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Managing for heterogeneity reduces fire risk in boreal forest landscapes—A model analysis

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ABSTRACT

Boreal forests are susceptible to direct and indirect human impacts, which are intensified by climate change. Amidst these challenges, wildfires are increasingly shaping forest dynamics. Boreal forest management must be adapted to mitigate such risks. It is still unclear whether management practices such as using smaller stands, species mixtures and avoiding ditching can effectively mitigate wildfires. We develop a cellular-automata simulation approach and couple it with minimalist litter and plant water balance models to examine the influence of stand size, species composition (conifers vs. broadleaf-deciduous species), and ditching impact on fire spread. We find that smaller forest stand sizes greatly reduce landscape fire risk, where for instance, decreasing stand size from 100 to 1 hectares leads to a fivefold reduction in fire risk. Diversifying tree species composition and avoiding landscape ditching can further reduce fire risk, especially when stand size is large. The effect of forest heterogeneity on fire risk is comparable to that of variability of environmental factors such as air humidity, highlighting the importance and potential of forest heterogeneity in fire mitigation. Our model allows us to explore the complex interactions that govern wildfire behavior in intensively managed boreal landscapes, supporting wildfire risk assessments and informing the development of more sustainable and climate resilient forest management strategies.

1. Introduction

Boreal forests, which account for roughly 30% of the world's forests, are undergoing stronger temperature change than other biomes (Soja et al., 2007; Gauthier et al., 2015; Boulanger and Pascual Puigdevall, 2021), and suffering from increasing range of threats including droughts, pest outbreaks, and wildfires (Kärvemo et al., 2023; Cunningham et al., 2024). Wildfires have become concerning in many biomes around the world, particularly in boreal forests (Pechony and Shindell, 2010; Pausas and Keeley, 2021; Cunningham et al., 2024). Fires are an important factor determining the natural dynamics of forests (Bergeron et al., 2001; Boulanger and Pascual Puigdevall, 2021). Although with wide variations between regions and in time (Bergeron et al., 2001; de Groot et al., 2013), millions of hectares of forest burn every year (Stephens et al., 2014). Climate change has increased the frequency and size of fires, and this trend is expected to continue (Senande-Rivera et al., 2022; Wang et al., 2022; Liang et al., 2024; Cunningham et al., 2024).

Many boreal forests, particularly in Northern Europe, are subject to intense management for timber and biomass production (Högberg et al., 2021), including frequent harvests, as well as landscape augmentation (e.g. ditching) (Gauthier et al., 2015; Sikström et al., 2020). Even-aged, single-species management has become the prevailing management regime in such regions, with several thinnings followed by clear-cutting harvest at age of 60 to 100 years (Högberg et al., 2021; Kellomäki, 2022). This management results in forest stands rather uniform in species, size and age, with little relation to the underlying landscape characteristics. These management practices enhance wood production (Kilpeläinen et al., 2016), but may also lead to more spatially homogeneous stands, which can be particularly susceptible to disturbances, including fires (Gustafson et al., 2004; Zald and Dunn, 2018). Similarly, ditching forest landscapes using excavation and drainage systems creates networks that lower water levels in peatlands, where slow tree growth limits productivity, thus enhancing forest yield (Sikström et al., 2020). Ditching has historically been widespread in boreal forests, particularly in Fennoscandia and Canada. However, there is a growing interest in restoring drained landscapes to support biodiversity and peatland ecosystem functions (Andersen et al., 2017; Elo et al., 2024). Moreover, unditched or restored areas retain more water, and can therefore naturally serve to slow down fires (Lohmus

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et al., 2015). We need to quantify which and to what extent specific management approaches can counteract the climate-change induced increase in fire risk.

To date, most of the research of boreal forest fires has focused on north American forests, where damaging forest wildfires have historically been common and are expected to increase in frequency and severity (Erni et al., 2017; de Groot et al., 2013; Wang et al., 2022). It is unclear whether these results are applicable to northern European forests, which are more intensively managed and generally more homogeneous within stands, relative to North American forests (Kuuluvainen et al., 2021). Additionally, these forests tend to have smaller stand sizes due to the highly diverse landownership and management practices (e.g. 2 hectares in Finland; Yrjölä (2002)). As wildfires are projected to become more frequent and severe in Eurasia (Soja et al., 2007; Senande-Rivera et al., 2022), it is necessary to understand fire dynamics also in these intensely managed forests.

The effect of wildfires on forests depends on their extent, intensity and frequency (Stephens et al., 2014). Fire extent can vary by several orders of magnitude, from small (<1 [ha]) fires to rare mega-fires of a million or more hectares (Soja et al., 2007; de Groot et al., 2013; Stephens et al., 2014). Fire intensity can range from ground fires, not causing significant damage to mature trees even if frequent, to intense crown fires, often leading to tree mortality and stand replacement (Michaletz and Johnson, 2007; de Groot et al., 2013). Among the fire properties, fire frequency and its dependence on forest attributes and landscape augmentation is the most elusive. The easy-to-observe large impact big fires have historically been relatively few, making generalizations difficult. At the same time, most fires are very small and therefore harder to observe (Cumming, 2001). The complex relations between fire extent, intensity and frequency mean that the effects of fires on forest, and the role of forest management, remain difficult to predict (Zald and Dunn, 2018; Descals et al., 2022). Moreover, in many regions such as northern Europe, fires are often started by intentional or accidental human activities, so that observed fire frequency may not be directly related to the attributes of forests and landscapes (Granström and Niklasson, 2008). The fire risk, measured as the average area burnt in a given landscape once a fire has started, allows synthesizing the effects of wildfire effects, irrespective of its ignition cause.

An active fire requires three components: fuel, heat, and oxygen. Consequently, wildfire behavior is determined by three main factors: (i) stand attributes, including the structure and prevalence of burnable fuel; (ii) fuel moisture, i.e. the amount of water in ignitable organic material (with water slowing the heat accumulation needed for ignition and spread), and (iii) wind speed and climatic conditions in general, which affect available oxygen and fuel moisture (Rothermel, 1972; Pastor et al., 2003). All of these attributes depend on the interaction between large-scale external conditions (e.g. air temperature and humidity, antecedent precipitation, wind speed and direction) and the internal landscape properties (e.g. forest structure, topography), which can vary considerably across small distances, and are heavily influenced by forest management. To a large degree, these complex interactions are the reason for inadequate understanding and predictions of wildfire dynamics (Pastor et al., 2003). Notably, aspects such as the role of live fuel (e.g. leaves) and the impact of soil moisture on both live and dead fuel are rarely the focus of fire studies (Rossa, 2017; Zhao et al., 2022).

