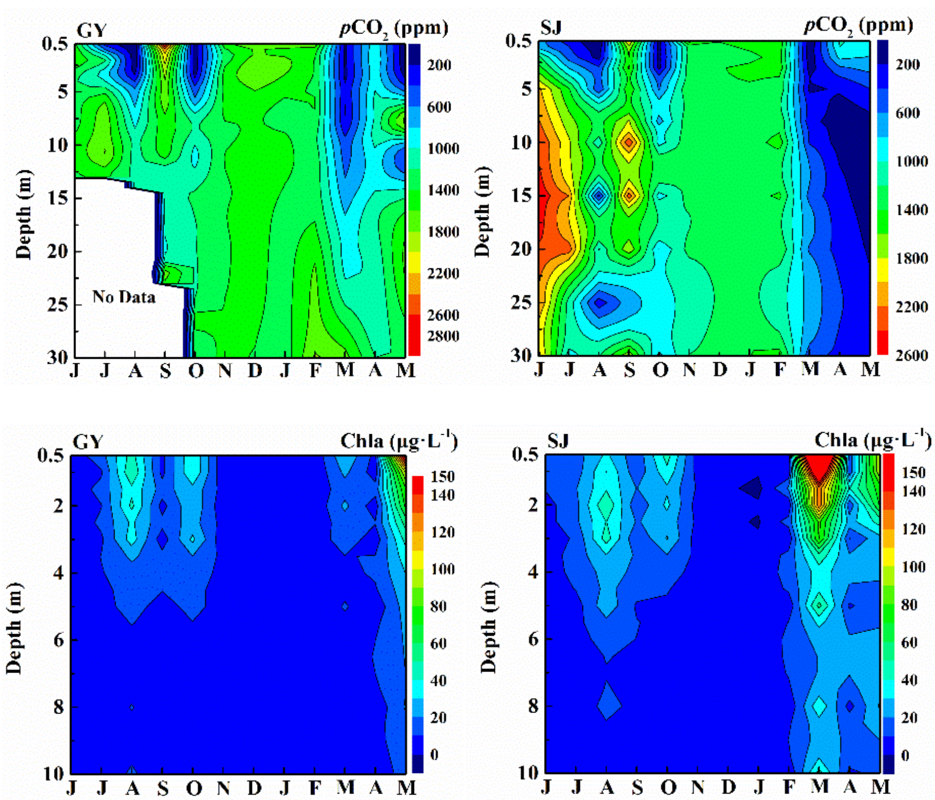

Figure 6-8: Vertical profiles of pCO_2 and Chl-a at GY and SJ locations in the PBA, Pengxi River.

6.3.3 Nutrients

Monthly variations of DOC, DIC, TN, DIN, TP, and SRP are shown in Figure 6-9. At WQ all nutrients concentrations were low compared to those in the FBA and PBA. Annual average TP at WQ was 0.034±0.006 mg·L^{-1}, while annual average TN and TOC at WQ were 1.016±0.085 mg·L^{-1} and 3.51±0.040 mg·L^{-1}, respectively. In the FBA concentrations of nutrients were much higher. KX had the highest concentrations of nutrients among all the sampling sites. Nutrients accumulated in the water column at KX during the LW period, especially in June and July. Concentrations of TP and TN at PBA sampling sites were greater than those in the FBA. However, there was an apparent decrease in TOC and DOC from FBA to PBA (i.e. downstream).

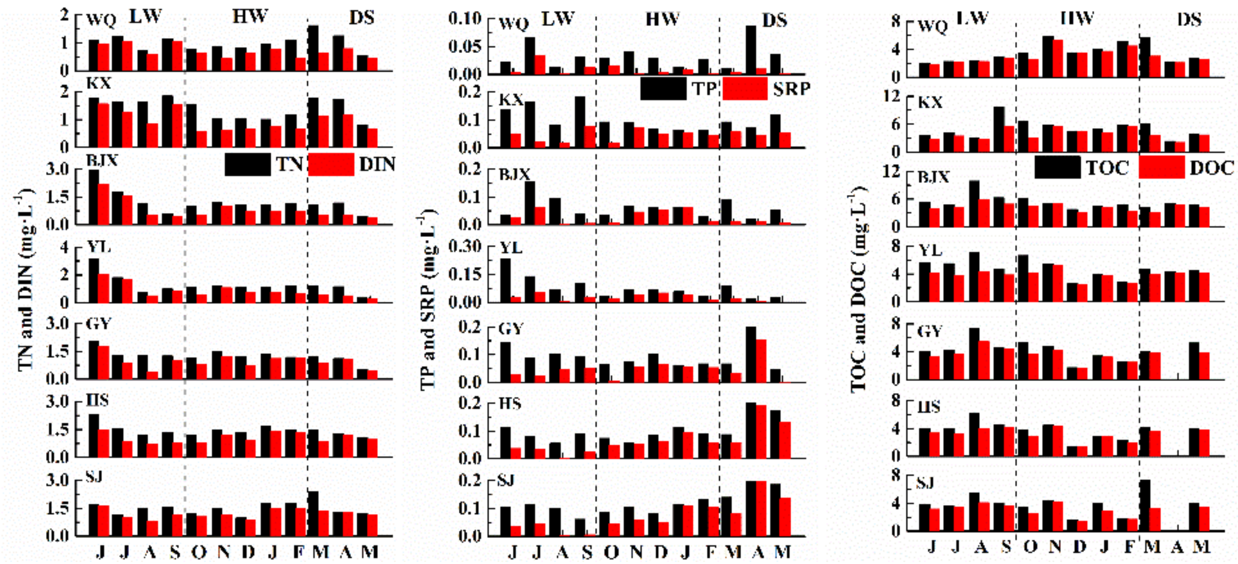

Figure 6-9: Variation in monthly TN, DIN, TP, SRP, TOC, and DOC at all sampling sites in the Pengxi River.

Temporal variations of nutrients were apparent. Generally, nitrogen and phosphorus were high in the first few months of the LW and DS periods, i.e. June, July, March and April. Decreases in nitrogen and phosphorus were obvious after the flood season in August and

September. At the KX location an obvious increase in nitrogen and phosphorus were detected at the end of the LW period and then again at the beginning of the HW period, indicating the effect of rising water levels and the extension of the backwater area to the KX location. However, carbon did not show the same monthly variations. TOC and DOC were generally high from August to October. Both TOC and DOC decreased during the HW period and increased in the first few months of the DS period, i.e. March and April. However, TOC and DOC data were missing from April measurements at locations GY, HS, and SJ; therefore, it is challenging to make a strong conclusion about April carbon levels. It is evident that the temporal variations of TOC and DOC were closely related to the growth of phytoplankton and the occurrence of algal blooms in both the FBA and PBA.

6.3.4 CO_2 and CH_4 fluxes

The air-water interface at the WQ riverine location acted as a source of CO_2 and CH_4 throughout the entire year. Effluxes of CO_2 ranged from 0.84 to 11.73 mmol h^{-1} m^{-2}, and CH_4 from 0.006 to 0.070 mmol h^{-1} m^{-2}. The annual average flux of CO_2 was 6.23 mmol h^{-1} m^{-2} and CH_4 was 0.025 mmol h^{-1} m^{-2}. In contrast to the riverine location, surface water in the backwater region showed great seasonal variations of CO_2 and CH_4 diffusion fluxes.

