If a Tree Falls in the Forest Does it Make a Splash? Forest Restoration in the Central Sierra Nevada Provides a Robust Hydrological Hedge Against Droughts

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Key Points:

- Restoring historic forest disturbance return intervals can increase water yield by 8-14% during dry years in the central Sierra Nevada.
- Understory largely compensates for reduced overstory transpiration, so 73% of streamflow gains are attributable to reduced interception.
- Thinner forests can increase headwaters peak flows, but climate uncertainty overwhelms this effect at the reservoir scale.

Abstract

Forest thinning and prescribed fire are expected to improve the climate resilience and water security of forests in the western U.S., but few studies have directly modeled the hydrological effects of multi-decadal landscape-scale forest restoration. By updating a distributed process-based hydrological model (DHSVM) with vegetation maps from a distributed forest ecosystem model (LANDIS-II), we simulate the water resource impacts of forest management scenarios targeting partial or full restoration of the historic disturbance return interval in the central Sierra Nevada mountains. In a fully restored disturbance regime, the models predict additional reservoir inflow of 4-9% total and 8-14% in dry years. At sub-watershed scales (10-100 km^2), thinning dense forests can increase streamflow generation by >20% in dry years. In a thinner forest, increased understory transpiration largely compensates for decreased overstory transpiration. Consequentially, 73% of streamflow gains are attributable to decreased overstory interception loss. Thinner forests can increase headwater peak flows, but reservoir-scale peak flows are almost exclusively influenced by climate projections. Uncertainty in the future precipitation volume causes high uncertainty in the future water yield, but the additional volume of water attributable to forest restoration is about five times less sensitive to annual precipitation uncertainty. This partial decoupling of streamflow generation from annual precipitation makes forest restoration especially valuable for water supply during dry years or in a drier future climate. Our study can increase confidence in the water resource benefits of forest restoration in the central Sierra Nevada mountains, and our modeling framework is widely applicable to forested mountain landscapes.

1 Introduction

Historical fire suppression (Stephens et al. 2016, van Wagtendonk et al. 2018) and anthropogenic climate change (Seidl et al. 2017, Tague and Dugger 2010) have combined to force North American conifer forests into an unstable ecosystem state (Marlon et al. 2012, Schoennagel et al. 2017). As a result of fire exclusion policies, forests in the Sierra Nevada mountains of California and Nevada are artificially homogenous and dense (Collins et al. 2011, Dolanc et al. 2014, Taylor et al. 2014), which increases vulnerability to damaging megafires (Skinner and Chang 1996, Steel et al. 2015, Goss et al. 2020, Safford et al. 2022) and negatively impacts water security (Boisramé et al. 2017, Stephens et al. 2021b). Forest restoration, including mechanical thinning and prescribed fire to restore historical disturbance frequencies (North et al. 2007, Scholl and Taylor 2010, Kalies and Yocom Kent 2016, Stephens et al. 2021a), can increase the resilience of forests (Hessburg et al. 2019, Knapp et al. 2021) and provide societal benefits including reduced catastrophic fire risk (Loudermilk et al. 2016), stabilized carbon storage (Liang et al. 2018, Cabiyo et al. 2021), wood products (Swezy et al. 2021, Elias et al. 2023), increased ecological diversity (Stephens et al. 2021b), and increased water yield (Guo et al. 2023, Chung et al. 2024).

Multi-benefit forest restoration frameworks may help fund restoration projects by bundling convergent societal goals and economic incentives to build partnerships (Stephens et al. 2021, Quesnel Seipp et al. 2023). The Tahoe Central Sierra Initiative (TCSI) is one such partnership, focusing on multi-benefit resilience-based forest management plans for the

Truckee, Yuba, Bear, and American River basins (Manley et al. 2023). As part of the TCSI, the U.S. Forest Service is assessing the feasibility of environmental markets for ecosystem services such as water supply, carbon storage, and wood products. Several environmental markets could potentially benefit from forest management tactics leading to more frequent tree removals and controlled fire, aligned with partial or full restoration of the pre-colonial disturbance return interval (van Wagtendonk et al. 2018, Maxwell et al. 2022). In this study, we investigate the potential impact of landscape-scale forest restoration scenarios developed by Maxwell et al. (2022) on water resources in the TCSI region.

The possibility of increased water yield has incentivized forest thinning efforts for decades, but the magnitude and even direction of post-disturbance streamflow changes vaires widely across western North America (Goeking and Tarboton 2022). It is broadly observed that streamflow may increase if forest cover is reduced because of decreased evapotranspiration (ET; Hibbert 1967, Bosch and Hewlett 1982, Troendle and King 1985). At the landscape scale, however, the hydrological response to vegetation disturbance is more nuanced (Andréassian 2004), and ET can increase with a thinner canopy (Goeking and Tarboton 2020) due to a combination of reduced shading (Morecroft et al. 1998), wetter soils (Boisramé et al. 2018), and vegetation regrowth (Perry and Jones 2016). In their analysis of forest disturbance for 159 watersheds in the western U.S., Goeking and Tarboton (2022) found that energy-limited watersheds are more likely to experience increases in streamflow associated with reductions to forest cover compared to wetter and cooler watersheds. This mediating effect of aridity on reductions in ET after vegetation disturbance is similarly observed locally in the Sierra Nevada mountains, where more northerly (wetter) watersheds experience larger streamflow gains following disturbance (Saksa et al. 2017). Previous studies in the TCSI region, which is part of the wetter northern Sierra Nevada zone, have predicted substantial increases in streamflow after wildfire or forest restoration (Saksa et al. 2020, Roche et al. 2018 and 2020, Guo et al. 2023). However, prior attempts to quantify landscape-scale streamflow changes usually rely on extrapolation from a few years of pre- and post-disturbance measurements at the plot or small catchment scale (e.g. Saksa et al. 2020, Roche et al. 2018). Landscape-scale forest restoration is not an instantaneous nor homogenous process, and the hydrological effects of proposed multi-decadal forest treatment plans (Maxwell et al. 2022) are further complicated by the role of climate change in mediating ecohydrology in the coming decades (Tague and Dugger 2010).

Seasonal snowmelt controls water resources in much of western North America (Bales et al. 2006), and there have been long-standing efforts to understand how forest management might enhance the snowpack (Troendle 1983, Kattelmann et al. 1983). A majority of studies analyzed by Varhola et al. (2010) and Goeking and Tarboton (2020) associate thinner forests with increased snow water equivalent (SWE). However, forest-snowpack relationships exhibit considerable variability due to complex mass and energy interactions that are mediated by climate and topographic controls. Reduced canopy interception tends to increase SWE, particularly in areas with snowfall events near or below the canopy storage capacity (Storck et al. 2002, Boon 2012, Winkler et al. 2012). However, increased shortwave radiation reaching the snowpack in thinner forests can accelerate ablation, particularly in warmer climates and on southern aspects, which results in reduced SWE after disturbance (Ellis et al. 2011, Harpold et al. 2014, Tennant et al. 2017). Darker snow albedo in post-fire forests can compound the effect of reduced shading, further accelerating snow ablation (Gleason et al. 2013). Due to the competing interaction of interception and shading effects, as well as additional factors including wind and thermal radiation from trees, many studies conclude that the snowpack response to disturbance is a complex function of the local conditions and fine-scale forest structure (e.g., Troendle 1983, Stevens 2017, Sun et al. 2018, Harpold et al. 2020). In the TCSI region considered in this study, high resolution lidar and modeling studies indicate that snow accumulation and meltwater inputs increase or decrease in different areas depending on the interaction of canopy structure with solar radiation and wind (Lewis et al. 2023), and opening gaps in dense canopies will promote snow accumulation and reduce ablation (Piske et al., in press).

The effect of forest disturbance on net ET, and hence water yield, depends on a cascade of complex factors (Moore and Heilman 2011, Adams et al. 2012). The different vapor loss components of ET (i.e., transpiration and evaporation or sublimation from interception storage) can be categorized depending on whether they are expected to increase or decrease in a thinner forest (c.f. Figure 1 of Goeking and Tarboton 2020). Declines in transpiration have been observed following tree mortality (Bales et al. 2018), which can reduce the soil moisture deficit and thus support additional streamflow generation from future precipitation (Troendle 1979, He et al. 2013, Boisramé et al. 2018). However, increased soil water availability sometimes supports higher transpiration rates from the remaining trees that would otherwise be water-limited, thereby capping streamflow gains during dry periods (Boisramé et al. 2019). Moreover, aerodynamic effects and reduced shading can create a more arid microclimate and contribute to increases in ET from remaining vegetation (Morecroft et al. 1998, Rambo and North 2009, Ma et al. 2010, Meili et al. 2024), further limiting the streamflow response to disturbance (Biederman et al. 2014, Meili et al. 2024). As a result, watershed-scale vapor loss can increase, decrease, or remain relatively unchanged following severe forest disturbance (Goeking and Tarboton 2020). Predicting the landscape-scale response to proposed management plans requires modeling the interaction of topography, soils, snow, and vegetation over decades of planned forest treatments under a changing climate.

Forest disturbance generally increases peak streamflow (Goeking and Tarboton 2020), but this response is dependent on the interaction of many processes. At small watershed scales, peak flows tend to increase after disturbance due to interactions between elevated antecedent water tables, reduced interception, increased snow accumulation, and faster snowmelt (Lewis et al. 2001, Moore and Wondzell 2005, Pomeroy et al. 2012). Additionally, natural or prescribed fires may contribute to peak flows through increased surface runoff from hydrophobic soils (Certini 2005). The ancillary effects of mechanical thinning such as soil compaction (Startsev and McNabb 2000) or the cutting of new forest roads can similarly alter flow paths and increase peak flows (King and Tennyson 1984, Bowling and Lettenmaier 2002). Peak flow increases on the order of 50% or more have been observed after forest disturbance in catchments up to a few square kilometers in area (Moore and Wondzell 2005), but peak flow impacts in larger-scale basins remain uncertain. Jones and Grant (1996) use the variable timing of forest harvest to propose a statistical argument that forest disturbance may contribute to substantially higher peak flows in watersheds as large as 600 km^2 in the Oregon Cascade Range, though this finding is contradicted by reanalysis of the same data by Thomas and Megahan (1998). Sierra Nevada watersheds face increased flood risks from anthropogenic climate change (Huang and Swain 2022). Consequentially, the possible effects of forest restoration on peak flows in the central Sierra Nevada requires consideration.

In this study, we address two of the unsolved problems in hydrology, namely the impact of land cover change on water fluxes and the spatial variability in hydrologic extremes in response to this change (Blöschl et al. 2019) by incorporating the outputs of a forest ecosystem model into a distributed hydrological model to simulate the spatially explicit hydrological effects of landscape-scale forest management scenarios. Forest management scenarios considered here are based on input from the broader TCSI partnership (Maxwell et al. 2022), so our hydrological investigation is uniquely grounded in detailed and plausible management alternatives in a landscape-scale forest planning exercise. Using a novel combination of two state-of-the-art models and a new Bayesian calibration method, we seek to answer the following questions: (1) How much additional water yield would result from partial or full restoration of the historic forest disturbance return interval in the central Sierra Nevada, and which factors control variability in the sub-watershed response? (2) Could forest restoration increase peak flows in ways that might accentuate flood risks to small- or large-scale infrastructure? (3) How do the sources of uncertainty inherent in multi-decadal simulations (climate, model uncertainty, and other unknowns) affect water resource planning?

2 Methods

Figure 1 outlines the three main components of this study. First, we set up and calibrate a hydrological model using multiple types of historical water data. Second, we set up and calibrate a forest ecosystem model to simulate vegetation responses to different forest management scenarios. Third, we run the hydrological model to the end of the century using climate projections and vegetation states from the forest ecosystem model. The following sections address each of these components in detail.