Theoretical models enable the exploration of interactions between external conditions and internal landscape properties and have thus been used extensively in wildfire research (Rothermel, 1972; Pastor et al., 2003; Sullivan, 2009b). Among modeling approaches, cellular-automata models are often employed, because they are computationally effective to simulate and interpret, but still describe the key phenomena and facilitate connections between observed patterns and the process that cause them (Sullivan, 2009b; Alexandridis et al., 2011). These models divide the landscape into equal-sized cells, where fire spread is the consequence of repeated iterations of fire propagation between neighboring cells (Pastor et al., 2003; Sullivan, 2009b; Alexandridis et al., 2011). They are thus particularly useful for exploring wildfire spread in spatially heterogeneous landscapes (Sullivan, 2009b; Trucchia et al., 2020), and examine the net effects of the interactions between management, environmental factors, and fire risk in the landscape.

Despite the abundance of modeling studies to predict and understand wildfire behavior (Sullivan, 2009a,b), two significant knowledge gaps still emerge. First, we have a limited mechanistic understanding of how fire dynamics is affected by environmental factors beyond the effects of wind and dead fuel moisture. Second, it is unclear to what extent fire dynamics is impacted by forest landscape structure, notably stand size and species composition, which in turn depend on management decisions.

To evaluate the role of environmental conditions and management choices, a fire risk model needs to: (i) incorporate the effect of stand properties such as density and tree type (which are directly impacted by management) on fire spread; (ii) be sensitive to the moisture dynamics of live and dead fuels, and how these depend on soil moisture, air humidity, and wind; (iii) consider the landscape properties resulting from the interaction between management and the environment, such as the impact of ditching on soil moisture, and the emergent relations between stand properties (in particular, stand age and stand density). We lack a model considering all these aspects. Yet, given their interactions, this is necessary to quantify their relative roles and net effects on fire risk. Including these aspects also allows us to fully account for the effects of climatic conditions, which affect live and dead fuel moisture dynamics directly and indirectly through soil moisture and the effects of ditching. This is necessary to evaluate the relative role of changes in management practices and climatic conditions.

In this study, we synthesize existing knowledge on moisture dynamics in dead fuels, live fuels, soils, and the effect of stand attributes (e.g. tree type and density), into a novel cellular-automata simulation model. The model is parameterized for fires in boreal forest landscapes in Fennoscandia, and quantifies the impact of forest and landscape structure on wildfire behavior. We focus on three management aspects: stand size, tree type (conifers vs. broadleaf-deciduous species), and landscape ditching altering soil and hence fuel moisture. We consider how fire spreads through small areas (1–100 [km^2]), relevant to land owners that manage the forest landscapes, and how such spread is affected by management practices under different external hydro-meteorological conditions. To these aims, we combine three components into a single model framework: a fire spread model that is based on simple environmental drivers; ecophysiological modeling of factors affecting fire spread; and properties of boreal forests under intense management. With this framework, we ask: (i) How is fire risk impacted by the size of forest stands? (ii) How do other landscape features interact with stand size to affect fire dynamics? (iii) How effectively can alternative forest management practices mitigate fire risks, and how do these mitigation actions compare to the effect of changing environmental conditions such as air humidity?

2. Methods

We developed a cellular-automata model of wildfire dynamics simulating the spread of fire across a forest landscape. The model explicitly accounts for the role of both forest and landscape structure, and external environmental factors. Fig. 1 shows the how these inputs impact fire spread. We focus on 5 × 5 [km^2] landscapes, and explore how management and hydro-meteorological conditions interact in defining fire spread. We do not explicitly consider the causes of fire ignition. Rather, fire can start within the landscape or reach its edge from an adjacent area – a likely event when modeling small landscapes as in this study. We assume the hydro-meteorological conditions and associated fuel moisture remain constant for the duration of the fire, (e.g. fires do not end due to a rain events), which simplifies the analysis and is reasonably realistic in small landscapes. We present here the model

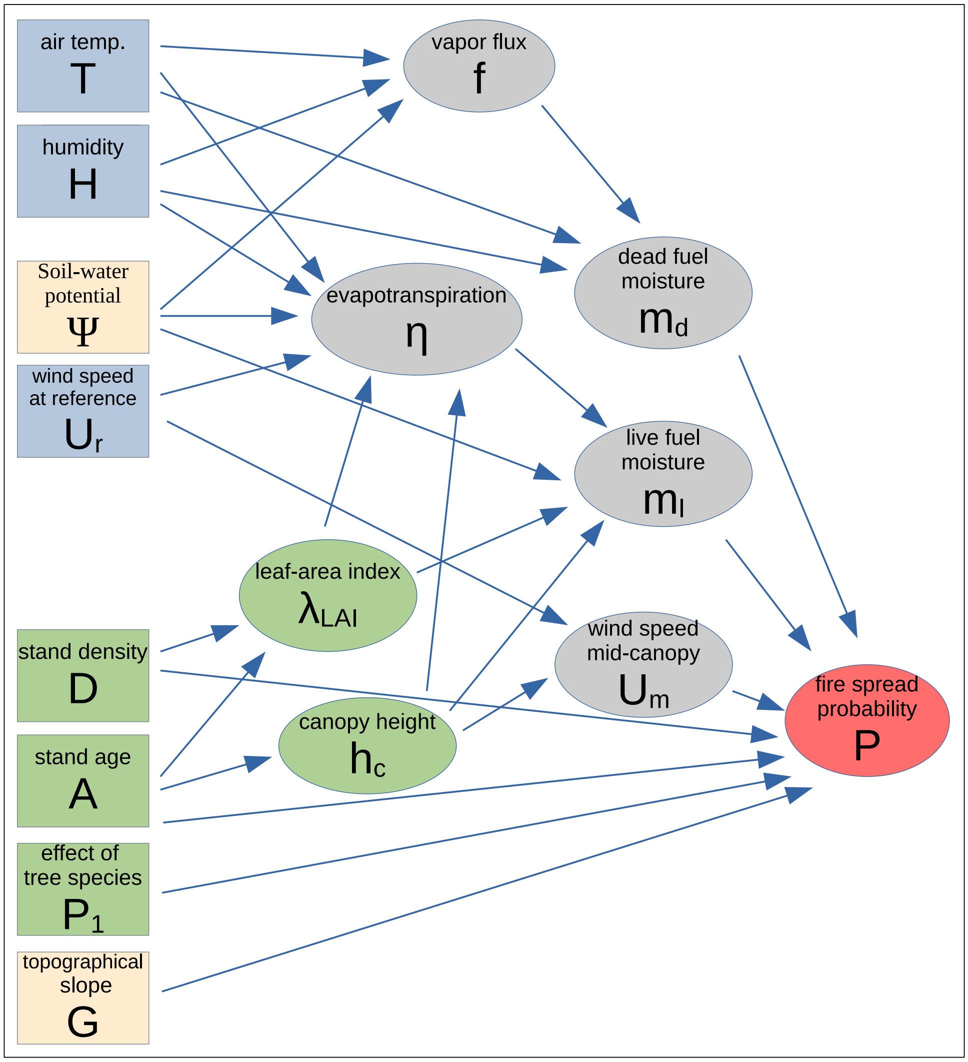

Fig. 1. Model structure. Boxes designate quantities that are given as input, while circles are calculated. Blue, green, and tan denote quantities related to meteorological conditions (uniform in space), tree properties, and physical landscape, respectively. Gray denotes mixed quantities (i.e. based on more than one category), and red is used for the key model output, fire spread probability P.

overview, detailing only the fire spread model. To focus on broad-scale links between hydro-meteorological factors, management, and fire, our model is deliberately simplified, capturing the net effects of small-scale complexities – such as fuel geometry, fine-scale topography, micro-climatic variability, and short-term weather fluctuations – through aggregated relationships. For the same reasons, we keep the environmental conditions fixed, including temperature and wind, during each simulated fire. Further details on the model are described in Supplementary Information (SI) A (full list of parameters), SI B (management and stand structure), and SI C (environmental factors).

