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Riparian post-fire response: factors influencing vegetation recovery and channel stability

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Natural Resources and Environmental Science

By

Camie M. Dencker

Sherman Swanson, Ph.D. / Thesis Advisor

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Sherman R. Swanson, Ph.D., Advisor

Jeanne C. Chambers, Ph.D., Committee Member

Matt L. Forister, Ph.D., Graduate School Representative

David W. Zeh, Ph.D., Dean, Graduate School

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Abstract

The BLM Emergency Stability and Rehabilitation Handbook suggests a rest from grazing following wildfire for two years or until objectives are met for the recovery of vegetation and key processes. However, both land users and managers dispute this policy because of economic, ecological and social implications and little supporting scientific evidence. Riparian areas are of particular concern because of concentrated grazing-use and importance for wildlife, humans, livestock production, and hydrologic functions. This research sought to quantify rates of change and variation post-fire in riparian condition and response across channel and watershed attributes, fire severity, and pre- and post-fire grazing-use. To quantify stream recovery, we used Multiple Indicator Monitoring (MIM) of Stream Channels and Streamside Vegetation (Burton et al. 2011) because it is becoming a standard method for quantifying if riparian objectives are met. We monitored 23 streams burned in 2012 wildfires on public lands in Nevada, focusing on reaches of greatest management concern, such as those classified as functional at-risk, or with threatened species habitat or aspen stands. Watershed and stream channel characteristics were quantified in ArcGIS with the exception of stream gradient, which was measured at site. We used MIM variables that had been measured over two years as indicators of riparian condition: greenline-to-greenline width, greenline plant composition, woody species cover and height, and streambank stability and vegetation cover. Winward greenline stability and wetland indicator rating were calculated from greenline plant composition and used as metrics of ecosystem functionality. Riparian species composition was most related to variables associated with watershed position, such as substrate size, gradient, and elevation. Wetland obligate species were found at sites with high sinuosity or bank stability and within watersheds characterized by high percentage volcanic bed material. Bank cover was associated with higher position in the watershed, Winward greenline stability rating, and streambank stability. Banks were more stable with increased bank cover and decreased percent fine substrate, stream gradient, and post-fire grazing-use. Over the two-year study, bank stability decreased from 2014 to 2015 with increased post-fire grazing duration at sites with higher percent fine substrates. Bank stability, species richness, and woody species cover and height class increased with duration of recovery periods and decreased with continuous, hot season grazing-use (July-September) prior to the fires, from 2006-2012. Woody species height increased with riparian width and recovery after grazing during the growing season and decreased with stream gradient and high burn severity. Sites lower in the watershed were grazed for longer duration with shorter recovery periods during the growing season and fewer years of rest. Lower-position sites also had the greatest percent fine substrate and lowest bank cover, making them more unstable. Two-year grazing deferment may not be adequate for recovery of riparian functionality at lower watershed sites if streambank cover and stability are compromised. Continued monitoring is necessary to ascertain the required bank cover and time for recovery for these lower reaches to be resilient to the pressures of post-fire livestock grazing.

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Introduction

Over the past few decades, the extent of fires on public rangelands has increased in the Western United States (Dennison et al. 2014, Dillon et al. 2011, Westerling et al. 2014,, Whisenant 1990). Following wildfire, the Burned Area Emergency Stabilization and Rehabilitation (ESR) Handbook (BLM 2007) mandates a deferment from grazing to allow short-term rehabilitation objectives to be met for burned area stabilization. This includes recovery of vegetation and stabilization of soils to prevent erosion. This is particularly important in riparian areas, where stability can moderate the devastating effects of episodic floods (Prichard et al. 1998). The ESR handbook suggests that native vegetation may require 2-years or more for reestablishment. Based on these recommendations, the Bureau of Land Management (BLM) has customarily followed a 2-year grazing deferral.

However, the implications of this policy and lack of supporting scientific research have many rangeland managers and permittees questioning the necessity of this policy. Grazing allotment closures associated with wildfire can severely affect local economies, with millions of revenue dollars lost through the livestock industry (Harris et al. 2002). In contrast, wildlife proponents, many of them ranchers and hunters, and environmentalists think longer periods of rest are better. Mule deer and other game animals depend on riparian areas for water, forage, protection and nutrients (Kauffman and Krueger 1984). Western Nevada streams and riparian areas are critical habitat for several at risk species, including Lahontan cutthroat trout, a federally listed (threatened) species under the Endangered Species Act (Dunham et al. 1999, USFWS 1994). Furthermore, riparian areas provide forage for livestock, quality water and recreation for humans, and support ecosystem functions (Gregory et al. 1991, Naiman et al. 1993, Dickard et al. 2015).

Riparian areas in the Great Basin differ significantly in resilience to disturbance (Miller and Germanoski 2005; Chambers et al. 2005). In Northern Nevada, Kozlowski et al. (2010) studied forty streams following wildfires. Most streams responded favorably to wildfire, with increases in both the presence and extent of hydrophilic vegetation, and improved stream bank stability and structure and little evidence suggesting fire resulted in stream degradation. Many streams showed improvement, but whether this resulted from the effects of fire or the changes in land management could not be ascertained. It should also be noted, the years following fires had below average precipitation, resulting in low flows not large enough in most locations to result in the expected damage had there been a flooding event.

Increased peak flows and sediment supply after fire from loss of upland vegetation can cause considerable degradation to riparian areas including erosion, sedimentation, and vegetation loss or burial (Germanoski and Miller 1995). This may result in altered channel morphology, floodplain characteristic like hydrologic function and plant succession, (Dwire and Kauffman 2003, Kozlowski et al. 2010, Moody and Martin 2001, Montgomery and Buffington 1998, Myers and Swanson 1996). Incised channels may be even more susceptible to deterioration because of unprotected banks exposed to erosion and lack of floodplain access to dissipate flood energy (Myers and Swanson 1996). Conversely, Wyman et al. (2006) describes high functioning riparian areas as having inherent resiliency as a result of adaptation to dynamic water regimes – flooding and drought.

Many hydrophilic plant species have evolved physiologies to tolerate frequent flood disturbance (Corenblit et al. 2009a, Dwire and Kauffman 2003, Naiman et al. 1993, Swanson et al. 2015). Adaptive mechanisms, like rhizomatous roots and re-sprouting root crowns allow for quick plant regeneration following above ground vegetation removal. These mechanisms can facilitate survival of riparian vegetation after disturbance including rapid post-fire recovery. An intact, productive riparian vegetation community stabilizes channel geomorphology and sediment transport, and maintains hydrologic function (Beschta and Platts 1986, Hession et al. 2003, Tabacchi et al. 1998).

Riparian vegetation reinforces streambanks by increasing soil strength and dissipating stream energy (Micheli and Kirchner 2002). Many hydric graminoids have rhizomatous roots that quickly colonize new sediment deposits and form thick, dense mats that reinforce bank material (Wyman et al. 2006). Roots of woody species anchor banks and resist high-energy flows (Gregory et al. 1991, Wyman et al. 2006). Riparian trees contribute woody debris to the stream channel, which dissipates stream energy, traps sediment and aids in channel formation (Beschta and Platts 1986, Montgomery and Buffington 1998). Roughness of above ground vegetation slows water movement and allows sediment deposition and nutrient capture (Corenblit et al. 2009b, Wyman et al. 2006). The multiple roles of colonizing and especially stabilizing riparian vegetation are well described by Dickard et al. (2015).

Vegetation removal or loss as a result of disturbance can result in bank destabilization (Belsky et al. 1999, Kauffman and Krueger 1984, Trimble and Mendel 1995). When disturbance effects of livestock grazing, fire, and hydrologic characteristics were examined on stream condition in Northern Nevada, grazing was found to be the most impactful by Dalldorf et al., (2013) who studied the same burned streams as Kozlowski et al. (2010) and an additional forty unburned streams over the same period. Banks became more unstable and channels widened with increased intensity and duration of use (Dalldorf et al. 2013). There was no statistically significant difference in burned and unburned stream riparian condition (Dalldorf et al. 2013). This suggests grazing management, as opposed to fire, has the most potential to elicit change in stream physical attributes. However, grazing and fire effects do not act independently.

Livestock grazing combined with ungulate browsing can have profound effects on woody growth and has been found to limit willow reproduction (Brookshire et al. 2002). After a fire, growing tissue of re-spouting shrubs is highly exposed and most above ground biomass is initially at browse height, making it particularly vulnerable to herbivory stress (Dwire et al. 2006, Mills 1983). Dwire et al. (2006) found that combined ungulate and livestock grazing following a fire suppressed crown area and volume development, and decreased shrub height for many common riparian shrub species. Virtually all burned areas are exposed to some herbivory and degree of growth repression may depend on the intensity, timing, and duration of use (Swanson et al. 2015).

Reaches composed of non-cohesive, fine-grained alluvium are susceptible to mass wasting and bank shearing in basins with high erosive capabilities (Germanoski and Miller 2004), so may be more dependent on vegetation to maintain bank integrity (Abernethy and Rutherfurd 1997, Gurnell 2014, Swanson et al. 2015). Once destabilization occurs, it may be difficult for vegetation to reestablish until banks stabilize (Corenblit et al. 2007). Channel incision is the downcutting of a stream channel into the valley alluvium from increased stream power relative to sediment load (Schumm 1979, Simon and Rinaldi 2006) and riparian functions that dissipate stream energy (Dickard et al. 2015). It often results in the lowering of the water table, eventual loss of riparian vegetation on terraced banks, and a decrease in riparian extent within the terraced walls of the incision (Jewett et al. 2004).

Initiation of channel incision and exacerbation of channel destabilization has been attributed to both natural and anthropogenic disturbance, such as climate change, wildfire, and land management practices (Germanoski and Miller 1995, Germanoski and Miller 2004). It is poorly understood how the combination of fire and varying degrees of livestock grazing effects influence post-fire condition, vegetation community response, and speed of recovery. Post-fire response probably differs among vegetation species in relation to local geomorphology and hydrology, and physical attributes that influence water availability, which may vary by watershed position and attributes.

Sensitivity to destabilization varies by watershed lithology, shape, and size. Watershed basins in the Great Basin underlain by volcanic bedrock are associated with high peak flows, short lag times, and shorter duration high flows, which results in greater stream power and more unstable channels (Germanoski and Miller 2004). Large, steep basins characterized with high hypsometric integrals, area-ruggedness, and stream power can carry large amounts of water quickly through the channel and are thought to be more sensitive to channel incision (Germonski and Miller 2004). Channels in elongated, narrow valleys with prograding alluvial fans are thought to be more sensitive to incision once the fan has been breached and knick points migrate up the channel (Germanoski and Miller 2004).

To quantify stream condition and change over time, we used *Multiple Indicator Monitoring of Stream Channels and Streamside Vegetation* (MIM) (Burton et al. 2011). This method was developed to monitor the impacts of livestock grazing and management decisions in an efficient, effective, and objective way. The US Bureau of Land Management implemented MIM to build upon qualitative methods and standardize quantitative riparian monitoring across regions. It focuses measurements on the greenline, "the first perennial vegetation that forms a lineal grouping of community types on or near the water's edge (Burton et al. 2011)," which also includes embedded rock, or anchored wood (Winward 2000). The greenline is the critical zone for maintaining bank stability and channel form and is highly stressed as an important focus of the Emergency Stabilization and Rehabilitation handbook (BLM 2007), Riparian Proper Functioning Condition Assessment (Prichard 1998, Dickard et al. 2015) and riparian grazing management guidelines (Wyman et al. 2006, Swanson et al. 2015). Notably, these methods do not consider the characteristics of the watershed or channel.

The objectives of this study were to:

- Quantify the rate of recovery for streams after wildfire using long-term riparian MIM indicators.
- Investigate how response differs in relation to season, duration, and rotation of grazing prior to the fire.

- Determine the influence of watershed position and characteristics on post-fire vegetation and streambank stability.
- Examine how post-fire grazing strategy impacts short-term riparian condition.

