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Do fuel treatments decrease forest mortality or increase streamflow? A case study from the Sierra Nevada (USA)

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Abstract

Reduced forest mortality and increased streamflow have been promoted as hydrologic benefits of forest fuel treatments in water-limited systems. These benefits, however, are often quite variable because they both rely upon the use of water made available through tree-scale reductions in evapotranspiration. In this study, we examined whether forest mortality benefits and streamflow benefits from fuel treatments offset one another, because the allocation of unused water for one benefit implies less availability of that water for the other benefit. To do this, we took advantage of two paired-watershed experiments in the Kings River Experimental Watersheds located in the southern Sierra Nevada. The fuel treatments, which included mechanical thinning and prescribed fire, began in 2012 and coincided with the 2012–2016 California drought and related forest mortality event. We found that in the higher-elevation Bull watersheds, fuel treatments decreased forest mortality but had no effect on annual streamflow. In the lower-elevation Providence watersheds, we observed the opposite result, as fuel treatments had no effect on forest mortality but depending on the model, increased annual streamflow at 95% and 90% confidence. The results suggest that the water made available from fuel treatments followed different hydrological pathways through the watersheds, with the water being taken up by neighbouring vegetation and decreasing water stress in Bull, and the water contributing to streamflow in Providence. These findings suggest that fuel treatments in water-limited systems may not provide full hydrologic benefits to both forest mortality and streamflow concurrently in a given watershed.

KEYWORDS

drought, forest mortality, fuel treatments, paired watershed, Sierra Nevada, streamflow

1 | INTRODUCTION

Forest fuel treatments (e.g., prescribed fire and mechanical thinning) have been shown to reduce wildfire intensities and burn severities and have been widely advocated for reducing fire risk in the Western United States (North et al., 2015; Omi & Martinson, 2002; Prichard, Peterson, & Jacobson, 2010). Forest fuel treatments may also have the potential hydrologic benefits of reducing drought-induced tree mortality in forests and augmenting streamflow. These latter benefits, however, are quite variable, in part because resistance to drought-related mortality and streamflow augmentation depend on competing water sources. Reduced forest mortality during drought occurs when water made available through treatment-induced reductions in tree-scale evapotranspiration (ET) is taken up by neighbouring trees, reducing tree water stress. On the other hand, greater streamflow generation occurs when water made available through treatment-induced reductions in tree-scale ET makes its way to a stream, increasing water yield. Understanding the partitioning between mortality resistance to drought and streamflow augmentation is important for predicting hydrologic and forest responses to forest fuel management, especially under future climate scenarios.

Since the early 1900s, policies of fire suppression and fire exclusion have caused many forests with high-frequency low-severity fire regimes in the Western U.S. to become overly dense (Chang, 1996; Fellows & Goulden, 2008). This condition has increased competition for water and increased forest mortality during droughts relative to lower-density forests (Gleason et al., 2017). Forest fuel treatments can counter these issues by reducing forest densities and removing forest undergrowth (e.g., young trees and shrubs) (Agee & Skinner, 2005; Sohn, Saha, & Bauhus, 2016). At the tree scale, fuel treatments that remove vegetation make available water that otherwise would have been transpired or intercepted and evaporated. At the watershed scale, this unused water may follow a number of pathways through the watershed (Figure 1). First, the water may directly return to the atmosphere via greater abiotic vapour fluxes such as ground evaporation or sublimation (Biederman et al., 2014). Second, the unused water may be transpired by remaining neighbouring vegetation (Tague & Moritz, 2019). Finally, the unused water may make its way to an aquifer or stream. From a management perspective, the latter two pathways are generally considered to be beneficial, as uptake by the remaining vegetation can support vegetation health and reduce forest mortality during drought (Grant, Tague, & Allen, 2013), whereas increased groundwater and streamflow can augment water supplies downstream. The challenge is that it is unclear when, where and under what conditions the latter pathways dominate, if at all.

Higher temperatures associated with climate change are exacerbating drought effects on forests, increasing vulnerability to hydraulic failure, carbon starvation and pests (Allen, Breshears, & McDowell, 2015; McDowell et al., 2008). Consequently, many forests, and particularly high-density stands, have become more vulnerable to widespread mortality events during drought (Allen et al., 2010). Forest fuel treatments can reduce water competition during hotter droughts (Park et al., 2018) and may decrease the likelihood of drought mortality (Restaino et al., 2019; Sohn et al., 2016; van Mantgem, Caprio, Stephenson, & Das, 2016). Still, posttreatment water uptake by remaining neighbouring trees may not always be sufficient to reduce mortality, and the water made available from fuel treatments may follow alternative pathways (e.g., abiotic evaporation and streamflow generation) through the watershed.

Streamflow response to vegetation change has a long history of study (Andréassian, 2004; Bosch & Hewlett, 1982; Brown, Zhang, McMahon, Western, & Vertessy, 2005; Goeking & Tarboton, 2020; Stednick, 1996); however, investigations of streamflow response to forest fuel reduction treatments, which often have small relative impacts on stand density or residual basal area, are more limited. Overall, studies of forest thinning have shown that streamflow response is highly variable, with some studies showing posttreatment increases (Dung et al., 2012; Lane & Mackay, 2001; Serengil et al., 2007), others showing no response (Gökbulak et al., 2016) and in some cases, reductions in streamflow (Hawthorne, Lane, Bren, & Sims, 2013). The effect of prescribed fire on streamflow has shown similar variability (Cawson, Sheridan, Smith, & Lane, 2012; Gottfried & DeBano, 1990). Some of this variability has been attributed to the presence of a vegetation change threshold, often estimated at 20% of basal area, below which streamflow changes are undetectable (Bosch & Hewlett, 1982). Others have noted that meteorological conditions (Bart, 2016) and soil depth (Tague & Moritz, 2019) can also affect streamflow response to vegetation change. In the Sierra Nevada, Saksa, Conklin, Battles, Tague, and Bales (2017) and Saksa et al. (2020) found that streamflow increased in response to low-intensity fuel treatments in wetter, more northern watersheds, but had minimal response in drier, more southern watersheds. Also, post-treatment vegetation recovery can affect streamflow response to fuel treatments, as rapid succession by grasses and shrubs can minimize increases in streamflow and in some cases reduce streamflow (Bennett et al., 2018).

Reduced drought mortality and enhanced streamflow have both been promoted as benefits of forest fuel treatments, yet to our knowledge, no empirical studies have tested for these benefits in combination at the watershed scale. In this study, we take advantage of a fuel treatment experiment in the Kings River Experimental Watersheds (KREW) that coincided with a drought that has been estimated to be

