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Could restoration of a landscape to a pre-European historical vegetation condition reduce burn probability?

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Abstract. Montane regions throughout western North America have experienced increases in forest canopy closure and forest encroachment into grasslands over the past century; this has been attributed to climate change and fire suppression/exclusion. These changes threaten ecological values and potentially increase probabilities of intense wildfire. Restoration of landscapes to historical conditions has been proposed as a potential solution. We used historical oblique photographs of an area in the Rocky Mountains of Alberta, Canada, to determine the vegetation composition in 1909 and then asked whether restoration to a historical vegetation condition would: (1) reduce the overall burn probability of fire; (2) reduce the probability of high-intensity fires; and (3) change the spatial pattern of burn probabilities, as compared to current conditions. We used the Burn-P3 model to calculate the overall and high-intensity burn probabilities in two scenarios: (1) the baseline (current (2014) vegetation composition) and (2) historical restoration (vegetation in the study area as of 1909 with the surrounding landscape in its current condition). In the baseline, the landscape had 50% less grassland and more coniferous forest than 100 yr ago. Except for the fuel grids, we ensured all input parameters (number and locations of ignitions, weather conditions, etc.) were identical between the two scenarios; therefore, any differences in outputs are solely attributable to the changed fuels. The historical restoration scenario reduced the overall burn probability by only 1.3%, but the probability of high-intensity wildfires was reduced by nearly half (44.2%), as compared to the baseline scenario. There were also differences in the spatial pattern of overall burn probabilities between the two scenarios. While 6.7% of the landscape burned with half (or less) the probability in the restoration scenario (compared to the baseline), other areas (3.2%) had burn probabilities two to five times higher. More than 21.5% had high-intensity burn probabilities that were 20% or less of those in the baseline scenario. Differences in burn probabilities between the two scenarios were largely attributable to the effects of the vegetation difference on rate of fire spread. Restoration to historical vegetation structure significantly lowered wildfire risk to the landscape.

Key words: burn probability; burn-P3 model; ecological restoration; forest encroachment; historical ecology; indicator kriging; landscape change; repeat photography; vegetation change; wildfire intensity.

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Introduction

Studies have shown that forest cover throughout many places in western North America is more homogenous and continuous in the early 2000s than it was at the time of large-scale European settlement in the late 1800s and early 1900s (Arno and Gruell 1983, Gruell 1983, Rhemtulla et al. 2002, Hessburg et al. 2005, Stockdale 2017, Stockdale et al., *unpublished manuscript*). Between roughly 1900 and 2000 AD, there have been significant shifts from grasslands and open canopy woodlands to closed canopy forests across the forest–grassland interface of the Canadian prairie regions (Strong 1977, Campbell et al. 1994, Stockdale 2017), Rocky Mountains (Gruell 1983, Brown et al. 1999, Rhemtulla et al. 2002, Stockdale 2017) and intermountain west in the United States and Canada (Hessburg et al. 2005). While it is not inherently problematic that vegetation change is occurring across large areas of the landscape, there is considerable evidence that suggests this shift away from open canopy forests, grasslands, and meadows is outside the natural range of variability (NRV) for these ecosystems (Fulé et al. 2002, Agee 2003, Hessburg and Povak 2015). These dramatic changes in vegetation have thus led to concerns regarding ecological values and processes on these landscapes. Encroaching forests are a threat to rangeland resources (Gruell 1983, Archer 1994) and threaten biodiversity of grasslands at lower elevations (Haugo and Halpern 2010) and of subalpine/alpine meadows at higher elevations (Franklin et al. 1971).

Large-scale vegetation changes have been observed in much of the intermountain west region of the United States where the fire regime condition class assessment tool (Barrett et al. 2010) has been used to show that nearly half of the currently forested landscape of Washington, Oregon, Montana, and Idaho is no longer considered to be within its natural range of variability with regard to its vegetation composition or fire regime (Haugo et al. 2015). In the Rocky Mountains of southern Alberta, Canada, 25% of grasslands and 39% of open canopy woodlands have converted to later successional stages over the last century, while forested area has increased dramatically (by 35–80%; Stockdale 2017). This stands in stark contrast to the expectation that forests should be shifting in favor of grasslands under future climates (Wang et al. 2012, Schneider 2013, Stralberg et al. 2018). A continuance of the current trend could result in loss of open canopy woodlands and grasslands of the Montane Natural Subregion and their associated flora and fauna, in favor of closed canopy lodgepole pine, Douglas-fir, and Engelmann spruce forest and their associated biota.

It is widely believed that 20th-century fire suppression and exclusion is one of several key factors (in addition to climate and land-use change, among others) causing these vegetation changes (Nelson and England 1971, Arno and Gruell 1983, Archer 1994, Wakimoto and Willard 2005). Fire regimes have been altered considerably, and the overall fire frequency and annual area burned have declined throughout the Alberta Rocky Mountains (Tande 1979, Hawkes 1980, Barrett 1996, Andison 1998, Rogeau 2005, 2009), British Columbia (Gray 2003, Kubian 2013), and the western United States (Arno 1980, Barrett et al. 1997, Hessburg et al. 2005, Prichard et al. 2009). Vegetation has changed because of this lack of fire (Arno and Gruell 1983, Arno et al. 2000, Hessburg et al. 2013, Davis et al. 2018), which has created a higher risk of increased numbers and intensities of future fire (Baker 1992, Arno et al. 2000, Gallant et al. 2003, Stephens and Ruth 2005). Indeed, we are now seeing evidence of recent sharp increases in area burned (Dennison et al. 2014) and fire severity (Agee 2002, Hessburg et al. 2016) in much of the western United States, partially due to increases in the overall length of fire seasons (Albert-Green et al. 2013) and shifts toward more fires burning in the spring (Westerling et al. 2006).

These ecological changes have led numerous jurisdictions throughout Canada and the United States to invest heavily in thinning forests, change silvicultural practices, and create landscape-scale ecosystem management plans with the intent of restoring forest age class distributions, species composition, and landscape patterns to historic conditions (White et al. 2003, Brown et al. 2004, Baker et al. 2007, Walkinshaw 2008, Barrett et al. 2010, Government of Alberta 2011, Hessburg et al. 2013). Choosing an appropriate reference condition for historical ecological restoration, whether that is a single point in time or a range of possible conditions (such as the NRV), presents numerous challenges; most notably that climate change can result in novel conditions to which historic ecosystems are ill-adapted (Veblen 2003, Klenk et al. 2008, Hall 2010, Flatley and Fulé 2016). Regardless of this concern, and recognizing that there is no single right reference point (or range), the pre-European settlement period is considered to be within the natural range of variability, and is widely used throughout North America as a reference for ecological restoration in forests (Brown et al. 2004, Baker et al. 2007, Barrett et al. 2010, Churchill et al. 2013), rangelands (Fuhlendorf and Engle 2001), and protected areas (White et al. 2003, Mawdsley et al. 2009, Bjorkman and Vellend 2010, Higgs et al. 2014). An important underlying assumption of plans to restore vegetation to a historical condition is that this will improve the ecological integrity of the landscape and reduce the risk of catastrophic wildfires (Shinneman et al. 2012). This assumption is largely untested and clearly may not hold true across all ecosystems. Our purpose herein is to address this gap.

