

Four decades of change: How woody encroachment, fire, grazing, and agricultural legacy shape hydrology, nutrients, and sediment in tallgrass prairie streams

by

Md Abu Raihan

B.S., University of Dhaka, 2017
M.S., Columbus State University, 2021

AN ABSTRACT OF DISSERTATION

submitted in partial fulfillment of the requirements for the degree

DOCTOR OF PHILOSOPHY

Division of Biology
College of Arts and Sciences

KANSAS STATE UNIVERSITY
Manhattan, Kansas

2026

Abstract

Grassland streams are among the most hydrologically variable and least monitored freshwater ecosystems on Earth. This dissertation uses up to four decades of continuous data from Konza Prairie Biological Station (KPBS), a tallgrass prairie Long-Term Ecological Research site in the Kansas Flint Hills, combined with rainfall simulation experiments in the riparian zone, to show that interacting changes in climate, vegetation, and land management are fundamentally transforming hydrology, nutrient biogeochemistry, and sediment dynamics of intermittent prairie streams. I analyzed long-term data on discharge, precipitation, temperature, and woody vegetation expansion to assess influences on hydrology between 1982 and 2020. Kings Creek's daily discharge decreased by over half and lost 55% of its annual flow days without a decline in precipitation. Riparian woody cover expanded two- to sevenfold across one- to four-year fire return intervals. This increase in woody cover, coupled with a 1.2°C temperature increase over the same period, increased evapotranspiration, with the strongest increases during the July to September period. Structural equation modeling confirmed that evapotranspiration is the primary pathway linking woody encroachment to streamflow decline.

Stream nutrient concentrations responded to a hierarchy of drivers operating over different timescales. Agricultural legacy exerted the strongest and most persistent influence: stream nitrate below restored croplands was initially high (450 $\mu\text{g N L}^{-1}$) and declined by 75% over 25 years as restoration progressed, but groundwater nitrate remained near 950 $\mu\text{g N L}^{-1}$ more than two decades after crop fertilization ceased. Bison grazing doubled stream nitrate relative to ungrazed watersheds, an effect larger than that observed during the first few years after bison were introduced. Cattle elevated both nitrate and total phosphorus through direct riparian disturbance, and these effects were strongest when cattle were actively present on the

pasture. Differences in fire frequency had minimal direct effects on nutrients but interacted with grazing to shape decadal trajectories.

Suspended sediment dynamics were governed by a dual-pathway framework in which organic and inorganic fractions responded to different controls. Integration of 15 years of watershed monitoring (2009–2023), rainfall simulation experiments with bromide tracer ($n = 10$), and streambank geomorphic surveys revealed that discharge drove organic sediment (volatile suspended solids, VSS) through dilution, while fire management drove inorganic sediment (inorganic suspended solids, ISS) through mineral soil exposure. An area-normalized erodibility index (Sed/m^2) revealed a nonlinear riparian vegetation cover threshold between approximately 60% and 70%, below which areal erodibility increased by approximately two orders of magnitude. Two high-cover sites (Exps 2 and 3) produced no detectable bromide breakthrough despite sub-annual storm depths, while bare bison-grazed soil was 168 times more erodible per unit area than an adjacent fully vegetated surface at matched slope. Generalized additive models (43–46% variance explained) and structural equation modeling confirmed that days since burn exerted the strongest direct effect on the VSS/ISS ratio, establishing prescribed fire frequency as the primary determinant of sediment composition. The VSS/ISS ratio was more sensitive to management than any individual fraction, with biennial burning under bison maintaining the most organic-dominated sediment regime and quadrennial burning producing the most inorganic-dominated. Streambank surveys corroborated the inorganic pathway, with woody cover as the strongest erosion predictor, and the watershed combining the longest fire interval with the lowest canopy cover exhibited the highest bank cut frequency.

These findings demonstrate that effective management of grassland streams requires integrated, long-term approaches that account for the coupled dynamics of water, nutrients, and

sediment under ongoing climate and vegetation change. Fire and grazing interactions regulate all three domains through nonlinear threshold responses, and temporal scales of years to decades are required to detect directional changes that shorter-term studies systematically underestimate. Prescribed fire frequency emerges as the most broadly effective management lever, while the prevalence of threshold behaviors underscores the importance of monitoring for proximity to critical transitions before they are crossed.

Four decades of change: How woody encroachment, fire, grazing, and agricultural legacy shape hydrology, nutrients, and sediment in tallgrass prairie streams

by

Md Abu Raihan

B.S., University of Dhaka, 2017
M.S., Columbus State University, 2021

A DISSERTATION

submitted in partial fulfillment of the requirements for the degree

DOCTOR OF PHILOSOPHY

Division of Biology
College of Arts and Sciences

KANSAS STATE UNIVERSITY
Manhattan, Kansas

2026

Approved by:

Major Professor
Walter K. Dodds

Copyright

© Md Abu Raihan 2026.

Abstract

Grassland streams are among the most hydrologically variable and least monitored freshwater ecosystems on Earth. This dissertation uses up to four decades of continuous data from Konza Prairie Biological Station (KPBS), a tallgrass prairie Long-Term Ecological Research site in the Kansas Flint Hills, combined with rainfall simulation experiments in the riparian zone, to show that interacting changes in climate, vegetation, and land management are fundamentally transforming hydrology, nutrient biogeochemistry, and sediment dynamics of intermittent prairie streams. I analyzed long-term data on discharge, precipitation, temperature, and woody vegetation expansion to assess influences on hydrology between 1982 and 2020. Kings Creek's daily discharge decreased by over half and lost 55% of its annual flow days without a decline in precipitation. Riparian woody covers expanded two- to sevenfold across one- to four-year fire return intervals. This increase in woody cover, coupled with a 1.2°C temperature increase over the same period, increased evapotranspiration, with the strongest increases during the July to September period. Structural equation modeling confirmed that evapotranspiration is the primary pathway linking woody encroachment to streamflow decline.

Stream nutrient concentrations responded to a hierarchy of drivers operating over different timescales. Agricultural legacy exerted the strongest and most persistent influence: stream nitrate below restored croplands was initially high (450 $\mu\text{g N L}^{-1}$) and declined 75% over 25 years, but groundwater nitrate remained near 950 $\mu\text{g N L}^{-1}$ more than two decades after crop fertilization ceased. Bison grazing doubled stream nitrate relative to ungrazed watersheds, an effect larger than that observed during the first few years after bison were introduced. Cattle elevated both nitrate and total phosphorus through direct riparian disturbance, and these effects

were strongest when cattle were actively present on the pasture. Fire frequency had minimal direct effects on nutrients but interacted with grazing to shape decadal trajectories.

Suspended sediment dynamics were governed by a dual-pathway framework in which organic and inorganic fractions responded to different controls. Integration of 15 years of watershed monitoring (2009–2023), rainfall simulation experiments with bromide tracer ($n = 10$), and streambank geomorphic surveys revealed that discharge drove organic sediment (volatile suspended solids, VSS) through dilution, while fire management drove inorganic sediment (inorganic suspended solids, ISS) through mineral soil exposure. An area-normalized erodibility index (Sed/m^2) revealed a nonlinear riparian vegetation cover threshold between approximately 60% and 70%, below which areal erodibility increased by approximately two orders of magnitude. Two high-cover sites (Exps 2 and 3) produced no detectable bromide breakthrough despite sub-annual storm depths, while bare bison-grazed soil was 168 times more erodible per unit area than an adjacent fully vegetated surface at matched slope. Generalized additive models (43–46% variance explained) and structural equation modeling confirmed that days since burn exerted the strongest direct effect on the VSS/ISS ratio, establishing prescribed fire frequency as the primary determinant of sediment composition. The VSS/ISS ratio was more sensitive to management than any individual fraction, with biennial burning under bison maintaining the most organic-dominated sediment regime and quadrennial burning producing the most inorganic-dominated. Streambank surveys corroborated the inorganic pathway, with woody cover as the strongest erosion predictor, and the watershed combining the longest fire interval with the lowest canopy cover exhibited the highest bank cut frequency.

These findings demonstrate that effective management of grassland streams requires integrated, long-term approaches that account for the coupled dynamics of water, nutrients, and

sediment under ongoing climate and vegetation change. Fire and grazing interactions regulate all three domains through nonlinear threshold responses, and temporal scales of years to decades are required to detect directional changes that shorter-term studies systematically underestimate. Prescribed fire frequency emerges as the most broadly effective management lever, while the prevalence of threshold behaviors underscores the importance of monitoring for proximity to critical transitions before they are crossed.

Table of Contents

List of Figures	xiv
List of Tables	xix
Acknowledgements	xx
Dedication	xxii
Preface	xxiii
Chapter 1 - Introduction.....	1
Woody encroachment and the drying of grassland streams.....	2
Fire, grazing, and agricultural legacy as controls on nutrient chemistry	4
Sediment composition as an indicator of watershed disturbance	6
Crosscutting themes and dissertation structure	8
Chapter 2 - Controls on Discharge and Drying in an Intermittent Grassland Stream: Temporal and Network Variability	9
Abstract	9
Introduction.....	11
Methods	15
Study area description	15
Watersheds analyzed and data sources	17
Calculation of evapotranspiration, excess water, and excess energy.....	22
Statistical Data Analysis	25
Results.....	29
Temporal trends of hydrology and climate at USGS Kings Creek gage.	29
Temporal autocorrelation in discharge	30
Factors influencing discharge and total flow days per season in the three sub-watersheds.	32
Correlations with discharge and the number of flow days and how they are changing over the years for USGS gage and upstream sub-basins.....	33
What might influence discharge and the number of flow days at the USGS station over the years by season?	34

Assessing evapotranspiration dynamics with wood across seasons	34
Discussion	41
Expanded woody vegetation plays a role in declines in stream discharge and flow days....	41
Temporal Dynamics of Evapotranspiration driven by the Interplay of Water and energy ...	44
How might this apply to other grasslands?	47
Ecological and water yield implications	49
Conclusions.....	52
Acknowledgements.....	54
Chapter 3 - Effects Of Ungulate Grazing, Burning, And Cropland on Stream Water Chemistry in	
Tallgrass Prairie Watersheds	55
Abstract	55
Introduction.....	56
Methods	59
Study Site locations and land use history	59
Water Quality Monitoring and Analytical Methods	64
Statistical Analyses	65
Results.....	67
Nutrient Trends in Bison-Grazed Sites Under Differing Burn Frequencies	67
Trends in Nutrients in Restored Cropland, Pristine Spring, and Downstream Sites	69
Changes in Management Practices Influencing Nutrient Concentrations	71
Trends in Nutrients in Groundwater Wells under Grassland or Sequential Restoration from	
Cropland.....	72
Bison and Cattle Effects on Water Nutrients	76
Discussion	78
Fire effects	78
Agricultural Legacy Creates Decades-Long Nutrient Elevation	80
Bison and Cattle Effects.....	81
Legacies and temporal and spatial heterogeneity	81
Conclusions.....	83
Acknowledgments	85

Chapter 4 - Vegetation Cover Thresholds and Fire-Grazing Interactions Regulate Organic and Inorganic Sediment Pathways in Tallgrass Prairie Streams	86
Abstract	86
Introduction	88
Methods	94
Study Site	94
Dataset 1: Long-Term Suspended Sediment Monitoring	95
Dataset 2: Rainfall Simulation Experiments	97
Dataset 3: Streambank Geomorphic Surveys	101
Data Availability	102
Results	103
Riparian Sediment Delivery During Simulated Rainfall Events	103
Long-term Suspended Sediment Dynamics (2009–2023)	110
Streambank Geomorphic Surveys	117
Discussion	120
Independent Regulation of Organic and Inorganic Sediment by Discharge and Fire Management	120
Nonlinear Erodibility Thresholds and the Multi-Gradient Structure of Riparian Function	122
Fire Frequency, Woody Encroachment, and Bank Instability	124
Limitations and Future Directions	126
Conclusions	128
Chapter 5 - Conclusions	131
Synthesis of Findings	131
Hydrological transformation driven by woody encroachment	131
Nutrient hierarchy: agricultural legacy, grazing, and fire	132
Dual-pathway sediment framework	133
Crosscutting findings	135
Management implications	137
Limitations	138
Future research directions	140
Closing	142

References	143
Appendix A- Supplemental Information for Chapter 2	174
Appendix B- Supplemental Information for Chapter 3	189
Appendix C- Supplemental Information for Chapter 4	199

List of Figures

- Figure 2.1: Conceptual model of our study. The purple dashed arrows represent the hypothesized positive influences, and the orange solid lines represent the hypothesized negative influences of the controlling and intermediate variables on discharge and number of flow days. 14
- Figure 2.2: USGS Kings Creek watershed within the Konza Prairie Biological Station, located in Kansas, USA, including the Long-Term Ecological Research (LTER) watersheds N1B, N2B, and N4D. The U.S. Geological Survey has consistently monitored the USGS weather and stream gaging station (red square) at this site since 1979 (Gage 06879650, red square). The yellow dots denote sub-watershed weirs, serving to measure discharge levels since 1987..... 16
- Figure 2.3: Annual mean daily discharge by year (A) and number of annual flow days by year (B), annual mean daily temperature (C), and annual total precipitation (D) at the USGS gage in the Kings Creek watershed. The solid salmon-red line illustrates a regression fit for each graph where $p < 0.05$ 30
- Figure 2.4: Discharge averaged for each day from 1982 - 2020 at the USGS Kings Creek gage. Boxes indicate the “seasons” we used for subsequent analyses, which were defined based on periods where autocorrelation was not significant..... 31
- Figure 2.5: $\log_{10}$ scaled proportion of shrub and tree cover in riparian zones over time at the USGS Kings Creek sub-basin watersheds with different burning frequencies. A) N1B every year, B) N2B burns every 2 years, and C) N4D burns every 4 years. All watersheds showed a significant increase in the total riparian cover. Note the vertical dashed line in panel B indicates to 2011 Riparian removal immediately following the whole watershed riparian removal experiment. Original riparian woody cover proportions ranged from 0.15-0.30 in N1B, 0.15-0.32 in N2B (with temporary reduction to 0.10 following removal), and 0.05-0.35 in N4D during the study period 32
- Figure 2.6: Seasonal variation in evapotranspiration (mm/3 months (cumulative seasonal total)) dynamics for Total watershed grass ET (A), Total watershed grass + riparian woody cover

ET (B), and Total watershed grass + total woody cover ET (C) in the USGS Kings Creek watershed	36
Figure 2.7: Comparative Plots of Excess Precipitation (Pex) vs. Excess Evaporative Demand (Eex) for Grass with Wood in the USGS Kings Creek Watershed from 2001 to 2020.....	37
Figure 2.8: Seasonal breakdown of Excess Precipitation (Pex) vs. Excess Evaporative Demand (Eex) for the USGS Kings Creek watershed from 2001-2020 (wet years only). Panels show: (A-D) Pex patterns and (E-H) Eex patterns for Winter (Jan-Mar), Spring (Apr-Apr-Jun), Summer (Jul-Sep), and Fall (Oct-Dec), respectively. Color gradient represents the year progression from 2001 (cool colors) to 2020 (warm colors). Red dashed lines indicate linear trends.....	38
Figure 2.9: Structural Equation Model illustrating significant relationships according to the conceptual model. Blue estimates are significant at $p < 0.002$, while the purple estimate indicates significance at $p = 0.032$. (Here, “Q”= log discharge, “ET”= actual evapotranspiration, “Precip”=precipitation, “Temp”=temperature, “Rip Cover”=log riparian cover). Solid arrows represent direct causal relationships, while dashed arrows represent covariances without assumed direct causality. Straight arrows indicate hypothesized directional effects, and curved arrows represent residual variances (self-loops) or correlations between variables. Numbers on paths are standardized path coefficients, with blue values indicating significance at $p < 0.002$ and purple values at $p < 0.05$	40
Figure 3.1: Map of Konza Prairie Biological Station (KPBS), Kansas, USA, showing watershed boundaries and sampling locations. Yellow outlined watersheds (SFKC and NFKC) show South Fork Kings Creek (bison-grazed) and North Fork Kings Creek (ungrazed), respectively. Yellow circles mark sampling sites N1B, N2B, and N4D in bison-grazed LTER watersheds. The orange outlined area shows the Shane Creek cattle-grazed watershed with the sampling location marked. Orange stars indicate key monitoring sites: EDLR (pristine spring), HIKX (downstream site below cropland), and HOKN (upstream of cropland). Blue triangles show groundwater well locations (ALC-A5 in former cropland, K01A-P5 in grassland). The red square marks the USGS gauging station. Cropland areas are visible in the lower left portion of the map. The inset shows the location within the United States.	62

Figure 3.2: Stream nitrate (A), soluble reactive phosphorus (B), and total phosphorus (C) concentrations at three bison-grazed prairie watersheds with different burn frequencies (1995-2022). Black circles show N1B (annual burn); purple squares show N2B (2-year burn frequency); orange diamonds show N4D (4-year burn frequency). Panel A: Linear regression for N2B: $\ln(\text{NO}_3^- - \text{N}, \mu\text{g N L}^{-1}) = \beta_0 + 0.021(\text{year} - 1995)$, $R^2 = 0.26$, $p = 0.009$ (equivalent to 2.1% increase per year). Panel C: TP showed no consistent long-term trend but exhibited elevated concentrations from 1995-2005 across all sites before returning to baseline levels. Lines show linear trends with 95% confidence intervals. 69

Figure 3.3: Stream nitrate (A), soluble reactive phosphorus (B), and total phosphorus (C) concentrations at three monitoring sites (1995-2022). Black circles show HIKX (downstream site below cropland); purple squares show HOKN (upstream of cropland); orange diamonds show EDLR (groundwater spring). Linear regression equations: Panel A (HIKX): $\ln(\text{NO}_3^- - \text{N}, \mu\text{g N L}^{-1}) = \beta_0 - 0.024(\text{year} - 1995)$, $R^2 = 0.68$, $p < 0.001$ (equivalent to -2.4%/yr decline). Panel B (HIKX): $\ln(\text{SRP}, \mu\text{g P L}^{-1}) = \beta_0 - 0.022(\text{year} - 1995)$, $R^2 = 0.54$, $p < 0.001$ (-2.2%/yr decline); (HOKN): $\ln(\text{SRP}, \mu\text{g P L}^{-1}) = \beta_0 - 0.021(\text{year} - 1995)$, $R^2 = 0.35$, $p < 0.001$ (-2.1%/yr decline). Panel C: No significant temporal trends detected. Lines show linear trends with 95% confidence intervals..... 71

Figure 3.4: Groundwater nitrate (A), soluble reactive phosphorus (B), and total phosphorus (C) concentrations at two monitoring wells (2000-2022). Black circles show ALC-A5 (agricultural well under sequentially restored cropland); purple diamonds show K01A-P5 (grassland reference well). Linear regression equations: Panel A (ALC-A5): $\ln(\text{NO}_3^- - \text{N}, \mu\text{g N L}^{-1}) = \beta_0 - 0.031(\text{year} - 2000)$, $R^2 = 0.72$, $p < 0.001$ (equivalent to -3.1%/yr decline); (K01A-P5): $\ln(\text{NO}_3^- - \text{N}, \mu\text{g N L}^{-1}) = \beta_0 - 0.018(\text{year} - 2000)$, $R^2 = 0.45$, $p = 0.003$ (-1.8%/yr decline). Panels B and C: No significant temporal trends detected. Lines show linear trends with 95% confidence intervals. 74

Figure 3.5: Total nitrogen and phosphorus fertilizer applications to croplands in the Kings Creek watershed (1978-2022). Panels A and B show fertilizer inputs to all cropland areas upstream of HIKX monitoring site; panels C and D show inputs to fields immediately adjacent to ALC-A5 groundwater wells. Blue circles connected by dashed lines represent annual applications; orange circles connected by solid lines represent 5-year running means. R^2

values indicate weak correlations for declining trends. Fertilizer applications ceased entirely by 2015 following sequential cropland restoration.....	75
Figure 3.6: Nutrient concentrations (nitrate, soluble reactive phosphorus (SRP), and total phosphorus (TP) between two sites - SFKC, in which bison graze the sub-watersheds, and NFKC, which is ungrazed (n=1044 paired samples for SFKC vs NFKC comparisons, from 1999 to 2022). The line in the box plot indicates the median, 25th and 75th percentiles are the top and bottom of the box, and the whiskers give the 95th and 5th percentiles.	76
Figure 3.7: Stream nutrient concentrations in Shane Creek watershed comparing periods when cattle were present versus absent (2009-2022): A) nitrate, B) soluble reactive phosphorus (SRP), and C) total phosphorus (TP) (n=697). Box plots show median (center line), 25th and 75th percentiles (box boundaries), and 5th and 95th percentiles (whiskers). Gray lighter boxes represent cattle present periods; purple darker boxes represent cattle absent periods.	77
Figure 4.1: Conceptual framework linking fire frequency, grazer identity, and riparian condition to organic and inorganic sediment delivery pathways in tallgrass prairie streams. Solid yellow lines indicate negative relationships; dashed blue lines indicate positive relationships.	93
Figure 4.2: Time series of background-corrected sediment (gold) and bromide (blue) concentrations at the downstream sampling point during ten rainfall simulation experiments. BDL panels (Exps 2, 3, 4, 7) display sediment data only; the bromide signal did not exceed the 95% confidence interval of background stream concentrations and is not plotted. Site characteristics (vegetation cover class, slope, clay content) are shown above each panel.	106
Figure 4.3: Nonlinear vegetation cover threshold on areal erodibility (Sed/m ²) across rainfall simulation experiments. log ₁₀ (Sed/m ²) plotted against vegetation cover (%) for the six experiments with detectable bromide transport. Point color denotes vegetation cover class; symbol shape distinguishes loamy sand (circles) from sandy loam (squares). Yellow band marks the 60–70% transition zone, across which areal erodibility shifts by approximately two orders of magnitude.	108
Figure 4.4: Divergent environmental controls on organic versus inorganic suspended sediment fractions revealed by generalized additive model (GAM) analysis. Heatmap of signed log ₁₀ (p-value) for each predictor (rows: log discharge, precipitation, days since burn,	

watershed area) across four response variables (columns: TSS, VSS, ISS, VSS/ISS ratio). Blue shading indicates significant positive effects; red shading indicates significant negative effects; color intensity is proportional to statistical significance. Upward triangles denote positive effects; downward triangles denote negative effects..... 112

Figure 4.5: Predicted concentrations from generalized additive models showing the effects of log-transformed discharge (a–d), 30-day cumulative precipitation (e–h), and days since last prescribed burn (i–l) on TSS (blue), VSS (green), ISS (red), and the VSS/ISS ratio (purple) by watershed. In panels (a–d), solid lines and shaded bands represent high-precipitation conditions (>75th percentile) and dashed lines represent low-precipitation conditions (<25th percentile), illustrating the precipitation $\times$ discharge interaction. Shaded bands represent 95% confidence intervals throughout. Watersheds: N02B (bison, 2-year burn; blue), N04D (bison, 4-year burn; green), Shane Creek (cattle, 3-year burn; pink). The significance of each smooth term is indicated in each panel..... 114

Figure 4.6: Structural equation model (SEM) depicting the pathway structure linking climate drivers, watershed properties, and fire management to organic (VSS) and inorganic (ISS) suspended sediment fractions and their compositional ratio (VSS/ISS). Standardized path coefficients (β) are shown along each arrow, with line thickness proportional to effect magnitude. Blue arrows indicate significant positive paths; red arrows indicate significant negative paths. Dashed green arcs represent residual covariances between response variables. All paths significant at $p < 0.001$ 117

Figure 4.7: Streambank geomorphic attributes across four KPBS watersheds under contrasting fire and grazing management. Top row: box plots of (a) bank slope (m/m), (b) weighted soil texture score, and (c) average bare soil score. Kruskal-Wallis test statistics and p-values are shown. Bottom row: bar plots of (d) bank cut frequency (%), (e) grazer trail frequency (%), and (f) canopy cover frequency (%), with chi-square test statistics and p-values. Sample sizes (n) are indicated above each bar. Error bars represent 95% confidence intervals. Watersheds: N04D (bison, 4-year burn), N02B (bison, 2-year burn), K2B (ungrazed, 2-year burn), C3SB (cattle, 3-year burn)..... 119

List of Tables

Table 2.1: Summary of gauged watersheds. Data range represents the first year and the last year of data analysis for each site. Burn frequency at USGS gage is listed as “mixed” because various upstream watersheds in this grassland area feed USGS gage. These sub-watersheds are grazed and ungrazed, and some have variable burn frequencies (data available at 10.6073/pasta/c7bc668437d3bd7035cff8aa85b567a8).	19
Table 2.2: Summary of data sources on factors that potentially influence discharge. Data range represents each site's first year and last year of data analysis.	22
Table 3.1: Summary of site characteristics. The data range represents the dates of each site's first and last analyzed samples. The denotation “variable” is used to describe sites that have irregular burn frequencies or multiple watersheds upstream with different burn frequencies. NFKC and SFKC are the two major tributaries of Kings Creek, and SFKC is bison-grazed and NFKC is ungrazed. The number of nitrate samples (n) at each site is indicated.	60
Table 4.1: Site characteristics, soil properties, precipitation, bromide mass balance, sediment export, and areal erodibility index (Sed/m^2) for ten rainfall simulation experiments at KPBS, 2023–2024. Br^- applied at 82.08 mg L^{-1} (100 g NaBr in 946 L deionized water). BDL = below detection limit (95% CI of background stream Br^-); Experiments 2, 3, 4, and 7 excluded from quantitative analyses.	105

Acknowledgements

This dissertation would not have been possible without the support, guidance, and generosity of many people across two continents.

First and foremost, I thank my advisor, Dr. Walter K. Dodds, for his mentorship over the past 5 years. Walter gave me the freedom to pursue my own questions while always being available with sharp feedback, patient guidance, and the kind of deep ecological thinking that shaped how I see watersheds and streams. I am a better scientist and a more careful thinker because of him. I also thank my advisory committee, Drs John Blair, Keith Gido, and Trisha Moore, for their time, expertise, and constructive challenges that improved every chapter of this work.

I am grateful to my collaborators and coauthors. Sam Zipper and Trisha Moore contributed essential hydrological and engineering perspectives to the streamflow work in Chapter 2. The long-term datasets that form the backbone of this dissertation exist because of decades of careful work by Konza LTER technicians, data managers, and field crews; their largely unseen labor makes research like mine possible.

My colleagues in the Dodds Lab made graduate school both productive and enjoyable. I thank Anne Schechner, James Guinnip, Molly Fisher, Lane Lundeen, Shaun Baughman, Emily Burnett, Brooke Burris, and Madison Moriello for field help, statistical arguments, feedback on drafts, and the camaraderie that sustains people through long Kansas winters and longer writing sessions. Tari Philips, Whitney Finto, Melissa Bruce, and Sara Smith in the Division of Biology kept everything running behind the scenes, and I appreciate their care for graduate students more than they know.

I owe a special debt to the undergraduate researchers I have had the privilege of mentoring. Their curiosity and hard work reminded me regularly of why this work matters.

I acknowledge that research at KPBS takes place on land with deep indigenous history, including the Kaáⁿze (Kaw/Kansa), Wah.Zha.Zhe (Osage), Očhéthi Šakówinj (Sioux), and Kiikaapoi (Kickapoo) peoples. I hope this work honors their long stewardship of this landscape.

I grew up in Bangladesh, far from the tallgrass prairie, and I would not be here without the people who helped me along the way. I thank my teachers at the University of Dhaka, particularly in the Department of Soil, Water, and Environment, for giving me a strong foundation. I thank Dr. Stacey Sloan Blersch and the faculty at Columbus State University for believing in me early and setting me on this path. I am grateful to the Bangladeshi community at Kansas State, and especially the Bangladeshi Student Association, for making Manhattan feel a little more like home.

To my family in Bangladesh, thank you for your unwavering support across the distance. You encouraged me to pursue this dream even when it meant years apart, and every conversation reminded me where I come from and why I keep going.

Dedication

I dedicate this work to my wife, Fahima Khatun, and my daughter, Aakifah Raihan, whose love, encouragement, and sacrifices made this journey possible. Their unwavering support gave me the strength to persevere through every challenge.

Preface

The contents of this dissertation are the product of collaboration with my major advisor, Walter Dodds, as well as coauthors from other institutions and disciplines. Chapter 2 is published in *Ecohydrology* with coauthors Walter Dodds, Sam Zipper, and Trisha Moore. Chapter 3 is under revision at *Freshwater Science* with coauthor Walter Dodds. Chapter 4 is in preparation for a journal with coauthor Walter Dodds. All authors have given consent for chapters to be reproduced within this dissertation.

Chapter 1 - Introduction

Grassland ecosystems cover approximately 40% of Earth's terrestrial surface and generate roughly a quarter of global runoff (Dodds, 1997a; Dodds et al., 2015), yet the streams draining these landscapes remain among the most understudied freshwater systems worldwide (Krabbenhoft et al., 2022). The majority of small stream channels are non-perennial, flowing intermittently in response to the tight coupling among precipitation, evapotranspiration, and subsurface storage that characterizes water-limited landscapes (Messenger et al., 2021; Zimmer et al., 2020a). Because intermittent streams integrate the cumulative effects of climate, vegetation, land use, and disturbance into measurable signals of water quantity, chemistry, and material transport, they function as sensitive indicators of watershed-scale change. Although intermittent streams have received some study, they are highly variable, and long-term records are necessary to assess hydrologic trends and their relationship to water quality and quantity.

Tallgrass prairie once stretched from Texas to Manitoba, but has been reduced to approximately 5% of its historical extent, with much of the remaining tallgrass habitat concentrated in the Flint Hills of Kansas and Oklahoma (Knapp et al., 1998). Streams in this remnant grassland are intermittent, support communities adapted to wetting-drying cycles, and serve as conduits of nutrient and sediment export to the Kansas River and ultimately the Missouri-Mississippi system. While tallgrass prairie is endangered, streams in this habitat are even more so because they require protection of entire watersheds to maintain their integrity (Dodds et al., 2004). Understanding how these streams are changing is therefore relevant not only to local ecology but to baselines for regional water quality and the resilience of one of North America's most imperiled biomes.

This dissertation examines the ecohydrology, biogeochemistry, and sediment dynamics of headwater streams at Konza Prairie Biological Station (KPBS), a 3,487-hectare tallgrass prairie Long-Term Ecological Research site in the Flint Hills of northeastern Kansas. KPBS maintains experimentally controlled treatments of prescribed fire frequency and ungulate grazing over more than four decades, with management boundaries corresponding to sub-watersheds on the site. The main stream, with its watershed bounded entirely within KPBS, is Kings Creek. This watershed is monitored by the only USGS benchmark gauge whose entire drainage lies within relatively pristine grassland (gauge 06879650) and by the Konza Prairie Long-Term Ecological Research program, which has monitored upstream sub-watersheds since the 1980s. This combination of long-term monitoring, experimental management, and benchmark status makes KPBS an unparalleled platform for investigating how interacting environmental drivers regulate grassland stream function.

The three studies comprising this dissertation address a single overarching question: how do climate variability, vegetation change, fire management, ungulate grazing, agricultural legacy, and their interactions influence the flow regime, nutrient chemistry, and sediment composition of tallgrass prairie streams over decadal timescales?

Woody encroachment and the drying of grassland streams

The flow regime of a stream is the master variable governing its ecological structure and biogeochemical function (Poff et al., 1997). In intermittent systems, the duration and extent of zero-flow periods impose additional constraints on biological communities, nutrient processing, and downstream connectivity (Allen et al., 2020; Datry et al., 2014). At KPBS, streamflow is sustained primarily by groundwater discharge through a merokarst aquifer system in which alternating limestone and mudstone layers create a network of fractures, perched aquifers, and

localized groundwater-surface water connections (Gambill et al., 2024; Sullivan et al., 2019, 2020; S. Zipper et al., 2025). Groundwater contributes over 95% of baseflow (Hatley et al., 2023), and the spatial pattern of flow persistence is controlled by subsurface connections operating at scales of tens to hundreds of meters (Swenson et al., 2024). Kings Creek, therefore, does not behave as a simple rainfall-runoff system: the relationship between precipitation and discharge is mediated by subsurface pathways with response times ranging from days to years.

Superimposed on this hydrogeological template is a transformation of the watershed's vegetation that is reshaping the water balance. The expansion of shrubs and trees into historically grass-dominated landscapes, commonly termed woody plant encroachment, is occurring in grasslands globally and is recognized as one of the most consequential ecological transitions of the past century (Archer et al., 2017; Briggs et al., 2005a; Ratajczak et al., 2012). At KPBS, riparian woody cover has increased dramatically, with expansion rates varying by fire frequency: approximately doubling in annually burned watersheds and increasing roughly sevenfold in watersheds burned every four years between 1982 and 2020 (Dodds et al., 2023; Keen et al., 2023). The dominant encroaching species include rough-leaf dogwood (*Cornus drummondii*) in uplands and American elm (*Ulmus americana*), hackberry (*Celtis occidentalis*), bur oak (*Quercus macrocarpa*), and green ash (*Fraxinus pennsylvanica*) in riparian corridors.

The hydrological consequences are substantial. Woody plants transpire at roughly twice the rate of the C₄ grasses they replace (O'Keefe et al., 2020), access deeper soil water and groundwater through extensive root systems (Keen et al., 2022; Nippert & Knapp, 2007), and maintain transpiration later into the growing season than grasses, which senesce earlier in fall. The net effect is an amplification of actual evapotranspiration that reduces water available for streamflow and groundwater recharge.

Chapter 2 of this dissertation (Raihan et al., (2025), published in Ecohydrology) documents these changes and establishes the mechanistic linkage between woody encroachment, amplified evapotranspiration, and declining streamflow. Using four decades of discharge data from the USGS gauge and three upstream LTER sub-watersheds, combined with remotely sensed vegetation time series, modeled ET, and structural equation modeling, this study revealed that daily discharge declined by over half and annual flow days declined by 55%. These declines occurred without a concomitant decrease in precipitation. Riparian woody cover was significantly associated with decreased flow days, and the effect was most pronounced during the July to September period when vegetation water demand peaks. Analysis of excess precipitation and energy following Tomer & Schilling, (2009) revealed temporal shifts in the water-energy balance consistent with progressive woody plant expansion, even in wet years. These findings position woody encroachment as a first-order control on tallgrass prairie stream hydrology and establish the hydrological baseline against which the biogeochemical and sedimentological findings of subsequent chapters must be interpreted.

Fire, grazing, and agricultural legacy as controls on nutrient chemistry

If hydrology defines the physical template of a stream ecosystem, nutrient chemistry defines its biogeochemical character. Nitrogen and phosphorus concentrations reflect the integrated effects of atmospheric deposition, soil processes, and plant uptake. In managed landscapes, the inputs and legacies of fertilization, grazing, and land use change further influence stream nutrient concentrations (Alexander et al., 2007; Dodds & Oakes, 2006). In tallgrass prairie, fire and grazing have co-evolved as dominant disturbance agents for millennia, but understanding their influences on stream nutrient dynamics requires long-term data spanning gradients of management intensity and land use history.

Fire maintains grass dominance by top-killing woody seedlings, volatilizes most aboveground-litter nitrogen while recycling phosphorus and other less-volatile elements through ash, and stimulates vigorous C₄ grass regrowth (Blair, 1997; S. L. Collins & Wallace, 1990; Knapp et al., 1999). Its effects on stream chemistry, however, are less straightforward. Repeated frequent burning at Konza reduces net soil nitrogen mineralization and enhances plant–microbial nitrogen retention (Blair, 1997; Turner et al., 1997), although individual burns can transiently elevate surface nitrate through short-term nitrification pulses (Wan et al., 2001).

Ungulate grazing adds further complexity through mechanistically distinct pathways. Both bison and cattle concentrate nutrients through waste deposition and modify vegetation in ways that alter runoff and nutrient delivery to streams (Frank et al., 2016; Hobbs, 1996). The two grazers differ, however, in their spatial behavior. Cattle preferentially occupy riparian zones during hot periods, physically disturbing banks and creating direct conduits for nutrient-laden runoff (Belsky et al., 1999; Grudzinski & Daniels, 2018; Trimble & Mendel, 1995). Bison move more broadly across the landscape with preferential grazing of recently burned patches, distributing impacts across upland and lowland positions (Allred et al., 2011a; Fuhlendorf & Engle, 2001). Prior to this dissertation, only two studies had compared bison and cattle effects on stream water chemistry (Grudzinski et al., 2018; D. M. Larson et al., 2013), neither spanning more than a few years.

A third influence, often the dominant one in the Great Plains, is the legacy of historical cropland agriculture. Fertilized cropland drives elevated stream nitrate and phosphorus (Banner et al., 2009; Dodds & Oakes, 2006, 2008), and recovery after agricultural abandonment is often neither immediate nor complete. Nutrient legacies persist for decades in soils, vadose zones, and shallow groundwater (Hamilton, 2012; Meals et al., 2010; Van Meter et al., 2016, 2017), but the

timescales of recovery are poorly constrained in grassland systems because few sites combine long-term, high-frequency water chemistry monitoring with documented, sequential land use change.

Chapter 3 of this dissertation (Raihan and Dodds, under revision at Freshwater Science) addresses these gaps using 37 years of stream and groundwater chemistry data from sites spanning gradients of burn frequency, grazer identity, and land use history within the Kings Creek and Shane Creek watersheds. The results revealed a clear hierarchy: agricultural legacy produced the largest and most persistent effects, with stream nitrate below restored croplands declining 75% over 25 years, but groundwater nitrate remained near $950 \mu\text{g N L}^{-1}$ more than two decades after fertilization ceased. These results imply subsurface residence times far exceeding the 4 to 5 year surface water response. Bison grazing roughly doubled stream nitrate relative to ungrazed watersheds, substantially exceeding the 35% increase reported over a 4-year record by Kemp et al. (2001). Cattle presence was associated with a 33% increase in nitrate and elevated total phosphorus, mediated through direct riparian disturbance. Fire frequency had minimal direct effects on stream nutrients but interacted with grazing to shape long-term trajectories.

Sediment composition as an indicator of watershed disturbance

Suspended sediment is the third major material flux linking terrestrial and aquatic environments in grassland watersheds. Elevated sediment, particularly inorganic suspended solids (ISS), can reduce light penetration, smother benthic habitat, impair invertebrate and fish communities, and serve as a vector for phosphorus transport (Bilotta & Brazier, 2008; Jones et al., 2012; P. Kemp et al., 2011; Wood & Armitage, 1997). In contrast, many stream invertebrates directly rely on suspended organic particles (e.g., filter feeders). Yet the mechanistic controls on sediment in grassland streams have received far less attention than nutrient dynamics, and most

work treats total suspended solids as a single metric without distinguishing its organic and inorganic components.

This distinction is ecologically meaningful. Volatile suspended solids (VSS), representing organic matter, derive primarily from in-channel and near-channel sources such as leaf litter, algal biomass, and decomposing riparian material. These materials may be diluted or washed away at high flows (Dodds et al., 2004; Webster & Meyer, 1997). Inorganic suspended solids (ISS) originate from physical disturbance: overland erosion, bank failure, and mineral soil exposure by fire, trampling, or vegetation loss (Pierson et al., 2011; Williams et al., 2020). The relative abundance of VSS and ISS therefore provides information about sediment source and delivery mechanism that is lost when only TSS is measured.

Riparian zones exert disproportionate control over sediment delivery relative to their small areal footprint. Intact corridors reduce sediment inputs by 50 to 90% compared to unvegetated margins (Naiman & Décamps, 1997; Sweeney & Newbold, 2014). In tallgrass prairie, riparian structure is shaped by the same fire-grazing interactions that regulate hydrology and nutrients, creating the potential for coupled responses across all material fluxes. Dodds et al., (2025) recently argued that riparian function in tallgrass prairie cannot be characterized by any single gradient but emerges from the interaction of vegetation cover, slope, soil properties, hydrology, and management history.

Chapter 4 of this dissertation (Raihan and Dodds, in preparation) tests and extends this multi-gradient framework using three complementary datasets: (1) long-term records of stream suspended sediments, (2) rainfall simulation experiments in the riparian zone, and (3) channel surveys in grazed and ungrazed streams to document the frequency of bare soil near streams. Rainfall simulation experiments with bromide tracer quantified overland sediment delivery

across gradients of vegetation cover, slope, and grazing history, revealing a nonlinear threshold of erodibility between 50% and 75% cover. The use of sediment and bromide calculation, developed in these experiments, separates material erodibility from hydrological connectivity, a distinction that previous grassland studies have been unable to make. Fifteen years of suspended sediment monitoring, analyzed with generalized additive models and structural equation models, confirmed that VSS and ISS respond to different controls: discharge drives organic sediment through dilution, while fire management drives inorganic sediment through mineral soil exposure. Streambank surveys documented systematic differences in bank geometry and erosion susceptibility linked to grazer identity and fire regime.

Crosscutting themes and dissertation structure

Although the three studies address distinct response variables, several themes emerge: fire–grazing interactions regulating stream function, the importance of multi-decade records for detecting slow directional change, threshold-like responses, and coupling among water, nutrient, and sediment fluxes. These crosscutting findings are developed fully in Chapter 5.

This dissertation is organized as follows. Chapters 2, 3, and 4 are stand-alone manuscripts corresponding to the three studies described above. Chapter 2 (Raihan et al., (2025); published in *Ecohydrology*) addresses streamflow and drying. Chapter 3 (Raihan and Dodds; under revision at *Freshwater Science*) addresses nutrient dynamics. Chapter 4 (Raihan and Dodds; in preparation) addresses sediment transport. Chapter 5 synthesizes the findings and discusses management implications, limitations, and future directions. A consolidated list of all works cited across the dissertation is provided at the end of the document.

Chapter 2 - Controls on Discharge and Drying in an Intermittent Grassland Stream: Temporal and Network Variability

Abstract

Intermittent streams are prevalent worldwide, yet the understanding of drivers of their changing flow patterns remains incomplete. We examined hydrological changes spanning four decades (1982-2020) in Kings Creek, an intermittent grassland stream within the Konza Prairie Biological Station (KPBS) in Kansas, USA. We analyzed streamflow data from a U.S. Geological Survey gage on Kings Creek and three upstream LTER sub-watersheds with annual, biennial, or quadrennial burn frequencies, and linked trajectories of woody encroachment to increased evapotranspiration and changes in streamflow. Riparian woody cover doubled in the annually and biannually burned sub-watersheds, and seven-fold in the quadrennially burned watersheds. We observed significant decreases (84%) in daily discharge and number of annual flow days (55%) at the downstream USGS Kings Creek gage, with similar changes in the LTER sub-watersheds. The changing riparian cover, propelled by the regional expansion of woody plants, contributed to decreased streamflow by amplifying actual evapotranspiration (ET). Seasonal assessments underscored the critical influence of late summer conditions (July-September), under which increases in ET were linked to rising temperatures and increased evapotranspiration by riparian cover. Our results highlight the significant hydrological impacts of woody encroachment in grasslands and emphasize the importance of long-term ecohydrological monitoring in unraveling the interplay between climate and vegetation as controls on the hyper-variable flow patterns in this intermittent stream. Predicting and managing hydrological impacts

on the flow of intermittent grassland rivers and streams worldwide requires accounting for the effects of accelerating woody encroachment.

Introduction

Non-perennial streams are common in the majority of river networks worldwide (Messenger et al., 2021) and are under-studied and under-monitored with respect to hydrology (Krabbenhoft et al., 2022). Recent studies have begun quantifying the intermittency of stream networks (Leigh et al., 2016; Shanafield et al., 2020), though less is known about changes in intermittency over time, particularly as related to global change (Sauquet et al., 2021; Trambly et al., 2021). Zipper et al., (2021) documented changes in intermittency in US streams over time using long-term USGS gaging station data and identified increased aridity as a primary factor driving increased stream drying. Across 540 sites, they demonstrated that streams in more arid regions exhibited trends towards longer dry periods over the past ~40 years. Their analysis controlled for watershed characteristics and land use changes, allowing them to isolate climate aridity as the dominant driver of increasing intermittency trends across the continental United States. However, other potential drivers of temporal change in stream intermittency, particularly the effect of vegetation mediated by evapotranspiration, are not as well-studied (McCabe & Wolock, 2002; Rodgers et al., 2020).

The timing and rate of stream drying have strong policy implications and heavily influence biogeochemical cycles (Von Schiller et al., 2011), as well as ecological characteristics (Price et al., 2024). Therefore, the timing and duration of “zero flow” (periods when water discharge is not measurable or falls below detection thresholds in otherwise flowing channels) are commonly used in studying intermittent water bodies (Zimmer et al., 2020a). Understanding the patterns and causes of intermittent stream flow is important since drying is becoming more prevalent, and drying patterns vary in both space and time (Allen, et al., 2020; Jaeger et al., 2014; Perkin et al., 2017). Although climate is a key factor influencing non-perennial stream

flow regimes (Dodds, 1997b; Hammond et al., 2021), other factors can also lead to stream drying, including anthropogenic activities (Datry et al., 2023; Eng et al., 2019), and human water use (Datry et al., 2014; Zipper et al., 2022).

Grassland rivers and streams are vital components of the Great Plains ecosystem and other grasslands worldwide, and are dominated by intermittent streams (Dodds et al., 2004). The Great Plains is well-suited for understanding drivers of stream drying due to its extensive stream network, long-term hydrological records, well-documented woody encroachment, and natural climate gradient that creates conditions ideal for examining interactions between changing vegetation and hydrology. However, woody plant encroachment is occurring in grasslands globally, altering infiltration rates, soil water storage, and streamflow, which can lead to reduced water availability and streamflow duration (Zou et al., 2014). The Kings Creek gage at the Konza Prairie Biological Station (KPBS) is the only “benchmark” gauging station in the United States Geological Survey (USGS) network with its entire watershed in a relatively pristine grassland. It therefore represents an important sentinel gage for estimating wider changes across grassland ecosystems.

Kings Creek has experienced the expansion of woody vegetation in its watershed (Veatch et al., 2014, 2015) as well as changes in hydrology over the past four decades. Riparian shrub cover increased by about 57% from 1978 to 2020 (Keen et al., 2023), while mean discharge and number of flow days decreased from 1980 to 2010, with no concomitant decrease in annual precipitation and a modest increase in temperature (Dodds et al., 2012). However, it remains unknown to what extent increasing stream intermittency can be attributed to vegetation changes and during which seasons these effects are most pronounced, largely because few datasets

simultaneously track both vegetation dynamics and hydrological metrics at appropriate scales and timeframes to capture these gradual ecological transitions.

Our overarching research goal was to investigate the relative importance of changes in vegetation and climate as controls over the changing hydrology in the Kings Creek watershed from 1982 to 2020. We hypothesized that discharge can be impacted within this watershed by interrelated changes in precipitation, temperature, and vegetative cover, with variable impacts as a function of position in the watershed (Fig. 2.1). We base our study on one United States Geological Survey (USGS) discharge measurement station at the bottom of the watershed and discharge measurements at three upstream sub-watersheds (N1B, N2B, and N4D) with different historical trajectories of woody vegetation encroachment. We also used additional long-term data on environmental factors collected by the Konza Long-Term Ecological Research program. We hypothesized that, along with precipitation and temperature, the variation in the vegetation composition of these upstream sub-watersheds would influence evapotranspiration and flow regimes (defined in terms of annual or seasonal mean discharge and number of zero flow days). As woody vegetation increases, we predict that higher evapotranspiration will lead to decreased discharge and fewer flow days in a year. Our research objectives were to: 1) characterize changing discharge patterns and identify during which seasonal meteorological factors have the greatest impact on changes in discharge; 2) assess changes in meteorological factors that could influence hydrology; and 3) link changes in meteorology and vegetative cover to discharge at the watershed outlet and upstream sub-watersheds.

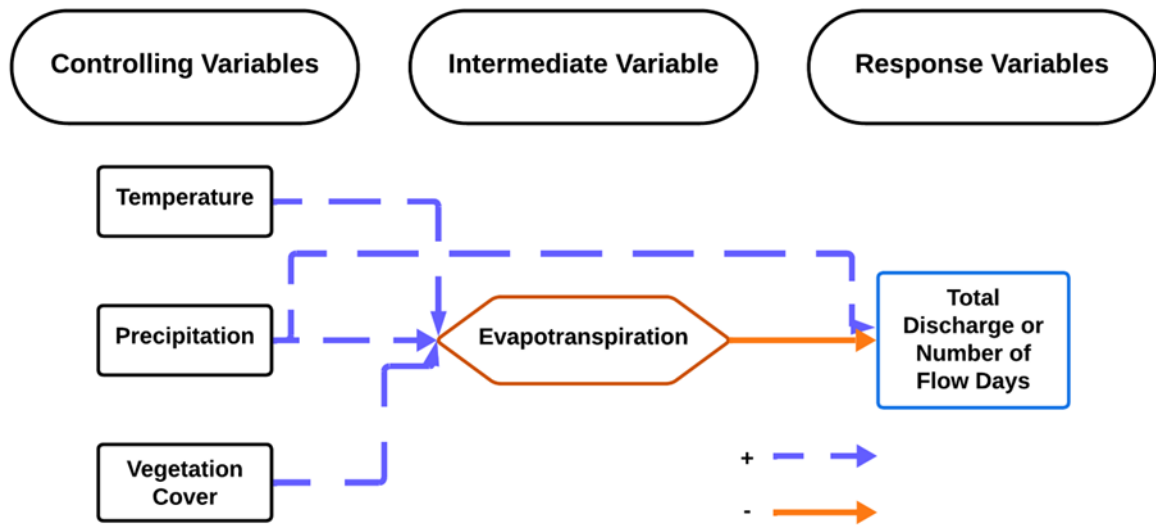


Figure 2.1: Conceptual model of our study. The purple dashed arrows represent the hypothesized positive influences, and the orange solid lines represent the hypothesized negative influences of the controlling and intermediate variables on discharge and number of flow days.