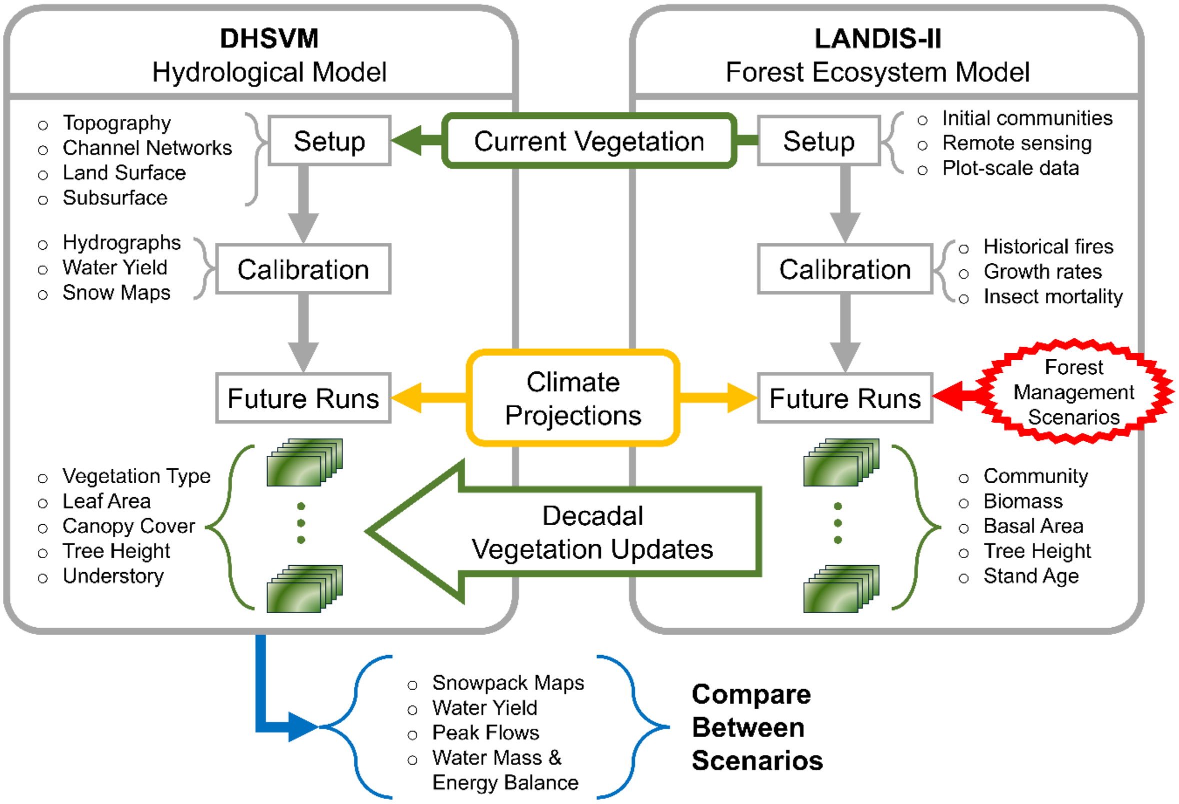

Figure 1. Methodological flowchart for simulation procedures used in this study. Vegetation states simulated by the LANDIS-II forest ecosystem model are used to update the DHSVM hydrological model under different forest management scenarios.

2.1 Hydrological Modeling

A physics-based, distributed-parameter approach to hydrological modeling enables us to quantify the interacting effects of forest restoration on hydrological processes and evaluate spatial heterogeneity in the watershed response to forest management scenarios. The Distributed Hydrology Soil Vegetation Model (DHSVM) was developed by Wigmosta et al. (1994, 2002) to simulate mountain watersheds with a particularly robust treatment of forest ecohydrological processes, such as overstory and understory interception and transpiration. DHSVM solves a water mass and energy balance on a two-dimensional grid, typically at high spatial resolution (90 m in this study) on a sub-daily timestep (3 hours in this study). The model represents vegetation with a two-layer “big leaf” approach that enables the separate calculation of overstory and/or understory interception storage, interception loss, and transpiration from three root zone soil layers in each grid cell. Streamflow in DHSVM is primarily generated from saturated subsurface discharge to a spatially explicit channel network, though overland flow paths are also included. Soil moisture is recharged by surface infiltration, and water in the saturated zone moves laterally through a shallow aquifer based on two-dimensional hydraulic gradients, approximated in this study from the local surface topography and spatially variable soil hydraulic conductivity. Compared to other process-based ecohydrology models, DHSVM is notable for the high grid-scale resolution of its water mass and energy balance and the high fidelity of its ecohydrological processes, which makes it particularly suitable for simulating the hydrological effects of heterogeneous forest disturbances (see 30-model intercomparison of Beckers et al. 2009).

2.1.1 Hydrological Model Setup

We combine literature review and a variety of spatial datasets to set up DHSVM for the study region in the central Sierra Nevada mountains. The TCSI area includes four adjoining watersheds, as shown in Figure 2: the Truckee River (2,273 km^2, mean elevation 2,100 m), the Yuba River (3,435 km^2, mean elevation 1,305 m), the Bear River (585 km^2, mean elevation 821 m), and the American River (4,812 km^2, mean elevation 1,350 m). Topography is represented using elevations from the Shuttle Radar Topography Mission (SRTM; Farr et al. 2007). Stream networks are initiated with a minimum catchment area of 0.1 km^2, which is consistent with imagery and the National Hydrography Database (NHD) channel network (U.S. Geological Survey 2019). Textural soil data required for DHSVM are derived from POLARIS (Chaney et al. 2019), a high-resolution probabilistic remapping of the Soil Survey Geographic (SSURGO) Database (Soil Survey Staff 2018). The multi-layer POLARIS data are vertically aggregated to three root-zone depth layers up to 1.2 m depth (Jackson et al. 1996), plus a deep layer. The Supporting Information includes an exhaustive description of inputs to DHSVM and their estimation for this study, and relevant code is available online (see Availability Statement).

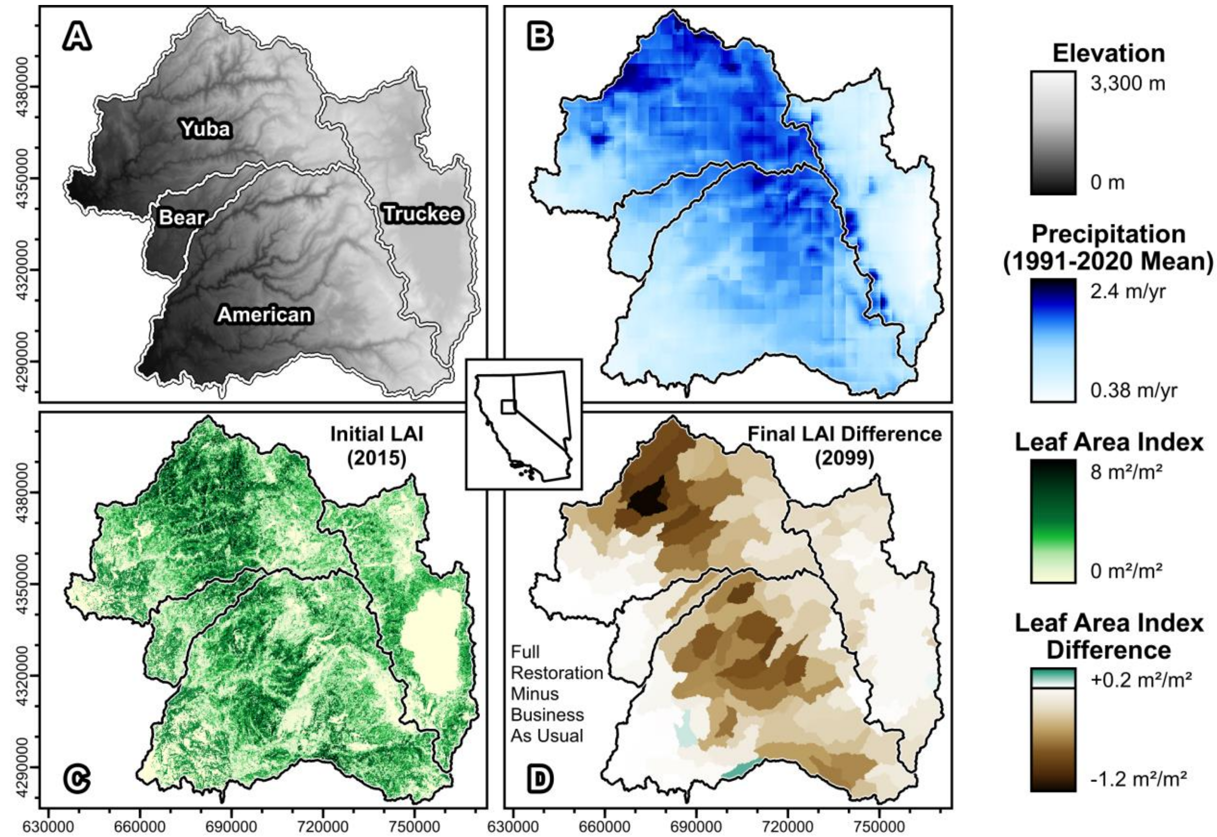

Figure 2. Maps of the Tahoe-Central Sierra project area: (A) digital elevation model; (B) mean yearly precipitation; (C) pre-restoration leaf area index (LAI); (D) final LAI difference between the full-restoration (S6) and business-as-usual (S2) management scenarios aggregated by sub-watershed. Inset map shows the study area location in the U.S. states of California and Nevada.

The parameterization of vegetation in DHSVM requires particular attention to robustly estimate the effects of forest restoration. Spatially explicit maps of vegetation type, canopy fractional cover, overstory monthly leaf area index (LAI), tree height, and dense understory presence are derived from outputs of the forest ecosystem model (Section 2.2). Where dense understory is not indicated, we assume a light understory presence beneath the forest canopy based on field experience in the study region, which indicates a typical lack of bare soil. Understory is parameterized with a constant LAI of 3 m^2/m^2 for dense understory or 1 m^2/m^2 for light understory based on field estimates of LAI for shrub ecosystems common to the study area (Hughes et al. 1987 and McMichael et al. 2004). Through manual sensitivity tests, we find that most vegetation parameters are insensitive to minor perturbations in the study region, so these parameters are maintained at baseline values used elsewhere in the DHSVM literature. One particularly sensitive parameter is the minimum stomatal resistance, which controls overstory transpiration for each vegetation type. We conduct a review of 18 species-level stomatal conductance field studies to estimate minimum resistance values and uncertainties for 15 distinct DHSVM vegetation types with at least two literature estimates per type (see Supporting Information). Canopy interception parameters are estimated from numerous field studies using the literature review summaries compiled by Link et al. (2004) for rain and Martin et al. (2013) for snow. Finally, abiotic land surfaces including water, urban, and rock types are mapped using the National Land Cover Database (NLCD; Dewitz and U.S. Geological Survey 2019) and represented in DHSVM with impermeable areas and detention storage (Cuo et al. 2008).

Meteorological forcing data for DHSVM are generated using a variety of spatiotemporal disaggregation techniques. Historic meteorological data are obtained from gridMET (Abatzoglou 2013), and future climate projections are obtained from the analogous Multivariate Adaptive Constructed Analogs (MACA) downscaled global circulation model (GCM) dataset (Abatzoglou and Brown 2012). Both climate datasets provide daily precipitation, minimum/maximum temperature, and wind speed at 4 km resolution. The model requires inputs of precipitation, air temperature, relative humidity, wind speed, incoming shortwave radiation, and incoming longwave radiation. We temporally disaggregate the daily data to the 3-hour DHSVM timestep and simulate the additional required variables using MetSim (Bennett et al. 2020), which is based on the MTCLIM model (Hungerford et al. 1989). Precipitation is assumed constant within each day, which is a reasonable assumption for the multi-day cold-season storms typical of the central Sierra Nevada (e.g. Huning and Margulis 2017), but we acknowledge that this assumption could affect assessments of rain-driven peak flow events. We spatially disaggregate 4 km gridMET precipitation data using monthly redistribution maps calculated from 800 m PRISM normals (PRISM Climate Group 2022) to exactly preserve the spatial mass balance of the 4 km data while redistributing precipitation approximately proportional to the monthly PRISM normals within each 4 km cell (Figure 2B). Preserving the exact gridMET precipitation mass balance is a priority in this study for the sake of consistency with the gridMET forcing data used for LANDIS-II, so we eschew common downscaling techniques that are not mass-preserving, such as bilinear interpolation. Temperature data are spatially disaggregated to the 90 m DHSVM grid scale using spatially variable monthly lapse rates calculated by linear regression of elevation versus PRISM monthly temperature normals for the 25 800 m cells within each 4 km gridMET cell. In the study region, there are 193 gridMET cells covering the Truckee watershed, 295 covering the Yuba watershed, 65 covering the Bear watershed, and 385 covering the American watershed. Baseline DHSVM parameters controlling snowpack accumulation and ablation are set based on the results of Sun et al. (2018) for the Sierra Nevada region.

