RUNOFF AND SEDIMENT RESPONSE TO MECHANICAL REMOVAL OF JUNIPER AND RESTORATION OF RANGELAND

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Abstract: This dissertation examines runoff and sediment responses to restoring woody plant encroached rangeland in the south-central Great Plains using the paired watershed approach, before-after control-impact (BACI) analysis, and soil and water assessment tool (SWAT) to answer three different but closely related research questions:

i) Will the soil moisture dynamics and runoff patterns recover after mechanical removal of juniper (*Juniperus virginiana* L., redcedar) and restoration to native prairie or establishment of switchgrass (*Panicum virgatum* L.)? Analysis based on a paired watershed approach showed that the root zone soil water storage increased by 1.6 and 1.9 times for restored prairie and switchgrass, respectively, after juniper removal. The restored prairie and switchgrass production system increased annual runoff by 4.46 and 4.54 times, respectively.

ii) What level of soil erosion will be caused by the mechanical removal of juniper, and how will the soil erosion of restored grasslands compare with the juniper woodland? The BACI analysis concluded that mechanical removal itself resulted in a limited but significant increase of sediment yield in the year after removal. However, a pulsed increase of sediment load occurred with the herbicide application to plant switchgrass in the following year. After grasslands were established, the sediment load was very low, but the switchgrass biomass production system had a lower average sediment concentration and sediment yield than the restored prairie.

iii) How will juniper removal followed by switchgrass planting affect runoff and sediment yield at a regional scale? The scenario simulation using SWAT model showed that the conversion of juniper to switchgrass had limited impacts on annual water budget and sediment yield at the basin level of the Lower Cimarron River (LCR) basin; however, this transformation had local effects on ET and streamflow in subbasin with high juniper cover (14% of LCR). Converting juniper and marginal rangelands to switchgrass resulted in 1.3 – 3.5% increase of annual ET and 5.4 – 13.5% decrease in annual streamflow, with 12.2 – 61.6% reduction in sediment yield. The increase of ET and reduction of streamflow were greatest during the switchgrass growing season.

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CHAPTER I

GENERAL INTRODUCTION

Globally, securing sufficient freshwater resources for sustainable development has increasingly become a more challenging issue since the late 20th century (Shiklomanov, 2000). Freshwater is an essential natural resource and is dynamic, renewable, and re-distributable through the water cycle. However, water resources are threatened by land use change and changing climate, such as increasing temperature and greater precipitation variability (Seager *et al.*, 2018). These impacts are imminent in the south-central Great Plains, where global warming is forecast to lead to more extreme intra-annual precipitation regimes characterized by larger rainfall events and longer intervals between events (Easterling *et al.*, 2000; IPCC, 2007). Therefore, adaptive land management is critical to ensure the environmental flow and adequate water supply to municipal water use in this region. Rangeland is a major land cover type in the southern Great Plains (Sutton, 1984; Bagley *et al.*, 2017). An increasing trend of rangeland acreage was documented since the 1970s (Dale *et al.*, 2015; Taylor *et al.*, 2015). Runoff and baseflow generated from rangeland are important water sources for ranchers and municipal water supplies and critical for stream and aquatic ecosystems. In addition to the high erosivity of the precipitation regime, this region is also characterized by highly erodible soils and the site disturbance by any land management practices will have the potential to significantly increase sediment loads in surface runoff and impair the beneficial uses of water (Fox and Wilson, 2010). Site specific measurements of runoff and sediment load and scenario simulation at the regional scale are critical before the implementation of land use change.

The quantity and quality of runoff from rangeland are highly variable and highly responsive to change in land surface conditions and are generally detrimentally affected by rangeland degradation (Munoth and Goyal, 2020). Rangeland degradation can be generically defined as a decrease in plant species diversity, vegetation cover, and plant productivity (Fenetahun *et al*., 2018). During the past decades, rangelands in the south-central Great Plains have been encroached heavily by a juniper species (*Juniperus virginiana* L., eastern redcedar, redcedar) (Wang *et al.*, 2017). When the juniper reaches full canopy cover, this physiognomic conversion has the potential to significantly reduce soil water and surface runoff (Liu *et al*., 2018; Zou *et al*., 2014). Switchgrass (*Panicum virgatum* L.) is a native species in tallgrass prairie ecosystems and is widely used to prevent soil erosion in restoration projects (Wu and Liu, 2012; Feng *et al.*, 2015). It is also selected as a dedicated feedstock for biofuel production (Wullschleger *et al.*, 2010) and the south-central Great Plains is a potential focal area for switchgrass biomass production because this region has adequate precipitation and sufficiently productive soils and has moderate to high suitability for switchgrass production (Wright and Thurhollow 2010). Most importantly, a large portion of these lands is under marginal use, highly susceptible to soil erosion, or encroached by woody vegetation.

Cumulative effects of land use and cover change and variable climate will profoundly impact the hydrological functions in the south-central Great Plains, such as soil water and surface runoff. Observations showed that herbaceous vegetation usually establishes fairly quickly after juniper is removed in the mixed and tallgrass prairie in the southern Great Plains. However, the soil moisture dynamic and runoff patterns after juniper removal and grassland establishment have not been quantitatively assessed.

An increase of juniper trees (*Juniperus osteosperma* [Torr.] Little) in the rangelands of intermountain west of the USA led to the rise in overland flow and sediment transport between canopies because bare soil area increases and become more interconnected (Pierson *et al*., 2010). However, further east to the grassland and forest transition zone, there are not enough studies to show the sediment yield response to the encroachment. Juniper encroachment increased the amount of bare soil and may have the potential to increase soil erosion (Thurow and Carlson, 1994; Thurow and Hester, 1997). Due to great rainfall variability and erosive soil type, the ambient rate of water erosion in the rangelands tends to be high in this transition zone. As a result, high turbidity in streams and reservoirs is currently a major water quality issue in the south-central Great Plains. Undoubtedly, mechanical removal of juniper and potential site preparation will incur substantial soil surface disturbance and expose soil to direct raindrop erosion. However, runoff and sediment response to mechanical removal of juniper and the effectiveness of the switchgrass feedstock production system compared with a natural restored prairie following mechanical removal of juniper has not been systematically assessed.

One of the critical challenges in watershed study and watershed management related to water budget and water quality is understanding the paradox of scale (Wilcox *et al*., 2006). The runoff and sediment responses with complete juniper removal may produce very significant runoff and sediment responses at the experimental watershed scale, but the diverse land use and patchy encroachment coverage make it difficult to extrapolate the experimental watershed scale result to the regional scale. Historically, most lands in the southern Great Plains were used for rangeland (Liuzzo *et al.*, 2009; Zou *et al.*, 2016). Even presently, rangeland still occupies a large proportion of land surface in this region (Dale *et al*. 2015), but its distribution has become fragmented, and many rangelands are under different levels of degradation. Physical-process-based, spatially explicit models can be used to assess the impact of proactive management practices, such as converting marginal rangeland to switchgrass biomass production system, on hydrological function and soil loss at a regional scale (Chen *et al*., 2015). Such models include Regional Hydro-Ecological Simulation System (RHESSys) (Tague *et al.*, 2004), Soil and Water Integrated Model (SWIM) (Krysanova *et al.*, 2005), Soil and Water Assessment Tool (SWAT) (Arnold *et al.*, 1998), and Variable Infiltration Capacity (VIC) (Christensen *et al.*, 2004). Among these models or model frameworks, SWAT has been successfully used to simulate the impact of different wooded systems (Bingner, 1996) and conversion of all rangeland to juniper woodland in the lower Cimarron River Basin (Qiao *et al.*, 2015) on water budget and sediment load. Policy and government programs generally support land management to curtail the expansion of juniper in rangeland and ranchers also prefer to remove encroached juniper to recover herbaceous productivity. However, it remains unquantified whether juniper removal may or may not have a measurable impact on the water budget at the basin scale.

This dissertation research aims to achieve the following three specific research objectives, with each objective addressing one of the three questions mentioned above: i). To understand whether and when the hydrological function associated with the prairie will recover after the encroached junipers are removed, ii). To quantify the level of soil erosion associated with the mechanical removal of juniper and the restored grasslands, and iii). To assess the impacts of converting marginal rangeland to switchgrass biomass production system on runoff and sediment yield at a regional scale. The dissertation has been structured to address the three objectives sequentially, with each objective being a standalone research paper. The second chapter of this dissertation assesses and contrasts the watershed hydrological function pre and post juniper removal. The third chapter of this dissertation quantifies sediment load associated with mechanical removal of juniper and establishment of restored prairie and of switchgrass biomass production system. The fourth chapter of this dissertation evaluates the water resource impact of juniper removal and restoration in rangelands using switchgrass at a regional scale by the SWAT model.

CHAPTER II

CONVERSION OF ENCROACHED JUNIPER WOODLAND BACK TO NATIVE PRAIRIE AND TO SWITCHGRASS INCREASES ROOT ZONE SOIL MOISTURE AND WATERSHED RUNOFF

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ABSTRACT

Mechanical removal of encroached juniper (*Juniperus* spp.) is a common practice to restore native grasslands. However, the hydrological responses to grassland restoration remain mostly unquantified for the climate transition zone in the southern Great Plains of the USA where ecosystem evapotranspiration is highly sensitive to the change of vegetation functional type. We used a paired watershed approach to directly quantify the impact of mechanical removal of eastern redcedar (*Juniperus virginiana* L., redcedar) and restoration to native prairie or establishment of switchgrass (*Panicum virgatum* L.) on root zone soil moisture and event-based runoff for eight years including three main phases - calibration, transition, and restored grassland in north-central Oklahoma, USA. Results showed that the root zone soil water storage on average increased 1.6 and 1.9

times for restored prairie and switchgrass, respectively, after juniper removal. The regression model estimation based on the relationships established in the calibration phases between event-based runoff from control watershed and the treatment watersheds found that the restored prairie and switchgrass production system increased annual runoff by 4.46 and 4.54 times, respectively. These results indicated that both soil moisture and runoff are highly responsive to land use change in the southern Great Plains. Reestablishment of herbaceous dominance by mechanically removing encroached woody species is closely followed by restoration of soil moisture dynamics and watershed runoff regime.

INTRODUCTION

Change in land surface condition and increased climate variability impose a great challenge on sustainable management of watersheds and water resources in the climate transition zone of the southern and central Great Plains, USA (Ojima *et al*., 1999; Sun *et al*., 2008). An important land surface change in this region is associated with the transition from herbaceous plant dominance to woody plant dominance characterized by a juniper species (*Juniperus virginiana* L., eastern redcedar or redcedar) encroachment in the prairie and oak savannas of the central Great Plains (Briggs *et al*., 2005; Hoff *et al*., 2018). This transition is associated mainly with a disruption of the historical fire regime and causes a decline or loss of prairie productivity (Norris *et al*., 2001; Ganguli *et al*., 2016), plant biodiversity (van Els *et al*., 2010), and avian and mammalian biodiversity (Horncastle *et al*., 2004). Importantly, the soil moisture dynamics are substantially altered and water yield is reduced (Zou *et al*., 2014; Qiao *et al*., 2017; Acharya *et al*., 2018). Mechanical removal of juniper trees is an effective approach to quickly reclaim encroached land parcels (Morton *et al*., 2010). However, site disturbance associated with mechanical operations could cause excessive surface runoff and potential soil erosion if not appropriately managed. Also, after juniper removal, the watershed will revegetate and can restore to prairie naturally or be planted with species such as switchgrass (*Panicum virgatum* L.) for biomass production. However, it is unknown how the root zone soil moisture condition and runoff regime will differ between pre- and post-treatment conditions, and between these two potential land use options that both restore herbaceous plant dominance.

Due to reduced evapotranspiration, reducing woody canopy can increase streamflow in the semiarid and subhumid regions in the USA (Zou *et al*., 2010; Madsen *et al*., 2011; Madsen *et al*., 2012; Roundy *et al*., 2014). However, there is a limitation regarding the vegetation-induced change in evapotranspiration (ET). The magnitude of vegetation-induced change is likely the greatest where precipitation approximately equals potential evapotranspiration (PET) (Zhang *et al*., 2001; Huxman *et al*., 2005). In wetter areas where precipitation exceeds PET, vegetation change may have less effect because excess precipitation causes runoff while in drier areas, a large deficit between precipitation and PET causes runoff events to be rare (Huxman *et al*., 2005). In low precipitation regions, several recent studies showed that woody canopy removal did not result in a measurable increase in streamflow (Guardiola-Claramonte *et al*., 2011; Biederman *et al*., 2015; Williams *et al*., 2018) and that streamflow did not increase in most basins following the loss of tree canopy due to tree die-off (Biederman *et al*., 2015). The current consensus is that the effect of tree or brush removal on the streamflow is minimal where annual precipitation is less than 500 mm (Archer and Predick, 2014). In contrast, deforestation in high precipitation regions has been a significant contributor of soil erosion, and large scale deforestation in the cloud forest and tropical rainforest may even reduce runoff by suppressing the positive feedback between soil moisture and precipitation (Junkermann *et al*., 2009; Lawrence and Vandecar, 2015).

Consequently, vegetation change should have maximal hydrological consequences in the forest and grassland transition zone such as the southern and central Great Plains of the USA encompassing a precipitation gradient from 500 mm to 1000 mm (Figure 1A). As the climate in the transition zone is becoming more variable and the demand for water continues to increase, control of invasive and encroaching woody species to increase streamflow is an important consideration. However, the effectiveness of woody vegetation removal within the forest and grassland transition zone (e.g., the southern and central Great Plains) remains unknown. For this region, increases in the runoff after woody vegetation removal have been generally estimated through hydrological modeling (Afinowicz *et al*., 2005), with few empirical data to verify modeling conclusions.

The impact and response of experiments measuring runoff after removing trees could be subject to three types of uncertainties. Failure to detect the change in runoff after tree removal could result from the mismatch in scale between the treatment area and the total area contributing to streamflow (Wilcox *et al*., 2006; Biederman *et al*., 2015; McDonald *et al*., 2015). Also, how the woody vegetation regrowth or regeneration is managed after harvesting could affect the runoff response (Biederman *et al*., 2015). If woody plants re-sprout and reoccupy the same area, the positive effects of tree removal on runoff could be transitory. Finally, the trends and variability of climate, particularly precipitation prior to and post-treatment, could confound the results. However, a paired watershed approach overcomes these shortfalls by building the potential difference of watershed behavior and climate variability during the calibration and treatment period into the analytical solution (Brown *et al*., 2005). At the experimental watershed scale, the land use can be experimentally controlled and its impact directly quantified, providing opportunities to disentangle the impact immediately after disturbance and the altered land use (Bosch and Hewlett, 1982; Clausen *et al*., 1996; Ali *et al*., 2017). Paired watershed designs have been widely used to determine the impacts of forest harvesting on the water yield (Bosch and Hewlett, 1982; Brown *et al*., 2005; Ochoa-Tocachi *et al*., 2018). However, their application for assessing the impact of woody encroachment and removal on hydrological processes is limited because multi-year, pre-treatment runoff data to establish the calibration regression line are rarely available.

Streamflow is usually a small component in the water budget and is highly variable in a water-limited system. However, overland flow and streamflow regimes are closely associated with root zone soil moisture of which the temporal dynamic is greatly influenced by the plant functional type (Huxman *et al*., 2005; Liu *et al*., 2017). Improvement in root zone soil moisture is indicative of the increased runoff and recharge potential at the local scale (Dekker and Ritsema, 1994; Doerr *et al*., 2000; Acharya *et al*., 2018). An increase in soil water storage and recharge was documented after removing western juniper (*Juniperus occidentalis*) in Idaho (Seyfried and Wilcox, 2006) and a 20% greater over-winter soil water accumulation after western juniper removal was reported in Oregon (Mollnau *et al*., 2014). Ultimately, the increase of root zone soil moisture may profoundly affect the responsiveness of the land surface to precipitation and increase the surface runoff coefficient (Sriwongsitanon and Taesombat, 2011; Qiao *et al*., 2017).

For prairie lands that have been encroached by woody vegetation in the Great Plains, the potential of recovery of streamflow depends on the re-vegetation and land use after woody vegetation removal. A common management approach is to leave the land to restore to native prairie for cattle grazing, although it is subject to encroachment by woody plants again without subsequent management. Alternatively, the land can be actively managed for hay and other biomass production. Switchgrass, an herbaceous species, has been promoted as a potential biofuel feedstock and is highly adapted to the southern United States. Switchgrass can be established by seeding with no-till drilling techniques along with herbicide application under non-drought conditions (Casler *et al*., 2015). These two conditions, i.e., native prairie and switchgrass production system, may differ in runoff generation as they vary in factors such as plant composition or species diversity, leaf area, rooting depth, and litter cover.

The overall goal of this study was to assess the hydrological response to mechanical removal of juniper followed by restoration to prairie or planted to switchgrass in the south-central Great Plains using a paired watershed approach. The specific objectives included: i) Assess the state change in root zone soil water storage during the land use transition and restored grassland phase (restored native prairie and established switchgrass); ii) Quantify runoff responses during the transition phase and the restored grassland phase; iii) Compare runoff between the juniper woodland converted to restored prairie and juniper converted to switchgrass production system.

