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Aspen also responds readily to canopy opening disturbances such as wildfire or coppice silviculture, providing critical forest resilience especially in regions where conifer regeneration is lacking due to compound disturbances (Andrus et al., 2021; W. D. Shepperd et al., 2015).

Simulated fire behavior in aspen stands of northern Utah showed that greater aspen dominance reduced crown-fire potential, particularly under moderate fire weather conditions (DeRose & Leffler, 2014). Similarly, an experimental study in the boreal forests of Canada showed that high-intensity crown fire in conifer-dominated stands failed to sustain itself in adjacent aspen-dominated stands (Alexander, 2010).

The expansion of deciduous forest species, which are associated with less flammable fuel profiles, has been proposed as one unique solution for reducing wildfire hazard without completely removing fuels (Fechner & Barrows, 1976; Girardin & Terrier, 2015; Nesbit et al., 2023). In particular, quaking aspen (*Populus tremuloides Michx.*), which is the most widely distributed tree species in North America (Rogers et al., 2020), has been characterized by potentially fire-moderating traits such as lower canopy bulk density, higher canopy base height, and greater leaf moisture content (DeByle & Winokur, 1985; W. Shepperd, 1990; Shinneman et al., 2013). Simulated potential fire behavior in aspen stands of northern Utah showed that greater aspen dominance reduced crown-fire potential, particularly under moderate fire weather conditions (DeRose & Leffler, 2014). Similarly, an experimental study in the boreal forests of Canada showed that high-intensity crown fire in conifer-dominated stands failed to sustain itself in adjacent aspen-dominated stands (Alexander, 2010). In addition, responding to canopy-opening events like wildfire, aspen are often the first to colonize disturbed areas, providing crucial forest resilience in post-fire landscapes (Bartos, 2001; Landhäusser et al., 2010; W. D. Shepperd et al., 2015). Aspen, with these fire-moderating traits, may offer a natural buffer to extreme fire behavior and post-fire ecosystem impacts. However, there remains a knowledge gap between desired management outcomes and scientific understanding of how, when, and where aspen forests moderate fire behavior, especially relative to forest composition and structure characteristics at regional-to-continental scales (Nesbit et al., 2023). Still fewer studies have been conducted in the Southern Rockies, where quaking aspen is overwhelmingly the dominant deciduous forest type.

In the Southern Rockies, forest types generally follow distinct fire regime characteristics that are tied to climate, dominant species, elevation, and fuel condition, and are often defined based on the frequency, intensity, and severity characteristics (Schoennagel et al., 2004; Tepley et al., 2018). For example, low elevation pPonderosa pine forests in the Southern Rockies are often defined as having historically high frequency, low intensity/severity fire behavior (*ref*), although this dynamic has been altered by a century of fire suppression, changing fire weather conditions, and increased development of the Wildland Urban Interface (WUI) (*refs*). Conversely, high-elevation lodgepole pine and or subalpine spruce-fir forests are typified by low frequency, high intensity/severity wildfire, often as stand-replacing events (Sibold et al., 2006). In the case of aspen, which co-exists with nearly all major forest types of the Southern Rockies across elevation ranges and site conditions (Bartos, 2001; Kulakowski et al., 2013; Landhäusser et al., 2010), fire activity is driven by functional type (*i.e.*, seral or stable), stand structure, and tree age (Morris et al., 2019; Shinneman et al., 2013). Where aspen persists in seral types, or having a conifer component, fire activity (*i.e.*, frequency, behavior) is generally higher than in stable or pure stands (Shinneman et al., 2013). While aspen are generally considered “fire-resistantresilient” because of their moderating effects on fire behavior and prolific response in post-fire landscapes (*ref*), they *will burn* under the right conditions (Alexander, 2010; DeRose & Leffler, 2014). The composition and structure of aspen stands and the conditions (e.g., fire weather, fuel moisture, topography) under which fires are burning likely have a profound effect on observed fire behavior, especially relative to co-occurring major forest types in the Southern Rockies. However, studies elucidating these relationships on observed fire behavior are severely limited, in part due to the lack of during-fire measurements of fire behavior. The widespread availability of remote sensing data before, during, and after a wildfire event presents an opportunity to address this need by relating satellite-derived metrics of fire intensity and severity to forest characteristics, fire weather, and landscape conditions.