Methods

Study area description

Kings Creek has a 10.6 km² drainage area entirely within KPBS. KPBS is a 3487-ha tallgrass prairie located in the Northern part of the Flint Hills region near the city of Manhattan, Kansas (39.1N and 96.9W), co-owned by Kansas State University and The Nature Conservancy (Fig. 2.2). The site experiences a mid-continental climate, receiving roughly 800 mm of precipitation annually as averaged from 1982 to 2020 (Table 2.2). Most precipitation falls from April to September. Silty-clay loam soils are over two meters deep in lowland areas and at the bottoms of stream valleys (Vinton, 1999). Apart from riparian corridors on larger streams and rivers, this area of the Flint Hills historically consisted mostly of native tallgrass prairie and little woody vegetation except. Although woody plant expansion is ongoing, the dominating perennial C₄ grasses (*Schizachyrium scoparium*, *Sorghastrum nutans*, *Andropogon gerardii*, *Panicum virgatum*) as well as sub-dominant grass, forb, and woody species, are the primary vegetation of KPBS's regularly burned uplands (Abrams, 1986; S. L. Collins & Calabrese, 2012). The dominant woody species in upland areas consists primarily of rough-leaf dogwood (*Cornus drummondii*), smooth sumac (*Rhus glabra*), and eastern redcedar (*Juniperus virginiana*), while riparian corridors are characterized by a more diverse woody community including American elm (*Ulmus americana*), hackberry (*Celtis occidentalis*), bur oak (*Quercus macrocarpa*), and green ash (*Fraxinus pennsylvanica*). For this study, riparian areas were delineated based on topographic position within 30 meters of stream channels, where greater soil moisture and protection from fire historically allowed for the establishment of woody vegetation even in otherwise grass-dominated landscapes.

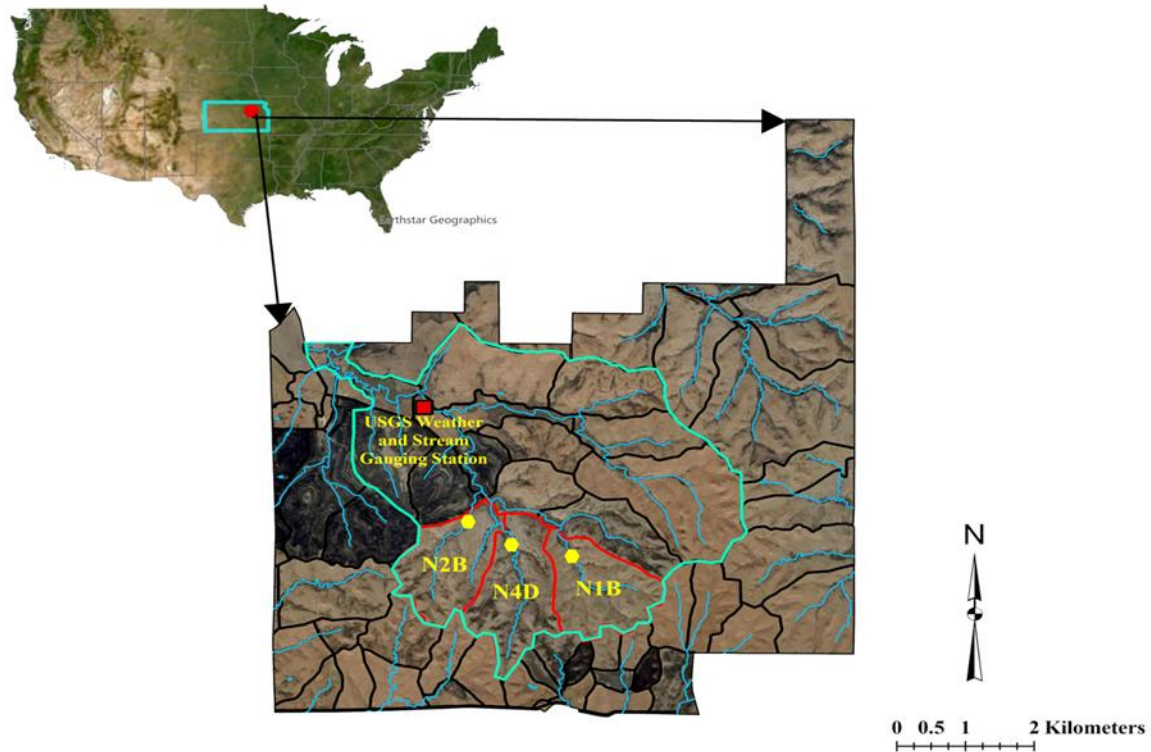


Figure 2.2: USGS Kings Creek watershed within the Konza Prairie Biological Station, located in Kansas, USA, including the Long-Term Ecological Research (LTER) watersheds N1B, N2B, and N4D. The U.S. Geological Survey has consistently monitored the USGS weather and stream gaging station (red square) at this site since 1979 (Gage 06879650, red square). The yellow dots denote sub-watershed weirs, serving to measure discharge levels since 1987.

KPBS features a merokarst geology where the weathering of limestone bedrock layers has produced a complex network of fractures, joints, and perched aquifers. Mudstone layers separate these weathered limestone bedrocks, creating a shallow groundwater table (typically 1-5 meters below the surface in valleys, and 5-15 meters in upland areas) that is well-connected to the Kings Creek stream network. Groundwater residence times vary from days to months in shallow flow paths to several years in deeper aquifers (Sullivan et al., 2019, 2020; Vero et al., 2018). However, connections between the groundwater system and stream network are highly localized (Gambill et al., 2024), typically occurring at scales of tens to hundreds of meters where permeable limestone layers intersect the surface. Recent modeling demonstrated that these

localized connections create discrete zones of groundwater discharge supporting baseflow in specific reaches while nearby stream segments remain dry (Swenson et al., 2024; Zipper et al., 2025). Smaller upstream watersheds typically experience more zero flow days, except for short spring-fed reaches (Costigan et al., 2015).

Watersheds analyzed and data sources

Kings Creek is monitored by a USGS stream gage (06879650 KINGS C NR MANHATTAN, KS) and has been serving as a hydrological benchmark since 1979. The watershed drains native tallgrass prairie and has an extensive oak riparian forest in its lower reaches. Approximately half of the watershed upstream of the gaging station is under bison grazing, and the rest is ungrazed. We compiled discharge data published on the USGS website for the period from 1982 to 2020, consistent with the available meteorological data from the nearby (< 1 km) Konza LTER weather station.

We also have three upstream sub-watersheds in which we have measured discharge using weirs since 1987. These weirs are in intermittent, third-order headwater streams (N1B, N2B, and N4D) and were used to characterize discharge variation in sub-basins relative to the downstream USGS station (Table 2.1). These three small watersheds have been bison-grazed since 1992 and burned every 1, 2, or 4 years (N1B, N2B, and N4D, respectively). Data for these three weirs are publicly available (see Table 2.1). Discharge at these weirs is estimated by water height in v-notched weirs using Druck pressure transducers from 1987 to 2013 and from 2013 to the present using a YSI WaterLOG Bubbler/Pressure Sensor H-3553. During flow periods, discharge is monitored at each site every five minutes, and the sensors are calibrated against manual height readings roughly three times a week during flow. We experienced a few occasions when our depth equipment malfunctioned. For example, about 1.2% of the 95,300 days of data

were missing from watershed N4D; however, since we used annual or three-month averages of daily values (see results) this minimizes the influence of missing data.

Table 2.1: Summary of gauged watersheds. Data range represents the first year and the last year of data analysis for each site. Burn frequency at USGS gage is listed as “mixed” because various upstream watersheds in this grassland area feed USGS gage. These sub-watersheds are grazed and ungrazed, and some have variable burn frequencies (data available at 10.6073/pasta/c7bc668437d3bd7035cff8aa85b567a8).

Site	Lat. and Long.	Data range	Data set name	Watershed area (Ha)	Target burn frequency	Grazers
USGS Gage	N 39.060747, W 96.354088	1982-2020	06879650	1060	Mixed	Bison and Ungrazed
N1B Weir	N 39.08656, W 96.57703	1987-2020	DOI: 10.6073/pasta/7b75a0efa4617f7d34dd9f2f583b686a	121	Every Year	Bison
N2B Weir	N 39.08995, W 96.58900	1987-2020	DOI: 10.6073/pasta/cff758a0736702b20d9b6338e9bbbc25	119	Every 2 Years	Bison
N4D Weir	N 39.08735, W 96.5844	1985 - 2020	DOI: 10.6073/pasta/c01278aaaac74572ee53a4a6ba3017d3	135	Every 4 Years	Bison

We also calculated the proportion of riparian woody cover over time in watersheds N1B, N2B, and N4D based on data for woody expansion from Dodds et al. (2023) and Keen et al. (2023). These data were created from a time series of remotely sensed images ground-truthed to plant surveys for three watersheds which bracket the burn frequencies of most watersheds in Kings Creek (1, 2, and 4-year burn frequencies). The aerial imagery was consistently collected at 1-meter spatial resolution across all time periods (1982-2020), with images standardized for seasonal timing (predominantly late spring/early summer acquisitions) to minimize phenological differences in vegetation appearance. The classification of woody vegetation was verified through extensive field validation points (>200 points per watershed) to ensure accuracy across the time series.

To extrapolate these measurements to the entire Kings Creek watershed, we applied a burn frequency-based approach. For sub-watersheds within Kings Creek that were burned annually, biennially, or quadrennially, we directly applied the woody cover values from N1B, N2B, and N4D, respectively. For sub-watersheds with intermediate burn frequencies (e.g., 3-year burns), we used linear interpolation between the nearest burn frequency categories. For approximately 15% of the Kings Creek watershed with irregular fire histories or burn frequencies outside the 1-4 year range, we used the values from the 4-year burn treatment (N4D) as a conservative estimate, as these areas typically experience greater woody encroachment. We then calculated a weighted average based on the proportional area of each burn frequency within the entire Kings Creek watershed to estimate overall woody cover. This approach accounts for the varying management histories across the watershed while being constrained by our detailed measurements from the representative sub-watersheds.

While our gauged sub-watersheds only covered 35.5% of the USGS watershed, the woody vegetation results are similar to an earlier study that assessed riparian woody expansion with aerial imagery of all watersheds feeding the USGS station over a shorter period (Veatch et al., 2014, 2015). The data we used covered a longer timespan than the earlier study and allowed the use of our full record of discharge data. The N2B watershed was selected for a riparian woody vegetation removal experiment beginning in 2010. Vegetation within 30 m of the Kings Creek streambed and 10 m within the side channel of the creek was removed mechanically (Dodds et al., 2023; Larson et al., 2019). Vegetation was cut along 4.8 km of stream channel, roughly 21% of the total watershed area, and cutting was repeated every 2 years. The cutting was not effective in decreasing woody vegetation cover, as the shrubs sprouted rapidly and had no observable effect on stream hydrology (Dodds et al., 2023).

We also collected meteorological data from the KPBS headquarters dataset AWE012, which monitors daily mean, maximum, and minimum air temperature, relative humidity, total precipitation, total solar radiation; mean, maximum, and minimum soil temperature, and mean wind speed. These data are mostly complete with very few days missing. When equipment malfunctions, it is repaired or replaced rapidly as many researchers rely on these data. We also used the Konza dataset AET01 which reports modeled daily grass-based reference evapotranspiration (ET) using meteorological data from AWE012 including wind speed, temperature, precipitation, and solar radiation under non-limiting water conditions for a hypothetical short grass (height of 0.12 meters, a surface resistance of 70 s m^{-1} , and an albedo of 0.23) based on meteorological conditions at KPBS from AWE012 (See Table 2.2).

Table 2.2: Summary of data sources on factors that potentially influence discharge. Data range represents each site's first year and last year of data analysis.

Sample type	Data set name	Locations	Data range	Data citation
Meteorological data	AWE012	Konza Prairie headquarters weather station	1982-2020	http://doi.org/10.6073/pasta/743c6b205e38a087bc54925ed258f549
Riparian tree and shrub cover	RIV06	Watershed N1B, N2B, and N4D	1982-2020	https://doi.org/10.6073/pasta/682b8d7a78aa814e96dba672cdabd34c
KPBS daily ET	AET011	Konza Prairie headquarters weather station	2000-2020	http://doi.org/10.6073/pasta/b5a57131dc37d035b7c14ec7a8a49604

Calculation of evapotranspiration, excess water, and excess energy

We calculated ET for every season for statistical analyses of effects of grass and wood ET. Grass ET was calculated by multiplying daily reference ET obtained from the AET01 dataset by a crop coefficient (K_c). Daily K_c values representing tallgrass prairie water use across the growing season used in this study were developed by Hutchinson et al. (2008) at KPBS and vary as a function of day of year. These values follow a seasonal pattern, ranging from 0.2 during the non-growing season (November-March) to 1.1 during peak water use (day of year 189-206, mid-July), with gradual increases during spring green-up and decreases during fall senescence. We applied these day-specific K_c values to the corresponding reference ET value rather than using a single coefficient for the entire growing season. We note that calculated grass ET

represents water use under non-limiting water conditions, since Hutchinson et al (2008) model was used to calculate the expected water use when soil water is not limited (Hervé-Fernández et al., 2023).

Woody vegetation has greater rates of ET than grasses at this site based on work by O’Keefe et al., (2020), who identified roughleaf dogwood (*Cornus drummondii*), as the primary woody encroacher in upland areas of this grassland and a common species in early woody riparian encroachment. Under field conditions at our site, rough-leaf dogwood demonstrates a notably higher canopy transpiration rate of 2.01 mm/day in contrast to the dominant grass species, big bluestem, which transpires at a rate of 0.91 mm/day during the growing season (O’Keefe et al., 2020). Based on these data we assumed woody vegetation has two times greater rates of ET than grass modeled in the ET dataset (AET01). This is a conservative estimate because the grasses go dormant earlier than the woody vegetation. We acknowledge this simplification introduces uncertainty, as actual ET ratios likely vary with vegetation age, density, topographic position, and seasonal conditions.

We employed exponential equation regression models to analyze the temporal changes in riparian and whole watershed vegetative cover based on remote-sensed, ground-truthed image analyses (Dodds et al., 2023; Keen et al., 2023). The independent variable was year, and the dependent variable was the proportion of vegetation cover. When we averaged the values across all three watersheds, an exponential model exhibited an excellent fit to the change in vegetative cover data over time ($R^2 = 0.97$). We then calculated three distinct ET values: one for grass ET, and second for grass + riparian woody cover ET, and a third for ET accounting for all woody cover in the entire watershed. We separated out woody riparian cover because riparian vegetation

may be able to access stream water as well as groundwater before it enters the streams (Keen et al., 2022) .

We also determined excess precipitation and energy according to Tomer & Schilling (2009). We note that this is a separate ET calculation from that for the data used for the statistical analyses. The calculation required estimating reference evapotranspiration (ET_0) with the Hargreaves method (Hargreaves & Allen, 2003; Hargreaves et al., 1985) for the Kings Creek watersheds using temperature data collected by the KPBS weather station. We used this method to remain consistent with the original Tomer & Shilling (2009) methodology to assess temporal trends in excess energy and precipitation with the expansion of woody vegetation.

$$PET = 0.0023R_a \left(\frac{T_{max} - T_{min}}{2} + 17.8 \right) \sqrt{T_{max} - T_{min}} \dots \dots \dots (1)$$

Where PET is in mm day⁻¹, R_a is the extraterrestrial radiation (MJ m⁻² day⁻¹), T_{max} is the daily maximum air temperature (°C), and T_{min} is the daily minimum air temperature (°C). ET_0 was calculated on a daily time step and then aggregated to seasonal and annual values for analysis.

The availability of unused water and energy are a function of precipitation (PPT) and ET, and can fluctuate based on vegetation conditions (Tomer & Schilling, 2009). Following Tomer & Schilling (2009), we normalized this unutilized portion on a scale from -1 to 1 where 1 is complete excess, 0 is balanced, and -1 is the complete deficit of precipitation (P_{ex}) or Excess Evaporative Demand (E_{ex}) according to the equations:

$$P_{ex} = \frac{(PPT-ET)}{PPT} \dots \dots \dots (2)$$

$$E_{ex} = \left(\frac{PET-ET}{PET} \right) \dots \dots \dots (3)$$

To ensure valid water balance calculations, we focused on wetter years (those exceeding the 60th percentile of annual precipitation, $n=12$ years between 2001-2020). We excluded specific drought years (2002, 2003, 2006, 2011, 2012, 2013, and 2018), which showed consistent patterns of low precipitation coinciding with decreased discharge and flow days. This selection was necessary because the Tomer & Schilling (2009) method assumes that soil water storage remains relatively constant between years. During dry years, a significant portion of subsequent precipitation replenishes soil moisture deficits and recharges shallow aquifers rather than contributing to streamflow or ET, which would confound the water balance calculations. By focusing on consistently wet years, we minimized the effects of year-to-year variations in soil and groundwater storage, allowing us to more directly analyze changes in precipitation-ET partitioning.

Statistical Data Analysis

We transformed non-normal data when necessary. Log10 transformation of discharge and riparian cover normalized the data for all statistical analyses, including regression, correlation, ANCOVA, and structural equation modeling. This transformation addressed non-normal distributions and unequal variances, meeting critical assumptions for our entire analytical framework. To minimize issues with zero-inflated data, we added 0.0001 cms (one unit less than our minimum detection limit) to each daily discharge value to retain information on daily mean discharge days with zero flow when using the log transformation.

We controlled for temporal autocorrelation in discharge data using an analysis mean daily discharge from the USGS gaging station over the entire period of record. We offset the temporal series of data against itself starting at ten days and increasing the sliding window offset by ten-day increments up to 90 days. We used this approach to find the time lag where temporal

autocorrelation was no longer statistically significant ($p > 0.05$). This threshold indicates the point at which past daily discharge no longer significantly influenced current daily discharge, meaning observations separated by this time lag could be considered statistically independent of each other.

Once we determined the length of time lag for autocorrelation, we used this window length for subsequent analyses. We moved the window by month through the year to see if varying the window start date influenced the relationship of change in discharge over time to assess if there were natural breaks or seasons with differences in discharge patterns. We then used the results from the USGS Kings Creek site to divide the data to account for seasonal factors for this and the upstream LTER sub-watersheds.

We focused on discharge and the number of flow days as our two key response (dependent) variables and their relationships with potential drivers (precipitation, temperature, riparian cover and ET) over time for both USGS Kings Creek and LTER sub watersheds for each period determined based on the autocorrelation analysis. We first explored the dataset with Pearson correlation for both USGS Kings Creek and individual LTER sub-watersheds to establish correlations among discharge and flow days with year, precipitation, temperature, and riparian cover by seasons. As the solar radiation and humidity are directly proportional to the temperature, we removed solar radiation and humidity from our further analysis. Precipitation and temperature were most strongly correlated with discharge, so we used Analysis of Covariance (ANCOVA) with year, precipitation, and temperature as the continuous variables, and season as an independent categorical variable to assess if the season interacted with the continuous variables to predict discharge or the number of flow days (i.e. if overall trends were

strongly influenced by specific seasons). We also analyzed temperature and precipitation as separate dependent variables to establish temporal trends in these variables with time.

We assessed the independent variables discharge and seasonal number of flow days against year, precipitation, temperature, and riparian cover with regression. Then we used ANCOVA to explore discharge and the number of flow days at LTER sub watersheds as a function of season (categorical) with continuous variables of year, temperature, and precipitation and interactions of all variables. We analyzed the proportion of woody riparian vegetation in each LTER sub-watershed to see how this impacted the discharge. Finally, we performed ANCOVA for discharge and flow days as a function of the sub-watersheds (N4D, N2B, and N1B) with continuous variables of year, temperature, and precipitation.

We also employed ANCOVA to assess the influence of vegetation on the USGS Kings Creek gage. Initially, we analyzed the evapotranspiration (ET) rates for the entire watershed, specifically focusing on grass ET measured in mm/3 months (ET_{grass}) with season as a categorical variable and year as a continuous variable. Subsequently, we expanded the analysis to include both grass ET and riparian area woody cover ($ET_{grass+wood}$), examining their combined effects with the same categorical and continuous variables. Finally, we conducted an ANCOVA to explore the influence of grass ET while incorporating the total proportion of woody cover across the entire watershed ($ET_{total\ proportion\ wood}$). For this analysis, we employed temporal interpolation using exponential regression modeling ($R^2 = 0.9691$) fitted to six measured time points of watershed-wide woody cover from 1980-2020. This highly significant model ($R^2 > 0.96$) allowed us to interpolate annual woody cover values for years between measurements. We then calculated ($ET_{total\ proportion\ wood}$) by applying the 2x woody ET multiplier to these annual

woody cover estimates, weighted by the proportional area of woody vegetation versus grassland across the entire watershed for each year.

We tested the model in Figure 2.1 with a structural equation model (SEM) to elucidate the interrelationships among key environmental variables. The SEM focused on two response variables: discharge and the number of flow days. Evapotranspiration was modeled as being influenced by three directly measured variables (precipitation, temperature, and riparian woody encroachment). Discharge and number of flow days were both modeled as being influenced by ET (grass with riparian wood cover) and precipitation, since precipitation can directly run off into the streams or be intercepted and lost through ET. All statistics were done with IBM SPSS Statistics 27, except structural equation modeling, which was done in R Studio version 2023.12.1+402.

Results

Temporal trends of hydrology and climate at USGS Kings Creek gage.

Kings Creek at KPBS is an intermittent stream that can flow or dry any month of the year. We found modest evidence for decreases in mean annual discharge, likely related to the high interannual variance; long-term data indicated a decrease in yearly mean discharge over time ($F = 2.289$, $p = 0.139$). However, using daily data resulted in a clearer decline in discharge; $\log_{10}$ (daily mean discharge + 0.0001 cms) decreased 84% from 1982 to 2020 ($F = 413.761$, $p < 0.001$). The annual number of flow days at the USGS Kings Creek gage showed a downward trend ($F = 8.973$, $p < 0.005$) with a moderate correlation ($R^2 = 0.20$), with predicted values from the regression model roughly halving from 240 to 120 days annually between 1982 and 2020 (Fig. 2.3B). Temperature demonstrates a warming trend, increasing by approximately 1.2°C over the 39-year period ($F = 19.27$, $p < 0.001$) (Fig. 2.3C). Yearly precipitation shows a slight visual increase but with high variability and low confidence ($R^2 = 0.03$, $F = 1.21$, $p = 0.278$) (Fig 2.3D, Tables S2.1-2.3). We found no individually significant contributions of season, year, and season*year interaction with precipitation patterns, but temperature dynamics are notably influenced by time (years) (Table S2.5).

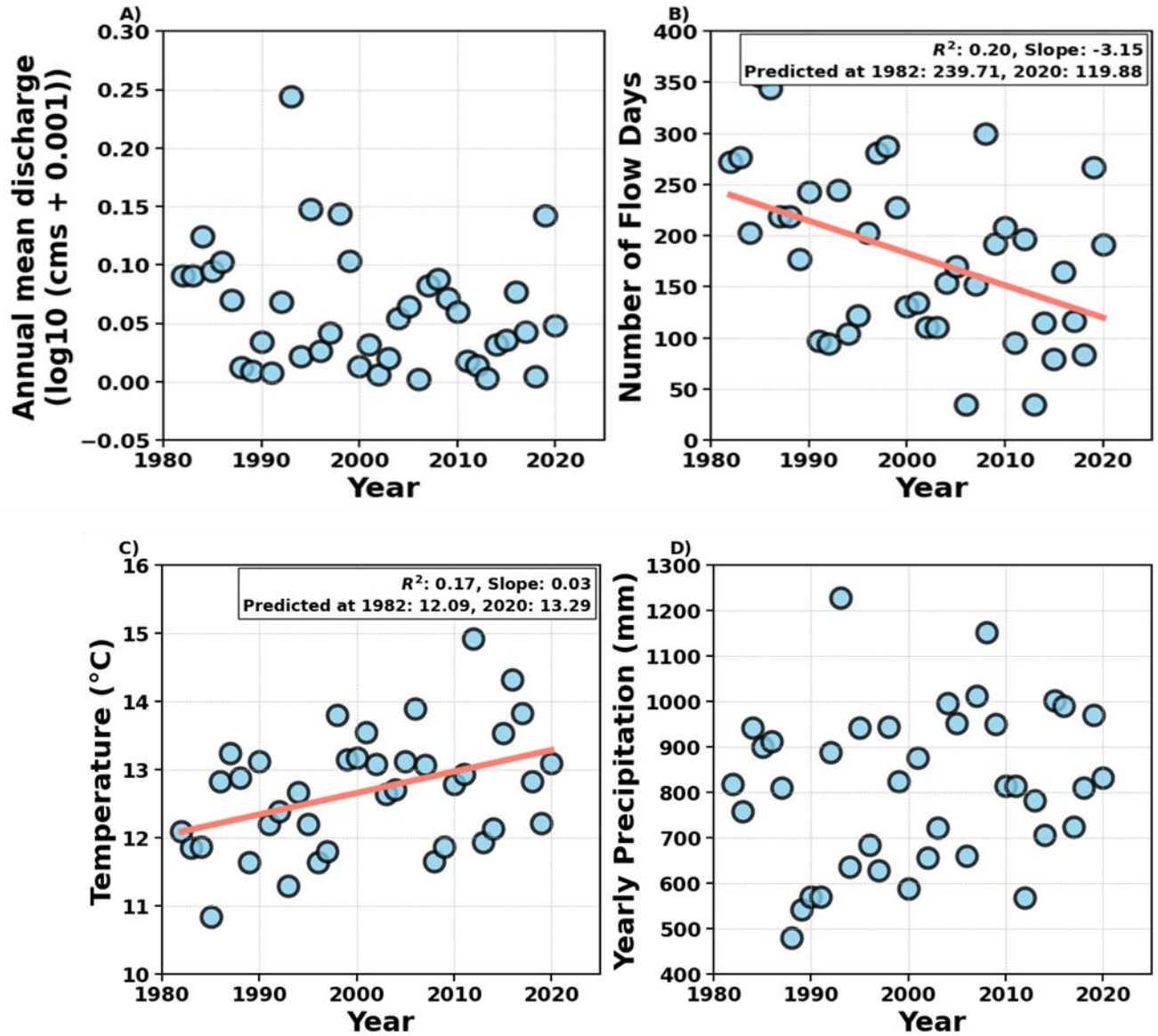


Figure 2.3: Annual mean daily discharge by year (A) and number of annual flow days by year (B), annual mean daily temperature (C), and annual total precipitation (D) at the USGS gage in the Kings Creek watershed. The solid salmon-red line illustrates a regression fit for each graph where $p < 0.05$

Temporal autocorrelation in discharge

We offset the USGS discharge using a sliding window of 10 days up to 90 days (increasing the window size by 10 days for each correlation) to estimate the temporal autocorrelation period and explore the times of the year that were most influential in the

observed decrease in discharge and the number of flow days. By the time we reached the 90-day sliding window, temporal autocorrelation of discharge became non-significant. We then used a 5-day window between 80 and 90 days and the temporal autocorrelation abruptly became non-significant with an 85-day time lag. Based on this we divided the data into four three-month seasonal periods in each year (Fig 2.4; see Table S2.4). We also tested if it mattered if we started the three-month periods in January, February, March, or April using temporal autocorrelation. It made little difference ($p > 0.05$) which of the months we started our three-month periods, so we analyzed seasonal effects with January to March as season one (Julian day 1-90), April to June as season two (Julian days 91- 181), July to September (Julian days 182 - 273) as season three, and October to December (Julian days 274 to 365 or 366 depending upon leap-year) as season four (Fig. 2.4).

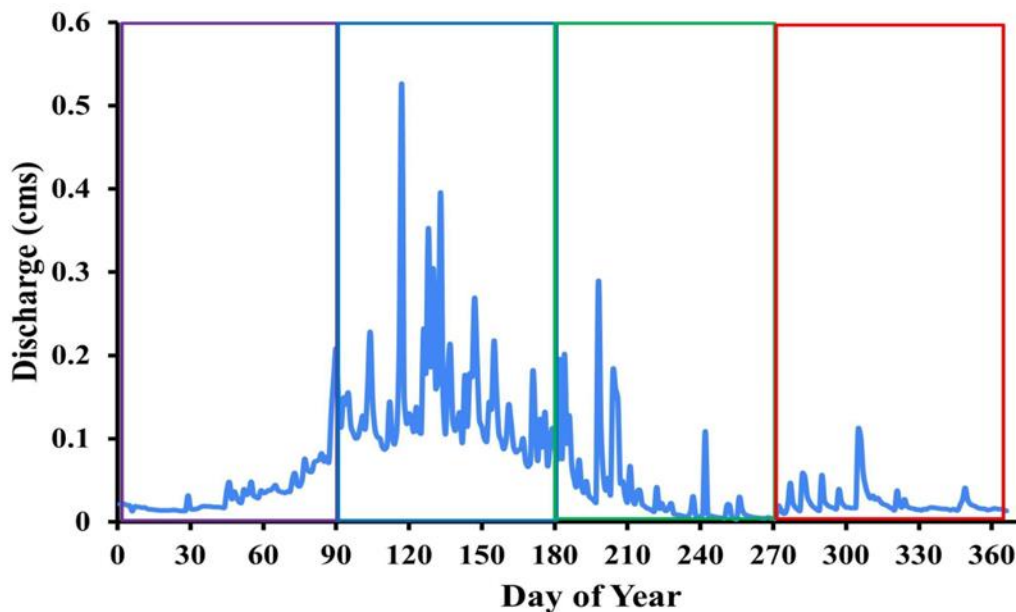


Figure 2.4: Discharge averaged for each day from 1982 - 2020 at the USGS Kings Creek gage. Boxes indicate the “seasons” we used for subsequent analyses, which were defined based on periods where autocorrelation was not significant.

Factors influencing discharge and total flow days per season in the three sub-watersheds.

Watershed N4D (burned every four years) had a ~7-fold increase in the proportion of riparian woody cover between 1980 and 2020 ($F = 8921$, $p < 0.001$, Fig 2.5C). At the N2B watershed which burns every 2 years, and excluding the years of manual removal, total riparian cover increased almost 200% over the same period (Fig 2.5B). The manual vegetation removal carried out in the N2B watershed during 2011-13 had minimal impact on the long-term trend, causing a short-term decrease during the removal experiment but rapid return to the trajectory of exponential increase. Annually burned N1B had a riparian cover increase of about two-fold from 1982 to 2020 (Fig. 2.5A). The riparian zones cover approximately 20% of each watershed.

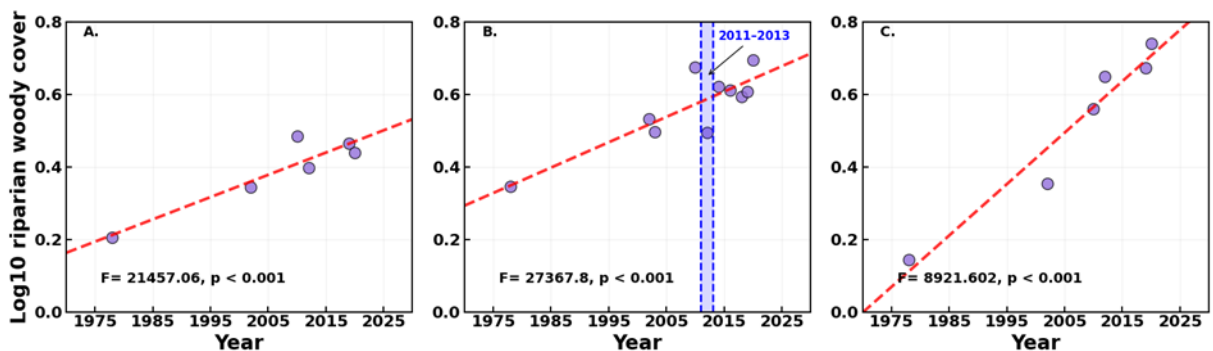


Figure 2.5: Log₁₀ scaled proportion of shrub and tree cover in riparian zones over time at the USGS Kings Creek sub-basin watersheds with different burning frequencies. A) N1B every year, B) N2B burns every 2 years, and C) N4D burns every 4 years. All watersheds showed a significant increase in the total riparian cover. Note the vertical dashed line in panel B indicates to 2011 Riparian removal immediately following the whole watershed riparian removal experiment. Original riparian woody cover proportions ranged from 0.15-0.30 in N1B, 0.15-0.32 in N2B (with temporary reduction to 0.10 following removal), and 0.05-0.35 in N4D during the study period

The ANCOVA for the sub-watersheds revealed that season, sites, precipitation, and riparian cover all influenced discharge ($p < 0.01$). Precipitation emerges as the dominant factor, with the highest F-value (72.410). Temperature, however, doesn't impact discharge ($p = 0.285$).

The interaction between sites and riparian cover ($p < 0.001$) suggests that riparian cover's influence on discharge varies across locations (Table S2.7). In contrast, for flow days, season and precipitation remain highly influential ($p < 0.001$), but watershed less so ($p = 0.309$).

Temperature and riparian cover also show no significant main effects. Notably, interaction terms indicate that temperature, precipitation, and riparian cover affect flow days consistently across sites and seasons (Table S2.8).

Correlations with discharge and the number of flow days and how they are changing over the years for USGS gage and upstream sub-basins.

At the USGS Kings Creek gage, discharge showed strong positive correlations with both annual precipitation and the number of flow days. The number of flow days exhibited negative relationships with both year (Fig. 2.3B) and temperature (not shown). Temperature displayed a positive relationship with year (Fig 2.3C, Table S2.9).

The flow characteristics (mean discharge and number of flow days) were very variable in the sub-watersheds, generally with fewer flow days than at the USGS gage. The LTER N4D watershed displayed strong positive correlations of both discharge (annual) and the number of flow days with precipitation. Unlike the USGS site, no significant correlation was found between discharge and temperature in this watershed. Over the study period, riparian cover steadily increased, yet this increase did not correlate strongly with discharge or the number of flow days (Table S2.10).

The N02B watershed demonstrated similar patterns to those observed in N4D, with discharge and the number of flow days positively correlated with precipitation amounts. Temperature had a weak relationship with discharge and flow days (Table S2.11). In contrast, the N01B watershed exhibited different patterns compared to N4D and N02B. Discharge did not

significantly relate to precipitation, and there was a positive relationship between the number of flow days and precipitation (Table S2.12).

What might influence discharge and the number of flow days at the USGS station over the years by season?

The ANCOVA for discharge at USGS Kings Creek gage reveals precipitation is a key driver affecting discharge ($p = 0.002$, $F = 10.264$). Season had weak relationships with discharge ($p = 0.623$) as did temperature and riparian cover. The absence of strong interaction terms (all $p > 0.05$) suggests consistent effects of temperature, precipitation, and riparian cover across seasons. At this site, precipitation drove discharge variation (see Table S2.13). Analysis of flow days presents a contrasting result to the discharge model. Increases in riparian cover were associated with a decrease in flow days ($F = 4.074$, $p = 0.048$), unlike in the discharge model. Surprisingly, neither season nor precipitation showed significant main effects at the USGS gage, diverging from sub-watershed results. Temperature remained weakly related. Again, interaction terms all had $p > 0.12$, indicating consistent variable effects across factor combinations (Table S2.14). In sum, these analyses suggest riparian cover is a key control in determining the number of flow days at USGS Kings Creek, outweighing precipitation and season.

Assessing evapotranspiration dynamics with wood across seasons

Our ANCOVA analyses revealed that woody vegetation substantially amplifies both seasonal ET patterns and long-term ET increases. For the grassland-only ET scenario, we found modest seasonal variation ($F_{3,72} = 2.562$, $p = 0.061$) and a weak constant yearly increase ($F_{1,72} = 4.245$, $p = 0.043$), with a marginal season-by-year interaction ($F_{3,72} = 2.696$, $p = 0.052$). When woody vegetation was included, these patterns became much more pronounced. For the ET with riparian woody vegetation scenario, both season ($F_{3,72} = 6.616$, $p = 0.001$) and year ($F_{1,72} =$

26.801, $p < 0.001$) had strong effects, with a powerful season-by-year interaction ($F_{3,72} = 7.096$, $p < 0.001$). Similar strong effects were observed for the ET with total watershed woody vegetation scenario, with notable effects of season ($F_{3,72} = 6.440$, $p = 0.001$), year ($F_{1,72} = 25.937$, $p < 0.001$), and their interaction ($F_{3,72} = 6.908$, $p < 0.001$). See Tables S2.15-2.17 for full ANCOVA results.

Our most striking finding is the significant increase in evapotranspiration (ET) with time due to woody encroachment, with this effect most pronounced during late summer months. Specifically, the July-September trendline for $ET_{grass + wood}$ and $ET_{total proportion wood}$ exhibited the steepest positive slopes, approximately 9.94 and 7.39 mm per decade, respectively, while ET_{grass} showed a more moderate slope of 6.36 mm per decade (Fig. 2.6). This indicates that woody vegetation amplified the increasing trend in summer ET over the study period. Over the entire 2001-2020 period, summer (July-September) ET increased dramatically by 56 mm (2.78 mm/year) in areas with riparian woody vegetation compared to just 28 mm (1.42 mm/year) in grass-only areas. Total ET during this critical period reached approximately 400 mm per three-month season in woody-influenced watersheds by 2020, representing a ~15% increase from the 350 mm seasonal totals observed in 2000. The spring (April-June) period also showed notable increases in ET with woody vegetation present, with positive slopes of 1.42 mm/decade for ET_{grass} , 3.44 mm/decade for $ET_{total proportion wood}$, and 1.42 mm/decade for $ET_{grass + wood}$ suggesting an upward trend, although not as steep as the Jul-Sept period (Fig. 2.6). Winter and fall periods showed minimal change over time, regardless of vegetation type. The predominant summer ET increase is particularly important for streamflow as it occurs during the period when water availability is limited and demand is high, creating a seasonal water deficit that contributes directly to fall stream drying.

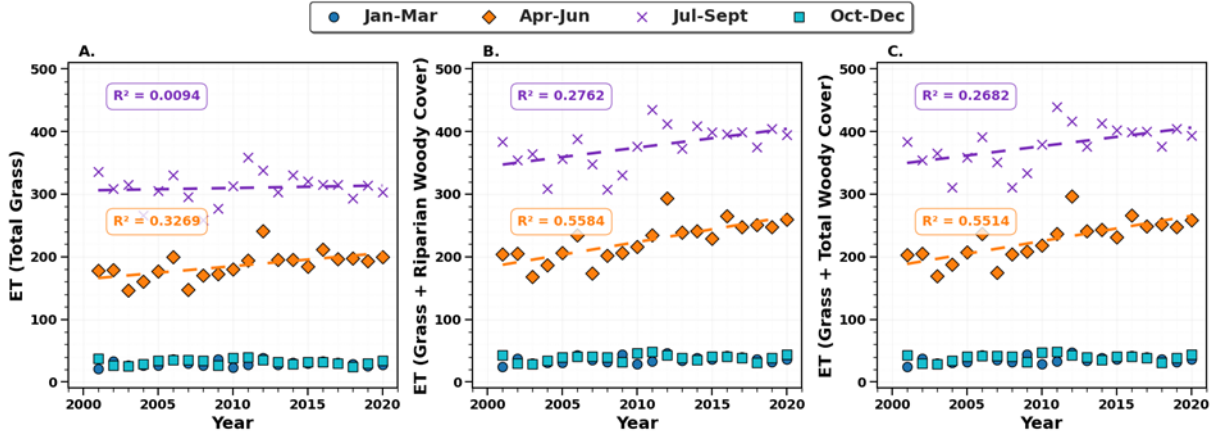


Figure 2.6: Seasonal variation in evapotranspiration (mm/3 months (cumulative seasonal total)) dynamics for Total watershed grass ET (A), Total watershed grass + riparian woody cover ET (B), and Total watershed grass + total woody cover ET (C) in the USGS Kings Creek watershed

We analyzed the relationship between precipitation excess (P_{ex}) and energy excess (E_{ex}) for the USGS Kings Creek watershed (Fig. 2.7) to understand temporal dynamics of these drivers. During these wet years, both E_{ex} and P_{ex} were generally positive (Fig. 2.7). During the spring (Apr-Jun) and fall (Oct-Dec) seasons, P_{ex} was generally high and there was variability primarily along the E_{ex} dimension among years, indicating that during these seasons availability of energy is the primary factor varying among years. In winter (Jan-Mar) and summer (Jul-Sept), the points generally shift downward and to the left through time, indicating both P_{ex} and E_{ex} are decreasing, which is a pattern that would be expected under woody vegetation encroachment (Tomer & Schilling, 2009).

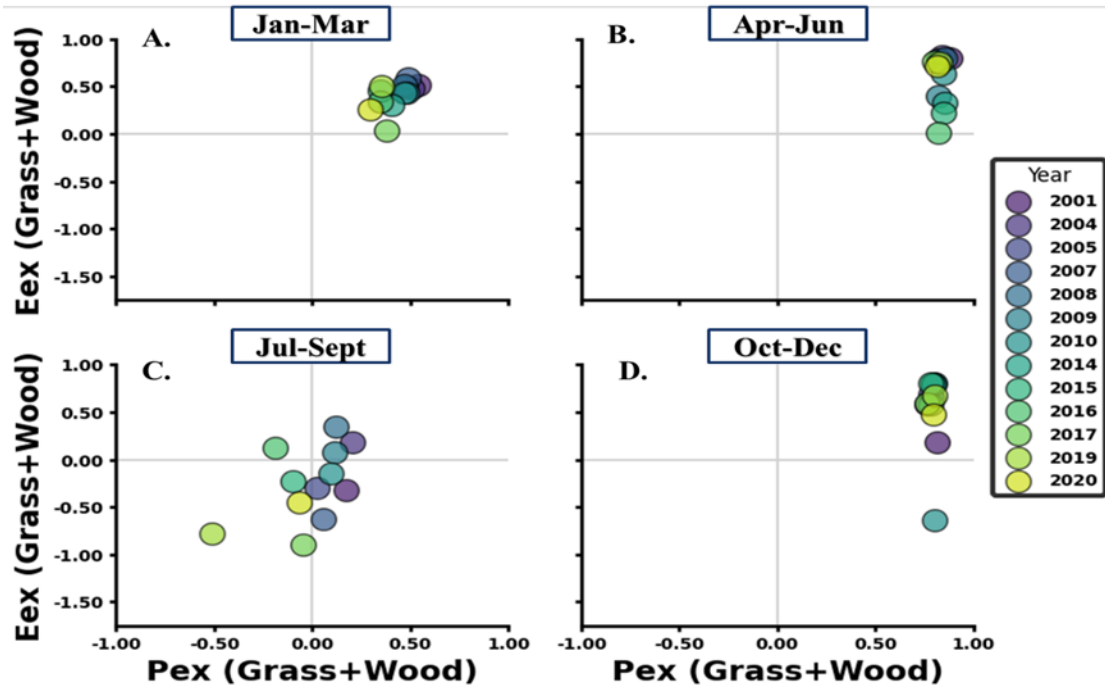


Figure 2.7: Comparative Plots of Excess Precipitation (Pex) vs. Excess Evaporative Demand (Eex) for Grass with Wood in the USGS Kings Creek Watershed from 2001 to 2020.

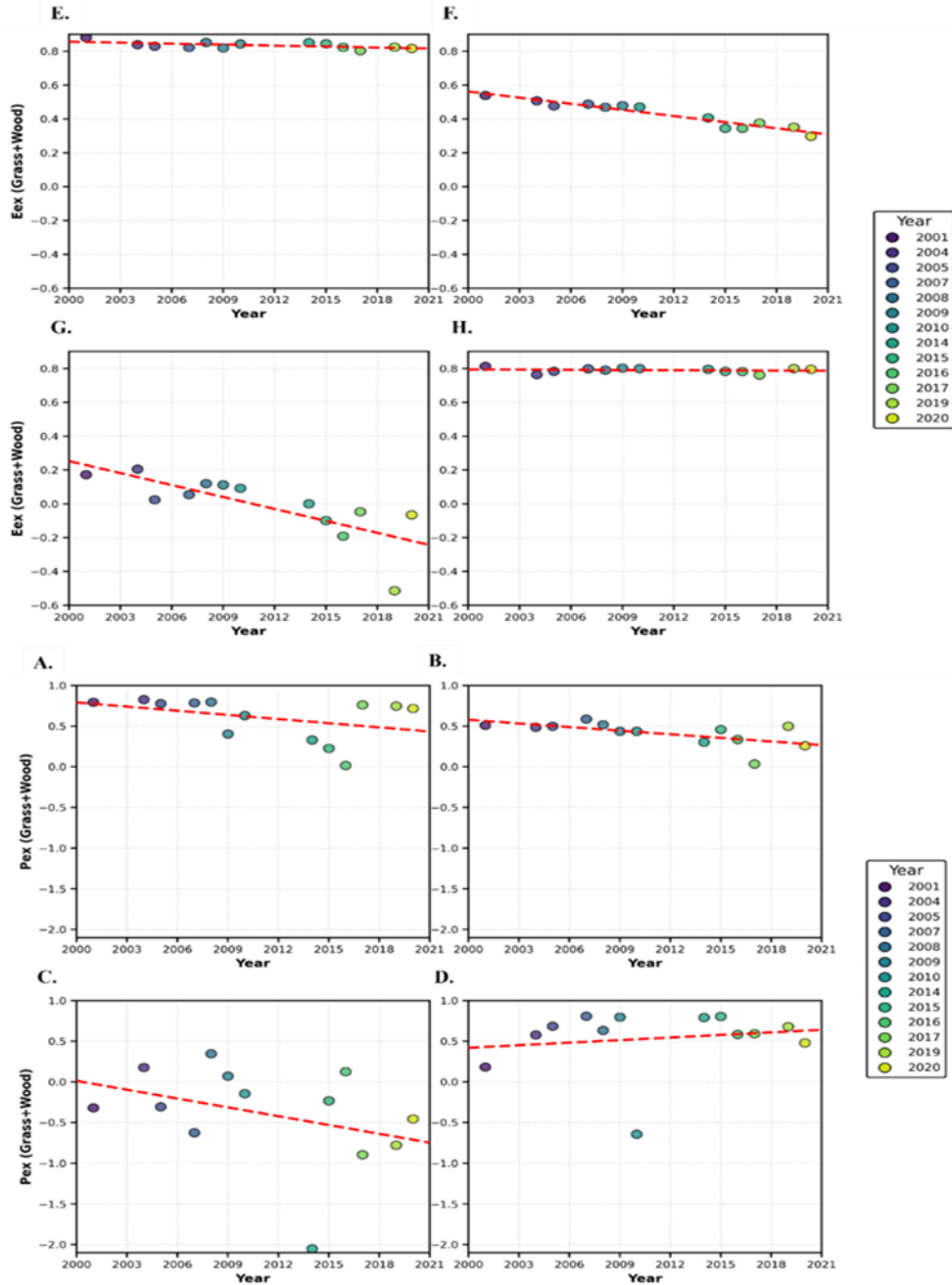


Figure 2.8: Seasonal breakdown of Excess Precipitation (Pex) vs. Excess Evaporative Demand (Eex) for the USGS Kings Creek watershed from 2001-2020 (wet years only). Panels show: (A-D) Pex patterns and (E-H) Eex patterns for Winter (Jan-Mar), Spring (Apr-Apr-Jun), Summer (Jul-Sep), and Fall (Oct-Dec), respectively. Color gradient represents the year progression from 2001 (cool colors) to 2020 (warm colors). Red dashed lines indicate linear trends.

The path coefficient of the structural equation modeling (Fig 2.9) indicates strong effects of precipitation on both discharge and the number of flow days and a very strong influence of ET on discharge, but not flow days. This modeling also shows that ET has a modest negative relationship with streamflow discharge (coefficient = -0.002, $p = 0.002$). While the p value is small, the magnitude is relatively small in standardized terms. Temperature (16.58) and riparian cover (219.29) show much larger direct effects on ET than ET shows on discharge. This explains the apparent contradiction in our earlier analyses: the relationship is statistically strong (highly significant) but modest in effect size. Precipitation maintains the dominant direct effect on discharge (0.006, $p < 0.001$), while ET's influence operates both directly and through complex indirect pathways. The relationship between ET and flow days is even weaker (-0.041, $p = 0.178$), suggesting different mechanisms control flow persistence. The large coefficients highlight the strong impacts of temperature and riparian areas, while the small coefficients suggest weaker effects for the other relationships in this watershed hydrological network, where evapotranspiration serves as a key mediating process.

