*Hydrological Processes*

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# Rapid Hydrological Responses Following Process-Based Restoration in a Degraded Sierra Nevada Meadow

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## ABSTRACT

Mountain meadows are ecologically important groundwater dependent ecosystems that retain and store water in upland forested landscapes. They tend to occur in low gradient, broad valleys where water slows and sediment accumulates, making them efficient locations for restoration. Over a century and a half of land use has degraded many meadows in the Sierra Nevada, reducing their hydrological and ecological functionality. Process-based restoration (PBR) is an ecosystem rehabilitation approach that utilises biogeomorphic processes to facilitate functional ecosystem recovery. Low-tech applications of PBR leverage fluvial processes, plant growth and the manipulation of onsite materials to increase structural and hydrological complexity. In meadows, typical goals associated with restoration are to increase groundwater elevations, expand wetted area, encourage sediment capture and create diffuse flow paths leading to improved ecological function over time. This study compares surface and groundwater conditions in a degraded riparian meadow in the Sierra Nevada, California, USA for 1 year before and after process-based restoration to understand initial changes in meadow hydrogeomorphic function. Restoration included the installation of 39 postless beaver dam analog structures in ~1 km of incised meadow channel. Stage-discharge data at the inlet and outlet of the project area were paired with groundwater data collected from 13 wells distributed across the meadow to estimate increased water storage of 3700 $m^3$ due to restoration. After the wet winter of 2023, we estimated that pools upstream of structures filled to over half their volume with fine sediment. We also applied hydrodynamic modelling to evaluate fluvial changes at high flows and found that restoration increased flow complexity and wetted surface area. These short-term responses highlight the potential speed and ability of low-tech, process-based restoration in achieving restoration outcomes.

## 1 | Introduction

Mountain meadows furnish valuable wetland habitat in upland forested landscapes and provide a variety of ecosystem services including retaining water (Loheide and Booth 2011), serving as carbon sinks (Norton et al. 2011; Reed et al. 2021) and providing habitat for a diverse suite of organisms (Allen-Diaz 1991; Oles et al. 2017; Campos et al. 2020). Herbaceous vegetation, relatively flat terrain and abundant water make meadows attractive to a diversity of wildlife and to humans. However, due to their overuse by humans and livestock, montane meadows have become one of the most altered, impacted and at-risk landscape features in the western US (Ratliff 1985; Kattlemann 1996). Meadows in the Sierra Nevada of California historically may have covered about three times more area than they currently occupy (~2% of the landscape) but desiccation via channel incision and associated forest encroachment have converted many wet meadows into upland habitats (Cummings, Pope, and Mak 2023). Most meadows are degraded due to past and current land use including removal of beaver, livestock overgrazing, filling, compacting or diking for railroad grades or roads, straightening channels for more efficient water conveyance, and draining for homesteading and other uses (Kattlemann 1996; Drew et al. 2016; Pope and Cummings 2023). These land uses often cause excess stream bank and bed erosion by consolidating flow paths and reducing resistance by wood and vegetation, resulting in oversized and incised channels (Ratliff 1985; Viers et al. 2013). Channel incision can lower groundwater elevation, disconnect the stream from its floodplain and reduce habitat diversity (Loheide II and Gorelick 2005, 2006, 2007; Hammersmark, Rains, and Mount 2008; Beechie et al. 2010; Pollock et al. 2014).

Because of their high ecological importance and extent of degradation, meadows are often targeted for ecological restoration. Meadows usually occur in low-gradient alluvial valleys with broad floodplains where water slows, spreads and infiltrates into fine-grained soils, resulting in deposition of sediment and particulate organic matter (Weixelman et al. 2011; Viers et al. 2013; Nash et al. 2018). Meadows are groundwater dependent ecosystems with meadow plants uniquely adapted to shallow water tables (usually less than 1 m) (Allen-Diaz 1991; Loheide II and Gorelick 2007; Weixelman et al. 2011). To successfully restore the ecological conditions of meadows, hydrologic and geomorphic goals of meadow restoration often include slowing water velocities to initiate deposition of sediment, increasing wetted surface area to encourage groundwater infiltration, and reducing the depth and volume of incised channels to reduce groundwater drainage, raise groundwater elevations and increase frequency of overbank flows. These changes often lead to increased meadow vegetation productivity (Hammersmark, Rains, and Mount 2008; Loheide II and Gorelick 2007; Tague, Valentine, and Kotchen 2008), increased wildlife usage (Campos et al. 2020), increased carbon sequestration in soils (Reed et al. 2022) and increased fire resistance (Fairfax and Whittle 2020; Pope and Cummings 2023).

Many meadow restoration efforts have focused on reconfiguring channel morphology using heavy machinery to divert streamflow onto the meadow floodplain. Approaches such as complete channel fill and pond-and-plug restoration can be effective at quickly raising groundwater elevation and rejuvenating meadow vegetation (Hammersmark, Rains, and Mount 2008; Loheide II and Gorelick 2007). However, concerns have been raised about the added disturbance due to construction vehicles used to remove riparian vegetation and fill or plug incised channels (Ciotti et al. 2021), addition of novel features such as ponds on the meadow floodplain (Pope et al. 2015) and not addressing the causal factors of degradation (Palmer, Filoso, and Fanelli 2014; Ciotti et al. 2021).

Alternative restoration approaches, here grouped under the term "process-based restoration" (e.g., Beechie et al. 2010), focus less on rapidly resetting channel conditions and more on encouraging positive system change over time. These techniques often rely on dynamic interactions among physical, fluvial, and biological processes to achieve ecosystem recovery while minimising the use of heavy machinery (Beechie et al. 2010; Kondolf et al. 2006; Ciotti et al. 2021). In many cases, instream biogenic structures, such as beaver dam analogs (BDAs), are built by hand to mimic the dam building activities of beavers (*Castor* spp.) to slow water and initiate floodplain connectivity (Pollock et al. 2014; Wheaton et al. 2019).

This relatively low-impact and "low-tech" application of the process-based approach (Wheaton et al. 2019) is gaining recognition as a restoration tool for degraded, low-gradient streams and meadows (Ciotti et al. 2021; Nash et al. 2021; Jordan and Fairfax 2022). The biogenic structures used to initiate changes to streamflow are made from locally sourced building materials such as conifers encroaching on the meadow or from adjacent slopes, willow or alder branches from within the meadow, sod from eroding banks and rocks from the channel bed. The structures are designed for specific purposes and are strategically and carefully placed. For example, some structures are built as channel-spanning dams in incised reaches to slow water velocity and aggrade the channel bed through sediment deposition, while other structures are placed to direct flow into relict channels or to erode a bank to encourage meandering and downstream channel aggradation (Wheaton et al. 2019). Multiple structures can be used together to increase channel complexity, activate multiple flow paths, increase the residence time of water moving through the reach and reconnect previously disconnected floodplains (Beechie et al. 2010; Pollock et al. 2014; Ciotti et al. 2021).

While mimicking beavers often has popular public appeal (e.g., Goldfarb 2018; Charnley 2019; Whitcomb 2022), debate remains as to what successful meadow restoration consists of, under what conditions these results should be expected, how quickly they might occur and how long they might last (Pilliod et al. 2018; Nash et al. 2021). Nash et al. (2021) described a conceptual framework that addresses the processes needed for this type of process-based restoration to be considered successful. The framework defines short-term hydrological responses to instream structures that are expected to link to positive longer term outcomes including persistent elevated groundwater tables, increased vegetative productivity and higher streamflows in late spring and summer. To achieve these outcomes, the instream structures must impound water and sediment to increase water storage, and initiate sediment aggradation and channel heterogeneity. The impounded water causes groundwater levels to rise, resulting in increases in wetland vegetation and delayed streamflow (Nash et al. 2021).

We conducted a study to track specific hydrological processes for the first year following process-based meadow restoration in the northern Sierra Nevada of California. Short-term goals of the restoration were to increase storage of surface and groundwater in the meadow, capture sediment upstream of instream structures and increase channel and floodplain heterogeneity. We used intensive streamflow and groundwater measurements, channel and structure surveys, and hydrodynamic modelling to examine how the treatment affected stream discharge, groundwater storage, surface water hydraulics, and sediment deposition. These results were used to assess whether hydrologic processes can be manipulated over short timescales to attain restoration goals. Longer term changes in surface and groundwater hydrology, geomorphology, and vegetation will be reported after additional data are collected.

## 2 | Methods

### 2.1 | Site Description

Middle Creek Meadow is a 6.4 ha riparian meadow located in the East Branch of the North Fork Feather River watershed in the Plumas National Forest, California, USA (Figure 1) (40.123° north latitude, 120.642° west longitude). Riparian meadows are defined as throughflow meadows with an inflow channel upstream, an outflow channel downstream, and a discernible channel through most of the meadow (Weixelman et al. 2011). Middle Creek Meadow has an average slope of 0.021 and a mean elevation of 1426 m. Rainfall makes up the majority of the 594 mm of mean annual precipitation (California Data Exchange Center 2024; Viers et al. 2013; Null, Viers, and Mount 2010). The Mediterranean climate (Köppen csb) brings cold and wet winters with December and January mean minimum temperatures of −5.1°C and warm and dry summers with July mean maximum temperature of 28.3°C (PRISM Climate Group 2004; Peel, Finlayson, and McMahon 2007).