In the FBA (i.e. KX, BJX and YL) CO_2 fluxes were amongst the highest in all the 7 sampling sites. During the winter HW period the maximum level of CO_2 fluxes (6.66±1.62 mmol h^{-1} m^{-2}) were witnessed. Negative values of CO_2 fluxes began in February and continued throughout the spring and summer seasons in the FBA, indicating that the area was a CO_2 sink during this period. Conversely, the highest CH_4 fluxes in the FBA occurred in May and June and the lowest CH_4 fluxes were generally in winter, except for KX in February when an extremely high emission of CH_4 was observed. Even in BJX and YL, some samples of CH_4 fluxes were below detection limit. The PBA (i.e. GY, HS, and SJ locations) had similar variations of air-water fluxes. CO_2 sinks were frequently observed during the DS and LW periods. Simultaneously, maximum levels of CH_4 effluxes were also detected during these periods in the PBA. Generally, there was significant increase in the level of CH_4 fluxes from WQ to the FBA and a gradual decrease from sampling sites in the FBA to the PBA. For CO_2 fluxes, the decreasing trend from WQ to downstream sampling sites was much more apparent.

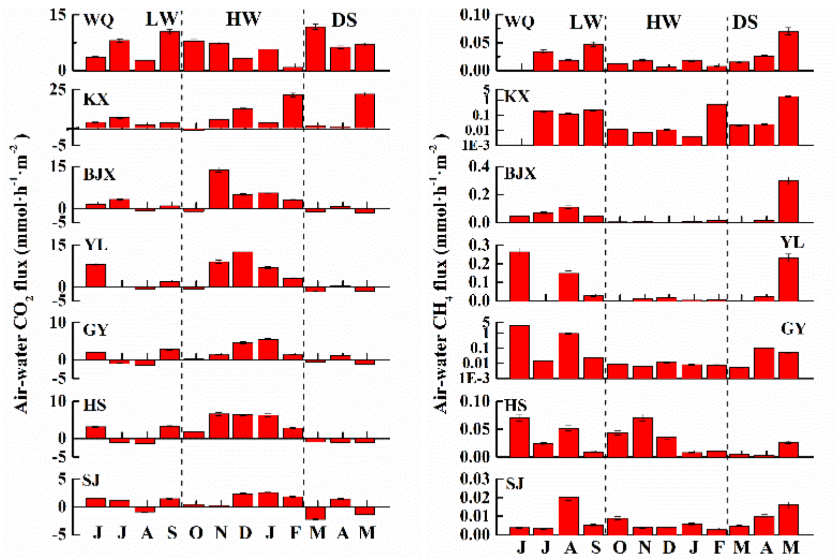

Figure 6-10: Monthly CO_2 and CH_4 fluxes at each sampling site in the Pengxi Tributary

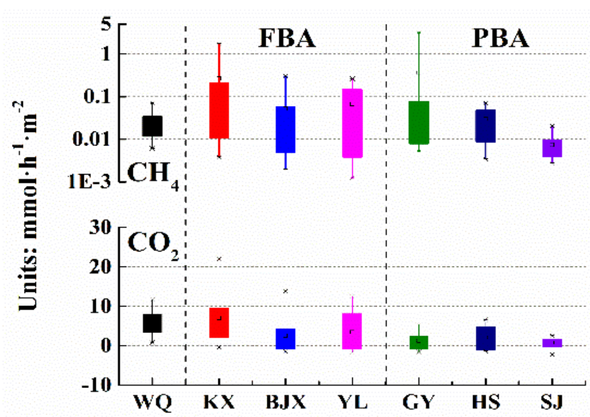

Figure 6-11: Boxplots of CO_2 and CH_4 fluxes at each sampling site in the Pengxi Tributary

6.3.5 CO_2 and CH_4 diffusive fluxes and environmental parameters

Spearman's rank correlations were examined between CO_2 and CH_4 diffusive fluxes and water environmental parameters to better understand possible mechanisms driving fluxes. In the WQ upstream river location there were no significant correlations between CO_2 diffusive fluxes and the independent variables water temperature, pH, DO, Chl-a, DTN, DTP, DIC, and DOC. During the LW period CO_2 diffusive fluxes were correlated significantly (at $p < 0.005$) with pH ($\rho = -0.70$), water temperature ($\rho = -0.71$), DO ($\rho = -0.76$), pCO_2 ($\rho = 0.79$), Chl-a ($\rho = -0.79$), and DIC ($\rho = 0.73$), as well as DTN ($\rho = 0.43$) and DOC ($\rho = -0.45$) at $p<0.05$. During the HW period, pH ($\rho = -0.68$), DO ($\rho = -0.55$), and pCO_2 ($\rho = 0.66$) were significantly correlated to CO_2 diffusive fluxes at $p<0.005$. Also, Chl-a ($\rho = -0.48$), and DIC ($\rho = 0.44$) had significant correlations at $p<0.05$. Water temperature and DTN were not significantly correlated with CO_2 diffusive fluxes during the HW period. During the DS period significant correlations were observed between CO_2 diffusive fluxes and pH ($\rho = -0.73$), DO ($\rho = -0.73$), and pCO_2 ($\rho = 0.79$), at $p<0.005$, as well as Chl-a ($\rho = -0.48$), and DIC ($\rho = 0.56$) at $p<0.05$; however, there was not a significant correlation with water temperature.

Table 6-3: Correlations between CO_2 diffusive fluxes and environmental parameters during the different storage periods

Operation Period	n	Water temperature	DO	pH	pCO_2	Chl-a	DTN	DTP	DIC	DOC
WQ Location	12	0.19	-0.19	-0.49	0.52	0.43	0.38	0.24	0.55	-0.05
Low	24	-0.71**	-0.76**	-0.70**	0.79**	-0.79**	0.43*	0.17	0.73**	-0.45*
High	30	-0.31	-0.55**	-0.68**	0.66**	-0.48*	-0.15	0.24	0.44*	0.26
Discharge	18	-0.18	-0.73**	-0.73**	0.79**	-0.48*	0.26	0.04	0.56*	-0.10
All	84	-0.30*	-0.67**	-0.77**	0.80**	-0.68**	0.03	-0.14	0.63**	-0.23*

(*) Indicates results are significant at the 0.05 level, (**) indicates results are significant at the 0.005 level.

During the LW and HW periods, CH_4 did not have any significant correlations with any variables. However, during the DS period CH_4 diffusive fluxes were correlated with water temperature (ρ =0.74, p=0.000), air temperature (ρ =0.68, p=0.002), and also with DTN (ρ =-0.57, p=0.014).

6.3.6 Estimate of Gross CO_2 and CH_4 Emissions from the Pengxi Water Body

Estimates of total monthly gross emissions from the entire Pengxi water body ranged from -791Mg CO_2 in March to 5120Mg CO_2 in December, and 1.44Mg CH_4 in March to 337Mg CH_4 in June. Average monthly CO_2 emissions were highest during the HW period at 2604Mg and lowest during the DS period at -89.3Mg CO_2. Total emissions of CO_2 were 2830Mg in the LW period, 13020Mg in the HW period, and -89.3Mg during the DS period. Average monthly CH_4 emissions were highest during the LW period at 119Mg and lowest during the HW period at 4.23Mg. Total emissions of CH_4 were 474Mg in the LW period, 21.2Mg in the HW period, and 51.3Mg in the DS period. Overall annual emissions were estimated to be 15600Mg CO_2 and 547Mg CH_4.