Methods

Study area

The study included 23 perennial and intermittent streams located on public land in the Great Basin region of Nevada. The Great Basin is characterized by North-South fault block ranges that drain internally into closed basins (Minsall et al. 1989). Precipitation occurs largely during winter months (Mock 1996), with highest peak annual flows occurring from March to June from snowmelt (Cayan 1996, Germanoski and Miller 2004). The rest of the year stream flows are low and sustained by groundwater, except when high precipitation events cause flood pulses (Jewett et al. 2004, USGS 2016). Streams typically occur within steep narrow valleys in the mountains but lessen in gradient as valleys open up into the basin (Chambers et al. 2004).

In the Great Basin, channel morphology, hydrology, sediment transport, vegetation community composition vary greatly within and among watersheds as a result of the complex mountain topography and geology (Chambers et al. 2004, Engelhardt et al. 2011, Engelhardt et al. 2015, Jewett et al. 2004). Across our study sites in the Trout Creek, Bilk Creek, Montana, Santa Rosa, Independence, Jarbridge, Roberts, and Schell Creek Mountains, the most common woody species were *Salix* L. species, *Populus tremuloides Michx,* and *Rosa woodsii* Lindl.. Dominant forbs included *Urtica dioica* L., *Epilobium ciliatum* Raf., *Veronica americana* Schwein. ex Benth. and *Achillea millefolium* L.. Common graminoids included *Poa* L., *Agrostis* L., *Carex* L., and *Juncus* L. species. Water year precipitation was 85% of average the year following the fire, 88% of average two years after and 82% in the third year (Lamance Creek SNOTEL station) (USDA-NRCS & NWCC 2016).

Data collection

On each stream burned in 2011 we selected reaches of greatest management concern, which included those rated as functional-at-risk using Proper Function Condition assessment (Prichard et al. 1998) or valued as wildlife habitat (e.g. Aspen stands). In the absence of other clear prioritizing criteria, a low gradient reach was selected, because of their increased sensitivity to management decisions (Montgomery and Buffington 1998, Winward 2000, Wyman et al. 2006, Swanson 2015, Dickard et al. 2015).

Plots were randomly established in accordance with Multiple Indicator Monitoring for Stream Channels and Streamside Vegetation (MIM). A designated monitoring area included 80 plots along 110 meters of stream length. Plots were placed 2.75 meters apart, with 40 plots per bank. Monitoring occurred during summer low flow for ease of accessibility and to ensure ease of plant identification.

Indicators of long term change - greenline composition by species, streambank stability and cover, woody species height class, and greenline to greenline width were measured in accordance to the MIM handbook (Burton et al. 2011). Greenline species composition was determined using a 40 cm by 50 cm quadrat placed with the longest side on the edge of the greenline. Greenline is the line of perennial vegetation, rocks, or embedded wood parallel to the waterline, commonly on the edge of the floodplain or bench above the water's edge (Burton et al. 2011). It is often continuous but can be discontinuous on sandbars and areas of new vegetation colonization (Burton et al. 2011). Metrics calculated from greenline composition included percent relative cover by species, wetland rating, and Winward stability rating (1-10).

Relative cover included perennial species comprising more than 10 percent of the total vegetative cover, and embedded rock and anchored wood greater than 15 cm diameter were measured in the field. Composition of the understory and overstory were calculated separately, and include any woody individual rooted in or overhanging a plot. When no greenline cover existed within 6 m of the water line, the plot was considered to have no greenline in place of percent cover and given an NA value.

Site wetland rating is the weighted average wetland indicator rating of plants present at the site. Species were classified as wetland obligate, facultative wetland, facultative, facultative upland, or upland based on the National Wetland Plant List wetland indicator status (U.S. Fish and Wildlife Service 1993, Lichvar et al. 2014, U.S. Army Corps of Engineers 2014). Wetland indicator ratings were converted to numerical values (upland=0, facultative upland=25, facultative= 50, facultative wetland=75, and obligate wetland=100), then weighted by plot species composition. Similarly, greenline stability was calculated by weighting Winward's (2000) species stability rating by plot species composition. The heights of each woody species overhanging the plot are estimated and grouped into height classes, and then expressed as a percentage across the site. Perennial species richness, evenness, and Simpson's diversity index were calculated for every site using perennial species composition (Simpson, 1949).

Greenline-to-greenline width is the average distance across the stream from one greenline to the other on the opposite bank. It is an indicator of change, usually widening with stream degradation or narrowing with recovery, but potential width at each site is relative to stream discharge. Vegetative islands with at least 25% foliar cover were subtracted from the distance, while non-vegetated islands were included. **S**treambank stability is measured on the streambank, not necessarily at the greenline and is calculated as the percentage of plots categorized as stable versus having evidence of erosion, sloughing, fracturing or slumping. Streambank cover is the percent of these plots covered by at least 25% vegetation, embedded rock or anchored wood.

Grazing-use variables were calculated using records of actual use and billing statements provided by the Bureau of Land Management for pastures corresponding to nineteen study streams. Data were collected for six years prior to the fire and three years after. Pre-fire grazing metrics included average days of continuous use, total days of recovery during the growing season over six-year period, mean days of recovery within the growing season before and after grazing-use (Thornton et al. 2014). Post-fire metrics included days of use between monitoring years (2014 and 2015) and total days grazed three years after fire. Growing season was calculated for each year of grazing data as the time between 6 consecutive days of minimum temperatures at or below freezing using Daymet data (Thornton et al. 2014).

Stream gradient was measured using a surveyor's rod and laser level along 100 meters of stream length. Sinuosity was calculated by digitizing the stream reach using NAIP aerial imagery (2013) in Geographic Information Systems (GIS), and then by dividing the visible channel length within the reach by the straight line distance between the two end points.

Substrate size was measured according to methods outlined in the MIM handbook (Burton et al. 2011). Ten substrate samples were measured across the width of the stream at each of twenty plots. The intermediate axis of each particle was measured in millimeters. In addition, fraction of fine substrate (less than 6 mm) and D50, the particle size that 50% of samples are equal or smaller than were calculated for each site.

Riparian extent, stream length above site, percent of bed rock types within the watershed, elevation, and hypsometric integral were calculated to determine site position in the watershed and watershed scale hydrographic influences.

Riparian extent, the average width of riparian vegetation lateral to the stream channel, was calculated in ArcGIS (ESRI 2011) using NAIP 2011 satellite imagery (USDA 2015). The reach was divided into 10 equal segments using the "divide line by length" add-in for ArcGIS (Jones

2012) and 10 lines perpendicular to the channel were created using the station line tool for ArcGIS (Dilts 2015). Lines were spliced to the edge of riparian vegetation and averaged for the reach.

Stream length above the site was calculated using stream flowline spatial data from the USGS National Hydrography Dataset (USGS 2013). The geometry calculator tool in ArcMap (ESRI 2011) was used to calculate the perennial stream length in kilometers contributing to each site location.

Contributing area of the watershed above a pour point (site location) was calculated using the hydrology tools within ArcGIS 10.0 software (ESRI 2011). Percent bedrock type within the watershed was derived using the geology geodatabase layer in ArcGIS 10.0 software (ESRI 2011). Percent intrusive igneous, volcanic, sedimentary, and siliciclastic were calculated for the watershed above each site. Bedrock lithology was included because it has been associated with hydrologic regimes and riparian vegetation composition (Engelhardt et al. 2011).

Percent watershed burned and percent of each burn severity class was calculated using Monitoring Trends of Burn Severity (MTBS) (USFS-USGS MTBS Project 2009) classification polygons, based on of pre- and post-fire NDVI values using Landsat imagery. MTBS includes only fires greater than 1000 acres.

Elevation was determined using a handheld GPS device. Elevation is related to position in the watershed, which is related to precipitation and temperature regimes, and evaporative loss. It has been negatively correlated to channel width and depth (Engelhardt et al. 2015).

Hypsometric integral (HI) is a dimensionless ratio that captures the stage in the erosional process of a watershed basin. It is calculated as “percentage area under a dimensionless curve produced as the ratio of h/H and a/A where h = elevation, H = watershed relief, a = planimetric area above h, and A = planimetric watershed area above the site (Engelhardt et al. 2011)”. A higher HI represents a less eroded basin with greater surface area in the higher elevations relative to the elevation range of the watershed basin. A lower HI is a more eroded basin with greater surface area in the lower elevations. At equivalent elevations, watersheds with higher HI are characterized by greater snow accumulation and retention, which is expected to have higher discharge events during spring run-off thus, is more prone to flood disturbance events (Germanoski and Miller 2004). Aspen stands (*Populus* spp.) are associated with basins with high HI, which has been attributed to their reliance on relatively high water availability supplied during spring floods and ground water recharges from snow pack (Engelhardt et al. 2011).

Statistical analysis

Non-metric multidimensional scaling (NMDS) of species community data was used investigate grouping of species by community and how those communities potentially related to disturbance, watershed and channel characteristics. Data were manipulated prior to analysis to remove excess noise, improve multivariate normality, and model performance. The Sorensen (Bray-Curtis) distance measure requires data to be proportional; so prior to analysis, species percent compositions were converted to proportions. Rare species (N=1) were removed from the species matrix. To correct for positive skewness in the community data, the dataset was transformed using an arcsine square root. Environmental variables were examined in histograms and scatterplots. Errors were detected and corrected and any variables not having adequate representation across sites were removed (e.g. % metamorphic rock with N=1). The remaining variables were assessed for correlation using a Pearson's correlation matrix. Highly correlated variables were removed, selecting for the variable of greatest interest or ecological importance. Highly skewed variables were identified by visual examination of histograms and calculated values for skewness and kurtosis. All variables with skewness values greater than 2 were transformed to improve normality of distributions. Finally, all variables were standardized to make variables measured in different units comparable.

Stress values from solution with 1-15 dimensions were compared to choose optimum dimensionality. NMDS was run five times using a random start to confirm consistent results.

Two-way cluster analysis was used to group sites by similarity in watershed position and bedrock geology (Appendix 1). Relative Euclidean distance with Ward's linkage method was used because it allows for negative values and is equivalent to Sorensen with flexible beta (McCune and Grace 2002). Sites were categorized into low, mid, and upper watershed position based on hypsometric integral, elevation, riparian extent, stream gradient, contributing stream length, and bankfull width values (Appendix 1). Sites were grouped by bedrock geology using percent rock volcanic, sedimentary, and intrusive igneous within the watershed. Sites were also categorized by their grazing strategy, determined by the duration, timing and variation in use. Three grazing strategies stood out as distinct, consistent hot season use (n = 7), variation in timing of use but included hot season (n = 8), and spring-only use (n = 4).

Graphical interpretation and one-way analyses of variance (ANOVA) were used to determine if MIM greenline indices (Winward greenline stability, wetland rating, bank stability, woody height class, percent woody cover, and bank cover), grazing-use, and fraction of fines varied by watershed position, grazing-use, and monitoring year. The assumption of homogeneity of variance was tested using Levene's test. If null hypothesis of equal variances failed to be rejected, then Welch's correction was used. For ANOVAs, a p-value of < 0.10 was accepted because of small in-group sample size (~ 10 samples per group). Tukey HSD for multiple comparisons test was used to compare group means after ANOVA.

Multiple regression models, scatterplots, and Pearson's correlation values were used to determine the relationship between MIM indices of greenline condition, and channel morphology and grazing history. We used multiple regression to predict post-fire grazing duration by greenline condition. Differences between monitoring years for MIM indices was examined using ANOVAs. Any significant change between years was modeled by post-fire grazing duration and ecologically relevant environmental variables.

In order to select the most parsimonious model, a subset of ecologically revelvant predictor variables were chosen for each model, and then the variable set was further reduced using stepwise regression (Barton 2012). Parameters that occurred in two or more of the top five models with the lowest AIC scores were included in the final model. Watershed position was included as a parameter in those models where the response variable was found to be signifcally different across positions. See Table 1 for full set of MIM indices, grazing, watershed and reach-scale variables and Appendix 2 for variables included in each of the full models before stepwise regression. Watershed-scale variables were included to account for top-down influences of watershed geology and morphology on riparian disturbance response. Reach-scale and site biophysical variables were included to account for site variablity and determine the contribution of local channel attributes on condition. Once variability in watershed and channel characteristics were accounted for, we included grazing variables to determine influences of management on riparian condition.