The meadow is at the outlet of a 958 ha watershed drained by Middle Creek and occurs within a zone of granodiorite bedrock with shallow soils of the Chaix soil series on the adjacent hillslopes (NRCS 1998). Middle Creek has no tributary streams or springs in the meadow. The Walker Fire burned the lower portion of the watershed in 2019 and the Dixie Fire burned the upper portion in 2021, resulting in the majority of the watershed being burned at high or moderate severity in the past 5 years (Figure 1). The forested upland primarily consists of Jeffrey pine (*Pinus jeffreyi*), ponderosa pine (*Pinus ponderosa*), and white fir (*Abies concolor*) with sparse occurrences of trees in the meadow. Herbaceous vegetation within the meadow includes wet, mesic, and dry meadow plant communities.

Middle Creek Meadow was identified as degraded by Plumas National Forest staff because of deep channel incision, active head cutting and upland vegetation encroachment. This ecological degradation was attributed to legacy land use practices within the Middle Creek watershed including mining, agriculture, livestock grazing and logging. Before restoration, Middle Creek had two distinct reaches differentiated by a > 1 m headcut. The upstream reach (Reach 1) was 490 m in length and extended through approximately 60% of the upper meadow area (Figure 2). The channel in Reach 1 averaged approximately 0.50 m wide and ranged from 0.15 to 0.80 m deep. The downstream reach (Reach 2) was 420 m long and extended to the outlet of the meadow. The channel in Reach 2 averaged 1.0 m wide, had several additional headcuts, and was three times more incised than Reach 1 with channel depths ranging from 0.50 to 2.60 m (Figure 2). The distinct channel conditions of Reach 1 and Reach 2 provided an opportunity to assess restoration treatments in two different hydrogeomorphic contexts. We measured stream discharge and groundwater elevations from fall 2021 through September 2023. The meadow was treated in late summer 2022 with minor additions in spring 2023.

### 2.2 | Restoration

Construction of restoration structures occurred between 30 August and 4 September 2022 following a design developed by Swift Water Design LLC. Six experienced practitioners from Swift Water Design implemented the design and additional labour was provided by USDA Forest Service staff including 40 wildland firefighters and volunteers. A total of 39 structures were constructed over a period of six days using handheld tools including chainsaws, loppers, levers, and shovels (Figure 2). Four structures were located upstream of Reach 1 and consisted primarily of large wood felled within the meadow. The remaining 35 instream structures consisted of postless BDAs (Wheaton et al. 2019) made from channel bed and bank material, sod, rock and wood from trees (pine, fir, willow, alder) in or adjacent to the meadow. Boughs with tips pointing upstream were interwoven and packed with sod, sand and gravel to seal contact points with the channel bed. Larger material was added to build up the structures over multiple layers in a 'lasagna' fashion with boughs serving as the "pasta" and sod, sand, gravel, and rock serving as the "sauce." The layers were canted downstream to create a gradual slope along the upstream edge of the structure. This surface was packed with sod to seal the structure and encourage vegetative growth. Larger tree trunks were integrated onto the downstream 'skirt' of structures and, where possible, locked against stable bank features, such as bank-embedded boulders or trees, to provide stability in high flows. No streamflow was diverted during BDA installation. Immediately post-treatment, newly activated surface water channels within the meadow were mapped to calculate added channel length and difluences for the post-treatment low flow condition.

In Reach 1, two structures were placed near the inlet of the meadow to divert a portion of the streamflow onto the meadow floodplain to bypass flow into the headcut between transects 30 and 50, which was also filled by a structure composed of wood from 8 conifers. Fourteen additional structures were installed in and adjacent to the channel to initiate and expand overbank flow or to slow water and activate alternative flow paths. Combined, the Reach 1 restoration resulted in a mean density of 3.5 structures per 100 m of channel (Figure 2), a mean width of 4 m (across the channel), a mean length of 2.3 m (along the channel) and a mean height of 0.8 m. Reach 1 structures were generally built to extend beyond the existing channel width to promote overbank flooding and lateral floodplain connection.

In Reach 2, the more incised portion of the meadow, 18 structures were installed within the channel at a mean density of 4.3 structures per 100 m of channel (Figure 2). The Reach 2 structures averaged 2.2 m wide, 4.3 m long and 1.1 m high. While the Reach 2 structures were taller than those built in Reach 1, the Reach 2 structures did not reach bank height (often close to 2 m), because taller structures would be unlikely to withstand expected high spring flows confined to the single-thread channel (Wheaton et al. 2019). Instead, goals were to impound water and aggrade the incised channel in the first year and enlarge the structures in the following years as sediment filled the pools and provided additional structural support (Pollock et al. 2014; Wheaton et al. 2019). Between 18 and 23 May 2023, four new structures were built and eight

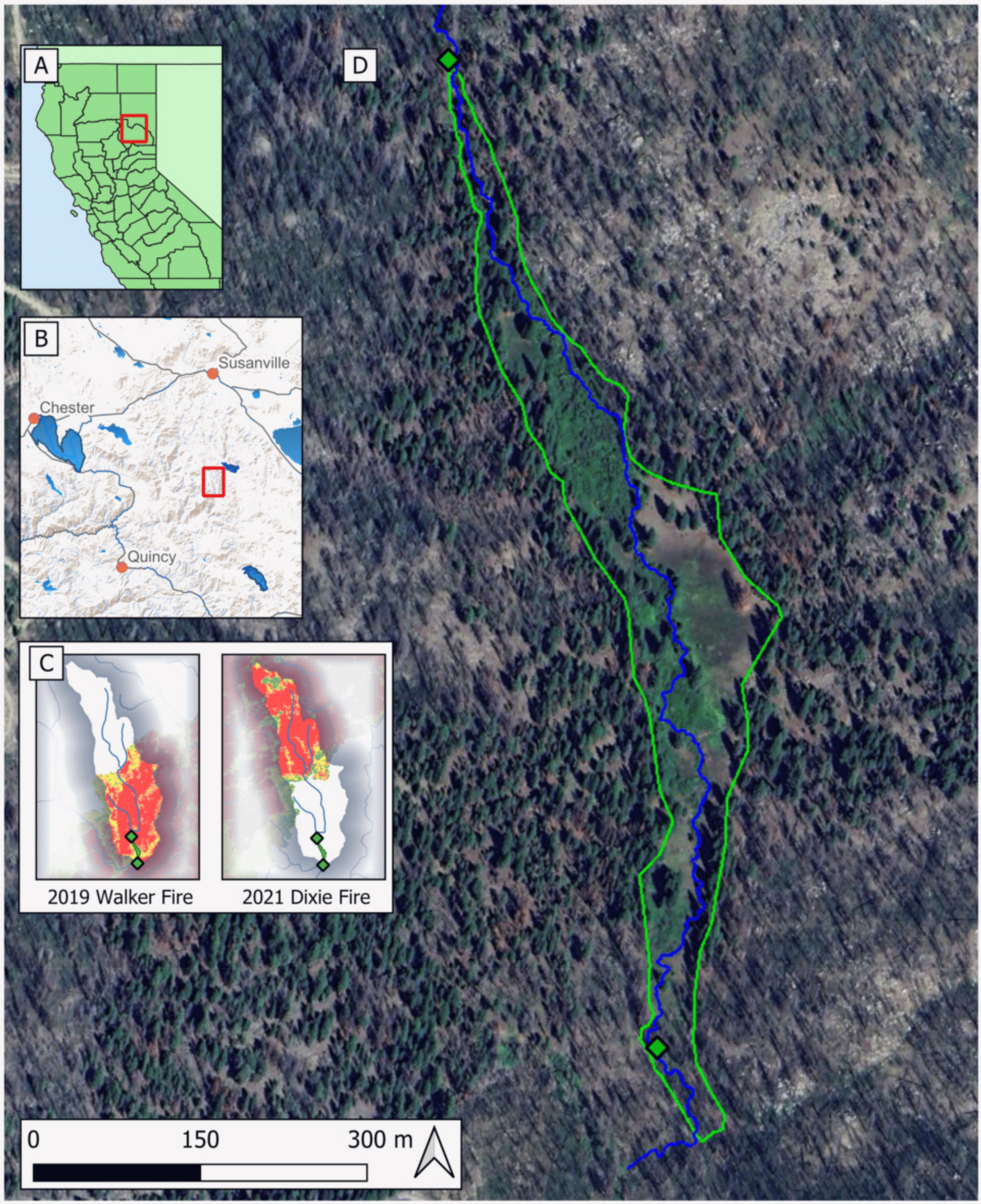


**FIGURE 1** | (A) Location of Middle Creek Meadow (red box) in California, USA and (B) in the northern Sierra Nevada in Plumas County (40.123 North latitude, 120.642 West longitude). (C) Parts of the watershed of Middle Creek Meadow burned in the Walker Fire in 2019 and Dixie Fire in 2021. Red, yellow and green shading represent CBI4 high, moderate and low burn severity categories, respectively. (D) Aerial imagery of Middle Creek Meadow outlined in green. Diamonds in (C) and (D) show locations of the upper (northern) and lower (southern) gaging stations.

were expanded in Reach 1 to further increase sediment storage or meet other objectives such as increasing the wetted area of the meadow surface. The installation team returned again in spring 2024. This study focused on the changes associated with the initial restoration treatment implemented in late summer 2022.