2.1.2 Hydrological Model Calibration

We refine our baseline setup of DHSVM with calibration of key parameters. Manual sensitivity tests reveal seven parameters that are sensitive to perturbation within the a priori uncertainty range of available data: mean soil depth [m], hydraulic conductivity [m/s], the exponential decrease in hydraulic conductivity with depth [-], porosity [%], minimum stomatal resistance [s/m], the maximum air temperature for snowfall [°C], and the melt-season albedo decay rate [-]. Soil depth is calibrated using an offset applied to a baseline pattern based on topographic curvature (Patton et al. 2018) to generate a mean soil depth between 1.3 and 4 m based on the minimum required to satisfy the rooting depth and upper-bound sensitivity tests. Hydraulic conductivity and porosity are calibrated between the 5th, 50th, and 95th percentile values for each grid cell and soil layer provided by the POLARIS data (Chaney et al. 2019). Single-valued parameters are calibrated over a prior range determined from variability observed in the literature or manual sensitivity tests, as outlined in the Supporting Information. All parameters and maps are maintained within plausible physical ranges, and all four major TCSI basins are calibrated together to improve generality.

To quantify uncertainty in key hydrological processes, we evaluate six objective functions targeting different hydrological signatures. The selected objective functions are daily streamflow Nash-Sutcliffe Efficiency (NSE), log-scale NSE for two sets of sub-watersheds exhibiting comparatively high or low baseflows, root mean squared error (RMSE) for yearly water yield at large scales, RMSE for 95th-percentile high flows at large scales, and pixel-wise RMSE for yearly peak SWE. The calibration period is defined as water years 2012-2017 for streamflow data (not counting one year of model spin-up) in order to capture a range of variability, including a multi-year drought (2013-2015) and one of the wettest years in recent decades (2017). Unimpaired or reconstructed daily streamflow timeseries are obtained for 10 stream gauges in the Truckee and Yuba watersheds (U.S. Geological Survey 2022; refer to calibration figures in Supporting Information for gauge numbers). Large-scale flows are constrained with reconstructions of natural streamflow at the YRS flow point (California Department of Water Resources 2022). Yearly pixel-wise peak SWE maps are calculated for the Yuba and Truckee watersheds from the Margulis et al. (2016) snow reanalysis for water years 2011-2016.

We apply multi-objective Bayesian optimization to the sensitive parameters identified previously in order to minimize errors in each objective function and quantify residual hydrological uncertainty. In Bayesian optimization, objective functions are modeled using stochastic processes as surrogates for the underlying model (DHSVM in our case), which boosts calibration efficiency (Jones et al. 1998). Starting with an initial minimax Latin hypercube sample of the seven-dimensional parameter space (Morris and Mitchell 1995, implemented by Dupuy et al. 2015), we use Gaussian Process regression (Rasmussen and Williams 2008, implemented by Roustant et al. 2012) to build surrogate models of each objective, and optimize the expected hypervolume improvement of subsequent parameter sets (Emmerich et al. 2011, implemented by Binois and Picheny 2019) with parallel particle swarm optimization (Kennedy and Eberhart 1995, implemented by Zambrano-Bigiarini et al. 2013). After testing a total of 300 unique parameter sets during 6 rounds of optimization, we identify Pareto-efficient designs, which constitute the set of models where improving one objective requires worsening another. From this Pareto set of calibrated models, we select three high-performing models that are representative of the tradeoffs between objective functions and the residual uncertainty in calibration parameters (SI Figures S1-S3). We validate the selected models over water years 2006-2011. During validation, we expand our objective functions to include two additional stream gages and reconstructed natural flows in the American River watershed at the AMF/NAT flow point below Folsom Lake (California Department of Water Resources 2022).

2.1.3 Hydrological Model Future Runs

We simulate the long-term hydrological effects of forest restoration by running DHSVM subject to future climate projections and updating vegetation maps to reflect spatiotemporal heterogeneity associated with forest disturbances, management, and regrowth. We select the CNRM-CM5 (Voldoire et al. 2013) and MIROC5 (Watanabe et al. 2010) climate projections since these models represent endmembers for fire weather in the central Sierra Nevada study region (Maxwell et al. 2022), with CNRM-CM5 representing a relatively wet scenario, and MIROC5 representing a relatively dry scenario. We select the RCP-8.5 pathway for both GCMs since this higher-emissions scenario fits likely behaviors in the near to mid-future (Schwalm et al. 2020). On average within the project area, the CNRM-CM5 climate projection has a temperature trend (Sen 1968) of 0.062 °C/yr, amounting to a 5.3 °C increase over the 85-year simulation period, with mean precipitation of 597 mm/yr. The MIROC5 climate has a temperature trend of 0.045 °C/yr, amounting to a 3.9 °C increase over 85 years, with 456 mm/yr mean precipitation (24% less than CNRM-CM5). GCM projections are further disaggregated and downscaled from the Abatzoglou and Brown (2012) MACA dataset using the same methodology outlined above for consistency with the forcing data on the historic calibration and validation periods.

The effect of ongoing forest restoration is represented in DHSVM with updated maps of vegetation type, canopy fractional cover, overstory LAI, tree height, and dense understory presence every ten years. Updated vegetation maps are ingested into DHSVM on October 1st five years prior to the date that they represent. Offsetting the dates with this approach enables DHSVM to simulate hydrological conditions for each set of vegetation maps over a 10-year period centered on the year corresponding to the updated vegetation maps. More frequent (e.g., yearly) updates of the DHSVM vegetation maps could better resolve rapid changes near the beginning of the treatment period, but we find that decadal updates can satisfactorily resolve relatively slow trends in the forest landscape after the first decade of treatment.

2.2 Forest Ecosystem Modeling

To model change in vegetation over time and in response to climate and management, we used the LANDIS-II forest landscape model (Scheller et al. 2007) with parameterization following Maxwell et al. (2022). LANDIS-II is a flexible modeling framework that allows for varying extensions to model vegetation dynamics and disturbance in a spatially explicit gridded format allowing communication among cells (e.g, by seed dispersal or fire spread). In this study, we also use the Net Ecosystem Carbon and Nitrogen (NECN v6.9) succession extension (Scheller et al. 2011). NECN is a mechanistic succession model which tracks cohorts of trees (each with associated age, species, and biomass) as they grow, reproduce, recruit, and senesce. Cohort growth and establishment depend on site conditions (e.g., climate, soils) and competition with other cohorts for water, growing space, and soil nitrogen. NECN tracks carbon and nitrogen through multiple biomass and soil compartments. In NECN, climate has emergent effects on ecosystem processes through its impact on vegetation growth, respiration, and soil carbon dynamics. All model parameters and installers needed to reproduce our TCSI forest ecosystem model are available online (see Open Research section).

2.2.1 Forest Ecosystem Model Setup and Calibration

The initial (year 0) landscape of the LANDIS-II model is derived from multiple data sources. Initial vegetation conditions are generated from Forest Inventory and Analysis (FIA) plots (Burrill et al. 2021) imputed from Landsat remote sensing products with soil data from SSURGO. We model several key disturbance processes, including fire (natural ignition and prescribed), insect pests, and harvest (through implementation of the management scenarios).

To model wildfire and prescribed fire, we use the Social-Climate-Related Pyrogenic Processes and their Landscape Effects (SCRPPLE v3.2.3) extension (Scheller et al. 2019), a data-driven empirical model of fire spread, fire intensity, and tree mortality. SCRPPLE simulates fire spread, intensity, and mortality depending on fuels, weather, and topography. The fire parameters are calibrated to ignitions data (Short 2021), daily fire perimeters from the National Interagency Fire Center (NIFC 2019) and fire severity maps from Monitoring Trends in Burn Severity (Eidenshink et al. 2007). Tree mortality is parameterized using the Fire and Tree Mortality Database (Cansler et al. 2020). For details, refer to the appendix of Maxwell et al. (2022).

Insect pests are simulated using the Biomass Biological Disturbance Agents (Biomass-BDA) extension, modified from BDA v2.1 (Sturtevant et al. 2004), which simulates outbreaks of pests and pathogens as a spatially contagious process dependent upon climate and host availability. We simulate fir engraver (Scolytus ventralis), Jeffrey pine beetle (Dendroctonus jeffreyi), mountain pine beetle (D. ponderosae), and western pine beetle (D. brevicomis), as well as white pine blister rust (Cronartium ribicola). We calibrate outbreak patterns using USFS Aerial Detection Survey and Ecosystem Disturbance and Recovery Tracker data (Koltunov et al. 2020). The extent and severity of outbreaks is an outcome of climate, host tree density, and spatial patterning, allowing for complex interactions among climate, vegetation, management, and hydrology (Scheller et al. 2018).

2.2.2 Forest Management Scenarios

We utilize several previously developed scenarios (Maxwell et al. 2022) to represent a range of management activities ranging from very little management to approximately full restoration of a natural disturbance return interval. The scenarios' overall objectives are to restore forest ecosystems to a state that is more similar to their character prior to fire exclusion. The scenarios attempt to restore a low- or mixed-severity fire regime by reintroducing disturbances in the form of prescribed fire or thinning from below. The proportion of the landscape treated per year depends upon the historical fire-return interval of the management zone, but ranges from ~1% to ~6% per year across the whole landscape. We implement harvests using the Biomass-Rank Biomass Harvest extension, a modification of Biomass Harvest that allows greater flexibility in selecting locations to harvest based on their biomass. Management zones are developed using land ownership and land use, slope steepness, and historical fire return interval data from LANDFIRE (U.S. Department of the Interior 2016). Within all scenarios, private lands are managed as business-as-usual, with pre-commercial thinning and clearcuts on private timberlands. The wildland-urban interface (WUI)

Defense zone (within 400 m of settlements) is also treated for fuel reduction in all scenarios. The scenarios are described in detail by Maxwell et al. (2022) and summarized here. Note that the scenario names given here are selected for interpretability with reference to our hydrological results, and the concept of "business-as-usual" is not prescriptive but rather reflects a baseline level of management in the simulations.

1. Reduced treatment. Management is restricted to fuel treatments within the WUI Defense zone (within 400 m of settlements) and private lands. This scenario represents a substantial reduction in general forest treatment compared to present-day management.

2. Business-as-usual (BAU). This scenario is designed to closely match management practices in the present and recent past, including private land management and management of general forests as recorded in USFS and CalFire databases.

3. Partial restoration with less fire. In this scenario, treatments are extended to general forest and roadless areas. Almost all treatments are either mechanical thinning or hand thinning, depending upon the slope steepness and land use category. Prescribed fire is used for 5% of treatments on general forest land and 20% of treatments in roadless areas.

4. Partial restoration. This scenario is similar to Scenario 3, but it replaces 20% of the thinning treatments in the WUI Threat zone with prescribed fire.

5. Full restoration with less fire. This scenario and Scenario 6 attempt to replicate the historical disturbance return interval (~6% of the landscape treated per year). The types of treatment and kinds of stands treated are identical to Scenario 4, but the area treated per year is greater.

6. Full restoration. Compared to Scenario 5, this scenario increases the amount of prescribed fire: 30% of treatments in general forest and roadless areas use fire rather than mechanical thinning treatments.

2.2.3 Forest Ecosystem and Hydrological Model Linkage

In order to assess the emergent effects of forest management and disturbance on water resources, we translate outputs from LANDIS-II into suitable inputs for DHSVM. LANDIS-II generates several outputs natively which can be directly used in DHSVM, including LAI (Figure 2C) and species composition (SI Figure S14), but other outputs require further processing. To create inputs for fractional canopy cover, we use regression models from FIA data. For details, refer to the appendix of Zeller et al. (2023). Because the LANDIS-II model is parameterized from forest inventory data, understory vegetation is underrepresented in the vegetation layers, which could bias the hydrological model. To impute understory vegetation cover, we use FIA data to create beta regression models predicting understory shrub cover as a function of tree biomass, tree age, canopy cover, and forest type. We classify sites as having dense understory if the regression model predicts understory shrub cover exceeding 20%. Sites without a shrub understory have an assumed light understory cover in DHSVHM, typical of grasses and forbs in the study area, as discussed in the hydrological modeling section.

2.3 Output Processing and Scenario Comparisons

Comparing simulated watershed behaviors between different restoration scenarios enables us to attribute variable hydrological dynamics to forest management actions. Differences in modeled maps of yearly pixel-wise peak SWE and snowmelt timeseries can quantify forest disturbance impacts on snowpack dynamics. We use simulated streamflow timeseries to estimate additional reservoir inflow volumes attributable to forest restoration, and we construct flow duration curves (Vogel and Fennessey 1995) to assess impacts on the high flow regime. Finally, aggregated timeseries of water mass balance fluxes enable us to calculate differences in the partitioning of landscape-average yearly overstory and understory transpiration, overstory and understory interception loss, and streamflow generation.

