

Temporal changes in soil water retention in burned and unburned areas after a wildfire and implications for slope stability

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Abstract

Temporal changes in the hydrologic and mechanical behavior of burned slopes result in changes in post-wildfire slope stability. The lack of long-term monitoring data on landscape recovery hinders our understanding of the interdependencies between wildfire-induced alterations in soil and vegetation properties, near-surface processes, and landslides. This study presents results from a 4-year field monitoring project of soil water retention and the corresponding long-term changes in landslide susceptibility after the 2019 Williams Flats Wildfire on the Colville Indian Reservation, WA, USA. Water retention was measured near both a burned and an unburned tree at multiple depths. The measured data were integrated into a suction stress-based infinite slope stability analysis to determine the corresponding temporal changes in shallow landslide susceptibility. The results showed that the lowest factor of safety (FS) near the burned tree was in June of the first year, which was attributed to macropores acting as preferential flow paths. The FS near the burned tree increased in the following years, which was attributed to vegetation regrowth. Vegetation recovery was quantified using normalized difference vegetation index mean values and soil temperature data. Field observations supported contributions of the macropore clogging on temporal changes in slope stability. The results suggest a dynamic interaction of vegetation recovery and soil physical changes and discuss the challenges of quantifying short- and long-term post-wildfire slope stability.

Key words: wildfire, water retention, shallow landslide, vegetation recovery

Résumé

Les changements temporels dans le comportement hydrologique et mécanique des pentes brûlées entraînent des modifications de la stabilité des pentes après un incendie de forêt. Le manque de données de surveillance à long terme sur la récupération des paysages entrave notre compréhension des interdépendances entre les altérations induites par les incendies de forêt dans les propriétés du sol et de la végétation, les processus de surface proche et les glissements de terrain. Cette étude présente les résultats d'un projet de surveillance sur le terrain de quatre ans sur la rétention d'eau du sol et les changements à long terme correspondants dans la susceptibilité aux glissements de terrain après l'incendie de forêt de Williams Flats en 2019 sur la réserve indienne de Colville, WA, États-Unis. La rétention d'eau a été mesurée près d'un arbre brûlé et d'un arbre non brûlé à plusieurs profondeurs. Les données mesurées ont été intégrées dans une analyse de stabilité de pente infinie basée sur la contrainte de succion pour déterminer les changements temporels correspondants dans la susceptibilité aux glissements de terrain superficiels. Les résultats ont montré que le facteur de sécurité (FS) le plus bas près de l'arbre brûlé était en juin de la première année, ce qui a été attribué aux macropores agissant comme des voies de circulation préférentielles de l'eau. Le FS près de l'arbre brûlé a augmenté au cours des années suivantes, ce qui a été attribué à la repousse de la végétation. La récupération de la végétation a été quantifiée en utilisant les valeurs moyennes de l'indice de végétation par différence normalisée et les données de température du sol. Les observations sur le terrain ont soutenu l'impact du colmatage des macropores sur les changements temporels de la stabilité des pentes. Les résultats suggèrent une interaction dynamique entre la récupération de la végétation et les changements physiques du sol et discutent des défis de la quantification de la stabilité des pentes à court et à long terme après un incendie de forêt.

Mots-clés : incendie de forêt, rétention d'eau, glissement de terrain superficiel, récupération de la végétation

Introduction

Wildfires are natural processes that have occurred on the planet for millions of years and affect most ecosystems and economies (Bixby et al. 2015; McLauchlan et al. 2020). According to the National Interagency Fire Center database (<https://www.nifc.gov/>), the area burned by wildfires between 2000 and 2022 is equivalent to 7.2% of the total land area of the U.S., and the suppression cost of wildfires just in 2022 was \$3.5 billion. In several regions, such as the Western U.S. and Australia, wildfire activities have increased over the past few decades, partly attributed to anthropogenic climate change (e.g., Abatzoglou and Williams 2016; Jones et al. 2022). The effects of wildfires are not only limited to the initial burn of vegetation and the associated suppression costs, but cascading effects of wildfires also pose a threat to communities as well as the natural and built environments (Parise and Cannon 2012; Vahedifard et al. 2024). Wildfires can alter the mechanical, hydraulic, chemical, biological, and thermal properties of surface and near-surface soils (Certini 2005; Moody et al. 2016). Further, wildfires also affect land cover and near-surface processes such as evapotranspiration, interception, vegetation cover and root vitality, sediment transport, erosion, infiltration, and runoff. These wildfire-induced changes in soil and land cover properties, coupled with the associated near-surface processes, can trigger a cascade of geohazards, including dry raveling, erosion, rockfalls, landslides, debris flows, and land subsidence (Akin et al. 2023; Vahedifard et al. 2024). Even though the immediate effects of wildfires on soil properties are documented, temporal effects are still not fully known or cannot be modeled.

Several studies have investigated post-wildfire surficial runoff-dominated erosion and associated debris flows (e.g., Parise and Cannon 2012; De Graff et al. 2015; De Graff 2018; Graber et al. 2023). However, the existing research on post-wildfire shallow landslide susceptibility is limited. Post-wildfire landslides can take place at varying times after a wildfire, and associating shallow landslide initiation with wildfires can be difficult (e.g., Meyer et al. 2001; Rengers et al. 2020; Lei et al. 2022). As a result, temporal changes in landslide susceptibility due to changes in soil water retention and vegetation of a burned slope are not fully known. The research efforts in the field of post-wildfire landslides have been concentrated on documenting shallow landslides and identifying the common trends (e.g., Rengers et al. 2020) and modeling post-wildfire landslide susceptibility (e.g., Abdollahi et al. 2023, 2024). Although these efforts are valuable, a wildfire-specific slope stability model cannot be developed without a complete understanding of the temporal changes in forest soil hydromechanical behavior, including temporal changes in soil water retention.

Wildfire has immediate and long-term effects on hillslopes. The combustion of above-ground biomass, intense soil drying, reduced soil porosity and permeability, and the formation or enhancement of a hydrophobic layer on the soil surface all decrease infiltration and increase surface runoff (Ebel and Moody 2020). Depending on the availability of

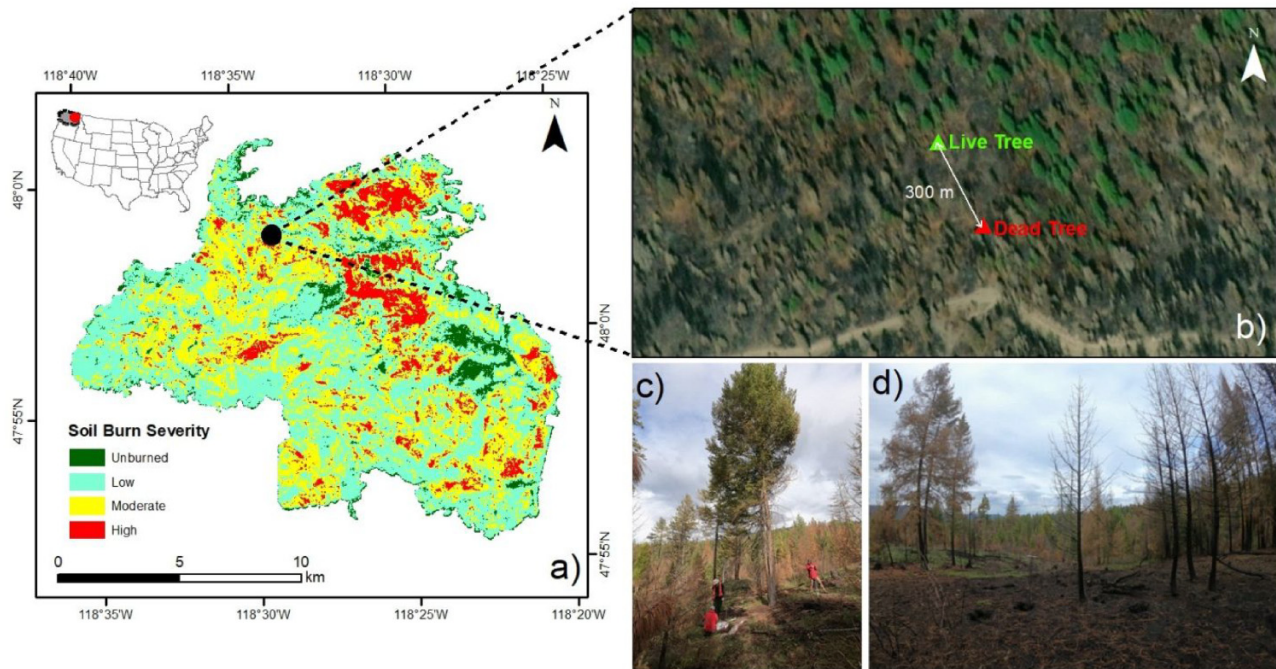
sediment, debris, the geologic and topographic characteristics of the burned area, this increased runoff can result in catastrophic debris flows shortly after the fire (Rengers et al. 2016). However, as the area transitions through the recovery phase and soil infiltration capacity is restored, mass wasting tends to shift from runoff-driven events like debris flows to infiltration-triggered shallow landslides (Rengers et al. 2020). For instance, in January 2019, a severe rainstorm caused widespread landslides across a 70 km² area of the San Gabriel Mountains in Southern California. The impacted areas had experienced wildfires within the previous 3–10 years, including the 2009 Morris Fire 2014 Colby Fire, and 2016 San Gabriel Complex Fire (Fish Fire and Reservoir Fire). Notably, over 90% of these landslides were within the boundaries of the San Gabriel Complex Fire (Rengers 2020).

