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Snag dynamics and surface fuel loads in the Sierra Nevada: Predicting the impact of the 2012–2016 drought

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ABSTRACT

Forest die-backs linked to extreme droughts are expected to increase as the climate dries and warms. An example is the 2012–2016 hotter drought in California that induced widespread tree mortality in the Sierra Nevada, California. The sudden increase in snags (i.e., standing dead trees) raised immediate concerns about their impact on wildfire hazard and longer-term questions about their effect on ecosystem structure and function. We quantified the likely progression of snag fall and fuel succession following the recent extensive mortality event in the southern Sierra Nevada mixed conifer forest. Our results used data from a long-term demography study to project trends in surface fuel loads at three study sites in Yosemite, Sequoia and Kings Canyon national parks. In the short term (2017–2021), fine woody debris and litter + duff significantly increased across all three sites (>145 % and >55 %, respectively); coarse woody debris increased significantly at one site (48.6 %); and total fuel loads increased significantly at two of the three sites (38 % and 69 %). Snag longevity increased with size, with the relationship varying by species. Yellow pine was a notable outlier: size played a small role in influencing its fall rates. Overall, species-specific snag fall rates in the southern Sierra Nevada were 20 % to 40 % slower than previously reported. By 2040, projected median cumulative inputs of biomass from future snag fall range from 49.4 Mg ha^{-1} to 136.1 Mg ha^{-1}across our three sites, which exceeds the amounts currently present (47.17–89.97 Mg ha^{-1}) and is well above estimates of historical coarse woody debris amounts in the Sierra Nevada (17.7 Mg ha $^{-1}$). These results provide a robust empirical basis to refine the snag fall algorithm in vegetation simulation models. Options to manage the impact of extreme number of snags and their large surface combustible biomass include salvage operations and prescribed burning, with both methods having operational, financial, and legal limitations that need to be considered.

1. Introduction

A warming and drying climate poses a global threat to the structure and function of forests (Brodribb et al., 2020). Tropical and temperate forests are particularly vulnerable to climate-induced declines in resilience (Forzieri et al., 2022). In a recent review, Hartmann et al. (2022) highlight the most compelling indicator of the forest health crisis – a proliferation of sudden and unexpected tree mortality events in tropical and temperate forests around the world. Moreover, these forest die-backs are linked to extreme climate events, in particular hotter droughts (Hammond et al., 2022). For example, between 2012 and 2016, California experienced one of the most severe droughts in its recorded history (Griffin & Anchukaitis, 2014; Robeson, 2015). What made this drought exceptional is that an extended period of low precipitation was accompanied by unusually high temperatures (Diffenbaugh et al., 2015). This hotter drought coincided with widespread tree mortality throughout the state (Axelson et al., 2019; Fettig et al., 2019).

The ecosystem consequences of accelerated tree death are profound but complex (Anderegg et al., 2013). For example, an immediate concern for the fire-prone forests of California is that pulsed increase in the number of snags (i.e., standing dead trees) will exacerbate wildfire risk due to their influence on wildfire behavior. Burning snags can

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produce embers that facilitate fire spread (Barrows, 1951) while decayed snags and coarse woody debris can provide a receptive surface for spot fires (Knapp, 2015; Stephens, 2004). Furthermore, the combustion of coarse woody material can result in long burnout times that release high amounts of energy and heat to the soil, increasing the severity of fire effects (Brown et al., 2003; Monsanto & Agee, 2008). The 2020 Creek Fire in the southern Sierra Nevada of California provides an example of how drought-induced additions to dead biomass can contribute to "mass fire" behavior (Stephens et al., 2022). By 2016, the combination of drought stress and bark beetle attacks increased the proportion of snags in this forest to 37 % of the total standing tree basal area. Notably, high severity fire occurred on 41 % of the Creek Fire footprint, despite moderate fire weather. The best predictor of the timing and location of these high severity patches was the amount of dead biomass. The dynamics of the Creek Fire are an example of how climate change can have a pervasive impact on forest disturbance agents (Seidl et al., 2017). The hotter drought in California not only imposed water stress on trees (direct effect) but also facilitated attacks by native bark beetles (indirect effect, Robbins et al., 2022). Moreover, a large increase in surface fuels has the potential to intensify the impact of wildfire (interactive effect).

Beyond fire behavior, snags also play important roles in forest ecosystems. They provide critical habitat for cavity-nesting birds and den sites for mammals, and the decaying wood supports many invertebrate and fungal species (Harmon et al., 1986; Raphael & White, 1984; Saab et al., 2014). The decomposition of snags and the coarse woody debris created when snags fall are key processes in the cycling of nutrients and carbon within ecosystems (Cousins et al., 2015; Harmon et al., 1986).

Despite the importance of snags, the dynamics of their persistence are not well-characterized (Oberle et al., 2018), even in the very well-studied forests of the Sierra Nevada. Raphael and Morrison (1987) reported that snag longevity varies based on species and that larger trees tend to fall at slower rates. Grayson et al. (2019) found similar patterns for snags killed in wildfires. But neither Raphael and Morrison (1987) nor a follow-up study by Morrison and Raphael (1993) provided species and size specific equations to predict fall rates. Although Grayson et al. (2019) did include predictive equations, given the expectation that the longevity of fire-killed snags is different than other causes of mortality (Raphael & Morrison, 1987; Gärtner et al., 2023), these equations must be generalized with caution.

While it is reasonable to assume that as snags decay and degrade they will contribute to surface fuel loads, the timing of fall rates and magnitude of fuel inputs are rarely quantified. Most managers in the United States rely on the Fire and Fuels Extension (FFE) to the Forest Vegetation Simulator (FVS) to inform their decisions. FFE includes a snag submodel to predict the impact of snags on fuel loads (Rebain, 2010). FFE has proven to be a vital addition to the widely used FVS (Rebain, 2010). However, key elements of FFE were built on the best-available, but sparse, information. Specifically for the western Sierra Nevada forests (WS variant), the trajectory of the FFE snag fall model was based on unpublished data and the timing for different species was set by expert knowledge (Rebain, 2010). And yet, an accurate snag fall model is important to predicting fuel load dynamics (Kennedy et al., 2021). Given that hotter droughts and associated forest die-backs are emerging characteristics of the Anthropocene (Allen et al., 2015) and that the consequences of these droughts interact with wildfire hazard (sensu Seidl et al., 2017), it is essential to understand the potential timing and magnitude of fuel inputs from snags.