We analyze the effects of forest restoration on local streamflow generation and peak flows using sub-watershed daily streamflow timeseries. During each DHSVM run, we save streamflow records at 139 selected pour points approximately corresponding to the HUC-12 watersheds represented in the NHD watershed boundary dataset (U.S. Geological Survey 2019). We calculate yearly local streamflow generation for each sub-watershed by subtracting streamflow contributions from any upstream tributaries. Comparing streamflow generation in different management scenarios isolates the effect of forest restoration. To analyze peak flow effects, we compute the Sen's slope (Sen 1968) for one-day yearly peak flows in each sub-watershed across the full 85-year period. By differencing the peak flow trends for each sub-watershed under different management scenarios, we ascertain the effect of forest restoration on one-day yearly peak flows. This approach isolates the effects of each management scenario by removing climate-induced peak flow trends from the business-as-usual scenario.

To investigate peak flow connections to forest restoration, we isolate major storm runoff events and compare contemporaneous water balance fluxes between different management scenarios. Since we model the study region with four separate watershed domains, three DHSVM models, and two GCMs, we obtain a total of 24 watershed-aggregated timeseries. In each timeseries, we identify yearly peak flow dates across the 85-year simulation period. For peak flow events preceded by multiple days of continuous precipitation, we define a storm period extending from the first day with more than 1 mm/d of precipitation through the day of the yearly peak flow. For each such event, we calculate precipitation intensity as the mean daily precipitation rate during the storm period. For the 10 highest-intensity precipitation events prior to yearly peak flows in each combination of watershed, DHSVM model, and GCM, we calculate storm-total interception by subtracting the total overstory and understory interception storage (rain plus snow) on the first day of the continuous storm period from the total interception storage on the peak flow date. We similarly calculate cumulative interception vapor loss, snowpack outflow, and snow energy balance fluxes over the same storm periods.

3 Results

3.1 Hydrological Model Calibration and Validation

We select three Pareto-efficient DHSVM parameter sets from our multi-objective Bayesian calibration based on tradeoffs between simulation accuracy for the snowpack, baseflows, yearly water yield, and high flows. Model A achieves the highest area-weighted daily NSE across all calibration watersheds, Model B has the lowest RMSE for large-scale high flows, and Model C has the lowest yearly water yield RMSE subject to the requirements of daily NSE > 0.8 and high-flow daily RMSE < 80 m^3/s. The three selected models are Pareto-efficient for all six calibration objective functions and representative of the residual uncertainty in the calibration parameter space. Model A is notable for having relatively low porosity and a slower decrease in transmissivity with depth (deep layer average porosity of 0.40 compared to 0.53 in Model B and 0.54 in Model C). Model B is notable for having relatively deep soil (3.8 m average compared to 2.8 m in Model A and 2.4 m in Model C). Model C is notable for having relatively low transpiration rates (average minimum stomatal resistance across conifer classes of 260 s/m compared to 189 s/m in Model A and 187 s/m in Model B). All three selected models have relatively high effective hydraulic conductivities, near the 95th percentile of the POLARIS data (Chaney et al. 2019), with snow parameters converging reasonably close to those estimated by Sun et al. (2018) for the Sierra Nevada region.

The ensemble of calibrated DHSVM models reproduces key hydrological signatures during historic calibration and validation periods. During the calibration period (water years 2012-2017), the three selected models achieve a mean area-weighted daily NSE of 0.82 across all 10 gauged watersheds, with an area-weighted NSE of 0.75 indicating a moderate decrease in skill on the validation period (water years 2006-2011). The decrease in NSE during the validation period could be caused in part by lower year-to-year variability in streamflow, which reduces the total sum of squares (denominator of NSE). In the North Yuba watershed (USGS station 11413000), which is the largest gauged basin without upstream flow regulation in our study area (650 km^2), the calibrated models produce a mean daily NSE of 0.87 over the calibration period and 0.80 over the validation period (Figure 3). In log-transformed space, the calibrated models produce calibration and validation NSEs of 0.94 and 0.92 in the North Yuba, indicating that DHSVM is satisfactorily reproducing both low-flow and high-flow regimes at this gauge, including major peak flows associated with rain-on-snow events in the winter of 2017 (Figure 3). Daily NSEs are more variable in smaller watersheds (11-130 km^2), typically in the range of 0.6 to 0.8 for both calibration and validation periods with occasional larger errors associated with incorrect rain-snow partitioning at the ~10 km^2 scale in steep terrain (refer to Supporting Information for a full set of hydrographs). At full watershed scales, DHSVM shows low bias in water yield, with bulk runoff errors of -8% to +2% at the YRS and AMF full natural flow points during calibration and validation periods (mean error -3% across all models and periods at both measurement points). The calibrated models also satisfactorily reproduce variability in large-scale peak flows, pixel-wise maximum yearly SWE, and pixel-wise maximum yearly SWE timing (SI Figures S4-S13).

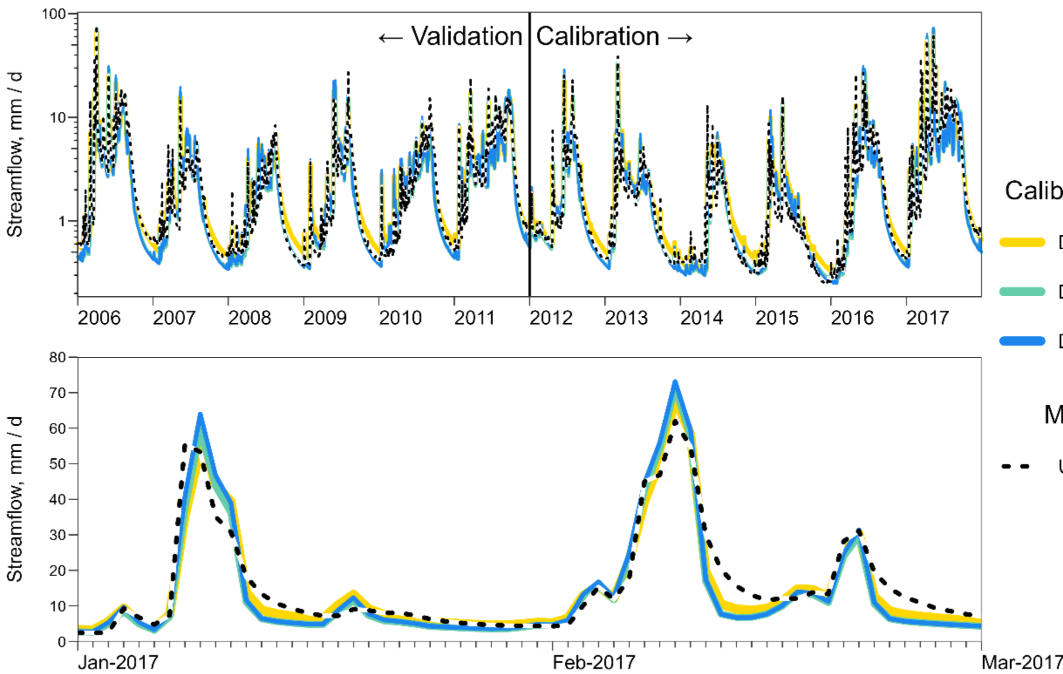

Figure 3. Example calibration and validation hydrographs for DHSVM. The North Yuba (USGS station 11413000) is the largest gauged watershed in the study area that has negligible upstream diversion or flow regulation. The model is calibrated on a total of 10 watersheds and validated on a total of 12 watersheds (refer to Supporting Information for additional calibration and validation hydrographs).

3.2 Forest Ecosystem Response to Disturbance

The LANDIS-II forest ecosystem model produces spatially explicit timeseries of vegetation characteristics across a spectrum of prescribed forest management scenarios. Conifer forests cover most of our central Sierra Nevada study region with the exception of high alpine regions and lakes, most notably Lake Tahoe. The historical baseline map from LANDIS-II, used for DHSVM calibration, has an area-averaged grid cell LAI of 2.3 m^2/m^2, with a median of 2.2 m^2/m^2 and a 90th percentile of 4.5 m^2/m^2. The densest forests are historically concentrated at mid-elevations on the west slope of the mountain range (Figure 2C).

Partial or full restoration of the historic forest disturbance return interval produces relatively thinner forests. Regardless of management scenario, LANDIS-II indicates a substantial decrease in forest density by the end of the century primarily from increased insect mortality (Maxwell et al. 2022). In the business-as-usual scenario (S2), LANDIS-II shows a mean LAI of 1.5 m^2/m^2 at the end of the century (median 1.3 m^2/m^2 and 90th percentile 3.4 m^2/m^2), which is a decrease of 33% relative to the pre-restoration mean (averaged across both climates). Comparatively, the reduced treatment scenario (S1) shows a 32% decrease in mean LAI, the partial restoration scenarios show a 36% decrease in mean LAI (both S3 and S4), and the full restoration scenarios show a 43% (S5, less fire) or 47% (S6, more fire) decrease in mean LAI over the same 85-year period. At the end of the century, mean LAI in the full restoration (S6) is 0.31 m^2/m^2 lower (-14%) compared to the business-as-usual scenario (S2). Scenario differences in post-restoration LAI are largest in the same mid-elevation west slope regions that have the densest initial forest cover (Figure 2C-D), up to a maximum sub-watershed difference of 1.2 m^2/m^2 (39%) between full restoration and business-as-usual scenarios. In all scenarios and both climates, LANDIS-II predicts significant species changes during the 85-year simulation period; most notably, white fir (A. concolor) cover substantially decreases and Douglas fir (P. menziesii) cover increases. Additionally, understory cover increases at higher elevations regardless of management scenario. Refer to SI Figure S14 for maps of initial and final vegetation types and understory LAI.

3.3 Simulated Effects on the Snowpack

In restoration scenarios with a relatively thinner forest canopy, DHSVM predicts relatively higher landscape-average snowpack accumulation. A pixel-wise average of the peak yearly SWE in all three models and all 85 years provides a spatial metric for snow accumulation during the simulation period (Figure 4). Compared to business-as-usual (S2), the mean pixel-wise peak SWE in the full restoration scenario (S6) is about 5% higher in the wetter CNRM-CM5 climate and about 6% higher in the drier MIROC5 climate. In both climates, the absolute magnitude of peak SWE differences between S6 and S2 varies between about -5 and +12 cm (1st and 99th percentiles). The spatial heterogeneity of SWE accumulation effects is only slightly lower (-4 to +13 cm) when only considering grid cells where LAI is lower in S6 than S2 to account for the effect of stochastic disturbance locations in LANDIS-II outputs. Thus, most of the modeled heterogeneity in the snowpack response to forest thinning (Figure 4) is attributable to meaningful differences in the model response to forest structure rather than merely spatial noise resulting from the stochasticity of the prescribed treatments. We observe that the percent change in pixel-wise peak SWE is relatively consistent throughout the simulation period, but the effect begins to attenuate during the 2070s-2090s as precipitation increasingly falls as rain in a warmer future climate (SI Figure S15). The pixel-wise snow ablation rate, calculated from the peak SWE date to the melt-out date, also increases by around 1-10% in most years, except late in the century when thinner forests melt out marginally slower (SI Figure S16).

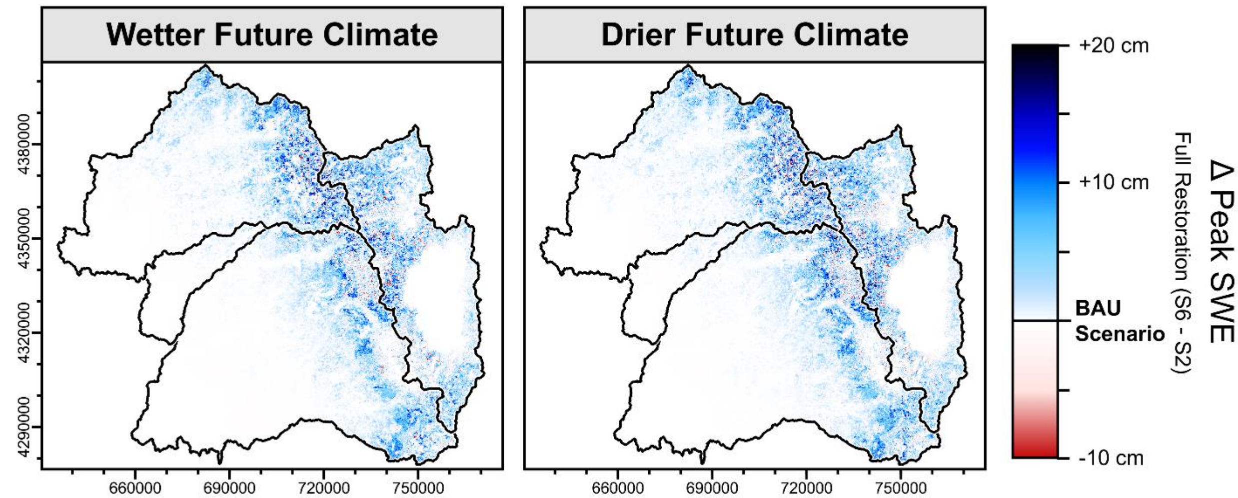

Figure 4. Difference in pixel-wise peak snow water equivalent (SWE) between the full restoration (S6) and business-as-usual (BAU, S2) forest management scenarios, calculated as an average over the full 85-year period under the CNRM-CM5 (wetter) and MIROC5 (drier) RCP 8.5 climate projections.