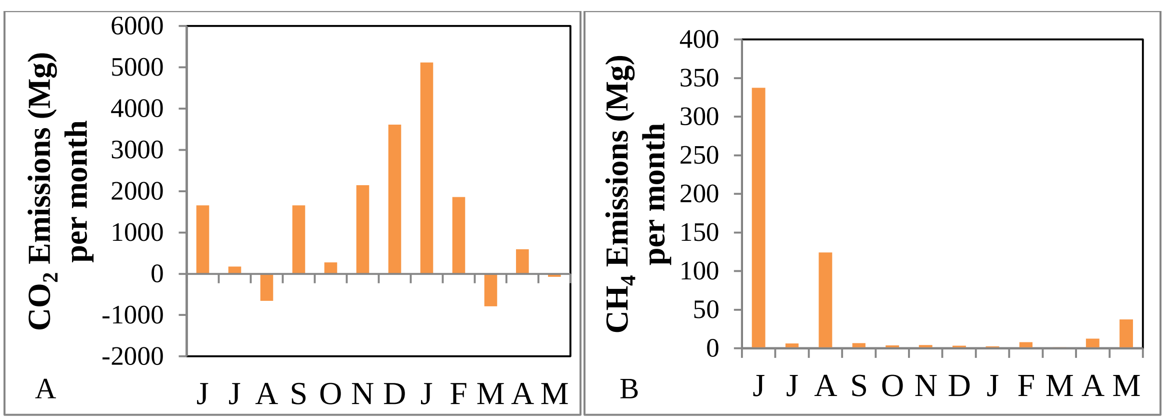

Figure 6-12: Monthly gross CO_2 emissions (Mg) are shown in A and monthly gross CH_4 emissions (Mg) are shown in B.

Table 6-4: Estimated monthly emissions of CO_2 and CH_4 (Mg) from the Pengxi Tributary

	Emissions (Mg)	
	CO_2	CH_4
June 2010	1660	333
July 2010	174	6.16
August 2010	-653	124
September 2010	1650	6.76
October 2010	275	3.59
November 2010	2150	4.03
December 2010	3610	3.33
January 2011	5120	2.52
February 2011	1860	7.70
March 2011	-791	1.44
April 2011	594	12.3
May 2011	-71	37.6

6.4 Discussion

6.4.1 Environmental conditions influencing CO_2 diffusive fluxes

At the WQ location, situated upstream of the backwater area, the DIC concentration seems to be largely governed by the buffering reactions of carbonic acid and the amount of bicarbonate and carbonate derived from the weathering of surrounding rocks. The quantity of phytoplankton in the upstream river was weak through the year, so the relative low levels of photosynthesis led to the supersaturation of CO_2 and subsequent efflux during the course of this study.

However, in the backwater area algae plays a large role in diffusive CO_2 emissions. During the LW period (June – September) the average water temperature (27.1±4.7 ºC), nutrient concentrations, and hydraulic residence time (~15-35 days) were adequate for algal growth. Consequently, photosynthetic activity from high algal biomass created an influx of CO_2 from the atmosphere into the aquatic environment (or a negative CO_2 efflux). There was a positive relationship between Chl-a concentrations and DO and pH and a negative relationship with pCO_2, DTN and DIC. During the HW period (October – February) algal production was not as critical to CO_2 fluxes. Although hydraulic retention time was long enough for phytoplankton growth (~100-160 days), and nutrient concentrations (DTN and DTP) were also adequate, the low water temperature (17.5±5.1 ºC) limited the growth of algae. During the HW period low Chl-a concentrations were observed, the relationships between CO_2 flux and both DO and Chl-a, are statistically significant but not as strong, while the relationship between DTN and CO_2 flux is no longer statistically significant. Phytoplankton dynamics still seem to be involved in regulating the CO_2 diffusive flux; however, other processes may be more influential during this time, such as decomposition of organic matter.

During the DS period algal growth again became the dominant factor regulating CO_2 efflux. Similarly to the LW period, there were highly negative correlations between CO_2 efflux and Chl-a, DO, and pH. Conversely there were strong positive correlations between DIC concentrations and CO_2 efflux. Additionally, several algal blooms occurred in the Pengxi River during the DS period (as can be observed in Figure 6-8), turning the backwater area into a carbon sink at all sampling locations except KX during March and May. However, many of these locations had positive CO_2 fluxes again in April. These results suggest there was a time between the two blooms when composition of dead algae led to pCO_2 supersaturated in the surface water, turning the Pengxi River backwater area into an atmospheric carbon source. There were no significant correlations between CO_2 diffusive fluxes and DOC during the discharge period, which is contrary to results from other studies (Sobek et al. 2005). However, it seems that the range of DOC values may play an important role in determining the effect on CO_2 diffusive fluxes. For example, Roehm et al. found a significant correlation (R^2= 0.41; $p < 0.001$) between DOC and pCO_2 in the Eastmain River region over a broad range of DOC concentrations (4.0 to 24.0 mg-C L^{-1}), (Roehm et al. 2009). Sobek et al. found a significant relationship between DOC and pCO_2 in a global analysis of data from 4555 lakes, also with a broad range of DOC concentrations (2005). In the Pengxi backwater area the DOC range is much narrower, 1.54 to 6.04 mg-C L^{-1}, which perhaps affected the statistical relationship between DOC and CO_2 diffusive fluxes.

Water level was used to represent the characteristics of the reservoir under different operating conditions in order to examine the impact of operations on CO_2 fluxes. The results indicated that the average CO_2 emissions of the backwater area were significantly positively related to water level ($\rho = 0.355$; $p < 0.002$). Reservoir storage operations changed the hydraulic residence time of the PBA, which in turn altered nutrient availability and turbidity, which regulated algal growth (Wetzel 1975). Each reservoir stage operation has a different dominant process regulating CO_2 fluxes in the backwater area. In the LW period, phytoplankton photosynthesis was the key ecological process that controlled CO_2 fluxes. However, during HW operation, algal growth was inhibited and was supplanted by carbon degradation, thus contributing 91% of the annual CO_2 efflux during this period. Then during the spring DS period, algal blooms created an alternating dominant state between CO_2 fixing and organic carbon synthesis, causing a net CO_2 influx, and organic carbon decomposition and carbon mineralization, causing a net CO_2 efflux. Our results are supported by Zhao et al.'s study of the Three Gorges Reservoir CO_2 production, which also demonstrated a substantial seasonal variation due to the photosynthetic drawdown in the spring and summer and the subsequent oxidation of organic carbon (Zhao et al. 2013).

6.4.2 Environmental conditions influencing CH_4 diffusive fluxes

Our estimates provide only a partial fraction of annual CH_4 released from the PBA because total ebullition and plant-mediated fluxes are not included. Other studies indicate that the nutrient concentrations, humic content, area and depth, are associated with the variation in CH_4 flux (Demarty et al. 2011; Lima 2005). Juutinen et al. showed that partial pressure of CH_4 in surface and hypolimnetic water is negatively correlated with dissolved oxygen concentrations, lake depth, and lake area (Juutinen et al. 2009). CH_4 emissions in the PBA had a significant negative correlation with water level ($\rho = -0.528$; $p < 0.000$), which supports the findings of Juutinen et al. To further support this relationship, the highest CH_4 effluxes were observed during the LW period, which also had the warmest average water temperature (27.4°C) and the lowest water depths. The average CH_4 diffusive flux during this period was 2.54×10^8 mmol/day, which is twice as high as the average flux during DS operation (1.01×10^8 mmol/day), and three times higher than the average flux during the HW period (8.42×10^7 mmol/day). Overall, the LW period produced approximately 59% of the total annual CH_4 emissions. However, we are unable to parse if the increased CH_4 is due to increased CH_4 production during the low water period, or if a higher percentage of the CH_4 is able to escape the water column without being oxidized due to shallower water depths. Most likely a combination of increased production of CH_4 in shallow sediments and enhanced transport to the water surface result in higher CH_4 emissions, or what has been called an "epilimnetic shortcut" (Bastviken et al. 2008).