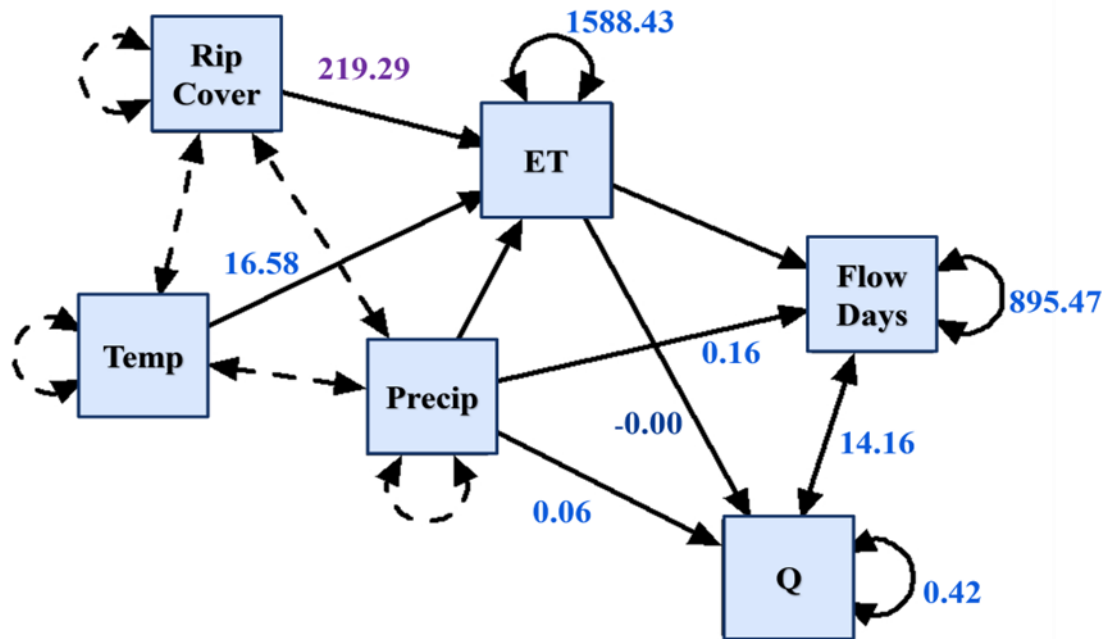


Figure 2.9: Structural Equation Model illustrating significant relationships according to the conceptual model. Blue estimates are significant at $p < 0.002$, while the purple estimate indicates significance at $p = 0.032$. (Here, “Q”= log discharge, “ET”= actual evapotranspiration, “Precip”=precipitation, “Temp”=temperature, “Rip Cover”=log riparian cover). Solid arrows represent direct causal relationships, while dashed arrows represent covariances without assumed direct causality. Straight arrows indicate hypothesized directional effects, and curved arrows represent residual variances (self-loops) or correlations between variables. Numbers on paths are standardized path coefficients, with blue values indicating significance at $p < 0.002$ and purple values at $p < 0.05$.

Discussion

Expanded woody vegetation plays a role in declines in stream discharge and flow days.

Our results highlight a decline in daily discharge by 84% and flow days by 55% at the USGS Kings Creek gage from 1982 to 2020. The USGS Kings Creek watershed basin is on a trajectory of change with riparian woody expansion occurring that could eventually spread to all streams in all watersheds at the site and replace all grass-dominated riparian areas (Dodds et al., 2023). This expansion was dramatic: near doubling of riparian cover in watersheds N1B and N2B, and a seven-fold increase in watershed N4D. This progressive increase in woody vegetation substantially expanded the total leaf area available for transpiration, particularly in watersheds with less frequent burning.

The negative association of riparian cover with discharge, as mediated by ET in our study, indicates that increased woody vegetation is associated with reduced discharge, as has been shown by others in catchment-scale hydrological systems (Bosch & Hewlett, 1982; Zhang et al., 2001). Our data suggest a nonlinear response where impacts intensify once woody cover exceeds critical thresholds (approximately 30% in watershed N4D). This supports recent work suggesting that stream intermittency can be characterized by tipping point behavior between wet and dry states (Zipper et al., 2022).

Encroachment of woody species in mesic grasslands is associated with increased evapotranspiration due to the generally higher transpiration rates exhibited by woody plants compared to grasses (Huxman et al., 2005; Keen et al., 2022; O’Keefe et al., 2020; Wilcox et al., 2022). In the KPBS riparian area, about 75% of the total precipitation is lost to ET (Steward et al., 2011). This proportion is well-supported by our P_{ex} analysis, which shows winter (Jan-Mar)

and fall (Oct-Dec) seasons with positive P_{ex} values around 0.2-0.5, indicating 50-80% of precipitation consumed by ET. Spring (Apr-Jun) shows similar patterns, while summer (Jul-Sept) shows P_{ex} values approaching or below zero, particularly in recent years, indicating nearly 100% of precipitation consumed by ET. The significant increase in ET we documented during July-September, particularly in models incorporating woody vegetation, reflects the capture of subsurface water by woody species. This is further supported by observed correlations between the spatial distribution of stream drying and seasonal to annual evapotranspiration rates in the watershed (Zipper et al., 2025).

The well-distributed and deeper root systems of woody vegetation compared to grasses enable them to access deeper soil water or groundwater, altering infiltration rates and flow paths (Nippert & Knapp, 2007). While grasses are predominantly dependent upon water within the top 30 cm of the soil, with a majority of grass root biomass in the upper soil profile (Dahlman & Kucera, 1968; Hintz, 1999), woody C_3 species access deeper soil water through extensive taproot systems. The use of stream water by shrubs along riparian areas, as indicated by water stable isotopes, shows substantial shrub consumption of stream water during portions of the growing season (Keen et al., 2022). When comparing our ET models, we found that riparian woody vegetation had a stronger per-area impact on streamflow than upland woody vegetation, likely due to riparian trees' direct access to stream water and shallow groundwater. While both riparian-only and total watershed woody vegetation models predicted significant increases in ET, the riparian model explained a slightly higher proportion of discharge variance.

Our findings on decreasing streamflow align with changing vegetation-water dynamics in this watershed. Even during wet periods, over 95% of precipitation travels through the subsurface rather than as direct runoff (Hatley et al., 2023), with increasingly long flow paths as

the stream network dries (Swenson et al., 2024). Our discharge and ET results suggest this subsurface flow path is increasingly exploited by woody vegetation. The differential response we observed in watersheds with varying woody encroachment levels (N1B vs. N4D) further supports this mechanism.

Rising atmospheric CO₂ and temperature will complicate vegetation-water relationships. Current CO₂ levels of 420 ppm are projected to reach 550-900 ppm by 2100 (Change, 2007), which will enhance woody C₃ species' dominance by increasing their water-use efficiency, allowing them to keep stomata closed more often without becoming carbon-limited (Anadón et al., 2014; Volder et al., 2013; Wang et al., 2018). Warming of 2.5-5.5°C by 2100 (RCP4.5-8.5) will increase evapotranspiration by 3-5% per °C (Acharya et al., 2017), potentially raising annual ET demand by 10-25%. Our data show a shift toward more intense, less frequent precipitation events—a trend projected to continue (Davenport & Diffenbaugh, 2021). These larger storms would likely lead to greater proportions of runoff and less infiltration, favoring woody vegetation with deeper roots that can tap into stored subsurface water.

Our results indicate that yearly burning in this grassland ecosystem could somewhat slow the expansion of woody vegetation, as the increase of woody riparian cover was slower in watershed N1B (burned annually) compared to N2B and N4D. However, even the annually burned watershed with slower encroachment still experienced discharge declines once sufficient woody cover was established. This threshold effect highlights important implications for targeted management interventions, suggesting that prevention of initial establishment may be more effective than managing established woody vegetation. Due to the complexity of stream intermittency, there might be many physical factors involved, each of which has a moderate effect (Snelder et al., 2013). However, our results and those of Keen et al., (2023) suggest that

the increase in gross water use associated with woody encroachment both inside and outside the riparian area leads to an increase in overall water use, making it more likely that woody cover decreases the amount of water that reaches streams and groundwater, regardless of the specific water source used by trees and shrubs.

Temporal Dynamics of Evapotranspiration driven by the Interplay of Water and energy

Precipitation and discharge usually display weak correlations due to interception by plants (Costigan et al., 2015; Wine & Zou, 2012). Our system is hyper-variable because average potential evapotranspiration over all of the years is roughly equal to annual precipitation (Acharya et al., 2017; Sadayappan et al., 2023). Thus, slightly wetter seasons or years can lead to streamflow, and modestly drier years or seasons lead to dry streams. The effect of woody ET is most strongly illustrated by our results from late summer when vegetation demand for water is high, the temperature is high, and even with periodic large storms, the streams often stop flowing.

The analysis of the excess precipitation (P_{ex}) vs. excess evaporative demand from 2001 to 2020 reveals additional evidence for vegetation-induced changes in the local hydrology as streamflow requires positive P_{ex} . The July-September period reveals the gradual transition towards increased woody vegetation within the watershed creates more evaporative demand as indicated by a shift downward and to the left on Fig 2.7C. The seasonal breakdown (Fig. 2.8) shows this pattern is most pronounced during summer (July-September) and winter (January-March) months with E_{ex} decreasing most strongly during July-September over time, while spring and fall seasons exhibit more variable patterns with less clear temporal trajectories. This pattern aligns with the watershed afforestation effects on the water and energy balance following woody

vegetation expansion in other watersheds (Bosch & Hewlett, 1982; Zhang et al., 2001). We note this trend was evident even though we restricted the analysis to wetter years, which was necessary to avoid the problem of precipitation renewing groundwater or soil moisture. The trend would be stronger if we included drier years. This trend is particularly significant given that July to September typically represents the warmest and driest part of the year.

The temporal shift of the energy and precipitation balance provides substantial evidence of potential consequences of an ecosystem transition from grassland to a more woody-dominated landscape, a change that can have profound implications for biodiversity, carbon storage, and local climate regulation (Banerjee et al., 2023; Dodds et al., 2023; Ekberzade et al., 2023; Feurdean et al., 2021; Keen et al., 2024; Liu et al., 2023). The trend could be part of a positive feedback loop, where initial increases in woody vegetation alter local surface water availability, further favoring woody encroachment (Wilcox et al., 2006, 2022). This self-reinforcing process aligns with documented threshold dynamics in grassland ecosystems (Ratajczak et al., 2012; Scheffer et al., 2015), where transitions accelerate once woody cover exceeds critical thresholds. D’Odorico et al., (2012) specifically identified hydrological feedbacks as key drivers of irreversible state changes in grasslands, where woody plants modify soil moisture regimes to their advantage, further suppressing grasses. It is important to consider these vegetation-induced changes in the context of broader climate change patterns, as rising temperatures and altered precipitation regimes could amplify the hydrologic effects of vegetation change.

Specific drought years (2002, 2003, 2006, 2011, 2012, 2013, and 2018) revealed a consistent pattern where low precipitation coincided with decreased discharge and reduced flow days. These drought years exhibited discharge and flow day values 35-50% lower than wet years, demonstrating how climate variability amplifies vegetation-driven hydrological changes. With

projected precipitation changes bringing more intense but infrequent storms that favor woody vegetation, our models suggest riparian woody cover may reach 70-90% by 2050, potentially reducing annual discharge by 30-50% and increasing zero-flow days by 20-40 days. Even with intensive management (annual burning, woody removal), discharge may still drop 15-25%. Only aggressive vegetation control combined with climate mitigation could maintain near-current streamflow patterns.

While our analysis focused on flow at watershed outlets, cumulative precipitation and evapotranspiration can also have significant impacts on the spatial distribution of flow within watersheds (Zipper et al., 2025), indicating potential widespread ecological impacts. The compound effects of below-average precipitation and increased ET demands from woody vegetation were particularly evident during these drought periods, resulting in more extensive stream drying than would be expected from precipitation deficits alone in a grass-dominated system.

The structural equation model in our study showed a high degree of unexplained variance in flow days, as evidenced by the substantial flow days-flow days coefficient. This finding underscores the complex nature of flow persistence in the USGS Kings Creek watershed system and suggests that factors beyond precipitation and evapotranspiration play crucial roles in determining the number of flow days. For example, the karstic landscape leads to complex hydrology as various geological layers connected by intricate flow paths respond to a variable climate (Costigan et al., 2015; Gambill et al., 2024; Zipper et al., 2025).

The largely unexplained decreases in flow days highlight a key distinction in hydrological drivers. While both our ANCOVA and structural equation modeling captured precipitation's positive and ET's negative effects on discharge, flow persistence remained poorly

explained. This suggests that unmeasured factors such as groundwater dynamics and subsurface characteristics play crucial roles in determining flow days, representing an important area for future research (Huxman et al., 2005; Wilcox et al., 2022). Additionally, the high unexplained variance may indicate the presence of non-linear relationships or time-lagged effects that our linear model does not account for (although accounting for temporal autocorrelation could obviate this problem).

How might this apply to other grasslands?

While a decrease in streamflow with increased woodiness is common, diverse responses have been observed, indicating context dependence. Alterations in streamflow patterns resulting from either woody encroachment or shifts in climate occur in diverse geographical contexts. For example, declines in streamflow occurred in encroached prairie regions, notably in Canada and the Great Plains of the United States (Starks & Moriasi, 2017; Zou et al., 2018). In the North Concho catchments of Texas, an up to 70% reduction in streamflow coincided with an increase in woody cover as degraded grassland recovered to savannah grassland, and was probably related to decreased stormflows (Wilcox et al., 2008). In the karst regions of the Matese and Picentini massifs in Southern Italy, spring discharge showed a significant declining trend despite a slightly positive (though not statistically significant) trend in annual precipitation, with temperature increases playing an important role in this decline. This was especially evident in the Caposele spring catchment, which is notably undisturbed by groundwater abstraction or land-use changes (Fiorillo et al., 2021).

At KPBS, groundwater storage is one of the most important sources of stream water. In a recent investigation within the N4D watershed, Hatley et al., (2023) elucidated that groundwater discharge played a predominant role, contributing over 95% to streamflow during their sampling

period from April to July 2021. The study found minimal contributions from soil water (below 1%) and direct surface runoff (below 4%) to the overall streamflow dynamics. These findings are consistent with the substantive influence of groundwater in sustaining flow across various aquatic ecosystems, encompassing both small headwater non-perennial streams (Costigan et al., 2014; Hatley et al., 2023; Warix et al., 2021) and larger intermittent rivers (Vu et al., 2018; Zipper et al., 2022). The observed cyclical patterns of evapotranspiration, both seasonally and annually, enhance our understanding of water and energy balance in grassland ecosystems and relate to stream drying at differing timescales (Zipper et al., 2025). This may contribute to the phenomenon observed by Swenson et al., (2024), wherein a transition to slower and deeper flow paths occurred as the USGS Kings Creek stream network dried out over the summer.

Our Kings Creek studies offer a key comparison for global grassland studies. In the Edwards Plateau's karst catchments, streamflow increased from 1925 to 2010 alongside rising woody cover, while an Oklahoma catchment (1938–1992) saw stable streamflow and increased baseflow due to riparian woody expansion and slight precipitation increases (Eldridge et al., 2015; Wine & Zou, 2012) found that in regions with similar precipitation (700–900 mm annually), woody cover generally reduced streamflow. However, Archer et al., (2017) observed increased dry season flow in South American grasslands following woody encroachment, attributed to reduced fire and enhanced hillslope infiltration. These global contrasts highlight how climate, hydrogeology, and management shape hydrological responses, with USGS Kings Creek providing critical insights into temperate grassland stream dynamics. Researchers and land managers in various grassland territories may consider adopting similar methodologies to scrutinize the ecological repercussions of evolving evapotranspiration patterns and integrate

these insights into the broader context of ecosystem dynamics (Cook et al., 2020; Ryu & Hayhoe, 2017).

These substantial increases in woody cover reflect broader regional trends in the Great Plains. Assessments of land cover change across Kansas show dramatic increases of 40-60% in riparian corridors (Briggs et al., 2005a). Similar patterns have been observed throughout the central Great Plains, where woody plant cover has increased by approximately 8-15% per decade in areas without active management (Symstad & Leis, 2017). The rates of woody encroachment in our more studied watersheds exceed these regional averages, particularly in the 4-year burn watershed (N4D), suggesting that local factors such as topography, hydrology, and specific management histories may not have been taken into account previously.

Ecological and water yield implications

Low and zero flow can impact various physical characteristics of stream habitats, including wetted area, maximum depth, and dissolved oxygen concentrations (Leopold & Maddock, 1953). The ecological consequences are multifaceted, affecting the availability and suitability of habitats for organisms reliant on streamflow. Macroinvertebrate density and diversity, critical components of aquatic ecosystems, may experience significant reductions in diverted reaches (McKay & King, 2006), posing challenges for species adapted to historical flow regimes (Maskey et al., 2022). Changes in discharge patterns also influence nutrient transport and cycling, with potential repercussions for both terrestrial and aquatic nutrient dynamics (Dodds et al., 2004; Von Schiller et al., 2011; von Schiller et al., 2017). The role of hydrology in shaping water quality is pivotal (Li, 2019), with studies establishing a correlation between low streamflow and elevated concentrations of solutes in rivers and streams on a continental scale (Li et al., 2022).

While heightened riparian cover corresponds to decreased discharge, ecological processes are also influenced by riparian vegetation, including nutrient retention, sediment filtration, habitat complexity, and autotrophic/ heterotrophic balance in grassland streams (Stagliano & Whiles, 2002), and stream biology (Vandermyde & Whiles, 2015; Veach et al., 2015). While woody vegetation might be more efficient at sediment and nutrient removal, the trade-off between these vegetation benefits and reduced streamflow raises crucial questions about the overall ecological balance and sustainability of the watershed.

Cascading effects of decreased discharge may extend to downstream areas, where vegetation relying on streamflow may undergo water stress, impacting growth, reproduction, and community composition (Stromberg et al., 2010). In agricultural and human contexts, the decrease in mean discharge and flow days holds significant water yield implications, potentially affecting downstream users such as agricultural irrigation and municipal water supply. Long-term hydrological analyses of the Kansas River (which USGS Kings Creek drains to), indicate that drought-like flows are becoming more common even with relatively constant annual precipitation (Putnam et al., 2008). Woody expansion in the region, combined with widespread agricultural water use, could partially explain this phenomenon. The characterization of seasonal influences on evapotranspiration dynamics provides insights into the water use patterns of different vegetation types. Assessment of seasonal variations in grass and grass with wood driven by factors like burning frequency and woody encroachment may continue shaping water availability and demand in the watershed. Although our analysis focused on surface hydrology, groundwater dynamics play a crucial role in stream intermittency within the USGS Kings Creek watershed's merokarst geology (Sullivan et al., 2020), as discussed in Section 4.1.

Woody encroachment likely affects groundwater-surface water dynamics in Kings Creek in three key ways: (1) Deep-rooted woody species, as opposed to grasses, tap groundwater, lowering water tables and reducing streamflow (Keen et al., 2022); (2) encroachment alters infiltration and subsurface flow, redistributing recharge patterns (Gambill et al., 2024.); and (3) as the stream network dries, baseflow becomes critical, but increased ET from woody plants accelerates depletion, extending zero-flow periods (Swenson et al., 2024). The uncorrelated discharge-precipitation pattern in the annually burned N1B watershed suggests that hydrogeological buffering, rather than immediate precipitation, governs streamflow. Future studies should integrate groundwater monitoring, seasonal water table dynamics, and isotopic analyses to refine our understanding of woody encroachment's impact on intermittent streams.

Conclusions

Our study demonstrates that the USGS Kings Creek watershed has experienced significant hydrological changes from 1982 to 2020, with an 84% decrease in daily discharge and a 55% reduction in annual flow days. These changes coincided with substantial increases in woody vegetation cover across all watersheds, ranging from a two-fold increase in annually burned areas to a seven-fold increase in areas burned every four years. Through structural equation modeling and seasonal analysis, we found that expanded riparian woody vegetation significantly influences stream discharge through increased evapotranspiration, particularly during summer months. While our SEM showed a weaker direct relationship between ET and flow days than between ET and discharge, our results indicated that riparian woody cover was significantly associated with decreased flow days. This suggests that woody vegetation affects flow persistence through complex pathways that may not be fully captured in our structural models. Notably, this impact showed pronounced seasonal variation, with the strongest effects during summer months (July-September) when increased evapotranspiration from woody vegetation combined with higher temperatures created conditions where water demand frequently exceeded availability, leading to more extensive stream drying.

This effect is amplified by rising temperatures, which showed a 1.2°C increase over the study period, and increased rates of ET. This warming rate of approximately 0.31°C per decade is consistent with broader regional warming trends across the Great Plains, where average temperatures have increased by 0.25-0.35°C per decade since the 1970s (Shafer et al., 2014; Tebaldi et al., 2021). However, the warming at Kings Creek is almost twice as high as the global average of approximately 0.18°C per decade over the same period (Change, 2007), indicating

that this grassland ecosystem may be experiencing accelerated warming compared to global means. Yearly precipitation shows a slight increase but with high variability and low confidence, which also aligns with regional climate assessments showing increased precipitation variability rather than consistent directional changes for the central Great Plains.

In the USGS Kings Creek watershed and similar regions, our findings indicate an increased need for adaptive management strategies to counteract ongoing vegetation changes and their hydrological impacts. These efforts may include intensified actions to maintain the historical grassland ecosystem. The observed trends provide insight into the current state of the watershed and raise critical questions about the ecosystem's future trajectory under continued climate change and vegetation shifts.

Our results emphasize the need for a more comprehensive approach to modeling flow persistence in ecohydrological systems. Future research should focus on incorporating additional variables such as soil moisture dynamics, groundwater table fluctuations, root depth distributions, karst connectivity patterns, and antecedent watershed conditions. Additionally, exploring non-linear modeling techniques and considering various spatial and temporal scales to better capture the intricate dynamics governing flow days. This finding not only highlights the complexity of eco-hydrological systems but also opens new avenues for investigating the factors that control flow persistence in intermittent streams.

While the number of studies of intermittent streams has greatly increased in the last decade, especially focused on its ecology (Allen et al., 2020; DelVecchia et al., 2022), much work is needed to understand the hydrology of intermittent streams. Our findings could extend beyond just the USGS Kings Creek watershed, forming a foundational basis for identifying potential strategies to enhance resilience in grassland regions regionally, and potentially globally.

Policymakers and managers, guided in part by the techniques used in this study, could potentially evaluate and predict how streams may be vulnerable to climate-induced intermittency, enabling more effective decision-making.

Acknowledgements

We acknowledge that research on KPBS uses land taken from those who lived here before us, including the Kaáⁿze (Kaw / Kansa, from who we have the site name), Wah.zha.zhe ᖃᕐᕈᓴᔨ (Osage), Očhéθi Šakówiŋ (Sioux), and Kiikaapoi (Kickapoo), and hope that our work honors their existence and those of their ancestors who remain. We thank the many workers for the upkeep and data curation for the monitoring equipment used in this research. The research was funded by the US National Science Foundation LTER program (DEB: 0823341, 1440484, and 2025849).

Chapter 3 - Effects Of Ungulate Grazing, Burning, And Cropland on Stream Water Chemistry in Tallgrass Prairie Watersheds

Abstract

Fire and ungulate grazing historically shaped tallgrass prairie, but their long-term effects on stream water quality and modern anthropogenic impacts remain poorly understood. We analyzed 37 years (1985-2022) of nitrate (NO_3^-), soluble reactive phosphorus (SRP), and total phosphorus (TP) concentrations from downstream cropland and upstream bison-grazed and ungrazed watersheds in Kings Creek and from 2009-2022 in Shane Creek with and without cattle at Konza Prairie Biological Station (KPBS), Kansas. We compared bison-grazed sites with varied fire frequencies, cattle-grazed areas, ungrazed sites, and cropland restoration areas. Grazing increased stream nutrients, with bison raising NO_3^- by 2.9% annually compared to ungrazed watersheds, while cattle caused larger increases (13.5% annually for NO_3^- , 3.7-4.9% for phosphorus). More frequent fires modestly increased nutrients. Stream NO_3^- downstream of restored croplands decreased from 450 to 110 $\mu\text{g N L}^{-1}$ over 25 years. Groundwater NO_3^- in former agricultural areas declined 70% but remained elevated at 950 $\mu\text{g N L}^{-1}$ after two decades, compared to 200 $\mu\text{g N L}^{-1}$ in comparable grassland wells. Water quality improvements followed fertilizer reduction with 4–5-year lag periods. We found that agricultural fertilization most strongly influenced water quality, with grazing having moderate effects, and fire having minimal effects. Agricultural nitrogen legacies persisted for decades in groundwater, demonstrating extended recovery time lags. Effective watershed management requires understanding cumulative effects and baseline conditions across multiple scales, with fertilizer reduction representing the most effective intervention for sustainable water quality improvement in this area.

Introduction

Grassland streams contribute approximately 25% of global runoff (Dodds, 1997b; Dodds et al., 2015), which regulate downstream water quality through nutrient processing and retention (Alexander et al., 2007; Wellen et al., 2015). Despite their global importance, factors influencing grassland stream water chemistry, baseline conditions, and responses to environmental change remain poorly understood because of limited long-term, watershed-scale data. Understanding reference conditions in grassland streams (best attainable conditions) is essential for water quality management. Strong empirical data will help establish baseline conditions in historically widespread tallgrass prairie systems.

Tallgrass prairie once stretched through central temperate North America, but the remaining ~5% is primarily found in the Flint Hills region, which is used for cattle production, with prescribed burning (Towne et al., 2005). Simultaneously, climate change and woody encroachment are shifting the ecosystem properties of grasslands (Dodds et al., 2023; Raihan et al., 2025). Understanding management effects on water chemistry requires long-term whole-watershed experiments that capture cumulative effects, legacy responses, and ecological thresholds not visible in short-term studies (Callaghan et al., 2013; Lindenmayer et al., 2010), and in the case of tallgrass prairie, requires an understanding of fire, grazing, and cropland conversion.

Tallgrass prairie is fire-adapted, and fires were common and played an important role in maintaining mesic grasslands. Repeated frequent fire in tallgrass prairie reduces net soil nitrogen mineralization and tightens plant–microbial N retention, although individual burns can transiently mobilize surface nitrogen through short-term nitrification pulses (Blair 1997, Turner et al. 1997, Wan et al. 2001). However, how this translates to water chemistry remains unclear.

Cattle have replaced the grazing regime in most remaining tallgrass prairie, potentially mimicking the closely related native bison. However, we know of only two studies that have compared the influence of cattle versus bison on water quality, particularly at the watershed level (Grudzinski & Daniels, 2018; Larson et al., 2016). Large ungulates (native bison or cattle) influence water quality through direct nutrient inputs, physical disturbance, and vegetation modification. Urine and feces deposited in or near streams directly add nitrogen and phosphorus, with cattle-grazed watersheds yielding 5 to 10 times more nutrients than ungrazed areas (Hubbard et al., 2004). Trampling and wallowing increase erosion, mobilizing particulate-bound nutrients (Trimble & Mendel, 1995) while selective grazing and physical damage can reduce riparian vegetation, decrease nutrient uptake, and increase runoff velocity (Belsky et al., 1999). Whether bison impact water quality differently than cattle remains debated, particularly under rotational fires and grazing (Allred et al., 2011; Kohl et al., 2013; Larson et al., 2016a).

Fertilized cropland causes high nutrient loads in low-order streams (Banner et al., 2009; Dodds & Oakes, 2006, 2008). Complete recovery from agricultural legacies can require 10 to 20 years for surface water and longer for groundwater (Meals et al., 2010; Van Meter et al., 2016, 2017). Legacy nutrients move through surface runoff, subsurface flow, and groundwater, each with different lag times (Hamilton, 2012). However, fertilization legacy effects in restored tallgrass prairie remain poorly understood.

Here, we use up to 37 years of water chemistry data from watershed and sub-watershed scales, sampled at sites nested within the Kings Creek and Shane Creek watersheds on the Konza Prairie Biological Station (KPBS), to investigate potential historical pre-European conditions and the influence of contemporary anthropogenic pressures on stream nutrients. We hypothesized (Fig S3.1) that: (a) frequent fire would reduce long-term stream NO_3^- concentrations through

repeated volatilization combined with reduced net mineralization and enhanced N retention, while increasing phosphorus transport through enhanced post-burn surface erosion (Blair 1997, Turner et al. 1997); (b) livestock grazing would increase all nutrients through direct inputs and physical disturbance, with cattle having greater impacts than bison due to concentrated riparian use (Larson et al., 2016b); and (c) agricultural legacy effects would persist for decades in both surface and groundwater despite restoration efforts, following documented patterns of nutrient accumulation and slow release (Van Meter et al., 2017). Few sites globally provide such extensive water quality measurements (1-3 times per week for multiple decades), particularly for whole watersheds in native grassland, and for intermittent streams where most nutrient transport occurs during limited flow periods. Our objectives were: 1) Analyze trends in water quality data (1983 - 2020) taken weekly or more frequently to examine how concentrations of stream NO_3^- , soluble reactive phosphorus (SRP), and total phosphorus (TP) have responded over time. 2) Understand the drivers of long-term water quality dynamics in the Kings Creek and Shane Creek drainage basins by examining the effects of grazing regimes (bison, cattle, or ungrazed), vegetation restoration from cropland to prairie, and prescribed fire frequencies on stream nutrient concentrations.

Methods

Study Site locations and land use history

Our study was conducted on two headwater intermittent drainage basins (Fig. 3.1), Kings Creek and Shane Creek, located at KPBS in northeastern Kansas, USA. KPBS is a 3,497-hectare native tallgrass prairie owned by the Nature Conservancy and managed by Kansas State University's Division of Biology. Watersheds are divided into experimental sub-watersheds with long-term fire and grazing treatments (Table 3.1). Vegetation includes unplowed native tallgrass prairie with riparian oak gallery forests (Veach et al., 2014). Geology is alternating limestone and mudstone with shallow upland soils and deeper alluvium. Both streams have non-perennial flow, with perennial flow observed only near the outlet of the Kings Creek drainage basin, and short spring-fed reaches higher up in the watershed. The non-perennial reaches typically flow from early spring until late summer (Raihan et al., 2025)

Table 3.1: Summary of site characteristics. The data range represents the dates of each site's first and last analyzed samples. The denotation “variable” is used to describe sites that have irregular burn frequencies or multiple watersheds upstream with different burn frequencies. NFKC and SFKC are the two major tributaries of Kings Creek, and SFKC is bison-grazed and NFKC is ungrazed. The number of nitrate samples (n) at each site is indicated.

Site	Date Range	n	Type	Watershed area ha	Burn frequency	Grazers	Latitude and Longitude
N1B	1985-2022	1859	Surface water	121	Annual	Bison	39° 5'11.87"N 96°34'38.70"W
N2B	1985-2022	1416	Surface water	119	Two Year	Bison	39° 5'23.10"N 96°35'19.32"W
N4D	1985-2022	2310	Surface water	135	Four Year	Bison	39° 5'15.25"N 96°35'5.23"W
Shane Creek	2009-2022	696	Surface water	435	Three Year	Cattle	39° 6'50.82"N 96°33'27.10"W
HIKX	1994-2022	3701	Surface water	1524	Variable	None	39° 6'24.63"N 96°36'24.61"W
EDLR	1994-2022	3653	Upland spring	karst Unknown	Mixed effect	None	39° 6'5.86"N

								96°36'21.96"W
HOKN		1994-2022	2107	Surface water	1139	Variable	None	39° 6'13.42"N
								96°35'42.94"W
NFKC		1999-2022	1044	Surface water	497	Variable	None	39° 6'5.47"N
								96°35'31.93"W
SFKC		1999-2022	1106	Surface water	627	Variable	Bison	39° 6'4.40"N
								96°35'32.41"W
Agricultural wells (ALC-A5)		1996-2022	297	Lowland groundwater	Unknown	Mixed effect	None	39° 6'13.44"N
								96°36'12.38"W
Grassland (K01A-P5)	Wells	1996-2022	214	Lowland groundwater	Unknown	Mixed effect	None	39° 6'8.20"N
								96°35'35.79"W

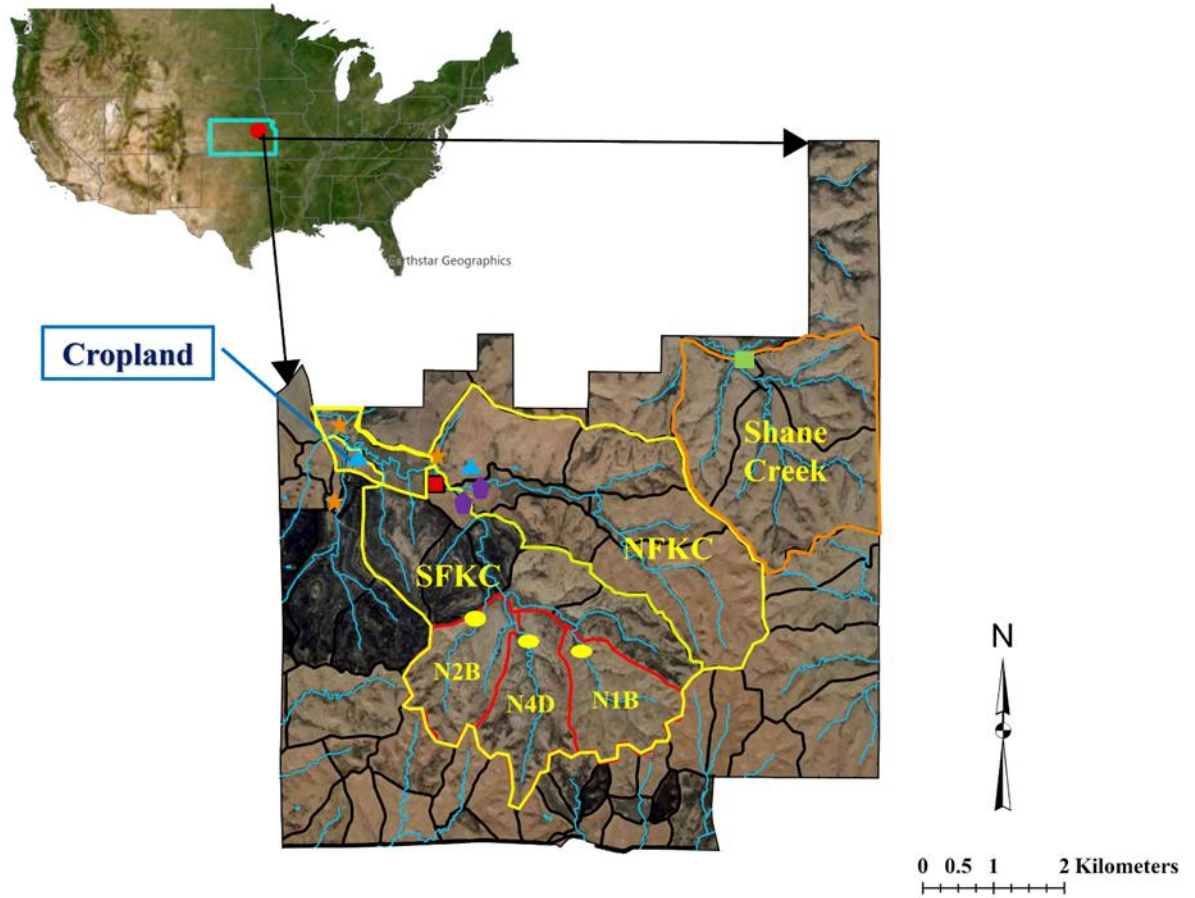


Figure 3.1: Map of Konza Prairie Biological Station (KPBS), Kansas, USA, showing watershed boundaries and sampling locations. Yellow outlined watersheds (SFKC and NFKC) show South Fork Kings Creek (bison-grazed) and North Fork Kings Creek (ungrazed), respectively. Yellow circles mark sampling sites N1B, N2B, and N4D in bison-grazed LTER watersheds. The orange outlined area shows the Shane Creek cattle-grazed watershed with the sampling location marked. Orange stars indicate key monitoring sites: EDLR (pristine spring), HIKX (downstream site below cropland), and HOKN (upstream of cropland). Blue triangles show groundwater well locations (ALC-A5 in former cropland, K01A-P5 in grassland). The red square marks the USGS gauging station. Cropland areas are visible in the lower left portion of the map. The inset shows the location within the United States.

The lower portion of Kings Creek flows through a flat area of deep alluvial soil with a history of cultivation for row-crop agriculture. Specific lowland areas near sites (HIKX and

ALC-A5) were converted to row-crop agriculture between 1939 and 1950. Sequential restoration began in 1995, with complete conversion to unfertilized prairie by 2023. The agricultural legacy represents 50 to 80 years of cultivation, rather than dating back to the 1870s, although we are uncertain of specific fertilization amounts and sources before the 1970s. Since becoming a biological research station, several new buildings with septic systems have been built in the lower Kings Creek watershed. These buildings include laboratory spaces, temporary housing for researchers, facilities for education, and workspaces for prescribed fire and maintenance activities. Buildings in the headquarters area with septic systems were improved in 2008 (Table S3.1). Sub-watersheds have variable target burning frequencies: every two, four, ten, or twenty years. Native bison graze South Fork Kings Creek (SFKC), and North Fork Kings Creek (NFKC) is ungrazed. In 1987, 30 bison were reintroduced to KPBS and allowed to graze year-round on a portion of the south branch, gradually increasing to approximately 300 bison grazing on 984 hectares in the mid-1990s across the entire watershed area encompassed by SFKC. The current herd is stocked at approximately 0.21 animal units (AU) per hectare (0.08 AU/acre) and remains on the prairie year-round. The Department of Agronomy at Kansas State University manages the remaining agricultural fields, following typical crop rotations of the region (corn, wheat, and soybeans). Fertilizer application varies yearly based on soil chemistry and crop nutrient requirements, typically ranging from 100-150 kg N/ha for corn, 50-80 kg N/ha for wheat, and 0-30 kg N/ha for soybeans (D. Ridder, personal communication).

The Shane Creek watershed was ungrazed from 1977 until 2011, when cattle were reintroduced. A third of the pasture was burned annually during our study. Cattle graze 150 days annually (1 May - 30 September). The Shane Creek watershed (435 ha) supports 203 cow-calf pairs at a stocking density of 0.47 animal units per hectare (0.19 AU/acre), 124% higher than the

bison stocking density (0.21 AU/ha year-round), yet both represent light to moderate grazing pressure typical of conservation management rather than intensive commercial production in the Flint Hills (0.5-0.6 AU/ha; Towne & Owensby, 1984). Cattle average 460 kg per adult (approximately 160 kg less than the regional average), with calves born on site reaching around 350 kg by season's end, resulting in approximately 215 kg biomass per hectare during the grazing period.

Water Quality Monitoring and Analytical Methods

Water quality data were collected from 1983 to 2022, with increased spatial scale and temporal frequency starting in 1994. (Table 3.1). NO_3^- , SRP, and TP were measured in streams draining watersheds with 1 (N01B), 2 (N02B), and 4 (N04D) year burn frequencies. Bison grazed these sites from May 1992, and they had no cattle from 1977 to 1992. Additional sites were added in 1992, including the NFKC, which is roughly the same size as the SFKC, Kings Creek below the United States Geological Survey (USGS) gauging station above the first agricultural fields (HOKN), a pristine prairie groundwater site (EDLR), and Kings Creek at the bottom of Konza as it leaves the agricultural land (HIKX). Shane Creek has been sampled since 2009, and cattle were introduced in 2011. Stream samples were collected weekly to three times weekly when flowing. All samples were collected in the morning, kept cool, divided into filtered and unfiltered samples, and frozen for storage

Groundwater wells were sampled monthly from 1996-2022 when groundwater was present (Dodds et al., 1996b). The native grassland well site (K01A-P5) has not been grazed for 25 years, has no cultivation history, and is burned in spring annually. The agricultural well site (ALC-A5) was cultivated from approximately 1940-1998 (~58 years), then sequentially restored to native prairie through 2023. Wells were drilled 13 to 15 m deep to bedrock. Samples were

collected monthly using a PVC bailer after removing twice the well volume, and then the samples were filtered and frozen.

Nitrogen and phosphorus were measured by colorimetry with a flow analyzer. Total phosphorus was determined by persulfate digestion (Ameel et al., 1993). Biannual cross-calibration was conducted using USGS Standard Reference Samples. <https://qsrb.usgs.gov/srs/>.

Statistical Analyses

Nutrient data were log-transformed before calculating annual means to stabilize variance and account for multiplicative effects across orders of magnitude in nutrient concentrations. First, we employed a time-lagging sliding window approach to test for temporal autocorrelation. Autocorrelation was weak at lags >60 days ($\rho < 0.3$), and we used annual averages for analyses to minimize autocorrelation except for grazing comparisons. We used Multivariate Analysis of Covariance (MANCOVA) to examine stream NO_3^- , SRP, and TP across sites with varied burn and grazing regimes. MANCOVA accounts for relationships between these three nutrients while controlling for Type I error. The MANCOVA model included log-transformed NO_3^- , SRP, and TP. Fixed effects included sites (N01B [annually burned, bison-grazed], N02B [burned every 2 years, bison-grazed], N04D [burned every 4 years, bison-grazed]). Our non-factorial experimental design prevented the isolation of individual treatment effects. We analyzed three distinct site groupings: burn frequency effects in bison-grazed watersheds (N01B, N02B, N04D), agricultural groundwater influence (ALC-A5, K01A-P5), and downstream effects (EDLR, HIKX, HOKN). Within each grouping, the 'site' term captures combined effects that cannot be statistically separated. We acknowledge that this limits causal inference, and results could represent associations with combined management practices rather than the mechanistic effects of isolated treatments.

We tested MANCOVA assumptions using Box's M test ($p = 0.089$), Levene's test ($p > 0.05$ for all variables), and Variance Inflation Factors ($VIF < 3$). For bison effects, we used t-tests with 1044 paired samples from 1999-2022 from SFKC (bison-grazed) and NFKC (ungrazed) watersheds. We used independent t-tests comparing times with cattle present versus absent. All statistical analyses were performed using the software packages IBM SPSS version 27.0 or Google Colab.

Results

Nutrient Trends in Bison-Grazed Sites Under Differing Burn Frequencies

Nitrate concentrations increased over time at bison-grazed watersheds with varying burn frequencies. The N1B site increased 47% from 30 to 44 $\mu\text{g NO}_3^- \text{-N L}^{-1}$ over 25 years. The N2B site showed a smaller increase of 25% (from 8 to 10 $\mu\text{g NO}_3^- \text{-N L}^{-1}$), while N4D maintained stable concentrations of 8 to 12 $\mu\text{g NO}_3^- \text{-N L}^{-1}$ (Fig. 3.2). Soluble reactive phosphorus concentrations remained constant across all sites, averaging 5 $\mu\text{g P L}^{-1}$ throughout the study period. Despite occasional outliers, particularly peaks in 2005 (25 to 27 $\mu\text{g P L}^{-1}$), no long-term trends were detected in SRP. Total phosphorus showed varying trends across sites. The N1B and N4D sites had increasing trends (15% and 12%), while N2B had a decreasing trend of 20% (from 15 to 12 $\mu\text{g P L}^{-1}$).

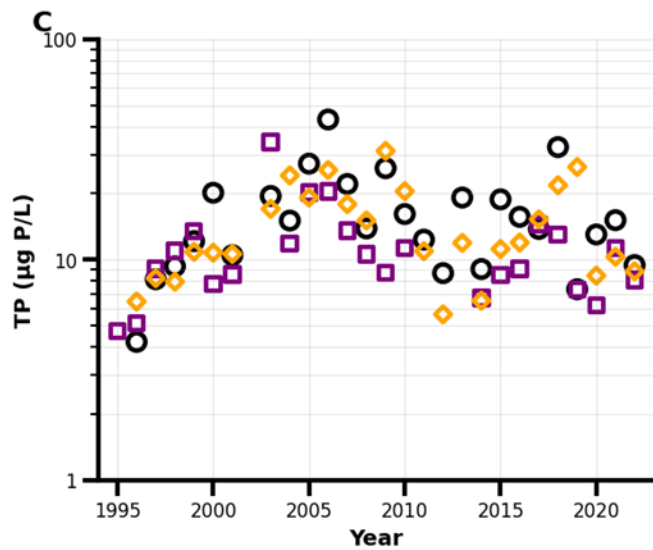
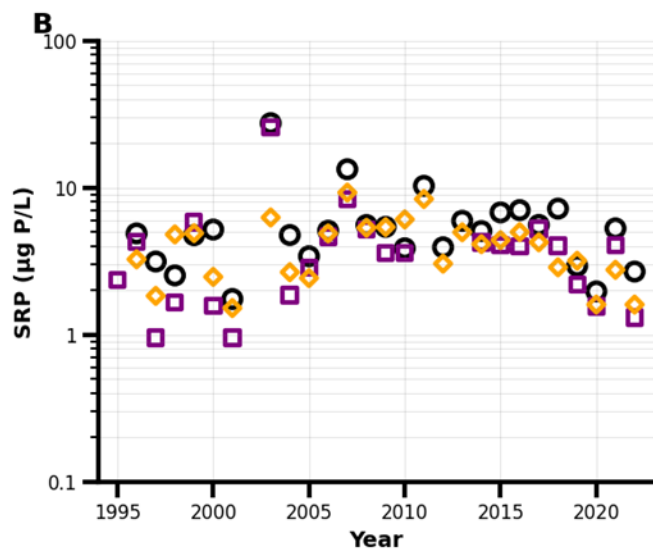
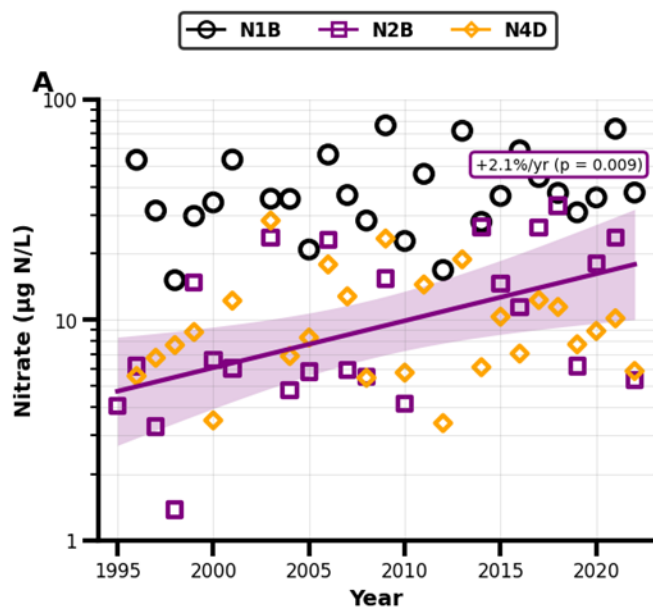


Figure 3.2: Stream nitrate (A), soluble reactive phosphorus (B), and total phosphorus (C) concentrations at three bison-grazed prairie watersheds with different burn frequencies (1995-2022). Black circles show N1B (annual burn); purple squares show N2B (2-year burn frequency); orange diamonds show N4D (4-year burn frequency). Panel A: Linear regression for N2B: $\ln(\text{NO}_3^- \text{-N}, \mu\text{g N L}^{-1}) = \beta_0 + 0.021(\text{year} - 1995)$, $R^2 = 0.26$, $p = 0.009$ (equivalent to 2.1% increase per year). Panel C: TP showed no consistent long-term trend but exhibited elevated concentrations from 1995-2005 across all sites before returning to baseline levels. Lines show linear trends with 95% confidence intervals.

Both year ($F(1,70) = 32.6$, $p < 0.001$) and site ($F(2,70) = 37.1$, $p < 0.001$) affected NO_3^- concentrations. Years since burn showed no clear effect on NO_3^- ($F(1,70) = 3.1$, $p = 0.08$) or SRP ($F(1,70) = 3.2$, $p = 0.08$). Neither phosphorus form (SRP or TP) showed strong effects from any factors or their interactions, indicating that spatial and temporal variations in phosphorus forms were less pronounced than those observed for NO_3^- (Table S3.2).

Trends in Nutrients in Restored Cropland, Pristine Spring, and Downstream Sites

Stream nutrients declined substantially below restored croplands. The HIKX site, below the sequentially restored croplands, exhibited the highest initial NO_3^- concentrations, with a 75% decrease from 450 to 110 $\mu\text{g NO}_3^- \text{-N L}^{-1}$ from 1995 to 2020. In contrast, groundwater from EDLR and upstream above cropland HOKN sites maintained relatively stable and lower NO_3^- concentrations, averaging 50 to 80 $\mu\text{g NO}_3^- \text{-N L}^{-1}$ throughout the study period (Fig. 3.3).