This study was designed to examine how the likelihood and intensity of fire would be altered in a location where the vegetation was restored to a historical condition. Using the Bob Creek Wildland in southern Alberta as a case study, we tested the following hypotheses:

1. Restoring landscape vegetation composition to a reasonable approximation of its pre-European settlement condition would:

   a. Alter the overall burn probability due to changes in both the composition and spatial arrangement of fuels.
   b. Change the spatial pattern of burn probabilities across the restored and surrounding landscape.
   c. Reduce the probability of high-intensity fire (≥4000 kW/m). This threshold represents the level at which fire transitions from surface to crown fire (causing significant overstory mortality), and direct attack suppression tactics become ineffective (Forestry Canada Fire Danger Group 1992).

This study compared two scenarios: (1) the baseline, which is the vegetation composition of the Bob Creek Wildland and surrounding area as of 2014; (2) the historical restoration, which is an approximation of the historical vegetation of the Bob Creek Wildland as it was in 1909 embedded in the matrix of the current landscape as of 2014. The latter represents a realistic restoration scenario in which vegetation in a specific target area would be restored while the surrounding landscape would not.

Methods

Study area

The Bob Creek Wildland (BCW) is a 20,775 ha Provincial Wildland Park (Fig. 1, Table 1) located in southern Alberta at an elevation ranging from 1345 m to 2210 m a.s.l. The area includes the Subalpine (8511 ha) and Montane Natural Subregions (12,264 ha; Natural Regions Committee 2006) and is at the boundary of the Cordilleran and Grassland ecoclimatic provinces. The Cordilleran ecoclimatic province has cold winters and very short cool summers (Natural Regions Committee 2006). The Grassland ecoclimatic province has cold winters and short hot summers; precipitation is overall low and the wettest month is June (Natural Regions Committee 2006).

Vegetation in the Subalpine consists primarily of forest dominated by lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), aspen (*Populus tremuloides*), and balsam poplar *(Populus balsamifera*) with lesser components of Engelmann spruce (*Picea engelmanii*) and subalpine fir (*Abies lasiocarpa*). Shrubs in this subregion include shrubby cinquefoil (*Potentilla fruticosa*) and creeping juniper (*Juniperus horizontalis*). The Montane has extensive fescue (*Festuceae* tribe) grasslands interspersed with forests dominated by Douglas-fir (*Pseudotsuga menziesii*), aspen, balsam poplar, with limber (*Pinus flexilis*) and whitebark pine (*Pinus albicaulis*) on the exposed rocky ridges. The common shrubs in this subregion are bog birch (*Betula glandulosa*) and several willow species (*Salix* spp.).

Fires between 1961 and 2003 were mostly human caused (75%), occurred in July and August, and exhibit a current fire cycle of roughly 400 yr (Rogeau 2005). This fire cycle is

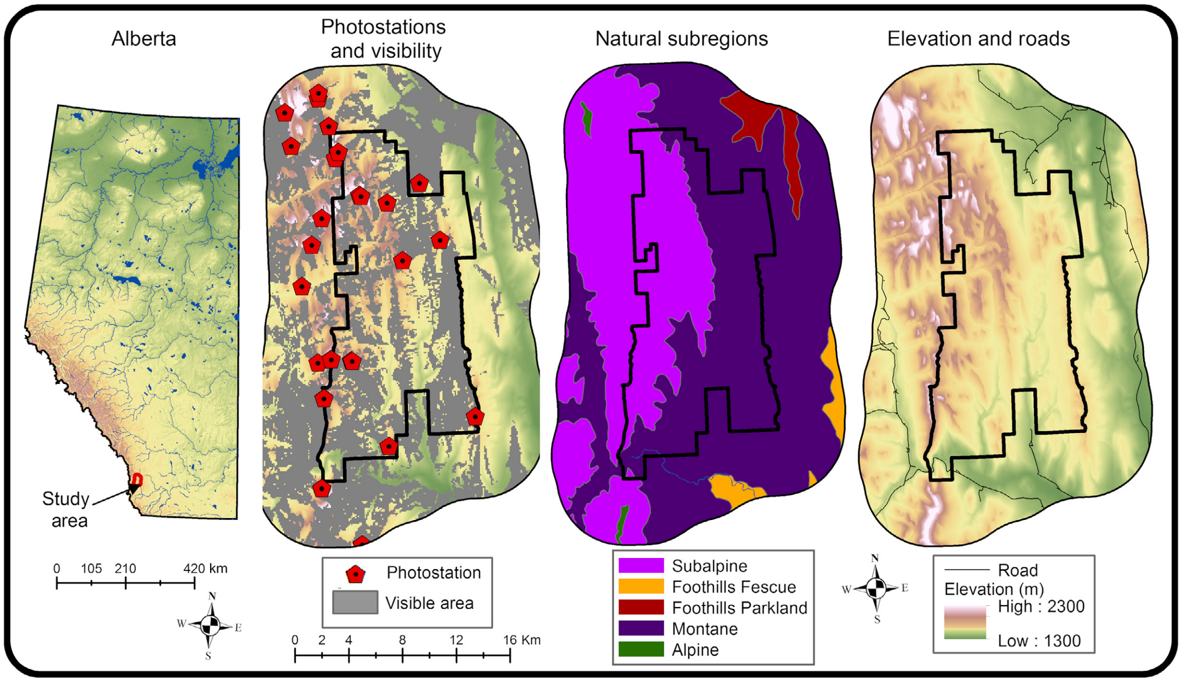

Fig. 1. Overview of the study area. Bob Creek Wildland (BCW) protected area (outlined in black) is located in the SW corner of Alberta, Canada, and the study area includes a 5-km buffer around the BCW. Photostations are shown on the map, as is the area visible from these photostations. Natural Subregions (Natural Regions Committee 2006) are shown in the third panel, and the fourth panel shows the elevation and roads. The Whaleback Ridge makes up the eastern border of the BCW.

Table 1. Study area size, number of photographs, and total area visible in the images used to reconstruct the vegetation of 1909 in the study area (Bob Creek Wildland and surrounding 5-km buffer zone).