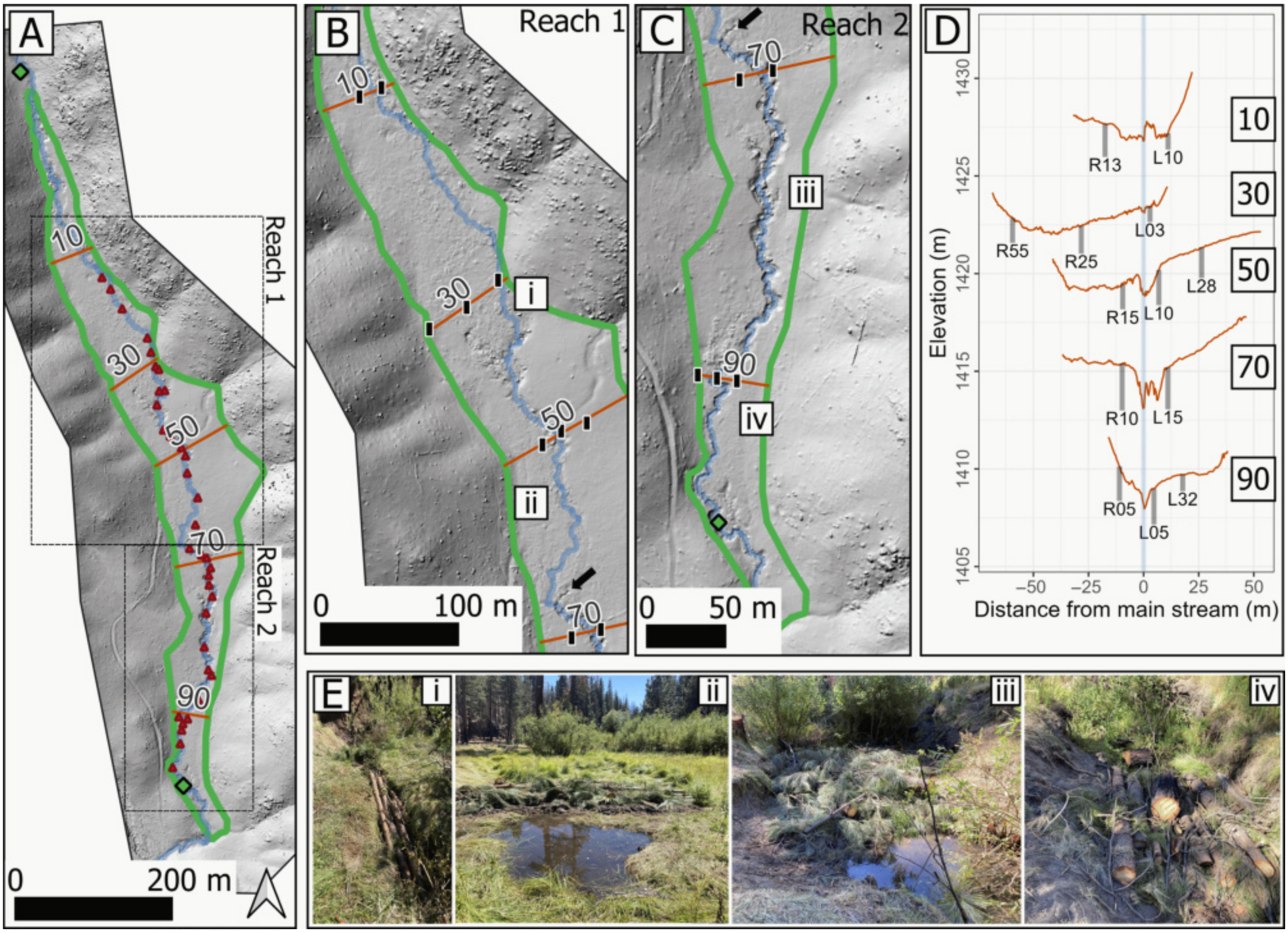


**FIGURE 2** | (A) LiDAR-derived hillshade showing Middle Creek Meadow outlined in green and Middle Creek shown in blue. The boxes show Reach 1 (490 m) and Reach 2 (420 m), groundwater transects (orange lines) labelled with the percent upstream meadow area, gaging stations designated by black diamonds, and 35 of 39 restoration structures installed in August 2022 as maroon triangles. Detailed view of (B) Reach 1 and (C) Reach 2. Vertical black bars indicate groundwater well locations. The black arrow indicates the location of the headcut separating Reach 1 and Reach 2. (D) Cross-section elevation profiles for Middle Creek at each transect. Lines are centred on the thalweg of the main channel prior to restoration. Grey bars indicate groundwater well locations and depths. (E) Photos showing restoration structures. Locations of photos are shown with corresponding Roman numerals in (B) and (C). From upstream (Reach 1) to downstream (Reach 2): (i) 8 conifers packed into substantial meadow headcut, (ii) 11 m wide channel-spanning structure, (iii) downed conifers placed into 1.3 m deep incised channel, and (iv) conifers placed into incised meadow channel.

## 2.3 | Groundwater

In November 2021, 13 groundwater wells were installed to measure groundwater storage dynamics in the meadow. Wells were spaced along five transects perpendicular to the main stream channel at specific intervals representing 10%, 30%, 50%, 70% and 90% of the meadow area (Figure 2). Boreholes were drilled to refusal with depths ranging from 1.4 to 3.6 m using hand augers and a rotary hammer rock drill. Boreholes were cased with 1-in. schedule 40 PVC pipe that was packed with sand along a 20–50 cm screened interval at the bottom of the well. The screened interval was hydraulically isolated using a bentonite and native soil mixture (Hahm et al. 2022). The wells were instrumented with Solinst Levelogger 5 (0.05% accuracy) or Levelogger 5 Junior (0.1% accuracy) unvented pressure transducers with 0.001 m resolution (Solinst, Georgetown, Ontario, Canada) that logged pressure at 15-min intervals. Water pressures were corrected for barometric pressure using a Solinst Barologger located at the downstream gaging station. Manual measurements of groundwater surface elevations were periodically made using a Solinst Model 102 Water Level Meter to verify pressure transducer measurements.

## 2.4 | Stream Discharge

In October 2021, stream gaging stations were installed at the inlet and outlet of Middle Creek Meadow (Figure 2), to compare discharge dynamics above and below the area to be restored before and after restoration. A vented pressure transducer (Ott PLS; Ott Hydromet GmbH, Kempten, Germany) was installed in a stilling well at each station and connected to a CR1000X data logger (Campbell Scientific, Logan, UT, USA) to record stream stage at 15-min intervals with 0.05% accuracy and 0.001 m resolution. Sheds were built on the banks to house the data loggers and batteries that were charged with solar panels. Staff plates were installed near the pressure transducers to allow for visual validation of the stream stage during regular site visits.

Discharge measurements were collected at each station starting in December 2021 to develop stage-discharge rating curves. Winter access to sites was limited, therefore, a majority of flow measurements were collected during the summer months. Discharge measurements were made using a Flowtracker Acoustic Doppler Velocimeter (YSI Incorporated, Yellow Springs, OH, USA). Stream velocities measured at 0.6 times the depth were used to calculate discharge using the mid-section method (SonTek/YSI Inc 2007). During periods of low streamflow, discharge was measured using a salt dilution gaging technique (Moore 2005). A rating curve of the following functional form was fit to stage-discharge data:

$$\text{Discharge} = c(\text{Stage} - a)^b \tag{1}$$

where $a$, $b$ and $c$ were coefficients unique to each station.

To estimate high-flow return intervals, we used the USGS regional regression equations that predict flow statistics as a function of drainage area, mean basin elevation and mean annual precipitation (Gotvald et al. 2012). The median annual exceedance flow (2-year return period; $Q_2$) for the Middle Creek gaging stations was determined to be $1.2\,\text{m}^3\,\text{s}^{-1}$.

