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Response of sediment concentration and load to removal of juniper woodland and subsequent establishment of grasslands – A paired experimental watershed study

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A R T I C L E I N F O

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A B S T R A C T

Encroachment of woody plants into grasslands is a global phenomenon of environmental concern. Mechanical removal is often necessary to re-establish herbaceous dominance for heavily encroached watersheds, but its impact on water quality and quantity of runoff into streams and reservoirs has not been vigorously studied. The sediment concentration and load following mechanical removal of juniper (*Juniperus virginiana*, L.) woodland and subsequent re-establishment of tallgrass prairie or switchgrass (*Panicum virgatum* L.) biomass production were quantified at the experimental watershed scale in the south-central Great Plains, USA. Impact analysis was used to evaluate the effects of watershed treatment and phase of land use conversion. The annual sediment yield from juniper woodland watersheds averaged <0.10 t ha^{-1} yr^{-1} before treatment and increased to 0.28 t ha^{-1} yr^{-1} after juniper were cut and left on site. In the second year, removing the dried trees increased the annual sediment yield to 1.14 t ha^{-1} yr^{-1} in the prairie restoration watershed and 13.29 t ha^{-1} yr^{-1} in the switchgrass watershed that was sprayed with herbicide in preparation for no-till planting. The annual sediment loads averaged 0.44 t ha^{-1} yr^{-1} from the restored prairie and 0.29 t ha^{-1} yr^{-1} from the established switchgrass, comparable to 0.73 t ha^{-1} yr^{-1} from the intact juniper woodland during the third and fourth years after initiation of treatments. While restored grassland watersheds had elevated peak flows and longer flow duration leading to greater runoff, lower sediment yields were due to reduced mean and peak sediment concentration compared to the juniper watershed. Therefore, restoring juniper woodland to native prairie or switchgrass biomass production systems may increase water yield without increasing sediment yield, especially in years with extreme storm events.

1. Introduction

Rangeland for cattle production is the primary land use in southern Great Plains, USA (Collins et al., 2014; Sohl et al., 2012), 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 by providing water for both municipal and livestock supplies (Berg et al., 2016a,b; Wine et al., 2012a; Zou et al., 2018). However, this region is characterized by relatively rapid land use and vegetation change and experienced substantial erosion related to row-crop agriculture and the Dust Bowl of the 1930 s (Baveye et al., 2011). Site disturbance and land use change can 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; Wine et al., 2012b).

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 (Munoth and Goyal, 2020; Wilcox et al., 1990). Reduction in herbaceous vegetative cover often leads to increased overland flow and sediment transport in grasslands (Belnap and Gillette, 1998; Field et al., 2011; Urgeghe et al., 2010). As a result, excessive loss of herbaceous vegetative cover from overgrazing, fire, and woody plant encroachment will exacerbate surface runoff and soil erosion (Field et al., 2011; Menzel et al., 1978;

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Pierson et al., 2007; West et al., 2016).

Woody plant encroachment is a phenomenon of global concern (Archer et al., 2017), which reduces the herbaceous vegetation, especially in ecosystems where competition for light and water is intense (Feltrin et al., 2016). An increase of juniper trees (*Juniperus osteosperma* [Torr.] Little) in the Intermountain West of the USA led to significant increases in overland flow and sediment transport down hillslopes (Pierson et al., 2007, 2010; Williams et al., 2018, 2020). 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 within the intercanopy areas of juniper woodlands (Pierson et al., 2010; Williams et al., 2020). Juniper removal in that system stimulated herbaceous plant recovery 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 reduced soil surface exposure to rain splash and decreased sediment transport (Cline et al., 2010).

Further east in the tallgrass prairie ecoregion, juniper (*Juniperus virginiana* L., eastern redcedar) reduced herbaceous cover mainly under their canopies during the early stages of encroachment (Limb et al., 2010). The economic losses of this encroachment were estimated at $218 million in 2001 for Oklahoma alone, with nearly half of the projected losses attributed to the reduction of forage yield, followed by catastrophic wildfires, loss of lease hunting, and reduced water yield (Drake and Todd, 2002). The rate of soil erosion under woody encroachment conditions is relatively low compared with soil erosion from the row crop systems. However, the level of sediments in the runoff could be great enough to result in high turbidity and water quality issues in streams and reservoirs (Dodds and Whiles, 2004). Mechanical removal of juniper trees resulted in a rapid recovery of prairie vegetation (Schmidt et al., 2021) and a substantial increase of runoff at the experimental watershed scale (Zhong et al., 2020). However, the sediment concentration and load at the edge of the hillslope or watershed outlet following mechanical removal of juniper trees and re-establishment of grasslands remains mostly unquantified.

Switchgrass (*Panicum virgatum* L.) is a native species of the tallgrass prairie and is widely used to control soil erosion in watershed management (Feng et al., 2015; Wu and Liu, 2012). It is also a United States Department of Agriculture (USDA) dedicated species for biofuel production, partially due to its high potential for reducing soil erosion (Wright., 2007; Wullschleger et al., 2010). While switchgrass production can improve water quality in marginally productive croplands (Acharya et al., 2019), no manipulative experiments have directly evaluated the water quality impact following converting juniper encroached rangelands to switchgrass production systems at the watershed scale. In practice, herbicides are widely used to kill herbaceous vegetation for establishing switchgrass using no-till drilling in this region. Due to this site preparation, the pulsed responses of sediment concentration in runoff and sediment load require thorough assessment.

Prairies are often dominated by intermittent streams with generally low sediment concentrations. However, sediment concentrations can increase by 3- to 12-fold in response to large rainfall events 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 prairie-dominated regions (Blanchard et al., 2011; Dodds and Whiles, 2004). The event-based sediment concentration, the peak sediment concentration, and sediment load measured at the outflow of upland watersheds are the most direct assessment of the effects of land use and management practices on sediment transport to streams (Grum et al., 2017) and stream turbidity (Rasmussen et al., 2009).