Post-fire grazing in 2015 was modeled by condition in 2014 using MIM indices and watershed position. High correlation among MIM indices in 2014 resulted in a much reduced model with only bank cover and watershed position included as parameters in the final model. Change in bank stability was modeled by an interaction between post-fire grazing and fraction of fines to determine whether post-fire grazing duration could alone account for the change in bank stability or whether there was an interactive effect at those sites most dependent on vegetation to prevent destabilization, i.e. site characterized by fine substrate.

Results

Relation of Vegetation to Biophysical Characteristics

A 3-dimensional NMDS was the optimal solution for reducing stress and ease of interpretation. The final solution had a stress of 0.12, non-metric fit R^2 = 0.99, and linear fit R^2 = 0.89. Axis 1 was strongly correlated with percent fraction of fines, elevation, and riparian extent (Table 2, Figure 1, and 2). It separated rocky, higher elevation sites from lower elevation sites characterized by fine substrates (Figure 2). Sites with fine-textured streambeds were associated with herbaceous species found commonly in meadows of the Intermountain West, such as mountain rush (*Juncus arcticus* Willd.), meadow barley (*Hordeum brachyantherum* Nevski), and Rocky Mountain iris (*Iris missouriensi* Nutt.) (Appendix 3, 4). High elevation, rocky sites with narrow riparian extents were associated with redosier dogwood (*Cornus sericea* L.), shortawn foxtail (*Alopecurus aequalis* Sobol.) and quaking aspen (*Populus tremuloides* Michx).

Axis 2 was strongly correlated with greenline-to-greenline width, woody height and cover, Winward stability rating, and fraction of fines, which separated sites with high woody species dominance from herbaceous-dominated sites and aspen stands (Table 2, Figure 1 & 3). Sites with greater woody cover and greenline-to-greenline widths were associated with willow species (*Salix exigua* Nutt., *S. bebbiana* Sarg., S. *lemmonii* Bebb), scouringrush horsetail (*Equisetum hyemale* L.), and curly dock (*Rumex crispus* L.) (Appendix 3, 4). Vegetation associated with high Winward stability rating and/or fraction of fines were meadow barley (*Hordeum brachyantherum* Nevski), creeping bentgrass (*Agrostis stolonifera* L.), Nebraska sedge (*Carex nebrascens*is Dewey), and quaking aspen (*Populus tremuloides* Michx).

Axis 3 was associated with wetland rating, stream length, and stream gradient (Table 2, Figure 2 & 3). Lower gradient sites with more contributing stream length were associated with wetland obligate species, presumably because of greater water residency time at lower gradient locations.

Species included hardstem bulrush (*Schoenoplectus acutus* (Muhl. ex Bigelow) Á. Löve & D. Löve), shortawn foxtail (*Alopecurus aequalis* Sobol.), and straightleaf rush (*Juncus orthophyllus* Coville). High gradient sites were associated with facultative wetland species, such as quaking aspen (*Populus tremuloides* Michx.), Douglas' sedge (*Carex dougl*asii Boott), and common yarrow (*Achillea millefolium* L.). Percent rock type and percent burn severity class were not significantly correlated with ordination axes (Table 2).

Biophysical characteristics

Lower watershed positioned sites were characterized by greater contributing stream length and greenline-to-greenline widths, and were located at lower elevations and stream gradients relative to mid- and upper positioned sites (Table 3). Conversely, upper positioned sites had the lowest values for stream length and greenline-to-greenline width, and were located at the highest elevations with steepest stream gradients. The mid-positioned sites were dispersed around the median values for stream length, greenline-to-greenline width, elevation and stream gradient, and had the greatest variation in mean elevation and riparian extent (Table 3).

One-way analysis of variance found a significant difference among watershed positions for bank cover ($F_{2, 20} = 3.05$, $p = 0.070$) and greenline-to-greenline width ($F_{2, 20} = 4.48$, $p = 0.025$) (Table 3). The difference lies mostly between lower-positioned sites and upstream sites, as mid- and upper-positioned sites had comparable means with similar distributions. Bank cover was lower, whereas greenline-to-greenline width was higher at lower-positioned sites (Table 3). Fraction of fines, total days grazed post-fire, bank stability, wetland rating, percent woody cover, woody height class, Winward greenline stability, and diversity indices did not differ significantly by watershed position (Table 3). Watershed position was included in multiple regression analyses where there was a significant variation in the response variable among watershed positions.

Grazing use and riparian condition

One-way analysis of variance showed that lower-positioned sites had significantly more days of continuous pre-fire grazing-use ($F_{2, 16} = 8.09$, $p = 0.004$) and fewer total days of recovery from grazing during the growing season ($F_{2, 16} = 5.88$, $p = 0.012$) (Figure 4). Most lower-positioned sites were grazed every year with little variation in season of use among years (Figure 4). Mid-positioned sites had much variation in use among years and many or most had full years of rest (Figure 4). Four of the nine mid-positioned sites were only grazed in the spring and had rest years for recovery. Upper-position sites were grazed for shorter duration; three of four saw little variation among years (Figure 4). Fifteen of the nineteen sites were grazed in the hot season of the year, when cattle are most prone to concentrate in riparian areas, and four were grazed in the hot season annually.

After the fire, the number of days grazed in 2015 was predicted by multiple regression to be higher at sites with higher percent bank cover in 2014 and lower at mid-watershed positioned sites relative to lower-positioned sites when holding all other variables constant (Table 4). Bank stability was highly correlated with bank cover in 2014, so was not included in that model ($r = 0.81$). Wetland rating, greenline-to-greenline width, and woody height and cover in 2014 did not significantly contribute to predicting the duration of grazing in 2015 and were removed for model parsimony.

One-way analysis of variance showed that lower bank stability significantly decreased from 80.1 ± 8.4 % in the first year of monitoring to 67.8 ± 8.5 % the second year ($F_{1, 43} = 4.056$, $p = 0.050$). The change in bank stability over the two monitoring years was predicted using multiple regression by a significant interaction between total days grazed two years post-fire and fraction of fines (Table 5). There was no significant difference in Winward stability rating, percent bank cover and stability for those sites that decreased in bank stability and those unchanged between 2014 and 2015. In ANOVA, there was no significant change in bank cover ($p = 0.679$), wetland rating ($p = 0.111$), woody height class ($p = 0.968$), percent woody species cover ($p = 0.512$), greenline-to-greenline width (p = 0.897) or hydric herbaceous species (p = 0.374) over the two monitoring years. Burned, re-sprouting woody vegetation average height class was 1.37 m in 2014 and 1.60 m in 2015, and remained between 0 and 1 m tall three years after fire.

Using multiple regression, wetland rating was positively predicted by bank stability, sinuosity, and percent volcanic bedrock (Table 6). Mean growing season days before or after grazing-use, total grazing-use during the growing season, and average annual duration of use prior to the fire did not significantly predict wetland rating. The strongest predictor for bank cover was position in the watershed (Table 7). Sites mid- and upper positioned in the watershed were predicted to be 9.26% and 30.46% more covered than lower positioned sites, respectively. There was a 0.98% decrease in bank cover with every percent increase in bank instability holding all other variables constant. Surprisingly, bank cover decreased with increase in recovery days during the growing season after accounting for all other predictor variables (Table 7).

Bank stability three years after fire was positively predicted by percent bank cover and negatively predicted by gradient, fraction of fines and post-fire grazing between monitoring years using multiple regression analysis (Table 8). Percent bank cover was the strongest predictor of bank stability; every percent increase in bank cover was predicted to increase bank stability 0.89%, holding all other variables constant. Bank stability was significantly greater for sites within grazing strategy 3, spring-only grazing and rest years versus sites with grazing strategy 1, continuous, hot season grazing ($F_{2, 16}$ = 5.03, p = 0.020, Figure 5). Surprisingly, Winward stability rating, an index of vegetation's ability to maintain bank stability did not significantly predict bank stability (r = -0.19, Table 8). Winward greenline stability had a small negative correlation with bank stability for both lower (N = 7, r = -0.25) and upper watershed position sites (N= 5, r = -0.31), and there was no correlation for mid-positioned sites (N = 11, r = -0.02).

Percent woody species cover was significantly higher for grazing strategy 3 ($F_{2, 16} = 3.63$, $p = 0.050$), but did not differ among grazing strategy 1 and 2. However, percent woody cover was significantly different among all grazing strategies when variation among watershed positions was accounted for in the model ($F_{2, 12} = 6.55$, $p = 0.012$, Figure 6). Lower and mid- positioned sites increased in woody cover with grazing strategies 2 and 3. Upper-positioned sites were under represented in grazing strategies 2 and 3, but had greater woody cover in category 1 than the other watershed positions. Across all sites, woody cover increased with mean recovery days after grazing-use during the growing season in the six years prior to the fires (Table 9).

Woody height class increased with grazing strategies 2 and 3 for all watershed positions ($F_{2, 12} = 7.55$, $p = 0.008$, Figure 7). Using multiple regression analysis, woody height class was positively predicted by riparian extent and mean recovery days after grazing-use during the growing season prior to fire and negatively predicted by stream gradient and percent of the watershed categorized as high burn severity (Table 10). Perennial species richness was significantly lower in grazing category 1 than grazing category 2 and 3 ($F_{2, 16} = 3.68$, $p = 0.048$, Figure 8). Pre-fire grazing variables were highly correlated, so only one was included in a single model and was chosen based on highest contributed variance explained.

Discussion

At the reach scale, riparian species composition is most influenced by variables associated with watershed position, more so than recent disturbance and local channel morphology. Similarly, Engelhardt et al. (2011) found that watershed variables corresponding to watershed position were most predictive of riparian vegetation community. This is not surprising considering sediment transport, hydrologic regime, and channel shape and gradient are dependent on location in the watershed (Bendix and Hupp 2000).

Steeper reaches with high energy and erosional flood regimes are dominated by rocky substrate and support woody vegetation, like willow (*Salix* spp.) and alder (*Alnus* spp.). This is attributed to *Salix* and other riparian woody species resistance to seasonal high energy floods, with flexible stems and root anchoring (Naiman and Decamps 1997). Large, rocky substrates are a result of high energy flows and high rates of sediment transport (Montogomery and Buffington 1997). Aspen stands (*Populus tremuloides Michx*) occurred at high gradient, headwater reaches characterized by cooler temperatures and greater water availability during spring snowmelt (Engelhardt et al. 2015). Reaches associated with fine textured sediment were dominated by dense stands of wetland graminoids, like *Carex* and *Juncus* species. This is thought to be a product of physiological adaptation to anoxia, which can occur in fine sediments saturated by water. Species in the *Cyperaceae* and *Juncaceae* families commonly have aerenchyma, soft porous tissue that allows for transportation of oxygen to roots (Naiman and Decamps 1997).

Wetland rating was associated with *Salix* and *Juncus* species and highest for sinuous sites with high bank stability and volcanic bed material. This is attributed to higher water residency time and low energy flows at more sinuous sites, allowing for sediment deposition and riparian vegetation colonization along the channel (Hupp and Osterkamp 1996). Watersheds associated with high percent volcanic bedrock are associated with riparian forests, dominated by *Salix* species and *Populus tremuloides Michx,* wetland facultative species (Chambers et al. 2004, Engelhardt et al. 2011). Reaches characterized by low wetland rating, and a mix of drought-adapted upland and ruderal species were thought to be driven by low water availability or unstable channels.

Bank stability was sensitive to pre-fire grazing strategy and benefitted from reduction in grazing use during the hot season (July-September) and increase in recovery time. Sites grazed only in the spring with full years of rest between grazing-use were the most stable. During the hot season, cattle spend more time in riparian areas especially in lower elevation pastures, because of coolers temperatures, ease of water access, and abundance of green forage relative to the senesced upland vegetation (Swanson et al., 2015). The concentrated use increases mechanical damage to banks through hoof action and consumption of bank vegetation, which may contribute to destabilization of banks.

