THESIS

EFFECTS OF POST-FIRE MULCH APPLICATIONS ON HILLSLOPE-SCALE EROSION

Submitted by

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ABSTRACT

EFFECTS OF POST-FIRE MULCH APPLICATIONS ON HILLSLOPE-SCALE EROSION

Wildfires are increasing in frequency and intensity, greatly altering the landscape and increasing risk of erosion. Mulching is a common restoration technique used after wildfire to enhance protective ground cover and reduce erosion, yet most studies are conducted at the plot-scale. This study applies an experimental approach to evaluate the impact of mulch treatments at the hillslope-scale using varying mulch levels. Similar adjacent hillslopes were chosen to minimize variability in landscape features. The objectives of this research are to 1) examine the effectiveness of post-fire mulching in reducing erosion at the hillslope-scale, and 2) identify landscape features and precipitation factors contributing to the occurrence and magnitude of sediment yield. Sediment fences were installed in convergent swales and planar hillslopes to quantify sediment yields before and after aerial wood mulch application. Rain gauges were installed to compute rainfall amount (mm), duration (hr), and maximum intensities (mm/hr) by storm event. Field observations, coupled with game camera footage, were utilized to evaluate whether each storm produced sediment in the fences. Surface cover surveys were conducted to assess cover changes over the season. Collectively these data were used to 1) identify rainfall intensity thresholds for erosion, 2) examine controls on sediment generation occurrence with a binomial distribution mixed-effects model, 3) examine controls on the magnitude of sediment yield using a gamma distribution mixed-effect model, and 4) assess relative importance of variables relating to sediment yield using random forest models. Threshold rainfall intensities for generating erosion at the study sites were 32-38 mm/hr for MI5, 11-18 mm/hr for MI15, 7-13 mm/hr for MI30, and 5-8 mm/hr for MI60. Across all models of erosion occurrence and magnitude of sediment yield, maximum rainfall intensity and total precipitation were primary drivers of erosion. There was no evidence of a mulch treatment effect on

sediment occurrence or magnitude, likely resulting from insufficient initial mulch cover and a high-intensity storm that removed much of the mulch shortly after it was dropped on the hillslopes. Contributing area, slope mean, and slope length showed no influence on sediment yield, likely due to limited variation in these variables between hillslopes. These results highlight the importance of mulch cover that will stay in place under extreme rainfall. Future hillslope-scale studies should consider dropping mulch during a time period that is unlikely to have high intensity rainfall and explore mulch materials and application methods that will better ensure adequate initial cover for reducing hillslope-scale erosion.

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1 INTRODUCTION

The western US is experiencing an increase in forest wildfire activity attributed to rising temperatures and earlier spring snowmelt, and ongoing climate change may extend the duration of fire seasons and intensify fire severity in the coming decades (Westerling, 2016; Parks & Abatzoglou, 2020). Wildfires have profound impacts on the environment, causing severe damage to vegetation and surface cover, while also altering soil properties (MacDonald & Stednick, 2003; MacDonald & Larsen, 2009; Morris & Moses, 1987). These changes contribute to increased soil water repellency and decreased surface roughness (Morris & Moses, 1987; Benavides-Solorio & MacDonald, 2005; Larsen et al., 2009). Consequently, the soil becomes more hydrophobic, impeding water infiltration, and promoting overland flow (Morris & Moses, 1987; Larsen et al., 2009). The combination of reduced surface cover and increased soil hydrophobicity poses a significant risk during high-intensity summer storms, leading to substantial hillslope erosion (Benavides-Solorio & MacDonald, 2005). The heightened risk of erosion, flash-flooding, and debris flows after wildfires can introduce safety hazards, while also resulting in substantial economic losses (Moody & Martin, 2001; Cameron Peak Fire BAER Executive Summary, 2020). Additionally, these events can adversely affect recreation, aquatic biota, and water quality and availability (Moody & Martin, 2001). Consequently, emergency response plans have been devised to quickly identify areas with heightened hazards, and these plans subsequently transition into long-term post-fire rehabilitation efforts (Larimer County Cameron Peak Fire Risk Assessment Summary, 2021).

After a wildfire, water resource managers have implemented various treatments to rehabilitate the landscape and mitigate impacts of fire. Wood-strand, wood-shred, wheat-straw mulching, seeding, erosion barriers and contour felling, are commonly used as post-fire restoration techniques to reduce hillslope erosion (Wagenbrenner et al., 2006; Robichaud et al. 2013a; Robichaud et al., 2020; Girona-García et al., 2021). These treatments aim to stabilize affected areas, enhance protective ground cover, and facilitate vegetation recovery by increasing soil infiltration and reducing runoff (Girona-García et al., 2021). These different types of treatments have shown variable effectiveness for reducing sediment yields (Wagenbrenner et al., 2006). Previous studies have shown that seeding techniques had no impact on either ground cover or sediment yield, making them ineffective due to their limited ability to increase ground cover (Wagenbrenner et al., 2006; Groen & Woods, 2008). Other literature indicates that post-fire seeding in forests provides minimal short-term soil protection, with inconclusive effects on non-native species invasion, though there is limited data on the long-term impacts of seeding (Peppin et al., 2010). Treatments that provide immediate cover are generally more effective for short-term erosion reduction because of the strong correlation between ground cover and sediment yield (Larsen et al., 2009), although the yields varied significantly depending on the treatment applied (Wagenbrenner et al., 2006).

Thus, the most effective treatments, such as mulching, are those that immediately increase ground cover, facilitate vegetative regrowth, and protect the soil from overland flow (Wagenbrenner et al., 2006; Robichaud et al., 2020). According to Wagenbrenner et al. (2006), mulching treatments resulted in a reduction of sediment yields by at least 95% compared to untreated areas. However, effectiveness of treatments may decrease over time, emphasizing the importance of considering the timing of implementation to mitigate post-fire impacts, and in the case of mulching, application rate (Robichaud et al., 2020; Girona-García et al., 2021).

Most of these prior studies on mulch effectiveness have been conducted at the plot-scale, with some studies extending to hillslope scales (Schmeer et al., 2018; Wilson et al., 2018; Girona-García et al 2021). Advantages of plot-scale studies include the ability to replicate plots with similar characteristics like areas, shapes, slopes and lengths, etc. A hillslope-scale experiment is challenging to implement because of the large areas to be treated and introducing complexities for comparison between varying mulch treatments. Prior research at the hillslope-scale has not been designed experimentally where replicate hillslopes received different levels of mulch treatments (Girona-García et al., 2021). The current study applies a replicated-controlled experimental approach at the hillslope-scale in which pairs of similar hillslopes were each treated with different levels of mulch. The objectives of this study were to 1) evaluate the impact of post-fire mulching in reducing erosion at the hillslope-scale, and 2) identify potential landscape features and precipitation metrics contributing to the occurrence and magnitude of sediment yield.

2 METHODS

2.1 Study site

The 2020 Cameron Peak Fire started August 13, 2020, until completely contained on December 12, 2020. It was the largest wildfire in Colorado's recorded history, burning a total of 208,913 acres in Larimer County with ~36% of the area having moderate to high soil burn severity (Larimer County Cameron Peak Fire Risk Assessment Summary, 2021; Cameron Peak Fire BAER Executive Summary, 2020). Extreme temperatures with low humidity, rugged terrain, and winds gusting at speeds reaching over 70 mph, were some of the main contributing factors leading to rapid fire spread (Cameron Peak Fire BAER Executive Summary, 2020). The fire primarily affected the Cache la Poudre (CLP) and Big Thompson River basins. The focus site for this study is in the Bennett Creek watershed within the Arapaho-Roosevelt National Forest. Bennett Creek is a tributary of the South Fork of the Poudre River in the CLP River Basin.

The study area has a temperate semi-arid climate, with mean annual temperature of 5°C, and mean annual precipitation of 470 mm (PRISM Climate Data). Precipitation during the summer months accounts for approximately 30% of the annual mean, and some of this precipitation comes from convective storms with potentially high-intensity rainfall. Swale elevation at the Bennett Creek MW study site ranges from 2,246-2,546 m (USGS 3D Elevation Program). Aspect ranges from 4-185 deg, and slope ranges from 10-26 deg (USGS 3D Elevation Program). The bedrock geology at this site is primarily equigranular granite (National Geologic Map Database, USGS/AASG). The soils in this area originate from colluvium and residuum parent material. The dominant soil type is Bullwark-Catamount, characterized by gravelly and cobbly sandy loam textures, with a soil profile depth ranging from 0-0.8 m (Soil Survey Staff, NRCS, USDA). Pre-fire vegetation was primarily comprised of lodgepole and ponderosa pine trees and grass and forb understory. Post-fire vegetation is regenerating primarily as grasses, forbs, and lodgepole pine.

In 2021, 850 acres of the Bennett Creek study site were mulched, and in July 2022, an additional 361 acres were mulched (Figure 1). The initial mulch applications in 2021 were designed to examine the effects of mulching at the catchment scale. The study area includes six adjacent sub-catchments that are tributary to Bennett Creek. These catchments were mostly burned at moderate severity (Figure 1). Three of the watersheds received mulch, and three remained unmulched (Figure 1). The additional mulching in 2022 was designed to examine the effects of varying application rates of mulch at the swale scale.

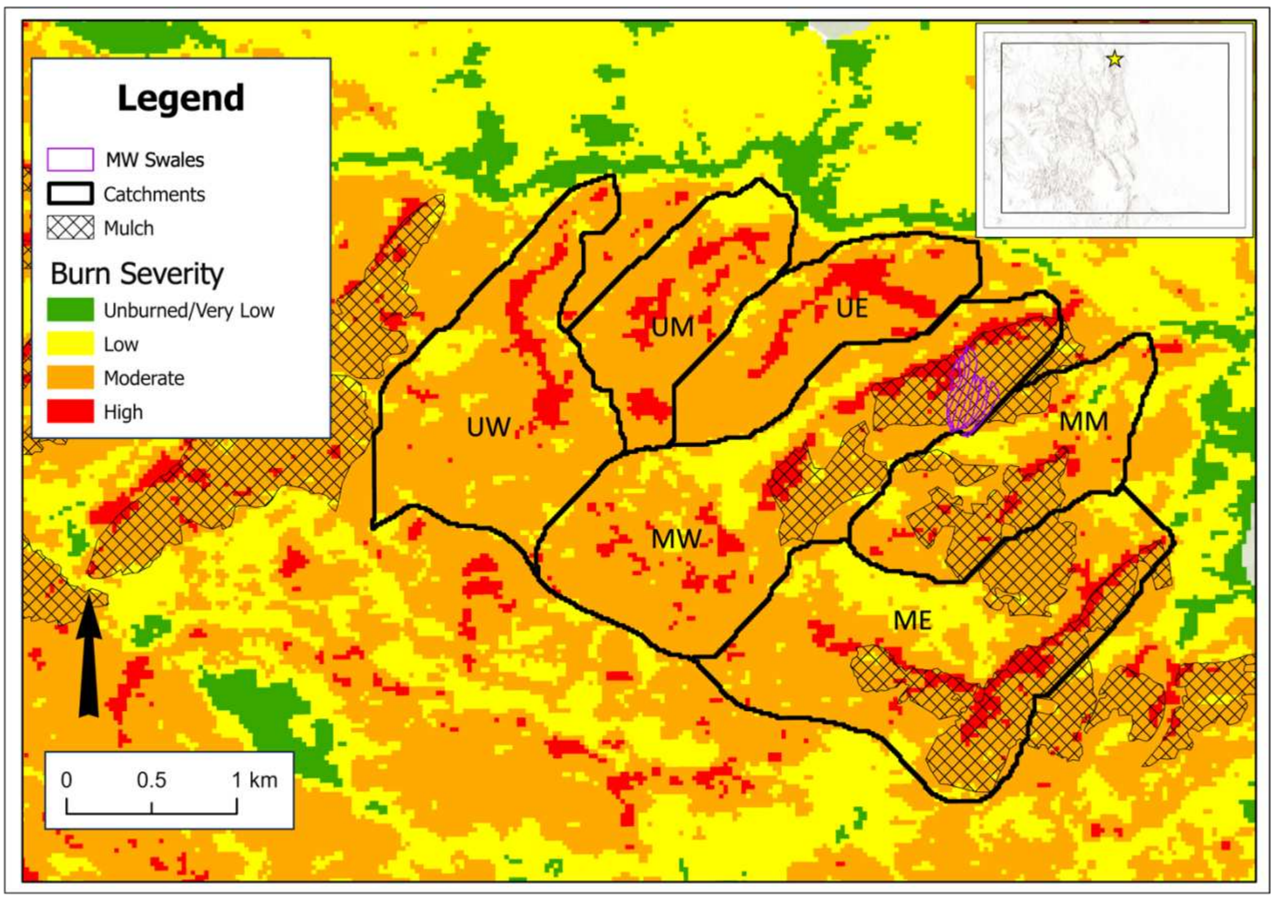

Figure 1. Tributary catchments to Bennett Creek with burn severity. Mulch was applied in the Mulch West (MW), Mulch Middle (MW), Mulch East (ME) watersheds in August 2021, with additional mulch applied July 2022.

Eight hillslopes were selected within the Mulch West (MW) sub-catchment for additional mulch treatments. Swales were chosen to be adjacent to one another with similar shapes and sizes, which was expected to minimize variability of rainfall and other factors affecting erosion. These hillslopes burned at moderate severity and had limited ground cover and mostly dead trees after fire. Three hillslopes were treated with two layers of mulch, two with one layer of mulch, and three hillslopes remained unmulched, control hillslopes (Table 1). Most of the hillslopes are convergent swales, but two planar hillslope sites were also added (Figure 2).

