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Large projected increases in area burned and wildfire frequency by 2050 in Utah, USA

Joseph D. Birch[1] · **Yoshimitsu Chikamoto**[2] · **James A. Lutz**[1]

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Abstract
Changes in wildfire regimes may disrupt ecosystem processes as wildfires burn larger areas or burn more frequently than the recent natural range of variability. The climatic drivers of wildfire behavior may change in strength, but these effects are not likely to be uniform across space and different vegetation types. Increased understanding of how weather and climate influence patterns of burn area and frequency across vegetation types may assist in better predicting and managing future wildfire regimes. We examined a dataset of all 1469 wildfires ≥40 ha from 1984 - 2021 in Utah, USA, and used antecedent daily weather data to analyze how temperature and aridity influenced fire area and frequency across forested and non-forested vegetation types. The number of days in a year when air temperature was ≥ 26.6 °C (80 °F) was the best predictor for area burned for forest ($R^2=0.31$) and non-forest ecosystems ($R^2=0.31$) in Utah. However, model skill was variable across vegetation types and models performed best for high-elevation forest ecosystems ($R^2=0.27$ to 0.32) relative to low-elevation, non-forest ecosystems ($R^2=0.11$ to 0.31). By 2050, warming trends in Utah may result in a 60% increase in area burned for forests and a 232% increase for non-forests. These results highlight a simple metric – temperature – that explains large portions of variability in burned area and correlates with fire season length. A simple temperature metric is a good match for the vegetation and fire season climate of Utah, which is generally dry. Our results suggest that a warmer future may bring widespread and vegetation-specific changes in wildfire regimes with large increases in area burned across most vegetation types.

Keywords Climate change · Wildfire behavior · Great Basin · Colorado Plateau · Area burned · Fire frequency

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1 Introduction

1.1 Past and future climatic drivers of wildfire behavior

The area burned by fire, and fire frequency, are key components of fire regimes (among others, McLauchlan et al. 2020) that interact with ecosystems to shape ecosystem function and composition. Although fire is a natural and beneficial process in many terrestrial ecosystems, uncharacteristic fire behavior, frequency, seasonality, or severity may cause long-term shifts in fire regimes and corresponding changes in ecosystem composition and function (Miller et al. 2019). For example, human-caused wildfires have been associated with more extreme rates of spread and up to three-fold greater tree mortality relative to natural ignitions (Hantson et al. 2022), potentially shifting forest composition away from historic conditions. Wildfire behaviors are jointly influenced by a combination of topography (Birch et al. 2023b), fuel availability (e.g., fuel mass and moisture), and fire weather (Brodie et al. 2024; Kane et al. 2015). Whereas topography is largely invariant, fuel availability and fire weather change over both long- and short-timescales as a result of ecosystem dynamics (Lutz et al. 2021; Terry et al. 2024) and climatic influences (e.g., El Niño Southern Oscillation; Brown et al. 2008; Chikamoto et al. 2017, 2020). Thus, shifts in fire regime characteristics, such as area burned or fire frequency, may occur under future, warmer climates.

Seasonal to decadal climatic variability influences fire behavior by changing fuel structure and availability for consumption by influencing vegetation productivity and mortality, phenology, and precipitation seasonality (Chikamoto et al. 2023; Germain and Lutz 2020; Stevens-Rumann et al. 2018; Zohner et al. 2020). In contrast, high-frequency and continuous weather, such as heat waves, wind gusts, air temperature, or relative humidity, alter fuel moisture and influence the direction and mode of fire and heat transport (e.g., wind-driven fire behavior and spotting), and the ease of fuel combustion (Dong et al. 2021; Mendes-Lopes et al. 2003; Zhang et al. 2025). The combination of low- and high-frequency weather acts as top-down control on fire behavior, influencing fire regime characteristics, and is likely to change under warmer and drier climates of the future (Abatzoglou and Williams 2016; Dai 2013). Despite substantial progress in understanding these multiscale controls, key uncertainties remain regarding which short-term, high-frequency weather metrics most effectively explain variability in fire activity. In particular, the factors directly influencing fire at smaller spatial scales (i.e., the area of one state in the western USA) may be different than those that reflect the mean response of much larger regions (i.e., the entire American West) owing to variation in topography and regional weather patterns. This raises the central question of whether simple metrics such as temperature or moisture thresholds may act as primary determinants of interannual variability in area burned or fire frequency.

1.2 Shifts in wildfire regimes – consequences and impacts on ecosystem resilience

Warming and more variable climates have already been implicated in widespread increases in the area burned by wildfires (Ayars et al. 2023), greater proportions of high severity (Hagmann et al. 2021), and longer fire seasons (Clarke et al. 2022). Contemporaneously, warming climates have increased vegetation mortality by increasing drought frequency and severity (Dai 2013; Furniss et al. 2022), altering insect lifecycles and ranges (Cudmore et al. 2010), or disrupting phenology (Birch et al. 2023a). In combination, these climate-induced disruptions may increase the availability of dead, dry fuels (Goodwin et al. 2021) and the window of time in which wildfire may carry through receptive fuels (e.g., the 'fire season'). Alterations to historic fire regimes may result in fire behavior and burn severity that cause shifts in forest structure or composition so severe as to result in a loss of ecosystem function or a stable-state shift towards non-forest (Seidl and Turner 2022). As climates continue to warm and become more variable, predicting wildfire behavior (e.g., area burned and fire frequency) may be more relevant to managers and policy makers seeking to pre-empt, where possible, unwanted fire effects which may shift ecosystems towards undesirable functional changes.

Existing models assessing climatic influences on fire in the western USA predict large increases in area burned, fire season length, and the area burned at higher severity (Abatzoglou et al. 2021; Parks et al. 2025a). These models frequently use vapor pressure deficit (VPD) or measures of aridity such as climatic water deficit or drought indices (e.g., Keetch-Byram Drought Index, Liu et al. 2013). Whereas these models often have considerable skill at regional scales, they possess several limitations owing to 1) their use of metrics that are assessed in arrears at monthly, seasonal, or yearly scales and are not easily calculated by managers during the fire season, and 2) being spatially coarse and agnostic of variation in topography and vegetation types and ecosystems. Managers and researchers seeking to understand potential changes in area burned, fire frequency, and fire severity in their respective ecosystems may thus be forced to rely on regional models averaging across many ecosystem types and which offer little guidance on potential fire behavior in the current fire season. These projected changes suggest that future wildfires will become more frequent and burn larger areas, particularly in forested ecosystems that are most sensitive to climatic constraints and typically have higher fuel loadings (and lower fuel constraints) than non-forested ecosystems.

