HYDROLOGIC RESPONSE OF MEADOW RESTORATION FOLLOWING

THE REMOVAL OF ENCROCHED CONIFERS

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ABSTRACT

Hydrologic Response of Meadow Restoration Following the Removal of Encroached Conifers

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Meadows are important within forest ecosystems because they provide diverse species habitats, facilitate water cycling, help with sediment capture, aid in carbon sequestration, and create natural fire breaks in forested regions. However, fire suppression, poor grazing practices, and climate change have accelerated the encroachment of conifers into historical meadow habitat. This has led to an extensive loss of meadow habitat within the Sierra Nevada and Cascade Mountain Ranges. Therefore, the purpose of this study is to quantify changes in percent soil moisture and groundwater levels following the removal of encroached lodgepole pine (*Pinus contorta)* in a historic meadow habitat near Lake Almanor, California.

A before-after control-intervention (BACI) study design was used, with Marian Meadow (MM) as the control and Rock Creek Meadow (RCM) as the restored meadow. Soil moisture and groundwater level data was collected one year before (water year 2019), and three years after (water years 2020-2023) the removal of lodgepole pine from RCM in the fall of 2020. This data was then analyzed using multiple linear regression and estimated marginal means (EMMs) models.

Percent soil moisture increased each year after restoration, with significant increases from pre-restoration values occurring in year 2 and year 3 post-restoration. The overall mean soil moisture content increased from 30.69% (pre-restoration) to 40.42% by the 3rd year post-restoration. Groundwater has had a much more mixed response to restoration, with the 1st year restoration seeing a significant decrease in groundwater availability. Years 2 and 3 showed gradual recovery of groundwater levels, although on average they were still less than pre-restoration groundwater levels. This can likely be contributed to moderate drought occurring in the 2020 and 2021 water years.

Sources of variability include the 2021 Dixie Fire which burned through both meadows at different severity levels, gaps in the data due to issues with the data loggers, differences in snowmelt timing, and differences in soil attributes. Collectively, however, all these factors converge toward a wetter meadow habitat. Hopefully, the results of this research will help promote a better understanding of how meadow restoration can improve California forestland management.

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1.0 Introduction

Montane meadows are associated with shallow water tables, high soil moisture levels, and high soil carbon content, which support the growth of herbaceous meadow vegetation (Pope et al., 2015; Weixelman et al., 2011). When meadows are associated with a stream they can stabilize channel banks, dissipate energy during high flow events, increase water quality by filtering sediment, and improve groundwater recharge (Pope et al., 2015). However, degradation of meadows can result in deeper water tables, changes in local plant communities, and cause them to become carbon sources (Pope et al., 2015). Conifer encroachment on historical meadows in the Sierra Nevada and Cascade Mountain ranges has caused the degradation of many meadows in the region. Lodgepole pine (*Pinus contorta*) in particular tends to invade areas with high soil moisture content, such as meadows (Vankat, 1982). Prior to the 20th century conifer invasion of montane meadows would have been managed by wildfires and/or grazing (Hamilton et al., 2019). However, increased fire suppression, poor grazing practices, and increased impacts from climate change starting in the 20th century, has allowed the expansion of these conifers into historical montane meadows.

Since a montane meadow's health is highly connected to its hydrology, this study aims to quantify the changes to meadow hydrologic conditions once conifers have been cleared from the meadow and identify if the environmental benefit outweighs the impacts of the tree removal. Conifer removal in this case is the mechanism being used to restore a montane meadow to historical/healthy conditions. The hypothesis is that the water availability of a montane meadow will improve in the long-term following conifer removal. This research was conducted on Rock Creek Meadow (RCM), which is a 75 ha (185.3 ac) meadow in the intersection of the Sierra Nevada and Cascade Mountain ranges near Chester, CA.

Objectives for this research are:

1. Quantify changes to the meadow hydrology following restoration.

2. Analyze if there were any significant changes to the meadow hydrology between pre-restoration and post-restoration values.
3. Determine if the removal of encroached conifer trees improved the amount of water available in the meadow's subsurface.

To achieve these objectives, a Before-After Control-Intervention (BACI) study design was used. One year of pre-restoration and three years of post-restoration data were collected at RCM and the control meadow, also known as Marian Meadow (MM). The hydrologic conditions measured at RCM and MM were soil moisture (m^3/m^3) and depth to groundwater (m). Data collection occurred from July of 2018 until September of 2023, with most of the encroached conifers being mechanically removed from RCM in the fall of 2020. Soil moisture and depth to groundwater data were collected by data loggers at thirty-minute intervals.

Seasonal changes in soil moisture and depth to groundwater were analyzed using time series graphs and monthly averages by year. Multiple linear regression and estimated marginal means (EMMs) analyses were used to examine the significance of changes between pre-restoration and post-restoration soil moisture and depth to groundwater values. Additionally, these models were used to compare differences in soil moisture and depth to groundwater values between RCM and MM, as MM illustrates what restored meadow conditions in this area should be.

In the fall of 2021, the Dixie Fire burned through both RCM and MM. As this was not anticipated as a part of this study, a simple multiple linear regression analysis was performed to compare RCM and MM hydrologic conditions pre- and post-fire.

2.0 Background

This chapter provides a review of the literature surrounding montane meadow vegetation, climate, and hydrology within the Sierra Nevada and Southern Cascade Mountain ranges. It additionally reviews knowledge on how forest management, stream channel restoration, and meadow restoration affect montane meadows.

2.1 Montane Meadow Characteristics

Montane meadows provide many ecosystem services including enhanced biodiversity, carbon sequestration, flood attenuation, fire breaks, and habitat for wildlife (Freitas et al., 2014). Vegetation, soil characteristics, and hydrology are used to define what is or is not a meadow. According to Weixelman et. al. (2011), montane meadows in California's Sierra Nevada and Southern Cascades are dominated by herbaceous plant species that rely on access to surface water and/or shallow groundwater and cannot be dominated by woody vegetation.

2.1.1 Vegetation

While meadows make up less than 1% of the Sierra Nevada Mountain range, they support highly diverse plant and animal communities including many rare and endangered species (McIlroy & Allen-Diaz, 2012; Viers et al., 2013). Various studies across meadows in the Sierra Nevada Mountain range have found that native perennial plant species dominate the meadows, while native conifers dominate the surrounding vegetation (McIlroy & Allen-Diaz, 2012; Ratliff, 1985; Weixelman et al., 1996). One study found that water table level was an indicator of plant community types and community classifications (McIlroy & Allen-Diaz, 2012). This conclusion illustrates how connected meadow vegetation is with the water table and soil moisture.

Besides herbaceous meadow vegetation, conifers from the surrounding forested area may also grow within meadows. These trees can grow and be dense within a meadow, but they cannot dominate the area (Weixelman et al., 2011). *Pinus contorta* (lodgepole pine), is a conifer that has higher tree densities where there is more soil moisture, such as meadow edges (Vankat, 1982). Due to the higher soil moisture content of meadows, lodgepole pines will often begin to invade the adjacent meadow. According to Vankat (1982), lodgepole pine encroachment on meadows within the Sierra Nevada Mountain range has been occurring since the early 20th century. Historically, this invasion was regulated by natural high intensity fires or selective grazing by livestock (Hamilton et al., 2019).

2.1.2 Soil Characteristics

Montane meadows are estimated to contain 12-31% of the soil organic carbon stocks in the Sierra Nevada mountains due to them being mineral soil wetlands (Reed et al., 2020). These meadows tend to form in alluvial plains and low-gradient valleys, where sediment from the surrounding uplands may be deposited above the soil in-situ (Reed et al., 2020; Weixelman et al., 2011; Blackburn et al., 2021). The soils are typically finely textured and rich in soil organic matter (Blackburn et al., 2021; Weixelman et al., 2011). The high soil organic content of these meadows results in high densities of carbon (C) and nitrogen (N). However, because much of the C and N is stored in anaerobic conditions, drying of meadow soil can result in the breakdown and release of these compounds into the atmosphere (Norton et al., 2011; Reed et al., 2020).

2.1.3 Hydrology

Montane meadows rely on water inputs from precipitation, snowmelt, subsurface flow, and overland flow (Lord et al., 2011). In the Cascade Mountain Range, snowmelt is the primary source for surface water and groundwater recharge. The amount of water available from snowmelt can be measured based on the snow's depth and density. The timing of snowmelt is associated with the changes in air temperature during the spring and early summer months. The size of the snowpack and timing of the snowmelt greatly affects the groundwater table elevation. Subsurface or groundwater flow is controlled by stratigraphy, hydraulic conductivity of subsurface materials, what the water source is, how the water source support the meadow complex, and how the groundwater interacts with stream channels (Lord et al., 2011). Since groundwater interacts with stream channels, the water levels in a stream are partially dependent on whether the groundwater flows to or away from the stream channel (Lord et al., 2011).

Montane meadows also output water due to overland runoff, groundwater seepage, and evapotranspiration (ET) (Viers et al., 2013). ET plays a major role in the water balance of a hydrologic system. The rate of ET is governed by many factors including relative humidity, air temperature, solar radiation, wind speed, local vegetation, soil moisture, rooting depth, and the distribution of near surface groundwater or water bodies (Allen et al., 1998). Potential evapotranspiration (PET) is a way to estimate meadow ET by assuming that there is uniform vegetation cover and soil moisture conditions (Allen et al., 1998). Therefore, actual evapotranspiration (AET) can be determined if local vegetation cover and soil moisture levels are taken. In general, the water balance for montane meadows can be determined by subtracting the outputs from the inputs.

Meadows aid in hydrologic processes by reducing peak flows, increasing groundwater infiltration, reducing sediment transport into water bodies, and protecting stream banks (Weixelman et al., 2011). This is because meadows provide a large floodplain for runoff to pond and percolate into the soil. Many tributaries in the Sierra Nevada Mountain range pass through one or more meadows, which helps improve water quality and control sediment discharge (Blackburn et al., 2021).

2.2 Forest Management Effects on Montane Meadows

In the Sierra Nevada and Southern Cascade Mountain ranges, decades of wildfire suppression and poor grazing practices have led to the development of dense conifer forest stands. In some areas, the density of pine trees increased 3 to 5 times historic stocking levels (USDA Forest Service, 2008).

2.2.1 Wildfire

Prior to the 1900s, mixed conifer forest stands within the Sierra Nevada experienced low intensity fires between 5-18 years (Miller & Urban, 1999; Kilgore & Taylor, 1979). Lodgepole pine forests within the Sierra Nevada were estimated to experience low intensity fires between 25-150 years (*Carson Range…*, 2008). Ignition sources during that time included lightning, burning by native Americans (prior to 1860) and burning by sheepherders (late 1800s). During the 20th century, changes in legislation resulted in intense fire suppression which altered fuel loading, vegetation patterns/composition, and the connectivity of landscapes. The removal of intentional periodic burning by humans resulted in an increase in woody plant density within various forest types (Vankat & Major, 1978). Higher stand densities led to higher tree mortality due to increased competition for nutrients and spread of pathogens and insects (USDA Forest Service, 2008). This mortality increased the accumulation of dead fuels and ladder fuels for more intense fires (USDA Forest Service, 2008). Beyond this, timber harvests that prioritize harvesting the oldest, largest, and most fire-resistant trees have led to the increased density of smaller trees which are more susceptible to wildfires (Sterner et al., 2022).

2.2.3 Grazing

Livestock herders brought cattle into the Sierra Nevada Mountain range around the mid-1800s (Vankat & Major, 1978; Freitas et al., 2014). Grazing practices rapidly increased, with domestic sheep eventually becoming more common than cattle (Vankat & Major, 1978). During a trip to Mount Whitney, Magee (1885) recounted that "each of these meadows is yearly cropped several times by various flocks of sheep, and the result is that, even where there was a genuine mountain meadow, there are now only shreds and patches. The sod and the verdure are gone - eaten and trodden out; the gravel is now in the ascendant" (as quoted in Vankat & Major, 1978). These grazing habits served to reduce the herbaceous cover and accelerate soil erosion of meadows. When meadows experience excessive grazing, there is a reduction in plant vigor, reproduction, and competitiveness, which can then trigger shifts in the plant community types (Freitas et al., 2014). Shifts in the plant community types can lead to reduced root complexity, soil stability, and resistance to soil erosion (Freitas et al., 2014; Ratliff, 1985). Since meadows are strongly associated with riparian areas, these changes in plant communities and soil stability can negatively impact the hydrologic functions of the area (Freitas et al., 2014). Improved grazing standards/practices, such as restricted herbaceous biomass consumption, restricted browse of riparian willow species, restricted access to streambanks, improved livestock distribution, and annually variable timing of grazing have been shown to greatly reduce the impact of livestock grazing on meadows and riparian areas (Freitas et al., 2014; Clary & Leininger, 2000).

2.3 Beaver Meadows

The North American beaver (*Castor canadensis*) is an ecosystem engineer that can drastically impact a watershed's dynamics by creating dams. The creation of beaver meadows depends on the river segment, water reliability, and food availability (Burchsted et al., 2010). Previous research has found that low stream gradients tend to increase the success and longevity of a beaver impoundment (Burchsted et al., 2010). Over time these beaver dams collect and trap nutrient rich sediment that can be released into the floodplain when a dam is abandoned and breached (Wright et al., 2002; Burchsted et al., 2010). Since beaver dams tend to collect finer sediment particles, the channel bed following a dam tends to become coarser with larger cobbles scattered across the bed (Burchsted et al., 2010).

Additionally, beavers alter the riparian zone and forested area around them by removing woody vegetation via herbivory, felling, and flooding (Wright et al., 2002). With patches of woody vegetation removed in the riparian zone, herbaceous plant species are able to regenerate which can lead to an increase in herbaceous plant species richness up to 25% (Wright et al., 2002). The combination of these behaviors can result in the formation of meadows that remain for more than 50 years (Wright et al., 2002). Beaver meadows tend to persist for decades to centuries due to re-colonization by beaver, which allows for the layering of beaver-created meadow patches (Westbrook, 2005).

2.4 Montane Meadow Restoration

Montane meadow restoration can occur through the removal of encroached trees from a historical meadow, or through restoration on an associated stream. Encroached trees are often removed using mechanical thinning, herbicides or prescribed fire (Halpern et al., 2012; Halpern & Antos, 2021). Tree removal using mechanical thinning has shown positive responses in meadow understory species cover and richness, although, this is highly reliant on whether there is an adequate seed bank in the soil (Halpern et al., 2012). Despite the historical use of prescribed fire in the Cascade Mountain range, there was little difference in the response of meadow vegetation from prescribed burning versus mechanical thinning (Halpern et al., 2012). Restoration of an associated stream is usually done to fix an incised stream channel and reconnect the floodplain within the montane meadow. Restoration can be accomplished using beaver dam analogs (BDA) or pond-and-plug restoration, which has become increasingly more popular (Hammersmark et al., 2008)

2.4.1 Pond and Plug Restoration

"Pond-and plug" restoration was first introduced in the early 1990s by Dave Rosgen and subsequently implemented in the Plumas National Forest in the Sierra Nevada Mountains in 1995 (Plumas National Forest, 2010). The goal of this restoration technique is to promote flooding of an incised stream channel to re-water and restore an associated meadow (Tennant et al., 2021). This is a low-tec restoration approach that involves the use of on-site material to create a series of earthen plugs (Plumas National Forest, 2010). If done successfully, these plugs will result in the reconnection of the degraded stream channel with its floodplain (Plumas National Forest, 2010). When a more intact floodplain is in place, some of the winter and spring runoff can be stored in adjacent meadows (Plumas National Forest, 2010).

A study of the effects of pond-and-plug structures on fish habitat in Red Clover Creek in Plumas National Forest found that the ponds offer moderate habitat quality for fish communities (Tennant et al., 2021). However, the dissolved oxygen and water temperature levels of these ponds during certain times of the year reached or exceeded levels associated with stress-related responses in fish (Tennant et al., 2021). Notably, the transformation of a creek from a lotic system to a primarily lentic system can affect and displace aquatic taxa associated with lotic systems (Tennant et al., 2021). These potential impacts on native species must be analyzed and considered before implementing pond-and-plug as a meadow restoration technique.

2.4.2 Conifer Re-Invasion

Once encroached conifers are removed from a restored meadow, re-establishment of conifers must be monitored. The longevity of meadow restoration via conifer removal often depends on the rate at which woody plants reestablish (Halpern & Antos, 2021). A study examining the rate of conifer re-invasion 15 years after restoration in Oregon's Cascade Mountain Range, found that there was not a consistent increase in the frequency of trees on either plots mechanically thinned or burned (Halpern & Antos, 2021). While the rate of re-invasion varied by plot, the rate on average was 9-10 trees ha^{-1} $year^{-1}$ with a range from -5 trees ha^{-1} $year^{-1}$ to 26 trees ha^{-1} $year^{-1}$ (Halpern & Antos, 2021). Notably, plots fully surrounded by forest saw higher rates of re-invasion than plots with forest absent or distant from the edge (Halpern & Antos, 2021). Due to the inevitability of conifer re-invasion, management of open meadow space is necessary (Kremer et al., 2014). When *Pinus* is the dominant invader, earlier and more frequent cutting may be necessary due to the faster-growing nature of these species (Kremer et al., 2014).

3.0 Methodology

This chapter describes the study area geography, climate, soils, surface hydrology, vegetation, meadow type, and wildfire impacts. This section also details the study design, instrument installation, field site visits, and data analysis of Marian Meadow (MM) and Rock Creek Meadow (RCM).

3.1 Study Area Description

3.1.1 Geography

The focus of this study was Rock Creek Meadow (RCM) which is located approximately 13 km (8 mi) to the east of Chester, California, USA (Figure 3-1). The site sits at an elevation of 1,524 m (5,000 ft) and covers approximately 75 ha (185.3 ac). Marian Meadow (MM) was used as a control for RCM and is located approximately 13 km (8 mi) to the west of Chester, California, USA (Figure 3-1). MM sits at an elevation of 1,370 m (4,495 ft) and covers approximately 18.2 ha (45 ac). Both meadows are located in the transitional zone between the Southern Cascade and Northern Sierra Nevada Mountain ranges on private forestland owned by Collins Pine Company.

Figure 3-1. Map of the linear distance (21 km) between RCM and MM, which are located just outside of Chester, CA.

3.1.2 Climate

RCM and MM are in a Mediterranean climate zone, which is characterized by cool wet winters and warm dry summers. Table 3-1 presents the monthly snowfall depth normals from data collected by the National Oceanic and Atmospheric Administration's (NOAA) climate station (station ID USC00041700) in Chester, CA. The normals indicate that Chester, CA retains at least 1 meter of snowfall from November to April. Similarly, Chester, CA receives the majority of its precipitation between November and May. Table 3-2 shows the total incremental precipitation by water year based on daily data collected by US Forest Service (USFS) climate station (station ID CHS) in Chester, CA. Notably, the USFS and NOAA climate stations are located at an elevation of 1,381 m while RCM is located at an elevation of 1,524 m. RCM's higher elevation indicates that the meadow may have higher snowfall and precipitation than recorded by the Chester station during the winter months.

Table 3-1. Monthly average (avg.) snowfall depth (cm) based on data collected at the NOAA climate station (station ID USC00041700) in Chester, CA between 1991-2020.

Month	Avg. Snowfall Depth (cm)
Jan	881.38
Feb	708.66
Mar	497.84
Apr	124.46
May	7.62
Jun	2.54
Jul	0.00
Aug	0.00
Sep	2.54
Oct	17.78
Nov	322.58
Dec	635.00

Table 3-2. Total precipitation (mm), minimum temperature (°C), maximum temperature (°C), and average temperature (°C) by water year (WY). Data was collected from the USFS's *Chester* climate station (40.283°, -121.233°) in Chester, CA (DWR, n.d.).

WY	Total Precipitation (mm)	Min Temp. (°C)	Max Temp. (°C)	Average Temp. (°C)
2019	1,019.05	-17.78	36.11	8.00
2020	210.31	-15.00	37.22	8.96
2021	350.52	-17.78	38.33	9.17
2022	500.89	-17.78	38.89	9.20
2023	1,035.30	-17.78	37.78	7.49

3.1.3 Geology and Soils

RCM and MM are located southeast of Mount Lassen, which is an active volcano in the Cascade Mountain Range (Figure 3-2). The primary soil parent material for RCM and MM are mapped to have volcanic flow rocks from minor pyroclastic deposits (Generalized Rock Types, 2010). The majority of RCM consists of the volcanic rock deposits from the quaternary period (Figure 3-2), which is the geologic timeframe from about 2.6 million years ago to the present. RCM also consists of the volcanic rock deposits from the tertiary period, which is also the primary parent material for MM (Figure 3-2). The tertiary period began about 66 million years and ended when the quaternary period began.

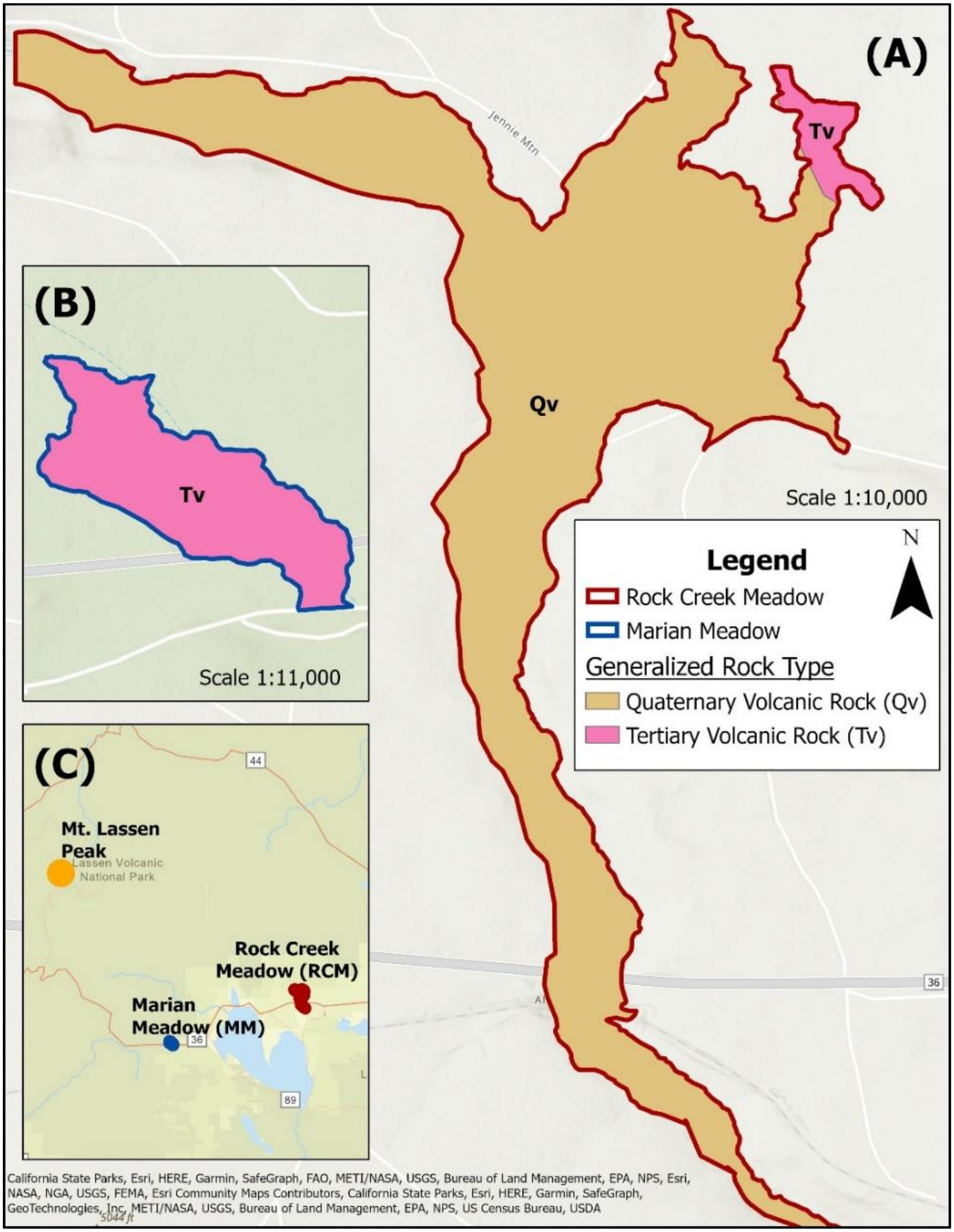


Figure 3-2. Distribution of generalized rock types for RCM (A) and MM (B), located southeast of Mt. Lassen (C). Generalized rock types are based on data from the California Department of Conservation (*Generalized Rock Types*, 2010).