2.1. Fire spread model

We model the dynamics of fire spread across a landscape, from its ignition to when it is extinguished, with a focus on capturing surface fires. These fires are the most relevant in many areas of intense management such as the boreal zone in Europe (Drobyshev et al., 2021). We use a stochastic cellular-automata (CA) model (Wolfram, 1984; Karafyllidis and Thanailakis, 1997), where the landscape is partitioned into a lattice of equal-sized cells, and the fire spreads between cells using a probability function. Following previous CA models, each cell (20 × 20 [m^2]) represents a small area in the landscape, but large enough to include many trees (Alexandridis et al., 2011; Trucchia et al., 2020). Stand attributes are uniformly distributed within each cell, but vary across the landscape (see the Simulations and Scenarios subsection for details on their spatial distribution). While the model is a simplified description of wildfire dynamics, it retains the key connections between physical mechanisms and fire spread, while accounting for variability in landscape features such as stand size and composition of species.

The model is run iteratively. At each simulation time-step the fire can potentially spread from each burning cell to its neighbors, and the previously burning cells are assumed to be extinguished. Although these time-steps can represent actual times during the fire, here we consider them simply as part of the simulation process, since we focus on small landscapes and are interested in the overall effect of the fire, rather than its duration. The simulation continues until the fire dies out because it has either burned the entire landscape or failed to spread to any new cells in a given time-step.

For each spread event, i.e. at every time-step for each of the cells that are adjacent to the burning cells, there is a probability P that the fire will spread to the adjacent cell. To determine P we use three main factors that represent the local conditions: the cell characteristics P_C (vegetation type, age/size and density), the factor α that encapsulates effects of wind speed and direction and topographical slope, and a factor summarizing the role of fuel moisture ρ_m. With these, we define

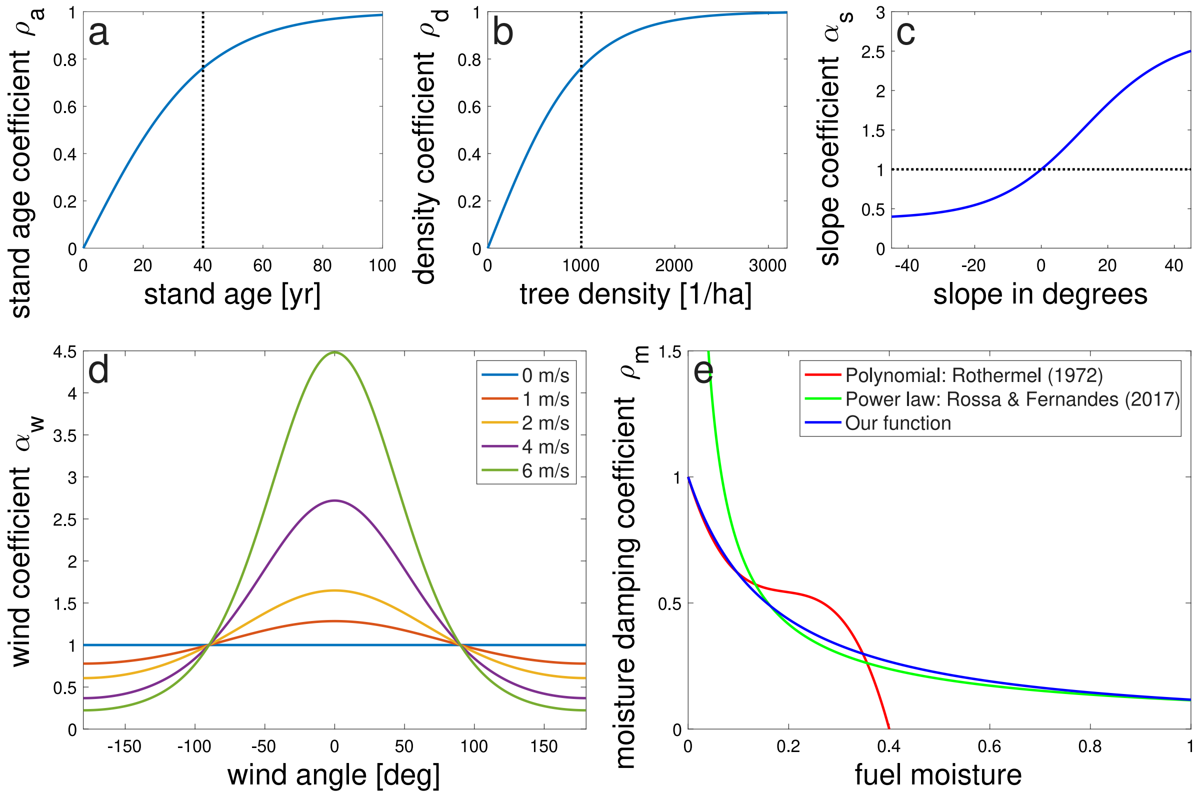

Fig. 2. Summary of the functional form of five factors that affect fire spread. Wind direction is measured as the angle from which the wind blows, and slope is negative when the fire spreads downhill, reflecting reduced spread rates compared to upslope fire movement.

P as (Trucchia et al., 2020):

$$P = (1 - (1 - P_C)^{\alpha}) \cdot \rho_m, \tag{1}$$

where P_C is the baseline fire spread probability of a cell, the coefficient α modifies the probability due to wind and slope, and the coefficient ρ_m dampens fire spread due to fuel moisture. These three parameters roughly correspond to the three aspects allowing fire spread: fuel, oxygen, and heat, respectively.