As expected, banks were less stable when not protected by vegetation, embedded rock or anchored wood, when substrates were fine textured, and stream energy was increased by gradient. Though vegetation is important, this indicates that bank stability is not dependent on vegetation alone but also influenced by site geomorphic characteristics and soils. Vegetation importance in maintaining bank structure and reducing erosive water energy through surface drag and roughness has been well established (Corenblit et al. 2009a, Dwire and Kauffman 2003, Naiman et al. 1993, Swanson et al. 2015). Rock and woody material incorporated into the bank material can also provide structural support and resist erosion (Swanson et al. 1982).

Winward et al. (2000) states that not all species are equally capable in maintaining bank stability. These functions depend on root mass, strength, depth, and density and above ground growth resistance to flow energy. These characteristics tend to be associated with late-seral, wetland obligate species. Surprisingly, in our model, Winward stability rating was not directly related to bank stability; however, the positive association between bank cover and both Winward stability rating and bank stability suggest there may be an indirect effect. Riparian plant species with high Winward stability ratings tend to cover more bank surface by nature of how they are rated. Rhizomatous species tend to form expansive vegetative mats along the greenline and woody species (e.g. *Salix* species) resist flood forces with large branching crowns.

The surprising negative correlation between Winward stability rating and percent bank stability for sites higher and lower in the watershed can be attributed to the high presence of *Populus tremuloides* and *Juncus articus respectively*. *Juncus articus* is associated with fine textured substrate where stream gradients are lower and sediment deposition occurs, and *Populus tremuloides* is associated with steep, high energy reaches. Both reach types are prone to erosion and bank instability. Both species persist after channel incision which increases bank instability. Furthermore, measuring species cover relative of all vegetative cover, as oppose to absolute cover may over inflate the cover of each species and not reflect the amount of bank covered by vegetation.

After fire, fine textured sites were more susceptible to bank instability and saw rapid destabilization with increased duration of post-fire grazing. In our study, a significant change was seen from year two to year three, suggesting a two-year rest from grazing was not adequate to maintain an upward trend towards recovery. Commonly, non-cohesive, fine-grained streambanks of low gradient, unconfined reaches are thought to have more potential for erosion without root masses to stabilize the non-cohesive soils (Montgomery and Buffington 1998, Winward 2000). In addition, finer substrate size tended to be lower in the watershed, where pre-fire grazing was longer duration with fewer days of recovery during the growing season. Furthermore, this change in bank stability did not seem to depend on site condition, as described by presence of Winward stability species, percent bank cover and stability, which was comparable to unchanged sites. These results suggest a complication with stages and process of channel incision and recovery that are shown in Dickard et al. (2015). Unfortunately, MIM (Burton et al. 2011) does not provide measurements of channel incision depth and width.

Moreover, when land managers were making the decision to return livestock to burned allotments, sites lower in the watershed were grazed for a longer duration. In our study, sites lower in the watershed also tended to have the lowest values for bank cover. However, sites selected for longer lengths of grazing tended to have higher bank cover, suggesting managers took into account percent bank cover when deciding when cattle were allowed to return and how long they were grazing. Despite higher rates of bank cover, these sites were less stable. This may be a result of watershed position, as discuss above. Alternatively, a threshold for bank cover may not have been met prior to the return of livestock to adequately protect the bank from grazing pressures or a combination of these factors.

More targeted research is required to determine the minimum bank cover required to maintain bank stability. Thresholds may vary by stream type, riparian vegetation community and degree of grazing pressure (Wyman et al. 2006, Dickard et al. 2015. To understand the role of grazing, future studies should focus on sites characterized by fine textured soils, where banks are more susceptible to instability. We can then parse out the effects of season, duration and intensity of grazing on riparian condition, the factors that determine site resiliency, and the time necessary for post-fire recovery.

Unlike in upland vegetation communities in the Great Basin, burn severity did not influence species composition or cover. This may reflect the use of adaptive mechanisms by riparian vegetation to tolerate frequent disturbance regime (Naiman and Decamps 1997). However, watersheds characterized by high burn severity tended to have lower woody heights, which may directly reflect loss of above ground plant material from fire. Herbaceous vegetation was observed to recover quickly, with regrowth happening within weeks after fire (observed by author); unfortunately, the year immediately post-fire was not captured. Drier sites characterized by upland vegetation are suspected to respond similarly to burned upland vegetation sites, with high rates of annual grass invasion, slower rates of recovery, and may be more susceptible to flood damage (Miller et al. 2013). Future research should target monitoring of drier, upland species dominated reaches to verify this hypothesis.

Our results suggest that pre-fire grazing strategy is important for the structure of the post-fire vegetation community, indicated by perennial species richness, woody species cover and height. Post-fire vegetation community response benefitted from pre-fire grazing strategies whose timing, duration, and variation-in-use allow for plant regrowth and recovery. Woody growth potential and recovery depend on root carbohydrate reserves (Loescher et al. 1990). Carbohydrate reserves increase throughout the growing season and are highest late in the growing season when they are most sensitive to stressors (Landhäusser and Lieffers 2002, Loescher et al. 1990). Woody plant survival during winter dormancy and next year's growth depend on adequate root carbohydrate reserves (Landhäusser and Lieffers 2002, Loescher et al. 1990).

Hot season grazing (July-September) and peak wildfire season in the Great Basin (July-August) occur when woody vegetation is most sensitive to depletion of reserves (Knapp 1998, Loescher et al. 1990). Excessive loss of photosynthetic material from full growing season or hot season grazing prior to a wildfire may leave the woody plant with inadequate reserves to recover, may reduce growth after disturbance, and delay terminal growth leaders reaching adequate height to escape browsing pressures (Clarke et al. 2013, Landhäusser and Lieffers 2002).

In our study, burned woody vegetation averaged between 0 and 1 meter with none exceeding 2 meters in height three years after fire. The terminal growth leaders had not escaped browse height for wildlife and livestock, assumed to be 2 meters. Continual and intense browsing stress could suppress woody species growth (Dwire et al. 2006, Mills 1983) and delay meeting BLM Emergency Stability and Rehabilitation objectives. Time for woody vegetation to reach browse escapement height may depend on species composition. Furthermore, repeated hot season grazing over time may reduce the extent of woody vegetation. Within watershed positions we found post-fire woody vegetation cover increased with average number of recovery days after grazing-use during the six growing seasons prior to fire, which is likely a reflection of pre-disturbance woody cover.

Prior to fire, grazing strategy within riparian pastures should consider woody growth physiology and allow for carbohydrate storage, particularly late in the growing season. This may encourage expansion of woody vegetation and help prepare woody vegetation for dormancy and resiliency to disturbance. After fire, terminal growth leaders should be allowed to escape browsing height for full recovery of the woody community. We will continue to monitor these locations to attempt to determine the time required for woody vegetation to escape browsing height after fire and how that relates to grazing strategy in burned streams.

It should be noted that we did not have experimental controls for grazing at any of our sites, so the effects of grazing on condition are not definitive. Completely excluding grazing from these pastures for the extent of the study would have been an economic loss for the permittee and would require legal justification by land managers, and thus was not deemed economically or politically feasible. There was also no experimental manipulation of grazing, which would have required permission from the permittee and Range Management Specialist, as well as extensive coordination between permittees, scientists, and land managers. Even if economically feasible, there is often an additional barrier of distrust between one of more of the parties that prevent experimental grazing manipulation.

In addition, many of our sites had burned and unmaintained fences, which allowed for cattle grazing at times and locations not normally allowed. These sites are remote and rarely visited by Range Specialists, whose responsibilities include managing millions of acres of public land, and trespass grazing is poorly documented. As a result, the post-fire gazing duration is potentially less than experienced and reported.

Management Implications

- Managers should consider watershed position, substrate size, and stream gradient when managing for particular species or vegetation communities. Willows, aspen and herbaceous dominated communities prefer different site characteristics, which should be considered when setting site objectives (Naiman and Decamps 1997, Chambers et al. 2005a, b, Wyman et al 2006, Swanson et al 2015, Dickard et al 2015).

- Lower positioned sites may be more sensitive to grazing than those higher in the watershed. A longer rest or recovery period from grazing after fire than received in this study is required to maintain site stability. Minimum values for MIM indices have not been determined and may depend on vegetation community, substrate texture, and bank condition prior to the fire.
- Site potential should be considered when managing for wetland rating, as it is dependent on hydrologic processes and placement within the watershed. Land managers should focus efforts on site characteristics that may be responsive to management decisions, asking the questions what plants are possible in this location, where and how will they influence recovery, and how can management influence their recovery (Dickard et al. 2015, Swanson et al. 2015, Wyman et al. 2006).
- MIM protocol currently measures relative cover of vegetation, rock and anchored wood, which does not capture percent bare or exposed soil. If the goal is to determine the minimum bank cover necessary to maintain bank stability, absolute cover should be considered in replacement of, or in addition to, relative cover. It is important to have stabilizer species on the greenline (Burton et al. 2011, Winward 2000), but it is also important to have those plants robust and vigorous (Dickard et al 2015, Swanson et al. 2015, Wyman et al. 2006).
- Grazing strategies that reduce hot season grazing (July-Sept), vary in seasonality of use, incorporate recovery during the growing season or occasional full seasons of rest, and allow for regrowth before winter promote growth of riparian species and bank stability. This may help meet post-fire objectives more quickly following disturbance.

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Tables

Table 1 *MIM indices and biophysical variables derived for each study site*

Model Parameters	Units	Explanation
MIM Indices		
Bank Stability	%	Percentage of plots along greenline with a deposition landform or perennial vegetation cover and without signs of sloughing, fracturing, or slumping
Bank Coverage	%	Percentage of plots along greenline covered by rock, anchored wood, or 25% perennial vegetation
Winward Stability Rating		Winward stability rating weighted by species composition at the site
Wetland Rating		Wetland indicator status weighted by species composition at the site
Woody Cover	%	Percentage woody species cover
Greenline-to-greenline Width	m	Nonvegetated distance between greenlines on opposite streambanks
Woody Height Class		Average woody species height class at site (see Methods for height classes)
Fraction of Fines	%	Percentage of fine substrate (less than 6 mm)
D50	cm	Substrate diameter at the 50th percentile
Reach-scale Parameters		
Sinuosity		Metric of stream meander
Elevation	m	Mean elevation of end points of designated monitoring area
Stream Length	km	Stream length above the monitoring site
Gradient	cm / m	Ratio of drop in streambed elevation over 100 m
Riparian Extent	m	Distance of riparian vegetation lateral to stream channel
Watershed-scale Parameters		
Volcanic Bedrock	%	Percentage of watershed surface area above site with volcanic bedrock
Sedimentary Bedrock	%	Percentage of watershed surface area above site with sedimentary bedrock
Intrusive Igneous Bedrock	%	Percentage of watershed surface area above site with intrusive igneous bedrock

Siliciclastic Bedrock	%	Percentage of watershed surface area above site with siliciclastic bedrock
Hypsometric Integral (HI)		HI = (elevation (E)mean - Emin) - (Emax - Emin)
Unburned to Low Severity	%	Percentage of unburned to low severity burn class in watershed above site
Low Burn Severity	%	Percentage of low severity burn class in watershed above site
High Burn Severity	%	Percentage of high severity burn class in watershed above site
Burned	%	Percentage of burned area in watershed above site
Livestock Grazing Variables		
Average Duration of Grazing	days	Average duration of grazing use for six years prior to fire (2006-2011)
Post-fire Grazing Use Between Years	days	Number of days grazed between two monitoring years (2014 & 2015)
Mean Growing Season Days After Grazing	days	Average days grazed during the growing season after grazing use (2006-2011)
Total Growing Season Recovery Days	days	Total days of recovery from grazing use during the six growing season prior to fire (2006-2011)

Table 2 *Mean vector scores for each non-metric multidimensional scaling (NMDS) axis, square root of correlation coefficient, and p-value (999 permutations).*