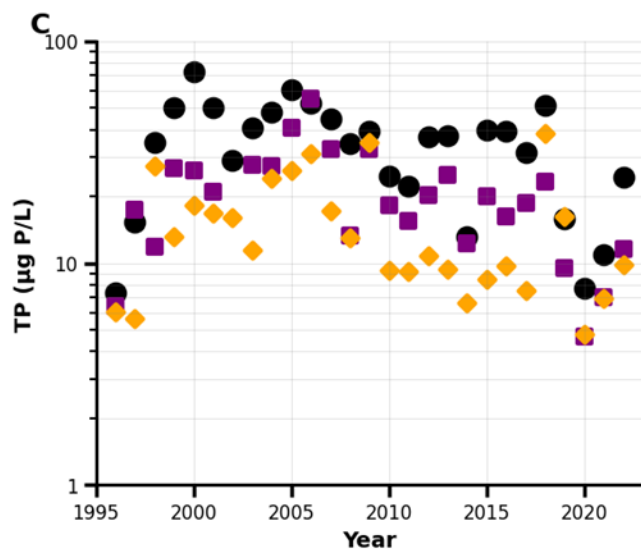
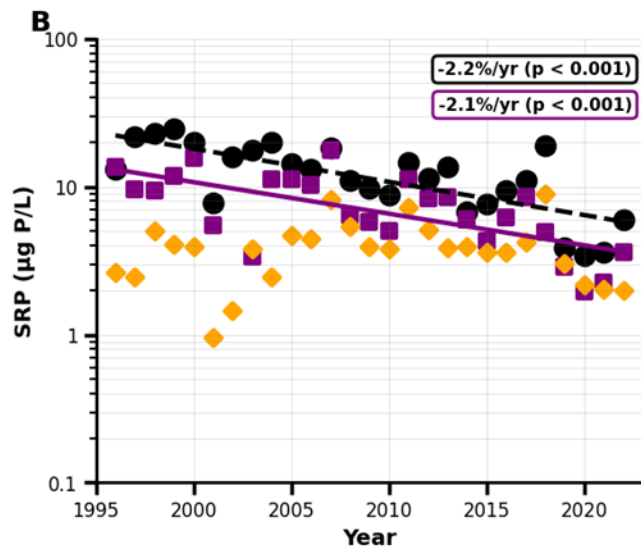
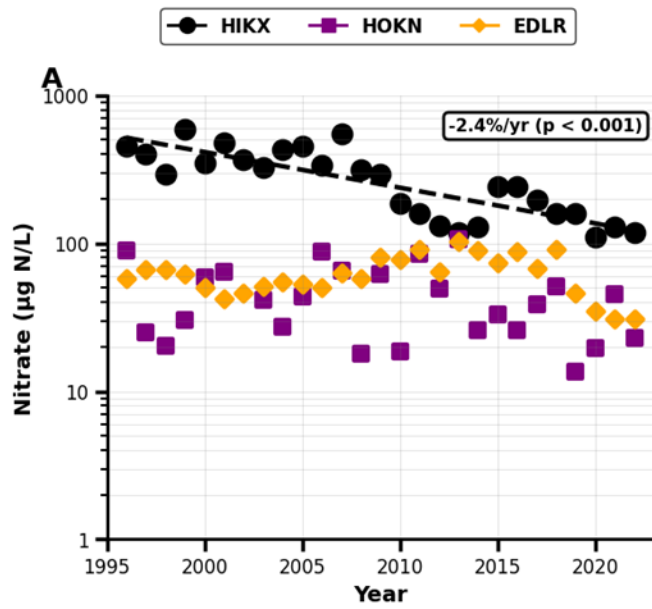


Figure 3.3: Stream nitrate (A), soluble reactive phosphorus (B), and total phosphorus (C) concentrations at three monitoring sites (1995-2022). Black circles show HIKX (downstream site below cropland); purple squares show HOKN (upstream of cropland); orange diamonds show EDLR (groundwater spring). Linear regression equations: Panel A (HIKX): $\ln(\text{NO}_3^- - \text{N}, \mu\text{g N L}^{-1}) = \beta_0 - 0.024(\text{year} - 1995)$, $R^2 = 0.68$, $p < 0.001$ (equivalent to -2.4%/yr decline). Panel B (HIKX): $\ln(\text{SRP}, \mu\text{g P L}^{-1}) = \beta_0 - 0.022(\text{year} - 1995)$, $R^2 = 0.54$, $p < 0.001$ (-2.2%/yr decline); (HOKN): $\ln(\text{SRP}, \mu\text{g P L}^{-1}) = \beta_0 - 0.021(\text{year} - 1995)$, $R^2 = 0.35$, $p < 0.001$ (-2.1%/yr decline). Panel C: No significant temporal trends detected. Lines show linear trends with 95% confidence intervals.

SRP showed strong declining trends at two sites. HIKX showed the most pronounced decrease, dropping 70% from 20 to 6 $\mu\text{g P L}^{-1}$ over the 25-year period. HOKN also exhibited a decline of 50% (from 12 to 6 $\mu\text{g P L}^{-1}$), while EDLR maintained consistently low SRP concentrations around 3 to 5 $\mu\text{g P L}^{-1}$. Total phosphorus (TP) showed similar declining patterns, with HIKX decreasing by 40% from initial values of 45 to 27 $\mu\text{g P L}^{-1}$ by 2020. HOKN showed a moderate decline (from 25 to 17 $\mu\text{g P L}^{-1}$), while EDLR maintained stable TP concentrations around 15 $\mu\text{g P L}^{-1}$ throughout the study period.

Year had strong effects on all nutrients: NO_3^- ($F(1,74) = 18.1$, $p < 0.001$), SRP ($F(1,74) = 18.8$, $p < 0.001$), and TP ($F(1,74) = 4.8$, $p = 0.03$), primarily driven by decreases at HIKX. Site effects occurred for NO_3^- ($F(2,74) = 10.4$, $p < 0.001$) and SRP ($F(2,74) = 6.4$, $p = 0.003$). The interactions between site and year were present for both NO_3^- ($F(2,74) = 10.1$, $p < 0.001$) and SRP ($F(2,74) = 6.2$, $p = 0.003$), largely attributed to the pronounced decreases at HIKX compared to other sites. Total phosphorus showed no site-specific effects, suggesting different controlling factors compared to other nutrient forms (Table S3.3).

Changes in Management Practices Influencing Nutrient Concentrations

Sequential restoration of cropland to unfertilized prairie began in 1995, resulting in a 75% reduction in fertilized area above the HIKX site between 1978 and 2022. The proportion of land cultivated for agriculture in the entire Kings Creek watershed declined from 1% in 1979 to

0.4% by 2019 (Fig. S3.2). Nitrogen inputs were highly variable, peaking in the mid-1980s and around 2000. Nitrogen inputs decreased by 67% over the study period, with the steepest declines after 2005 ($8.3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, $p < 0.01$). Phosphorus inputs dropped sharply after 1985 and continued declining, with phosphorus applications ceasing entirely by 2015 (Fig. 3.5). These reductions in fertilizer application corresponded temporally with observed NO_3^- declines at HIKX, which decreased from 450 to 110 $\mu\text{g N L}^{-1}$ following a lag period of 4-5 years post-fertilizer reduction. However, the nearest upstream site to HIKX, HOKN, had concentrations of roughly 50-80 $\mu\text{g NO}_3^- \text{-N L}^{-1}$, indicating that a longer period than 4-5 years could be required to reach levels seen in areas with no cropland immediately upstream.

Trends in Nutrients in Groundwater Wells under Grassland or Sequential Restoration from Cropland

Groundwater NO_3^- declined in the cropland well but remained elevated compared to the grassland well. The cropland well (ALC-A5) exhibited high initial NO_3^- concentrations followed by a pronounced decline of 70% from 3,200 to 950 $\mu\text{g NO}_3^- \text{-N L}^{-1}$ between 2000 and 2020. In contrast, the grassland well (K01A-P5) maintained consistently lower NO_3^- concentrations, averaging 200 $\mu\text{g NO}_3^- \text{-N L}^{-1}$ throughout the study period. However, these values were greater than Kings Creek at the nearby HOKN site (Fig. 3.4).

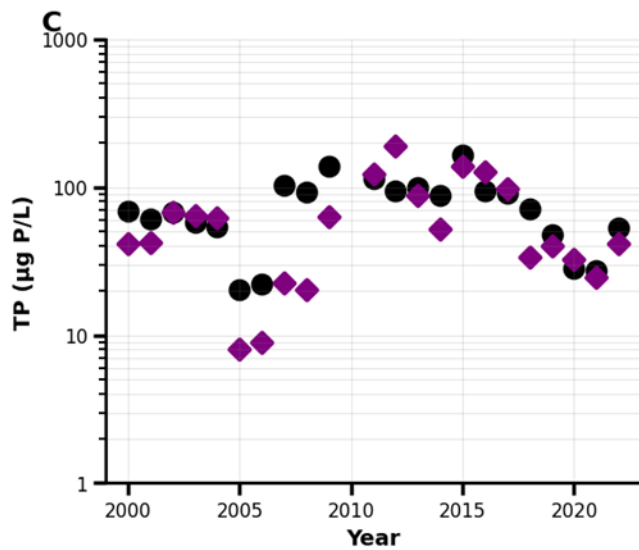
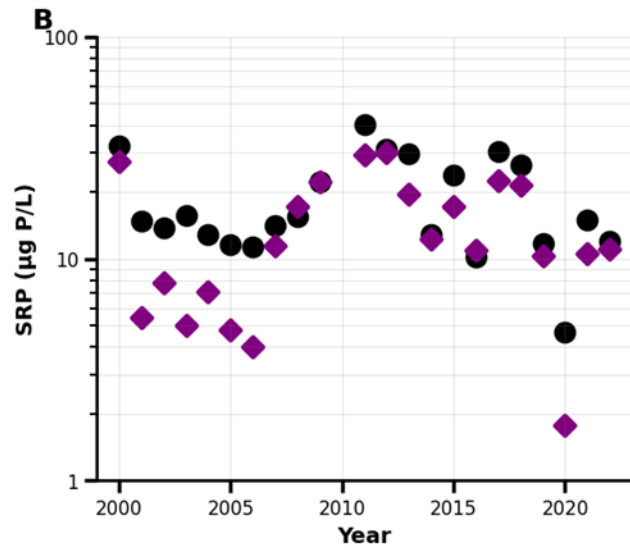
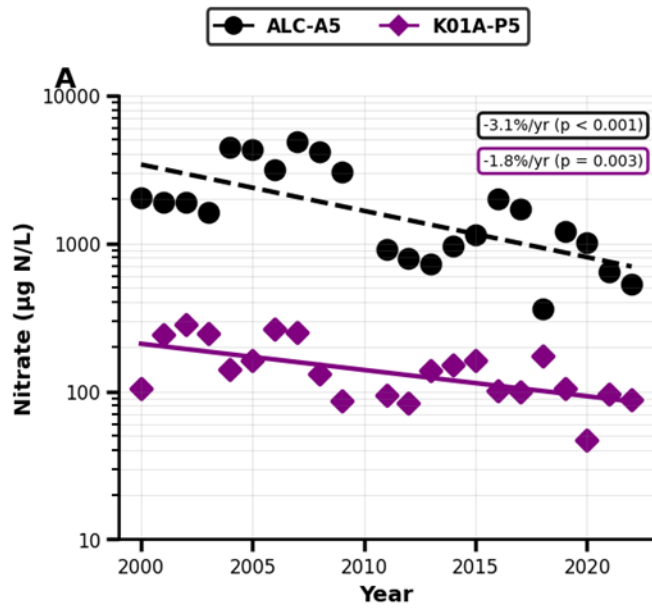


Figure 3.4: Groundwater nitrate (A), soluble reactive phosphorus (B), and total phosphorus (C) concentrations at two monitoring wells (2000-2022). Black circles show ALC-A5 (agricultural well under sequentially restored cropland); purple diamonds show K01A-P5 (grassland reference well). Linear regression equations: Panel A (ALC-A5): $\ln(\text{NO}_3^- \text{-N}, \mu\text{g N L}^{-1}) = \beta_0 - 0.031(\text{year} - 2000)$, $R^2 = 0.72$, $p < 0.001$ (equivalent to -3.1%/yr decline); (K01A-P5): $\ln(\text{NO}_3^- \text{-N}, \mu\text{g N L}^{-1}) = \beta_0 - 0.018(\text{year} - 2000)$, $R^2 = 0.45$, $p = 0.003$ (-1.8%/yr decline). Panels B and C: No significant temporal trends detected. Lines show linear trends with 95% confidence intervals.

Soluble reactive phosphorus (SRP) concentrations showed distinct temporal phases at both sites. At ALC-A5, SRP concentrations began at $15 \mu\text{g P L}^{-1}$ in 2000, increased to $40 \mu\text{g P L}^{-1}$ during 2010-2012, before declining to $15 \mu\text{g P L}^{-1}$ by 2020. K01A-P5 displayed a similar temporal pattern but with a lower magnitude, ranging from 5 to $30 \mu\text{g P L}^{-1}$ with peak concentrations occurring during the same period. Total phosphorus (TP) exhibited a more pronounced temporal pattern at both sites. Both wells showed relatively low initial concentrations (40 to $60 \mu\text{g P L}^{-1}$) in the early 2000s, followed by substantial increases during 2010-2015 (reaching $170 \mu\text{g P L}^{-1}$ for K01A-P5 and $140 \mu\text{g P L}^{-1}$ for ALC-A5), before returning to baseline levels by 2020.

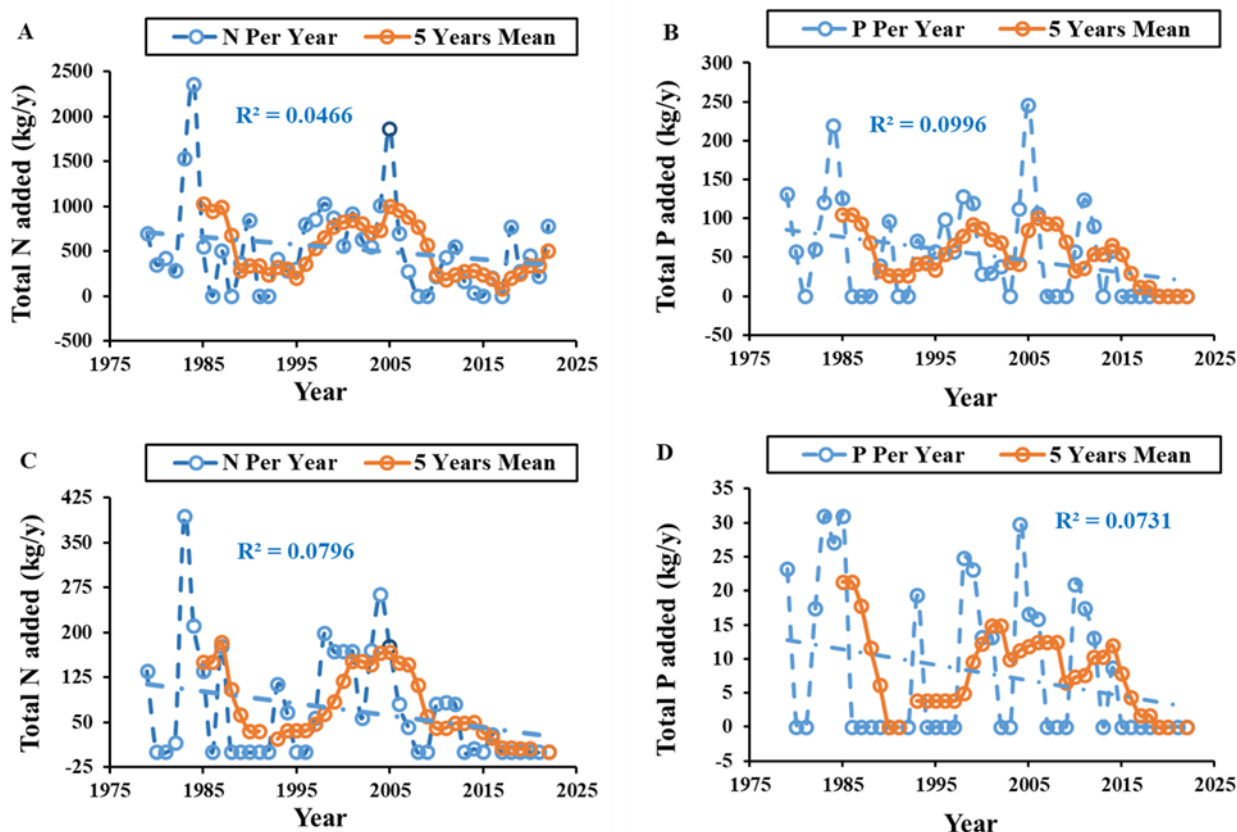


Figure 3.5: Total nitrogen and phosphorus fertilizer applications to croplands in the Kings Creek watershed (1978-2022). Panels A and B show fertilizer inputs to all cropland areas upstream of HIKX monitoring site; panels C and D show inputs to fields immediately adjacent to ALC-A5 groundwater wells. Blue circles connected by dashed lines represent annual applications; orange circles connected by solid lines represent 5-year running means. R^2 values indicate weak correlations for declining trends. Fertilizer applications ceased entirely by 2015 following sequential cropland restoration.

Site $\times$ year interaction and site effects for NO_3^- yielded F-statistics of 2.9 and 3.1, respectively ($p = 0.09$ and 0.08). Neither SRP nor TP showed effects of year, site, or their interaction ($p > 0.05$ for all tests), suggesting that phosphorus dynamics in these groundwater wells were governed by different factors than those controlling nitrate concentrations (Table S3.4).

Bison and Cattle Effects on Water Nutrients

Bison grazing roughly doubled stream NO_3^- concentrations. Nitrate concentrations in the bison-grazed South Fork Kings Creek (SFKC) watershed were two times higher than in the ungrazed North Fork Kings Creek (NFKC) ($p < 0.001$) (Fig. 3.6a). SRP concentrations were 40% higher in SFKC compared to NFKC ($p = 0.08$) (Fig. 3.6b). Mean TP levels were 11.3% higher in the bison-grazed SFKC watershed compared to NFKC ($p < 0.001$) (Fig. 3.6c) (Table S3.5).

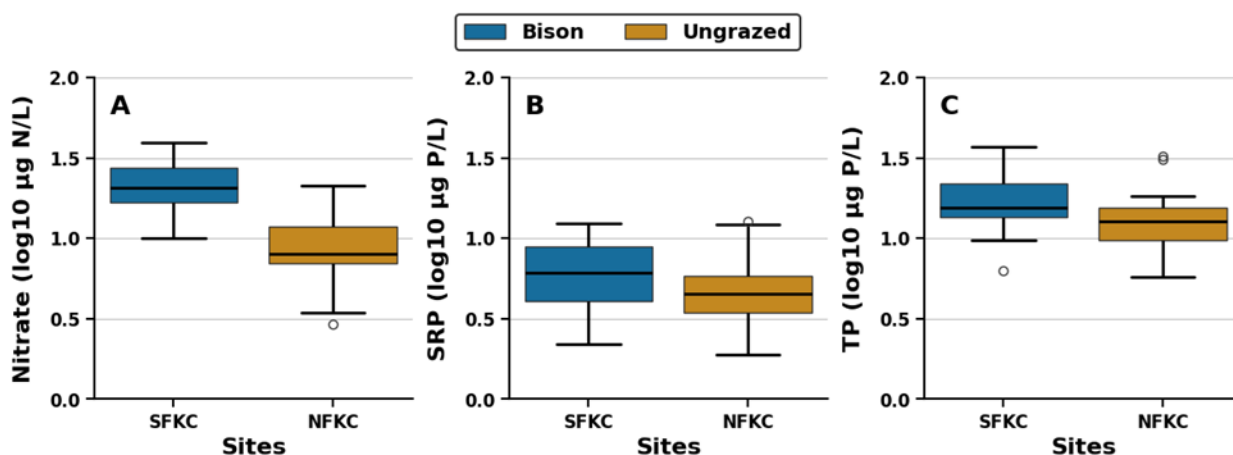


Figure 3.6: Nutrient concentrations (nitrate, soluble reactive phosphorus (SRP), and total phosphorus (TP) between two sites - SFKC, in which bison graze the sub-watersheds, and NFKC, which is ungrazed (n=1044 paired samples for SFKC vs NFKC comparisons, from 1999 to 2022). The line in the box plot indicates the median, 25th and 75th percentiles are the top and bottom of the box, and the whiskers give the 95th and 5th percentiles.

Cattle presence increased stream nutrients more than bison. In the Shane Creek watershed, where cattle graze for half of the year and a third of the area is burned annually, nitrate concentrations were 33% higher during cattle presence compared to periods without cattle. SRP levels were 12% higher and TP was 9% higher when cattle were present, though these differences showed high variability (Fig. 3.7, Table S3.6)

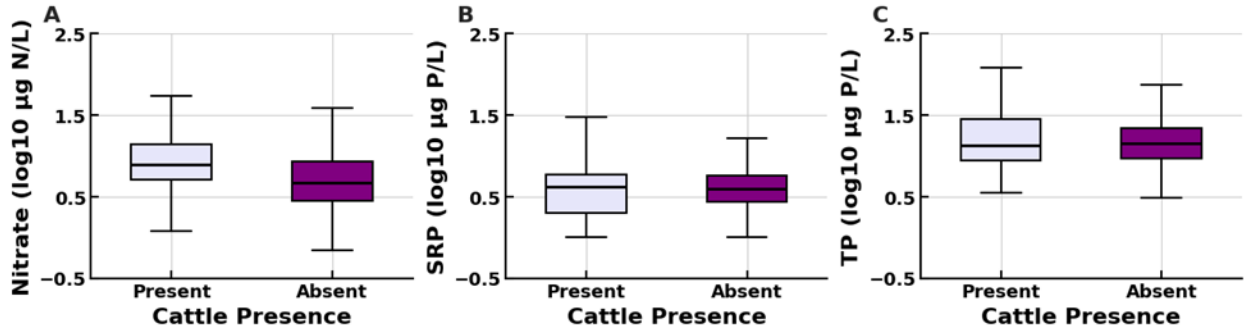


Figure 3.7: Stream nutrient concentrations in Shane Creek watershed comparing periods when cattle were present versus absent (2009-2022): A) nitrate, B) soluble reactive phosphorus (SRP), and C) total phosphorus (TP) (n=697). Box plots show median (center line), 25th and 75th percentiles (box boundaries), and 5th and 95th percentiles (whiskers). Gray lighter boxes represent cattle present periods; purple darker boxes represent cattle absent periods.

Discussion

Agricultural fertilization dominated water quality effects, followed by grazing, while fire had minimal direct impacts. Stream NO_3^- below restored croplands declined 75% over 25 years, but groundwater recovery remained incomplete after two decades. Bison doubled stream nitrate compared to ungrazed watersheds, while cattle had stronger seasonal effects.

Fire effects

Nitrate concentrations were 3-4 times higher at N1B (annually burned with bison grazing) than N4D and N2B generally which had lower nutrient levels than downstream areas, consistent with Kemp & Dodds (2001). However, strong temporal and spatial variations in NO_3^- concentrations were observed across N1B, N2B, and N4D watersheds, with annually burned sites having 3-4 times higher levels. Previous findings from tallgrass prairie show that fire and bison grazing together increase stream nitrogen concentrations, with short-term (weeks) elevations following burns (Larson et al., 2013). The proximate mechanisms include short-term post-burn nitrification pulses (Wan et al. 2001), elevated soil temperatures during the early post-burn growing season, and a brief regrowth lag in plant uptake, effects that dissipate as regrowing vegetation re-establishes nutrient demand (Blair 1997, Turner et al. 1997). While fire frequency showed weak effects on NO_3^- , the 4-to-5-fold higher concentrations at N1B (annual burn + bison) versus N2B/N4D (less frequent burn + bison) likely reflect concentrated bison use at a spring above the N1B sampling site rather than fire alone. Our design cannot statistically separate fire $\times$ grazing interactions, but the strong site effects versus weak fire effects suggest spatial heterogeneity in grazing intensity overrides burning frequency as the primary driver. Years since fire were not well correlated with stream nutrient concentrations. Our design cannot separate fire effects from grazing because bison prefer recently burned watersheds, particularly

when it has been longer since the previous burn (Larson et al., 2013). This pyric herbivory makes isolating fire effects challenging (Allred et al., 2011b).

Soluble reactive phosphorus levels remained stable across the three watersheds with both bison grazing and varied fire frequencies, contrary to our initial hypotheses. This stability suggests that phosphorus dynamics may be more resilient to burning regimes, possibly due to strong soil sorption or efficient biological uptake. SRP levels were slightly higher in prescribed burn areas, likely due to higher soil phosphorus relative to volatilized nitrogen post-fire. Total phosphorus trends varied across sites, with increases in N1B and N4D but decreases in N2B, suggesting factors other than burning frequency control phosphorus dynamics.

Nitrogen concentrations increased over decades in bison-grazed watersheds. The most pronounced NO_3^- increases over time occurred in the N1B watershed, although N2B also had an increase. The sampling point in N1B is downstream from a spring-fed reach frequented by bison, and the increase suggests gradual nitrogen saturation. Although NH_4^+ data were not analyzed in this study, previous work at KPBS documented elevated NH_4^+ in bison urine patches (150-300 mg N/kg soil) compared to background soils (5-10 mg N/kg), with rapid nitrification to NO_3^- within 7-14 days (Frank et al., 2016). The elevated NO_3^- in grazed areas aligns with this enhanced N mineralization (Frank et al., 2016). However, alternative explanations including atmospheric nitrogen deposition (Porter et al., 2013; Stevens et al., 2015) and changes in discharge which has decreased over time at Kings Creek (Keen et al., 2023; Raihan et al., 2025), which could increase concentrations through reduced dilution (Sadayappan et al., 2026). The analysis of Sadayappan et al. (2026) of concentration-discharge patterns in Kings Creek indicated that NO_3^- concentrations responded either not at all or with increased concentrations to short-term increases in discharge.

Agricultural Legacy Creates Decades-Long Nutrient Elevation

Riparian buffer zones can intercept nitrogen and phosphorus (Husk et al., 2023; Larson et al., 2024). However, at site HIKX, deep channel incision allows nutrients to bypass riparian roots, as documented for some other sites (Hill, 2019; Pacific et al., 2010). In contrast, at HOKN, which is not incised with oak gallery forest, nutrient concentrations in grassland wells (K01A-P5) were several-fold higher than in the stream, indicating riparian vegetation intercepted and retained nitrogen.

Declining NO_3^- at ALC-A5 and HIKX demonstrate that grassland restoration reduces nitrate loading in streams (Dubrovsky et al. 2010); the 4–5 year lag and ongoing decline likely reflect both the cessation of fertilizer inputs and the progressive re-establishment of plant–microbial N demand as soil carbon accumulates in restored fields (Baer & Blair, 2008; Rosenzweig et al., 2016). Phosphorus dynamics appeared decoupled from surface waters. Limited P mobility to groundwater reflects strong sorption and precipitation with calcium in limestone-derived soils, consistent with similar calcareous systems (Holman et al., 2008; Domagalski & Johnson, 2012).

HOKN had homesteads and then a house with a septic tank until the 1980s. Ongoing declining TP may reflect decades-long recovery. Stable concentrations at EDLR suggested that uplands did not influence other sites. A 75% reduction in stream NO_3^- over 25 years downstream of restored croplands aligns with other agricultural watersheds (Meals et al., 2010; Sebilo et al., 2013).. However, groundwater NO_3^- remained elevated at $950 \mu\text{g N L}^{-1}$ even 20+ years post-restoration, compared to $200 \mu\text{g N L}^{-1}$ in the grassland reference well, reflecting 'biogeochemical stationarity' noted by Basu et al. (2010).

Bison and Cattle Effects

Bison grazing doubled stream NO_3^- over decades. The bison-grazed SFKC watershed had two times higher NO_3^- levels compared to the ungrazed NFKC area, substantially higher than the 35% increase observed in earlier 4-year studies (Kemp & Dodds, 2001). This larger difference may reflect grazing-related disturbances and accelerated nitrogen mineralization, influencing plant-soil interactions that have accumulated over time (Frank & Evans, 1997; Hobbs, 1996). TP levels were modestly higher in bison-grazed areas, likely due to enhanced erosion from physical disturbances (Zhang et al., 2017).

In the Shane Creek watershed, NO_3^- levels were 33% higher during periods of cattle presence, likely due to nitrogen inputs from urine, feces, and enhanced erosion from riparian disturbances (Line et al. 2000, Malan et al. 2018). Cattle reintroduction to this watershed increased erosion 2-fold (Grudzinski et al., 2018), contributing to modest increases in TP levels during grazing periods.

Bison and cattle affect watersheds through different mechanisms. Cattle physically disturb riparian zones, leading to enhanced erosion and phosphorus mobilization (McDowell et al., 2003), while bison influence nutrient cycling primarily through selective grazing patterns, more broadly distributed across watersheds (Frank & Evans, 1997). Both species contribute nitrogen through localized urine and feces deposition, but cattle preferentially use riparian areas while bison prefers uplands and access streams via specific trails.

Legacies and temporal and spatial heterogeneity

Agricultural legacies create the strongest and most persistent water quality impacts. Most striking was the persistence of elevated groundwater NO_3^- concentrations ($950 \mu\text{g N L}^{-1}$) in areas previously used for agriculture, a pattern that underscores the lasting impact of historical farming

practices. This accumulated nitrogen creates a biogeochemical inheritance that persists long after changes in land management (Alexander et al., 2008; Van Meter et al., 2016).

Conclusions

Our data reveal a clear hierarchy of anthropogenic effects on stream nutrients. Early in the time series, NO_3^- concentrations at HIKX were high relative to upstream tributaries (1-2 orders of magnitude), despite the croplands making up only 1% of the upstream watershed area. The declines in NO_3^- at HIKX correlate with the removal of fertilized cropland, confirming that restoration to grassland led to recovery. This contrasts with cattle and bison effects (doubled NO_3^- in bison areas over decades and increased by 33% when cattle were on the pasture), which were modest even though both ungulates had access to the entire watershed. The use of fire, necessary to maintain grassland, also had more modest influences on stream nutrients, which showed <50% differences between treatments with inconsistent responses to burn frequencies.

Management priorities to improve water quality should focus on fertilizer reduction and grazing management. Historical cropland use creates the strongest and most persistent effects (NO_3^- elevated 4 to 5 fold for >20 years), followed by grazing impacts (bison increased NO_3^- by 100%, cattle by 33%). Fire frequency effects were weak at the watershed scale, likely because of rapid vegetation recovery, precipitation timing, and fire-grazing interactions that override direct fire effects. However, fire remains essential for maintaining grassland vegetation structure. The observed decrease in discharge at Kings Creek (Keen et al., 2023; Raihan et al., 2025) may reflect broader climate trends affecting Great Plains hydrology. Regional climate projections suggest increased precipitation variability, longer dry periods, and more intense rainfall events (Garbrecht et al., 2004; Rahmani et al., 2016). These changes could fundamentally alter nutrient transport by: (1) concentrating nutrients during low flows, (2) increasing erosional P export during intense events, and (3) reducing denitrification potential as streams become more

intermittent (Arango et al., 2007; Zimmer et al., 2020b). The interaction between land management and climate variability will likely become increasingly important. For instance, Brookshire et al. (2011) found that temperature effects on forest nitrogen cycling can be revealed through long-term watershed records, suggesting that multi-decadal studies like ours are essential for detecting climate-driven changes in nutrient dynamics.

Acknowledgments

We acknowledge that research on KPBS uses land taken from those who lived here before us, including the Kaá'ze (Kaw / Kansa, from who we have the site name), Wah.zha.zhe ᖃᖃᖃᖃᖃᖃ (Osage), Očhéthi Šakówinj (Sioux), and Kiikaapoi (Kickapoo), and hope that our work honors their existence and those of their ancestors who remain. We thank James Guinnip for conceptualizing this research and compiling the comprehensive management history records that were essential to interpreting our long-term water quality data. We thank the many workers for the upkeep and data curation for the monitoring equipment used in this research, particularly the analytical chemists (Rosemary Ramundo, Courtney Tobler, and Janaye Figge). We are grateful to the anonymous reviewers and the associate editor for their constructive comments and valuable suggestions, which significantly improved this manuscript. The research was funded by the US National Science Foundation LTER program (DEB: 0823341, 1440484, and 2025849). The data are publicly available through the LTER program at KPBS and the Environmental Data Initiative (stream chemistry: NWC01 doi:10.6073/pasta/bb6b065e5b25234dd1bb80ff476933e0; groundwater chemistry: AGW021, doi:10.6073/pasta/0d2a03409c57f11905e4ae073720b8e7).

Chapter 4 - Vegetation Cover Thresholds and Fire-Grazing Interactions Regulate Organic and Inorganic Sediment Pathways in Tallgrass Prairie Streams

Abstract

Elevated suspended sediment degrades freshwater ecosystems globally, yet how organic and inorganic fractions respond to grassland fire and grazing management remain unresolved. We integrated 10 rainfall simulation experiments with bromide tracer, 15 years of watershed monitoring (2009–2023), and streambank geomorphic surveys at Konza Prairie Biological Station (KPBS) (Kansas, USA) to test whether fire frequency and grazer identity (bison or cattle grazed, or ungrazed) differentially control volatile organic suspended solids (VSS) and inorganic suspended solids (ISS) in streams. An area-normalized erodibility index (Sed/m^2) revealed a nonlinear riparian vegetation cover threshold between approximately 60% and 70%, below which areal erodibility increased by approximately two orders of magnitude. Two high-cover sites (Exps 2 and 3) produced no detectable bromide breakthrough despite strong simulated storms, while bare bison-grazed soil was 168 times more erodible per unit area than an adjacent fully vegetated surface at a similar bank slope. Generalized additive models (43–46% variance explained) demonstrated that discharge exerted strong negative effects on VSS but weaker effects on ISS, while days since burn acted in opposing directions on the two fractions, positive for ISS, negative for VSS, establishing fire management as a determinant of sediment composition. Precipitation and discharge operated in opposite directions within the same model, revealing a two-phase storm process of mobilization followed by dilution not previously documented in grassland headwaters. Structural equation modeling confirmed the directionality

of these pathways, with days since burn exerting the strongest direct effect on the VSS/ISS ratio ($\beta = -0.405$). The VSS/ISS ratio was more sensitive to management than any individual fraction: biennial burning under bison maintained the most organic-dominated regime, while quadrennial burning produced the most inorganic-dominated, with persistent watershed-specific sediment regimes sustained across the 15-year record. Streambank surveys corroborated the inorganic pathway, with woody cover as the strongest erosion predictor ($OR = 4.19$, $p = 0.020$), and the watershed combining the longest fire interval with the lowest canopy cover (N04D; 15%) exhibited the highest bank cut frequency. The unexpected finding that the cattle-grazed watershed (C3SB) had the lowest bank erosion (25%) despite intensive grazing, attributable to its dense riparian canopy (70%), demonstrates that vegetation conditions can override expected grazer identity effects. These findings establish that organic and inorganic sediment inputs are governed by different controls operating through distinguishable pathways, challenging the adequacy of total suspended solids as a stand-alone metric controlling sediment composition in tallgrass prairie streams.

Introduction

Sediment transport in stream ecosystems represents one of the most consequential, yet mechanistically complex material fluxes linking terrestrial and aquatic environments. Suspended sediment affects light attenuation, benthic habitat quality, nutrient sorption and release, and the metabolic rates of stream communities (Bilotta & Brazier, 2008). In agricultural and grazed landscapes, accelerated sediment delivery from uplands and eroding streambanks constitutes a leading cause of freshwater quality degradation globally (Horowitz, 2009; Walling, 2005). Riparian zones, the narrow ecotonal strips between terrestrial and aquatic ecosystems, exert disproportionate influence on stream water quality relative to their small areal footprint. Through physical filtration, biotic uptake, dissimilatory processes (e.g., denitrification), and geomorphic stabilization, intact riparian corridors intercept suspended and dissolved materials moving from uplands toward stream channels, reducing the delivery of sediment, nutrients, and other contaminants to receiving waters (Lowrance et al., 1984; Mayer et al., 2007; Peterjohn & Correll, 1984; Sweeney & Newbold, 2014).

Meta-analyses confirm that riparian buffers as narrow as 30 m can retain more than 75% of incoming sediment and nutrient loads under typical agricultural conditions (Lind et al., 2019). However, this filtering capacity is neither fixed nor uniform: it emerges from multiple interacting physical and ecological gradients: vegetation structure, soil hydrology, root architecture, and geomorphic position all can determine the functional properties of the riparian zone under any given management regime (Bernhardt et al., 2017; Dodds et al., 2025). Understanding what drives variability in this capacity is essential for predicting how management decisions influence erosion and downstream water quality.

The structure of riparian vegetation and the functional condition of streambanks in grasslands are shaped by the co-evolutionary interaction of fire and ungulate grazing, two disturbance agents that have structured tallgrass prairie ecosystems for millennia (Collins & Calabrese, 2012; Knapp, et al., 1999). Fire removes accumulated litter and woody encroachment, maintaining grass dominance and modulating nutrient cycling; ungulate grazing controls biomass and selectively removes plant tissue, altering vegetation structure, soil compaction, and hydrological connectivity between uplands and streams (Naeth et al., 1990; Trimble & Mendel, 1995).

The identity of the large ungulate herbivore matters as well: American bison (*Bison bison*) and domestic cattle (*Bos taurus*) differ in their grazing behavior, movement patterns, and effects on vegetation structure and soil properties (Fuhlendorf & Engle, 2004; Knapp, et al., 1999). Bison tend to create spatially heterogeneous grazing patterns with preferential use of upland areas, while cattle concentrate near water sources and riparian zones, compacting soils, reducing vegetative cover, and increasing bare ground exposure along streambanks, conditions known to accelerate both overland sediment delivery and direct bank erosion (Belsky et al., 1999; Trimble & Mendel, 1995; Zaimes et al., 2019).

At Konza Prairie Biological Station (KPBS), fire and bison grazing directly influence stream suspended sediment concentrations, with bison increasing sediment loads linked to greater bare ground coverage in grazed watersheds (Grudzinski et al., 2016; Larson et al., 2013). More broadly, land-use transitions from native prairie to cropland alter stream water chemistry over decadal timescales (Chapter 3). However, most empirical work has examined nutrients rather than sediment, and has treated fire and grazing as independent rather than interacting drivers. The degree to which fire and grazing interact to control suspended sediment

concentrations and fractions remains poorly quantified, particularly because sediment dynamics integrate multiple source and transport processes operating at different spatial and temporal scales that are difficult to capture with any single dataset or method.

The two components of stream suspended sediment, volatile suspended solids (VSS) and inorganic suspended solids (ISS), respond to distinct environmental controls (Bilotta & Brazier, 2008; Dodds & Whiles, 2004). ISS is predominantly mineral in origin and reflects physical disturbance processes, including soil detachment by raindrop impact, overland flow erosion, and streambank failure; it is therefore most sensitive to bare ground cover, soil erodibility, and bank geomorphic condition. VSS is composed mostly of organic material (e.g., litter inputs, algal biomass, and organic matter resuspension) and is potentially coupled to seasonal patterns of in-stream primary production and riparian litter fall. The sum of these fractions constitutes total suspended solids (TSS), but because the two components arise from different source processes and respond to different drivers, failure to distinguish them may obscure important consequences of management. Furthermore, concentrations of suspended sediment are highly variable across whole watersheds, necessitating long-term, whole-watershed monitoring to assess the dominant factors influencing VSS and ISS sources.

While substantial work has examined riparian sediment filtration in cropland and urban settings (Mayer et al., 2007; Sweeney & Newbold, 2014), the role of riparian vegetation structure in native grasslands in mediating overland sediment delivery during high-intensity precipitation events remains underexplored. Plant cover intercepts raindrop energy, increases surface roughness, promotes infiltration, and physically traps sediment in transit, collectively reducing both the detachment and transport capacity of overland flow (Nearing et al., 2017; Prosser et al., 1995). In tallgrass prairie, where convective thunderstorms regularly generate intense rainfall,

the condition of riparian vegetation during a storm may be as important as antecedent soil moisture or storm magnitude in determining whether runoff reaches the stream laden with sediment or is filtered through a vegetated buffer (Sweeney & Newbold, 2014). Dodds et al., (2022) demonstrated with rainfall simulation at KPBS that a well-vegetated ungrazed riparian zone on Ivan silty clay loam retained more than 50% of bromide tracer applied during an 80 mm simulated event, with minimal sediment mobilization despite extreme rainfall intensity. Whether this buffering capacity is maintained under cattle or bison grazing, which reduces plant cover, alters litter architecture, and compacts soil with foot traffic within riparian zones has not been experimentally tested.

Streambank erosion constitutes a third mechanism of sediment input to grassland streams. Bank failure and lateral retreat depend on the interplay of soil texture, pore water pressure, bank geometry, and the mechanical reinforcement provided by plant roots (Simon & Collison, 2002; Zhao et al., 2022). Cattle trampling near riparian margins destabilizes banks, creates preferential flow paths, and increases porewater pressure fluctuations that amplify failure probability (Belsky et al., 1999; Trimble & Mendel, 1995). Fire frequency could modulate these effects by influencing aboveground biomass and root turnover rates, potentially altering the mechanical properties of the root–soil matrix on timescales corresponding to typical burn rotation intervals. Despite the mechanistic plausibility of these interactions, geomorphic surveys quantifying how fire–grazing management shapes bank erosion susceptibility across prairie watershed networks are absent from the literature.

KPBS provides an unparalleled landscape for addressing these questions. With over four decades of continuous hydrological and biogeochemical monitoring under factorial combinations of burn frequency (1-, 2-, 4-, and 20-year intervals) and grazer identity (bison, cattle, ungrazed),

KPBS offers the long-term context necessary to separate management effects from interannual climate variability. The transition of Shane Creek subwatersheds (C3SA, C3SB, C3SC) from ungrazed conditions to cattle production in 2011, combined with an active woody vegetation removal program in bison-grazed N02B beginning that same year (Dodds et al., 2023; Larson et al., 2019), created conditions in which shifts in management regime can be tracked against established pre-transition baselines, enabling stronger causal inference.

We take a three-pronged approach to resolve the mechanistic controls on suspended sediment dynamics in these prairie headwater streams. Using 15 years of suspended sediment monitoring (2009–2023) across watersheds with contrasting burn frequency and grazer identity, we characterize long-term temporal trends and test for fire $\times$ grazer interactions in TSS, VSS, and ISS concentrations. Using rainfall simulation experiments with a bromide tracer across riparian plots spanning gradients of vegetation cover, slope, and grazing intensity, we experimentally quantify overland sediment delivery as a function of riparian condition. With standardized geomorphic bank surveys across the watershed network, we assess how management regimes shape the structural condition of streambanks as a direct sediment source. By integrating these three lines of evidence, we test how fire and grazer identity could interact to produce sediment outcomes that neither driver would generate independently.

Our specific predictions, derived from the multi-gradient conceptual framework for riparian zone function advanced by Dodds et al. (2025; Fig. 4.1), are:

(i) Grazer identity is a stronger predictor of ISS than burn frequency, with cattle-grazed watersheds showing higher inorganic sediment concentrations than bison-grazed watersheds at equivalent burn intervals, reflecting greater bare ground exposure and riparian soil compaction under cattle management.

(ii) Vegetation cover is the dominant control on overland sediment delivery in rainfall simulation experiments, with delivery rates increasing nonlinearly with decreases in cover.

(iii) Cattle-grazed watersheds exhibit taller, steeper, more geometrically unstable banks with greater bare-face exposure and lower root density than bison-grazed or ungrazed watersheds, reflecting chronic physical disturbance by cattle near water.

(iv) Fire frequency and grazer identity interact nonlinearly: frequent burning amplifies inorganic sediment concentrations in cattle-grazed watersheds by reducing cover between burn events but exerts minimal or even negative effects in bison-grazed watersheds where patchy grazing maintains heterogeneous cover even in recently burned areas.

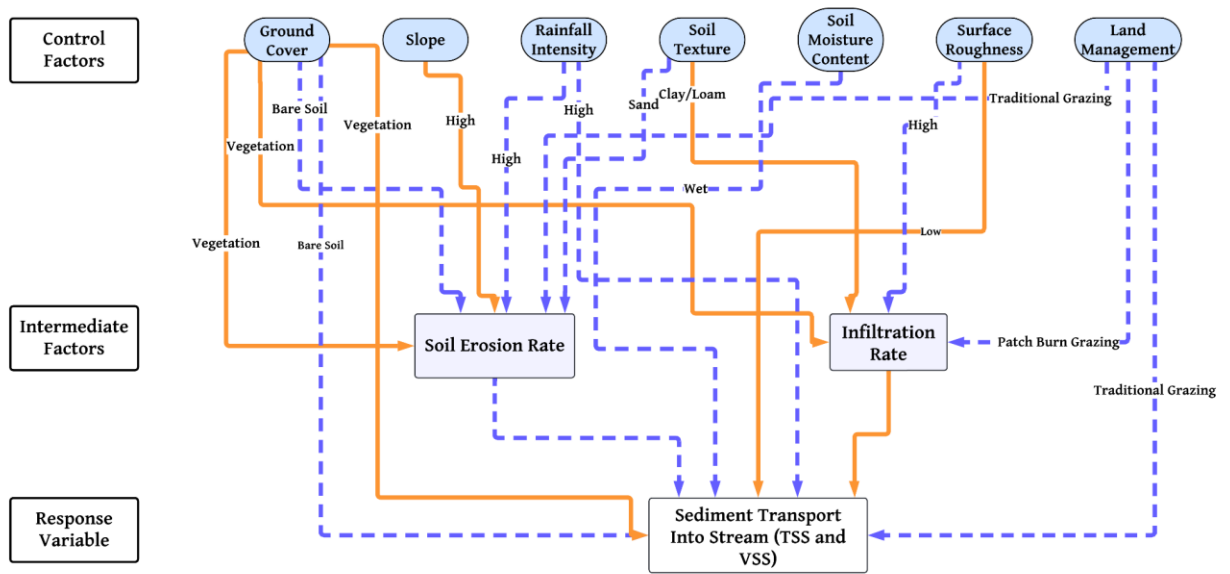


Figure 4.1: Conceptual framework linking fire frequency, grazer identity, and riparian condition to organic and inorganic sediment delivery pathways in tallgrass prairie streams. Solid yellow lines indicate negative relationships; dashed blue lines indicate positive relationships.

Methods

Study Site

This study was conducted at KPBS, a 3,487-ha native tallgrass prairie Long-Term Ecological Research (LTER) site located in the Flint Hills of northeastern Kansas (39°05'N, 96°35'W). KPBS is underlain by alternating chert-bearing limestone and shale, generating a mosaic of uplands dominated by C₄ grasses (*Andropogon gerardii* and *Sorghastrum nutans* on deeper soils, with *Schizachyrium scoparium* and *Bouteloua spp.* on shallow ridges) and lower-slope woody draws. Mean annual precipitation is approximately 835 mm, with substantial interannual variability (coefficient of variation ~25%) driven by convective thunderstorm activity concentrated in spring and early summer (Raihan et al., 2025). The Kings Creek watershed and associated experimental sub-watersheds, and Shane Creek immediately to the north, drain intermittent and perennial headwater streams that are the focus of the KPBS hydrology and biogeochemistry monitoring program.

The KPBS watershed-scale experimental treatments include factorial combinations of burn frequency and ungulate grazer identity. For this study, primary Kings Creek watersheds included: N02B (2-year burn interval, American bison grazing, with riparian woody vegetation removal beginning in 2011; Larson et al., 2018; Dodds et al., 2023); N04D (4-year burn interval, bison grazing); and at the confluence of three Shane Creek sub watersheds (C3SA, C3SB, C3SC). This C3 sampling site is at the bottom of the watershed, where one of the three sub-watersheds is burned each year, rotating so the entire watershed is on a three-year burn frequency. The burn regime started in 2009, and cattle grazing began in 2011. Additional Kings Creek watersheds K02A (2-year burn, ungrazed) and K2B (Kings Creek main stem, 2-year burn, bison) were included in the streambank geomorphic surveys to provide further contrast in grazer

influences on riparian condition. Watershed areas range from approximately 54 to 273 ha.

Riparian zones are characterized by Ivan silty clay loam soils, with dense grass cover dominated by *Phalaris arundinacea* and *Spartina pectinata* in minimally disturbed reaches, and variable bare-ground exposure in cattle-grazed areas.

Dataset 1: Long-Term Suspended Sediment Monitoring

Sample Collection

Water samples for suspended sediment analysis were collected from stream gauging stations at watersheds N02B, N04D, and Shane Creek (C3SB) at a frequency of 2–3 times per week during periods of active streamflow. Sampling occurred from 2009 through 2023, yielding a 14-year monitoring record ($n > 800$ samples per watershed). Grab samples (4 L) were collected from the thalweg using pre-rinsed high-density polyethylene bottles and transported on ice to the laboratory within 4 hours of collection. Samples were rarely collected during intense storms, but efforts were made to sample as soon as possible after such storms when safe conditions existed for field work. The range and frequency of sample collection mean the dataset represents a broad range of flow conditions under varied storm intensities and different levels of soil moisture.

Laboratory Analysis

Total suspended solids (TSS), volatile suspended solids (VSS), and inorganic suspended solids (ISS) were determined using pre-combusted (550°C, 4 h) and pre-weighed Whatman glass microfiber filters (GF/C, 1.2 μm nominal pore size) upon return to the laboratory. Known volumes of thoroughly mixed samples were filtered under gentle vacuum, and filters were dried at 105°C for 24 h before weighing ($\text{TSS} = \text{dry weight} - \text{initial filter weight}$). Filters were then combusted at 550°C for 1 h and reweighed to obtain ash-free dry mass ($\text{VSS} = \text{TSS} - \text{final ash weight}$; $\text{ISS} = \text{TSS} - \text{VSS}$). All weights were recorded to ± 0.01 mg on a Mettler Toledo

analytical balance. Ashed samples were periodically re-wetted, dried, and re-weighed to correct for water of hydration, but this was discontinued because it yielded negligible weight changes. Stream discharge data for N04D and N02B were obtained from co-located v-notch flume gauging stations with continuous stage recording at 15-min intervals, converted to discharge using established stage–discharge rating curves (Konza LTER datasets ASD01, ASD02; available at <https://lter.konza.ksu.edu>).