Expanding Eq. (1) to explicitly include the different factors (see Table 1) yields

$$\begin{aligned} P &= (1 - (1 - [P_0 + \Delta_P \rho_a \rho_d])^{\alpha_w \alpha_s})\rho_m \\ \rho_m(m) &= \rho_m(m_d + \sigma \cdot m_l) \end{aligned} \tag{2}$$

Here, P_0 is the fire spread probability in open ground and $\Delta_P = P_1 - P_0$, where P_1 is the fire spread probability in a dense old forest. By definition, the probability cannot be larger than 1, so that the factors ρ_a and ρ_d lower the spread probability due to (young) stand age or small tree size and (low) tree density, respectively. The parameters α_w and α_s can either increase or decrease the fire spread probability, due to wind and elevation change (slope), respectively. The factor ρ_m is an additive function of dead (m_d) and live fuel (m_l) moisture (Rossa, 2017), where σ ($0 \leq \sigma \leq 1$) quantifies the importance of live fuel moisture on fire spread.

To determine the functional relationships governing the different factors, we use simple functional forms based on general trends reported in the literature (Table 1), as detailed next.

2.2. Quantification of fire spread probability

Six factors impact fire spread in our model (Table 1, Fig. 2). For each factor, we first motivate the functional form selected, and then describe how we determined the parameter values. In SI E we use three empirical patterns from previous studies to validate that our model exhibits the expected behavior regarding fire size distribution (Fig. S9), decidious trees (Fig. S10) and soil moisture gradient (Fig. S11). We note that much of the empirical literature of fire spread and its dependence on other factors focuses on the rate of spread, rather than the probability of spread between adjacent areas. However, these two concepts are strongly related (Alexandridis et al., 2011), and therefore we make use of both notions to quantify the fire spread factors.

2.2.1. Role of ground cover and species

We use discrete values of baseline probability of fire spread for different ground cover types. The properties of ground cover are embedded in P_0, the baseline spread probability over a bare ground, and in P_1, the baseline spread probability of a dense old forest (baseline conditions are of 0 fuel moisture, as well as no slope and no wind). Together with stand age and density (see Sections 2.2.2 and 2.2.3), P_0 and P_1 describe the overall local cell conditions.

In general, the fire spread probability for bare ground (P_0) is substantially lower than for forests (Li et al., 1997; Trucchia et al., 2020). We use a probability value of $P_0 = 0.5$.

The fire spread probability of dense mature forests, P_1 depends on the dominant tree species (Rogers et al., 2015; Tymstra et al., 2010). In boreal forests, broadleaf-deciduous trees burned significantly less than conifers (Cumming, 2001; Ouarmim et al., 2016; Felton et al., 2016). Among conifers, pines have higher risk of fires than spruce (Tanskanen et al., 2005; Kuuluvainen et al., 2017; Cumming, 2001), with the exception of black spruce forests for which similar fire spread rates to pine have been reported (Cumming, 2001; Rogers et al., 2015). Thus, we set higher values of fire spread probability for conifers ($P_1 = 1$) than for broadleaf-deciduous species ($P_1 = 0.7$, see Fig. S10). Note that despite the high value of P_1 for conifers, the baseline fire spread probability of the cell, P_C, is rarely over 0.8 due to stand age and density (Fig. S2). The specific values chosen for P_0 and P_1 are not based on data as it was unavailable, but were chosen to avoid unreasonable results (e.g. persistent fires with high fuel moisture, or conversely, no

Table 1
Factors affecting fire spread (see Fig. 2).

Factor	Higher spread for	Functional form	References
cover type	conifer (vs. broadleaf)	discrete values	Li et al. (1997), Ouarmim et al. (2016)
stand age	older trees	saturating	Erni et al. (2017), Hargrove et al. (2000)
stand density	denser forests	saturating	Johnston et al. (2021), Butler et al. (2013)
wind	stronger wind	exponential	Trucchia et al. (2020), Alexandridis et al. (2011)
slope	upslope (vs. downslope)	sigmoid	Trucchia et al. (2020)
fuel moisture	drier fuel	sub-exponential	Rothermel (1972), Rossa (2017)

fires even with very low fuel moisture). A sensitivity analysis shows that both P_0 and P_1 strongly affect fire spread (with larger values leading to more fire), but without any impact on qualitative behavior (Figs. S13 and S14).

2.2.2. Role of stand age

The amount of live and dead fuel tends to increase with forest age and stand density (Hély et al., 2000; Thompson et al., 2017). The model takes into account the average tree age, which we assume is representative of the mean tree size of a forest cell. Focusing on average tree age/size is appropriate especially in even-age managed forests, which have narrow size distribution (Kuuluvainen et al., 2021).

The fuel availability follows complex trajectories as a result of forest disturbances and management practices (Sturtevant et al., 1997; Schimmel and Granström, 1997; Thompson et al., 2017; Johnston et al., 2021). Observations show a general trend of faster and more substantial fire spread with increasing stand age, and hence tree biomass, leading to accumulation of both dead and live fuel (Schimmel and Granström, 1997; Hargrove et al., 2000; Parks et al., 2015; Hart et al., 2019). However, the age-fire spread relation plateaus or even decreases in older (50+ years) forests (Héon et al., 2014; Erni et al., 2017; Kitzberger et al., 2012).

Following from our definition of P_1 as the fire spread probability for a dense old forest, we assume a stand age dependency ρ_a, with a maximum value of 1. We use a hyperbolic tangent function of stand age A (Fig. 2a):

$$\rho_a = \tanh(A/c_a) \tag{3}$$

As such, ρ_a grows roughly linearly with A up to the critical stand age c_a, but the function is largely flat and near its maximum value of 1 around $2c_a$. The critical stand age c_a likely varies with tree species, growth conditions and other factors. We set $c_a = 40$ [yr], which is roughly the stand age when observed burn rates start plateauing (Héon et al., 2014; Erni et al., 2017).

2.2.3. Role of stand density

The stand density D [trees ha^{-1}] is a key stand attribute for fire spread, and one that can be effectively controlled by forest management. Particularly, forest thinning reduces fire spread, as consistently evidenced by fire experiments (Schroeder, 2010; Butler et al., 2013; Wilkinson et al., 2018; Johnston et al., 2021). Therefore we assume that fire spread increases with D.

However, the literature is inconclusive on the functional shape of the density-fire spread relation (Cavard et al., 2015). We assume that fire spread increases with increasing D at low densities, as the fire needs to make increasingly smaller jumps between trees. However, when the crowns overlap at high tree density, the probability is likely to level off as such jumps are no longer necessary. We thus formulate the density dependency ρ_d as (Fig. 2b):

$$\rho_d = \tanh(D/c_d). \tag{4}$$

We set the critical value as $c_d = 1000$ [trees ha^{-1}], i.e. twice as high as the average post-thinning density of fire-thinning experiments (Schroeder, 2010; Butler et al., 2013; Wilkinson et al., 2018). This is equivalent to assuming that at a density of 1000 [trees ha^{-1}], a further increase in density no longer has a strong effect on fire spread, so that up to c_d the relationship is roughly linear. We note that managed forests do not typically have densities exceeding 3000 [trees ha^{-1}] (see SI B), so that the specific shape of the relationship at high D does not play a major role in the model outcomes for realistic forests.