MATERIALS AND METHODS

Study area and land use history

The study was conducted in the Cross Timbers Research Range (CTER), which is managed by Oklahoma State University (Figure 2.1). The study site is located 15 km southwest of Stillwater, Payne County, Oklahoma, USA (36°3’46.73” N, 97°11’3.33” W). The soils are well drained, consisting predominately of the Stephenville–Darnell complex, Coyle soil series, and Grainola–Lucien complex. The average depth of soil is approximately 1 m underlain by sandstone substrates. The site was previously cultivated and allowed to recover to prairie after agricultural abandonment in the 1950s. The land is currently being managed for research purposes related to prescribed fire, grazing, and watershed hydrology. Fire and grazing interactions through patch-burning have been studied in parts of the research range since 1983 (Fuhlendorf and Engle 2004). Native tallgrass prairie has been maintained due to burning treatments whereas juniper was present in some areas before the prescribed fire and has heavily encroached many other areas of CTER where burning has been excluded. The site also contains areas of post oak (*Quercus stellata*) dominated woodland as well as prairie lightly encroached by woody vegetation. In this study, we chose to use three previously established juniper-encroached experimental watersheds which have been monitored since 2010 (Qiao *et al*., 2015). These watersheds range from 1.34 to 3.79 ha (Table 2.1; Figure 2.1).

Data Collection

Meteorological observations

Meteorological observations were continuous since 2011. A weather station was located approximately 5 m east of the H-flume of watershed J-SG (Figure 2.1). Precipitation was measured with a TB3 siphoning tipping bucket rain gauge with 0.254 mm per tip (Hydrological Services America, Lake Worth, FL, USA). Air temperature and relative humidity were measured at 2 m using a Vaisala Temperature/RH Probe (HMP50, Campbell Scientific Inc, Logan, UT, USA) with an RM Young 6-plate radiation shield. Wind speed and direction (RM Young Company Wind Sentry Set, Traverse City, MI, USA) and solar radiation (Apogee SP-110, Apogee Instruments, Inc., Logan, UT, USA) were measured at 3 m height.

Root zone soil water

Root zone soil water for each watershed was measured using three soil moisture arrays established in 2010 and distributed at the upper, middle, and lower position of each watershed (Figure 2.1). Each array was composed of one EM50 datalogger and four ECH_2O EC-5 capacitance probes (Meter Group Inc., Pullman, WA, USA) installed at depths of 50, 200, 450, and 800 mm to capture the mean condition of soil volumetric water content (θ) every 15 minutes for the soil layers of 0–100, 100–300, 300–600, and 600–1000 mm, respectively. ECH_2O probes measured the dielectric constant of the soil, which was converted to volumetric water content using the manufacturer's calibration relationship. The EC-5 sensors run at a high measurement frequency which allows a single calibration equation to be used for mineral soils of various textures and electrical conductivities. The accumulated root zone soil water storage (SWS, mm) in the upper 1 m soil profile was calculated based on soil water in each soil layer (Zou *et al*., 2014). The average of SWS from the three arrays was used to represent the average root zone soil water storage for the watershed.

Runoff

Runoff from each watershed was gauged using a 0.9 m H-flumes. The stage level in the flume was measured at 5-minute intervals using an optical shaft encoder with a minimum stage reading and resolution of 3.0 mm (50386SE-105, HydroLynx, West Sacramento, CA, USA). Stage level readings were converted to discharge values using the known stage-discharge relationship for the given H-flume and discharge was converted to runoff depth using the watershed areas. Annual runoff values, treatment period values, and event-based values were generated by summing the 5-minute data for the period of interest. Two conditions were used to determine the end of runoff events. Most commonly, event end was considered as the last flow record with no further flow reading for a minimum of 24 hours in any of the watersheds. For rare cases where the flow did not completely cease before the next precipitation event, the end time of runoff was set as 5 minutes before the start of the next precipitation event to separate runoff events.

Treatment design

Paired watershed studies have been widely used to determine the impact of changes in land use or vegetation change on water yield (Bosch and Hewlett, 1982; Hibbert, 1983; Brown *et al*., 2005; Neary, 2016; Ochoa-Tocachi *et al*., 2018). This approach involves the use of two nearby watersheds similar in size, topography, soils, and land use. The quantitative relationship of runoff from these two similar, paired watersheds established when the land use or land cover was relatively stable or unchanged for multiple years, is considered the calibration phase. One of the watersheds is subsequently subjected to a treatment (i.e., land use change or land cover change) and the other remains as a control. This allows the yearly climatic variability to be accounted for in the analysis such that change in the quantitative runoff relationship between the paired watersheds is attributed to changes in land use or vegetation type (Brown *et al*., 2005).

During July 2015, all juniper trees in two adjacent watersheds south of the control watershed were cut using a skid steer equipped with a Marshall tree saw (Tulsa, OK, USA) (Table 2.2). Cut trees were left to dry on site until January 2016 when they were ground and hauled off site. Grinding and removal were completed by the end of February 2016. One treatment watershed was left to revegetate without further treatment and assigned “juniper to restored prairie” watershed and abbreviated as J-RP (Figure 2.1). The other cut watershed was further treated with glyphosate herbicide beginning in May

2016 to kill vegetation until May 2017 when an upland switchgrass variety, Alamo, was seeded using a large drilling machine. This watershed was assigned as "juniper to switchgrass" watershed and abbreviated as J-SG (Figure 2.1). The treated watersheds were fenced to prevent cattle grazing and trampling. The watershed north of the treatment watersheds was not treated and was used as the control (juniper) watershed and abbreviated as J (Figure 2.1).

Collectively, there were two control vs treatment pairs (J vs. J-RP and J vs. J-SG) and the effects of the treatments were compared using the paired-watershed approach. In the pre-treatment period, J had the highest juniper basal area, while J-SG had the highest density (Table 2.1). The calibration phase was four years and nine months from Oct 2010 through June 2015 with no management activities on any of the three watersheds (Table 2.2). To address our objectives, the treatment period was further separated into a transition phase and a restored grassland phase. The transition phase was defined as the period after juniper cutting and before the new vegetation cover was fully established. The restored grassland phase includes restored prairie and established switchgrass. The first year after cutting of the J-RP (May 2016 to April 2017) was assigned to the transition phase because of little vegetation cover and biomass growth in the first year of recovery as compared to the second year when more herbaceous vegetation occurred. Likewise, the first year after juniper cutting for the J-SG watershed was considered transition as switchgrass was not planted until the following spring.

Data analysis and statistics

Daily reference evapotranspiration (ET_0) was calculated based on local, daily weather station data using the FAO Penman-Monteith method (Cai *et al*., 2007). A generalized additive model (GAM) was used as a scatterplot smoothing function for climate variables, which gave a general idea of the trend for weather variables. The Mann-Kendall test was used to quantify the trend of climate variation during the study period (Mann, 1945; Mondal *et al*., 2012). In the Mann-Kendall test, τ represented the measure of the strength of the trend of climate variation in a time series, with a range from -1 to 1. This value represented the strength and direction of association that exists between climate and time series. The larger absolute value means larger strength between time and climate. A negative value stands for a decreasing trend, while a positive value stands for an increasing trend. We used $\alpha = 0.05$ in this research. We used the time series of SWS of treatment watersheds (J-RP and J-SG) minus that of the control watershed J to calculate the time series of ΔSWS. Then change point detection (Cho and Fryzlewicz, 2015) was applied to check whether there was an increase or decrease in the time series of ΔSWS following treatments. Both the Mann-Kendall test and change point detection were performed in RStudio.

We determined the treatment impacts by testing whether the linear relationship of event-based runoff depths between control watershed and the treatment watershed significantly changed in the transition phase and the restored grassland phase (we tested the significance of changes in slope and intercept between the appropriate sets of regression lines in different phases). Runoff in our experimental watersheds was predominantly from stormflows. Some small rainfall events produced runoff for one watershed but not for the others. For this analysis, we dropped runoff events when there was zero runoff in any of the paired watersheds (J vs. J-RP; J vs. J-SG) which allowed the square root transformation of the event-based runoff data to satisfy the linear regression analysis assumptions: the error terms are normally distributed and the paired data have equal variance. Impact of mechanical removal of juniper on runoff was calculated as the ratio of observed annual runoff to estimated annual runoff in the treatment phase (Clausen *et al.*, 1996). The estimated runoff for the manipulated watersheds was modeled based on the relationship of square root event-based runoff established during the calibration phase assuming the juniper trees were not removed.

RESULTS

Climatic dynamics

The mean annual precipitation was 832 ± 143 (mean ± standard deviation, SD) mm for the eight water years from October 2010 to September 2018. The highest daily precipitation of 112 mm occurred in May 2017 (Figure 2.2). The precipitation during the study trended up ($p = 0.027$) with a positive but small τ value. There was no significant trend in the daily temperature ($p = 0.18$). The highest daily ET_0 of 11 mm occurred in May 2014 and the mean daily ET_0 was 3.46 ± 2.16 mm for the study period. The daily ET_0 trended down ($p<0.001$) with a negative small τ value.

Root zone soil water storage

The SWS within the 1000 mm soil profile in the control watershed (J) was generally less than that in J-RP and J-SG for the entire study period. During the calibration phase, mean daily SWS in J was 143 ± 45 mm (mean ± SD), 267 ± 55 mm in J-RP and 216 ± 52 mm in J-SG (Figure 2.3a1 and 2.3b1). In the transition and restored grassland phases, mean daily SWS was 163 ± 42 mm in J, 356 ± 53 mm in J-RP, and 303 ± 37 mm in J-SG

(Figure 2.3a1 and 2.3b1). The difference in SWS (ΔSWS) increased for both comparisons (Figure 2.3a2 and Figure 2.3b2). For the J – J-RP pair, a change point in the ΔSWS was detected on August 22, 2015. The ΔSWS averaged 125 ± 21 mm before August 22, 2015 and increased to an average value of 195 ± 40 mm after that date (Figure 2.3a2). For the J – J-SG pair, a change point in the ΔSWS was detected right after cutting of juniper. It averaged 74 ± 28 mm before July 6, 2015 and increased to an average value of 141 ± 36 mm after that date (Figure 2.3b2). There was no change point detected for ΔSWS between the transition phases and the restored grassland phases.

Annual runoff and hydrograph

Annual runoff depth varied greatly among water years and the type of land use, ranging from negligible (<1 mm) runoff in 2011 to over 150 mm in 2017. Annual runoff depth in control (J) was 21 ± 12 (mean ± SD) mm, 9 ± 5 mm in J-RP and 16 ± 10 in J-SG during the calibration phase (2011-2015) (Figure 2.4). After juniper cutting in July 2015, the runoff of the treated watersheds increased relative to the control and remained elevated throughout the transition and grassland recovery phases. In 2016 - 2018, the average annual runoff depth in J was 27 ± 22 mm, in J-RP was 94 ± 49 mm and in J-SG was 131 ± 73 mm.

For all time periods except the cut and drying phase for the J-SG watershed, the slopes of the relationship between the treated and control watersheds for event-based runoff were significant (Table 2.3, Fig 2.5). During the calibration phase, the regressions comparing J to J-RP and J to J-SG had strong, linear relationships with high coefficients of determination (R^2) (Table 2.3, Figure 2.5). For the comparisons between the J-RP and J watersheds, the slopes during the cut and drying, recovery to the prairie, and restored prairie phases were all significantly greater than during the calibration phase, indicating an increase in runoff depth with treatment (Table 2.4, Fig 2.5a, c, e) The greatest slope occurred during the recovery to prairie phase (Table 2.3). Intercepts were significantly greater for regressions during the cut and drying and during the restored prairie phases than during the calibration phase (Table 2.4). For the J-SG vs. J comparison, the slopes were greater during the herbicide and during the established switchgrass phases than during the calibration phase (Table 2.4, Fig 2.5d, f). The greatest slope occurred during the herbicide phase when vegetation was absent (Table 2.3). Intercepts were significantly greater for regressions during the herbicide and the established switchgrass phases than during the calibration phase (Table 2.4). During the established grassland phases (May 2017 through September 2018), the total runoff depth based on the regression model increased by 4.46 fold for the restored prairie and 4.54 fold for the switchgrass production system compared to the noncut control watershed.

During the calibration phase, the hydrographs in all three watersheds resembled each other in terms of peak flow and runoff duration. Following a 62 mm storm, for example, event peak flow in J was the highest, followed by J-SG and J-RP (Figure 2.6a). The flow duration was comparable and ranged from 35 to 45 hours. In the restored grassland phases, the hydrographs of J-PR and J-SG diverged from that of J. For example, a storm event of 27 mm had a peak flow in J that was the lowest and lasted for only 2 hours while the peak flows and flow durations of J-RP and J-SG were much higher with much longer, delayed flow (Figure 2.6b).

DISCUSSION

Increase in soil moisture and runoff under transition phase

Cutting juniper reduces the tree transpiration component of the water budget. Although an increase in soil moisture after shrub and tree removal is often attributed to reduced canopy interception (Taucer *et al*., 2008), the increase in soil moisture as observed in this study was likely primarily due to reduced transpiration since the cut trees were initially left on site and could continue to intercept rainfall. Caterina *et al*. (2014) found that juniper can transpire almost all throughfall during a year of normal precipitation. Initially, after cutting, there was little herbaceous component as it had been previously suppressed by the tree canopies. In addition, prior to removal, the cut juniper continued to shade the soil and suppress herbaceous plant development.

After February 2016, the removal of the dead juniper from the cut watersheds allowed existing grasses and herbaceous plants to expand and new plants to establish throughout the watershed which increased transpiration. From May 2016, the J-RP watershed started to recover to native prairie, while herbicide was applied to the J-SG watershed. For this period, the different dynamics of soil moisture for J-RP and J-SG reflected the transpiration induced soil moisture response. Suppression of transpiration by the herbicide in J-SG resulted in a high and less variable root zone soil moisture storage while J-RP experienced minimum root zone soil moisture in the late growing season corresponding to maximum leaf area of recovering vegetation.

Overland flow is an important component of the annual runoff in this study region (Qiao *et al*., 2017). Therefore, the antecedent soil water before and during the rainy season influences the total runoff depth for a year. In the transition phases, an apparent trend was that the root zone soil water storage more frequently approached the peak storage value, leading to potentially higher overland flow.

The application of herbicide to kill herbaceous plants in preparation for planting switchgrass is a common practice. Eliminating all vegetation by herbicide at hillslope and watershed scales always raises a great concern regarding soil erosion, especially if an application is followed by large rainfall events. In our study, the largest runoff depth for a single event for J-SG during the herbicide period was 28.0 mm, which was three times greater than 8.9 mm in J-RP and 85 times higher than 0.3 mm in J for the same precipitation event. It is important to take the potential rainfall condition into consideration when determining the timing of herbicide application for switchgrass planting.

Increase in root zone soil moisture and runoff under restored grassland phase

In our study, the state change in root zone soil moisture storage was detected only for juniper removal but no further change was detected due to prairie restoration or switchgrass establishment. Based on the statistical result, the intercept of the linear regressions increased significantly following grassland restoration compared to the calibration phase, which suggests the juniper woodland has a much higher precipitation threshold needed to produce runoff and that conversion from juniper to herbaceous vegetation decreases the precipitation threshold for runoff generation. The increase in the slope of the regression line during the transition and restored grassland phases means a greater increase in runoff depth per unit increase in precipitation, suggesting a quick and steady rise of runoff in restored grasslands relative to the pre-treatment phase. During the established grassland phase, the total runoff depth for the restored prairie and switchgrass production system increased by 4.46 and 4.54 fold, respectively. In the switchgrass production system, the majority of aboveground biomass was removed following the first frost in 2017 and 2018, leaving a site with less canopy cover during the dormant season. Even for the restored prairie, change in species composition, and management approaches will likely alter the runoff regime in the future.

Juniper removal and water recovery potential

Juniper has encroached into millions of hectares in Oklahoma, Kansas, and Nebraska (McKinley 2017) and it is getting worse. However, most encroached areas have very low canopy cover and the areas with high juniper density are still relatively small (Wang *et al.*, 2017). Given its current status, large scale or regional campaigns of juniper removal will not be effective solely for water recovery. Instead, our results show that treating heavily encroached parcels can result in a localized increase in runoff, which could be meaningful for increasing the water for retention ponds and upland reservoirs in the southern and central Great Plains. Those ponds and upland retention structures collect flow from the overland flow and intermittent streams. In most cases, juniper removal to regain space and restore soil moisture condition for enhancing grass production may be a more direct motivation for ranchers. Restoration of the runoff regime to sustain streams and refill reservoirs are added benefits.

A large swath of the Great Plains is climatically adapted for the cultivation of switchgrass. Although the planting and harvesting requirements prevent switchgrass application in many rangeland sites with rocky outcrops, our results show that switchgrass production provides an alternative land use option which could provide a hydrological function similar to restored native prairie.

CONCLUSION

Our study concludes that the paired watershed approach is an effective method to detect the impacts of vegetation manipulation on hydrological processes such as the change in rooting zone soil moisture storage and runoff dynamics to address field-scale issues on relatively flat rangeland watersheds. However, this method requires long term monitoring on the runoff and soil moisture data. Mechanical removal of juniper trees increases watershed soil moisture storage and decreases the threshold of precipitation to generate surface runoff. Converting juniper woodland to prairie resulted in a four to five fold increase in total runoff at the experimental watershed scale in the southcentral Great Plains. The hydrograph analyses suggest more extensive and delayed flow in restored prairie than that of established switchgrass. These observations and the conclusions are based on well defined, small upland watersheds in the tallgrass prairie climate transition zone in the southern Great Plains of the United States. Further work should explore whether such results apply to the mixed prairie where woody encroachment is rapidly transitioning the landscape to a woody state.

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Table 2.1 Watershed characteristics of the intact control watershed (J), the cut watershed allowed to restore to native prairie (J-RP), and the cut watershed converted to switchgrass (J-SG). The J watershed was measured in January 2016 and the J-RP and J-SG were measured in January 2015.