Statistical Analysis Long-Term Trends

Suspended sediment data were aggregated to monthly means prior to modeling to improve signal-to-noise ratio and avoid temporal autocorrelation; daily resolution models yielded pseudo- $R^2 < 0.03$, while monthly aggregation consistently produced pseudo- $R^2 > 0.40$, reflecting the dominant role of storm-event variability at sub-monthly scales. Temporal trends in TSS, VSS, and ISS were analyzed using Generalized Additive Models (GAMs) with a Gamma error distribution and log-link function, implemented in the *mgcv* package in R (S. N. Wood, 2016). Each model included penalized thin-plate regression splines for stream discharge (log-transformed), antecedent precipitation (30-day cumulative), and days since last burn; cyclic cubic regression splines for month of year to capture seasonal patterns; and factor terms for burn frequency and grazer type as fixed categorical predictors.

To decompose the direct versus indirect pathways through which management affects sediment, Structural Equation Models (SEMs) were fitted using the *lavaan* package in R (Rosseel, 2012). Five competing model structures were evaluated across two families: Family A models included N02B and Shane Creek with channel cross-sectional area as a physical predictor; Family B models incorporated all three monitoring watersheds with drainage area as the scaling variable. Because TSS equals the sum of VSS and ISS (confirmed by correlation of r

> 0.97 between monthly values), only VSS and ISS were modeled as separate response endpoints to avoid structural multicollinearity; TSS was not included as a SEM variable. Final SEM structures treated ISS and VSS as separate response pathways with discharge and management as shared antecedents. Model fit was evaluated using CFI, RMSEA, and SRMR, with good fit defined as CFI > 0.95, RMSEA < 0.06, and SRMR < 0.08.

Dataset 2: Rainfall Simulation Experiments

Experimental Design

Rainfall simulation experiments were conducted during the summers of 2023 and 2024 to quantify overland sediment delivery from riparian plots to the stream channel across a gradient of vegetation cover, slope, and grazing history. The experimental design follows the approach of Dodds et al. (2022), who used the same precipitation simulation system at KPBS to characterize riparian zone retention of nitrate and bromide under a single high-intensity event. The present study extends this framework to 10 replicate trials, systematically varying biotic and abiotic conditions to enable analysis of controls on sediment delivery.

Water (946 L) was processed through mixed-bed deionizer resin to reduce conductivity by >100-fold (to <1 $\mu\text{S cm}^{-1}$) and lower all major ion concentrations below analytical detection limits, ensuring that bromide additions constituted the dominant anion signal in runoff. Sodium bromide (100 g NaBr) was dissolved and mixed thoroughly into the deionized water prior to each simulation.

The rainfall simulator consisted of two Conservation Demonstration (Salina, KS) overhead sprinkler heads mounted on tripods over plots to achieve roughly 10×3 m coverage immediately adjacent to the streamside but not falling directly into the stream water. Each sprinkler head was equipped with a VeeJet H1/2 U nozzle driven by an oscillating motor to

generate spatially uniform simulated rainfall. Water was delivered by two pumps from the main tank. Sprinkler heads were positioned at approximately 3 m above the soil surface (variable due to uneven topography), sufficient to approach the terminal velocity attained by natural raindrops. Approximately 20 rain gauges were distributed across each plot to measure average storm depth and characterize spatial variability arising from uneven topography, variable wind, and vegetation interception. Once the simulated storm was underway, the exact area receiving rainfall was flagged, and after the experiment, the length of each long edge and five widths across the narrower dimension of the approximate rectangle were measured to document the storm footprint.

Rainfall intensity was targeted to approximate the 95th-percentile regional annual storm intensity for the Flint Hills region ($\sim 80 \text{ mm h}^{-1}$), consistent with Dodds et al. (2022). Trial durations ranged from 30 to 73 minutes, depending on the time to steady-state runoff conditions.

Bromide Tracer Monitoring and Sample Collection

Each experiment was conducted adjacent to a relatively constrained stream reach. Baffles (wing dikes) were placed in the stream to assist lateral mixing as water passed the simulation plot. Natural or constructed constriction points below each experimental plot served as sampling locations. A Br^- probe was placed at each collection point to provide a real-time assessment of tracer arrival. Samples were collected every 5 minutes once the simulation began. When the Br^- signal reached the sampling point, TSS and Br^- samples were collected at a higher frequency as concentrations rose rapidly. Once Br^- concentrations reached an approximate steady state, precipitation was terminated. Sampling continued progressively longer intervals as the values from the Br^- probe returned more closely to values at pre-experiment levels, and values did not

change over at least 5 minutes. Background samples for Br⁻ and TSS were collected from upstream of the experiment approximately five times across each trial duration.

Strict care was taken to avoid disturbing the stream or bank during experiments. Stream samples were collected from deeper, quickly flowing areas without disturbance to the stream bottom. No personnel entered the riparian plot or stream channel within the runoff pathway during simulation.

Bromide concentrations were measured by ion chromatography (Dionex ICS-1100) in the laboratory of Dr. Matt Kirk (Department of Geology, Kansas State University). Discharge was gauged using NaCl constant-release dilution conducted the day before or after each simulation during baseflow conditions; discharge was assumed stable within this window. Suspended sediment analysis followed the same filtration protocol described for long-term monitoring.

Mass Balance Calculation

Mass export of Br⁻, ISS, and VSS was calculated by stepwise integration of background-corrected concentrations over the duration of each tracer pulse:

$$C_{\text{mass}} = \Sigma(t_0 \text{ to } t_n) [C_{\text{bkd}}] \times Q \times \Delta t$$

where C_{mass} is the total exported mass of the constituent, $[C_{\text{bkd}}]$ is the background-corrected concentration at each time point (corrected using pre-experiment downstream and during-experiment upstream background concentrations), Q is discharge, t_0 is the first sample, t_n is the last sample, and Δt is the interval between successive samples.

Bromide mass applied was calculated from the corrected solution chemistry: 100 g NaBr dissolved in 946 L deionized water yields 82.08 mg L⁻¹ Br⁻ (Br⁻ mass fraction in NaBr = 0.7765), depth of precipitation added and area of footprint. The minimum detectable bromide increase was determined as the upper limit of the 95% confidence interval of background

concentrations measured upstream of each plot and at the downstream sampling point prior to the simulation. Experiments in which the integrated downstream Br^- signal did not exceed this threshold were classified as below detection limit (BDL) and excluded from quantitative analyses, because sediment collected at the sampling point during these trials cannot be attributed to overland transport from the experimental plot in the absence of a detectable conservative tracer. Four of the ten experiments (2, 3, 4, and 7) were classified as BDL on this basis.

To enable comparison across plots that differed in storm depth and wetted footprint, sediment export was normalized by wetted plot area to yield an areal erodibility index (Sed/m^2 ; mg m^{-2}). Plot area was calculated from post-experiment field measurements: the length of each long edge and five widths across the narrower dimension of the rainfall footprint were averaged to compute a planar area. The areal erodibility index isolates the per-unit-area sediment response of the riparian surface from site-to-site variation in plot size and storm magnitude and was the primary erodibility metric used in subsequent analyses. Bromide recovery, expressed as a percentage of total Br^- mass applied, was retained as an independent indicator of hydrological connectivity between the plot and the channel.

Plot Characterization

After each simulation trial, each plot was characterized for: percent vegetation cover, estimated using a 1- m^2 quadrat grid method at 2.5-m intervals along three longitudinal transects; percent bare ground; litter depth (mm) measured at 10 random points; and surface slope determined using a stadia rod, level, and tape measure. Soil texture was determined by the hydrometer method (ASTM D422) at nine time intervals (0.5, 1, 3, 10, 30, 60, 90, 120, and 1440

minutes after stirring) from composite soil samples collected pre- and post-simulation, following Dodds et al., (2022).

Dataset 3: Streambank Geomorphic Surveys

Survey Design

Streambank geomorphic condition was assessed across four watersheds (N02B, N04D, K02A, and C3SB). Within each watershed, 10 survey locations were established at equal spacing along the main stem stream channel, extending from the headwater source to the watershed outlet. At each location, cross-sectional transects were laid perpendicular to the channel and extended 10 m from each bank at 2.5-m intervals, yielding five measurement points per bank per transect. This design characterizes the full lateral gradient of the riparian zone with respect to eroded banks and animal trails near the stream channel. Each set of transects was sampled twice (in 2023 and 2024), using GPS to ensure relocation. Paired measurements across years were averaged to account for small-scale interannual variance and differences among field crew members without inflating sample size.

Field Measurements

At each transect location, the following variables were measured: bank height (m) and angle (°) using an automatic surveying level (± 1 cm vertical precision) and clinometer ($\pm 0.5^\circ$); bank face condition scored on an ordinal scale from 1 (fully vegetated, no erosion) to 5 (fully bare, active undercutting or slumping); occurrence of bank-cut features (binary: present/absent), including overhanging banks, tension cracks, and recent slump scars; vegetation cover was visually estimated on a scale of 0 to 4 at the bank surface (1-m² quadrat); canopy cover from 0.5-5 m high was visually estimated in 10x10 m transects, 2.5, 5, 7.5 and 10 m above the ground (U.S. EPA, 2022); root density and depth estimated by visual inspection of exposed bank face

profiles; and surface cover composition (grass, forb, litter, bare mineral) recorded by line-intercept method at 10-cm intervals along each transect. Soil texture was determined by the hydrometer method as described for the rainfall simulations

Statistical Analysis Bank Surveys

Given that bank survey data included ordinal, binary, and continuous response variables, a suite of approaches was employed. Kruskal–Wallis tests with pairwise Wilcoxon post-hoc comparisons were used to compare bank height, angle, and bare ground across watersheds. Binary bank-cut occurrence was modeled using Generalized Estimating Equations (GEEs) with a logistic link and exchangeable correlation structure to account for spatial autocorrelation among transects within watersheds (Liang & Zeger, 1986), implemented in the *geepack* package in R. Continuous bank condition scores were modeled using Linear Mixed Models (LMMs) with watershed as a fixed factor and transect location as a random effect. Site-specific GAMs with thin-plate splines were fitted for each watershed to characterize longitudinal gradients in bank condition from headwater to outlet, using stream distance (m) as the smoothing variable. All analyses were conducted in R version 4.3.1.

Data Availability

All long-term suspended sediment monitoring data are archived in the KPBS LTER database and are available at <https://lter.konza.ksu.edu>.

Results

Riparian Sediment Delivery During Simulated Rainfall Events

Sediment export and breakthrough curve dynamics

Rainfall simulations produced sediment export spanning more than two orders of magnitude across the ten plots (Table 4.1). Four experiments yielded no bromide signal above the 95% confidence interval of background concentrations and were excluded from quantitative analyses: Experiments 2 (75–100% cover) and 3 (50–75% cover, gentle slope), where intact vegetation drove near-complete infiltration; and Experiments 4 (high-cliff bank cut) and 7 (rocky bank), where vertical infiltration into fractured limestone-derived substrate dominated over lateral transport despite low to moderate cover. Sediment collected downstream during these four trials could not be attributed to overland delivery from the experimental plot in the absence of a detectable conservative tracer.

Among the six experiments with detectable transport, total sediment export ranged from 27 mg at a fully vegetated bank in the bison-grazed area (Exp 9) to 4,678 mg at a knickpoint 100 m below the same area (Exp 8); bromide recovery ranged from 0.005% (Exp 10) to 0.503% (Exp 1) of applied mass (Table 4.1). Sediment export and bromide recovery were not proportional across sites: Exp 8 produced the largest sediment mass yet ranked third in Br recovery (0.091%), and Exp 1 recovered the most bromide but exported less than half the sediment of Exp 8. Bare and trail-disturbed bison sites consistently produced the highest hydrological connectivity (highest % Br recovery), while bank-cut sites generated high sediment mass with lower connectivity, reflecting infiltration-dominated water pathways at steep, exposed positions (Fig. 4.2; Fig. S4.2). Because sediment exports integrated both site-specific erodibility and differences

in plot footprint and storm depth, we normalized export by wetted plot area to yield an areal erodibility index (Sed/m^2 ; mg m^{-2}) for the remaining analyses

Table 4.1: Site characteristics, soil properties, precipitation, bromide mass balance, sediment export, and areal erodibility index (Sed/m²) for ten rainfall simulation experiments at KPBS, 2023–2024. Br⁻ applied at 82.08 mg L⁻¹ (100 g NaBr in 946 L deionized water). BDL = below detection limit (95% CI of background stream Br⁻); Experiments 2, 3, 4, and 7 excluded from quantitative analyses.

Exp	Site Characteristic	Veg Cover (%)	Slope (m/m)	Sand (%)	Silt (%)	Clay (%)	Texture	Precip (cm)	Br Applied (mg)	Br Recovered (mg)	% Br Rec	Sed Exp (mg)	Sed/m ² (mg/m ²)	Br Status
1	Bison grazed bare soil	0-25%	0.150	85.3	10.0	4.7	Loamy Sand	1.77	13526	68.0	0.503	2263	243.10	Detected
2	Bison grazed Vegetated cover	75-100%	0.075	84.3	11.0	4.7	Loamy Sand	2.00	44323	BDL	BDL	nan	-	BDL
3	Ungrazed vegetated	50-75%	0.856	78.7	16.0	5.3	Loamy Sand	1.70	22083	BDL	BDL	nan	-	BDL
4	High cliff ungrazed bank cut	0-25%	2.620	77.3	12.7	10.0	Sandy Loam	0.99	23450	BDL	BDL	nan	-	BDL
5	Bottom of high cliff ungrazed bank	0-25%	0.328	74.0	14.7	11.3	Sandy Loam	3.73	37198	23.1	0.062	1224	100.72	Detected
6	Bison grazed with bison trail	50-75%	0.162	74.7	18.7	6.7	Loamy Sand	5.58	38885	40.1	0.103	988	116.36	Detected
7	Bison grazed with rocky bank	50-75%	0.980	74.0	14.7	11.3	Sandy Loam	3.83	42942	BDL	BDL	nan	-	BDL
8	Bison grazed below fall	50-75%	2.440	72.0	18.7	9.3	Sandy Loam	2.78	40954	37.2	0.091	4678	261.03	Detected
9	Bison grazed above fall	75-100%	0.149	80.0	14.0	6.0	Loamy Sand	2.40	36694	25.9	0.071	27	1.44	Detected
10	Bison grazed with bison trail	25-50%	0.422	77.3	13.3	9.3	Sandy Loam	6.76	73214	3.5	0.005	57	4.34	Detected

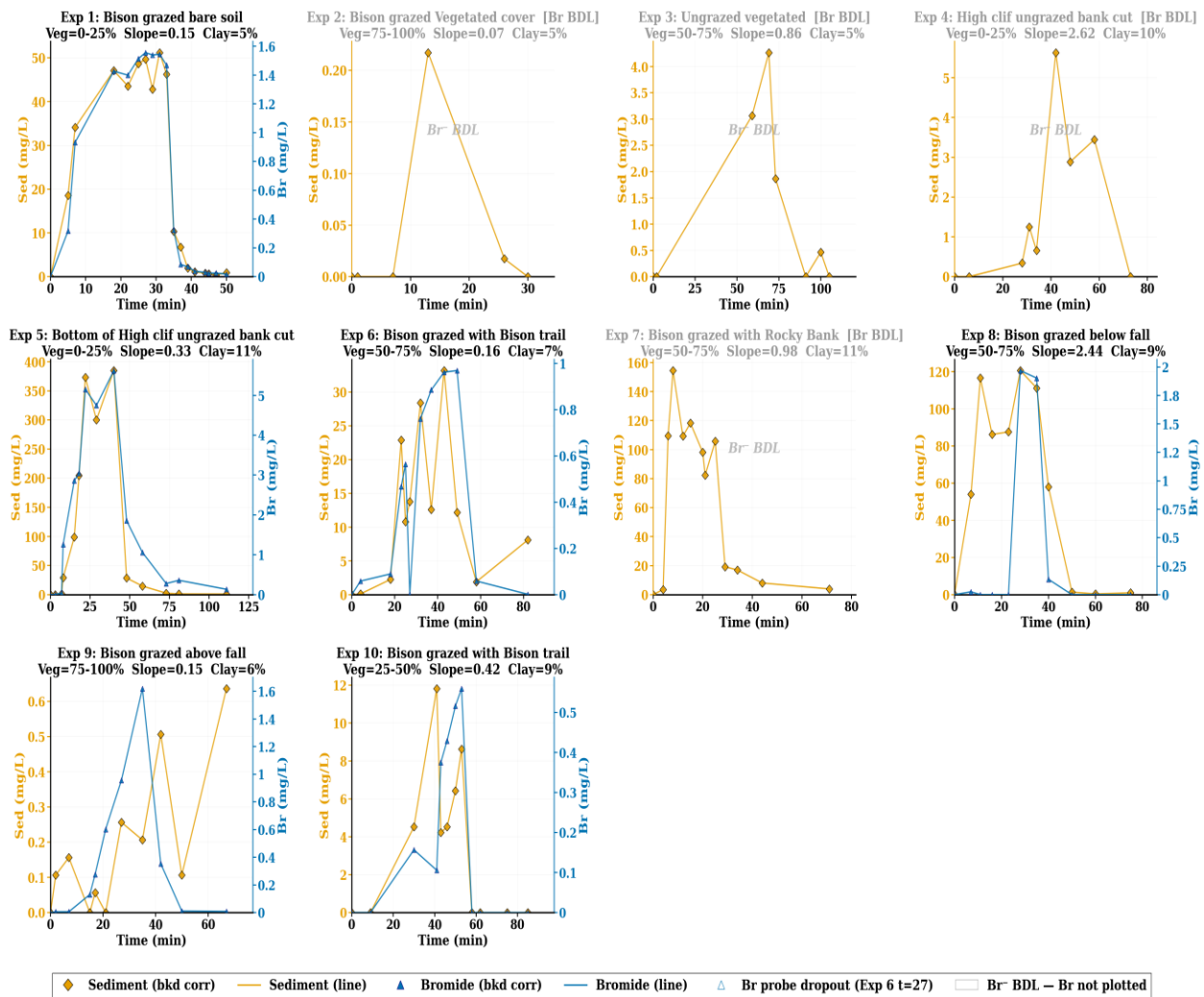


Figure 4.2: Time series of background-corrected sediment (gold) and bromide (blue) concentrations at the downstream sampling point during ten rainfall simulation experiments. BDL panels (Exps 2, 3, 4, 7) display sediment data only; the bromide signal did not exceed the 95% confidence interval of background stream concentrations and is not plotted. Site characteristics (vegetation cover class, slope, clay content) are shown above each panel.

Vegetation threshold on erodibility

Sed/m² varied by approximately 170-fold across the cover gradient and revealed a nonlinear transition between approximately 60% and 70% vegetation cover (Fig. 4.3). The 0–

25% class produced the highest values (Exp 1 = 243.10; Exp 5 = 100.72), and the single 75–100% experiment with detectable transport (Exp 9) produced the lowest (1.44) — an approximately two-order-of-magnitude shift across the threshold. Within the 50–75% class, slope was the dominant modifier: Exp 8 (slope = 2.44) produced $\text{Sed/m}^2 = 261.03$ versus 116.36 at Exp 6 (slope = 0.16). The vegetation $\times$ slope bivariate space defined two zones (Fig. S4.3A): a high-erodibility zone of low cover combined with steep slope (Exps 1, 5, 8), and a low-erodibility zone where either adequate cover or gentle slope was individually sufficient. Clay content was not a useful predictor of areal erodibility (Spearman $r = 0.07$, $p = 0.901$, $n = 6$; Fig. S4.3B). Experiment 10 (bison trail, 25–50% cover, sandy loam) produced an unexpectedly low Sed/m^2 (4.34) despite low cover, likely because its uncompacted sandy loam substrate favored infiltration over surface detachment.

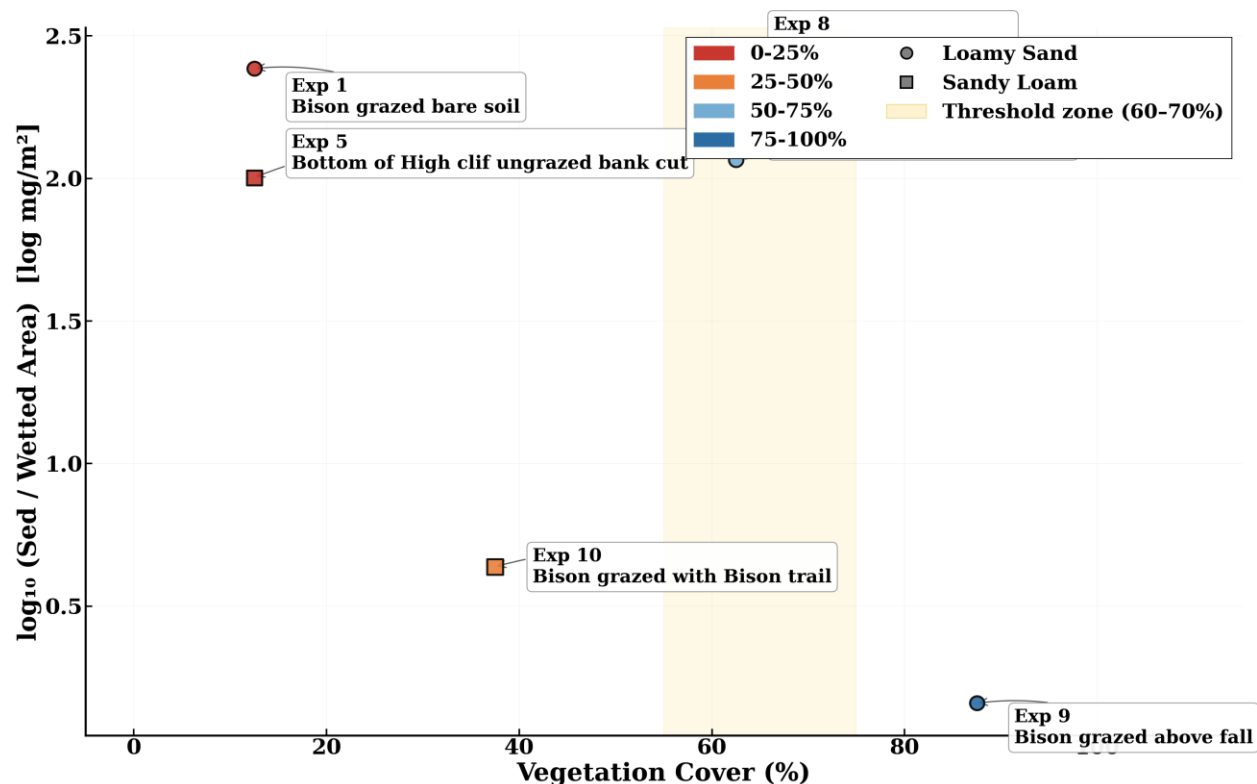


Figure 4.3: Nonlinear vegetation cover threshold on areal erodibility (Sed/m²) across rainfall simulation experiments. $\log_{10}(\text{Sed}/\text{m}^2)$ plotted against vegetation cover (%) for the six experiments with detectable bromide transport. Point color denotes vegetation cover class; symbol shape distinguishes loamy sand (circles) from sandy loam (squares). Yellow band marks the 60–70% transition zone, across which areal erodibility shifts by approximately two orders of magnitude.

Paired site contrasts

Four paired comparisons isolated individual controls while holding other attributes approximately constant (Fig. S4.1). Pair A contrasted vegetation cover within the same bison-grazed terrain at matched slope: bare soil (Exp 1; $\text{Sed}/\text{m}^2 = 243.10$) was 168 $\times$ more erodible per unit area than a fully vegetated surface above a waterfall (Exp 9; $\text{Sed}/\text{m}^2 = 1.44$), isolating cover as a first-order control. Pair B compared bare bison soil (Exp 1) to a partially vegetated bison trail (Exp 10; $\text{Sed}/\text{m}^2 = 4.34$), yielding a 56 $\times$ contrast and a 19-fold difference in Br recovery

(68.0 vs. 3.5 mg), showing that bare soil maximizes both erodibility and connectivity. Pair C contrasted two high-export bank-erosion contexts: an ungrazed cliff base (Exp 5; $\text{Sed}/\text{m}^2 = 100.72$) and a bison-grazed below-fall site (Exp 8; $\text{Sed}/\text{m}^2 = 261.03$), a $2.6\times$ difference indicating that slope-driven transport energy can override the protection afforded by intermediate cover. Pair D, comparing bare bison soil (Exp 1) to a vegetated bison trail at near-identical slope (Exp 6; $\text{Sed}/\text{m}^2 = 116.36$), produced a $2.1\times$ contrast substantially smaller than Pair A, indicating that even partial vegetation recovery on disturbed bison surfaces attenuates a large fraction of the bare-soil erodibility signal. Grouping by geomorphic site type confirmed these patterns (Fig. S4.4): bison bare/trail sites had the highest connectivity, bank-cut/below-fall sites had the highest areal erodibility with lower connectivity, and the fully vegetated bison site (Exp 9) had the lowest Sed/m^2 among detected experiments.

Precipitation recurrence context

Simulated rainfall depths were compared against Kansas precipitation-frequency curves for 30–60 minute storm durations to evaluate the climatological representativeness of the experimental conditions (Fig. S4.5). Six experiments (1, 2, 3, 4, 8, 9) fell within or below the sub-annual to ~1-year return period, four (5, 6, 7, 10) corresponded to ~2- to 30-year return periods, and none exceeded the 100-year event. Notably, the two BDL high-cover experiments (2 and 3) received sub-annual depths yet produced no measurable transport, while Exp 1 generated the highest areal erodibility under a comparable sub-annual event — demonstrating that degraded riparian surfaces deliver substantial sediment during storms occurring multiple times per year.

Long-term Suspended Sediment Dynamics (2009–2023)

GAM model performance

Generalized additive models fitted to monthly log-transformed sediment concentrations explained 43–46% of total variance across the four response variables (TSS: $R^2 = 0.459$, adj. $R^2 = 0.408$; VSS: $R^2 = 0.428$, adj. $R^2 = 0.374$; ISS: $R^2 = 0.441$, adj. $R^2 = 0.389$; VSS/ISS: $R^2 = 0.435$, adj. $R^2 = 0.394$; $n = 235$ site-months for TSS, VSS, and ISS; $n = 234$ for VSS/ISS).

Residual-versus-fitted plots showed no systematic heteroscedasticity across the range of fitted values, though smoothed residual trends indicated slight under-prediction at extreme values, particularly for ISS (Fig. S4.6). Predicted-versus-observed plots confirmed reasonable calibration for all four responses, with Pearson correlation coefficients of 0.66–0.68 and root mean squared errors of 0.49–1.08 on the log scale (Fig. S4.6), consistent with the expected performance of monthly sediment models at watersheds subject to event-driven variability.

Differential driver control of organic versus inorganic sediment

The GAM driver importance heatmap (Fig. 4.4), which displays the signed $\log_{10}(\text{p-value})$ for each predictor across the four response variables, revealed a coherent pattern of shared and divergent environmental controls on suspended sediment fractions. Three smooth terms (discharge, precipitation, days since burn) and one linear term (watershed area) were evaluated.

Log-transformed discharge exerted significant negative effects on TSS ($p < 0.001$), VSS ($p < 0.001$), and ISS ($p = 0.014$), with predicted concentrations declining nonlinearly across roughly four orders of magnitude in flow (Fig. 4.5a–c). The magnitude of the discharge effect was substantially greater for VSS than for ISS, indicating that dilution disproportionately affects organic material. Conditioning the discharge response on precipitation revealed that high-precipitation periods (>75th percentile) sustained higher predicted concentrations than low-

precipitation periods (<25th percentile) at every flow, with the offset between the two precipitation conditions evident across the discharge range and persisting at high flows (Fig. 4.5a–c). The discharge effect on the VSS/ISS ratio was not significant ($p = 0.19$; Fig. 4.4), meaning that while high flows reduce both fractions, they do not systematically shift the compositional balance between them.

Antecedent precipitation (30-day cumulative) had significant positive effects on TSS ($p = 0.001$), VSS ($p < 0.001$), and ISS ($p = 0.015$), but was not significant for VSS/ISS ($p = 0.259$). The positive direction of the precipitation effect, opposite to the negative direction of the discharge effect, indicates that wetter antecedent conditions promote sediment mobilization through increased soil saturation and overland flow connectivity, a process temporally and mechanistically distinct from the dilution that occurs at high instantaneous discharge.



Figure 4.4: Divergent environmental controls on organic versus inorganic suspended sediment fractions revealed by generalized additive model (GAM) analysis. Heatmap of signed $\log_{10}(p\text{-value})$ for each predictor (rows: log discharge, precipitation, days since burn, watershed area) across four response variables (columns: TSS, VSS, ISS, VSS/ISS ratio). Blue shading indicates significant positive effects; red shading indicates significant negative effects; color intensity is proportional to statistical significance. Upward triangles denote positive effects; downward triangles denote negative effects.

Days since last prescribed burn was significant for all four responses ($p < 0.001$ each), but with contrasting effect directions that constitute the most distinctive finding of the GAM analysis. Days since burn had positive effects on TSS and ISS, meaning that inorganic sediment concentrations increased with greater time elapsed since fire. Conversely, days since burn had negative effects on VSS and the VSS/ISS ratio, meaning that organic sediment concentrations and the organic-to-inorganic compositional balance both declined as the post-fire interval lengthened (Fig. 4.4). Predicted concentration curves for days since burn showed that ISS rose

sharply within approximately 200 days post-fire, declined modestly through ~600 days, and then climbed to an elevated plateau extending beyond 1,000 days (Fig. 4.5k). VSS exhibited a similar early post-fire peak but declined gradually thereafter (Fig. 4.5j), and the VSS/ISS ratio showed a corresponding monotonic decline that became most pronounced after ~700 days (Fig. 4.5l), providing direct visual support for the SEM finding of days since burn as the strongest single predictor of sediment composition.

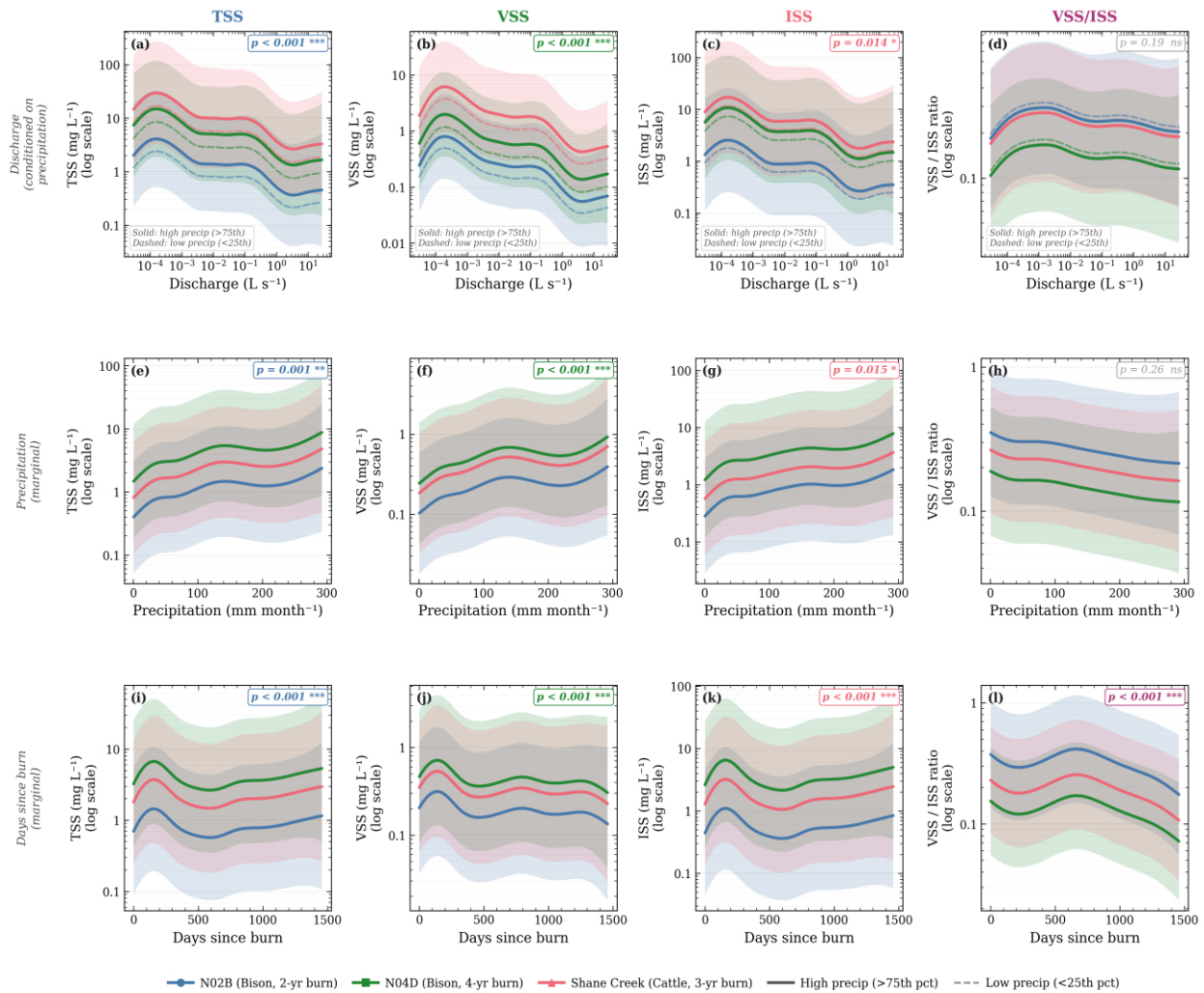


Figure 4.5: Predicted concentrations from generalized additive models showing the effects of log-transformed discharge (a–d), 30-day cumulative precipitation (e–h), and days since last prescribed burn (i–l) on TSS (blue), VSS (green), ISS (red), and the VSS/ISS ratio (purple) by watershed. In panels (a–d), solid lines and shaded bands represent high-precipitation conditions (>75th percentile) and dashed lines represent low-precipitation conditions (<25th percentile), illustrating the precipitation $\times$ discharge interaction. Shaded bands represent 95% confidence intervals throughout. Watersheds: N02B (bison, 2-year burn; blue), N04D (bison, 4-year burn; green), Shane Creek (cattle, 3-year burn; pink). The significance of each smooth term is indicated in each panel.

Watershed area, included as a linear predictor to account for among-site differences in drainage area and associated geomorphic characteristics, was significant only for VSS ($p = 0.015$) and non-significant for TSS ($p = 0.218$), ISS ($p = 0.414$), and VSS/ISS ($p = 0.224$).

Watershed-scale temporal patterns

Predicted monthly concentrations from the GAMs (Fig. 4.5) and observed log-transformed values over the 15-year record (2009–2023) revealed persistent divergence among the three monitoring watersheds consistent with their management regimes. N04D (bison, quadrennial fire) maintained the highest TSS and ISS concentrations across the full record, while N02B (bison, biennial fire) sustained the lowest. Shane Creek (cattle, triennial fire) was intermediate for TSS and ISS but periodically exceeded both bison watersheds in VSS. Its temporal trajectory was distinctive: low sediment during the pre-cattle baseline (2009–2010), an increase after cattle introduction in 2011, and a subsequent multi-year decline.

The VSS/ISS ratio provided the clearest separation among management regimes. N02B sustained the highest VSS/ISS values (most organic-dominated sediment composition), N04D the lowest (most inorganic-dominated), and Shane Creek was intermediate. Rolling median lines through the time series revealed multi-year oscillations in both ISS concentrations and the VSS/ISS ratio that corresponded to the timing of prescribed burn events, with ISS concentrations rising in the one to two years following fire and declining during subsequent vegetative recovery.

Causal pathway modeling

A structural equation model was fitted to evaluate the directional structure of pathways linking climate drivers, watershed properties, and fire management to the two sediment fractions and their ratio (Fig. 4.6; Table 4.1). Discharge was well predicted by watershed area ($\beta = 0.823$, $p < 0.001$), precipitation ($\beta = 0.162$, $p < 0.001$), and days since burn ($\beta = -0.027$, $p < 0.001$),

yielding $R^2 = 0.738$. Path coefficients from discharge to the sediment endpoints were consistent with the GAM findings: the negative effect of discharge was stronger on VSS ($\beta = -0.788$) than on ISS ($\beta = -0.542$), and days since burn had a nearly twice as large positive direct effect on ISS ($\beta = 0.296$) as on VSS ($\beta = 0.154$). The strongest single direct path to the VSS/ISS ratio was from days since burn ($\beta = -0.405$). Residual covariance between VSS and ISS was high ($r = 0.88$, $p < 0.001$), indicating substantial shared event-scale variation not captured by the modeled predictors.

Model fit indices, however, did not meet conventional thresholds for adequate structural representation of the covariance matrix (CFI = 0.718, TLI = -0.127, RMSEA = 1.335, $\chi^2 = 2497.53$, $p < 0.001$). This poor fit is consistent with the curvilinear relationships identified by the GAMs, although missing pathways or heteroscedasticity could also contribute; we therefore use the SEM as a directional decomposition rather than as a confirmatory test. The SEM results are therefore interpreted here as corroborating the direction and relative magnitude of effects identified by the nonparametric GAM analysis, rather than as an independent causal model.

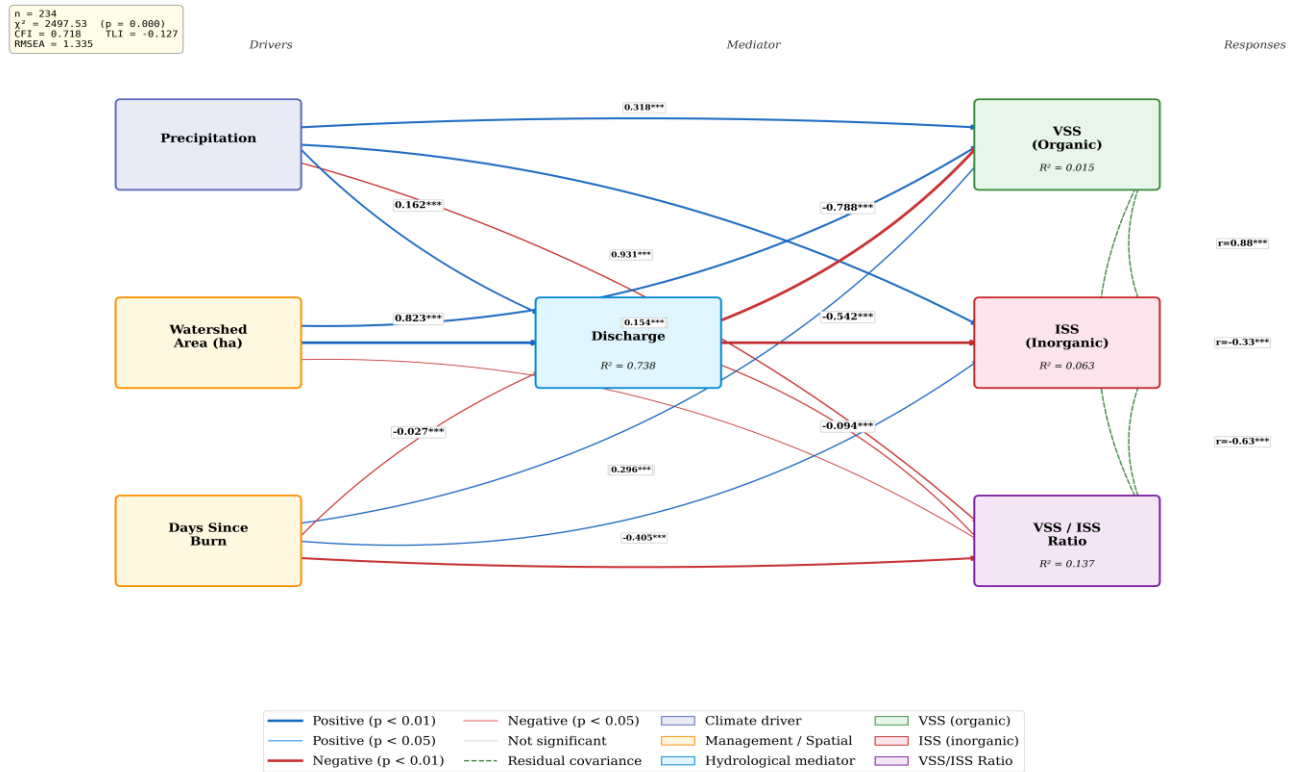


Figure 4.6: Structural equation model (SEM) depicting the pathway structure linking climate drivers, watershed properties, and fire management to organic (VSS) and inorganic (ISS) suspended sediment fractions and their compositional ratio (VSS/ISS). Standardized path coefficients (β) are shown along each arrow, with line thickness proportional to effect magnitude. Blue arrows indicate significant positive paths; red arrows indicate significant negative paths. Dashed green arcs represent residual covariances between response variables. All paths significant at $p < 0.001$.

Streambank Geomorphic Surveys

Streambank geomorphic attributes varied significantly among watersheds under different fire and grazing management (Fig. 4.7). Bank slope differed among the four surveyed watersheds (Kruskal-Wallis $H = 11.2$, $p = 0.011$), with N04D exhibiting the steepest median banks and C3SB the most variable. Weighted soil texture score also differed significantly ($H = 14.5$, $p = 0.002$), with N02B having the finest-textured soils on average. Average bare soil scores did not

differ among watersheds ($H = 3.3$, $p = 0.341$), although C3SB showed the greatest range, with occasional high bare soil values absent from other sites.

Bank cut frequency, calculated as the mean proportion of surveyed bank positions exhibiting active erosion features across the two survey years, was 50% at both N04D and K2B, 30% at N02B, and 25% at C3SB (Fig. 4.7d). These differences were not statistically significant ($\chi^2 = 4.5$, $p = 0.216$). In contrast, grazer trail presence varied strongly among watersheds ($\chi^2 = 34.4$, $p < 0.001$; Fig. 4.7e): trails were recorded at 80% of N04D positions, 75% of N02B positions, 45% of C3SB positions, and 0% at ungrazed K2B. Canopy cover also differed significantly ($\chi^2 = 13.9$, $p = 0.003$; Fig. 4.7f), with C3SB exhibiting the highest frequency of canopy-covered bank positions (70%), followed by N02B (50%), K2B (32%), and N04D (15%).

Generalized estimating equation models with a logistic link identified woody cover presence as the strongest statistically significant predictor of bank cut occurrence (OR = 4.19, $p = 0.020$; Fig S. 4.7), indicating that banks with woody vegetation were more than four times as likely to exhibit erosion features as those without. Among watershed-level contrasts relative to the K2B reference, C3SB had significantly lower bank cut probability (OR = 0.14, $p = 0.005$) and N04D marginally lower (OR = 0.30, $p = 0.026$). N02B showed a similar but non-significant trend (OR = 0.37, $p = 0.088$). Bank slope had the largest point estimate among continuous predictors (OR = 17.00), but its wide confidence interval precluded statistical significance ($p = 0.160$). Neither grazer trail presence (OR = 2.53, $p = 0.108$), soil texture (OR = 1.10, $p = 0.507$), survey year (OR = 1.08, $p = 0.897$), nor longitudinal position (OR = 0.69, $p = 0.273$) was significant.

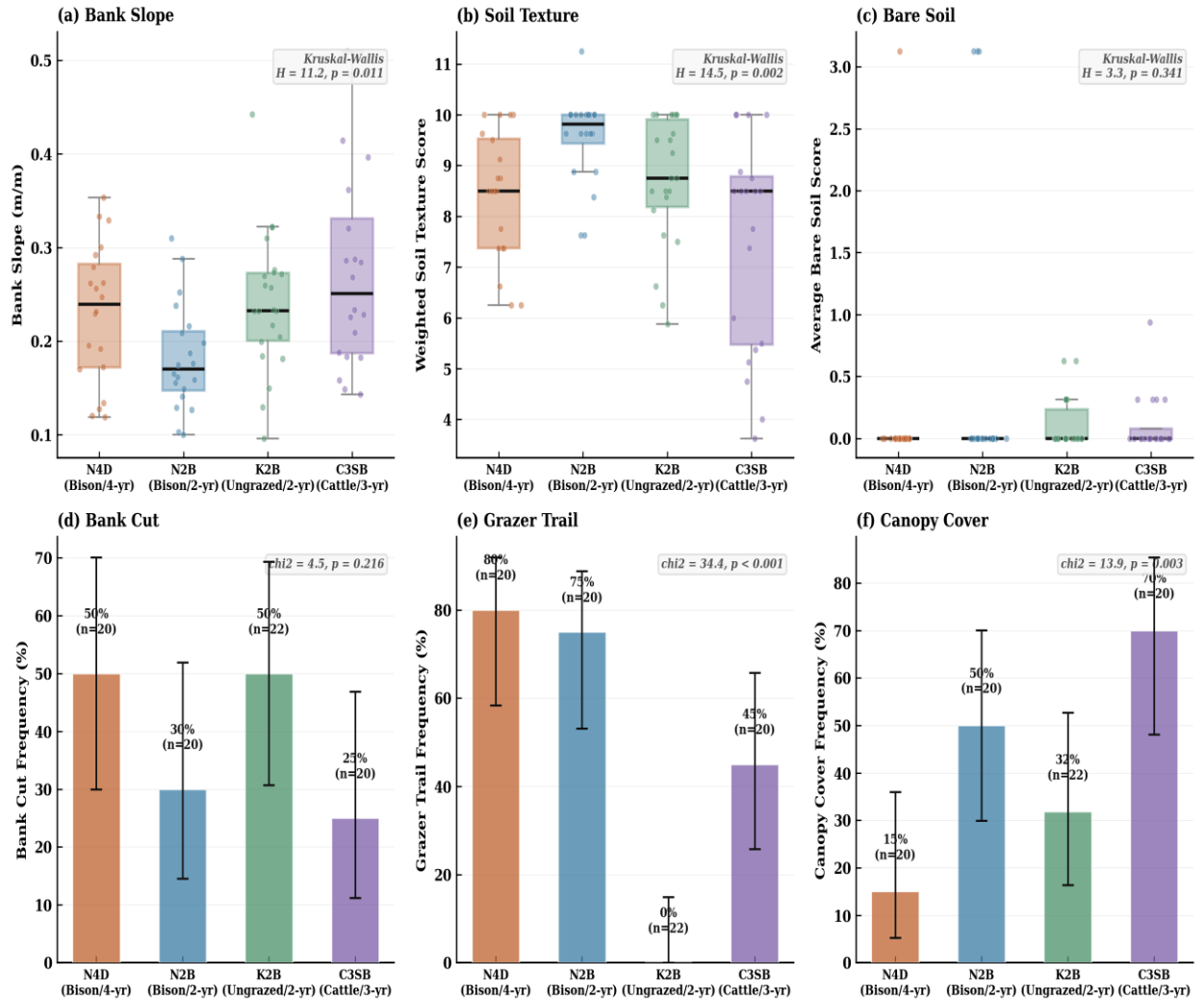


Figure 4.7: Streambank geomorphic attributes across four KPBS watersheds under contrasting fire and grazing management. Top row: box plots of (a) bank slope (m/m), (b) weighted soil texture score, and (c) average bare soil score. Kruskal-Wallis test statistics and p-values are shown. Bottom row: bar plots of (d) bank cut frequency (%), (e) grazer trail frequency (%), and (f) canopy cover frequency (%), with chi-square test statistics and p-values. Sample sizes (n) are indicated above each bar. Error bars represent 95% confidence intervals. Watersheds: N04D (bison, 4-year burn), N02B (bison, 2-year burn), K2B (ungrazed, 2-year burn), C3SB (cattle, 3-year burn).

A Spearman rank correlation matrix among the measured bank attributes revealed uniformly weak pairwise associations ($|\rho| \leq 0.19$; Fig. S4.8). Bank slope was weakly positively correlated with bank cut occurrence ($\rho = 0.19$) and bare soil ($\rho = 0.18$) and negatively correlated

with canopy cover ($\rho = -0.15$). The general weakness of these correlations indicates that bank erosion susceptibility is influenced by multiple largely independent factors, none of which dominates across the watershed network.

Taken together, N04D combined the highest grazer trail frequency (80%), the lowest canopy cover (15%), and the highest bank cut frequency (50%) of any watershed, a profile consistent with this watershed sustaining the highest ISS concentrations and lowest VSS/ISS ratios in the 15-year monitoring record. Conversely, C3SB had the highest canopy cover (70%) and the lowest bank cut frequency (25%) despite the presence of cattle grazing, a finding that bears on prediction (iii) and is considered further in the Discussion.

Discussion

Independent Regulation of Organic and Inorganic Sediment by Discharge and Fire Management

A growing body of work calls for composition-aware approaches to suspended sediment in streams, arguing that the ecological consequences of mineral versus organic particles differ fundamentally for benthic habitat quality, nutrient dynamics, and stream metabolism (Bilotta & Brazier, 2008; Collins et al., 2017; Gellis et al., 2024). However, empirical demonstrations that the two fractions respond to different environmental drivers within a single system remain limited, particularly in grasslands. Our results provide this demonstration with three independent lines of evidence.