The LW period also coincides with the period of high algal production, which would provide a large carbon source for methane production. However, there was no significant relationship between CH_4 and DO, nor CH_4 and DIC or DOC or Chl-a during any of the water operation periods. However, all relationships were tested using surface water measurements. Yet it is clear that there can be a great difference between surface and subsurface DO. For example, at locations GY and SJ that there were relatively high DO measurements at the surface during the LW period (June – August); however, at a depth of approximately 5m there is a drastic decrease in DO from 1-4 mg-L^{-1}. Therefore, it seems likely that low DO values near the sediment-water interface may be the cause of higher CH_4 emissions (Figure 6-5).

While this study found that summer months produced the highest CH_4 emissions, a previous study of the Three Gorges Reservoir determined that CH_4 emissions were higher in the winter (the HW period) compared to spring (DS period) and summer (LW period) (Chen et al. 2011b). Chen et al. suggested that the higher water levels in the winter could allow for more phytoplankton production and a greater amount of substrate for CH_4 production. Yet, the Chl-a measurements from our study show that algal production in the Pengxi tributary is actually much higher during the LW period (21.11±24.75 μg L^{-1}) and the DS period (47.82±71.60 μg L^{-1}), compared to the HW period (11.90±17.40 μg L^{-1}). This difference is most likely due to the differing dynamics of a shallow tributary system compared to the deeper reservoir system.

In addition, other studies of methane emissions from the Three Gorges Reservoir determined that CH_4 emissions were higher from the drawdown area than the permanently flooded sites (Chen et al. 2011b). However, our results from the Pengxi area suggest the opposite, average (0.125 mmol h^{-1} m^{-2}) and total (4.24 mmol h^{-1} m^{-2}) fluxes from the three drawdown sites (KX, BJX and YL) were somewhat lower than average (0.133 mmol h^{-1} m^{-2}) and total (4.81 mmol h^{-1} m^{-2}) fluxes from the permanently flooded sites (GY, HS, SJ). However, another study of the Three Gorges Reservoir and tributaries found that there are significant spatial variations in littoral CH_4 fluxes and therefore caution should be used when making comparisons and generalizations of CH_4 fluxes (Zhao et al. 2013).

6.4.3 Comparison and Regional Clustering of Reservoirs

In total, 155 estimates of CO_2 emissions and 103 estimates of CH_4 fluxes were assembled (Barros et al. 2011b; Demarty et al. 2009; Roland et al. 2010; Soumis et al. 2004). Most of the reservoirs are situated in the northern temperate and tropical climate zones, in part reflecting the global distribution of reservoirs, and very few studies were focused on subtropical and warm temperate climates. In this data set, mean CO_2 and CH_4 flux for all 92 reservoirs was 481.7±555.4 mg-C m^{-2} d^{-1}, 48.2±131.5 mg-C m^{-2} d^{-1}, respectively. All the reservoirs were sources of CH_4 to the atmosphere except one reservoir in Canada, and 90% were also a source of CO_2, only about 10% of reservoirs were net sinks of CO_2, and in these cases the effect was small. The highest CO_2 influx was -325.9 mg-C m^{-2} d^{-1} in a temperate reservoir compared to a maximum efflux value of 2845.4 mg-C m^{-2} d^{-1} in a tropical reservoir. Figure 6-13 shows the distribution of CO_2 and CH_4 reservoir emissions of different climatic regions. Barros et al. conclude that the areal emissions of both CO_2 and CH_4 from hydroelectric reservoirs were significantly negatively correlated to latitude, with highest emission rates near the tropics and lowest emission rates at high latitudes.

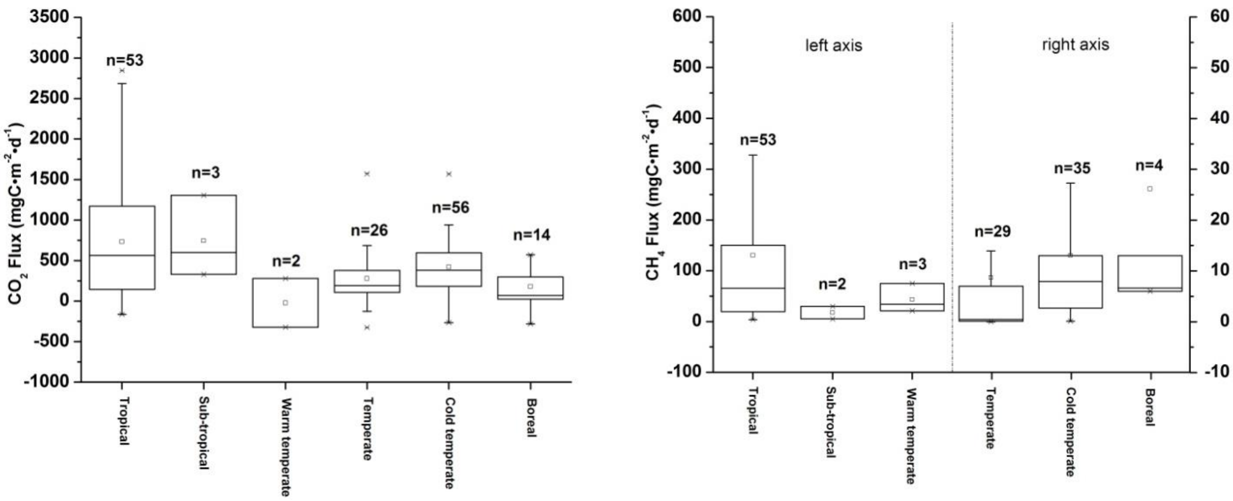

Figure 6-13: Distribution of flux of CO2 and CH4 in different climatic regions, as defined by latitude (tropical: 0°-23°26′; sub-tropical: 23°26′-35°; warm temperate: 35°-40°; temperate: 40°-48°; cold-temperate: 48°-56°; Boreal: >56°). The diamond depict the average value, the boxes show the quartiles, and the whiskers mark the 10% and 90% percentiles. The number of reservoirs in each climatic region is shown.

The CO_2 and CH_4 fluxes in the Pengxi backwater area ranged from -26.8 to 263.4 mg-C $m^{-2}\,h^{-1}$ and from 0.015 to 38.07 mg-C $m^{-2}\,h^{-1}$, respectively. If we assume that the gas flux in our research region was consistent throughout the day, then we can calculate a mean daily CO_2 flux of 946.9 mg-C $m^{-2}\,d^{-1}$ and a mean daily CH_4 flux of 33.12 mg-C $m^{-2}\,d^{-1}$. The Pengxi mean CO_2 flux is well above the mean reported values in the literature, and is in fact near the upper end of the range of values reported for tropical and subtropical reservoirs. Pengxi CH_4 fluxes are below the global average and similar to values reported for subtropical and temperate reservoirs. It is important to note that the total CH_4 ebullition flux is not represented in these values, as it remains a challenge to quantify total ebullition over the entire water area. Therefore, the estimates put forth here are conservative and the actual gross CH_4 fluxes are most likely higher.