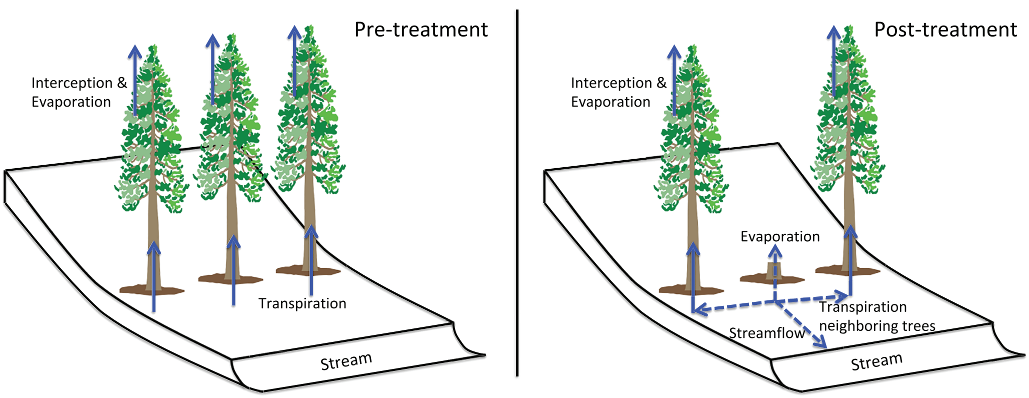

FIGURE 1 Conceptual model of pre- and post-treatment water pathways

the most extreme in over 1,200 years (Griffin & Anchukaitis, 2014). Using a paired watershed framework, we addressed two questions: (a) What are the effects of forest fuel treatments on forest mortality during subsequent drought? and (b) What are the effects of vegetation changes (e.g., fuel treatments and forest mortality) on annual streamflow? We hypothesized that streamflow is more likely to increase in response to vegetation reductions in watersheds where forest fuel treatments did not affect forest mortality during the drought (Figure 1). This research will provide insight into the pathways that water made available from fuel treatments follows and will be useful in assessing the impacts of future similar fuel treatments.

2 | STUDY WATERSHEDS AND THE CALIFORNIA DROUGHT

2.1 | Physical watershed characteristics

This study was conducted in the KREW, located in the southern Sierra Nevada southeast of Shaver Lake, California, USA. The KREW consist of two groups of four long-term research watersheds, Bull (B201, B203, B204, T003) and Providence (P301, P303, P304, D102), operated by the U.S. Forest Service Pacific Southwest Research Station since 2002 (Figure 2). The areas of the Bull watersheds range from 53 to 228 ha and have mean elevations between 2,257 and 2,373 m

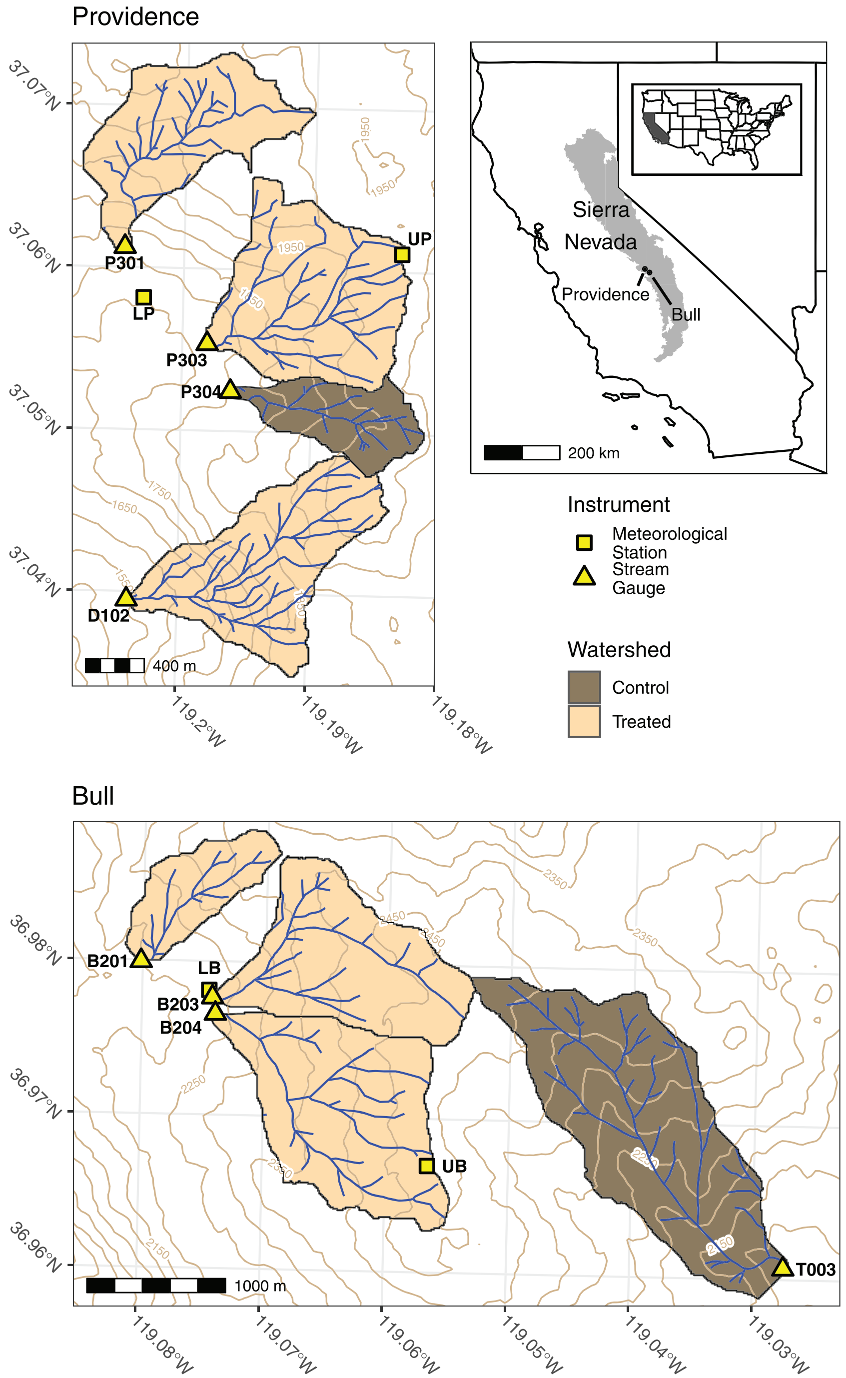

FIGURE 2 Map of the Kings River Experimental Watersheds. Meteorological stations UP, LP, UB and LB correspond to Upper Providence, Lower Providence, Upper Bull and Lower Bull, respectively

TABLE 1 Kings River Experimental Watersheds (KREW) characteristics and treatment timing

Watershed	Thinning (Season year)	Prescribed fire (Season year)	Area (ha)	Mean elevation (m)	Mean aspect (degrees)	Mean slope (%)	Pretreatment live tree basal area (m^2ha^{-1})	Pretreatment live tree density (ha^{-1})	Mean annual streamflow (mm)	Annual streamflow coefficient of variation
Bull										
B201	Summer 2012	-	53	2,257	228	18	54.1 (0.5–123.5)	648 (100–2,225)	453	0.90
B203	-	Fall 2013	138	2,373	235	18	46.2 (0–116.3)	548 (0–2,463)	678	0.81
B204	Summer 2012	Fall 2013	167	2,365	235	17	54.8 (0–158.2)	592 (0–2,425)	589	0.87
T003	-	-	228	2,289	142	24	79.2 (0–211.7)	522 (0–2050)	531	0.85
Providence										
P301	Summer 2012	Fall 2016	99	1979	208	19	63.4 (3–142.1)	614 (125–1,500)	435	1.08
P303	-	Fall 2016	132	1905	233	20	43.8 (0–97)	613 (0–2038)	303	1.14
P304	-	-	49	1899	249	22	44.8 (15.1–147.2)	598 (150–2075)	411	0.84
D102	Summer 2012	-	121	1782	246	27	50 (4.5–119.6)	812 (88–1750)	318	1.07

Note: Pretreatment live tree basal area and density are mean values (ranges in parentheses) from plots surveyed from 2003 to 2006 and obtained from Lydersen et al. (2019).