1.3 The role of vegetation and types of fuel on wildfire behavior

Fuel availability for ignition and consumption depends on its physical and chemical composition (e.g., terpenoid content; Della Rocca et al. 2017), orientation in space (e.g., fuel density or suspension), moisture content (Popović et al. 2021), and the absolute mass of fuel (e.g., 'loading'). Thus, wildfire behavior varies widely between vegetation types owing to both abiotic and biotic controls. For example, a dry, low density, *Bromus tectorum* L. (cheatgrass) fuel-bed may be expected to ignite and combust more completely than a wet, high density forest floor derived from *Abies bifolia* (Hook.) Nutt. (Rocky Mountain subalpine fir) litter (Brown 1981; Diamond et al. 2009). Consequently, the type of fuels in an ecosystem may influence fire behavior and frequency in combination with climatic controls. The relative influence of climate, fire weather, or fuel loadings will thus play a large role in predicting how future climates will influence fire regimes in specific vegetation types. For example, fuel-limited ecosystems (e.g., salt desert) may have smaller changes in fire regimes under future climates than ecosystems currently limited by climatic controls (e.g., spruce-fir forest). Therefore, changes in climate or weather will not affect all areas equally, and predictions of future fire behavior should account for vegetation-specific differences in the relative strength of climatic- or fuel-driven controls. Given these contrasts among vegetation types, we hypothesize that forests will exhibit a stronger response to climatic drivers than non-forested ecosystems, potentially owing to greater fuel moisture retention and the influence of remnant snowpack.

The state of Utah (~219,887 km^2) in the western USA is a microcosm of the vegetation types - and fire regimes - found across much of the western US, including the Great Basin (the western half of Utah), the Colorado Plateau (the eastern half of Utah), and sub-alpine and montane forests in western North America (Birch and Lutz 2023a; Parry 2016). Utah has considerable variation in fire regimes, owing to the diversity in vegetation types, but have an increasing trend through time (1984–2021) of a few large fires burning increasing proportions of the annual area burned (Birch and Lutz 2023a).

The goal of this study is to identify ecosystems types most at-risk for future shifts in fire regimes in Utah. In this study we leveraged a dataset of 1469 wildfires in Utah, USA in combination with daily weather data in the period 1984–2021. We sought to identify simple climatic metrics that could be easily calculated during the fire season and that would explain controls on annual area burned and fire frequency so as to assess past and future trends in these metrics for the most common vegetation types. By developing vegetation-specific models of the climatic-controls on area burned and fire frequency, we sought to identify ecosystems types most at-risk for future shifts in fire regimes. We focus on simple temperature and precipitation metrics to provide a robust set of measurements that can be easily calculated by land managers during the fire season and for specific ecosystems in Utah. We used these models to test the hypotheses:

- **H1:** The number of days above a specific temperature threshold will be the primary determinant of area burned and fire frequency.
- **H2:** Projected future wildfires will be more numerous and burn larger areas, particularly in forested ecosystems which are frequently climate-limited.
- **H3:** Forests will have a greater response to climatic drivers than non-forested ecosystems possibly owing to greater fuel moisture and remnant snowpack.

2 Methods

2.1 Study area

Our study area exists at the confluence of multiple geographic regions, and climatic zones, and with an elevational range of 664–4120 m, sufficient to support vegetation ranging from low-productivity desert ecosystems to productive, closed-canopy forests (Birch and Lutz 2023a; Ramsey and West 2009). Utah has a semi-arid climate shaped by its mountainous topography, with colder temperatures prevailing in the northern regions and higher elevations, and warmer conditions in the southern and lower elevations. Annual precipitation totals range from approximately 120 to 1,000 mm, with a seasonal cycle characterized by dry summers and even drier conditions in the southern part of the state. Consequently, summers are typically hot and dry across the state but can occasionally experience increased rainfall due to active North American monsoon, particularly in southern Utah (Meyer and Jin 2017; Nauslar et al. 2019). Snowfall is the primary contributor to annual precipitation in the northern regions and higher elevations, accounting for more than 80% of winter precipitation (Gillies et al. 2012; Wang et al. 2009). Over the past 60 years, spring temperatures have shown a significant increasing trend, leading to a shift in precipitation type from snow to rain (Gillies et al. 2012).

2.2 Fire ignitions and footprints

To characterize the starting dates and seasonality of fires in Utah, we sourced ignition data and location from the inFORM fire occurrence dataset, an ongoing consolidation of wild and prescribed fires (National Interagency Fire Center, 2023). We clipped the inFORM dataset to the contiguous boundary of Utah ($n=18{,}414$ ignitions) and joined each ignition point to a corresponding fire footprint (≥40 ha; 1984–2022). We sourced fire footprints from state and federal land managers and the Wildland Fire Interagency Geospatial Services (WFIGS; 2022). We excluded prescribed fires from our dataset and analyzed pixels only for areas that burned inside the boundary of Utah – excluding portions of fires that burned into neighboring states (Birch and Lutz 2023a). We analyzed escaped prescribed burns as wildfires. As described in Birch and Lutz (2023a), we used high-resolution satellite imagery and remotely sensed burn severity to manually correct sourced fire footprints only in cases where burned vegetation was visible outside the adjacent fire footprint. We classified ignitions into four categories: 1) all ignitions, 2) ignitions that led to a fire ≥ 40 ha, 3) ignitions that led to fires (≥40 ha) that burned > 50% of their footprint in forested landscapes, or 4) ignitions that led to fires (≥40 ha) that burned > 50% of their footprint in non-forested landscapes.

2.3 Vegetation classification

To assess differences in fire patterns across vegetation types we classified all burned areas in 30 × 30 m pixels using the LANDFIRE existing vegetation type classification system (EVT; LANDFIRE 2018). The LANDFIRE EVT dataset categorizes vegetation associations that frequently co-occur on the landscape and are mapped using a combination of remote sensing, topographic data, ground truthing, and modeling (2018). Because the EVT dataset had greater specificity than desired for our analyses (e.g., “Rocky Mountain Subalpine-Montane Limber-Bristlecone Pine Woodland” *versus* “Inter-Mountain Basins Subalpine Limber-Bristlecone Pine Woodland”), we reduced complexity in our dataset by aggregating the 102 EVT layers into 19 broader vegetation categories (Table 1, *sensu* Birch and Lutz 2023a). To avoid modeling all 19 vegetation categories, some of which had low replication (e.g., “Riparian”), we further aggregated all vegetation categories into a ‘Forest’ and ‘Non-forest’ category based on the primary type of fuels present in each ecosystem. We chose a forest and non-forest comparison because forested ecosystems are more likely to contain fuels with high moisture time-lags (e.g., 100-h and 1000-h fuels) than non-forest types and forested ecosystems typically exist at higher elevations in Utah where cooler temperatures and lingering snowpack may alter the timing of ignitions and fire behavior. To calculate the elevation of each pixel we extracted elevation using the raster 3.6–11 R package (Hijmans 2022) from a 30 × 30 m digital elevation model (Utah Geospatial Resource Center 2023). We excluded burned perennial grassland that were > 2000 m in elevation from vegetation-type specific analyses to minimize the impact of the uniquely bimodal distribution of this vegetation type. When calculating the frequency of annual fires by vegetation type, we counted all fires in a given year that burned at least one pixel of a vegetation type (e.g., “Douglas-fir”) or a fuel category (e.g., “forest”).