The goal of this research was to improve our understanding of snag dynamics in drought-prone forests like the western Sierra Nevada. We took advantage of a rare, multidecadal effort to track the fate of trees on an annual basis to develop robust, predictive equations of treefall rates. This extensive dataset allowed us to test alternative trajectories of snag fall timing. Moreover, we evaluated the practical implications of our results by comparing our predictions with results from the prevailing management standard. Given the threat posed by high severity wildfire in the Sierra Nevada, we focused on the consequences of snag fall on surface fuel loading. Moreover, we placed our results in the larger regional context to help build the empirical foundation needed to construct the next generation of applied forest simulation models. To accomplish these objectives, we combined recent surface fuel measurement with data from a long-term tree demography study to quantify the consequence of drought-induced mortality in Yosemite, Sequoia, and Kings Canyon national parks. We answered the following questions: 1) Did drought-induced mortality lead to near-term increases in fuel loads? 2) How do snag fall rates differ among species and size classes? 3) What is the timing and magnitude of snag inputs to surface fuel loads? 4) Do our site-specific, empirically based models offer management-relevant improvements over existing models?

2. Methods

2.1. Study sites

We used data from two complementary studies in Yosemite, Sequoia, and Kings Canyon national parks: the Drought Mortality Network and the Forest Demography Study (Fig. S1). The Drought Mortality Network installed tree and fuel inventory plots across the Sierra Nevada to monitor the consequences of the 2012–2016 drought mortality event (Axelson et al., 2019). A major objective of this monitoring effort was to quantify how recently dead trees influence fuel loads and fire hazard. The Forest Demography Study includes long-term forest plots where trees are censused annually for mortality and periodically for diameter growth (Das et al., 2016). To answer the questions posed, we integrated results from both datasets. We used the extensive, long-term snag database from the Forest Demography Study to develop high quality snag fall models. Results from the Drought Mortality Network allowed us to document the immediate and predict future impacts of drought-induced tree die-back.

2.1.1. Site description: Drought Mortality Network

For this study, we focused on the three Drought Mortality sites in the southern Sierra Nevada where drought-induced tree mortality was most extensive (Fettig et al., 2019; Preisler et al., 2017).

The site in Sequoia and Kings Canyon national parks (SEKI) is in the Crystal Cave area of Sequoia National Park in Tulare County, CA (Fig. S1A). The 1,700-ha study area is managed by the National Park Service. There is no history of timber harvests but some of the area has burned over the last few decades.[1] In 1969, 542 ha burned as part of a prescribed fire (Schuft, 1973); smaller fire impacts occurred in 1980 (121 ha burned) and 1992 (139 ha burned). Elevation ranges from 1,524 to 1,829 m (mean = 1,687 m). Average annual temperature is 7 °C (January: 0 °C; July: 17 °C) and average annual precipitation is 1143 mm (PRISM Climate Group, 2020). White fir (*Abies concolor,* Gord & Glend) and incense-cedar (*Calocedrus decurrens,* Torr) dominate the site (Table S1).

The Yosemite Mixed-Conifer (YOMI) site is situated near the Hodgdon Meadow Campground at the Big Oak Flat Entrance station in Tuolumne County, CA (Fig. S1C). The 260-ha study site is managed by the National Park Service. Elevation ranges from 1,323 to 1,535 m (mean = 1,422 m). Average annual temperature for the site is 11 °C (January: 4 °C; July: 20 °C) and average annual precipitation is 991 mm (PRISM Climate Group, 2020). The site is dominated by white fir, incense-cedar, and sugar pine (*Pinus lambertiana,* Dougl., Table S1). No recent timber harvests have been conducted at the site. However, prescribed fire was applied across the study area in 2011 and 2012, and back burning was

[1] Note: In September 2021 after the 2021 field inventories were completed, the entire Drought Mortality study area was burned in the KNP Complex wildfire. Based on the Rapid Assessment of Vegetation Condition after Wildfire, the KNP Fire burned 30% of the study area at high severity (RAVG 2023).

utilized within the study area during the Rim Fire in 2013.

The Yosemite Pine (YOPI) site is located along Wawona Rd. 11 km from the southern entrance to Yosemite National Park in Mariposa County, CA (Fig. S1D). The site spans 397 ha and is also managed by the National Park Service. Elevation ranges from 1,569 to 1,891 m (mean = 1722 m). Average annual temperature is 10 °C (January: 3 °C; July: 19 °C) and average annual precipitation is 1,042 mm (PRISM Climate Group, 2020). Ponderosa pine (*Pinus ponderosa* Dougl. Ex Laws), and incense-cedar dominate the site (Table S1). No recent timber harvests or fires have affected the study site.

2.1.2. Site Description: Forest Demography study

We used data from the Forest Demography Study located in old-growth mixed conifer and montane conifer forests in Sequoia, Kings Canyon, and Yosemite national parks (Das et al., 2016). These 23 plots ranged from 0.9 ha to 2.5 ha and were established from 1982 to 2001. The study areas had never been logged and experienced frequent low to moderate severity surface fires prior to Euro-American settlement (Caprio & Swetnam, 1993). We used demography plot data through 2021 (collected before the extensive KNP Complex wildfire burned many of the plots). Before 2021, four of the 23 demography plots had experienced relatively recent prescribed burns or wildfires. The demography plots encompass common conifer species in the southern Sierra Nevada, such as ponderosa pine, sugar pine, white fir, incense-cedar, Jeffrey pine (*Pinus jeffreyi,* Grev. and Balf), red fir (*Abies magnifica,* Pitcher), and black oak (*Quercus kelloggii,* Newb.). The elevations of plots range from 1,500 to 2,576 m. Two of the demography plots are near the SEKI and YOMI Drought Mortality study areas (Fig. S1).