6.5 Conclusions

This study observed CO_2 and CH_4 fluxes in surface waters of the Pengxi Tributary, which is a part of the Three Gorges Reservoir system, during an annual cycle of reservoir storage operation. There were great seasonal variations in CO_2 and CH_4 fluxes, with CO_2 fluxes particularly influenced by water level and algal production. Due to algal photosynthesis, pCO_2 decreased greatly during the low water operation and discharge periods, at times even lower than the atmospheric level of CO_2 causing the water to become a sink for CO_2. During the high water operation period, there were large effluxes from the water into the atmosphere contributing 91% of the annual CO_2 emissions from the Pengxi area. Methane production was highest during the low water level operation and lowest during the discharge period. From this study, we were not able to statistically determine the main environmental conditions controlling methane production. However, it is clear from the vertical DO profiles that low DO levels near the substrate-water interface is most likely a strong factor in high methane emissions during the LW period. In addition, high fluxes during the LW period may be due to a combination of increased carbon production from algal growth and increased ability to escape the water column due to low water depths. This study suggests that hydropower management has a large role in the fluxes of reservoir greenhouse gas emissions by altering the water depth and hydraulic residence time.

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Chapter 7 – Conclusions and Recommendations

7.1 Conclusions

The overall goal of this dissertation was to evaluate the impacts of climate change and land-use change on water resource management using an interdisciplinary approach. Tools utilized include an ecohydrological model (ArcSWAT), geospatial analysis, literature review, and a social survey. There was an overall geographical focus on Kansas in chapters 2 – 5; however, study conclusions are more broadly applicable to other states and regions. In particular, the conclusions from the land-use study may be applied to other Great Plains states that have the potential to grow grain sorghum as a biofuel feedstock. Also, the review of climate change impacts on reservoir systems in chapter 4 can be applied to most reservoir systems, but may have the greatest relevance to mid-size reservoirs in agricultural watersheds.

The first study presented the development and calibration of two SWAT models that represent the Perry Lake and the Kanopolis lake watersheds. This study showed that SWAT was able to successfully simulate streamflow and sediment yield over a wide range of hydrologic conditions. Crop yield was also simulated with reasonable accuracy in both watersheds. The SWAT model performed very well in the Perry Lake watershed predicting annual and monthly streamflow with low error. However, SWAT was less accurate predicting streamflow and crop yields in the Kanopolis Lake watershed. The Kanopolis Lake watershed is large, with little relief, and with significant groundwater – surface water interactions. In addition, annual average precipitation varies greatly across the watershed with less than 483mm in the west and up to 711mm near the watershed outlet. For these reasons, SWAT was not able to simulate the crop growth and hydrology with a great deal of accuracy. Overall model performance was still acceptable and within recommended metrics (Moriasi et al. 2007).

The SWAT models were then used to explore the impacts of biofuel-based land-use change in the same two Kansas watersheds. Land-use change scenarios focused on increasing grain sorghum and corn land-use in exchange for either winter wheat, hay, or CRP land-use. Simulations demonstrated that increasing both corn and grain sorghum resulted in higher total nitrogen, total phosphorus, and sediment loads in both watersheds. Specifically, extensification of corn or grain sorghum cropland to hay or CRP land-uses resulted in the highest water quality impacts. Intensification of winter wheat cropland to either corn or grain sorghum produced changes in water quality indicators that were not statistically different from the baseline scenario. Corn-based scenarios produced statistically greater water quality impacts than grain sorghum scenarios. However, corn had a higher yield potential per area, which was demonstrably higher in the Perry Lake watershed. The higher yield resulted in better land, nutrient, and water use efficiencies in comparison to grain sorghum in Perry Lake watershed. In Kanopolis Lake watershed both crops had similar land, nutrient, and water use efficiencies. Therefore, this study demonstrated that grain sorghum is a more environmentally sustainable choice as a biofuel feedstock in central and western Kansas, as well as other areas of the Great Plains with low average annual precipitation.

Chapter 4 provided an assessment of the impacts of climate change for reservoir systems, and a review of watershed and in-reservoir management strategies that have potential to mitigate the impacts of climate change. Reservoirs provide many services to regional populations, but the sustainability of reservoir services is threatened by climate change. Current reservoir issues such as sedimentation, algal blooms, and water supply shortages will be further complicated by climate change. A review of management strategies suggested that climate adaptation may require thinking beyond current practices and employing simulation modeling to estimate nutrient, water, and sediment loads. However, the tools available require large amounts of data and a degree of technical expertise that may make them of limited applicability for day-to-day management efforts. Nonetheless, collaborations between reservoir managers and climate scientists may help develop regional simulation modeling platforms that can explore and virtually test adaptive management strategies in the context of altered climate patterns.

The issue of climate change and water resource management was also explored from the managers' perspectives using a survey. Respondents were asked about their personal perspectives towards climate change and its integration into state-based water planning. Respondents were targeted at three agencies: the Kansas Department of Health and Environment, the Kansas Water Office, and the Kansas Department of Agriculture – Division of Water Resources; 37 of 64 respondents finished the survey. Survey results suggest that Kansas water managers are interested in including climate change into state-planning efforts. However, barriers such as lack of funding and staff, as well as technical complexity stand in the way of climate integration. These barriers may be ameliorated through a top-down initiative outlined in the Kansas Water Vision, which provides a 50-year plan for Kansas water resources. The Kansas Water Vision has the potential to bring together researchers and planners in collaborative work that can produce management tools and knowledge that can make the state more resilient to future climate change.

Finally, the last study in this dissertation flips the perspective of climate change and water management presented in earlier chapters by examining how water management can influence greenhouse gas emissions. Water chemistry and greenhouse gas emissions from the Pengxi Tributary of the Three Gorges Reservoir were measured for one annual management cycle. Both CO_2 and CH_4 fluxes were influenced by water levels and exhibited distinct patterns that correspond to the reservoir operation cycle. Over 90% of CO_2 efflux occurred during the high water period, whereas the 58% of CH_4 efflux occurred during the low water period.

7.2 Recommendations

1. Developing a hydrologic model that represents past and current conditions is a difficult task. The SWAT models developed in this dissertation have the potential to be improved with the following recommendations:
    a. The SWAT models used in this study were developed using stationary land-use data from 2005 and then data from 2006-2008 to develop crop rotations. However, a more realistic model would include dynamic land-use change over time. As land-use data becomes more available for more years, this will become possible. In addition, BMPs were not modeled in the SWAT watersheds and could have an impact on sediment and nutrient export.
    b. Small impoundments, such as farm ponds, small reservoirs, and water detention structures, were not included in the watershed models. These structures retain water, nutrients, and sediment within the watershed and therefore are likely to affect peak flow, nutrient, and sediment export (Renwick et al. 2005, Bosch 2008). For a more detailed analysis of watershed processes, more of these structures should be included in future SWAT modeling.
    c. As these models were developed for large watersheds, STATSGO soil data was used. SSURGO soil data is available at a finer resolution, but there are still many gaps in the data. The use of SSURGO soil data could have the potential to improve modeling results, but this would also increase the number of HRUs in the

model and increase computational time. SSURGO could be used along with HRU thresholds to optimize the number of HRUs within each watershed.

   d. The SWAT model demonstrated shortcomings in accurately predicting streamflow and crop yield in a drier climate. The model should be further developed to improve the simulation of groundwater-surface water interactions and crop growth in semi-arid environments.