Table 1 The modified vegetation classification schema from Birch and Lutz (2023a) derived from LANDFIRE EVT (2018) for fires ≥40 ha between 1984 and 2021 in Utah, USA. Mean elevation ±1 sd for each vegetation type

Vegetation type	Fuel type	Mean elevation (m)	Area burned (ha)
Alpine	Non-forest	2416±328	13,779
Agriculture	-	-	-
Annual grassland	Non-forest	2525±318	13,035
Aspen	Forest	2684±228	63,902
Chaparral	Forest	1585±282	14,709
Developed	-	-	-
Douglas-fir	Forest	2697±217	58,174
Five-needle pine	Forest	2659±418	20,855
Lodgepole	Forest	2882±174	9,791
Mountain Mahogany	Forest	2346±244	14,549
Piñón - Juniper	Forest	2005±331	249,141
Perennial grassland	Non-forest	1506±212	205,165
Ponderosa Pine	Forest	2450±210	23,197
Riparian	Forest	1842±699	4,938
Riparian-hardwood	Forest	1800±403	7,102
Sagebrush	Non-forest	1830±329	776,511
Shrubland	Non-forest	1457±217	371,557
Snow	-	-	-
Sparse	Non-forest	2279±630	27,989
Spruce-fir	Forest	2980±236	46,055
Water	-	-	-
WUI Shrub	Non-forest	2047±350	22,857
WUI Woodland	Forest	2055±246	149,082
Forests	Forest	1800–2882	661,495
Non-forests	Non-forest	1506–2525	1,430,893

2.4 Climatic data and grouping of burned pixels

To identify simple climatic predictors of area burned and fire frequency we selected the number of days in which air temperature exceeded a threshold temperature, and the number of days in which precipitation was < 0.1 mm. These indices provide intuitive, easy-to-calculate values that have empirical connections to fire behavior through influencing fuel moisture and the availability of fuels. Whereas other fire models frequently use the vapor pressure deficit (VPD, Abatzoglou et al. 2021; Goss et al. 2020; Parks et al. 2025a), the VPD is often used to represent fuel aridity, and is a function of both air temperature and humidity, and thus not independent of temperature. Moreover, humidity observations are typically more uncertain and spatially limited, especially in mountainous regions. In contrast, precipitation is observed independently from temperature using different instruments and procedures. As a result, defining our indices based on temperature and precipitation combines two independently observed variables, providing a more robust and physically consistent framework for analysis – particularly in the mountainous topography of Utah.

We retrieved daily weather data for the state of Utah from the Global Historical Climatology Network (GHCN, 2024) via the Utah Climate Center. The GHCN is a comprehensive database of daily climate summaries from land-based weather stations, providing observations of precipitation, snowfall, snow depth, maximum and minimum temperatures, evapotranspiration, and multi-day precipitation totals. Among these variables, air temperature and precipitation offer the longest continuous records, with data extending back to the early 1900s in many locations. We categorized GHCN climate stations into higher elevations (ranging from 1950 to 2500 m, Table S1) and lower elevations (ranging from 1400 to 1950 m, Table S2) to match the patterns of forested and non-forested vegetation (Table 1) and analyzed climatic data for the periods 1984 to 2021 and 1961 to 2021, respectively. We selected only climate stations that had data availability > 90% over the period of record 1961–2021. For the high elevation stations, we used a > 80% coverage threshold to increase sample size (Table S1).

Using the compiled weather stations, we first calculated the station averages for daily precipitation and maximum air temperature for both lower and higher elevations separately. We then calculated indices based on temperature and temperature plus precipitation, or 'hot days' and 'dry days', respectively. To capture a broad range of fire-conducive temperature conditions, we tested several temperature thresholds: 21.1 °C, 23.9 °C, 26.7 °C, 29.4 °C, 32.2 °C, 35.0 °C, and 37.8 °C (corresponding to 70°F to 100°F, in 5°F increments). We calculated hot days by flagging each day at each station when the daily maximum temperature exceeded the threshold of interest (e.g., ≥ 26.6 °C [80°F]) and assigning a value of '1' if more than half of the stations exceeded the threshold. We assessed dry days as the total number of days in each year with daily precipitation < 0.1 mm/day. This criterion excludes trace amounts of precipitation, thereby minimizing the misclassification of dry days as wet days caused by blowing snow or fog, which may register as small precipitation values in gauges. Hot and dry days are the total number of days in each year when both the daily precipitation is < 0.1 mm/day and the daily maximum air temperature exceeds the specified thresholds. Finally, we calculated the cumulative sum for each calendar year of days that exceeded the threshold. These indices were selected in part for their ease of calculation for land managers, policy makers, and stakeholders and because setting a temperature threshold that is too high would lead to very few exceedances at high elevations, whereas a threshold that is too low would not represent hot days associated with elevated fire risk.

2.5 Analyses

All analyses were conducted using the statistical program R 4.2.1 in the graphical user interface R Studio 2024.04.2 (R Core Team 2021; R Studio Team 2020). We made graphs using ggplot2 3.5.1, ggpubr 0.6.0, and the plotmo 3.6.3 R packages (Kassambara 2023; Milborrow 2022; Wickham 2016). To detect trends in the frequency of hot days and hot and dry days through time, we separately analyzed low and high-elevation climate data for the period 1960–2021 using a non-parametric Mann-Kendall test and the Theil-Sen method (*sensu* Chikamoto et al. 2023) in the Kendall 2.2.1 R package (McLeod 2022). We calculated the mean number of hot days for the mean of the period 1984–2021 and used the projected trend through time to estimate the mean number of hot days in 2050 separately for forest and non-forest ecosystems.

We used a generalized additive model (GAM) approach to model non-linear relationships between variable climates associated with annual patterns of wildfire for the period 1984–2021. Our independent variables included our climatic indices of hot days and hot-dry days whereas our dependent variables included annual area burned, annual frequency of wildfires, and the mean area burned by each fire in a given year (area/frequency, Fig. 1).