An opportunity may exist in the southern Great Plains to supply feedstock for a vibrant cellulosic biofuel industry while also enhancing ecosystem services, particularly water resources in marginal cropland and rangelands (Wagner et al., 2017). However, much of these lands have been under rapid conversion to woody cover, particularly juniper species. *Juniperus virginiana* has encroached into grasslands over large areas in Oklahoma (approximately 5 million ha) and across millions of hectares of Texas, Kansas, and Nebraska (Smith and Johnson, 2003; Kaskie et al., 2019). This conversion is detrimental to the ecological and economic value of the land, reducing ecosystem water provisioning in particular (Zou et al., 2018). Therefore, the integration of woody biomass into the biofuel production strategy is needed and would likely add value by restoring degraded rangeland while sustaining or increasing the water resources for this region. In addition, management of woody plant encroachment is a regional issue in the Great Plains and common concern in the rangeland in the west USA and rangelands in Africa, Asia, and South America.

Given the history of the region, including the Dust Bowl or the 1930 s, people are very concerned with the soil erosion associated with vegetation removal at the hillslope or watershed scale. These concerns arise partly due to the lack of manipulative, statistically vigorous, multi-year studies to capture the climate variability critical for understanding runoff and sediment processes. Also, site feasibility and financial constraints limit research using replicates at the landscape or watershed level. The objectives of this study were to quantify the impact of mechanical removal of juniper woodland and subsequent natural recovery to prairie or the establishment of a switchgrass stand on sediment concentration and loads at the watershed scale. The results were based on a paired experimental watershed study (Clausen and Spooner, 1993). This study lasted five years from 2014 to 2019, including three sequential phases: Calibration, Transition, and Alternative (Table 1). The Calibration was the pretreatment period when all watersheds were heavily encroached by juniper. The Transition phase included tree removal and grassland recovery for one watershed or herbicide application and switchgrass planting for the other watershed. The Alternative phase was defined as when the prairie recovered or the switchgrass was established in the impact watersheds. A Before-After Control-Impact (BACI) analytical approach was used to test the interactive effect of the watershed and the three phases and contrast the sediment metrics in the Alternative phase. This is the first watershed scale manipulative study quantifying sediment concentration and load response to mechanical removal of encroached juniper woodland in tallgrass prairie.

2. Materials and methods

2.1. Study area

The research was conducted in three juniper-encroached watersheds at 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) (Fig. 1). Most of the area covered by these watersheds was cultivated after the 1889 Land Run to grow crops and later returned to grass-dominated ecosystems in the 1940 s (Booth, 1941; Lewis, 1989). The main soil types in this study area include Stephenville-Darnell

Table 1
Timeline of treatments for the watershed J-RP (juniper to restored prairie) and the watershed J-SG (juniper to switchgrass) from water years 2015 through 2019.

Phase	Time	J-RP	J-SG
Calibration	Oct. 2014 – Jun. 2015	Pretreatment	Pretreatment
Transition	Jul. 2015 – Jan. 2016	Tree cut, dried on site	Tree cut, dried on site
	Feb. 2016 – Mar. 2017	Tree removal, recovery to prairie	Tree removal, herbicide spray
	Apr. 2017 – Sep. 2017	Recovery to prairie	Planting and establishing switchgrass
Alternative	Oct. 2017 – Sep. 2019	Restored prairie	Established switchgrass