	NMDS1	**NMDS2**	**NMDS3**	**R^2**	**Pr(>r)**
Bank Stability (%)	0.14	-0.83	0.55	0.31	0.082
Bank Coverage (%)	0.83	0.41	0.36	0.12	0.483
Winward Stability Rating	-0.14	0.96	0.23	0.32	0.062
Wetland Rating	-0.21	-0.06	0.98	0.46	**0.007**
Woody Cover (%)	0.29	-0.71	-0.64	0.44	**0.010**
Greenline-to-greenline Width	-0.20	-0.96	0.18	0.40	**0.024**
Woody Height Class	0.39	-0.71	0.59	0.52	**0.003**
Fraction of Fines (< 6 mm)	-0.64	0.76	0.06	0.67	**0.001**
Volcanic Bedrock (%)	0.27	0.74	0.61	0.30	0.077
Sedimentary Bedrock (%)	-0.25	-0.65	-0.71	0.21	0.243
Intrusive Igneous Bedrock (%)	-0.39	-0.56	-0.73	0.11	0.493
Siliciclastic Bedrock (%)	-0.62	-0.30	-0.72	0.24	0.147
Sinuosity	-0.88	0.34	0.34	0.19	0.225
Hypsometric Integral	-0.08	1.00	-0.04	0.06	0.742
Elevation (m)	0.96	-0.24	-0.15	0.38	**0.022**
Stream Length (km)	-0.17	-0.65	0.74	0.52	**0.001**
Burned (%)	-0.56	0.82	-0.13	0.31	0.070
Unburned to Low Severity (%)	-0.45	0.61	0.65	0.04	0.848
Low Burn Severity (%)	-0.42	0.45	0.79	0.28	0.092
High Burn Severity (%)	0.73	0.01	-0.68	0.28	0.099
Gradient	0.52	0.42	-0.74	0.60	**0.001**
Riparian Extent	0.99	-0.03	0.11	0.44	**0.011**

Table 3 *Mean values, standard deviation, f-value, and p-value for MIM vegetation and geomorphic variables, and elevation by watershed position in 2015. An asterisk denotes a significant difference among watershed positions, P < 0.10 based on analysis of variance test with 2 and 20 degrees of freedom.*

	Lower	Mid	Upper	F-value	*P*
Species Richness	18.29 ± 7.39	21.27 ± 5.02	20.00 ± 1.87	0.64	0.536
Species Evenness	0.65 ± 0.12	0.73 ± 0.13	0.75 ± 0.07	0.64	0.53
Simpson's Diversity Index	0.70 ± 0.14	0.70 ± 0.14	0.57 ± 0.18	1.41	0.267
Wetland Rating	66.23 ± 6.42	59.13 ± 6.02	58.54 ± 14.71	1.76	0.198
Winward stability rating	5.44 ± 0.94	5.40 ± 0.89	5.98 ± 0.74	0.81	0.459
Bank Stability (%)	65.46 ± 17.26	73.74 ± 20.22	58.06 ± 26.11	1.05	0.369
Bank Coverage (%) *	63.67 ± 20.37	81.23 ± 17.96	86.15 ± 9.66	3.05	**0.070**
Woody Height Class	1.63 ± 0.92	1.65 ± 0.74	1.49 ± 0.46	0.26	0.774
Woody Cover (%)	21.75 ± 13.95	22.43 ± 14.68	25.84 ± 10.22	0.57	0.573
Greenline-to-greenline width *	3.10 ± 1.98	1.80 ± 0.64	1.15 ± 0.24	4.48	**0.025**
Stream length (km) *	98.78 ± 81.85	16.93 ± 17.67	5.74 ± 7.42	8.25	**0.002**
Elevation (m)	1961 ± 251	2042 ± 185	2138 ± 123	1.18	0.328
Gradient *	1.32 ± 1.59	5.69 ± 2.39	9.70 ± 7.70	6.76	**0.006**

Table 4 *Multiple regression analysis for total number of days grazed 3-years after fire*

Variable	β	b	SE	*t*	*P*
Bank cover (%)	2.11	0.13	0.70	3.00	**0.010**
Mid-watershed position	-3.22	-3.22	1.52	-2.12	**0.052**
Upper watershed position	-0.20	-0.20	2.01	-0.10	0.923

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values

[2] Dependent variable: total number of days grazed 3-years after fire

[3] Multiple R^2: 0.462, adjusted R^2: 0.347, $F_{3,14}$: 4.01, P: 0.030.

***Table 5** Multiple regression analysis for change in bank stability between 2014 and 2015. Total post-fire days grazed included three years of data, 2013-2015.*

Variable	β	SE	*t*	*P*
Total Days of Post-fire Grazing	0.53	0.23	2.29	**0.037**
Fraction of Fines	-0.63	0.25	-2.54	**0.023**
Total Days Grazed Post-fire * Fraction of Fines	-0.73	0.24	-3.03	**0.008**

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values
[2] Dependent variable: change in bank stability between 2014 and 2015
[3] Multiple R^2: 0.503, adjusted R^2: 0.4034, $F_{3, 15}$: 5.056, P: 0.013

***Table 6** Multiple regression analysis for wetland rating. Mean growing season days before grazing is for the six year prior to fire (2006-2011).*

Variable	β	b	SE	*t*	*P*
Mean Growing Season Days Before Grazing-use	1.32	0.04	1.37	0.96	0.355
Gradient	-2.72	-53.26	1.55	-1.76	0.104
Sinuosity	4.03	20.71	1.44	2.80	**0.016**
Fraction of Fines	3.90	21.37	1.92	2.04	0.065
Volcanic Bedrock (%)	3.46	8.06	1.10	3.14	**0.009**
Bank Stability (%)	6.54	29.65	1.94	3.37	**0.006**

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values
[2] Dependent variable: wetland rating
[3] Multiple R^2: 0.823, adjusted R^2: 0.734, $F_{6, 12:}$ 9.268, $P < 0.001$

***Table 7** Multiple regression analysis for the inverse of percent bank cover (1-ln(bank cover)). Total growing season recovery is the sum of days not grazed during the six growing seasons prior to fire (2006-2011). Bank instability is the inverse (1-ln(bank stability)) of bank stability.*

Variable	β	b	SE	*t*	*P*
Total Growing Season Recovery Days	5.21	0.02	1.68	3.09	**0.010**
Mid-watershed Position	-9.26	-9.26	3.18	-2.92	**0.014**
Upper Watershed Position	-30.46	-30.46	4.60	-6.62	**< 0.001**
Winward Stability Rating	-4.10	-4.76	1.23	-3.32	**0.007**
Fraction of Fines (> 6 mm)	0.08	4.22E-03	1.64	0.05	0.963
Bank Instability (%)	15.98	0.98	1.60	9.97	**< 0.001**

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values
[2] Dependent variable: inversion of percent bank cover (1-ln(bank cover))
[3] Multiple R^2: 0.937, adjusted R^2: 0.903, $F_{6, 11:}$ 27.27, $P < 0.001$

Table 8 *Multiple regression analysis for percent bank stability. Post-fire grazing use between years includes the days of grazing-use between 2014 and 2015 monitoring, and total growing season recovery is the sum of days not grazed during the six growing seasons prior to fire (2006-2011).*

Variable	β	b	SE	*t*	*P*
Bank Cover (%)	15.69	0.89	0.02	7.15	< **0.001**
Fraction of Fines	-8.00	-0.44	0.03	-3.10	**0.009**
Gradient	-12.31	-2.41	0.02	-5.71	< **0.001**
Post-fire Grazing Use Between Years	-5.36	-0.12	0.02	-2.68	**0.020**
Winward Stability Rating	-4.33	-5.06	0.02	-1.95	0.075
Total Growing Season Recovery Days	0.78	3.43E-03	0.02	0.36	0.726

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values
[2] Dependent variable: percent bank stability
[3] Multiple R^2: 0.926, adjusted R^2: 0.889, $F_{6, 16:}$ 25.03, P < 0.001

Table 9 *Multiple regression analysis for percent woody species cover. D50 is the particle size that 50% of samples are equal or smaller. Mean growing season days after grazing is for six years prior to fire (2006-2011).*

Variable	β	b	SE	*t*	*P*
Mean Growing Season Days After Grazing	10.01	0.22	2.49	4.02	> 0.001

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values
[2] Dependent variable: percent woody species cover
[3] Multiple R^2: 0.488, adjusted R^2: 0.458, $F_{1, 17}$: 16.18, P > 0.001

Table 10 *Multiple regression analysis for woody height class. Post-fire grazing use between years includes the days of grazing-use between 2014 and 2015 monitoring, and mean growing season days after grazing is for six years prior to fire (2006-2011). High burn severity is the percent of the watershed area classified as a high severity burn.*

Variable	β	b	SE	*t*	*P*
Gradient	-1.30	-0.24	0.12	-10.69	< **0.001**
Riparian Extent	1.29	0.54	0.12	10.70	< **0.001**
Post-fire Grazing Use Between Years	-0.10	-0.02	0.05	-1.82	0.129
Mean Growing Season Days After Grazing	0.35	0.01	0.09	3.97	**0.011**
High Burn Severity (% of area)	-0.31	-0.05	0.09	-3.62	**0.015**

[1] Columns represent independent variables, standardized coefficients (β), unstandardized coefficients (b), standard errors, t-values, and p-values
[2] Dependent variable: woody height class
[3] Multiple R^2: 0.965, adjusted R^2: 0.930, $F_{3, 15:}$ 27.55, P: 0.001

Figures

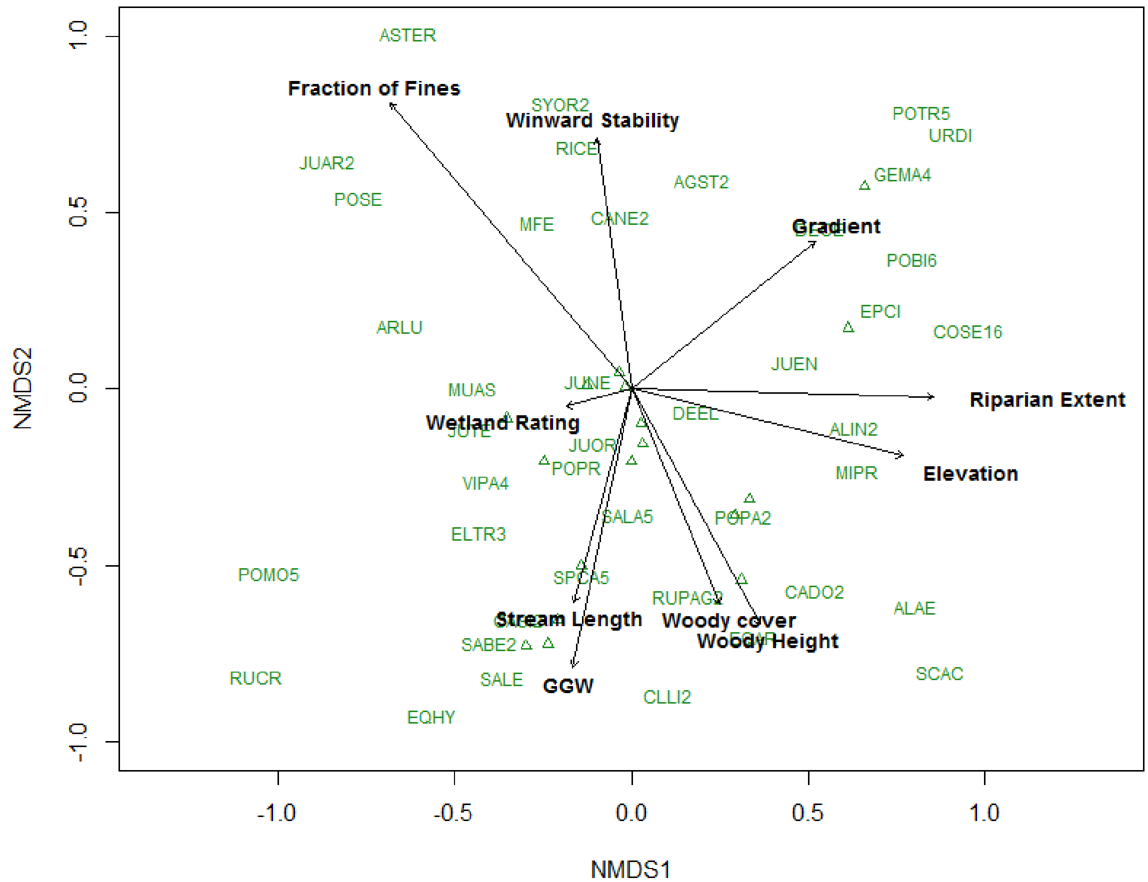


Figure 1 *Bi-plot of NMDS axes 1 and 2 of community species data and correlated environmental vectors (p-values ≤ 0.05). See Appendix 3 for species codes.*