Watershed	Area (ha)	Slope (%)	Juniper basal area (m^2 ha^{-1})	Juniper density (trees ha^{-1})	Soil types
J	1.34	0-5	17.97	1103	Coyle and Zaneis soils (55.75%); Stephenville-Darnell complex (22.42%) Grainola-Lucien complex (21.83%);
J-RP	2.57	0-5	13.18	863	Stephenville-Darnell complex (77.75%); Renfrow and Grainola soils (11.00%)
J-SG	3.79	0-4	8.66	1533	Stephenville-Darnell complex (29.31%); Renfrow and Grainola (29.02%); Coyle-Lucien complex (20.08%); Grainola-Lucien complex (13.04%)

Table 2.2 Timeline of treatments for the cut watershed allowed to restore to native prairie (J- RP) and the cut watershed converted to switchgrass (J-SG) for the water years 2011 to 2018 (* indicates a time period used in the regression analyses testing difference for runoff events).

Phase	Time	J-RP	J-SG
Calibration	Oct. 2010 — Jun. 2015*	Pretreatment	Pretreatment
Transition	Jul. 2015*	Cut	Cut
	Aug. 2015 — Jan. 2016*	Dry	Dry
	Feb. 2016 — Apr. 2016	Juniper removal and land idle	Juniper removal and land idle
	May 2016 — Mar. 2017*	Recovery to prairie	Herbicide spraying
	Apr. 2017	Recovery to prairie	Plant switchgrass
Restored grassland	May, 2017— Sep., 2018*	Restored prairie	Established switchgrass

Table 2.3 Statistical results of regression analyses for event-based runoff (square root transformed) between J and J-RP, and between J and J-SG during calibration phase (from October 2010 through June 2015), cut and drying phase (from July 2015 through January 2016), recovery or herbicide phase (from May 2016 through March 2017) and restored prairie or established switchgrass phase (from May 2017 through September 2018). Coefficients for slope and intercepts are presented ± S.E.

Regressions*	Phases	Slope		Intercept		R^2
		Coefficient	P values	Coefficient	P values	
J vs. J-RP	Calibration	0.62±0.02	<0.001	0.003±0.02	0.902	0.91
	Cut juniper, drying	1.27±0.37	0.009	0.68±0.41	0.132	0.59
	Recovery to prairie	2.82±0.35	<0.001	0.12±0.15	0.425	0.74
	Restored prairie	1.19±0.11	<0.001	0.28±0.11	0.013	0.79
J vs. J-SG	Calibration	0.90±0.04	<0.001	-0.11±0.06	0.049	0.91
	Cut juniper, drying	0.77±0.04	0.098	1.05±0.46	0.057	0.34
	Herbicide	2.96±0.64	<0.001	0.64±0.29	0.039	0.51
	Established switchgrass	1.41±0.16	<0.001	0.73±0.16	<0.001	0.77

*J: the intact control watershed of juniper; J-RP: the cut watershed allowed to restore to native prairie (J-RP); J-SG: the cut watershed converted to switchgrass (J-SG).

Table 2.4 Statistical results of the comparison of the regression line of slope and intercept between calibration phase (from October 2010 through June 2015) with cut and drying phase (from July 2015 through January 2016), recovery and herbicide phases (from May 2016 through March 2017), and restored grassland phases (from May 2017 through September 2018).

Phase Contrast	Runoff regression pair*	P values	
		Slope	Intercept
Calibration vs. Cut juniper, drying	J vs. J-RP	<0.001	<0.001
Calibration vs. Recovery to prairie	J vs. J-RP	<0.001	0.21
Calibration vs. Restored prairie	J vs. J-RP	<0.001	0.001
Calibration vs. Cut juniper, drying	J vs. J-SG	0.52	<0.001
Calibration vs. Herbicide	J vs. J-SG	<0.001	<0.001
Calibration vs. Est. switchgrass	J vs. J-SG	<0.001	<0.001

* J: the intact control watershed of juniper; J-RP: the cut watershed allowed to restore to native prairie (J-RP); J-SG: the cut watershed converted to switchgrass (J-SG).

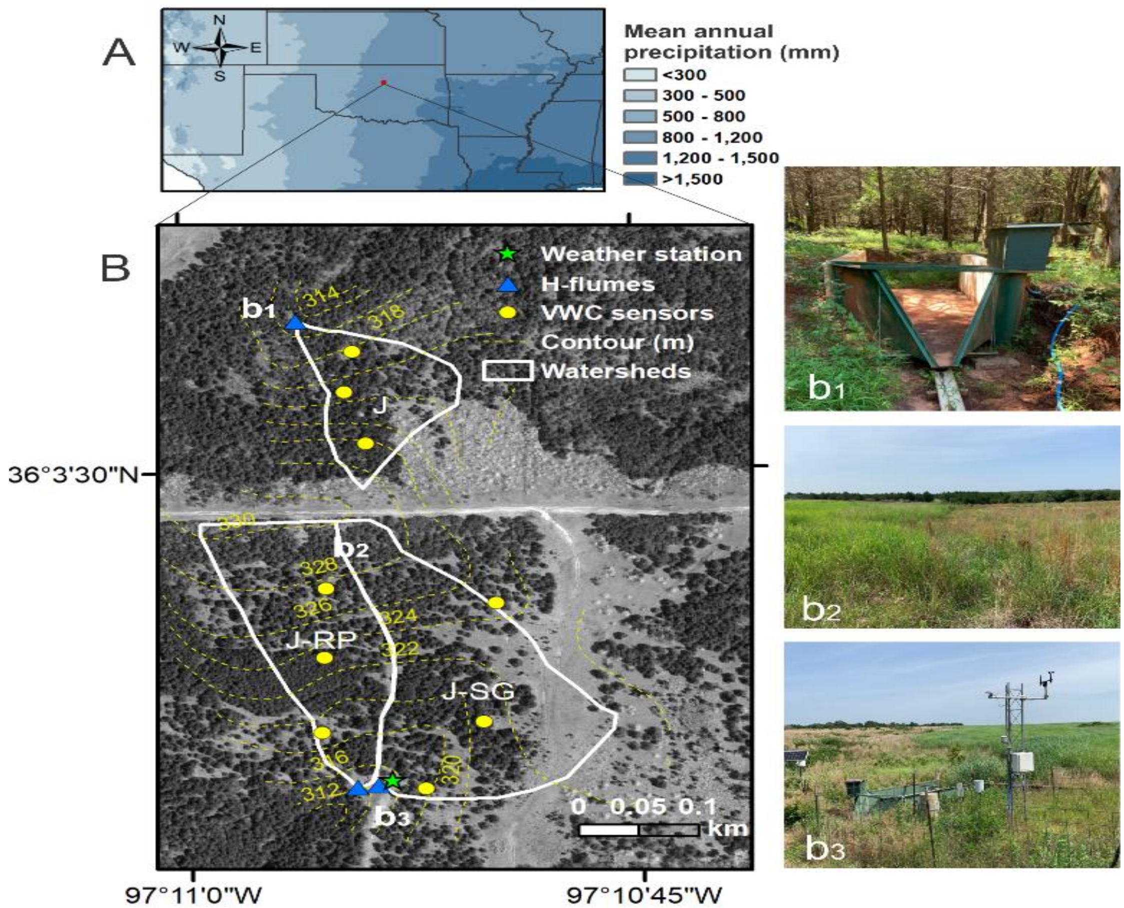

Figure 2.1 Mean annual precipitation from 1981 to 2010 across Oklahoma, USA (precipitation raster GIS data were obtained from the PRISM climate group), and the approximate location of the study site in the Cross Timbers of north-central Oklahoma (A). The juniper cover of the study site before treatment (based on Google Earth image taken in February 2014) (B). The three experimental watersheds (J, J-RP, J-SG) were in upland locations (B). The watershed boundaries were delineated manually using a land surveying laser level, GPS units, and ground-truthing by manual observation of surface flow paths. The contour lines shown were generated in ArcGIS 10.5 by using the Lidar 2-meter data for the visualization purpose. The soil moisture arrays with four volumetric water content (VWC) sensors were distributed at the upper, middle, and lower positions of each watershed. The intact control juniper watershed (J) had no treatment (b_1). J – RP watershed was restored to native prairie, and J-SG watershed was converted to switchgrass (b_2). The weather station was close to the H-flume in the watershed J-SG (b_3).

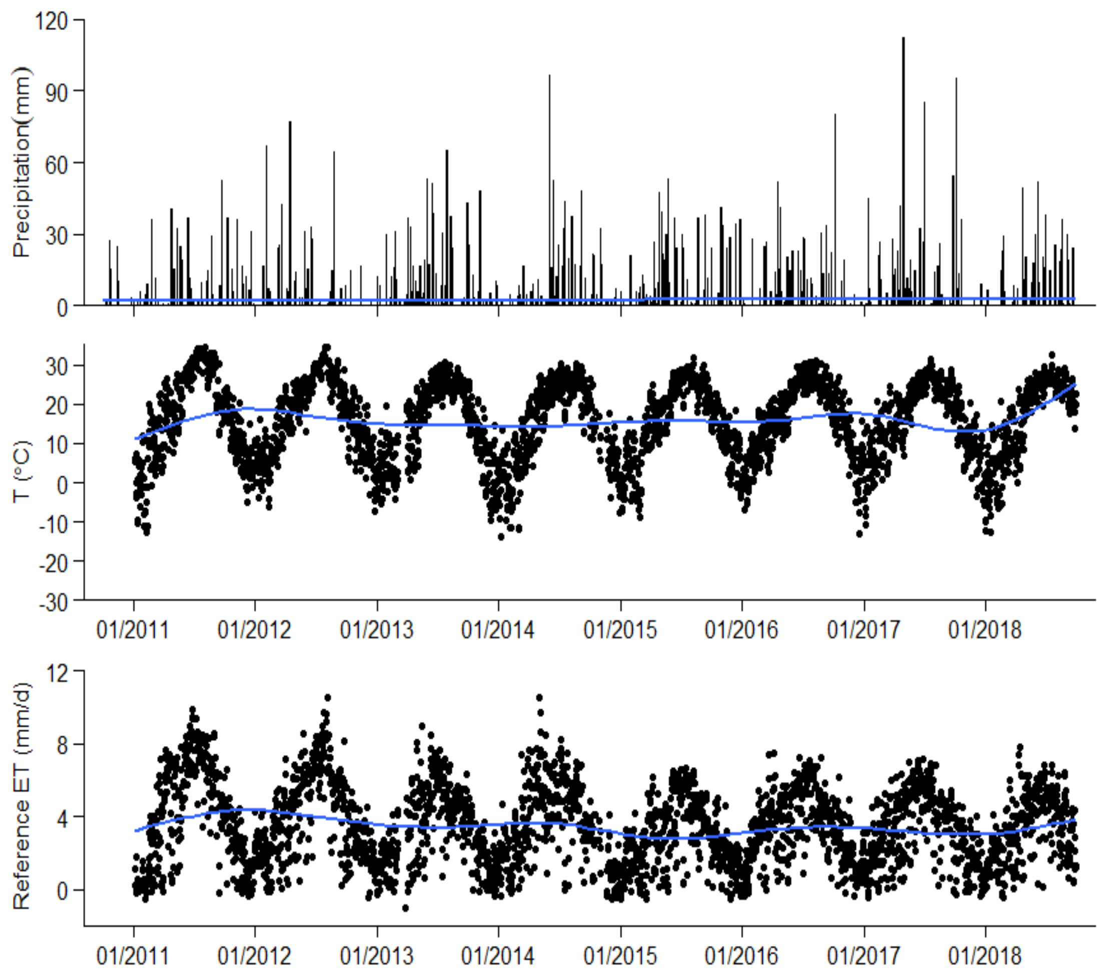

Figure 2.2 Daily precipitation, daily mean temperature, and daily reference evapotranspiration from the water years 2011 to 2018. The blue lines are the weighted regression trend lines. The daily precipitation trended up (τ =0.031, $p = 0.027$), there was no significant trend in the daily temperature ($p = 0.18$), and the daily reference ET trended down (τ = -0.066, $p < 0.001$).

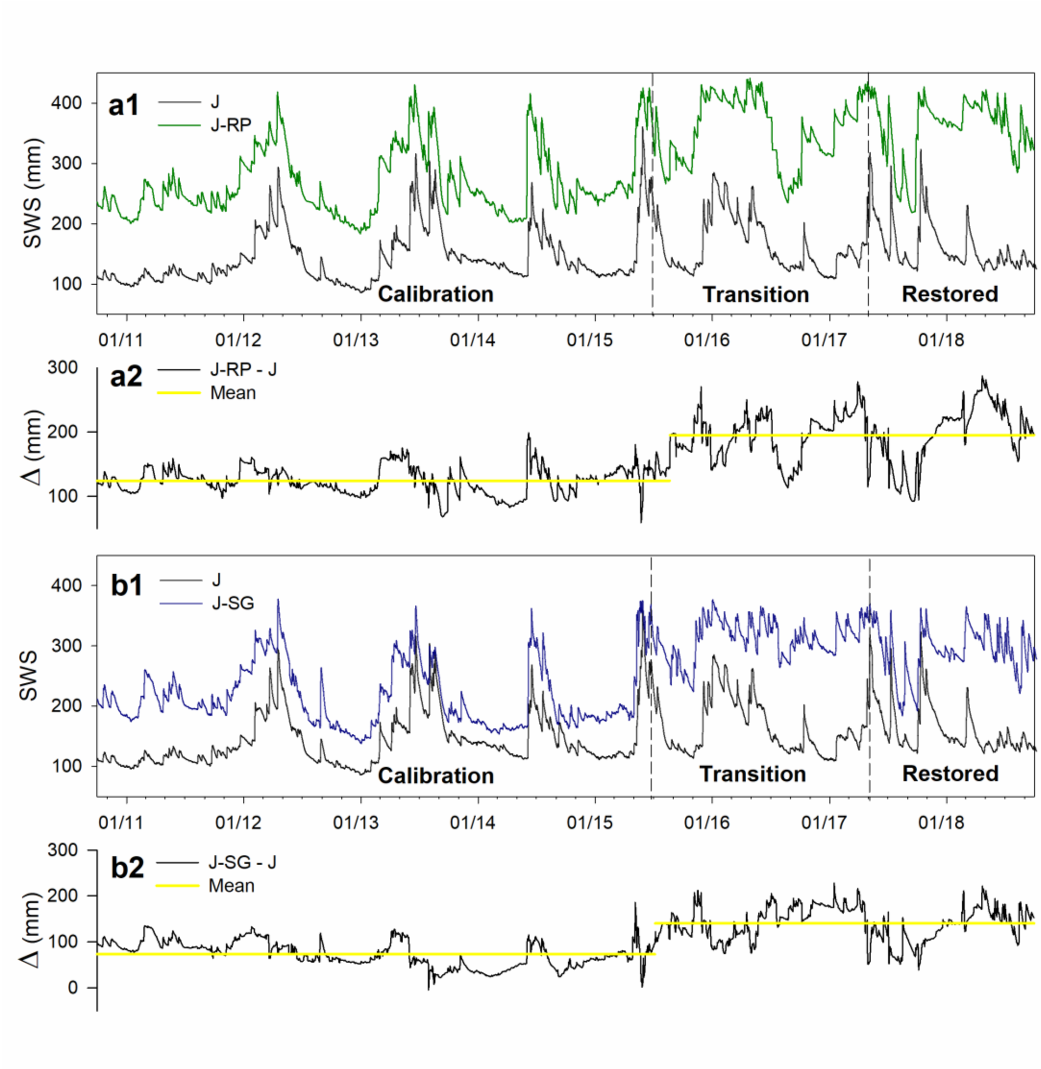

Figure 2.3 Daily variation of root zone soil water storage (SWS) in the soil profile (1000 mm) of the paired watersheds; J vs. J-RP (a1) and J vs. J-SG (b1), and the difference in daily SWS (Δ) between paired watersheds; J-RP – J (a2) and J-SG – J (b2) during the study period (water years 2011 to 2018). Yellow lines represent the mean values of the ΔSWS in different periods separated by the change point. J: the intact control watershed of juniper; J-RP: the cut watershed allowed to restore to native prairie (J-RP); J-SG: the cut watershed converted to switchgrass (J-SG).

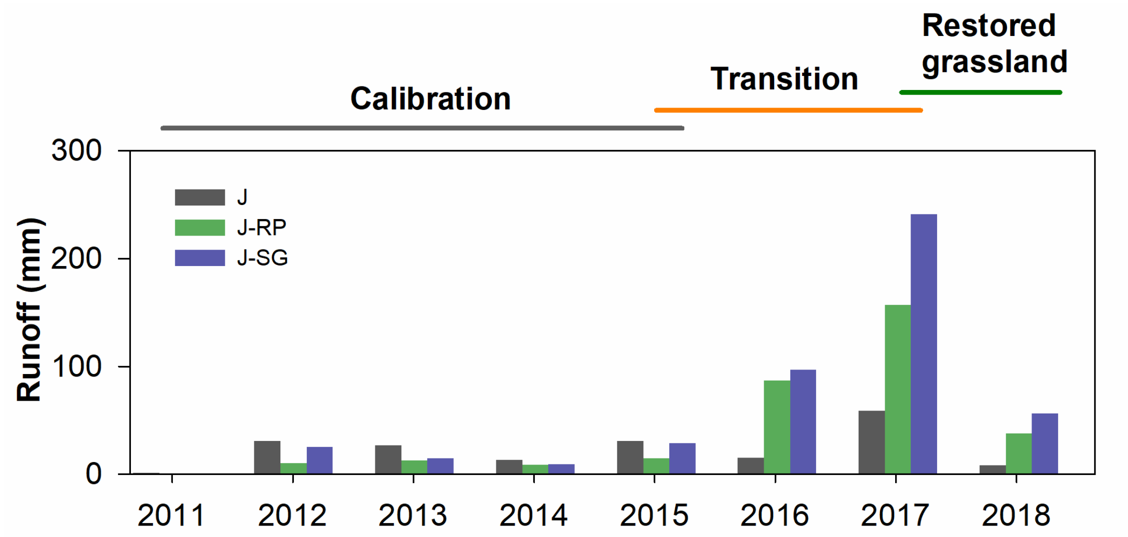

Figure 2.4 Annual runoff depth from J, J-RP and J-SG watersheds during the calibration, transition, and restored grassland phases for water years 2011 to 2018. J: the intact control watershed of juniper; J-RP: the cut watershed allowed to restore to native prairie (J-RP); J-SG: the cut watershed converted to switchgrass (J-SG).