Table 1. Hillslope information and characteristics for the Bennett Mulch West study site. Length is defined from the top of the hillslope down to the fences.

Fence	Swale Name	Swale ID	Mean Slope (%)	Length (m)	Area (ha)	Treatment	Number of Mulch Drops
1	Mulch West 1	MW1	40.7	350	0.9267	Control	0
2	Mulch West 2	MW2	39.9	300	1.081	Two layers	23 total
3	Mulch West 3	MW3	43.5	260	0.3157	Two layers	
4	Mulch West 3 Planar 1	MW3p1		235	0.0075	Two layers	
5	Mulch West 4	MW4	44.6	250	0.4654	One layer	11 total
6	Mulch West 5	MW5	45.6	245	0.4816	One layer	
7	Mulch West 7	MW7	40.2	285	0.3076	Control	0
8	Mulch West 8 Planar 2	MW8p2		100	0.0020	Control	0

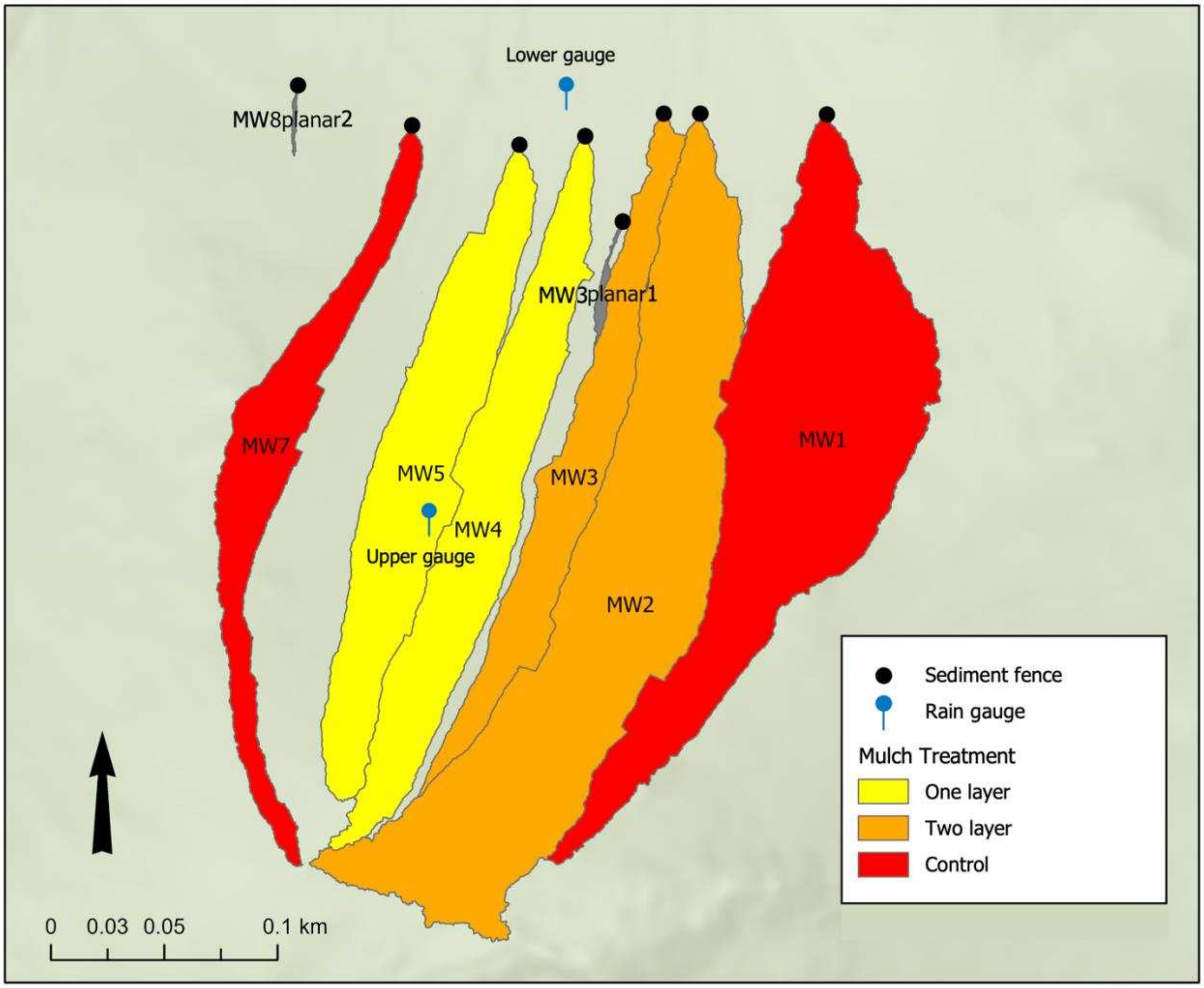

Figure 2. Bennett Mulch West catchment swales with associated mulch treatments, sediment fence locations, and rain gauge locations.

2.2 Instrumentation and data collection

Eight sediment fences were installed amongst the swales; six fences positioned to follow flow paths of swales on the convergent hillslope, and two fences in the planar plots for comparison between planar and convergent hillslope erosion (Figure 2). Contributing areas were then delineated based on fence location using Digital Elevation Models (DEMs) derived from drone imagery and Structure from Motion (SfM) methods, which had a spatial resolution of 6.4 cm. The size and dimensions of sediment fences were determined based on the width of each swale. The fences were constructed in a U-shape design to capture sediment eroded from the hillslope, and subsequently transported by runoff (Figure

3a). The U-shaped dimensions of the fence were established by vertically posting wooden stakes on the exterior of the landscaping fabric. Bailing wire was used to reinforce the wooden stakes. Although other studies used rebar to support the wooden stakes (Schmeer, 2014; Benavides-Solorio & MacDonald, 2005), underlying soil within swales at the MW study site was too rocky. Instead, two additional wooden stakes were placed in an "X" shape to strengthen fence posts (Figure 3b). To secure the landscaping within the fence, several eight-gauge landscape staples were driven into the fabric. Eight-gauge landscape staples were also placed along the fabric's edge, the chord of the semi-circle fence shape, at the soil surface. This placement aimed to minimize potential sediment transport from going underneath the fabric during high-intensity storm events.

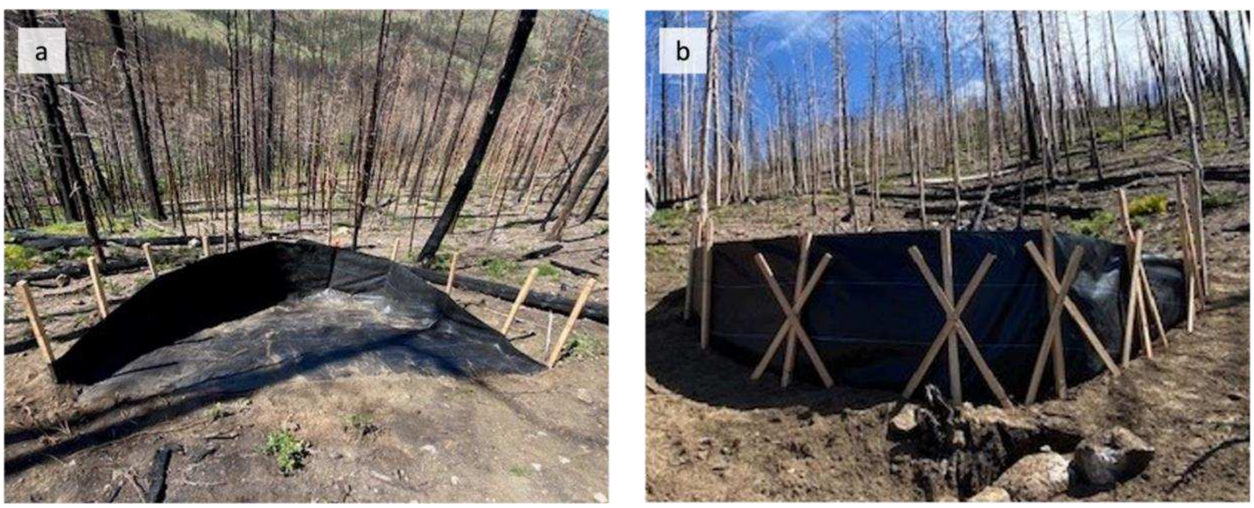

Figure 3. View of sediment fences (a) from above, looking downslope (b) from below, looking upslope.

Game cameras were installed facing each fence, to document sediment movement during storm events. Three game cameras were installed at MW1, MW4, and MW7 on 7/11/2022. All other fences had cameras installed on 8/10/2022. Pictures were taken every five minutes. However, not all footage for each fence throughout the summer 2022 season is available due to technology issues and limited personnel capacity.

To measure rainfall, two Onset Smart Sensor Rain Gauges, were installed on 7/1/2022, and were mounted on T-posts 1 m from the ground. The rain gauges measured intensity with 0.3 mm resolution, and 1% accuracy for rainfall rates up to 0.025 m/ hr (Onset). HOBO Pendant Event Data Loggers were attached to each rain gauge and recorded tips during storm events to measure rainfall. Rain gauges provided information to compute rainfall rates, times, and duration. One rain gauge was placed down by fences at the convergent swales (lower gauge) and the other further up the hillslope at the (upper gauge) (Figure 2). The lower gauge was installed approximately mid-way across the hillslope, and the upper gauge near fence 4 (MW3p1).

Following major storm events, the sediment fences were cleared to measure the amount of sediment that had been deposited. Sediment from each fence was shoveled into 19 L buckets, and the wet mass measured with a PESOLA hanging scale, recorded at least to the nearest 0.05 kg (PESOLA Präzisionswaagen AG, Switzerland). The sediment from the buckets was dumped downslope beside the fence, to prevent it from re-entering the fence during subsequent storms. This process was repeated until each fence was completely emptied.

For each sediment generating storm event, sediment samples weighing a minimum of 0.1 kg were collected from each fence and placed in Ziploc bags. The samples were taken every meter along the central axis of the fence, starting from downslope (1 m inside the fence) and progressing upslope towards the edge of the fabric, or until the end of sediment deposition from the specific storm event (Figure 4). Due to the rocky nature of the sediment, samples were collected without known volumes. Instead, total depth of each sample was recorded. Samples were later brought to the lab to determine water content.

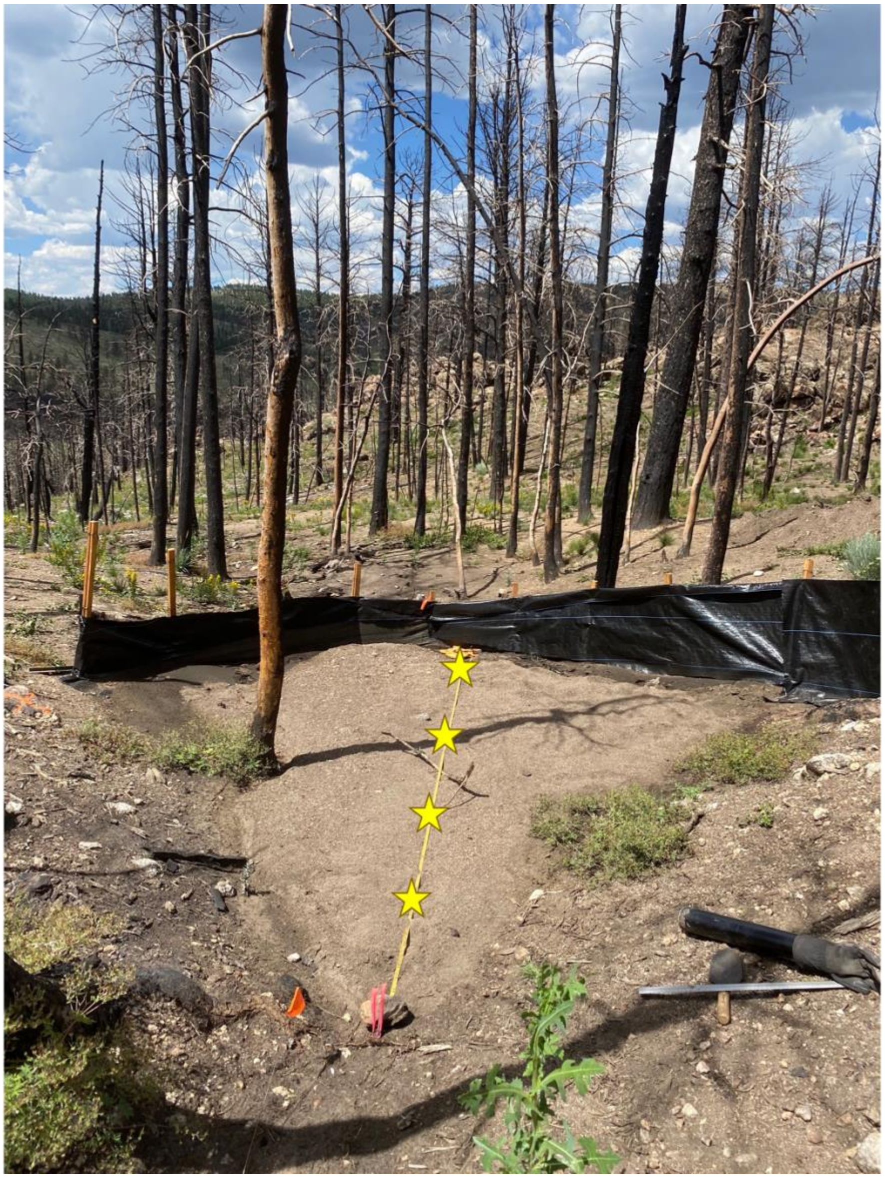

Figure 4. Location of samples collected at each fence at 1 m, 2 m, 3 m, and 4 m, where sediment was present.