2.2.4. Role of wind

Wind has a strong effect on fire spread probability, rate, and direction, where strong winds within and above the canopy significantly increase fire spread in the downwind direction, while inhibiting it in the upwind direction (Weise and Biging, 1997; Hoffman et al., 2016). Numerous models have been proposed for the dependence of fire spread on wind speed, based on data showing a highly nonlinear relationship (Weise and Biging, 1997; Nelson, 2002; Boboulos and Purvis, 2009; Hoffman et al., 2016; Trucchia et al., 2020). Here we use a simple exponential function, acting on the cosine of the wind direction, so that the exponent is applied on the effective wind speed inside the canopy in a given direction (Fig. 2d):

$$\alpha_w = \exp((w/c_w) \cdot \cos(\theta)) \tag{5}$$

We set $c_w = 4$ [m/s], which largely follows the formulation by Trucchia et al. (2020), where within-canopy wind speeds of 5 [m/s] lead to α_w values of approximately 3.5 and 0.29, with and against the wind, respectively. Note that the original formulation assumes above-canopy wind speed while here we refer to wind inside the canopy, which is substantially lower but more important for the fire spread (see SI C.3).

2.2.5. Role of slope

The topographical slope affects fire spread similarly to wind speed, with fire spread being stronger uphill (rising warm air plume) and weaker downhill (Rothermel, 1972; FCFDG, 1992; Finney, 1998). We use the factor (Fig. 2c):

$$\alpha_s = \exp(\tanh(G/c_s)), \tag{6}$$

with $c_s = 0.5$ [rad] and G [rad] being the local slope angle in the direction of fire spread. This function is similar to the formulation by Trucchia et al. (2020) in that it levels off for steep slopes (over $\pi/6$), where empirical data is not available (FCFDG, 1992). Nevertheless, the results below focus on gentle slopes where the effect is roughly linear (Fig. 2c).

2.2.6. Role of dead and live fuel moisture

The fuel moisture, i.e. the amount of water in the organic material that burns during fires, is the key factor for the occurrence and dynamics of wildfires (Rothermel, 1972; Van Wagner et al., 1987). Fuel flammability is drastically reduced at higher moisture content, leading to slower fire spread. It is often assumed that above a moisture threshold fires can no longer spread (Rothermel, 1972; Wilson, 1985), although some formulations assume no such threshold (Rossa, 2017).

On these bases, to account for the effect of fuel moisture on fire spread, we set

$$\rho_m = (\frac{c_m}{m + c_m})^{\Omega}, \tag{7}$$

where $m[gH_2O/gDW]$ is the effective fuel moisture and the parameters c_m and Ω control the shape of the function. With $c_m = 0.2$ and $\Omega = 1.2$, Eq. (7) gives values similar to Rothermel (1972) (parameterized for fire extinction occurring at moisture content of 0.4 (Dyrness and Norum, 1983; Masinda et al., 2021)), but avoids its abrupt threshold and non-monotonous derivative. It also retains the possibility of fire spread at high moisture, as in a power-law formulation (Rossa, 2017), while avoiding the exploding term around zero moisture content (Fig. 2e). We use this intermediate form to balance realism and simplicity. While Rothermel's model is widely used, its structure is harder to interpret. Our function keeps a clear mathematical form and can be applied to both live and dead fuels, similar to Rossa (2017).

The effective fuel moisture m depends on the amount and quality of flammable components, but an established approach to categorize fuel type and how that affects flammability is still lacking (Rothermel, 1972; Pastor et al., 2003). Nevertheless, a clear distinction can be made between live fine fuels (living plant organs such as leaves, needles and branches) and dead fuels (dead plant parts and plant litter on the ground) (Sullivan, 2009a; Rossa and Fernandes, 2017). While the importance of dead fuel for fire dynamics is widely accepted, the role of live fuel is still debated (Alexander and Cruz, 2013; Rossa and Fernandes, 2017). To account for the potential role of both live and dead fuel, we determine effective gravimetric fuel moisture m as a weighted sum of dead (m_d) and live (m_l) fuel moisture contents: $m = (m_d + \sigma \cdot m_l)/(1+\sigma)$. The weighing parameter σ quantifies the importance of live fuel for the fire dynamics, with $\sigma = 0$ corresponding to no effect and $\sigma = 1$ to equal importance to dead fuel. Although live biomass in a forest stand can be significantly larger than dead biomass, we assume that σ is lower than 1, given that much of the relevant fire fuel (i.e. fuel with high surface to volume ratio that easily burns) is in the form of litter and other dead debris (Thompson et al., 2017), as well as mosses which behave similarly to fine dead fuels in that they dry quickly and readily ignite (Tanskanen et al., 2006). In line with most fire spread models, m does not explicitly depend on amount or type of fuels. This effect is indirectly accounted for through the stand age (ρ_a; Eq. (3)) and stand density (ρ_d; Eq. (4)) functions (Sullivan, 2009b; Alexandridis et al., 2011).

Both dead and live fuel moisture content depend on the local atmospheric conditions and soil moisture. The dead fuel moisture is determined as a combination of two processes: equilibration with atmospheric humidity (i.e. affected by relative humidity and temperature) and vapor diffusion from the soil, considering the typical equilibration timescale of fuel τ (see SI C.1). The live fuel moisture is quantified based on a minimalist plant water transport model (Couvreur et al., 2018), providing the tree water potential as a function of soil water potential, transpiration rate and tree hydraulic traits (see SI C.2). In turn, the tree water potential is transformed into moisture content via empirical pressure volume curves (SI C.2).

2.3. Landscape and environment affecting fire spread

Given the six factors defining the fire spread probability and their formulations (See Table 1), the following landscape attributes and hydro-meteorological variables determine the probability of fire spread into a given cell (See Fig. 1):

1. cover type of the cell adjacent to a burning cell (hereafter, focal cell)
2. stand attributes (tree age/size and tree density) of the focal cell
3. slope due to elevation difference between burning and focal cell
4. wind conditions inside canopy during fire event
5. dead and live fuel moisture in the focal cell

We consider the first three attributes static for a given landscape scenario. Among them, stand age and density affect fire spread both directly and indirectly via environmental factors. The relationship between age and density is determined based on literature-derived data on stand structure of managed boreal forests (see SI B). The last two attributes, wind speed and direction inside the canopy, and fuel moisture, vary and depend on the weather conditions as well as the landscape attributes.

2.4. Simulations and scenarios

We consider different scenarios of landscapes and forest structures, and simulate fire dynamics over these landscapes. We focus on landscapes of size 5 × 5 [km^2], comprising 250 × 250 equal-sized cells of size 20 × 20 [m^2]. For simplicity we consider relatively flat landscapes and neglect the direct effect of slope on fire spread, because this effect is small when topography is gentle (see sensitivity analysis in SI F; Fig. S15). The landscapes can be drained by a single channel (Fig. 3c), which affects soil and thereby fuel moisture.