2.2. Field measurements

2.2.1. Fuels

In 2017, a collection of 29, 35, and 37 circular 0.05-hectare fixed radius (12.62 m) plots were established at SEKI, YOMI, and YOPI, respectively. These plots were located randomly using a spatially balanced design (Stevens & Olsen, 2003). When establishing plots in 2017, crews monumented plot centers with a painted rebar stake for later remeasurements. In 2021, field crews located and re-measured all plots established in 2017. Crews recorded the diameter at breast height (height = 1.37 m, DBH), species, height, and status (live or dead) of all standing trees with a DBH equal to or greater than 10 cm in 2017. Crews also recorded the decay status of snags in 2017, which classifies snag decay on a scale of 1 to 5 based on the condition of the tree's top, branches, twigs, sapwood, and heartwood (FIA, 2021, Table S2). At each plot, fuels, defined as downed woody material consisting of twigs, branches, stems, and tree boles that are severed from the original source of growth, were recorded along three line transects with azimuths of 90, 210, and 330 degrees following the Brown's fuel protocol (Brown, 1974). Per the protocol, crews tallied 1-hour (<0.64 cm diameter) and 10-hour (0.64–2.54 cm) fuels from 3 to 5 m along each transect, and 100-hour (2.54–7.62 cm) fuels from 3 to 6 m along each transect. Depths of litter and duff were measured at 3 and 10 m along each transect. Litter was defined as undecomposed organic material that can be readily identified such as freshly fallen leaves, needles, bark flakes, cone scales, dead matted grass, and miscellaneous vegetative parts. Duff was defined as the layer between mineral soil and litter, which contains organic material decomposed to the point that there are no clearly identifiable whole materials. Crews measured the diameter and decay status (sound or rotten) of all 1000-hour (>7.62 cm) fuels along the entire length (12.62 m) of each transect. Fuels were classified as sound or rotten based on the condition of the heartwood, color of the wood, and ability of the material to support its own weight. Fuels were measured in both 2017 and 2021.

2.2.2. Snag fall

At the time of plot establishment, all live trees taller than 1.37 m in height in the Forest Demography plots (described in 2.1.2 above) were mapped, tagged, identified to species, and measured for DBH. Crews survey plots annually to document tree mortality, to note the status of snags, and to tag and measure new (ingrowth) trees (Das et al., 2016). Prior to 2013, crews included a comment in the notes when a snag fell. In 2013, researchers combed through the annual plot notes to document the year snags fell (Battles et al., 2015). Beginning in 2013, field crews explicitly recorded the year when snags fell. Snags were considered "fallen" if their main stem, or bole, broke at a height less than or equal to 1.37 m off the ground or if they were leaning at more than a 45-degree angle (FIA, 2021). We limited our analysis to snags greater than 12.7 cm in DBH.

2.3. Data analysis

2.3.1. Fuels

We used the RFuels package (Foster et al., 2018) to calculate fuel loads for each fuel category (1-hour, 10-hour, 100-hour, 1000-hour, litter, duff). Transect level estimates were averaged to the plot level, then we used a paired *t*-test to determine if there were significant differences in fuel loads for each site from 2017 to 2021. The distributions of fuel loads were visually inspected to ensure the assumption of normality for the paired t-tests was met for all fuel categories. Prior to testing for significant differences, we aggregated the fuel calculations into four fuels categories: total fuel loads, fine woody debris fuel loads (1, 10, 100 h), coarse woody debris fuel loads (1000-hour fuels), and litter + duff fuel loads.

2.3.2. Snag fall

To quantify snag fall rates, we applied survival analysis that uses the time of occurrence of an event to estimate the probability that the event has occurred by a specific time (Whitlock & Schluter, 2020). In our case, the event was snag fall, and we were interested in assessing how long snags stood until they fell. Thus, in this framework, "snag survival" means persistence as a standing dead tree.

Using the demography dataset, we were able to determine the time each snag was standing. Many of the snags in our dataset had not fallen by the time of the most recent sampling and were therefore treated as right-censored. Some of the demography plots had experienced fire in recent decades. Trees killed by fire often fall at different rates relative to snags created by other causes (Morrison & Raphael, 1993; Gärtner et al., 2023). Additionally, snags exposed to fire typically experience heightened fall rates compared to unburned snags (Landram et al., 2002). Since we wanted to understand the fall rates of unburned snags, we right-censored snags that were exposed to fire. Research at SEKI found that tree mortality following prescribed fire was comparable to the background rate after about five years (van Mantgem et al., 2011). Given this result, we excluded trees that died up to five years after fire to ensure that snags created afterward likely died from non-fire related causes and therefore would not bias our fall rate estimates. Given the small sample of Jeffrey pine snags (n = 44), and the functional and phylogenetic similarity between Jeffrey pine and ponderosa pine (Fulé et al., 2012; Haller, 1959), we combined snags of these species into the species category of yellow pine.

Given the potential for non-constant rates and strongly skewed distributions when considering survival times, we built parametric survival models with care. For example, in addition to testing the fit of the model against observations, we also calculated non-parametric (i.e., distribution free) estimates of snag fall rates to check for potential biases introduced by model selection. All analyses were conducted in the R statistical software (R Core Team, 2019); survival analysis used the "survival" package in R (Therneau, 2015).

We quantified the observed snag fall rate for each species using the Kaplan-Meier estimator, a non-parametric technique for estimating and visualizing survival probability over time (Kaplan & Meier, 1958). We also checked for differences in fall rates based on species using the log rank test, which tests the null hypothesis that survival curves for two or more groups are the same (Andersen & Keiding, 2014). We then used accelerated failure time (AFT) models to project snag survival time.

We applied an information theoretic approach to select the best AFT model (Burnham and Anderson 2002). We evaluated the fit of five potential underlying distributions that are commonly used when modeling snag fall (Parish et al., 2010; Grayson et al., 2019) rates: Weibull, exponential, log-normal, log-logistic, and gaussian distributions. Each distribution was fit with all possible (5) combinations of covariates: null (no covariates), species only, DBH only, additive effects of species and DBH, and interactive effects of species and DBH. The criteria for model selection were based on the Akaike Information Criterion (AIC). For this set of 25 potential models, we calculated the difference in AIC between every model and the model with the lowest AIC (ΔAIC) and the weight of evidence for each model .

We assessed the fit of the best model by comparing binned predicted survival probabilities to observed survival probabilities (Das et al., 2022). To do this, we predicted survival probabilities of each tree using our model, binned results in ten evenly divided classes (i.e., 0.1, 0.2, 0.3, up to 1.0), and then aggregated adjacent bins until there were at least 50 trees in each bin. We then compared the average predicted persistence probability within each bin to the average observed persistence probability within each bin. We also assessed model performance by calculating the concordance statistic, which is a measure of how well a model discriminates between dying and surviving objects (where a value of 1 indicates perfect discrimination and a value of 0.5 indicates discrimination that is no better than random). We summarized the effect size of species and DBH by generating predicted survival curves using the best fitting AFT.

2.3.3. *Snag fall rate and biomass*

Upon determining the best fitting model to describe snag fall rates, we used the 2017 inventory of snags at SEKI, YOMI, and YOPI to forecast the input of biomass to the surface from falling snags at each site. We used regional biomass equations (Zhou and Hemstrom, 2009) to estimate live tree biomass that was originally represented by each snag when it was alive. We then discounted bole biomass (stem + bark) in each snag using ratios based on decay class (Cousins et al., 2015). Branch biomass in snags was also reduced based on decay class with no degradation in recently dead trees (decay class 1), 0.9 for young snags (decay class 2); 0.4 for middle-aged snags (decay classes 3 and 4), and 0 for old snags (decay class 5). The AFT allowed us to predict the year each snag is expected to fall. Knowing the amount of biomass and expected fall date for each snag, we forecasted snag input to surface fuel loads at each site. To characterize the uncertainty associated with our model predictions, we generated forecasts using fall rates in 3 different percentiles, 25th, 50th (median), and 75th percentiles, which correspond to slower or faster overall fall rates.