The wet samples were weighed within 24-48 hr, using a Mettler PC 4400 scale. Accuracy of the scale was verified using calibration weights of 2 g, 50 g, 100 g, 200 g, and 500 g. Wet mass sample masses were recorded and adjusted to account for the additional mass of the Ziploc bags. The samples were dried in an oven at 105°C for 28 hr, and then re-measured. To calculate gravimetric water content, wet mass was subtracted from dry mass, and divided by the wet mass. The arithmetic average water content for each fence was calculated using the values from each sample taken along the central axis of the fence (Figure 4). This calculation was performed for each storm event, allowing for comparisons of average water content across different storms. The sample water content was used to adjust the wet sediment mass to a dry mass for sediment load (Mg). The sample's dry mass was normalized by swale contributing area to get sediment yield (Mg/ha).

To evaluate the effect of wood mulching treatments, surface cover surveys were conducted from 7/1/2022 to 10/12/2022. A total of 20 surveys were completed across all eight swales to capture changes in cover over time, both before and after the application of mulching treatments. Measurements were collected pre- and post-mulching and split into four groups: pre-mulch (7/1 to 7/13), post-mulch 1 (7/27), post-mulch 2 (8/10), and post-mulch 3 (9/21 and 10/12). Transects were established along the central axis of each swale, and across the entire length of the swale up to the hillslope ridge. The swales varied in length, ranging from 100 m to 350 m with an average length of approximately 253 m (Table 1). Measurements were collected at 5 m intervals, starting at the base of the swale above the sediment fence. This sampling approach ensured that there were enough data points, at least 100 points per transect, to capture a comprehensive representation of the swale's surface cover. A tape measure was used to mark survey points within each transect. Pin flags were placed to mark locations of the points in the transect and ensure that cover could accurately be re-measured at the same points in subsequent surveys. A precise location with latitude/longitude with < 0.5 m accuracy was recorded for each point along the transect using a Juniper Systems Geode GNSSS Receiver (Juniper Systems, Logan, Utah). A 36.5 cm by 30.5 cm survey grid, split up by 4.4 cm by 4.4 cm squares, was used at each point to measure cover. For each point in the transect, measurements were taken from nine predetermined squares of the grid and randomly selected within each square (Figure 5). The nine measurements were tallied by surface cover type. Cover classes were categorized as bedrock, rock > 10 cm diameter, rock 1-10 cm, rock < 1 cm diameter, bare soil, tree standing, tree fallen, live vegetation, litter, mulch, and root (Schmeer, 2014). Ground cover is defined as the total surface cover types (vegetation, litter, and mulch), excluding rock and bare soil (other cover).

Figure 5. Grid used for ground cover surveys. The red squares, 4.4 cm by 4.4 cm outline the squares where measurements were taken (1 measurement/square).

Drone imagery and supervised classification techniques were applied to analyze spectral characteristics from pixels in the drone images, specifically by using RGB bands to distinguish between ground cover and vegetation, in order to quantify vegetation cover over the season. Two sets of binary classification rasters were created with one set classifying live vegetation, and another classifying dead vegetation. The live and dead vegetation rasters were then combined to create a single set of binary total vegetation (live and dead) rasters. The live vegetation rasters were created by calculating the spectral index, RGBVI (Equation 2.2), for each pixel, then classifying all pixels with index values greater than 0.08 as live vegetation. The dead vegetation rasters were created by calculating a dead vegetation spectral index (Equation 2.3), then classifying all pixels with an index value greater than 0.75 as dead vegetation. The total vegetation rasters were validated against a set of 862 manually classified test points using Cohen's kappa, which yielded a kappa value of 0.63.

$$\frac{green^2 - (\text{red} \times \text{blue})}{green^2 + (\text{red} \times \text{blue})} \tag{2.2}$$

$$\frac{red^2}{(blue^2) + (green^2)} \tag{2.3}$$

2.3 Data analysis and modeling

2.3.1 Precipitation

Rain data recorded from the data loggers were separated into single storm events, with a new event starting once > six hours without rain had elapsed. Events with less than 1 mm of precipitation were excluded. For each event the following metrics were calculated: total precipitation depth (mm), storm duration (hr), and maximum rainfall intensities (MIs) at 5-minute, 15-minute, 30-minute, and 60-minute intervals.

2.3.2 Sediment producing events and thresholds

Each precipitation event was evaluated to determine if the storm produced sediment. Game camera images for the time period of each event were reviewed to see if sediment entered the fence. A sediment producing event is defined as any event where enough sediment was deposited in the fence to collect a minimum 0.1 kg sample. Where fence data for a single storm could not be obtained, the sediment producing response was determined by comparing MIs to the 7/5 event, which had the lowest sediment load. If an event with missing sediment load data had MIs lower than those of the 7/5 event, it was assumed that no sediment was produced. If there was a response (sediment produced) camera images were then used to evaluate whether the fence overtopped with water and/or sediment. This information was used to determine when events had under-catch of sediment, meaning the amount of sediment measured was likely less than the actual quantity eroded.

For each site, the values of MIs for sediment-producing and non-producing events were evaluated to determine whether there was a clear intensity threshold that led to erosion. A clear threshold is where all storms above the threshold produced sediment, and all storms below the threshold did not.

2.3.3 Controls on sediment yield

The sediment fence sampling yielded a relatively small sample size (n = 27) for sediment producing events, compared to non-sediment producing events (n = 130). Considering the small sample size, we took a multifaceted approach to assess the impact of landscape features and precipitation factors on the occurrence and magnitude of sediment yield (mass of sediment divided by contributing area). First, we analyzed univariate correlations for all potential predictors of sediment yield: precipitation metrics, pre/post mulching (categorical), vegetation (percent cover), treatment (categorical no mulch, 1 drop, 2 drops), contributing area, slope mean, and slope length. Then, to assess multivariate relationships and potential predictor interactions, we developed separate linear models to explain sediment generation occurrence and quantity of sediment generated. Sediment yield values were not normally distributed (Shapiro-Wilks $p > 0.8$), and log-transforming the data did not lead to a normal distribution either ($p = 0.1$). Additionally, random variation among swales and precipitation events justified using a mixed-effects modeling approach with precipitation event nested within swale ID.

To model sediment generation occurrence, we developed a mixed-effects model with a binomial distribution and logit link, estimating the probability of yes sediment or no sediment (presence of sediment or absence of sediment). For sediment yield, we developed a mixed-effects model with a gamma distribution and inverse link (sediment yield with a minimum value of zero, increasing to the highest sediment yield value). Because all precipitation intensity metrics were highly correlated with sediment yield in the univariate correlations, and MI15 was the highest correlated, we used MI15 as the only precipitation metric during mixed-effects model development. All mixed-effects modeling was completed using the base 'stats' package in R. Models were compared using estimates of pseudo R^2 (1-deviance/null deviance) and likelihood ratio tests.

We used random forest modeling as a third line of modeling support, and to test all potential predictors against the dependent variable, sediment yield. This approach is important to assess how all parameters are related to the dependent variable, considering interactions, without also risking the so-called "large *P*, small *n* problem" (Barnard et al., 2019) where an excess of potential predictors without sufficient observations can lead to model overfitting and unreliable coefficient estimation. Similar to the mixed-effects modeling, we developed separate random forest models for occurrence of sediment generation and for sediment yield. Random forest models were developed using the 'randomForest' package in R. Models were developed to test two variables for each tree on over 1000 trees and evaluated using variable importance plots and out-of-box error estimates for the occurrence model, and percent variance explained for the regression model of sediment yield.

3 RESULTS

3.1 Precipitation

The thirty-year normal (1991-2020) of mean monthly precipitation for the study site in July is 54 mm, followed by 50 mm in August, and 38 mm in September, totaling 142 mm for the summer season (PRISM Climate Data). From the rain gauges at Bennett Creek MW, during the time period of this study in summer 2022 (7/1/2022, to 9/27/2022), precipitation totaled 76 mm in July, 43 mm in August, and 24 mm in September, for a total of 143 mm for the summer season. This is similar in comparison to the observed long-term average annual summer precipitation collected from PRISM.

The Estes Park station was selected to compare the highest maximum rainfall intensities to the Bennett Creek MW study site because of their similar elevations. In general, Bennett Creek MW rain gauge maximum intensities (MIs) were higher than the Estes Park station's MIs for the 1-yr recurrence interval storms and aligned more closely with 5-yr and 10-yr intervals (Table 2). During the summer of 2022, at the Bennett Creek MW rain gauge, the maximum 60-minute rainfall intensity (MI60) was 17 mm/hr, close to the MI60 of 15 mm/hr at the Estes Park station for a 1-yr recurrence interval. MI30 at the Bennett Creek MW rain gauge was 33 mm/hr, which is most similar to MI30 at the Estes Park station at a 5-yr recurrence interval, 34 mm/hr. However, shorter rainfall MI durations, MI5 and MI15 at the study site were closer to 10-yr recurrence interval MIs at Estes Park. The 10-yr MI5 at Estes Park was 113 mm/hr compared to Bennett's 103 mm/hr, and MI15 68 mm/hr compared to Bennett's 66 mm/hr. This suggest that rainfall occurring at Bennett Creek during the summer of 2022 included more extreme storm events than there would have been in an average year.

Table 2. Comparison of highest maximum intensities (MI) for summer 2022 at the study area to 1, 5, and 10-yr recurrence interval storm at Estes Park station (NOAA Atlas 14, Volume 8).

Precipitation Station/Gauge	MI5 (mm/hr)	MI15 (mm/hr)	MI30 (mm/hr)	MI60 (mm/hr)
Estes Park station (1-yr)	61	36	23	15
Estes Park station (5-yr)	90	54	34	20
Estes Park station (10-yr)	113	68	42	25
Bennett Creek MW 2022	103	66	33	17

During the 2022 season there were 22 rain events at the fence sites. Rain data collected between the upper and lower gauges were highly correlated (Pearson's correlation coefficients ranging from $r = 0.84$ to $r = 0.99$ for the various MI metrics), so gauge averages were used for precipitation analyses. Total precipitation for the summer 2022 storm events ranged from 1 mm to 31 mm (Figure 6). The event with the highest total precipitation took place on 8/15 with 31 mm of rainfall recorded.

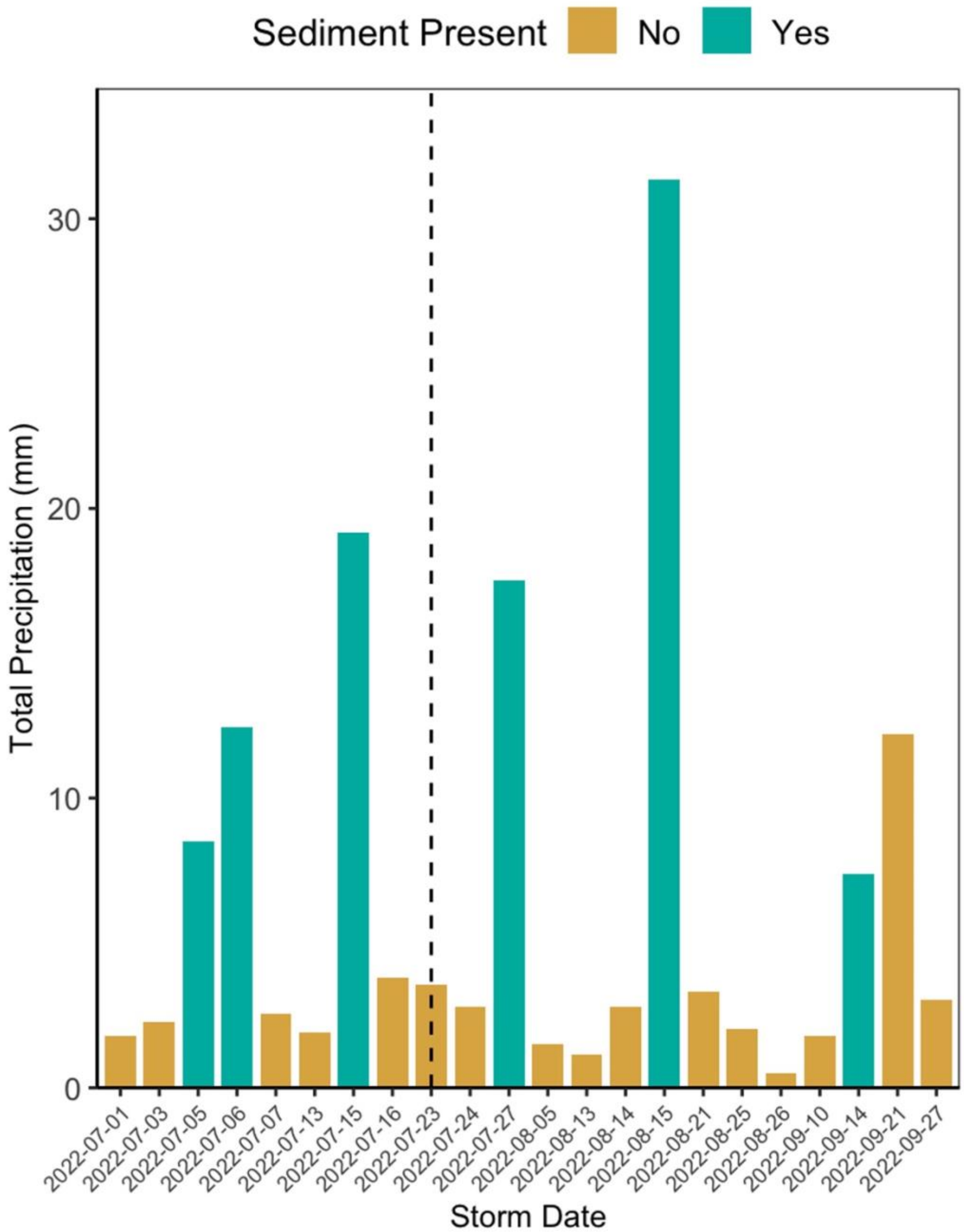

Figure 6. Total precipitation for each storm event, with colors indicating whether the event produced sediment. Black dashed line indicates date of mulching (7/22).