3.4 Simulated Effects on Streamflow Generation

DHSVM predicts an increase in streamflow generation from forested sub-watersheds in the central Sierra Nevada with an increased pace of forest restoration (Figure 5). On average over the 85-year simulation period across the whole study area and all three models, DHSVM predicts 4.3% more total streamflow generation in S6 relative to S2 under the wetter CNRM-CM5 climate, and 5.7% more total streamflow generation under the drier MIROC5 climate. However, the effect of forest restoration on local streamflow generation is heterogeneous in space and time. In certain sub-watersheds, the average 85-year streamflow change is as high as +27% in the drier climate and +22% in the wetter climate. Conversely, the effect is near zero in sub-watersheds with low initial forest cover. Decreases in streamflow generation (as low as -4%) are observed where forest cover is locally denser in S6 than in S2, a result of stochasticity in the spatial distribution of fires between different LANDIS-II runs. Averaging over only the driest 10 years in each climate projection, sub-watershed streamflow generation increases by up to +35% under the drier MIROC5 climate (median = +6%, 90th percentile = +18%) and up to +27% in the wetter CNRM-CM5 climate (median = +7%, 90th percentile = +18%). In summary, restoring a more frequent disturbance return interval to Sierra Nevada forests has the greatest relative effect on streamflow generation during dry years.

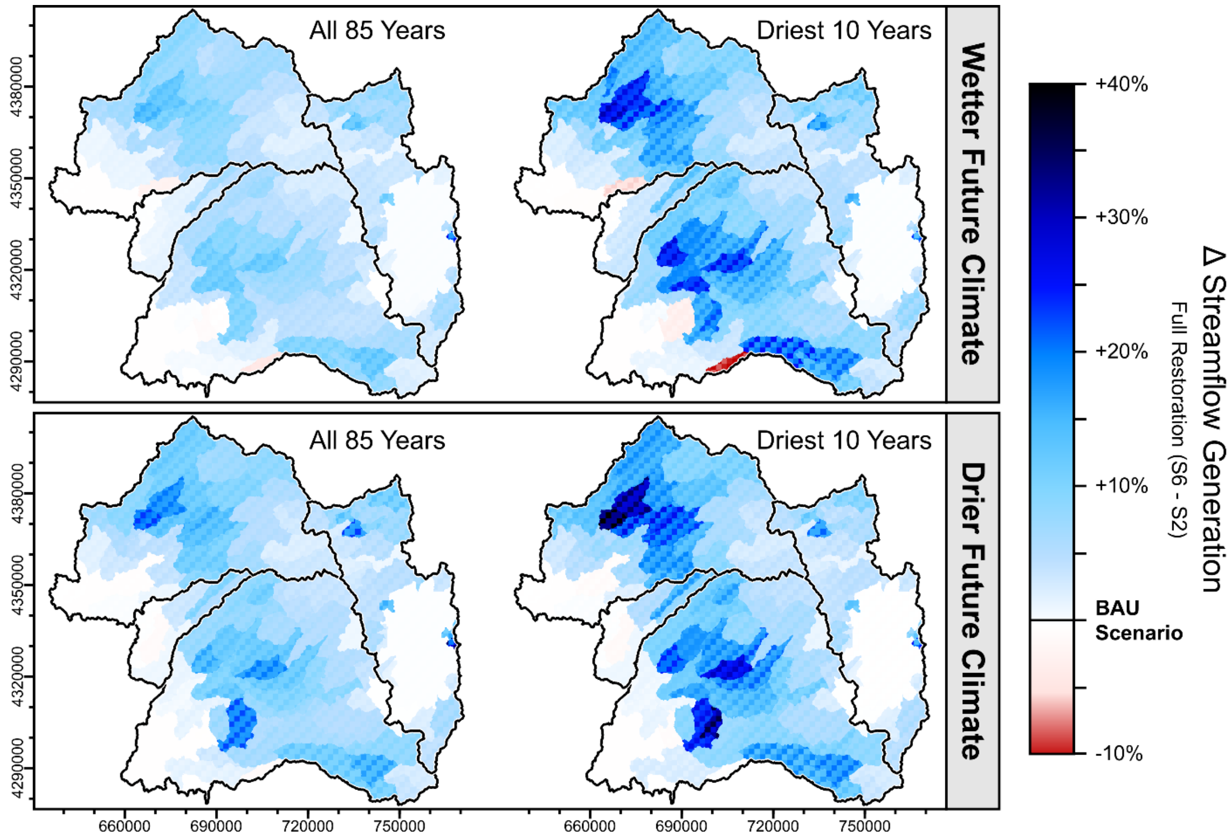

Figure 5. Percent difference in local streamflow generation for 139 sub-watersheds in the full-restoration (S6) forest management scenario relative to the business-as-usual (BAU, S2) scenario, calculated as an average over the full 85-year period and as an average over the driest 10 years in each climate. The checkerboard pattern is produced by interweaving results from three calibrated DHSVM models, thus representing uncertainty in the hydrological model.

Sub-watersheds show significantly different streamflow generation responses depending on the local climate and initial forest conditions. The absolute streamflow generation response to forest restoration (S6 – S2) in units of water depth (SI Figure S17) has a median of 45 mm/yr or 39 mm/yr, a 90th percentile of 93 mm/yr or 82 mm/yr, and a maximum of 151 mm/yr or 142 mm/yr in the wetter (CNRM-CM5) and drier (MIROC5) climates, respectively. Spatial heterogeneity in the absolute (not percentage) effect on streamflow generation correlates with the sub-watershed mean historical LAI (Pearson correlation $r = 0.69$), the LAI difference between scenarios at the end of the century ($r = 0.93$), and precipitation ($r = 0.64$), with no linear correlation to elevation ($r = 0.14$). Comparing Figures 2 and 5, we note that the largest percentage gains in streamflow occur in watersheds at low- to mid-elevations on the west slope of the Sierra Nevada where the forest is initially dense and precipitation is relatively low. In this relatively arid zone, pre-restoration streamflow generation is low, so small changes to the water balance can lead to large percentage streamflow gains. Considering absolute differences in area-normalized streamflow instead of percentage differences, forest restoration has the largest effect in sub-watersheds with a combination of relatively high precipitation and dense pre-restoration forests. Most of the landscape-scale increase in water yield is attributable to forest disturbance in relatively wet regions, but the largest relative impact on streamflow generation occurs in relatively dry regions.

3.5 Simulated Effects on Water Balance Partitioning

Precipitation inputs to DHSVM are identical in all management scenarios, so changes in other water balance terms must sum to zero. Calculating the difference in water balance fluxes between forest management scenarios reveals the impact of forest restoration on water balance partitioning. Yearly storage changes and soil evaporation only show negligible differences between management scenarios, so these terms are excluded from comparison here. The negligible change in simulated soil evaporation is partially a result of our assumption that light understory is present in all forested grid cells (Section 2.1).

Systematic increases in water yield from forest restoration are primarily attributable to decreased canopy interception loss, because increases in understory transpiration largely compensate for decreases in overstory transpiration (Figure 6). For the remainder of this section, analogous numeric values are given first for the wetter future climate (CNRM-CM5) and second for the drier future climate (MIROC5). Mean overstory transpiration is 42 or 40 mm/yr lower on average across all years in the full restoration scenario relative to business-as-usual. However, mean annual understory transpiration is 31 or 29 mm/yr higher in the full restoration scenario, which compensates for 72% or 73% of the reduction in overstory transpiration. Mean interception loss from the canopy is 41 or 35 mm/yr lower in the full restoration scenario, while understory interception loss increases by only 9 or 7 mm/yr, a smaller compensation of 21% in both climates. Since increases in understory ET do not fully compensate for decreases in overstory ET, the full restoration scenario generates 45 or 40 mm/yr more streamflow on average relative to the business-as-usual scenario. About 27% or 28% of the increased streamflow generation in the full restoration scenario is attributable to decreased transpiration and about 73% or 72% of increased streamflow generation is attributable to decreased interception loss.

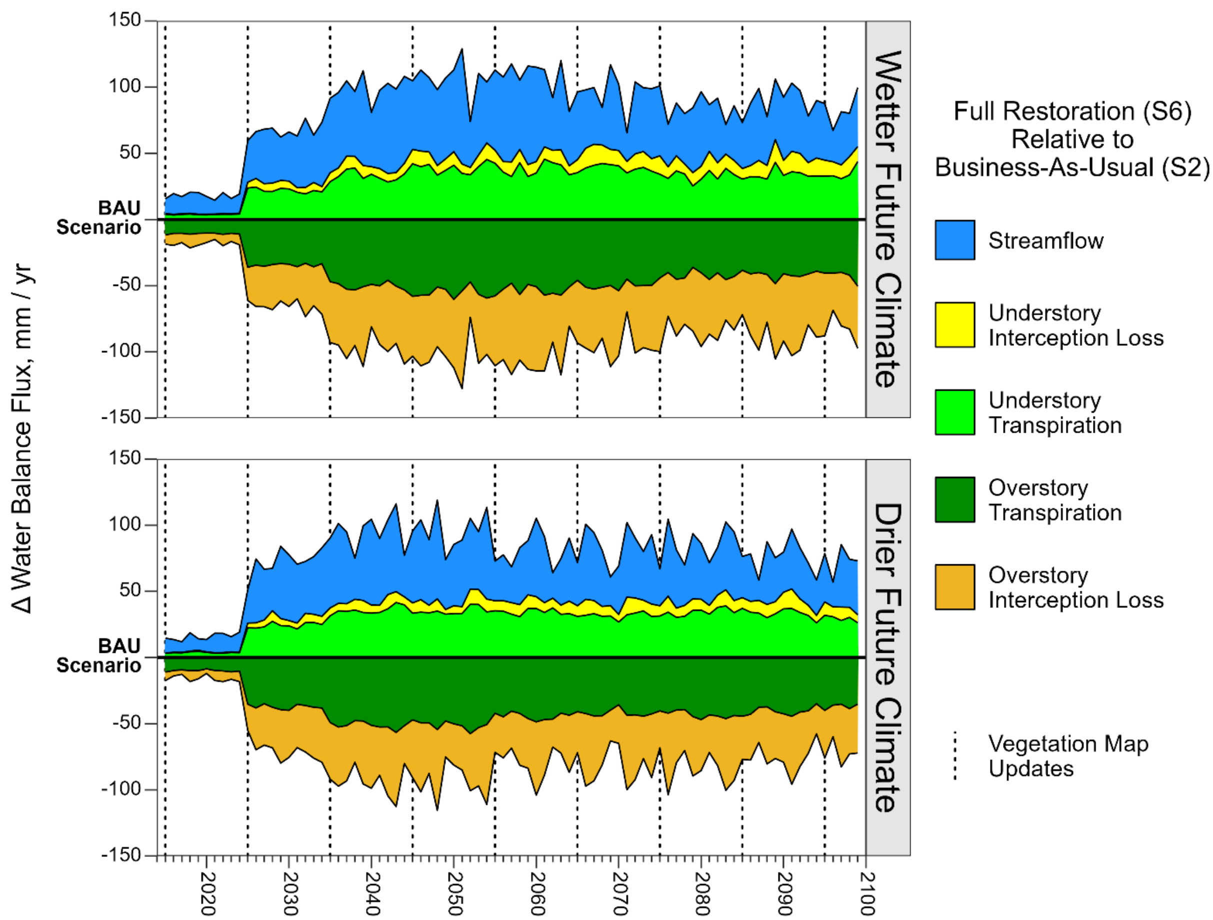

Figure 6. Timeseries of differences in yearly water balance fluxes between full restoration (S6) and business-as-usual (BAU, S2) forest management scenarios, averaged across the entire study area. All terms are calculated as the difference between forest management scenarios with equal precipitation, so all changes in the water balance fluxes visualized here approximately sum to zero each year. Fluxes that are larger component of the water balance in S6 relative to S2 are positive here, and vice versa. Dashed vertical lines indicate dates where vegetation maps in DHSVM are updated using outputs from LANDIS-II.