Digital soil mapping by the USDA's Natural Resources Conservation Service (NRCS) shows that most of the entrance to RCM and the area to the west of the main access road is a part of the Mountmed loam, 0 to 2 percent slopes soil series (Appendix A-1). This soil series is characterized by a thick clayey C horizon with gleying due to frequent flooding and the soil having a very poorly drained drainage classification (Table 3-3) (NRCS, n.d.). The most limiting layer of this soil series typically has a hydraulic conductivity (Ksat) rating of moderately low to moderately high (0.06 to 0.20 in/hr) (NRCS, n.d.). The area of RCM to the east of the main access road primarily consists of the Inville very gravelly sandy loam, 0 to 5 percent slopes (Appendix A-1). This soil series is characterized by very gravelly loamy soil horizons that are well drained with a Ksat rating of moderately low to moderately high (0.06 to 0.20 in/hr) in the most limiting layer (NRCS, n.d.). The sandier soil texture and well drained characteristics of the eastern portion of RCM may provide some explanation for why it tends to be dryer than the clayey and very poorly drained western portion.

Table 3-3. Distribution of soil types across RCM based on digital soil maps from NRCS's Web Soil Survey.

Soil Series Name	Percent of Meadow	Typical Soil Profile
Inville very gravelly sandy loam, 0 to 5 percent slopes	23.4%	*A - 0 to 10 inches:* very gravelly sandy loam *Bt - 10 to 21 inches:* very cobbly loam *2Bt - 21 to 30 inches:* extremely gravelly loam *3C - 30 to 60 inches:* very gravelly loam
Mountmed loam, 0 to 2 percent slopes	57.0%	*A - 0 to 6 inches:* loam *Cg - 6 to 35 inches:* clay *2C - 35 to 60 inches:* stratified sand to very gravelly sandy clay loam
Redriver-Weste complex, 2 to 9 percent slopes	8.4%	*A - 0 to 3 inches:* very gravelly sandy loam *AB - 3 to 19 inches:* extremely cobbly sandy loam *Bw - 19 to 36 inches:* extremely gravelly sandy loam *R - 36 to 46 inches:* unweathered bedrock
Redriver-Woodwest-Wafla Complex, 0 to 9 percent slopes	9.4%	*A - 0 to 6 inches:* very gravelly sandy loam *AB - 6 to 16 inches:* extremely cobbly sandy loam *Bw - 16 to 31 inches:* extremely gravelly sandy loam *R - 31 to 41 inches:* unweathered bedrock
Swainow- Almanor complex, 15 to 30 percent slopes	1.8%	*A - 0 to 3 inches:* extremely stony sandy loam *AB - 3 to 18 inches:* extremely stony sandy loam *2Bt - 18 to 35 inches:* very gravelly loam *2BC - 35 to 44 inches:* extremely cobbly loam *2Cr - 44 to 60 inches:* weathered bedrock

Digital soil mapping of MM indicates that over 96% of the meadow is a part of the Childs-Chummy complex, 1 to 5 percent slopes soil series (Appendix A-2). The Childs soil series is characterized by very fine sandy loam with a moderately well drained drainage class (Table 3-4). The most limiting layer of this soil series typically has a hydraulic conductivity (Ksat) rating of moderately high to high (0.60 to 2.00 in/hr) (NRCS, n.d.). The Chummy soil series is characterized by an herbaceous mucky peat O horizon with a very poorly drained drainage class (Table 3-4). The most limiting layer of this soil series typically has a hydraulic conductivity (Ksat) rating of moderately high to high (0.20 to 1.98 in/hr) (NRCS, n.d.). The difference hydraulic conductivity rating across MM and RCM may impact the timing it takes for rainfall and snowfall to percolate deeper into the meadows' subsurface.

Table 3-4. Distribution of soil types across MM based on digital soil maps from NRCS's Web Soil Survey.

Soil Series Name	Typical Soil Profile
Childs Soil Series, 1 to 5 percent slopes	*A1 - 0 to 1 inches:* very fine sandy loam *A2 - 1 to 7 inches:* very fine sandy loam *AB - 7 to 13 inches:* very fine sandy loam *Bw1 - 13 to 18 inches:* very fine sandy loam *Bw2 - 18 to 28 inches:* very fine sandy loam *Bw3 - 28 to 37 inches:* very fine sandy loam *Bw4 - 37 to 51 inches:* very fine sandy loam *Bw5 - 51 to 60 inches:* very fine sandy loam
Chummy Soil Series, 1 to 2 percent slopes	*Oe - 0 to 3 inches*: herbaceous mucky peat *A1 - 3 to 12 inches:* mucky silt loam *A2 - 12 to 22 inches:* silt loam *Bw - 22 to 35 inches:* very gravelly sandy clay loam *C - 35 to 39 inches:* gravelly sandy loam *Cg1 - 39 to 49 inches:* gravelly sandy loam *Cg2 - 49 to 63 inches:* very gravelly sandy clay loam

3.1.4 Vegetation

As a part of RCM's 2017 timber harvest plan (THP), Collins Pine Company staff conducted a plant survey in Rock Creek's riparian corridor and adjacent meadow openings. The survey recorded 6 tree species, 19 shrub species, 35 graminoid species, and 111 forbs species (Appendix Table B.1). RCM is surrounded by a mixed conifer forest that is dominated by *Pinus contorta ssp. Murrayana* (lodgepole pine), *Abies concolor* (white fir), *Pinus jeffreyi* (jeffery pine), and *Pinus lambertiana* (sugar pine). Additionally, the riparian areas around Rock Creek are dominated by *Populus tremuloides* (quaking aspen), *Populus trichocarpa* (black cottonwood), *Salix lasiandra* (shining willow), and *Salix lemmonii* (Lemmon's willow). Prior to restoration, these conifer species were concentrated more densely on the west side of the main access road (29.54 m^2/ha) than the east side (22.34 m^2/ha) (Marks, 2021).

3.1.5 Surface Hydrology

RCM and MM are located within the Upper Feather River Watershed, which spans from the headwaters of the North Fork Feather River to Lake Oroville (Figure 3-3). The sub-watershed associated with RCM spans from the headwaters of Rock Creek to Lake Almanor, covering approximately 75.25 km^2 (18,595 acres) (Figure 3-3). Adjacent to RCM is Rock Creek, which is an intermittent stream that begins to flow at the start of the annual snow melt (around March or April) and dries up when the snow has completely melted. A streamflow gauge in the southern portion of RCM that is managed by the Plumas Corporation (Quincy, CA, USA) recorded peak hourly average flow rates in 2017, 2018, and 2019 of 1.4, 0.7, and 6.4 m^3/s respectively. The sub-watershed associated with MM spans approximately 16.90 km^2 (4,176 acres) and contains the intermittent stream Marian Creek (Figure 3-3).

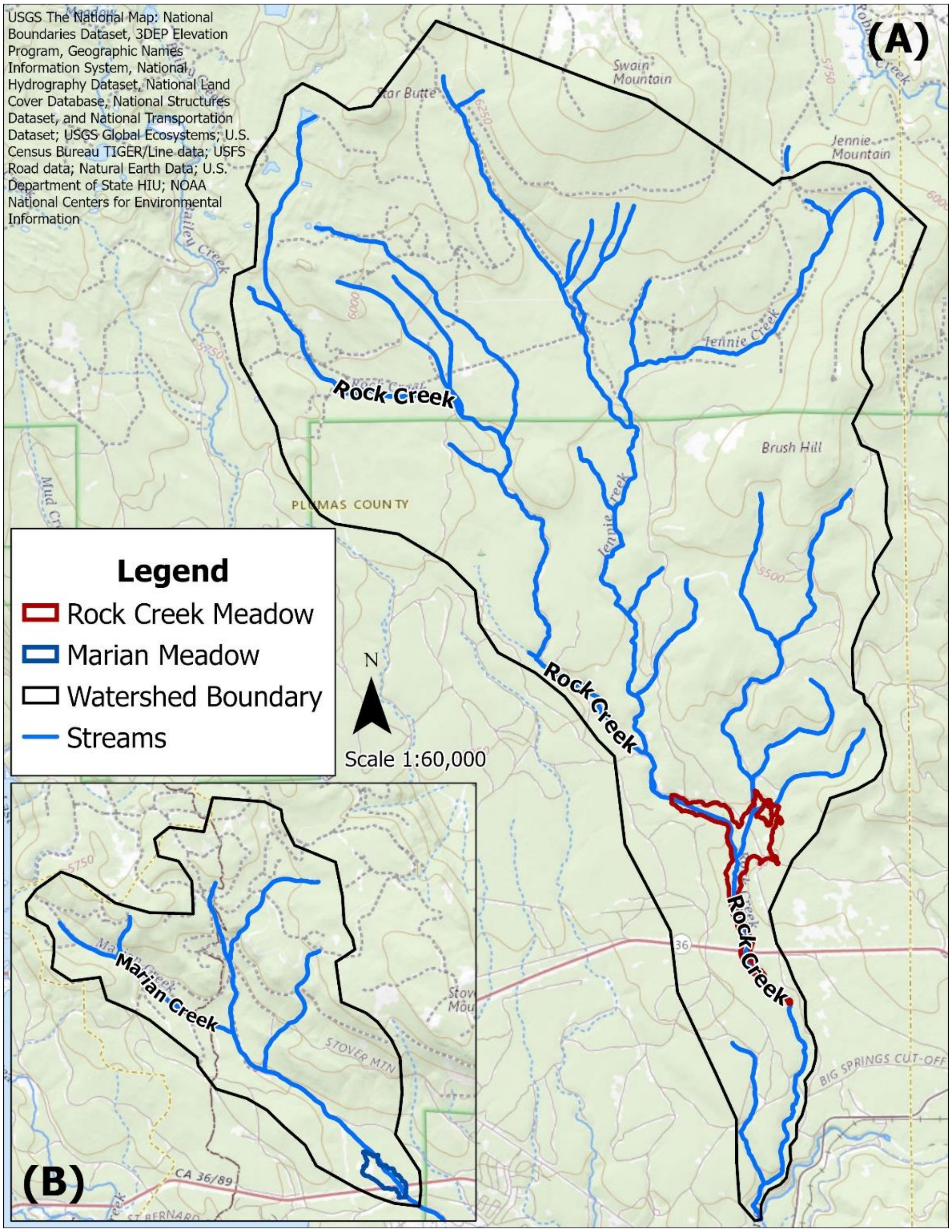


Figure 3-3. Watercourses and watershed boundaries of RCM (A) and MM (B), which are sub-watersheds within the North Fork River Watershed.

3.1.6 Meadow Types

Many prior studies have attempted to classify Sierra Nevada meadows by plant communities, general topography, or elevation range, however, differentiating meadows based primarily on hydrology and geomorphology makes more sense in the context of this study. In 2011, the USDA released a field key to “Meadow Hydrogeomorphic Types for the Sierra Nevada and Southern Cascade Ranges in California,” which differentiates Sierra Nevada and Southern Cascades meadows into fourteen types based on landscape, position, water sources, flow direction and plant species information (Weixelman et. al., 2011). Based on this field key, both the eastern portion of RCM and the entirety of MM are characteristic of a dry meadow (Weixelman et. al., 2011; Surfleet et al., 2020). However, the western portion of RCM is more characteristic of a subsurface low gradient meadow based on its location relative to Rock Creek, different soil qualities, and more hydric vegetation (Weixelman et. al., 2011). When compared to the other types of meadows from this field key, “dry” and “subsurface” meadows fluctuate much more between having high water availability in the subsurface and being dry (Viers et al., 2013). See Appendix C for in-depth description of dry and subsurface low gradient meadows.

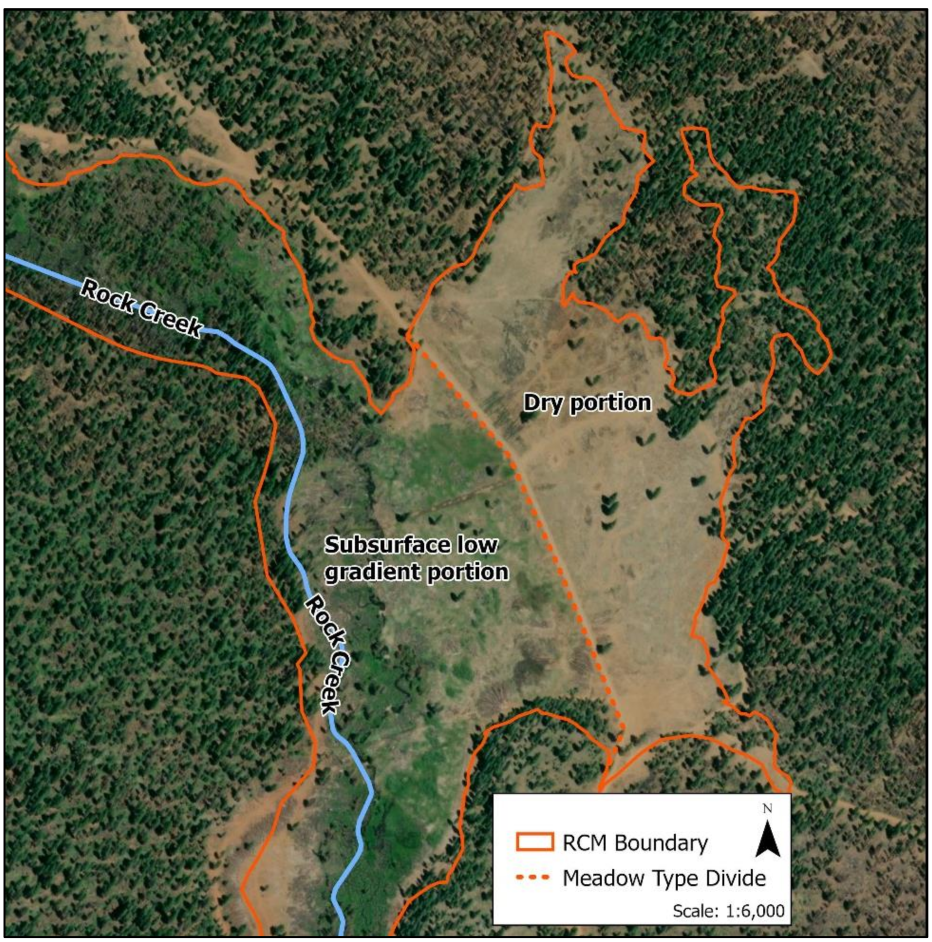


Figure 3-4. Division of RCM based on hydrogeomorphic meadow types. Satellite imagery shows visual differences between each meadow type during the dry summertime season.

3.1.7 Wildfire

Between July and September 2021, the Dixie Fire burned 963,309 acres in California's Butte, Plumas, Shasta, Lassen, and Tehama counties (Cal Fire, 2022). The watersheds associated with RCM and MM were burned during the wildfire at varying intensities. The majority of MM's watershed was burned at a moderate and high intensity, while a little more than half of RCM's watershed was burned at a moderate to high intensity (Table 3-5: Figure 3-5). It is likely that the consumption of herbaceous meadow vegetation and the surrounding forested area will influence the meadow hydrology. For instance, wildfire can alter the snowmelt timing due to canopy loss and blackened trees affecting long-wave radiation (Boisramé et al., 2018). Reduced shading can also lead to increased soil evaporation, while the loss of mature trees can lead to increased infiltration. Additionally, soil moisture levels can be altered due to an increase in water demands from meadow vegetation regrowth (Boisramé et al., 2018). These impacts were not anticipated in the original study design.

Table 3-5. Description of the 2021 Dixie Fire's impacts on the study meadows' vegetation and burn severity within the greater watershed.

Meadow	Watershed Contributing Area km^2 ($mile^2$)	Percentage Moderate and High Burn Severity in Watershed	Meadow Vegetation Post Fire
Rock Creek Meadow (RCM)	70.3 (27.2)	57%	Patches of burned vegetation with varied burn severity.
Marian Meadow (MM)	13.5 (5.2)	78%	Moderate to high burn severity in the meadow.

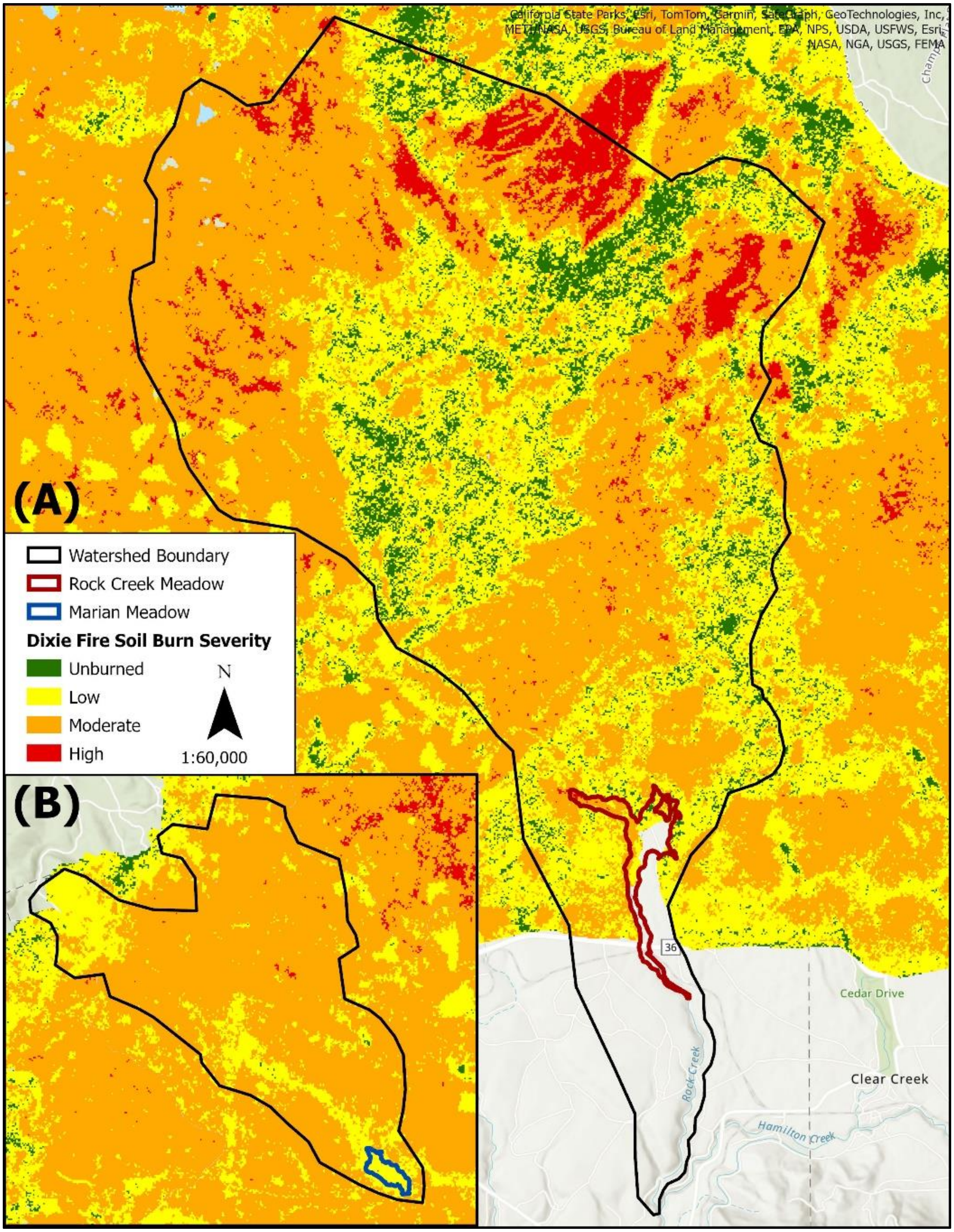


Figure 3-5. Soil burn severity for the watersheds associated with RCM (A) and MM (B).

3.2 Study Design

A Before-After Control-Intervention (BACI) study design was used to assess changes in hydrologic conditions at MM and RCM before and after the removal of encroached conifers. The BACI design was chosen to help account for year-to-year or seasonal climactic variation on the study sites, while also being able to compare hydrologic conditions before and after restoration. One year of pre-restoration and three years of post-restoration data was collected at the meadows in this study (Figure 3-6). MM was used as the control for RCM due to its proximity, similar climate, and similar plant species. Pre-restoration comparison of the recession of groundwater were more between RCM and MM than with the control meadow used with MM. Restoration at MM occurred approximately five years before restoration at RCM. Analysis of MM groundwater and soil moisture conditions suggested that the hydrologic conditions of MM following restoration had stabilized enough to be used as a control for RCM (Surfleet et al., 2020).

Hydrologic conditions play an important role in whether a meadow can return to a stable hydrologic and vegetative state post-restoration (Ratliff, 1985). The hydrologic conditions measured at RCM and MM were soil moisture (m^3/m^3) and depth to groundwater (m). Soil moisture is useful for measuring changes in meadow hydrologic conditions because it represents the balance of precipitation, evapotranspiration, and water percolation within a localized area (Boisramé et al., 2018). Depth to groundwater is another useful metric because shallow water tables are important for meadow development (Ratliff, 1985).

Climatic data such as precipitation, temperature, solar radiation, and barometric pressure were also collected at two sites on or near the meadows. Barometric pressure was used to adjust the pressure head in the groundwater wells.

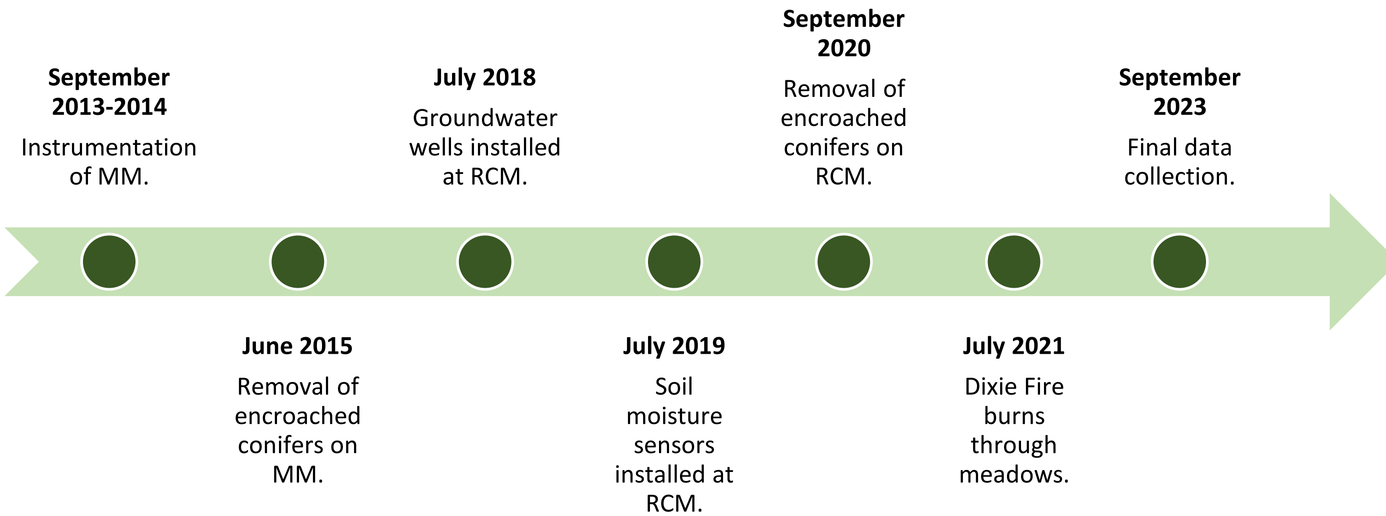


Figure 3-6. Timeline of key events on Marian Meadow and Rock Creek Meadow before and after restoration (adapted from Fie, 2018).