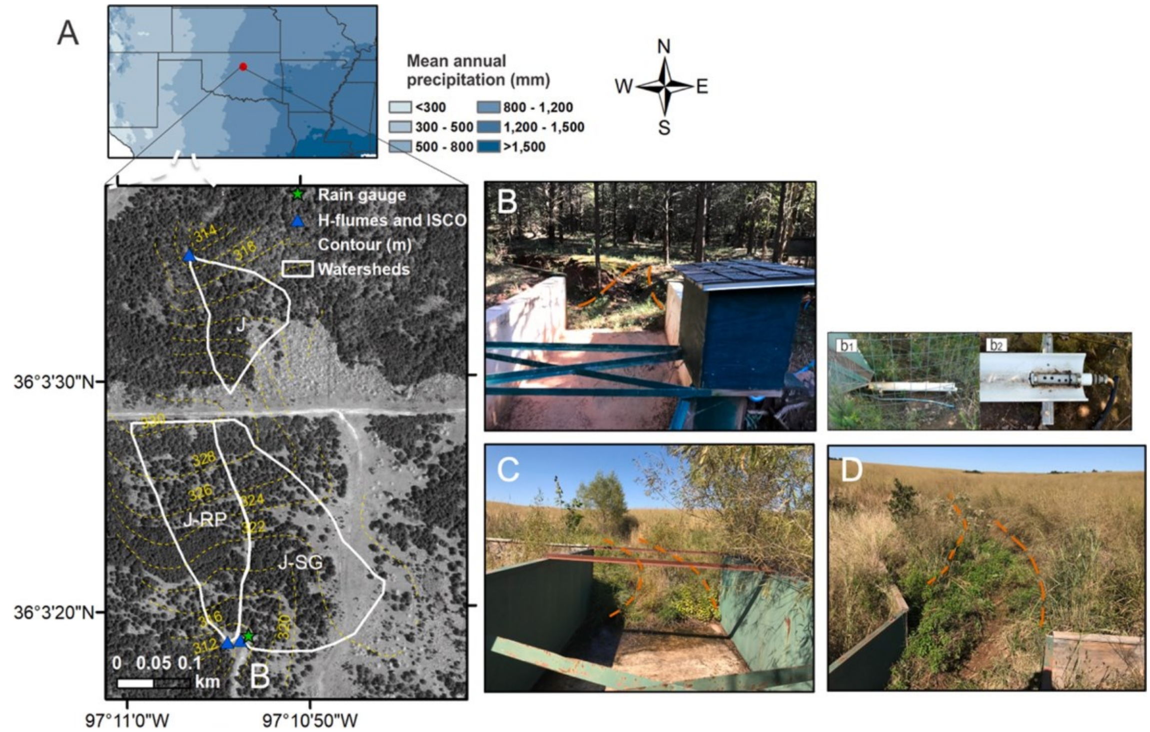

Fig. 1. The three experimental watersheds in OSU-RRS, north-central Oklahoma, USA. The aerial photo was taken before treatment (Google Earth, February 2014). The contour lines were generated by 2 m resolution Lidar data (A). Juniper control watershed (J) is to the north (B). The restored prairie watershed (J-RP) (C) is adjacent to the switchgrass watershed (J-SG) (D). The location of troughs relative to H-flume discharge (b1); the strainer's location within trough (b2). The orange dash lines illustrate the well-vegetated, ephemeral stream channels approaching H flumes.

complex, Renfrow and Grainola, Coyle and Zaneis soil, and Grainola – Lucien complex. All four soil types have loam soils in the top 10 cm, varying from fine sandy loam in Stephenville-Darnell complex, silt loam in Renfrow and Grainola, loam in Coyle and Zaneis soil, and clay loam in Grainola –Lucien complex. The areas were 1.3, 2.6, and 3.8 ha for the juniper control watershed (J), the restored prairie watershed (J-RP), and the switchgrass watershed (J-SG), respectively (Fig. 1). The J watershed includes Coyle and Zaneis soils (55.75%), Stephenville–Darnell complex (22.42%), and Grainola –Lucien complex (21.83%). The J-RP watershed includes primarily Stephenville-Darnell complex (75.42%), Renfrow and Grainola soil (11.86%), and Grainola –Lucien complex (3.14%). The J-SG watershed is composed of Renfrow and Grainola soil (33.87%), Stephenville-Darnell complex (30.39%), and Grainola –Lucien complex (8.45%). The slopes of the watersheds are from 0 to 5%. The average soil depth is < 1 m underlain by sandstone substrates (Zou et al., 2014) and all watersheds are well drained.

2.2. 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 (Smith, 2002; Underwood, 1992), such as the change in precipitation pattern (Brown et al., 2005), which also directly affect runoff and sediment load (Stewart-Oaten et al., 1986). The precipitation in the study region has very high temporal variation. The paired experimental watershed approach using BACI was specifically selected to address the variability of rainfall and other climatic variabilities. In the study, all three watersheds were initially heavily encroached by juniper (Fig. 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 Alternative phase following the initial Calibration phase (Table 1). Analysis of variance (ANOVA) using a linear mixed-effect model for the BACI design evaluated the main effects from watershed treatment and phase of land use conversion (or phase) and 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 and 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) (Fig. 1). The other Impact watershed was further treated with glyphosate herbicide during 2016 to prepare the site for planting. The lowland 'Alamo' switchgrass cultivar was seeded at a rate of 7.8 kg ha^{-1} and depth of 0.64 cm using a Truax no-till drill machine in April 2017. This watershed was assigned as the "juniper to switchgrass" (J-SG) (Fig. 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 years (2018 and 2019) following the Transition phase (Table 1). Switchgrass was cut at approximately 10 cm in height, baled, and removed from the J-SG watershed every November.

2.3. Precipitation and runoff

Precipitation was measured using a tipping bucket rain gauges (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. In these watersheds, the runoff was mainly from overland flows in direct response to storm events. As a result, it was important to separate one storm from another when there are multiple rainfall events in a day. For most of the storm flow, the runoff ended < 6 h after the storm event. A precipitation event was considered completed when there was no further precipitation reading for a minimum of 6 h. The definition and separation of a runoff event and associated values were described in Zhong et al. (2020).

2.4. 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) (Fig. 1) to analyze total suspended solids. Runoff samples were collected using an intake strainer 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 (Fig. 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, 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, allowing better characterization of flashy versus long-duration runoff events. The sediment concentrations were gap-filled using the 30- or 40-minute concentration data 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 °C using a VWR Horizontal Air Flow Oven for a minimum of 72 h. Then samples were placed in a desiccator to prevent any atmospheric moisture from re-entering the samples as they cooled.

The event-based sediment load included all suspended solids accumulated from all sediment values calculated for each 5-minute interval during each runoff event. The unit area sediment load (g m^{-2}) was calculated using the area of each watershed. For instance, when sediments gradually build up on the H-flume floors after multiple runoff events, this deposit was shoveled into buckets and weighed in the lab. This load was not included in the event-based sediment load but added to the accumulated sediment load on the annual timescale.

2.5. Data analysis and statistics

The effects of phase, watershed, and interaction between phase and watershed were tested using a linear mixed-effect (LME) model (Smith, 2002). Data were log10 transformed to reduce heteroscedasticity. 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: J-RP vs. Control watershed (J), J-

Table 2
Date, precipitation, sediment load, average concentration, and peak concentration of 34 large rainfall events from watersheds, J: Juniper; J-RP: juniper to restored prairie; and J-SG: juniper to switchgrass.