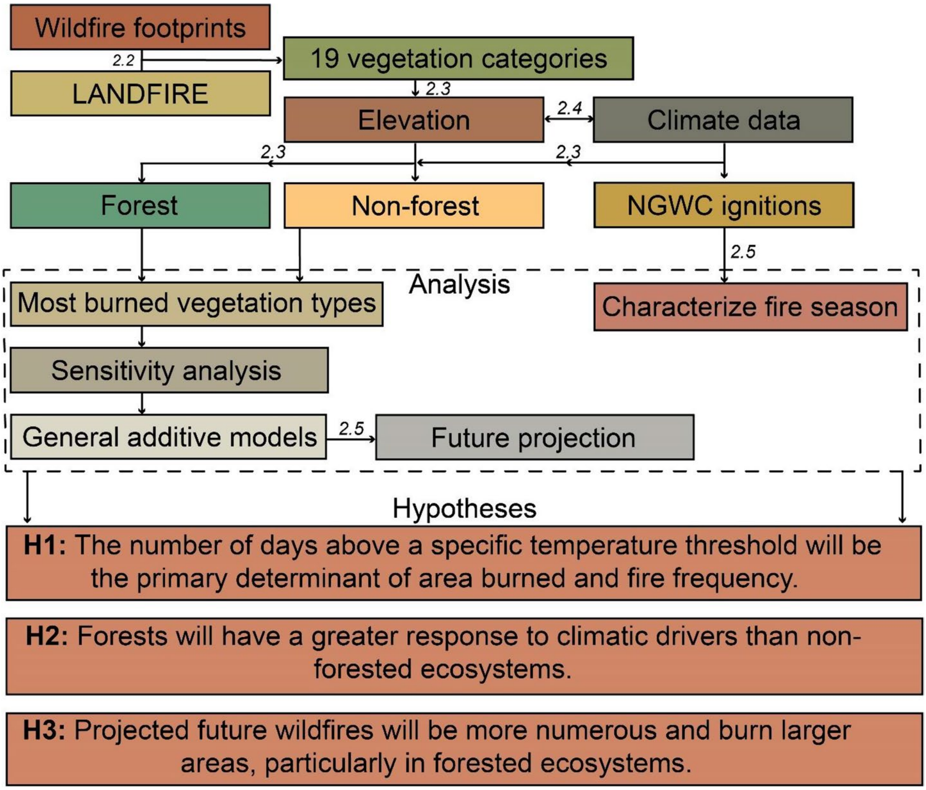

Fig. 1 Diagram of the analytical flow used for combining and processing data. Directional arrows are labeled with the section of the methods describing how data was manipulated

We truncated all analyses to 1984–2021, the period for which we had the most complete fire footprint data and for which high-quality climatic data was readily available. We built separate models for forest and non-forest pixels and each of the five forested and non-forested vegetation types with the greatest area burned as well as two wildland-urban interface (WUI) categories which included WUI shrubland and WUI woodland.

For each model, we used the mgcv 1.8–40 R package (Wood 2010, 2023), with smoothing parameters selected by restricted maximum likelihood. We used the 'gam.check' function to check smoothing fits and adjusted smoothing parameters as needed. We used the Gamma family for all models which had a lower AIC than the default gaussian model and modeled the variance scaling with the square of the mean. To identify the most parsimonious climate variables for our final models we calculated the Akaike Information Criterion (AIC) for all possible variable combinations and chose the best (lowest) AIC model. For each of the final models, we graphically denoted the mean number of hot days for the period 1984–2021 (represented by the year 2002) and the projected mean number of hot days projected for the year 2050 using the existing linear trends present at our suite of stations.

3 Results

3.1 Identifying the climatic drivers of area burned and fire frequency

The number of days ≥ 26.6 °C (hereafter 'hot days') had the lowest (best) AIC model for both forest and non-forest area burned, though the frequency models responded to higher temperature thresholds (Fig. 2). The number of hot dry days consistently underperformed the simpler metric of hot days alone. Variation explained was slightly better for higher temperature thresholds (Fig. S1).

3.2 Trend of hot day frequency through time and timing of ignitions

Between 1960 and 2020 there were positive, increasing trends of hot day frequency through time for both high-elevation forests (0.214 days year^{-1}; $p = 0.012$; Fig. 3a) and low-elevation non-forested ecosystems (0.199 days year^{-1}; $p = 0.004$; Fig. 3b).

The majority (79%) of ignitions in the period 1984–2021 occurred between May-August, with ignitions peaking in July (33%) and August (25%, Fig. 4). The peak date for ignitions was July 24th (a celebratory holiday in Utah associated with fireworks) whereas the median date of ignition for the largest fire in each year was July 13th (Table S3; Table S4). Only

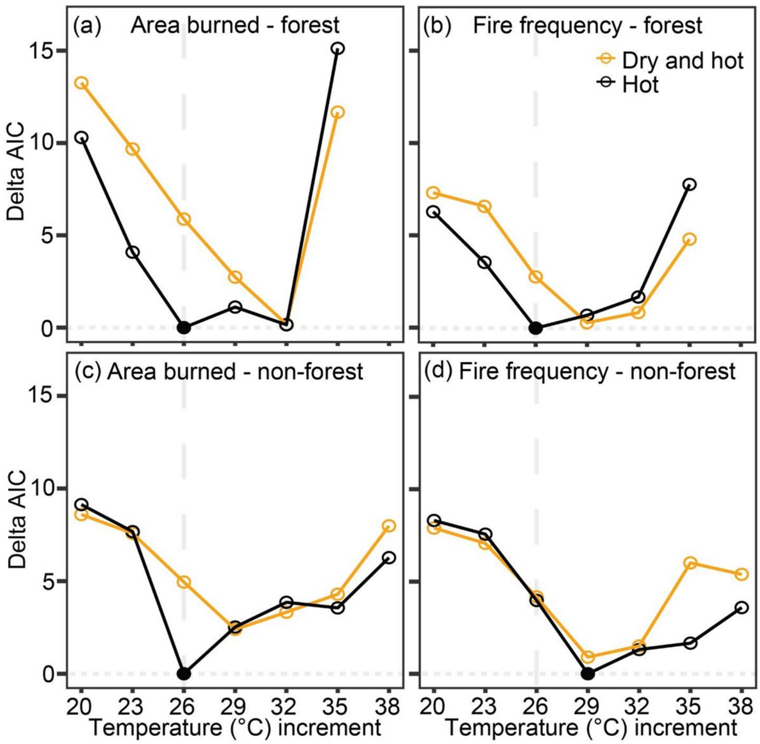

Fig. 2 The Akaike Information Criterion (AIC) for each model for area burned (**a,b**) and fire frequency (**c;d**) for forest (**a;c**) and non-forest (**b;d**) vegetation types in Utah, USA. A vertical long-dashed line denotes the temperature threshold (26.6 °C, 80 °F) selected for further analysis