2.3.4. *Snag fall rate comparison*

We compared predictions of the fall rate of snags in the 2017 inventory at SEKI, YOMI, and YOPI. For this comparison, only recently dead snags (decay class = 1) were included. From the best AFT model (i.e., locally informed model), we predicted time to fall in years from the 1st to 99th quantile and then calculated the probability of falling in 5, 10, 15, and 20 years for each snag. We ran 1,000 random simulations where each realization started with a random probability draw between 0 and 1. Every tree with a fall probability greater than the random draw for each time interval was counted as falling. Snag fall counts were reported as medians with interquartile ranges. A similar approach was taken to estimate fall rates from the FVS's FFE snag submodel (Rebain, 2010). We re-created the snag submodel logic (see Supplementary Materials for details) and simulated time to fall for trees based on a random probability draw. For each of 1,000 realizations, the number of trees falling in 5, 10, 15, and 20 years was calculated. Simulation results were summarized with the median and interquartile range.

To extend the empirical basis for managing snag fall for species common in the Sierra Nevada, we compiled fall rates from relevant studies conducted in similar montane forests. We extended the review in Grayson et al. (2019) by parsing results in the older publications, recreating the survivorship data, and then fitting the data to one of three distributions – linear, exponential, and Weibull. As above, the distribution that had the lowest AIC score was considered the best model. Parsing the results was necessary to refine the comparisons. For example, only recently dead trees were included in the analysis of results from Morrison and Raphael (1993) to match the snag cohort tracked in the Forest Demography Study.

3. Results

3.1. *Surface fuel load changes*

Total fuel loads increased significantly from 2017 to 2021 at SEKI (+55.4 Mg ha^{-1}; 38 % increase) and YOPI (+90.7 Mg ha^{-1}; 69 %), but there was no significant change at YOMI (Fig. 1, Table S3). Litter and duff loads increased significantly at all sites with increases of 48.6 (55 %), 58.4 (68 %), and 50.5 Mg ha^{-1} (101.5 %) for SEKI, YOPI, and YOMI, respectively ($p < 0.05$). Fine woody debris fuel loads also increased significantly at all 3 sites by more than 145 %. In contrast, the only significant change in coarse woody debris occurred at YOPI ($p = 0.032$) where the load increased by 18.96 Mg ha^{-1} (48.6 %).

3.2. *Snag fall rates*

The Forest Demography Study recorded 3,059 snags and 1,187 snag falls. Of the remaining 1,872 snags, 372 were right-censored due to fire and 1,500 did not fall during the observation period (Table 1). DBH of snags ranged from 12.7 to 251 cm. Diameter distributions for each species followed the general reverse J-shaped curve (Fig. S2).

Median survival times (Table 1) (Kaplan-Meier estimator), and the results from the log-rank test using species as groups ($\chi^2 = 47.8, df = 5$, $p < 0.01$) indicate that snag fall rates vary by species. Yellow pine snags had the shortest survival time with a median of 9 years while incense-cedar had the longest survival time with a median of 20 years. For all snags in the dataset, the median Kaplan-Meir estimate of survival time was 13 years.

Survival probabilities of snags declined monotonically for the first ten years after death for all species (Fig. 2). However, there was a short, initial delay in snag fall for all species except yellow pine. For example, less than 2 % of white fir and sugar pine snags fell in the first two years. In contrast, 12.5 % of the yellow pine snags fell in the first two years (Fig. 2). For five of the six species, the decline in survival probabilities slows after 20 years. The exception was red fir where snag survival continued to decline roughly linearly for 30 years.

The best performing AFT model used a log-logistic distribution and included the interaction of species and diameter as covariates. This model yielded the lowest AIC score (Table S4), demonstrated good model performance with a concordance statistic of 0.70 (SE = 0.009), and revealed a strong fit to the data (Fig. S3).

Model results reveal a strong effect of both species and size, and they also highlight key interactions between them (Fig. 3, Table S5 and Fig. S4). Species was the most influential covariate (Fig. S5). Across all species, snag longevity increased with size; however, the effect varied by species. Size played a small role in influencing yellow pine fall rates but a very large role in red fir and sugar pine longevity. Size played an important, but less pronounced role for white fir, incense-cedar, and black oak. When size is held constant, yellow pines fall the fastest while incense-cedars fall the slowest.