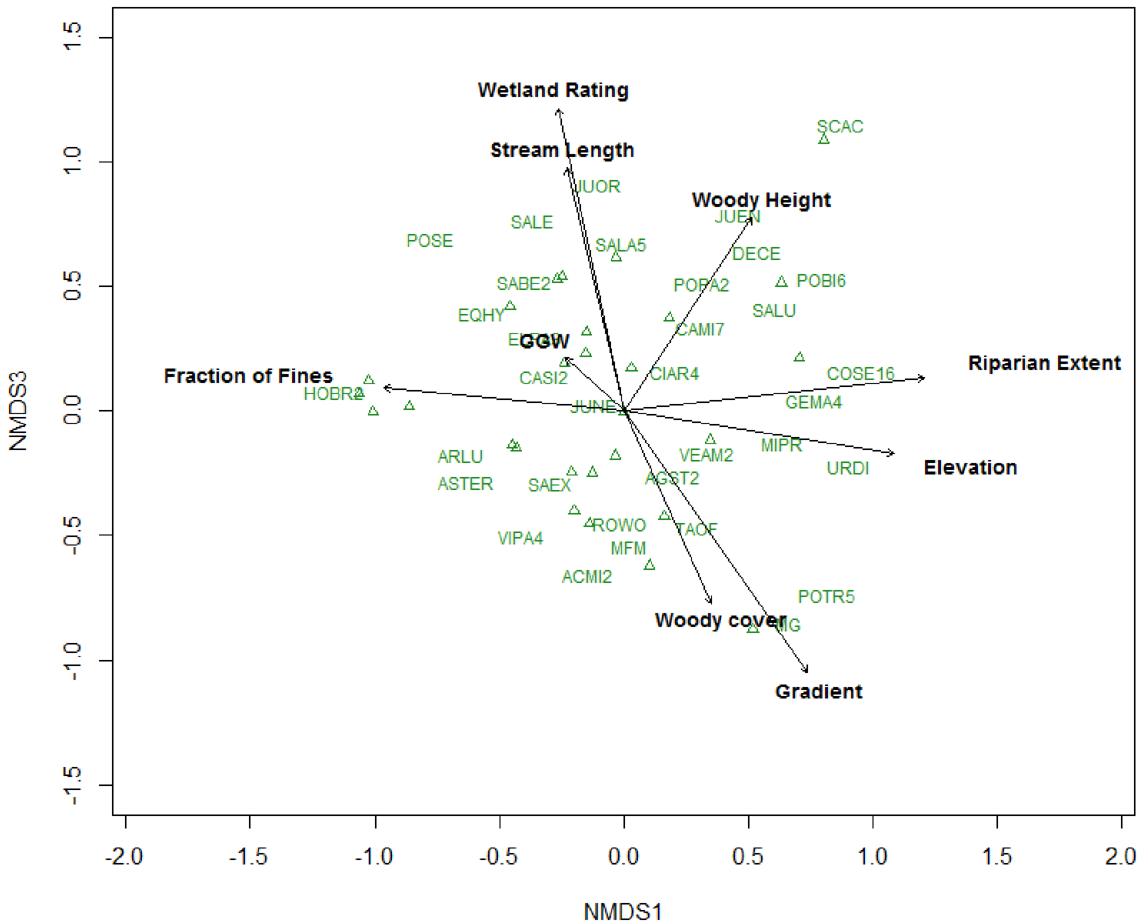


Figure 2 *Bi-plot of NMDS axes 1 and 3 of community species data and correlated environmental vectors (p-values ≤ 0.05). Species codes are listed in table 10.*

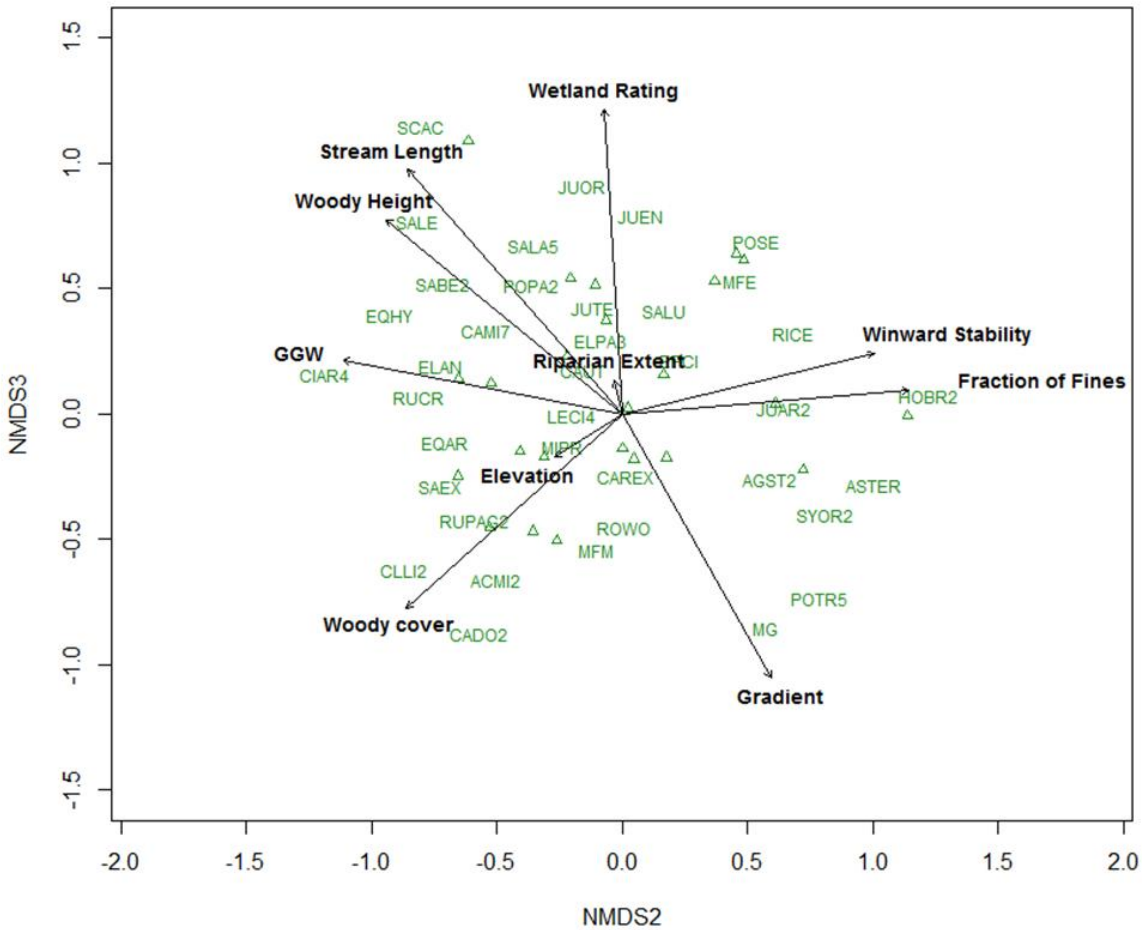


Figure 3 *Bi-plot of NMDS axes 2 and 3 of community species data and correlated environmental vectors (p-values ≤ 0.05). Species codes are listed in table 10.*

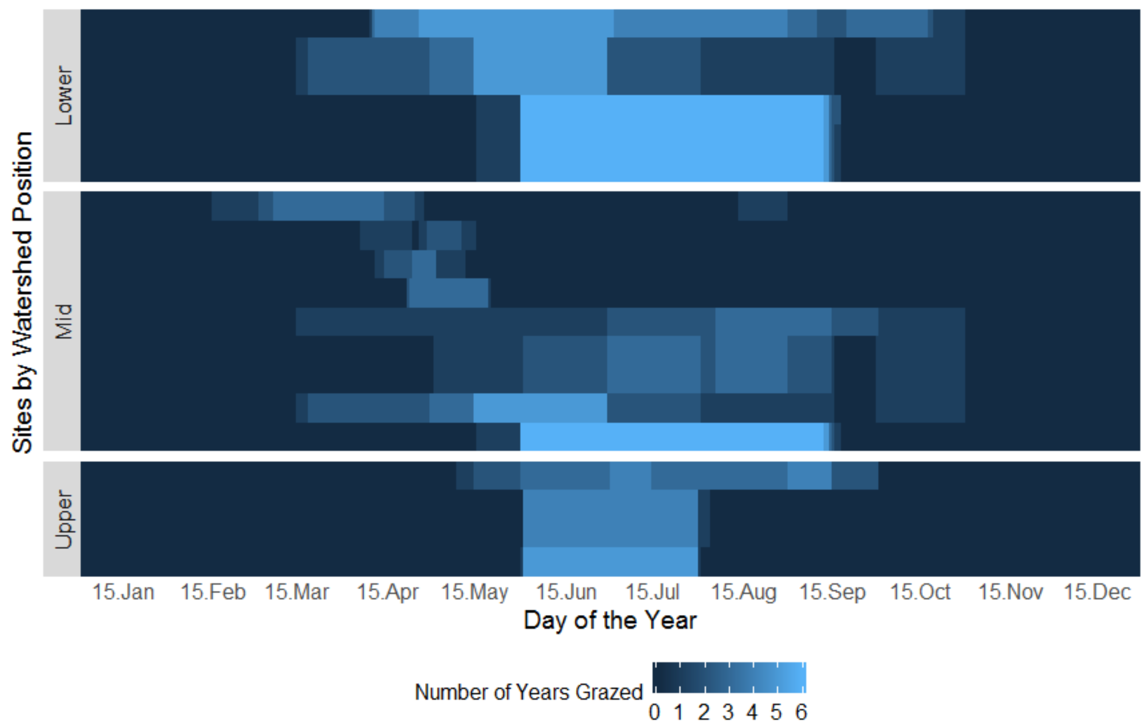


Figure 4 *Number of years grazed for each day of the year for 6-years prior to the wildfire for each site grouped by watershed position (i.e. lower, mid, upper).*

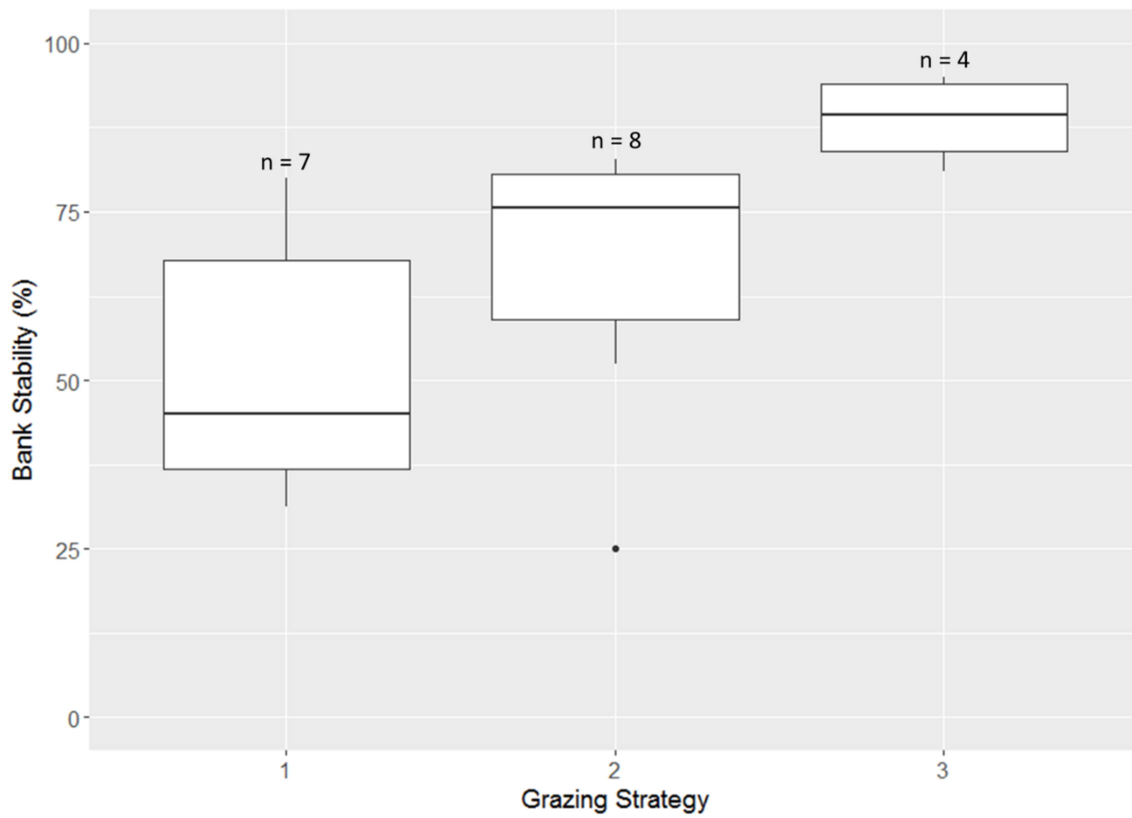


Figure 5 *Bank stability by grazing category, with category 1 representing intense, consistent hot season grazing, category 2 representing variation in grazing time among years and some rest during the hot season, and category 3 representing spring only grazing use.*