Wildfires can heighten the risk of shallow landslides by weakening root structures, lowering evapotranspiration rates, modifying vegetation cover and canopy interception, and altering the soil's mechanical and hydraulic properties (Rengers et al. 2020; Akin et al. 2023; Vahedifard et al. 2024). The immediate combustion of roots results in an increase in landslide susceptibility because of (i) loss of root cohesion (e.g., Wondzell and King 2003; Lei et al. 2022) and (ii) reduction in suction stress with the loss of evapotranspiration (e.g., Ng et al. 2013; Akin et al. 2023). Recent studies also showed that macropores left behind by burned roots change the hydrology of the slope, acting as preferential pathways for water (e.g., Leslie et al. 2014) and wildfire ash which can affect short-term slope stability after a wildfire (e.g., Akin and Akinleye 2021; Akin et al. 2023). The long-term effects of the burn are associated with progressive root decay, which results in landslides taking place years after a wildfire (Gehring et al. 2019). In addition, temporal changes in post-wildfire landslide susceptibility are related to vegetation recovery as newly grown vegetation provides root cohesion and an increase in suction stress. Even though the general mechanisms that control short- and long-term post-wildfire slope stability were investigated in discrete studies, there is no long-term data on temporal changes in soil water retention or corresponding landslide susceptibility during the recovery period.

Despite the large number of studies performed on post-wildfire geohazards, based on a critical review of the literature, the central research gap is the lack of long-term understanding of how post-wildfire soil and vegetation dynamics influence landslide susceptibility, particularly through temporal changes in soil water retention. From this, the main scientific research questions are as follows:

- How do the mechanical and hydrological properties of soil, particularly soil water retention, evolve over time in wildfire-affected areas during the recovery period?
- What are the temporal changes in shallow landslide susceptibility in burned versus unburned slopes, and how are these changes influenced by the evolution of soil water retention and vegetation recovery?

Fig. 1. Study area: (a) soil burn severity map of the 2019 Williams Flats Fire (the black circle shows the study site); (b) locations of the dead and live trees in the study site (the image was acquired from ESRI's World Imagery—Wayback archive, 22 July 2020); photography of live tree (c) and dead tree (d).



- How can long-term, field-monitored soil suction, and saturation data be integrated with vegetation indices (e.g., normalized difference vegetation index (NDVI)) to better model and predict slope stability in post-wildfire environments?

To answer these questions and fill the research gap, this study presents results from over 4 years of field monitoring of soil water retention and the corresponding long-term changes in landslide susceptibility in an area burned by the 2019 Williams Flats Wildfire on the Colville Indian Reservation, WA, USA. The changes in the hydrologic behavior of the slope were monitored through in situ soil water content and suction sensors installed under a burned tree, and the results were compared with those measured near an unburned tree. The NDVI was used to evaluate the post-wildfire vegetation recovery, which was combined with the temperature data from field sensors to explain the yearly trends in soil water retention. The temporal changes in soil water retention were incorporated into an unsaturated infinite slope stability model to evaluate the long-term slope stability.

Methods and materials

Study site and field instrumentation

The Williams Flats Fire started on 2 August 2019, approximately 12 km southeast of Keller, Washington, on the Colville Indian Reservation, and burned 18 100 ha before it was contained on 25 August 2019. The Burn Area Emergency Response teams classified 1060 ha as unburned to low soil burn severity, 8800 ha as low soil burn severity, 6360 ha as moder-

ate soil burn severity, and 1840 ha as high soil burn severity areas, defined by Parsons et al. (2010; Fig. 1a). The burned vegetation was ponderosa pine (*Pinus ponderosa*), mixed conifer, light logging slash, brush and grass species, such as antelope bitterbrush (*Purshia tridentata*) and *Ceanothus*. The Colville Indian Reservation is located in northeastern Washington State and spans a diverse topography ranging from river valleys to mountainous terrain. The area has an arid to semi-arid climate with cold, snowy winters, and warm, dry summers (Bureau of Indian Affairs 2014). Geologically, the area consists of volcanic and metasedimentary bedrock overlain by a heterogeneous mixture of glacial till, colluvium, and volcanic ash derived soils (Pardee 1981). Soils at the site belong to the ash-derived Nevine Series, which is a well-drained sandy soil, underlain by glacial till bedrock (Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture 2016). The study site was selected from a high-burn severity area next to a moderately burn severity area. The surrounding waterbodies are the South Fork Ninemile Creek in the north and Brody Creek in the south, which drains into the Colombia River.

The site was instrumented with water content (TEROS 11) and matric suction (TEROS 21) sensors (Meter Group, Pullman, WA) on 2 October 2019, as described in Akin et al. (2023). In addition to matric suction or water content, both sensors measure soil temperature using thermistors. The range of matric suction sensors was between near-saturation (3–6 kPa) and 2000 kPa with 10% accuracy and the range of the water content sensors was 0 and 0.7 m³/m³ with approximately 3% accuracy. Two locations were selected for sensor instrumentation: one near a burned tree in the high soil burn severity

area, referred as the dead tree, and one near an unburned tree in the moderate soil burn severity area, 300 m away from the dead tree location, referred as the live tree (Fig. 1b). All the trees and understory vegetation at the dead tree location were burned, leaving wildfire ash visible on the soil surface, and macropores present near the base of the burned tree. Burned, partially burned, and unburned trees and understory vegetation were present at the live tree location, and no visible wildfire ash or macropores were found on the soil surface. An unburned tree with unburned understory vegetation was selected as the live tree location for sensor installation to provide the baseline data for comparison. The sensors were installed 2 m away from the tree base to avoid large roots during drilling for sensor placement while still capturing the changes in water retention caused by transpiration. The sensors were placed at 0.3, 0.5, and 1.0 m depths. The bedrock was at 1.2 m. Sensor depths were selected to cover the entire soil layer above the bedrock, while considering ease of installation. A hand auger was used to drill a 1.2 m deep borehole, and the sensors were placed on the hole starting from 1.0 m depth. The water content sensors have stainless steel rods that ensure a good contact between the sensor and the soil. The matric suction sensors are made of ceramic disks, requiring good contact with soil for accurate measurement. Soil collected from the targeted depth was molded and packed to the field void ratio in a plastic cup where the matric suction sensor was placed to ensure good contact between the sensor and soil. The molded soil and the matric suction sensor were extracted and then placed back into the borehole at the targeted depth. After installing the sensors at each depth, the original soil was carefully replaced to match the original soil layers, ensuring minimal disturbance to the soil profile. The sensors were connected to ZL6 data loggers (Meter Group, Pullman, WA), which recorded the data hourly.

In addition, a tipping bucket rain gauge was installed in May 2020, approximately located 20 m from the dead tree, to record site-specific rainfall. Snow accumulation at the study site was monitored using Landsat 8-9 satellite images acquired from the United States Geological Survey (USGS) EarthExplorer, providing a 16-day temporal resolution for tracking snow cover. The total snowfall was obtained from the nearest weather station, Coulee Dam 1 SW, located 35 km from the study site at an altitude of 524 m.

Soil characterization

Soil properties were determined by Akin and Akinleye (2021) for the samples collected in October 2019 from the surface and at 0.5 m depth around the dead tree. The tests included standard soil classification tests (ASTM D2487), field void ratio measurements, water vapor sorption (WVS) isotherms, and WVS-based specific surface area (SSA) measurements (Table 1). The WVS isotherms were measured as described in Akin and Likos (2014) using a fully automated vapor sorption analyzer (Meter Group, Pullman, WA) operated in dynamic dewpoint mode. Hysteretic sorption isotherms were obtained by increasing the relative humidity (RH) of the sample chamber from 3% to 95%, followed by a decrease back down to 3% in 1% increments while taking mass mea-

Table 1. Properties of surface and 0.5 m soil of the 2019 Williams Flats Fire for October 2019.

	Soil depth	
	Surface	0.5 m
USCS soil classification	SM	SM
Fine content passing #200 sieve (%)	47	31
Organic content (%)	9.3	2.8
Void ratio (e)	1.57	0.57
Specific surface area (m ² /g)	54	24
Specific surface area of October 2019 ash (m ² /g)	76	

Note: USCS, Unified Soil Classification System; SM, silty sand.