3.3. *Snag fall rates and fuel loads*

Our projections indicate large inputs of biomass from snag fall in the

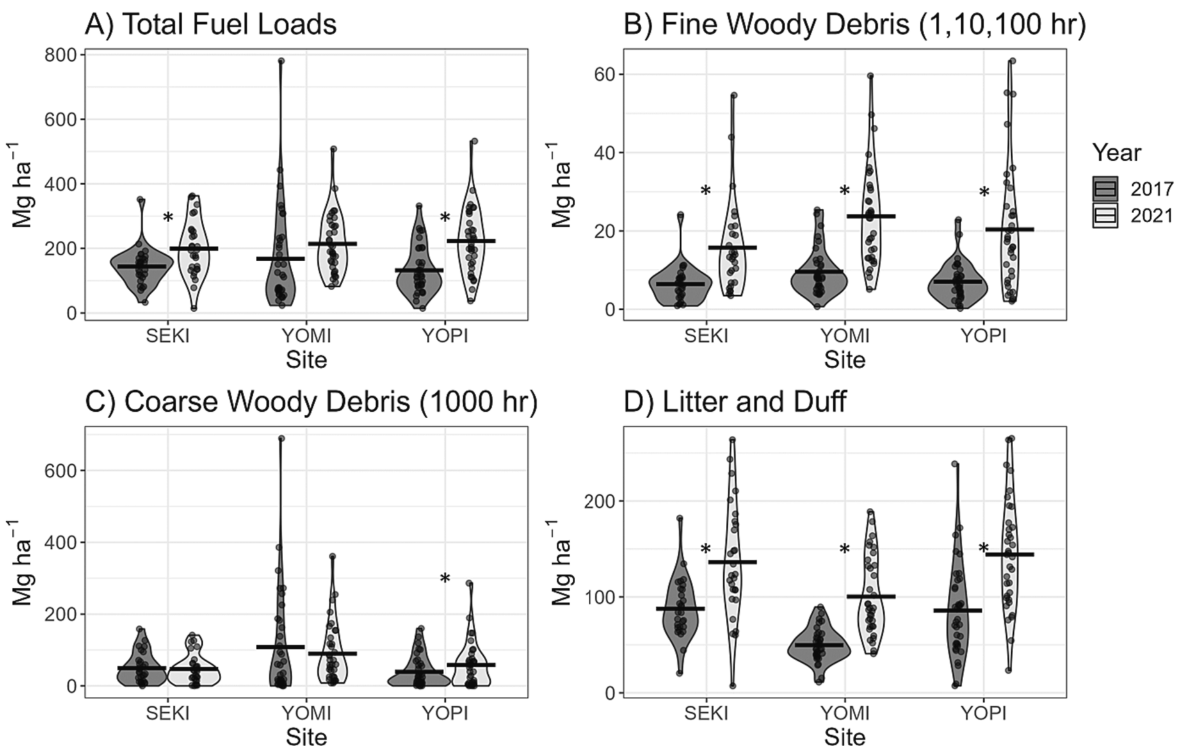

Fig. 1. Violin plots of the fuel load estimates within each stratum at SEKI, YOMI, and YOPI measured in 2017 (dark grey) and 2021 (light grey). Horizontal lines represent mean fuel loads for a given site and year. The width of the shaded filled region around each violin plot illustrates the proportion of data located at that given value and the dots represent actual measurements for a given plot. Groups of plots marked by * denote a significant change ($p < 0.05$) from 2017 to 2021.

Table 1
List of species included in the model, number of snags of each species, number of snags that fell for each species, median survival time using the Kaplan-Meier estimator, and mean DBH. DBH = diameter at breast height (BH = 1.37 m).

Species	Number of snags	Number of snags that fell	Median Survival Time (years)	Mean DBH (cm)
White fir	1418	604	13 (8–26)[1]	43 (12.7–162)[3]
Red fir	412	147	17 (9–27)	49.4 (12.7 – 251)
Incense-cedar	412	71	20 (13-und)[2]	25.2 (12.7–184)
Sugar pine	483	211	13 (9–25)	41.1 (12.7–98.1)
Yellow pine	175	76	9 (6–16)	49.9 (12.7–162)
Black oak	159	78	14 (8–23)	23.6 (12.7–98.1)

[1] Values in parentheses represent the 25th – 75th quartile range; [2] und = undefined, the 75th percentile is undefined due to skewedness of data; [3] DBH values in parentheses represent the minimum and maximum of observed values.

coming decades in these Sierra Nevada mixed conifer forests. Based on 2017 field measurements, SEKI, YOMI, and YOPI had an average of 162.4 (SE = 31.07), 236.8 (64.05), and 165.08 (24.39) Mg ha^{-1} of standing dead biomass in the form of snags, respectively. However, the rate of input from snag fall varied across our three sites with YOPI having the fastest input of snag material, with projections of 136.1 Mg ha^{-1} (115.6–150.6, 25th and 75th percentile values) of biomass input by 2040 (Fig. 4). YOMI was predicted to have a fast initial deposition of material, with 92.3 Mg ha^{-1} (64.4–109.3) falling by 2040 followed by a slower but steady input for the decades afterward (Fig. 4). SEKI was projected to have the slowest biomass deposition with 49.4 Mg ha^{-1} (30.9–80.3) predicted to fall by 2040, and the cumulative biomass curve revealed a monotonic increase in cumulative inputs to 2070 (Fig. 4).

For snags in our Drought Mortality study, the Fire and Fuels Extension (FFE) of the Forest Vegetation Simulator (FVS) predicted a median fall time that was 30 % longer than our best model (Table 2). The timing of predicted snag falls diverged even more between models – FVS-FFE estimated 69 % more snag falls in 5 years and 17 % less snag falls in 20 years. In other words, FVS-FFE expected more snags to fall soon after dying and more snags to persist long after dying.

Species-specific snag fall rates in old-growth forests of the southern Sierra Nevada were considerably slower that reported for comparable studies (Table 3). Median fall times tended to be 2–3 years slower regardless of the cause of mortality or the design of the study. However, four of the five studies had survivorship distribution curves (i.e., log-logistic and Weibull) that suggest an initially slow rate of snag fall that then accelerates with time since tree death.

4. Discussion

4.1. Short-term changes in fuel loads

Across all three sites, there was a significant increase in the amount of fine woody material and litter and duff from 2017 to 2021. These results align well with theoretical timelines of fuel loads following bark beetle mortality (Hicke et al., 2012) and match empirical observations of fuel loads following similar mortality events in Rocky Mountain lodgepole pine (*Pinus contorta* var. *latifolia* Englem. Ex Wats) forests (Klutsch

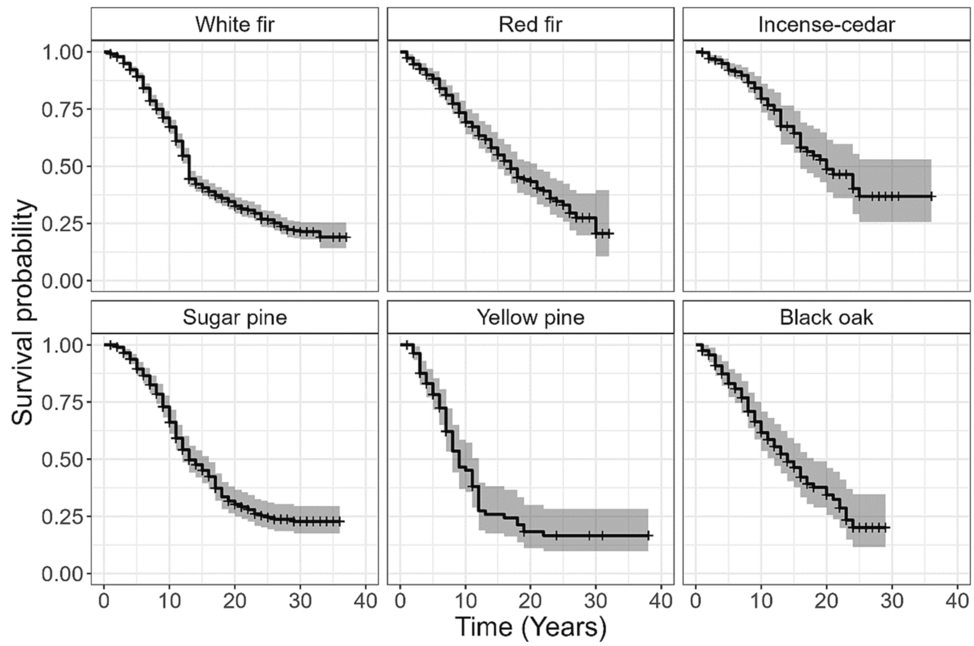

Fig. 2. Observed survival curves using Kaplan-Meier estimator for each species with 95% confidence intervals (shaded areas). Curves depict survival probability of snags over time, by species.