Area	Bob Creek Wildland Only (BCW)	Bob Creek Wildland + 5-km buffer (BCW5K)
Total area	20,775 ha	66,053 ha
Visible area	12,051 ha (58%)	33,338 ha (50.4%)
No. of images used	43	61

likely considerably longer than it would have been historically; in nearby areas with similar vegetation, climate, and topography, the pre-settlement fire return interval was 15–30 yr (Montane) to 30–150 yr (Subalpine; Hawkes 1979, Arno 1980, Barrett 1996, Rogeau 2005).

Fire modeling

For both the baseline (BLS) and historical restoration (HRS) scenarios, we modeled the burn probability and expected fire intensities across the provincial fire management unit containing the BCW. The 4000 kW/m threshold indicating high-intensity fire was chosen for two reasons. First, this represents Intensity Class 5, which in the Canadian Forest Fire Behaviour Prediction (FBP) System indicates levels at which direct attack suppression tactics become ineffective (Forestry Canada Fire Danger Group 1992), and therefore, the chances of a fire escaping initial attack are higher. The second reason is ecologically based and refers to the level of mortality expected (severity) from fires burning beyond this threshold: C2 (spruce) fuel = active crown fires (full overstory mortality); C3, C4, M1–M2 fuel (mature and immature lodgepole pine, aspen) = intermittent crown fires (patchy mortality of the overstory); C7 fuel (Douglas-fir, open canopy pine) = intense surface fire, transitioning to intermittent crown (some mortality expected), D1/2 fuel (aspen) = intense surface fire, significant mortality expected; O1 fuel (grass) = intense surface fires with extreme rates of spread expected. We only evaluated model outputs within the BCW plus a 5-km buffer to capture the effect of fires igniting within the park and burning beyond the park boundary. Hereafter, this area of the BCW plus 5-km buffer is referred to as BCW5K (Table 1). The BLS represents the landscape as of 2014 and used the Government of Alberta (GOA) fuel grid (2014 baseline grid). For the HRS, we changed the BCW (not the buffered area or surrounding landscape) to a reasonable approximation of its condition as of 1909 (historical restoration grid) based on analysis of historical photographs (Stockdale 2017) taken as part of MP Bridgland's 1913–1914 Survey (Trant et al. 2015).

To model burn probability and fire intensity, we used Burn-P3, which is a Monte Carlo simulation model based on the Prometheus fire growth engine (Tymstra et al. 2010), and simulates ignition and spread of fires across the landscape. Burn-P3 combines deterministic fire growth (influenced by fuels, topography, and weather) with probabilistic fire ignition locations, fire duration, and weather (Parisien et al. 2005). We assembled Burn-P3 inputs using methods described in detail by Parisien et al. (2013). Static and stochastic inputs used to model burn probability are described in Appendix S1: Table S1. Vegetation in the Burn-P3 model is represented by FBP System fuel types (Stocks et al. 1989, Forestry Canada Fire Danger Group 1992). Except for the fuel grids, we ensured all input parameters (number and locations of ignitions, weather conditions, etc.) were identical between the two scenarios, and therefore, any differences in outputs are solely attributable to the changed fuels; however, this ignores the fact that ignition probability can be influenced by fuel type. We examined whether the Burn-P3 inputs were realistic and calibrated fire outputs to match fire history information from the period 1961 to 2014 (Appendix S1: Fig. S4a). We ran 110,000 iterations of the Burn-P3 model for the two scenarios to ensure we would have local stability in burn probability outputs for the BCW5K (Appendix S1: Fig. S4b).

Baseline and historic restoration scenario landscapes.— For the baseline scenario (BLS) landscape fuel map, we used the Alberta Provincial 2014 baseline grid. In the HRS, the vegetation within the BCW was changed to a reasonable representation of its pre-settlement condition while the surrounding landscape was the same as for the BLS above. The fuel composition within the BCW (historical restoration grid) was based on vegetation classes interpreted from historical photographs taken in 1913 (Higgs et al. 2009; Fig. 2). Large fires in 1910 (Pyne and Maclean 2008) affected roughly 8% of the total landscape surrounding and including the BCW (Stockdale 2017); we wished to avoid the transient effect of these, and thus erased the large fire by back-casting the forest structure to what it likely was in 1909 (more details in Stockdale 2017).

We used the WSL (Swiss Federal Institute for Forest, Snow and Landscape Research) Monoplotting Tool (Bozzini et al. 2012) to georeference the images and followed the procedures outlined in Stockdale et al. (2015) to extract raster data. We overlaid a spatially referenced grid over the photographs and classified the vegetation at a resolution of 1-ha/cell into one of seven vegetation classes: conifer (CF), broadleaf deciduous (BD) or mixedwood (MX) forest, shrubland, open canopy woodland (WD), grassland (MG), or non-vegetated. The accuracy of our historic landscape reconstruction is addressed in Stockdale et al., *unpublished manuscript*.

For back-casting forest structure to 1909, we determined the 1909 vegetation category by the density and form of the standing dead timber (in areas where the 1910 fire was clear and obvious) in the 1913 images; areas with dense coniferous snags were classified as CF, mixed BD and CF snags as MX, BD snags as BD, and low-density snags as WD. This method did not detect any fires that burned: (1) at low severity causing no visible overstory mortality; (2) at high severity that burned all dead wood away completely; or (3) through grasslands leaving no evidence.

1. Interpolation of Non-visible Areas.—Only 58% of the BCW landscape was visible in the historic images we selected (Table 1); this area is referred to as the historic visible landscape, and the grid cells within this area are referred to as the historic visible grid. To create a continuous fuel layer from the historic visible grid, we chose indicator kriging to interpolate the non-visible portions of the landscape. Indicator kriging is one of the only kriging variants for categorical data and is used in ecology (Wang 2007, Martinez 2013)

Fig. 2. Examples of Mountain Legacy Project paired photographs from the 1913–1914 MP Bridgland Survey repeated in 2008. The 1913 photographs were used to classify historical vegetation to create the historical restoration scenario, and the modern photographs are included here only to show an example of the degree of vegetation change between the two time periods. The image pair in the top row shows the Whaleback Ridge as seen from the north-north-east.

and geology (Solow 1986, Marinoni 2003) to predict discrete boundaries between vegetation categories and mineral deposits, respectively. The indicator kriging procedure tends to perform better than simpler nearest neighbor analyses (Solow 1986, Marinoni 2003, Li and Heap 2014), but requires binary response data rather than multiple categories. Further details regarding the interpolation method and accuracy in filling in the non-visible portions of the landscape are provided in Appendix S2. Our objective was to create a reasonable approximation of what the historic landscape would have looked like in 1909 in terms of fuels. However, a visual examination of the modern imagery of the area and the Mountain Legacy Photographs show that most of the grasslands in this area had aspen and willow copses in the areas with higher moisture (depressions, swales, north and east aspects). The interpolation we used did not produce these features, mainly because they were smaller than the 100 m resolution at which we conducted the study.