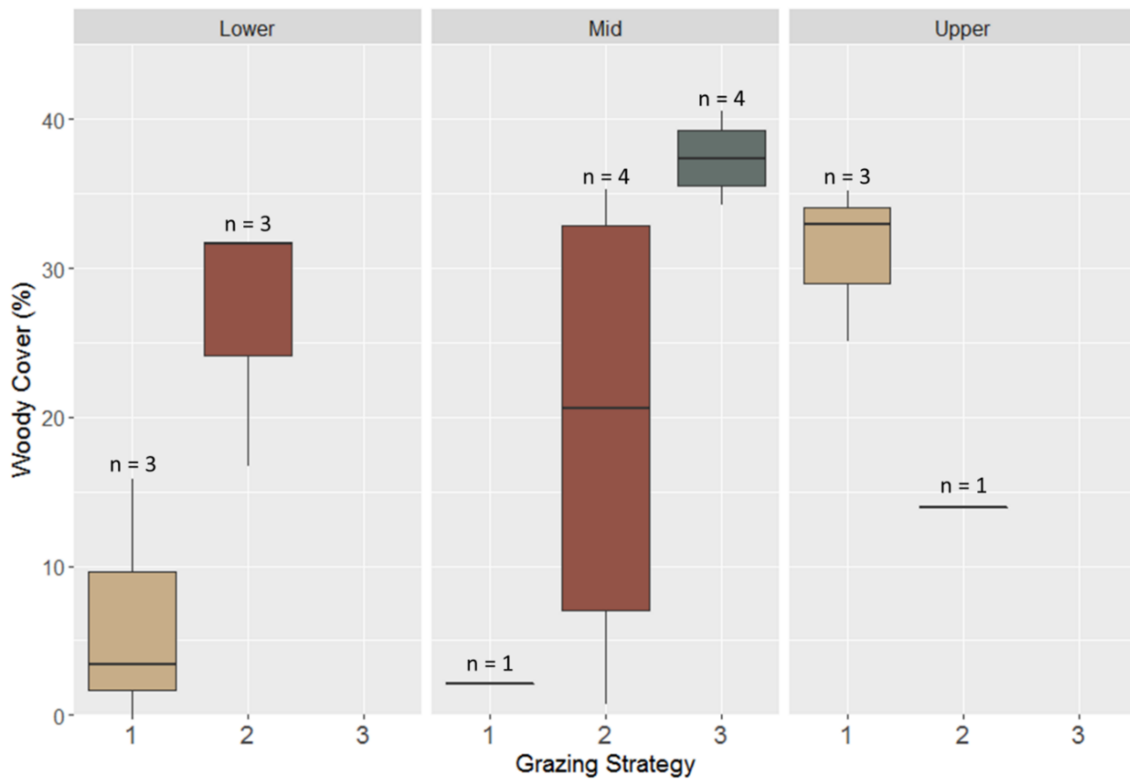


Figure 6 *Percent greenline woody cover among grazing categories and watershed positions. Watershed positions include lower-, mid-, and upper-positioned sites. Grazing category 1 represents intense, consistent hot season grazing, category 2 represents variation in grazing time among years and some rest during the hot season, and category 3 represents spring only grazing use.*

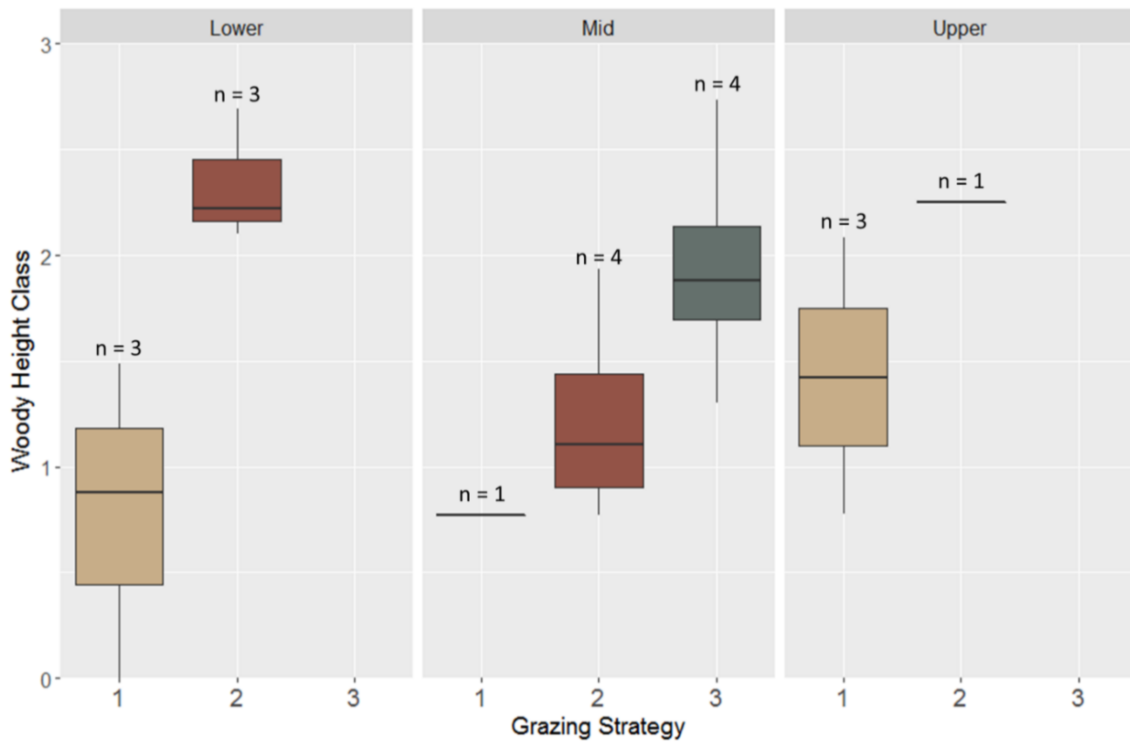


Figure 7 *Greenline woody height class among grazing categories and watershed positions. Watershed positions include lower-, mid-, and upper-positioned sites. Grazing category 1 represents intense, consistent hot season grazing, category 2 represents variation in grazing time among years and some rest during the hot season, and category 3 represents spring only grazing use. Woody height class 1 ranges from 0.-0.5 meters, class 2 ranges from 0.5-1 meters, and class 3 ranges from 1-2 meters.*

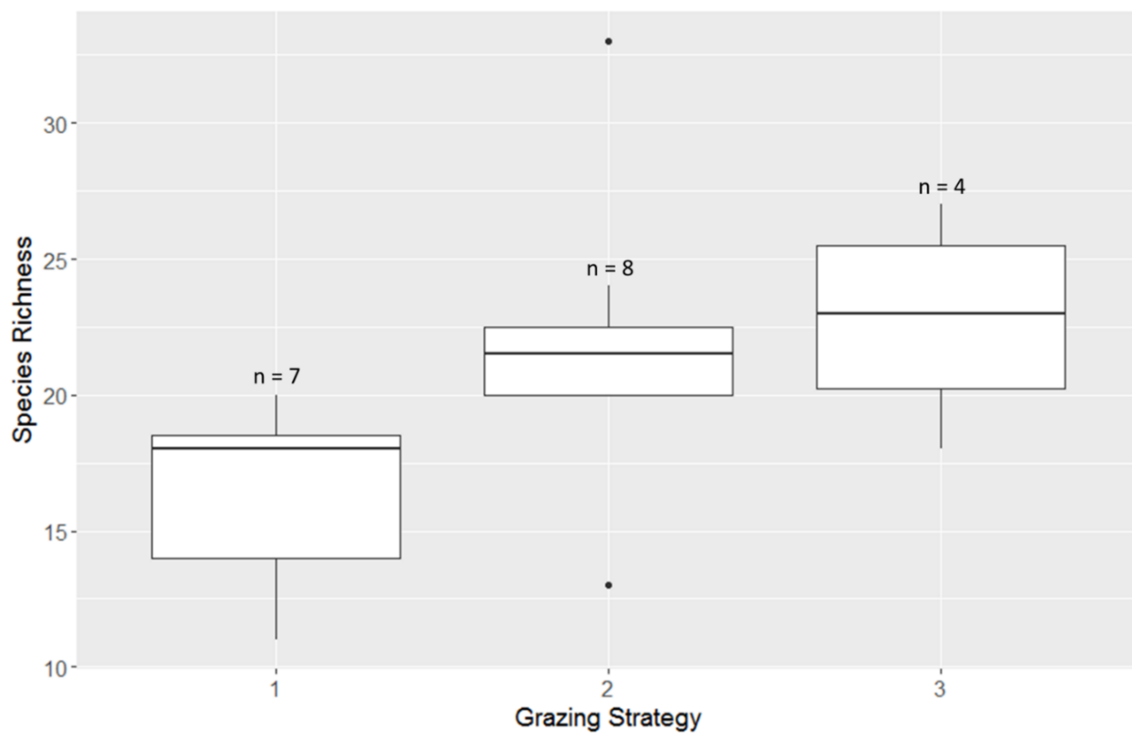
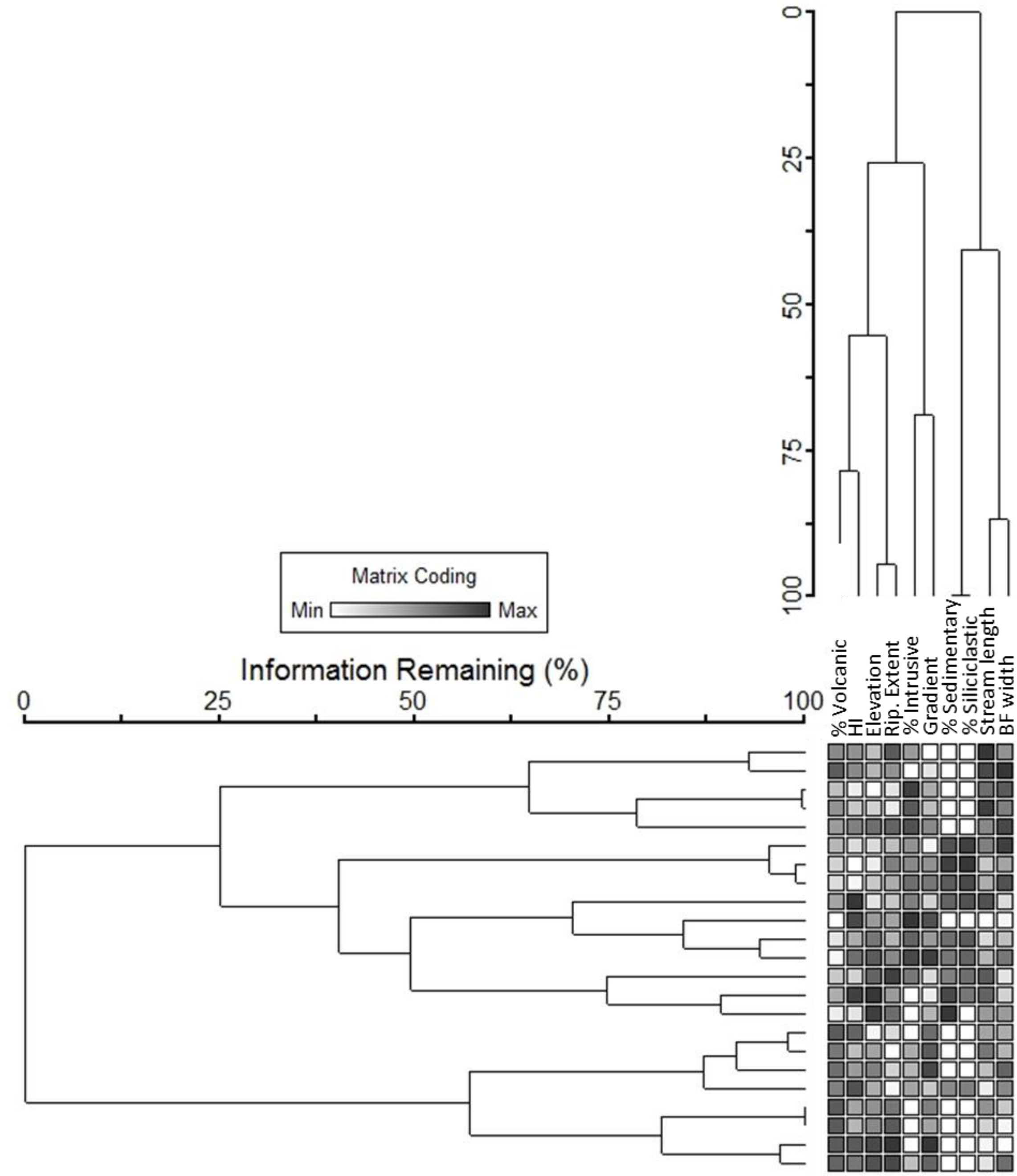