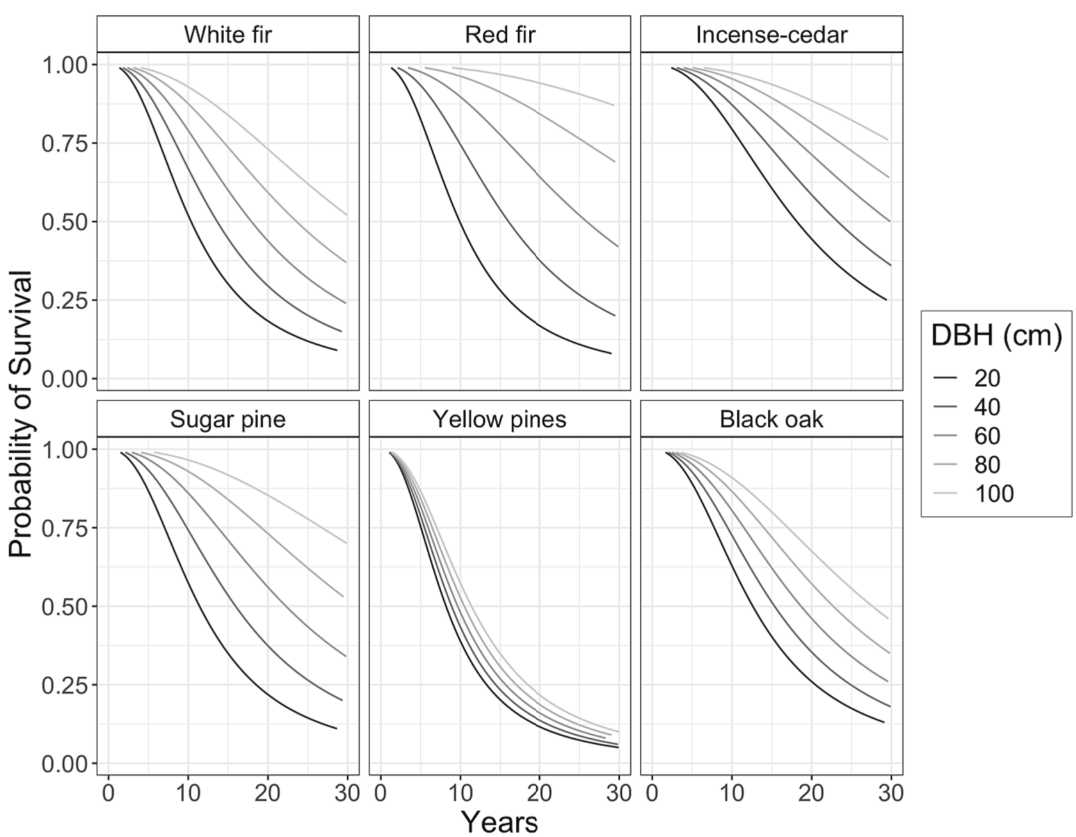

Fig. 3. Predicted conifer snag survival times for each species of different sizes. Predictions based on results from the best performing accelerated failure time (AFT) model.

et al., 2009; Page & Jenkins, 2007; Schoennagel et al., 2012). Clearly, the loss of leaves and twigs from recently killed trees leads to the rapid accumulation of fine woody material and litter loads. For the Sierra Nevada, the timing documented by Raphael and Morrison (1987), who found that pine and fir snags in the Sierra Nevada lost most of their foliage and small branches within 5 years of mortality, suggests that the decay of snags created by the 2012–2016 drought are likely the primary source of the increases we observed.

The only significant change in coarse woody debris loads was the increase between 2017 and 2021 at YOPI, a site whose overstory is

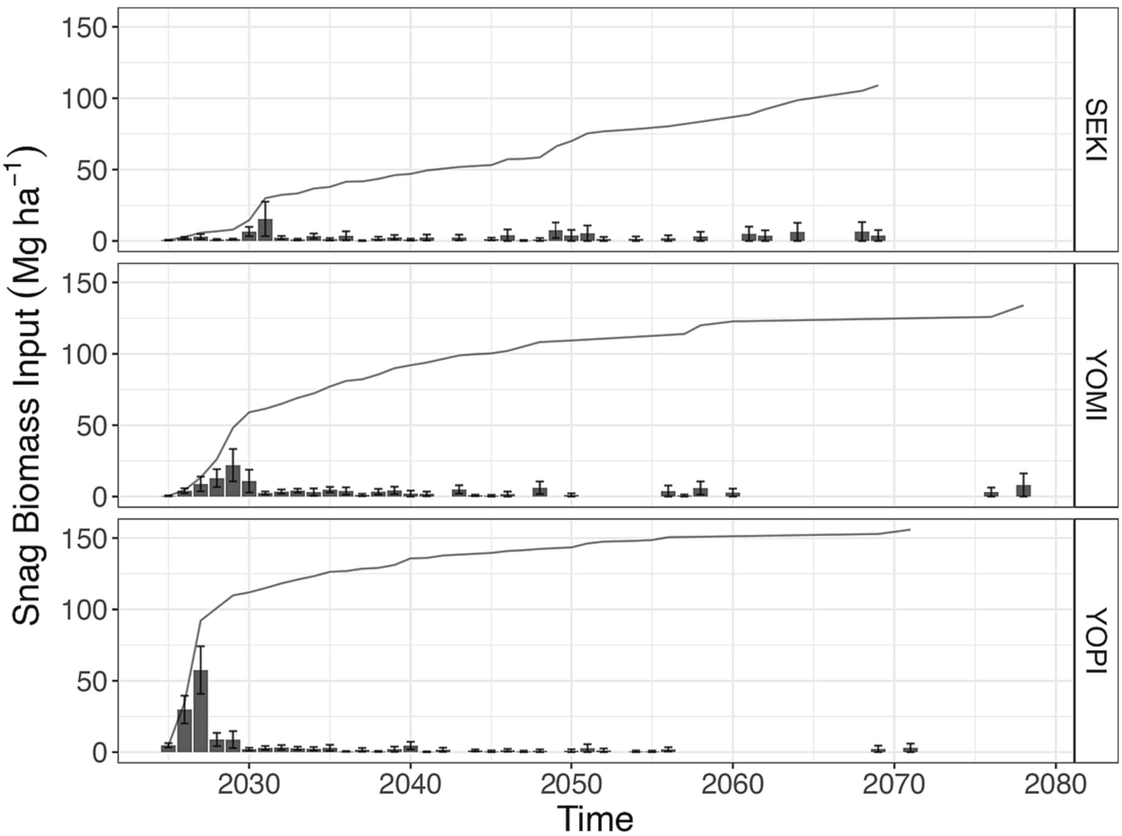

Fig. 4. Forecasted biomass inputs over time based on standing snags in 2017 and model predictions at SEKI, YOPI, and YOMI, based on median modeled fall rates. Bars depict forecasted annual biomass input from snag fall, the error bars depict standard error of annual biomass input, and the lines depict cumulative biomass input from snag fall.