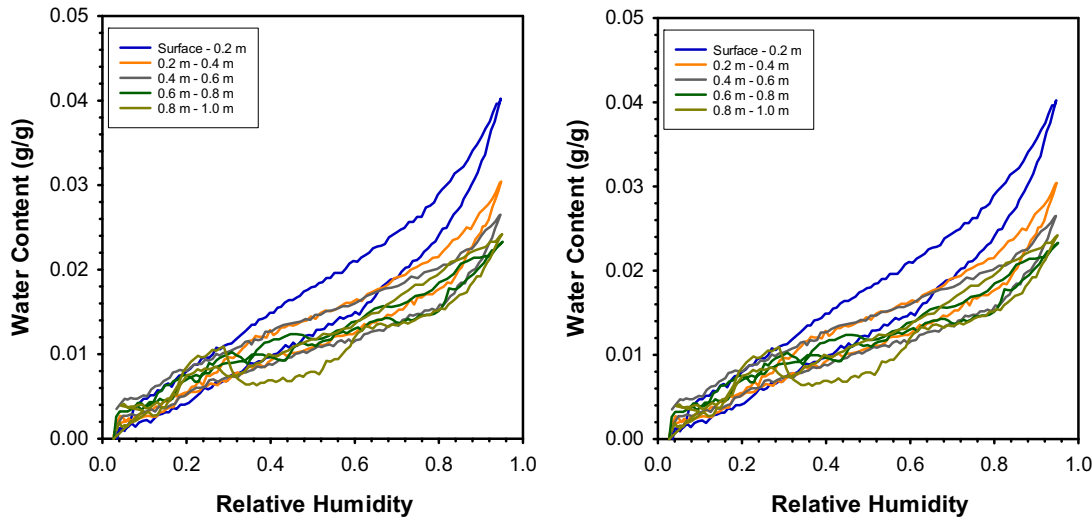
surements. The desorption curve was used to calculate SSA between 10% and 30% RH, which corresponds to monolayer adsorption (Akin and Likos 2020a). In addition, loss on ignition tests were conducted as described in Scalia et al. (2014), where the samples were kept in a 550 °C oven for 4 h, and the mass difference was reported as organic content.

The soil type was classified as silty sand at all depths according to the Unified Soil Classification System. However, the differences in organic content, fine content, and WVS behavior (Fig. 2) indicated that soils in the three sensor depths varied from one another. The representative depth of 0.5 m was selected to determine soil properties and conduct the subsequent slope stability analyses. The reason for selecting soil properties at 0.5 m depth for the slope stability analysis was to eliminate the effect of organic matter on the top layer. In addition, nearly identical WVS isotherms between 0.5 and 1.0 m indicated that soil properties remained relatively unchanged in this range. This depth selection aligns with the depths of post-wildfire shallow landslides reported in previous studies. For instance, Rengers (2020) indicated that the 2019 landslides in the San Gabriel Complex Fire area generally had depths hardly exceeding 0.5 m.

Direct shear tests were conducted using intact samples collected at 0.5 m depth, as described in Akin et al. (2023). A trench of 0.5 m depth was dug, and 6.3 cm diameter, 2.5 cm height sampling rings were used to collect the intact soil samples. In the laboratory, the intact samples were extruded, saturated, consolidated, and sheared with a 0.5 mm/min displacement. In addition, soil water retention curve (SWRC) and Brazilian tensile strength (BTS) tests were conducted. The SWRC was measured with a chilled-mirror dew point technique (WP4C, Meter Group, Pullman, WA) as described in Akin and Akinleye (2021), where specimens were compacted in stainless steel cups at predetermined degrees of saturation and a constant void ratio and tested for suction. The BTS tests were conducted on disk-shaped specimens with a 3 cm diameter and 1 cm thickness compacted using a custom-made mold and compressed between two curved (14°) bearing blocks using a small-scale setup, as described in Akin and Likos (2017).

Water droplet penetration time (WDPT) tests (Woudt 1959) were also conducted on the soil samples collected from the dead tree location at 0.5 m depth. The soil samples were collected, using 6.3 cm diameter and 2.5 cm height sampling rings and tested in the laboratory by placing 16 equally

Fig. 2. Water vapor sorption isotherms from surface to 1.0 m depth measured at 0.2 m intervals.



spaced deionized water droplets on the samples, as described in [Akin and Akinleye \(2021\)](#). The WDPT was recorded by measuring the average time required for each sample to absorb the droplets, and the water repellency was classified based on [King \(1981\)](#) and [Chenu et al. \(2000\)](#). [Akin et al. \(2023\)](#) found that the soil at a 0.5 m depth exhibited nonrepellent behavior, thereby eliminating any potential effects of hydrophobicity for this study.

Slope stability analysis

Landslide susceptibility was evaluated using the infinite slope stability equation ([eq. 1](#)) modified for unsaturated conditions, as proposed by [Lu and Godt \(2008\)](#). The infinite slope stability model was selected because of the shallow soil depth (1.2 m). Previous studies have shown that the translational failure represented by the infinite slope stability method can effectively model shallow landslides, including those influenced by wildfires (e.g., [Godt et al. 2012](#); [Santos et al. 2020](#); [Abdollahi et al. 2023, 2024](#); [Akin et al. 2023](#)).

$$(1) \quad FS(z_u) = \frac{\tan \phi'}{\tan \beta} + \frac{2C_r}{\gamma (H_{wt} - z_u) \sin 2\beta} - \frac{1}{\gamma (H_{wt} - z_u)} (\tan \beta + \cot \beta) \tan \phi'$$

where FS = factor of safety, z_u = vertical depth of failure surface from the water table in the unsaturated zone, ϕ' = friction angle, β = slope angle, C_r = root cohesion, γ = total unit weight, H_{wt} = vertical depth of water table from the ground surface, and σ^s = suction stress. The [Akin and Likos \(2020b\)](#) model ([eq. 2](#)) was used to calculate suction stress as follows:

$$(2a) \quad \sigma^s(\psi) = \sigma_a^s(\psi) + \sigma_c^s(\psi)$$

$$(2b) \quad \sigma_a^s = \left(\frac{\psi}{10^6} \right) \times \sigma_{dry}^s$$

$$(2c) \quad \sigma_c^s = \left(\frac{\psi_{ae}}{\psi_{ae} + \psi} \right) \times \psi \times \left(\frac{S_e}{S_{e,ae}} \right)$$

where σ_{dry}^s is the suction stress at oven-dry (105 °C) conditions, 10^6 is the theoretical maximum suction (kPa), ψ_{ae} is the air entry pressure, and $S_{e,ae}$ is the effective saturation at air entry pressure, which is determined using the [van Genuchten \(1980\)](#) SWRC model.

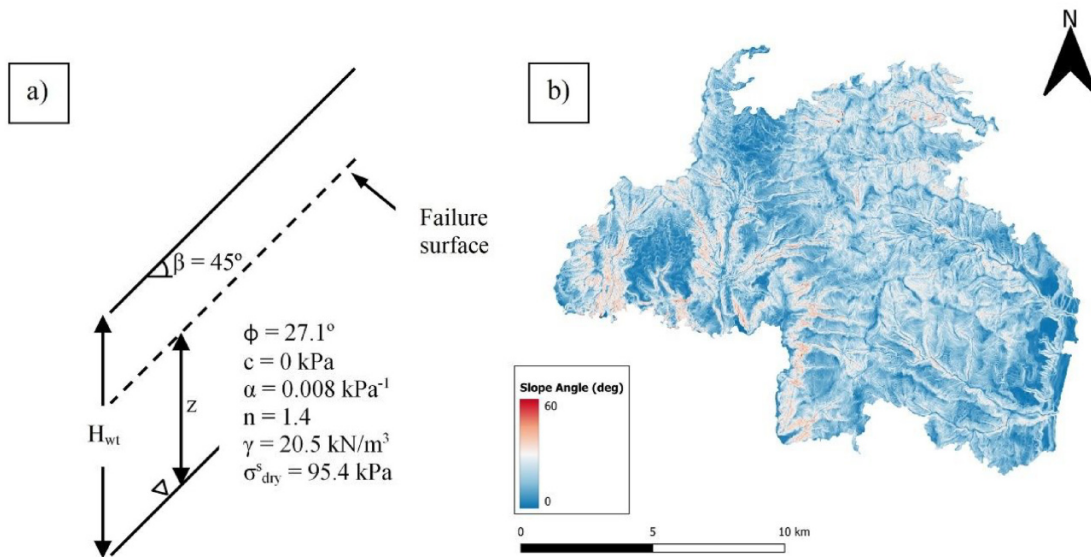
The soil was defined with constant soil properties to demonstrate the effects of post-wildfire water retention on slope stability. The SWRC parameters of $\alpha = 0.008 \text{ kPa}^{-1}$, $n = 1.4$, $S_r = 0$, and the friction angle of 27.1° were used as constant parameters for this study ([Akin et al. 2023](#)). Two separate analyses were performed to examine the effect of root cohesion. First, the root cohesion was assumed to be zero to demonstrate only the effects of temporal evolution in water retention on slope stability. A second analysis was performed by assuming an initial root cohesion of 6 kPa ([Sidle 1991](#)) to reflect the vegetation at the study site (i.e., ponderosa pine and mixed conifer), which decreased exponentially with time due to root decay. The root decay equation used in this analysis is defined as ([Abdollahi et al. 2024](#)):

$$(3) \quad C_r = C_{r0} \times e^{-wt}$$

where C_r is root cohesion, C_{r0} is root cohesion at pre-fire conditions, w is an empirical constant, and t is time. The empirical constant (w) was assumed to be 0.25, indicating a root loss of 50% over 3 years ([Abdollahi et al. 2024](#)).