Phase	Date	Precip. (mm)	Sediment Load (g m^{-2})			Av. Concentration (g L^{-1})			Peak Concentration (g L^{-1})		
			J	J-RP	J-SG	J	J-RP	J-SG	J	J-RP	J-SG
Calibration	5/5/15	59.2	0.85	0.07	0.43	0.41	0.14	0.40	1.70	0.33	0.40
	5/8/15	21.6	0.25	0.03	0.12	0.42	0.14	0.28	1.31	0.20	0.28
	5/19/15	29.9	0.58	0.15	0.64	0.20	0.11	0.21	0.40	0.23	1.30
	5/23/15	62.5	7.58	2.57	5.17	0.16	0.13	0.19	2.11	0.70	0.59
Transition	8/22/15	36.6	0.10	0.40	0.16	0.71	0.34	0.66	0.71	0.78	0.99
	11/5/15	33.3	0.12	1.50	1.52	1.11	0.25	0.24	1.11	0.51	0.75
	11/26/15	52.3	0.09	1.08	2.01	0.09	0.19	0.20	0.45	0.30	0.22
	7/3/16	14.2	0.04	0.65	4.28	0.89	0.60	0.76	0.89	1.83	5.18
	7/8/16	27.7	0.04	1.41	10.97	0.48	0.48	0.61	0.48	1.92	4.22
	10/6/16	57.9	0.23	11.54	136.77	0.68	0.75	1.38	0.72	3.11	15.38
	4/29/17	113.8	9.34	60.10	722.99	0.18	0.45	6.01	1.53	3.98	26.40
	7/2/17	30.7	0.06	1.60	20.49	0.98	0.44	1.05	0.98	1.49	7.95
	7/3/17	85.3	11.81	21.91	233.18	0.36	0.23	7.26	1.97	1.74	9.85
	8/16/17	25.9	0.10	0.18	4.41	1.20	0.32	0.73	1.20	0.60	2.12
Alternative	10/4/17	95.5	6.37	1.63	4.75	0.75	0.17	0.14	3.96	0.89	1.25
	10/21/17	36.1	0.25	0.51	1.48	1.27	0.15	0.14	1.27	0.96	0.72
	5/2/18	26.7	0.05	1.20	0.59	1.04	0.35	0.24	1.04	0.48	0.77
	7/1/18	37.8	0.08	0.25	0.41	0.25	0.17	0.10	0.56	0.27	0.21
	10/8/18	53.1	0.13	0.56	1.77	0.55	0.47	0.16	0.60	0.64	0.36
	4/23/19	42.4	0.35	5.14	2.15	0.22	0.21	0.15	0.50	0.41	0.53
	4/30/19	43.2	0.80	3.70	1.92	0.09	0.19	0.13	0.66	0.45	0.43
	5/3/19	24.6	1.18	1.82	1.66	0.22	0.14	0.18	0.73	0.36	0.51
	5/7/19	60.2	10.85	3.76	3.25	0.22	0.11	0.12	1.12	0.37	0.37
	5/20/19	157.2	34.97	26.58	8.12	0.16	0.20	0.06	3.03	0.48	0.27
	5/22/19	5.8	0.23	0.39	0.20	2.44	0.24	0.14	2.44	0.53	0.40
	5/24/19	35.3	4.00	3.90	4.73	0.32	0.20	0.44	1.85	0.28	0.52
	5/25/19	55.1	41.67	5.44	3.10	0.41	0.09	0.11	3.48	0.24	0.39
	5/28/19	10.7	0.04	0.53	0.28	2.02	0.29	0.18	2.02	0.36	0.56
	6/6/19	77.2	16.44	8.08	1.96	0.21	0.24	0.06	2.47	0.79	0.26
	6/15/19	33.5	0.07	0.91	0.50	1.38	0.15	0.09	1.38	0.32	0.18
	6/23/19	18.3	0.07	0.14	0.04	1.11	0.07	0.02	1.11	1.01	0.02
	8/22/19	55.6	0.12	0.03	0.11	0.90	0.50	0.03	0.90	0.70	0.05
	8/30/19	65.5	3.14	1.75	0.64	0.56	0.12	0.03	1.84	0.61	0.11
	9/12/19	74.9	1.15	0.96	0.73	0.43	0.15	0.04	1.46	0.70	0.10

SG vs. Control watershed (J), and J-RP vs. J-SG 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 (Table 2). Error terms were the differences between observed values and estimated values. The model was:

$$X_{ijk} = \hat{A}\,\mu + \hat{I}\pm_{i} + \tau_{k(i)} + \beta_{j} + (a\beta)_{ij} + \hat{I}\mu_{ijk} \quad (1)$$

where X_{ijk} was the dependent variable: event-based sediment load (g m^{-2}), or average sediment concentration (g L^{-1}), or peak sediment concentration (g L^{-1}); μ was the overall mean; a_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 ($k_{(1)} = 1, 2,\dots, 4$; $k_{(2)} = 1, 2,\dots, 10$; $k_{(3)} = 1, 2,\dots, 20$); β_j was the effect of watershed (j = impact, control; $j = 1, 2$); $(a\beta)_{ij}$ was the interaction between phase and watershed; and ε_{ijk} was the remaining error.

The interaction between phase and watershed was the main interest of the BACI method (Underwood, 1992). It tests whether the relative difference in sediment loads among watersheds significantly changed at different phases of land use. No interaction indicates the main effects of phase and watershed were independent. In other words, the changes in sediment variables among phases were similar among the different watersheds. A significant main effect of 'phase' would mean that sediment load, sediment concentration, or peak sediment concentration was greater in one period than another after controlling for watershed differences. A significant main effect of 'watershed' would indicate that one watershed had greater sediment load, sediment concentration, or peak sediment concentration than another regardless of phase.