Table 2
Comparison of predicted snag fall rates for snags from 2017 dead tree survey (761 snags) and using predictions from the Forest Vegetation Simulator tool (Reinhardt & Crookston, 2003). Medians (25th – 75th quartile range) reported for the number of snags predicted to fall.

Model	Predicted median fall time (years)	Cumulative number of snags predicted to fall			
		5 years	10 years	15 years	20 years
Our top model	13	89 (82–94)	275 (267–284)	424 (415–432)	522 (514–530)
FVS-FFE	17	150 (143–157)	266 (258–275)	358 (350–367)	431 (423–440)

predominantly ponderosa pine (Table S1). This result is consistent with our fall rate model (Fig. 3) and fuel projections (Fig. 4) that predict faster fall rates in yellow pines and the fastest input of biomass from snag fall at the YOPI site. Regardless of the near-term changes in coarse woody debris, it is important to note that fuel loads for all three sites are well above the historical baseline of 17.7 Mg ha^{-1} (Safford &Stevens, 2017). Similarly, our 2021 fuel loads for fine woody material (1, 10, and 100-hr fuels) are well above historical estimates of 3.6 Mg ha^{-1} in the Sierra Nevada (Safford & Stevens 2017).

4.2. *Snag fall rates*

The observed patterns in snag fall rates generally align with prior estimates of snag survival for mixed conifer species. Several studies report that ponderosa pine snags fall faster than fir snags and that incense-cedar snags often fall at the slowest rates (Grayson et al., 2019; Raphael & Morrison, 1987; Ritchie et al., 2013) – results consistent with our findings. While our observation that snag longevity increases with DBH is also common (Keen, 1955; Parish et al., 2010), an interactive effect between species and size has not been reported. Larger trees tend to have a greater volume of rot-resistant heartwood which provides structural support to standing trees (Harmon et al., 1986), and would explain the general trend that larger snags remain standing longer relative to smaller snags. The interaction of size and species is likely driven by varying amounts of sapwood and heartwood within each species and the processes by which each species decays. Yellow pines trees decay much more rapidly than other species (Raphael & Morrison, 1987), so perhaps size is less important for these species since extra stability-providing heartwood simply degrades quickly, meaning larger trees only provide marginally more stability and snag survival time, as suggested in Fig. 3. Ultimately, snag failure is most often a result of decay that has removed structural stability. The most common agents of wood decay are insects and fungi which sometimes have species-specific interactions with snags (Wood et al., 2003). Furthermore, rate of bark loss, aspect, and elevation also play a large role in decay (Audley et al., 2021; Harmon et al., 1986; Kimmey, 1955). So, variation in those qualities likely contributed to the observed differences in the effects of species and size on fall rates.

Table 3
Summary of snag fall rates reported for common Sierra Nevada tree species. Yellow pine includes both ponderosa pine and Jeffrey-pine; DBH is diameter at breast height (breast height = 1.37 m). Note: und = undefined.

DESIGN DETAILS	This study	Morrison[1]	Landram[2]	Grayson[3]	Keen[4]
Location	Sierra Nevada	Sierra Nevada	Modoc/Cascades	Sierra Nevada	Modoc
Monitoring regime	annual	periodic	annual	annual	mixed
Length of record (years)	38	10	9	10	30
Number of snags observed	3,059	220	3,734	1,140	6,657
Cause of death	multiple; not fire	multiple; not fire	mixed	wildfire	beetles
Age of snag	recently dead	recently dead	includes older snags	recently dead	recently dead
Size range (DBH, cm)	13–251	13 to 53+	13–76+	24–136	25–127+
Snag minimum height (m)	1.37	1.37	< 6	3.65	6
Survivorship distribution	log-logistic	exponential	Weibull	Weibull	Weibull
SPECIES	Median (interquartile) Fall Time				
White fir	13 (8–26)	8 (3–16)	8 (5–13)	10	–
Yellow pine	9 (6–16)	6 (2–12)	6 (4-und)	7	7 (5–10)
Incense-cedar	20 (13-und)	–	–	14	–
Sugar pine	13 (9–25)	–	–	11	–

[1] Morrison & Raphael, 1993. Only included snags that were in decay stage 1 (recently dead) and measured in unburned stands. Snags were tracked across two, five-year intervals.
[2] Landram et al., 2002. Snags in all decayed stages were included in the initial inventory. Snags in harvested, burned, and unburned stands were included. Snags were tracked annually for nine years.
[3] Grayson et al., 2022. Snags recently killed by wildfires were tracked annually for 10 years. No model fitting was necessary since the manuscript reported fall rate medians and survivorship distribution.
[4] Keen, 1955. Only included beetle-killed snags from unburned stands on loam soil. Snags fall was tracked annually for 15 years and then remeasured at 30 years.

4.3. Projected trends in fuel loads

At all three study sites, coarse woody material was expected to increase dramatically as a result of the 2012–2016 drought (Fig. 4). The YOPI site was projected to have the fastest inputs. The more than 100 Mg ha^{-1} of coarse woody debris expected to fall by 2030 is almost double the current amount (Fig. 4, Table S2). This rapid accumulation was driven by the fast fall rates of yellow pine snags. In contrast, the YOMI and SEKI sites had more gradual inputs over time. YOMI would not reach 100 Mg ha^{-1} of input until 2045, and SEKI in 2060 (Fig. 4). Regions of the Sierra Nevada that contain high densities of yellow pine snags will likely see rapid fall rates and heavy coarse woody material surface fuels more quickly; regions with more fir and cedar snags will likely experience a more gradual deposition of heavy fuels. These forecasts only include inputs of woody material from snags present in 2017. Future studies may make projections that incorporate not only cumulative inputs from snag fall but also model fuel decomposition over time to formulate more realistic forecasts of total biomass over time.