The actual slope of the site is 17°. However, slope stability analysis was conducted on a wide range of slope angles to represent the slopes within the burn perimeter. The slope angle does not affect the trends of FS but influences its magnitude ([Cha and Kim 2011](#); [Kokutse et al. 2016](#); [Shiferaw 2021](#)). The mean slope angle in the burn perimeter was 18° with a standard deviation of 9° based on USGS digital elevation model data (<https://apps.nationalmap.gov/lidar-explorer/>) that has 10 m spatial resolution ([Fig. 3b](#)). Therefore, the stability analysis was conducted on slopes with 18° (mean), 27° (mean plus one standard deviation), and 45° (mean plus three standard deviation) angles.

Fig. 3. (a) Soil properties of the modeled slope and (b) slope angle map of the burn perimeter.



Vegetation recovery

The NDVI was calculated annually to evaluate the changes in vegetation cover. NDVI is one of the most widely used indices to monitor regional vegetation changes (e.g., [Pettorelli et al. 2005](#); [Guo et al. 2014](#); [Jamali et al. 2014](#)). The NDVI map was produced using red and near-infrared (NIR) bands of satellite imagery. Vegetation absorbs most of the red wavelength and reflects most of the NIR wavelength. Therefore, the difference between these two bands correlates with chlorophyll density (e.g., [Gandhi et al. 2015](#)).

Landsat 8-9 satellite images were used to determine the NDVI at 30 m spatial resolution. The NDVI of the area was calculated by

$$(4) \quad NDVI = \frac{(NIR - RED)}{(NIR + RED)}$$

where NIR is the near-infrared wavelength (Band 5) and RED is the red wavelength (Band 4). NDVI maps were produced for different pre-fire and post-fire dates. [Table 2](#) lists the dates of the acquired images and the mean NDVI values in each burn severity class (low, moderate, high). The dates were selected based on available datasets and the closest dates to eliminate the effect of any seasonal changes.

Results

Rainfall and snow cover

A total of 1421 mm of rainfall was measured on site between May 2020 and October 2023 ([Fig. 4](#)). The dashed lines delineate the data into 4 years (i.e., post-fire year 1, year 2, year 3, and year 4), allowing for an annual comparison of the rainfall patterns. The total rainfall amount that exceeds 15 mm was classified as a large rain event for comparison. The year 1 dataset was incomplete because of the late installation of the rain gauge in May 2020. After May 2020, there were two large rain events in year 1, one in June

(18 mm) and one in September (26 mm). Year 2 had five large rain events: one in October (24 mm), one in November (15 mm), and three in January (20, 24, and 27 mm). The cumulative rainfall in year 2 was 327 mm. In year 3, six large rain events were distributed over the year: two were in October (15 and 25 mm), one in January (20 mm), one in February (21 mm), one in March (15 mm), and one in June (21 mm). A cumulative rainfall of 449 mm occurred in year 3. In year 4, there were 13 large rain events: 1 in November (18 mm), 3 in December (17, 26, and 38 mm), 2 in January (17 and 23 mm), 3 in April (20, 26, and 41 mm), 1 in May (21 mm), 1 in June (18 mm), and 2 in August (15 and 26 mm). The large rain events are shown with blue arrows in [Fig. 4](#). Combined with the other rain events, year 4 was the wettest year over the data collection period, having a cumulative rainfall of 523 mm.

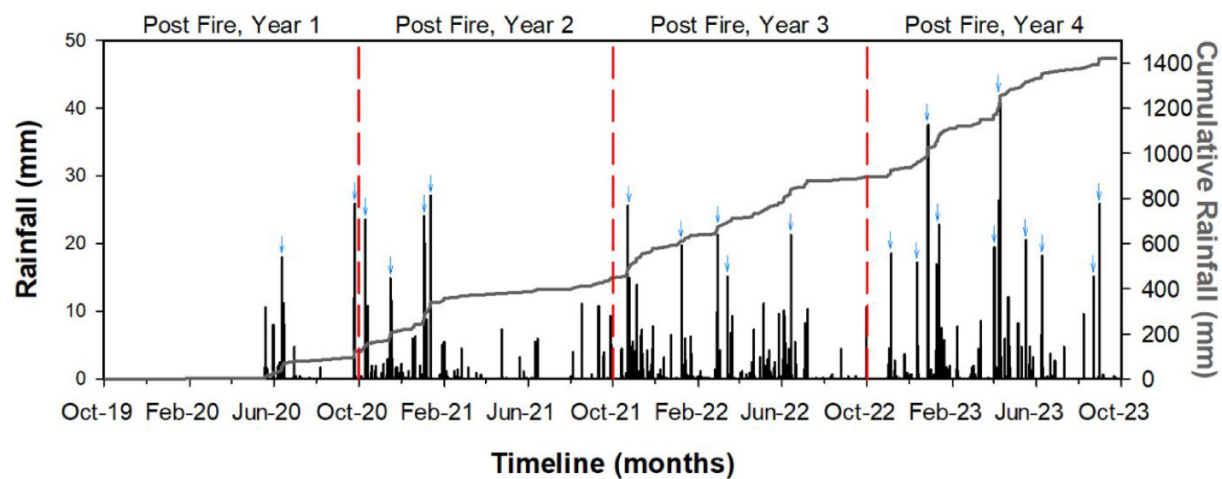
The acquired Landsat 8-9 satellite images were also used to track the snow cover at the site with QGIS (<https://qgis.org>). The site was under snow cover between December and April in year 1, November and April in year 2, December and March in year 3, and November and April in year 4. The total snowfall varied between 295 and 312 mm in winter months between October 2019 and October 2023, according to National Oceanic and Atmospheric Administration (NOAA) Online Weather Data for the nearby Coulee Dam 1 SW weather station (<https://www.weather.gov/wrh/Climate?wfo=sew>). The nearby weather station indicates that the total snowfall over 4 years was 1473 mm, while the total precipitation was 1053 mm.

Water retention

Both monitoring locations (i.e., dead and live trees) showed wide fluctuations in suction and degree of saturation. The fluctuations in the 0.3 m sensors were greater because the shallow surfaces are affected by surface moisture fluxes more. The measured saturation fluctuated between 0.24 and 1.0,

Table 2. The acquired image timeline and normalized difference vegetation index statistics of the 2019 Williams Flats Fire.

Date	Timeline	Burn severity class					
		Low		Moderate		High	
		Mean	Standard deviation	Mean	Standard deviation	Mean	Standard deviation
5 September 2018	Pre-fire	0.207	0.055	0.219	0.044	0.260	0.042
10 October 2019	Immediately after fire	0.158	0.043	0.108	0.033	0.060	0.026
10 September 2020	Post-fire, 1 year	0.169	0.047	0.145	0.035	0.145	0.036
28 August 2021	Post-fire, 2 years	0.168	0.051	0.142	0.038	0.160	0.040
8 September 2022	Post-fire 3 years	0.179	0.052	0.165	0.047	0.198	0.055
26 August 2023	Post-fire, 4 years	0.183	0.051	0.166	0.045	0.198	0.050

Fig. 4. Rainfall data of the study site (blue arrows indicate large rain events; red dashed lines separate the post-fire years).