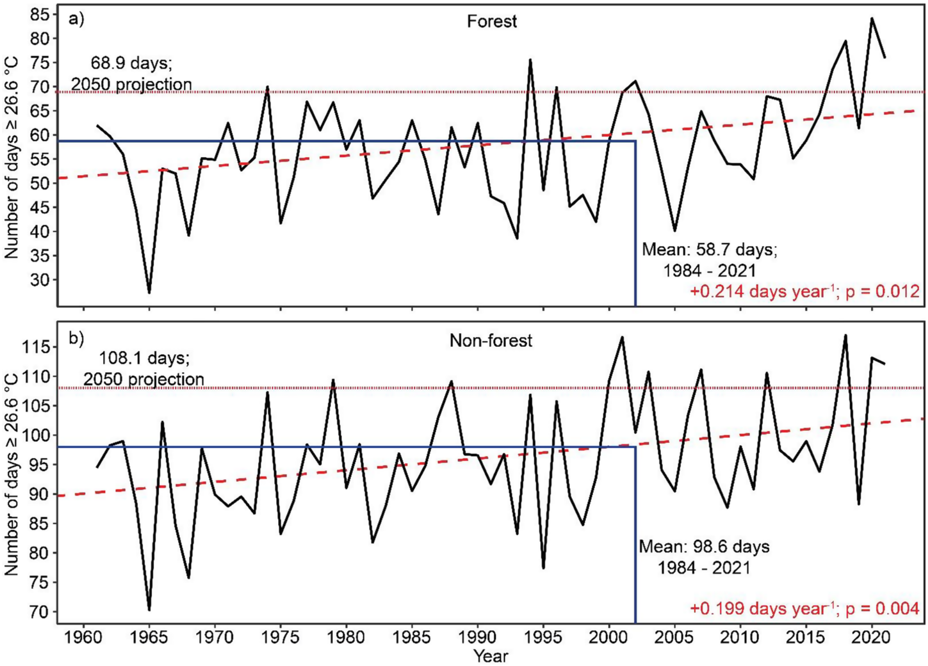

Fig. 3 Trends in hot days (≥26.6 °C) for forest (**a**) and non-forest (**b**) ecosystems in Utah, USA for the period 1960–2020. Blue lines denote the mean number of hot days for the period 1984–2021 and red fine-dashed lines indicate the projected mean in the year 2050 based on the existing linear trend (red, long-dash line) in hot days

6% of ignitions resulted in a fire ≥ 40 ha. Over the period of record, hot day frequency was largely restricted to the period of June – Sept before declining rapidly in frequency by October (Fig. 4).

3.3 Climatic drivers of fire frequency and size

Area burned was well-explained by hot day frequency for forests ($R^2=0.31$; $p<0.001$) and non-forests ($R^2=0.31$; $p=0.021$) with a positive, non-linear trend of increasing area burned with more hot days (Fig. 5a:b). The frequency of fires was moderately well-explained for forests ($R^2=0.17$; $p=0.009$) and poorly for non-forests ($R^2=0.11$; $p=0.034$) with a positive, linear trend of increasing fire frequency with more days > 26.6 °C (Fig. 5c:d). The mean area burned by each fire was well-explained for forests ($R^2=0.41$; $p<0.001$) and for non-forests ($R^2=0.27$; $p=0.021$) with a positive, non-linear trend of increasing area burned per fire with more days > 26.6 °C (Fig. 5e:f).

3.4 Vegetation type models of climatic drivers of fire frequency and size

Area burned in all forested ecosystem types was well explained by the number of hot days > 26.6 °C (Fig. 6, Table S5) with strong trends of increasing area burned with greater hot day frequency. Fire frequency in all forested ecosystem types was well explained by

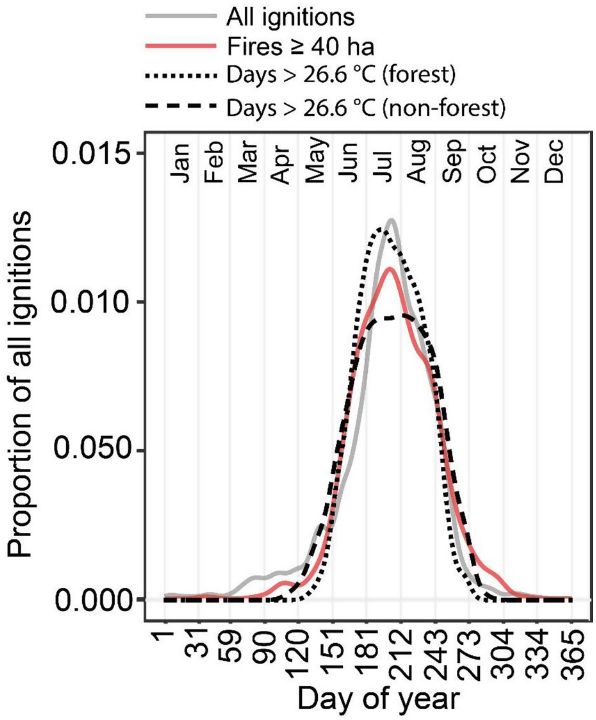

Fig. 4 Annual pattern of ignitions, wildfires burning ≥ 40 ha, and abundance of hot days (> 26.6 °C) in Utah, USA from 1984 - 2021. The median peak of ignitions and wildfire occurs on July 24th

number of days > 26.6 °C (Fig. 6) with trends of increasing area burned with greater hot day frequency. In contrast, non-forested ecosystem types had considerable variation and significant increasing trends for area burned in only sagebrush steppe and WUI woodland (Fig. 6c, o, Table S5). Fire frequency had significant increasing trends for all but shrubland ecosystem types (Fig. 6).

3.5 Projected changes in burn area and frequency by 2050

Between the median year of the period 1984–2021 (2002) and 2050, hot days are expected to increase by 10.2 days for forest ecosystems (17% increase) and 9.5 days in non-forest ecosystems (9% increase; Fig. 2). The increase in hot day frequency are predicted to lead to a 60% increase in area burned for forest ecosystems and a 232% in area burned for non-forests (Fig. 5; Table 2). Fire frequency is predicted to increase by 24% for forests and 25% for non-forests (Fig. 5; Table 2).