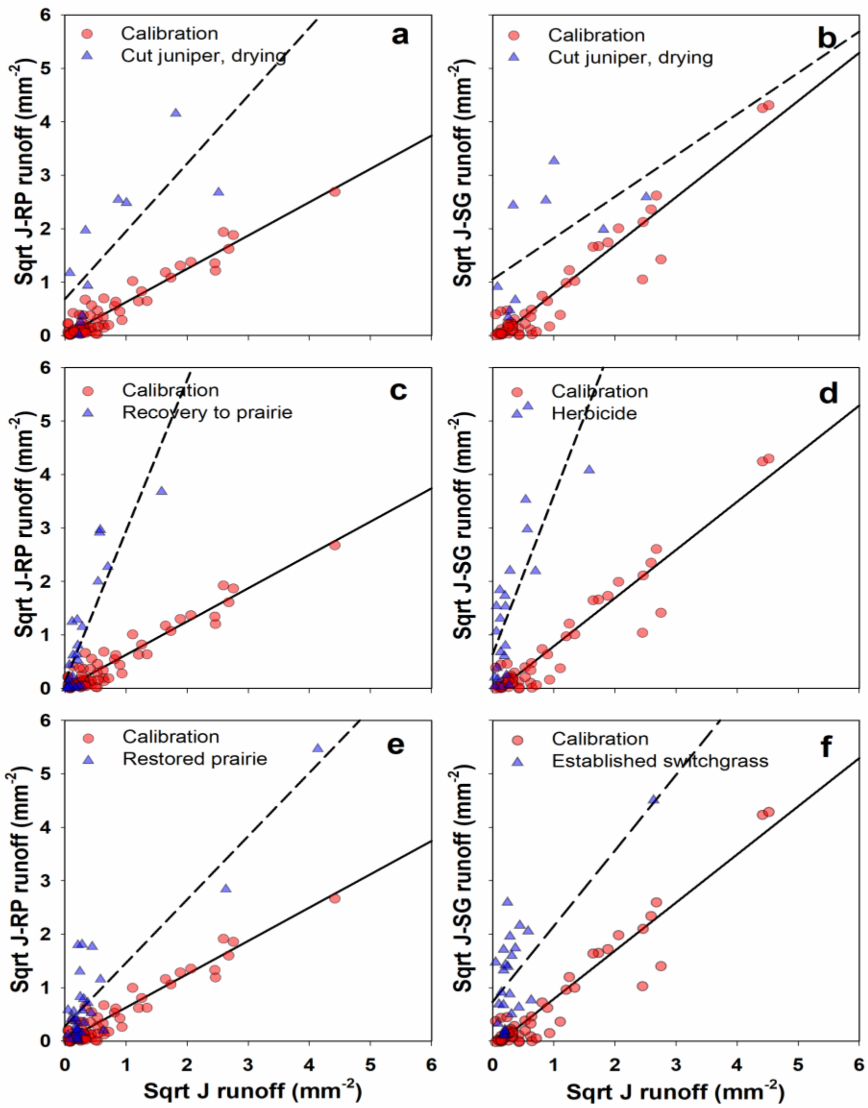

Figure 2.5 Event-based linear relationship of square root transformed runoff depth between the control juniper watershed (J) and the treated watersheds in three phases: calibration; transition (juniper cut and drying, recovery to prairie or herbicide treated); restored grassland. The calibration phase was from October 2010 through June 2015, the juniper cut and drying phase (a, b) was from July 2015 through January 2016, the recovery to prairie phase for J-RP (c) and the herbicide phase for J-SG (d) were from May 2016 through March 2017, and the restored prairie phase for J-RP (e) and the established switchgrass phase for J-SG (f) were from May 2017 through September 2018.

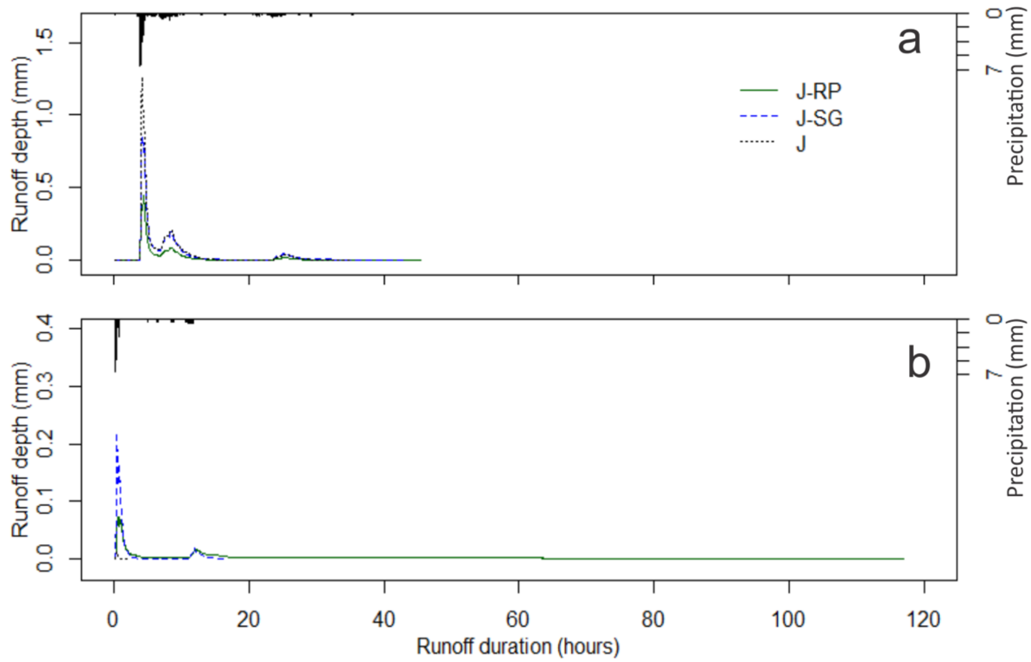

Figure 2.6 Comparison of hydrograph of control watershed J and treated watersheds J-RP and J-SG for a storm event on May 23rd, 2015 during the calibration phase (a) and for a storm event on May 2nd, 2018 during the restored grassland phase (b).

CHAPTER III

SEDIMENT RESPONSES AFTER JUNIPER REMOVAL AND ESTABLISHMENT OF PRAIRIE OR SWITCHGRASS BIOMASS PRODUCTION SYSTEM

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ABSTRACT

Tallgrass prairie in the southern and south-central Great Plains of the USA has been encroached by juniper (*Junipers virginiana* L., eastern redcedar), which has altered hydrological functions. Mechanical removal of juniper woodlands to reverse this landscape transition, followed by re-establishing herbaceous species, is encouraged and practiced by some landowners. However, there has been no systematic assessment of the sediment responses to the land surface disturbances associated with the juniper removal and the re-establishment of prairie or herbaceous bioenergy crop systems. In this study, the Before-After Control-Impact (BACI) design was used to 1) quantify the impact of mechanical removal of juniper and restoration to prairie or switchgrass (*Panicum virgatum* L.) biomass production system on sedimentation processes, including average sediment concentration, peak sediment concentration, and event-based sediment load; 2) compare the sedimentation process between restored prairie and established switchgrass.

Results showed that the ambient sediment load was low for juniper woodlands averaging 10 g m^{-2} in 2015. Mechanical removal of juniper increased annual sediment load to approximately 30 g m^{-2} in 2016. However, a pulsed increase of sediment load (1,330 g m^{-2}) occurred with herbicide application to plant switchgrass compared to 114 g m^{-2} for regenerating prairie and 23 g m^{-2} for the intact juniper woodland in 2017. The average sediment concentration in the runoff and the event-based sediment load in the fully established switchgrass biomass production system were relatively lower than the restored prairie. The mean sediment load for the first two years after grassland establishment (2018 and 2019) were 44 $\pm$24 s.e. g m^{-2} and 29 $\pm$9 s.e. g m^{-2} from the restored prairie and the switchgrass biomass production system, respectively, significantly lower than 73 $\pm$47 s.e. g m^{-2} for the intact juniper woodland. The results suggest that hydrological function can quickly recuperate by mechanically removing juniper woodland following the establishment of prairie or switchgrass biomass production system.

INTRODUCTION

Due to marginal and often variable rainfall in the south-central Great Plains, managing surface water is critical to ensure adequate supply during drought and to prevent flooding during extreme rainfall events. As a result, the region has the highest density of surface impoundments in the USA constructed to store and confine runoff (Berg *et al.*, 2016b). Rangeland for cattle production is the primary land use in this region (Bagley *et al.*, 2017), and the surface runoff generated from these grass-dominated ecosystems serves as an essential water source and supports a diverse network of ephemeral and intermittent streams, farm ponds, and reservoirs, which are critical for ranching communities providing water for both municipal and livestock supplies (Wine *et al.*, 2012; Berg *et al.*, 2016a). However, this region is characterized by highly erosive soils and sparse vegetation cover. Site disturbance and land use change can significantly increase sediment concentration in surface runoff, impairing streams, and reducing the storage capacity of surface impoundments, especially for flood control reservoirs (Fox and Wilson, 2010; McAlister *et al.*, 2013).

The quantity and quality of surface runoff from grasslands are highly responsive to decreasing herbaceous vegetation cover associated with low soil productivity or rangeland degradation (Fenetahun *et al.*, 2018; Munoth and Goyal, 2020). Reduction in herbaceous vegetative cover often leads to increased overland flow and sediment transport in grasslands (Belnap and Gillette, 1998; Urgeghe *et al.*, 2010; Field *et al.*, 2011). As a result, excessive loss of herbaceous vegetative cover due to high cattle stocking rate or high-intensity fire will exacerbate surface runoff and soil erosion (Menzel *et al.*, 1978; Field *et al.*, 2011). Even a moderate stocking rate can reduce litter cover and cause soil compaction which can result in a significant increase in surface runoff and sediment yield in grassland watersheds with very erodible soil (Wine *et al.*, 2012).

In addition to grazing and fire, woody plant encroachment can reduce the herbaceous vegetation cover in prairies. An increase of juniper trees (*Juniperus osteosperma* [Torr.] Little) in the grasslands of the Intermountain West of the USA led to an increase in overland flow and sediment transport down hillslopes (Pierson et al., 2010). This increase in surface runoff and sediment concentration at the edge of the hillslope or watershed outlet was related to reduced herbaceous cover and increased soil compaction associated with the intercanopy areas of juniper woodlands (Pierson *et al.*, 2010), with the largest increase of runoff and soil erosion occurring during large thunderstorm events (Wilcox *et al.*, 2003). Juniper removal in that system stimulated herbaceous plants to recover and improved soil infiltration capacity of the intercanopy patches, which protected the soil surface from direct rain splash erosion (Pierson *et al.*, 2007). Leaving residues from shredding junipers on-site also decreased soil surface exposure and sediment transport (Cline *et al.*, 2010). Further east, restoration of prairie by mechanically removing juniper (*Juniperus virginiana* L., eastern redcedar) resulted in a rapid recovery of prairie vegetation (Schmidt et al. In Press) substantial increase of runoff at the experimental watershed scale (Zhong *et al*., 2020). However, the sediment response to mechanical removal and the restored prairie remains mostly unquantified.

Switchgrass (*Panicum virgatum* L.) is a native species in tallgrass prairie and is widely used to prevent soil erosion in restoration projects (Wu and Liu, 2012; Feng *et al.*, 2015). It is also a dedicated feedstock for biofuel production (Wullschleger *et al.*, 2010).

However, the effectiveness of switchgrass feedstock production systems on sediment load, especially compared with the restored prairie, has not been systematically assessed. While switchgrass stands can reduce soil erosion, the soil loss associated with the site disturbance from the conversion process needs to be quantified. Mechanical removal of juniper and associated machine trafficking disturbs topsoil and litter cover, and herbicide application used as site preparation for no-till drilling for establishing switchgrass production systems temporarily reduces vegetative cover. It is essential to assess the magnitude of the pulsed increase of surface runoff and soil erosion during the transition from one vegetation type to another.

Prairies are often dominated by intermittent streams with generally low sediment concentrations, but the sediment concentrations can increase by 3- to 12-fold in response to large rainfall storms and disturbances (Larson et al., 2013). As a result, high water turbidity is a common water quality impairment for ephemeral and intermittent streams and farm ponds in the prairie-dominated regions (Dodds and Whiles, 2004; Blanchard *et al.*, 2011). The peak sediment concentration is very responsive to bare soils and is a good indicator of soil erodibility. Event-based sediment load for a watershed can be estimated based on runoff depth and sediment concentration (Defersha and Melesse, 2012). Event-based sediment load measured at the outflow of upland watersheds with well-defined land use or vegetation type is the most direct assessment of the effects of land use and management practices on soil erosion (Grum *et al.*, 2017). Accumulated sediment yield at a monthly or annual scale can be calculated from event-based load measurements to compare results from other land uses and regions (Walling, 1994).

The objectives of this study were to quantify the impact of mechanical removal of eastern redcedar and subsequent restoration to tallgrass prairie or switchgrass on runoff and sediment processes and the difference in sediment processes between restored prairie and established switchgrass. The runoff and sediment results are reported based on an experimental watershed study over five years, which included three phases: calibration, transition, alternative. The transition phase included tree removal, herbicide application, and grassland recovery. The alternative phase was defined as when the switchgrass was established or prairie restored. A Before-After Control-Impact (BACI) analytical approach was used to compare the sediment load during the three phases and contrast the sediment metrics in the alternative phase.

MATERIALS AND METHODS

Study area

The research was conducted in the OSU-Range Research Station (OSU-RRS) situated 15 km southwest of Stillwater, Payne County, Oklahoma, USA (36°3’46.73” N, 97°11’3.33” W) (Figure 3.1). Most of this area was cultivated after the 1889 Land Run to grow cotton and later abandoned in the 1940s (Booth, 1941). In this study, three juniper encroached watersheds were used (Figure 3.1). The soils are well drained, consisting predominantly of the Stephenville (Fine-loamy, siliceous, active, thermic Ultic Haplustalfs) –Darnell complex (Loamy, siliceous, active, thermic, shallow Udic Argiustolls), Coyle (Fine-loamy, siliceous, active, thermic Udic Argiustolls), and Grainola–Lucien complex (Fine, mixed, active, thermic Udertic Haplustalfs; Loamy, mixed, superactive, thermic, shallow

Udic Haplustolls). The average depth of soil is less than 1 m underlain by sandstone substrates (Zou *et al.*, 2014). The slopes of the watersheds are from 0 to 5%, and the area was 1.3 ha, 2.6 ha, and 3.8 ha for each watershed (Zhong *et al.*, 2020).

Experimental design and treatment implementation

The Before-After Control-Impact (BACI) experimental design (Green, 1979) was used in this study. In comparison with the Before and After (BA) design, the BACI design accounts for the effects of temporal variation of environmental variables (Underwood, 1992; Smith, 2002), such as the change in precipitation pattern (Brown *et al.*, 2005), which also directly affects runoff and sediment load (Stewart-Oaten *et al.*, 1986). In the study, all three watersheds were initially heavily encroached by juniper (Table 3.1). The watershed to the north was selected as the Control watershed (J), and the two watersheds to the south were selected as the Impact watersheds, i.e., land use conversion. Based on the research objectives, the “Impact” was further divided into the transition phase and the alternative phase after the calibration phase (Table 3.1). Analysis of variance (ANOVA) through a linear mixed-effect model for the BACI design was used to evaluate the main effects from watershed treatment and phase of land use conversion (or phase), but also adequately address the interactive effects of watershed treatment and phase.

All the juniper trees in the two Impact watersheds were cut in July 2015. Cut trees were left to dry on site, then removed by the end of February 2016. One Impact watershed was left to revegetate naturally and was assigned as the “juniper to restored prairie” watershed (J-RP) (Figure 3.1). The other Impact watershed was further treated with glyphosate herbicide during 2016, and the lowland ‘Alamo’ switchgrass cultivar was planted at a rate of 7.8 kg/ha and depth of 0.64 cm using a Truax no-till drill machine in April 2017. This watershed was assigned as "juniper to switchgrass" (J-SG) (Figure 3.1). The two treated watersheds were fenced to prevent cattle grazing and trampling. More details of juniper removal and watershed treatment were described in Zhong *et al.* (2020). For the Impact watersheds, the transition phase was defined as the period after juniper cutting (July 2015) and before the new vegetation cover was fully established (October 2017). The alternative phase included two water years (2018 and 2019) following the transition phase (Table 3.1). Switchgrass cut at approximately 10 cm in height, baled, and removed from the J-SG watershed every November.

Data collection

Precipitation and Runoff

Precipitation was measured using a tipping bucket rain gauge (TB3, Hydrological Service America, Lake Worth, FL, USA) installed near the outlets of the two Impact watersheds. Runoff from each watershed was gauged using a 0.9 m prefabricated USDA H-flume at each watershed outlet (Zhong *et al.*, 2020). A precipitation event was considered ended when there was no further precipitation reading for a minimum of six hours. The definition and separation of a runoff event and associated values were described in Zhong *et al.* (2020).