2. The land-use updater tool within the SWAT model worked well to conduct a relatively quick analysis of many land-use scenarios. However, it poses some limitations as it does not provide any control over the spatial nature of land-use change and it cannot reach the targeted change percentages indicated in the tool. Programming the SWAT model to change land-use using the Access database or SWAT input files, without relying on the updater tool, may result in more control in the spatial and temporal dynamics of land-use change simulations.
3. The survey of Kansas water managers provided an important advancement in the understanding of manager perspectives towards climate change. As Kansas is embarking on a new plan in the Kansas Water Vision it would be useful to come back to this topic in several years to see if managers have changed their perspectives, or to see how much progress has been made in the area of climate-water integration.

7.3 References

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Chapter 8 – Appendices

Appendix A. Crop management inputs for SWAT models

Perry Lake Watershed	
Alfalfa	
Planting Date	September 15
Harvesting Date	May 10, June 15, August 1 (for 4 years before replanting)
Fertilizer (Date, Amount) Nitrogen Phosphorus	 September 15, 112 (on year of planting) September 15, 134 (on year of planting)
Tillage	Disc + 4 field cultivations
Corn	
Planting Date	April 30
Harvesting Date	October 10
Fertilizer (Date, Amount) Nitrogen[1] Phosphorus[2]	 May 1, 170 kg ha^{-1} May 1, 45 kg ha^{-1}
Tillage	Conventional fall tillage, disc +3-4 field cultivators
Grain Sorghum	
Planting Date	May 20
Harvesting Date	October, 15
Fertilizer (Date, Amount) Nitrogen Phosphorus	 May 23,121kg May 23, 32kg
Tillage	Disc + 2 field cultivations
Soybean	
Planting Date	May 15
Harvesting Date	October 10
Fertilizer (Date, Amount) Nitrogen Phosphorus[3]	 None May 18, 23 kg ha^{-1}
Tillage	Disc + 4 field cultivators
Winter Wheat	
Planting Date	September 30
Harvesting Date	June 20
Fertilizer (Date, Amount) Nitrogen Phosphorus	 September 30, 32kg; January 10, 64kg September 30, 28kg

[1] 1.25 lb-N bu^{-1} corn x (county average corn yield [bu ha^{-1}] +10% for yield goal)
[2] 0.33 lb-P_2O_5 bu^{-1} corn x (county average corn yield [bu ha^{-1}] +10% for yield goal)
[3] 0.5 lb-P_2O_5 bu^{-1} soybean (county average soybean yield [bu ha^{-1}] +10% for yield goal)

Tillage	5 field cultivations
Kanopolis Lake Watershed	
Corn/Irrigated corn	
Planting Date	April 25
Harvesting Date	October 1
Fertilizer (Date, Amount) Nitrogen[i] Phosphorus[ii]	 April 26, 101 kg ha^{-1} (dry)/ 260 kg ha^{-1} (irr) April 26, 27 kg ha^{-1} (dry)/ 69 kg ha^{-1} (irr)
Tillage	Conventional fall tillage, disc + 3-4 field cultivations
Grain Sorghum	
Planting Date	June 1
Harvesting Date	October 15
Fertilizer (Date, Amount) Nitrogen[4] Phosphorus[5]	 June 2, 59 kg ha^{-1} June 2, 16 kg ha^{-1}
Tillage	7 field cultivations after wheat
Winter Wheat	
Planting Date	September 10
Harvesting Date	June 30
Fertilizer (Date, Amount) Nitrogen[6] Phosphorus[7]	 September 10, 75 kg ha^{-1} (1/3 at time of planting, 2/3 in following January) September 10, 22 kg ha^{-1}
Tillage	After wheat is harvested disc till, then 5-7 field cultivations if next crop is wheat; when grown in rotation with other crops, no till before planting wheat crop

[4] 1.25 lb-N bu^{-1} grain sorghum x (county average grain sorghum yield [bu ha^{-1}] +10% for yield goal)
[5] 0.33 lb-P_2O_5 bu^{-1} grain sorghum x (county average grain sorghum yield [bu ha^{-1}] +10% for yield goal)
[6] 1.7 lb-N bu^{-1} winter wheat x (county average winter wheat yield [bu ha^{-1}] +10% for yield goal)
[7] 0.5 lb-P_2O_5 bu^{-1} winter wheat (county average winter wheat yield [bu ha^{-1}] +10% for yield goal)

Appendix B. SWAT model inputs

Swat Model Input	Source
Weather station data	National Climatic Data Center (NCDC) weather station data http://www.ncdc.noaa.gov/cdo-web/
Digital Elevation Models (DEM) (30m)	U.S.Geological Survey http://nationalmap.gov/viewer.html
Soils	STATSGO Soil Database, Natural Resources Conservation service (NRCS) http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm
Land Cover Land Use Maps 2005	2005 Kansas Level IV map Kansas Applied Remote Sensing (KARS) Program, Kansas Biological Survey, KU http://kars.ku.edu/ 2006-2010 Can be found at USDA Data Gateway http://datagateway.nrcs.usda.gov/
Point source Municipal and Industrial discharges	Kansas Department of Health and Environment (KDHE) (Open records request) Environmental Protection Agency (EPA) Clean watersheds http://water.epa.gov/scitech/datait/databases/cwns/ EPA ECHO Databasehttp://www.epa-echo.gov/echo/
Irrigation	2005 KARS Irrigated Land-use maps, Kansas Applied Remote Sensing (KARS) Program, Kansas Biological Survey, KU http://kars.ku.edu/
Land management practices - planting and harvesting dates - fertilizer application rates and timing	Personnel communication Dr. Nathan Nelson Department of Agronomy, KSU Agricultural publications, Extension literature Department of Agronomy, KSU http://www.agronomy.ksu.edu/extension/p.aspx?tabid=55
Stream channels	National Hydrography Data (NHD) http://www.horizon-systems.com/nhdplus/
Reservoir outflow	Army Corps of Engineers and Bureau of Reclamation (personal communication)
Large reservoir parameters	Bureau of Reclamation http://www.usbr.gov/gp/lakes_reservoirs/kansas_lakes.htm
Location of small reservoirs and all water quality data	EPA Storet Database http://www.epa.gov/storet/dbtop.html

Appendix C – Land-use change scenarios

		Perry Watershed		Kanopolis Watershed	
Original Land-use Type	New Land-use Type	Targeted Land-use Change Percentage	Actual Land-use Change Percentages	Targeted Land-use Change Percentage	Actual Land-use Change Percentages
Winter Wheat	Grain Sorghum	10	7.0	2	1.2
		20	14	4	2.5
		30	21	6	3.7
		40	28	8	4.9
		50	35	10	6.2
		60	42	12	7.4
		70	49	14	8.7
		80	56	16	10
		90	63	18	11
		100	69	20	12
Winter Wheat	Corn	10	10	2	0.5
		20	20	4	0.9
		30	29	6	1.4
		40	39	8	1.9
		50	49	10	2.3
		60	59	12	2.8
		70	68	14	3.3
		80	78	16	3.7
		90	88	18	4.2
		100	97	20	4.7
Hay	Grain Sorghum	2	1.3	10	8.9
		4	2.7	20	18
		6	4.0	30	27
		8	5.4	40	36
		10	6.7	50	44
		12	8.1	60	53
		14	9.4	70	62
		16	11	80	71
		18	12	90	80
		20	13	100	88
Hay	Corn	2	2.0	10	8.9
		4	3.9	20	18
		6	5.9	30	27
		8	7.8	40	36
		10	9.8	50	45
		12	12	60	53
		14	14	70	62
		16	16	80	71
		18	18	90	80