3.3. Instrument Installation

3.3.1. Marian Meadow

The initial soil moisture sensors installed at MM in September 2013 were manufactured by Odyssey Dataflow Systems Limited (Van Oosbree, 2015). These sensors were installed 30 cm (1 ft) deep from the soil surface. The sensors' raw values were converted to gravimetric wetness using a two-point calibration (See Sanford, 2016 for full calibration description). In August 2015, four EC5 Decagon Devices soil moisture sensors were installed at MM at depths of 10 cm (0.33 ft), 30 cm (1 ft), and 1 m (3.28 ft) to determine soil moisture deeper within the soil profile (Sanford, 2016). These devices were pre-calibrated by Decagon Devices and attached to an Onset Computer Corporation HOBO Micro Station Data Logger (Sanford, 2016).

Additionally, in September 2013, Odyssey Dataflow Systems Limited water level loggers were installed in shallow (1.3-1.5 m) groundwater wells at MM (Sanford, 2016). To prevent soil from entering the well, while allowing groundwater to flow into the well, the loggers were placed in 1.5-inch diameter PVC well casings with small holes covered with a screen near the bottom. The wells also had capped tops to prevent precipitation or surface water from entering the well (Van Oosbree, 2015). Additional water level loggers were deployed at MM in September 2014. This included deep (3 m) groundwater wells, and multiple blank or non-instrumented wells (Sanford, 2016). The water level sensors within the wells were calibrated using manually sounded well values and their raw values (See Sanford, 2016 for full calibration description). Figure 3-7 illustrates where all soil moisture sensors and groundwater wells were installed in MM.

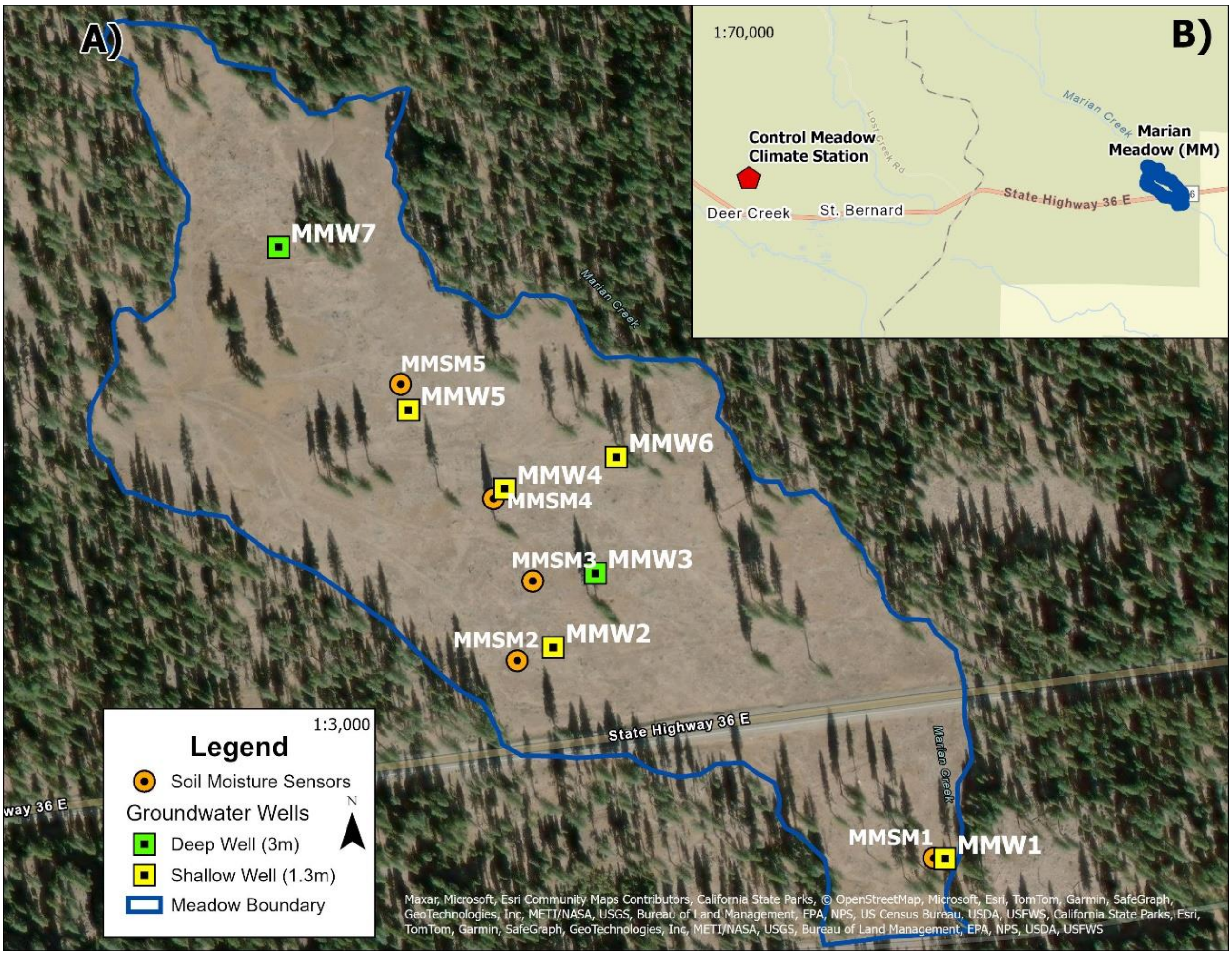


Figure 3-7. Map (A) is of MM soil moisture sensors and groundwater wells. Map (B) shows the location of the climate station, used for correcting the groundwater well pressure, relative to MM.

Soil moisture sensors and groundwater wells were installed in MM using a spatially balanced random sampling approach (adapted from Stevens and Olsen, 2004). At Marian Meadow a 1,250-foot line bisecting the meadow running N 45° W was established with ten equally spaced (125 ft) perpendicular lines (Van Oosbree, 2015). Four of the perpendicular lines were randomly selected and further split into 25 ft intervals spanning 500 ft starting from the western meadow edge (Van Oosbree, 2015). Finally, a random number generator was used to identify four subsections for instrumentation on each of the four perpendicular lines (Van Oosbree, 2015).

3.3.2. Rock Creek Meadow

In July 2018, four Onset Computer Corporation U20L-04 Water Level Data Loggers were installed in groundwater wells at RCM (Marks, 2021). These wells were installed at depths between 1.41 m (4.63 ft) and 2.90 m (9.51 ft) (Table 3-6). The loggers were placed in PVC well casings with perforated bottoms to allow groundwater to flow into the well. The wells also had capped tops to prevent precipitation or surface water from entering the well. These well pressure transducers were not vented, so the groundwater pressure needed to be adjusted by atmospheric pressure. Additionally, two In-Situ Level TROLL 500 Data Loggers were installed in groundwater wells by Plumas Corporation (Quincy, CA, USA) in July 2017 (Marks, 2021). The Plumas Corporation's wells had vented pressure transducers.

Table 3-6. RCM well codes, depths, and riser heights. The Plumas Corporation wells have ID's ending with a "P". Adapted from Marks, 2021.

Well ID	Depth (m)	Riser Height (m)	Depth from Surface to Bottom of Well (m)
RCW1	2.90	0.15	2.75
RCW2	1.41	0.09	1.32
RCW3	2.63	0.42	2.21
RCW6	2.90	0.15	2.75
RCW3P	2.38	0.58	1.80
RCW4P	3.05	1.10	1.95

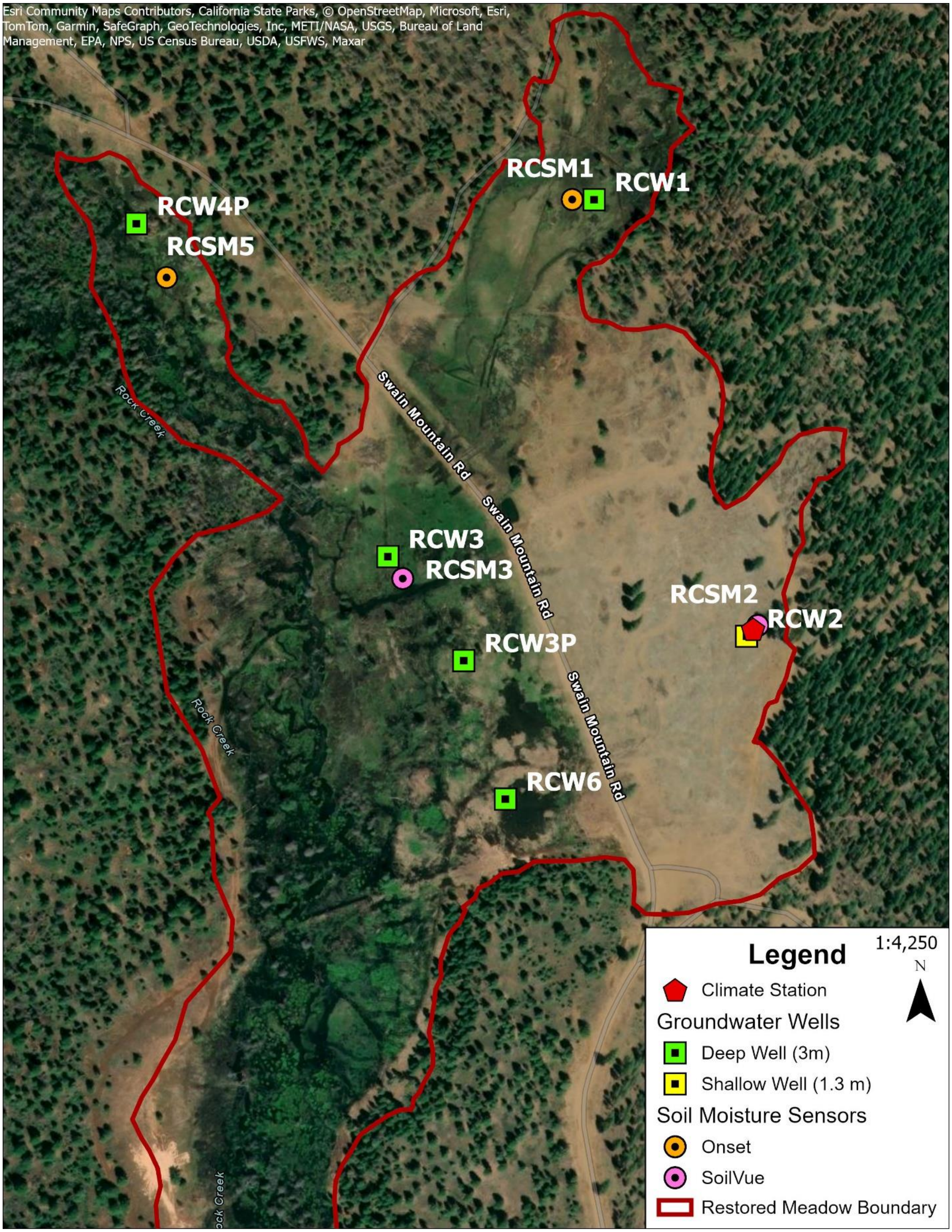


Figure 3-8. Map of RCM soil moisture sensor, groundwater well, and climate station locations.

In July 2019, two soil moisture sensors were installed at RCM. These soil moisture sensors were manufactured by Decagon Devices and attached to an Onset Computer Corporation HOBO Micro Station Data Logger. The sensors were installed at depths of 10 cm (0.33 ft), 30 cm (1 ft), and 1 m (3.28 ft) from the soil surface (Marks, 2021). Due to limitations from instrument availability and the larger size of RCM, the spatially balanced random sampling approach used at MM was not employed at RCM. Instead, a stratified random sample was used to capture the variations in meadow type and plant communities (Figure 3-8). No site-specific time domain reflectometry (TDR) calibration was conducted on the soil moisture probes (see Marks, 2021).

In September 2019 and December 2019 two SoilVue, Campbell Scientific soil moisture sensors were installed at RCM. The SoilVue sensors provide soil moisture measurements at nine depths 5, 10, 20, 30, 40, 50, 60, 75, and 100 cm. These additional sensors were attached to CR1000 and CR300 Control Dataloggers manufactured by Campbell Scientific. However, the SoilVue sensors had firmware errors and did not have adequate connection with the soil, so they were removed, repaired, and re-installed in May 2021.

3.3.3 Climate Stations

Prior to the beginning of this study, a weather station had been installed in a meadow adjacent to MM called Control Meadow (CM) (Figure 3-7b). Then in September 2019, an Onset Computer Corporation Weather Station was installed on the eastern half of RCM near RCW2 and RCSM2 (Figure 3-8). The stations had sensors to measure air temperature, relative humidity, wind speed, wind direction, barometric pressure, shortwave solar radiation, and precipitation (Marks, 2021). Measurements were recorded by an Onset Computer Corporation HOBO U30 USB Weather Station Data Logger in 30-minute intervals. The barometric pressure (psi) recorded by these weather stations were used to calibrate/correct the well pressure collected at the meadows.

3.4 Data Collection

3.4.1 Soil Moisture Monitoring

The soil moisture data loggers were set to record soil moisture measurements in 30-minute intervals. Field site visits were conducted at a variable rate due to travel distance, schedule availability, and weather, however, they generally occurred every two to three months. Due to the time gap between field visits (Table 3-7), if there were any issues with the data loggers recording data, those issues would not be resolved for that time, resulting in data gaps. Soil moisture sensors that failed throughout the study were either replaced or removed from the study. Failure was caused by things such as poor soil connection, dead batteries, exposure to the elements, wildlife, vandalism, and wildfire.

Table 3-7. Timing and frequency of field site visits to RCM and MM between 2019 and 2023.

	2019	2020	2021	2022	2023
January					X
February					
March					
April		X		X	X
May	X		X		
June				X	X
July	X	X	X		X
August		X			
September	X		X	X	X
October					
November		X			
December	X		X		

From its installation in September 2019 until May 2021, the 1 m probe of RCSM2 had a poor connection with the soil. This resulted in poor soil moisture readings that could not be used in the final analysis. The two SoilVue sensors (RCSM2 and RCSM3) were removed and reinstalled in May 2021, due to issues with the probes following a field site visit in November 2020. During a site visit in June 2022, the 10 cm soil moisture probe on RCSM5 was replaced. However, following this visit the 30 cm and 1 m probes at RCSM5 also began recording poor data. Therefore, in June 2023 all the probes at RCSM5 were replaced.

Other data gaps recorded in Figure 3-9 were resolved by replacing dead batteries and/or relaunching the data logger.

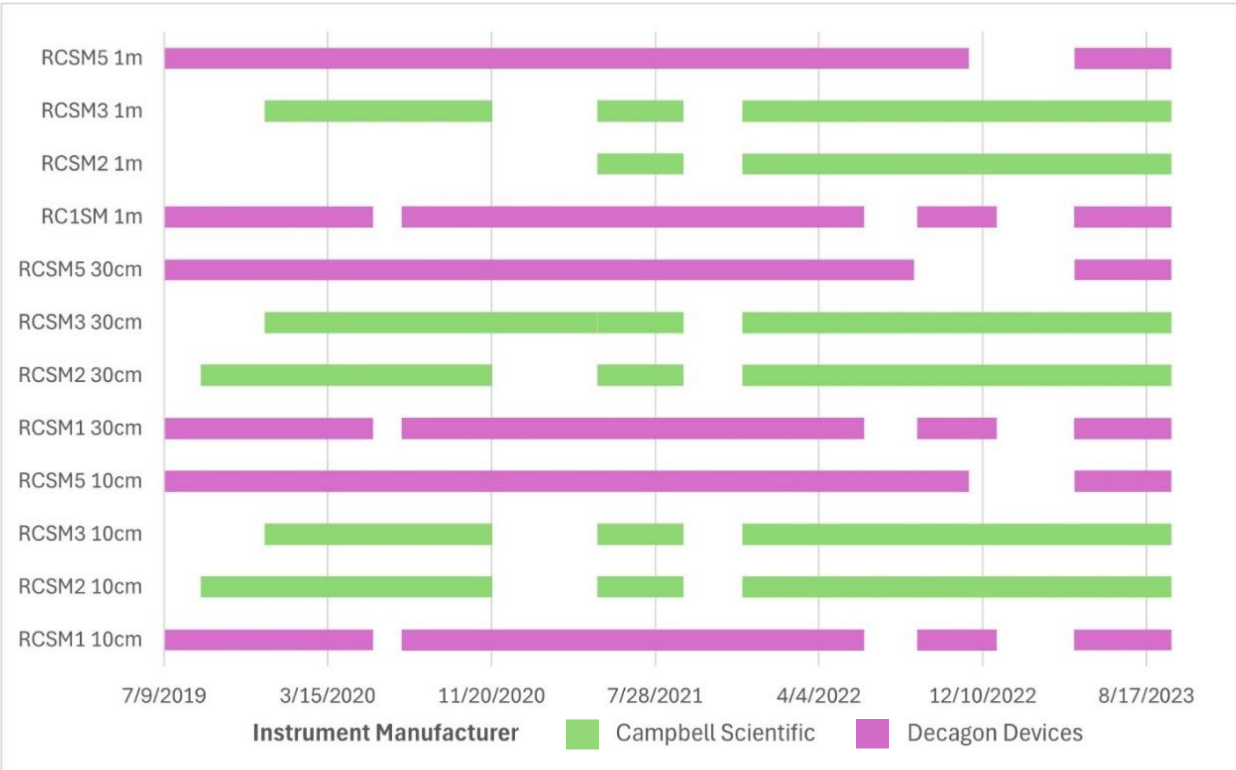


Figure 3-9. Timeline of soil moisture sensors operating at RCM. The 1m probe on RCSM2 had connection issues that were not resolved until May 2021. Other gaps were due to failures caused by exposure to the elements, dead batteries, and issues with connectivity between the data logger and probes.

3.4.2 Groundwater Depth Monitoring

Like the soil moisture loggers, the groundwater probes were set to record water level in 30-minute intervals. Field site visits generally occurred every two to three months (Table 3-7). The well probes failed less frequently than the soil moisture probes. Low or missing water level values were typically the result of a dry well, rather than instrumental error.

RCW1 had data issues but was relaunched in December 2019 and no other issues were experienced with this probe (Figure 3-10). RCW2 had a full memory in July 2020, so it was relaunched. Additionally, in April 2022 data could not be downloaded off the probe so it was replaced. Due to extensive snow during the 2022-2023 winter, wells were not serviced between September 2022 and June 2023.

RCW3 stopped recording in December 2022, so it was replaced and launched in June 2023. The PVC well casing for RCW6 was crushed during site clean-up following the lodgepole pine removal, but data was able to be retrieved from the probe up to mid-August 2021. Due to the adequate distribution of wells within the meadow, including the two Plumas Corp. wells, RCW6 was not re-installed.

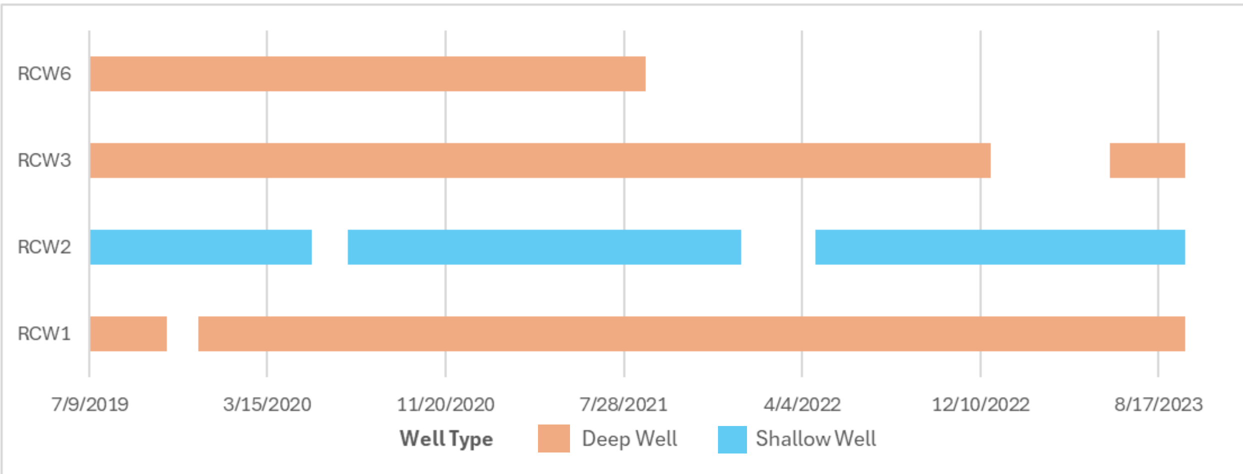


Figure 3-10. Timeline of the operationality of groundwater well probes at RCM. RCW6 was run over by logging equipment following the 2021 Dixie Fire and was not re-installed.

3.4.3 Climate Monitoring

Prior to September 2019, barometric pressure (psi) data recorded at the CM weather station was used to calibrate/correct well pressure data collected at both MM and RCM. Once the RCM weather station began recording barometric pressure (psi) in September 2019, data from this station was used to calibrate/correct well pressure data collected at RCM. Meanwhile, MM well pressure data continued to be calibrated/corrected by the barometric pressure (psi) recorded at the CM weather station. Onset Computer Corporation's *HOBOware Pro* software contains a barometric compensation assistant which uses barometric data collected by the weather stations to correct water pressure data collected by the HOBO Micro Station Data Logger in the meadow wells (Onset Computer Corporation, n.d.). RCW3P and RCW4P had vented pressure transducers, so barometric pressure data was not needed to correct these well pressure values.

On 9/30/2022 the CM barometric pressure probe stopped recording and on 10/1/2022 the RCM barometric pressure probe did the same. At RCM this was not noticed during the January 2023 field site visit, so the barometric pressure probe was not relaunched until 4/29/2023. The MM station had a chain link fence around it that could not be opened until the snow melted. During the April 2023 field visit, it was not noticed that the probe had stopped recording, so it was not replaced until a site visit on 6/10/2023.

During these gaps in reliable barometric pressure data, a barometric formula that relates altitude and air temperature to barometric pressure (Equation 1) was used. Altitude was determined by the elevation (m) of the meadows and temperature (°K) data was used from the climate stations at the meadows. The calculated barometric pressure values were converted from in-Hg to psi, which is what the well pressure values were recorded in. Equation 2 was used to calculate the predicted depth to groundwater in feet.

$$P_a = P_0 e^{-gMh/RT} \tag{1}$$

Where:

P_a = Barometric pressure (psi)
P_0 = Air pressure at sea level (29.92 in-Hg)
g = Acceleration due to gravity (9.80665 m/s^2)
M = Molar mass of air (0.0289644 kg/mol)
h = Altitude (m)
R = Gas constant (8.31432 J/mol*K)
T = Temperature (°K)

$$\widehat{D_{gw}} = (d_w - (P_w - P_a) * C) + O \tag{2}$$

Where:

$\widehat{D_{gw}}$ = Predicted depth to groundwater (m)
d_w = Well depth (m)
P_w = Well pressure (psi)
C = Conversion factor from psi to m of water (0.704 m/psi)
O = Average offset (m)

The predicted depth to groundwater (m) values were compared to observed depth to groundwater (m) values that were calculated using barometric data values recorded from the climate stations between 4/3/2022 and 9/1/2022 at 30-minute intervals. On average, the predicted barometric values underpredicted depth to groundwater by 0.06 to 0.11 m (Table 3-8). To account for this difference, the average offset was included in Equation 2. The offset after correction (Table 3-8) was considered small enough to use this method to correct well pressure data at RCM between 10/1/2022 and 4/29/2023, and at MM between 9/30/2022 and 6/10/2023.

Table 3-8. Table of the average offset between observed depth to groundwater (m) and predicted depth to groundwater (m), as well as the average offset after equation 3 was used to correct the predicted depth to groundwater (m).