Runoff sample and event-based sediment load

All runoff events between 2014 and 2019 were sampled using ISCO samplers (Model 3700C, Teledyne ISCO, Lincoln NE, USA) (Figure 3.1) to analyze total suspended solids. Runoff samples were collected using an intake strainer located at the bottom of a 16 cm polyvinyl chloride (PVC) trough. Each trough was placed approximately 15 cm beneath each flume outlet, and each strainer was made using a 2.5 cm PVC pipe with 10 mm diameter holes and wire screening. The wire screening prevented the intake strainer from collecting debris and clogging the flexible ISCO intake tubing, while the trough prevented the intake strainer from sitting inside the H-flume and disturbing the H-flume stage-discharge relationship (Figure 3.1). Samples were collected based on a flow-weighted and time-weighted sampling strategy to trigger runoff sample collection. In this sampling strategy, if the runoff depth converted from the H-Flume stage reading was greater than 21 mm, the sampler was triggered to collect an initial 250 mL runoff sample. Subsequently, CR200 or CR1000 dataloggers (Campbell Scientific, Logan, UT, USA) calculated the absolute difference between the initial and next five-minute runoff depth. If the absolute difference within the five-minute interval was greater than 21 mm, then the sampler would take another runoff sample. If not, the sampler would continue to calculate the absolute difference between the previous and current runoff depth until the 40-minute maximum time (for J-RP and J-SG) or 30-minute maximum time (for J) between samples was reached, and then another runoff sample would be taken (Lisenbee *et al.*, 2015). This sampling strategy allowed the sampler to capture more samples when the runoff increased significantly, allowing better characterization of flashy versus long duration runoff events. The sediment concentrations were gap filled using the 30- or 40-minute concentration to match runoff data collected at the five-minute interval.

Total suspended solids were analyzed in the lab, according to ASTM Standard D3977-97 (ASTM, 2000). Samples were dried at 105 ℃ using a VWR Horizontal Air Flow Oven for a minimum of 72 hours. Then samples were placed in a desiccator to prevent any atmospheric moisture from re-entering the samples as they cooled. Samples were quickly weighed, and the data were recorded. Sediment concentration (g/L) was calculated as the mass of sediment per unit volume of runoff.

The event-based sediment yield (g) was accumulated from all sediment values calculated for each 5-minute interval during each single runoff event. The unit area sediment load (g m^{-2}) was calculated using the area of each watershed. Sediments gradually built up within the H-flumes after multiple runoff events. This deposit was shoved into buckets and weighted in the lab. This load was not included in the event-based sediment load, but it was added to the accumulated sediment load on an annual scale.

Data analysis and statistics

The effects of phase, site (i.e., watershed), and phase and site interaction were tested using a linear mixed-effect (LME) model (Smith, 2002). Pairwise comparisons were made among the three watersheds. In the LME model, three independent variables were incorporated as fixed factors: 1) Phase: 'calibration' vs. 'transition', 'transition' vs. 'alternative', and 'calibration' vs. 'alternative'; 2) Site: 'impact' vs. 'control' watershed, 'impact' vs. 'impact' watershed; 3) Sampling times: time of sampling was treated as a categorical factor (repeated measurements in the BACI design) nested within the phase, allowing the time series structure to be taken into account (there were 34 sampling times that had sufficient flow in all three watersheds to be included in this analysis). Error terms are the differences between observed values and estimated values. The model was:

$$X_{ijk} = \mu + \alpha_i + \tau_{k(i)} + \beta_j + (\alpha\beta)_{ij} + \varepsilon_{ijk} \qquad (2)$$

Where X_{ijk} was the dependent variable: event-based sediment load (g m^{-2}), or average sediment concentration (g/L), or peak sediment concentration (g/L); μ was the overall mean; α_i was the effect of phase (i = calibration, transition, or alternative; i = 1, 2, 3); $\tau_{k(i)}$ represented time within the phase, $k(i)$ was the $k(i)$ times for each i ($k_{(1)}$ = 1, 2,…, 4; $k_{(2)}$= 1, 2,…, 10; $k_{(3)}$= 1, 2,…, 20); β_j was the effect of site (j = impact, control; j = 1, 2); $(\alpha\beta)_{ij}$ was the interaction between phase and site; and ε_{ijk} was the remaining error. Event-based sediment data were log-base 10 transformed to meet the assumptions of the LME model.

The interaction between phase and site was the main interest of the BACI method (Underwood, 1992). No interaction indicates the main effects of phase and site were independent. In other words, the changes in sediment variables over phases were similar among the different sites; likewise, any differences in sediment variables among sites were consistent among the different phases. A phase effect would mean that sediment load or sediment concentration was greater in one period than another after controlling for the effect from the site. A site effect would indicate that one site has more sediment load or sediment concentration than another after controlling for the effect of phase.

During the water year 2015 through 2019, there were 318 rainfall events, 251 runoff events, and 123 sediment events. Event-based sediment data were log-base 10 transformed to meet the model assumption (i.e., the errors are normally distributed). However, for most small runoff events, runoff or sediment loads were small and occurred in one or two watersheds. Only 34 significant sediment events occurred across all watersheds and were used in the statistical analysis. The accumulated sediment load from the 34 events accounted for 96%, 76%, and 85% of the total sediment load for J, J-RP, and J-SG, respectively. The sediment loads beyond the 34 events were included in the annual sediment yield.

If the terms were significant in the statistical test, the Contrast method (it can test a specific set of hypotheses among the means) was applied to estimate the difference of least-square means of different phases (three sets of comparison: Transition vs. Calibration; Alternative vs. Transition; Alternative vs. Calibration) (Lane *et al.*, 1999; Dąbrowska *et al.*, 2017). Since the data were unbalanced among phases, the least-square means were applied. The estimated means in each phase (calibration, transition, and alternative) were the differences between the site for each dependent variable. The LME model and the contrast of least-square means were run in the RStudio, with the R version 4.0.0.

RESULTS

Event-based sediment load and sediment concentration

During the significant rainfall events that produced sufficient runoff and sediment load for inclusion in the analysis, all three pairwise comparisons among watersheds had significant interactions between phase and site (Table 3.2, Figure 3.2a), indicating that the differences in sediment load between watersheds varied depending on the phase. This interaction was primarily caused by the large and significant increase in sediment load for the Impact watersheds during the transition phase compared to the calibration phase (Table 3.3). The largest increase in sediment load during the transition phase was for the J-SG watershed. For all pairwise comparisons, the differences among sites during the alternative phase were significantly less than during the transition phase. Comparing the calibration phase to the alternative phase, the relative sediment load for restored prairie was lower than the transition phase but remained elevated compared to the juniper woodland (Table 3.3). Also, comparing the alternative phase to the calibration phase, the J-SG watershed had a significantly lower sediment load than the J-RP watershed (Table 3.3). The J vs. J-SG difference between the calibration and alternative phase was not significantly different (Table 3.3).

For average sediment concentration, only the comparisons that involved conversion to switchgrass had a significant interaction between phase and site (Table 3.2, Figure 3.2b). This interaction occurred because the difference in average sediment concentration between the switchgrass watershed and the Control watershed decreased for the alternative phase (Table 3.3).

All three comparisons of peak sediment concentrations had significant interactions between phase and site (Table 3.2, Figure 3.2c). Similar to the effects on sediment load, the interactions occurred because the peak sediment concentration for the Impact watersheds significantly increased during the transition phase compared to the calibration phase (significant for J vs. J-RP and J vs. J-SG) (Table 3.3). For the alternative state, the differences were smaller than for the transition phase and no longer significant than the calibration phase (all three comparisons) (Table 3.3).

Annual precipitation, runoff, sediment yield

Thirty-year average annual precipitation was 939 mm (Figure 3.3a). The annual precipitation was above the 30-year average annual precipitation for the water year 2017 (8%) and 2019 (62%). Substantial annual runoff and sediment load were observed during 2019. Although 2017 had a slightly above average precipitation, the annual runoff was much greater in the treated watersheds that were mostly devoid of vegetation (Figure 3.3b and 3.3c). Before treatment (in the water year 2015), annual runoff from the three juniper watersheds was 25 mm (Figure 3.3b). In the transition phase (2016 and 2017), the annual runoff was 37 mm from J compared with 122 mm and 169 mm from J-RP and J-SG. In the alternative phase (2018 and 2019), the mean annual runoff was 156 mm for juniper woodland, 274 mm for the restored prairie, and 257 mm for the switchgrass.

In 2015 during the calibration phase, the annual precipitation was below the 30-year average precipitation and the annual sediment load was very low for all watersheds (Figure 3.3c). However, the annual sediment load was the greatest from J (13 g m^{-2}) (Figure 3.3c, Table 3.3). In the first part of the transition phase (2016) when juniper was cut but not yet removed, sediment load was slightly increased and greater in the Impact watersheds (24 g m^{-2} for J-RP and 33 g m^{-2} for J-SG) than the juniper watershed (1 g m^{-2}). During the second part of the transition phase (2017), when junipers were removed from both Impact watersheds and the J-SG had been sprayed with herbicide, the J-SG had the largest sediment load (1,330 g m^{-2}), followed by the J-RP (114 g m^{-2}), and J (23 g m^{-2}) (Figure 3.3). In the alternative phase (2018 and 2019), the mean annual sediment yield was 73 ± 47 g m^{-2} for juniper, 44 ± 24 g m^{-2} for the restored prairie, and 29 ± 9 g m^{-2} for the switchgrass.

Mechanism underlying runoff and sediment response

The hydrograph and the sediment concentration of a typical runoff event with similar rainfall were selected from each phase to examine the sedimentation processes for different watersheds. During the calibration phase, the J-RP watershed had a relatively lower peak flow rate but a similar flow duration than the J and J-SG watersheds (Figure

3.4A). However, no runoff was generated from the J watershed during the transition phase with a similar rainfall amount (Figure 3.4B). The J-SG watershed produced greater peak flow compared to J-RP, and the flow duration was similar. During the alternative phase, the J-SG watershed had a relatively greater peak flow than the J-RP watershed. Both J-RP and J-SG had relatively greater peak flows and longer flow duration than J during the alternative phase (Figure 3.4C). During the transition phase, sediment concentration was positively related to the flow rate, especially for J-SG (Figure 3.4B). The sediment concentration showed a reverse relationship with the flow rate during the calibration and alternative phases (Figure 3.4A and 3.4C).

DISCUSSION

Sediment concentration response to juniper removal

For both impact watersheds, peak sediment concentration significantly increased after juniper removal during the transition phase, while average concentration was the same from calibration to the transition phase. After the juniper stems were removed, the bare soil and the connectivity among bare soil patches increased, and the resistance that litter or standing herbaceous vegetation provided by overland flow decreased (West *et al.*, 2016). Peak sediment concentration is more responsive than average sediment concentration to site disturbance and increased bare soil areas associated with removing juniper. During the calibration phase, the average sediment concentration for three juniper watersheds was 0.32 g/L, substantially lower than 0.50 g/L for the rivers in the south-central Great Plains (Dodds and Whiles, 2004). During the transition phase, the average sediment concentration was 0.62 g/L for J and 0.39 g/L for J-RP, similar to the level during the calibration period. However, the average sediment concentration from the J-SG watershed during the transition phase increased to 1.5 g/L, three times the average sediment concentration for the south-central Great Plains. During the calibration phases and the alternative phase, the sediment concentration was high at the onset of runoff. However, the sediment concentration tended to be synchronized with the peak flow rate during the transition phase.

Sediment load response to juniper woodland removal during the transition phase

Sediment load increased after juniper removal during the transition phase for both impact watersheds. The threshold to generate runoff in the transition phase was lower than the calibration phase (Zhong *et al.*, 2020); therefore, more runoff led to more sediment load with similar or even greater sediment concentration in the transition phase. In the J-RP watershed, the quick recovery of native grasses (Schmidt *et al*., In Press) helped reduce sediment load. Native prairie started to recover beginning in May 2016 and continued throughout the 2017 water year, such that the annual sediment yield for J-RP (max of 114 g m^{-2}) was in the range of tolerable soil loss to sustain soil resources (less than 200 g m^{-2}) (FAO 2019). Sediment load had a significant increase in the J-SG watershed following the site preparation with herbicide. The annual sediment yield in the water year 2017 from the J-SG watershed was 1,330 g m^{-2}, nearly twice of water-driven sediment yield from cropland in southern Great Plains (USDA, 2009). This high annual loss primarily resulted from the pulsed response of sedimentation to relatively high precipitation in April 2017, soon after herbicide treatment.

Based on jet erosion tests (JETs), Lisenbee *et al*. (2015) predicted an approximately three-fold increase in average annual runoff and a one to two orders of magnitude increase in average sediment load immediately after juniper removal for this site. This suggested that the loss of herbaceous vegetation associated with decades-long juniper cover and the topsoil disturbance due to the machine traffic for mechanically removing trees make the watershed vulnerable to a significant water erosion risk. In this study, trees were cut in July 2015 but not immediately removed. They were removed from the site by May 2016. Our results suggest that leaving cut trees to dry in place might help protect the soil surface from rain splash erosion (Wilcox *et al.*, 2003), and the annual soil loss in the water year 2016 was only moderately elevated. However, leaving trees on site for an extended time might slow down the natural re-establishment of herbaceous cover. Shredding junipers and leaving debris on-site was reported to be effective to reduce sediment transport (Cline *et al.*, 2010), and further study is needed to understand whether leaving part of redcedar debris on-site will reduce sediment load, especially if the site is to be followed by herbicide for planting switchgrass for biomass production.

Sediment concentration response of restored prairie and switchgrass

The average sediment concentration was lower in the alternative phase than the calibration phase for the J-SG watershed, while it was the same for the J-RP watershed. The average concentration was 0.73 g/L for J, 0.25 g/L for J-RP, and 0.14 g/L for J-SG during the alternative phase. Naturally, the forested systems tend to have a lower sediment concentration than prairie systems (Dodds and Whiles, 2004). However, this study showed that the ambient sediment concentration from a juniper woodland was relatively higher than the well-established grassland, although the value was within the reported range of 0.008 to 3.0 g/L for streams in the tallgrass prairie of the United States (Larson *et al.*, 2013).

Sediment load response of restored prairie and switchgrass

When the prairie and switchgrass were fully established in the alternative phase, the restored prairie had more soil loss than the calibration phase (previous juniper woodland) from the statistical answer, and switchgrass had a similar sediment load compared to the calibration phase. The annual sediment load from the grasslands ranged from 30 – 50 g m^{-2}, which is in the range of tolerable soil loss to sustain soil resources range (FAO 2019) and significantly less than the average annual water erosion soil loss (673 g m^{-2}) from the cropland in this region (USDA, 2009). Comparing sediment responses from the two impacts watersheds, J-SG, and J-RP had a similar surface runoff depth after the grassland was established in the water year 2018 (Zhong *et al.*, 2020). The J-SG watershed had a lower sediment yield and lower average sediment concentration compared to the J-RP watershed. Planting of switchgrass was reported to increase soil macroporosity and saturated hydraulic conductivity (Zaibon *et al.*, 2016), reducing the overland flow and soil erosion (Wu and Liu, 2012). Therefore, planting switchgrass for biomass production may also reduce soil erosion.

CONCLUSION

Mechanical removal of juniper is a common management practice in rangeland restoration and protection. The Before-After Control-Impact (BACI) accounts for the effects of natural variability and proves to effectively detect the impact of land management practices on hydrological functions. Immediately after the juniper removal and before grasslands were fully established, the watersheds were vulnerable to sedimentation processes and pulsed increases in sediment load in response to site disturbance and storm events. The loss of sediments could reach a level similar to or greater than the average load from cropland in this region. However, sediment load declined considerably and returned to a level similar to the juniper woodland one year after the grasslands are established. The switchgrass production system was more effective than naturally restored prairie in reducing the total sediment yield. Planting switchgrass biofuel production system can be considered as a part of management strategies to curtail and reverse juniper expansion and prevent the rangeland from further degradation in the mesic region of the southern Great Plains.

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Table 3.1 Timeline of treatments for the watershed J-RP (Juniper → Restored Prairie) and the watershed J-SG (Juniper → Switchgrass) from water years 2015 through 2019.