Over the summer 2022 season, MIs for sediment producing events varied from 3 mm/hr to 102 mm/hr (MI5), 1 mm/hr to 66 mm/hr (MI15), 0.5 mm/hr to 33 mm/hr (MI30), and 0.3 mm/hr to 17 mm/hr (MI60). The highest storm intensities occurred during the storm on 7/27, followed by storms on

7/15 and 7/6 (Figure 7). The 7/27 storm would have been approximately a 10-yr recurrence interval storm for MI5 and MI15, 5-yr storm for MI30, and 1-yr storm for MI60 (Table 2). The second highest intensity storm occurred on 7/15 and was approximately a 1-yr recurrence interval storm for almost all intensities, MI5 of 61 mm/hr; MI30 of 24 mm/hr, and MI60 of 12 mm/hr. For 7/6 storm (third highest intensity event), all MIs recorded at the Bennett Creek MW site were below those at the 1-yr recurrence interval at the Estes Park station.

Of the 22 storm events, six produced sediment in at least one of the eight fences. There were five storms where precipitation exceeded 10 mm total (Figure 6). The storms with precipitation greater than 10 mm almost always yielded sediment, aside from the 9/21 storm. The 9/21 storm had lower maximum 5-minute and 15-minute intensities (MI5, MI15) compared to all other sediment producing storms (Figure 7).

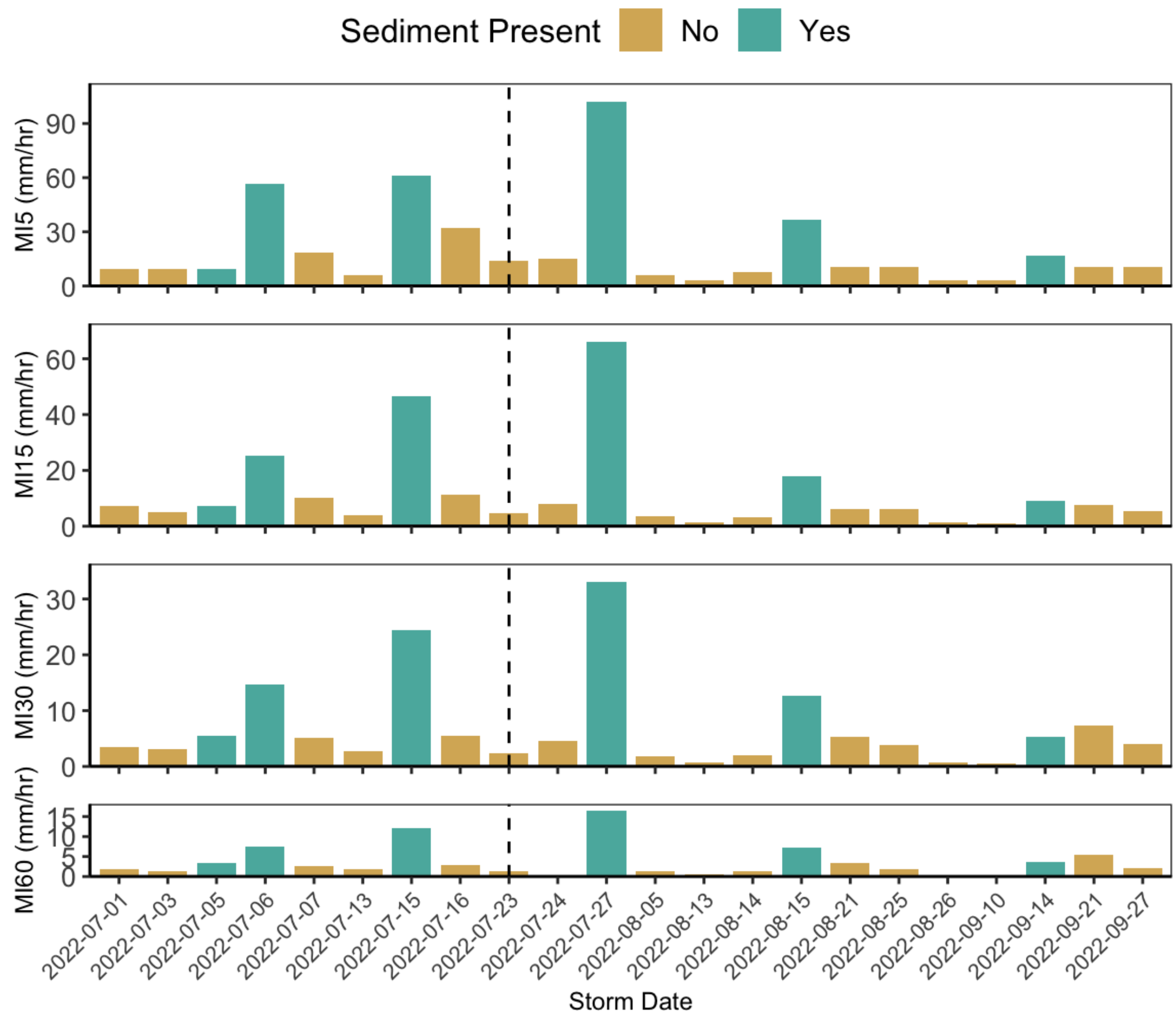

Figure 7. Maximum intensities (MI) for precipitation events over the summer 2022 season. MIs are calculated at 5-minute, 15-minute, 30-minute, and 60-minute intervals. Black dashed line indicates date of mulching.

3.2 Ground cover

Over the summer 2022 season, bare soil was the majority of the surface cover across swales (44-76%), followed by mulch (8-39%), with vegetation only representing 5-18% (Figure 8). Mulch cover was highest for the first surface cover survey on 7/27, conducted five days after mulching: 23-44% across all mulched swales, and 39-44% for swales treated with two layers of mulch (MW2 and MW3). For the following 8/10 survey, percent mulch cover dropped almost in half for the swales treated with two layers of mulch (18-19%), likely resulting from the 7/27 storm, which had the highest intensities for the summer 2022 season. From the first to the third survey, where mulch decreased, vegetation increases for swales treated with one layer of mulch (MW4 and MW5). The 7/27 surface cover survey was conducted during the day, right before evening 7/27 storm event took place.

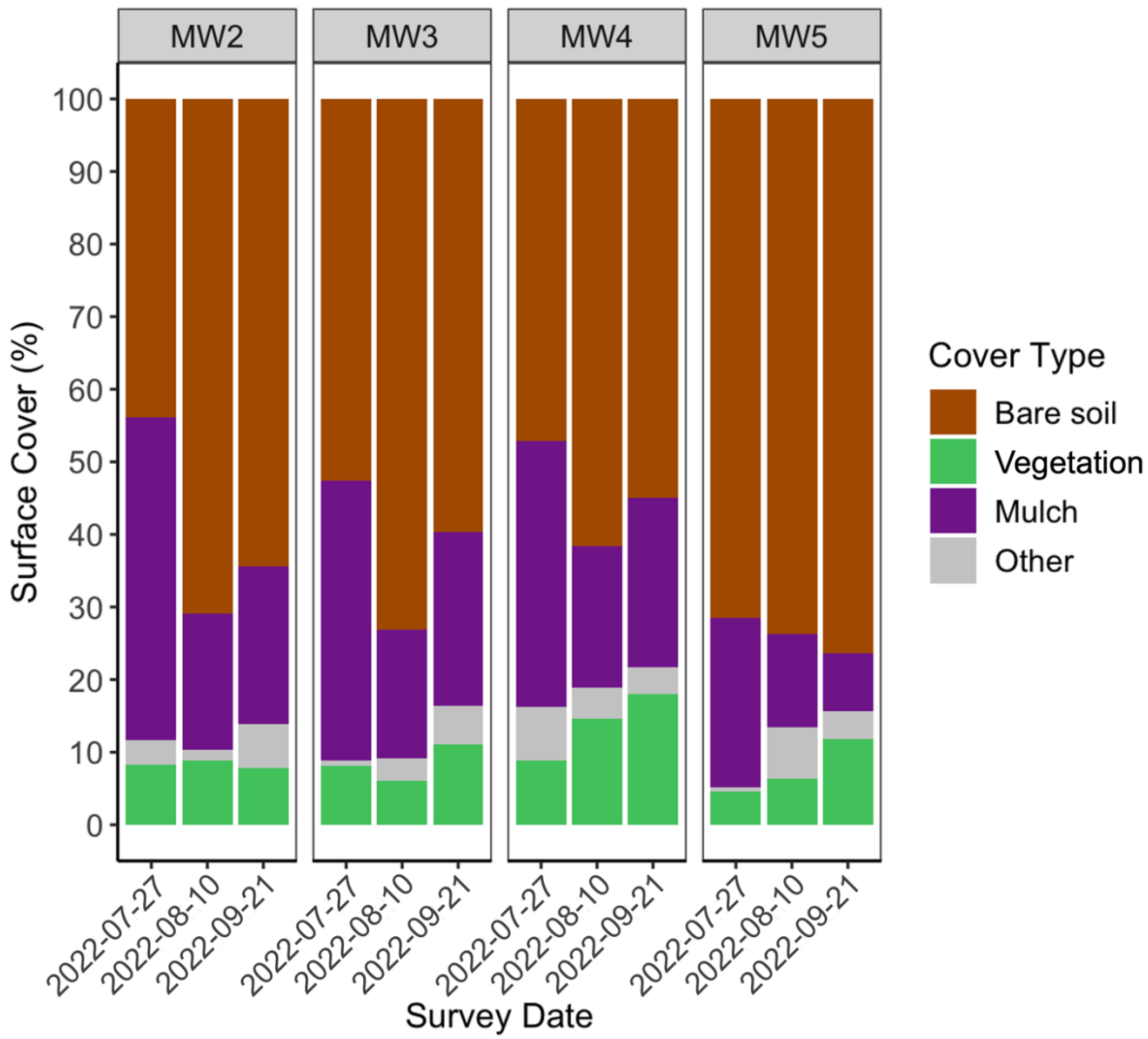

Figure 8. Surface cover percent over the summer 2022 season. Surveys were conducted three times post mulching. MW2 and MW3 were treated with two layers of mulch, and MW4 and MW5 with one layer of mulch.

3.3 Sediment load and sediment yield

Across all swales, the total sediment load was the highest in MW1, a control swale (11.0 Mg, almost double the load compared to other swales). Sediment load for MW5 (one layer of mulch) was 5.8 Mg, the second highest load amongst swales, and 5.6 Mg for MW3 (two layers of mulch), the third highest load (Table 3). Compared to convergent swales, planar swales had the lowest sediment loads (0.15-0.31 Mg).

When sediment loads were normalized by drainage area, MW1 was no longer the highest sediment producer. The two control swales, MW1 and MW7, had 11.9 Mg/ha and 14.7 Mg/ha sediment yields, respectively. For convergent swales treated with two layers of mulch, MW3 had the highest total sediment yield (17.8 Mg/ha), while MW2 had the lowest total sediment yield (3.2 Mg/ha). The swales treated with one layer of mulch, M4 and MW5, had the second and third lowest sediment yields, at 6.5 Mg/ha and 12.0 Mg/ha respectively. The drainage areas of the planar swales are uncertain because they cannot be easily delineated with a standard GIS algorithm; therefore, they are not included in sediment yield comparisons.

Apart from the planar swales, the values of sediment load and sediment yield are under-estimates because the fences overtopped multiple times during storms, causing sediment loss. MW1, a control swale, overtopped for the greatest number of events. Of the four events with fence over-topping, MW1 overtopped at least 22 times according to time lapse camera photos. For most events, MW1 overtopped with water (sediment overtop), leading to underestimates of sediment yield. MW1 overtopped with water only during the 8/15 storm, which was the storm with the fourth highest maximum intensity (Figure 7). MW2 and MW4 were knocked out during the 7/27 storm, and data could not be collected, making comparisons inaccurate across the summer 2022 season (Figure 9).

Table 3. Summary sediment fence data from 7/1 – 9/27.