(Table 1). The Providence watersheds are smaller and lower in elevation, with areas ranging from 49 to 132 ha and mean elevations ranging from 1,782 to 1,979 m.

The lithology for all the watersheds is granite. Soil series in the Providence watersheds are composed of Gerle-Cagwin for P301 and Shaver for P303, P304 and D102, whereas the Bull watersheds are primarily composed of Cagwin soils (Johnson, Hunsaker, Glass, Rau, & Roath, 2011). The rooting depth associated with the Gerle-Cagwin and Shaver soils in Providence are 76–127 cm and 102–203 cm, respectively, whereas the depth of the Cagwin soils in Bull are shallower at 50–102 cm (Hunsaker, Whitaker, & Bales, 2012). Vegetation in the Bull watersheds consists primarily of red fir (*Abies magnifica*) and white fir (*Abies concolor*), along with some Jeffrey pine (*Pinus jeffreyi*), lodgepole pine (*Pinus contorta*) and a small amount of sugar pine (*Pinus lambertiana*) at lower elevations. Vegetation in the Providence watersheds is primarily mixed-conifer forest consisting of white fir, incense cedar (*Calocedrus decurrens*), ponderosa pine (*Pinus ponderosa*), Jeffrey pine and sugar pine. Riparian meadows are present in all watersheds and range in area from 0.1 ha (P303 and P304) to 8.3 ha (B204) (Hunsaker, Adair, Auman, Weidick, & Whitaker, 2007). The KREW, similar to many forested areas in the southern Sierra Nevada, have relatively high forest stand densities, averaging ~577 stems per hectare across the Bull watersheds and ~659 stems per hectare across the Providence watersheds (Table 1) (Lydersen, Collins, & Hunsaker, 2019).

The lower montane forest in the southern Sierra Nevada historically had a low-severity fire regime with fire return intervals ranging from 5 to 20 years (Kilgore & Taylor, 1979; Scholl & Taylor, 2010). Wildfire has been essentially excluded in the KREW, with only 4 ha affected by wildfire since 1911 (Lydersen et al., 2019). Prior to the establishment of the KREW study, timber harvests have occurred in all watersheds except T003, with small harvests occurring most recently on privately owned land in the Providence group.

2.2 | Hydro-climatology

The KREW have Koppen-Geiger Mediterranean Csa or Csb climates with most precipitation occurring between the late fall and early spring (Kottek, Grieser, Beck, Rudolf, & Rubel, 2006). Summers are exceptionally dry. Mean annual precipitation in Bull and Providence for water years 2005 to 2017 were 1,409 and 1,326 mm, respectively, with water year defined as October 1 of the previous year to September 30 of the present year. Mean annual temperatures were 7.6°C and 9.9°C in Bull and Providence, respectively. Historically, the watersheds in Providence were in the winter rain-snow transition zone for the southern Sierra Nevada, whereas the watersheds in Bull were located above the rain-snow transition zone. Precipitation and temperature data were measured at four meteorological stations, with one located near the lower elevation of each watershed group, close to the stream gauging stations and one near the upper elevation of each watershed group. Full details on the equipment and procedures used to collect and process precipitation and temperature data in the KREW are provided in Hunsaker et al. (2012) and Safeeq and Hunsaker (2016), with data from 2002 to 2017 available in Hunsaker and Safeeq (2018).

Stream stage for each of the watersheds is measured with Parshall Montana flumes, except T003 which has a compound v-notch and rectangular weir, and streamflow is computed using standardized stage-discharge relationships (Hunsaker & Safeeq, 2017). Mean annual streamflow in the KREW ranged from 303 (P303) to 678 mm (B203) for water years 2004 to 2017 and was highly variable on an annual basis (Table 1). Hydrographs in both sets of watersheds were snowmelt dominated, though to a greater extent in Bull than Providence. The Bull watersheds had higher streamflow than the Providence watersheds, which has been attributed to a higher proportion of snowfall and lower ET rates (Hunsaker et al., 2012). Further details on the streamflow gauging can be found in Hunsaker et al. (2012) and Safeeq and Hunsaker (2016). Streamflow data from 2002 to 2015 are available in Hunsaker and Safeeq (2017).

2.3 | 2012–2016 California drought

California experienced a severe drought that began in 2012 and continued through 2016. This drought was notable for near-record low precipitation levels coinciding with record high temperatures (Shukla, Safeeq, AghaKouchak, Guan, & Funk, 2015). The 'hot drought' had many consequences on both the hydrology (Bales et al., 2018; Goulden & Bales, 2019) and ecology (Roberts, Burnett, Tietz, & Veloz, 2019; Young et al., 2020) of the southern Sierra Nevada. Snowpack and streamflow in the KREW were at record lows and included the first recorded summer cessation of streamflow in P301 in 2013. The drought also contributed to widespread tree mortality in the southern Sierra Nevada, with the period of greatest forest mortality occurring from 2015 to 2017. The U.S. Forest Service has estimated that over 129 million trees in California have died from the onset of the drought through 2017 (Buluç, Fischer, Ko, Balachowski, & Ostoja, 2017). The KREW are located near the centre of the forest mortality impact area and given the coincidental timing of the fuel treatments, provide an opportunity to evaluate differences in forest mortality and subsequent changes in streamflow among the watersheds.

3 | METHODS

We used a paired-watershed framework (Andréassian, 2004; Brown et al., 2005; Stednick, 1996) as the basis for examining changes in forest mortality and annual streamflow during drought. In addition, we included a longitudinal time-series analysis of streamflow change to assess the robustness of the results from the paired streamflow analysis.

3.1 | Fuel treatment experiment

Beginning in 2012, forest fuel treatments were conducted in the KREW with the objective to understand how forest fuel reduction treatments affect watershed hydrology, watershed ecology and biogeochemical processes. For each group of four watersheds, one watershed was mechanically thinned (B201 and D102); one watershed was burned by prescribed fire (B203 and P303); one watershed was mechanically thinned and followed by prescribed fire (B204 and P301) and one watershed was left as an experimental control with no recent treatment (T003 and P304) (Table 1). Mechanical thinning was conducted using feller-bunchers and hand felling combined with ground-based skidding in summer and fall 2012. Shrub mastication also occurred in stands with greater than 50% shrub cover to less than 10% shrub cover (Lydersen et al., 2019). Prescribed fires in the Bull watersheds were conducted in fall 2013. Prescribed fires in the Providence watersheds were delayed because of unacceptable ignition conditions until fall 2016. Approximately 1/3 of the P301 watershed was salvage logged in summer 2017, with additional removal of dead timber using ground-based equipment.

The intensity of the thinning treatments was low (see Lydersen et al., 2019 for full details about treatments in the KREW). Thinning was limited to trees with a maximum diameter at breast height (DBH) of 76 cm, with a few exceptions. Between 10% and 25% of the area permitted for treatment in each thinned watershed was eventually excluded due to inaccessibility and/or steep slopes that made thinning not economically viable. Thinning was allowed to within 15 m of streams. Based on posttreatment surveys of soil disturbance, thinning occurred in 52% and 29% of the area in the thinned Bull and Providence watersheds, respectively. Averaged over the thinned watersheds, 33 and 23 trees greater than 25.4 cm DBH were removed per hectare for Bull and Providence, respectively. This was equivalent to a basal area of 4.6 and 4.5 m^2 ha^{-1}. These values were based on timber sale records and do not include trees smaller than 25.4 cm DBH, which would increase the total number of trees and basal area removed. Based on field surveys prior to thinning (2003 to 2006) and after thinning (2013 and 2014), Lydersen et al. (2019) did not observe a significant change in basal area or tree density from the thinning, although we note the 7-year gap between presurvey and postsurvey may have masked growth in the watersheds prior to the fuel treatments.