### 2.5 | Linking Stream Discharge and Water Storage

Stage-discharge data from the two gaging stations were paired with groundwater and surface water data to explore the role of surface water interactions with groundwater before and after the restoration. Specifically, we hypothesised that an observed decrease in outlet stream discharge immediately following restoration could be directly linked to groundwater storage increases. To calculate the volume of post-restoration 'missing' outlet discharge, we first fit a power-law model that predicts outlet discharge as a function of inlet discharge using data from the two-month period prior to restoration:

$$Q_{\text{outlet}} = gQ_{\text{inlet}}{}^{f} \tag{2}$$

where $Q_{\text{outlet}}$ and $Q_{\text{inlet}}$ were the discharge values at the lower and upper gaging stations, respectively, and $g$ and $f$ were fitted coefficients. We used Equation (2) to extrapolate the counterfactual outlet discharge over the post-restoration period as though restoration had not occurred ($Q_{\text{outlet,unrestored}}$). With this counterfactual $Q_{\text{outlet,unrestored}}$, we analysed the groundwater mass balance equations for the restored and unrestored scenarios:

$$\Delta S_{\text{w,restored}} = \int \left(Q_{\text{inlet}} - Q_{\text{outlet,restored}} - \text{ET}_{\text{restored}} + R_{\text{hillslope,restored}}\right) dt \tag{3}$$

$$\Delta S_{\text{w,unrestored}} = \int \left(Q_{\text{inlet}} - Q_{\text{outlet,unrestored}} - \text{ET}_{\text{unrestored}} + R_{\text{hillslope,unrestored}}\right) dt \tag{4}$$

where $\Delta S_{\text{w}}$ was the change in water storage over the integration period, the integral was taken over the 30 days immediately preceding and following restoration, ET was evapotranspiration sourced from the meadow and $R_{\text{hillslope}}$ was recharge to the meadow from hillslopes at the meadow flanks. Additional storage owing to restoration ($V_{\text{w}}$) was then estimated as the difference in $\Delta S_{\text{w}}$ between the restored and unrestored cases:

$$V_{\text{w}} = \Delta S_{\text{w,restored}} - \Delta S_{\text{w,unrestored}} = \int \left(Q_{\text{outlet,unrestored}} - Q_{\text{outlet,restored}}\right) dt + \int \left(\text{ET}_{\text{unrestored}} - \text{ET}_{\text{restored}}\right) dt + \int \left(R_{\text{hillslope,unrestored}} - R_{\text{hillslope,restored}}\right) dt \tag{5}$$

Assuming recharge to the meadow groundwater system from adjacent hillslopes during this precipitation-free period and evapotranspiration from groundwater were not significantly altered by restoration actions in the short window immediately following restoration, we neglected the $R$ and ET difference terms in Equation (5). In this case, the integrated difference between the counterfactual predicted $Q_{\text{outlet,unrestored}}$ and the observed $Q_{\text{outlet,restored}}$ provided an estimate of the total volume of water that recharged the meadow groundwater system relative to a hypothetical unrestored scenario:

$$V_{\text{w}} = \int \left(Q_{\text{outlet,unrestored}} - Q_{\text{outlet,restored}}\right) dt \tag{6}$$

Importantly, this analysis did not assume that evapotranspiration from the groundwater system was negligible, only that the change in evapotranspiration in the immediate post-restoration period relative to the unrestored counterfactual was small relative to the difference in observed discharge versus the counterfactual unrestored discharge.

Since some of $V_{\text{w}}$ includes surface detention storage behind structures, we also quantified the volume of surface water stored upstream of in-channel restoration structures ($V_s$) using a digital elevation model (DEM) depression filling algorithm, which allowed us to calculate a subsurface detention storage ($V_{\text{g}}$) as follows:

$$V_{\text{g}} = V_{\text{w}} - V_{\text{s}} \tag{7}$$

First, we applied the algorithm to an unmodified DEM to establish baseline conditions. Then, we 'burned' the restoration structures into the DEM at their measured locations and heights and reapplied the depression-filling algorithm. The difference between these two calculated rasters represented the depression space created by the restoration structures, and we estimated the volume of ponded water entrapped by the structures post-restoration ($V_s$) as the sum of the difference raster multiplied by the pixel area. This method did not account for water forced into new flow paths outside the main channel and assumed that the structures were filled to the top and no water passed through the structures, which provided a rough approximation of entrapped surface water. Finally, we converted $V_g$ to a storage depth by dividing by the meadow area less the new inundated surface area, which enabled us to also estimate meadow drainable porosity, which is a key source of uncertainty in groundwater and hydraulic modelling, by dividing this depth-normalised $V_g$ by the average groundwater level increase over the restoration period.

We also investigated whether the relationship between groundwater storage in the meadow and stream discharge changed in response to restoration. Storage and discharge in many watersheds are tightly coupled (Kirchner 2009), and shifts in the storage-discharge relationship can be indicative of underlying shifts in hydrologic function (Cheng et al. 2017). Because the precipitation and streamflow in 2022 were different than in 2023, we subset groundwater levels and outlet discharge from a pre-restoration period between 1 May 2022 and 15 August 2022 and identified similar discharge levels during the post-restoration period between 1 May 2023 and 1 October 2023. For all shared values of outlet discharge within these two periods, we plotted the average depth to groundwater across all wells and tested whether there was a change in groundwater elevation in the post-restoration period using a linear mixed-effects model developed with the *statsmodels* package (Seabold and Perktold 2010) in Python 3.8 (Python Software Foundation 2019). The model estimated the depth to groundwater as a function of the logarithm of discharge plus random intercepts for the pre- (2022) and post-restoration (2023) periods.

## 2.6 | Sediment

We measured in-channel sediment deposition in pools in summer 2023 to determine if aggradation was occurring upstream of structures. We defined pools as mostly still water with > 3 times the depth of the surrounding channel and a clearly defined riffle crest. We measured the volume of fine sediment and particulate that accumulated in every third pool encountered while walking up the main channel through the meadow using the *V* star ($V^*$) method (Figure 3; Lisle and Hilton 1992). $V^*$ is the fraction of the total pool volume occupied by fine sediment and was calculated for 10 pools, nine of which were formed upstream of restoration structures and one was a naturally formed pool. We calculated the mean residual width of each pool by measuring three to seven equally spaced cross sections that extended from the water's edge where the depth of water plus fine sediment equalled the mean water depth at the pool outlet, which was usually the top of the structure (Figure 3). The mean residual pool width was multiplied by the measured length to estimate the residual surface area at base flow conditions. At 20 equally spaced points along each cross section, we sampled the depth of fine sediment to resistance using a 1.2 cm diameter graduated rod and the total depth of sediment plus water ($D_t$). We calculated the residual pool depth ($D_r$) by subtracting the mean water depth at the pool outlet (i.e., riffle crest) from the total depths and multiplying $D_r$ by residual pool area to obtain the residual volume ($V_r$). We determined the volume of fine sediment ($V_f$) from the sediment depths and residual pool area and calculated $V^*$ as follows:

$$V^* = \frac{V_f}{(V_f + V_r)} \tag{8}$$

## 2.7 | Hydrodynamic Modelling

Flow conditions in Middle Creek Meadow were modelled in Sedimentation and River Hydraulics-Two Dimension (SRH-2D) Version 11.6 (Aquaveo LLC Provo, UT, USA), which solves 2-dimensional, depth-averaged, dynamic wave equations to predict water surface elevation, water depth and depth-averaged velocity (Lai 2009). Model inputs included materials coverage, mesh and boundary conditions, which were developed from LiDAR-derived DEMs, geomorphic mapping and topographic surveying with a total station.

Locations of geomorphic features and materials were mapped using ArcGIS Field Maps and included wet and dry channels, headcuts greater than 0.2 m, in-channel wood greater than 0.2 m diameter and 2 m length, vegetation cover, channel bed grain size, the presence and type of overbank vegetation, and dimensions and positions of restoration structures. Grain size median diameter was measured using a gravelometer or facies card. Vegetation conditions outside the channel were not recorded as part of the geomorphic mapping, and were estimated from the aerial RGB imagery. Vegetation roughness values were 0.08 for conifers, 0.035 for grass and 0.15 for willow (Chow 1959). Channel bed substrate was primarily coarse sand to fine gravel

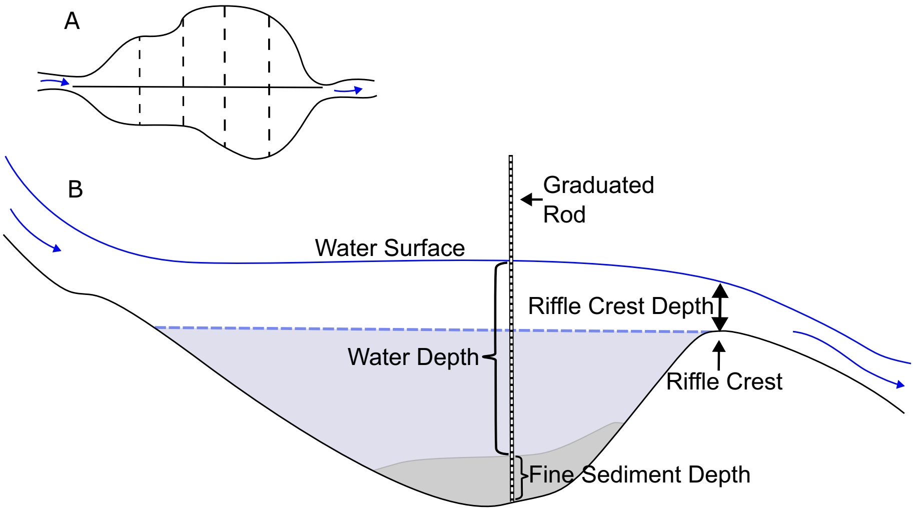


**FIGURE 3** | The volume of residual sediment and water in pools was estimated using the *V* star method described by Lisle and Hilton (1992). The length and distance between 3–7 equally spaced cross sections in each pool were measured to estimate pool area (A). Total water depth and depth of fine sediment were measured along the pool cross sections at ~20 equally spaced points, and riffle crest depth was subtracted from pool depth measurements to estimate the residual pool volume (B). Figure modified from Lisle and Hilton (1992).

(0.5 mm to 8 mm) and was assigned a roughness value of 0.02 consistent with this grain size class (Coon 1997). These roughness values were assigned to the materials coverage as input for the hydrodynamic model.