Vegetation classification and fuel type harmonization.—For the area outside of the BCW (including the 5-km buffer zone), we used the GOA fuel grid to define fuel types in both scenarios. For the historical reconstruction scenario, we interpreted fuel types within the BCW from vegetation as seen in the historical photographs. These photographs lacked sufficient detail to separate some of the vegetation categories into distinct fuel types, so we simplified some of the GOA fuel grid FBP fuel types for the BCW area for both scenarios (Table 2) as follows. All closed canopy coniferous forests were put into a single fuel type (C3 or mature lodgepole pine, which was the most common coniferous fuel type on the landscape). This would have simplified the fire behavior of the landscape somewhat in these model runs; however, these results should be unbiased, as we simplified the fuel grids in both scenarios. The WD class represents a wide range of possible fuel types and was therefore the most difficult to harmonize with the FBP fuel types. Areas in the visible historic landscape that were

Table 2. Alignment of fuel types with vegetation categories from the historical photography analysis.

Historical photography vegetation category	Final fuel type for burn-P3 modeling	Government of Alberta Fuel Grid 2014
Open canopy woodland that was not O1 in 2014	C7	C7
Conifer	C3	C1,2,3,4,5M1/2 < 10%
Broadleaf deciduous	D1/D2	D1/D2M1/2 > 90%
Mixedwood	M1-50%	M1/2 20-80%
Grassland Open canopy woodland that was O1 in 2014	O1	O1
Non-vegetated	Non-fuel (NF)	Non-fuel

Notes: Given are the vegetation categories (left column), fuel types from the Government of Alberta grid of Canadian Forest Fire Behaviour Prediction System (Forestry Canada Fire Danger Group 1992; right column), and the final fuel type used for the Bob Creek Wildland in the Burn-P3 modeling runs (middle column). C7 = Ponderosa pine/Douglas-fir, C1 = spruce lichen woodland, C2 = boreal spruce, C3 = mature lodgepole pine, C4 = immature lodgepole pine, C5 = red or white pine, D1/2 = broadleaf deciduous (1 is leafless, 2 is leaf-on), M1/2 = mixedwood leafless (1) and leaf-on (2) (% indicates proportion of broadleaf deciduous in the mix, remainder is conifer), O1 = grass.

categorized as WD were classified as O1 (grass) where the trees were widely spaced, or C7 if the trees were close together. The C7 (Ponderosa pine and Douglas-fir, which indicates widely spaced trees with little ladder fuels and grass on the forest floor) fuel type was not merged into the C3 fuel type as it was readily identifiable as Douglas-fir and limber pine growing on rocky ridges and outcroppings. We did not differentiate conifer–broadleaf deciduous ratios, so the mixedwood leaf-on and leaf-off (M1 or M2) fuel type cells were set at 50% conifer: 50% broadleaf deciduous.

Analysis methods

Burn probability, fire intensity, and fire size outputs from Burn-P3 were analyzed only within the BCW5K. The overall burn probability of each cell (BP_o) is the number of times each cell burned (burn count) divided by 110,000 (the number of iterations), and the high-intensity burn probability (BP_h) of each cell was the number of times it burned at an intensity of >4000 kW/m. We calculated the difference in the overall burn probabilities (ΔBP_o) between the two scenarios by dividing the historical burn probability by the baseline burn probability as follows: (historical burn probability)/(baseline burn probability), and in the case of the ΔBP_h, this equation was (historical burn probability + 1 fire)/(baseline burn fire + 1 fire) to prevent divide by zero errors.

We created a Δfire_size metric for each pair of fires as follows: (historical fire size – baseline fire size)/(historical fire size + baseline fire size). This variable ranged from −1 (historical much smaller than baseline) to +1 (historical much larger than baseline). The historical and baseline fire size were added in the denominator instead of using either scenario alone to prevent divide by zero errors (the fire sizes were sometimes 0 in one scenario but not in the other). We also compared the fuel grids between the two scenarios and calculated how changing the fuel types would affect the rate of spread (ROS); this was done using the REDApp Universal Fire Behaviour Calculator (McLoughlin 2016) with a weighted mean ROS for each fuel type (each season's value was weighted by its proportion of average area burned; spring = 0.03, summer = 0.75, fall = 0.22), and using Fire Weather Index (FWI) indices representing high fire danger in Alberta, which for this location were a Fine Fuel Moisture Code of 90, Build Up Index of 75, and a Wind Speed (WS) of 20 km/h; grass curing percentages for spring, summer, and fall were 75, 40, and 60, respectively.

Results

There were substantial differences in the fuel grid for the Bob Creek Wildland (BCW) between the two scenarios (Table 3). The HRS had more area in grass (O1) and broadleaf deciduous (M1/M2) vegetation with less coniferous forest cover (C3, C7 and M1).

A total of 10,881 modeled fires originated within the BCW5K (3059 in the BCW and 7221 within the 5-km buffer zone). The largest fires originating in the BCW grew to 8259 and 6538 ha in the baseline (BLS) and HRS,

Table 3. Matrix showing differences in fuel types in the Bob Creek Wildland between the 1909 historical restoration and 2014 baseline scenarios.

Historical restoration scenario fuels	Baseline scenario fuels						
	Non-fuel	D1–D2	O1	C7	C3	M1–M2	Totals historical
Non-fuel	0.2	0	0.1	<0.1	0.1	<0.1	0.4
	0.8	0	*<0.1*	<0.1	*0.1*	<0.1	*0.9*
D1–D2	<0.1	4.4	2.7	1.6	2.9	4.4	16.0
	<0.1	*3.0*	*2.0*	*1.3*	*2.5*	*3.3*	*12.1*
O1	0.4	4.1	13.7	7.9	9.4	4.9	40.3
	0.3	4.2	*13.7*	*7.6*	*12.4*	*4.8*	*43.0*
C7	0	0	0	7.9	0	0	7.9
	0	0	*0*	*7.5*	*0*	*0*	*7.5*
C3	0.3	1.5	3.5	0	21.1	3.2	29.9
	0.2	1.4	4.0	0	*22.8*	*2.9*	*31.3*
M1–M2	0	0.1	0.3	0.7	02.9	0.7	5.7
	<0.1	0.7	0.4	*0.6*	*2.7*	*0.6*	*5.0*
Totals baseline	0.9	10.9	20.4	18.1	36.4	13.3	100

Notes: Numbers in the central block of cells indicate the percentage of the total landscape that was a given fuel type in 1909 (rows) and which fuel type it was in 2014 (columns). Bolded values along the diagonal represent the percentage of the total area with no change between the two scenarios. "Totals" (bottom row, right column) indicate the percentage of the landscape covered by each fuel type in each time period. Non-italicized numbers in the top of each cell are for the visible portion of the landscape (observed), and the italicized numbers at the bottom of each cell are for the total landscape (visible plus interpolated). D1–D2 = leafless/leafy aspen, O1 = grassy, C7 = Douglas-fir, C3 = lodgepole pine (and all other conifers), M1–M2 = leafless/green mixedwood (Forestry Canada Fire Danger Group 1992). See also Table 2.

respectively. Fires originating in the 5-km buffer zone grew to 32,219 and 20,276 ha in the BLS and HRS, respectively. These sizes include the full perimeter of the fire even if it spread outside the BCW5K boundary. The mean number of times any given grid cell on the landscape burned was 22.6 in the HRS (standard deviation [SD] = 11.9, range = 1–94) and 22.8 in the BLS (SD = 12.5, range = 1–92).