The prescribed fires were of low intensity and were not intended to change the structure of the forest, but rather reduce surface and ground fuels (Lydersen et al., 2019). The fires were conducted during cool temperatures (less than 12°C) with low to moderate humidity (range 10% to 77%) and moderate fuel moisture (range 6% to 21%). Flame lengths were limited to less than 1.8 m and burning was permitted to within 1.5 m of the stream channels. Posttreatment surveys of soil disturbance found that 39% and 52% of the areas where prescribed fire was applied showed signs of burning in Bull and Providence, respectively.

3.2 | NDVI model

For this study, we evaluated forest drought mortality using the Normalized Difference Vegetation Index (NDVI), a watershed-scale measure of vegetation greenness (Pettorelli et al., 2005). NDVI has two advantages as compared with the field-collected plot measurements in the KREW as a proxy for forest mortality (Dong et al., 2019). First, NDVI data are available over the entire study period and can provide details about vegetation changes during the years that were not sampled in the field. In this regard, the effects of treatments and tree mortality can be better quantified and separated. Second, while NDVI cannot discern subwatershed characteristics on the forest (i.e., basal area, species) that can be gained from plot-scale measurements, we expect that a spatially lumped metric of vegetation may better correspond to the integrated watershed measure of streamflow.

NDVI data for each of the watersheds were obtained from Landsat 7 for the period from 1999 to 2017. The imagery was obtained from the U.S. Geological Survey Landsat Surface Reflectance collection (Masek et al., 2006) via Google Earth Engine (Gorelick et al., 2017). NDVI was calculated using Bands 3 and 4 from all cloud-free Landsat scenes between May and September of a given year, following a similar approach to Su et al. (2017). These months generally correspond to the snow free periods in the KREW. All scenes with pixel values greater than 1.0 were removed, as these values were indicative of processing errors. Of the remaining scenes, we used the maximum-value composite (MVC) approach to generate a single image for each year (Holben, 1986). The MVC approach selects the highest NDVI value for each pixel from all scenes in a period, creating a single composite that represents the maximum NDVI for each pixel. The MVC approach has the added benefit of negating the Scan Line Corrector Failure issue in Landsat 7 that produced striations with missing data in the Bull watersheds (Andrefouet, Bindschadler, & Brown de Colstoun, 2003). For each annual composite, the mean NDVI value across the watershed was computed to provide a watershed-scale measure of NDVI.

Posttreatment changes in NDVI in the treated watersheds ($NDVI_t$) relative to the control watershed ($NDVI_c$) were evaluated using a mixed-effects model with treatment period (T) as a fixed effect (with levels of pretreatment and posttreatment as 0 and 1) and watershed as a random effect (u):

$$NDVI_t = \beta_0 + \beta_1 NDVI_c + \beta_2 T + \beta_3 NDVI_c * T + u + e, \tag{1}$$

where e is the model error. The interaction between $NDVI_c$ and T was also included in the model. For the thinned watersheds and thinned/prescribed fire watersheds, pretreatment was designated as 2002–2012, and posttreatment was designated as 2013–2017. The pretreatment period for the watersheds with only prescribed fire was designated as 2002–2013 in Bull and 2002–2016 in Providence. For this component of the study, we were primarily interested in the interaction variable between $NDVI_c$ and T, as a significant interaction term would signify that the relation between $NDVI_t$ and $NDVI_c$ differed before and after the treatments, which we use as an indication of drought mortality.

We estimated the parameters in the mixed-effects model using a Bayesian estimation procedure. Mixed-effects modelling was performed in the R programing environment (R Core Team, 2019) using the rstanarm package (Goodrich, Gabry, Ali, & Brilleman, 2020) which allows Bayesian computation in the Stan Probabilistic Programming Language (Gelman, Lee, & Guo, 2015). The Stan language uses a Hamiltonian Monte Carlo sampling algorithm for evaluating model parameters. Separate mixed-effects models were developed for the Bull and Providence groups using a weakly informative prior. Each model generated a posterior sample size of 10,000, and we visually assessed model convergence. Autocorrelation in the model residuals was assessed using the partial autocorrelation function (PACF).

3.3 | Streamflow model

We were interested in evaluating annual streamflow responses to forest fuel treatments in the KREW. However, an implicit assumption of the paired watershed framework when evaluating changes in streamflow is that land cover in the control watershed is stable during the entire study period, providing an experimental control to compare the treated watershed against. This assumption, however, was not valid in this study. The severe drought during the posttreatment period generated widespread mortality throughout the southern Sierra Nevada region, including in the control and treated watersheds (Lydersen et al., 2019). Consequently, the use of a fixed effect for pretreatment and posttreatment periods, as was done in the paired NDVI model, would have been inadequate. In the paired NDVI models, we were able to use a fixed effect for period because, relative to the control watershed, NDVI in the treated watersheds were only differentially affected by fuel treatments because the intensity of the drought was assumed to be equal for all watersheds. Streamflow in the treated watersheds, in contrast, were differentially affected by both fuel treatments and differences in mortality between the treated and control watersheds. Thus, we chose to use the difference in watershed-averaged annual maximum NDVI between the control and the treated watershed ($NDVI_{diff}$) as a covariate in the paired streamflow model to account for both treatment effects and tree mortality effects. This variable cannot be used to directly evaluate the effects of the fuel treatments and mortality on streamflow, as it represents vegetation differences (e.g., treatment effects, drought stress effects and tree mortality effects) between the treated and control watersheds across the entire time-series. Nonetheless, as the fuel treatments and mortality produced the largest changes in NDVI, these events were implicitly represented. The mixed-effects model to estimate changes in annual streamflow (log-transformed) in the treated watersheds (Q_t) relative to the control watershed (Q_c) included $NDVI_{diff}$ as a fixed effect and watershed as a random effect (u):

$$Q_t = \beta_0 + \beta_1 Q_c + \beta_2 NDVI_{diff} + u + e. \tag{2}$$

We assessed the robustness of paired streamflow model results by including a longitudinal analysis of streamflow change. To predict annual streamflow for year t in the treated watersheds (Q_t), the longitudinal model included annual precipitation (P_t), a measure of antecedent storage (P_{t_lag}), and the amount of vegetation ($NDVI_t$) in the treated watershed as fixed effects, with watershed as a random effect (u):

$$Q_t = \beta_0 + \beta_1 P_t + \beta_2 P_{t\_lag} + \beta_3 NDVI_t + u + e. \tag{3}$$

Both precipitation variables were log-transformed. P_{t_lag} was the sum of annual precipitation from the prior two water years and was used to account for carryover storage effects in the soil and regolith that may affect hydrologic response.

We estimated the potential percentage change in annual streamflow for different levels of NDVI change based on the calibrated paired streamflow and longitudinal models. Because the observed variability in watershed-scale NDVI spanned approximately 0.1 units over the period of record for any given watershed, we limited the modelled streamflow change estimates to this range. Both the paired streamflow model and the longitudinal model were calibrated using the same procedures as the paired NDVI model.