Fence	**Swale ID**	**Treatment**	**Total Sediment Load (Mg)**	**Total Sediment Yield (Mg/ha)**	**Number of Events with Sediment Produced**	**Number of Events with Fence Sediment Overtop**
1	MW1	Control	11.0	11.9	5	3
2	MW2	Two layers	3.49	3.2	6	1
3	MW3	Two layers	5.63	17.8	6	2
4	MW3p1	Two layers, planar	0.31	*41.9*	4	0
5	MW4	One layer	3.01	6.5	5	2
6	MW5	One layer	5.78	12.0	6	2
7	MW7	Control	4.52	14.7	6	2
8	MW8p2	Control, planar	0.15	*75.6*	5	0

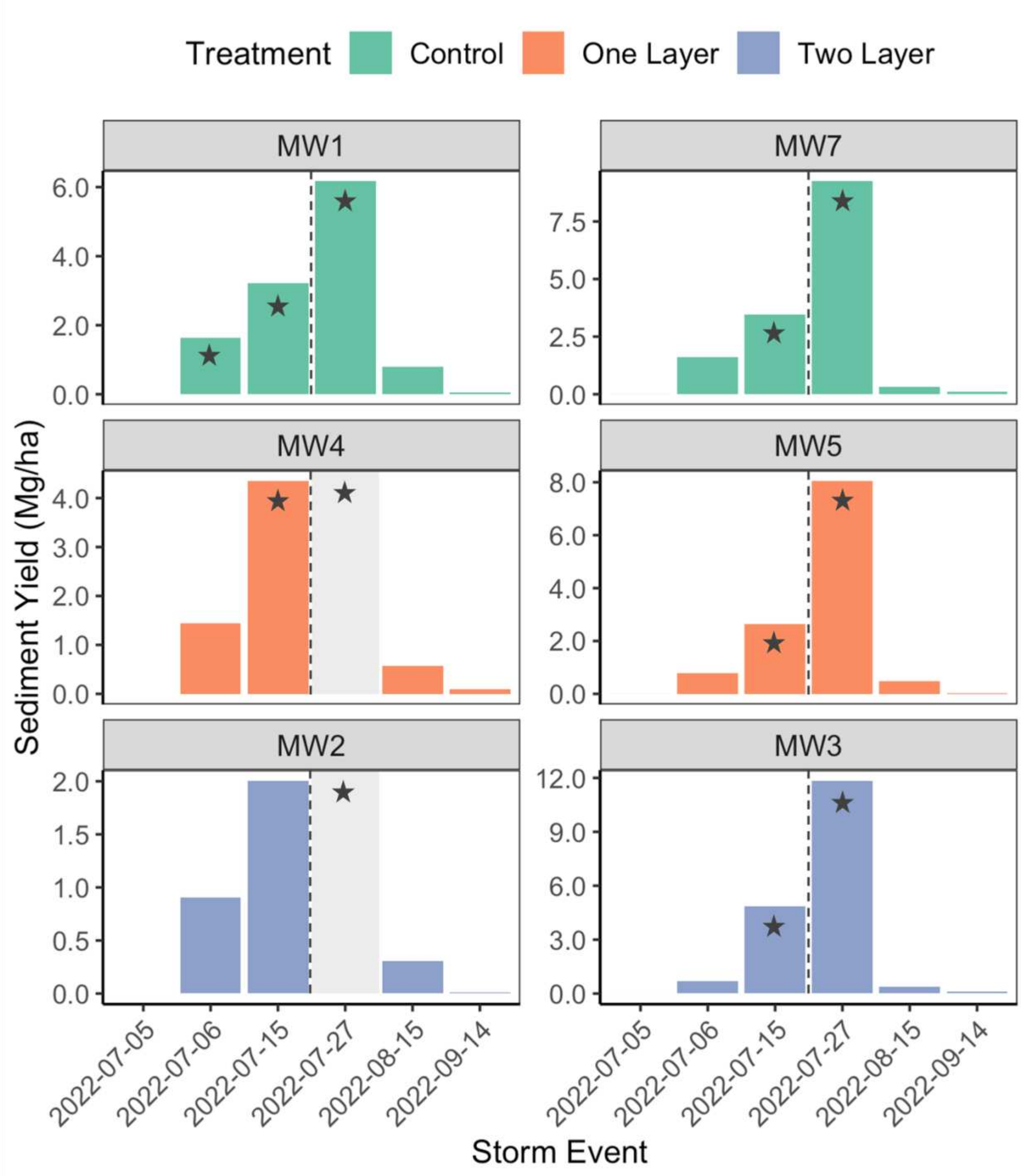

Figure 9. Sediment yield by swale and treatment type. Black dashed line indicates date of mulching and stars represent fence overtopped with sediment. Grey bars indicate fences that were broken by the sediment from the 7/27 storm; because the sediment load was high enough to make the fences fail, the sediment yield was likely at least as high as the sediment yield from 7/15, when the fences did not fail.

For convergent swales, 23-27% of rainfall events produced sediment. Most of the sediment-producing storms (7/6, 7/15, 7/26, and 8/15) produced sediment in all fences. Exceptions are the 7/5 storm, which produced sediment in five out of eight fences and the September 14 storm, which produced sediment in six out of eight fences – and all convergent swales (Figure 10). Four rain events produced most of the sediment load in summer 2022; two events prior to mulching in July (7/6, 7/15), and two events after mulching (7/27, 8/15). The 7/27 storm, the first storm with greater than 10 mm of precipitation storm post-mulching, produced the highest sediment yields out of all storm events. The next highest sediment producing events were 7/15 and 7/6 (Figure 9). Despite being the event with the highest precipitation total of the season, the 8/15 event had lower intensity and did not produce as much sediments as the July storms (Figures 7 and 9).

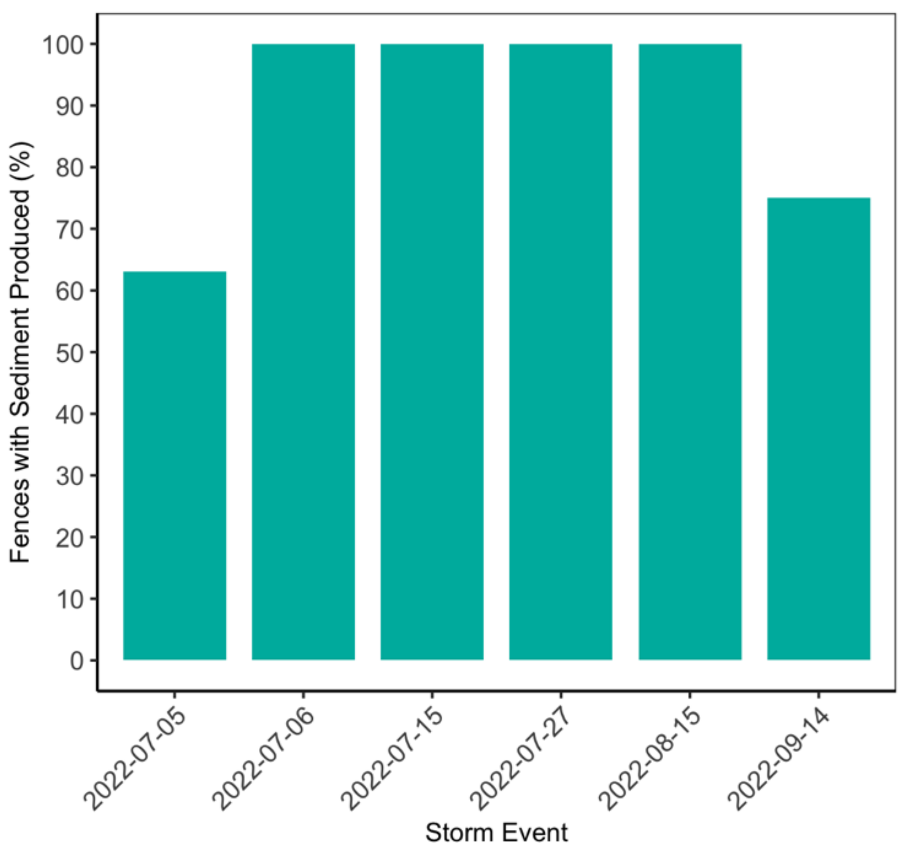

Figure 10. Percent of fences with sediment produced by storm event.

3.3.1 Sediment producing thresholds

The maximum intensities of storms were evaluated to determine whether a threshold rainfall intensity causes erosion at the study area. All sites produced sediment for the storms with the four highest intensities, whereas no sites produced sediment for the storm with the fifth highest intensity. The range of maximum intensities between these two storms (4^{th} and 5^{th} highest intensity storms) is marked as a threshold range in Figure 11. This is not a perfect threshold because for all convergent swales at all intensities, there are two sediment-producing events below the threshold. Of those, one event occurred prior to mulching, and one post-mulching (7/5 and 9/14) (Figure 11). Those two events had the lowest sediment yields across swales, compared to other sediment producing storms. Sediment yield was less than or equal to 0.1 Mg/ha for each fence during the 7/5 and 9/14 storm events. For example, sediment yield at MW7 for the 7/5 event was only 0.004 Mg/ha, 99% below MW7's season average sediment yield at 2.45 Mg/ha, and 0.09 Mg/ha for the 9/14 event, 96% below MW7's season average.

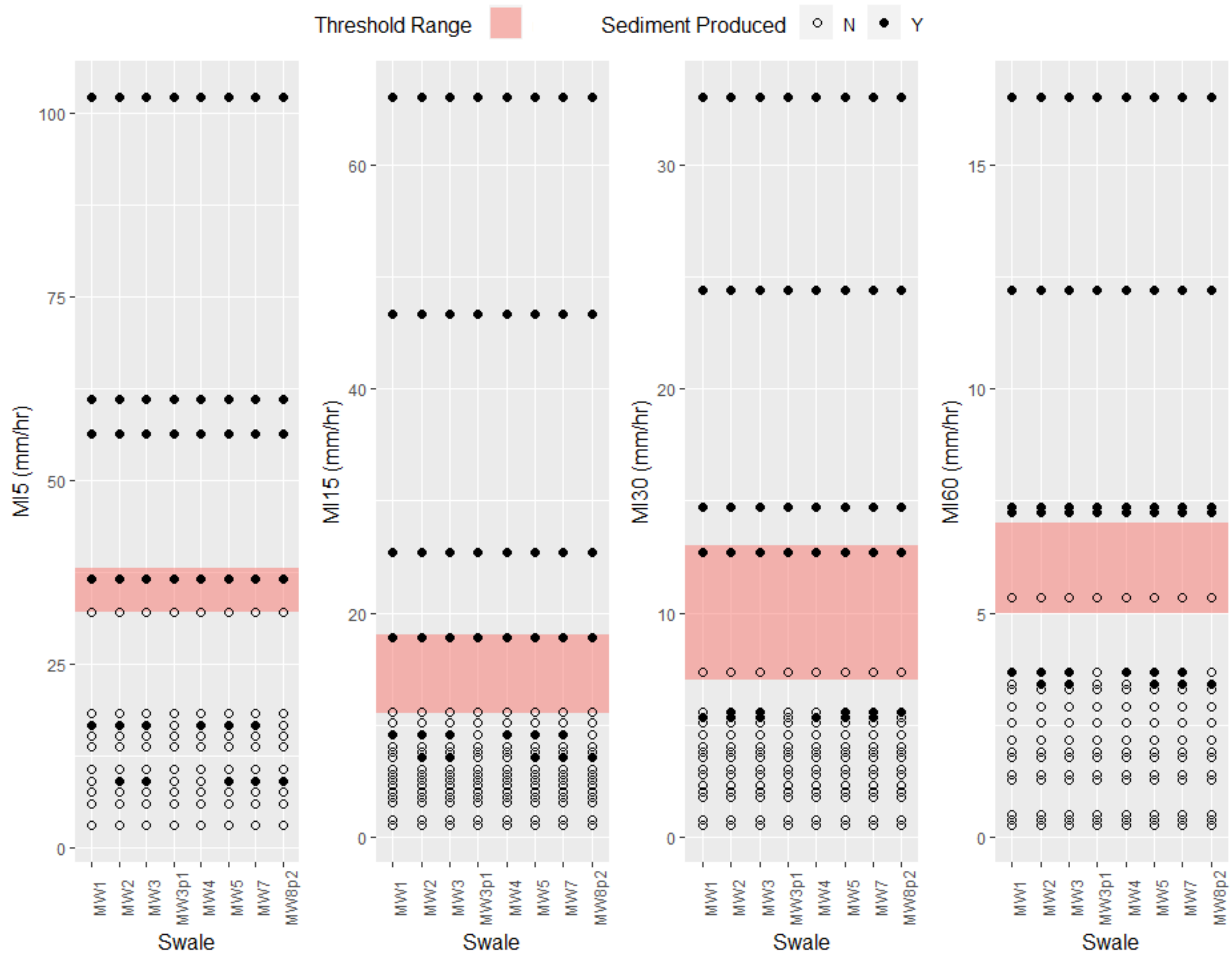

Figure 11. Sediment-producing and non-producing storms and precipitation threshold ranges by swale for 5-minute, 15-minute, 30-minute, and 60-minute maximum intensities. Open circles represent storms that did not produce sediment, and filled circles represent storms that did produce sediment.

Table 4. Precipitation thresholds for 5-minute, 15-minute, 30-minute, and 60-minute intensities to predict the producing events.