The spatial heterogeneity in a landscape is modeled via three key internal factors: (i) stand size and the age and density of trees within the stand; (ii) planting of different tree species (conifers, broadleaf-deciduous species, and their mixtures); (iii) ditching of the landscape to prevent soil water logging. These stand attributes are spatially varied across the individual cells within the landscape, influenced by management decisions. Fig. 3a,b illustrates the size and arrangement of stands in a typical simulated landscape.

We consider a variety of weather conditions (Section 2.4.3) and management type (e.g. same average stand size and species composition; Section 2.4.1). For each set of weather conditions and management type, we randomly generate 500 different landscapes. For each such landscape, we run 500 fire spread simulations with a different randomization in wind direction, initial fire location, and fire progression. In each fire spread simulation, we start by burning 10 cells at the western edge of the landscape (representing a fire that spread from outside the landscape). We limit the wind direction to always be towards the east (between north-east and south-east, with a uniform distribution of the angle), to ensure that the landscape area we focus on is where the fire will spread into. We let each simulation run until the fire is extinguished. In the main text, among the variables included in the fire spread model, we vary in the simulations humidity, wind, soil moisture, and vegetation, while others, such as slope and ignition pattern, are held constant (see SI F for slope sensitivity).

The differences between simulation settings per figure are described in Table 2. We measure the fire size in hectares by multiplying the number of burned cells by the cell area (0.04 [ha]). For each landscape realization we also consider the effect of fire by defining a metric of **fire risk** as the average fire size measured in hectares (averaged over 500 fire simulations) of a given landscape realization. We also consider the distribution of **fire size** over all 500 landscape realizations and all 500 simulations per landscape, making for 250,000 fires overall.

2.4.1. Stand attributes

A forest stand is a combination of neighboring cells with the same forest attributes. The stand sizes ranges from 0.16 [ha] (represented by 4 cells in the simulation) to 1024 [ha] (nearly the entire landscape). We create a random arrangement of stands by iteratively choosing a random location (x, y) in the landscape, and a random stand size of N cells from a uniform distribution with its average designated as the "average stand size". We then define a rectangle of size N, centered on the coordinates (x, y). If this rectangle is empty (i.e. no cells have been populated by other stands so far) then we define it as a new stand (with age and density randomly chosen, see below). Otherwise we increase the size of the rectangle until enough previously free cells are within the rectangle. We choose N free cells in the rectangle, shedding cells randomly from the edges if necessary, and designate this as a new stand. As a result of this procedure, the stand can have non-rectangular shape. This process is repeated until all the cells in the landscape are designated as part of a stand (see Fig. 3a,b). We tested several spatial arrangement of stands (SI D; See Fig. S8), but use the delineated method to explore the main effects of stand size and spatial arrangement (Fig. 3a,b).

We assume that the tree age, density and species composition within each stand is uniform. In the case of mixed-species (conifer and broadleaf-deciduous) stands, we assume that the mixed stand P_1

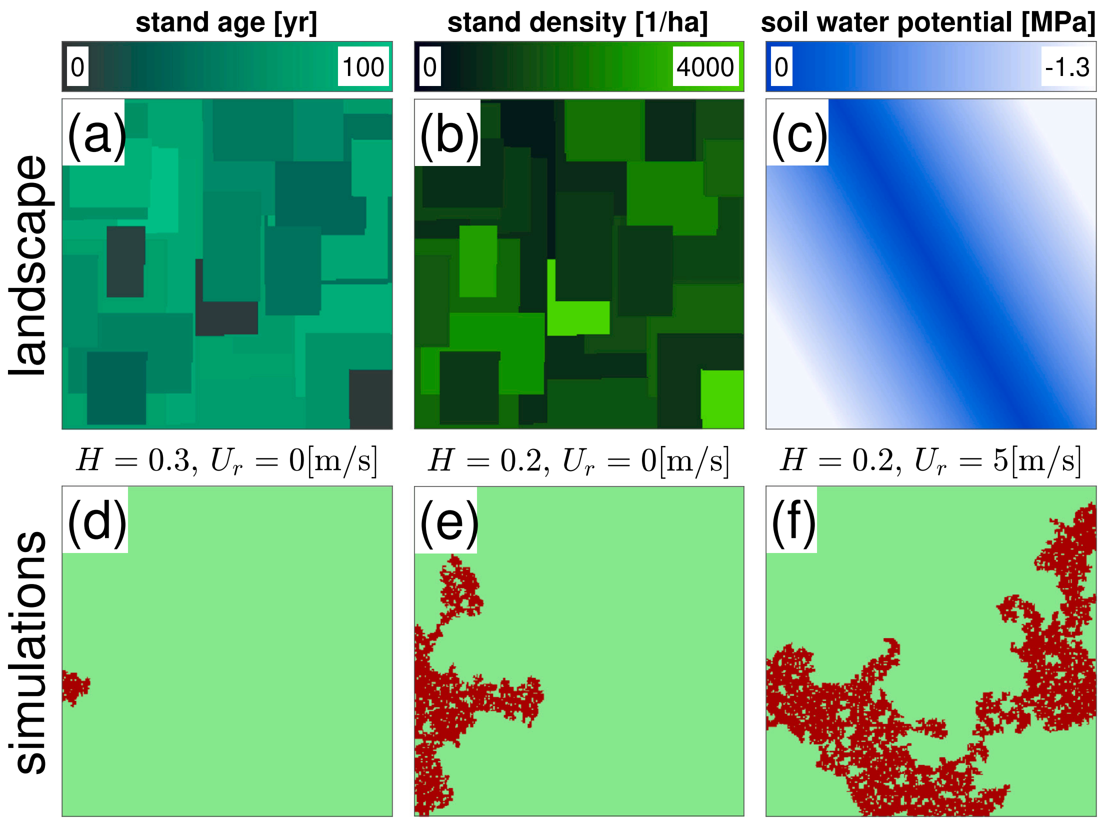

Fig. 3. Features of an example landscape (top) and simulated fire spread on this landscape under different conditions (bottom). (a,b) age and density of the different forest stands in the 5 × 5 [km^2] landscape, with average stand size of 100 [ha]. **(c)** soil water potential for non-augmented landscape, with a gradient of soil water conditions (from wetter, in dark blue, to drier, in white). **(d,e,f)** instances of fire spread through the landscape (burnt regions in red), under different conditions. Two levels of air relative humidity (H) are considered: very dry ($H = 0.2$ (e,f) and moderate ($H = 0.3$ (d)). Wind speed (U_r) is high in (f), and negligible in (d,e).