Well ID	Offset (m)	Offset After Correction (m)
RCW1	0.06	5.0E-17
RCW2	0.06	2.0E-16
RCW3	0.10	8.5E-17
MMW2	0.11	2.7E-16
MMW3	0.10	1.8E-16
MMW4	0.10	2.5E-18
MMW5	0.10	1.9E-18
MMW7	0.10	2.0E-16

3.5 Data Analysis

3.5.1 Soil Moisture

Each 30-minute interval of soil moisture data was filtered out for values less than 0.02 since values below this typically indicate that the soil is dry or there is an error with the sensor. For each individual soil moisture probe and depth (10 cm, 30 cm, and 100 cm), weekly averages were taken of these cleaned 30-minute interval values. For both RCM and MM, these averages went from Saturday 7/13/2019 to the week of 9/16/2023.

Weekly averages were then averaged across all soil moisture sensor probes for each respective depth. For all of RCM, these averages were based on RCSM 1, RCSM 2, RCSM 3, and RCSM 5. For RCM West only RCSM 3 and RCSM 5 were considered, and for RCM East only RCSM 1 and RCSM5. Time series visualizations were created using these weekly averages and compared to weekly precipitation totals for both RCM and MM. Additionally, these weekly averages were averaged across all depths for each week to get the overall average soil moisture between 0-100 cm.

These weekly averages were averaged by month and restoration year to illustrate the average monthly soil moisture value pre-restoration and each year post restoration. The month the weekly average started in was the month the average value was included in. There should be little difference in soil moisture values between the end of one month and the beginning of another month. MM weekly averages were just averaged by month because the meadow's soil moisture conditions remained relatively stable throughout the study.

For statistical analysis, one weekly average data point was taken every three weeks, effectively skipping two weeks of data between each data point. This was done to reduce the impact of serial autocorrelation typically associated with time series data. Using these data points, a multiple linear regression analysis was performed to compare RCM soil moisture to MM soil moisture for each year pre- and post-restoration. The relationship between RCM soil moisture and MM was included in this analysis because MM soil moisture values are being treated as the baseline conditions for what a restored meadow's soil moisture levels should be. Additionally, the analysis included an interaction term between MM and restoration year. Restoration year was a categorical variable used to group RCM and MM soil moisture values by pre-restoration, year 1 post-restoration, year 2 post-restoration, and year 3 post-restoration. Grouping the data by restoration year should account for climatic variations such as snow and rain precipitation between the years. Equation 3 illustrates the linear regression model used to perform this analysis in RStudio

$$RCM\ soil\ moisture \sim MM\ soil\ moisture * Restoration\ Year \tag{3}$$

Using the same model in RStudio (equation 3), an estimated marginal means (EMMs) analysis was performed. EMMs is an analysis that reports the mean response for a group based on the mean of a covariate (Grace-Martin, 2021). In this study, the mean RC soil moisture value was grouped by restoration year and based on the mean MM soil moisture value. The intention of this EMMs model is to see if restoration year influences RC mean soil moisture beyond the effect of MM mean soil moisture.

Finally, a multiple linear regression analysis was performed in RStudio to compare RCM soil moisture to MM soil moisture for pre- and post- Dixie Fire. The analysis included an interaction term between MM and fire year. Similar to restoration year, fire year was a categorical variable used to group RCM and MM soil moisture values by pre- or post- Dixie Fire. This analysis was intended to determine if the 2021 Dixie Fire accounted for a significant amount of variability, with MM soil moisture values still being treated as the baseline conditions for what a restored meadow's soil moisture levels should be.

3.5.2 Depth to Groundwater

Each 30-minute interval of depth to groundwater data was filtered out for values less than 0.2 since values below this typically indicate that the well is dry or there is an error with the sensor. For each individual depth to groundwater probe, weekly averages were taken of these cleaned 30-minute interval values. For both RCM and MM, these averages went from Saturday 4/27/2019 to the week of 9/16/2023.

Weekly averages were then averaged across all depth to groundwater probes. For all of RCM, these averages were based on RCW 1, RCW 2, RCW 3, RCW 3P, RCW 4P, and RCW 6. For RCM West only RCW 3, RCW 3P, RCW 4P, and RCW 6 were considered, and for RCM East only RCW 1 and RCW 2. Time series visualizations were created using these weekly averages and compared to weekly precipitation totals for both RCM and MM.

Like soil moisture, these weekly averages were averaged by month and restoration year to illustrate the average monthly depth to groundwater value pre-restoration and each year post restoration. MM weekly averages were just averaged by month because the meadow's depth to groundwater conditions remained relatively stable throughout the study. Whatever month the weekly average started in was the month the average value was included in.

Statistical analysis on depth to groundwater was performed in the same way as soil moisture, where RCM depth to groundwater was regressed against MM depth to groundwater for each year pre- and post-restoration. This analysis also included an interaction term between MM and restoration year and was analyzed in RStudio using the linear regression model illustrated by equation 4. For the EMMs analysis, the mean RC depth to groundwater value was grouped by restoration year and based on the mean MM depth to groundwater value, to see if restoration year influences RC mean depth to groundwater beyond the effect of MM mean depth to groundwater.

$$RCM\ depth\ to\ groundwater \sim MM\ depth\ to\ groundwater\ * Restoration\ Year \tag{4}$$

Again, a multiple linear regression analysis was performed in RStudio to compare RCM depth to groundwater to MM depth to groundwater for pre- and post- Dixie Fire. As with the soil moisture analysis, an interaction term between MM and fire year was included.

4.0 Results

4.1 Seasonal Differences

This chapter includes time series graphics of the changes to volumetric soil moisture content (%) (Figure 4-1 and Figure 4-2) and average changes in volumetric soil moisture content of years (Table 4-1) and months (Figure 4-3). It also includes time series graphics of the changes to depth to groundwater (m) (Figure 4-4 and Figure 4-5) and average monthly changes in depth to groundwater (Figure 4-6 and Table 4-2). Marian Meadow (MM) volumetric soil moisture content (%) and depth to groundwater (m) were used as a control to account for seasonal variation.

4.1.1 Changes in Volumetric Soil Moisture

While soil moisture measurements were taken at depths of 10 cm, 30 cm, and 100 cm, the focus of the time series graphics are on 30 cm. In areas dominated by herbaceous plants, the upper 30 cm of soil contains most of the total root biomass (Mueller et al., 2013). Additionally, the soil moisture data taken for 10 cm and 100 cm had changes that were consistent with the changes in 30 cm.

The soil moisture time series graphics illustrate an increase in soil moisture content over time following the removal of encroached conifers. In both Figure 4-1 and Figure 4-2, weekly average RCM soil moisture values were slightly lower than weekly average MM soil moisture values prior to conifer removal. The first year after conifer removal, RCM soil moisture values continued to be slightly lower than MM values, however, in year 2 and 3 RCM's soil moisture values appear larger than that of MM. Figure 4-1 is an average of RCM soil moisture content across the meadow, while Figure 4-2 shows the difference in soil moisture content between RCM West and RCM East.

In both figures there appears to be a lag between some peaks in MM volumetric soil moisture content and RCM volumetric soil moisture content. Dips in soil moisture content also appear in both meadows during the late summer and fall when there is little to no precipitation.

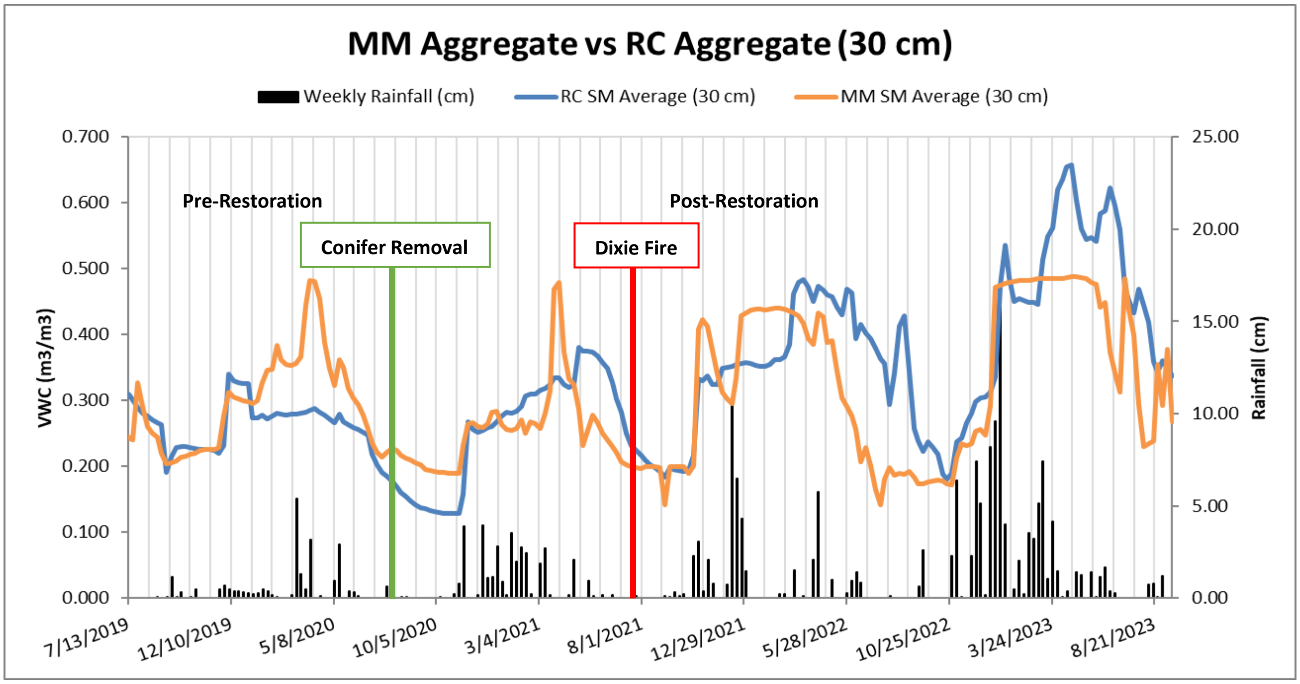


Figure 4-1. Time series of volumetric soil moisture content (m^3/ m^3) at a depth of 30 cm for RCM and MM from July 2019 to September 2023. Weekly rainfall (cm) included to show where influxes in soil moisture may correspond to precipitation.

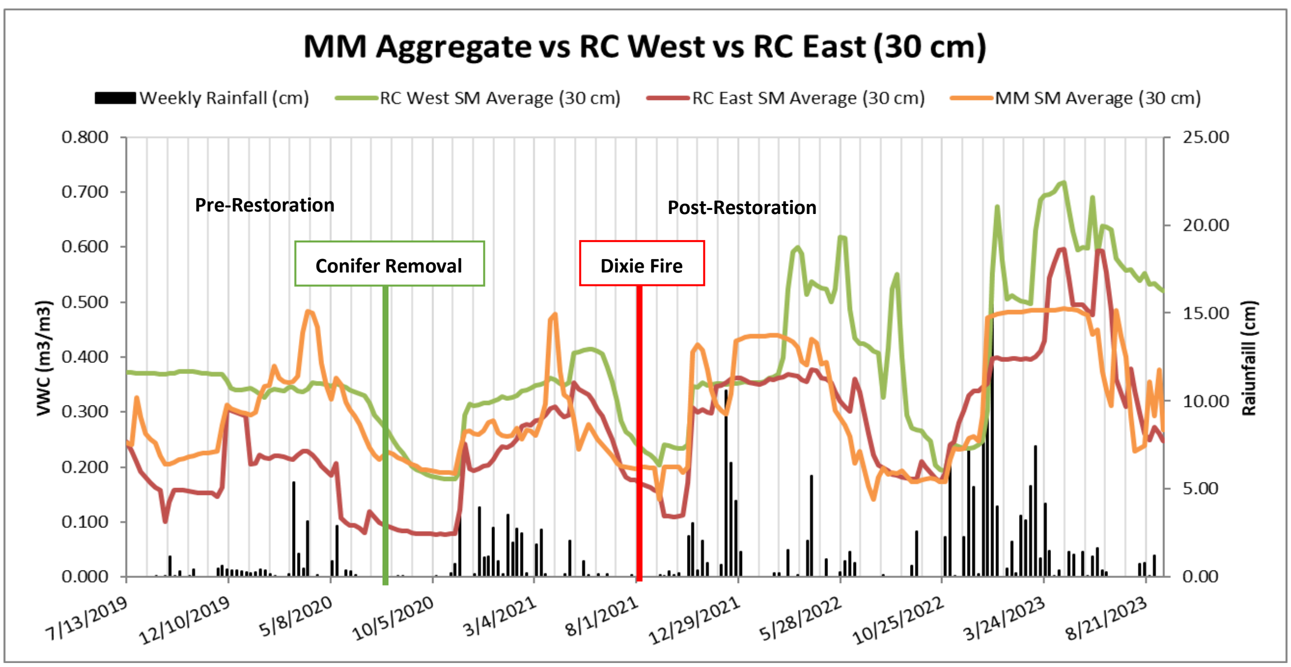


Figure 4-2. Time series of volumetric soil moisture content (m^3/ m^3) at a depth of 30 cm for RCM and MM from July 2019 to September 2023. RCM is split between the east and west portions to illustrate differences in soil moisture levels. Weekly rainfall (cm) included to show where influxes in soil moisture may correspond to precipitation.

Aggregated averages of soil moisture content from 0-100 cm deep also show increases in soil moisture content at RCM over MM following restoration (Table 4-1 and Figure 4-3). WY 2019 and 2020 are pre-restoration values, WY 2021 is 1 year post-restoration, WY 2022 is 2 years post-restoration, and WY 2023 is 3 years post-restoration. In WY 2019 RCM soil moisture ranges from 27% to 33% while MM soil moisture ranges from 17% to 31%, and in WY 2020 RCM soil moisture ranges from 19% to 36% while MM soil moisture ranges from 20% to 46%. RCM's average soil moisture value was higher than that of MM in WY 2021, however MM had a higher maximum soil moisture content of 47% over RCM's 39%. In WY 2022 RCM's average soil moisture value was higher than MM and the maximum value was the same as MM, however, RCM had a higher minimum soil moisture content of 23% over MM's 14%. By WY 2023 RCM's minimum, average, and maximum soil moisture content values were all higher than that of MM.

Table 4-1 also shows the difference between pre-restoration and post-restoration values where post-restoration values are an average of year 1, year 2, and year 3 post-restoration. Pre-restoration RCM's average and minimum soil moisture values were larger than MM's, however, MM had a higher maximum soil moisture content of 46% over RCM's 36%. Post-restoration RCM's minimum, average, and maximum soil moisture values were larger than MM's, with RCM's maximum soil moisture value being 65% and MM's being 49%.

Aggregated RCM and MM soil moisture content varied by month. From June to November MM volumetric soil moisture content varied from 20% to 30%, and from December to May volumetric soil moisture content varied from 32% to 44%. Soil moisture at RCM was lower than MM between January and May, but from June to December RCM soil moisture is higher than MM. The first-year post-restoration RCM soil moisture values stayed fairly similar to pre-restoration value. The second-year post-restoration RCM soil moisture values between March to August were higher than MM and RCM pre-restoration values. The third-year post-restoration RCM soil moisture values between January to September were higher than MM and RCM pre-restoration values, while the October to December values were slightly lower than RCM pre-restoration values.

Table 4-1. Minimum, maximum, and average volumetric soil moisture content (%) for RCM and MM between the 2019-2023 water years and by restoration years. Values are based on an average of measurements from depths of 10, 30, and 100 cm.

		Marian Meadow (m3/m3)			RC Meadow (m3/m3)		
		Min.	Avg.	Max.	Min.	Avg.	Max.
WY	2019	0.17	0.23	0.31	0.27	0.31	0.33
	2020	0.20	0.30	0.46	0.19	0.30	0.36
	2021	0.17	0.27	0.47	0.18	0.30	0.39
	2022	0.14	0.33	0.46	0.23	0.37	0.46
	2023	0.19	0.37	0.49	0.22	0.45	0.65
Pre-Restoration		0.17	0.30	0.46	0.20	0.31	0.36
Post-Restoration		0.14	0.32	0.49	0.18	0.37	0.65

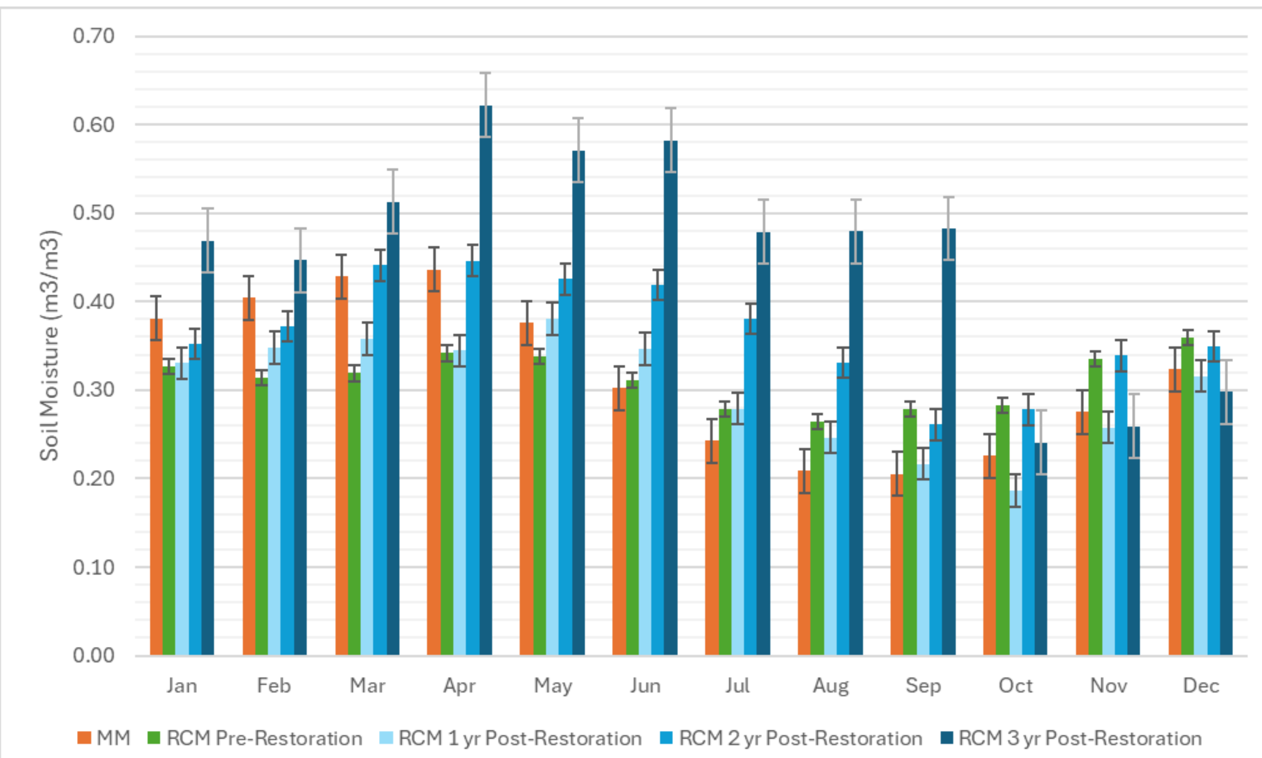


Figure 4-3. Average monthly volumetric soil moisture content (m^3/m^3) for MM, RCM Pre-restoration, and RCM for each year post-restoration (WY 2021-2023) with standard error bars. Values are based on an average of measurements taken from depths of 10, 30, and 100 cm.

4.1.2 Changes in Depth to Groundwater

The depth to groundwater time series illustrates the change in RCM depth to groundwater content over time in the context of seasonal variations shown by MM depth to groundwater (Figures 4-4 and 4-5). Lower depth to groundwater values indicates a higher water level or more groundwater, while higher depth to groundwater values indicates a lower water level or less groundwater. Figure 4-4 shows the weekly average depth to groundwater across RCM while Figure 4-5 shows the difference in water level between RCM West and RCM East. The depth to groundwater (m) y-axis for both Figures 4-4 and 4-5 is inverted to display trends in the water table, where 0 m is the surface. Dips in depth to groundwater content also appear in both meadows during the late summer and fall when there is little to no precipitation. Gaps in RCM East data are due to the wells being dry or issues with the data logger as noted in Figure 3-10.

Prior to conifer removal, RCM depth to groundwater values appear similar if not slightly lower than weekly average MM depth to groundwater values especially when looking at just RCM West. The first year after conifer removal, RCM depth to groundwater values appear to be slightly higher than MM values. Again, in year 2 and the beginning of year 3 RCMs depth to groundwater values appear larger than that of MM. However, towards the end of year 3 RCM's average depth to groundwater values are lower than MM (Figure 4-4). Similar to soil moisture, Figure 4-4 and Figure 4-5 appear to have a lag between some peaks in MM depth to groundwater content and RCM volumetric depth to groundwater content, with RCM trailing behind MM.

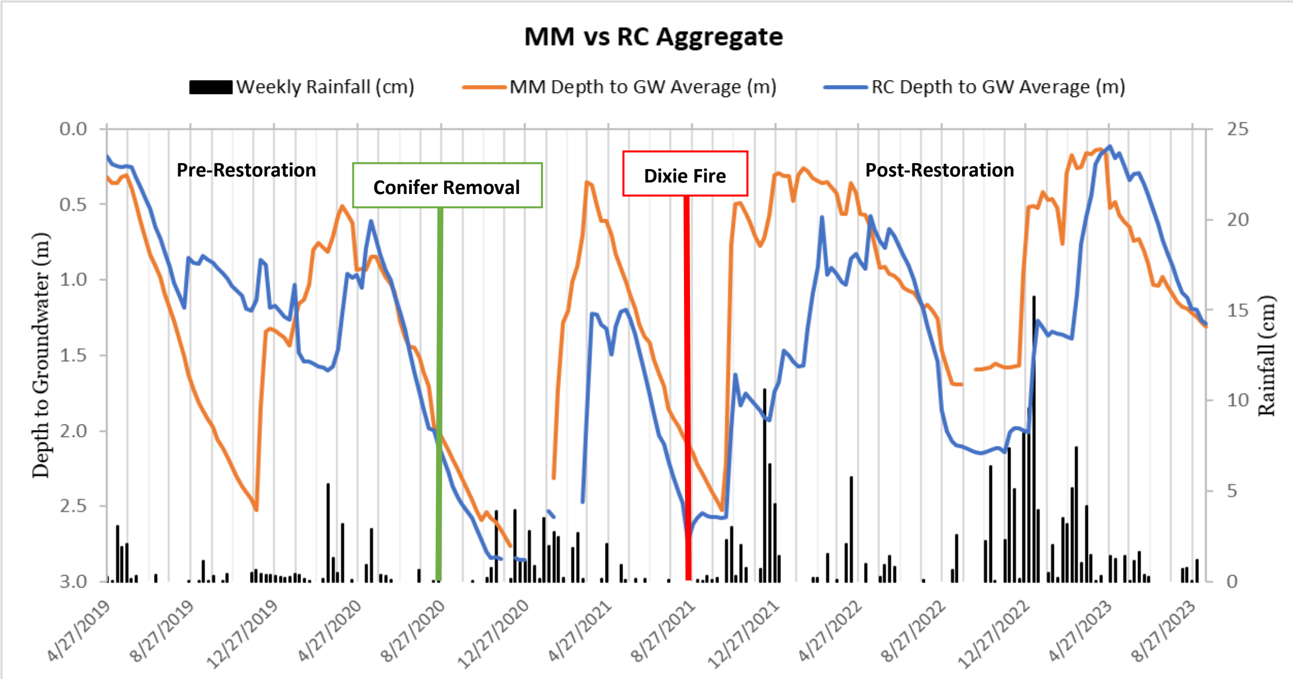


Figure 4-4. Time series of depth to groundwater for RCM and MM from April 2019 to September 2023. Weekly rainfall (cm) is included to show where decreases in depth to groundwater may correspond to precipitation.