Maximum Intensity Interval	Threshold Range (mm/hr)
MI5	32-38
MI15	11-18
MI30	7-13
MI60	5-8

3.4 Controls on sediment yield

3.4.1. Univariate analysis

Across all swales, sediment yield was best correlated with maximum rainfall intensities (r = 0.84-0.90, Table 5), and MI15 had the most significant relationship ($p = 7.42 \times 10^{-49}$). Sediment yield correlated with total event precipitation, but with a lower correlation coefficient (r = 0.46). It did not have a significant correlation with precipitation duration, pre/post-mulching, treatment, slope mean, nor slope length. Correlations between sediment yield and surface cover could not be determined because surface cover measurements from on-the-ground surveys were not available at all swales for all events. However, vegetation cover derived from drone imagery was significantly correlated with sediment yield (p = 0.001; r = 0.28).

Table 5. Univariate correlations between predictors and sediment yield response.

Predictor Variable	**Correlation Coefficient**	**P-value**
MI5	0.85	9.87×10^{-38}*
MI15	0.90	7.42×10^{-49}*
MI30	0.87	9.15×10^{-41}*
MI60	0.84	1.68×10^{-35}*
Total precipitation (mm)	0.46	2.52×10^{-8}*
Duration precipitation (hr)	0.04	0.62
Pre/post-mulch	-0.03	0.74
Vegetation	0.28	0.001*
Treatment	-0.03	0.76
Slope mean	0.02	0.82
Slope length	-0.01	0.87

Because rainfall intensities were best correlated with sediment yield, these relationships are examined in more detail. For lower intensity events (MI5 < 25 mm/hr), sediment yields were low with no fence overtopping (Figure 12). Events with higher intensities had higher sediment yields and often overtopped with sediment, apart from MW2. Information on overtopping was not available for all fences from the 7/6 storm (MW2, MW3, MW4, MW5). For that same storm, MW1 overtopped with sediment, whereas MW7 did not. This event was the third highest intensity event; most fences overtopped with sediment for the two highest intensity storms, whereas fences tended to overtop with water only for the 4th highest intensity storms.

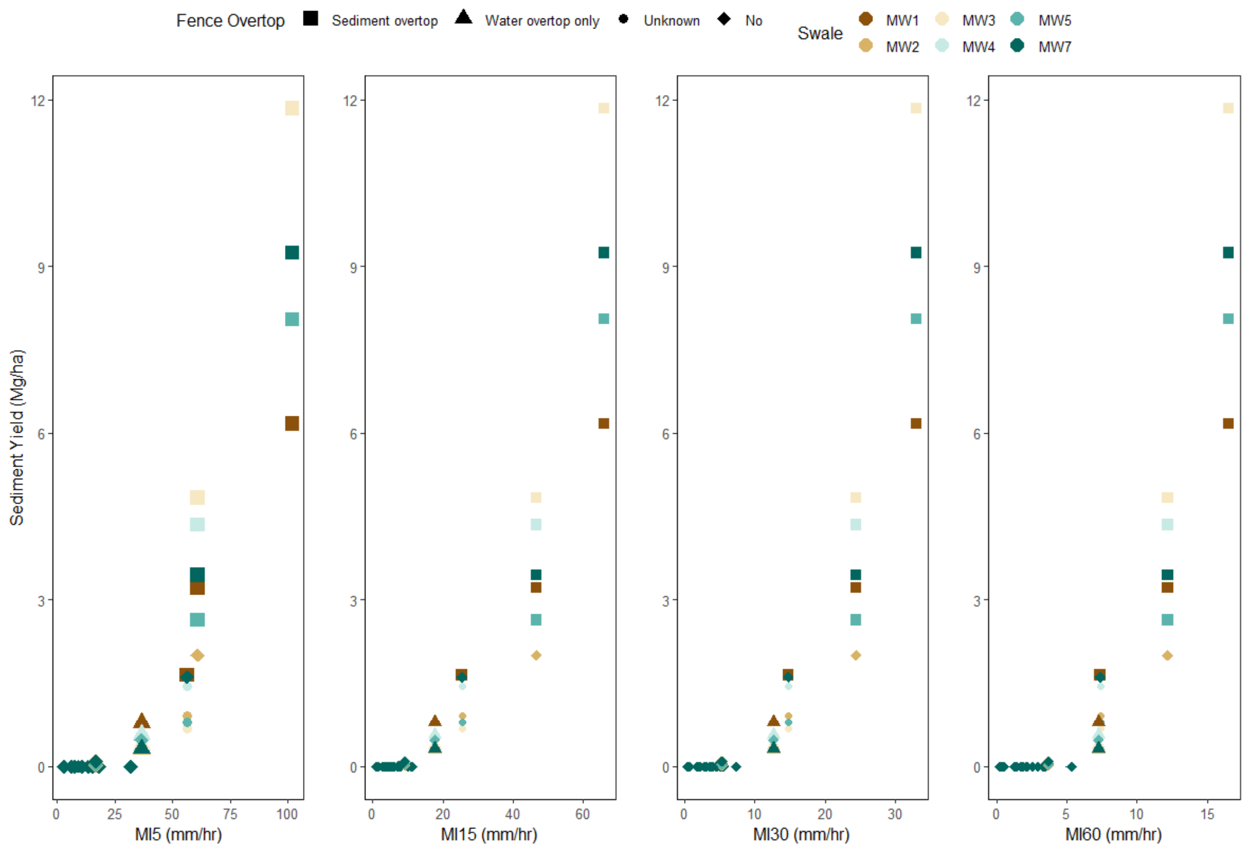

Figure 12. Relationship between sediment yield and rainfall maximum intensities at 5-minute, 15-minute, 30-minute, and 60-minute intervals. Colors indicate swale, and shapes indicate whether or not the fence overtopped.

However, differences between the treatments were not evident in plots of sediment yield vs. maximum intensities (Figure 13). At the highest intensity, across all rainfall MI durations (102 mm/hr for MI5, 66 mm/hr for MI15, 33 mm/hr for MI30, and 17 mm/hr for MI60), sediment yield was highest for swales treated with two layers of mulch. Generally, the fraction of events with sediment overtop do not differ between treatments.

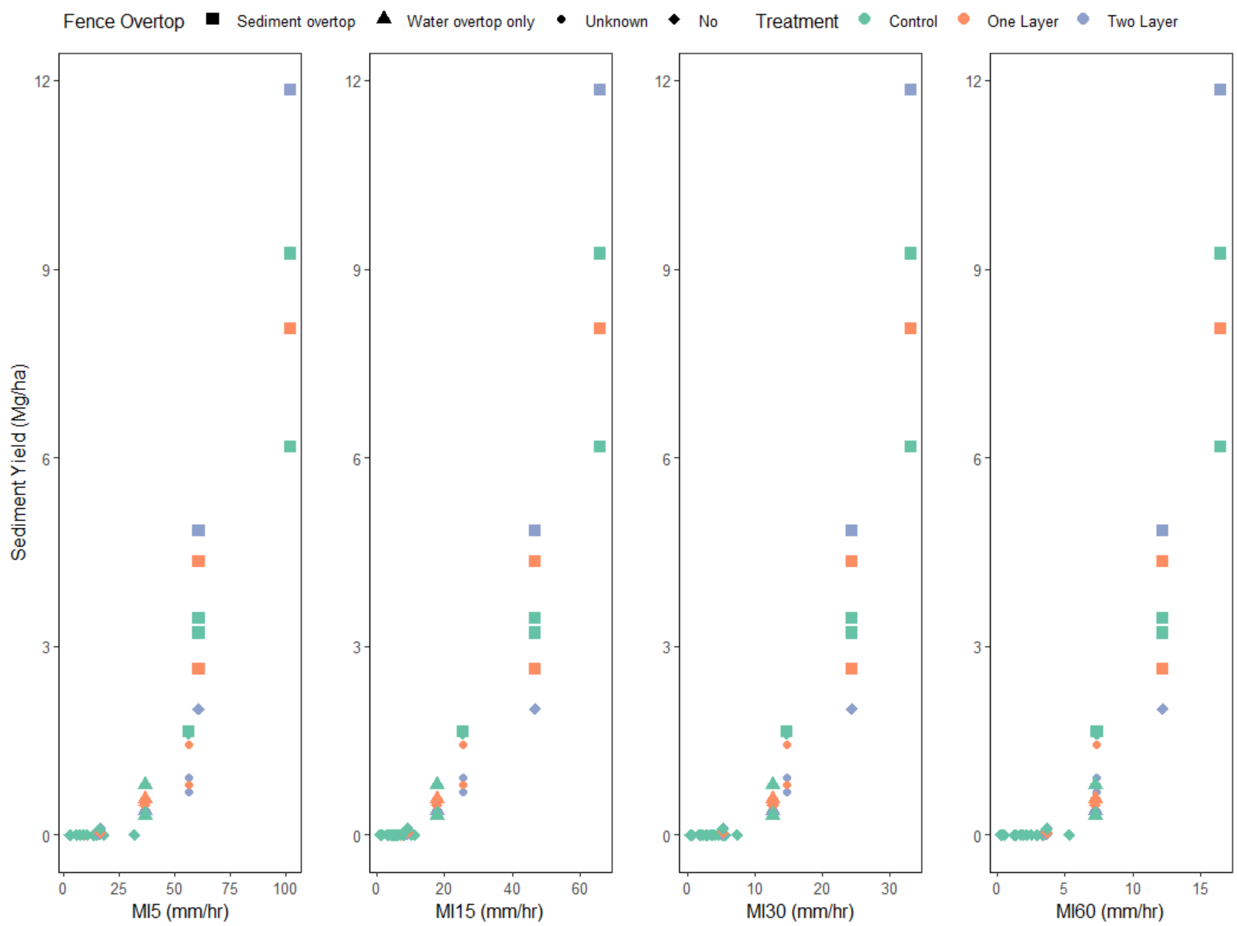

Figure 13. Relationship between sediment yield fence overtop and rainfall maximum intensities at 5-minute, 15-minute, 30-minute, and 60-minute intervals. Colors indicate type of mulch treatment.

3.4.2 Generalized linear mixed-effects models

Full binomial modeling showed that out of all potential predictors, total precipitation was the only significant driver of sediment generation occurrence (Table 6). The GLM with gamma distribution showed sediment yield differed among storms across the summer 2022 season, indicating that precipitation is a driving factor of sediment yield ($p < 0.0005$, Table 7). There was no significant difference in sediment yield across swales for the summer 2022 season and therefore no direct evidence of treatment effect. In addition, there was no indication of a change in sediment yield before and after mulching ($p = 0.49$). Both results are confirmed when looking at different treatment types (control, one layer mulch, two layers mulch), which show direct evidence of no treatment effect on sediment yield.

Treatment effect decreases even further when MI15 is considered. Neither contributing area, slope mean, nor slope length, affected yield in the GLM with gamma distribution. The amount of vegetation cover did significantly affect sediment generation, with increasing vegetation cover decreasing sediment yield by -3.3 Mg/ha ($p = 0.0009$). Vegetation and MI15 were the only significant drivers of sediment yield.

Table 6. Binomial mixed-effects model for sedimentation generation occurrence with all predictor variables without interactions.

Predictor	P-value
MI15	0.08
Total precipitation (mm)	0.003*
Pre/post-mulch	0.94
Vegetation	0.62
Treatment	0.78
Contributing area	0.95
Slope mean	0.71
Slope length	0.78

Table 7. Gamma distribution mixed-effects model of sediment yield when sediment yield is greater than zero.

Parameters	P-value
MI15	1.73×10^{-5}*
Total precipitation (mm)	0.42
Duration precipitation (hr)	0.18
Among storms	< 0.0005*
Pre/post-mulch	0.49
Vegetation	9.70×10^{-4}*
Treatment	0.71
Treatment + MI15	0.79
Contributing area	0.32
Slope mean	0.79
Slope length	0.92
Among swales	0.82
Among swales + MI15	0.96

Aside from precipitation metrics, the only other parameter showing significance in any model was vegetation cover. When considering interactions between vegetation cover and MI15 on sediment yield, the significance of vegetation diminished, indicating the explanatory power of precipitation intensity potentially overshadowing the significance of vegetation. Specifically, the full model with MI15 and vegetation variables, yielded a pseudo R^2 = 0. 4042, and the reduced model with only the MI15 variable resulted in a pseudo R^2 = 0.3999. The likelihood ratio test comparing the full and reduced model did not indicate a significant difference between models, implying that vegetation did not provide additional explanatory power ($p = 0.77$), and a simple, parsimonious model is the best model.

3.4.3 Random-forest model

Precipitation metrics, followed by vegetation cover, were ranked as the most important predictor variables in random forest models, with 92% predictive accuracy (out-of-box error 8.4%) (Table 8). Treatment type was the lowest ranked in predictor importance. Random forest models explain the observed data better than the mixed-effects models (R^2 = 0.74 versus 0.40, and 0.16, respectively). However, results from random forests models present an additional line of evidence supporting precipitation metrics as keys predictors of sediment yield.

Table 8. Random-forest modeling importance rankings as predictors of sediment yield.