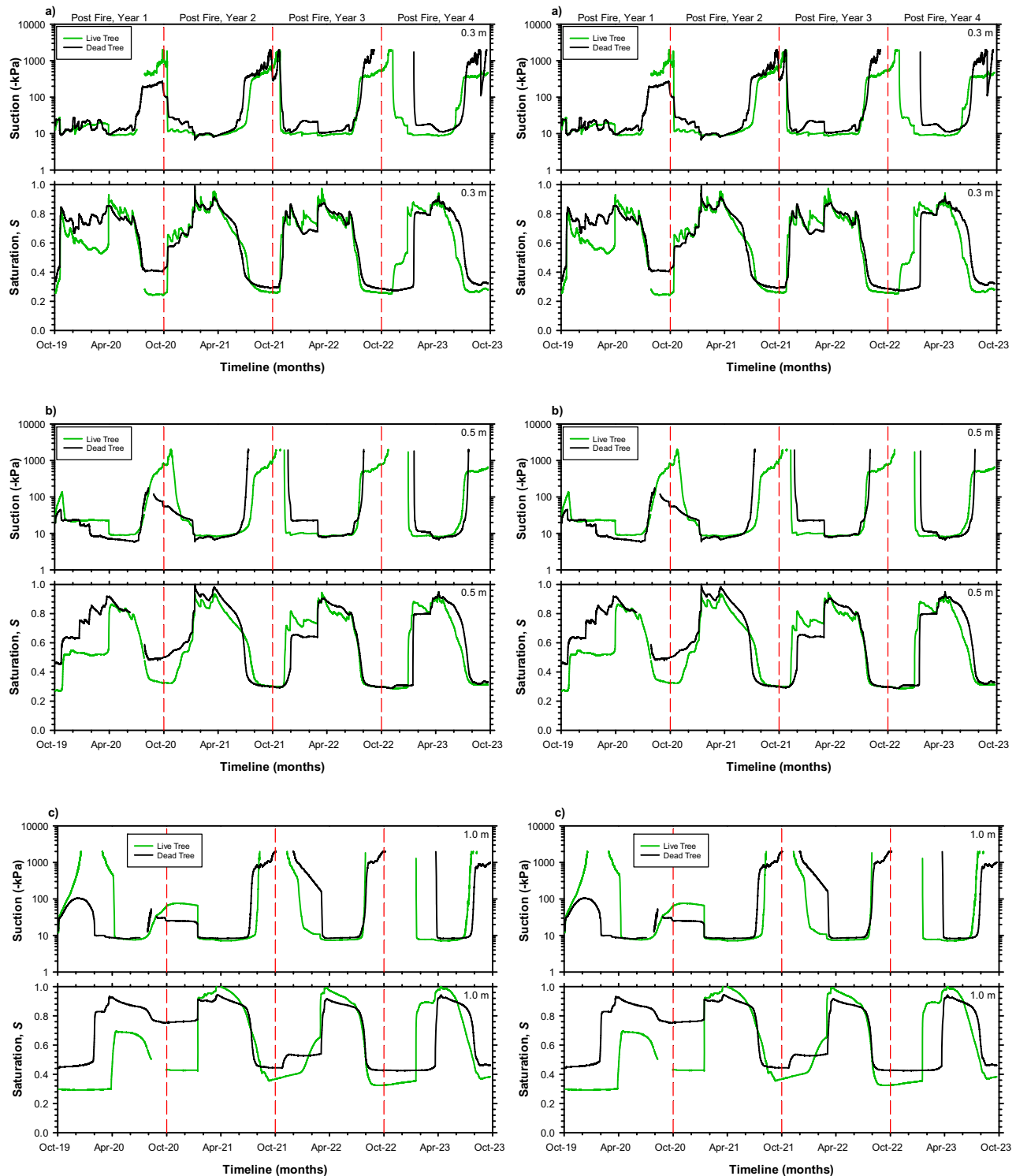
and the suction fluctuated between 5.9 and 2000 kPa (Fig. 5). The actual range of suction experienced in the field is expected to be wider than what was measured due to the limited capacity (2000 kPa) of the TEROs 21 suction sensor. The limited sensor capacity resulted in a discontinuity in the suction data in dry months. The discontinuity in water content and suction data in the summer of the first year was due to two bear attacks that cut the sensor cables or displaced the data loggers. Over the years, at both locations, fluctuations in water retention decreased with increasing depth (Fig. 5), consistent with the trend reported in the literature (e.g., Dorner et al. 2015; Zhou et al. 2018).

Rapid decreases in suction and increases in saturation were observed along the depth, except at five cases: at the dead tree at 0.5 m depth between January and June of year 1, at the dead tree at 0.3 m depth between October and January of year 2, at both the dead and live trees at 0.5 m depth between October and January of year 2, at both the dead and live trees at 1.0 m between December and March of year 3, and at the live tree at 0.3 m depth between November and December of year 4. The difference in water retention at the dead and live trees was the highest in year 1 and progressively decreased over time. Toward the end of year 2, the dead tree reacted to steep

suction increases in the dry season faster than the live tree at all depths. Still, similar temporal trends in suction were observed in both locations with a lag of approximately 1 month. Throughout years 3 and 4, the dead and live trees had rapid increases in suction during the dry season, corresponding to steep decreases in saturation. Seasonal variations are visible in the suction and saturation at all depths over 4 years. At all locations, higher suction and lower saturation were generally observed from summer to late fall, while lower suction and higher saturation were generally observed from late fall to the end of spring.

A rapid decrease in suction and a corresponding increase in saturation was generally observed following large rain events. For example, in October 2020, at 0.3 m depth, suction decreased rapidly at the live tree upon a rapid increase in the degree of saturation after a series of rain events between 10 and 13 October that produced a cumulative rainfall of 35 mm. The same series of rain events resulted in more gradual and delayed reductions in suction at 0.5 and 1.0 m depths at the live tree. The events also resulted in a rapid decrease in suction at the dead tree, although the drop in suction was nearly two orders of magnitude smaller at 0.3 m. Moreover, a rain event on 4 November 2022 produced 18 mm

Fig. 5. Suction and saturation data at (a) 0.3 m depth, (b) 0.5 m depth, and (c) 1.0 m depth.



of cumulative rainfall that resulted in a rapid decrease in suction and an increase in saturation at the live tree. The rain events between 26 and 28 December produced 63 mm of cumulative rainfall, which caused a rapid decrease in suction and increased saturation at 0.3 m depth for dead and live trees.

Overall, the main trends in measured suction and saturation values over 4 years were as follows: In year 1, higher suctions and lower saturations were observed at the live tree, with a rapid increase in saturation (and corresponding rapid decrease in suction) in April. In year 2, similar suction and saturation trends were noted at both locations. In year 3, lower

suctions and higher saturations were recorded at the live tree. In year 4, lower suctions and higher saturations were again observed at the live tree, along with a longer wet period compared to previous years.

Slope stability

The fluctuations in water retention over the years were mirrored in the trends in FS at 0.5 m depth regardless of the root cohesion assumption (Figs. 6 and 7). The trends were same for modeled slope angles (18°, 27°, 45°), but the magnitudes were different. The 18° slope had the highest FS values, while the 45° slope had the lowest FS over 4 years. Generally, the changes in FS were rapid at both the dead and live tree locations, with exceptions observed: from January to June in year 1 and from August of year 1 to January of year 2 at the dead tree, where the changes in FS were more gradual. In year 1, FS at the live tree was generally higher than the dead tree, with differences of up to 1.6 times for the 45° slope. However, toward the end of year 2 and continuing throughout years 3 and 4, the FS at the dead tree was higher than the live tree, with differences reaching up to 4.2 times for the 45° slope. This shift indicates a significant change in slope stability dynamics between the two locations over the study period.

Similar temporal FS trends were observed in both dead and live trees. The analysis with zero cohesion (Fig. 6) showed a minimum FS in June of year 1 at the dead tree, nearing a value of 1.0 for the 45° slope, while the minimum FS at the live tree over the 4 years was 1.3 for the 45° slope. A transition in FS occurred in year 2, where the FS at the dead tree began to exceed that at the live tree by the year's end. Subsequently, higher FS values were calculated at the dead tree in years 3 and 4 for all slope angles. In years 3 and 4, higher FS values were consistently observed at the dead tree compared to the live tree.

The analysis incorporating root cohesion and its temporal decay (Fig. 7) resulted in overall higher FS values at both the live and dead tree locations compared to the analysis assuming zero root cohesion. The root cohesion at the live tree was kept at 6 kPa over 4 years, so the trends were the same as zero root cohesion but with higher magnitudes. For example, the lowest FS for 45° slope angle at the live tree was 1.25 when the root cohesion was assumed to be zero but was 2.43 when the root cohesion was assumed to be 6 kPa. The trends in FS at the dead tree differed from the zero-cohesion analysis because of the exponential decay in root cohesion. The minimum FS for the 45° slope was calculated as 1.7 in April of year 4.

Normalized difference vegetation index (NDVI)

The mean NDVI value in each burn severity class decreased after the fire from their initial values. The pre-fire mean NDVI value in the entire burn perimeter was 0.218, which reduced to 0.133 1 month after the fire and increased to 0.161 after 1 year (Fig. 8). The mean NDVI value remained at 0.162 in year 2, increased to 0.179 in year 3 and 0.182 in year 4.

Soil temperature

Temperature data obtained from the water content sensors showed that soil temperatures fluctuated between ~1 and 24 °C. Soil temperatures increased with depth during winter months due to surface cooling and decreased with depth during summer months due to surface heating. In year 1, temperature was higher at the dead tree than at the live tree at all depths, except during winter (Fig. 9). At 0.3 m depth, the temperature at the dead tree reached up to 23.6 °C, whereas at the live tree, the maximum temperature was 20.4 °C. In year 4, this trend had reversed, with temperatures at the live tree being higher than those at the dead tree at all depths. The highest temperature at the live tree at 0.3 m depth was 21.9 °C, compared to 20.4 °C at the dead tree.

Discussion

Temporal changes in water retention, forest system dynamics, and corresponding slope stability

The observed trends of water retention and saturation versus time can be attributed to several underlying factors. Higher suctions and lower saturations at the live tree than the dead tree, which lasted until October of the first year, were attributed to evapotranspiration at the live tree (e.g., Ng et al. 2013). This is consistent with previous studies that quantified the temporal decrease in evapotranspiration after a wildfire. For example, Ma et al. (2020) quantified post-wildfire evapotranspiration in Sierra Nevada, CA, by using remote sensing and found that average evapotranspiration reduced to 36% of pre-wildfire evapotranspiration level 1 year after the wildfire, which reduced to 50% of pre-wildfire evapotranspiration level in high burn severity regions 1 year after the wildfire. The burn of vegetation and the trees in the high burn severity area resulted in the loss of evapotranspiration in the dead tree location. Even though the live tree was in a moderate burn severity area, the tree and its surroundings were not affected by the fire, so evapotranspiration was expected to continue in the live tree location. In addition, root activity likely contributed to the rapid responses in suction and saturation following rainfall events at the live tree. The live tree's active root system enhances soil structure and water retention capacity, leading to quicker changes in soil moisture conditions.