Satellite remote sensing has been widely applied to study fire regime characteristics, active fire behavior, and post-fire impacts (Szpakowski & Jensen, 2019). Active fire detection has become a crucial tool for fire managers and researchers to detect, monitor, and assess global fire activity (Wooster et al., 2021). Aggregation of active fire pixels has been used to delineate fire events and recreate daily fire progression maps, providing crucial data to understand the drivers of fire growth patterns and changes in fire regimes (Balch et al., 2020b, 2024; Coop et al., 2022; Parks, 2014; Scholten et al., 2024). Where moderate resolution optical imagery is considered, spectral channels in the mid-infrared (MIR; 4 µm) are sensitive to actively burning fires, capturing a majority of the radiometric energy released by both cool/smoldering (450-850 K) and hot/active (800-1200 K) fires (*ref*). Measurements in this spectral range are also highly correlated with the rate of biomass consumption per unit time (Kaufman et al., 1998; Wooster et al., 2003). Middle infrared radiance is thus employed to calculate the fire radiative power (FRP), a measure of energy released, often in megawatts (MW), by actively burning fires. Studies have applied satellite-derived FRP to, for example, track wildfire smoke emissions (Li et al., 2019) and investigate the relationship between energy released and fire size (Laurent et al., 2019). Other studies have used FRP to measure relationships between biomass consumption rates and fuel moisture content (Smith et al., 2013), canopy interception of out-going radiation (Roberts et al., 2018), and land cover type (K. Barrett & Kasischke, 2013; Fu et al., 2020). Despite this growing body of work, satellite-derived FRP has thus far been understudied as it relates to forest composition, fire weather, and landscape factors, especially in the Southern Rockies.

While studies relating vegetation type to fire intensity may be limited, remote sensing has been widely applied for decades to map and characterize burn severity in post-fire assessments. Severity is a measure of ecosystem impacts and is often characterized by the loss of organic matter on the landscape (Keeley, 2009). Measurements of fire severity from satellite remote sensing have been applied widely to understand fire effects to ecosystems (*ref*), effectiveness of fuels treatments (*ref*), and post-fire vegetation trajectories (*ref*). Satellite-derived fire severity metrics such as the difference normalized burn ratio (dNBR) use the spectral differences between pre- and post-fire multispectral imagery and have been shown to correlate strongly with field-based measurements (Roy et al., 2006). However, these metrics often use non-standardized units and can be difficult to interpret across different biomes and vegetation types (Morgan et al., 2014; Parks et al., 2019a). New satellite-derived severity metrics, such as the composite burn index (CBI), has recently been developed to link field-based measurements of CBI to Landsat imagery (Parks et al., 2019b). The CBI provides an ecologically-relevant and standardized measurement of fire severity which can improve comparisons made between fire events of different biomes. Thus far, studies quantifying the relationship between satellite retrievals of FRP and subsequent burn severity are lacking, though field based measurements suggest there should be a strong relationship (*ref*).

Middle infrared spectral channels are sensitive to actively burning fires and highly correlated with the rate of biomass consumption per unit time (Kaufman et al., 1998; Wooster et al., 2003). Middle infrared radiance is thus employed to calculate the fire radiative power (FRP), a measure of energy released, often in megawatts (MW), by actively burning fires. Studies have applied satellite-derived FRP to, for example, track wildfire smoke emissions (Li et al., 2019) and investigate the relationship between energy released and fire size (Laurent et al., 2019). Other studies have used FRP to measure relationships between biomass consumption rates and fuel moisture content (Smith et al., 2013), canopy interception of out-going radiation (Roberts et al., 2018), and land cover type (K. Barrett & Kasischke, 2013; Fu et al., 2020). Despite this growing body of work, satellite-derived FRP has thus far been understudied as it relates to forest composition, fire weather, and landscape factors, especially in the Southern Rockies.

Satellite RS has been extensively applied to study fire regimes, active fire behavior and post-fire impacts (Szpakowski & Jensen, 2019).

In the western U.S., fire regimes are changing in response to increasing human ignitions (*ref*), a more fire-conducive climate (*ref*), and over a century of fire exclusion in fire-adapted landscapes (*ref*). These changes have altered fire behavior (*ref*), forest composition (*ref*) and in some cases have ushered ecosystem transformations such as the conversion of forest to non-forest following high severity wildfire (*ref*). In response, varied adaptive management strategies for reducing wildfire hazard and improving forest resilience are urgently needed [(Hessburg et al., 2021)](https://www.zotero.org/google-docs/?broken=lGNp8A). The direct modification of forest composition and structure is a viable management tool often leveraged to reduce wildfire risk to communities and ecosystems [(Prichard et al., 2021; Ziegler et al., 2017)](https://www.zotero.org/google-docs/?broken=qYM0r1). These management actions often involve the reduction of fuels through mechanical treatment or prescribed fire, thereby reducing extreme fire intensity and severity potential [(Agee & Skinner, 2005; Stephens et al., 2009)](https://www.zotero.org/google-docs/?broken=6XHu6r). However, the reduction of fuels may in some cases be

Composition and structure (e.g., size, biomass, canopy cover, etc.) together influence *fire intensity*, or how hot a fire burns, and *fire severity*, or the consumption of organic matter (Hagmann et al., 2021; Keeley, 2009; Meigs & Krawchuk, 2018). Fire intensity and severity, in turn, influence suppression difficulty, during and post-fire impacts to the built environment (e.g., structure loss, water quality, and air quality)during and post-fire community impacts, and ecosystem functioning (Keeley, 2009; Plucinski, 2019). Thus, the direct modification of forest composition and structure is a viable management tool often leveraged to reduce wildfire risk to communities and ecosystems (Prichard et al., 2021; Ziegler et al., 2017). These management actions often involve the reduction of fuels through mechanical treatment or prescribed fire, thereby reducing extreme fire behavior potential (Agee & Skinner, 2005; Stephens et al., 2009). However, as fire activity and impacts continue to rise in the western U.S. (Abatzoglou et al., 2021; Schoennagel et al., 2017), varied adaptive management strategies for reducing hazard and improving forest resilience to wildfire are urgently needed (Hessburg et al., 2021).