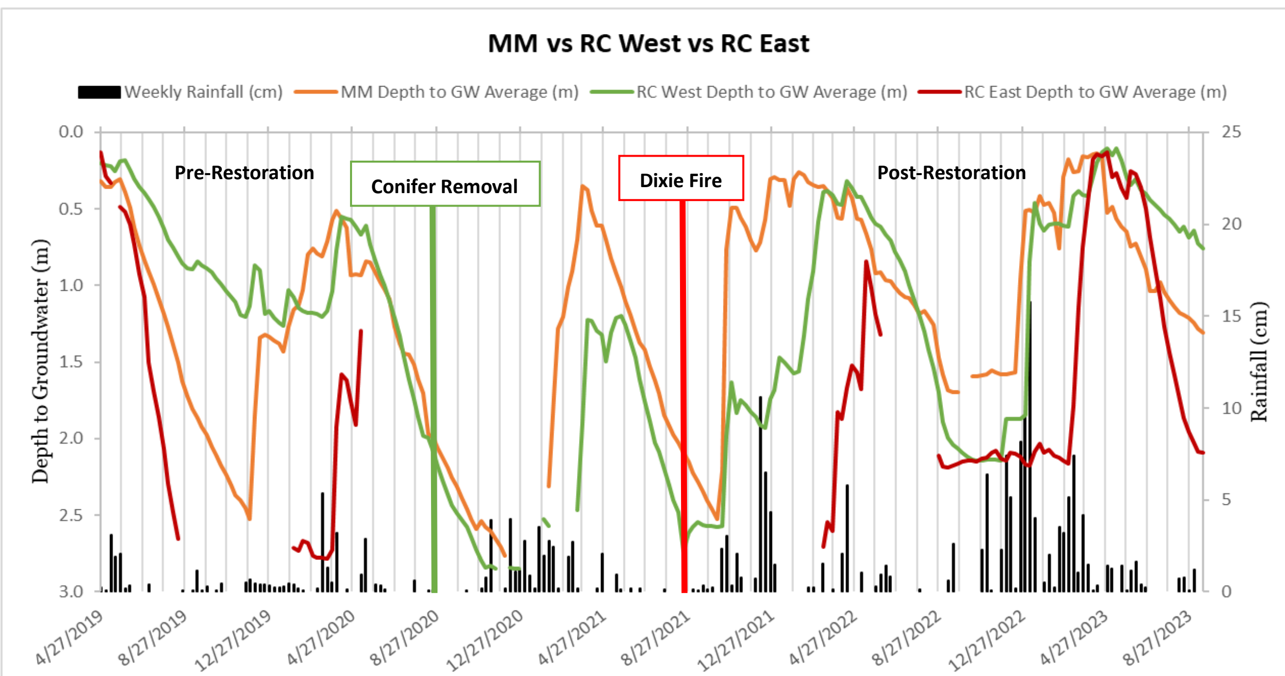


Figure 4-5. Time series of depth to groundwater for RCM West, RCM East, and MM from April 2019 to September 2023. Weekly rainfall (cm) is included to show where decreases in depth to groundwater may correspond to precipitation.

Figure 4-6 illustrates how RCM and MM depth to groundwater content varied by month. Prior to restoration, RCM's depth to groundwater was lower than MM from May to December with the lowest depth (0.55 m) in May and the highest depth (1.63 m) in September. All months during the first-year post-restoration RCM had a greater depth to groundwater than MM with RCM, indicating lower levels of groundwater in RCM. RCM wells were dry in January, February, and December. By the second-year post-restoration RCM had lower depth to groundwater values than MM in June through August, with the lowest value being 0.73 m in May. In the third-year post-restoration, April through September had lower depth to groundwater values than MM with RCM groundwater being 0.59 m shallower than MM groundwater in June (Table 4-2).

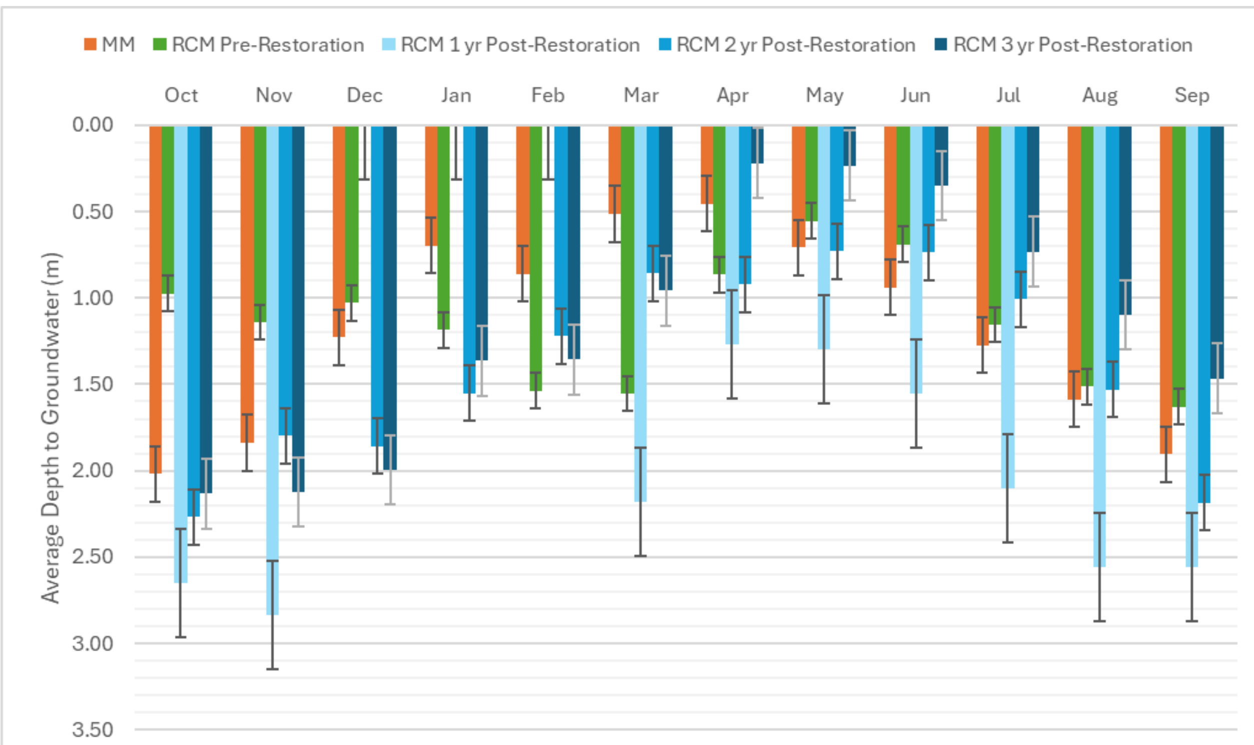


Figure 4-6. Average monthly depth to groundwater (m) for MM, RCM Pre-restoration, and RCM for each year post-restoration (WY 2021-2023) with standard error bars. The absence of data for RCM 1-year post-restoration during January, February, and December is due to the wells being dry, meaning the average depth to groundwater was greater than 2.8 m.

Table 4-2. Average monthly depth to groundwater (m) for MM, RCM Pre-restoration, and RCM for each year post-restoration (WY 2021-2023). Red values indicate that the depth to groundwater is deeper than MM levels, while green values indicate that the depth to groundwater is shallower than MM levels.

	Average Depth to Groundwater (m)				
Month	MM	RCM Pre-Restoration	RCM 1 Year Post-Restoration	RCM 2 Years Post-Restoration	RCM 3 Years Post-Restoration
Oct	2.02	0.97	2.65	2.27	2.13
Nov	1.84	1.14	2.84	1.80	2.12
Dec	1.23	1.03	Dry	1.86	1.99
Jan	0.70	1.19	Dry	1.55	1.37
Feb	0.86	1.54	Dry	1.22	1.36
Mar	0.51	1.55	2.18	0.86	0.96
Apr	0.46	0.87	1.27	0.92	0.22
May	0.71	0.55	1.29	0.73	0.23
Jun	0.94	0.69	1.55	0.74	0.35
Jul	1.27	1.16	2.10	1.01	0.73
Aug	1.59	1.51	2.56	1.53	1.10
Sep	1.90	1.63	2.56	2.19	1.47

4.3 Statistical Analysis

This section includes a multiple linear regression analysis between RCM and MM volumetric soil moisture content for 0-100 cm by restoration year (Figure 4-7, Table 4-3, and Table 4-4), and an estimated marginal means (EMMs) analysis using the same model (Figure 4-8, Table 4-5, and Table 4-6). It also includes a multiple linear regression analysis (Figure 4-9, Table 4-7, and Table 4-8) and EMMs analysis (Figure 4-10, Table 4-9, and Table 4-10) between RCM and MM volumetric soil moisture content for 30 cm by restoration year. Like soil moisture, a multiple linear regression analysis was performed between RCM and MM depth to groundwater (Figure 4-11, Table 4-11, and Table 4-12), as well as an EMMs analysis (Figure 4-12, Table 4-13, and Table 4-14). An additional analysis was performed between RCM West and MM depth to groundwater using multiple linear regression (Figure 4-13, Table 4-15, and Table 4-16) and EMMs (Figure 4-14, Table 4-17, and Table 4-18). MM is used in all of these analyses as a baseline for restored meadow conditions. The lack of replication within this study on other meadows besides RCM indicates that any interpretation of these results cannot be applied to the broader population of montane meadows in the Sierra Nevada and Southern Cascade Mountains ranges.

This section also includes an analysis of how the 2021 Dixie Fire impacted soil moisture content and depth to groundwater. Table 4-19 shows RCM versus MM volumetric soil moisture content at 0-100 cm by fire year, while Table 4-20 shows RCM versus MM volumetric soil moisture content at 30 cm by fire year. Additionally, RCM versus MM depth to groundwater by fire year (Table 4-21) and RCM West versus MM depth to groundwater by fire year (Table 4-22) are displayed.

4.3.1 Soil Moisture Statistical Analysis

Going forward "aggregated soil moisture" indicates that an average soil moisture was derived from measurements at depths of 10, 30, and 100 cms. Figure 4-7 illustrates that RCM aggregated soil moisture and MM aggregated soil moisture have a positive linear relationship by restoration year, meaning as MM's soil moisture content increases, so does RCM's. This aligns with the trends seen in Figures 4-1 and 4-2. "Restoration year" is a categorical variable that groups the data as pre-restoration, 1 year post-restoration, 2 years post-restoration, or 3 years post-restoration values. When used as an interaction term, pre-restoration becomes the base case that each other year is compared against.

The analysis of variance (ANOVA) table (Table 4-3) shows how including each independent variable (MM soil moisture content and restoration year) is significant to the model. Individually, MM soil moisture content and restoration year have a statistically significant relationship with RCM soil moisture content. Additionally, the inclusion of an interaction term between these two independent variables is necessary (p-value of 0.0163). The estimate value associated with each interaction term displays the difference between the pre-restoration slope and post-restoration slopes for each year. Therefore, the slope of year 3 post-restoration significantly increased by 0.498 from the pre-restoration slope.

The intercept estimate is the value of RCM soil moisture when MM's soil moisture is at 0% for this model pre-restoration (Table 4-4), while the term *MM Average Soil Moisture content* displays the overall slope of the model pre-restoration. The terms (*Pre-Restoration) - Year 1,* (*Pre-Restoration) - Year 2,* and (*Pre-Restoration) - Year 3* display the difference between pre-restoration values and the post-restoration values for the model when MM soil moisture content is equal to 0%. For example, if the value of MM soil moisture content is 0%, the average value of RCM soil moisture content 1-year post-restoration is 4.59% lower than pre-restoration (Table 4-4). The p-values of 0.5551, 0.1478, and 0.4502 for 1-year post-restoration, 2 years post-restoration, and 3 years post-restoration (respectively), suggests that these years are not significantly different from pre-restoration (Table 4-4).

Despite skipping two weeks between data points, a Durbin-Watson Test statistic of 0.434 indicated that autocorrelation still existed between data points. The prevalence of autocorrelation within the data violates the independence assumption for linear regression analysis. Autocorrelation could result in some bias in the standard error and p-values, which could lead to a type I error. A type I error occurs when a significant difference is concluded, when there is no actual significant difference. Despite this, autocorrelation should not impact the estimate values or overall trend of the linear regression lines.

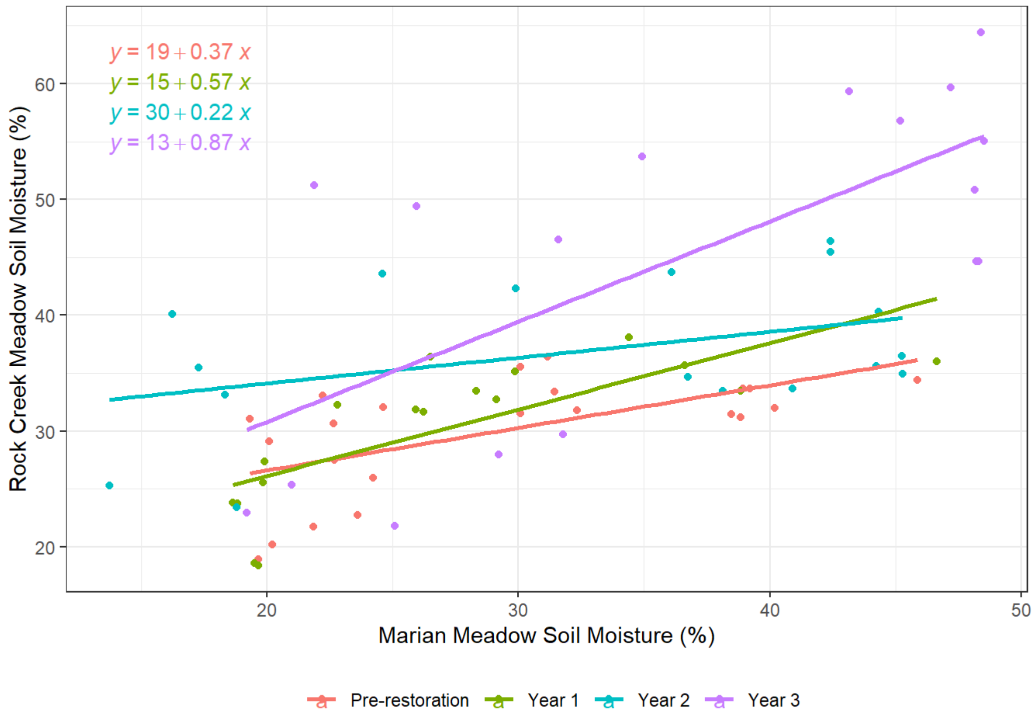


Figure 4-7. Multiple linear regression graph between RCM aggregated soil moisture content (%) and MM aggregated soil moisture content (%) by Restoration Year without any interaction terms. Soil moisture content in this model is averaged across the 10, 30, and 100 cm depths.

Table 4-3. ANOVA from a multiple linear regression model between RCM aggregated soil moisture content (%) and MM aggregated soil moisture content (%) with Restoration Year as an interaction term. Soil moisture content in this model is averaged across the 10, 30, and 100 cm depths.

	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Soil Moisture Content	3120.92	3120.92	72.97	<0.001
Restoration Year	1273.85	424.62	9.93	<0.001
MM Average Soil Moisture Content * Restoration Year	472.88	157.63	3.68	0.0163

Table 4-4. Table of coefficients from a multiple linear regression model between RCM aggregated soil moisture content (%) and MM aggregated soil moisture content (%) with Restoration Year as an interaction term. Soil moisture content in this model is averaged across the 10, 30, and 100 cm depths.

Term	Estimate	Standard Error	t Statistic	P-Value (t-test)
(Intercept)	19.247	5.244	3.670	<0.001
MM Average Soil Moisture Content	0.368	0.174	2.109	0.0388
(Pre-restoration) - Year 1	-4.593	7.742	-0.593	0.5551
(Pre-restoration) - Year 2	10.397	7.098	1.465	0.1478
(Pre-restoration) - Year 3	-5.779	7.608	-0.760	0.4502
MM Average Soil Moisture Content * (Pre-restoration) - Year 1	0.206	0.266	0.774	0.4419
MM Average Soil Moisture Content * (Pre-restoration) - Year 2	-0.144	0.223	-0.648	0.5195
MM Average Soil Moisture Content * (Pre-restoration) - Year 3	0.498	0.227	2.196	0.0316

The EMMs analysis uses the same model as the linear regression analysis to further explore the relationship between RCM aggregated soil moisture pre-restoration and each year post-restoration. The black dot in the model represents the overall mean for RCM soil moisture for each restoration year, while blue areas around it illustrate the confidence interval (Figure 4-8). Since MM aggregated soil moisture is a covariate, these mean values are based on when MM is at its overall mean aggregated soil moisture content. Additionally, when two red arrows overlap, it indicates that the difference between the two corresponding mean RCM soil moisture values is not significant (Figure 4-8).

The small p-values of 0.0383 and <0.001 for years 2 and 3 post-restoration (respectively) suggest that RCM's mean soil moisture content for these years is significantly different from RCM's pre-restoration mean soil moisture content when MM is at its average soil moisture content (Table 4-5). Based on Table 4-6, RCM's mean soil moisture content significantly increased from 30.69% pre-restoration to 36.60% 2 years post-restoration and 40.42% 3 years post-restoration. Additionally, the difference between the means of years 1 and 3 post-restoration was statistically significant (Figure 4-8; Table 4-5). Although RCM's mean soil moisture value for 1 year post-restoration is higher than the pre-restoration value (Table 4-6), the p-value greater than 0.05 (Table 4-5) suggests there is not a significant increase in mean soil moisture in year 1.

While the EMMs analysis uses the same model as the linear regression analysis, independence is not an assumption made in the EMMs analysis. Therefore, autocorrelation within this dataset will not impact the significance of the findings of the EMMs analysis.

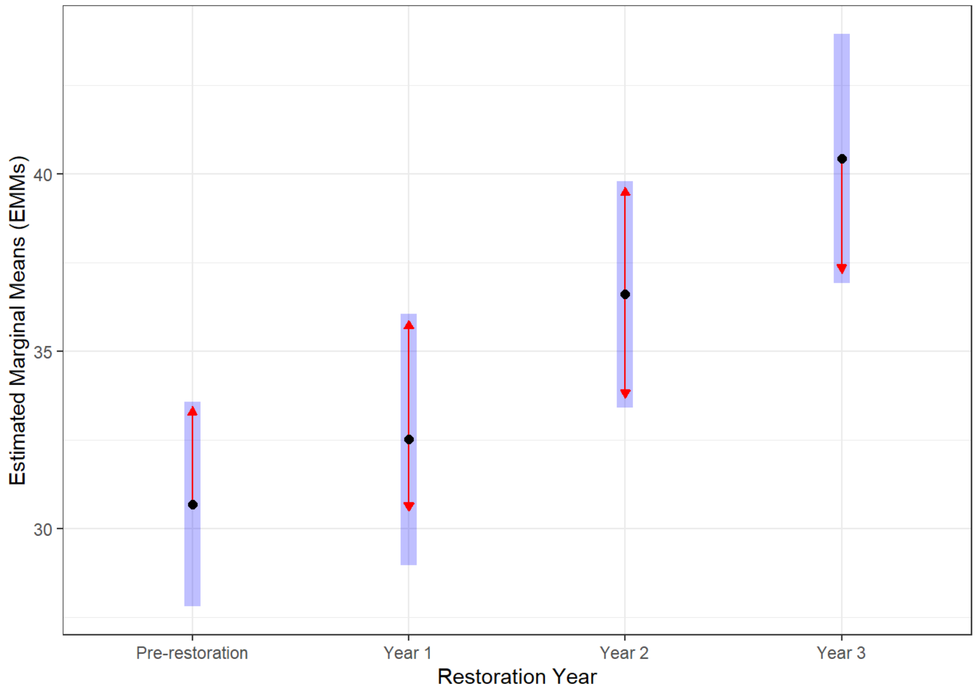


Figure 4-8. EMMs analysis of RCM aggregated soil moisture content by Restoration Year with MM aggregated soil moisture content as a covariate. Soil moisture content in this model is averaged across the 10, 30, and 100 cm depths. Black dots are the mean RCM aggregated soil moisture content, the blue boxes are the confidence interval, and the red arrows display the direction of an insignificant relationship.

Table 4-5. Table of coefficients from EMMs analysis of RCM aggregated soil moisture content by Restoration Year with MM aggregated soil moisture content as a covariate. Soil moisture content in this model is averaged across the 10, 30, and 100 cm depths.

Comparison	Estimate	Standard Error	P-Value (t-test)
(Pre-restoration) - Year 1	-1.823	2.288	0.8557
(Pre-restoration) - Year 2	-5.911	2.154	0.0383
(Pre-restoration) - Year 3	-9.733	2.274	<0.001
Year 1 - Year 2	-4.088	2.390	0.3268
Year 1 - Year 3	-7.910	2.499	0.0123
Year 2 - Year 3	-3.823	2.377	0.3811

Table 4-6. Table of EMMs values and standard error (SE) for RCM aggregated soil moisture content by Restoration Year with MM aggregated soil moisture content as a covariate. Soil moisture content in this model is averaged across the 10, 30, and 100 cm depths.

Year	EMMs (%)	Standard Error
Pre-restoration	30.69	1.443
Year 1	32.51	1.776
Year 2	36.60	1.600
Year 3	40.42	1.758

An additional analysis was conducted on RCM soil moisture content at 30 cm deep. Figure 4-9 illustrates that RCM soil moisture and MM soil moisture at 30 cm have a positive linear relationship for each year pre- and post-restoration. The p-values of 0.9394 and 0.3649 for 1 year post-restoration and 3 years post-restoration (respectively), suggests that the these years are not significantly different from pre-restoration when MM soil moisture at 30 cm is equal to 0% (Table 4-7). While the intercept of year 3 post-restoration was not statistically different from pre-restoration, based on the interaction term, the slope of year 3 post-restoration significantly increased by 0.569 from the pre-restoration slope (Table 4-7). Additionally, the p-value of 0.0198 for year 2 post-restoration indicates that there is a significant difference from pre-restoration soil moisture. Therefore, if the value of MM soil moisture content is 0%, the average value of RCM soil moisture content is 17.51% higher 2 years post-restoration than pre-restoration (Table 4-5). Similar to the regression analysis between RCM and MM aggregated soil moisture, the ANOVA table indicates the significance of including an interaction term between MM soil moisture and restoration year (Table 4-6).

Additionally, a Durbin-Watson Test statistic of 1.04 indicated that autocorrelation still existed between data points. Again, the prevalence of autocorrelation within the data violates the independence assumption for linear regression analysis, which could lead to a type I error. Despite this, autocorrelation should not impact the estimate values or overall trend of the linear regression lines.

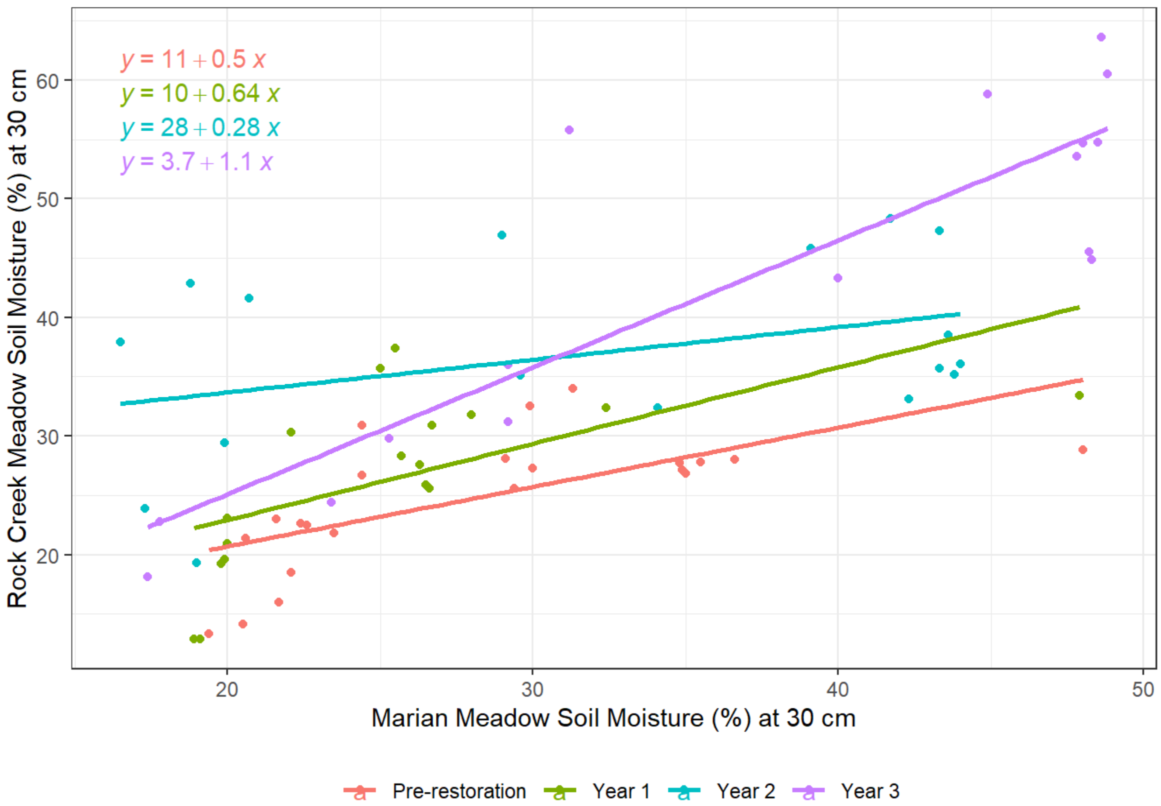


Figure 4-9. Multiple linear regression graph between RCM soil moisture content (%) and MM soil moisture content (%) for 30 cm by Restoration Year without any interaction terms.