Water year is defined as the 12-month period from October 1 of the previous year through September 30 of the current year. During the water years 2015 through 2019, there were 318 rainfall events, 251 runoff events, and 123 events where sediments were recorded. However, runoff or sediment loads were minimal for most small runoff events and occurred in one or two watersheds. Only 34 large rainfall events

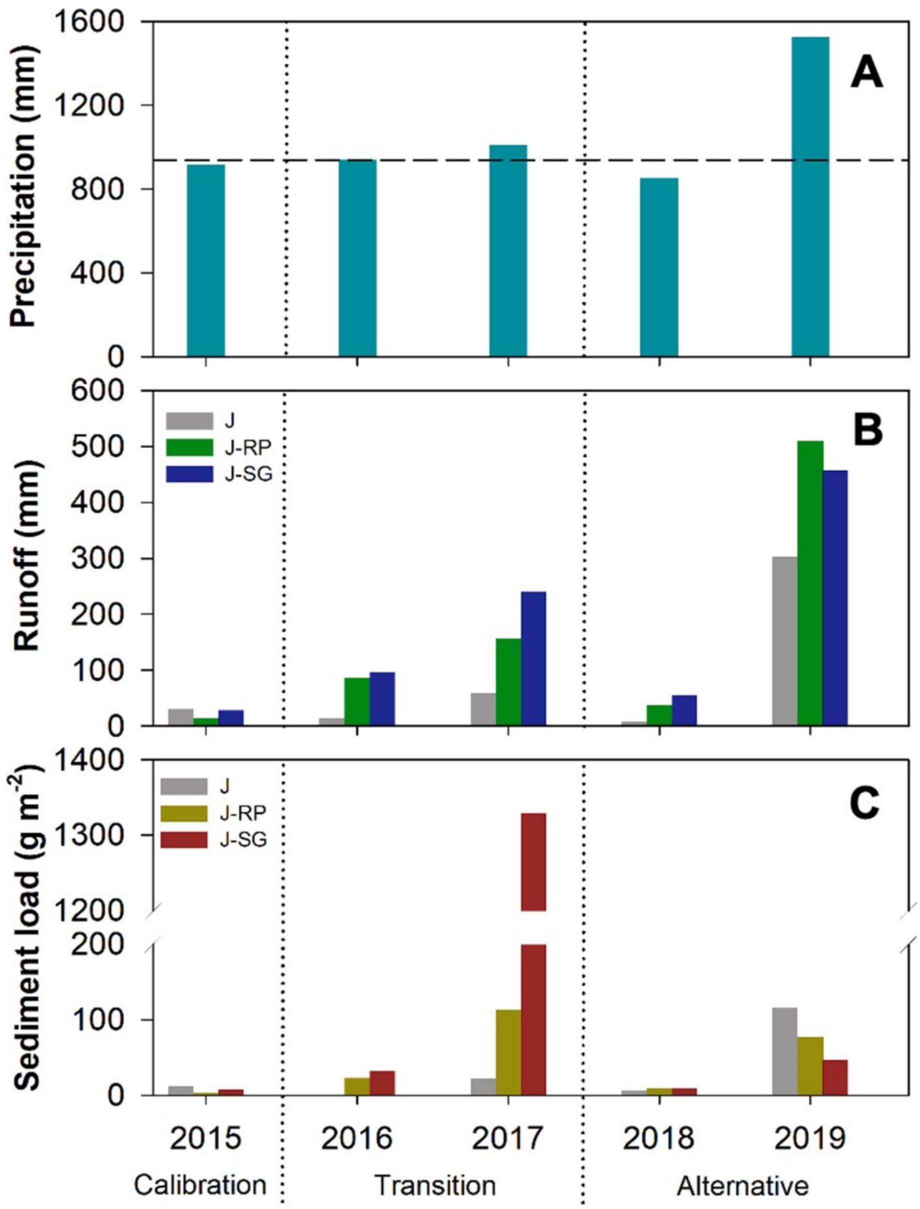

Fig. 2. a) Annual precipitation during the water years 2015 through 2019. The dashed line indicates the 30-year annual mean precipitation between 1981 and 2010 from the nearby Marena Mesonet station; b) Annual runoff depth from the three watersheds; c) Annual sediment load from the three watersheds. J: Juniper; J-RP: juniper to restored prairie; and J-SG: juniper to switchgrass. Note: 100 g m^{-2} = 1 t ha^{-1}.

produced sediment across all watersheds (4, 10, and 20 in Calibration, Transition, and Alternative phase, respectively) and were used in the statistical analysis (Table 2). 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 calculation of annual sediment yield.

When the interaction terms were significant in the statistical test, the Contrast method was applied to estimate the difference of least-square means of different phases for each pair of watersheds (three sets of comparison: Transition vs. Calibration; Alternative vs. Transition; Alternative vs. Calibration) (Dąbrowska et al., 2017; Lane et al., 1999). Since the data were unbalanced among phases, the least-square means were applied. The LME model and the contrast of least-square means were run in the RStudio, with the R version 4.0.0.

3. Results

3.1. Annual precipitation, runoff, sediment yield

The average precipitation during the study was 1050 mm, approximately 12% greater than the long-term mean precipitation of 939 mm, with 2019 being an extremely wet year (62% above the long-term mean) (Fig. 2A). Before treatment (2015), annual runoff from the three juniper watersheds was comparable and low, averaging 25 mm (Fig. 2B). In the Transition phase (2016 and 2017), the annual runoff averaged 37 mm from the J watershed, compared with 122 mm and 169 mm from J-RP and J-SG watersheds. In the Alternative phase (2018 and 2019), the annual runoff averaged 156 mm for the intact juniper watershed, 274 mm for the restored prairie, and 257 mm for the switchgrass.

Before treatment, the annual sediment load for all three watersheds averaged < 0.10 t ha^{-1} yr^{-1} in 2015 (Fig. 2C). In the first part of the Transition phase (2016), after the juniper trees were cut and left to dry on site, the average sediment load for the Impact watersheds increased to 0.28 t ha^{-1} yr^{-1}. In comparison, the sediment load from the intact juniper watershed was 0.01 t ha^{-1} yr^{-1}. In the second part of the Transition phase (2017), when the dried trees were removed from both Impact watersheds, and the J-SG watershed was sprayed with herbicide, the J-SG had the largest sediment load (13.29 t ha^{-1} yr^{-1}), followed by the J-RP (1.14 t ha^{-1} yr^{-1}) while the J watershed had 0.23 t ha^{-1} yr^{-1}. In the Alternative phase (2018 and 2019), the mean annual sediment load averaged 0.44 t ha^{-1} yr^{-1} for the restored prairie, 0.28 t ha^{-1} yr^{-1} for the switchgrass, and 0.73 t ha^{-1} yr^{-1} for the intact juniper.