***Figure 8** Greenline species richness among grazing categories, with category 1 representing intense, consistent hot season grazing, category 2 representing variation in grazing time among years and some rest during the hot season, and category 3 representing spring only grazing use.*

Appendices

Appendix 1 *Two-way cluster analysis of variables related to watershed position by site (McCune and Mefford 2011).*



Appendix 2 *Set of variables include in stepwise and final multiple regression models for wetland rating, bank cover and stability, and woody height and cover. The number 1 denotes the variable was included in the full model; a 2 denotes the variable was selected in two or more of the top five models with the lowest AIC using stepwise regression and was thus included in the final model.*

	Full Model				
	Bank Stability	**Wetland Rating**	**Bank Cover**	**Woody Cover**	**Woody Height**
Variable	**Table 4 or 7**	**Table 5**	**Table 6**	**Table 8**	**Table 9**
Total Growing Season Recovery Days	2	1	2	2	
Gradient	2	2		1	2
Stream Length		1	1		
Sinuosity		2			
D50		1	1	2	1
Fraction of Fines	2	2	2	1	
Volcanic Bedrock		2		1	1
Greenline-to-greenline Width		1		2	1
Hypsometric Integral (HI)		1			
Riparian Extent		1	1	2	2
Average Duration of Grazing	1	1		1	
Bank Stability (%)		2			
Wetland Rating	1		1		
Winward Stability Rating	2		2		
Watershed Position			2		
Bank Cover	2				
Elevation				1	1
Post-fire Grazing Use Between Years	2		1	1	2
Mean Growing Season Days After Grazing					2
High Burn Severity (%)				1	2
Burned (%)				1	

Appendix 3 *Plant species included in NMDS with growth characteristics. Species sampled with only a single occurrence were not included in the NMDS analysis.*

Species Code	**Scientific Name**	**Common Name**	**Woody**	**Hydric**	**Forb**	**Wet-land Rating**	**Success -ional Status**	**Win-ward stability rating**
ACMI2	Achillea millefolium	Common yarrow	n	n	y	25	E	2
AGST2	Agrostis stolonifera	Creeping bentgrass	n	y	n	75	E	2
ALAE	Alopecurus aequalis	Short-awned foxtail	n	y	n	100	E	2
ALIN2	Alnus incana	Mountain Alder	y	y	n	75	L	8.5
ARLU	Artemisia ludoviciana	White sagebrush	n	n	y	25	E	2
ASTER	Aster spp.	Aster	n	n	y	25	E	2
CADO2	Carex douglasii	Douglas' sedge	n	n	n	50	M	2
CAMI7	Carex microptera	Small-winged sedge	n	n	n	50	M	5
CANE2	Carex nebrascensis	Nebraska sedge	n	y	n	100	L	8.5
CAREX	Carex	Sedge	n	y	n	75	M	5
CASI2	Carex simulata	Short-beaked sedge	n	y	n	100	L	8.5
CAUT	Carex utriculata	Beaked sedge	n	y	n	100	L	8.5
CIAR4	Cirsium arvense	Canada thistle	n	n	y	25	E	2
CLLI2	Clematis ligusticifolia	White Clematis	y	n	n	50	E	5
COSE16	Cornus sericea	Redosier dogwood	y	y	n	75	L	8.5
DECE	Deschampsia cespitosa	Tufted hairgrass	n	y	n	75	L	5
DEEL	Deschampsia elongata	Slender Hairgrass	n	y	n	75	E	2
ELAN	Elaeagnus angustifolia	Russian olive	y	n	n	50	E	5
ELPA3	Eleocharis palustris	Common spikerush	n	y	n	100	E	5
ELTR3	Elymus triticoides	Creeping wildrye	n	n	n	50	E	5
EPCI	Epilobium ciliatum	Fringed willowherb	n	y	y	75	M	2
EQAR	Equisetum arvense	Field horsetail	n	n	y	50	M	5

EQHY	Equisetum hyemale	Scouringrush horsetail	n	y	y	75	M	5
ERNA10	Ericameria nauseosa	Rubber rabbitbrush	y	n	n	0	E	2
GEMA4	Geum macrophyllum	Largeleaf avens	n	y	y	75	E	2
HOBR2	Hordeum brachyantherum	Meadow barley	n	y	n	75	E	2
IRMI	Iris missouriensis	Rocky Mountain iris	n	y	y	75	M	5
JUAR2	Juncus arcticus	Artic rush	n	y	n	75	L	8.5
JUEN	Juncus ensifolius	Swordleaf rush	n	y	n	75	M	5
JUNE	Juncus nevadensis	Nevada rush	n	y	n	75	L	5
JUOR	Juncus orthophyllus	Straightleaf rush	n	y	n	75	M	2
JUTE	Juncus tenuis	Povery rush	n	y	n	75	E	2
LECI4	Leymus cinereus	Basin Wildrye	n	n	n	50	M	5
MEAR4	Mentha arvensis	Wild mint	n	y	y	75	M	2
MIPR	Mimulus primuloides	Primrose monkeyflower	n	y	y	75	M	2
MUAS	Muhlenbergia asperifolia	Alkali muhly	n	y	n	75	E	2
PLMA2	Plantago major	Common plantain	n	n	y	50	E	2
POBI6	Polygonum bistortoides	American bistort	n	y	y	75	M	2
POMO5	Polypogon monspeliensis	Annual rabbitsfoot grass	n	y	n	75	E	2
POPA2	Poa palustris	Fowl bluegrass	n	n	n	50	E	2
POPR	Poa pratensis	Kentucky bluegrass	n	n	n	50	E	2
POSE	Poa secunda	Sandberg bluegrass	n	n	n	25	E	2
POTR5	Populus tremuloides	Quaking aspen	y	n	n	25	L	8.5
RICE	Ribes cereum	Wax currant	y	n	n	0	E	2
ROWO	Rosa woodsii	Woods' rose	y	n	n	25	E	5

RUCR	Rumex crispus	Curly dock	n	n	y	50	E	2
RUPAG2	Rumex paucifolius	Alpine sheep sorrel	n	n	y	50	E	2
SABE2	Salix bebbiana	Bebb willow	y	y	n	75	L	8.5
SAEX	Salix exigua	Narrowleaf willow	y	y	n	75	E	5
SALA5	Salix lasiandra	Whiplash willow	y	y	n	75	L	8.5
SALE	Salix lemmonii	Lemon's willow	y	y	n	75	L	8.5
SALU	Salix lucida	Shining willow	y	y	n	75	L	8.5
SCAC	Scirpus acutus	Hardstem bulrush	n	y	n	100	L	8.5
SPCA5	Sphenosciadium capitellatum	Woollyhead parsnip	n	n	y	25	E	2
SYOR2	Symphoricarpos oreophilus	Mountain snowberry	y	n	n	25	M	5
TAOF	Taraxacum officinale	Common Dandelion	n	n	y	25	E	2
URDI	Urtica dioica	Stinging nettle	n	n	y	50	M	8.5
VEAM2	Veronica americana	American speedwell	n	y	y	100	M	5
VIPA4	Viola palustris	Marsh violet	n	y	y	100	M	2

Appendix 4 *Species scores for NMDS axes 1-3. See Appendix 3 for species codes.*

	MDS1	MDS2	MDS3
ACMI2	-0.14	-0.50	-0.66
AGST2	0.20	0.59	-0.26
ALAE	0.80	-0.61	1.09
ALIN2	0.63	-0.11	0.52
ARLU	-0.65	0.18	-0.17
ASTER	-0.63	1.01	-0.28
CADO2	0.52	-0.57	-0.88
CAMI7	0.31	-0.54	0.33
CANE2	-0.03	0.49	0.62
CAREX	-0.13	0.01	-0.25
CASI2	-0.32	-0.65	0.14
CAUT	0.03	-0.15	0.17
CIAR4	0.21	-1.18	0.16
CLLI2	0.10	-0.87	-0.62
COSE16	0.95	0.17	0.16
DECE	0.53	0.46	0.64
DEEL	0.18	-0.07	0.37
ELAN	-0.24	-0.72	0.19
ELPA3	-0.35	-0.08	0.29
ELTR3	-0.43	-0.41	-0.15
EPCI	0.71	0.23	0.21
EQAR	0.35	-0.70	-0.12
EQHY	-0.56	-0.93	0.39
ERNA10	-0.03	0.05	-0.18
GEMA4	0.77	0.61	0.04
HOBR2	-1.17	1.22	0.08
IRMI	-1.01	1.14	-0.01
JUAR2	-0.86	0.64	0.02
JUEN	0.46	0.07	0.79
JUNE	-0.12	0.02	0.02
JUOR	-0.10	-0.16	0.91
JUTE	-0.46	-0.11	0.42
LECI4	0.00	-0.20	-0.01
MEAR4	-0.25	-0.21	0.54
MFE	-0.27	0.47	0.53
MFM	0.03	-0.10	-0.54
MG	0.66	0.57	-0.85
MIPR	0.64	-0.23	-0.13
MUAS	-0.45	0.00	-0.14

PLMA2	-0.21	-0.65	-0.25
POBI6	0.80	0.37	0.53
POMO5	-1.02	-0.52	0.12
POPA2	0.32	-0.36	0.51
POPR	-0.15	-0.22	0.23
POSE	-0.77	0.54	0.69
POTR5	0.82	0.78	-0.74
RICE	-0.15	0.69	0.32
ROWO	-0.02	0.00	-0.45
RUCR	-1.06	-0.81	0.07
RUPAG2	0.16	-0.59	-0.42
SABE2	-0.40	-0.72	0.52
SAEX	-0.30	-0.73	-0.29
SALA5	-0.01	-0.36	0.67
SALE	-0.36	-0.82	0.77
SALU	0.61	0.17	0.41
SCAC	0.88	-0.80	1.15
SPCA5	-0.14	-0.53	-0.45
SYOR2	-0.20	0.81	-0.40
TAOF	0.29	-0.36	-0.47
URDI	0.90	0.72	-0.22
VEAM2	0.33	-0.31	-0.17
VIPA4	-0.41	-0.26	-0.50