4 Discussion

We identified a simple climatic metric, days above ≥ 26.6 °C, that well-explained patterns of burn area and fire frequency for differing vegetation types in Utah, USA. This metric is simple to track during the year with reference to easily available information. Our results supported our hypotheses H1) that hot days would strongly relate to area burned and that H3) models would explain more variation in forests than non-forests. However, our results

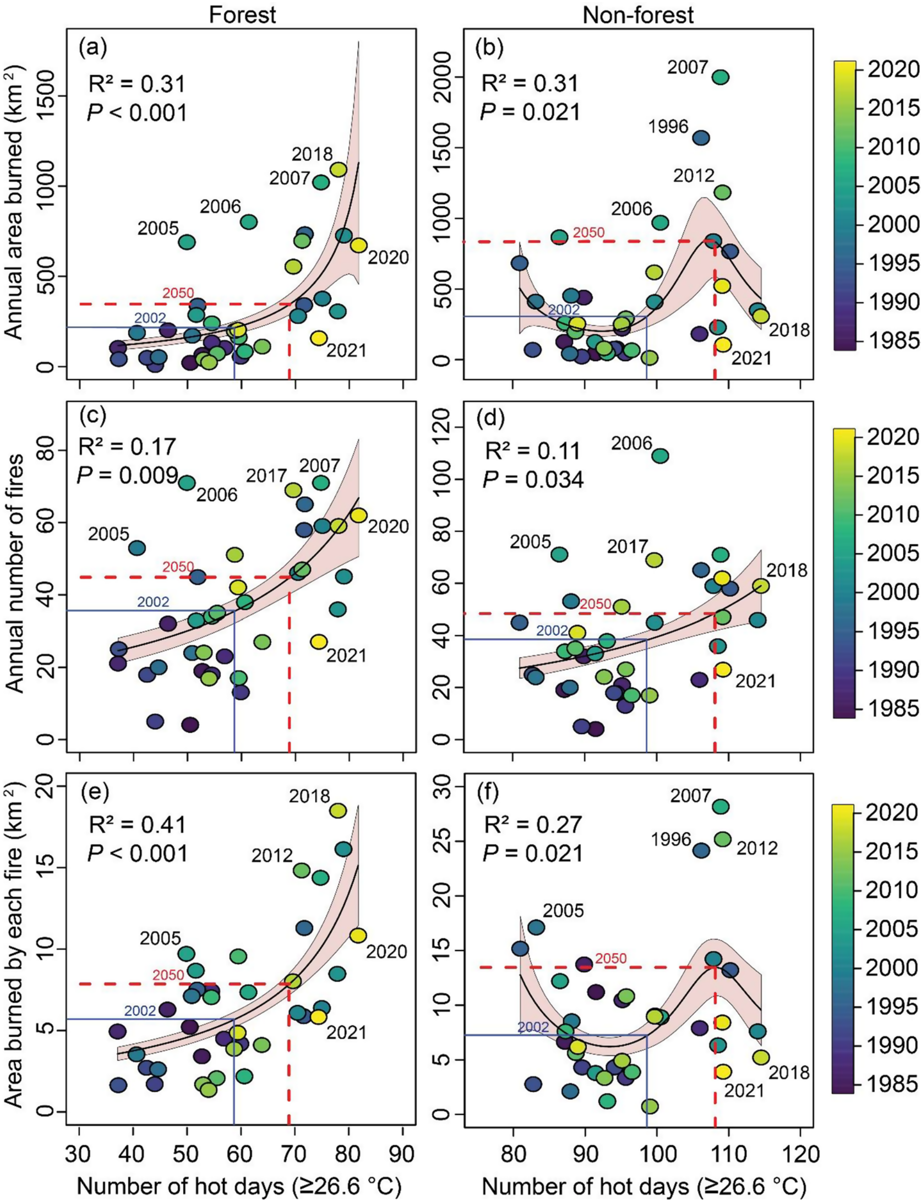

Fig. 5 Model output of the annual area burned **(a:b)**, fire frequency **(c:d)**, and area burned by each fire **(e:f)** for forested and non-forest ecosystems in Utah, USA from 1984 to 2021. Points are colored according to their respective year. Blue lines denote the mean number of hot days (x-axis, days ≥ 26.6 °C) for the period 1984–2021 (median year 2002). Red dashed lines denote the projected number of hot days in 2050. Yearly labels denote notable outliers

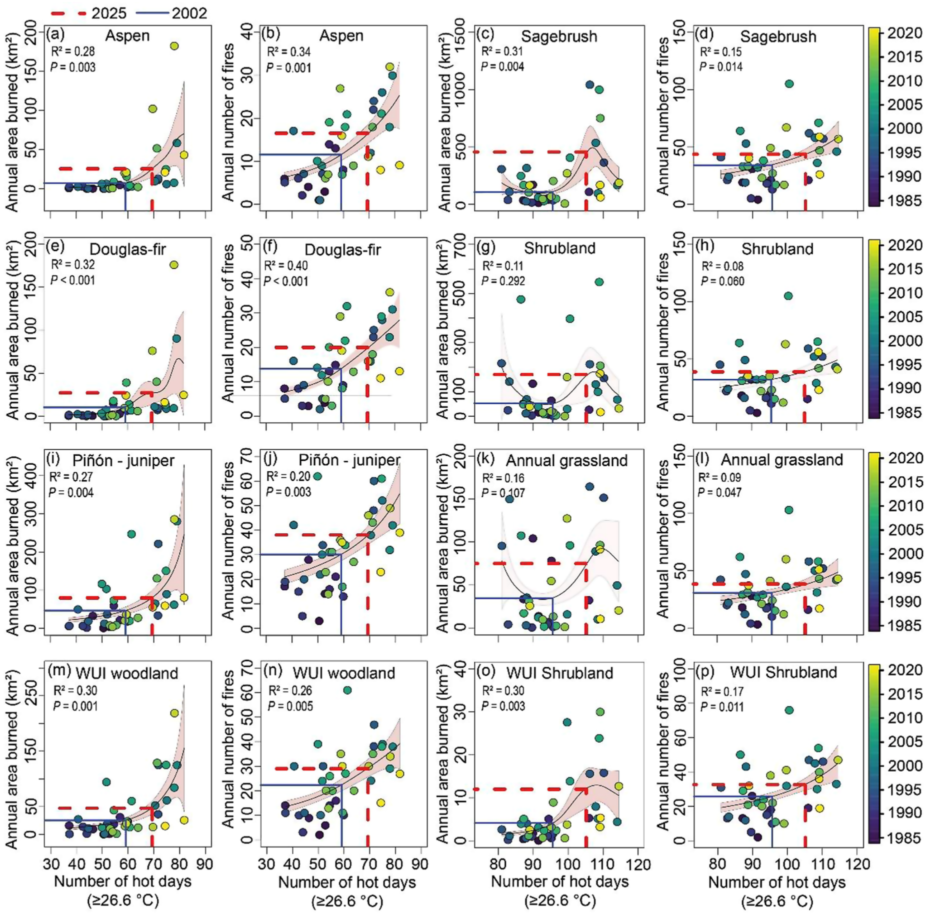

Fig. 6 Model output for forest and non-forest ecosystems for the annual area burned and fire frequency in Utah, USA from 1984 to 2021. Points are colored according to their respective year. Blue lines denote the mean number of hot days (x-axis, days ≥ 26.6 °C) for the period 1984–2021 (median year 2002). Red dashed lines denote the projected number of hot days in 2050. Non-significant patterns are marked with reduced opacity

are mixed in supporting H2) that future wildfires may burn more area in forests than non-forests. Despite high uncertainty in non-forests, the projected increases in area burned and frequency (Table 2) are greater than in forests. Whereas more complicated models incorporating fine-scale modeling of terrain, fuel, or fire weather may improve model accuracy, our metric of hot days is easily calculatable and effectively estimates wildfire season length in Utah, making for an accessible and interpretable metric for managers, stakeholders, and researchers.