Table 4-7. Table of coefficients from a multiple linear regression model between RCM soil moisture content (%) and MM soil moisture content (%) for 30 cm with Restoration Year as an interaction term.

Term	Estimate	Standard Error	t Statistic	P-Value (t-test)
(Intercept)	10.674	5.537	1.928	0.0583
MM Average Soil Moisture Content	0.501	0.191	2.622	0.0109
(Pre-restoration) - Year 1	-0.620	8.132	-0.076	0.9394
(Pre-restoration) - Year 2	17.510	7.323	2.391	0.0198
(Pre-restoration) - Year 3	-6.989	7.659	-0.912	0.3649
MM Average Soil Moisture Content * (Pre-restoration) - Year 1	0.142	0.297	0.479	0.6337
MM Average Soil Moisture Content * (Pre-restoration) - Year 2	-0.226	0.238	-0.951	0.3452
MM Average Soil Moisture Content * (Pre-restoration) - Year 3	0.569	0.234	2.430	0.0179

Table 4-8. ANOVA from a multiple linear regression model between RCM soil moisture content (%) and MM soil moisture content (%) for 30 cm with Restoration Year as an interaction term.

	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Soil Moisture Content	5229.1	5229.1	129.81	<0.001
Restoration Year	1628.8	542.9	13.48	<0.001
MM Average Soil Moisture Content * Restoration Year	696.6	232.2	5.76	<0.001

Figure 4-10 illustrates the EMMs analysis of RCM soil moisture content at 30 cm by restoration year with MM soil moisture content at 30 cm as a covariate. The small p-values of <0.001 for years 2 and 3 post-restoration suggests that their mean soil moisture values are also significantly different from the pre-restoration mean soil moisture value at 30 cm beyond the effect of MM mean soil moisture at 30 cm (Table 4-9). There was no difference detected between post-restoration year means (Table 4-9). Based on Table 4-10, the mean soil moisture values significantly increased from 25.93% pre-restoration to 36.56% 2 years post-restoration and 36.27% 3 years post-restoration. Again, while the mean soil moisture value for 1 year post-restoration is higher than the pre-restoration value (Table 4-10), the p-value greater than 0.05 (Table 4-9) does not communicate a significant increase in mean soil moisture.

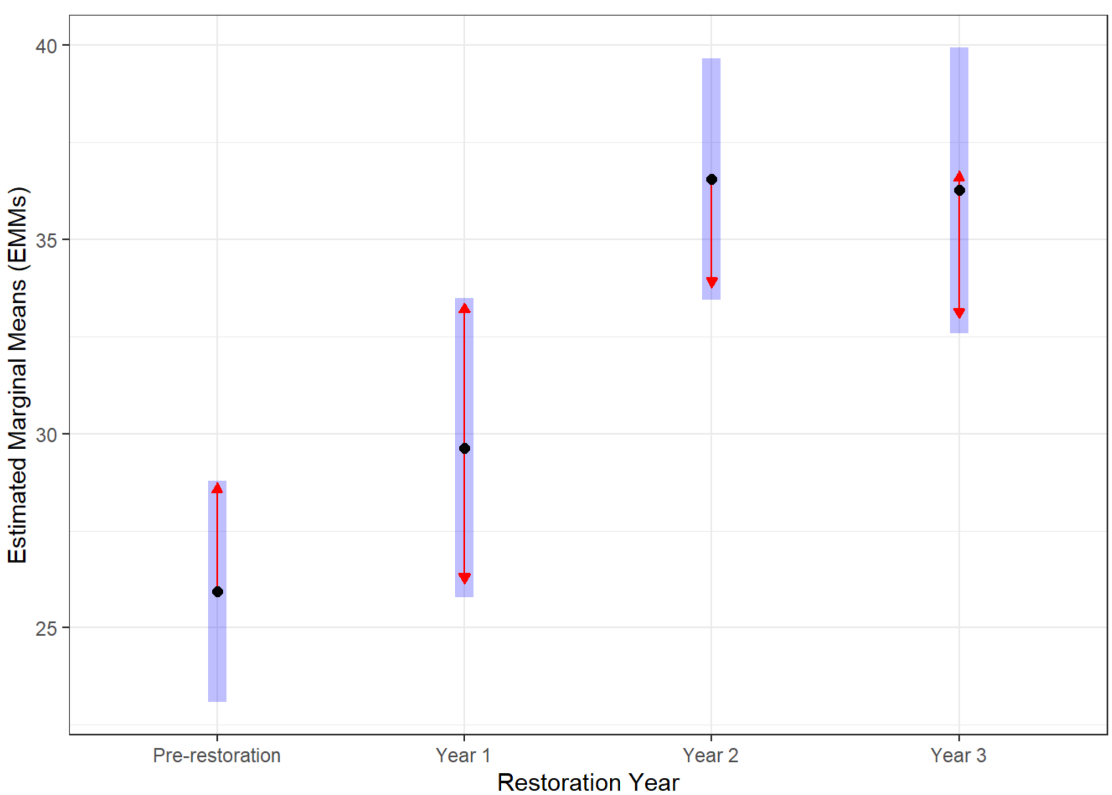


Figure 4-10. EMMs analysis of RCM soil moisture content at 30 cm by Restoration Year with MM soil moisture content at 30 cm as a covariate. Black dots are the mean RCM aggregated soil moisture content, the blue boxes are the confidence interval, and the red arrows display the direction of an insignificant relationship.

Table 4-9. Table of coefficients from EMMs analysis of RCM soil moisture content at 30 cm by Restoration Year with MM soil moisture content 30 cm as a covariate.

Comparison	Estimate	Standard Error	P-Value (t-test)
(Pre-restoration) - Year 1	-3.707	2.398	0.4166
(Pre-restoration) - Year 2	-10.630	2.112	<0.001
(Pre-restoration) - Year 3	-10.338	2.327	<0.001
Year 1 - Year 2	-6.923	2.479	0.0339
Year 1 - Year 3	-6.631	2.664	0.0713
Year 2 - Year 3	0.292	2.410	0.9994

Table 4-10. Table of EMMs values and SE for RCM soil moisture content at 30 cm by Restoration Year with MM soil moisture content at 30 cm as a covariate.

Year	EMMs (%)	Standard Error
Pre-restoration	25.93	1.426
Year 1	29.63	1.928
Year 2	36.56	1.558
Year 3	36.27	1.839

4.3.2. Depth to Groundwater Statistical Analysis

RCM depth to groundwater and MM depth to groundwater have a positive linear relationship by restoration year (Figure 4-11), meaning as MM's depth to groundwater increases, so does RCM's. In other terms, as MM's water level decreases, so does RCM's, which aligns with the trends seen in Figures 4-4 and 4-5. The p-values of 0.1798, 0.2105, and 0.9585 for 1-year post-restoration, 2 years post-restoration, and 3 years post-restoration (respectively), suggests that these years are not significantly different from pre-restoration when MM depth to groundwater equals 0 m (Table 4-11). Additionally, the ANOVA table illustrates the significance of MM depth to groundwater and restoration year to RCM depth to groundwater individually, but not together as an interaction term (Table 4-12). Despite the lack of statistical difference, the interaction term is important to analyze because RCM depth to groundwater changed each year post-restoration whether it was significantly different to pre-restoration values or not. Similar to the regression analysis between RCM and MM aggregated soil moisture, a Durbin-Watson Test indicated that autocorrelation still existed between data points.

Additionally, a Durbin-Watson Test statistic of 0.523 indicated that autocorrelation still existed between data points. Again, the prevalence of autocorrelation within the data violates the independence assumption for linear regression analysis, which could lead to a type I error. Despite this, autocorrelation should not impact the estimate values or overall trend of the linear regression lines.

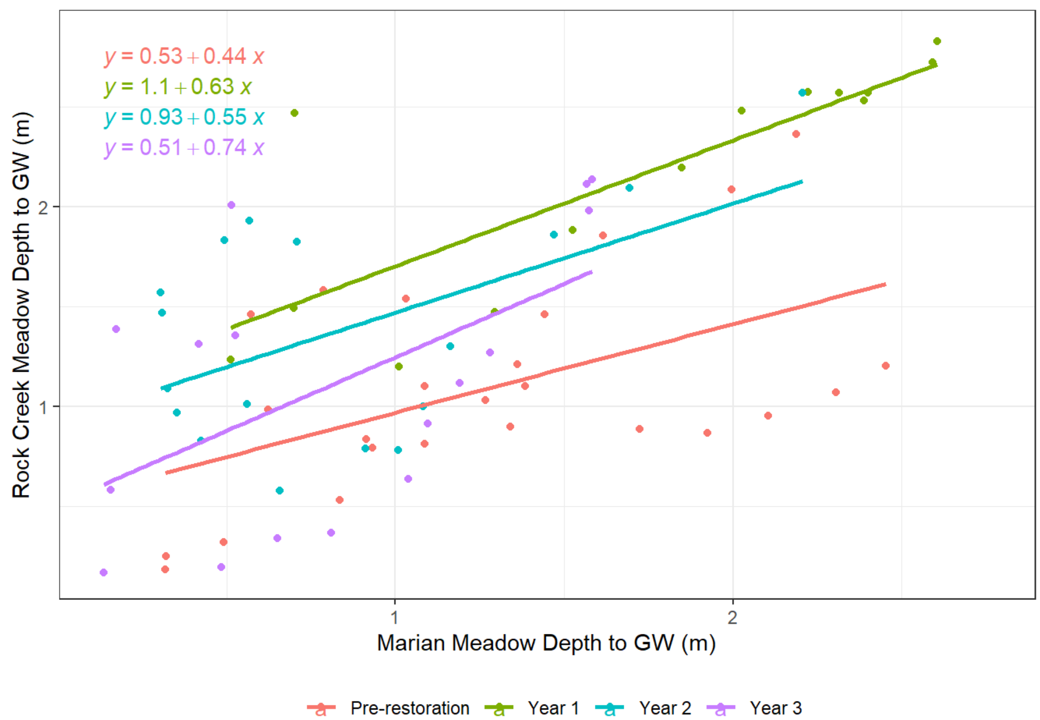


Figure 4-11. Multiple linear regression graph between RCM depth to groundwater (GW) (m) and MM depth to groundwater (m) by Restoration Year without any interaction terms.

Table 4-11. Table of coefficients from a multiple linear regression model between RCM depth to groundwater (m) and MM depth to groundwater (m) with Restoration Year as an interaction term.

Terms	Estimate	Standard Error	t Statistic	P-Value (t-test)
(Intercept)	0.526	0.225	2.338	0.0225
MM Average Depth to Groundwater (m)	0.443	0.158	2.802	0.0067
(Pre-restoration) - Year 1	0.546	0.403	1.356	0.1798
(Pre-restoration) - Year 2	0.399	0.315	1.265	0.2105
(Pre-restoration) - Year 3	-0.017	0.325	-0.052	0.9585
MM Average Depth to Groundwater * (Pre-restoration) - Year 1	0.187	0.238	0.785	0.4351
MM Average Depth to Groundwater * (Pre-restoration) - Year 2	0.103	0.273	0.376	0.7085
MM Average Depth to Groundwater * (Pre-restoration) - Year 3	0.294	0.290	1.014	0.3146

Table 4-12. ANOVA from a multiple linear regression model RCM depth to groundwater (m) and MM depth to groundwater (m) with Restoration Year as an interaction term.

	Sum Sq	**Mean Sq**	**F value**	**P-value (F-test)**
MM Average Depth to Groundwater (m)	13.057	13.057	55.244	<0.001
Restoration Year	6.682	2.227	9.424	<0.001
MM Average Depth to Groundwater (m) * Restoration Year	0.292	0.097	0.412	0.745

As illustrated by Figure 4-12, an EMMs analysis was conducted using the same model. The small p-value of <0.001 and 0.0173 for 1 year post-restoration and 2 years post-restoration (respectively), suggests that these mean depth to groundwater values are significantly different from the pre-restoration mean depth to groundwater value (Table 4-13). Based on Table 4-14, the mean depth to groundwater value significantly increased from 1.04 m pre-restoration to 1.81 m year 1 post-restoration and 1.56 m year 2 post-restoration, beyond the effect of MM mean depth to groundwater. Meaning that the amount of groundwater available in RCM year 1 and 2 post-restoration decreased significantly from pre-restoration. The p-value for year 3 post-restoration does not communicate a significant change in depth to groundwater from pre-restoration conditions. Year 3 has a value of 1.37 m, which is greater than the mean pre-restoration depth to groundwater value.

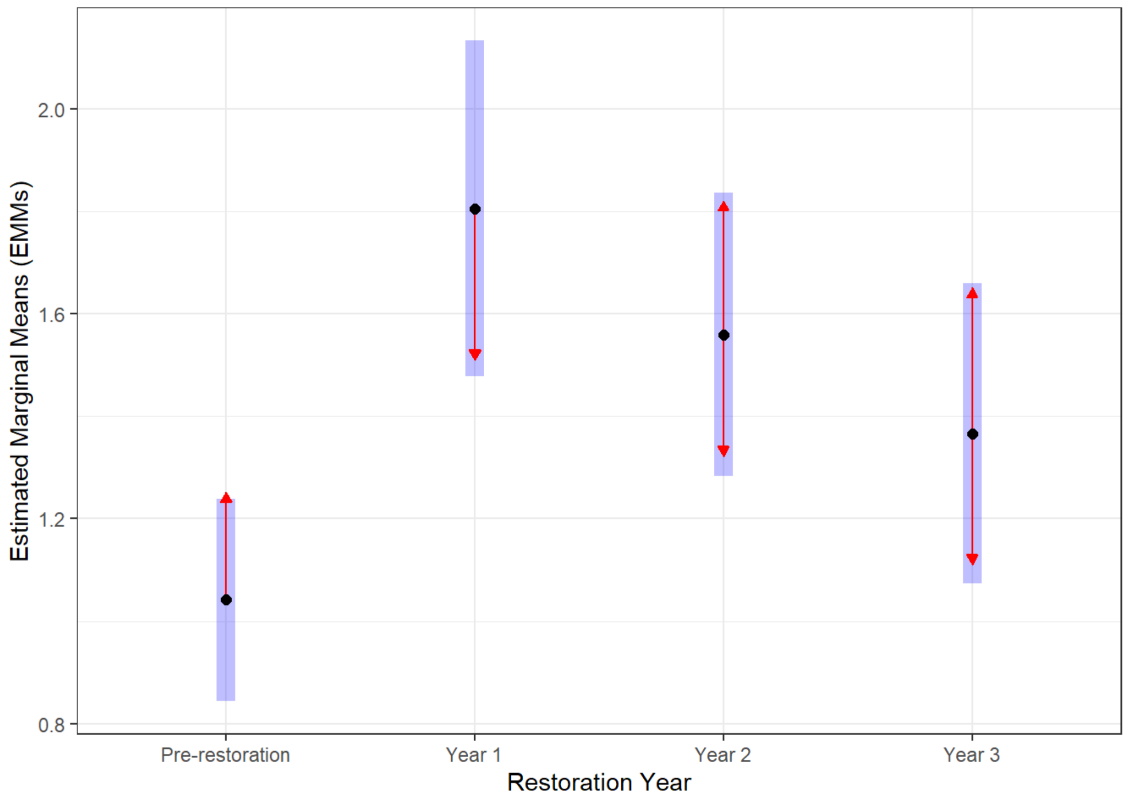


Figure 4-12. EMMs analysis of RCM depth to groundwater (m) by Restoration Year with MM depth to groundwater (m) as a covariate. Black dots are the mean RCM depth to groundwater (m), the blue boxes are the confidence interval, and the red arrows display the direction of an insignificant relationship.

Table 4-13. Table of coefficients from EMMs analysis of RCM depth to groundwater (m) by Restoration Year with MM depth to groundwater (m) as a covariate.

Comparison	Estimate	Standard Error	P-Value (t-test)
(Pre-restoration) - Year 1	-0.764	0.192	<0.001
(Pre-restoration) - Year 2	-0.518	0.170	0.0173
(Pre-restoration) - Year 3	-0.325	0.177	0.2664
Year 1 - Year 2	0.246	0.215	0.6637
Year 1 - Year 3	0.439	0.220	0.2009
Year 2 - Year 3	0.193	0.202	0.7731

Table 4-14. Table of EMMs values and SE for RCM depth to groundwater (m) by Restoration Year with MM depth to groundwater (m) as a covariate.

Year	EMMs (m)	Standard Error
Pre-restoration	1.04	0.099
Year 1	1.81	0.164
Year 2	1.56	0.138
Year 3	1.37	0.147

An additional analysis was conducted on RCM West depth to groundwater due to the data gaps in RCM East. Figure 4-13 illustrates that RCM depth to groundwater and MM depth to groundwater have a positive linear relationship by each year pre- and post-restoration. The p-values of 0.0549, 0.1722, and 0.6222 for 1 year post-restoration, 2 years post-restoration, and 3 years post-restoration (respectively), suggests that the these years are not significantly different from pre-restoration when MM depth to groundwater equals 0 m (Table 4-15). Like with depth to groundwater across RCM, the ANOVA table illustrates the significance of MM depth to groundwater and restoration year to RCM depth to groundwater individually, but not together as an interaction term (Table 4-16). Like the previous depth to groundwater analysis, a Durbin-Watson Test statistic of 0.523 indicated that autocorrelation still existed between data points. Despite this, autocorrelation should not impact the estimate values or overall trend of the linear regression lines.

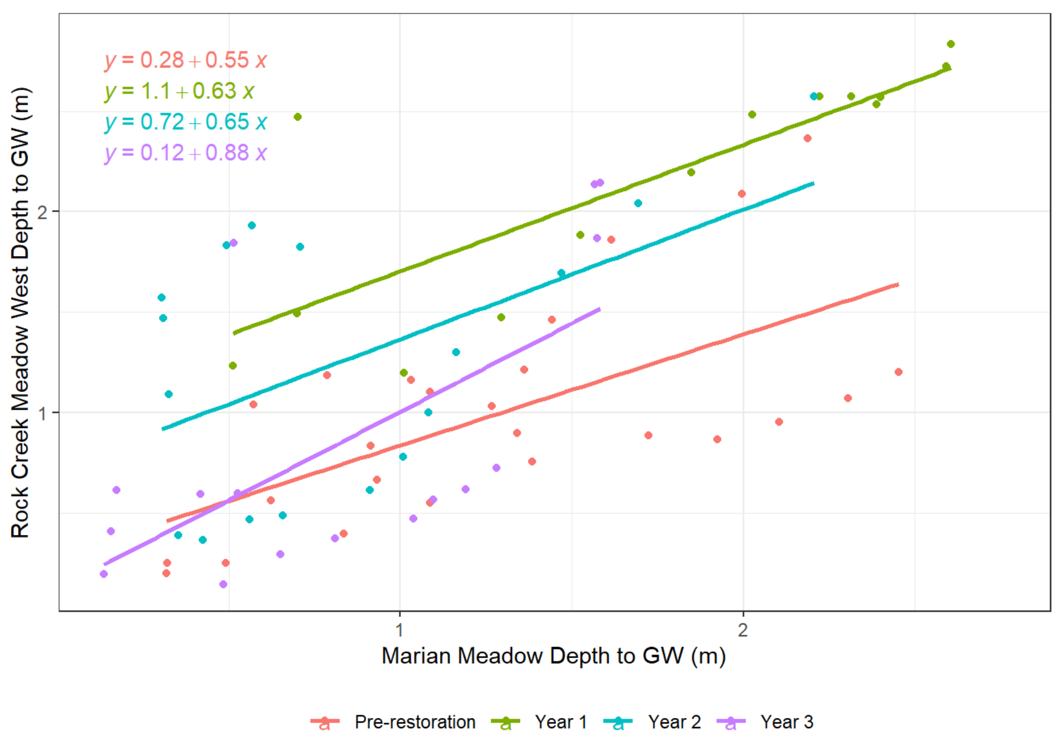


Figure 4-13. Multiple linear regression graph between RCM West depth to groundwater (m) and MM depth to groundwater (m) by Restoration Year without any interaction terms.

Table 4-15. Table of coefficients from a multiple linear regression model between RCM West depth to groundwater (m) and MM depth to groundwater (m) with Restoration Year as an interaction term.

Terms	Estimate	Standard Error	t Statistic	P-Value (t-test)
(Intercept)	0.284	0.226	1.259	0.2124
MM Average Depth to Groundwater (m)	0.553	0.158	3.490	<0.001
(Pre-restoration) - Year 1	0.789	0.403	1.956	0.0549
(Pre-restoration) - Year 2	0.436	0.316	1.381	0.1722
(Pre-restoration) - Year 3	-0.161	0.326	-0.495	0.6222
MM Average Depth to Groundwater * (Pre-restoration) - Year 1	0.078	0.239	0.325	0.7460
MM Average Depth to Groundwater * (Pre-restoration) - Year 2	0.093	0.274	0.339	0.7360
MM Average Depth to Groundwater * (Pre-restoration) - Year 3	0.328	0.291	1.129	0.2632

Table 4-16. ANOVA from a multiple linear regression model RCM West depth to groundwater (m) and MM depth to groundwater (m) with Restoration Year as an interaction term.

	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Depth to Groundwater (m)	18.327	18.327	77.329	<0.001
Restoration Year	8.088	2.696	11.375	<0.001
MM Average Depth to Groundwater (m) * Restoration Year	0.304	0.101	0.427	0.734

As illustrated by Figure 4-14, an EMMs analysis was conducted using the same model. The small p-values of <0.001 and 0.0115 for 1-year post-restoration and 2-years post-restoration (respectively) suggests that the mean depth to groundwater value is significantly different from the pre-restoration mean depth to groundwater value at MM's mean depth to groundwater value (Table 4-17). Based on Table 4-18, the mean depth to groundwater value significantly increased from 0.93 m pre-restoration to 1.81 m 1-year post-restoration and 1.47 m 2 years post-restoration. Meaning that the amount of groundwater available in RCM west 1 to 2 years post-restoration decreased significantly from pre-restoration. The p-value for 3 years post-restoration did not communicate a significant change in depth to groundwater from pre-restoration conditions. Year 3 has a mean depth to groundwater value of 1.15 m which is greater than the mean pre-restoration depth to groundwater value.