Variable Importance Ranking	Predictor
1	Total precipitation (mm)
2	MI15 (mm/hr)
3	MI30 (mm/hr)
4	MI60 (mm/hr)
5	MI5 (mm/hr)
6	Precipitation duration (hr)
7	Vegetation
8	Pre/post-mulch
9	Slope length
10	Contributing area
11	Slope mean
12	Swale
13	Treatment

4 DISCUSSION

4.1 Effectiveness of mulching

This study examined the impact of mulching as a restoration technique to reduce erosion during a hillslope-scale experiment implemented two years post-fire. Results did not demonstrate a mulch effect on the occurrence or magnitude of sediment production at the hillslope-scale, in the initial months post-mulching. This finding is inconsistent with previous studies that have documented erosion reduction after mulch applications. A meta-analysis on post-fire erosion restoration techniques indicated that cover treatments, such as straw and wood mulch, significantly reduced post-fire erosion across multiple studies (Girona-García et al., 2021). Mulch can reduce sediment yield the first few years after fire (Schmeer et al. 2018; Díaz et al., 2022; Lucas-Borja et al., 2021; Wagenbrenner et al., 2006; Robichaud et al., 2020) by 60-90% (Lucas-Borja et al., 2021; Fernández et al., 2016; Prats et al., 2016). Many studies have attributed this short-term success to an immediate increase in ground cover and found that the added cover from mulch was effective when there was > 70% cover, applied within the "window of disturbance," when erosion is accelerated in post-fire conditions (Prosser & Williams, 1998; Prats et al., 2016; Díaz et al., 2022). Such studies observed treatment efficacy with initial mulch cover as low as 47%, up to 95% cover (Table 9). A possible reason why the Bennett Creek MW study did not identify a mulch impact may have been that cover was not as high as in prior studies. Initial mulch cover ranged from 39-44% for convergent swales treated with two layers of mulch, and 23-37% mulch cover for swales treated with one layer of mulch, resulting in approximately 36% mulch cover. The optimal cover levels for reducing erosion are typically between 60% and 80% mulch cover (Robichaud et al., 2000; Girona-García et al., 2021). Therefore, the mulch applications in our study area may not have been sufficiently high to be effective in reducing hillslope erosion.

Table 9. Mulch application rates and cover study comparison.

Study	Material and Treatment	Application Rate (Mg/ha)	Initial Mulch Cover
Bennett Creek MW 2022	Wood mulch – one layer	11 (± 5.5-22)	23-37%
	Wood mulch – two layers	15 (± 7.5-30)	39-44%
Lucas-Borja et al., 2021	Straw mulch	1.8	95%
Prats et al., 2016	Forest-residue mulch (micro-plots)	10.8	87%
	Forest-residue mulch (slope-scale plots)	13.6	77%
Díaz et al., 2022	Straw mulch	3.0	70-75%
	Wood mulch	20.0	47-50%
Girona-García et al., 2021 meta-analysis effective rates and cover	Straw mulch	> 2.0	60-80%
	Wood mulch	< 13.0	60-80%
Robichaud et al., 2013b	Wood mulch	13.0	60%
Jonas et al., 2019	Wood mulch – standard rate	13.0	70%
	Wood mulch – high rate	19.5	70%

Along with lower-than-optimal mulch cover, our study site experienced its highest intensity rainstorm only five days after mulching. After the high-intensity storm immediately post mulch, mulch cover dropped to 17%, almost half of the initial mulch cover. Lucas-Borja et al. (2021) conducted a similar plot-scale study after a wildfire in a Mediterranean pine forest, but with straw instead of wood mulch. Their plots started out with higher cover (95%), with 3 cm in thickness, and the mulch was in place for a month before the first high-intensity rainfall. These conditions led to greatly reduced sediment yield (Table 10). The high intensity storm at Bennett Creek MW study site within a few days of mulch application likely allowed little time for the mulch to settle before being carried away by storm overland flow.

Table 10. Treatment longevity and effectiveness study comparison to Lucas-Borja et al., 2021.

First Rainfall Post-mulch	**Bennett Creek MW**	**Lucas-Borja et al., 2021**
Time after fire	2 yr	2 mo
Time after application	4 days	1 mo
MI5	103 mm/hr	12 mm/hr
Soil loss mulched	> 20 Mg/ha	0.002 Mg/ha (± 0.0006 Mg/ha)
Soil loss unmulched	15 Mg/ha	0.006 Mg/ha (± 0.006 Mg/ha)

Several other factors may have limited the effectiveness of mulch applications in this study. First, the hillslopes are very narrow, and this made it difficult for the helicopter operators to place the mulch exactly within hillslope boundaries. The mulch was dropped in clumps and was not evenly spread across the full hillslope areas (Figure 14). Initial wood mulch distribution that promotes even coverage is favorable, however, helimulching can result in uneven distribution and clumping (Robichaud et al., 2013b; Robichaud et al., 2010).


Figure 14. Initial mulch cover distribution drone imagery (left) and on-the-ground (right).

Treatment performance can also be affected by the materials used in the treatment, size of wood strands, and mixture of fine and coarse strands. Our study used a mix of large and small wood mulch strands. Larger wood mulch particles primarily reduce sediment yield, while smaller particles absorb rainfall and reduce runoff. Long-strand wood mulch create small debris dams or mats by interlocking mulch fibers, which slow down overland flow, decreasing flow velocity, and increasing sediment retention on hillslopes (Faucette et al., 2007). Long-strand wood mulch is more resistant to displacement compared to short-strand, fine shred, wood mulch, which is more easily moved by overland flow (Robichaud et al., 2010; Robichaud et al., 2013b; Foltz & Copeland, 2009).

If mulch material is initially clumped before being spread evenly, it may not have the opportunity to form "mini-debris dams" or “mats” during intense rainfall events. Foltz & Copeland (2009) suggested that effective erosion control with wood mulch using a blend of mulch sizes, mixed with fewer pieces smaller than 25 mm and larger than 200 mm. Sampling wood mulch shreds before application was recommended to establish the desired and effective mass-cover relationship. Our study was short duration, but over time mulch can decay, impacting its long-term effectiveness, and the treatment material selected for erosion reduction is important for mulch treatment longevity.

In previous studies, hay/straw mulch treatments were found to be effective in reducing erosion at application rates ranging from approximately 2 Mg/ha to 3 Mg/ha (Table 9). Wood mulch treatments showed effectiveness at application rates ranging from approximately 11 Mg/ha to 20 Mg/ha. At Bennett Creek MW, the one-layer mulch treatment with an application rate of 11 Mg/ha was at the lower end of the range previously found effective in other studies. Whereas the two-layer mulch treatment at 15 Mg/ha aligns precisely with the average application rate observed in studies where wood mulch treatments were effective. However, the two-layer mulch treatment application rate falls within an estimated error range extending from approximately 8 Mg/ha to 30 Mg/ha. This error range includes application rates exceeding the recommended wood mulching rates from Girona-García et al., (2021), as effectiveness tends to vary rates surpass 18 Mg/ha, despite some studies observing effectiveness up to 20 Mg/ha (e.g., Díaz et al., 2022). This suggests that the 30 Mg/ha application rate may introduce significant uncertainty.

While it is possible for a 50% error in our application rates, with a rate of 15 Mg/ha there may have been potential for erosion reduction based on findings from other studies, yet it remains challenging to determine if the mulch application rates in our study directly impacted the effectiveness of the mulch. Our study's application rates were consistent with those observed in studies demonstrating treatment effects but fell short of initial mulch cover in comparison, especially for swales treated with one layer of mulch, where initial cover was less than half of that observed in a study with a similar application rate. Some studies emphasized initial cover may be more important than application rate or material (Foltz & Copeland, 2010), as application rate does not guarantee sufficient cover (Robichaud et al., 2013b).

Our study experienced several sources of variability during the treatment application, which possibly led to low initial mulch cover. Factors such as variations in fuel load, moisture content of the wood chips, and flight path may have led to fluctuations in the amount of mulch dropped on the hillslope. To ensure aerial mulching is effective, it is necessary to assess ground cover at the initiation of the treatment application process, to allow for adjustments to be made in factors such as like flight altitude and speed to achieve desired cover (Robichaud et al., 2013b). However, given these uncertainties, our findings highlight the possibility that the mulch application rate may have not been sufficient to achieve desired cover for treatment effect. Yet, determining the degree to which this factor influenced our results remains uncertain due to the presence of multiple sources of variability.

4.2 Drivers of sediment yield

4.2.1 Rainfall intensity thresholds

A primary finding of this study is that rainfall intensity was the most significant predictor and driving factor on both sediment generation occurrence and magnitude, with MI15 being the intensity with the highest correlation to sediment yield. This aligns with several other studies, where rainfall intensity was highly correlated to sediment generation, particularly from high-intensity, convective summer storms (Wilson et al., 2018; Robichaud et al., 2013c; Benavides-Solorio & MacDonald, 2005). We found that the MI60 threshold range for sediment generation was between 5-7.5 mm/hr, similar to Wilson et al. (2018), where MI60 exceeding 8 mm/hr would generate sediment. Wilson et al. (2018) also found that there was not a mulch effect on MI60 thresholds, which aligns with our findings across intensity intervals. In contrast, Prats et al. (2016) found that, with forest-residue mulching, there was a difference in thresholds for sediment generation for untreated versus treated plots – for the second-year post-fire, untreated plots MI30 threshold was between 15-18 mm/hr and mulched 20-25 mm/hr. In our study, thresholds were consistent across all, treated and untreated swales, with a lower MI30 threshold range between 7-13 mm/hr.

In the mixed model of sediment generation occurrence and the random forest model of sediment yield, total precipitation was a more significant predictor than MI15. This indicates that total precipitation is an important predictor, but its role is not as evident without statistically controlling for rainfall intensity.

4.2.2 Slope and length

In this study, there was no significant impact of slope length on sediment generation ($p = 0.78$; $p = 0.92$). In a nearby study area, Schmeer et al. (2018) also observed a weak to insignificant correlation between slope length on sediment yield. In contrast, other studies have found that increasing slope lengths lead to greater sediment yield (Robichaud et al., 2010; Miller et al., 2011) where slopes are shorter than 260 m, but when slopes get longer than this sediment yield starts to decline due to deposition along the flow path (Miller et al., 2011). The hillslopes in this study were mostly longer than 260 m (245 m to 300 m) and probably did not have enough variability in length to be able to detect a length effect on sediment yield.

Generally, steep slopes have greater sediment yield than gentle slopes. In a recent study (Díaz et al., 2022), conducted in a pre-fire Mediterranean pine forest, post-fire wood and hay mulching treatments were compared in plots with lower slope (30%) and high slope (50%), which falls within the mean slope percent range of the Bennett Creek MW swales (40-46%). They found that straw mulching with 70-75% initial cover significantly and almost completely reduced erosion on steeper slopes, though wood mulch had a lower reduction comparatively to straw mulch treatment, with 47-50% initial cover, suggesting that wood mulching might be less effective as protecting the hillslope, regardless of slope steepness, if initial cover is not high enough. On the other hand, in a study area near Bennett Creek, Schmeer et al. (2018) found a weak correlation between slope and sediment yield, which is consistent with the lack of a significant relationship between mean slope percent and sediment production we found. In our case, the limited range of slopes was probably not sufficient to detect a slope influence on sediment yield.

4.2.3. Ground cover

Although the mulch unfortunately washed away shortly after placement, vegetation cover did begin to return over summer 2022. This added vegetation appeared to have some erosion reduction benefit (Table 5). More vegetative cover provides a protective cover against rain splash, adding surface roughness, slowing down overland flow, allowing water to pool to infiltrate the soil (Morris & Moses, 1987; Larsen et al., 2009). This allows for greater soil infiltration capacity, for higher levels of organic matter, increasing porosity improving soil structure, to contribute to hillslope stabilization (Morris & Moses, 1987; Larsen et al., 2009; Ebel & Martin, 2017). Ecological and hydrologic recovery can take 10 years post-fire, with vegetation playing a crucial role (Jonas et al., 2019; Ebel & Martin, 2017). Reestablishment of vegetation takes 1-6 years in the study region (Wilson et al., 2018; Jonas et al., 2019). Most sediment movement occurs before vegetation is established in the first-year or two post-fire but decreases with vegetation recovery (Robichaud et al., 2013c). Mulch provides protection to the soil, which may create favorable conditions, such as increased soil moisture, to support vegetation recovery. However, studies on mulch impact on plant recovery yield mixed results. In some cases, mulching treatment material and application rate have had minimal impacts on vegetation recovery and do not hinder establishment (Jonas et al., 2019; Dodson & Peterson, 2010). Straw mulch and seeding treatments can introduce invasives species, which may outcompete regrowth of native vegetation. In contrast, wood mulch provides long-term soil protection, increases soil moisture, conditions supporting seedling germination, observed to increase lodgepole pine regeneration (Dodson & Peterson, 2010; Jonas et al., 2019). More long-term research is needed to assess the impact of vegetation on sediment yield, and if mulching treatments enhance vegetation growth.

Vegetation recovery takes time after a fire and does not offer benefits of immediate cover like mulching treatments. However, because of low cover and rapid displacement, in this study mulch had no significant effect on sediment yield, and was ranked the lowest of all predictors, while vegetation was a significant driver of sediment yield, independent of other parameters. However, the influence of vegetation cover on sediment yield may have been masked by the extreme effect of rainfall intensity, due to the limited changes in vegetation over a single season. We noticed that as vegetation increased over the season, sediment yield tended to decrease, but this effect cannot be quantified due to the limited number of sediment-producing events. Conducting a long-term study could reveal a more substantial impact of vegetation on erosion reduction.