3.6 Simulated Effects on Peak Flows

DHSVM predicts a trend toward relatively higher one-day yearly peak flows in sub-watersheds that are subject to a more frequent forest disturbance regime. Both the CNRM-CM5 and MIROC5 climate projections cause trends towards higher peak flows in at least half of the 139 sub-watersheds in the project area, calculated as the Sen's slope of one-day yearly peak flows. The peak flow trend is stronger in the relatively wet CNRM-CM5 climate, with a median sub-watershed peak flow trend of +0.75% per year in the business-as-usual management scenario. The analogous median trend in the drier MIROC5 climate is +0.33% per year. In both climates, scenarios with a more frequent forest disturbance return interval can produce hydrographs with accelerated peak flow trends in certain sub-watersheds (SI Figure

S18). Relative to business-as-usual (S2), yearly one-day peak flows in the full restoration scenario (S6) are 3% or 6% higher on average across all sub-watersheds in the wetter and drier climates, respectively. There is a large degree of variation in the relative sub-watershed peak flow response (SI Figure S19), and the maximum difference between mean annual peak flows for particular sub-watersheds in these two scenarios is as high as 23% in the wetter climate or 39% in the drier climate. However, percentage-based metrics can overemphasize sub-watersheds with relatively low streamflow, so it is more informative to compare Sen's slope trends in area-normalized specific discharge units (mm/d/yr), as in Figure 7. Note that the streamflow generation effect is calculated after subtracting upstream watershed contributions, but the peak flow trend is calculated from raw hydrographs to represent the actual peak flows in a particular channel reach (including upstream contributions).

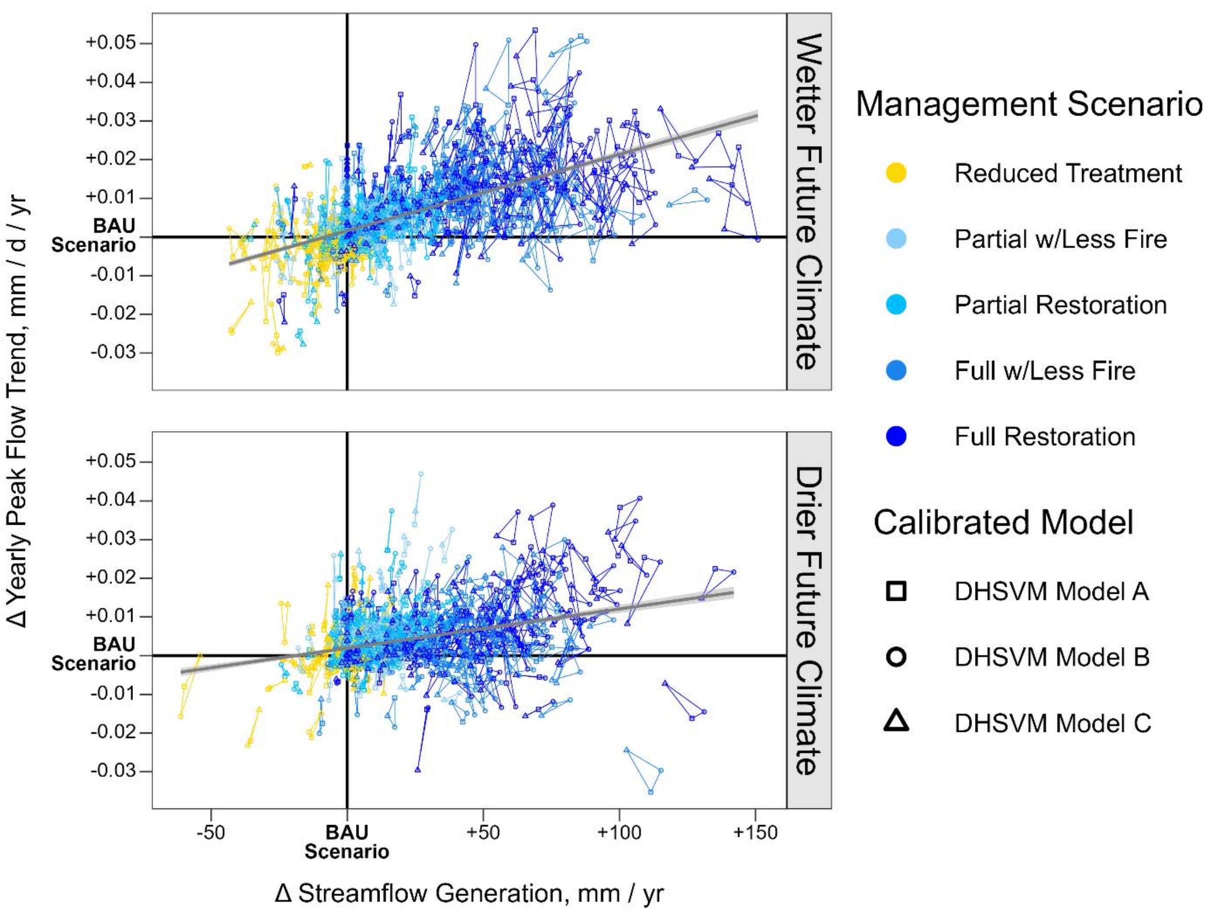

Figure 7. Relationship between increased streamflow generation and increased one-day yearly peak flows for sub-watersheds subject to different forest restoration scenarios. Differences in streamflow generation and peak flow trend are shown for the full-restoration scenario (S6) relative to the business-as-usual (BAU, S2) scenario. Each point represents a single sub-watershed simulated with a particular DHSVM model. The results from all three calibrated DHSVM models are connected with line segments, creating a triangular region that represents the uncertainty in the response of each sub-watershed.

We observe a tradeoff between increased streamflow generation and elevated peak flow trends in scenarios with more frequent forest disturbance (Figure 7). Considering all sub-watersheds across all management scenarios, the correlation between additional water yield and a higher peak flow trend is stronger in the wetter future climate (r = 0.60 in CNRM-CM5 and r = 0.33 in MIROC5). Relative to the business-as-usual scenario (S2) and averaged across all three models, both climates, and all 139 sub-watersheds (area-weighted), the reduced treatment scenario (S1) has 2 mm/yr less streamflow generation and no clear pattern of peak flow change. The scenarios with partial restoration of the disturbance return interval have increased streamflow generation of 8 mm/yr (S3, less fire) or 9 mm/yr (S4, more fire), with peak flow trends higher by 0.0036 mm/d/yr (S3) or 0.0037 mm/d/yr (S4). Finally, full restoration of the disturbance return interval produces increased streamflow generation of 34 mm/yr (S5, less fire) or 42 mm/yr (S6, more fire), with peak flow trends higher by 0.0062 mm/d/yr (S5) or 0.0094 mm/d/yr (S6). Compared to the partial restoration scenarios (S3 and S4), the full restoration scenarios (S5 and S6) are about 4.5 times as efficacious at producing additional streamflow.

At watershed scales that are relevant for reservoir operations, the effect of forest restoration on peak flows is overwhelmed by the uncertainty of future climate projections. There are two major artificial reservoirs in the project domain: New Bullards Bar (capacity 966,00 acre-ft. / 1.19 km^3) in the North Yuba River watershed, and Folsom Lake (capacity 976,000 acre-ft. / 1.20 km^3) at the outlet of the American River watershed. Comparing daily flow duration curves for both of these reservoirs derived from DHSVM shows that forest management only has potential to exert a negligible impact on the high flow regime compared to the uncertainty in future precipitation trends (Figure 8). We note that the reservoir-scale peak flow statistics presented here are calculated from raw modeled hydrographs, not accounting for upstream diversions or artificial storage, and are thus not suitable for comparison with historical flow records. Over all 85 years in the simulation period, the mean yearly one-day peak flow is 80% higher for New Bullards Bar and 94% higher for Folsom Lake in CNRM-CM5 compared to MIROC5. Comparing the same statistic between full restoration (S6) and business-as-usual (S2) scenarios, the mean yearly one-day peak flow increases by 4% or 7% for New Bullards Bar and 4% or 6% for Folsom Lake in the CNRM-CM5 and MIROC5 climates, respectively. Thus, uncertainty in the future climate is about 14 times or 20 times larger than the potential impact of forest thinning on annual peak flows into New Bullards Bar and Folsom Lake, respectively.

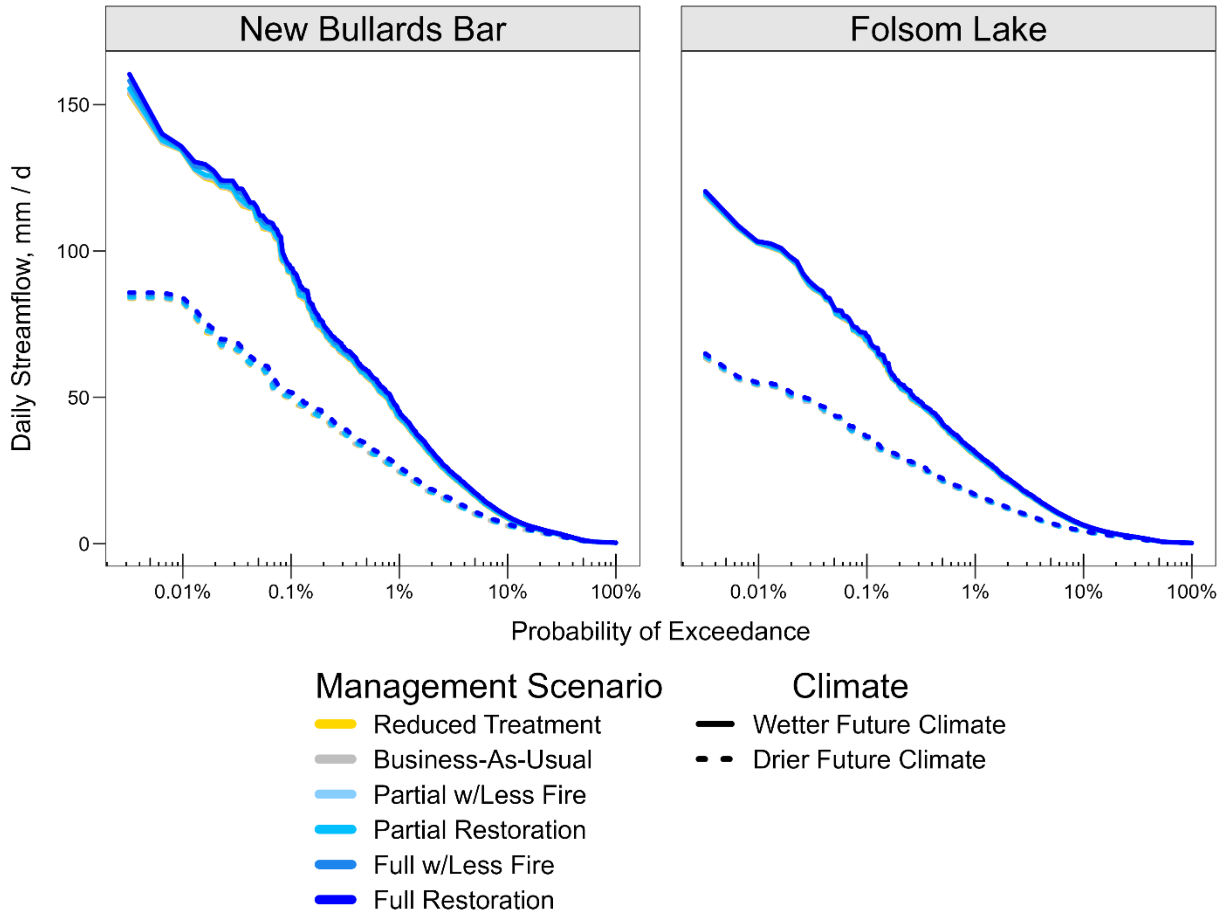

Figure 8. Flow duration curves for two major reservoirs in the project area, calculated using daily natural (unregulated) inflows simulated by DHSVM for the full 85-year period. Note the large difference in the high flow regime between climate models compared to the small difference in high flows between forest restoration scenarios. The probability of exceedance is logarithmically scaled to emphasize variations in the high flow regime.