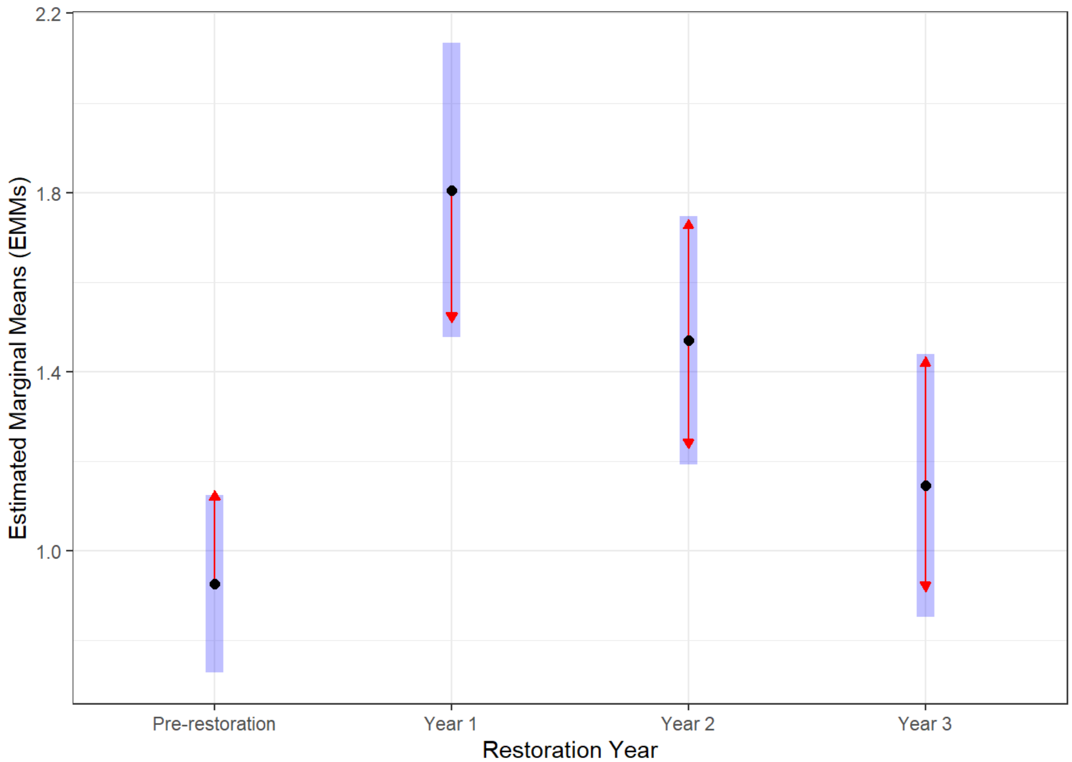


Figure 4-14. EMMs analysis of RCM West depth to groundwater (m) by Restoration Year with MM depth to groundwater (m) as a covariate. Black dots are the mean RCM West depth to groundwater (m), the blue boxes are the confidence interval, and the red arrows display the direction of an insignificant relationship.

Table 4-17. Table of coefficients from EMMs analysis of RCM West depth to groundwater (m) by Restoration Year with MM depth to groundwater (m) as a covariate.

Comparison	Estimate	Standard Error	P-Value (t-test)
(Pre-restoration) - Year 1	-0.879	0.192	<0.001
(Pre-restoration) - Year 2	-0.544	0.170	0.0115
(Pre-restoration) - Year 3	-0.220	0.177	0.6033
Year 1 - Year 2	0.335	0.215	0.4087
Year 1 - Year 3	0.659	0.220	0.0202
Year 2 - Year 3	0.324	0.202	0.3841

Table 4-18. Table of EMMs values and SE for RCM West depth to groundwater (m) by Restoration Year with MM depth to groundwater (m) as a covariate.

Year	EMMs (m)	Standard Error
Pre-restoration	0.93	0.099
Year 1	1.81	0.164
Year 2	1.47	0.139
Year 3	1.15	0.147

4.3.3 Dixie Fire Analysis

For this analysis, “Fire Year” is a categorical variable that groups the data as pre-fire or post-fire values. The purpose of this analysis is to determine if the 2021 Dixie Fire accounted for a significant amount of variability in RCM data, with MM values still being treated as the baseline conditions for what a restored meadow’s soil moisture levels should be. The p-value of Fire Year term in the ANOVA tables below indicates the term explains a significant amount of the variance in RCM values.

The p-value of <0.001 for Fire Year in Table 4-19 indicates that the Dixie Fire accounted for a significant amount of the variance in RCM aggregated soil moisture content. Similarly, the p-value of <0.001 for Fire Year in Table 4-20 indicates that the Dixie Fire accounted for a significant amount of the variance in RCM soil moisture content at 30 cm. On the other hand, the p-value of 0.1987 for Fire Year in Table 4-21 indicates that the Dixie Fire did not account for a significant amount of the variance in RCM depth to groundwater (m). Finally, the p-value of 0.4376 for Fire Year in Table 4-22 indicates that the Dixie Fire did not account for a significant amount of the variance in RCM West depth to groundwater (m)

Table 4-19. ANOVA from a multiple linear regression model between RCM aggregated soil moisture content and MM aggregated soil moisture content with Fire Year as an interaction term.

Terms	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Soil Moisture Content	3,120.92	3,120.92	61.79	<0.001
Fire Year	1,030.56	1,030.56	20.40	<0.001
MM Average Soil Moisture Content * Restoration Year	10.93	10.93	0.22	0.6433

Table 4-20. ANOVA from a multiple linear regression model between RCM soil moisture content at 30 cm and MM soil moisture content at 30 cm with Fire Year as an interaction term.

Terms	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Soil Moisture Content	5,229.07	5,229.07	107.20	<0.001
Fire Year	1,553.28	1,553.28	31.84	<0.001
MM Average Soil Moisture Content * Restoration Year	33.17	33.17	0.68	0.413

Table 4-21. ANOVA from a multiple linear regression model between RCM depth to groundwater (m) and MM depth to groundwater (m) with Fire Year as an interaction term.

Terms	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Depth to Groundwater (m)	13.057	13.057	41.218	<0.001
Fire Year	0.534	0.534	1.684	0.1987
MM Average Depth to Groundwater (m) * Restoration Year	0.026	0.026	0.082	0.7754

Table 4-22. ANOVA from a multiple linear regression model between RCM West depth to groundwater (m) and MM depth to groundwater (m) with Fire Year as an interaction term.

Terms	Sum Sq	Mean Sq	F value	P-value (F-test)
MM Average Depth to Groundwater (m)	18.327	18.327	53.377	<0.001
Fire Year	0.209	0.209	0.610	0.4376
MM Average Depth to Groundwater (m) * Restoration Year	0.003	0.003	0.008	0.9274

5.0 Discussion

The results of the EMMs analysis indicate that soil moisture at RCM significantly increased from pre-restoration values starting 2 years after restoration. This indicates that there is higher water availability in the root zone of herbaceous plants at RCM post-restoration. The most significant increases in soil moisture during year 3 post-restoration (from pre-restoration values) are between April and September. This indicates that water in the vadose zone is much more available during the warm growing months despite evapotranspiration occurring. This could also indicate that the removed conifers were using and transpiring more water than the plant species that have re-established in RCM. This increased water availability in the vadose zone could accelerate a shift in plant communities from woody and grassland species to more wetland associated species. Since RCM's West side is a subsurface low gradient meadow, mesic (moist) and hydric (wet) herbaceous perennial plant communities are expected to repopulate the area (Viers et al., 2013). On the other hand, the dry nature of RCM's East side will likely continue as grassland community similar to that of an upland vegetation community (Viers et al., 2013).

While there was a significant decrease in groundwater availability in the first year following restoration, the reduced amount of water available in year 2 and 3 post-restoration from pre-restoration values was not significantly different. However, this could be the result of a drought occurring in WY 2020 and 2021 (Table 3-2). Additionally, the water level continued to improve between year 2 and year 3 which could indicate that water level may require more years to return or surpass pre-restoration conditions. Figure 4-6 and Table 4-2 shows that water levels were higher in year 3 post-restoration than RCM pre-restoration conditions between April and September which is the dry growing season. This was further illustrated by figures 4-11 and 4-13 which indicate that some data points from 3 years post-restoration have lower depth to groundwater values than pre-restoration data points. This means that despite the overall decrease in groundwater at the meadow post-restoration, more groundwater is being stored in the summer when plants are relying on water in the subsurface to grow.

Due to the unexpected nature and timing of the 2021 Dixie Fire, it is not possible to separate any specific effects the fire may have had on soil moisture and groundwater availability beyond the effects of RCM's restoration. The presence of a significant relationship between the fire year variable, and RCM aggregated soil moisture and RCM soil moisture at 30 cm variables, indicates the Dixie Fire may have had an impact on the soil moisture values at RCM. Despite this, RCM soil moisture values increased in all three years' post-restoration. On the other hand, the lack of a significant relationship between the fire year variable, and RCM depth to groundwater and RCM West depth to groundwater variables, indicates that the fire should have little impact on the overall analysis. Additionally, because most of RCM and MM were burned at a moderate to high burn severity, the fire should not have impacted MM's ability to be a control for RCM.

5.1 Limitations

While MM was a reasonable control for RCM due to relative proximity and similar vegetation communities, a better control would have been at the same elevation as RCM. The 500 ft elevation difference between these two meadows meant substantial differences in temperature, snowfall, and precipitation. Since snowmelt is the primary source for surface water and groundwater recharge in this region, any differences in timing could have had an impact on soil moisture and groundwater level timing. Unfortunately, the climate instruments installed on the meadows could not effectively differentiate between rain and snow precipitation, so these could not be used to account for the precipitation type differences. Additionally, the lack of replication within this study on other meadows besides RCM indicates that any interpretation of these results cannot be applied to the broader population of montane meadows in the Sierra Nevada and Southern Cascade Mountains ranges.

Furthermore, only one year of pre-restoration data may not be fully representative of soil moisture and depth to groundwater conditions. This is especially evident in Table 3-2, which shows that the 2019 WY saw significantly more rainfall than WYs 2020, 2021, and 2022. This could mean that the pre-restoration data is showing RCM as being wetter than it typically would have been. Likewise, it is possible that an additional year of post-restoration data collection would have reflected an increase in groundwater, as WY 2023 precipitation levels were getting closer to that of WY 2019.

The use of time series data in the statistical analysis resulted in serial autocorrelation within the data. Despite steps taken to reduce autocorrelation, it still was not removed, which violated the independence assumption of linear regression analysis. Therefore, additional time series statistical analysis should be done on this data in the future. This could include the use of autoregressive (AR) and moving-average (MA) terms to remove autocorrelation from the data points for a more accurate linear regression analyses. Forecasting could also be done on the depth to groundwater data to predict if an extra year of post-restoration data collection would have resulted in lower depth to groundwater values. Finally, an analysis could be done on the lag between RCM and MM data points, which could be impacting the accuracy of the multiple linear regression analyses.

5.2 Management Implications

Following the RCM restoration by removal of lodgepole pine, regular forest management needs to occur due to lodgepole pines' association with intermediately wet areas (Boisramé et al., 2018). It is ideal that tree removal happens when the trees are younger due to their fast-growing nature (Kremer et al., 2014). The perimeter of the meadow will need maintenance as conifer re-invasion tends to occur at higher rates around the forested edge (Halpern & Antos, 2021; Helms, 1987). Notably, invasion of lodgepole pines into meadows tends to occur during drier conditions when there is higher seed fall and less flooding to wash away the seeds (Helms, 1987). Additional improvements could also be made to the Rock Creek stream channel to re-connect the channel to the meadow as its floodplain. With a more intact floodplain, more runoff can be stored in RCM during the winter and spring (Plumas National Forest, 2010).

6.0 Conclusions

The removal of conifers from Rock Creek Meadow (RCM) for restoration had mixed impacts on the hydrologic response of the meadow. The first year after restoration there was an insignificant increase in soil moisture content and a significant decrease in groundwater availability. However, years 2 and 3 post-restoration saw a significant increase in soil moisture content at RCM from 30.69% to 36.60% and 40.42% (respectively). This indicates long-term improvement in soil moisture availability at RCM post-restoration. Groundwater, on the other hand, had not significantly increased beyond the pre-restoration values by year 3 post-restoration. While there was not a significant increase based on the statistical models, the months of April through September in year 3 post-restoration had higher water levels than pre-restoration values. Due to the gradual recovery in water level each year post-restoration, it is unclear whether an extra year of post-restoration data collection would have seen an increase in groundwater level.

Despite the mixed results of the depth to groundwater analysis, the improved soil moisture content and lower depth to groundwater values in the summer of year 3 post-restoration suggest that the environmental benefit outweighs the impacts of the tree removal. Further analysis of the vegetative response to restoration could give more insight into the environmental benefits of meadow restoration. To enhance the ecological function of RCM, periodic removal of conifers may be necessary as lodgepole pines tend to invade areas of higher soil moisture (Vankat 1982; Boisramé et al., 2018). Additionally, improved connection between RCM and Rock Creek could improve subsurface water storage in the meadow (Plumas National Forest, 2010).

WORKS CITED

Allen, R., Pereira, L., Raes, D., & Smith, M. (1998). Crop evapotranspiration guidelines for computing crop requirements. FAO Irrig. Drain. Report modeling and application. *Journal of Hydrology*, *285*.

American Rivers. (n.d.). *Mountain Meadow Restoration in California*. American Rivers. https://www.americanrivers.org/threats-solutions/restoring-damaged-rivers/mountains-meadows/

Blackburn, D. A., Oliphant, A. J., & Davis, J. D. (2021). Carbon and Water Exchanges in a Mountain Meadow Ecosystem, Sierra Nevada, California. *Wetlands*, *41*(3). https://doi.org/10.1007/s13157-021-01437-2

Boisramé, G., Thompson, S., & Stephens, S. (2018). Hydrologic responses to restored wildfire regimes revealed by soil moisture-vegetation relationships. *Advances in Water Resources*, *112*, 124–146. https://doi.org/10.1016/j.advwatres.2017.12.009

Burchsted, D., Daniels, M., Thorson, R., & Vokoun, J. (2010). The River Discontinuum: Applying Beaver Modifications to Baseline Conditions for Restoration of Forested Headwaters. *BioScience*, *60*(11), 908–922. https://doi.org/10.1525/bio.2010.60.11.7

Cal Fire. (2022). *Dixie Fire*. The Department of Forestry and Fire Protection. https://www.fire.ca.gov/incidents/2021/7/13/dixie-fire/

CDOC. (2010). Generalized Rock Types. California Department of Conservation: Open Data and Maps. https://maps-cadoc.opendata.arcgis.com/datasets/cadoc::generalized-rock-types-2/about

Clary, W. P., & Leininger, W. C. (2000). Stubble Height as a Tool for Management of Riparian Areas. *Journal of Range Management*, *53*(6), 562–573. https://doi.org/10.2307/4003148

DWR. (n.d.). CHESTER. California Data Exchange Center; California Department of Water Resources (DWR). https://cdec.water.ca.gov/dynamicapp/staMeta?station_id=CHS

Fie, N. (2018). *Hydrologic response from conifer removal and upslope harvest from an encroached montane meadow*. Master's thesis, California Polytechnic State University, San Luis Obispo. https://doi.org/10.15368/theses.2018.52

Freitas, M. R., Roche, L. M., Weixelman, D., & Tate, K. W. (2014). Montane meadow plant community response to livestock grazing. *Environmental Management*, *54*(2), 301–308. https://doi.org/10.1007/s00267-014-0294-y

Grace-Martin, K. (2021). *Why report estimated marginal means?* The Analysis Factor. https://www.theanalysisfactor.com/why-report-estimated-marginal-means-in-spss-glm/

Halpern, C. B., & Antos, J. A. (2021). Rates, patterns, and drivers of tree reinvasion 15 years after large-scale meadow-restoration treatments. *Restoration Ecology*, *29*(5). https://doi.org/10.1111/rec.13377

Halpern, C. B., Haugo, R. D., Antos, J. A., Kaas, S. S., & Kilanowski, A. L. (2012). Grassland restoration with and without fire: evidence from a tree-removal experiment. *Ecological Applications*, *22*(2), 425–441. https://doi.org/10.1890/11-1061.1

Hamilton, B. T., Roeder, B. L., & Horner, M. A. (2019). Effects of sagebrush restoration and conifer encroachment on small mammal diversity in sagebrush ecosystem. *Rangeland Ecology & Management*, *72*(1), 13–22. https://doi.org/10.1016/j.rama.2018.08.004

Hammersmark, C. T., Rains, M. C., & Mount, J. F. (2008). Quantifying the hydrological effects of stream restoration in a montane meadow, northern California, USA. *River Research and Applications*, *24*(6), 735–753. https://doi.org/10.1002/rra.1077

Helms, J. A. (1987). Invasion of Pinus Contorta Var. Murrayana (pinaceae) into Mountain Meadows at Yosemite National Park, California. *Madroño*, *34*(2), 91–97. https://www.jstor.org/stable/41424621

Kilgore, B. M., & Taylor, D. (1979). Fire history of a sequoia-mixed conifer forest. *Ecology*, *60*(1), 129–142. https://doi.org/10.2307/1936475

Kremer, N. J., Halpern, C. B., & Antos, J. A. (2014). Conifer reinvasion of montane meadows following experimental tree removal and prescribed burning. *Forest Ecology and Management*, *319*, 128–137. https://doi.org/10.1016/j.foreco.2014.02.002

Lord, M. L., Jewett, D. G., Miller, J. R., Germanoski, D., & Chambers, J. C. (2011). *Chapter 4: Hydrologic processes influencing meadow ecosystems* (pp. 44–67). Gen. Tech. Rep. RMRS-GTR-258. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-258

Magee, T. (1885). *Ascent of Mt. Whitney.* In: Tulare County Scrapbook.

Marks, S. (2021). *Estimating and modeling transpiration of a mountain meadow encroached by conifers using sap flow measurements*. Master's thesis, California Polytechnic State University, San Luis Obispo. https://doi.org/10.15368/theses.2021.167

McIlroy, S. K., & Allen-Diaz, B. H. (2012). Plant community distribution along water table and grazing gradients in montane meadows of the Sierra Nevada Range (California, USA). *Wetlands Ecology and Management*, *20*(4), 287–296. https://doi.org/10.1007/s11273-012-9253-7

Miller, C., & Urban, D. L. (1999). A model of surface fire, climate and forest pattern in the Sierra Nevada, California. *Ecological Modelling*, *114*(2-3), 113–135. https://doi.org/10.1016/s0304-3800(98)00119-7

Mueller, K. E., Tilman, D., Fornara, D. A., & Hobbie, S. E. (2013). Root depth distribution and the diversity—productivity relationship in a long-term grassland experiment. *Ecology*, *94*(4), 787–793. https://www.jstor.org/stable/23436291

NOAA. (2023). NOAA NCEI U.S. Climate Normals Quick Access. National Oceanic and Atmospheric Administration. https://www.ncei.noaa.gov/access/us-climate-normals/#dataset=normals-monthly&timeframe=15

Onset Computer Corporation. (n.d.). HOBOware® Pro Barometric Compensation Assistant User's Guide. https://www.onsetcomp.com/sites/default/files/resources-documents/10572-J%20Barometric%20Compensation%20Assistant%20User%27s%20Guide.pdf

Plumas National Forest. (2010). *The pond-and-plug treatment for stream and meadow restoration: resource effects and design considerations*. A Briefing Paper for Plumas National Forest Resource Specialists and Managers . https://olis.oregonlegislature.gov/liz/2020R1/Downloads/CommitteeMeetingDocument/213892

Pope, K. L., Montoya, D. S., Brownlee, J. N., Dierks, J., & Lisle, T. E. (2015). Habitat Conditions of Montane Meadows associated with Restored and Unrestored Stream Channels of California. *Ecological Restoration*, *33*(1), 61–73. https://doi.org/10.3368/er.33.1.61

Ratliff, R. D. (1985). *Meadows in the Sierra Nevada of California: state of knowledge*. Gen. Tech. Rep. PSW-GTR-84. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture. https://doi.org/10.2737/psw-gtr-84

Reed, C. C., Merrill, A. G., Drew, W. M., Christman, B., Hutchinson, R. A., Keszey, L., Odell, M., Swanson, S., Verburg, P. S. J., Wilcox, J., Hart, S. C., & Sullivan, B. W. (2020). Montane Meadows: A Soil Carbon Sink or Source? *Ecosystems*, *24*(5), 1125–1141. https://doi.org/10.1007/s10021-020-00572-x

Sanford, T. (2016). *The water table and soil moisture response following the removal of conifers from an encroached meadow*. Master's thesis, California Polytechnic State University, San Luis Obispo. https://doi.org/10.15368/theses.2016.154

Soil Survey Staff. (n.d.). Web Soil Survey. United States Department of Agriculture, Natural Resources Conservation Service. https://websoilsurvey.nrcs.usda.gov/app/HomePage.htm?TARGET_APP=Web_Soil_Survey_application_ckrqnvj2avdulnxckazvm4dz.

Sterner, S., Aslan, C., Best, R., & Chaudhry, T. (2022). Forest management effects on vegetation regeneration after a high severity wildfire: A case study in the southern Cascade range. *Forest Ecology and Management*, *520*(0378-1127), 120394. https://doi.org/10.1016/j.foreco.2022.120394

Surfleet, C. (2023). *Final report to the California State Board of Forestry and Fire Protection Monitoring Effectiveness Committee EMC -2018-003 Alternative Meadow Restoration*. Board of Forestry and Fire Protection. https://bof.fire.ca.gov/media/p4udw32d/emc-2018-003-alternative-meadow-restoration-final-report_ada.pdf

Surfleet, C., Fie, N., & Jasbinsek, J. (2020). Hydrologic response of a montane meadow from conifer removal and upslope forest thinning. *Water*, *12*(1), 293. https://doi.org/10.3390/w12010293

Surfleet, C., Sanford, T., Van Oosbree, G., & Jasbinsek, J. (2019). Hydrologic response of meadow restoration the first year following removal of encroached conifers. *Water*, *11*(3), 428. https://doi.org/10.3390/w11030428

Tennant, L. B., Eagles-Smith, C. A., Willacker, J. J., & Johnson, M. (2021). Fish habitat use and food web structure following pond-and-plug restoration of a montane meadow in the Sierra Nevada, California. *Northwestern Naturalist*, *102*(1). https://doi.org/10.1898/1051-1733-102.1.30

USFS. (2008). *Carson Range multi-jurisdictional fuel reduction and wildfire prevention strategy* (pp. 37–40). USDA Forest Service. https://scholarsbank.uoregon.edu/xmlui/bitstream/handle/1794/17741/NV_510_CarsonRange_2008.pdf?sequence=1&isAllowed=y

Van Oosbree, G. (2015). *Hydrologic response to conifer removal from an encroached mountain meadow*. Master's thesis, California Polytechnic State University, San Luis Obispo. https://doi.org/10.15368/theses.2015.60

Vankat, J. L. (1982). A gradient perspective on the vegetation of Sequoia National Park, California. *Madroño*, *29*(3), 200–214. https://www.jstor.org/stable/41424371

Vankat, J. L., & Major, J. (1978). Vegetation changes in Sequoia National Park, California. *Journal of Biogeography*, *5*(4), 377–402. https://doi.org/10.2307/3038030

Viers, J., Purdy, S., Peek, R., Fryjoff-Hung, A., Santos, N., Katz, J., Emmons, J., Dolan, D., & Yarnell, S. (2013). *Montane meadows in the Sierra Nevada: changing hydroclimatic conditions and concepts for vulnerability assessment*. Center for Watershed Sciences Technical Report prepared for National Fish & Wildlife Foundation and Resources Legacy Fund. http://climate.calcommons.org/sites/default/files/CWSMeadowsVulnerabilityWhitePaper_2013-1-1_FinalReport_0.pdf

Warner, T. E. (1980). *Fire history in the yellow pine forest of Kings Canyon National Park* (pp. 89–92). In: Proceedings of the Fire History Workshop, Tucson, AZ, October 20–24, 1980. USDA For. Serv. Gen. Tech. Rep. RM-81. http://npshistory.com/publications/seki/fire/gtr-rm-81.pdf

Weixelman, D. A., Hill, B., Cooper, D. J., Berlow, E. L., Viers, J. H., Purdy, S. E., Merrill, A. G., & Gross, S. E. (2011). *A field key to meadow hydrogeomorphic types for the Sierra Nevada and Southern Cascade Ranges in California*. Gen. Tech. Rep. R5-TP-034. Vallejo, CA. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region. http://dx.doi.org/10.13140/RG.2.1.1574.4088