The mean area per model iteration that burned at any intensity (BP_o) within the BCW declined very slightly (1.3%) from 3.71 ha in the BLS to 3.66 ha in the HRS (see Fig. 3c, d). In contrast, the mean area per iteration that burned as high-intensity fires (BP_h, intensities exceeding 4,000 kW/m) declined by 44.8% from 1.05 ha in the BLS to 0.58 ha in the HRS (Fig. 3e, f). The high-intensity burn probability was, however, considerably lower than the overall burn probability in both scenarios, and much of the BLS and HRS landscapes did not burn at all at this intensity (see Fig. 3c, d vs. e, f).

There were distinct patterns of burn probability across the landscape that were driven by combinations of fuel type and topography. The areas of the landscape with the highest BP_o in both scenarios were along the eastern edge of the BCW (Fig. 3c, d) where the burn probabilities in both scenarios were two to three times higher in the valley bottom, and two to six times higher along the ridge than in the western portion of the BCW. These differences are largely due to fires having faster rates of spread in grasses in the valley bottoms, and the momentum of these fires carried up to the top of the ridge to the east (being driven by westerly winds). Higher BP_o was also observed within the buffer zone to the east where it was double that of the western zone. Some areas within the eastern buffer had burn probabilities up to seven times higher than the western zone.

While the mean BP_o did not differ much between the two scenarios, there were visible differences in the patterns of burning between the two scenarios (see Fig. 4b). Overall 49.8% of the area within the BCW showed higher BP_o in the HRS than the BLS, 9.6% was the same in both scenarios, and 40.6% had lower BP_o in the HRS. Along parts of the eastern BCW, the BP_o was considerably lower (~half) in the HRS than the BLS (Fig. 4b). These lower probabilities mainly occurred in areas where the fuels were broadleaf deciduous (D1) in the HRS but were conifer (M1, C3, C7) in the BLS (see Fig. 3a, b,

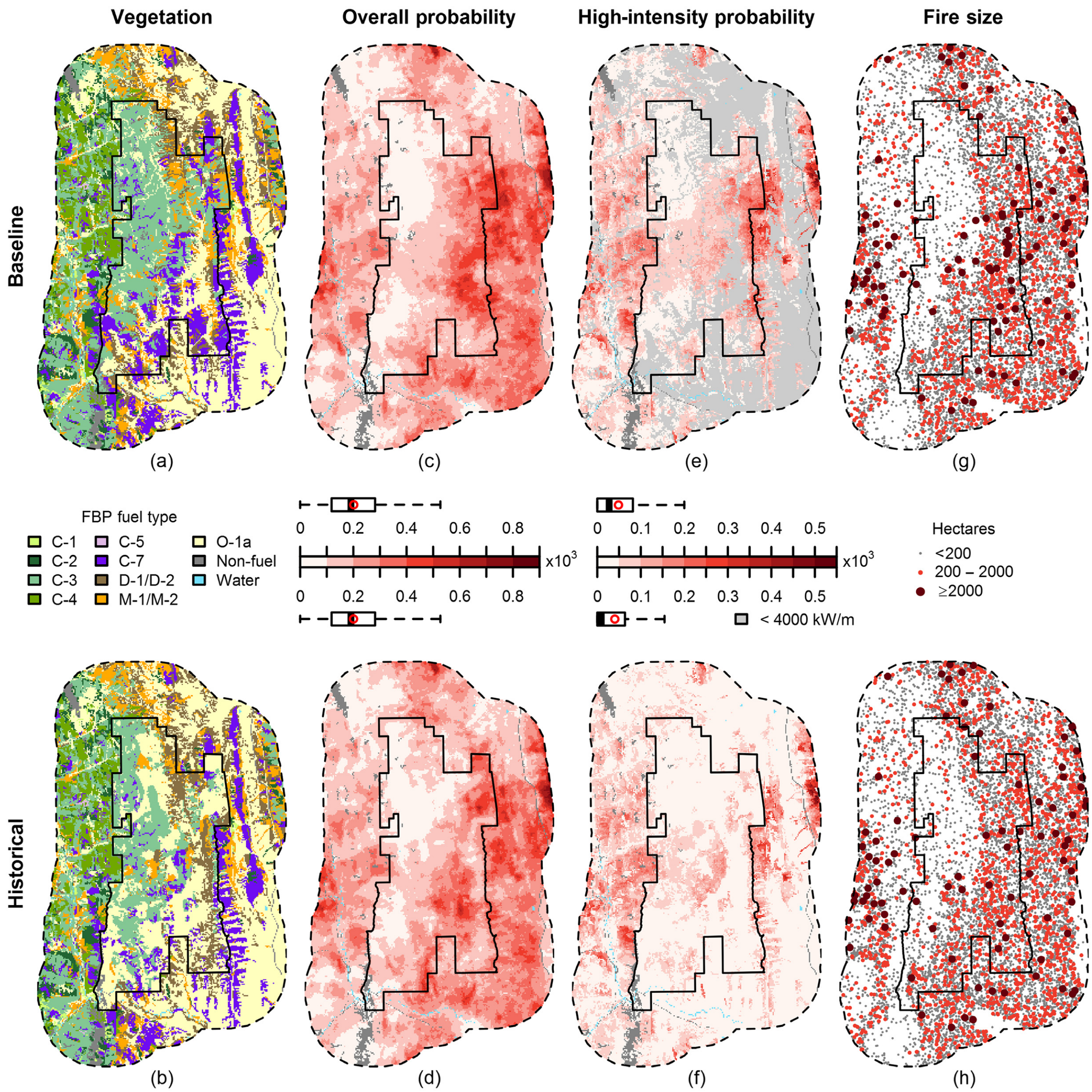

Fig. 3. Maps of the study area (in the Bob Creek Wildland plus 5-km buffer zone [BCW5K]) showing vegetation (FBP fuel) types and outputs from the Burn-P3 model for the baseline (upper row) and historical restoration (lower row) scenarios. Panels (a–b) show the input fuel maps classified as Canadian Forest Fire Behaviour Prediction (FBP) System fuel types (Forestry Canada Fire Danger Group 1992; see Table 2 for FBP—vegetation associations). Panels (c–d) show the overall probability (fire at any intensity). Panels (e–f) show the high-intensity burn probability (≥4,000 kW/m intensity). Panels (g–h) show each modeled ignition within the study area and the size class of the associated fire. Shown between the two burn probability maps are box and whisker plots showing the mean probability (red circle), the median (line in box), 25th and 75th percentiles (ends of the box), and the 1.5× interquartile range (dotted lines).