In the varied landscapes of Utah and the broader western United States the forcing of wildfire behavior by temperature alone should not be underestimated, particularly for productive high-elevation ecosystems where fire – for the foreseeable future – is likely to be climate-limited and not fuel-limited. Importantly, extrapolating to other regions (e.g., California, or outside North America) will require identifying region-specific temperature or

Table 2 Modeled area burned and fire frequency, by vegetation type, for Utah, USA. Values for 2002 are model estimations based on the number of hot days (days ≥ 26.6 °C) and its relationship with area burned and fire frequency 1984–2021. Projections for 2050 are based on projected increases in the number of hot days by 2050. Values in 2002 are based on the median of our modeled period of record, and do not reflect actual area burned nor fire frequency in 2002

Vegetation type	**Area burned (2002; km^2)**	**Area burned (2050; km^2)**	Increase in area (%)	Fire Frequency (2002)	Fire Frequency (2050)	Increase in fire frequency (%)
Forest	219	352	60	36	44	24
Aspen	7	25	257	11	16	45
Douglas-fir	10.5	26	147	14	19	35
Piñón-Juniper	47	80	70	30	36	20
WUI Woodland	25	47	88	22	29	31
Non-Forest	228	759	232	37	46	25
Sagebrush	108	443	310	34	47	38
Shrubland	51	166	225	32	38	18
Annual grassland	35	72	105	31	37	19
WUI shrubland	4	12	200	26	32	23

moisture thresholds that reflect local conditions and fire regimes. Region-specific hot-days may have value as a metric to be paired with local knowledge and fire-danger rating systems for in-season assessment of potential fire behavior.

4.1 Summer temperature acts as a primary control on wildfire in Utah

Air temperature acts as a primary driver of fuel moisture with higher temperatures being associated with more rapid drying of fuels and greater vapor pressure deficits, though relative humidity, wind, and solar radiation also play key roles in fuel curing (Matthews 2013). Higher temperatures also reduce the amount of energy needed for ignition and fire propagation. Because Utah experiences most precipitation during the winter months, most precipitation events during fire season are likely insufficient to fully wet fuels and this is likely the primary reason that hot dry days underperformed, relative to hot days alone. As well, precipitation events can induce a time-lagged influence on fuels – such as rapid growth of *B. tectorum* – which influence fire behavior in subsequent years after new growth has senesced and become readily available for fire propagation (Pilliod et al. 2017; Terry et al. 2024). Our results contrast with Brown et al. (2008) who reconstructed Utah fire regimes 1630–1900 from tree rings and identified regional fire synchronicity with drier summer conditions and a non-significant trend of more fires during warmer summers. Historical reconstructions of summer soil moisture (1750–1880; Margolis et al. 2025) are spatially correlated with fire frequency in Utah and western North America. However, this apparent disagreement between our results and existing studies may be due to the inherent limitations of reconstructing pre-instrumental climate from proxies, such as tree rings (Briffa et al. 1992; Brown et al. 2008), the presence of precipitation time-lags through soil moisture or influence on vegetation productivity, or because of past indigenous use of cultural fire influencing historical fire regimes (e.g., Carter et al. 2021; Roos et al., 2021) and changes in the spatial and temporal pattern of modern-day ignitions.

Our projected increases in area burned for forests (60%, Table 2) were comparable to the lower end of the modeled increases of 46–107% identified by Abatzoglou et al. (2021) which focused on the entire western USA. Similar to their approach, our metric captures the influence of increasing atmospheric dryness on fire activity, conceptually aligning with vapor pressure deficit (VPD) and climatic water deficit (CWD) as indicators of moisture demand under warming conditions. However, whereas VPD and CWD are typically estimated from monthly or seasonal data and require additional variables such as humidity and evapotranspiration, our daily temperature-based metric explicitly incorporates the cumulative number and timing of hot days within the fire season. This temporal framework allows us to identify not only baseline increases in aridity but also the seasonal expansion of conditions conducive to burning. One pathway in which more fire may occur under future climates is through a lengthening fire season – a trend already in evidence across the western USA (Abatzoglou and Williams 2016). Longer fire seasons in Utah may shift into September and October rather than March or April, when the influence of seasonal and ephemeral snowpack may still be wetting fuels (Petersky and Harpold 2018, Fig. 4). For example, the 2024 Yellow Lake fire started late in the fire season (Sept 28th, Fig. 4) and burned 13,378 ha of high-elevation forest – far surpassing area burned for all earlier fires of 2024 (Incident Information System 2025). As the number of hot days increase in the future, we may expect that longer fire seasons will result in fires occurring into months outside of the natural range of variability for much of Utah.

4.2 Potential long-term impacts to Utah's forests from shifting fire regimes

Forests are slow to develop and have long successional times ranging from centuries to millennia. These projected increases in area burned and fire frequency may shift many forests towards early-seral systems or even promote vegetation type change to non-forested systems (McKenzie et al. 2004). Fewer, larger fires are increasingly burning larger proportions of Utah's landscape at higher severities than smaller fires (Birch and Lutz 2023a) which may not only kill and injure mature trees but also remove snags and deadwood needed for the successful recruitment of new trees (Birch and Lutz 2023; Swanson et al. 2023) and preservation of late seral conditions. Although projected increases in burned area and fire frequency are higher for non-forests, relative to forests, the uncertainty in the confidence interval is large. Some outlying years were not well-explained by our models (e.g., 2007, 2018), but these anomalies in Utah match broader trends in the USA (Fig. S2, National Interagency Coordination Center 2024) suggesting a regional climatic forcing, possibly from ENSO, the Pacific Decadal Oscillation, or yet-unknown factors, such as fire suppression operations or other climatic drivers. These outlying years suggest that it is important to continue to refine models at both the scale of the entire West as well as the scale of individual vegetation types within states. Fire suppression over the period of record may have confounded or weakened some of our results owing to successful suppression efforts minimizing fire frequency ≥ 40 ha as well as area burned. Future research would benefit from accessible, centralized information on suppression operations which may more clearly define how tactical decision making alters wildfire frequency, area burned, and severity across ecosystem types.

4.3 Future fire may act unequally depending on vegetation and feedbacks with fuels

Future increases in fire frequency and area burned will likely be dampened by fuel limitations owing to declining ecosystem productivity (and thus fuel accumulation) under warmer and drier climates (Anderson-Teixeira et al. 2022; Ning et al. 2024). However, fuel limitations on fire behavior are likely to be many decades away in most high-elevation forests (e.g., spruce-fir or Douglas-fir) which have substantial fuel loadings that are unlikely to be consumed in a single fire event (Birch and Lutz, 2023b; Lutz et al. 2020). Changes in species distributions under future climates could further alter fire regimes as hot and dry-adapted species become more abundant at higher elevations and more northerly locations.