3.2. Flow rate, peak flow, and sediment concentration

The hydrographs and sediment graphs for runoff events with similar rainfall amounts (20–30 mm) during each phase of the experiment illustrate the runoff and erosion processes for the different watersheds (Fig. 3). During the Calibration phase, the runoff was low, and there was no substantial increase in sediment concentration during the peak flow. The J-RP watershed had a relatively low peak flow rate but a similar flow duration compared with the J and J-SG watersheds (Fig. 3A). During the second part of the Transition phase, the J-SG watershed produced greater peak flow than the J-RP watershed with a similar flow duration, and a large peak in sediment concentration occurred simultaneously with the peak flow rate. In contrast, negligible runoff and sediment load were generated from the J watershed (Fig. 3B). During the Alternative phase, both the J-RP and J-SG watersheds had elevated peak flows and longer flow duration than the J watershed, but the sediment concentrations were lower than in the J watershed and did not strongly respond to increasing flow rate (Fig. 3C).

3.3. 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 watershed (Table 3), indicating the differences in sedimentation

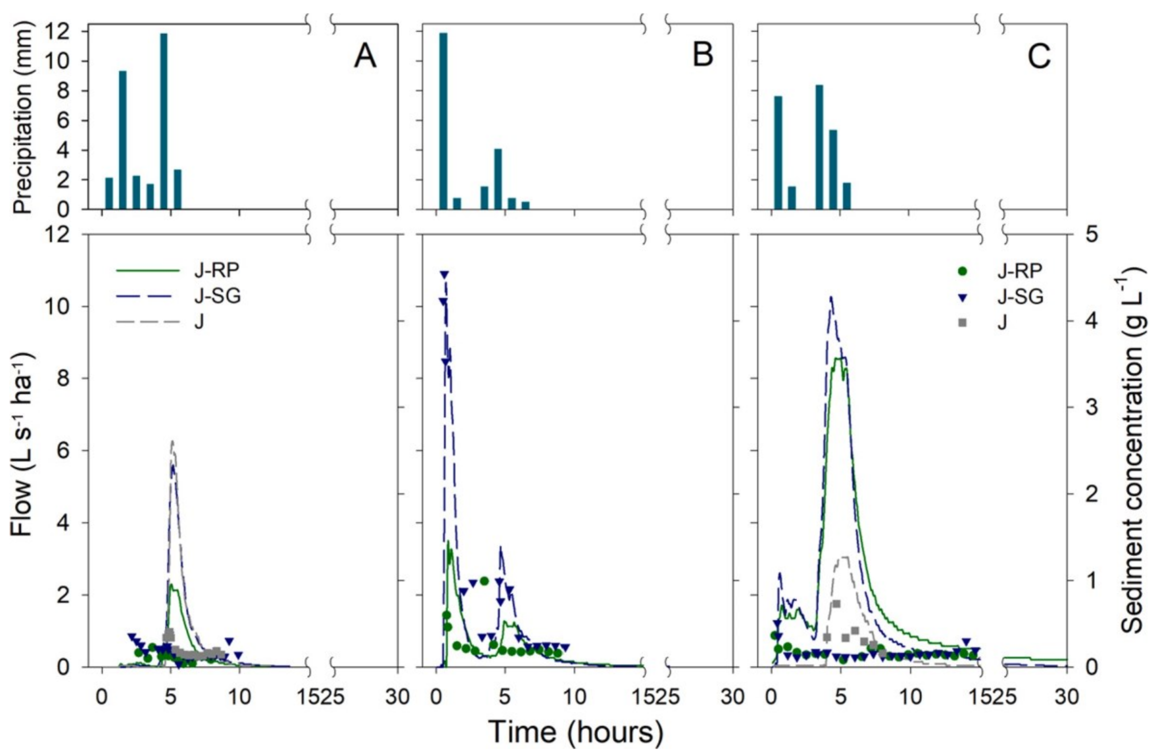

Fig. 3. Comparison of flow rates (lines) and sediment concentrations (symbols) for the control watershed J (Juniper) and impact watersheds, J-RP (juniper to restored prairie) and J-SG (juniper to switchgrass) from a rainfall event of 30 mm on May 19th, 2015, during the Calibration phase (A), a rainfall 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).

Table 3
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 to restored prairie; and J-SG: juniper to switchgrass).

Pairs	Terms	Sediment load	Average sediment concentration	Peak sediment concentration
J vs. J-RP	Phase	0.648	**<0.001**	0.131
	Site	0.229	**0.013**	**<0.001**
	Phase × Site	**<0.001**	**0.040**	**<0.001**
J vs. J-SG	Phase	0.677	**<0.001**	**0.002**
	Site	**0.001**	0.556	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**	0.053	0.131
	Phase × Site	**<0.001**	**<0.001**	**0.001**

processes between watersheds varied depending on the phase. When comparing the Impact watersheds to the untreated J watershed, these interactions were caused by much greater relative sediment load, sediment concentration, and peak sediment concentration from the Impact watersheds during the Transition phase than during the Calibration or Alternative phases (Table 4). When comparing the Alternative phase to the Calibration phase, the sediment variables decreased in the Impact watersheds relative to the J watershed (Table 4). However, only the relative difference in average sediment concentration for the J-SG vs. J comparison was significantly lower when comparing the Alternative phase to the Calibration phase.

Comparing the J-SG to the J-RP watersheds, sediment variables were much greater in the J-SG watershed than the J-RP watershed during the Transition phase (Table 4). However, large variation resulted in nonsignificant results among differences during the Calibration and Transition phases. In contrast, relative differences in sediment variables were lower for the J-SG than the J-RP during the Alternative phase compared to either the Calibration or Transition phases.