Phase	Time	J-RP	J-SG
Calibration	Oct. 2014 — Jun. 2015	Pretreatment	Pretreatment
Transition	Jul. 2015	Cut	Cut
	Aug. 2015 — Jan. 2016	Dry	Dry
	Feb. 2016 — Apr. 2016	Juniper removal and land idle	Juniper removal and land idle
	May. 2016 — Mar. 2017	Recovery to prairie	Herbicide spray
	Apr. 2017	Recovery to prairie	Plant switchgrass
	May. 2017— Sep. 2017	Recovery to prairie	Establishing switchgrass
Alternative	Oct. 2017— Sep. 2019	Restored prairie	Established switchgrass

Table 3.2 P values related to results of the BACI model of event-based sediment load, average concentration, and peak sediment concentration during 34 large rainfall events from watershed pairs: J vs. J-RP; J vs. J-SG and J-RP vs. J-SG. (J: Juniper; J-RP: Juniper → Restored Prairie; and J-SG: Juniper → Switchgrass)

Pairs	Terms	Sediment load	Average sediment concentration	Peak sediment concentration
J vs. J-RP	Phase	0.648	**0.029**	0.131
	Site	0.229	**0.001**	**<0.001**
	Phase × Site	**<0.001**	0.181	**<0.001**
J vs. J-SG	Phase	0.677	**<0.001**	**0.002**
	Site	**0.001**	0.178	0.195
	Phase × Site	**<0.001**	**<0.001**	**<0.001**
J-RP vs. J-SG	Phase	**0.019**	**<0.001**	**<0.001**
	Site	**<0.001**	0.906	0.131
	Phase × Site	**<0.001**	**<0.001**	**0.001**

Table 3.3 The difference in mean values (mean ± S.E. back-transformed from log10 values) between every two watersheds (the former minus the latter) on event-based sediment load, average sediment concentration, and peak sediment concentration during each phase (calibration, transition, and alternative) during 34 large rainfall events. Note: within pairwise comparisons, means that do not share a common letter are statistically different ($p < 0.05$). Statistical analyses were conducted on log10 transformed data. (J: Juniper; J-RP: Juniper → Restored Prairie; and J-SG: Juniper → Switchgrass)

Pairs	Phase	Sediment load ($g\ m^{-2}$)	Average sediment concentration (g/L)	Peak sediment concentration (g/L)
J-RP – J	Calibration	-1.61 ± 5.41 [a]	-0.17 ± 0.27 [a]	-1.02 ± 0.51 [a]
	Transition	7.84 ± 3.42 [c]	-0.26 ± 0.17 [a]	0.62 ± 0.32 [b]
	Alternative	-2.74 ± 2.42 [b]	-0.52 ± 0.12 [a]	-1.08 ± 0.23 [a]
J-SG – J	Calibration	-0.73 ± 60.60 [a]	-0.03 ± 0.74 [b]	-0.74 ± 2.20 [a]
	Transition	111.49 ± 38.30 [b]	1.22 ± 0.47 [b]	6.30 ± 1.39 [b]
	Alternative	-4.18 ± 27.10 [a]	-0.60 ± 0.33 [a]	-1.22 ± 0.99 [a]
J-SG – J-RP	Calibration	0.88 ± 56.30 [b]	0.14 ± 0.69 [b]	0.28 ± 1.93 [ab]
	Transition	103.64 ± 35.60 [b]	1.48 ± 0.44 [b]	5.68 ± 1.22 [b]
	Alternative	-1.45 ± 25.20 [a]	-0.08 ± 0.31 [a]	-0.14 ± 0.86 [a]

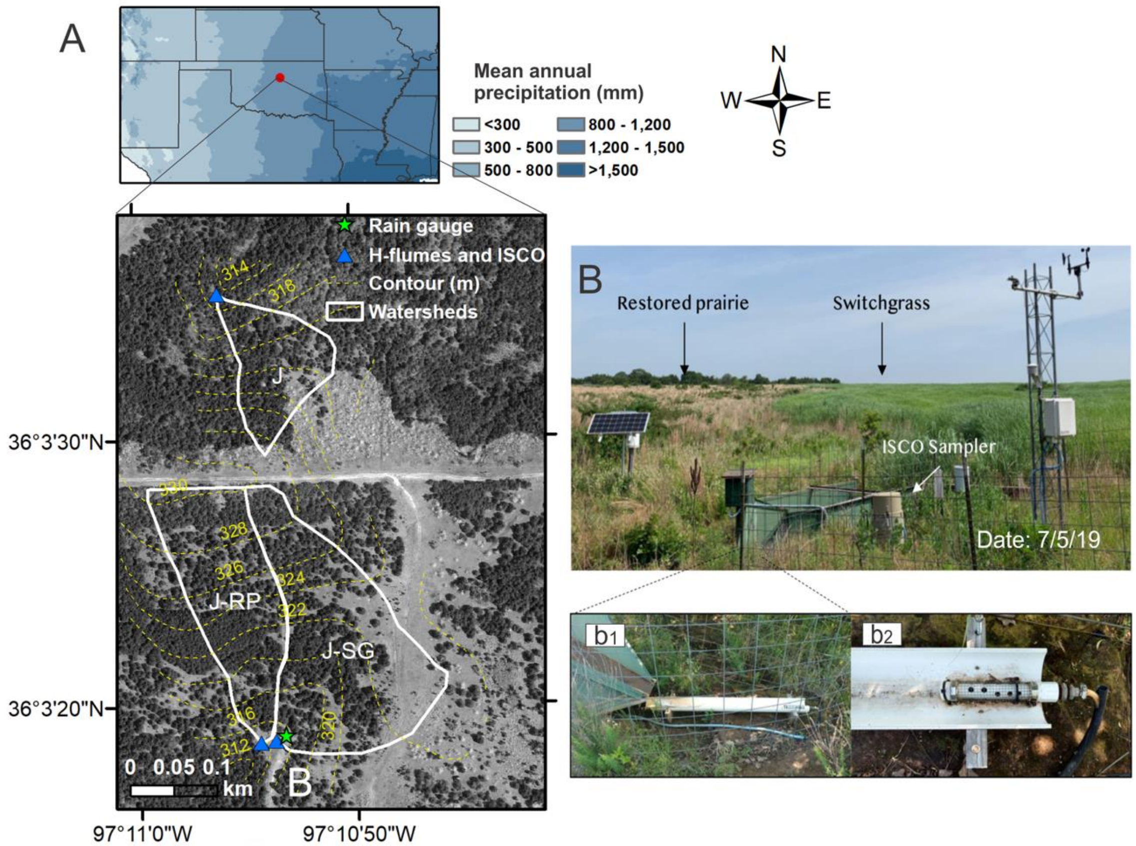

Figure 3.1 The three experimental watersheds in OSU-RRS, north-central Oklahoma, USA. The aerial photo was taken before treatment (Google Earth, February 2014). The contour lines were generated by 2 m resolution Lidar data (A). The restored prairie watershed (J-RP) is adjacent to the switchgrass watershed (J-SG) (B). H-flume, ISCO sampler, tipping bucket rain gauge, USDA standard rain gauge, and meteorological station for the switchgrass watershed are pictured. The location of troughs relative to H-flume discharge (b1); the strainer's location within trough (b2).

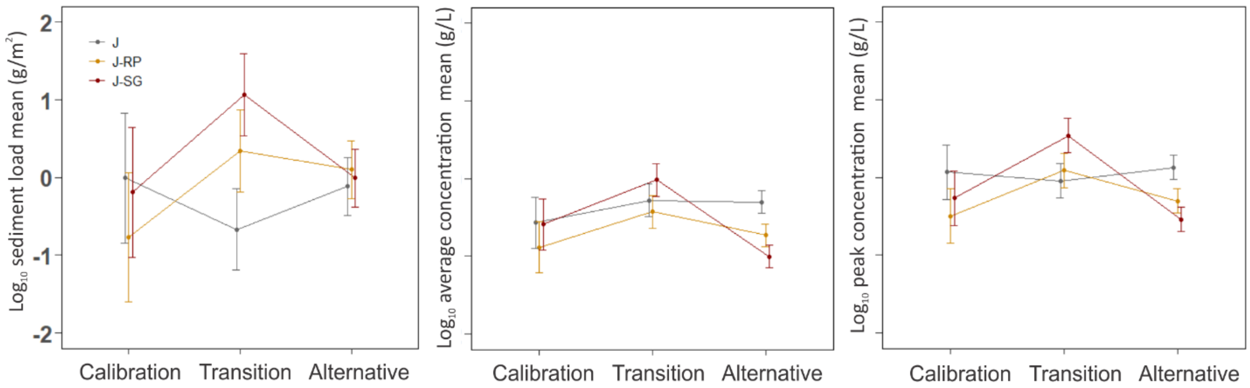

Figure 3.2 Means of log-base 10 event-based sediment load, average sediment concentration, and peak sediment concentration among watershed J (Juniper), J-RP (Juniper→ Restored Prairie), and J-SG (Juniper → Switchgrass) along with three phases: calibration, transition, and alternative.

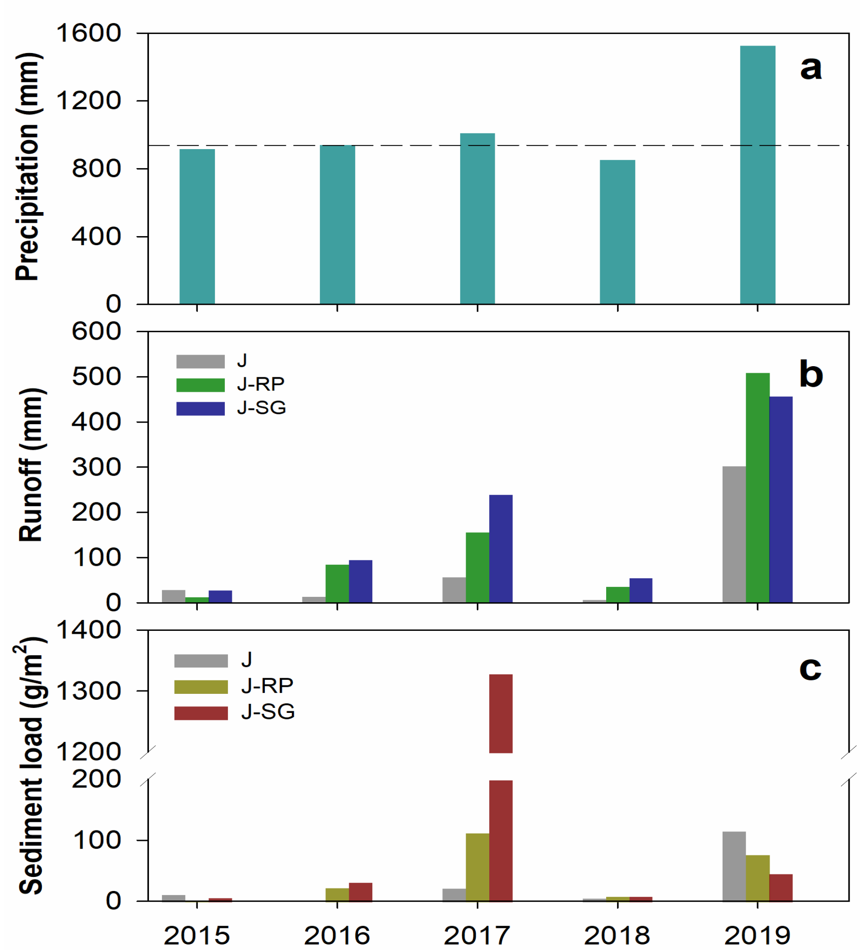

Figure 3.3 a) Annual precipitation during the water year 2015 through 2019 and the dashed line donates 30-year annual mean precipitation between 1981 to 2010 from the near Mesonet Marena station; b) Annual runoff depth from three watersheds; c) Annual sediment load from three watersheds.

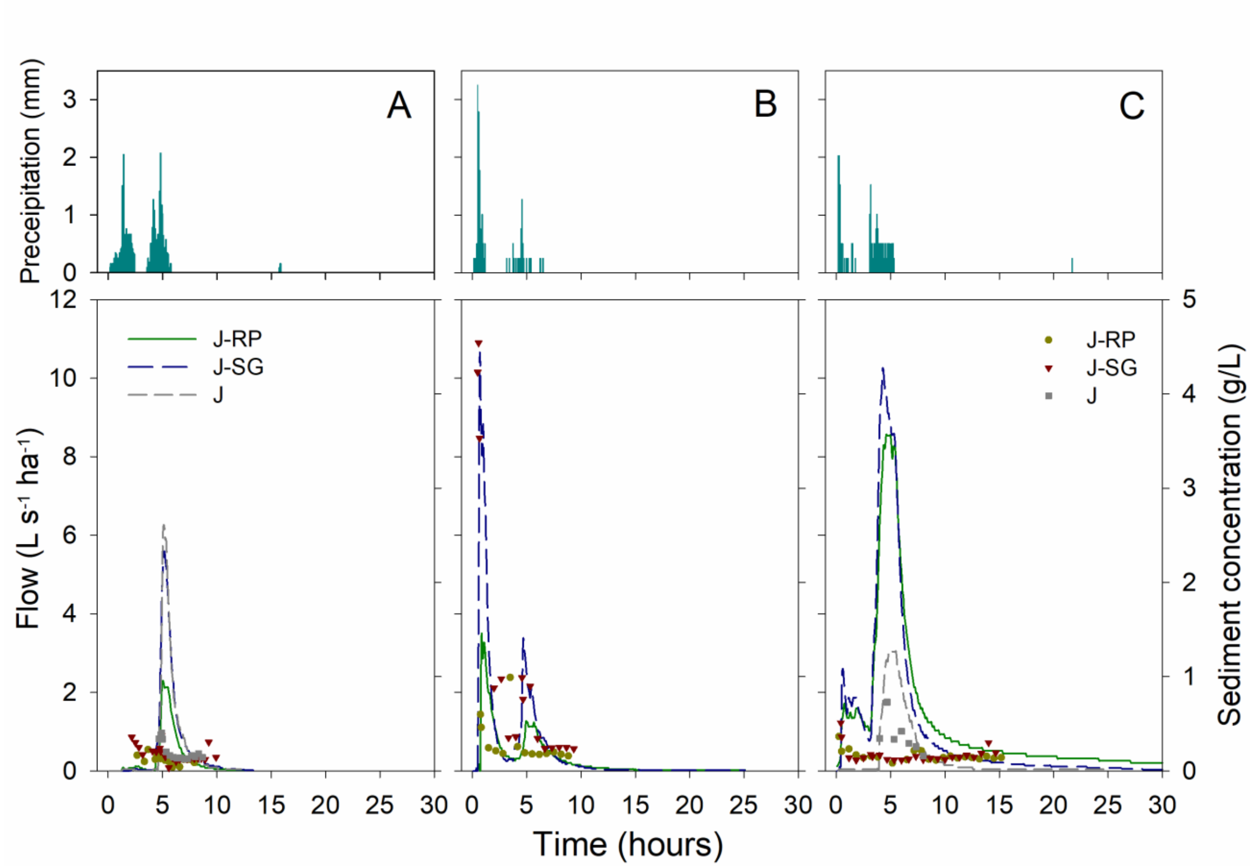

Figure 3.4 Comparison of flow rate, flow duration, and sediment concentration of control watershed J (Juniper), impact watershed J-RP (Juniper → Restored Prairie) and impact watershed J-SG (Juniper → Switchgrass) from a rainfall event of 30 mm on May 19th, 2015 during the calibration phase (A), a rainfall event of 20 mm on April 2nd, 2017 during the transition phase (B), and a rainfall event of 25 mm on May 3rd, 2019 during the alternative phase (C).

CHAPTER IV

HYDROLOGICAL IMPACTS OF CONVERTING WOODY ENCROACHED AND MARGINAL RANGELANDS TO SWITCHGRASS IN NORTH-CENTRAL OKLAHOMA

ABSTRACT

One of the significant constraints for the biomass-based biofuel industry is the availability of suitable land to produce biomass sustainably. Marginal rangelands constitute a large percentage of the potential area for conversion to switchgrass (*Panicum virgatum* L.) production in the south-central Great Plains. However, a significant barrier is the uncertainty of such conversion on hydrological impacts (such as streamflow and sediment yields). In this study, the Lower Cimarron River (LCR) basin's rangelands were categorized into different productivity classes based on the soil productivity index. The SWAT (Soil and Water Assessment Tool) was used to assess the impacts of four grassland conversion scenarios on water budgets and sediment load in the basin. These scenarios include I) conversion of existing juniper woodlands (*Junipers virginiana* L., eastern redcedar) to switchgrass; II) conversion of unproductive marginal rangelands to switchgrass; III) conversion of unproductive and moderately productive marginal rangelands to switchgrass; and IV) conversion of all rangelands to switchgrass. Results

showed that for the baseline model, mean annual precipitation, ET, streamflow, and baseflow was 766 $\pm$38 mm, 600 $\pm$13 mm, 74 $\pm$10 mm, and 47 $\pm$6 mm, respectively. Conversion of existing juniper woodlands, 3.7% of LCR, to switchgrass biomass production system (Scenario I) had limited impacts on water and sediment yields; however, a 2-3% increase in streamflow was predicted for the sub-basin with a juniper coverage of just around 14%. In contrast, the conversion of grasslands to switchgrass increased annual ET, which led to a decrease in streamflow and baseflow. Mean annual ET increased by 1.3%, 2.6%, and 3.5% leading to a decrease in annual streamflow by 5.4%, 10.8%, and 13.5% for scenarios II, III, and IV, respectively. Compared to the baseline model (270 $\pm$70 g m^{-2} of annual sediment yield), annual sediment yield decreased by 12.2%, 39.2%, and 61.6% for scenarios II, III, and IV. Monthly ET increased the most during the switchgrass growing season, concurred with the greatest reduction of streamflow during the same season. Results indicated that conversion of marginal rangelands to switchgrass based feedstock production systems could moderately decrease streamflow but substantially reduce the soil loss.

INTRODUCTION

The southern Plains states of Kansas, Oklahoma, and Texas include approximately 156.6 million acres of rangeland (National Agricultural Statistics Service, 2014), representing roughly 30 percent of the privately-owned grazing land in the United States. These rangelands primarily support ruminant livestock production. However, this livestock production system is under threat due to woody plant encroachment dominated by juniper species, which reduces herbaceous productivity and increases the risk of wildfire (Steiner *et al*., 2015). In central and western Oklahoma, juniper (*Junipers virginiana* L., eastern redcedar) cover has increased at an average annual rate of ~8% between 1984–2010 (Wang *et al.*, 2017).

Besides reducing the carrying capacity for livestock, juniper encroachment also alters the watershed hydrological function (Zou *et al.*, 2018). Conversion of rangeland to juniper woodland results in decreased soil moisture, surface runoff, and groundwater recharge in the moist grassland in the southcentral Great Plains (Zou *et al.*, 2014; Acharya *et al.*, 2018; Zou *et al.*, 2018). The increase of juniper trees (*Juniperus osteosperma* [Torr.] Little) in the rangelands of intermountain west of the USA was documented to increase sediment transport (Pierson *et al*., 2010). Sediment concentration in streams and reservoirs in the southcentral Great Plains is highly variable but generally high. High turbidity is a major water quality concern in the state of Oklahoma. If removing juniper woodlands or converting rangelands to switchgrass production systems leads to reduced sediment yield, they can be considered in restoring impaired watersheds.