The observed differences in soil suction at various depths can be attributed to the differences in soil structure and porosity. The surface soil (0.3 m depth) is more prone to rapid changes in moisture content due to its proximity to the surface and higher organic matter content, which enhances water infiltration and retention. The intensity and duration of rainfall events significantly influenced the observed trends. High-intensity rainfalls resulted in rapid saturation increases and corresponding suction decreases, particularly at the surface levels, while longer-duration events caused more gradual changes at greater depths. Seasonal variations, including wet and dry periods, also impacted the soil moisture dynamics. For instance, longer wet periods in year 4 led to sustained higher saturation levels and lower suction values,

Fig. 6. Factor of safety at the dead and live tree at 0.5 m depth for different slope angles and zero root cohesion.

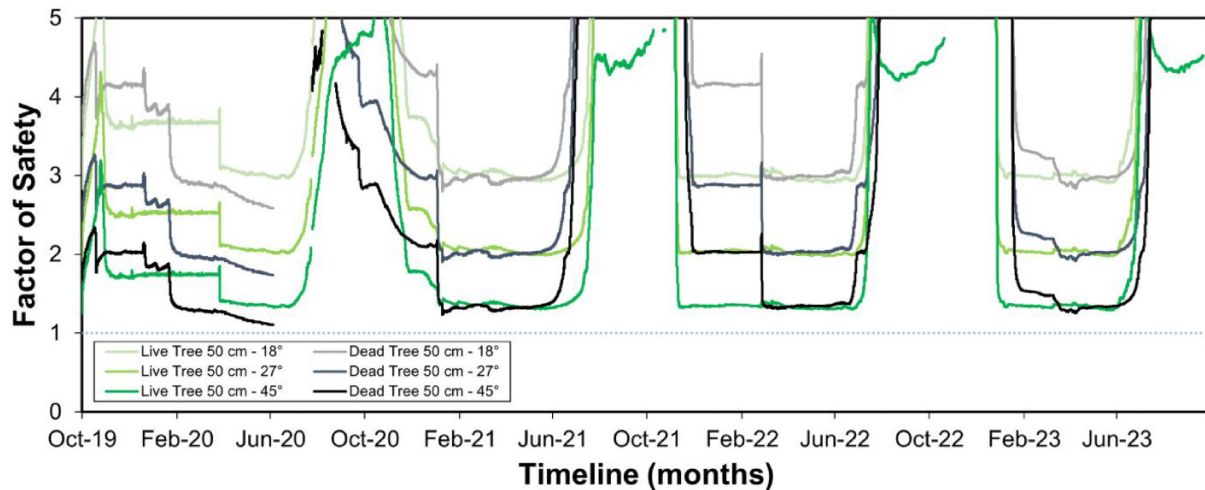
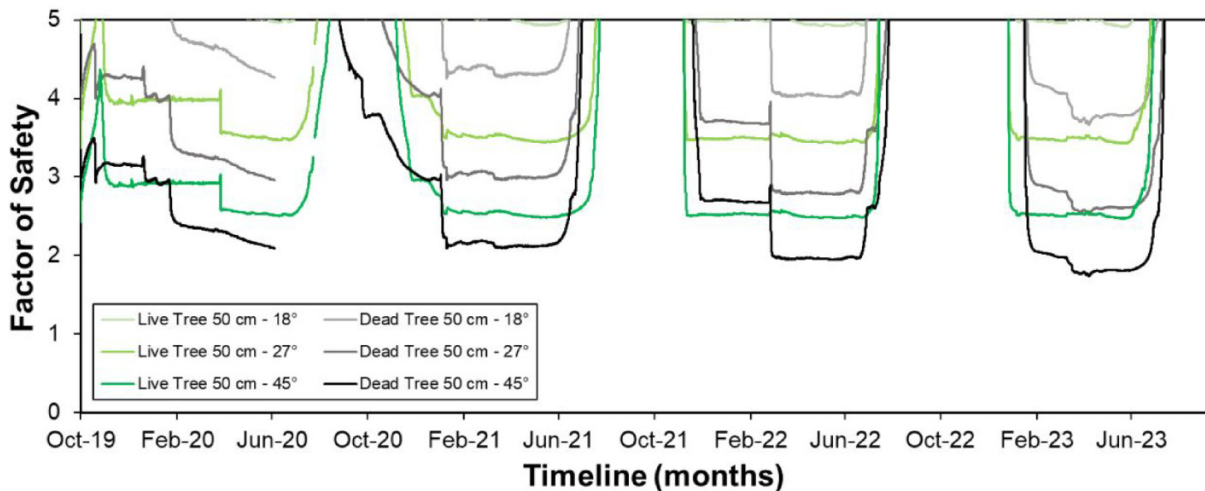


Fig. 7. Factor of safety of the dead and live tree at 0.5 m depth for different slope angles with contribution of root cohesion.



reflecting the cumulative effect of consistent rainfall over time.

The slower and more gradual increase in suction at the live tree, in response to changes in surface moisture flux toward the end of year 2 and throughout years 3 and 4, resulted in opposite FS trends compared to year 1 in the zero-cohesion analysis. In year 1, FS at the live tree was higher than the dead tree. In contrast, in year 4, FS at the dead tree was higher than at the live tree (Fig. 6). The inverse trend in the slope stability in year 4 compared to year 1 could partially be explained by vegetation recovery. Vegetation recovery was evident through changes in NDVI values, and soil temperature data. The temporal changes in the mean NDVI values in each burn severity class suggest that the high burn severity areas, which had the highest pre-fire NDVI values showed the fastest post-fire vegetation recovery (Fig. 10). The mean NDVI in the high burn severity areas, where the dead tree was located, exceeded the NDVI of low and moderate burn severity areas, where the live tree was located, by the end of 4 years. Pre-fire

NDVI levels were not reached in any burn severity class by the end of 4 years. The vegetation recovery analysis through NDVI maps does not necessarily reflect the recovery of trees, but an increase in NDVI can be due to the growth of herbaceous vegetation. The field observations indicate that NDVI values mainly reflected the growth of herbaceous vegetation in this study.

The vegetation recovery patterns were closely aligned with rainfall patterns in high and moderate severity burn areas. The mean NDVI in high burn severity areas increased from 0.060 to 0.145 after 1 year, reflecting a significant vegetation recovery. Two years after the fire, the mean NDVI in the high burn severity area was 0.160, indicating a slower recovery in year 2 compared to year 1. This slower recovery rate in year 2 was attributed to the difference in rainfall patterns, particularly in the summer months (Fig. 4). Only 28 mm of rainfall was recorded in the summer of year 2, while the summer of year 1 saw nearly double that amount (55 mm). The lower summer rainfall in year 2 delayed the death of seasonal

Fig. 8. Normalized difference vegetation index (NDVI) images of Williams Flats Fire burn perimeter for (a) pre-fire, (b) immediately after fire, (c) post-fire 1 year, (d) post-fire 2 years, (e) post-fire 3 years, and (f) post-fire 4 years.