Table 2
Overview of simulation scenarios for Figs. 4–8.

Figure	Tree species	Relative Humidity (H)	Soil Water Potential (Ψ [MPa])
Fig. 4	Only conifers $P_1 = 1$	$H = 0.3$ (left) or $H = 0.2$ (right)	Uniform: $\Psi = -1.25$ or $\Psi = -0.05$
Fig. 5	Only conifers $P_1 = 1$	H=0.2	Varying $-0.05 > \Psi > -1.25$
Fig. 6	Only conifers ($P_1 = 1$) or mix 30% deciduous, i.e. $P_1 = 1$ & $P_1 = 0.7$	$H = 0.2$	Uniform: $\Psi = -1.25$
Fig. 7	Only conifers $P_1 = 1$	$H = 0.2$ (left) or $H = 0.3$ (right)	Varying ($-0.05 > \Psi > -1.25$) or Uniform ($\Psi = -1.25/-0.05$)
Fig. 8	Both, as for Fig. 6	Both ($H = 0.2, H = 0.3$)	Both, as for Fig. 7

is given by the weighted average of P_1 values from each species type relative to its abundance, consistent with empirical results (Hart et al., 2019). The age of each stand is extracted from a uniform distribution between 1 and 99 years. The stand density is determined based on the age, accounting for the variability in age-density relationship, due to the interactions of many factors, such as soil type, management history, and climatic history. Specifically, we set the baseline stand density D_0 based on stand age A, via a power-law ($D_0 = c_1 a^{c_2}$ with $c_1 = 2 \cdot 10^4; c_2 = -0.75$). The actual stand density, D is then determined as $D = 2^{\zeta} D_0$, where ζ is a random factor taken from a uniform random distribution between −1 and 1. See more details and derivation in SI B.

2.4.2. Landscape augmentation (ditching)

The shape of the non-augmented landscape is assumed to be such that a gradient of soil water potential (Ψ) emerges, with the highest moisture in the lowest areas of the landscape — a straight line that cuts through the landscape at a randomly chosen angle, always going through the center of the landscape (Fig. 3c). The result is an upside-down V shaped gradient of Ψ, with the most negative values (driest soils) at higher elevation at the edge of the landscape to least negative Ψ in the lower areas in the center of the landscape. Landscape augmentation (ditching) can substantially decrease the soil water potential gradient that naturally occurs in the landscape. For simplicity, we consider an idealized scenario where ditching leads to no gradient, i.e. a spatially uniform soil water potential. While actual ditching is implemented via discrete channels, often targeting wetter, low-lying areas (e.g. Ronkainen et al. (2013)), its net effect is to reduce moisture variation across the landscape — an effect we capture through this homogenized representation. We compare the non-augmented landscape with a ditched one, with a uniform soil water potential throughout the landscape. We consider two ditched scenarios: uniformly dry and uniformly wet. Uniformly dry, with $\Psi = -1.25$ [Pa], represents the ditched landscape under the same external conditions as non-augmented landscape. For comparison, we also consider a uniformly wet landscape,

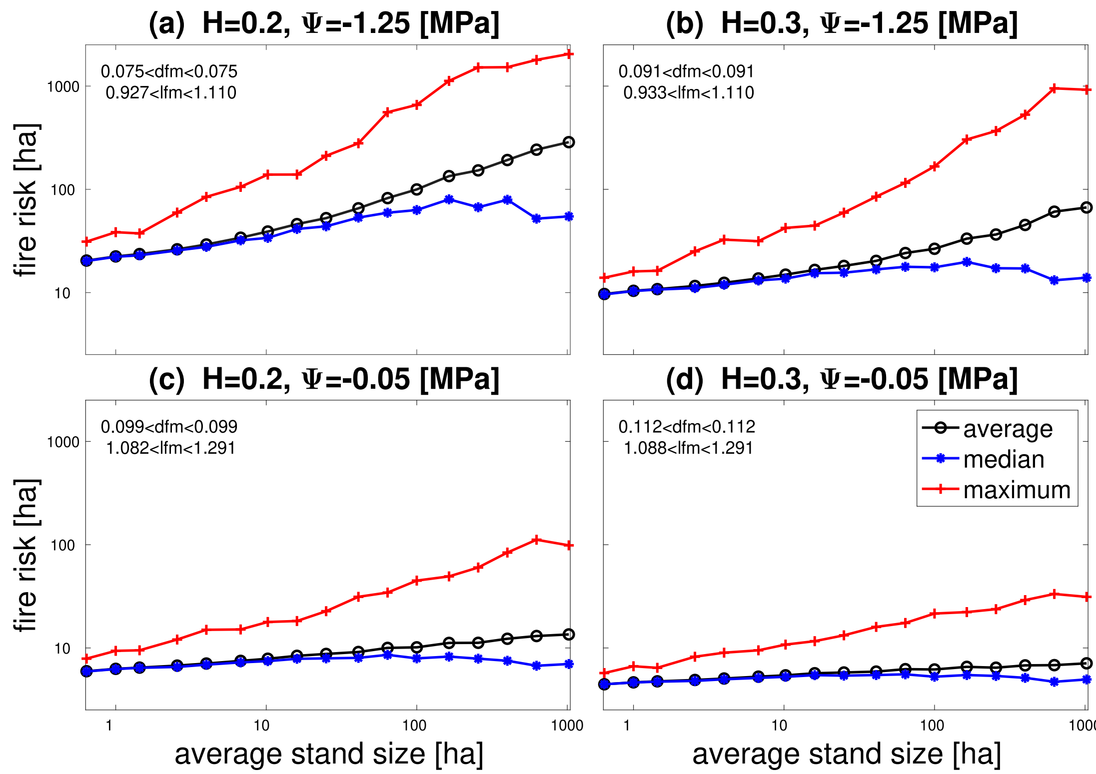

Fig. 4. Statistics of fire risk in different landscapes, for different air relative humidity H and soil water potential Ψ. Different curves show the average, median and maximum (in black, blue, and red, respectively) fire risk. Results are calculated over 500 landscapes with a given average stand area, and 500 fire simulations for each landscape. Left panels show fire risk for extremely dry conditions (relative humidity $H = 0.2$), while right panels for dry conditions ($H = 0.3$). Top (bottom) panels correspond to dry ($\Psi = -1.25$ [MPa]) and moist soil ($\Psi = -0.05$ [MPa]), respectively. Fire risk is calculated per landscape by averaging over the 500 fire spread simulations. Text in the top-right corner refers to the range of values of fuel moisture in the landscapes, both dead (dfm) and live (lfm).

with $\Psi = -0.05$ [Pa], representing moist landscape shortly after heavy rains.