In the western U.S., fire regimes are changing in response to increasing human ignitions (*ref*), a more fire-conducive climate (*ref*), and over a century of fire exclusion in fire-adapted landscapes (*ref*). These changes have increased the wildfire hazard (*ref*), altered fire behavior (*ref*), forest composition and structure (*ref*) and in some cases have ushered ecosystem transformations such as the conversion of forest to non-forest following high severity wildfire (*ref*).

A close-up of a graph

AI-generated content may be incorrect.

**Figure S3.** (A)Maximum FRP summarized across varying levels of aspen/conifer co-occurrence. Conifer here groups all other evergreen forest types in one category. (B) Distribution of percent cover overlap between aspen and conifer.

**A screenshot of a graph

AI-generated content may be incorrect.**

**Figure 5.** Effects of species co-occurrence on maximum FRP. Column (A) two-way interactions between aspen and co-occurring conifer percent cover where the aspen + co-occurring cover is >50%. Column (B) distribution of aspen and co-occurring species percent cover, again where at least 50% of forested area is made up of those two types.

### 2.5.1. Majority forest cover

We calculated the dominant forest type in each gridcell based on the proportion of TreeMap pixels.

The algorithmic forest type code (*FORTYPCD*) assigns a single forest type to each TreeMap pixel based on the imputed FIA plot information. The FORTYPCD closely aligns with LANDFIRE existing vegetation type (EVT), providing an assignment of the most likely forest type for that pixel. Based on common co-occurring species in the Southern Rockies, we collapsed some *FORTYPCD* into groups. For example, Engelmann spruce and subalpine fir were grouped into *Spruce-fir* and pinon and juniper types into *Pinon-juniper* (**Table SX**). For each gridcell, we calculated the percent cover of all forest types measured as the proportion of pixels in each type.

We calculated the percent cover for all forest types within each gridcell and identified forest types making up 97% of the total forested area by calculating the proportion of forest types in each grid cell.

The predominant forest types included ponderosa (28%), lodgepole (26%), spruce-fir (16%), quaking aspen (11%), mixed-conifer (10%), pinon-juniper (8%), and oak-woodland (6%) (**Figure S1a**). For each grid, the percent cover of each of these species groups was calculated from pixel counts within grids, and the majority type was identified as predominant. The distribution of grid cells with dominant species varies relative to the landscape proportion, where, for example, grids dominated by mixed conifer, oak-woodland and pinon-juniper are rare and lodgepole or ponderosa are more common (**Figure S1a**).

### 2.5.2. Species-specific structural metrics

Species-specific structural metrics were derived for all species present in a gridcell based on the TreeMap *Tree Table*, which provides rich information about the size, species, and status (live/dead) of all measured trees based on the imputed FIA plot data (Gray et al., 2012; Riley et al., 2021, 2022). These summaries thus include potential understory composition and mixed-forest pixels, which are important for understanding fire behavior and other ecological processes (Krebs et al., 2019). Instead of the *FORTYPCD*, we counted the frequency of FIA plot identifiers (*TMID*) from TreeMap pixels then linked these to the *Tree Table* to retrieve the individual tree information. We limit forest species to those which align with our predominant forest types described in *Section 2.5.1* and which would be commonly observed in Southern Rockies forests. We calculated the live and dead TPP and BA summed across individual species for each FIA plot and multiplied by the number of pixels assigned that plot identifier in the grid. The TPP was estimated by multiplying the *TPA\_UNADJ* attribute in the *Tree Table* for each species by a conversion factor for acres/pixel (Riley et al., 2022). BA was calculated by using the diameter (in) of individual trees in the *Tree Table* multiplied by the number of trees it represents per pixel (*i.e.*, TPP), per the FIA sampling design (Gray et al., 2012; Riley et al., 2022). Using BA and TPP, we also calculated the quadratic mean diameter (QMD) for each plot and species. The QMD represents a measure of average tree diameter that is commonly used in forestry practices and is useful for characterizing the size structure of trees (Curtis & Marshall, 2000). High values of QMD represent forests which are dominated by a few large individuals whereas small values represent forests dominated by a larger number of small trees. These metrics were then aggregated to the grid level to determine the contributing proportion of each species to the total dominance (BA) and abundance (TPP) of forest types within each grid (**Figure S1b**). Finally, each grid was assigned a diversity score using the Shannon diversity index based on abundance (H-TPP) to assess the structural richness and evenness of forest types (Neumann & Starlinger, 2001). We calculated the H-TPP by summing the product of each species' TPP proportion within the grid and the natural logarithm of that proportion, giving a higher value when there is a greater variety of species.