and look at the locations of the D1 fuel type in Fig. 3b). This was observed primarily in the north-central zone of the BCW, and along the western aspects of the Whaleback Ridge, which runs along the eastern border of the BCW. The fires that originated in these areas (Fig. 3g, h) were smaller in the HRS as compared to the BLS (Fig. 4d); however, these reductions were

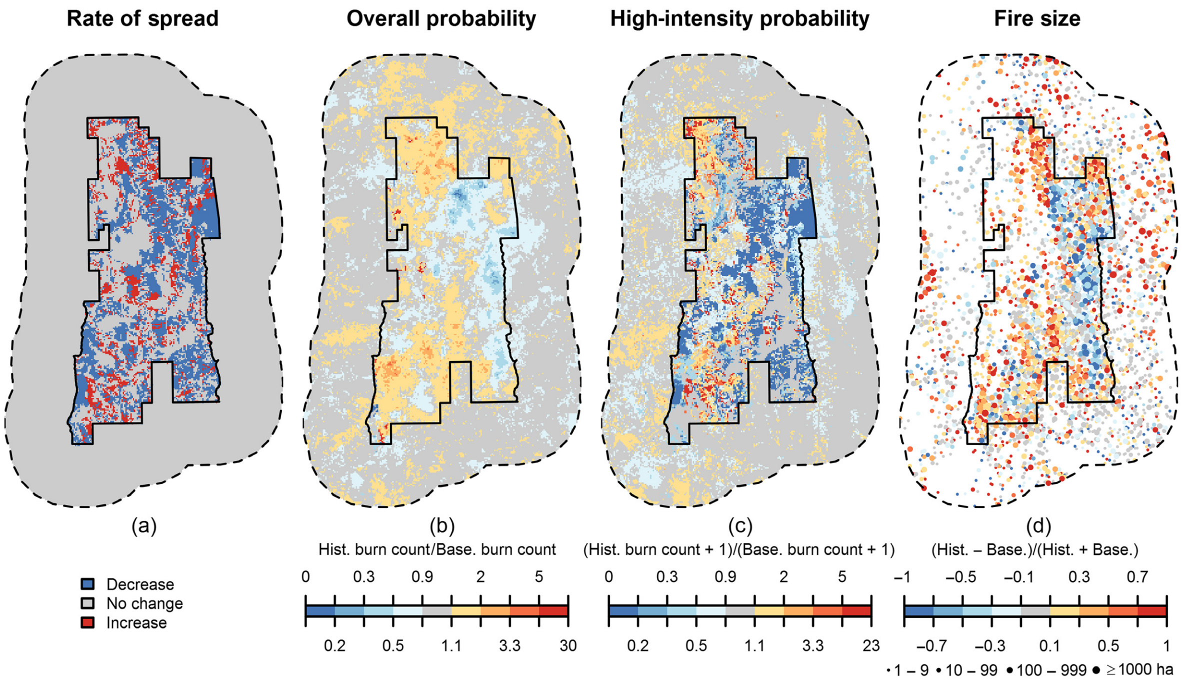

Fig. 4. Maps showing the differences between the 1909 historical restoration scenario and the 2014 baseline scenario. Cooler colors (blue) indicate lower values, and warmer colors (orange and red) indicate higher values in the historical scenario relative to the baseline scenario. Panel (a) shows differences in rate of spread associated with particular fuel type changes. Panels (b) and (c) show the ratio of probabilities between historical and baseline scenarios (historical burn count divided by baseline burn count) for fire at any intensity and at intensity ≥4,000 kW/m, respectively. In panel (c), "1" has been added to all burn counts to control for divide by zero errors. Notes: (b and c) are plotted on different scales, and gray background indicates no change. Panel (d) shows differences in fire sizes for each ignition ((historical − baseline)/(historical + baseline) with points scaled in size relative to the absolute value of historical − baseline fire sizes.

offset by increases in BP_o in other areas. Increased BP_o occurred in areas that changed from other fuel types in the BLS to grass (O1) for the HRS (see areas in Fig. 3a that are turquoise (C3) or purple (C7) that became large areas of tan (O1) in Fig. 3b).

The differences in high-intensity burn probabilities between the scenarios were much greater than for overall burn probabilities (Fig. 4b vs. c). Overall, 22% of the area of the BCW showed a higher BP_h in the HRS as compared to the BLS while 30.4% showed no change in BP_h and 47.6% of the landscape had a lower BP_h in the HRS than in the BLS (Fig. 4c). These reductions in BP_h were concentrated in the north central valley, along the Whaleback Ridge, and in the southeast corner where fuels had been changed from coniferous or mixedwood (C3, C7, or M1) in the BLS to broadleaf deciduous (D1) in the HRS, or from other fuel types in the BLS to grass (O1) in the HRS.

In general, fire sizes were smaller for the HRS than the BLS (mean, median, and maximum = 146.6, 13, and 8739 ha, vs. 192.6, 23, and 10,070 ha, respectively). Fire sizes were larger in areas that were coniferous (C3, C7) in the BLS but grass (O1) in the HRS (see areas in Fig. 3a that are turquoise (C3) or purple (C7) that became tan (O1) in Fig. 3b; red and orange dots in Fig. 4d correspond to the areas of expanded O1 fuels in Fig. 3b).

The effect of changing the fuels inside the BCW in the HRS also affected the burn probability outside the BCW, where the fuels had not been changed. In the 5-km buffer zone around the park, there was a considerable area with a higher BP_o and BP_h (Fig. 4b, c) in the HRS than in the BLS. This influence likely extended well beyond this 5-km buffer, but we did not evaluate this.

Fuel differences between the two scenarios affected the expected rate of spread of fires (Table 4). Of the 51.6% of the landscape that had changed fuels between 1909 and 2014, 37% would burn at lower rate of spread while 14.6% would burn with a higher rate of spread in the HRS as compared to the BLS (Table 4; Fig. 4a).