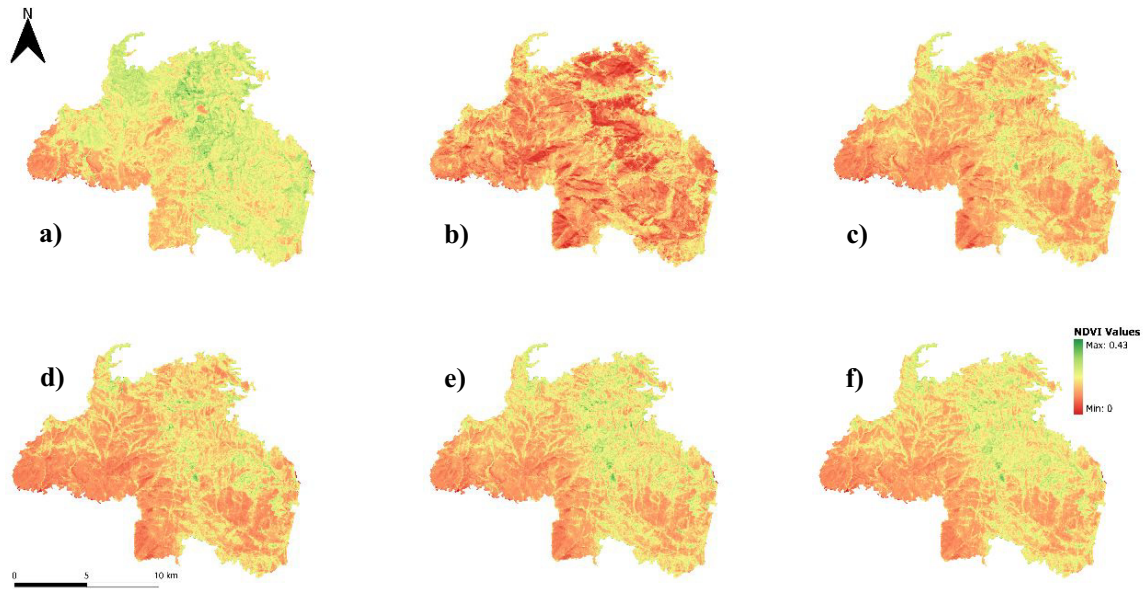
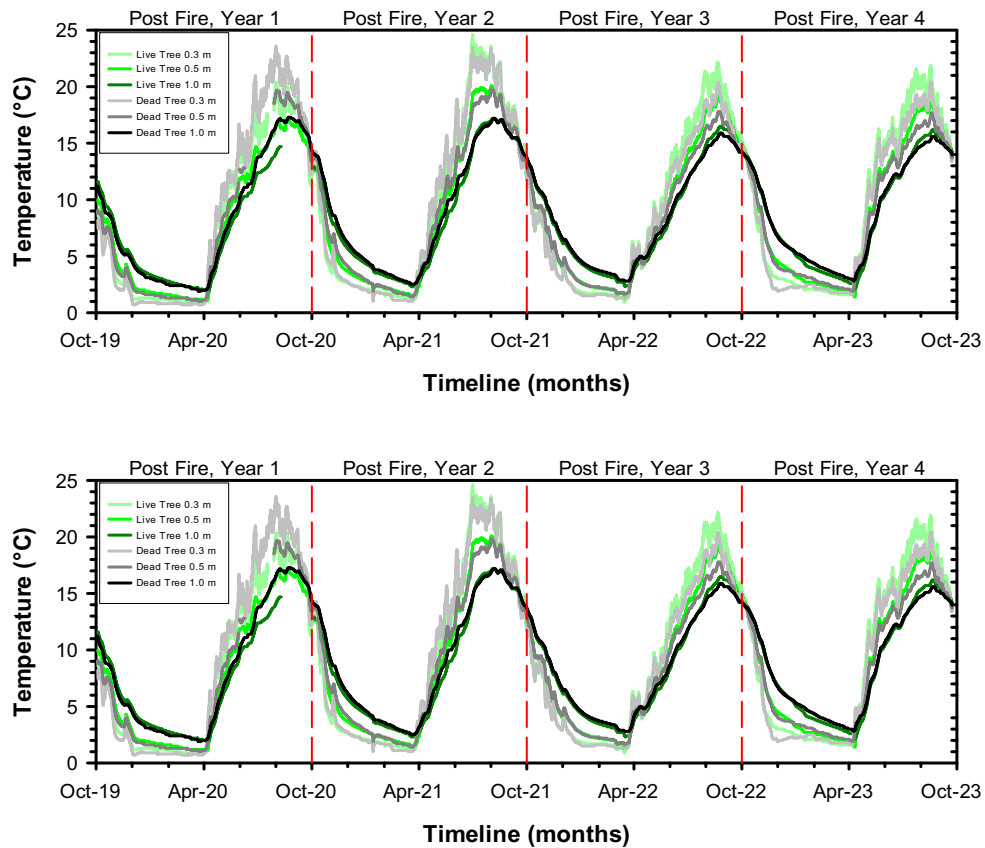
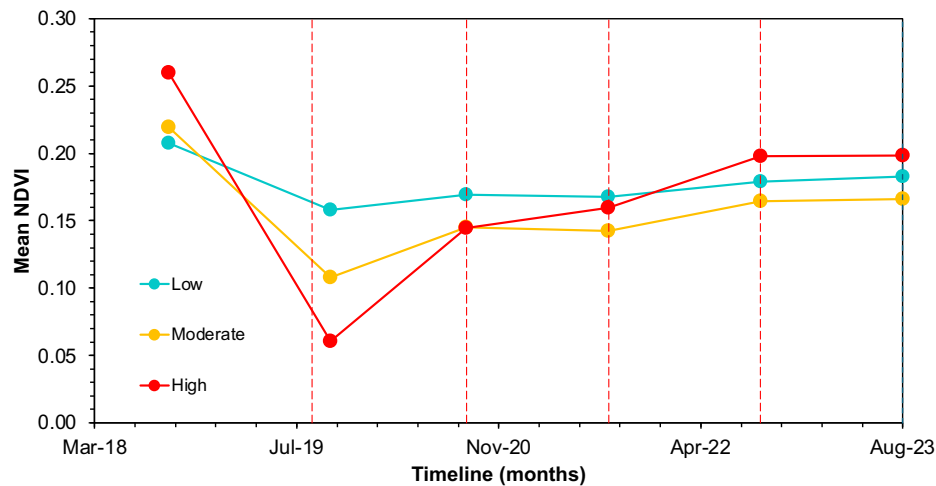


Fig. 9. Soil temperature data from the water content sensors for all depths.



vegetation (i.e., grasses, forbs, and shrubs), resulting in similar NDVI values 1 and 2 years after the fire. Three years after the fire, NDVI values increased to a mean value of 0.198, supported by a wetter summer with 104 mm of rainfall. In year 4,

the mean NDVI remained at 0.198, reflecting a dry summer with only 59 mm of rainfall up to the date of the satellite image acquisition. The NDVI values in the moderate severity areas followed a similar pattern (Fig. 10). While pre-fire mean

Fig. 10. Temporal changes in the mean normalized difference vegetation index (NDVI) in each burn severity class.

NDVI values were not reached within 4 years, vegetation recovery patterns were closely aligned with rainfall patterns. These observations highlight the complex interplay between vegetation recovery, soil moisture dynamics, and slope stability following a wildfire.

After the wildfire, the forest floor lacks vegetation and has a darker color, resulting in lower albedo (e.g., Govaerts et al. 2002; Yoshikawa et al. 2003). This leads to higher incoming solar radiation to the forest floor and an increase in the soil temperature. Additionally, loss of canopies and transpiration increase soil temperatures (e.g., Gupta et al. 2018). As vegetation regrows, albedo increases, resulting in less solar radiation reaching the forest floor and, consequently, lower soil temperatures. Therefore, the temperature change at the dead tree is expected to decrease over the years due to vegetation growth. Vegetation recovery was also examined through the temperature data collected from the water content sensors from the field (Fig. 9). At both the dead and live tree locations, the amplitude of temperature fluctuation decreased with depth, as expected (Hillel 2003). In year 1, temperature at the dead tree location was 3.2 °C higher at 0.3 m depth. This trend had reversed in year 4, where the live tree location had 1.5 °C higher temperature than the dead tree location. The decrease in temperature at the dead tree site over 4 years compared to the live tree site was attributed to vegetation regrowth at the study site. The regrowth increased albedo, reducing solar radiation absorption, increased transpiration cooling, and thereby lowering soil temperatures.

Higher temperatures at the dead tree location compared to the live tree location can also explain the early onset of the increase in degree of saturation in the winter months of year 1, compared to the live tree location. Degree of saturation gradually increased with the start of snowfall in the dead tree location, which was attributed to the presence of macropores (Akin et al. 2023). Higher soil temperatures at the dead tree location also support that snowmelt could happen earlier at the dead tree location compared to the live tree location.

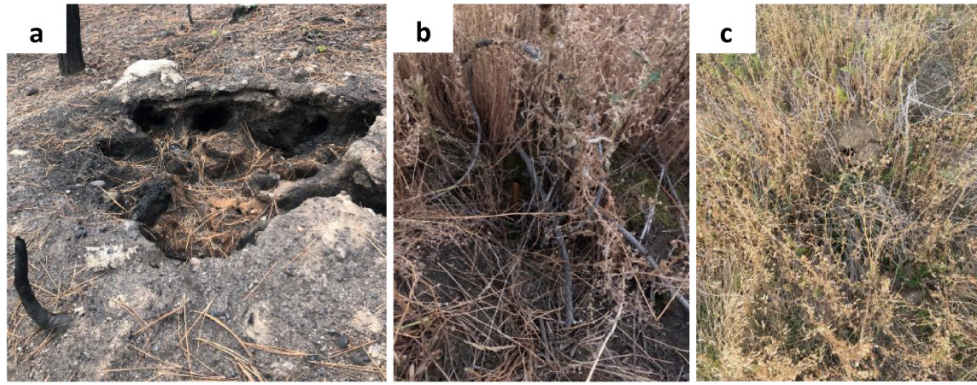
The analysis that considered an initial root cohesion value of 6 kPa that subsequently decreased with root decay showed consistently lower FS at the dead tree compared to the live tree in the wet months (Fig. 7), where suction stress reached a minimum. As a result of the exponential root decay, the lowest FS at the dead tree was calculated in year 4. This trend was consistent with the generally accepted understanding that post-wildfire landslide susceptibility decreases over time due to root decay, reaching a minimum of approximately 3 years after the fire (De Graff 2018; Rengers et al. 2020). However, the contribution of herbaceous vegetation regrowth, which would result in some increase in root cohesion over time, was not included in the analysis due to lack of data. It is also important to note that the root cohesion and root decay functions are empirical and are not specific to our study site. This study assumed an initial root cohesion of 6 kPa and a decay rate of 50% after 3 years. However, field-specific measurements might reveal different values that would change the temporal FS trends.