4 | RESULTS

4.1 | Fuel treatment effects on mortality

Overall, the watersheds in Bull had a lower NDVI than the watersheds in Providence due to vegetation productivity being more cold-limited in Bull (Figure 3). In both Bull and Providence, the control watersheds had the highest NDVI of their respective groups. During the pre-drought period, the NDVI time-series was characterized as being generally steady (Bull) or slightly increasing (Providence). NDVI for all the watersheds peaked in 2011, which was the second wettest year during the monitoring period and just prior to the treatments and the onset of the drought. During the drought, NDVI decreased in all of the watersheds, albeit at varying rates, regardless of treatments (Figure 3).

For the Bull watersheds, the slope of the pretreatment relation between NDVI in the control watershed and NDVI in the treated watershed was approximately one, indicating that changes in NDVI in the treated watersheds were matched by similar changes in NDVI in the control watershed (Figure 4). The 2012 thinning event in B201 and B204 produced a notable decrease in the 2013 NDVI relative to the control watershed. In 2014, a small decrease in NDVI was observed in both B203 and B204 from the prescribed fires the previous fall. These results indicate that the fuel treatments produced observable changes in NDVI at the watershed scale. During the combined drought and mortality period (2012–2017), NDVI in the Bull control watershed (T003) showed a much larger decrease than the treated watersheds (Figure 4). The lower slope signifies that the fuel treatments may have reduced drought-enhanced mortality in the Bull watersheds.

In Providence, the slope of the pretreatment relation between NDVI in the control and treated watersheds was less than one, indicating that the NDVI in the control watershed had a greater range than the NDVI in the treated watersheds under pretreatment conditions (Figure 4). Following thinning treatments in 2012, a large decrease in NDVI was observed in P301, whereas a much smaller decrease was observed in D102. The small NDVI decrease in D102 was due to a large proportion of the watershed being excluded during

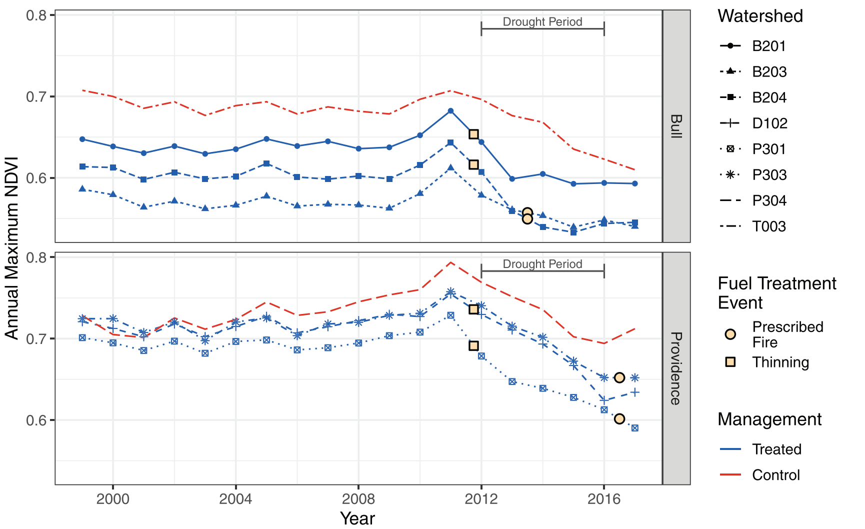

FIGURE 3 Time-series of the annual May to September maximum-value Normalized Difference Vegetation Index (NDVI) for each watershed, separated by Bull and Providence groupings. Thinning events are shown as filled squares and prescribed fire events are shown as filled circles

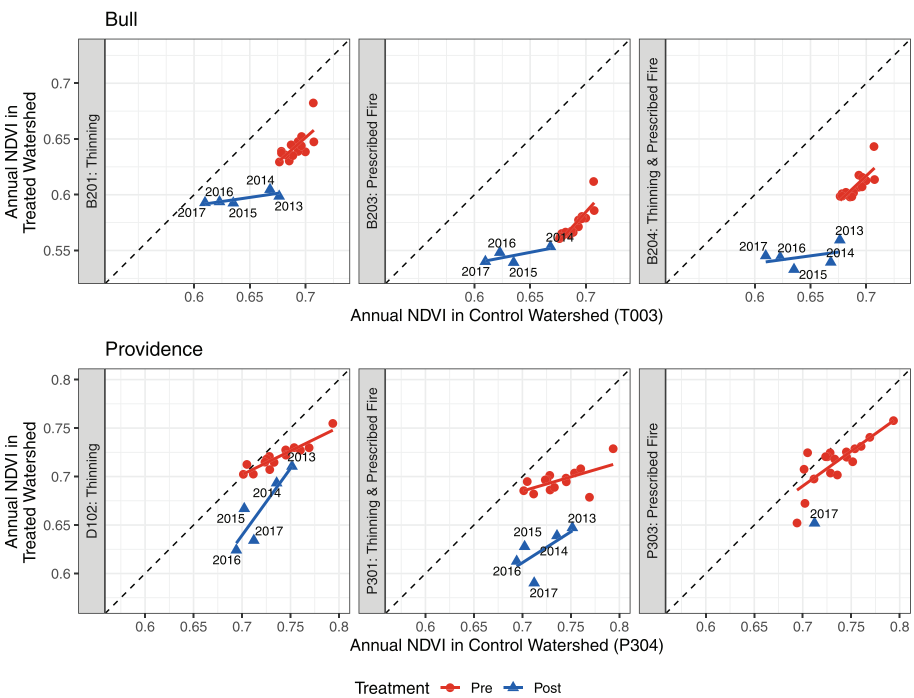

FIGURE 4 Relation between the watershed-averaged annual May–September maximum Normalized Difference Vegetation Index (NDVI) in the treated watershed ($NDVI_t$) and the control watershed ($NDVI_c$). Linear regression lines were separately fit to pretreatment and posttreatment values. Corresponding years are displayed for all posttreatment values. Dashed line designates 1:1 line

thinning operations (Lydersen et al., 2019) because the steep slopes in the watershed precluded access by the heavy equipment used in the mechanical thinning treatment (Table 1). After thinning, the slope of the relation between NDVI in the control watershed and NDVI in the treated watersheds was slightly steeper for P301 and D102, suggesting that the treated watersheds may have been more susceptible to forest mortality during the drought than the control watershed (Figure 4). This increase in slope was opposite of what was expected. The prescribed fire treatments in Providence occurred in fall 2016 and are only reflected in the year 2017. In the two watersheds that were burned, P301 and P303, NDVI in 2017 decreased relative to 2016, whereas in the two unburned watersheds, D102 and the control P304, NDVI in 2017 increased relative to 2016.

The results from the mixed-effects model confirm that NDVI in the treated Bull watersheds showed smaller reductions during the posttreatment period (i.e., less forest mortality) relative to NDVI in the control watershed, as the 95% credible intervals for the interaction variable beta coefficient do not overlap with 0 (Figure 5). In Providence, no posttreatment difference in the relation between NDVI in the treated and control watersheds was observed, as the beta coefficient was well within the uncertainty of the model. Measures of model performance and trace plots for the mixed-effects models may be found in the Supporting information (Tables S1, S2, and S3, Figure S1).