Discussion

By using historical photographs to determine what the historical vegetation composition on the landscape was at the turn of the 20th century, we were able to test the assumption held by many fire management agencies today that restoring to this vegetation structure would reduce the probability of intense wildfire on the landscape. While we found that the historical restoration scenario resulted in very little difference in the overall burn probability, there were distinct changes in the pattern of burn probability across the landscape, with localized increases and decreases in burn probability being associated with areas where vegetation differences would have resulted in increased or decreased rates of spread, respectively. For high-intensity fires, however, the historical restoration resulted in dramatically lower mean probability (nearly half) and a smaller reduction in the mean fire size (76% of that in the baseline). In roughly 9% of the landscape, the probability of high-intensity burn was reduced to <1/10th that of the baseline scenario. The only explanation for differences between the two scenarios is changes in the speed at which fires moved across the landscape (rate of spread), which is exclusively attributable to changes in the vegetation. This is because we held constant the number, location, and timing of ignitions; duration of burning; and the weather conditions under which the fires burned. While ignition probability is dependent upon fuel type (Krawchuk et al. 2006), we created ignition grids equally from both scenarios to control for this influence.

The largest fuel change was that nearly a third of the landscape changed from forested vegetation in the baseline scenario to grassland in the historical restoration scenario, and the overall effect of this on burn probability is complex and difficult to predict. We know that spring and fall fires in grass can burn very large areas (Rowe 1969, Bailey and Anderson 1980, Brown et al. 2005). Cured grassy fuels (O1) have higher rates of spread in the spring than any of the forest

Table 4. Changes in expected rates of fire spread (Δm/min) associated with fuel type differences in the Bob Creek Wildland between the 1909 historical restoration and the 2014 baseline scenarios.

	Baseline scenario fuels				
Historical restoration scenario fuels	D1–D2 (Δm/min)	O1 (Δm/min)	C7 (Δm/min)	C3 (Δm/min)	M1–M2 (Δm/min)
D1–D2	0	−2.21 −14.69/−1.22/−3.87	−2.64 −0.64/−3.31/−0.64	−6.44 −4.44/−7.11/−4.44	−8.27 −7.26/−8.61/−7.26
O1	2.21 14.69/1.22/3.87	0	−0.44 14.05/−2.09/3.23	−4.24 10.25/−5.89/−0.57	−6.07 7.43/−7.39/−3.39
C7	2.64 0.64/3.31/0.64	0.44 −14.05/2.09/−3.23	0	−3.80 −3.80/−3.80/−3.80	−5.63 −6.62/−5.30/−6.62
C3	6.44 4.44/7.11/4.44	4.24 −10.25/3.80/0.57	3.80 3.80/3.80/3.80	0	−1.83 −2.82/−1.50/−2.82
M1–M2	8.27 7.26/8.61/7.26	6.07 −7.43/7.39/3.39	5.63 6.62/5.30/6.62	1.83 2.82/1.50/2.82	0
Rate of spread	**1.34** 3.34/0.67/3.34	**3.54** 18.03/1.89/7.21	**3.98** 3.98/3.98/3.98	**7.78** 7.78/7.78/7.78	**9.61** 10.60/9.28/10.60

Notes: Bold numbers at the top of each cell indicate the mean change in rate of spread for all fires weighted by the proportion of fires occurring in each season, and the bottom row of numbers in each cell indicates changes in rate of spread associated with spring/summer/fall fires. D1–D2 = leafless/leafy aspen, O1 = grassy, C7 = Douglas-fir, C3 = lodgepole pine (and all other conifers), M1–M2 = leafless/leafy mixedwood (Forestry Canada Fire Danger Group 1992). The bottom row of the table gives the actual mean rate of spread (m/min) in each fuel type (top bold value) weighted by the proportion of fires in each season, with values for spring/summer/fall below.

fuels present in the study area (Rothermel 1983, Taylor et al. 1997, Wotton et al. 2009), and the net effect of increased grassland area was that fires grew larger than they would have if they occurred in forested fuels (Finney 2001). In the summer, however, when grass (O1) is actively growing and green, it would suppress the rate of spread resulting in smaller fires than if they occurred in mature lodgepole pine (C3), Douglas-fir and open grown pine (C7), or leafed-out mixedwood (M2). In the fall, the rate of spread for grass would still be lower than for lodgepole pine (C3) or leaf-off mixedwoods (M1), but higher than Douglas-fir or open grown pine (C7). Thus, how this shift of nearly a third of the landscape to grass would affect net burn probability depended strongly upon the spatial location of these changes relative to other fuel types (Miller and Urban 2000) and how much larger the spring and fall fires burned relative to summer fires (see below). It must also be noted that Fire Weather Index values have a large effect on these relative rates of spread, and specifically, O1 would have a higher rate of spread in the fall than all other fuel types if the percent curing value was 65% (we parameterized the model at 60% fall curing). While downscaling fuels (combining numerous fuel types into the C3 category) undoubtedly changed the fire behavior in some locations, we chose the C3 fuel type as it would generate the most extreme fire behavior, and this bias was consistent between both scenarios.

The spatial arrangement of fuel types influences burn probability and intensity, and areas of the landscape where the historical restoration caused increases in the rate of spread were offset by other areas that reduced the rate of spread. Areas of the landscape that showed increased probability of burning in the historical restoration scenario were largely in or near the areas that were changed to grasslands (O1) or other vegetation types that increased the rate of spread. These changes had disproportionate effects; while less than 15% of the landscape vegetation changes in the historical restoration scenario resulted in increases in the rate of spread, nearly 50% of the landscape had a higher overall burn probability. That large-scale shifts to increased grassland cover can have a disproportionate effect on area burned has also been demonstrated by Miller and Urban (2000), who modeled the relationship between area burned and fuel connectivity, and found that total area burned was strongly and positively correlated to the amount of grass (accelerants) in the fuel bed.

The location of vegetation changes that slowed the rate of spread appears to have been critical in offsetting the increased burn probability associated with the increase in grass cover. Areas that were converted to broadleaf deciduous (D1/D2) in the historical restoration scenario only covered ~9% of the landscape, but these primarily occurred on the downwind (eastern) side of the BCW. Thus, fires starting in or upwind of these areas would have been considerably smaller in the historical restoration scenario than fires starting in those same locations in the baseline scenario where they would have run into more flammable fuels. Although the change in size may not seem ecologically consequential (a reduction from 192 to 146 ha), the locations where the reductions occurred may well be important. In essence, these stands of deciduous forests acted as brakes on many fires relative to the baseline condition (Shinneman et al. 2012). To ultimately determine the effects of vegetation changes that increase or decrease rates of spread on burn probability, we would have to partition our analysis by season, and also examine the effects of these vegetation changes by modeling only fuel changes that increase rates of spread, fuel changes that decrease rates of spread, and both combined.

Moreover, to consider the broader issue of wildfire risk, which is defined as a combination of the likelihood (burn probability), intensity, and impacts of a fire (Miller and Ager 2013), we considered fire intensity by partitioning our analysis into the overall burn probability (all fires) and the high-intensity burn probability (fires >4000 kW/m). Whereas the increase of grassland cover in the historical restoration increased the overall burn probability in many parts of the landscape, the high-intensity burn probability was reduced in these same areas as well as in the areas restored to broadleaf deciduous. Other studies have also found overall reductions in burn probability and intensity due to fuel changes that had reduced rates of spread and lower ignition probability (Finney 2001, Fulé et al. 2004, Wang et al. 2016).