The GAM analysis of sediment export over 14 years revealed that discharge exerted a strong negative effect on VSS but a substantially weaker, though still significant, negative effect on ISS, while days since burn acted in opposing directions on the two fractions, positive for ISS,

negative for VSS. This pattern of differential control is consistent with the conceptual model of Dodds & Whiles (2004), in which organic material in prairie streams derives primarily from near-channel sources (algal biomass, riparian litter, resuspended particulate organic matter) that are mobilized at low to moderate flows and progressively diluted as discharge increases. Inorganic sediment, by contrast, originates from mineral soil detachment and bank erosion processes whose supply to the channel is less dependent on instantaneous flow magnitude and more dependent on antecedent surface condition set by fire management. The rainfall simulation experiments reinforce this interpretation at the process scale: the two high-cover BDL sites (Exps 2 and 3) produced no measurable overland transport, indicating complete attenuation through intact root mats, while bare and trail-disturbed surfaces generated concentrated sheet flow bearing mineral sediment even during sub-annual storm events (Fig. S4.5). Two additional BDL experiments at sparsely vegetated bank positions (Exps 4 and 7) showed that vertical infiltration into fractured substrates can also disconnect riparian surfaces from the channel, indicating that hydrological connectivity is jointly controlled by surface cover and substrate permeability.

The finding that precipitation and discharge act in opposite directions on sediment concentrations, positive and negative, respectively, within the same model has not, to our knowledge, been demonstrated previously in grassland headwaters. This divergence reveals a two-phase storm process in which antecedent rainfall primes sediment mobilization through soil saturation and enhanced overland connectivity, followed by dilution as sustained high flows expand the water volume relative to the entrained material. This temporal sequence has been recognized conceptually in the broader sediment literature (Vercruysse et al., 2017) but is typically inferred from event-scale concentration–discharge hysteresis loops rather than from the contrasting signs of two distinct predictors in a multi-year GAM, as demonstrated here..

The significance of the opposing days-since-burn effects on VSS and ISS extends beyond a statistical pattern to a management-relevant mechanism. As the post-fire interval lengthens, two concurrent processes shift the sediment regime toward inorganic dominance: mineral soil exposure increases as litter and herbaceous regrowth diminish, and woody vegetation progressively colonizes riparian zones, altering root architecture in ways that reduce surface binding capacity while potentially increasing bank instability (discussed further below). The net result is a monotonic decline in the VSS/ISS ratio with increasing time since fire, a trend that was the single strongest predictive relationship in the GAM for sediment composition ($p < 0.001$) and was confirmed directionally by the SEM ($\beta = -0.405$). These converging results from nonparametric and parametric approaches establish days since burn as the dominant lever through which management shapes not only total sediment load but the qualitative character of the suspended material entering these streams, with direct implications for benthic habitat: inorganic sediment smothers interstitial spaces and reduces macroinvertebrate diversity (Owens et al., 2024; Zweig & Rabeni, 2001), while organic sediment can subsidize heterotrophic food webs when delivered in moderate quantities (Webster & Meyer, 1997).

Nonlinear Erodibility Thresholds and the Multi-Gradient Structure of Riparian Function

Threshold responses of erosion to vegetation cover have long been predicted by hillslope erosion theory (Gyssels et al., 2005; Prosser et al., 1995) and documented in rangeland settings (Pierson et al., 2011; Williams et al., 2020), but to our knowledge, the present study provides the first quantification of such a threshold in tallgrass prairie riparian zones using a process-based metric that separates erodibility from connectivity. The areal erodibility index (Sed/m^2) enabled

detection of this threshold at approximately 60–70% cover with greater confidence than raw sediment export alone, which varied by more than 170-fold across experiments and confounded site susceptibility with differences in plot size and storm depth.

The complete absence of detectable bromide transport at the two highest-cover sites (Experiments 2 and 3) deserves particular emphasis. These experiments demonstrate that fully vegetated tallgrass prairie riparian zones can completely attenuate overland flow and associated particle transport during rainfall events well within the normal range of regional storm variability. This capacity has been documented for a single trial at Konza by Dodds et al. (2022), and our results confirm and extend that finding by extending the protection downward into the upper end of the 50–75% cover class on gentle slopes, substantially strengthening the inference that vegetation cover above ~70% confers near-complete erosion protection in this landscape.

The strong modifying role of slope within the vegetation transition zone, a 2.6-fold areal erodibility contrast between Experiments 5 and 8 in the high-export bank-erosion comparison, illustrates the multi-gradient structure of riparian function formalized by Dodds et al. (2025). Importantly, the paired comparisons reveal that the relative importance of different gradients shifts with the condition of the dominant one. When cover is very high, lateral transport ceases entirely (BDL at Exps 2 and 3). When cover is intermediate, slope and geomorphic position become decisive, and energy-concentrating features such as knickpoints can override the protective effect of moderate vegetation. When cover is very low, additional differences in cover and substrate permeability produce large changes in areal export, as shown by the 168-fold contrast between bare and fully vegetated bison-grazed surfaces at matched slope (Pair A) and the 56-fold contrast between bare soil and a partially vegetated bison trail (Pair B). This hierarchy implies that management responses should be spatially targeted: steep, intermediate-

cover positions represent the zone of greatest vulnerability and the zone where vegetation restoration would yield the largest marginal reduction in sediment delivery, consistent with precision conservation frameworks (Herrero et al., 2024; Tomer et al., 2013).

The spatial heterogeneity of bison-created disturbance features adds a dimension not captured by watershed-average vegetation metrics. The 168-fold areal erodibility contrast in Pair A, from bare soil to a fully vegetated surface at matched slope in the same watershed type, demonstrates that bison generate extreme point sources of both erodibility and connectivity at bare patches, while adjacent trail and vegetated surfaces remain orders of magnitude less erodible. This mosaic is characteristic of the pyric herbivory interaction at Konza, where bison selectively graze recently burned areas and create patchy disturbance (Fuhlendorf & Engle, 2004; Knapp, et al., 1999; Raynor et al., 2021). Our results suggest that it is the spatial configuration of these disturbance features relative to hydrological flow paths, not their aggregate areal extent, that governs watershed-scale sediment delivery, a conclusion with implications for how grazing intensity metrics are defined and monitored.

Fire Frequency, Woody Encroachment, and Bank Instability

The emergence of fire frequency as a strong determinant of sediment composition across all three datasets modifies the prediction, which anticipated that grazer identity would be the stronger control on ISS. Instead, the days-since-burn variable dominated both the GAM and SEM for the VSS/ISS ratio, and the among-watershed divergence in sediment composition tracked fire return interval more closely than grazer type. This finding extends the growing recognition that prescribed fire frequency structures ecosystem processes at KPBS beyond the direct effects of combustion, shaping vegetation trajectories, soil properties, and hydrological function over decadal timescales (Collins et al., 2022; Raihan et al., 2025).

The mechanistic link between fire frequency and inorganic sediment appears to operate through the trajectory of woody encroachment in riparian zones. N04D, which had the longest fire return interval, exhibited the lowest canopy cover at surveyed bank positions (15% bank-position mean, with substantial within-watershed variation) yet sustained the highest bank cut frequency (50%) and the highest ISS concentrations in the long-term record. This apparent paradox, low woody canopy coinciding with high bank erosion, resolves when the successional context is considered. Quadrennial burning allows sufficient fire-free intervals for initial woody establishment, but the resulting woody plants do not develop the dense, fibrous root mats characteristic of grass-dominated riparian zones. Woody root architecture provides deep anchoring but less surface-level soil binding than the diffuse, fine root networks of tallgrass prairie grasses (Abernethy & Rutherford, 2001; Simon & Collison, 2002). Woody stems may create turbulent flow paths during overbank events that accelerate bank scour (Briggs et al., 2005b; Keen et al., 2023). The GEE finding that woody cover was the strongest significant predictor of bank erosion ($OR = 4.19$, $p = 0.020$; Fig. 4.7) is consistent with this mechanism and with broader evidence that woody encroachment at KPBS alters both streamflow (Raihan et al., 2025; Sadayappan et al., 2023) and riparian structure (Dodds et al., 2023; Heidari et al., 2024).

The 15-year persistence of watershed-specific sediment regimes, N02B consistently organic-dominated, N04D consistently inorganic-dominated, despite shared climate, suggests a reinforcing sequence in which longer fire-return intervals permit woody establishment and bank destabilization that sustain elevated ISS, while shorter intervals maintain grass cover and organic-rich sediment (Fuhlendorf et al., 2009; McGranahan et al., 2024). Chapter 3 documented analogous multi-year persistence in stream nutrient dynamics at these same watersheds, with land use legacies in water chemistry persisting for 4–5 years following management transitions.

The timescale of these nutrient legacies is comparable to the multi-year oscillations in ISS and VSS/ISS observed here, suggesting that prescribed-fire periodicity imposes a multi-year forcing on both systems.

The finding that C3SB (cattle, triennial fire) exhibited the lowest bank cut frequency (25%) despite cattle being the dominant grazer challenges prediction (iii) and illustrates the importance of local riparian condition in modifying expected grazer effects. C3SB had by far the highest canopy cover (70%) of any surveyed watershed. Although woody root networks generally provide less fine-scale soil binding than prairie-grass root mats, at C3SB, the combination of continuous canopy and above-ground flow resistance appears sufficient to compensate for cattle trampling, a context-dependent exception to the general grass-superior pattern. This suggests that the interaction between vegetation structure and grazer type is more predictive of bank stability than grazer identity alone, an important nuance for management prescriptions that tend to categorize grazer effects based on species identity rather than site-specific riparian condition.

Limitations and Future Directions

Several limitations should be considered when interpreting these results. The rainfall simulations capture single-event responses and cannot represent cumulative effects of sequential storms, seasonal soil moisture variation, or multi-year vegetation recovery following disturbance. Under the corrected bromide mass balance ($82.08 \text{ mg L}^{-1} \text{ Br}^-$; 100 g NaBr in 946 L), Experiments 2, 3, 4, and 7 fell below the 95% confidence interval of background concentrations and were excluded. The resulting sample of six detected experiments constrains the precision with which the cover threshold can be located, and additional simulations spanning the 60–80%

range would refine its position. Independent validation of Sed/m² against erosion pins or radionuclide tracers would enhance transferability beyond KPBS.

The GAMs explain 43–46% of the variance in monthly sediment concentrations, a level comparable to other studies of event-driven sediment systems at monthly resolution (Horowitz, 2009; Vercruysse et al., 2017). The remaining variance likely reflects sub-monthly storm-event dynamics that are smoothed by temporal aggregation. The SEM did not achieve adequate fit (CFI = 0.718, RMSEA = 1.335), confirming that linear path models cannot represent the threshold and curvilinear relationships identified by GAMs in this system. Future work should explore piecewise or nonlinear structural equation approaches. Most critically, with only three long-term monitoring watersheds, management effects remain partially confounded with site identity, and the management contrasts, while long-standing and well-characterized, are unreplicated. Incorporation of additional KPBS watersheds with different fire grazing factorial combinations, deployment of high-frequency turbidity sensors to capture event-scale dynamics (Sadayappan et al., 2023), and repeated rainfall simulations across multiple burn cycles and seasonal conditions would strengthen causal inference. Independent validation of the Sed and Br measurement against erosion pin measurements or radionuclide tracer approaches would enhance its transferability beyond the KPBS ecosystems.

Conclusions

This study demonstrates that fire frequency and grazer identity control not only the quantity but the composition of suspended sediment entering tallgrass prairie streams, operating through two distinct and independently regulated delivery pathways. By integrating plot-scale rainfall simulations, 15 years of watershed monitoring, and streambank geomorphic surveys, we identified an organic pathway (VSS) mediated by discharge and sensitive to flow dilution, and an inorganic pathway (ISS) governed by fire management and mineral soil exposure. These pathways respond to different environmental drivers, a finding that suggests modification of the prevailing approach of managing grassland stream sediment as a single undifferentiated metric could allow for more fine-tuning of management results.

The areal erodibility index (Sed/m^2) introduced here provides a process-based, area-normalized metric for comparing riparian sediment delivery across plots differing in size and storm depth. Rainfall simulations revealed a vegetation cover threshold between approximately 60% and 70%, below which erodibility increased nonlinearly by approximately two orders of magnitude, modified by slope and geomorphic position consistent with the multi-gradient framework of Dodds et al. (2025). Bare bison-grazed soil and partially vegetated trails were 56- to 168-fold more erodible per unit area than fully vegetated surfaces in matched comparisons, identifying grazer-created disturbance features as disproportionate contributors to the riparian sediment budget.

At the watershed scale, GAMs and structural equation models confirmed that organic and inorganic sediment respond to different controls: discharge drives organic sediment through dilution, while fire management drives inorganic sediment through mineral soil exposure and

bank instability. The VSS/ISS ratio, more sensitive to management differences than any individual fraction, was most strongly predicted by days since burn and site identity, establishing fire frequency as a primary determinant of suspended sediment composition over decadal timescales. Streambank surveys corroborated the inorganic pathway, linking bank cut probability to woody cover presence and management regime, with the dramatic increase in bank erosion at N04D (5% to 45% between survey years) providing direct geomorphic evidence for the fire–encroachment–instability mechanism.

Three practical implications emerge from this work. First, sediment monitoring programs in grassland streams should incorporate VSS/ISS fractionation to distinguish between management scenarios that produce identical TSS concentrations but fundamentally different ecological outcomes for benthic habitat, nutrient cycling, and stream metabolism. Second, prescribed fire frequency was the strongest management lever in our analysis for controlling sediment composition in bison-grazed tallgrass prairie at KPBS: biennial burning maintained the most stable, organic-dominated sediment regime with the least bank erosion across all three datasets, while quadrennial burning produced the most inorganic-dominated and geomorphically unstable conditions. Third, riparian condition assessment should prioritize steep, low-cover positions where vegetation loss below the 50–75% threshold produces disproportionate increases in mineral sediment delivery, a spatial targeting approach consistent with emerging precision conservation frameworks.

The convergence of experimental, observational, and geomorphic evidence on a coherent dual-pathway framework provides a mechanistic foundation for predicting how grassland stream sediment regimes will respond to changing fire, grazing, and climate management across the

tallgrass prairie and potentially across the broader grassland biome where fire–grazing interactions structure ecosystem function.

Chapter 5 - Conclusions

Synthesis of Findings

This dissertation examined three dimensions of stream ecosystem function (hydrology, nutrient biogeochemistry, and sediment dynamics) across temporal scales ranging from individual storm events to four decades of continuous monitoring in tallgrass prairie headwater streams at KPBS. The overarching finding is that grassland streams are not static features of the landscape but are undergoing directional, interacting changes driven by the simultaneous pressures of woody plant encroachment, climate warming, contrasting fire and grazing management, and the persistent legacy of historical agriculture. No single driver or response variable captures the scope of this transformation; understanding it requires the integrated, long-term investigation that the KPBS LTER infrastructure uniquely enables.

Hydrological transformation driven by woody encroachment

Chapter 2 established that the Kings Creek watershed has lost a substantial portion of its streamflow over the past four decades. Daily discharge declined by over half, and the number of annual flow days declined 55% between 1982 and 2020, changes unaccompanied by any decrease in annual precipitation. We attribute this to a progressive vegetation shift: riparian woody cover expanded two- to sevenfold depending on fire frequency, amplifying actual evapotranspiration and intercepting water that would otherwise have reached streams through subsurface pathways. Structural equation modeling confirmed evapotranspiration as the mechanistic link between woody cover and streamflow loss. The effect was most pronounced during the July to September period, when vegetation water demand peaks and streams are most vulnerable to drying, creating a seasonal water deficit that propagates into fall stream drying.

Temperature increased approximately 1.2°C over the study period, compounding the vegetation-driven losses by raising evaporative demand. Analysis of excess precipitation and energy revealed temporal shifts in the watershed water balance consistent with afforestation, even when restricted to wet years.

These results carry two important implications. First, they demonstrate that a relatively pristine grassland watershed can experience dramatic hydrological change driven by vegetation transition alone. This challenges the assumption that undeveloped watersheds are hydrologically stationary and underscores the need to account for vegetation dynamics in water resource planning for grassland regions. Second, the finding that fire frequency modulates the rate of woody encroachment and thus the pace of streamflow decline establishes prescribed burning as the most direct management lever for maintaining streamflow in tallgrass prairie. However, even annual burning did not prevent encroachment entirely.

Nutrient hierarchy: agricultural legacy, grazing, and fire

Chapter 3 demonstrated that stream nutrient concentrations at KPBS respond to a hierarchy of anthropogenic effects, each differing in magnitude, mechanism, and temporal persistence. Agricultural legacy stood at the top: stream nitrate concentrations below sequentially restored croplands declined by 75% over 25 years, but groundwater nitrate concentrations persisted near 950 $\mu\text{g N L}^{-1}$ more than two decades after fertilization ceased. The 4–5 year lag reflects both nitrogen transit through the soil profile and the progressive re-establishment of plant–microbial N demand as soil carbon accumulates (Baer & Blair, 2008; Rosenzweig et al., 2016), while the sustained elevation in deeper groundwater points to legacy pools with residence times extending well beyond the monitoring record. These findings are directly relevant to the millions of hectares across the Great Plains and Midwest where formerly cultivated lands are

being restored to grassland under conservation programs: surface water quality improvements will come, but full groundwater recovery and the cessation of legacy nutrient contributions to streams will require patience measured in decades, not years.

Bison grazing produced the second-strongest effect, roughly doubling stream nitrate relative to ungrazed watersheds over the 25-year monitoring record. This effect is substantially larger than the 35% increase reported over a 4-year period by Kemp and Dodds (2001), illustrating how gradual, cumulative changes in nutrient cycling can be missed by investigations spanning only a few years. The mechanism likely involves sustained nitrogen enrichment of soils through urine and fecal deposition, combined with the landscape-scale redistribution of nutrients driven by bison movement patterns. Cattle presence was associated with a 33% increase in nitrate and elevated total phosphorus, driven primarily by direct physical disturbance of riparian zones rather than the landscape-scale nutrient redistribution characteristic of bison.

Fire frequency had minimal direct effects on stream nutrient concentrations across the burn frequency gradient examined. This finding is consistent with earlier work at KPBS, suggesting that the nutrient pulses generated by individual burns are transient and rapidly recaptured by regrowing vegetation (Dodds, Blair, et al., 1996). However, fire interacted with grazing to shape long-term nutrient trajectories. This pattern is only visible over multi-decadal timescales and reinforces the need to evaluate fire and grazing jointly rather than as independent management factors.

Dual-pathway sediment framework

Chapter 4 resolved a fundamental but previously unrecognized distinction in how suspended sediment enters and moves through tallgrass prairie streams. Organic sediment (VSS) and inorganic sediment (ISS) respond to different environmental controls, and their relative

abundance encodes information about sediment source and delivery mechanism that is lost when only total suspended solids are measured.

Discharge governed organic sediment through dilution. VSS concentrations declined at high flows as in-channel and near-channel organic matter sources were diluted by increasing water volume, a pattern consistent with the limited, locally sourced nature of organic material in headwater prairie streams. Fire management governed inorganic sediment through mineral soil exposure. ISS concentrations were highest shortly after burns and declined as vegetation recovered, reflecting the supply-limited nature of mineral sediment delivery from disturbed land surfaces. The VSS/ISS ratio integrated both pathways into a single compositional metric that proved more sensitive to management differences than either fraction alone or total suspended solids, providing the clearest separation among the three monitored watersheds.

At the plot scale, rainfall simulation experiments revealed that riparian erodibility responds nonlinearly to vegetation cover, with a critical transition between 50% and 75% cover. Below this threshold, bare-soil connectivity increased sharply, and sediment delivery rose by orders of magnitude. The sediment-to-bromide ratio developed in these experiments separated material erodibility from hydrological connectivity for the first time in grassland systems, extending the multi-gradient riparian framework of Dodds et al. (2025) by providing a quantitative, process-based metric applicable across geomorphic settings. Streambank surveys documented that cattle-grazed watersheds exhibited taller, steeper, more geometrically unstable banks with greater bare-face exposure than bison-grazed or ungrazed watersheds, linking grazer identity to geomorphic condition and confirming the spatial pattern of disturbance inferred from the monitoring and simulation data.

Crosscutting findings

Four themes occur across the three studies and constitute the integrative contribution of this dissertation.

Fire-grazing interactions are pervasive and nonlinear. In every dimension of stream function examined, fire frequency and grazer identity interacted to produce outcomes unpredictable from either factor alone. Fire modulated the rate of woody encroachment and thus the trajectory of evapotranspiration-driven streamflow declines (Chapter 2). Fire interacted with grazing to shape nutrient trajectories that accumulated over decades (Chapter 3). Fire controlled sediment composition while grazer identity determined the spatial pattern and intensity of riparian disturbance (Chapter 4). These interactions reflect the co-evolutionary history of fire and herbivory in tallgrass prairie and validate the pyric herbivory framework (Fuhlendorf & Engle, 2001, 2004) as a unifying paradigm for grassland stream ecology. Management strategies that treat fire and grazing as additive, independent factors will fail to predict stream ecosystem responses.

Temporal scale determines detectability. The hydrological effects of woody encroachment emerged only over four decades of monitoring (Chapter 2). The full magnitude of bison effects on stream nitrate became apparent only after 25 years, during which concentrations approximately doubled, a signal that would register as noise or a modest trend in a five-year study (Chapter 3). Sediment regime differences among management treatments required 15 years to distinguish from interannual variability driven by precipitation and discharge fluctuations (Chapter 4). These are not merely methodological observations; they mean that our collective understanding of grassland stream dynamics, built predominantly on studies spanning two to five years, is systematically biased toward short-term variability and underestimates the magnitude of

slow, directional change (Dodds et al., 2012; Lindenmayer et al., 2010). The Konza LTER dataset is among the very few in the world capable of resolving these decadal trajectories in grassland streams.

Thresholds structure system responses. Woody cover impacts on streamflow intensified once riparian cover exceeded approximately 30%, consistent with tipping-point behavior between wet and dry hydrological states (Chapter 2). Nutrient recovery after cropland restoration exhibited a multi-year lag before rapid decline, reflecting threshold-like flushing of accumulated soil nitrogen (Chapter 3). Sediment erodibility increased sharply below approximately 60–70% riparian vegetation cover (Chapter 4). The management consequence is clear: incremental changes in drivers can produce abrupt, large-magnitude responses once transitions are crossed, and reversing these changes is likely to be far more difficult than preventing initial degradation.

Material fluxes are coupled. Declining discharge concentrates solutes and reduces the capacity of streams to dilute and process nutrient inputs, potentially amplifying the biogeochemical effects of grazing and legacy fertilization even as some nutrient sources decline. Reduced flow days compress the temporal window during which nutrient spiraling, sediment transport, and biological processing occur, concentrating material fluxes into fewer, more intense pulses. Changes in sediment composition alter light availability and benthic habitat quality, which in turn affect nutrient uptake and retention by stream biofilms and macrophytes. Woody encroachment acts as a master driver connecting all three domains: it reduces streamflow (Chapter 2), modifies the riparian pathways through which nutrients reach streams (Chapter 3), and shifts the organic-inorganic sediment balance (Chapter 4). These coupled responses mean that managing any single flux in isolation will likely produce incomplete and potentially misleading assessments of stream condition. The findings extend beyond KPBS: woody

encroachment is occurring on every inhabited continent (Archer et al., 2017; Heidari et al., 2024), agricultural nutrient legacies affect vast areas of the Great Plains and Midwest, and the dual-pathway sediment framework offers a transferable basis for rethinking monitoring in grassland streams worldwide.

Management implications

The findings point toward several concrete management recommendations for tallgrass prairie streams and, by extension, grassland watersheds globally.

Prescribed fire should be maintained or intensified to slow woody encroachment. Across all three dimensions of stream function, fire frequency emerged as the most broadly effective management lever. Annual or biennial burning slowed encroachment rates, maintained streamflow, limited nutrient delivery, and sustained the organic-dominated sediment signature of intact grassland streams. However, fire alone is insufficient: even annually burned watersheds experienced woody expansion and discharge decline once encroachment reached sufficient density, and rising atmospheric CO₂ and temperatures are projected to further favor woody C₃ species over C₄ grasses. Complementary strategies such as mechanical removal, targeted herbicide application, or strategic grazing to suppress woody seedlings will likely be necessary to maintain grassland stream function under future climate scenarios.

Grazer identity should be considered as a water quality management variable. The consistent differences between bison-grazed and cattle-grazed watersheds across nutrients and sediment metrics are large enough to have practical significance. Where stream ecosystem conservation is a priority, bison grazing or cattle management systems that restrict riparian access and rotate grazing pressure across the landscape would better approximate the historical disturbance regime under which these streams evolved. The growing interest in bison restoration

across the Great Plains creates an opportunity to implement this recommendation at meaningful scales.

Expectations for nutrient recovery after cropland restoration should be calibrated to subsurface residence times. Surface water quality improvements can be expected within 4 to 5 years, but groundwater nitrate will remain elevated for decades. Regulatory and planning frameworks should account for these lag times rather than interpreting slow groundwater recovery as evidence that restoration has failed. Monitoring strategies should include groundwater sampling alongside surface water to track the full trajectory of recovery.

Sediment monitoring should incorporate VSS/ISS fractionation. The additional analytical effort is minimal, but the information gain is substantial. Streams with identical TSS concentrations can have fundamentally different sediment compositions, one reflecting healthy in-channel productivity and the other reflecting physical watershed disturbance. Incorporating the VSS/ISS ratio into monitoring programs would provide a sensitive, low-cost indicator of management effects and riparian condition.

Vegetation cover in riparian zones should be monitored against the 50 to 75% erodibility threshold. This threshold, identified through controlled rainfall simulations and consistent with theoretical erosion models, provides a field-measurable early warning indicator for riparian degradation. Sites approaching the lower end of this range should be flagged for intervention before large-scale erosion events occur.

Limitations

Several limitations of these studies warrant acknowledgment. This dissertation is based on a single research site. While KPBS is broadly representative of Flint Hills tallgrass prairie, its specific merokarst geology, long fire management history, and bison population may limit direct

transferability to other grassland systems underlain by different lithologies, managed under different regimes, or grazed by different assemblages. The specific rates and thresholds identified here should be treated as hypotheses to be tested in other settings rather than as universal constants.

The long-term datasets, though exceptionally valuable, are observational at the watershed scale. Management effects are confounded with site identity in the sediment monitoring dataset, and even the fire frequency gradient does not constitute a fully randomized experiment. Causal inference in this dissertation relies on the convergence of multiple analytical approaches, including correlation, ANCOVA, structural equation modeling, generalized additive modeling, and experimental manipulation, rather than on any single definitive test. This triangulation strengthens confidence in the major findings but does not eliminate the possibility of unmeasured confounding factors.

The evapotranspiration estimates in Chapter 2 used a simplified 2x multiplier for woody vegetation relative to grass, based on site-specific sap flow measurements. While conservative and empirically grounded, actual ET ratios likely vary with species composition, stand age and density, topographic position, and seasonal soil moisture availability. Spatially explicit ET modeling incorporating remote sensing of canopy structure would refine these estimates and better capture within-watershed heterogeneity.

The rainfall simulation experiments in Chapter 4 represent single-event responses at ten sites and cannot capture the cumulative effects of repeated storms, seasonal antecedent moisture variation, or multi-year vegetation recovery trajectories. The Sed and Br metric, while demonstrably useful, has not been validated against independent erosion measurements such as erosion pins, repeat cross-sections, or radionuclide tracers.

The structural equation models in both Chapters 2 and 4 explained substantial but incomplete portions of variance in key response variables, particularly flow days, where unmeasured factors, including groundwater dynamics and karst connectivity, likely play important roles. The high residual variance in flow persistence highlights the complexity of intermittent stream hydrology and the need for additional predictive variables in future models.

Future research directions

The findings and limitations of this dissertation point toward several high-priority areas for future work.

Integrated hydrological-biogeochemical-sediment modeling. The coupling among water, nutrients, and sediment documented here calls for process-based models that represent feedback across domains. Such models could address how declining discharge concentrates nutrients, how changing sediment loads alter light and thus nutrient uptake, and how woody encroachment simultaneously modifies flow paths, nutrient processing, and sediment character. The multi-decadal datasets from Konza provide an unusually rich resource for calibrating and validating such models.

Groundwater monitoring under vegetation transition. The unexplained variance in flow days (Chapter 2) and the persistent groundwater nutrient legacy (Chapter 3) both point to subsurface processes as a critical frontier. Seasonal water table monitoring, isotopic partitioning of water sources between grass and woody vegetation, and spatially distributed groundwater models would clarify how woody encroachment affects recharge, storage, and discharge, the subsurface counterpart to the surface water changes documented here.

Early warning indicators for threshold behavior. The nonlinear responses identified across all three chapters suggest that grassland streams may be susceptible to abrupt regime

shifts. Developing statistical, ecological, or remote-sensing-based indicators of proximity to critical thresholds (the woody cover level at which baseflow is permanently lost, the vegetation cover below which erosion accelerates, the nutrient loading rate above which processing capacity is overwhelmed) would have direct management value. Time-series analysis of the Konza monitoring data for signatures of critical slowing down, increased variance, or flickering between states could test whether the theoretical predictions of early warning signals apply to real grassland stream systems.

Cross-site validation. The mechanisms and thresholds identified at Konza should be tested at other grassland sites in the southern Great Plains, the Edwards Plateau, South African savannas, Australian rangelands, and Eurasian steppes to determine which findings are site-specific and which are general features of grassland stream response to woody encroachment, fire suppression, and grazer substitution. The global scope of these drivers suggests broad relevance, but this hypothesis requires explicit empirical testing.

Climate change projections. Rising CO₂ will favor woody C₃ species by increasing their water-use efficiency, potentially accelerating encroachment beyond historical rates. Warming of 2.5 to 5.5°C by 2100 will increase ET demand by an estimated 10 to 25%. Shifts toward more intense but less frequent precipitation will alter infiltration-runoff partitioning in ways that may further favor deep-rooted woody vegetation. The interactions among these climate drivers and the vegetation, nutrient, and sediment dynamics documented in this dissertation are likely to be nonlinear and difficult to predict from any single factor. Scenario modeling that couples climate projections with vegetation dynamics and hydrological response, parameterized against the long-term Konza data, would provide the most robust basis for anticipating the future trajectory of grassland stream ecosystems.

Closing

The streams of KPBS are sentinels. Over four decades, they have registered a progressive decline in flow, a reorganization of nutrient sources and sinks, and a shift in the character of the sediment they carry, all driven by forces that are reshaping grassland ecosystems worldwide. The value of this dissertation lies not in documenting any one of these changes alone, but in showing how they connect: how woody encroachment reduces streamflow and alters the material that streams transport, how fire frequency modulates the pace and trajectory of change across all three domains, how grazer identity shapes the spatial pattern of watershed disturbance, and how agricultural legacies echo through groundwater for decades after the land surface has been restored.

The challenge is that effective management of grassland streams requires attention to the entire system over timescales that exceed typical planning horizons. The opportunity is that the same long-term monitoring that reveals the problem also reveals the mechanisms, and mechanisms are the foundation of prediction. Prescribed fire slows encroachment. Grazer management reduces riparian disturbance. Vegetation cover thresholds provide measurable targets. Agricultural legacies, though slow to attenuate, eventually decline. The tallgrass prairie is one of the most endangered ecosystems on Earth, and maintaining the ecological integrity of its streams will require sustained commitment to both the science of long-term monitoring and the practice of adaptive, mechanism-informed management. This dissertation contributes to both.

References

- Abernethy, B., & Rutherford, I. D. (2001). The distribution and strength of riparian tree roots in relation to riverbank reinforcement. *Hydrological Processes*, 15(1), 63–79.
- Abrams, M. D. (1986). Historical development of gallery forests in northeast Kansas. *Vegetatio*, 65(1), 29–37.
- Acharya, B. S., Hao, Y., Ochsner, T. E., & Zou, C. B. (2017). Woody plant encroachment alters soil hydrological properties and reduces downward flux of water in tallgrass prairie. *Plant and Soil*, 414(1–2), 379–391. <https://doi.org/10.1007/s11104-016-3138-0>
- Alexander, R. B., Boyer, E. W., Smith, R. A., Schwarz, G. E., & Moore, R. B. (2007). The role of headwater streams in downstream water quality. *JAWRA Journal of the American Water Resources Association*, 43(1), 41–59. <https://doi.org/10.1111/j.1752-1688.2007.00005.x>
- Alexander, R. B., Smith, R. A., Schwarz, G. E., Boyer, E. W., Nolan, J. V., & Brakebill, J. W. (2008). Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi river basin. *Environmental Science & Technology*, 42(3), 822–830. <https://doi.org/10.1021/es0716103>
- Allen, D. C., Datry, T., Boersma, K. S., Bogan, M. T., Boulton, A. J., Bruno, D., Busch, M. H., Costigan, K. H., Dodds, W. K., Fritz, K. M., Godsey, S. E., Jones, J. B., Kaletova, T., Kampf, S. K., Mims, M. C., Neeson, T. M., Olden, J. D., Pastor, A. V., Poff, N. L., & Zimmer, M. (2020). River ecosystem conceptual models and non-perennial rivers: A critical review. *WIREs Water*, 7(5), e1473. <https://doi.org/10.1002/wat2.1473>
- Allen, D. C., Datry, T., Boersma, K. S., Bogan, M. T., Boulton, A. J., Bruno, D., Busch, M. H., Costigan, K. H., Dodds, W. K., Fritz, K. M., Godsey, S. E., Jones, J. B., Kaletova, T.,

- Kampf, S. K., Mims, M. C., Neeson, T. M., Olden, J. D., Pastor, A. V., Poff, N. L., ... Zimmer, M. (2020). River ecosystem conceptual models and non-perennial rivers: A critical review. *WIREs Water*, 7(5), e1473. <https://doi.org/10.1002/wat2.1473>
- Allred, B. W., Fuhlendorf, S. D., Engle, D. M., & Elmore, R. D. (2011a). Ungulate preference for burned patches reveals strength of fire–grazing interaction. *Ecology and Evolution*, 1(2), 132–144.
- Allred, B. W., Fuhlendorf, S. D., Engle, D. M., & Elmore, R. D. (2011b). Ungulate preference for burned patches reveals strength of fire–grazing interaction. *Ecology and Evolution*, 1(2), 132–144. <https://doi.org/10.1002/ece3.12>
- Ameel, J. J., Axler, R. P., & Owen, C. J. (1993). Persulfate digestion for determination of total nitrogen and phosphorus in low-nutrient waters. *American Environmental Laboratory*, 10(93), 1–11.
- Anadón, J. D., Sala, O. E., & Maestre, F. T. (2014). Climate change will increase savannas at the expense of forests and treeless vegetation in tropical and subtropical Americas. *Journal of Ecology*, 102(6), 1363–1373. <https://doi.org/10.1111/1365-2745.12325>
- Arango, C. P., Tank, J. L., Schaller, J. L., Royer, T. V., Bernot, M. J., & David, M. B. (2007). Benthic organic carbon influences denitrification in streams with high nitrate concentration. *Freshwater Biology*, 52(7), 1210–1222. <https://doi.org/10.1111/j.1365-2427.2007.01758.x>
- Archer, S. R., Andersen, E. M., Predick, K. I., Schwinning, S., Steidl, R. J., & Woods, S. R. (2017). Woody plant encroachment: Causes and consequences. *Rangeland Systems: Processes, Management and Challenges*, 25–84.

- Baer, S. G., & Blair, J. M. (2008). GRASSLAND ESTABLISHMENT UNDER VARYING RESOURCE AVAILABILITY: A TEST OF POSITIVE AND NEGATIVE FEEDBACK. *Ecology*, 89(7), 1859–1871. <https://doi.org/10.1890/07-0417.1>
- Banerjee, S., Das, D., Zhang, H., & John, R. (2023). Grassland-woodland transitions over decadal timescales in the Terai-Duar savanna and grasslands of the Indian subcontinent. *Forest Ecology and Management*, 530, 120764.
- Banner, E. B. K., Stahl, A. J., & Dodds, W. K. (2009). Stream discharge and riparian land use influence in-stream concentrations and loads of phosphorus from central plains watersheds. *Environmental Management*, 44(3), 552–565. <https://doi.org/10.1007/s00267-009-9332-6>
- Basu, N. B., Destouni, G., Jawitz, J. W., Thompson, S. E., Loukinova, N. V., Darracq, A., Zanardo, S., Yaeger, M., Sivapalan, M., Rinaldo, A., & Rao, P. S. C. (2010). Nutrient loads exported from managed catchments reveal emergent biogeochemical stationarity. *Geophysical Research Letters*, 37(23), 2010GL045168. <https://doi.org/10.1029/2010GL045168>
- Belsky, A. J., Matzke, A., & Uselman, S. (1999). Survey of livestock influences on stream and riparian ecosystems in the Western United States. *Journal of Soil and Water Conservation*, 54(1), 419–431. <https://doi.org/10.1080/00224561.1999.12457258>
- Bernhardt, E. S., Blaszcak, J. R., Ficken, C. D., Fork, M. L., Kaiser, K. E., & Seybold, E. C. (2017). Control points in ecosystems: Moving beyond the hot spot hot moment concept. *Ecosystems*, 20(4), 665–682. <https://doi.org/10.1007/s10021-016-0103-y>

- Bilotta, G. S., & Brazier, R. E. (2008). Understanding the influence of suspended solids on water quality and aquatic biota. *Water Research*, 42(12), 2849–2861.
<https://doi.org/10.1016/j.watres.2008.03.018>
- Blair, J. M. (1997). Fire, N availability, and plant response in grasslands: A test of the transient maxima hypothesis. *Ecology*, 78(8), 2359–2368. [https://doi.org/10.1890/0012-9658\(1997\)078%5B2359:FNAAPR%5D2.0.CO;2](https://doi.org/10.1890/0012-9658(1997)078%5B2359:FNAAPR%5D2.0.CO;2)
- Bosch, J. M., & Hewlett, J. D. (1982). A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, 55(1–4), 3–23.
- Briggs, J. M., Knapp, A. K., Blair, J. M., Heisler, J. L., Hoch, G. A., Lett, M. S., & McCarron, J. K. (2005a). An ecosystem in transition: Causes and consequences of the conversion of mesic grassland to shrubland. *BioScience*, 55(3), 243–254.
- Briggs, J. M., Knapp, A. K., Blair, J. M., Heisler, J. L., Hoch, G. A., Lett, M. S., & McCarron, J. K. (2005b). An ecosystem in transition: Causes and consequences of the conversion of mesic grassland to shrubland. *BioScience*, 55(3), 243–254.
- Buttle, J. M., Boon, S., Peters, D. L., Spence, C., (Ilja) Van Meerveld, H. J., & Whitfield, P. H. (2012). An Overview of Temporary Stream Hydrology in Canada. *Canadian Water Resources Journal / Revue Canadienne Des Ressources Hydriques*, 37(4), 279–310.
<https://doi.org/10.4296/cwrj2011-903>
- Callaghan, T. V., Jonasson, C., Thierfelder, T., Yang, Z., Hedenås, H., Johansson, M., Molau, U., Van Bogaert, R., Michelsen, A., Olofsson, J., Gwynn-Jones, D., Bokhorst, S., Phoenix, G., Bjerke, J. W., Tømmervik, H., Christensen, T. R., Hanna, E., Koller, E. K., & Sloan, V. L. (2013). Ecosystem change and stability over multiple decades in the

- Swedish subarctic: Complex processes and multiple drivers. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368(1624), 20120488.
<https://doi.org/10.1098/rstb.2012.0488>
- Change, I. P. O. C. (2007). Climate change 2007: The physical science basis. *Agenda*, 6(07), 333.
- Collins, A. L., Pulley, S., Foster, I. D. L., Gellis, A., Porto, P., & Horowitz, A. J. (2017). Sediment source fingerprinting as an aid to catchment management: A review of the current state of knowledge and a methodological decision-tree for end-users. *Journal of Environmental Management*, 194, 86–108.
- Collins, S. L., & Calabrese, L. B. (2012). Effects of fire, grazing and topographic variation on vegetation structure in tallgrass prairie. *Journal of Vegetation Science*, 23(3), 563–575.
<https://doi.org/10.1111/j.1654-1103.2011.01369.x>
- Collins, S. L., Nippert, J. B., Blair, J. M., Briggs, J. M., Blackmore, P., & Ratajczak, Z. (2022). Fire frequency, state change and hysteresis in tallgrass prairie. *Ecology Letters*, 24(4), 636–647.
- Collins, S. L., & Wallace, L. L. (1990). *Fire in North American Tallgrass Prairies*. University of Oklahoma Press.
- Cook, B. I., Mankin, J. S., Marvel, K., Williams, A. P., Smerdon, J. E., & Anchukaitis, K. J. (2020). Twenty-First Century Drought Projections in the CMIP6 Forcing Scenarios. *Earth's Future*, 8(6), e2019EF001461. <https://doi.org/10.1029/2019EF001461>
- Costigan, K. H., Daniels, M. D., & Dodds, W. K. (2015). Fundamental spatial and temporal disconnections in the hydrology of an intermittent prairie headwater network. *Journal of Hydrology*, 522, 305–316.

- Costigan, K. H., Daniels, M. D., Perkin, J. S., & Gido, K. B. (2014). Longitudinal variability in hydraulic geometry and substrate characteristics of a Great Plains sand-bed river. *Geomorphology*, 210, 48–58.
- Dahlman, R. C., & Kucera, C. L. (1968). Tagging Native Grassland Vegetation with Carbon-14. *Ecology*, 49(6), 1199–1203. <https://doi.org/10.2307/1934516>
- Datry, T., Larned, S. T., & Tockner, K. (2014). Intermittent rivers: A challenge for freshwater ecology. *BioScience*, 64(3), 229–235.
- Datry, T., Truchy, A., Olden, J. D., Busch, M. H., Stubbington, R., Dodds, W. K., Zipper, S., Yu, S., Messenger, M. L., & Tonkin, J. D. (2023). Causes, responses, and implications of anthropogenic versus natural flow intermittence in river networks. *BioScience*, 73(1), 9–22.
- Davenport, F. V., & Diffenbaugh, N. S. (2021). Using Machine Learning to Analyze Physical Causes of Climate Change: A Case Study of U.S. Midwest Extreme Precipitation. *Geophysical Research Letters*, 48(15), e2021GL093787. <https://doi.org/10.1029/2021GL093787>
- DelVecchia, A. G., Shanafield, M., Zimmer, M. A., Busch, M. H., Krabbenhoft, C. A., Stubbington, R., Kaiser, K. E., Burrows, R. M., Hosen, J., Datry, T., Kampf, S. K., Zipper, S. C., Fritz, K., Costigan, K., & Allen, D. C. (2022). Reconceptualizing the hyporheic zone for nonperennial rivers and streams. *Freshwater Science*, 41(2), 167–182. <https://doi.org/10.1086/720071>
- Dodds, W. K. (1997a). Distribution of runoff and rivers related to vegetative characteristics, latitude, and slope: A global perspective. *Journal of the North American Benthological Society*, 16(1), 162–168. <https://doi.org/10.2307/1468248>

- Dodds, W. K. (1997b). Distribution of Runoff and Rivers Related to Vegetative Characteristics, Latitude, and Slope: A Global Perspective. *Journal of the North American Benthological Society*, 16(1), 162–168. <https://doi.org/10.2307/1468248>
- Dodds, W. K., Banks, M. K., Clenan, C. S., Rice, C. W., Sotomayor, D., Strauss, E. A., & Yu, W. (1996). Biological properties of soil and subsurface sediments under abandoned pasture and cropland. *Soil Biology and Biochemistry*, 28(7), 837–846.
- Dodds, W. K., Barmuta, L. A., Bernal, S., Corman, J., Harms, T. K., Johnson, S. L., Li, L., Fernandes Cunha, D. G., Olden, J. D., Riis, T., Silva, L. C. R., Stella, J. C., Sullivan, P., & Wohl, E. (2025). Defining stream riparian zones across multidimensional environmental gradients. *Journal of Environmental Quality*, 54, 1674–1697. <https://doi.org/10.1002/jeq2.70080>
- Dodds, W. K., Blair, J. M., Henebry, G. M., Koelliker, J. K., Ramundo, R., & Tate, C. M. (1996). Nitrogen transport from tallgrass prairie watersheds. *Journal of Environmental Quality*, 25(5), 973–981. <https://doi.org/10.2134/jeq1996.00472425002500050007x>
- Dodds, W. K., Gido, K., Whiles, M. R., Daniels, M. D., & Grudzinski, B. P. (2015). The stream biome gradient concept: Factors controlling lotic systems across broad biogeographic scales. *Freshwater Science*, 34(1), 1–19. <https://doi.org/10.1086/679756>
- Dodds, W. K., Gido, K., Whiles, M. R., Fritz, K. M., & Matthews, W. J. (2004). Life on the edge: The ecology of Great Plains prairie streams. *BioScience*, 54(3), 205–216.
- Dodds, W. K., & Oakes, R. M. (2006). Controls on nutrients across a prairie stream watershed: Land use and riparian cover effects. *Environmental Management*, 37(5), 634–646. <https://doi.org/10.1007/s00267-004-0072-3>

- Dodds, W. K., & Oakes, R. M. (2008). Headwater influences on downstream water quality. *Environmental Management*, 41(3), 367–377. <https://doi.org/10.1007/s00267-007-9033-y>
- Dodds, W. K., Ratajczak, Z., Keen, R. M., Nippert, J. B., Grudzinski, B., Veatch, A., Taylor, J. H., & Kuhl, A. (2023). Trajectories and state changes of a grassland stream and riparian zone after a decade of woody vegetation removal. *Ecological Applications*, 33(4), e2830. <https://doi.org/10.1002/eap.2830>
- Dodds, W. K., Robinson, C. T., Gaiser, E. E., Hansen, G. J., Powell, H., Smith, J. M., Morse, N. B., Johnson, S. L., Gregory, S. V., & Bell, T. (2012). Surprises and insights from long-term aquatic data sets and experiments. *BioScience*, 62(8), 709–721.
- Dodds, W. K., & Whiles, M. R. (2004). Quality and quantity of suspended particles in rivers: Continent-scale patterns in the United States. *Environmental Management*, 33(3), 355–367.
- Dodds, W. K., Wichman, G., Guinnip, J. P., Corman, J. R., & Blair, J. M. (2022). Assessing transport and retention of nitrate and other materials through the riparian zone and stream channel with simulated precipitation. *Methods in Ecology and Evolution*, 13(3), 757–766. <https://doi.org/10.1111/2041-210X.13791>
- D’Odorico, P., Okin, G. S., & Bestelmeyer, B. T. (2012). A synthetic review of feedbacks and drivers of shrub encroachment in arid grasslands. *Ecohydrology*, 5(5), 520–530. <https://doi.org/10.1002/eco.259>
- Ekberzade, B., Yetemen, O., Lutfi Sen, O., & Nuzhet Dalfes, H. (2023). Transitioning: The role of disturbances on instigating cross-overs of vegetation zones (a biome perspective). *EGU General Assembly Conference Abstracts*, EGU-141. <https://ui.adsabs.harvard.edu/abs/2023EGUGA..25..141E/abstract>

- Eldridge, D. J., Wang, L., & Ruiz-Colmenero, M. (2015). Shrub encroachment alters the spatial patterns of infiltration. *Ecohydrology*, 8(1), 83–93. <https://doi.org/10.1002/eco.1490>
- Eng, K., Carlisle, D., Grantham, T. E., Wolock, D. M., & Eng, R. L. (2019). *Severity and extent of alterations to natural streamflow regimes based on hydrologic metrics in the conterminous United States, 1980–2014*. US Geological Survey. <https://pubs.usgs.gov/publication/sir20195001>
- Feurdean, A., Grindean, R., Florescu, G., Tanțău, I., Niedermeyer, E. M., Diaconu, A.-C., Hutchinson, S. M., Nielsen, A. B., Sava, T., & Panait, A. (2021). The transformation of the forest steppe in the lower Danube Plain of southeastern Europe: 6000 years of vegetation and land use dynamics. *Biogeosciences*, 18(3), 1081–1103.
- Fiorillo, F., Leone, G., Pagnozzi, M., & Esposito, L. (2021). Long-term trends in karst spring discharge and relation to climate factors and changes. *Hydrogeology Journal*, 29(1), 347–377. <https://doi.org/10.1007/s10040-020-02265-0>
- Frank, D. A., & Evans, R. D. (1997). Effects of native grazers on grassland N cycling in Yellowstone National Park. *Ecology*, 78(7), 2238–2248. [https://doi.org/10.1890/0012-9658\(1997\)078%5B2238:EONGOG%5D2.0.CO;2](https://doi.org/10.1890/0012-9658(1997)078%5B2238:EONGOG%5D2.0.CO;2)
- Frank, D. A., Wallen, R. L., & White, P. J. (2016). Ungulate control of grassland production: Grazing intensity and ungulate species composition in Yellowstone Park. *Ecosphere*, 7(11), e01603. <https://doi.org/10.1002/ecs2.1603>
- Fuhlendorf, S. D., & Engle, D. M. (2001). Restoring heterogeneity on rangelands: Ecosystem management based on evolutionary grazing patterns. *BioScience*, 51(8), 625–632.