Topographic surveys were completed in summer 2022 using a Trimble M3 DR total station (Trimble Inc. Westminster, CO, USA). Survey data were transformed in Trimble Business Centre using a 7-point parameter transformation based upon the northing, easting, and elevation data of three known benchmarks and projected to UTM Zone 10 N NAD83. We surveyed the coordinates of the channel thalweg, channel banks where the channel width was less than 1 m, and specific locations of headcuts, in-stream wood and pools with surface area larger than 1 $m^2$ and depth of at least 0.20 m. In areas where dense willow thickets were present, the channel width was measured using a flexible tape. Cross-sectional surveys were also conducted at five established transects (Figure 2) and detailed the thalweg, channel bed, banks and floodplain terraces.

LiDAR data and RGB aerial imagery were collected using an unoccupied aerial system in fall 2021. Data were post-processed to generate DEMs and orthomosaics gridded to 0.50 m resolution. Topographic survey data were integrated with the LiDAR-derived DEM to create a modified DEM that included dry and submerged surface elevations using R (R Core Team 2021) and the *sf* package (Pebesma and Bivand 2023; Pebesma 2018). Channel depths were measured as the difference between the meadow surface elevation and thalweg. Channel widths were identified from topographic bank data, field measurements of channel width and where channel width was greater than 1 m, estimated from RGB imagery. The DEM data along the channels were then modified using the channel depths and widths assuming a rectangular channel cross-sectional shape. In cases where the LiDAR-derived DEM did not accurately capture channel topography, stream channel elevations were smoothed so that step changes in the thalweg profile did not exceed 0.1 m.

Pre- and post-restoration meshes were developed for the meadow. 1-m wide triangular elements were used for the meadow channel and element size increased gradually to 20-m wide near the meadow's edge. The elements were sized to capture changes in channel conditions and the mesh was visually checked to ensure it followed land surface contours. The surveyed left and right banks were used as channel breaklines. The pre-restoration and post-restoration meshes consisted of 91 554 elements and nodes of the pre-restoration mesh at the structure locations were modified to account for the structure height. Structures were added iteratively to the model simulation and monitoring line output was verified to ensure flow continuity. Structure simulation was constrained by the triangular element geometry (1-m resolution) and the limitations of the model mesh. Structures were simulated as solid features and structure permeability was not accounted for. Calibration data to inform structure representation were not obtained during the period of study.

Pre- and post-restoration simulations were run for $Q_2$ (1.2 $m^3 s^{-1}$) and half the 2-year discharge ($Q_2/2$) (0.6 $m^3 s^{-1}$) assuming steady and subcritical flow. Both simulations represent runoff dynamics associated with storm events and spring thaw. Streamflow exceeded $Q_2$ on 10–13 April, 16 April and 13 June 2023 and exceeded $Q_2/2$ 8–28 April and 13–14 June 2023. We modelled the higher flows to simulate conditions that are difficult to observe in person due to their relative rarity and the remote site location. The inlet boundary condition was the upstream extent of the topographic survey, which was 27 m downstream from the upper gaging station. Water surface at the outlet boundary was calculated based on channel slope and composite Manning's $n$. Difference maps showing changes from pre-restoration to post-restoration were developed for water depth and velocity for $Q_2$ and $Q_2/2$ and used to assess restoration responses. Wetted surface area was computed from predicted depth output for the pre- and post-restoration scenarios using the modified DEM.

## 3 | Results

### 3.1 | Groundwater

The meadow groundwater system was highly dynamic, responded on the timescale of individual storms and appeared tightly coupled to streamflow. Prior to restoration, groundwater recession was rapid following the cessation of wet season storms and the spring snowmelt period, with groundwater levels dropping below the maximum well depth in some boreholes by mid-summer. We saw a rapid increase in groundwater levels during restoration across most of the 13 wells (Figure 4), with a mean increase in water level of 32 cm between 30 August and 20 September 2022 (Table 1).

While four wells demonstrated a rapid and sustained increase in groundwater table elevation (MID30-L03, MID30-R55, MID50-R15 and MID70-R10), others demonstrated a smaller increase which was sustained (MID50-L10, MID90-L32 and MID90-R05) (Table 1). Wells with the largest increases were generally closer to the stream or closer to restoration structures with significant retention of surface water. We did not observe surface water pouring into dry wells and recharging the unsaturated zone; instead, the existing water table surface smoothly rose in the days and weeks following restoration, likely due to infiltration of water from in-stream retention structures into the meadow aquifer. There was a large range in responses, from a 115.7 cm increase in MID70-R10 to a 1.7 cm decrease in MID70-L15 (Table 1). In the case of MID70-L15, the rate of groundwater level decrease appeared to have slowed as compared with the drawdown period prior to restoration (Figure 4). There was no clear difference in groundwater response to restoration between Reach 1 and Reach 2; both had wells that were very responsive (e.g., MID30-L03 with 62.8 cm of change and MID70R-10 with 115.7 cm of change respectively) and some that were less responsive (e.g., MID50-L10 with approximately 6 cm of change in Reach 1 and MID70-L15 with an approximate 1.7 cm decrease in Reach 2).

### 3.2 | Stream Discharge

The restoration occurred near the end of a multi-year drought. The 2022 water year precipitation at the Antelope Reservoir gage, located 7.3 km northeast of the outlet gaging station, was only 84% of the 1991–2021 mean value (594 mm) (California Data Exchange Center 2024). In contrast, the winter and spring of 2022–2023 were wetter than normal, and the total

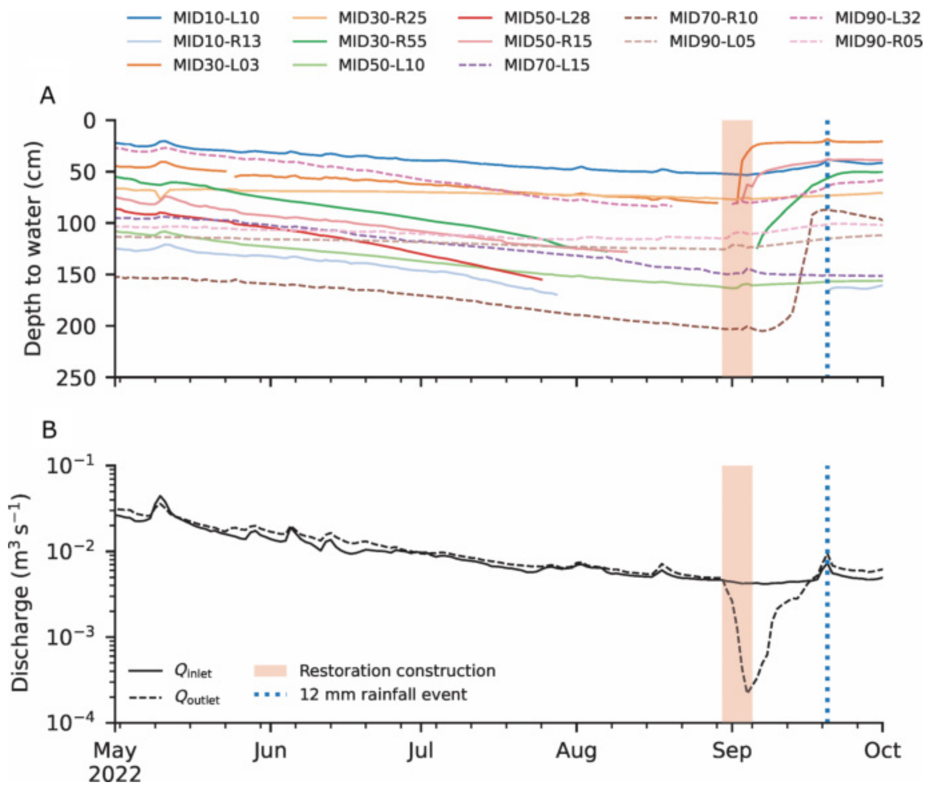


**FIGURE 4** | Groundwater depths (A) and streamflow for the upper ($Q_{inlet}$) and lower ($Q_{outlet}$) gaging stations (B) before, during (red shading) and after restoration. Groundwater wells in (A) are plotted with dashed lines for Reach 1 and solid lines for Reach 2. Groundwater data gaps occurred when water levels dropped below the pressure transducer. Well labels describe their position, with the first two numbers representing the transect (10%, 30%, 50%, 70% or 90% of meadow area), the L or R indicating to the left or right of the channel looking downstream, and the last two digits representing the distance from the left or right bank in metres (e.g., MID10-L10 is on the 10% transect, 10 m from the stream's left bank). Transects 10, 30, and 50 were in Reach 1, and transects 70 and 90 were in Reach 2.

precipitation for the 2023 water year was 1002 mm (169% of normal). Water detained upstream of structures following restoration resulted in a temporary but significant decrease in discharge at the meadow outlet gage (Figure 4), presumably due to increases in meadow storage. Over the restoration period, stream discharge at the meadow inlet decreased by a factor of 1.3, versus a decrease by a factor of 15.6 at the meadow outlet. The 'no restoration' power-law regression model of outlet discharge based on inlet discharge (Equation 2) resulted in an $R^2$ of 0.94 with $g = 1.71$ and $f = 1.09$. Relative to this modelled outlet discharge, we observed a decrease in discharge at the meadow's outlet that resulted in an estimated additional storage ($V_w$) of approximately 3701 m$^3$ (Figure 5). Surface detention storage ($V_s$) estimated via the DEM filling method equalled 542 m$^3$. Storage in the meadow groundwater system ($V_g$) was 3159 m$^3$, and the stream adjusted rapidly to altered channel conditions, with streamflow returning to expected values within several days of the completion of the restoration (Figure 5). With a mean groundwater level increase of 32 cm over the meadow area of 60 702 m$^2$ less the new inundated surface area of 2019.5 m$^2$, we calculated a meadow-average drainable porosity of $V_g/((60\,702–2019.5)\times 0.32) = 0.17$.