4.4. Model comparisons

Our review of snag fall rates for common species in the Sierra Nevada identified three consistent patterns within the region: smaller trees fall faster; yellow pine falls the fastest and incense-cedar falls the slowest; and fire-killed trees fall faster than trees killed by other factors (Table 2, Table 3). However, the timing of snag fall varied among the studies, and these variations have management implications. For example, snags started falling immediately after dying in the FVS-FFE model (Table 2) while all but one of the empirical projections (Table 3, Morrison & Rafael 1993) included a lagged fall response. The Weibull and log-logistic distributions that best fit the observed fall rates implied a non-constant process with an initially slow rate of fall that accelerates a few years after death (Table 3). In other words, the empirical models suggested a longer window to safely treat fuels or remove hazard trees. For our case study (Table 2), FVS-FFE also over-predicted long-term snag persistence. Again, this deviation from the empirical model has management implications – retaining large snags is an important criterion for wildlife habitat.

This mismatch between our results and FVS-FFE model outputs must be interpreted in context. The species-specific fall rates observed in the Forest Demography study were consistently slower than comparable studies (Table 3). And despite our efforts to make consistent comparison, there were several confounding factors. For example, we did not have sufficient information to control for differences in DBH. Other differences that may influence snag fall include (Audley et al., 2021; Gärtner et al., 2023, Keen, 1955; Rhoades et al., 2020; Ritchie et al., 2013): cause of death, age of snag, and snag minimum height (Table 3), as well as differences among sites (e.g., climate, terrain, management history, and soil fertility). Thus, our results from old-growth forests in the southern Sierra Nevada may represent the upper range of snag persistence for the region.

At the same time, these results were derived from a remarkably robust empirical basis. The Forest Demography Study documented the annual fate of more than 3,000 snags for almost four decades. The log-logistic distribution was the clear model choice (weight of evidence = 1; Table S3), the predicted results tracked observations (Fig. S3), and the non-parametric and parametric models produced very similar results (Fig. 2, Fig. 3). The resolution these data provide is exceptional. For example, all snag species except for yellow pine had an initial lag in snag fall (Fig. 3). Detecting this lag depended on precisely documenting the time of death and annual measurements – two features of the Forest Demography Study. In contrast, the data used in Morrison and Raphael (1993) cannot detect this initial delay. There was uncertainty with the time of death and the snag survival was surveyed at five-year intervals. In terms of study design, Grayson et al. (2019) shares similar strengths to the Forest Demography Study – large sample size, known time of death, and annual monitoring of snag fall. They also detected an initial lag in snag fall and their model parameters indicated that snag persistence to 100 years was an exceedingly rare event (e.g., 1 in 10,000 for a white fir with a DBH = 100 cm). Together these two studies provide a valuable, evidentiary basis to revise the snag fall model in the WS variant of FVS-FFE (Rebain, 2010).

4.5. Management implications and conclusion

At our sites and likely in other forests in the western Sierra Nevada that experienced drought-related mortality, fine fuel loads have already increased, and coarse woody debris loads will rise in the coming 10–20 years as many snags fall to the ground. This increase in fuel loads from snag fall, coupled with already heavy surface fuel loads (Vilanova et al., 2023), will increase the risk of high severity fire and potential for mass fire (Arno, 2000; Brown et al., 2003; Coppoletta et al., 2016; Stephens et al., 2018; Stephens et al., 2022). There will be added difficulties for wildland firefighting operations as snags falling represent a considerable danger to fire crews and downed snags can hinder fireline construction and block vehicle access (Dunn et al., 2019; Plucinski, 2019). Managers should anticipate particularly fast fall rates and high coarse woody debris loads in areas containing high densities of large yellow pine snags, which were one of the most common targets of the recent bark beetle and drought mortality (Axelson et al., 2019; Fettig et al., 2019; Pile et al., 2019).

Managers have some options for mitigating the increase in surface fuel loads associated with extreme snag densities. If conducted soon enough, salvage logging could recover some of the economic value in the snags while also removing some of the snag biomass to lower future fuel input from snags (Prestemon et al., 2006). However, commercially viable salvage operations are both time-limited and access-limited. Timber value declines rapidly following tree mortality and many stands are inaccessible to logging operations due to federal regulations and logistical constraints (North et al., 2015). Salvage operations remain an option for privately-owned forests in the Sierra Nevada (Stewart et al., 2016). Although expensive, salvage operations that do not remove timber but focus on falling, piling, and burning snags can effectively reduce future fuel inputs. Chipping snags and large coarse woody debris and sending this fuel to an electrical power plant is another option but is limited in extent because of hauling costs.

Prescribed fire represents another management option. Controlled burns in areas of high mortality would consume recently deposited fine fuels and any coarse woody debris already present from snag fall (Knapp et al., 2005; Stephens et al., 2018). Our results suggest most snags will fall within the next 10–20 years (Fig. 4, Table 2). Since operating in areas of high snag densities can be dangerous, it would be best to burn soon after snag creation before the period of frequent snag fall or to burn after most snags have fallen in 20 or more years. Burning prior to snag fall has the advantage of reducing the high surface fuel loads already present so that future coarse woody debris from snag fall does not add to the risk of mass fires (Stephens et al., 2018). Nonetheless, burn operations may be difficult in stands with high log densities and coarse woody debris consumption may be limited under prescribed burning conditions (Knapp et al., 2005). Despite the many regulatory, operational, and political barriers to implementing prescribed burning (Miller et al., 2020; York et al., 2020), it is likely the most feasible option available to managers to address fuel inputs associated with episodic mortality events. Regardless of the method, the reduction of snag densities is best implemented in a way that not only considers their impact on wildfire hazard but also their contributions to ecosystem structure and function (e.g., wildlife habitat and nutrient cycling).

Managing forests in the aftermath of extensive mortality is difficult, particularly given the novel conditions generated by climate change (Anderegg et al., 2022). The increased abundance of snags associated with hotter droughts is one immediate consequence and a pressing management challenge (Allen et al., 2015). Not only did this study provide local managers with relevant information regarding snag longevity and fuel load accumulation in the southern Sierra Nevada, but also it generated essential, empirically based models that can improve the performance of the decision support tools used by managers throughout the region. Restoring forests before they experience severe drought and subsequent mortality could prevent many of the adverse effects documented in this study.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data are available at the USGS ScienceBase Data Catalog (https://doi.org/10.5066/P938EGYD).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2023.121521.

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