3.7 Simulated Effects on Hydrological Processes During Major Storms

In scenarios with a thinner forest canopy, decreased interception and increased snowmelt may both contribute to increased runoff during major storms (Figure 9). As discussed in Section 2.3, we define the 10 highest-intensity storms immediately preceding yearly peak flow events in each of the four main watersheds for all three DHSVM models, thus identifying 120 storm runoff simulations in each scenario for each climate projection. In scenarios with a thinner forest canopy, interception is generally decreased while snowpack outflow is generally increased during these storm runoff periods, although some combinations of watershed and hydrological model produce outlying results for certain storms. In our simulations, forest restoration has a larger effect on cumulative storm-total interception vapor loss compared to net storm-total interception storage, because the same storage capacity may be filled and evaporated repeatedly during multi-day storms. Snowpack outflow from DHSVM includes both snowmelt and rain percolation, so some or all of the increased snowpack outflow in scenarios with a thinner canopy may be attributable to increased rain throughfall caused by reduced canopy interception.

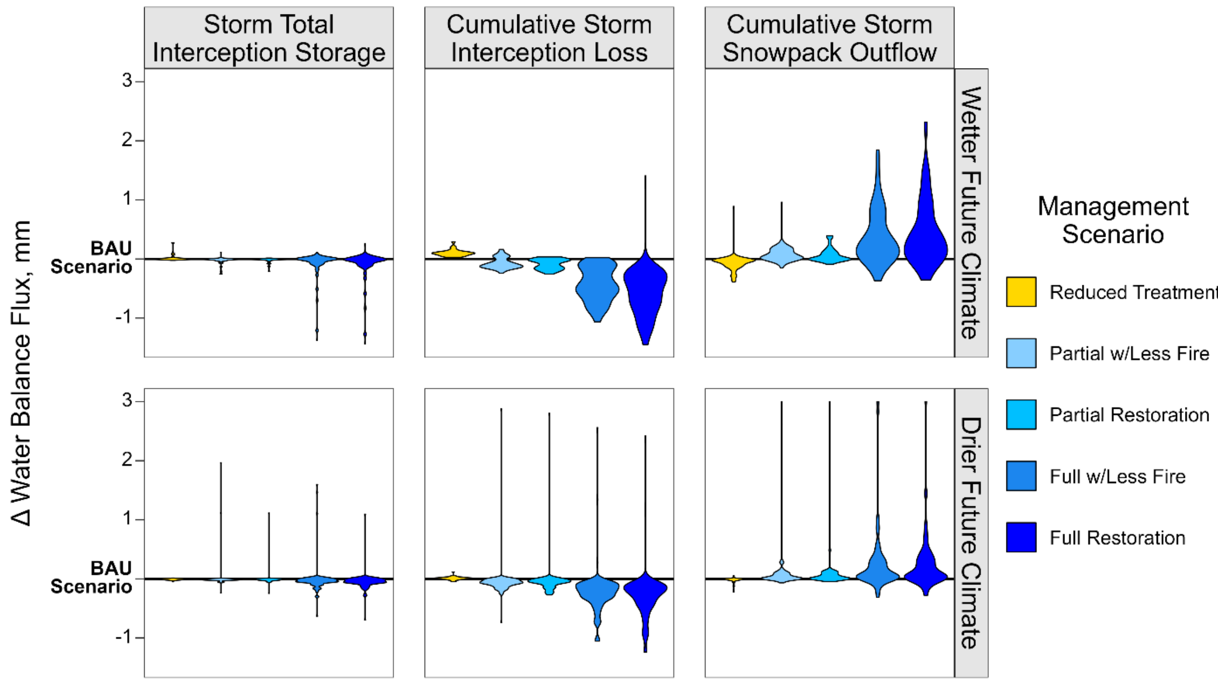

Figure 9. Forest restoration effect on storm-total interception storage, storm-total cumulative interception loss, and storm-total snowpack outflow during intense precipitation events prior to yearly peak flows, calculated as a landscape average for each of four major watersheds simulated by each of three DHSVM models. Results are shown for the full-restoration scenario (S6) relative to the business-as-usual (BAU, S2) scenario.

During storm events associated with yearly peak flows, the effect of a thinner forest canopy on the snowpack energy balance varies in space and time. In scenarios with a thinner forest canopy, the snowpack generally receives increased shortwave radiation, increased advected energy (from precipitation throughfall), and decreased longwave radiation (Figure 10). However, as a result of differing positive or negative signs between the shortwave, longwave, and advected energy effects, the net snowpack energy budget in a thinner forest may increase or decrease depending on the specific circumstances of each storm. Simulated sensible and latent heat fluxes do not show a clear pattern of change. We note that the small magnitude of energy balance changes in Figure 10 (on the order of 1 W/m^2) are area-averaged values for the entire TCSI area, while snow may exist only in limited areas during these storm events, especially in later decades. Thus, Figure 10 quantifies relative patterns of change but does not represent the absolute magnitude of changes to the snowpack energy balance.

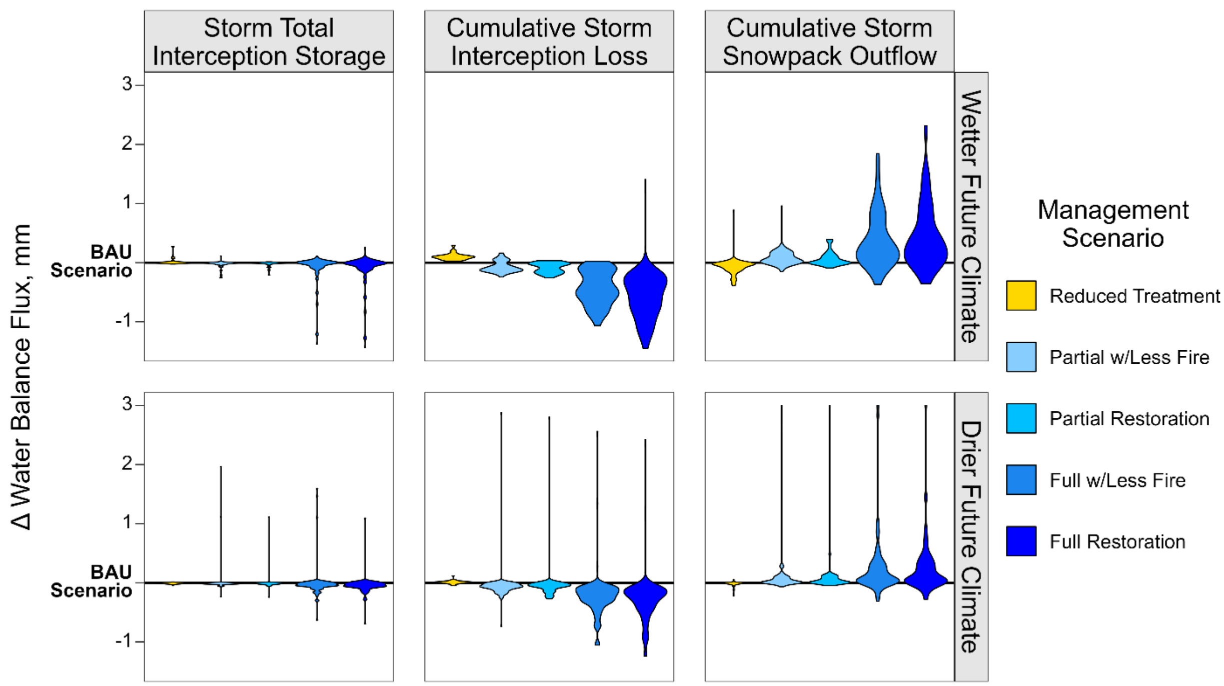

Figure 10. Forest restoration effects on the landscape-average snowpack energy balance during intense precipitation events prior to yearly peak flow events in each of four major watersheds simulated by each of three DHSVM models relative to the business-as-usual scenario.

3.8 Uncertainty in Sub-Watershed Streamflow Responses

The uncertainty of the calibrated hydrological model ensemble is considerably smaller than both the average magnitude and spatial heterogeneity of the predicted streamflow generation effects. In Figure 5, which shows the additional streamflow generation attributable to forest restoration, analogous results from all three selected DHSVM models are interwoven using a checkerboard pattern, where every third square is representative of the value from a single calibrated model. The resolution of the checkerboard is selected for ease of visualization, and the actual grid size of DHSVM is much smaller (90 m). Greater uniformity in the checkerboard pattern indicates proportionally lower uncertainty in the hydrological model ensemble. Overall, the uncertainty between different DHSVM ensemble members is about an order of magnitude smaller than the size of the predicted effects attributable to forest management.

The peak flow response to forest thinning is more uncertain than the streamflow generation response, but both responses show significant patterns of spatial heterogeneity. In Figure 6, hydrological model uncertainty is visualized with triangles defined by the results from all three calibrated DHSVM models in each sub-watershed. Although the true response of a given sub-watershed could fall outside the bounds of its modeled range (enclosed triangle), the overall spread of the three ensemble members gives an estimate of model uncertainty. In the full restoration (S6) scenario relative to business-as-usual (S2), the sub-watershed streamflow generation response has an average model uncertainty range of 5.3 mm/yr among the three

DHSVM models and an average difference of 7.9 mm/yr between both climate models. The mean sub-watershed streamflow response to forest restoration (S6 vs. S2) is 44 mm/yr with a standard deviation among sub-watersheds of 32 mm/yr. Analogously, the S6 vs. S2 peak flow trend difference in each sub-watershed varies by 0.0075 mm/d/yr on average among DHSVM models and 0.0090 mm/d/yr on average between climates, with a mean across sub-watersheds of 0.0097 mm/d/yr and a standard deviation among sub-watersheds of 0.0083 mm/d/yr. For most sub-watersheds, the magnitude of the predicted streamflow generation response considerably outweighs both the uncertainty of the model (~12%) and the mediating effect of climate (~18%), but the peak flow response has much higher model uncertainty (~78%) and much greater dependence on climate mediation (~93%). Nevertheless, we observe that the largest sub-watershed peak flow responses are relatively well-constrained based on the individual confidence regions in Figure 6. The maximum response for any one sub-watershed averaged across models and climates is 140 mm/yr of additional streamflow generation or a 0.037 mm/d/yr higher peak flow trend. Thus, locally strong sub-watershed responses are well-resolved relative to the model uncertainty.

3.9 Uncertainty in Reservoir-Scale Water Yield

Despite uncertainty in the future climate, the absolute magnitude of additional water yield into reservoirs under different forest restoration scenarios is well-constrained. Here, we consider mean yearly runoff into each reservoir over the full 85-year simulation period in the full restoration scenario (S6) relative to the business-as-usual scenario (S2). For the New Bullards Bar watershed, DHSVM predicts an additional 99,800 acre-ft/yr (0.123 km^3) of yearly runoff in the wetter CNRM-CM5 climate or an additional 90,700 acre-ft/yr (0.112 km^3) of yearly runoff in the drier MIROC5 climate. For Folsom Lake, the additional inflow is 176,000 acre-ft/yr (0.217 km^3) under CNRM-CM5 or 158,000 acre-ft/yr (0.195 km^3) under MIROC5. The total mean annual runoff between wetter and drier climate projections varies by 46% for New Bullards Bar and 55% for Folsom Lake, but the volume of additional runoff attributable to forest restoration varies by only 10% or 12% between climates for the same reservoirs. Thus, the additional water yield from forest restoration is about five times less sensitive to uncertainty in future precipitation trends compared to the total future water yield (46% vs. 10% uncertainty for New Bullards Bar and 55% vs. 12% uncertainty for Folsom Lake).

The relative contribution of additional runoff from forest restoration as a percentage of the total yearly volume is dependent on future climate trends and interannual variability. Relative to the business-as-usual scenario, additional water yield from the full restoration scenario amounts to 7% or 9% of the mean annual inflow for New Bullards Bar and 4% or 6% for Folsom Lake in the wetter or drier climate projections, respectively. Considering only the 10 driest years (by annual precipitation), the absolute additional runoff in the full restoration scenario relative to business-as-usual is 71,900 acre-ft/yr (0.0887 km^3) or 52,300 acre-ft/yr (0.0645 km^3) for New Bullards Bar and 113,000 acre-ft/yr (0.139 km^3) or 78,400 acre-ft/yr (0.0967 km^3) for Folsom Lake in the wetter and drier climate projections, respectively. This additional runoff amounts to 12% or 14% of the total annual inflow for New Bullards Bar and 8% or 9% of the total annual inflow for Folsom Lake during the same 10 driest years, again in the wetter or drier climates. While the absolute volume of additional water yield attributable to forest restoration decreases during dry climate conditions, the percent difference between business-as-usual and full restoration scenarios increases. Therefore, forest restoration would have the largest relative impact on reservoir-scale water yield during drought years, particularly in a drier climate.