Subsequent to restoration, the mean depth to groundwater across the meadow (Figure 6) varied across a range of dry-seaon discharges. The mixed-effects model indicated a strong and significant dependence of depth to groundwater on the logarithm of discharge ($p < 0.0001$), with a significant difference in intercepts between the pre- and post-restoration groups ($p < 0.0001$). The difference between the random intercepts indicated an average difference in groundwater levels across the observed range of discharges equal to 18.7 cm.

### 3.3 | Sediment

During the relatively wet winter and spring 2023, a large amount of fine sediment was captured in pools (Figure 7). Over half of the total pool volume (~90 m$^3$) in the 10 sampled pools was filled with fine sediment (56 m$^3$). $V^*$ in the nine structure-formed pools ranged from 0.39 to 0.85 (mean = 0.65) whereas the $V^*$ for the one natural pool was 0.51 (Table 2). Reach 1 and Reach 2 structure-formed pool volumes were similar, with mean pool volumes of 9.5 and 9.0 m$^3$, respectively. However, in general, structure-formed pools in Reach 1 captured less sediment (mean $V^* = 0.56$ and mean stored sediment volume of 5.0 m$^3$) than in Reach 2 (mean $V^* = 0.75$ and mean stored sediment of 7.0 m$^3$) (Table 2).

### 3.4 | Hydrodynamic Modelling

The hydrodynamic modelling at the $Q_2/2$ discharge resulted in consistent decreases in surface water velocity and increases in water depth in the primary channel through the meadow following restoration due to pooling upstream of structures. Model results suggest the meadow restoration resulted in 17% more wetted area (2.12 ha total) as compared with the unrestored scenario (1.81 ha) as the structures increased pool area and created overbank flooding. By increasing floodplain activation, the restoration treatment activated approximately 320 m of relict flow paths within Reach 1. These modelled results were generally supported by field measurements following restoration: we measured a 20% increase in channel length through the meadow and 307 m of newly activated channels and four new channel difluences.

**TABLE 1** | Post-restoration groundwater elevation change between 30 August and 20 September 2022 in the 13 wells.

| Well | Change in groundwater elevation (cm) |
|---|---|
| MID10-L10 | 13.7 |
| MID10-R13 | 6.5 |
| MID30-L03 | 62.8 |
| MID30-R25 | 2.8 |
| MID30-R55 | 74 |
| MID50-L10 | 5.6 |
| MID50-L28 | −0.3 |
| MID50-R15 | 93.1 |
| MID70-L15 | −1.7 |
| MID70-R10 | 115.7 |
| MID90-L05 | 10.4 |
| MID90-L32 | 20.5 |
| MID90-R05 | 13.7 |

The stream velocity decreased substantially in the meadow as a result of the restoration. In Reach 1, modelled velocities commonly decreased by 0.8 m s$^{-1}$ just upstream of the structures (Figure 8). Additionally, discharge was diverted onto the floodplain where activation of flow paths that were not occupied prior to restoration resulted in modelled velocity increases from 0 to 0.8 m s$^{-1}$ (Figure 8). The instream structures in Reach 2 also resulted in decreased velocities upstream of the structures, but there were only minimal increases in wetted area and overbank flooding. The primary modelled change observed in Reach 2 was an increase in water depth: predicted water surface elevations increased at cross sections 70 and 90 by 0.35 and 0.74 m, respectively, for the post-restoration scenario compared with the pre-restoration scenario (Figure 9).

Hydrodynamic modelling at the $Q_2$ flow rate (1.2 m$^3$ s$^{-1}$) similarly resulted in decreases in surface water velocity and increases in water depth in the primary meadow channel following restoration due to pooling upstream of structures. For the most part, patterns observed between Reach 1 and Reach 2 were similar to the $Q_2/2$ condition in which overbank flooding primarily occurred in Reach 1 and flow remained laterally confined in Reach 2. For the $Q_2$ flow rate, restoration structures were predicted to add 11% additional wetted area across the meadow (2.61 ha total) compared with pre-restoration conditions (2.34 ha). We did not observe stream or wetted area conditions at flow rates near $Q_2$.

## 4 | Discussion

### 4.1 | Short-Term Effects

This study tracked changes in hydrological processes immediately following the implementation of process-based restoration at a degraded meadow in the northern Sierra Nevada. The instream structures impounded water and sediment, raised groundwater elevations and increased hydrologic complexity as measured by increases in wetted area and stream channel

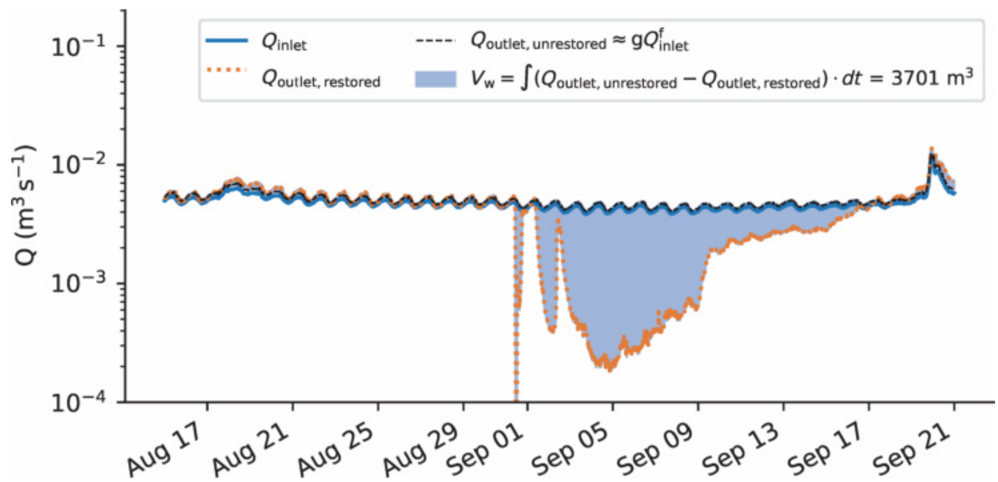


**FIGURE 5** | Discharge during the restoration period, which began 30 August 2022. The black dashed line plots the outlet discharge ($Q_{\text{outlet,unrestored}}$) predicted from the power-law equation derived during the pre-restoration period, and represents the counterfactual discharge that would have occurred without restoration. The inlet gage discharge (blue trace) was unaffected by restoration activities, whereas the actual outlet gage discharge ($Q_{\text{outlet,restored}}$; orange trace) dropped significantly below the counterfactual discharge during and immediately after restoration. The integrated difference between the $Q_{\text{outlet,unrestored}}$ curve and the $Q_{\text{outlet,restored}}$ (blue shading) provided an estimate for the increase in meadow water storage due to restoration.

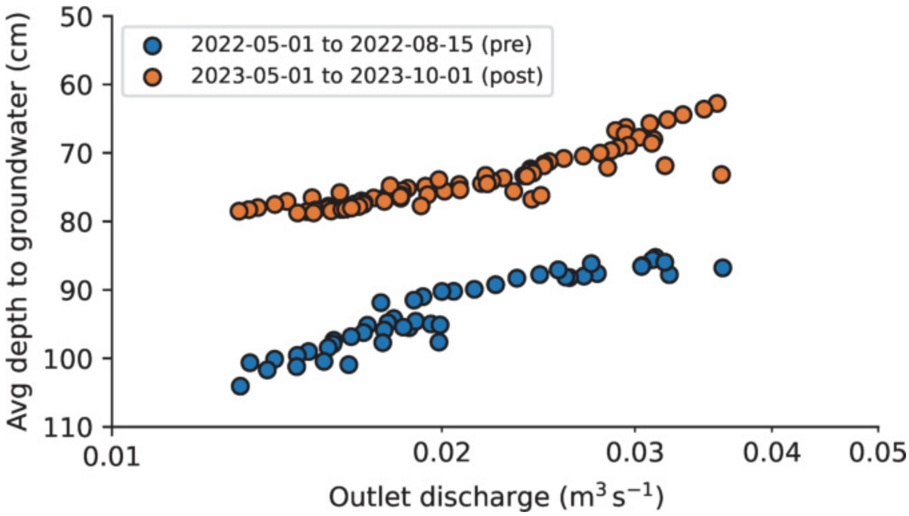


**FIGURE 6** | Average depth to groundwater across all instrumented meadow wells before (blue) and after restoration (orange) as a function of shared outlet discharge (log scale). The 2023-shared discharge values were later in the summer because of the wetter winter and longer recession limb and possibly the influence of the higher groundwater elevations on baseflow.