Most studies of broad spatial extent use gridded climate data, and those grids overlap many diverse vegetation types. More than half of all area burned in Utah 1984–2021 was in sagebrush and shrubland ecosystems, where fire behavior was poorly explained (Fig. 6) by hot days, highlighting potentially uneven changes in future fire behavior. It is likely that fuel-limitations and topographic controls are partially responsible for the lower model performance for non-forest ecosystems. Invasive annual grasses have recently altered fire seasonality, increased fire frequency, and area burned in many of the non-forest ecosystems of Utah (Balch et al. 2013; Bradley et al. 2018) which may have contributed to the lower model performances in these ecosystems, relative to forests. Suppression operations may also be more effective in limiting fire size in non-forest ecosystems which are generally easier to access than high-elevation forests.

The variation in model performance and trends with climate highlight the utility of developing vegetation-specific models. A single model for the state of Utah would have been heavily weighted towards sagebrush and shrubland (~50% of area burned), which had poorly performing models, and would have masked the climate-fire relationships in forest ecosystems. Where feasible, other regional or global models predicting area burned or fire frequency may benefit from different models for different vegetation types to improve accuracy and relevancy to managers.

4.4 Looking to the past: is there an analog in the historical record?

Are future temperatures outside of the historic range of variability for Utah? Maximum summer temperature reconstructions by Heeter et al. (2021; their Fig. 2) over the previous 1250 years noted several, multi-decadal warming events and temperatures several degrees centigrade above the mean – most notably between 1050 – 1070. However, recent warming trends are on track to eclipse historical warming events (Heeter et al. 2021) and will likely push Utah into climatic extremes not seen in the last millennia. Whereas past climate has been reasonably reconstructed in the western USA (e.g., Briffa et al. 1992; Heeter et al. 2021), much less is known about past fire regimes in Utah, particularly in non-forested ecosystems where fire scars on trees are absent. What evidence does exist (Brown et al. 2008; Heyerdahl et al. 2011; Parks et al., 2025b) suggests that past fire regimes in Utah were influenced by a combination of both low-frequency variation in climate (e.g., El Niño-Southern Oscillation or Pacific Decadal Oscillation) and annual and sub-annual variation depending on the ecosystem type. Fire frequency in forests pre-1880 was likely much higher (e.g., four-fold; Parks et al., 2025b) than modern fire regimes.

Much like today, historical fire was likely more frequent adjacent to human habitation (Carter et al. 2021; Roos et al. 2021) either as a deliberate action or due to inadvertent ignitions. Whereas the cessation of cultural burning and fuel management has resulted in forest densification and an increase in fuel loadings in some western forests (Markwith and Paudel 2022; Steel et al. 2015), it is unclear the extent to which fire suppression and accordant policies have influenced modern fire regimes in Utah, particularly in non-forested ecosystems. Future increases in fire frequency in some ecosystems may be inside historical ranges of variability, pre fire suppression (Parks et al. 2025b), but could still see uncharacteristically severe fire effects owing to legacies of fire suppression (e.g., fuel accumulation), changes in climatic drivers, or novel disturbance regimes (Abatzoglou et al. 2019; Barbero et al. 2015). As well, our projections for future fire do not account for extreme fire weather events which often have outsized influence on large fire spread and whose frequency could change unpredictably under future climates.

4.5 Management interventions and implications

As temperatures increase in Utah over the next 25 years, we should expect more fire in both forested and non-forested vegetation types until – and if – fuel loadings becomes the chief control on fire propagation. Increases in fire in forested areas are likely to have long-lasting effects. Much of Utah's forests burn at mixed severities (Birch and Lutz 2023a), and high severity patches within those fire footprints will likely take many years to decades to recover to pre-disturbance states. Fire frequently provides opportunity for non-native species to establish or increase prevalence (Kerns and Day 2017) which may in turn alter future fire regimes through changes to fuel structure and availability (Balch et al. 2013). Owing to the continued invasion of post-fire ecosystems by undesirable species (e.g., *Tamarix* spp., Busch and Smith 1993; *B. tectorum*; Peeler and Smithwick 2018), our models likely under-predict the consequences of future changes in fire frequency owing to changes in invasive species that generally increase fire.

As the trend in area burned increases, fuel treatments, like mastication, pile burning, but especially prescribed burns and wildfire managed for resource benefits, may reduce the risk of excessive tree mortality (Birch et al. 2023c; Brodie et al. 2024) and decrease landscape-scale fuel continuity and the risk of catastrophically large fires. Reducing landscape fuel continuity and promoting fire hardening and defensible spaces may be especially important in WUI ecosystems which have experienced increasingly catastrophic wildfire (Cunningham et al. 2025), and for which we predict 88–200% increases in area burned by 2050 (Table 2).

Post-fire restoration efforts, such as seeding with native species, preserving and promoting micro-scale habitats (e.g., nurse logs; Birch and Lutz 2023b; Swanson et al. 2023), and long-term fuel reduction efforts will likely be needed to promote ecosystem resilience under future fire regimes (Brodie et al. 2024). Increased prescribed fire in non-forest areas such as sagebrush steppe may be a practical tool to mitigate potential increases in fire. As well, restoration of native species to reduce invasive annual grass cover may limit fuel continuity and reduce the risks of overly large fires.

5 Conclusion

Warming climates will drive dramatic and unequal shifts in wildfire that will disproportionately impact high-elevation forests. The relationship between temperature and area burned in Utah provides a single relationship which is intuitive, simple to calculate, and has considerable skill in predicting area burned and fire frequency in many of the ecosystems in Utah. These models may provide estimates of future fire effects and guide research priorities and management policy in expectation of increased fire in the next several decades. Even in Utah, with relatively less expansive forests than other areas of the West, increases in summer temperatures alone are likely to result in increases of 160% area burned by 2050. Management action, whether through prescribed burning or other fuel reduction treatments will likely be needed to prevent the loss of ecosystem resilience, preserve human habitation in the WUI, and mitigate the risk of mega-fires.

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Data availability Data are available at the Utah State University Libraries (Birch et al. 2025).

Declarations

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Authors and Affiliations

Joseph D. Birch[1] · Yoshimitsu Chikamoto[2] · James A. Lutz[1]

Joseph D. Birch
joseph.cooper@usu.edu

Yoshimitsu Chikamoto
yoshi.chikamoto@usu.edu

James A. Lutz
james.lutz@usu.edu

1 Department of Wildland Resources and the Ecology Center, Utah State University, Logan, Utah 84321, USA

2 Department of Plants, Soils and Climate, Utah State University, Logan, Utah 84321, USA