4.2 | Vegetation effects on streamflow

In this section, we examine the response of streamflow to changes in NDVI. The interannual pattern for streamflow was similar for all of the

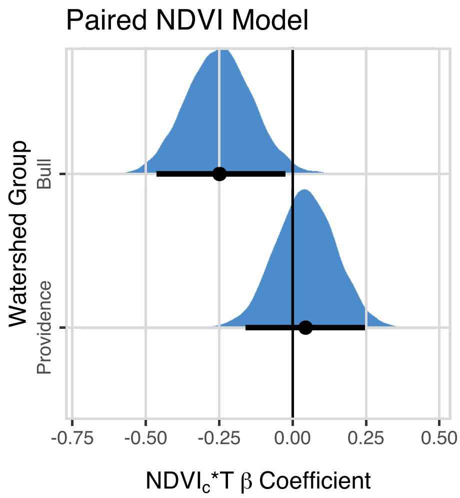

FIGURE 5 Mean β coefficient values (black dots) and 95% credible intervals (black lines) for the interaction variable $NDVI_c$*T in the paired Normalized Difference Vegetation Index (NDVI) model (Equation 1), separated by watershed grouping. Negative β values indicate that the $NDVI_t$ vs $NDVI_c$ relation had a lower slope in the posttreatment period, which we attribute to less forest mortality in the treated watersheds relative to the control watershed. Blue area represents density plot of 10,000 individual model runs

watersheds within their respective group (Figure 6). Overall, streamflow totals were higher in Bull than in Providence. Lowest annual streamflows were observed during the drought.

The relation between annual streamflow in the control watershed and annual streamflow in the treated watershed was very strong (Figure 7), indicating that precipitation is the first-order control on streamflow in these watersheds. Regarding the effect of differences in NDVI on the annual streamflow, the mixed-effects model results suggest that there was little streamflow response to changes in NDVI for the Bull watersheds (Figure 8a, Tables S4 and S5, Figure S2). In contrast, in Providence, the beta coefficient for $NDVI_{diff}$ fell outside the 90% credible intervals, providing moderate evidence that streamflow in the treated watersheds increased in response to NDVI reductions in the treated watershed relative to the control watershed (Figure 8a).

A longitudinal model was used to provide a second evaluation of streamflow response to changes in NDVI. Similar to the paired streamflow model, we found that NDVI did not have an effect on streamflow in the Bull watersheds (Figure 8b, Tables S6 and S7, Figure S3). However, in Providence, the coefficient for $NDVI_t$ fell outside the 95% credible intervals (Figure 8b). This result provides greater evidence that streamflow in Providence was responsive to vegetation change.

The potential percentage change in annual streamflow was estimated for a given decrease in NDVI (Figure 9). In Bull, the results showed minimal change in streamflow as NDVI deceased, though there was considerable uncertainty in the estimates. As an example, the estimated response in Bull ranges from a 32% increase to a 25% decrease in annual streamflow following a 0.05 reduction in NDVI. In Providence, the projections generally showed an increase in annual streamflow associated with a reduction in NDVI. In this case, the paired streamflow model estimated that annual streamflow would increase 24% (range −5% to 59% at 95% uncertainty) and the longitudinal model estimated that annual streamflow would increase 18% (range 3% to 35% at 95% uncertainty) following a 0.05 reduction in NDVI.

5 | DISCUSSION

5.1 | Evaluating changes in forest mortality

We found that reductions in NDVI during the drought were moderated by fuel treatments in the Bull watersheds but not in the Providence watersheds. Because we used NDVI as a proxy for mortality, these results imply that fuel treatments reduced forest mortality in Bull, but not in Providence. We recognize that processes other than forest mortality may reduce NDVI, such as overstory and understory declines and reduced vigour related to water stress in the absence of mortality. However, we contend that a large proportion of the observed posttreatment changes in NDVI were related to mortality and this contention is supported by plot-scale measurements in the watersheds (Lydersen et al., 2019).

Measurements of plot-scale understory canopy cover in the KREW suggest that changes in understory vegetation did not contribute to declines in NDVI during the posttreatment period (Lydersen et al., 2019). As an example, we observed that the Bull control watershed, T003, had the largest year-to-year reduction in NDVI in 2015, with further NDVI reductions in 2016 and 2017 (Figure 4). Lydersen et al. (2019) reported that in 2014, plot-scale canopy cover for woody vegetation under 2 m tall in T003 was at a sampling minimum for the watershed, with a canopy cover of approximately 5%. This amount of understory vegetation likely had only a small contribution to watershed-scale estimates of NDVI. Further, Lydersen et al. (2019) showed that understory canopy cover increased from 2014 to 2017 for all watersheds except P301, which was burned by prescribed fire in 2016. This expansion of understory canopy cover during the mortality period is counter to what would be expected if understory dieback were driving the reductions in NDVI. In contrast, the expansion of understory canopy cover during the mortality period may be explained by increased tree mortality, because tree mortality opens up the overstory canopy, enabling the release of understory.

Forest water stress can reduce NDVI even in the absence of mortality through reductions in leaf area. However, plot-scale measurements of forest mortality in the KREW support the assessment that mortality was the primary driver of posttreatment NDVI changes (Lydersen et al., 2019). Field surveys of live versus dead trees in 2017 showed that the control watershed in Bull (T003) had much greater riparian and upland mortality than the three treated watersheds, matching our observations of NDVI. In Providence, the surveys showed no clear difference in watershed mortality levels in response to fuel treatments, once again matching our observations. Our attribution of decreased NDVI to forest mortality is further supported by

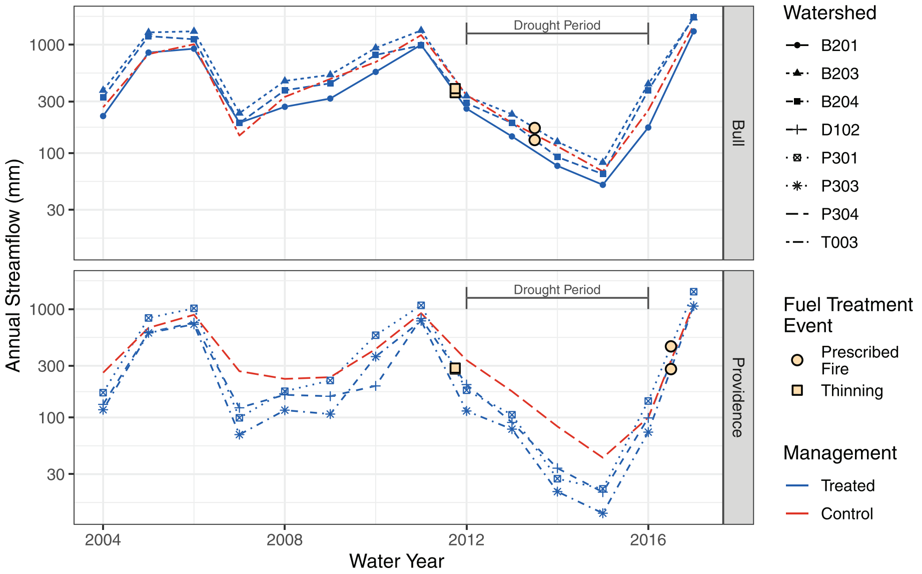

FIGURE 6 Time-series of observed annual streamflow for each watershed, separated by Bull and Providence groupings. Thinning events are shown as filled squares and prescribed fire events are shown as filled circles