- Fuhlendorf, S. D., & Engle, D. M. (2004). Application of the fire–grazing interaction to restore a shifting mosaic on tallgrass prairie. *Journal of Applied Ecology*, 41(4), 604–614.
<https://doi.org/10.1111/j.0021-8901.2004.00937.x>
- Fuhlendorf, S. D., Engle, D. M., Kerby, J., & Hamilton, R. (2009). Pyric herbivory: Rewilding landscapes through the recoupling of fire and grazing. *Conservation Biology*, 23(3), 588–598.
- Gambill, I., Zipper, S., Kirk, M., & Seybold, E. (2024). *Exploring drivers of groundwater recharge at Konza Prairie (Flint Hills region, Kansas, USA) using transfer function noise models*. <https://www.kgs.ku.edu/Publications/OFR/2024/OFR2024-6.pdf>
- Garbrecht, J., Van Liew, M., & Brown, G. O. (2004). Trends in precipitation, streamflow, and evapotranspiration in the Great Plains of the United States. *Journal of Hydrologic Engineering*, 9(5), 360–367. [https://doi.org/10.1061/\(ASCE\)1084-0699\(2004\)9:5\(360\)](https://doi.org/10.1061/(ASCE)1084-0699(2004)9:5(360))
- Gellis, A. C., Fitzpatrick, F., & Stout, J. (2024). Advances in sediment fingerprinting for watershed management. *Journal of Soils and Sediments*, 24, 1056–1072.
- Grudzinski, B. P., & Daniels, M. D. (2018). Bison and cattle grazing impacts on grassland stream morphology in the Flint Hills of Kansas. *Rangeland Ecology & Management*, 71(6), 783–791.
- Grudzinski, B. P., Daniels, M. D., & Dodds, W. K. (2016). Does tallgrass prairie riparian forest buffer strip removal affect stream bank erosion rates? *Geomorphology*, 266, 1–11.
- Grudzinski, B. P., Ruffing, C. M., Daniels, M. D., & Rawitch, M. (2018). Bison and cattle grazing impacts on baseflow suspended sediment concentrations within grassland streams. *Annals of the American Association of Geographers*, 108(6), 1570–1581.
<https://doi.org/10.1080/24694452.2018.1457430>

- Gyssels, G., Poesen, J., Bochet, E., & Li, Y. (2005). Impact of plant roots on the resistance of soils to erosion by water: A review. *Progress in Physical Geography*, 29(2), 189–217.
- Hamilton, S. K. (2012). Biogeochemical time lags may delay responses of streams to ecological restoration. *Freshwater Biology*, 57(s1), 43–57. <https://doi.org/10.1111/j.1365-2427.2011.02685.x>
- Hammond, J. C., Zimmer, M., Shanafield, M., Kaiser, K., Godsey, S. E., Mims, M. C., Zipper, S. C., Burrows, R. M., Kampf, S. K., Dodds, W., Jones, C. N., Krabbenhoft, C. A., Boersma, K. S., Datry, T., Olden, J. D., Allen, G. H., Price, A. N., Costigan, K., Hale, R., ... Allen, D. C. (2021). Spatial Patterns and Drivers of Nonperennial Flow Regimes in the Contiguous United States. *Geophysical Research Letters*, 48(2), e2020GL090794. <https://doi.org/10.1029/2020GL090794>
- Hargreaves, G. H., & Allen, R. G. (2003). History and Evaluation of Hargreaves Evapotranspiration Equation. *Journal of Irrigation and Drainage Engineering*, 129(1), 53–63. [https://doi.org/10.1061/\(ASCE\)0733-9437\(2003\)129:1\(53\)](https://doi.org/10.1061/(ASCE)0733-9437(2003)129:1(53))
- Hargreaves, G. L., Hargreaves, G. H., & Riley, J. P. (1985). Irrigation Water Requirements for Senegal River Basin. *Journal of Irrigation and Drainage Engineering*, 111(3), 265–275. [https://doi.org/10.1061/\(ASCE\)0733-9437\(1985\)111:3\(265\)](https://doi.org/10.1061/(ASCE)0733-9437(1985)111:3(265))
- Hatley, C. M., Armijo, B., Andrews, K., Anhold, C., Nippert, J. B., & Kirk, M. F. (2023). Intermittent streamflow generation in a merokarst headwater catchment. *Environmental Science: Advances*, 2(1), 115–131.
- Heidari, A., Nippert, J. B., & Ratajczak, Z. (2024). Remote sensing of woody encroachment in grasslands: Advances, challenges, and management applications. *Remote Sensing of Environment*, 301, 113952.

- Herrero, M., Henderson, B., & Thornton, P. K. (2024). Precision conservation in agricultural landscapes: Targeting interventions where they matter most. *Nature Food*, 5, 201–213.
- Hervé-Fernández, P., Muñoz-Arriagada, R., Glucevic-Almonacid, C., Bahamonde-Vidal, L., & Radic-Schilling, S. (2023). Influence of rangeland land cover on infiltration rates, field-saturated hydraulic conductivity, and soil water repellency in southern Patagonia. *Rangeland Ecology & Management*, 90, 92–100.
- Hill, A. R. (2019). Groundwater nitrate removal in riparian buffer zones: A review of research progress in the past 20 years. *Biogeochemistry*, 143(3), 347–369.
<https://doi.org/10.1007/s10533-019-00566-5>
- Hintz, H. W. (1999). Grassland dynamics: Long-term ecological research in tallgrass prairie. *Journal of Environmental Quality*, 28(5), 1685.
- Hobbs, N. T. (1996). Modification of ecosystems by ungulates. *The Journal of Wildlife Management*, 695–713.
- Holman, I. P., Whelan, M. J., Howden, N. J. K., Bellamy, P. H., Willby, N. J., Rivas-Casado, M., & McConvey, P. (2008). Phosphorus in groundwater—An overlooked contributor to eutrophication? *Hydrological Processes*, 22(26), 5121–5127.
<https://doi.org/10.1002/hyp.7198>
- Horowitz, A. J. (2009). Determining annual suspended sediment and sediment-associated trace element and nutrient fluxes. *Science of the Total Environment*, 407(14), 3849–3870.
- Hubbard, R. K., Newton, G. L., & Hill, G. M. (2004). Water quality and the grazing animal. *Journal of Animal Science*, 82(suppl_13), E255–E263.
- Husk, B., Julian, P., Tromas, N., Phan, D., Painter, K., Baulch, H. M., & Sauvé, S. (2023). Improving water quality in a hypereutrophic lake and tributary through agricultural

- nutrient mitigation: A multi-year monitoring analysis. *Available at SSRN 4572355*.
https://papers.ssrn.com/sol3/papers.cfm?abstract_id=4572355
- Hutchinson, S. L., Koelliker, J. K., & Knapp, A. K. (2008). Development of water usage coefficients for a fully watered tallgrass prairie. *Transactions of the ASABE*, 51(1), 153–159.
- Huxman, T. E., Wilcox, B. P., Breshears, D. D., Scott, R. L., Snyder, K. A., Small, E. E., Hultine, K., Pockman, W. T., & Jackson, R. B. (2005). ECOHYDROLOGICAL IMPLICATIONS OF WOODY PLANT ENCROACHMENT. *Ecology*, 86(2), 308–319.
<https://doi.org/10.1890/03-0583>
- Jack Brookshire, E. N., Gerber, S., Webster, J. R., Vose, J. M., & Swank, W. T. (2011). Direct effects of temperature on forest nitrogen cycling revealed through analysis of long-term watershed records: Direct effects of temperature on forest nitrogen cycling. *Global Change Biology*, 17(1), 297–308. <https://doi.org/10.1111/j.1365-2486.2010.02245.x>
- Jaeger, K. L., Olden, J. D., & Pelland, N. A. (2014). Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. *Proceedings of the National Academy of Sciences*, 111(38), 13894–13899.
<https://doi.org/10.1073/pnas.1320890111>
- Jones, J. I., Murphy, J. F., Collins, A. L., Sear, D. A., Naden, P. S., & Armitage, P. D. (2012). The impact of fine sediments on macro-invertebrates. *River Research and Applications*, 28(8), 1055–1071.
- Keen, R. M., Nippert, J. B., Sullivan, P. L., Ratajczak, Z., Ritchey, B., O’Keefe, K., & Dodds, W. K. (2023). Impacts of Riparian and Non-riparian Woody Encroachment on Tallgrass

- Prairie Ecohydrology. *Ecosystems*, 26(2), 290–301. <https://doi.org/10.1007/s10021-022-00756-7>
- Keen, R. M., Sadayappan, K., Jarecke, K. M., Li, L., Kirk, M. F., Sullivan, P. L., & Nippert, J. B. (2024). Unexpected hydrologic response to ecosystem state change in tallgrass prairie. *Journal of Hydrology*, 643, 131937.
- Keen, R. M., Voelker, S. L., Wang, S. -Y. S., Bentz, B. J., Goulden, M. L., Dangerfield, C. R., Reed, C. C., Hood, S. M., Csank, A. Z., Dawson, T. E., Merschel, A. G., & Still, C. J. (2022). Changes in tree drought sensitivity provided early warning signals to the California drought and forest mortality event. *Global Change Biology*, 28(3), 1119–1132. <https://doi.org/10.1111/gcb.15973>
- Kemp, M. J., & Dodds, W. K. (2001). Spatial and temporal patterns of nitrogen concentrations in pristine and agriculturally-influenced prairie streams. *Biogeochemistry*, 53(2), 125–141. <https://doi.org/10.1023/A:1010707632340>
- Kemp, P., Sear, D., Collins, A., Naden, P., & Jones, I. (2011). The impacts of fine sediment on riverine fish. *Hydrological Processes*, 25(11), 1800–1821.
- Knapp, A. K., Blair, J. M., Briggs, J. M., Collins, S. L., Hartnett, D. C., Johnson, L. C., & Towne, E. G. (1999). The keystone role of bison in North American tallgrass prairie. *BioScience*, 49(1), 39–50. <https://doi.org/10.2307/1313492>
- Knapp, A. K., Blair, J. M., Briggs, J. M., Collins, S. L., Hartnett, D. C., Johnson, L. C., & Towne, E. G. (1999). The keystone role of bison in North American tallgrass prairie: Bison increase habitat heterogeneity and alter a broad array of plant, community, and ecosystem processes. *Bioscience*, 49, 39–50.

- Knapp, A. K., Briggs, J. M., Hartnett, D. C., & Collins, S. L. (1998). *Grassland Dynamics: Long-Term Ecological Research in Tallgrass Prairie*. Oxford University Press.
- Kohl, M. T., Krausman, P. R., Kunkel, K., & Williams, D. M. (2013). Bison versus cattle: Are they ecologically synonymous? *Rangeland Ecology & Management*, 66(6), 721–731.
- Krabbenhoft, C. A., Allen, G. H., Lin, P., Godsey, S. E., Allen, D. C., Burrows, R. M., DelVecchia, A. G., Fritz, K. M., Shanafield, M., & Burgin, A. J. (2022). Assessing placement bias of the global river gauge network. *Nature Sustainability*, 5(7), 586–592.
- Larson, D. M., Dodds, W. K., & Veach, A. M. (2019). Removal of Woody Riparian Vegetation Substantially Altered a Stream Ecosystem in an Otherwise Undisturbed Grassland Watershed. *Ecosystems*, 22(1), 64–76. <https://doi.org/10.1007/s10021-018-0252-2>
- Larson, D. M., Dodds, W. K., Whiles, M. R., Fulgoni, J. N., & Thompson, T. R. (2016a). A before-and-after assessment of patch-burn grazing and riparian fencing along headwater streams. *Journal of Applied Ecology*, 53(5), 1543–1553. <https://doi.org/10.1111/1365-2664.12692>
- Larson, D. M., Dodds, W. K., Whiles, M. R., Fulgoni, J. N., & Thompson, T. R. (2016b). A before-and-after assessment of patch-burn grazing and riparian fencing along headwater streams. *Journal of Applied Ecology*, 53(5), 1543–1553. <https://doi.org/10.1111/1365-2664.12692>
- Larson, D. M., Grudzinski, B. P., Dodds, W. K., Daniels, M. D., Skibbe, A., & Joern, A. (2013). Blazing and grazing: Influences of fire and bison on tallgrass prairie stream water quality. *Freshwater Science*, 32(3), 779–791. <https://doi.org/10.1899/12-118.1>
- Larson, J. H., Bailey, S. W., Kreiling, R. M., Bartsch, L. A., Frost, P. C., Xenopoulos, M. A., Pearce, N. J. T., & Evans, M. A. (2024). Comparison of sediment and water column

- nutrient processing rates in agricultural streams of contrasting buffer land use. *Ecosphere*, 15(3), e4808. <https://doi.org/10.1002/ecs2.4808>
- Leigh, C., Boulton, A. J., Courtwright, J. L., Fritz, K., May, C. L., Walker, R. H., & Datry, T. (2016). Ecological research and management of intermittent rivers: An historical review and future directions. *Freshwater Biology*, 61(8), 1181–1199. <https://doi.org/10.1111/fwb.12646>
- Leopold, L. B., & Maddock, T. (1953). *Leopold, L. B., & Maddock, T. (1953). The hydraulic geometry of stream channels and some physiographic implications.* (Vol. 252). US Government Printing Office.
- Li, L. (2019). Watershed reactive transport. *Reviews in Mineralogy and Geochemistry*, 85(1), 381–418.
- Li, L., Stewart, B., Zhi, W., Sadayappan, K., Ramesh, S., Kerins, D., Sterle, G., Harpold, A., & Perdrial, J. (2022). Climate Controls on River Chemistry. *Earth's Future*, 10(6), e2021EF002603. <https://doi.org/10.1029/2021EF002603>
- Liang, K.-Y., & Zeger, S. L. (1986). Longitudinal data analysis using generalized linear models. *Biometrika*, 13–22.
- Lind, L., Hasselquist, E. M., & Laudon, H. (2019). Towards ecologically functional riparian zones: A meta-analysis to develop guidelines for protecting ecosystem functions and biodiversity in agricultural landscapes. *Journal of Environmental Management*, 249, 109391. <https://doi.org/10.1016/j.jenvman.2019.109391>
- Lindenmayer, D. B., Likens, G. E., Krebs, C. J., & Hobbs, R. J. (2010). Improved probability of detection of ecological “surprises.” *Proceedings of the National Academy of Sciences*, 107(51), 21957–21962. <https://doi.org/10.1073/pnas.1015696107>

- Line, D. E., Harman, W. A., Jennings, G. D., Thompson, E. J., & Osmond, D. L. (2000). Nonpoint-source pollutant load reductions associated with livestock exclusion. *Journal of Environmental Quality*, 29(6), 1882–1890.
<https://doi.org/10.2134/jeq2000.00472425002900060022x>
- Liu, H., Banfield, C., Gomes, S. I., Gube, M., Weig, A., & Pausch, J. (2023). Vegetation transition from meadow to forest reduces priming effect on SOM decomposition. *Soil Biology and Biochemistry*, 184, 109123.
- Lowrance, R., Todd, R., Fail, J., Jr., Hendrickson, O., Jr., Leonard, R., & Asmussen, L. (1984). Riparian forests as nutrient filters in agricultural watersheds. *BioScience*, 34(6), 374–377.
<https://doi.org/10.2307/1309729>
- Malan, J.-A. C., Flint, N., Jackson, E. L., Irving, A. D., & Swain, D. L. (2018). Offstream watering points for cattle: Protecting riparian ecosystems and improving water quality? *Agriculture, Ecosystems & Environment*, 256, 144–152.
- Maskey, M. L., Facincani Dourado, G., Rallings, A. M., Rheinheimer, D. E., Medellín-Azuara, J., & Viers, J. H. (2022). Assessing hydrological alteration caused by climate change and reservoir operations in the San Joaquin River Basin, California. *Frontiers in Environmental Science*, 10, 765426.
- Mayer, P. M., Reynolds, S. K., Jr., McCutchen, M. D., & Canfield, T. J. (2007). Meta-analysis of nitrogen removal in riparian buffers. *Journal of Environmental Quality*, 36(4), 1172–1180. <https://doi.org/10.2134/jeq2006.0462>
- McCabe, G. J., & Wolock, D. M. (2002). Trends and temperature sensitivity of moisture conditions in the conterminous United States. *Climate Research*, 20(1), 19–29.

- McDowell, R. W., Monaghan, R. M., & Carey, P. L. (2003). Potential phosphorus losses in overland flow from pastoral soils receiving long-term applications of either superphosphate or reactive phosphate rock. *New Zealand Journal of Agricultural Research*, 46(4), 329–337. <https://doi.org/10.1080/00288233.2003.9513561>
- McGranahan, D. A., Hovick, T. J., & Fuhlendorf, S. D. (2024). Fire and grazing as coupled processes: Advancing the fire–grazing interaction paradigm. *Trends in Ecology & Evolution*, 39(2), 143–153.
- McKay, S. F., & King, A. J. (2006). Potential ecological effects of water extraction in small, unregulated streams. *River Research and Applications*, 22(9), 1023–1037. <https://doi.org/10.1002/rra.958>
- McNew, L. B., Winder, V. L., Pitman, J. C., & Sandercock, B. K. (2015). Alternative rangeland management strategies and the nesting ecology of greater prairie-chickens. *Rangeland Ecology & Management*, 68(3), 298–304.
- Meals, D. W., Dressing, S. A., & Davenport, T. E. (2010). Lag time in water quality response to best management practices: A review. *Journal of Environmental Quality*, 39(1), 85–96. <https://doi.org/10.2134/jeq2009.0108>
- Messenger, M. L., Lehner, B., Cockburn, C., Lamouroux, N., Pella, H., Snelder, T., Tockner, K., Trautmann, T., Watt, C., & Datry, T. (2021). Global prevalence of non-perennial rivers and streams. *Nature*, 594(7863), 391–397.
- Naeth, M. A., Bailey, A. W., Pluth, D. J., Chanasyk, D. S., & Hardin, R. T. (1990). Grazing impacts on litter and soil organic matter in mixed and fescue prairie within a Canadian rangeland. *Journal of Range Management*, 43(1), 102–109.

- Naiman, R. J., & Décamps, H. (1997). The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics*, 28(1), 621–658.
- Nearing, M. A., Xie, Y., Liu, B., & Ye, Y. (2017). Natural and anthropogenic rates of soil erosion. *International Soil and Water Conservation Research*, 5(2), 77–84.
- Nippert, J. B., & Knapp, A. K. (2007). Soil water partitioning contributes to species coexistence in tallgrass prairie. *Oikos*, 116(6), 1017–1029. <https://doi.org/10.1111/j.0030-1299.2007.15630.x>
- O’Keefe, K., Bell, D. M., McCulloh, K. A., & Nippert, J. B. (2020). Bridging the Flux Gap: Sap Flow Measurements Reveal Species-Specific Patterns of Water Use in a Tallgrass Prairie. *Journal of Geophysical Research: Biogeosciences*, 125(2), e2019JG005446. <https://doi.org/10.1029/2019JG005446>
- Owens, P. N., Collins, A. L., Mayergantz, M., & Petticrew, E. L. (2024). Fine sediment and aquatic ecosystems: New perspectives on processes, impacts, and management. *Science of the Total Environment*, 912, 169187.
- Pacific, V. J., Jencso, K. G., & McGlynn, B. L. (2010). Variable flushing mechanisms and landscape structure control stream DOC export during snowmelt in a set of nested catchments. *Biogeochemistry*, 99(1–3), 193–211. <https://doi.org/10.1007/s10533-009-9401-1>
- Perkin, J. S., Gido, K. B., Falke, J. A., Fausch, K. D., Crockett, H., Johnson, E. R., & Sanderson, J. (2017). Groundwater declines are linked to changes in Great Plains stream fish assemblages. *Proceedings of the National Academy of Sciences*, 114(28), 7373–7378. <https://doi.org/10.1073/pnas.1618936114>

- Peterjohn, W. T., & Correll, D. L. (1984). Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology*, 65(5), 1466–1475.
<https://doi.org/10.2307/1939127>
- Pierson, F. B., Moffet, C. A., Williams, C. J., Hardegree, S. P., & Clark, P. E. (2011). Prescribed-fire effects on rill and interrill runoff and erosion in a mountainous sagebrush landscape. *Earth Surface Processes and Landforms*, 36(6), 750–759.
- Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegard, K. L., Richter, B. D., Sparks, R. E., & Stromberg, J. C. (1997). The natural flow regime. *BioScience*, 47(11), 769–784.
- Porter, E. M., Bowman, W. D., Clark, C. M., Compton, J. E., Pardo, L. H., & Soong, J. L. (2013). Interactive effects of anthropogenic nitrogen enrichment and climate change on terrestrial and aquatic biodiversity. *Biogeochemistry*, 114(1–3), 93–120.
<https://doi.org/10.1007/s10533-012-9803-3>
- Price, A. N., Zimmer, M. A., Bergstrom, A., Burgin, A. J., Seybold, E. C., Krabbenhoft, C. A., Zipper, S., Busch, M. H., Dodds, W. K., & Walters, A. (2024). Biogeochemical and community ecology responses to the wetting of non-perennial streams. *Nature Water*, 2(9), 815–826.
- Prosser, I. P., Dietrich, W. E., & Stevenson, J. (1995). Flow resistance and sediment transport by concentrated overland flow in a grassland valley. *Geomorphology*, 13(1–4), 71–86.
- Putnam, J. E., Perry, C. A., & Wolock, D. M. (2008). *Hydrologic Droughts in Kansas: Are They Becoming Worse?* US Department of the Interior, US Geological Survey.
<https://pubs.usgs.gov/fs/2008/3034/pdf/fs2008-3034.pdf>
- Rahmani, V., Hutchinson, S. L., Harrington, J. A., & Hutchinson, J. M. S. (2016). Analysis of frequency and magnitude of extreme rainfall events with potential impacts on flooding: A

- case study from the central united states. *International Journal of Climatology*, 36(10), 3578–3587. <https://doi.org/10.1002/joc.4577>
- Raihan, M. A., Dodds, W., Zipper, S., & Moore, T. (2025). *Controls on discharge and drying in an intermittent grassland stream: Temporal and network variability*. Press.
<https://doi.org/10.22541/au.173773026.68627327/v1>
- Ratajczak, Z., Nippert, J. B., & Collins, S. L. (2012). Woody encroachment decreases diversity across North American grasslands and savannas. *Ecology*, 93(4), 697–703.
<https://doi.org/10.1890/11-1199.1>
- Raynor, E. J., Joern, A., Nippert, J. B., & Briggs, J. M. (2021). Foraging decisions underlying restricted space use: Effects of fire and forage maturation on large herbivore nutrient uptake. *Ecology and Evolution*, 6(11), 3841–3853.
- Rodgers, K., Roland, V., Hoos, A., Crowley-Ornelas, E., & Knight, R. (2020). An analysis of streamflow trends in the southern and southeastern US from 1950–2015. *Water*, 12(12), 3345.
- Rosenzweig, S. T., Carson, M. A., Baer, S. G., & Blair, J. M. (2016). Changes in soil properties, microbial biomass, and fluxes of C and N in soil following post-agricultural grassland restoration. *Applied Soil Ecology*, 100, 186–194.
- Ryu, J.-H., & Hayhoe, K. (2017). Observed and CMIP5 modeled influence of large-scale circulation on summer precipitation and drought in the South-Central United States. *Climate Dynamics*, 49(11–12), 4293–4310. <https://doi.org/10.1007/s00382-017-3534-z>
- Sadayappan, K., Jarecke, K. M., Keen, R. M., Puthalalth, S., Dodds, W. K., Kirk, M. F., Sullivan, P. L., Nippert, J. B., Seybold, E. C., & Li, L. (2026). Flow or No-Flow: Does

- Discharge Regulate Water Chemistry in Intermittent Streams? *Water Resources Research*, 62(1), e2025WR040321. <https://doi.org/10.1029/2025WR040321>
- Sadayappan, K., Keen, R., Jarecke, K. M., Moreno, V., Nippert, J. B., Kirk, M. F., Sullivan, P. L., & Li, L. (2023). Drier streams despite a wetter climate in woody-encroached grasslands. *Journal of Hydrology*, 627, 130388.
- Sauquet, E., Shanafield, M., Hammond, J. C., Sefton, C., Leigh, C., & Datry, T. (2021). Classification and trends in intermittent river flow regimes in Australia, northwestern Europe and USA: A global perspective. *Journal of Hydrology*, 597, 126170.
- Scheffer, M., Carpenter, S. R., Dakos, V., & Van Nes, E. H. (2015). Generic Indicators of Ecological Resilience: Inferring the Chance of a Critical Transition. *Annual Review of Ecology, Evolution, and Systematics*, 46(1), 145–167. <https://doi.org/10.1146/annurev-eolsys-112414-054242>
- Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., & Mariotti, A. (2013). Long-term fate of nitrate fertilizer in agricultural soils. *Proceedings of the National Academy of Sciences*, 110(45), 18185–18189. <https://doi.org/10.1073/pnas.1305372110>
- Shafer, M., Ojima, D., Antle, J. M., Kluck, D., McPherson, R. A., Petersen, S., Scanlon, B., & Sherman, K. (2014). Ch. 19: Great plains. *Climate Change Impacts in the United States: The Third National Climate Assessment*, 441–461.
- Shanafield, M., Godsey, S., Datry, T., Hale, R., Zipper, S., Costigan, K., Krabbenhoft, C., Dodds, W., Zimmer, M., & Allen, D. (2020). Science gets up to speed on dry rivers. *Eos*, 101. <https://par.nsf.gov/biblio/10252961>

- Simon, A., & Collison, A. J. (2002). Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. *Earth Surface Processes and Landforms*, 27(5), 527–546. <https://doi.org/10.1002/esp.325>
- Snelder, T. H., Datry, T., Lamouroux, N., Larned, S. T., Sauquet, E., Pella, H., & Catalogne, C. (2013). Regionalization of patterns of flow intermittence from gauging station records. *Hydrology and Earth System Sciences*, 17(7), 2685–2699.
- Stagliano, D. M., & Whiles, M. R. (2002). Macroinvertebrate Production and Trophic Structure in a Tallgrass Prairie Headwater Stream. *Journal of the North American Benthological Society*, 21(1), 97–113. <https://doi.org/10.2307/1468303>
- Starks, P. J., & Moriasi, D. N. (2017). Impact of Eastern redcedar encroachment on stream discharge in the North Canadian River basin. *Journal of Soil and Water Conservation*, 72(1), 12–25.
- Stevens, C. J., Lind, E. M., Hautier, Y., Harpole, W. S., Borer, E. T., Hobbie, S., Seabloom, E. W., Ladwig, L., Bakker, J. D., Chu, C., Collins, S., Davies, K. F., Firn, J., Hillebrand, H., Pierre, K. J. L., MacDougall, A., Melbourne, B., McCulley, R. L., Morgan, J., ... Wragg, P. D. (2015). Anthropogenic nitrogen deposition predicts local grassland primary production worldwide. *Ecology*, 96(6), 1459–1465. <https://doi.org/10.1890/14-1902.1>
- Steward, D. R., Yang, X., Lauwo, S. Y., Staggenborg, S. A., Macpherson, G. L., & Welch, S. M. (2011). From precipitation to groundwater baseflow in a native prairie ecosystem: A regional study of the Konza LTER in the Flint Hills of Kansas, USA. *Hydrology and Earth System Sciences*, 15(10), 3181–3194.

- Stromberg, J. C., Lite, S. J., & Dixon, M. D. (2010). Effects of stream flow patterns on riparian vegetation of a semiarid river: Implications for a changing climate. *River Research and Applications*, 26(6), 712–729. <https://doi.org/10.1002/rra.1272>
- Sullivan, P. L., Stops, M. W., Macpherson, G. L., Li, L., Hirmas, D. R., & Dodds, W. K. (2019). How landscape heterogeneity governs stream water concentration-discharge behavior in carbonate terrains (Konza Prairie, USA). *Chemical Geology*, 527, 118989.
- Sullivan, P. L., Zhang, C., Behm, M., Zhang, F., & Macpherson, G. L. (2020). Toward a new conceptual model for groundwater flow in merokarst systems: Insights from multiple geophysical approaches. *Hydrological Processes*, 34(24), 4697–4711. <https://doi.org/10.1002/hyp.13898>
- Sweeney, B. W., & Newbold, J. D. (2014). Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: A literature review. *JAWRA Journal of the American Water Resources Association*, 50(3), 560–584. <https://doi.org/10.1111/jawr.12203>
- Swenson, L. J., Zipper, S., Peterson, D. M., Jones, C. N., Burgin, A. J., Seybold, E., Kirk, M. F., & Hatley, C. (2024). Changes in Water Age During Dry-Down of a Non-Perennial Stream. *Water Resources Research*, 60(1), e2023WR034623. <https://doi.org/10.1029/2023WR034623>
- Symstad, A. J., & Leis, S. A. (2017). Woody encroachment in northern Great Plains grasslands: Perceptions, actions, and needs. *Natural Areas Journal*, 37(1), 118–127.
- Tebaldi, C., Debeire, K., Eyring, V., Fischer, E., Fyfe, J., Friedlingstein, P., Knutti, R., Lowe, J., O'Neill, B., & Sanderson, B. (2021). Climate model projections from the scenario model

- intercomparison project (ScenarioMIP) of CMIP6. *Earth System Dynamics*, 12(1), 253–293.
- Tomer, M. D., Sadler, E. J., Lizotte, R. E., Bryant, R. B., Potter, T. L., Moore, M. T., & Locke, M. A. (2013). A decade of conservation effects assessment research by the USDA Agricultural Research Service. *Journal of Soil and Water Conservation*, 69(6), 365–373.
- Tomer, M. D., & Schilling, K. E. (2009). A simple approach to distinguish land-use and climate-change effects on watershed hydrology. *Journal of Hydrology*, 376(1–2), 24–33.
- Towne, E. G., Hartnett, D. C., & Cochran, R. C. (2005). Vegetation trends in tallgrass prairie from bison and cattle grazing. *Ecological Applications*, 15(5), 1550–1559.
<https://doi.org/10.1890/04-1958>
- Tramblay, Y., Rutkowska, A., Sauquet, E., Sefton, C., Laaha, G., Osuch, M., Albuquerque, T., Alves, M. H., Banasik, K., Beaufort, A., Brocca, L., Camici, S., Csabai, Z., Dakhlaoui, H., DeGirolamo, A. M., Dörflinger, G., Gallart, F., Gauster, T., Hanich, L., ... Datry, T. (2021). Trends in flow intermittence for European rivers. *Hydrological Sciences Journal*, 66(1), 37–49. <https://doi.org/10.1080/02626667.2020.1849708>
- Trimble, S. W., & Mendel, A. C. (1995). The cow as a geomorphic agent—A critical review. *Geomorphology*, 13(1–4), 233–253.
- Turner, C. L., Blair, J. M., Scharitz, R. J., & Neel, J. C. (1997). Soil N and plant responses to fire, topography, and supplemental N in tallgrass prairie. *Ecology*, 78(6), 1832–1843.
[https://doi.org/10.1890/0012-9658\(1997\)078%5B1832:SNAPRT%5D2.0.CO;2](https://doi.org/10.1890/0012-9658(1997)078%5B1832:SNAPRT%5D2.0.CO;2)
- USEPA: National Rivers and Streams Assessment 2023/24: Field Operations Manual – Wadeable. edn version 1. Edited by U.S. Environmental Protection Agency OoW. Washington, DC; 2022.

- Van Meter, K. J., Basu, N. B., & Van Cappellen, P. (2017). Two centuries of nitrogen dynamics: Legacy sources and sinks in the Mississippi and Susquehanna River Basins. *Global Biogeochemical Cycles*, 31(1), 2–23. <https://doi.org/10.1002/2016GB005498>
- Van Meter, K. J., Basu, N. B., Veenstra, J. J., & Burras, C. L. (2016). The nitrogen legacy: Emerging evidence of nitrogen accumulation in anthropogenic landscapes. *Environmental Research Letters*, 11(3), 035014.
- Vandermyde, J. M., & Whiles, M. R. (2015). Effects of experimental forest removal on macroinvertebrate production and functional structure in tallgrass prairie streams. *Freshwater Science*, 34(2), 519–534. <https://doi.org/10.1086/680442>
- Veatch, A. M., Dodds, W. K., & Skibbe, A. (2014). Fire and grazing influences on rates of riparian woody plant expansion along grassland streams. *PLoS One*, 9(9), e106922.
- Veatch, A. M., Dodds, W. K., & Skibbe, A. (2015). Correction: Fire and grazing influences on rates of riparian woody plant expansion along grassland streams. *PloS One*, 10(5), e0129409.
- Vercruysse, K., Grabowski, R. C., & Rickson, R. J. (2017). Suspended sediment transport dynamics in rivers: Multi-scale drivers of temporal variation. *Earth-Science Reviews*, 166, 38–52.
- Vero, S. E., Macpherson, G. L., Sullivan, P. L., Brookfield, A. E., Nippert, J. B., Kirk, M. F., Datta, S., & Kempton, P. (2018). Developing a Conceptual Framework of Landscape and Hydrology on Tallgrass Prairie: A Critical Zone Approach. *Vadose Zone Journal*, 17(1), 1–11. <https://doi.org/10.2136/vzj2017.03.0069>

- Vinton, M. A. (1999). Review of Grassland Dynamics: Long-Term Ecological Research in Tallgrass Prairie Edited by Alan K. Knapp, John M. Briggs, David C. Hartnett, and Scott L. Collins. *Great Plains Research: A Journal of Natural and Social Sciences*, 470.
- Volder, A., Briske, D. D., & Tjoelker, M. G. (2013). Climate warming and precipitation redistribution modify tree–grass interactions and tree species establishment in a warm-temperate savanna. *Global Change Biology*, 19(3), 843–857.
<https://doi.org/10.1111/gcb.12068>
- Von Schiller, D., Acuña, V., Graeber, D., Martí, E., Ribot, M., Sabater, S., Timoner, X., & Tockner, K. (2011). Contraction, fragmentation and expansion dynamics determine nutrient availability in a Mediterranean forest stream. *Aquatic Sciences*, 73(4), 485–497.
<https://doi.org/10.1007/s00027-011-0195-6>
- von Schiller, D., Bernal, S., Dahm, C. N., & Martí, E. (2017). Nutrient and organic matter dynamics in intermittent rivers and ephemeral streams. In *Intermittent rivers and ephemeral streams* (pp. 135–160). Elsevier.
<https://www.sciencedirect.com/science/article/pii/B9780128038352000061>
- Vu, H. M., Shanafield, M., & Batelaan, O. (2018). Flux dynamics at the groundwater-surface water interface in a tropical catchment. *Limnologica*, 68, 36–45.
- Walling, D. E. (2005). Tracing suspended sediment sources in catchments and river systems. *Science of the Total Environment*, 344(1–3), 159–184.
- Wan, S., Hui, D., & Luo, Y. (2001). Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: A meta-analysis. *Ecological Applications*, 11(5), 1349–1365.
[https://doi.org/10.1890/1051-0761\(2001\)011%5B1349:FEONPA%5D2.0.CO;2](https://doi.org/10.1890/1051-0761(2001)011%5B1349:FEONPA%5D2.0.CO;2)

- Wang, J., Xiao, X., Zhang, Y., Qin, Y., Doughty, R. B., Wu, X., Bajgain, R., & Du, L. (2018). Enhanced gross primary production and evapotranspiration in juniper-encroached grasslands. *Global Change Biology*, 24(12), 5655–5667.
<https://doi.org/10.1111/gcb.14441>
- Warix, S. R., Godsey, S. E., Lohse, K. A., & Hale, R. L. (2021). Influence of groundwater and topography on stream drying in semi-arid headwater streams. *Hydrological Processes*, 35(5), e14185. <https://doi.org/10.1002/hyp.14185>
- Webster, J. R., & Meyer, J. L. (1997). Organic matter budgets for streams: A synthesis. *Journal of the North American Benthological Society*, 16(1), 141–161.
- Wellen, C., Kamran-Disfani, A.-R., & Arhonditsis, G. B. (2015). Evaluation of the Current State of Distributed Watershed Nutrient Water Quality Modeling. *Environmental Science & Technology*, 49(6), 3278–3290. <https://doi.org/10.1021/es5049557>
- Wilcox, B. P., Basant, S., Olariu, H., & Leite, P. A. (2022). Ecohydrological connectivity: A unifying framework for understanding how woody plant encroachment alters the water cycle in drylands. *Frontiers in Environmental Science*, 10, 934535.
- Wilcox, B. P., Huang, Y., & Walker, J. W. (2008). Long-term trends in streamflow from semiarid rangelands: Uncovering drivers of change. *Global Change Biology*, 14(7), 1676–1689. <https://doi.org/10.1111/j.1365-2486.2008.01578.x>
- Wilcox, B. P., Owens, M. K., Dugas, W. A., Ueckert, D. N., & Hart, C. R. (2006). Shrubs, streamflow, and the paradox of scale. *Hydrological Processes*, 20(15), 3245–3259.
<https://doi.org/10.1002/hyp.6330>

- Williams, C. J., Pierson, F. B., Kormos, P. R., Al-Hamdan, O. Z., Nouwakpo, S. K., & Weltz, M. A. (2020). Vegetation, hydrologic, and erosion responses of sagebrush steppe 9 yr following mechanical tree removal. *Rangeland Ecology & Management*, 73(2), 234–246.
- Wine, M. L., & Zou, C. B. (2012). Long-term streamflow relations with riparian gallery forest expansion into tallgrass prairie in the Southern Great Plains, USA. *Forest Ecology and Management*, 266, 170–179.
- Wood, P. J., & Armitage, P. D. (1997). Biological effects of fine sediment in the lotic environment. *Environmental Management*, 21(2), 203–217.
- Wood, S. N. (2016). Just another Gibbs additive modeler: Interfacing JAGS and mgcv. *Journal of Statistical Software*, 75, 1–15.
- Zaimes, G. N., Schultz, R. C., & Isenhardt, T. M. (2019). Riparian land-use impacts on stream bank and gully erosion in agricultural watersheds: What we know and need to know. *Agriculture*, 9(5), 116. <https://doi.org/10.3390/agriculture9050116>
- Zhang, L., Dawes, W. R., & Walker, G. R. (2001). Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research*, 37(3), 701–708. <https://doi.org/10.1029/2000WR900325>
- Zhang, T., Xu, H., Ru, S.-H., & Su, D.-C. (2017). Distribution of phosphorus in soil profiles after continuous application of different fertilizers. *Huan Jing Ke Xue= Huanjing Kexue*, 38(12), 5247–5255.
- Zhao, K., Coco, G., Gong, Z., Darby, S. E., Lanzoni, S., Xu, F., Zhang, K., & Townend, I. (2022). A review on bank retreat: Mechanisms, observations, and modeling. *Reviews of Geophysics*, 60(2), e2021RG000761. <https://doi.org/10.1029/2021RG000761>

- Zimmer, M. A., Kaiser, K. E., Blaszcak, J. R., Zipper, S. C., Hammond, J. C., Fritz, K. M., Costigan, K. H., Hosen, J., Godsey, S. E., Allen, G. H., Kampf, S., Burrows, R. M., Krabbenhoft, C. A., Dodds, W., Hale, R., Olden, J. D., Shanafield, M., DelVecchia, A. G., Ward, A. S., ... Allen, D. C. (2020a). Zero or not? Causes and consequences of zero-flow stream gage readings. *WIREs Water*, 7(3), e1436. <https://doi.org/10.1002/wat2.1436>
- Zimmer, M. A., Kaiser, K. E., Blaszcak, J. R., Zipper, S. C., Hammond, J. C., Fritz, K. M., Costigan, K. H., Hosen, J., Godsey, S. E., Allen, G. H., Kampf, S., Burrows, R. M., Krabbenhoft, C. A., Dodds, W., Hale, R., Olden, J. D., Shanafield, M., DelVecchia, A. G., Ward, A. S., ... Allen, D. C. (2020b). Zero or not? Causes and consequences of zero-flow stream gage readings. *WIREs Water*, 7(3), e1436. <https://doi.org/10.1002/wat2.1436>
- Zipper, S. C., Hammond, J. C., Shanafield, M., Zimmer, M., Datry, T., Jones, C. N., Kaiser, K. E., Godsey, S. E., Burrows, R. M., & Blaszcak, J. R. (2021). Pervasive changes in stream intermittency across the United States. *Environmental Research Letters*, 16(8), 084033.
- Zipper, S., Popescu, I., Compare, K., Zhang, C., & Seybold, E. C. (2022). Alternative stable states and hydrological regime shifts in a large intermittent river. *Environmental Research Letters*, 17(7), 074005.
- Zipper, S., Wheeler, C. T., Peterson, D. M., Cook, S. C., Godsey, S. E., & Aho, K. (2025). STICr: An open-source package and workflow for stream temperature, intermittency, and conductivity (STIC) data. *Environmental Modelling & Software*, 106484.
- Zou, C. B., Turton, D. J., Will, R. E., Engle, D. M., & Fuhlendorf, S. D. (2014). Alteration of hydrological processes and streamflow with juniper (*Juniperus virginiana*) encroachment in a mesic grassland catchment: ALTERATION OF HYDROLOGICAL

- PROCESSES AFTER JUNIPER ENCROACHMENT. *Hydrological Processes*, 28(26), 6173–6182. <https://doi.org/10.1002/hyp.10102>
- Zou, C. B., Twidwell, D., Bielski, C. H., Fogarty, D. T., Mittelstet, A. R., Starks, P. J., Will, R. E., Zhong, Y., & Acharya, B. S. (2018). Impact of eastern redcedar proliferation on water resources in the Great Plains USA—current state of knowledge. *Water*, 10(12), 1768.
- Zweig, L. D., & Rabeni, C. F. (2001). Biomonitoring for deposited sediment using benthic invertebrates: A test on four Missouri streams. *Journal of the North American Benthological Society*, 20(4), 643–657.

Appendix A- Supplemental Information for Chapter 2

Supplementary Table 2.1: Regression analysis of USGS stream gage yearly mean discharge as a function of year.

Model		df	F	p
1	Regression	1	2.289	.139 ^b
	Residual	37		
	Total	38		

a. Dependent Variable: log10 discharge

b. Predictors: (Constant), Year

Supplementary Table 2.2: Regression analysis of USGS stream gage daily mean log10 discharge as a function of year

Model		df	F	p
1	Regression	1	413.761	.000 ^b
	Residual	14244		
	Total	14245		

a. Dependent Variable: log10 discharge

b. Predictors: (Constant), Year

Supplementary Table 2.3: Regression analysis of USGS stream gage number of flow days by year as the independent variables.

Model		df	F	p
1	Regression	1	8.973	.005 ^b
	Residual	37		
	Total	38		
a. Dependent Variable: Flow Days				
b. Predictors: (Constant), Year				

Supplementary Table 2.4: Offset the USGS daily log10 discharge data by the periods from 10 days up to 90 days to estimate the autocorrelation period. ** Correlation $p < 0.01$ (2-tailed). * Correlation $p < 0.05$ level (2-tailed). $n = 15349$

Number of days offset		Zero	Ten	Twenty	Thirty	Forty	Fifty	Sixty	Sixtyfive	Seventy	Seventyfive	Eighty	Eightyfive	Ninety
Zero	Pearson Correlation	1	.134*	.086**	.070**	.049**	.032*	.025*	.021*	.061**	.029**	.016*	.009	.007
	p (2-tailed)		.000	.000	.000	.000	.000	.002	.011	.000	.000	.047	.272	.380
Ten	Pearson Correlation	.134*	1	.157**	.089**	.064**	.045*	.030*	.043**	.030**	.019*	.069**	.051**	.012
	p (2-tailed)	.000		.000	.000	.000	.000	.000	.000	.000	.017	.000	.000	.133
Twenty	Pearson Correlation	.086*	.157*	1	.157**	.089**	.064*	.045*	.036**	.030**	.043**	.030**	.019*	.068**
	p (2-tailed)	.000	.000		.000	.000	.000	.000	.000	.000	.000	.000	.017	.000
Thirty	Pearson Correlation	.070*	.089*	.157**	1	.157**	.089*	.065*	.050**	.045**	.036**	.030**	.043**	.030**
	p (2-tailed)	.000	.000	.000		.000	.000	.000	.000	.000	.000	.000	.000	.000
Forty	Pearson Correlation	.049*	.064*	.089**	.157**	1	.156*	.089*	.065**	.065**	.050**	.045**	.036**	.030**
	p (2-tailed)	.000	.000	.000	.000		.000	.000	.000	.000	.000	.000	.000	.000
Fifty	Pearson Correlation	.032*	.045*	.064**	.089**	.156**	1	.157*	.111**	.089**	.065**	.065**	.050**	.045**
	p (2-tailed)	.000	.000	.000	.000	.000		.000	.000	.000	.000	.000	.000	.000

Sixty	Pearson Correlation	.025*	.030*	.045**	.065**	.089**	.157*	1	.200**	.157**	.112**	.089**	.065**	.065**
	p (2-tailed)	.002	.000	.000	.000	.000	.000		.000	.000	.000	.000	.000	.000
Sixtyfive	Pearson Correlation	.021*	.043*	.036**	.050**	.065**	.111*	.200*	1	.200**	.158**	.112**	.089**	.065**
	p (2-tailed)	.011	.000	.000	.000	.000	.000	.000		.000	.000	.000	.000	.000
Seventy	Pearson Correlation	.061*	.030*	.030**	.045**	.065**	.089*	.157*	.200**	1	.200**	.158**	.112**	.090**
	p (2-tailed)	.000	.000	.000	.000	.000	.000	.000	.000		.000	.000	.000	.000
SeventyFive	Pearson Correlation	.029*	.019*	.043**	.036**	.050**	.065*	.112*	.158**	.200**	1	.200**	.158**	.112**
	p (2-tailed)	.000	.017	.000	.000	.000	.000	.000	.000	.000		.000	.000	.000
Eighty	Pearson Correlation	.016*	.069*	.030**	.030**	.045**	.065*	.089*	.112**	.158**	.200**	1	.200**	.158**
	p (2-tailed)	.047	.000	.000	.000	.000	.000	.000	.000	.000	.000		.000	.000
EightyFive	Pearson Correlation	.009	.051*	.019*	.043**	.036**	.050*	.065*	.089**	.112**	.158**	.200**	1	.200**
	p (2-tailed)	.272	.000	.017	.000	.000	.000	.000	.000	.000	.000	.000		.000
Ninety	Pearson Correlation	.007	.012	.068**	.030**	.030**	.045*	.065*	.065**	.090**	.112**	.158**	.200**	1
	p (2-tailed)	.380	.133	.000	.000	.000	.000	.000	.000	.000	.000	.000	.000	

Supplementary Table 2.5: Precipitation at Konza Prairie Biological Station as a function of the season as an independent categorical variable and year as a continuous variable. R Squared = .554 (Adjusted R Squared = .532)

Source	df	F	p
Corrected Model	7	26.211	.000
Intercept	1	.875	.351
Season	3	.331	.803
Year	1	1.204	.274
Season * Year	3	.378	.769
Error	148		
Total	156		
Corrected Total	155		

Supplementary Table 2.6: Temperature at Konza Prairie Biological Station as a function of the season as an independent categorical variable and year as a continuous variable. R Squared = .976 (Adjusted R Squared = .975)

Source	df	F	p
Corrected Model	7	866.342	.000
Intercept	1	6.367	.013
Season	3	.812	.489
Year	1	9.990	.002
Season * Year	3	.597	.618
Error	148		
Total	156		
Corrected Total	155		

Supplementary Table 2.7: ANCOVA for log10 discharge at LTER watersheds (N4D, N2B, and N1B) as a function of Season (categorical) and site (N4D, N2B, and N1B) with continuous variables of temperature, precipitation, and riparian cover. R Squared = .585 (Adjusted R Squared = .567)

Source	df	F	p
Corrected Model	17	32.339	.000
Intercept	1	207.567	.000
Season	3	16.750	.000
Sites	2	6.529	.002
Temperature	1	1.146	.285
Precipitation	1	72.410	.000
Riparian Cover	1	20.943	.000
Sites * Temperature	2	.132	.877
Sites * Precipitation	2	.683	.506
Sites * Riparian Cover	2	7.903	.000
Sites * Temperature * Precipitation *Riparian Cover	3	.306	.821
Error	390		
Total	408		
Corrected Total	407		

Supplementary Table 2.8: ANCOVA for the number of flow days at LTER watersheds (N4D, N2B, and N1B) as a function of Season (categorical) and site (N4D, N2B, and N1B) with continuous variables of temperature, precipitation, and riparian cover. R Squared = .486 (Adjusted R Squared = .463)

Source	df	F	p
Corrected Model	17	21.652	.000
Intercept	1	.197	.658
Season	3	22.641	.000
Sites	2	1.179	.309
Temperature	1	.224	.636
Precipitation	1	50.228	.000
Riparian Cover	1	2.931	.088
Sites * Temperature	2	.096	.909
Sites * Precipitation	2	.372	.689
Sites * Riparian Cover	2	.231	.794
Sites * Temperature * Precipitation * Riparian Cover	3	.809	.489
Error	390		
Total	408		
Corrected Total	407		

Supplementary Table 2.9: Pearson Correlation for USGS Kings Creek log10 discharge and associated variables. from 1982 to 2020. **. Indicates that correlation is significant at the 0.01 level (2-tailed) where *. Indicates correlation is significant at the 0.05 level (2-tailed). Mean daily temperature (Temp), log10 discharge, Number of flow days (flow days), and total precipitation (Precip) within each season. The riparian cover is the average of the three LTER watersheds. The discharge and riparian cover were log-transformed. N= 39.