In the historical restoration scenario, having vegetation that slows the rate of spread on the downwind side of the wildland appeared to be effective in limiting large fires, as the location of different fuel types relative to one another is of the utmost importance to reducing fire spread and suppression effectiveness. However, the shift to increased grassland cover is not without risks too, because while fires in grasslands are generally easier to suppress or manage due to lower intensities, the increased rate of spread in these fuel types in spring and fall can offset this very quickly. Under conditions of extreme winds and low humidity, grassland fires can be virtually impossible to contain and will rapidly spread to more intense-burning fuels (coniferous forests) or to nearby values at risk. The placement of downwind vegetation brakes that generally reduce rates of spread may not be effective under all weather conditions. Under extreme weather conditions and with low fuel moisture, the time since the last fire (which affects the amount of fuel available) and the spatial arrangement of fuels can become irrelevant and the landscape is almost completely burnable (Miller and Urban 2000).

The third and final component of wildfire risk to consider is the impacts of fires, which are dependent upon the values at risk (societal, economic, ecological, and others) in a particular landscape, and will always be site-specific. In the case of the BCW, one of the most ecologically significant areas is the Whaleback Ridge along the eastern edge of the park. This area is classified as an *Environmentally Sensitive Area* of national importance (Government of Alberta 2011), with numerous stands of whitebark (*Pinus albicaulis*) and limber pine (*P. flexilis*), both of which are listed as endangered species in Alberta. The Whaleback Ridge is also considered one of the two most important winter ranges for elk and other ungulates in Alberta (Government of Alberta 2011). Our modeling scenarios showed that this ridge, and the valleys to the east had the highest burn probability, was most likely to burn at the highest intensity (Fig. 3). This is also the area where the historical restoration scenario resulted in the greatest reductions in the overall and high-intensity burn probabilities.

In the coming decades, climate suitability envelopes are expected to move upslope and to higher latitudes (Loarie et al. 2009), and recent work by Schneider (2013) has shown that the climate most suitable to the Foothills Fescue Natural Subregion (Natural Regions Committee 2006) will move into higher elevation areas currently classified as Montane (i.e., the Whaleback Ridge), which in turn will displace the Subalpine Natural Subregion. Climate change alone, however, does not immediately change the vegetation composition of the landscape, and ecosystem "memory" (as per Johnstone et al. 2016) tends to maintain current vegetation until a disturbance forces the change. Depending on which vegetation type the climate at the time of the disturbance event is suitable for, we might see re-emergence of the pre-disturbance vegetation; or a state change to a different type (Johnstone et al. 2016, Stralberg et al. 2018). In our study, fire occurring in the baseline (as modeled here) landscape is more likely to trigger a large-scale shift in vegetation type from forest cover to open grasslands due to high levels of mortality resulting from large crown fires, whereas the historical restoration scenario would likely preserve more of these forest features for a longer period of time due to a lower likelihood of fire. The fire environment of the landscape has changed considerably since the time of European settlement of the region. Based on studies nearby, historical (pre-1900) mean annual burn rates were probably 3.33–6.67% of the landscape in the Montane and 0.67–3.3% of the landscape in the Subalpine (Hawkes 1979, Arno 1980, Barrett 1996, Rogeau 2005). Modern burn rates (post-1960) are dramatically lower (mean annual burn rates <0.02%; Tymstra et al. 2005). Stockdale (2017) found that the mean annual area burned since 1913 was 0.075%, which is higher than the Burn-P3 model outputs (0.0179%, baseline, and 0.0176%, restoration) and the 1960–2002 burn rate (Tymstra et al. 2005), but still well below the pre-1900 burn rates. Evidently, the burn rate of the landscape has declined considerably in the latter half as compared to the first half of the 20th century. While the absolute burn probabilities of the landscape were very low in these simulations, these merely reflect the burn rates observed on the landscape over the last 53 yr. We emphasize the large relative differences in burn probabilities between the two scenarios.

The results of this study provide useful information for managers interested in restoring historical vegetation with the aim of reducing the probability and severity of future wildfire. Sites at greatest risk of loss (the highest BP_o and BP_h values) could be triaged and prioritized for either treatment or protection. For instance, prescribed burns could be lit in the grassy valleys and allowed to burn into neighboring forested stands, effectively pushing back the forest edge. Vegetation conversion that reduced the rate of spread could be strategically placed to reduce the probability of high-intensity fires around areas of high value or in locations designed to prevent fires from escaping the area targeted for restoration. Vegetation changes that reduce the rate of spread should be located on the downwind side of the restoration area prior to increasing the landscape abundance of vegetation that would increase the rate of spread, such as removing forests and converting to grasslands.

In this historical restoration simulation, we saw only minor reductions in the overall burn probability of the landscape, but large reductions in the likelihood of high-intensity fire. This suggests that restoration to historical vegetation conditions could result in future fires that would be easier to manage, but this would depend on the relative locations of vegetation changes and the effects they have on the rate of spread of fire. While some might argue that active intervention to preserve wilderness is undesirable or dangerous (see review by Pimbert and Pretty 1995), in the absence of intervention, many montane regions will continue to experience forest encroachment and increased risk of high severity wildfire (Agee 2002). However, restoring historical vegetation using a single point in time comes with its own set of risks, as the vegetation will be interacting with a new climate and disturbance regime. Like us, Flatley and Fulé (2016) found that historical vegetation structure (in the Grand Canyon from the 1900s) was better suited to projected future climates, and had lower wildfire risk than the current vegetation, but they found that the altered fire regime would not support the historical vegetation into the future. Flatley and Fulé (2016) concluded that this would likely result in the emergence of novel ecosystems, and the potential for this to occur in any restoration exercise must be weighed carefully (Jackson and Hobbs 2009). Rather than simply reconstructing a single point in time, an ideal solution would be to determine a range of ecologically sustainable conditions and choose the best reference point within that range that will achieve the landscape objectives (Keane et al. 2009).

Using the approach outlined here, managers could evaluate the benefits of historical restoration in areas of ecological concern wherever deep historical records exist. By using burn probability modeling and testing different vegetation configurations, managers can determine which parts of their landscapes are at greatest risk and then use this to prioritize areas for restoration treatments. Careful selection of input parameters is vital to conduct a burn probability assessment, and we caution against over-interpretation of these results as fire behavior is tightly linked to fuel and weather conditions; what we present here is based on a particular set of vegetation and FWI parameters and as these values change, so too would some of the relative rates of spread and intensity values.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2584/full