Curtailing juniper encroachment is commonly considered a general objective in rangeland management in the Great Plains (Twidwell *et al.*, 2013). Policy and government programs generally support land management and land use which will remove juniper or curtail the expansion of juniper in rangeland. Prescribe fire is a cost-effective management tool and has been recommended and widely used to suppress juniper encroachment in rangeland. However, mechanical removal is a common practice to reclaim and restore rangelands where juniper height and fuel load make fire management ineffective (Stritzke and Bidwell, 1990).

Recent studies demonstrated that conversion from juniper woodland to grasslands increased the runoff and decreased the sediment yield at the experimental watershed scale (2 – 4 ha in area) (Zhong *et al*., 2020). After mechanical removal of juniper, the herbaceous vegetation can be established or re-established fairly quickly (Zhong *et al*., 2020), and so can the juniper seedlings if the repeated prescribed fire is not applied. Establishing switchgrass following juniper removal might be used as a proactive management approach to address woody encroachment and provide an alternative income for ranchers as biofuel production and bio-based economy develop (Link *et al.*, 2017). Switchgrass (*Panicum virgatum* L.) is a native species in the tallgrass prairie and is commonly used to reduce soil erosion (Wu and Liu, 2012; Feng *et al*., 2015). It is recommended as a dedicated species for feedstock production for biofuels (Parrish and Fike, 2005; Sanderson *et al.*, 2006). The annual harvest of switchgrass as feedstock can prevent the establishment of perennial woody species, such as juniper, into the site. A field study in the experimental watershed scale showed that switchgrass could be readily established after mechanical removal using a no-till drill with the herbicide application (Zhong *et al*., 2020). Zhong *et al*. (2020) found that converting juniper woodland to grassland generally improved soil moisture and increased runoff. The sediment yield from the switchgrass watershed was comparable to the un-treated juniper woodland. This suggests that switchgrass-based feedstock production system could be a potential, environmentally friendly land use alternative to utilize the juniper encroached rangelands as well marginal rangelands with limited livestock production potentials in the region.

The environmental impact associated with the switchgrass feedstock system has been evaluated in this region. In a modeling study, Wu and Liu (2012) estimated 1.2 – 3.2% decrease in water yield by converting native grassland to switchgrass in the Midwest U.S. Wang *et al*. (2020) reported 3.2 – 12.1% decrease in surface runoff and 43.7 – 95.5% decrease in soil loss by converting cropland to switchgrass in the Midwest United States using the Daily Erosion Project modeling system. Using SWAT, Yimam *et al*. (2017) found a 27.7% decrease in average annual streamflow after converting grassland to switchgrass in north-central Oklahoma. Reduction in surface runoff is widely promoted in the cropping system to reduce the loss of soil and nutrient in the Midwest United States. However, a substantial streamflow reduction may be undesirable in the semiarid arid rangelands in the southern and southcentral Great Plains. It could stress the aquatic ecosystem and water availability to livestock, ponds, reservoirs, and municipal water supplies.

Runoff and sediment responses to land use change can be directly quantified at the experimental watershed scale (Zou *et al.,* 2014; Zhong *et al.,* 2020), but it is difficult to extrapolate experimental watershed scale results to large watershed due to the patchy and sparse canopy covers characterizing the juniper encroachment in rangelands. Therefore, there is a need to systematically assess the hydrological impact of converting juniper woodland without or with surrounding grasslands to switchgrass biomass production on a large watershed scale. Marginal rangelands can be divided into different marginal classes based on the Soil Productivity Index (SPI) (Larson *et al.*, 1988; Schaetzl *et al.*, 2012). These classes of marginal rangelands and the existing juniper encroachment data were used to develop land use change scenarios and were used to model their associated impacts on basin hydrology in the Lower Cimarron River (LCR) basin, Oklahoma. Assessment of land use change on hydrological processes in large areas requires model simulations (Goldstein and Tarhule, 2015). Many models have been developed for ecological and hydrological assessments, such as Soil and Water Assessment Tool (SWAT) (Arnold *et al.*, 1998), Regional Hydro-Ecological Simulation System (RHESSys) (Tague *et al.*, 2004), Soil and Water Integrated Model (SWIM) (Krysanova *et al.*, 2005) and WEPP model (Flanagan *et al.*, 2007). Among these models, SWAT has been relatively widely used for agricultural and rangeland watersheds to assess the hydrological impacts of land-use change (Ghoraba, 2015; Zou *et al.*, 2016). SWAT is a spatially explicit GIS-based semi-distributed model and is ideal for watersheds with diverse land use and cover. It has been successfully used for areas ranging from experimental watersheds to river basins (Qiao *et al.*, 2015; Zou *et al.*, 2016; Starks and Moriasi, 2017).

The main objective of this study is to model the hydrological impacts of the conversion of juniper woodlands and marginal rangelands into switchgrass biomass production systems in the LCR basin by using the SWAT model platform.

METHODS AND MATERIALS

Study area

The Lower Cimarron River basin

The Lower Cimarron River (LCR) basin is located in north-central Oklahoma, United States (Figure 4.1a), with a total area of around 18,231 km^2. The LCR is comprised of three HUC-8 watersheds (the upper – HUC11050001, the middle – HUC11050002, and the lower – HUC11050003) with markedly different vegetation cover – grassland, cropland, and woodland, respectively. Historically, the basin was predominantly grassland, but the majority of the basin was converted to cropland with European settlement since the 1830s (Samson and Knopf, 1994). In the 1970s, the grassland area started to recover and had a high percentage among the major vegetation covers (Dale *et al.*, 2015). Lately, juniper cover has increased in the grassland because of fire exclusion (DeSantis *et al.*, 2011; Wang *et al.*, 2017).

Data acquisition and model implementation

SWAT version 2012/ Revision 670 (Arnold *et al.*, 1998) was used to assess the impacts of land-use land cover (LULC) change on streamflow in the LCR basin. There are six streamflow gaging stations managed by the United States Geological Survey (USGS) in the basin. The Ripley station (USGS # 07161450) was set as the basin outlet, which is drained by 87% (15,802 km^2) of the LCR basin (Figure 4.1a). The basin was delineated based on the 30-meter digital elevation model (DEM) (Gesch *et al.*, 2009), resulting in 27 sub-basins. The sub-basins area ranged from 6.95 to 1801.69 km^2 with an average area of 585.27 km^2. Then the sub-basins were overlaid with three different map layers: land cover, soil, and slope to generate hydrological response units (HRUs), the smallest building block of the SWAT model to estimate water, nutrient, and sediment routings. For the land cover layer, the vegetation map, including the spatial distribution of eastern redcedar, was obtained from the Oklahoma Department of Wildlife Conservation (Diamond and Elliott, 2015) and was merged with the National Land Cover Database (2011) (Homer *et al.*, 2015). The modeled basin, therefore, is comprised of 47.5% grassland, 37.3% cropland, 6.2% urban areas, 3.7% eastern redcedar woodland, 4.0% oak woodland, and 1.3% water. The basin soil properties were based on the SSURGO soil database obtained from the USDA web soil survey (USDA, 2011). The basin is comprised of 22.6%. 27.2%, 20.2%, and 30.0% hydrologic soil group A, B, C, and D, respectively. The basin was divided into three slope classes: 0-2%, 2-5%, and >5%, representing 61.4%, 32.6% and 6.0% of the basin area, respectively (Figure 4.1b). The unique combination of land, soil, and slope resulted in 2863 HRUs.

The model was then driven by the 20-year period (1999 – 2018) daily climate data, including precipitation, minimum temperature, and maximum temperature, obtained from the Oklahoma Mesonet climate data portal (Brock *et al.*, 1995). Twelve Mesonet stations were used to represent the spatial coverage of the basin (Figure 4.1a). For the days with missing values in the Mesonet data, daily climate data interpolated to each Mesonet station were obtained from the PRISM climate group (Daly *et al.*, 1997). In the 1999 – 2018 period, the basin received annual average precipitation of 776 mm, displaying a clear east-west precipitation gradient with 550 mm in the western part of the basin to about 900 mm in the eastern part (Figure 4.1a). Then, the model was run using the Hargreaves method (Hargreaves *et al.*, 1985) for potential evapotranspiration calculation, a variable storage coefficient method (Williams, 1969) for routing of water, and the modified Soil Conservation Service Curve Number (CN) method for surface runoff.

Streamflow calibration and validation

To calibrate and validate the LCR basin model, the calibration software, called SWAT-CUP (Abbaspour, 2013), was used for two different time periods: 2002-2010 for model calibration, and 2011-2018 for model validation. Also, to account for initial model stabilization and hydrological conditioning, a warm-up period of three years was used in calibration and validation. The regionalization approach was used to calibrate the model. For this, the uppermost contributing sub-basins were first calibrated and validated, followed by the lower sub-basins (Table 4.1) using 25 hydrological parameters that are considered important for simulating watershed evapotranspiration, surface runoff, and baseflow in SWAT (Table 4.2) (Chen *et al.*, 2016; Kharel *et al.*, 2016). The model simulated monthly streamflow data were compared with the monthly measured streamflow data obtained from USGS for five different streamflow monitoring locations within the basin (Table 4.1). Also, the simulated baseflow was compared with the baseflow derived from the measured USGS streamflow using the recursive digital filter baseflow separation method (Arnold and Allen, 1999; Eckhardt, 2005).

SWAT model performance

Model performance was evaluated by using three statistical measurements: percent of bias (PBIAS), the square of correlation coefficient (R^2 or ρ^2), and Nash–Sutcliffe Efficiency index (NSE) (Nash and Sutcliffe, 1970). PBIAS (equation 4.1) measures the average tendency of simulated data to be larger or smaller than the observed data (Gupta *et al*., 1999). Smaller PBIAS values close to zero are preferred. Values below and above zero indicate model overestimation and underestimation bias, respectively (Gupta *et al*., 1999). R^2 (equation 4.2) has a range from 0 to 1, with 1 indicating a perfect relationship between the simulated and observed variables. NSE (equation 4.3) is a normalized statistic method to estimate the relative magnitude of the residual variances between the measured and simulated data. The NSE value ranges from -∞ to 1 with the value of 1 corresponding to a perfect match between the observed and simulated data. According to the performance ratings provided by Moriasi *et al*., 2007, model performance is good when NSE is greater than 0.65 and PBIAS < ±15%, and very good when the NSE is > 0.75 and PBIAS < ±10% (Moriasi *et al.*, 2007).

$$\text{PBIAS} = \frac{\sum_{t=1}^{T}(y_t - \hat{y}_t)}{\sum_{t=1}^{T} y_t} \times 100\% \tag{4.1}$$

$$\rho = \sqrt{R^2} = \frac{Cov(Y,\hat{Y})}{\sqrt{\text{Var(Y)Var}(\hat{\text{Y}})}} \tag{4.2}$$

$$\text{NSE} = 1 - \frac{\sum_{t=1}^{T}(y_t - \hat{y}_t)^2}{\sum_{t=1}^{T}(\hat{y}_t - \bar{y})^2} \tag{4.3}$$

where Y is the observed variable and $\hat{Y}$ is the simulated variable. y_t is the observed data (t = 0, 1, 2 … T) and $\hat{y}_t$ is the simulated data (t = 0, 1, 2 … T).

Land use change scenarios, implementation

In this study, four land use change scenarios (maps) based on the juniper encroachment map (Diamond and Elliott, 2015) and soil productivity data were developed for the basin and compared with the baseline scenario for any changes in evapotranspiration, streamflow, baseflow, and sediment load. The level of productivity of rangeland within the basin was first estimated using the Soil Productivity Index (SPI) based on the USDA-NRCS soil taxonomic database (Schaetzl *et al.*, 2012). This database provides the soil productivity capability of the land in the U.S. based on 20 ranked categories of productivity, with the rank of 0 being the least productive to 19 being the most productive. The primary variables used in the SPI classification are based on soil taxonomy, such as organic matter content, cation exchange capacity, and clay mineralogy. For this study, the SPI was grouped into three categories: unproductive rangeland (UR) with lower levels of productivity (0 – 7), moderately productive rangeland (MR) with mid-levels of productivity (8 –12), and highly productive rangeland (HR) with higher levels of productivity (13 – 19). Then, the SPI layer was overlaid with the basin land layer to generate land classes with three productivity levels. This process led to the new classification of the basin rangeland into three classes: unproductive rangeland (11.3%), moderately productive rangeland (21.5%), and highly productive rangeland (14.7%). The four scenarios included in this study are I) conversion of juniper to switchgrass (J-SG); II) conversion of unproductive rangeland to switchgrass (UR-SG); III) conversion of unproductive and moderately productive rangelands to switchgrass (URMR-SG); and IV) conversion of all rangelands to switchgrass (R-SG).

The four land use change scenarios (maps) were fed into the calibrated and validated model one at a time with their associated parameter values for redcedar and Alamo switchgrass obtained from Qiao *et al*. (2015) and Starks and Moriasi (2017). Then, the model was run to generate evapotranspiration, streamflow, and sediment yield for each scenario.

RESULTS

Observed monthly mean streamflow varied from 0 to 60 $m^3 s^{-1}$ for upper land gauges Waynoka and Lovell and from 0 to 500 $m^3 s^{-1}$ for downstream gauges Dover, Guthrie, and Ripley (Figure 4.1 and 4.2). Simulated monthly mean streamflow matched well with the observed monthly mean streamflow from five streamflow gauges. It generally captured all peak flows, baseflow, and the streamflow variation trend from the observed data (Figure 4.2). The values of PBIAS, NSE, and R^2 from the calibration and validation period for all five gauges were <10%, >0.76, and >0.77, respectively (Figure 4.2). For the simulated baseflow, PBIAS was 9.1%, R^2 was 0.76, and NSE was 0.75 at the basin outlet. With reference to the Moriasi (2007) recommended values, the performance of this model was deemed very good. Therefore, it can be assumed that the model could be used to estimate monthly and annual streamflow for testing and evaluating different land use change scenarios in the LCR basin. Although the model was not calibrated and validated for sediment yield due to the lack of sediment yield data in the basin, the annual sediment yield, as estimated by the model, was also presented here to provide a general reference for comparing the impact of different land use scenarios on sediment yield.

Average annual values of evapotranspiration, streamflow, baseflow, and sediment yield resulted from the four land use change scenarios were compared with the baseline condition where no land use change was imposed. Scenario I (J-SG), in which existing juniper woodlands occupying 3.7% of the basin were replaced with the switchgrass, showed negligible impacts in water budget and sediment yield at the LCR basin scale (Table 4.3). However, for sub-basin (#19) with the highest juniper presence, removal of existing juniper woodlands occupying 14% of the sub-basin resulted in an overall increase in streamflow (2.4%) with no detectable change in sediment yield (Table 4.4). In the other three scenarios (II – IV) where switchgrass was planted in rangelands with different levels of productivity, an increase of ET and a decrease of streamflow, baseflow, and sediment load, were projected compared to the baseline scenario (Table 4.3). As the conversion area increased, the absolute change percentage increased. Compared to the baseline scenario, average annual ET increased by 1.3%, 2.6%, and 3.5% leading to a decrease in streamflow by 5.4%, 10.8%, and 13.5% for scenario II (UR-SG), III (URMR-SG), and IV (R-SG), respectively. Average annual baseflow had a similar decrease trend for all scenarios (Table 4.3). The difference in annual sediment yield between the baseline and scenario I (J-SG) was minimal. However, annual sediment yield decreased by 12.2% in scenario II, 39.2% in scenario III, and 61.6% in scenario IV (Table 4.3).

The impact on the ET and streamflow varied among the months. After converting juniper woodland to switchgrass biomass production (scenario I), the mean monthly ET and streamflow had limited change (Figure 4.3). After converting grassland to switchgrass (Scenarios II – IV), a seasonal response to the water budget was observed (Figure 4.3). Mean monthly ET increased during the growing season from May to August, and it decreased from September to December. The greatest increase of monthly ET was in June (4.1% for scenario II, 8.6% for scenario III, and 11.8% for scenario IV). Increased ET in summer led to decreased streamflow with the greatest decrease of monthly ET by 11.0% in scenario II (in September), by 21.0% in scenario III (in August), and by 27.9% in scenario IV (in August). The impact on the sediment yield varied among the months as well, and the mean monthly sediment yield decreased the most in fall. In September, it decreased by 56.6% for scenario II, 75.6% for scenario III, and 84.2% for scenario IV.

DISCUSSION

Hydrological impacts of converting encroached juniper land to switchgrass

One of the critical challenges in watershed studies and watershed management is understanding the paradox of scale (Wilcox *et al*., 2006). Removing nearly 100% juniper cover and converting to switchgrass biomass production at the experimental watershed produced significant runoff and sediment responses (Zou *et al*., 2014; Zhong *et al*., 2020), but in the current study, converting less than 4% of the basin with juniper to switchgrass production showed negligible impacts on annual water budget and sediment load on the basin scale. However, a 2-3% increase in streamflow was predicted for the sub-basin with a juniper coverage of just around 14%. These results partially explain why isolated shrub control efforts sometimes fail to augment streamflow on the basin scale (Wilcox *et al.*, 2003). In addition, it suggests that juniper removal solely for water resource consideration may not be justified for the LCR basin at this point. The effect of early encroachment on water resources may be negligible at the basin scale; however, the risk of doing nothing can be high. Complete conversion of the rangelands to juniper woodlands could result in reductions of up to 40% in annual streamflow for the drier, upper portion of the basin, and approximately 20% for the entire basin (Zou *et al*., 2016). Early control of juniper encroachment using fire should be encouraged. Alternative land use may be explored for low productivity, marginal rangelands, which are more vulnerable to woody plant encroachment.