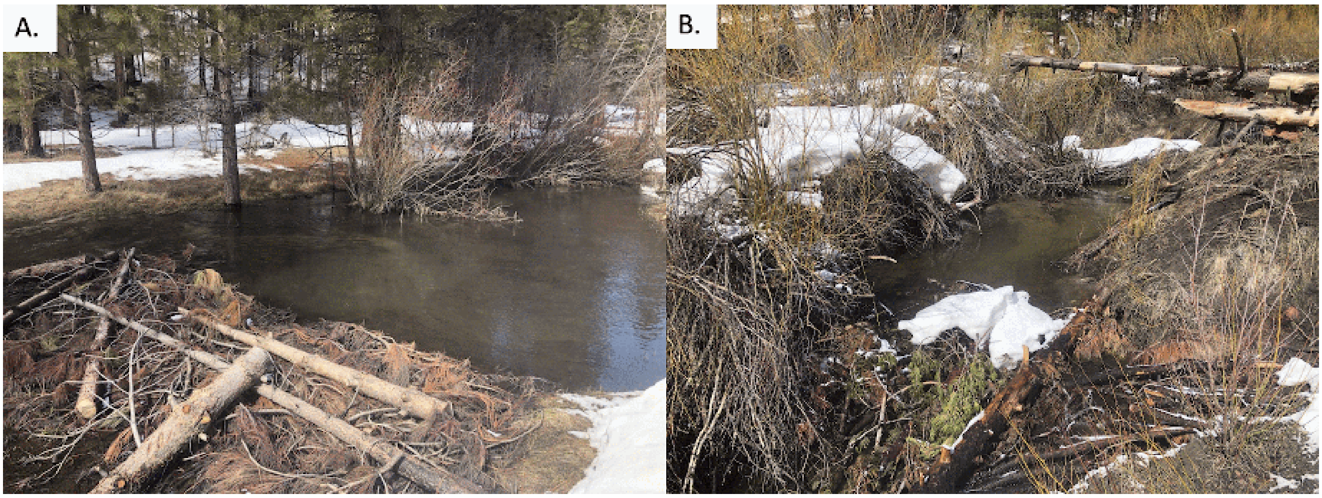


**FIGURE 7** | (A) Beaver dam analog (BDA) in Reach 1 with the top of the dam at the same elevation as the top of channel banks. (B) A BDA in Reach 2 with the top of the dam below incised channel banks. Both structures captured sediment following the first winter post-restoration. Photo credit: Patrick Jarrett, California Department of Water Resources.

length. Together, these short-term monitoring results show that the process-based restoration applied in Middle Creek Meadow initiated hydrologic changes necessary to activate longer term meadow recovery processes, such as increasing the area of wet meadow vegetation (Nash et al. 2021). The restoration design includes plans to continue encouraging these recovery processes through periodic structure assessment, modification and supplementation until the structures are integrated into the meadow ecosystem through sedimentation and vegetative growth, making them secure under high flows.

By diverting flow from the incised channel onto the meadow floodplain, increasing water depth upstream of in-channel restoration structures, and reducing water velocity through the incised channel, the structures increased the meadow's groundwater storage. Coincident with the rapid storage gains, we saw a distinct temporary reduction in discharge at the lower stream gage during and following restoration. Through hydrograph analysis, we estimated the immediate gains in surface and subsurface water storage due to restoration as 3701 m$^3$. This volume calculation along with an estimate of surface detention storage also enabled the estimation of the meadow aquifer's drainable porosity, a key but uncertain parameter in groundwater modelling (e.g., Nash et al. 2018). Typically, porosity estimates are obtained through measurement of soil water retention properties from core samples. Such point-scale estimates may not accurately represent larger areas. The presented mass-balance method provides a meadow-scale estimate of drainable porosity, which is potentially more useful for groundwater modelling and related base flow analyses.

Before restoration, the meadow supported two distinct reaches where the upstream Reach 1 was less incised than the downstream Reach 2. These differences led to different goals for restoration and different hydrologic responses to the restoration. In Reach 1, structures were designed to divert streamflow to reactivate relic

**TABLE 2** | Volumes of residual (base flow) water and fine sediment stored in 10 pools (5 in Reach 1 and 5 in Reach 2) in Middle Creek Meadow.

| Pool[a] | Reach | Water vol. ($m^3$) | Sediment vol. ($m^3$) | Total vol. ($m^3$) | $V^*$ |
|---|---|---|---|---|---|
| N1 | 2 | 2.94 | 3.09 | 6.03 | 0.51 |
| S2 | 2 | 3.24 | 18.54 | 21.77 | 0.85 |
| S3 | 2 | 4.01 | 5.88 | 9.89 | 0.59 |
| S4 | 2 | 0.56 | 2.01 | 2.57 | 0.78 |
| S5 | 2 | 0.4 | 1.42 | 1.81 | 0.78 |
| S6 | 1 | 9.2 | 9.9 | 19.1 | 0.52 |
| S7 | 1 | 2.34 | 3.04 | 5.39 | 0.57 |
| S8 | 1 | 7.37 | 4.67 | 12.03 | 0.39 |
| S9 | 1 | 1.61 | 3.28 | 4.88 | 0.67 |
| S10 | 1 | 2.03 | 4.01 | 6.05 | 0.66 |

*Note:* $V^*$ represents the fraction of the total pool volume occupied by fine sediment.
[a]N = natural pool, S = structure-formed pool.

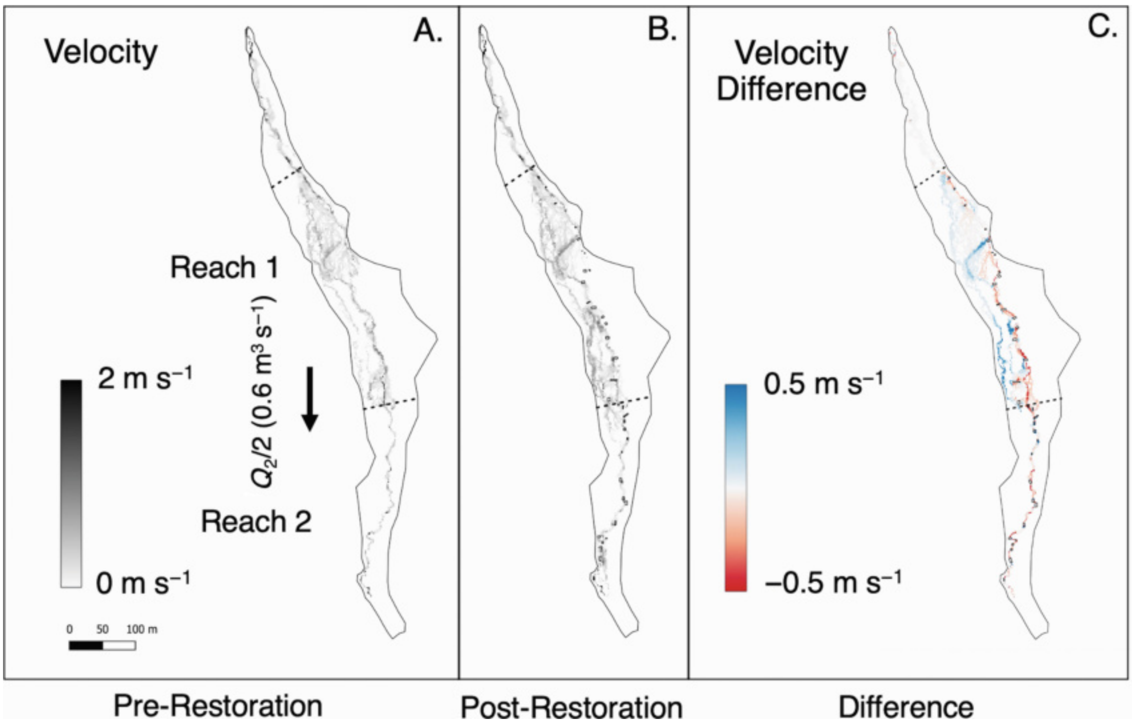


**FIGURE 8** | Predicted velocities for $Q_2/2$ ($0.6\,m^3\,s^{-1}$) for (A) the pre-restoration scenario, (B) the post-restoration scenario using the same scale bar as (A), and (C) the difference in velocity between the restored scenario and the pre-restoration scenario. Dashed black lines indicate the upstream extent of each reach and black outlines show locations of restoration structures.

channels and flow paths on the meadow surface and resulted in the development of multiple low velocity, sinuous channels and large areas of floodplain surface inundation. This was possible because structures could be built tall enough during the initial restoration period to reconnect streamflow with the floodplain (Figure 7).

In Reach 2, the channel depth was much greater than flow depths under observed flow conditions. Restoration in Reach 2 focused on building structures that would withstand high flows in the first year and reconnect with the floodplain in a phased approach over a few years (Pollock et al. 2014; Wheaton et al. 2019). The goals in Reach 2 for the first year were to increase sediment deposition and channel bed elevation within the channel and water storage upstream of structures and in the groundwater in the surrounding meadow. The treatment met these initial goals after one winter and spring, suggesting that the multi-year phased approach may lead to regaining access to the floodplain in Reach 2.