		Year	Discharge	Flow Days	Temperature	Precipitation	Riparian Cover
Year	Pearson Correlation	1	-.241	-.442**	.415**	.181	1.000**
	p (2-tailed)		.139	.005	.009	.271	.000
Discharge	Pearson Correlation	-.241	1	.556**	-.299	.732**	-.241
	p (2-tailed)	.139		.000	.064	.000	.139
Flow Days	Pearson Correlation	-.442**	.556**	1	-.316*	.233	-.442**
	p (2-tailed)	.005	.000		.050	.154	.005
Temperature	Pearson Correlation	.415**	-.299	-.316*	1	-.173	.415**
	p (2-tailed)	.009	.064	.050		.293	.009
Precipitation	Pearson Correlation	.181	.732**	.233	-.173	1	.181
	p (2-tailed)	.271	.000	.154	.293		.271
Riparian Cover	Pearson Correlation	1.000*	-.241	-.442**	.415**	.181	1
	p (2-tailed)	.000	.139	.005	.009	.271	

Supplementary Table 2.10: Pearson Correlation for LTER watershed N4D log10 discharge and associated variables. from 1985 to 2020. **. indicates that correlation is significant at the 0.01 level (2-tailed) where *. indicates correlation is significant at the 0.05 level (2-tailed). Variables are similar to Table 2. N=34

		Year	Discharge	Flow days	Temperature	Precipitation	Riparian Cover
Year	Pearson Correlation	1	.041	.049	.285	.308	.933**
	p (2-tailed)		.818	.783	.102	.076	.000
Discharge	Pearson Correlation	.041	1	.715**	-.299	.704**	-.002
	p (2-tailed)	.818		.000	.085	.000	.992
Flow days	Pearson Correlation	.049	.715**	1	-.039	.540**	.054
	p (2-tailed)	.783	.000		.829	.001	.763
Temperature	Pearson Correlation	.285	-.299	-.039	1	-.147	.239
	p (2-tailed)	.102	.085	.829		.407	.173
Precipitation	Pearson Correlation	.308	.704**	.540**	-.147	1	.233
	p (2-tailed)	.076	.000	.001	.407		.185
Riparian Cover	Pearson Correlation	.933**	-.002	.054	.239	.233	1
	p (2-tailed)	.000	.992	.763	.173	.185	

Supplementary Table 2.11: Pearson Correlation for LTER watershed N2B log10 discharge and associated variables. from 1987 to 2020. **. indicates that correlation is significant at the 0.01 level (2-tailed) where *. indicates correlation is significant at the 0.05 level (2-tailed). Variables are the same to table 2. N=34

		Year	Discharge	Flow days	Precipitation	Temperature	Riparian Cover
Year	Pearson Correlation	1	-.085	.059	.308	.285	1.000**
	p (2-tailed)		.634	.741	.076	.102	.000
Discharge	Pearson Correlation	-.085	1	.704**	.636**	-.006	-.085
	p (2-tailed)	.634		.000	.000	.974	.634
Flow days	Pearson Correlation	.059	.704**	1	.701**	-.270	.059
	p (2-tailed)	.741	.000		.000	.123	.741
Precipitation	Pearson Correlation	.308	.636**	.701**	1	-.147	.308
	p (2-tailed)	.076	.000	.000		.407	.076
Temperature	Pearson Correlation	.285	-.006	-.270	-.147	1	.285
	p (2-tailed)	.102	.974	.123	.407		.102
Riparian Cover	Pearson Correlation	1.000**	-.085	.059	.308	.285	1
	p (2-tailed)	.000	.634	.741	.076	.102	

Supplementary Table 2.12: Pearson Correlation for LTER watershed N1B log10 discharge and associated variables. from 1987 to 2020. **. indicates that the correlation is significant at the 0.01 level (2-tailed). Variables are same as table 2. N=34

		Year	Discharge	Flow days	Temperature	Precipitation	Riparian cover
Year	Pearson Correlation	1	-.274	.055	.285	.308	1.000**
	p (2-tailed)		.117	.759	.102	.076	.000
Discharge	Pearson Correlation	-.274	1	.181	.029	.302	-.274
	p (2-tailed)	.117		.306	.869	.082	.117
Flow days	Pearson Correlation	.055	.181	1	-.184	.577**	.055
	p (2-tailed)	.759	.306		.298	.000	.759
Temperature	Pearson Correlation	.285	.029	-.184	1	-.147	.285
	p (2-tailed)	.102	.869	.298		.407	.102
Precipitation	Pearson Correlation	.308	.302	.577**	-.147	1	.308
	p (2-tailed)	.076	.082	.000	.407		.076
Riparian Cover	Pearson Correlation	1.000**	-.274	.055	.285	.308	1
	p (2-tailed)	.000	.117	.759	.102	.076	

Supplementary Table 2.13: ANCOVA for discharge at USGS Kings Creek gage as a function of Season (categorical) with continuous variables of temperature, precipitation, and riparian cover. R Squared = .586 (Adjusted R Squared = .528).

Source	df	F	p
Corrected Model	19	10.124	.000
Intercept	1	4.925	.028
Season	3	.589	.623
Temperature	1	.137	.712
Precipitation	1	10.264	.002
Riparian Cover	1	1.840	.177
Season * Temperature	3	.639	.591
Season * Precipitation	3	1.312	.273
Season * Riparian Cover	3	.938	.424
Season * Temperature * Precipitation * Riparian Cover	4	1.085	.367
Error	136		
Total	156		
Corrected Total	155		

Supplementary Table 2.14: ANCOVA for the number of flow days at USGS Kings Creek gage as a function of Season (categorical) with continuous variables of temperature, precipitation, and riparian cover. R Squared = .412 (Adjusted R Squared = .330)

Source	df	F	p
Corrected Model	19	5.022	.000
Intercept	1	3.603	.060
Season	3	.155	.927
Temperature	1	1.524	.219
Precipitation	1	.196	.659
Riparian Cover	1	3.983	.048
Season * Temperature	3	.432	.730
Season * Precipitation	3	.566	.639
Season * Riparian Cover	3	.325	.807
Season * Temperature * Precipitation * Riparian Cover	4	.477	.752
Error	136		
Total	156		
Corrected Total	155		

Supplementary Table 2.15: ANCOVA for the whole watershed grass ET (mm/season) at USGS Kings Creek gage as a function of Season (categorical) with continuous variables of year. R Squared = .961 (Adjusted R Squared = .957).

Source	df	F	p
Corrected Model	7	633.794	.000
Intercept	1	3.363	.071
Season	3	2.562	.061
Year	1	4.245	.043
Season * Year	3	2.696	.052
Error	72		
Total	80		
Corrected Total	79		

Supplementary Table 2.16: ANCOVA for the whole watershed grass ET and riparian area woody cover (mm/season) at USGS Kings Creek gage as a function of Season (categorical) with continuous variables of year. R Squared = .962 (Adjusted R Squared = .958).

Source	df	F	p
Corrected Model	7	662.283	.000
Intercept	1	24.465	.000
Season	3	6.616	.001
Year	1	26.801	.000
Season * Year	3	7.096	.000
Error	72		
Total	80		
Corrected Total	79		

Supplementary Table 2.17: ANCOVA for whole watershed grass ET including whole watershed total prop wood (mm/season) at USGS Kings Creek gage as a function of Season (categorical) with continuous variables of year. R Squared = .962 (Adjusted R Squared = .958).

Source	df	F	p
Corrected Model	7	655.177	.000
Intercept	1	23.652	.000
Season	3	6.440	.001
Year	1	25.937	.000
Season * Year	3	6.908	.000
Error	72		
Total	80		
Corrected Total	79		

Appendix B- Supplemental Information for Chapter 3

Supplementary Table 3.1. Chronological record of management activities, infrastructure modifications, and experimental treatments at Konza Prairie Biological Station (1987-2018). Data collected by James Guinnip.

Year	Month	Activity Type	Description	Area/Details	Notes
1987	October	Bison reintroduction	30 bison introduced from Fort Riley	450 ha	Initial bison area, portion of current grazing area
1988	-	Waste management	Lab building completed with bathroom/showe r	Single facility	Plans from June 1987, septic east of building
1992	-	Bison management	Bison area expanded	450 → 984 ha	Now covers south fork of Kings Creek
1995	-	Waste management	Began organic matter removal from bison corral	Bison corral	Every 3-4 years, stored in "bone yard" south of staff house
1998	-	Prairie restoration	Agricultural plot restored	3.29 ha	Former livestock facilities, now experimental streams/ag wells
1998	July	Waste management	New septic for staff house	1 full bath	Lateral field north of house, sized for future firehall/shop
1998	-	Waste management	Two manufactured cottages installed	2 cottages, 1 bath each	West side of creek, demolished old doublewide trailer
1999	-	Infrastructure	Firehall/shop constructed	3/4 bath	Connected to staff house septic system
2000	-	Prairie restoration	Agricultural plot restored	3.2 ha	Current Schoolyard LTER plot between cottages and HQ
2005	-	Prairie restoration	Agricultural plot restored	0.35 ha	Small plot behind refueling station

2008	-	Waste management	Barn renovation with new septic	1500 gal tank	2 urinals, 4 commodes, 4 sinks; lateral field north of staff house
2008	-	Waste management	Hulbert Center new septic system	3 bathrooms total	Full, 3/4, and 1/2 bath; lateral field north of barn
2008	-	Waste management	Risers installed on septic tanks	All tanks except lab	Ground level access for inspection
2010	-	Prairie restoration	Two agricultural plots restored	1.84 ha total	Around 2005 plot + smaller plot near chronosequence shed
2011	-	Infrastructure	Staff house moved to new basement	Existing septic	Septic tank pumped during move
2012	-	Infrastructure	Large Green House installed	2 full + 1 3/4 bath	Lateral field north of cottages, geo/thermal wells on north end
2012	-	Prairie restoration	Agricultural plot restored	0.25 ha	First plot in restoration chronosequence
2014	-	Prairie restoration	Agricultural plot restored	0.25 ha	Continuation of restoration chronosequence
2016	-	Prairie restoration	Agricultural plot restored	0.25 ha	Continuation of restoration chronosequence
2017	February	Waste management	All septic tanks pumped	All except lab building	Major maintenance event
2018	-	Prairie restoration	Agricultural plot restored	0.25 ha	Continuation of restoration chronosequence

Supplementary Table 3. 2. MANCOVA of the yearly mean of NO_3^- , TP, and SRP by Year and Year Since Burn for N1B, N2B, and N4D watersheds. All the continuous data were log-transformed. Data from 2011-2013 were removed from all the analyses for the N2B watershed due to the riparian removal experiment in this watershed.

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	NO_3^-	9.6a	5	1.91	34.58	0.00
	SRP	.8b	5	0.16	2.20	0.06
	TP	.29c	5	0.06	1.58	0.18
Intercept	NO_3^-	1.72	1	1.72	31.10	0.00
	SRP	0.15	1	0.15	2.09	0.15
	TP	0.07	1	0.07	1.91	0.17
Year	NO_3^-	1.80	1	1.80	32.58	0.00
	SRP	0.17	1	0.17	2.28	0.14
	TP	0.09	1	0.09	2.45	0.12
Sites	NO_3^-	4.10	2	2.05	37.09	0.00
	SRP	0.06	2	0.03	0.41	0.67
	TP	0.07	2	0.04	1.01	0.37
Year Since Burn	NO_3^-	0.17	1	0.17	3.09	0.08
	SRP	0.23	1	0.23	3.15	0.08
	TP	0.02	1	0.02	0.56	0.46
Sites * Year Since Burn	NO_3^-	0.06	1	0.06	1.13	0.29
	SRP	0.24	1	0.24	3.30	0.07
	TP	0.00	1	0.00	0.01	0.92
Error	NO_3^-	3.87	70	0.055		
	SRP	5.11	70	0.073		
	TP	2.53	70	0.036		
Total	NO_3^-	67.18	76			
	SRP	23.30	76			
	TP	75.05	76			
Corrected Total	NO_3^-	13.42	75			
	SRP	5.91	75			
	TP	2.81	75			
a. R Squared = .712 (Adjusted R Squared = .691)						
b. R Squared = .136 (Adjusted R Squared = .074)						
c. R Squared = .102 (Adjusted R Squared = .037)						

Supplementary Table 3. 3. MANCOVA on the yearly mean of NO₃⁻, SRP, and TP by Year and Sites for EDLR, HIKX, and HOKN watersheds. All the data were log-transformed.

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	NO ₃ ⁻	12.12a	5	2.42	86.38	0.00
	SRP	3.72b	5	0.74	19.25	0.00
	TP	2.276c	5	0.46	7.73	0.00
Intercept	NO ₃ ⁻	0.60	1	0.60	21.39	0.00
	SRP	0.77	1	0.77	19.90	0.00
	TP	0.33	1	0.33	5.55	0.02
Year	NO ₃ ⁻	0.51	1	0.51	18.07	0.00
	SRP	0.72	1	0.72	18.77	0.00
	TP	0.28	1	0.28	4.77	0.03
Sites	NO ₃ ⁻	0.58	2	0.29	10.39	0.00
	SRP	0.49	2	0.25	6.35	0.00
	TP	0.02	2	0.01	0.15	0.86
Sites * Year	NO ₃ ⁻	0.57	2	0.28	10.08	0.00
	SRP	0.48	2	0.24	6.24	0.00
	TP	0.02	2	0.01	0.15	0.86
Error	NO ₃ ⁻	2.08	74	0.03		
	SRP	2.86	74	0.04		
	TP	4.36	74	0.06		
Total	NO ₃ ⁻	275.36	80			
	SRP	48.68	80			
	TP	121.70	80			
Corrected Total	NO ₃ ⁻	14.19	79			
	SRP	6.57	79			
	TP	6.63	79			
a. R Squared = .854 (Adjusted R Squared = .844)						
b. R Squared = .565 (Adjusted R Squared = .536)						
c. R Squared = .343 (Adjusted R Squared = .299)						

Supplementary Table 3. 4. MANCOVA on the yearly means of NO₃⁻, SRP, and TP by Year and Sites for groundwater wells sites ALC-A5, and K01A-P5 watersheds. All the data were log-transformed.

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	NO ₃ ⁻	13.43a	3	4.477	101.95	0.00
	SRP	.39b	3	0.130	1.85	0.15
	TP	.32c	3	0.105	1.10	0.36
Intercept	NO ₃ ⁻	1.23	1	1.225	27.90	0.00
	SRP	0.01	1	0.008	0.12	0.73
	TP	0.05	1	0.047	0.49	0.49
Year	NO ₃ ⁻	1.10	1	1.098	25.01	0.00
	SRP	0.01	1	0.005	0.07	0.80
	TP	0.07	1	0.065	0.68	0.42
Sites	NO ₃ ⁻	0.14	1	0.137	3.13	0.09
	SRP	0.10	1	0.096	1.37	0.25
	TP	0.05	1	0.050	0.52	0.47
Sites * Year	NO ₃ ⁻	0.13	1	0.129	2.93	0.10
	SRP	0.10	1	0.095	1.35	0.25
	TP	0.05	1	0.049	0.52	0.48
Error	NO ₃ ⁻	1.76	40	0.044		
	SRP	2.82	40	0.070		
	TP	3.83	40	0.096		
Total	NO ₃ ⁻	318.74	44			
	SRP	54.76	44			
	TP	129.30	44			
Corrected Total	NO ₃ ⁻	15.19	43			
	SRP	3.21	43			
	TP	4.14	43			
a. R Squared = .884 (Adjusted R Squared = .876)						
b. R Squared = .122 (Adjusted R Squared = .056)						
c. R Squared = .076 (Adjusted R Squared = .007)						

Supplementary Table 3.5. A paired sample t-test to estimate the true difference between SFKC (Bison grazed) and NFKC (Ungrazed) for NO_3^- , SRP, and TP. NFKC nutrients data were denoted with N and SFKC data were denoted with S.

Paired Samples Correlations				
		N	Correlation	Sig.
Pair 1	NO3-S & NO3-N	1044	-.02	.91
Pair 2	SRP-S & SRP-N	1044	.79	.00
Pair 3	TP-S & TP-N	1044	.80	.00

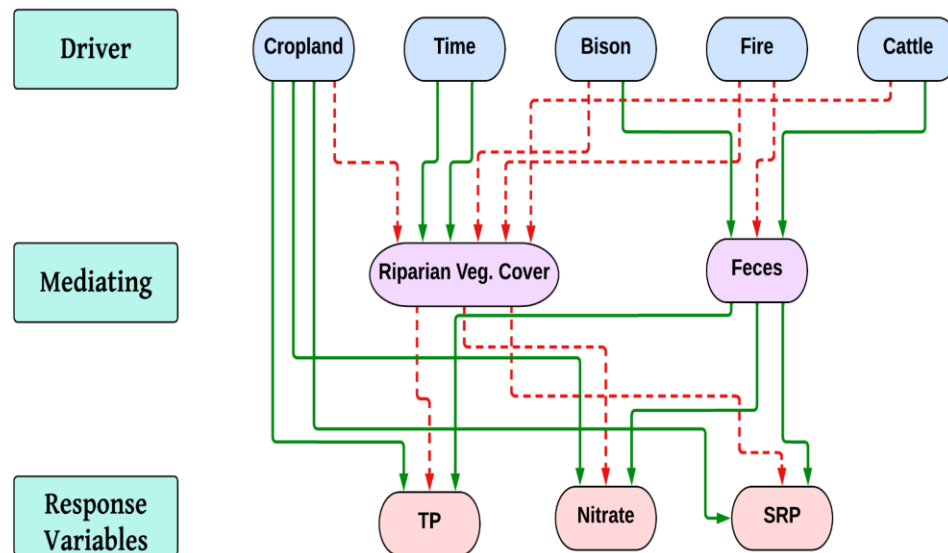
		Paired Differences					t	d f	Sig. (2-tailed)
		Mean	Std. Deviation	Std. Error Mean	95% Confidence Interval of the Difference				
					Lower	Upper			
Pair 1	NO ₃ ⁻ -S -NO ₃ ⁻ -N	0.44	0.31	0.06	0.31	0.57	7.05	23	0.00
Pair 2	SRP-S - SRP-N	0.05	0.14	0.03	-0.01	0.12	1.80	22	0.09
Pair 3	TP-S - TP-N	0.12	0.10	0.02	0.07	0.16	5.56	23	0.00

Supplementary Table 3.6. An independent samples t-test conducted on the water samples obtained from the Shane Creek watershed to discern distinctions between samples derived from watersheds characterized by the presence and absence of cattle for the 365 days of the year from 2009 to 2022. Notably, during the initial two years of the sampling period, cattle were absent, whereas, in the subsequent years, the watershed had cattle present for roughly half of the year.

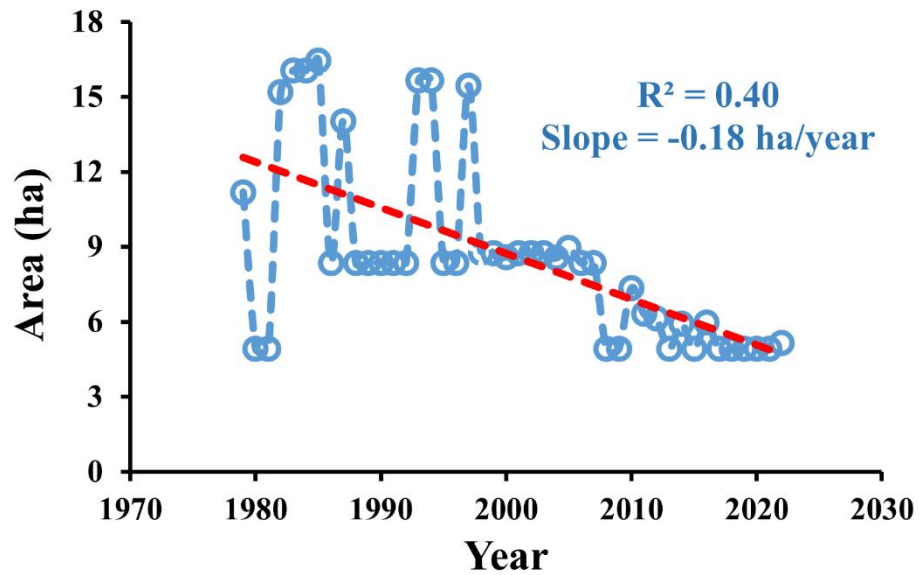
	Cattle present	N	Mean	Std. Deviation	Std. Error Mean
NO₃⁻	Absent	388	0.70	0.39	0.02
	Present	307	0.93	0.36	0.02
SRP	Absent	385	0.63	0.35	0.02
	Present	274	0.61	0.38	0.02
TP	Absent	370	1.18	0.35	0.02
	Present	289	1.22	0.36	0.02

Levene's test for equality of variances and t-test for equality of means										
		F	Sig.	t	df	p (2-tailed)	Mean Difference	Std. Error Difference	95% Confidence Interval of the Difference	
									Lower	Upper
NO₃⁻	Equal variances assumed	1.48	0.23	-8.10	693.0	0.00	-0.23	0.03	-0.29	-0.18
	Equal variances not assumed			-8.18	676.8	0.00	-0.23	0.03	-0.29	-0.18
SRP	Equal variances assumed	6.63	0.01	0.68	657.0	0.49	0.02	0.03	-0.04	0.08
	Equal variances not assumed			0.68	567.4	0.50	0.02	0.03	-0.04	0.08
TP	Equal variances assumed	6.44	0.01	-1.29	657.0	0.20	-0.04	0.03	-0.09	0.02
	Equal variances not assumed			-1.29	616.5	0.20	-0.04	0.03	-0.09	0.02

Supplementary Figure 3. 1. Conceptual model of our study, which includes our initial hypotheses. The green solid arrows represent the positive, and the red dashed arrows represent the negative relations of drivers with the mediating and response variables: TP = total phosphorus, and SRP = soluble reactive phosphorus.

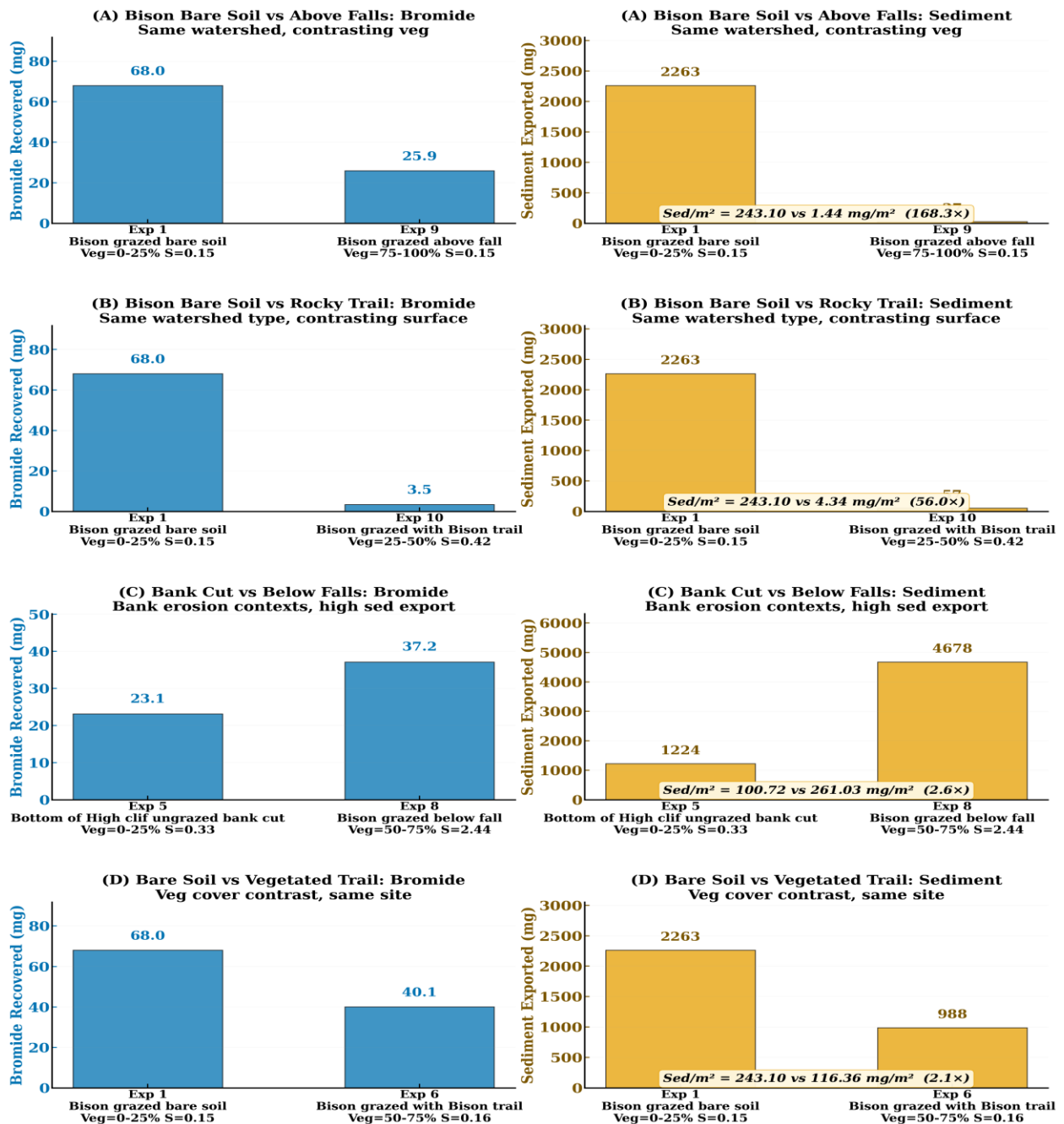


Supplementary Figure 3. 2. Decline in fertilized cropland area within the Kings Creek watershed at Konza Prairie Biological Station (1978-2022). Blue circles connected by dashed lines show annual cropland area; solid line shows linear regression of cropland area on year: $\text{Cropland area (ha)} = 364.7 - 0.18(\text{year} - 1978)$, $R^2 = 0.40$, $p < 0.001$. Cropland area decreased 75% over the study period as fields were sequentially restored to native prairie.

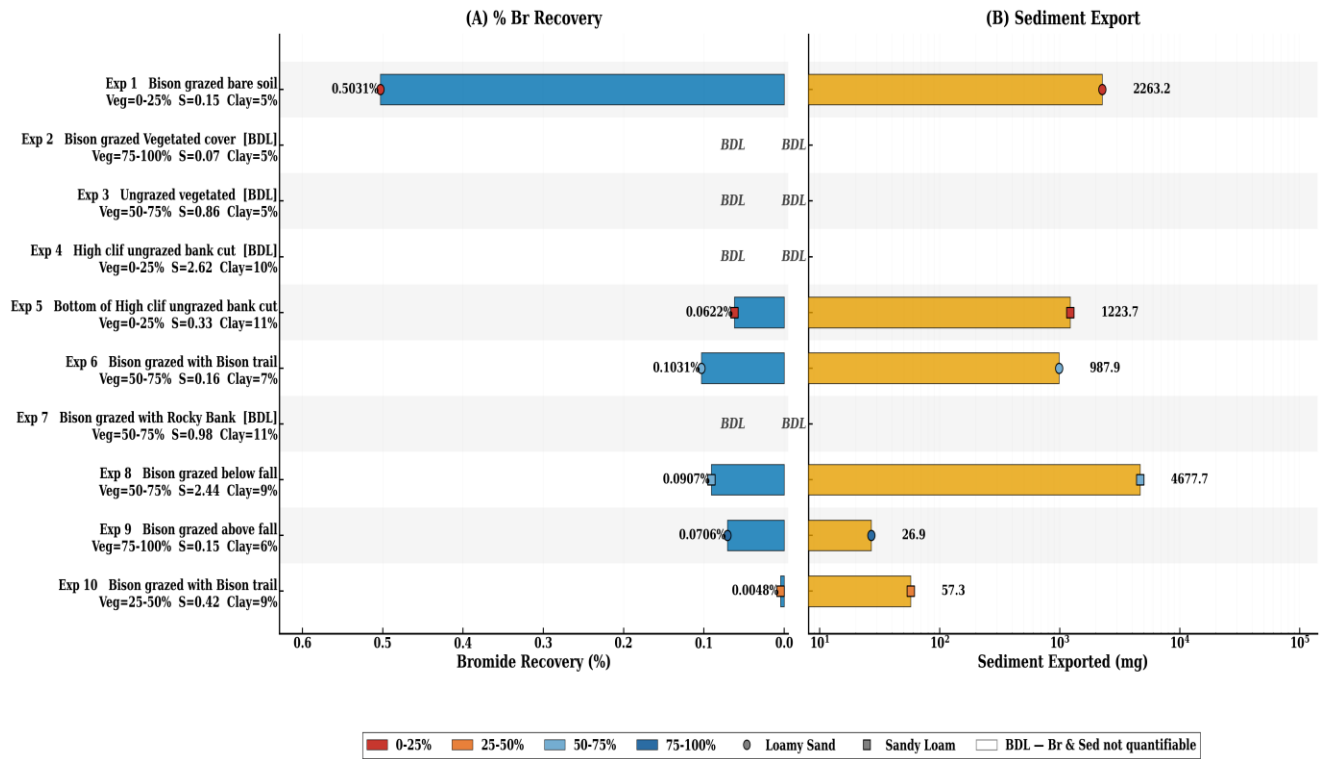


Appendix C- Supplemental Information for Chapter 4

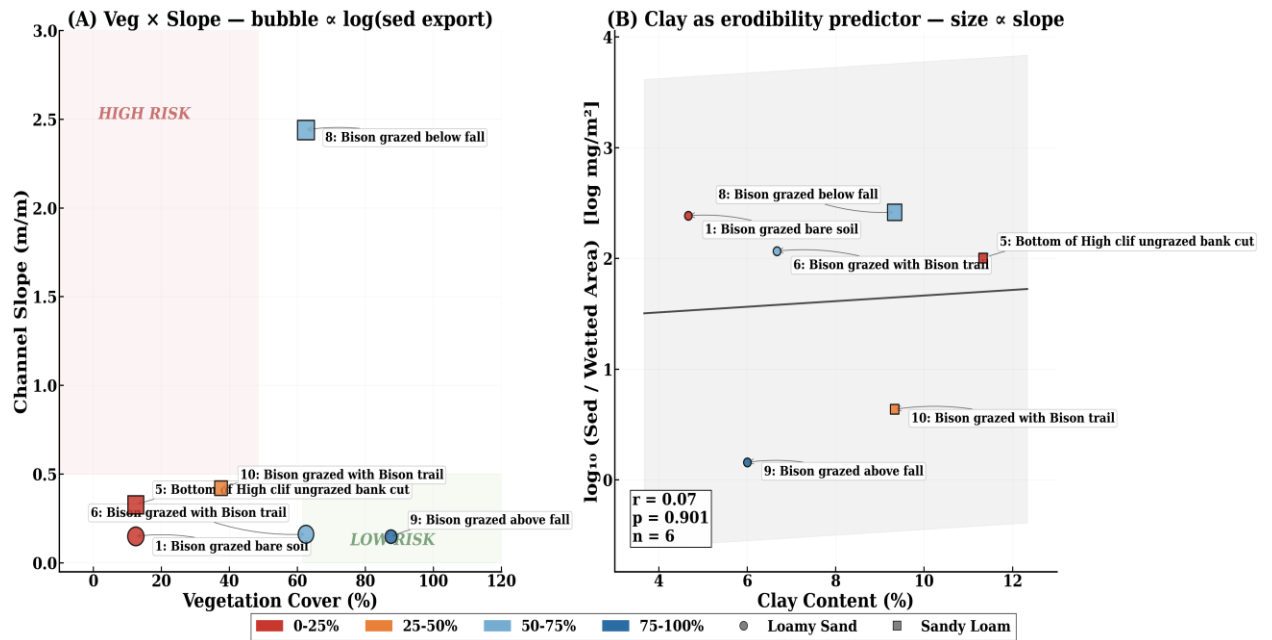
Supplementary Figure 4.1. Paired site contrasts isolating individual controls on riparian sediment delivery, expressed as bromide recovery (left, blue bars) and sediment export (right, gold bars). Pair A: vegetation cover at matched slope within bison-grazed terrain (Exp 1 vs. Exp 9; $168.3\times$ Sed/ m^2). Pair B: bare soil vs. partially vegetated bison trail (Exp 1 vs. Exp 10; $56.0\times$ Sed/ m^2). Pair C: cliff-base bank cut vs. below-fall knickpoint, contrasting bank-erosion contexts (Exp 5 vs. Exp 8; $2.6\times$ Sed/ m^2). Pair D: bare bison soil vs. vegetated bison trail at near-identical slope (Exp 1 vs. Exp 6; $2.1\times$ Sed/ m^2). Sed/ m^2 fold differences annotated on each sediment panel.



Supplementary Figure 4.2. Bromide recovery (A; % of applied) and sediment export (B; mg, log scale) across ten rainfall simulation experiments. Four BDL experiments (2, 3, 4, 7) are shown as empty rows. Point color denotes vegetation cover class; symbol shape distinguishes loamy sand (circles) from sandy loam (squares).



Supplementary Figure 4.3. Joint controls of vegetation cover, slope, and soil texture on riparian sediment delivery. (A) Vegetation cover (%) versus channel slope (m/m) for the six experiments with detectable transport; bubble size is proportional to $\log_{10}(\text{sediment export})$. Red shading delineates the high-erodibility zone (low cover, steep slope); green shading delineates the low-erodibility zone. (B) Clay content (%) as a predictor of areal erodibility ($\log_{10} \text{Sed}/\text{m}^2$), with point size proportional to slope (Spearman $r = 0.07$, $p = 0.901$, $n = 6$).



(A) Sediment Export

Exp	Site Description	Total Sediment Exported (mg)
Exp 1	Bison grazed bare soil	2263.2
Exp 6	Bison grazed with Bison trail	987.9
Exp 9	Bison grazed above fall	26.9
Exp 10	Bison grazed with Bison trail	57.3
Exp 8	Bison grazed below fall	4677.7
Exp 5	Bottom of High cliff ungrazed bank cut	1223.7
Exp 2	Bison grazed Vegetated cover [BDL]	BDL
Exp 3	Ungrazed vegetated [BDL]	BDL
Exp 4	High cliff ungrazed bank cut [BDL]	BDL
Exp 7	Bison grazed with Rocky Bank [BDL]	BDL

(B) Areal Erodibility Index

Exp	Site Description	Sed per Wetted Area (mg/m²)
Exp 1	Bison grazed bare soil	243.10
Exp 6	Bison grazed with Bison trail	116.36
Exp 9	Bison grazed above fall	1.44
Exp 10	Bison grazed with Bison trail	4.34
Exp 8	Bison grazed below fall	261.03
Exp 5	Bottom of High cliff ungrazed bank cut	100.72
Exp 2	Bison grazed Vegetated cover [BDL]	BDL
Exp 3	Ungrazed vegetated [BDL]	BDL
Exp 4	High cliff ungrazed bank cut [BDL]	BDL
Exp 7	Bison grazed with Rocky Bank [BDL]	BDL

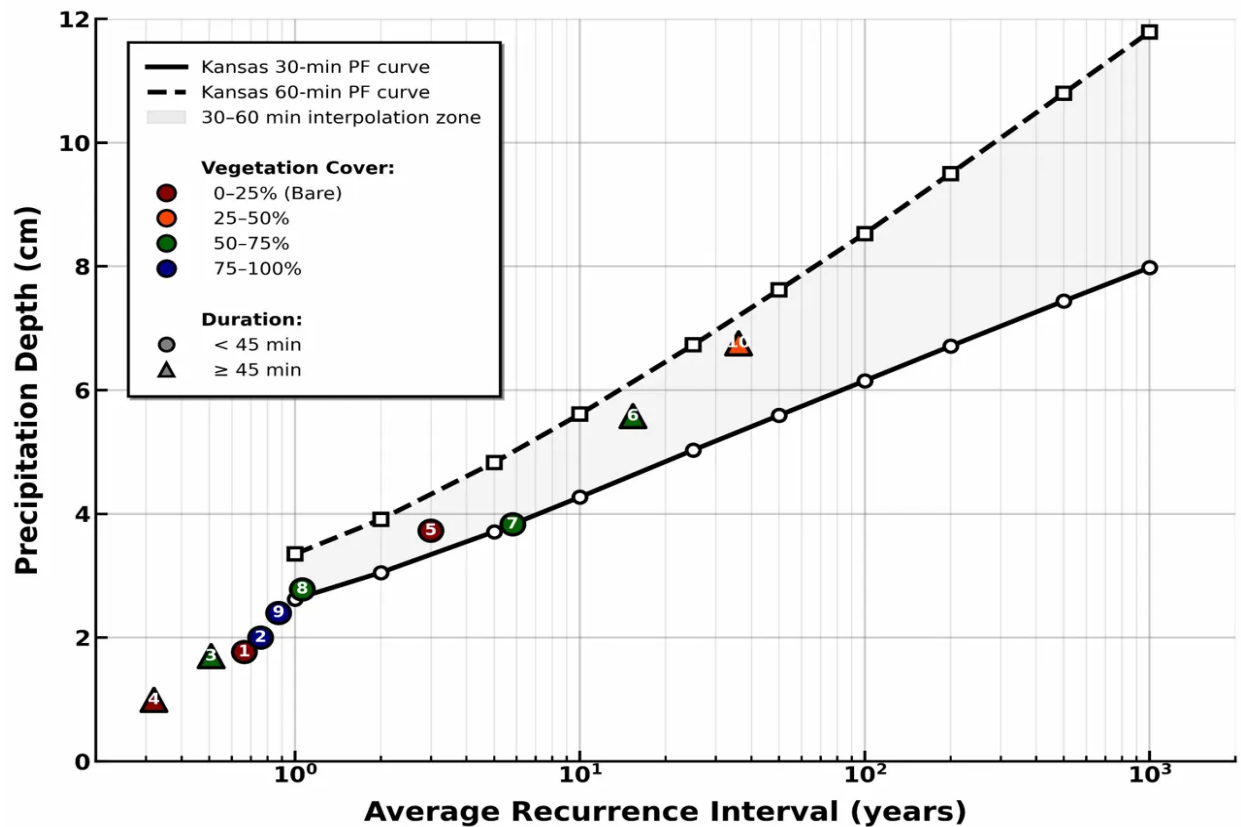
(C) Hydraulic Connectivity

Exp	Site Description	Br Recovery (%)
Exp 1	Bison grazed bare soil	0.503
Exp 6	Bison grazed with Bison trail	0.103
Exp 9	Bison grazed above fall	0.071
Exp 10	Bison grazed with Bison trail	0.005
Exp 8	Bison grazed below fall	0.091
Exp 5	Bottom of High cliff ungrazed bank cut	0.062
Exp 2	Bison grazed Vegetated cover [BDL]	BDL
Exp 3	Ungrazed vegetated [BDL]	BDL
Exp 4	High cliff ungrazed bank cut [BDL]	BDL
Exp 7	Bison grazed with Rocky Bank [BDL]	BDL

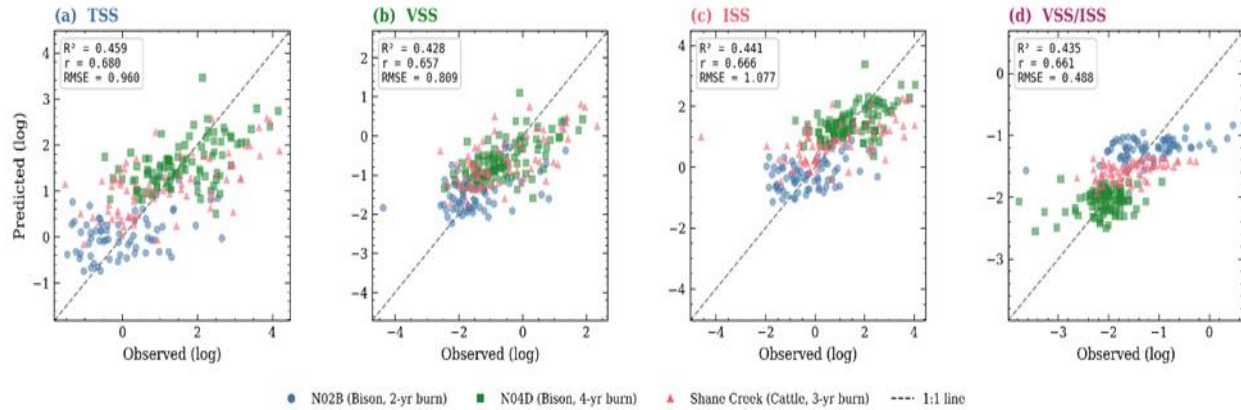
Legend:

- Bison bare/trail
- Bison geomorphic
- Bank cut/cliff
- BDL (excluded)
- † DL-based min. estimate (BDL exps)

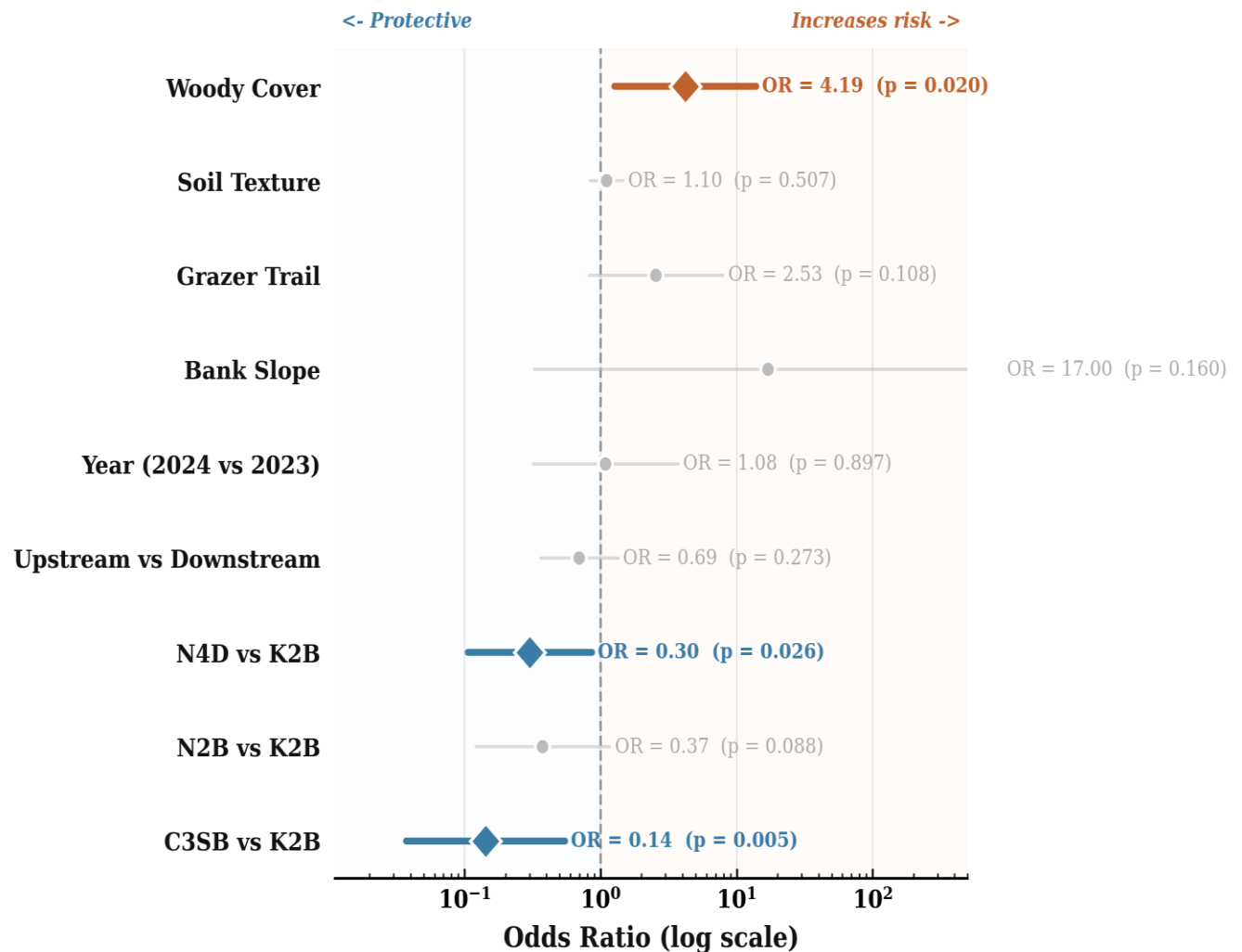
Supplementary Figure 4.5. Precipitation recurrence context for rainfall simulation experiments. Simulated precipitation depths (cm) plotted against average recurrence intervals (years) derived from Kansas precipitation-frequency curves for 30-minute (solid line) and 60-minute (dashed line) durations. Gray shading indicates the 30–60 minute interpolation zone. Numbered symbols represent individual experiments; symbol color indicates vegetation cover class (red = 0–25%, orange = 25–50%, green = 50–75%, blue = 75–100%); symbol shape indicates experiment duration (circles = < 45 min, triangles = ≥ 45 min).



Supplementary Figure 4.6. Generalized additive model diagnostics for monthly suspended sediment concentrations. Top row (a–d): residual-versus-fitted value plots for log-transformed TSS, VSS, ISS, and VSS/ISS ratio, with smoothed trend lines (black) and colored points by watershed (blue = N02B, green = N04D, pink = Shane Creek). Bottom row (e–h): predicted-versus-observed plots with 1:1 reference lines (dashed). R^2 , Pearson r , and RMSE values are annotated in each panel. $n = 235$ (234 for VSS/ISS).



Supplementary Figure 4.7. Forest plot of odds ratios from generalized estimating equation (GEE) logistic regression predicting bank cut occurrence across four Konza Prairie watersheds (N04D, N02B, K2B, C3SB). Points represent odds ratios on a log scale; horizontal lines indicate 95% confidence intervals. Orange-filled diamonds denote significant predictors ($p < 0.05$); blue-filled diamonds denote significant watershed contrasts; gray circles denote non-significant predictors. The reference watershed for site contrasts is K2B (ungrazed, 2-year burn). Woody cover presence was the strongest significant predictor of bank erosion ($OR = 4.19$, $p = 0.020$).



Supplementary Figure 4.8. Spearman rank correlation matrix among streambank geomorphic variables measured across four Konza Prairie watersheds. Variables: bank slope, soil texture score, bank cut occurrence (binary), bare soil score, grazer trail presence (binary), and canopy cover presence (binary). Cell values show Spearman ρ coefficients; color intensity is proportional to correlation magnitude (red = positive, blue = negative). All pairwise correlations were weak ($|\rho| \leq 0.19$), indicating that the measured bank attributes are largely independent predictors of erosion susceptibility.

Spearman Rank Correlations

Slope	1.00	-0.17	0.19	0.18	-0.09	-0.15
Soil Tex.	-0.17	1.00	0.12	-0.10	-0.12	0.05
Bank Cut	0.19	0.12	1.00	0.07	0.07	-0.01
Bare Soil	0.18	-0.10	0.07	1.00	-0.17	-0.15
Grazer Trail	-0.09	-0.12	0.07	-0.17	1.00	0.02
Canopy	-0.15	0.05	-0.01	-0.15	0.02	1.00
	Slope	Soil Tex.	Bank Cut	Bare Soil	Grazer Trail	Canopy

Supplementary Table 4.1: Standardized path coefficients from the structural equation model (SEM) for suspended sediment dynamics across Konza Prairie watersheds. All paths are significant at $p < 0.001$ (***). VSS = volatile suspended solids; ISS = inorganic suspended solids.

Path	β	SE	z	p	Sig.
Precipitation → Discharge	0.162	0.000	109560110217.40	0.0000	***
Watershed Area → Discharge	0.823	0.000	691412370731.50	0.0000	***
Days Since Burn → Discharge	-0.027	0.000	-21109491118.21	0.0000	***
Discharge → VSS	-0.788	0.000	-4151230.22	0.0000	***
Precipitation → VSS	0.318	0.000	3117390.87	0.0000	***
Watershed Area → VSS	0.931	0.000	4751333.97	0.0000	***
Days Since Burn → VSS	0.154	0.000	1299150.14	0.0000	***
Discharge → ISS	-0.542	0.000	-2119476.31	0.0000	***
Precipitation → ISS	0.311	0.000	2262206.94	0.0000	***
Watershed Area → ISS	0.727	0.000	2753111.33	0.0000	***
Days Since Burn → ISS	0.296	0.000	1858121.78	0.0000	***
Discharge → VSS/ISS	-0.094	0.000	-813481.21	0.0000	***
Precipitation → VSS/ISS	-0.167	0.000	-2692522.11	0.0000	***
Watershed Area → VSS/ISS	-0.081	0.000	-681026.23	0.0000	***
Days Since Burn → VSS/ISS	-0.405	0.000	-5631415.68	0.0000	***