Table 4
The mean difference (mean ± S.E.; back-transformed from log10 values) between paired events in each comparison of watersheds (the former minus the latter) for sediment load, average sediment concentration, and peak sediment concentration during each phase (Calibration, Transition, and Alternative). 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 to restored prairie; and J-SG: juniper to switchgrass).

Pairs	Phase	Sediment load ($g\ m^{-2}$)*	Average sediment concentration ($g\ L^{-1}$)	Peak sediment concentration ($g\ L^{-1}$)
J-RP – J	Calibration	−1.67 ± 1.17 [a]	−0.17 ± 0.05 [ab]	−1.02 ± 0.29 [a]
	Transition	6.11 ± 3.68[b]	0.01 ± 0.15[b]	0.62 ± 0.36[b]
	Alternative	−7.11 ± 5.57 [a]	−0.32 ± 0.07 [a]	−1.08 ± 0.23 [a]
J-SG – J	Calibration	−0.76 ± 0.59 [a]	−0.05 ± 0.02[b]	−0.74 ± 0.55 [a]
	Transition	42.27 ± 28.36[b]	2.21 ± 1.02[b]	6.30 ± 2.54[b]
	Alternative	−9.38 ± 4.57 [a]	−0.37 ± 0.07 [a]	−1.22 ± 0.21 [a]
J-SG – J-RP	Calibration	0.86 ± 0.62[b]	0.12 ± 0.05[b]	0.28 ± 0.27 [ab]
	Transition	31.00 ± 22.36[b]	2.20 ± 0.93[b]	5.68 ± 2.25[b]
	Alternative	−3.21 ± 2.46 [a]	−0.05 ± 0.02 [a]	−0.14 ± 0.08 [a]

* $100\ g\ m^{-2} = 1\ t\ ha^{-1}$

4. Discussion

4.1. Sediment concentration and juniper removal

Woody plant encroachment in semiarid rangelands can increase overland flow and sediment transport (Pierson et al., 2007; Williams et al., 2020), particularly during large thunderstorm events (Wilcox et al., 2003). But, less is known about the soil erosion and water quality impacts of juniper encroachment into the subhumid tallgrass prairie. This study addresses that knowledge gap. We found that the average flow-weighted sediment concentration in runoff from three juniper watersheds before treatment was 0.27 $g\ L^{-1}$ (Fig. 4), substantially lower than 0.60 $g\ L^{-1}$ reported for the rivers in the Great Plains (Dodds and Whiles, 2004). Therefore, encroachment and invasion of woody plants into moist grasslands are likely not the main culprits of increased turbidity of stream water and reservoirs observed in this region. Encroachment may lead to less soil erosion and water quality concerns in subhumid grassland than in arid and semiarid grasslands.

Although encroached watersheds were not major sediment sources in this study, the remediation of those watersheds by removing juniper and subsequent re-establishment of grassland created a transition period during which peak sediment concentrations were significantly increased. The elevated sediment concentrations tended to be synchronized to peak flow rate, resulting in a pulsed efflux of sediments that could be detrimental to streams and reservoirs (Sadeghi et al., 2008). We speculate that after the cut juniper trees were removed, the extent of bare soil patches and the connectivity among bare soil patches increased, leading to increased sediment in runoff (West et al., 2016). The maximum sediment concentration measured after juniper removal was 3.98 $g\ L^{-1}$ for the prairie restoration watershed (J-RP) and 26.40 $g\ L^{-1}$ for the switchgrass planting watershed (J-SG) (Fig. 4). The high peak concentrations in the switchgrass watershed mainly occurred during a few storm events after herbicide application and before switchgrass establishment (Table 2). Herbicide application is necessary for no-till switchgrass planting following juniper removal (Parrish and Fike, 2009), but it enhances the risk of a pulsed increase of sediment efflux.

4.2. Sediment load and juniper woodland removal

The loss of herbaceous vegetation due to juniper encroachment followed by the topsoil disturbance by machine traffic while mechanically removing trees makes the watershed vulnerable to water-caused soil erosion. During the study period, the annual sediment yield from intact juniper woodland varied from negligible to 1.39 $t\ ha^{-1}\ yr^{-1}$. Sediment yield increased after juniper removal during the *Transition* phase for both impact watersheds. In the prairie restoration watershed, the quick recovery of native grasses and forbs (Schmidt et al., 2021) moderated the increase in sediment load. Native prairie started to recover beginning in May 2016 and continued throughout the 2017 water year, and the annual sediment yield for this watershed was 1.14 $t\ ha^{-1}\ yr^{-1}$ during the *Transition* period. This load was substantially lower than the 1.95 $t\ ha^{-1}\ yr^{-1}$ reported the first year after clear-cutting and site preparation subsoiling of a loblolly pine (*Pinus taeda*) forest watershed in southeastern Oklahoma (Naseer, 1992) and was in the range of tolerable soil loss to sustain soil resources (<2.00 $t\ ha^{-1}\ yr^{-1}$) (FAO 2019).