4.3 Limitations and recommendations

We designed the study to identify the effects of mulch and minimize the variability in other factors between hillslopes. Therefore, the lengths, slopes, soil types, bedrock geology, and vegetation were similar for all hillslopes, and allowed for comparison between treatment application rates. Because the hillslopes were adjacent to one another, they also experienced similar rainfall. That is likely the reason that rainfall was the only variable besides vegetation cover to have a significant correlation with sediment yield. Terrain/topographic variables were all lower in importance compared to precipitation metrics. If the hillslopes had been distributed across a larger area, they would have had more variability in characteristics and rainfall, and therefore we might have found greater influence of other hillslope variables on sediment yield (e.g., Schmeer et al., 2018). The sample size of only six hillslopes also limited the utility of statistical modeling for identifying drivers of sediment yield.

In the random forest models, swale and treatment were ranked the least important predictors. This may have been influenced by missing sediment yield data, specifically when sediment overtopped fences resulting in undercatch, and in two cases where fences completely broke. Gaps in swale data limits our ability to compare different treatments. During the event most likely to demonstrate a mulch effect, right after mulch application, a large amount of mulch was displaced, decreasing our sample size by 30%. This loss of mulch led us to base a primary assumption on absence of any observed mulch effect over the season, particularly relying on observations from only four swales, only two treated with mulch. Constructing bigger and sturdier fences, and installing double fences, have the potential to increase storage capacity and reduce the risk of fences overtopping or breaking.

The large influence of precipitation on hillslope erosion highlights the importance of implementing post-fire restoration techniques that consider mulch material type, amount, and treatment longevity. Future studies may consider a blend of wood mulch with mixed strand lengths, providing > 70% initial mulch cover. Mulch material should be sampled prior to application, to ensure sufficient mass to cover ratio. At the beginning of aerial mulching, ground cover should be assessed from at the ground-level across treatment sites, adjusting flight speed or altitude needed for desired cover, and assessed throughout for even distribution across site. Lastly, mulch application should be implemented prior to summer months that experience peak annual rainfall.

5 CONCLUSIONS

The goal of this study was to assess the effectiveness of wood mulching treatments in reducing post-fire erosion at the hillslope-scale. We used an experimental approach with varying levels of mulch treatments between similar, adjacent hillslopes. Insufficient initial mulch cover and a high-intensity storm shortly after mulching led to low (< 44%) mulch cover in the experiment, hindering our ability to detect effects of mulching treatments at this spatial scale. Like other studies, precipitation intensity was the primary driver of erosion occurrence and also impacted the magnitude of sediment yield. Nearly all sediment generated during the season (98%) came during storms that exceeded the maximum 15-minute rainfall intensities of 11-18 mm/hr. With increasing occurrence and magnitude of wildfires, there is an urgent need for post-fire rehabilitation treatments that lessen impacts of erosion, flooding, debris flows, that threaten watershed and ecosystem functionality, and human life and safety. Future studies must consider application timing, mulch material, and adequate mulch cover for treatment effectiveness, that can be applied at larger, hillslope-scales.

REFERENCES

Barnard, D.M., Germino, M. J., Pilliod, D. S., Arkle, R. S., Applestein, C., Davidson, B. E., & Fisk, M. R. (2019). Cannot see the random forest for the decision trees: selecting predictive models for restoration ecology. Restoration Ecology, 27(5), 1053–1063. https://doi.org/10.1111/rec.12938

Benavides-Solorio, & MacDonald, L. (2005). Measurement and prediction of post-fire erosion at the hillslope scale, Colorado Front Range. International Journal of Wildland Fire, 14(4), 457–474. https://doi.org/10.1071/WF05042

Cameron Peak Fire Forest Service Burned Area Emergency Response (BAER) Executive Summary Arapaho Roosevelt National Forest December 15, 2020.

Díaz, Lucas-Borja, M. E., Gonzalez-Romero, J., Plaza-Alvarez, P. A., Navidi, M., Liu, Y.-F., Wu, G.-L., & Zema, D. A. (2022). Effects of post-fire mulching with straw and wood chips on soil hydrology in pine forests under Mediterranean conditions. Ecological Engineering, 182, 106720–. https://doi.org/10.1016/j.ecoleng.2022.106720

Dodson, & Peterson, D. W. (2010). Mulching effects on vegetation recovery following high severity wildfire in north-central Washington State, USA. Forest Ecology and Management, 260(10), 1816–1823. https://doi.org/10.1016/j.foreco.2010.08.026

Ebel, & Martin, D. A. (2017). Meta-analysis of field-saturated hydraulic conductivity recovery following wildland fire; applications for hydrologic model parameterization and resilience assessment. Hydrological Processes, 31(21), 3682–3696. https://doi.org/10.1002/hyp.11288

Faucette, Governo, J., Jordan, C. ., Lockaby, B. ., Carino, H. ., & Governo, R. (2007). Erosion control and storm water quality from straw with PAM, mulch, and compost blankets of varying particle sizes. Journal of Soil and Water Conservation, 62(6), 404–413.

Fernández, Vega, J. A., & Fontúrbel, T. (2016). Reducing post-fire soil erosion from the air; performance of heli-mulching in a mountainous area on the coast of NW Spain. Catena (Giessen), 147, 489–495. https://doi.org/10.1016/j.catena.2016.08.005

Foltz, & Copeland, N. S. (2009). Evaluating the efficacy of wood shreds for mitigating erosion. Journal of Environmental Management, 90(2), 779–785. https://doi.org/10.1016/j.jenvman.2008.01.006

Girona-García, Vieira, D. C. S., Silva, J., Fernández, C., Robichaud, P. R., & Keizer, J. J. (2021). Effectiveness of post-fire soil erosion mitigation treatments; a systematic review and meta-analysis. Earth-Science Reviews, 217, 103611–. https://doi.org/10.1016/j.earscirev.2021.103611

Groen, & Woods, S. W. (2008). Effectiveness of aerial seeding and straw mulch for reducing post-wildfire erosion, north-western Montana, USA. International Journal of Wildland Fire, 17(5), 559–571. https://doi.org/10.1071/WF07062

Jonas, Berryman, E., Wolk, B., Morgan, P., & Robichaud, P. R. (2019). Post-fire wood mulch for reducing erosion potential increases tree seedlings with few impacts on understory plants and soil nitrogen. Forest Ecology and Management, 453, 117567–. https://doi.org/10.1016/j.foreco.2019.117567

Juniper Systems, Logan, Utah. Geode GNSS Receiver: https://junipersys.com/products/geode

Cameron Peak Fire Forest Service Burned Area Emergency Response (BAER) Executive Summary Arapaho Roosevelt National Forest December 15, 2020.

Cameron Peak Fire Risk Assessment Summary Report Larimer County Office of Emergency Management, May 24, 2021. https://www.larimer.gov/sites/default/files/uploads/2021/cpf_risk_assessment_overview_5.24.2021.pdf

Larsen, MacDonald, L. H., Brown, E., Rough, D., Welsh, M. J., Pietraszek, J. H., Libohova, Z., Dios Benavides-Solorio, J., & Schaffrath, K. (2009). Causes of Post-Fire Runoff and Erosion: Water Repellency, Cover, or Soil Sealing? Soil Science Society of America Journal, 73(4), 1393–1407. https://doi.org/10.2136/sssaj2007.0432

Lucas-Borja, Parhizkar, M., & Zema, D. A. (2021). Short-Term Changes in Erosion Dynamics and Quality of Soils Affected by a Wildfire and Mulched with Straw in a Mediterranean Forest. Soil Systems, 5(3), 40–. https://doi.org/10.3390/soilsystems5030040

MacDonald, & Stednick, J. D. (2003). Forests and water: a state-of-the-art review for Colorado. Colorado Water Resources Research Institute, Colorado State University.

MacDonald, Lee H.; Larsen, Isaac J. 2009. Effects of forest fires and post-fire rehabilitation: A Colorado, USA case study. In: Cerd , Artemi; Robichaud, Peter R., eds. 2009. Fire effects on soils and restoration strategies. Land Reconstruction and Management Series, Volume 5. Science Publishers, Enfield, NH. pp. 423-452

Miller, MacDonald, L. H., Robichaud, P. R., & Elliot, W. J. (2011). Predicting post-fire hillslope erosion in forest lands of the western United States. International Journal of Wildland Fire, 20(8), 982–999. https://doi.org/10.1071/WF09142

Moody, & Martin, D. A. (2001). Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms, 26(10), 1049–1070. https://doi.org/10.1002/esp.253

Morris, & Moses, T. A. (1987). Forest fire and the natural soil erosion regime in the Colorado Front Range. Annals of the Association of American Geographers, 77(2), 245–254. https://doi.org/10.1111/j.1467-8306.1987.tb00156.x

National Geologic Map Database, USGS/AASG. Geologic map of the Rustic quadrangle, Larimer County, Colorado. Author(s): Shaver, K.C., Nesse, W.D., and Braddock, W.A.: https://ngmdb.usgs.gov/mapview/?center=-105.541,40.651&zoom=15, https://ngmdb.usgs.gov/Prodesc/proddesc_1143.htm

NOAA Atlas 14, Volume 8, Version 2, Point Precipitation Frequency Estimates: https://hdsc.nws.noaa.gov/pfds/pfds_map_cont.html?bkmrk=co

Onset Smart Sensor Rain Gauges and Data Loggers: https://www.onsetcomp.com/products/data-loggers/ua-003-64

Parks, & Abatzoglou, J. T. (2020). Warmer and Drier Fire Seasons Contribute to Increases in Area Burned at High Severity in Western US Forests From 1985 to 2017. Geophysical Research Letters, 47(22). https://doi.org/10.1029/2020GL089858

Peppin, Fulé, P. Z., Sieg, C. H., Beyers, J. L., & Hunter, M. E. (2010). Post-wildfire seeding in forests of the western United States: An evidence-based review. Forest Ecology and Management, 260(5), 573–586. https://doi.org/10.1016/j.foreco.2010.06.004

PESOLA Präzisionswaagen AG Digital Hanging Scale: https://www.pesola.com/E/digital-scales/digital-hanging-scales---dynamometers/digital-hanging-scale,-black-phs040.htm

Prats, Wagenbrenner, J. W., Martins, M. A. S., Malvar, M. C., & Keizer, J. J. (2016). Mid-term and scaling effects of forest residue mulching on post-fire runoff and soil erosion. The Science of the Total Environment, 573, 1242–1254. https://doi.org/10.1016/j.scitotenv.2016.04.064

PRISM Climate Data, (dataset) USDA Natural Resources Conservation Service (2023). PRISM Climate Group. NRCS. https://data.nal.usda.gov/dataset/prism. Accessed 2023-05-31

Prosser, & Williams, L. (1998). The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrological Processes, 12(2), 251–265. https://doi.org/10.1002/(SICI)1099-1085(199802)12:23.0.CO;2-4

Robichaud, Beyers, J. L., & Neary, D. G. (2000). Evaluating the effectiveness of postfire rehabilitation treatments. U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Research Station.

Robichaud, Ashmun, L. E., & Sims, B. D. (2010). Post-fire treatment effectiveness for hillslope stabilization. U.S. Dept of Agriculture, Forest Service, Rocky Mountain Research Station.

Robichaud, Wagenbrenner, J. W., Lewis, S. A., Ashmun, L. E., Brown, R. E., & Wohlgemuth, P. M. (2013a). Post-fire mulching for runoff and erosion mitigation; Part II, Effectiveness in reducing runoff and sediment yields from small catchments. Catena (Giessen), 105, 93–111. https://doi.org/10.1016/j.catena.2012.11.016

Robichaud. (2013b). *Production and aerial application of wood shreds as a post-fire hillslope erosion mitigation treatment*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Robichaud, Lewis, S. A., Wagenbrenner, J. W., Ashmun, L. E., & Brown, R. E. (2013c). Post-fire mulching for runoff and erosion mitigation: Part I: Effectiveness at reducing hillslope erosion rates. *Catena (Giessen)*, *105*, 75–92. https://doi.org/10.1016/j.catena.2012.11.015

Robichaud, Lewis, S. A., Wagenbrenner, J. W., Brown, R. E., & Pierson, F. B. (2020). Quantifying long-term post-fire sediment delivery and erosion mitigation effectiveness. Earth Surface Processes and Landforms, 45(3), 771–782. https://doi.org/10.1002/esp.4755

Schmeer, S.R. (2014). Post-fire erosion response and recovery, High Park Fire, Colorado. Master's Thesis, Colorado State University.

Schmeer, Kampf, S. K., MacDonald, L. H., Hewitt, J., & Wilson, C. (2018). Empirical models of annual post-fire erosion on mulched and unmulched hillslopes. *Catena (Giessen), 163,* 276–287. https://doi.org/10.1016/j.catena.2017.12.029

Soil Survey Staff, Natural Resources Conservation Service (NRCS), United States Department of Agriculture (USDA). Official Soil Series Descriptions. Available online. Accessed 5/31/2023.

U.S. Geological Survey, 2019, USGS 3D Elevation Program Digital Elevation Model, accessed May 16, 2023: https://apps.nationalmap.gov/3depdem

Wagenbrenner, MacDonald, L. H., & Rough, D. (2006). Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. Hydrological Processes, 20(14), 2989–3006. https://doi.org/10.1002/hyp.6146

Westerling. (2016). Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. *Philosophical Transactions of the Royal Society B: Biological Sciences, 371*(1696), 20150178–. https://doi.org/10.1098/rstb.2015.0178

Wilson, Kampf, S. K., Wagenbrenner, J. W., & MacDonald, L. H. (2018). Rainfall thresholds for post-fire runoff and sediment delivery from plot to watershed scales. Forest Ecology and Management, 430, 346–356. https://doi.org/10.1016/j.foreco.2018.08.025