4 Discussion

4.1 Reservoir-Scale Drought Hedge

Our results suggest that increased water yields from landscape-scale forest restoration can provide a hedge against future droughts in the central Sierra Nevada region, on the order of 8-14% increases in reservoir water yield during dry years. Compared to the annual water yield, streamflow gains from forest restoration are about five times less sensitive to uncertainty in the annual precipitation volume. This reduced sensitivity of streamflow gains follows from the partial decoupling of precipitation and forest ET in the relatively energy-limited study region, as previously shown by Saksa et al. (2017) for the same geographic area. This result is hydrologically intuitive because the impact of forest thinning is bounded by the maximum interception loss and transpiration rates supported by the initial forest. For example, trees have a minimum stomatal resistance beyond which additional soil moisture does not increase transpiration rates. Similarly, once interception storage becomes saturated during a storm, additional precipitation is less affected by canopy structure. From a management perspective, the environmental and economic value of additional streamflow generated from a thinner forest is likely to be contingent upon trends in future precipitation (see Guo et al. 2023 for a discussion of the marginal price of water in the study region). Even in a wetter future climate, certain years will likely still qualify as droughts, and the partial decoupling of additional runoff from precipitation could benefit the water supply in those dry years. Thus, the value proposition of forest restoration for water resources in the central Sierra Nevada region may best be understood as a hedge against a possible drier future climate and/or drought years in any future climate.

The magnitude of additional water yield from forest restoration as predicted by DHSVM is generally supported by prior findings in similar environments. In a small sub-basin of the American River watershed, included in our study area, Saksa et al. (2017) used a combination of process-based modeling and field data to estimate a 14% increase in streamflow generation after forest thinning. While a direct comparison is not possible due to differences in scale and treatment intensity, the empirical findings of Saksa et al. (2017) align with our model results in the same geographic region (Figure 5). Using statistical approaches based on relationships between measured ET and maps of the normalized difference vegetation index (NDVI) from remote sensing, Roche et al. (2020) estimated a 150-200 mm/yr potential streamflow gain from forest fires or thinning in the American and Yuba River watersheds. This streamflow gain is larger than predicted by our simulation for almost all sub-watersheds (Figure 7). While individual fires might cause extreme changes in the local water balance, our results indicate that it would be overly optimistic to expect such a large streamflow response from forest restoration at the landscape or even sub-watershed scale. Similarly, Guo et al. (2023) estimate reductions in ET as high as 361-371 mm/yr for high-severity wildfire or 269-277 mm/yr for medium-severity wildfire for small forest treatment areas (15.6-23.1 km^2) at the headwaters of the Yuba and American watersheds. Our results similarly suggest that these high ET reductions are unlikely to be realized from forest restoration at larger landscape scales, and we estimate a mean landscape ET reduction (overstory and understory transpiration plus interception loss) of only 41 mm/yr in the full restoration scenario. Nevertheless, extrapolation of empirical NDVI-based methods can produce similar water yield results at the landscape scale, provided that enough recent fires have occurred in the region of interest to constrain variable vegetation responses to disturbance: in the American River watershed, Roche et al. (2018) estimate a 5% (all years) or 10% (dry years) increase in reservoir inflow under a restored disturbance regime, remarkably close to the increase of 4-6% (all years) and 8-9% (10 driest years) modeled for the same basin in this study.

4.2 Sub-Watershed Heterogeneity

The additional water yield from forest restoration could be maximized by targeting treatments to sub-watersheds with particularly dense forests and high average precipitation. Forest management planning could prioritize mechanical thinning and prescribed fire in sub-watersheds with the greatest potential for increased streamflow generation (Figure 5). Additionally, a machine learning model such as Random Forest could potentially be trained on the process-based DHSVM results in a meta-model decision support framework (Mijic et al. 2024), such as the tool implemented by Lewis et al. (2023) for the snowpack response to forest restoration in the same geographical region. Other studies have demonstrated the ranges of forest density and canopy gap size that can best promote snow accumulation (Piske et al., in press), but these fine scales are not explored in the present study. With knowledge of the potential tradeoff between increased streamflow generation and higher peak flow trends (Figure 7), it may be desirable to target forest restoration in sub-watersheds that balance the utility of additional water yield against the risk of damage to small-scale infrastructure from high flows. In the TCSI region, the elevation head and capacity of downstream hydroelectric plants further mediates the potential benefit of additional streamflow generation (Guo et al. 2023). From a water resources perspective, the ideal areas for forest restoration could be dense forests that are situated above power-generating reservoirs, since these reservoirs can benefit from increased inflow and mediate increases in peak flows.

Although forest restoration may cause locally higher peak flows, any significant impacts are limited to headwaters and are unlikely to affect reservoir operations. DHSVM predicts peak flow increases on the order of 10-30% in certain sub-watersheds, but these are small headwaters catchments with relatively low absolute streamflow magnitudes, on the order of 0.1 to 10 m3/s (SI Figure S20). Increases in small-scale peak flows could accentuate risks to hydraulic infrastructure such as road culverts, many of which are only engineered for 25-year peak flows and are now aging and vulnerable to washouts (Halofsky et al. 2021). Additionally, elevated surface runoff in headwaters regions could exacerbate erosion and sediment loading from historic hydraulic mining sites, which are prevalent in the study area (Gilbert 1917 and

Curtis et al. 2005). The effect of forest restoration on peak flows attenuates rapidly at larger scales, and reservoir operations are unlikely to be significantly impacted by landscape-scale forest restoration in the central Sierra Nevada region. One key uncertainty in our results is the lack of changes in the DHSVM soil and snow properties that could contribute to faster runoff or faster snowmelt, such as increased soil hydrophobicity (Certini 2005) or darkened snow albedo from pyrogenic carbon (e.g., Gleason et al. 2013). Because uncertain future precipitation trends almost exclusively control the high flow regime, flood management planning will likely become increasingly motivated by the potential for extreme precipitation and rapid snowmelt events (Harpold and Kohler 2017, Hou et al. 2019). For example, a project is currently planned to add a second spillway to the New Bullards Bar Reservoir to reduce flood risk associated with atmospheric river storms (Yuba Water Agency 2023).

4.3 Overstory-Understory Compensation

A process-based modeling approach is necessary to untangle the compensating responses of overstory and understory vegetation to forest restoration. Reduced canopy interception can lead to greater terrestrial water input (rain + snowmelt), and lower overstory transpiration can lead to higher soil moisture. Not all excess terrestrial water input necessarily becomes streamflow though, as noted empirically after widespread insect-driven forest mortality (e.g., Biederman et al. 2015). Wetter soil and reduced canopy shading can both contribute to increases in ET from remaining vegetation and soil evaporation (Boisramé et al. 2019). Moreover, elevated soil moisture may encourage regrowth of both understory and trees, so continued treatment by mechanical thinning or fire is necessary to maintain a thinner forest state, as implemented in the scenarios tested here (Maxwell et al. 2022).

We observe a strong compensation between overstory and understory transpiration in addition to a weaker tradeoff between overstory and understory interception loss (Figure 6). Neglecting increases in understory ET during forest restoration would lead to about an 86% overestimation of the predicted streamflow response. About 73% of the streamflow response that we do predict is attributable to reductions in canopy interception loss. Understory interception has only a limited capacity for compensation due to lower height and the sheltering effect of remaining trees, both of which contribute to limitations on solar insolation and turbulent vapor transport. Additionally, DHSVM assumes that understory is buried when snow is present in a grid cell, so cold-season precipitation can only be intercepted by the overstory canopy. One limitation of our approach is that the understory maps in DHSVM are updated using a regression model coupled to the LANDIS-II outputs instead of by direct simulation of shrub and herb communities in LANDIS-II. Our results show the importance of improving simulations of multiple vegetation layers and understory regrowth to better constrain compensating ET effects following forest disturbance.

4.4 Uncertainty

A model ensemble derived from multi-objective Bayesian calibration enables the propagation of model parameter uncertainty into our analysis of hydrological responses to forest restoration. The structure of fully distributed, process-based models like DHSVM is well-suited for extrapolating hydrological interactions to spatially heterogeneous vegetation and climate conditions outside the range of historical observations. However, process-based hydrological models are typically deterministic and challenging to fully constrain with observational data, making uncertainty quantification difficult and threatening the integrity of predictions (e.g., Beven 1993). By applying a multi-objective Bayesian calibration and selecting an ensemble of Pareto-efficient parameter sets, we are able to estimate model uncertainty and propagate it into our final results (Figures 5 and 7). Despite considerable uncertainty remaining in landscape-scale subsurface parameters (e.g., soil depth and porosity), differences in ensemble model predictions at the sub-watershed scale are roughly an order of magnitude smaller than the size of streamflow generation effects attributable to forest restoration.

Not all forms of uncertainty can be explicitly represented in our model calibration, and we choose to focus on key processes and parameters identified as sensitive in previous DHSVM studies (e.g., Du et al. 2014). Although our conclusions are robust across two GCMs that are endmembers of fire weather (Maxwell et al. 2022), both GCMs are downscaled using the same MACA technique (Abatzoglou and Brown 2012), which may lead to underestimation of future climate uncertainty (Alder and Hostetler 2018). The MACA downscaling technique, which is based on historical analogues, also may not fully capture the dynamics of future atmospheric river storms (e.g., Gershunov et al. 2019, Huang et al. 2020). There is also the possibility of unforeseen “black swan” events like invasive insects or megafires that could drastically alter the trajectory of forest ecosystems outside the LANDIS-II simulation scope. Despite these modeling limitations, the consistency of our water balance results across multiple calibrated parameter sets and future climate projections should increase confidence in the potential water resource impacts of central Sierra Nevada forest restoration scenarios.

5 Conclusions

Landscape-scale forest restoration shows promise as a hedge against future droughts. In the central Sierra Nevada mountains, distributed hydrological modeling predicts that full restoration of the historic disturbance return interval could produce 8-14% more inflow into major reservoirs during dry years. Increased streamflow can benefit aquatic and riparian ecosystems, hydropower operations, and municipal or agricultural water customers. In the context of recent Sierra Nevada multi-year droughts (e.g., 2013-2015 and 2020-2022), these benefits may help incentivize investment in central Sierra Nevada forest restoration. Despite considerable climate-driven uncertainty in the total future water supply (46-55%), the effect of forest restoration on water yield is relatively well-constrained. In the relatively energy-limited central Sierra Nevada hydroclimate, streamflow gains from forest restoration are partially decoupled from yearly precipitation. Combined with the higher value of water in dry years, this reduced climate sensitivity enhances the value of forest restoration as a potential drought hedge. In a thinner forest, reduced canopy interception and increased snowpack outflow during major storms can increase peak flows in headwaters catchments, but this risk is effectively limited to the scale of smaller road culverts rather than reservoirs. Densely forested sub-watersheds immediately upstream of reservoirs appear most favorable for targeted forest restoration due to the tradeoff between increased water yield and higher peak flows.

Our study demonstrates the value of linking process-based ecosystem and hydrological models to predict the water resource impacts of landscape-scale forest restoration. However, extrapolating our results across the western U.S., or even across the Sierra Nevada mountains, is challenging due to the mediating role of aridity, forest type, land cover history, and other factors. Applying similar methods across a wider range of regional climates and forest conditions may help constrain the sensitivity of our results and help prioritize forest restoration priorities.

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Availability Statement

DHSVM and LANDIS-II are both open-source models, with ongoing development by the Pacific Northwest National Laboratory (https://github.com/pnnl/DHSVM-PNNL) and the LANDIS-II Foundation (https://github.com/LANDIS-II-Foundation), respectively. Code related to the hydrological modeling for this project is available at https://github.com/eli-mtnhydro/TCSI-ForestHydrology. Code related to the forest ecosystem modeling for this project is available at https://github.com/LANDIS-II-Foundation/Project-Tahoe-Central-Sierra-2019. The model inputs, outputs, processed results, and other data needed to recreate the results and figures in this study are archived at https://doi.org/10.5281/zenodo.13984265.

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