Sediment load had a significant but short-term increase in the switchgrass planting watershed following the watershed preparation with herbicide. The annual sediment yield in the water year 2017 was 13.30 $t\ ha^{-1}\ yr^{-1}$, nearly twice the average water-driven sediment yield from cropland in southern Great Plains (Menzel et al., 1978; USDA, 2009). This high annual loss was primarily resulted from the pulsed response of sediment to relatively high precipitation following herbicide treatment in April 2017. The measured increase in the sediment load falls within the range predicted by Lisnebee et al. (2015), who, based on jet erosion tests, predicted a one to two orders of magnitude increase in average sediment load immediately after juniper removal for this

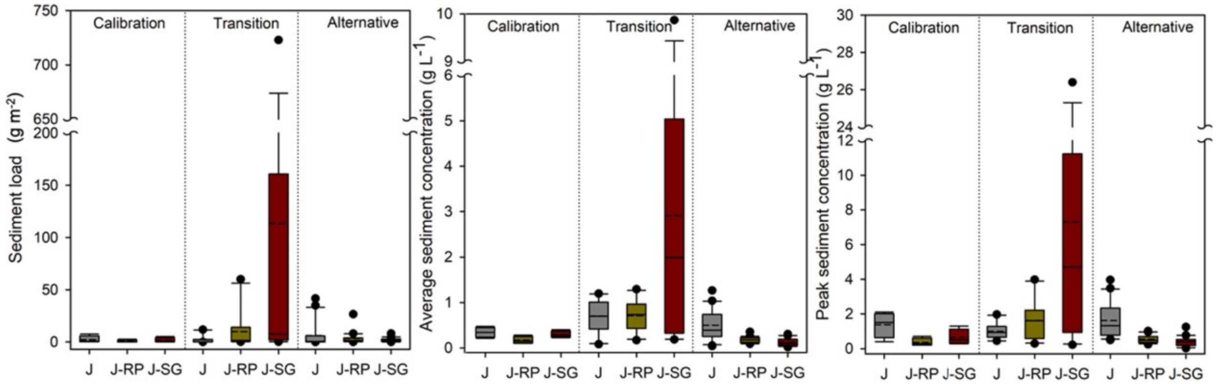

Fig. 4. Mean event-based sediment load, average sediment concentration, and peak sediment concentration among watershed J (Juniper), J-RP (juniper to restored prairie), and J-SG (juniper to switchgrass) along with three phases: Calibration, Transition, and Alternative. Note: 100 g m^{-2} = 1 t ha^{-1}.

watershed.

4.3. Restored prairie and switchgrass and water quality improvement

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 juniper woodland could be higher than the well-established switchgrass (Table 4; Fig. 4). The event-based average concentrations for 2018 and 2019 were 0.50 g L^{-1} for the juniper watershed, 0.18 g L^{-1} for the restored prairie watershed, and 0.13 g L^{-1} for the switchgrass watershed (Fig. 4). It is important to note that 2019 was an extremely wet year that caused a substantial increase in overland flow and forest floor material movement in the juniper woodland watershed, which rarely occurs in this system. More data are needed to compare and contrast the sediment processes between juniper woodland and restored grassland.

The annual sediment load from the established grasslands ranged from 0.10 to 0.80 t ha^{-1} yr^{-1}, which is in the range of tolerable soil loss to sustain soil resources for rangeland (FAO 2019) and significantly less than the average annual water erosion soil loss (6.73 t ha^{-1} yr^{-1}) from the cropland in this region (USDA, 2009). The established switchgrass watershed and restored prairie watershed had similar surface runoff (Zhong et al., 2020), but the former produced lower sediment concentration in runoff and lower sediment yield. Converting cropland to 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 as a biofuel feedstock may also reduce soil erosion and enhance water quality in the long term.

4.4. Implication of juniper management on soil erosion

The canopy cover of juniper woodland in our study area was over 75% (Zou et al., 2014). The hydrological alteration due to mechanical removal is analogous to clear-cutting in forest management and produced sediment load at a level similar to, or in the case where herbicide is used for site preparation, even higher than that reported from the clear-cutting loblolly pine forest in Oklahoma (Heh, 1988; Nassier, 1992). However, most encroached grasslands in the region are still in the early phase of canopy formation, with canopy cover < 50% (Wang et al., 2017). There is no clear evidence that the encroachment of juniper in moist tallgrass prairie causes a significant reduction in herbaceous cover between trees or patches of trees during the early stages of encroachment (Engle et al., 1987; Limb et al., 2010), which is in contrast to the findings from more xeric grassland (Ansley et al., 2006; Miller et al., 2000). Therefore, mechanical removal of isolated juniper trees or juniper patches will be less of a concern than removing juniper woodlands in terms of its impact on sediment and potential nutrient efflux.

The rapid decline in the sediment loads following the grassland recovery suggests that soil erosion is mostly a short-term, pulsed response to the juniper removal. Restoring juniper woodlands to grassland through re-establishment of prairie and switchgrass biomass production systems tends to increase water quality in the long term. Future research should explore alternative approaches to reduce soil erosion and sediment loss during the Transition phase.

We speculate that leaving the cut trees to dry in place might have helped to protect the soil surface from rain splash erosion (Wilcox et al., 2003), and the annual soil loss in the water year 2016, while the cut trees remained, was only moderately elevated relative to the baseline. However, leaving trees on site for an extended time might slow down the natural re-establishment of herbaceous cover, and further study is needed to understand the tradeoff between soil erosion prevention and prairie recovery. If the watershed is to be planted with switchgrass, a plausible plan might be to remove cut trees immediately and then follow with no-till drilling soon after to speed up the conversion process and potentially avoid the herbicide application used in this project.

5. Conclusion

Mechanical removal of encroached and invaded woody vegetation is a common management practice in rangeland. In this paired watershed study, sediment concentration and yield from juniper woodland watersheds were relatively low and increased with mechanical removal of juniper from the watersheds. Herbicide application following tree removal (in preparation for planting switchgrass) created a transition period with high erosion potential and requires careful management. Sediment loads returned to levels similar to or lower than those in the juniper woodland one year after the grasslands were established. After full establishment, the runoff from the switchgrass production system carried less sediment than that from the naturally restored prairie and produced less sediment load. The inclusion of switchgrass into grassland restoration projects may help remediate woody plant encroachment and enhance water quality and water quality in the mesic region of the southern Great Plains. Further study should explore the possibility of establishing a switchgrass production system without herbicide application and the impact of fertilizer application on sediment, nutrient, and bacteria efflux into streams and reservoirs.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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