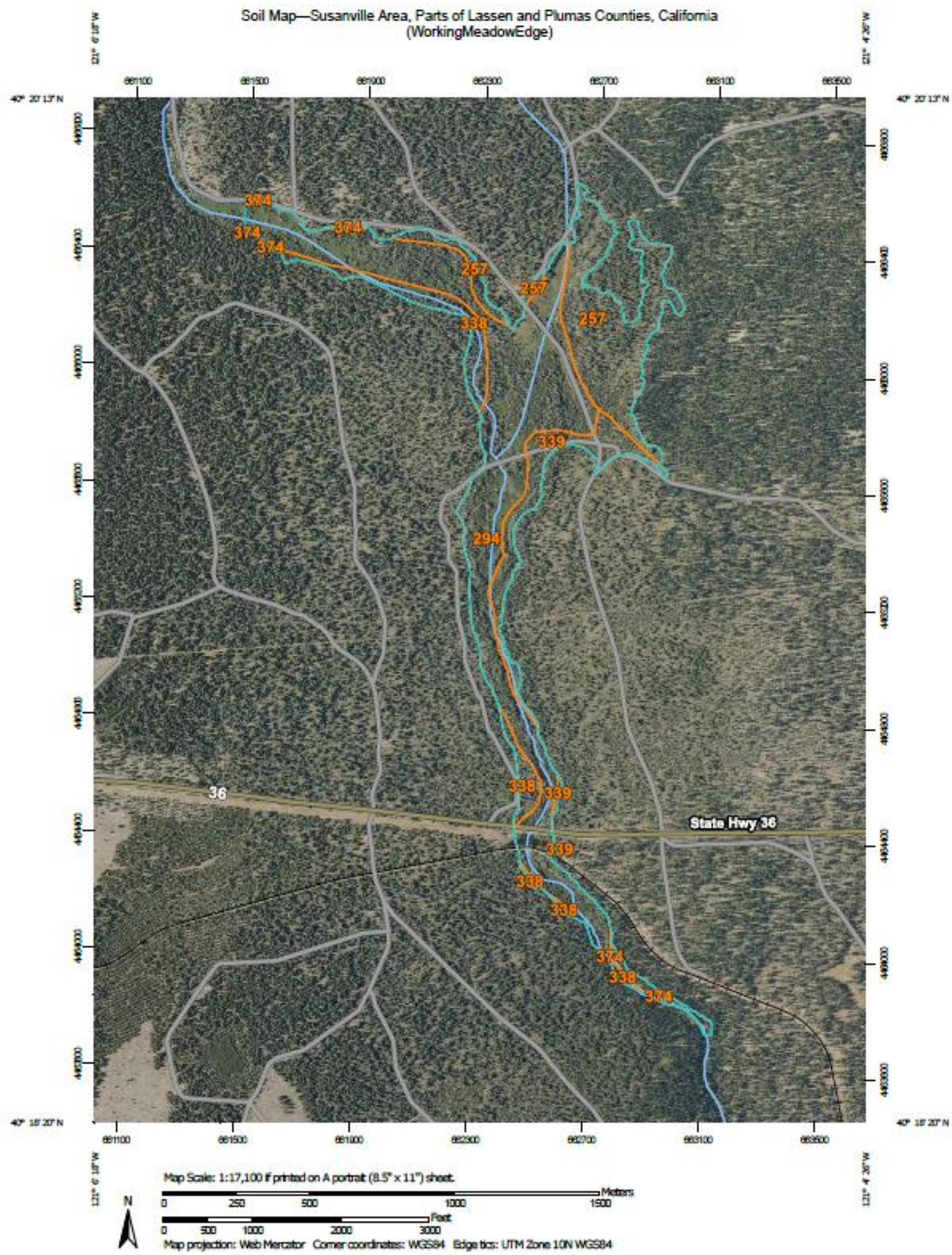
Weixelman, D. A., Zamudio, D. C., & Zamudio, K. A. (1996). *Central Nevada riparian field guide*. USDA Forest Service, Intermountain Region, Ogden. R4-ECOL-96-01.

Westbrook, C. J. (2005). *Beaver as drivers of hydrogeomorphic and ecological processes in a mountain valley* [Dissertation]. http://www.pacificwolves.org/wp-content/uploads/2019/06/Westbrook_2005.pdf

Wright, J. P., Jones, C. G., & Flecker, A. S. (2002). An Ecosystem Engineer, the Beaver, Increases Species Richness at the Landscape Scale. *Oecologia*, *132*(1), 96–101. https://www.jstor.org/stable/4223313

APENDICIES

Appendix A. Distribution of Soil Types



Map Unit Symbol	Map Unit Name	Acres in AOI	Percent of AOI
257	Inville very gravelly sandy loam, 0 to 5 percent slopes	45.5	23.4%
294	Mountmed loam, 0 to 2 percent slopes	110.8	57.0%
338	Redriver-Weste complex, 2 to 9 percent slopes	16.3	8.4%
339	Redriver-Woodwest-Wafla complex, 0 to 9 percent slopes	18.2	9.4%
374	Swainow-Almanor complex, 15 to 30 percent slopes	3.5	1.8%
Totals for Area of Interest		194.3	100.0%

MAP LEGEND

Area of Interest (AOI)
- Area of Interest (AOI)

Soils
- Soil Map Unit Polygons
- Soil Map Unit Lines
- Soil Map Unit Points

Special Point Features
- Blowout
- Borrow Pit
- Clay Spot
- Closed Depression
- Gravel Pit
- Gravelly Spot
- Landfill
- Lava Flow
- Marsh or swamp
- Mine or Quarry
- Miscellaneous Water
- Perennial Water
- Rock Outcrop
- Saline Spot
- Sandy Spot
- Severely Eroded Spot
- Sinkhole
- Slide or Slip
- Sodic Spot
- Spoil Area
- Stony Spot
- Very Stony Spot
- Wet Spot
- Other
- Special Line Features

Water Features
- Streams and Canals

Transportation
- Rails
- Interstate Highways
- US Routes
- Major Roads
- Local Roads

Background
- Aerial Photography

MAP INFORMATION

The soil surveys that comprise your AOI were mapped at 1:24,000.

Please rely on the bar scale on each map sheet for map measurements.

Source of Map: Natural Resources Conservation Service
Web Soil Survey URL:
Coordinate System: Web Mercator (EPSG:3857)

Maps from the Web Soil Survey are based on the Web Mercator projection, which preserves direction and shape but distorts distance and area. A projection that preserves area, such as the Albers equal-area conic projection, should be used if more accurate calculations of distance or area are required.

This product is generated from the USDA-NRCS certified data as of the version date(s) listed below.

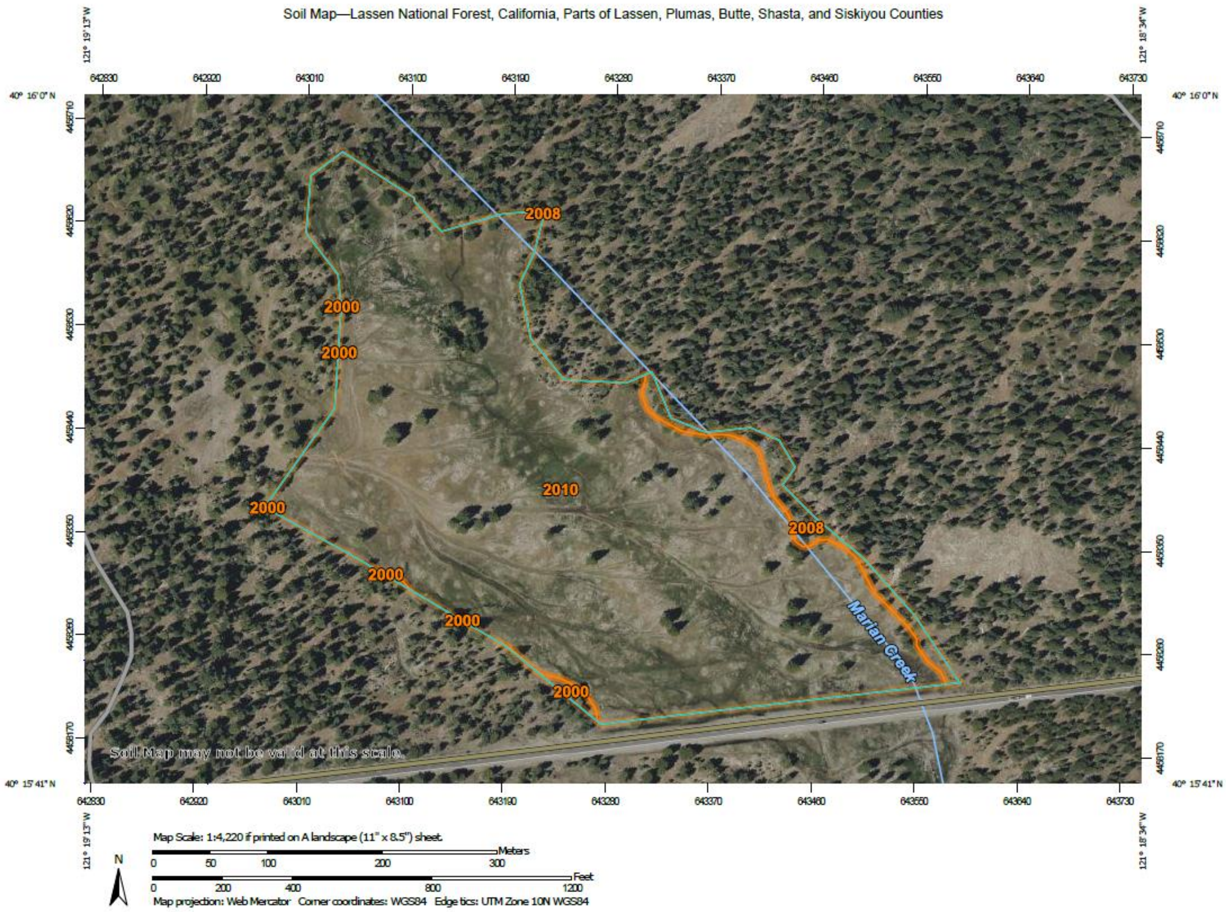
Soil Survey Area: Susanville Area, Parts of Lassen and Plumas Counties, California
Survey Area Data: Version 14, Sep 2, 2022

Soil map units are labeled (as space allows) for map scales 1:50,000 or larger.

Date(s) aerial images were photographed: Jun 8, 2019—Jun 21, 2019

The orthophoto or other base map on which the soil lines were compiled and digitized probably differs from the background imagery displayed on these maps. As a result, some minor shifting of map unit boundaries may be evident.

Figure A-1. Distribution of soil types within RCM based on digital soil mapping from Web Soil Survey (NRCS, n.d.).



Map Unit Symbol	Map Unit Name	Acres in AOI	Percent of AOI
2000	Almanor-Whorled-Inville, lacustrine complex, 3 to 15 percent slopes	0.2	0.5%
2008	Tahand-Swainow complex, 3 to 15 percent slopes	1.2	3.4%
2010	Childs-Chummy complex, 1 to 5 percent slopes	33.5	96.1%
Totals for Area of Interest		**34.9**	**100.0%**

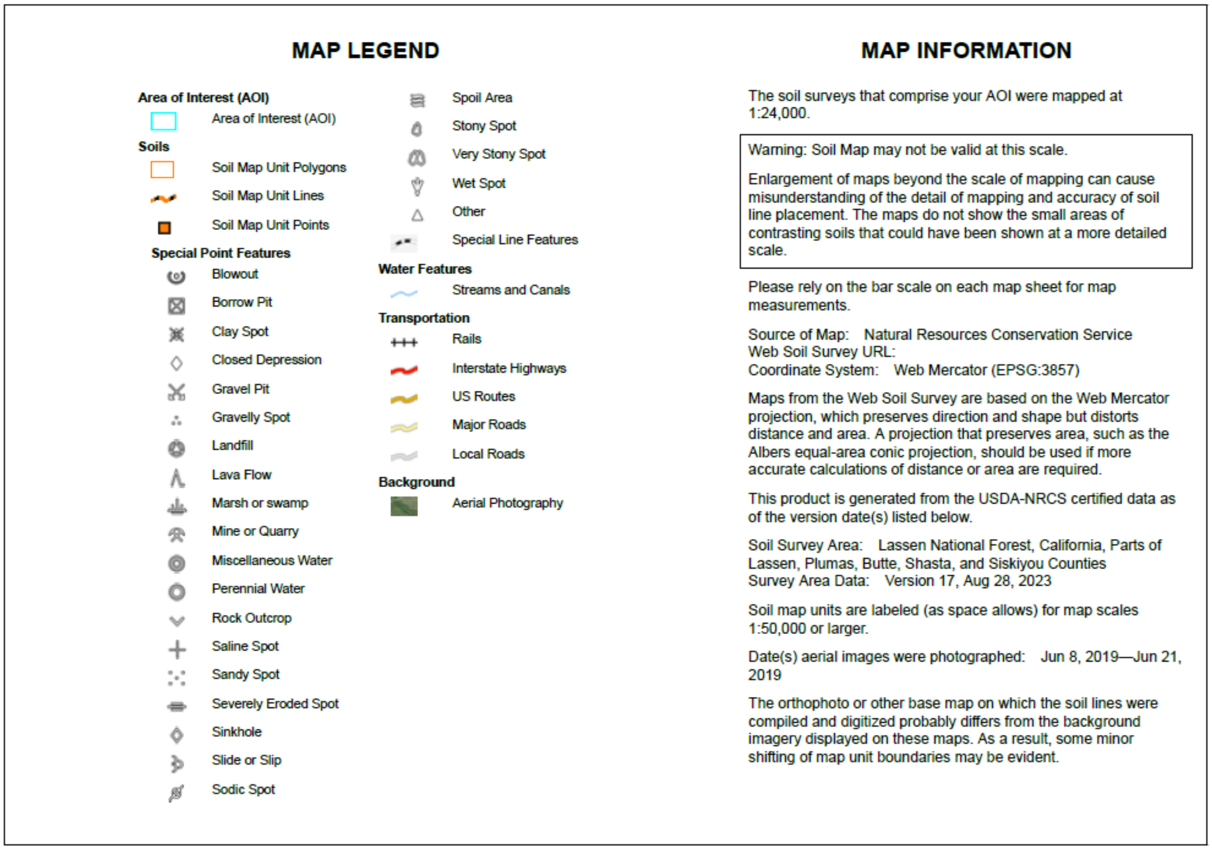


Figure A-2. Distribution of soil types within MM based on digital soil mapping from Web Soil Survey (NRCS, n.d.).

Appendix B. RCM Plant Species List

Table B-1. Rock Creek Meadow plant species list adapted from Collins Almanor Forest Timber Harvest Plan by surveyors: K. Bovee and B. Johnson.

Collins Pine Co. Rock Creek Plant Survey		
Trees	**Forbs**	*Navarretia intertexta ssp. intertexta*
Abies concolor	*Achillea millefolium*	*Navarretia sinistra*
Pinus jeffreyi	*Aquifolium repens*	*Osmorhiza berteroi*
Pinus lambertiana	*Aquilegia formosa*	*Packera pauciflora*
Pinus contorta ssp. murrayana	*Arnica nevadensis*	*Paeonia brownii*
Populus tremuloides	*Barbarea orthocera*	*Pedicularis densiflora*
Populus trichocarpa	*Bistorta bistortoides*	*Penstemon deustus*
Shrubs	*Calochortus nudus*	*Penstemon heterodoxus var. shastensis*
Alnus incana ssp. Tenuifolia	*Calyptridium umbellatum*	*Penstemon neotericus*
Amelanchier utahensis	*Calystegia occidentalis*	*Penstemon rydbergii*
Arctostaphylos nevadensis	*Camassia quamash*	*Phleum pratense*
Arctostaphylos patula	*Castilleja applegatei*	*Plagiobothrys (cognatus)*
Ceanothus cordulatus	*Castilleja lacera*	*Plantago major*
Ceanothus integerrimus	*Castilleja tenuis*	*Platanthera dilatata var. leucostachys*
Ceanothus prostratus	*Centaurea diffusa*	*Polygonum sawatchense*
Ceanothus velutinus	*Chamaesaracha nana*	*Potentilla gracilis*
Chrysolepis sempervirens	*Chimaphila menziesii*	*Potentilla millefolium*
Cornus sericea	*Cirsium andersonii*	*Poteridium annuum*
Ericameria bloomeri	*Cirsium scariosum*	*Prunella vulgaris*
Prunus virginiana	*Cirsium vulgare*	*Pterospora andromedea*
Ribes roezlii	*Clarkia sp.*	*Pyrola picta*
Rosa californica	*Claytonia rubra*	*Ranunculus aquatilis*
Rubus parviflorus	*Collomia grandiflora*	*Ranunculus occidentalis*
Salix lasiandra	*Collomia tinctoria*	*Ranunculus orthorhynchus*
Salix lemmonii	*Crepis modocensis*	*Rumex acetosella*
Spiraea douglasii	*Cryptantha intermedia*	*Sarcodes sanguinea*

Symphoricarpos mollis	*Cynoglossum occidentale*	*Senecio aronicoides*
Graminoids	*Dieteria canescens*	*Senecio triangularis*
Agrostis pallens	*Elytrigia repens*	*Sidalcea glaucescens*
Anthoxanthum aristatum	*Epilobium brachycarpum*	*Sidalcea oregana*
Bromus carinatus	*Epilobium pallidum*	*Silene lemmonii*
Bromus racemosus	*Equisetum arvense*	*Sisyrinchium idahoense*
Carex athrostachya	*Erigeron eatonii*	*Solidago lepida var. salebrosa*
Carex davyi	*Erigeron inornatus var. calidipetris*	*Stachys rigida ssp. rigida*
Carex douglasii	*Erigeron inornatus var. inornatus*	*Stellaria longipes*
Carex fracta	*Eriogonum nudum*	*Stephanomeria lactucina*
Carex integra	*Fragaria vesca*	*Symphyotrichum spathulatum*
Carex lenticularis var. impressa	*Galium aparine*	*Taraxacum officinale*
Carex leporinella	*Galium (boreale)*	*Taraxia tanacetifolia*
Carex nebrascensis	*Gayophytum diffusum*	*Thalictrum fendleri*
Carex pellita	*Geum macrophyllum*	*Tragopogon dubius*
Carex subfusca	*Gnaphalium palustre*	*Trifolium longipes ssp. hansenii*
Carex whitneyi	*Hackelia californica*	*Trifolium productum*
Cyperus squarrosus	*Heterocodon rariflorum*	*Triteleia hyacinthina*
Dactylis glomerata	*Hieracium albiflorum*	*Veratrum californicum*
Danthonia californica	*Horkelia fusca*	*Verbascum thapsus*
Deschampsia cespitosa	*Hosackia oblongifolia*	*Veronica peregrina ssp. xalapensis*
Deschampsia danthanioides	*Hypericum anagalloides*	*Vicia americana*
Eleocharis macrostachya	*Hypericum perforatum*	*Viola sp.*
Elymus elymoides	*Hypericum scouleri*	
Festuca idahoensis	*Kelloggia galioides*	
Hordeum brachyantherum	*Lactuca serriola*	
Juncus acuminatus	*Leucanthemum vulgare*	
Juncus balticus	*Ligusticum grayi*	
Juncus bufonius	*Lilium pardalinum*	
Juncus nevadensis	*Lupinus lepidus var. sellulus*	
Luzula comosa	*Lupinus polyphyllus var. burkei*	

Melica subulata	*Maianthemum racemosum*	
Muhlenbergia filiformis	*Maianthemum stellatum*	
Poa palustris	*Mentha arvensis*	
Poa pratensis	*Microsteris gracilis*	
Poa secunda	*Mimulus primuloides*	
Stipa occidentalis	*Monardella odoratissima*	

Appendix C. A Field Key to Meadow Hydrogeomorphic Types for the Sierra Nevada and Southern Cascade Ranges in California (Weixelman et al., 2011).

Dry

Setting

Dry meadows occur where the main source of water is precipitation or runoff. Dry meadows are located on benches, swales, drainways, terraces, slopes, and gentle summit ridges where soil has become relatively stabilized. Groundwater is generally deeper than 1 m for most or all of the growing season. They may occur adjacent to a wetter meadow that receives groundwater. These sites may resemble the *depressional seasonal* meadow type in general appearance, but are differentiated by not being located on a depressional landform.

Hydrology

Dry meadows occur at all elevations and on a variety of landforms. Dry meadows occur on sites where water from rains, snow, or snowmelt is concentrated near the soil surface and provides early season moisture sufficient for establishment of perennial graminoids and herbaceous dicots. They lose water by evapotranspiration, overland flow, and seepage to the underlying groundwater. At higher elevations, dry meadows are common where cool temperatures and snowmelt allow soil moisture to linger long enough for graminoid species and herbaceous dicots to flower and reproduce before the dry season comes. This type may mix with other meadow, forest and woodland types at a fine scale. Above treeline, this type may intergrade with alpine fell field communities in areas with decreasing levels of soil development.

Vegetation

Dry meadow vegetation is generally dominated by grasses (*Poaceae* family), dryland sedges (*Carex* spp.), or dryland rushes (*Juncus* spp.). This type includes a very broad range of moisture conditions from low elevation stream terraces and swales to subalpine and alpine areas on gravelly slopes. High elevation sites on gravelly soils may be dominated by shorthair sedge (*Carex exserta*) or Parry's rush (*Juncus Parryi*). High elevation sites that are saturated early in the season may be dominated by shorthair reedgrass (*Calamagrostis breweri*) or Sierra ricegrass (*Ptilagrostis kingii*).

Figure 15. *Photo of a dry meadow type occurring on a stream terrace on the Stanislaus National Forest.*

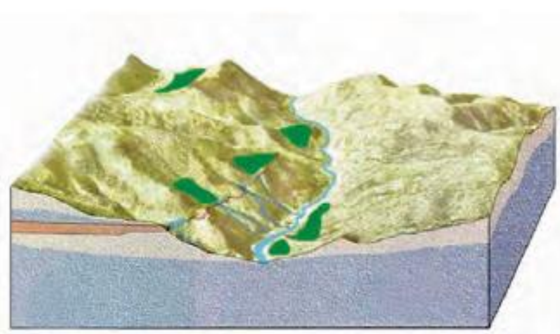


Figure 16. *Typical landscape positions for the dry meadow type.*

Subsurface Low Gradient

Setting

Subsurface low gradient meadows contain no stream or river channel, or a majority of the meadow does not contain a stream or river channel with a discernable bed and bank morphology. Ditches or rills may be present. This meadow type occurs on alluvium or colluvium and typically is connected to a distinct topographic flow line (perennial or intermittent channel) above and below the meadow. This type occurs where the down valley slope averages less than 2 percent.

Hydrology

The dominant water sources are surface water and groundwater throughflow. Additional water sources are groundwater inputs or overland flow from surrounding uplands, tributary inflow, and precipitation. Inflow and outflow stream channels (perennial or intermittent) are typically visible at the top end of the meadow and bottom end of the meadow. Subsurface low gradient types are distinguished from the groundwater discharge type by being a throughflow groundwater/interflow system with a water source (surface or subsurface) at a higher elevation and outflow (surface and/or subsurface) at the bottom of the meadow.

Vegetation

Vegetation is generally dominated by hydric meadow graminoid species (obligate and facultative wetland plant species). Occasionally scattered riparian shrubs including willows (*Salix* spp.) and coniferous tree species are present. Sites often exhibit a strong zonation from hydric to drier soils going from the center of the meadow toward the uplands.

***Figure 25**. Photo of a subsurface low gradient meadow occurring on the Stanislaus National Forest.*

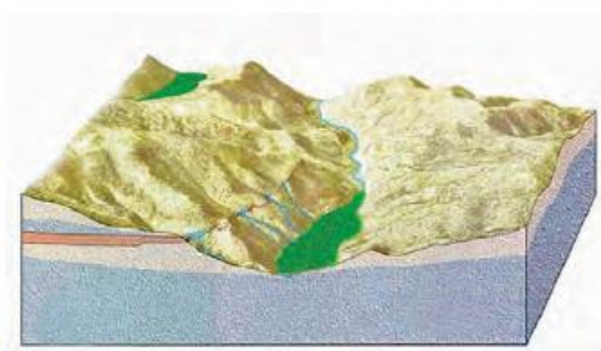


***Figure 26.** Typical landscape positions for subsurface low gradient meadows occurring in low gradient valleys.*