Hydrological impacts of converting marginal rangeland to switchgrass

Converting marginal rangelands to switchgrass had significant impacts on the water budget in the LCR basin. Average ET increased the most during the summer, leading to decreased streamflow and baseflow. The variation of ET was similar to previous researches that converted grassland to switchgrass in one of the upper sections of the LCR basin (Goldstein and Tarhule, 2015; Yimam *et al.*, 2017). A decrease in streamflow and baseflow may lead to water stress for aquatic ecosystems and municipal water use, especially during the drought years in north-central Oklahoma (DeSantis *et al.*, 2011). Since the late spring and early summer are usually the high flow seasons in this river basin, and reduction in streamflow in this period may have less impact on water resources. However, Yimam *et al*. (2017) showed the greatest change of streamflow occurred in winter rather than the summer, and further research is needed to understand the change of streamflow regime in response to the conversion of rangelands to switchgrass production systems.

The basin-wide average annual sediment load of 270 $\pm$70 g m^{-2} under the current land use was much lower than the mean annual soil loss estimated for the Midwest U.S. (400 to 700 g m^{-2} from regional models (Wu and Liu, 2012). However, it is still over the upper limit of the rate of tolerable soil loss to sustain soil resources in the long term (20 to 200 g m^{-2} yr^{-1}) (FAO 2019). Conversion of unproductive and moderately productive rangelands to switchgrass was predicted to reduce the basin level sediment yield to 164 g m^{-2}, accounting for a 39.2% reduction in the total sediment yield. Average annual sediment yield substantially decreased to 104 g m^{-2} by converting all rangelands to switchgrass. This decrease in sediment yield can be attributed to the reduced surface runoff as sediment loading is highly related to streamflow discharge in the streams (Dodds and Whiles, 2004). The desynchronization of ET and runoff impacts after converting marginal rangelands to the switchgrass production suggests that soil moisture dynamics may play an essential role in regulating the hydrological processes in this basin. Further studies are needed to understand the evapotranspiration and soil moisture dynamics associated with the land use change and how these changes alter surface runoff, subsurface flow, and sedimentation processes.

CONCLUSION

Conversion of currently existing encroached juniper woodland occupying 3.7% of the LCR basin to switchgrass biomass production was predicted to have negligible impacts on the basin-scale water budget and sediment yield. Conversion of marginal rangelands of the LCR basin into switchgrass biomass production system was predicted to increase ET leading to a reduction in streamflow and baseflow, primarily in the summer months, with a substantial reduction in annual sediment yield. Switchgrass-based feedstock production systems could be considered a potential land use alternative to address the juniper encroached grassland or marginal rangelands with limited livestock production potentials but vulnerable to woody plant encroachment in the southcentral region of the Great Plains.

Table 4.1 Streamflow stations with the USGS station number and its contribution sub-basins in the LCR model.

USGS station and code	Sub-basin location	All contributed sub-basins	Unique contributed sub-basins
Waynoka (07158000)	6	1,2,3,4,5,6	1,2,3,4,5,6
Dover (07159100)	18	1-9,12,14-18	7-9,12,14-18
Lovell (07160500)	10	10	10
Guthrie (07160000)	22	1-9,12,14-18, 20, 22-27	20,22-27
Ripley (07161450)	19	1-27	11,13,19,21

Table 4.2 The 25 selected hydrological parameters and their descriptions. “r” stands for relative change or multiplication, and “v” stands for replacement.

Parameter Name	Description
r_CN2.mgt	Runoff curve number
v_ESCO.hru	Soil evaporation compensation factor
r_SOL_AWC(..).sol	Available water capacity of the soil layer
v_CH_K1.sub	Effective hydraulic conductivity in tributary channel alluvium
r_SLSUBBSN.hru	Average slope length
v_EPCO.hru	Plant uptake compensation factor
r_HRU_SLP.hru	Average slope steepness
v_ALPHA_BNK.rte	Baseflow alpha factor for bank storage
v_DEEPST.gw	Initial depth of water in the shallow aquifer (mm)
v_GW_DELAY.gw	Groundwater delay (days)
v_RCHRG_DP.gw	Deep aquifer percolation fraction
v_REVAPMN.gw	Threshold depth of water in the shallow aquifer for “revap” to occur (mm)
v_GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to occur (mm)
v_SHALLST.gw	Initial depth of water in the shallow aquifer (mm)
v_SURLAG.hru	Surface runoff lagtime (days)
v_CH_N1.sub	Manning’s “n” value for tributary channels (m)
r_CH_S2.rte	Average slope of main channel
v_DEP_IMP.hru	Depth to impervious layer for modeling perched water tables
v_CH_N2.rte	Manning’s “n” value for the main channel
v_GW_REVAP.gw	Groundwater “revap” coefficient
v_ALPHA_BF.gw	Baseflow alpha factor (days)
r_OV_N.hru	Manning’s “n” value for overland flow
v_DIS_STREAM.hru	Average distance to stream (m)
v_CH_K2.rte	Effective hydraulic conductivity in main channel alluvium
r_CH_S1.sub	Average slope of tributary channels

Table 4.3 Mean annual precipitation and evapotranspiration (ET), streamflow, baseflow (in mm, mean $\pm$S.E.), and annual sediment yield (in g m^{-2}, mean $\pm$S.E.) in the LCR basin during the model simulation period (2002 – 2018) under the baseline, scenario I (juniper woodland to switchgrass: J-SG), scenario II (unproductive rangelands to switchgrass: UR-SG), scenario III (unproductive and moderately productive rangelands to switchgrass: URMR-SG), and scenario IV (all rangelands to switchgrass: R-SG) along with the area converted (km^2).

Scenarios	Precipitation	ET	Streamflow	Baseflow	Sediment yield	Converted area
Baseline	766 $\pm$38	600 $\pm$13	74 $\pm$10	47 $\pm$6	270 $\pm$70	0
Scenario I	766 $\pm$38	599 $\pm$12	75 $\pm$10	48 $\pm$6	271 $\pm$71	585
Scenario II	766 $\pm$38	608 $\pm$13	70 $\pm$10	44 $\pm$6	237 $\pm$71	2366
Scenario III	766 $\pm$38	616 $\pm$13	66 $\pm$9	42 $\pm$6	164 $\pm$50	5762
Scenario IV	766 $\pm$38	621 $\pm$14	64 $\pm$9	41 $\pm$6	104 $\pm$28	8083

Table 4.4 Mean annual precipitation and evapotranspiration (ET), streamflow (mm, mean ± S.E.) and annual sediment yield (g m^{-2}, mean ± S.E.) in sub-basin (#19) with the highest juniper cover percentage (14%) in the Lower Cimarron River basin during the model simulation period (2002 – 2018) for the baseline simulation and scenario I (converting juniper to switchgrass: J-SG).

Scenarios	Precipitation	ET	Streamflow	Sediment yield
Baseline	872 ± 47	656 ± 16	205 ± 30	619 ± 177
Scenario I	872 ± 47	651 ± 16	210 ± 30	620 ± 183

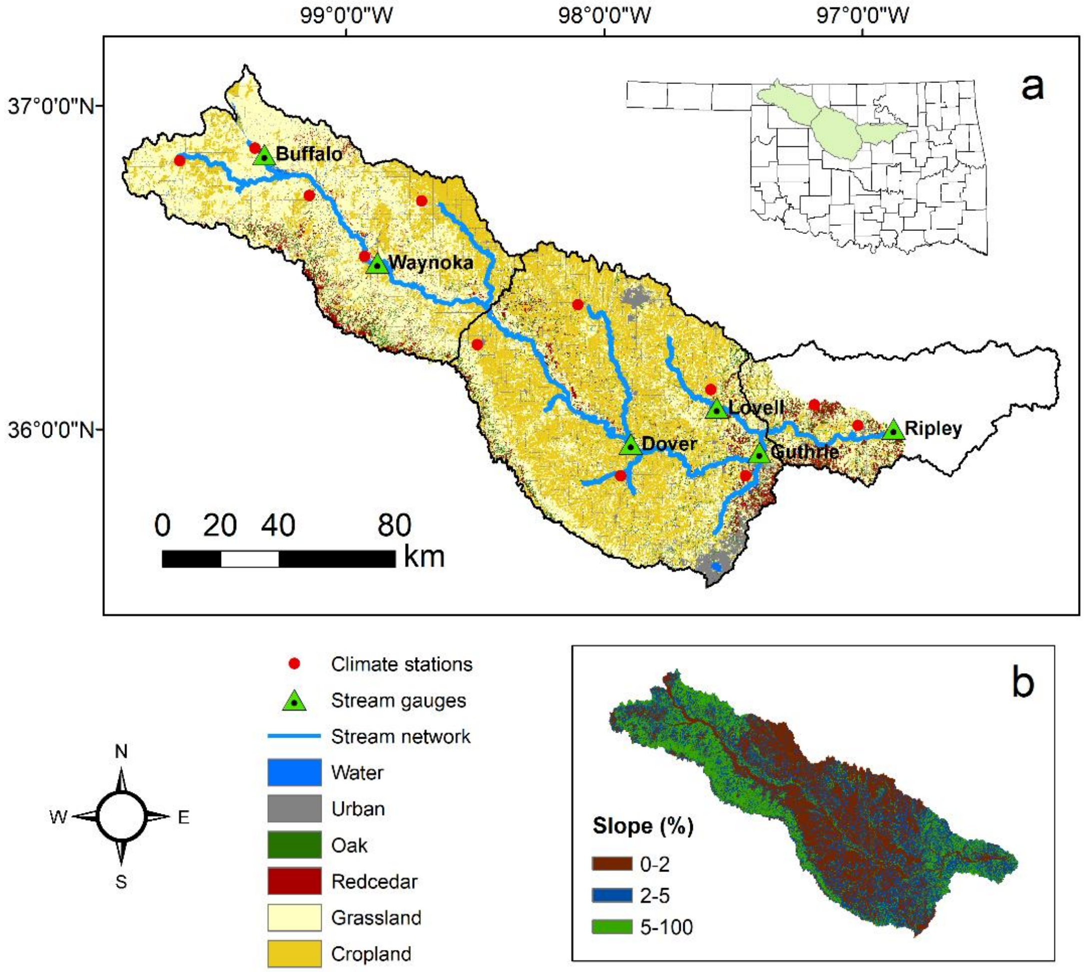

Figure 4.1 Land cover and land use, locations of streamflow gauges and climate stations (a), and spatial distribution of slope categories (0-2%, 2-5% and >5%) (b) of the Lower Cimarron River basin, north-central Oklahoma, USA.

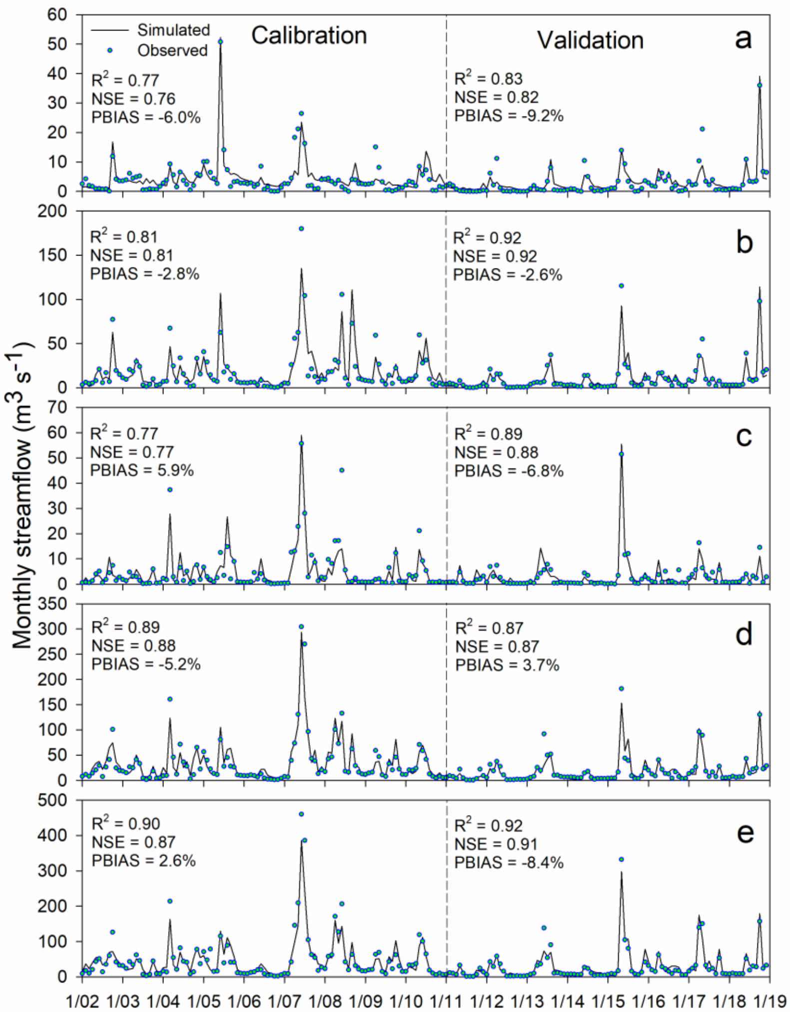

Figure 4.2 Comparison of observed and simulated monthly mean streamflow at Waynoka (a), Dover (b), Lovell (c), Guthrie (d), and Ripley (e) during calibration (2002–2010) and validation (2011–2018) in the Lower Cimarron River basin, north-central Oklahoma, USA.

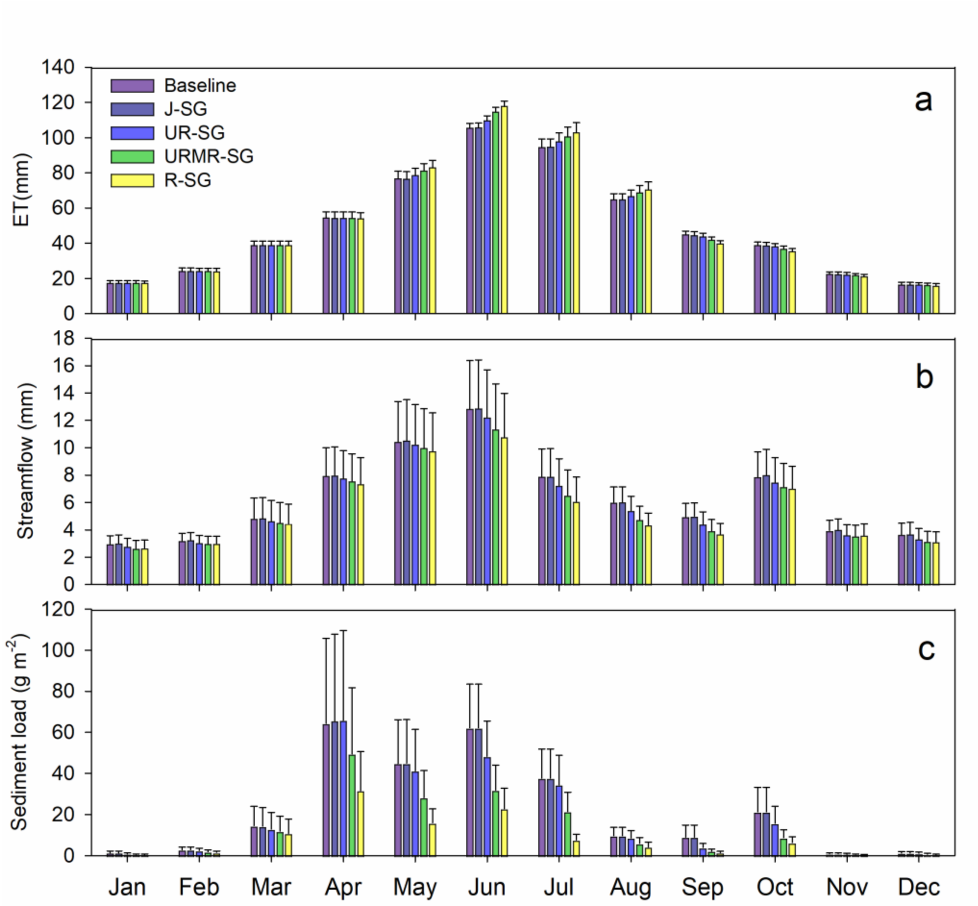

Figure 4.3 Average monthly evapotranspiration (ET) (a), streamflow (b), and sediment load (c) during the model simulation period (2002 – 2018) for the baseline and four land use change scenarios in the Lower Cimarron River basin, north-central Oklahoma, USA.

CHAPTER V

GENERAL CONCLUSION

In this study, at the experimental watershed scale, the study documented that mechanical removal of juniper trees increased watershed soil moisture storage and decreased the threshold of precipitation to generate surface runoff. Converting juniper woodland to prairie resulted in a four to five-fold increase in total runoff at the experimental watershed scale in the southcentral Great Plains. In addition, after the juniper removal and before grassland was fully established, the watersheds were vulnerable to sedimentation processes, and pulsed increases in sediment load in response to site disturbance and storm events were documented. However, sediment load declined considerably and returned to a level similar to the juniper woodland one year after the grasslands were established. The switchgrass production system was more effective than naturally restored prairie in reducing the total sediment yield. At the regional scale, conversion of existing juniper woodlands, which occupy approximately 3.7% of the LCR basin, to switchgrass biomass production was predicted to have negligible impacts on the basin-scale water budget and sediment yield. The conversion of marginal rangelands of the LCR basin into switchgrass biomass production system was predicted to increase ET leading to a reduction in streamflow and baseflow, primarily in the summer months, with a substantial reduction in annual sediment yield.

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Graduate Teaching Assistant, Department of Natural Resource Ecology and Management, Oklahoma State University, Stillwater, Oklahoma, 2018-2020.

Professional Memberships:

American Geophysical Union (2019)
Ecology Society of America (2018)
American Meteorological Society (2018)