The relatively wet winter of 2022–2023 likely caused higher sediment inputs and delivery than most years, particularly given the burned condition of the watershed (Wagenbrenner and Robichaud 2014; Cole et al. 2020). This observation supports the hypothesis that the time required for post-restoration

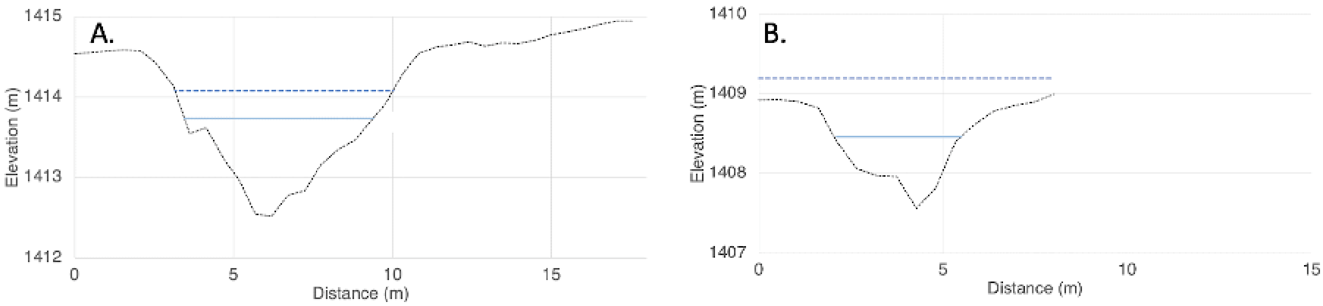


**FIGURE 9** | Predicted water surface elevations for $Q_2/2$ ($0.6\,m^3\,s^{-1}$) for the pre-restoration scenario (solid light blue line) and the post-restoration scenario (dashed dark blue line) using the modified DEM (dashed black line) as input topography for (A) cross section 70 and (B) cross section 90, both located in Reach 2. The modified DEM includes dry and submerged surface elevations.

recovery of deeply incised meadow channels may be reduced in recently burned catchments due to the potential for elevated sediment delivery rates (Pope and Cummings 2023). The increased depositional potential of meadows in recently burned watersheds may serve as a form of mitigation of enhanced post-fire downstream sediment delivery (Moody and Martin 2009; Robichaud, Wagenbrenner, and Brown 2010; Moody et al. 2013) by capturing the sediment where it can be useful and reducing downstream sediment delivery where it can be a problem for aquatic life and human infrastructure, such as reservoirs.

We collected sub-daily discharge and groundwater measurements before, during and after the restoration treatment to assess the relationship between flux and storage of water in the meadow. Groundwater elevations were raised by an average of 32 cm in the meadow almost immediately following restoration, likely as a result of temporary water loss from the stream into the meadow aquifer (Nash et al. 2021; Pearce et al. 2021). Mixed-effects statistical modelling also indicated a longer term increase in groundwater levels of 18 cm across a range of discharges typically observed during low flow conditions (Figure 6). Outlet discharge integrates changes in precipitation and demonstrates that there is a significant difference in depth to groundwater pre- and post-restoration despite elevated precipitation in winter 2023.

Our groundwater results support findings from other studies (e.g., Bouwes et al. 2016; Pearce et al. 2021), but tend to show greater gains over a greater floodplain area than most recent BDA studies (Munir and Westbrook 2021; Pearce et al. 2021; Scamardo and Wohl 2020). Differences among our and previous findings are likely due to restoration design whereby Middle Creek received a higher density of large structures compared with other restoration projects (e.g., 3.8 structures/100 m in Middle Creek versus 0.6/100 m in Pine Creek [Munir and Westbrook 2021]), and specific stream conditions of the studies. The installation of BDAs in these other studies occurred along stream reaches not associated with meadows where coarse valley substrate may have been unable to hold water for long periods (Burns and McDonnell 1998) or where clay strata prevented connection to floodplain aquifers (Scamardo and Wohl 2020). Middle Creek Meadow is a riparian meadow, a geomorphic type known to have a strong hydrologic connection between the stream channel and floodplain groundwater (Loheide II and Gorelick 2007; Weixelman et al. 2011) where one would expect site-specific conditions that support a strong response of shallow groundwater to changes in stream flow dynamics. Factors pertaining to BDA construction including density, size, intended function, construction materials, overall restoration design and quality of construction may additionally influence site response.

The modelling allowed us to predict the effects of the restoration structures on complex surface water flow patterns at flow conditions that were not visually observed during our study at the meadow scale. For example, we calculated increases in wetted meadow surface area of approximately 17% following restoration at the $Q_2/2$ flow rate with increases in wetted channel length, channel complexity and altered flow velocity. Over time, these more diverse hydraulic conditions may enhance the quality of aquatic habitat (Flitcroft et al. 2022). The effects of restoration structures on hydraulic conditions and aquatic habitat are controlled in part by channel morphology, where the channel depth determines whether a structure and flow condition may simply increase flow depth within the incised channel or create depths of a magnitude capable of creating lateral floodplain connection. For the post-restoration simulation, overbank flooding was more frequent in the less incised Reach 1, which supported the re-activation of multiple low velocity, sinuous channels. This finding was also observed in the field at low flow conditions. We anticipate that the reductions in flow velocity predicted through the highly incised Reach 2 will continue to promote aggradation, resulting in gains in channel bed elevation over time (Pollock et al. 2014). Ultimately, our modelling suggested that the restoration treatment initiated geomorphic processes that will support longer term ecosystem recovery (Ciotti et al. 2021).

## 4.2 | Longer Term Effects

This study focused on the short-term responses of the stream and groundwater conditions after the restoration structures were installed. The longer term effects, including physical and ecological changes over time, have not yet been determined. The restoration team will continue to use changes that occur in the meadow and watershed to enlarge and expand restoration structures (Ciotti et al. 2021). We will also continue to monitor the hydrological and geomorphic conditions, and assess ecological changes, to determine if meeting the initial restoration goals resulted in longer term ecosystem service benefits to water quality, summer base flow and carbon sequestration. We expect that wet meadow vegetation productivity will increase due to the increased water table elevations and hydrological connectivity (Hammersmark, Rains, and Mount 2008; Silverman et al. 2019).

A continued trajectory toward restoration will depend on the persistence of geomorphic and biological processes (e.g., sediment deposition, overbank flooding, vegetative growth) that are influenced by evolving restoration structures over time (Pollock et al. 2014; Silverman et al. 2019; Ciotti et al. 2021). More extreme precipitation and increased wildfire extent, which are expected in northern California as a result of climate change, may additionally impact outcomes of process-based meadow restoration (Berg and Hall 2015; Williams et al. 2019). Structures can be modified, repaired or enlarged following these expected changes. Structure failure and a reversion to pre-treatment conditions could occur if structures are not appropriately scaled or stabilised to withstand winter flows or if structures are poorly built (e.g., not packed properly at the streambed). Assuming quality restoration work, regardless of the restoration technique, long-term chances of meeting restoration goals improve with attentiveness to understanding system needs and adjusting accordingly (Moore and Rutherfurd 2017).

Disturbances such as intensive livestock grazing could slow meadow recovery by compacting soils, stunting root growth, destabilising banks, and damaging structures (Bohn and Buckhouse 1985; Wheeler et al. 2002; Clary 1995; Vernon, Campos, and Burnett 2022). As within most meadows on National Forest System lands, grazing is permitted in Middle Creek Meadow. Understanding the influence of grazing on meadow recovery following process-based restoration is an important interaction that is not addressed in the current study. A recent review of the literature on the effects of livestock grazing on Sierra Nevada meadows concluded that restoring functional ecological conditions in meadows used by livestock may be challenging but possible in certain conditions and management regimes (Vernon, Campos, and Burnett 2022). We will continue to monitor livestock use and effects on hydrological and ecological responses to treatment. The long-term stability and success of the restoration, particularly in the more incised Reach 2, will depend on whether the balance of restoration and disturbance processes weighs toward structure decay, release of sediment, and channel degradation or structure and channel stabilisation, sediment capture, continued channel aggradation, and expanded streamflow-meadow surface connectivity.

## 5 | Conclusions

Through intensive short-term monitoring, we showed how meadow restoration using low tech, process-based approaches can increase floodplain connectivity, increase stream complexity, raise groundwater elevation and capture sediment. We explained a short-term loss in discharge at the outlet of the meadow by showing a commensurate and rapid increase in groundwater storage following the installation of restoration structures that remained into the following year's low flow season. These rapid, positive hydrologic and geomorphic responses were likely due to a combination of planned and unplanned but fortuitous factors including (1) working in a groundwater dependent riparian meadow where paired surface and groundwater fluxes are expected, (2) applying a restoration design tailored to specific channel and flow conditions, (3) working in a recently burned catchment with high expected sediment delivery rates and (4) experiencing a wetter than average winter and spring that delivered the stream energy and sediment needed for rapid change. Sediment aggradation during the first post-restoration snowmelt helped stabilise the structures and will allow for continued expansion of the structures in the lower, more incised reach over the next 2 or 3 years as intended by the design team.

These results demonstrate the immediate short-term effects of process-based restoration in a degraded meadow in the Sierra Nevada. Though longer term monitoring is needed to determine the effects of the restoration on seasonal low flows, vegetation, biota and durability of the structures, the initial change in hydrologic functioning offers promise for longer term benefits. Our results highlight how a properly applied, low-impact approach conducted by a professional crew in only 6 days can produce rapid and foundational hydrological changes in a degraded meadow.

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### Conflicts of Interest

The authors declare no conflicts of interest.

### Data Availability Statement

Data, model files, and python and R scripts can be requested from the corresponding author. Additional materials may be accessed via the following HydroShare repository: Sevier et al. (2024).

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