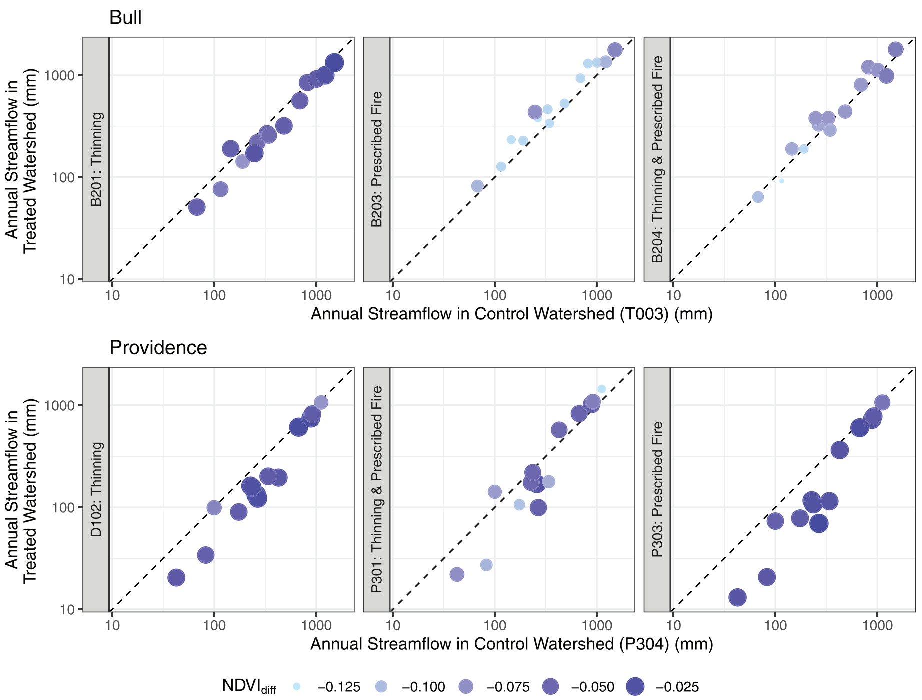

FIGURE 7 Relation between annual streamflow in the control watershed (Q_c) and the annual streamflow in the treated watershed (Q_t). Smaller and lighter blue circles correspond to lower Normalized Difference Vegetation Index (NDVI) values in the treated watershed relative to the control watershed (i.e., more negative $NDVI_{diff}$). Dashed line designates 1:1 line

posttreatment NDVI values being mostly lower than their pre-treatment ranges, with the exception of NDVI in the Providence control watershed (P304) (Figure 4). These results indicate that the processes causing reductions in NDVI were likely physical changes in vegetation, because they exceeded typical interannual variability.

The findings from our analysis using NDVI as a proxy for forest mortality are consistent with the findings from the earlier plot-scale analysis (Lydersen et al., 2019). Yet despite these similarities, Lydersen et al. (2019) concluded that tree mortality was not significantly influenced by fuel treatments. This discrepancy is likely due to differences in the way that the models assessed treatment effects on mortality. In this study, we examined whether there was a change in the relation (i.e., slope) between NDVI in the control water and NDVI in the treated watershed before and after treatment. Lydersen et al. (2019) examined the factors that could explain the proportion and number of dead trees in the watersheds during a 2017 field survey. The differences in the respective datasets and their effect on statistical inference highlight the importance of including multiple datasets, when possible, for analysing changes in vegetation.

5.2 | Where does water go following fuels treatments?

The results of this study suggest two different hydrologic responses to fuel treatments in the KREW. In the higher elevation Bull

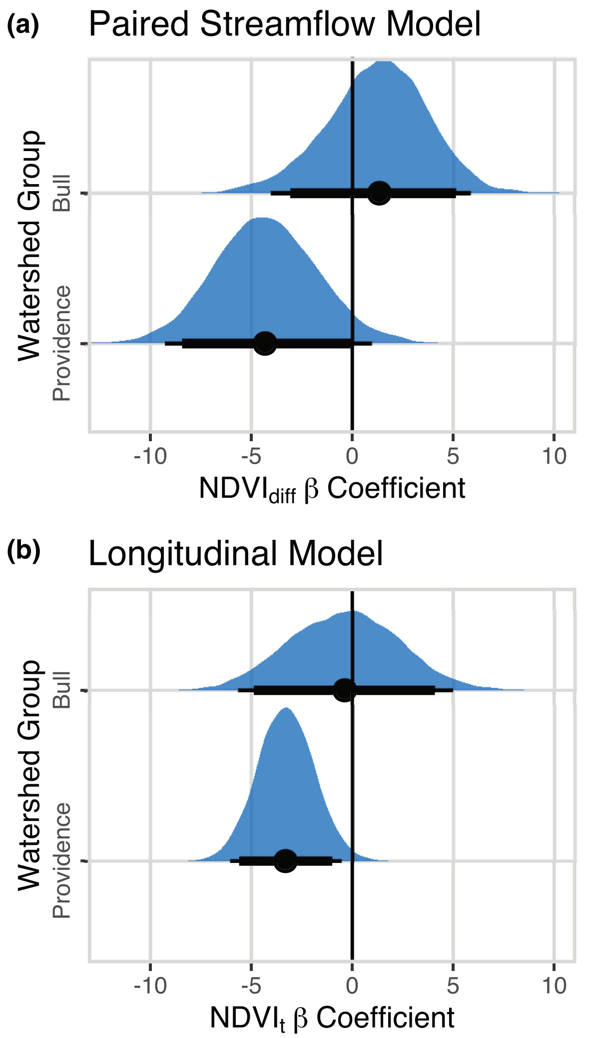

FIGURE 8 Mean β coefficient values (black dots), 90% credible intervals (thick line) and 95% credible intervals (thin line) for the variables $NDVI_{diff}$ and $NDVI_t$ in the paired streamflow model (Equation 2) and the longitudinal model (Equation 3), respectively. Negative β values indicate an increase in annual streamflow in the treated watershed for (a) a corresponding decrease in Normalized Difference Vegetation Index (NDVI) of the treated watershed relative to the control watershed (Paired Streamflow Model) or (b) a decrease in NDVI in the treated watershed (Longitudinal Model). Blue area represents density plot of 10,000 individual model runs

watersheds, fuel treatments reduced forest mortality during the drought, but did not have an effect on streamflow. In contrast, fuel treatments in the lower elevation Providence watersheds did not affect forest mortality during the drought but depending on the model, increased streamflow at 95% and 90% confidence. If we assume that the decrease in forest mortality in Bull was due to lower water stress associated with less competition (Sohn et al., 2016), then the results suggest different pathways for the water made available by fuel treatments (Figure 1). We identified three pathways that the unused water can follow after fuels treatments: abiotic evaporation, transpiration by neighbouring vegetation and augmentation of streamflow (Figure 1). The dominant pathway will depend on a variety of factors that are likely to vary within watersheds, across watersheds and through time. While a comprehensive evaluation of the water made available after fuel treatments includes all three mechanisms, in this section, we focus only on the watershed characteristics and conditions that may affect the partitioning of unused water to neighbouring vegetation or to streamflow.