		20	20	100	88
CRP	Grain Sorghum	5	3.7	5	4.5
		10	7.5	10	8.9
		15	11	15	13
		20	15	20	18
		25	19	25	22
		30	22	30	27
		35	26	35	31
		40	30	40	36
		45	34	45	40
		50	37	50	45
CRP	Corn	5	4.9	5	3.7
		10	10	10	7.4
		15	15	15	11
		20	20	20	15
		25	25	25	19
		30	30	30	22
		35	35	35	26
		40	40	40	30
		45	45	45	34
		50	49	50	37

Appendix D. Ethanol production potential from land-use scenarios

Table D-1: Increase in grain yield and subsequent ethanol production resulting from land-use scenarios substituting winter wheat, hay, and CRP for grain sorghum or corn in the Perry Lake watershed.

Added cropland area (km^2)	**Grain Yield (tons)**	**Ethanol Produced (L x 1000)**	**Added cropland area (km2)**	**Grain Yield (tons)**	**Ethanol Produced (L x 1000)**	**Added cropland area (km2)**	**Grain Yield (tons)**	**Ethanol Produced (L x 1000)**
Winter Wheat to Grain Sorghum			Hay to Grain Sorghum			CRP to Grain Sorghum		
2	1,061	376	13	5,507	1,953	5	2,410	855
5	2,122	752	25	11,010	3,906	11	4,819	1,709
7	3,183	1,129	38	16,520	5,859	16	7,229	2,564
10	4,244	1,505	51	22,030	7,812	21	9,639	3,418
12	5,304	1,881	64	27,540	9,765	27	12,050	4,273
15	6,365	2,257	76	33,040	11,718	32	14,460	5,127
17	7,426	2,634	89	38,550	13,671	38	16,870	5,982
19	8,487	3,010	102	44,060	15,624	43	19,280	6,837
22	9,547	3,386	115	49,560	17,577	48	21,690	7,691
24	10,610	3,762	127	55,070	19,530	54	24,100	8,546
Winter Wheat to Corn			Hay to Corn			CRP to Corn		
2	1,615	652	9	8,654	3,494	4	3,486	1,408
3	3,231	1,305	19	17,310	6,989	7	6,973	2,816
5	4,846	1,957	28	25,960	10,480	11	10,460	4,223
7	6,460	2,609	37	34,610	13,980	14	13,940	5,631
8	8,076	3,261	46	43,270	17,470	18	17,430	7,039
10	9,691	3,913	56	51,920	20,970	21	20,920	8,446
12	11,310	4,565	65	60,570	24,460	25	24,400	9,854
14	12,920	5,218	74	69,230	27,950	28	27,890	11,260
15	14,540	5,870	83	77,880	31,450	32	31,380	12,670
17	16,150	6,522	93	86,530	34,940	36	34,860	14,080

Table D-2: Increase in grain yield and subsequent ethanol production resulting from land-use scenarios substituting winter wheat, hay, and CRP land-use with grain sorghum or corn in the Kanopolis Lake Watershed.

Crop area changed (km2)	**Grain Yield (tons)**	**Ethanol Produced (L x 1000)**	**Crop area changed (km2)**	**Grain Yield (tons)**	**Ethanol Produced (L x 1000)**	**Crop area changed (km2)**	**Grain Yield (tons)**	**Ethanol Produced (L x 1000)**
Winter Wheat to Grain Sorghum			Hay to Grain Sorghum			CRP to Grain Sorghum		
82	26340	9342	10	2998	1063	39	12452	4416
165	52690	18684	19	5995	2126	79	24903	8832
247	79030	28026	29	8992	3189	118	37354	13247
330	105400	37368	39	11990	4252	158	49806	17663
412	131700	46710	48	14990	5315	197	62257	22079
494	158100	56052	58	17980	6378	237	74708	26494
577	184400	65394	68	20980	7441	276	87160	30910
659	210700	74735	77	23980	8504	315	99611	35326
742	237100	84078	87	26980	9567	355	112062	39741
824	263400	93420	96	29970	10630	394	124515	44158
Winter Wheat to Corn			Hay to Corn			CRP to Corn		
31	13340	5389	4	1404	5670	14	5712	2307
62	26690	10780	7	2807	1133	28	11423	4613
94	40030	16160	11	4209	1700	43	17135	6919
125	53370	21550	14	5611	2266	57	22846	9226
156	66710	26940	18	7014	2832	71	28558	11530
187	80060	32330	21	8417	3399	85	34268	13840
219	93400	37720	25	9819	3965	100	39980	16140
250	106700	43100	28	11220	4531	114	45691	18450
281	120100	48490	32	12630	5098	128	51403	20760
312	133400	53880	35	14030	5664	142	57114	23060

Appendix E. Statistical significant of water quality changes from land-use change scenarios

Table E-1: The p-value scores of paired t-tests performed on water quality output time series from land-use change scenarios and the baseline scenario in the Perry Lake watershed. Values in italics are statistically significant with a p-value < 0.05. A1-A10 refers to winter wheat to grain sorghum; B1-B10 refers to hay to grain sorghum; C1-C10 refers to CRP to grain sorghum; D1-D10 refers to winter wheat to corn; E1-E10 refers to hay to corn; and F1-F10 refers to CRP to corn scenarios.