The results of this study demonstrate that temporal changes in soil water retention—driven by seasonal precipitation patterns, vegetation recovery, and macropore dynamics—play a primary role in governing slope stability in post-wildfire environments. The close correspondence between suction stress evolution and FS trends confirms that slope stability is highly sensitive to time-varying hydrological conditions. These findings highlight that FS is not static in burned areas but evolves significantly during the recovery period. As such, accurate post-wildfire landslide prediction requires incorporating dynamic water retention behavior, rather than relying on static or laboratory-based estimates. This reinforces the importance of long-term field monitoring and offers a new pathway to improve slope stability modeling under post-disturbance conditions.

Limitations and challenges of post-wildfire slope stability analysis

Quantifying site-specific root decay and regrowth parameters remains a significant challenge. One of the primary dif-

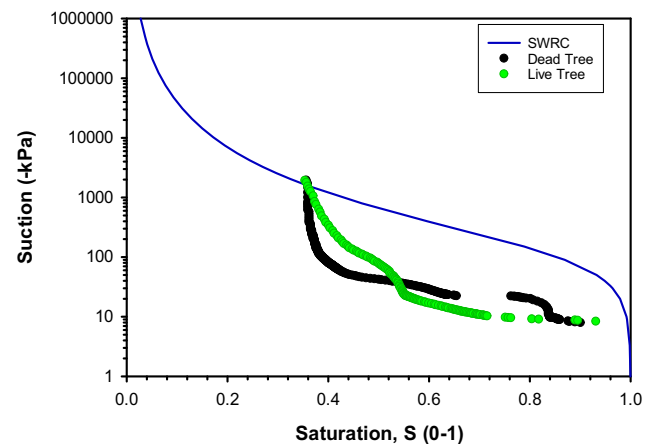
Fig. 11. Example macropore from a burned tree (a) 1 month after the fire, (b) 3 years after the fire, and (c) 4 years after the fire.



difficulties lies in accurately identifying new vegetation, including the brush and grass species, and assigning appropriate root cohesion values, as both regrowth and root decay occur concurrently. During the recovery period, new vegetation roots will be shallow, while the progressively decaying tree roots are deeper. Therefore, landslides are initially expected to be deeper, until new roots grow. Additionally, various tree and plant species contribute to root cohesion, which varies significantly depending on vegetation type, age, and environmental conditions (Schmidt et al. 2001). This complexity suggests that a probabilistic approach is necessary—one that accounts for the spatial and temporal variability in vegetation type, age, and corresponding root cohesion values. However, such detailed data are currently lacking. Developing a comprehensive dataset would require extensive field measurements and long-term monitoring across diverse ecosystems to capture the full range of variability. In the absence of this information, it is challenging to produce accurate models that predict root cohesion and its effects on soil stability.

In addition to the changes in water retention and root decay and regrowth, field observations (Fig. 11) showed the evolution of the macropores around the dead tree over time. One month after the fire, macropores around the dead tree were wide open, with vegetation completely burned. However, 3–4 years after the fire, the macropores became clogged with new vegetation. This suggests a gradual recovery process, where vegetation regrowth contributes to the infilling and stabilization of soil structure around the tree roots over time. Our previous study showed evidence of vertical ash movement at the site over a year after the fire (Akin and Akinleye 2021), which likely contributed to macropore clogging. An increase in ash concentration in soil may decrease the soil's thermal conductivity (Rubio 2014). Therefore, macropore clogging due to vegetation and ash movement over time could be contributing factors that further increase FS and decrease the temperature at the dead tree. Stoof et al. (2016) found ash movement into pores but did not observe the pore-clogging of ash for pure sand. However, unlike Stoof et al. (2016), this study site has 53% sand content at the surface and 68% sand content at 0.5 m depth (Akin and Akinleye 2021), which may not show similar behavior observed for the pure sand. Further research is

Fig. 12. Laboratory soil water retention curve (SWRC) versus field-measured water retention data at year 3.



needed to quantify the effects of macropores on post-wildfire slope stability.

In addition, in this study, the contribution of water retention to slope stability was calculated using direct measurements of suction and water content in the field. However, in practice, the common approach is to measure the soil water retention data of the soil in the laboratory along with a curve-fitting model, such as van Genuchten (1980) or Fredlund and Xing (1994) models, and then use the curve-fitting parameters to calculate the contribution of water retention to slope stability. This approach may be misleading in post-wildfire forest environments. For example, the field-measured water retention data of year 3 at 0.5 m depth was compared to the laboratory-measured water retention curve from van Genuchten (1980) model (Fig. 12). To generate field-measured water retention data, suction and saturation pairs were randomly selected in year 3. The field water retention data were lower than the laboratory-derived water retention curve and showed nearly up to two orders of magnitude difference in suction values. The suction and saturation at the dead and live trees converged when suction reached 2000 kPa, but the suction above 2000 kPa was not measured in the field due to the limited capacity of the sensors. This

discrepancy highlights the potential limitations of relying solely on laboratory measurements for understanding water retention and its impact on slope stability in post-wildfire environments. For the random suction-saturation pairs selected in year 3 (Fig. 12), the laboratory-measured SWRC would overestimate suction stress and therefore FS at high degree of saturations. This suggests that using laboratory-derived parameters in place of field-monitored data could lead to overly conservative estimates of slope stability, particularly in post-wildfire environments where in situ conditions are highly variable and influenced by factors such as vegetation dynamics, macropores, and ash infiltration. Additionally, transpiration and the SWRC of rooted soils depend on temperature (e.g., Ni et al. 2024). Therefore, laboratory-measured SWRCs should be adjusted for representative field temperatures. Another challenge in modeling post-wildfire slope stability using laboratory-measured SWRCs is the temporal changes in void ratio. Both vegetation regrowth and root decay can alter void ratio and therefore hydraulic conductivity and the SWRC (e.g., Ni et al. 2018). Slope stability models based solely on laboratory data must account for these temporal changes in hydraulic conductivity and the SWRC. In contrast, direct field measurements provide a more realistic representation of in situ conditions, which are influenced by factors such as soil structure, temperature, vegetation recovery, root decay, and ash movement, highlighting the importance of field-based assessments in post-wildfire environments.

The differences between field and laboratory water retention curves are studied by various researchers independent of wildfires (e.g., Morgan et al. 2001; Pachepsky et al. 2001; Askarinejad et al. 2011; Mosquera et al. 2021). Similar results were observed in other studies where the laboratory curve yielded higher suction values than the field curve (e.g., Eching et al. 1994; Iiyama 2016). The primary reason for this discrepancy is that complete wet-dry cycles are often not achieved in field tests. Other studies found that field suctions were greater than laboratory suctions, attributing the differences to factors such as sample disturbance (Schuh et al. 1988), the presence of wormholes and root holes (Klute 1986), overburden pressure (Klute 1986), macropores (Field et al. 1984), or vegetation (Leung et al. 2015). In both scenarios, using the laboratory-measured SWRC results in a deviation from the field FS.

Conclusion

The long-term stability of a slope burned by the 2019 Williams Flats wildfire was evaluated over 4 years with respect to the changes in suction, saturation, and corresponding calculated FS. Field suction and water content sensors installed near a dead and a live tree were monitored for 4 years. The results demonstrated that the vegetation regrowth over time leads to temporal changes in post-wildfire slope stability, high soil burn severity areas showed more vegetation regrowth, and regrowth rates aligned with rainfall patterns. Additionally, soil water retention aligned with regrowth rates, and soil temperature patterns. In year 1, the live tree location exhibited up to an order of magnitude higher suction

and correspondingly lower saturations. After June 2021, the dead tree suction started to exceed the live tree suction, with differences reaching up to two orders of magnitude, as the dead tree reacted more quickly to steep increases in suction during the dry season. These changes in suction and saturation resulted in changes in FS. The lowest FS at the dead tree was obtained in June, following snowmelt and precipitation (rain) in winter and spring. In the subsequent years, FS at the dead tree location increased and exceeded the FS at the live tree location. The changes in the FS of the dead tree site were attributed to the vegetation recovery quantified by NDVI values and soil temperature data. The trends in the mean NDVI aligned with rainfall patterns. Immediately after the fire, summer temperatures at the dead tree were higher than the live tree, but by the end of 4 years, summer temperatures at the live tree were higher than the dead tree due to vegetation by 1.5 °C.

The results emphasize the importance of considering both vegetation dynamics and soil physical changes when assessing post-wildfire slope stability. The implications of this study are significant for post-wildfire management and mitigation strategies, as they highlight the need for long-term monitoring of both vegetation recovery and soil conditions to accurately map landslide susceptibility. The insights gained can inform the development of more effective rehabilitation practices and early warning systems to mitigate the risks of landslides in post-wildfire landscapes.

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Data availability

All data of this study are presented in the manuscript.

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Competing interests

The authors do not have any competing interests.

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