Two of the primary controls on the partitioning of unused water to neighbouring vegetation are (a) the neighbouring vegetation being water limited and (b) the neighbouring vegetation having access to the water that was made available because of fuel treatments. Both of these components must be fulfilled for the neighbouring vegetation to transpire the water, otherwise the unused water will contribute to one of the alternate pathways. The findings of this study suggest that vegetation in Bull was both water-limited and that subsurface water stores were accessible among the remaining vegetation, thus there was no increase in streamflow. On the other hand, the increase in streamflow in Providence suggests that, one, or both, of these requirements may not have been met.

Both Bull and Providence faced water limitations during the drought (Goulden & Bales, 2019), as evidenced by the drought-enhanced mortality in both of their respective control watersheds. However, the causes for this water limitation differed for each watershed group. Regolith thickness at Providence has been shown to be highly variable; in some cases exceeding 10 m (O'Geen et al., 2018). While no similar measurements have been made in the Bull watersheds, measurements along a nearby elevation gradient indicate that regolith thickness decreases as elevation increases above ~1,100 m (O'Geen et al., 2018). Further, the effective rooting depth associated with the soil series at Providence range from 0.75 to 2 m, whereas at Bull they range from 0.5 to 1 m (Hunsaker et al., 2012). These measurements suggest the combined soil and regolith water storage is higher in Providence than in Bull, and this combined storage would provide a larger reservoir to draw down during drought. Nevertheless, Bull may not require as much subsurface storage as Providence to meet the evaporative demand of the vegetation because the higher elevation Bull watersheds have lower potential evaporation rates, shorter growing seasons and lower average tree densities (Lydersen et al., 2019).

We do not have explicit knowledge of the rooting networks or the exact mechanisms of access to subsurface water for vegetation at either Bull or Providence. As such, evaluating the capability of vegetation to access water made available by the removal of neighbouring vegetation is challenging. However, deeper water storage in Providence would indicate that trees in those watersheds likely devote more carbon resources to producing deep roots than shallow, horizontal roots (Fan, Miguez-Macho, Jobbágy, Jackson, & Otero-Casal, 2017). Consequently, rooting systems in Bull may be shallower and contain more overlap with neighbouring trees than in Providence. This overlap could make it easier to uptake unused water when neighbouring vegetation is removed. A recent modelling study showed that fuel treatments in watersheds with low water storage capacity and high overlap of rooting systems resulted in an increase in forest water use by the neighbouring trees (Tague & Moritz, 2019). The study also showed that the greatest posttreatment increases in streamflow

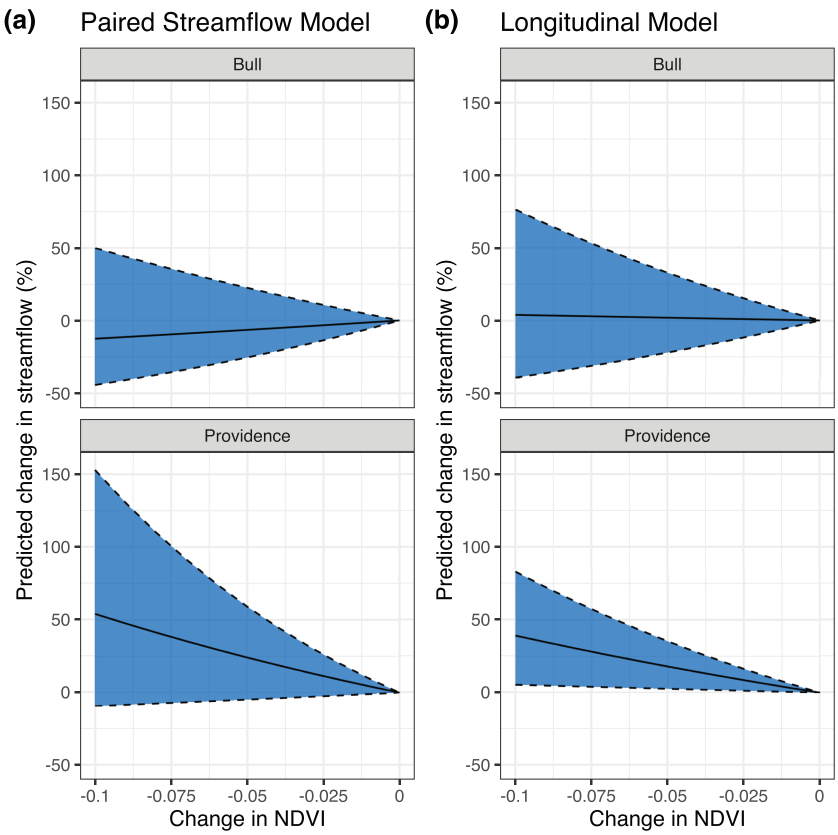

FIGURE 9 Predicted change in annual streamflow for a given change in Normalized Difference Vegetation Index (NDVI) based on the calibrated paired streamflow and longitudinal models. The blue area indicates 95% uncertainty intervals

occurred in watersheds with high water storage capacity and non-overlapping roots. Our empirical results are consistent with this explanation.

At the watershed scale, hydrologic responses to fuel treatments are likely to follow a combination of all three pathways that we have identified (Figure 1). Depending on the watershed and the post-treatment meteorological conditions, one of the pathways may dominate, or the unused water may be split across all three pathways. For example, abiotic evaporation and transpiration by remaining neighbouring vegetation may be more prominent under dry conditions when the amount of unused water is insufficient to move beyond the rooting zone. Under wetter conditions, when neighbouring vegetation is less water stressed, and there is ample water to drain through the rooting zone, we may expect an increase in streamflow. Because the unused water can be partitioned along multiple pathways, it can also lead to the appearance of a muted hydrologic response to fuel treatments when only examining one outcome pathway. This effect could partly explain the variability in streamflow and forest mortality responses to fuel treatments that have been reported in the literature (Bosch & Hewlett, 1982; Sohn et al., 2016). All three pathways will need to be assessed to accurately understand the hydrologic response to fuel treatments in any given watershed.

6 | CONCLUSIONS

Reduced forest mortality and increased streamflow are frequently promoted as hydrologic benefits of fuel treatments in water-limited systems. However, both of these benefits rely on water made available from vegetation removal. Hence, the benefits can offset one another. In this study of two groups of watersheds in the Sierra Nevada, we found empirical evidence of this tradeoff. In the higher-elevation Bull watersheds, fuel treatments were found to reduce forest mortality, but no changes in annual streamflow were observed. In the Providence watersheds, fuel treatments did not affect forest mortality, but increases in annual streamflow were observed at 95% and 90% confidence, depending on the model. These results imply that the water made available from implementing low-intensity fuel treatments followed different hydrological pathways through the watershed, with uptake by neighbouring vegetation dominating in Bull and augmentation of streamflow dominating in Providence.

Our findings have important implications for both science and management, as the water made available by reducing stand density is finite and cannot be fully allocated to multiple potential benefits. From a scientific perspective, we need to better understand the processes that control when, where and under what conditions unused water from fuel treatments will follow a given pathway. From a management perspective, there will be inherent uncertainty in the allocation of water benefits following fuel treatments due to a lack of process understanding. Fuel treatments in water-limited forests may reduce water stress during a subsequent drought, increase streamflow or provide a combination of these two benefits, but will not provide the full benefits across these competing outcomes simultaneously. Finally, the findings in this study highlight the benefits of simultaneously examining multiple responses to fuel reduction treatments as a means for providing greater understanding of ecohydrologic processes.

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CONFLICT OF INTEREST

The authors declare that there is no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available in the supplementary material of this article (Data S1).

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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