Sediment											
A1	*0.0005*	B1	*0.0008*	C1	*0.0000*	D1	*0.0022*	E1	*0.0002*	F1	*0.0022*
A2	*0.0023*	B2	*0.0000*	C2	*0.0000*	D2	*0.0310*	E2	*0.0000*	F2	*0.0019*
A3	*0.0119*	B3	*0.0001*	C3	*0.0000*	D3	*0.0194*	E3	*0.0007*	F3	*0.0024*
A4	*0.0072*	B4	*0.0014*	C4	*0.0000*	D4	*0.0069*	E4	*0.0002*	F4	*0.0003*
A5	*0.0027*	B5	*0.0001*	C5	*0.0000*	D5	*0.0039*	E5	*0.0000*	F5	*0.0012*
A6	*0.0030*	B6	*0.0000*	C6	*0.0000*	D6	*0.0021*	E6	*0.0001*	F6	*0.0019*
A7	*0.0015*	B7	*0.0000*	C7	*0.0002*	D7	*0.0014*	E7	*0.0000*	F7	*0.0025*
A8	*0.0012*	B8	*0.0001*	C8	*0.0001*	D8	*0.0004*	E8	*0.0000*	F8	*0.0015*
A9	*0.0016*	B9	*0.0000*	C9	*0.0001*	D9	*0.0003*	E9	*0.0001*	F9	*0.0008*
A10	*0.0010*	B10	*0.0000*	C10	*0.0001*	D10	*0.0002*	E10	*0.0000*	F10	*0.0009*
TN											
A1	0.0649	B1	*0.0001*	C1	*0.0000*	D1	0.2859	E1	*0.0002*	F1	*0.0003*
A2	0.1113	B2	*0.0001*	C2	*0.0004*	D2	0.3564	E2	*0.0001*	F2	*0.0001*
A3	0.1116	B3	*0.0002*	C3	*0.0001*	D3	0.5322	E3	*0.0000*	F3	*0.0000*
A4	0.1016	B4	*0.0000*	C4	*0.0001*	D4	0.6714	E4	*0.0000*	F4	*0.0000*
A5	0.0814	B5	*0.0000*	C5	*0.0000*	D5	0.6792	E5	*0.0000*	F5	*0.0000*
A6	0.0791	B6	*0.0000*	C6	*0.0000*	D6	0.5317	E6	*0.0000*	F6	*0.0000*
A7	0.0890	B7	*0.0000*	C7	*0.0001*	D7	0.4505	E7	*0.0000*	F7	*0.0000*
A8	0.1407	B8	*0.0000*	C8	*0.0000*	D8	0.2183	E8	*0.0000*	F8	*0.0000*
A9	0.1222	B9	*0.0000*	C9	*0.0000*	D9	0.1345	E9	*0.0000*	F9	*0.0000*
A10	0.2389	B10	*0.0000*	C10	*0.0000*	D10	0.0620	E10	*0.0000*	F10	*0.0000*
TP											
A1	0.0684	B1	*0.0000*	C1	*0.0000*	D1	0.1232	E1	*0.0000*	F1	*0.0001*
A2	0.0663	B2	*0.0000*	C2	*0.0002*	D2	*0.0507*	E2	*0.0000*	F2	*0.0000*
A3	0.0529	B3	*0.0000*	C3	*0.0000*	D3	*0.0414*	E3	*0.0000*	F3	*0.0000*

A4	*0.0369*	B4	*0.0000*	C4	*0.0000*	D4	*0.0327*	E4	*0.0000*	F4	*0.0000*
A5	*0.0320*	B5	*0.0000*	C5	*0.0000*	D5	*0.0427*	E5	*0.0000*	F5	*0.0000*
A6	*0.0247*	B6	*0.0000*	C6	*0.0000*	D6	0.1486	E6	*0.0000*	F6	*0.0000*
A7	*0.0215*	B7	*0.0000*	C7	*0.0000*	D7	0.1087	E7	*0.0000*	F7	*0.0000*
A8	*0.0235*	B8	*0.0000*	C8	*0.0000*	D8	0.1955	E8	*0.0000*	F8	*0.0000*
A9	*0.0129*	B9	*0.0000*	C9	*0.0000*	D9	0.1345	E9	*0.0000*	F9	*0.0000*
A10	*0.0111*	B10	*0.0000*	C10	*0.0000*	D10	0.1142	E10	*0.0000*	F10	*0.0000*

Table E-2: The p-value scores of paired t-tests performed on water quality output time series from land-use change scenarios and the baseline scenario in the Kanopolis Lake watershed. Values in italics are statistically significant with a p-value < 0.05. G1-G10 refers to winter wheat to grain sorghum; H1-H10 refers to hay to grain sorghum; I1-I10 refers to CRP to grain sorghum; J1-J10 refers to winter wheat to corn; K1-K10 refers to hay to corn; and L1-L10 refers to CRP to corn scenarios.

Sediment											
G1	0.4917	H1	0.3073	I1	0.8357	J1	0.3297	K1	0.4317	L1	0.9326
G2	0.8397	H2	0.3946	I2	0.6666	J2	0.9709	K2	0.2923	L2	0.6534
G3	0.2304	H3	0.3356	I3	0.5626	J3	0.1664	K3	0.5038	L3	0.8173
G4	0.1109	H4	0.3960	I4	0.5808	J4	0.5630	K4	0.1426	L4	0.4071
G5	*0.0004*	H5	0.1569	I5	0.5303	J5	0.7421	K5	0.6899	L5	0.6625
G6	*0.0038*	H6	0.1582	I6	0.9912	J6	0.1729	K6	*0.0335*	L6	0.8078
G7	*0.0075*	H7	*0.0266*	I7	0.8017	J7	*0.0068*	K7	0.1133	L7	0.9605
G8	*0.0003*	H8	0.2954	I8	0.1803	J8	*0.0146*	K8	0.7140	L8	0.7813
G9	0.1985	H9	0.1393	I9	0.2062	J9	*0.0042*	K9	0.5673	L9	0.7173
G10	0.1476	H10	0.0624	I10	0.1764	J10	*0.0343*	K10	0.5377	L10	0.8554
TN											
G1	0.6620	H1	0.2287	I1	*0.0469*	J1	*0.0436*	K1	*0.0023*	L1	*0.0120*
G2	0.4307	H2	0.0861	I2	0.0573	J2	0.3765	K2	*0.0002*	L2	*0.0114*
G3	0.5374	H3	0.0731	I3	0.0532	J3	0.1323	K3	*0.0018*	L3	*0.0095*
G4	0.4735	H4	*0.0490*	I4	0.0999	J4	0.0910	K4	*0.0039*	L4	*0.0073*
G5	0.6909	H5	0.0634	I5	0.1143	J5	0.1563	K5	*0.0024*	L5	*0.0194*
G6	0.5187	H6	*0.0480*	I6	0.0941	J6	0.2011	K6	*0.0013*	L6	*0.0079*
G7	0.3508	H7	0.0661	I7	0.0922	J7	0.1395	K7	*0.0104*	L7	*0.0076*
G8	0.4682	H8	0.0709	I8	0.0879	J8	0.1915	K8	*0.0075*	L8	*0.0081*
G9	0.4269	H9	0.0582	I9	0.0839	J9	0.1701	K9	*0.0077*	L9	*0.0078*

G10	0.4438	H10	0.0522	I10	0.0815	J10	0.2019	K10	*0.0066*	L10	*0.0085*
TP											
G1	0.5102	H1	*0.0138*	I1	0.0565	J1	0.4386	K1	*0.0009*	L1	*0.0066*
G2	0.5762	H2	*0.0122*	I2	0.0564	J2	0.3567	K2	*0.0011*	L2	*0.0062*
G3	0.6378	H3	*0.0116*	I3	0.0553	J3	0.3193	K3	*0.0012*	L3	*0.0064*
G4	0.6739	H4	*0.0121*	I4	0.0555	J4	0.2718	K4	*0.0012*	L4	*0.0065*
G5	0.6971	H5	*0.0126*	I5	0.0552	J5	0.2251	K5	*0.0012*	L5	*0.0065*
G6	0.7362	H6	*0.0126*	I6	0.0550	J6	0.1948	K6	*0.0012*	L6	*0.0066*
G7	0.7700	H7	*0.0124*	I7	0.0547	J7	0.1705	K7	*0.0013*	L7	*0.0064*
G8	0.7977	H8	*0.0120*	I8	0.0546	J8	0.1493	K8	*0.0012*	L8	*0.0063*
G9	0.8372	H9	*0.0122*	I9	0.0544	J9	0.1307	K9	*0.0012*	L9	*0.0062*
G10	0.8606	H10	*0.0123*	I10	0.0539	J10	0.1203	K10	*0.0013*	L10	*0.0061*