2.4.3. Hydro-meteorological conditions

Unless otherwise noted, simulations have low air relative humidity $H = 0.2$ (dry conditions), temperature of $T = 20$ [^oC], and weak wind speed $U_r = 1$ [m/s]. Soil water potential is set to range between −0.05 and −1.25 [MPa] in the case of no ditching (non-augmented landscape), with the least negative potential along the channel. When soil water potential is assumed uniform due to ditching, both a wet and a dry cases are considered, corresponding to the two extreme values in the non-augmented landscape (−0.05 and −1.25 [MPa] respectively). Table 2 describes the differences in conditions between different simulations.

3. Results

Fire spread through the landscape depends on the prevailing conditions (wind, relative humidity, soil moisture) and landscape and stand attributes. For a given fire event in the landscape, lower fuel moisture, caused for example by low air humidity and/or low soil moisture, leads to much larger fires (Fig. 3d,e). Strong winds (Fig. 3f) allow the fires to penetrate into farther stands. Stands with very low stand age or stand density are less likely to burn (Fig. 3a,b).

Fire risk, measured as the average number of hectares that burn in a given landscape, generally increases with mean stand size (Fig. 4), for instance increasing over ten-fold when stand size changes from 1 to 1000 [ha] in very dry conditions (Fig. 4a). When the landscape comprises mainly small stands (left side of each panel), the fire risk is low, since the fire frequently encounters stands with low spread probability regardless of the specific spatial configuration (i.e. location and shape of each stand). Increasing stand size increases average (black), median (blue) and maximum (red) fire risk in a similar way, as fires are more likely to readily spread when encountering a large-sized stand that is more fire prone. At very large stand sizes, up to the entire landscape, the average and maximum fire risk further increase due to super-sized fires that engulf the landscape. However, the median fire risk decreases, since some landscapes consist of large stands that have very low fire risks (e.g. young stands). The general trend of increasing fire risk with average stand size persist with increasing fuel moisture (top to bottom row) and air humidity (left to right). The fire risk is overall markedly higher under drier atmospheric and soil conditions.

The differences between average, median, and maximum fire risk (Fig. 4) can be better understood by looking at the distribution of fire risk for each landscape (Fig. 5). For very large stand size (approximately 10^3 [ha]), the distribution is extremely wide, ranging from small fire risk, i.e. <1 [ha] to landscape-scale fire risk > 10^3 [ha]. The difference between average (solid vertical line) and the median (dotted line) is substantial (e.g. for the very large stand size there is almost an order of magnitude difference between average and median), since most landscapes have a small fire risk, but a few landscapes have very large fire risk. The average fire risk is dominated by these few landscapes of very large fire risk, while the median can be quite low since most landscapes have small fire risk. As average stand size is decreased, the distribution shifts to the left and becomes more constrained. The largest fire risk only reaches 200 [ha] for intermediate stand size and 50 [ha] for small average stand size (here 4 [ha]). Very small fire risks become unlikely due to the self-averaging effect that comes with landscapes being made up of many small stands. The distinction between many small fires and few very large fires is even more pronounced if we look at the distribution of fire size in itself (as opposed to fire risk, the average fire size per landscape; Fig. S16). If conditions are conducive to large fires, with low air humidity and large forest stands, the fire size distribution becomes bimodal, signaling a critical threshold — fires either engulf the landscape or quickly die out, with little middle ground.

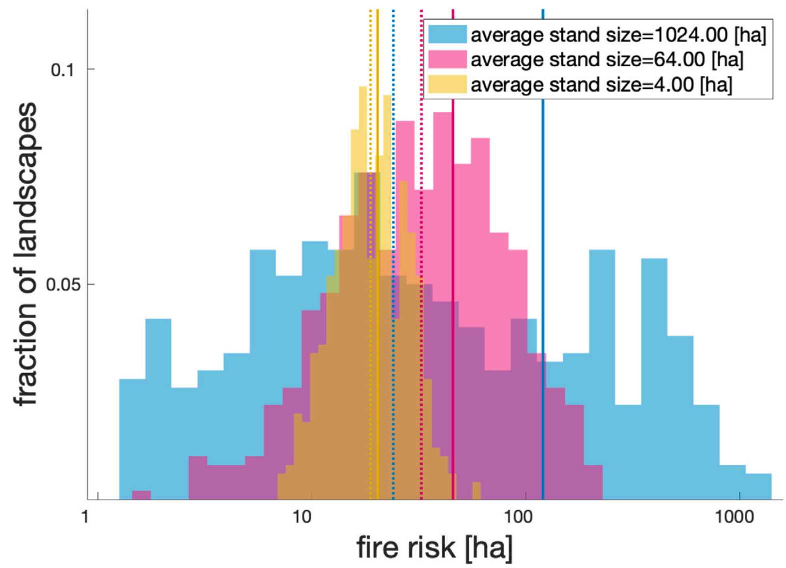

Fig. 5. Distributions of fire risk in different landscape configurations, with vertical lines showing the average (solid line) and median (dotted line) values of each distribution (note the log scale of the x-axis). Results are calculated over 500 landscapes with a given average stand area, and 500 fire simulations for each landscape, with extremely dry conditions (H = 0.2), and landscape that has not been ditched (soil water potential varying in range $-0.05 > \Psi > -1.25$ [MPa]). Colors correspond to 3 groups of landscape configurations, differing by their average stand size. The distributions show that several landscapes are responsible for a large fraction of burned area (so called fat tail), also leading to large divergence between average and median. This mainly occurs when average stand size is large (cyan), and that for smaller stands size (yellow) landscape variability is effectively smaller.

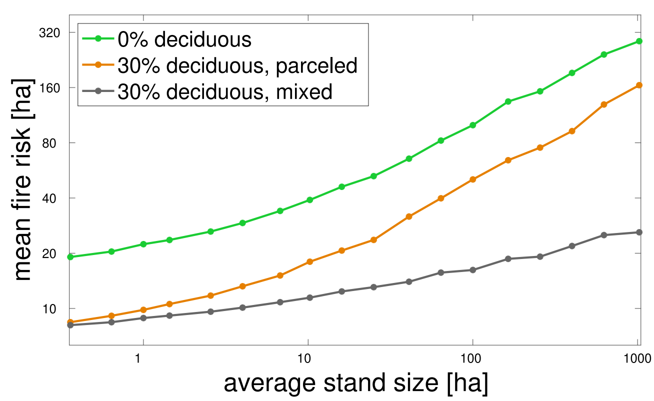

Fig. 6. Effect of introducing broadleaf-deciduous or mixed-species stands in the landscape. We compare the fire risk for all conifer landscape (green) with that of two landscapes where 30% of stands are broadleaf-deciduous, either as pure broadleaf-deciduous stands (orange) or trees mixed across all stands (gray). Results are calculated over 500 landscapes with a given average stand area, and 500 fire simulations for each landscape. We assume that for broadleaf-deciduous trees the baseline fire probability $P_1 = 0.7$ (as opposed to $P_1 = 1.0$ for conifers, see Fig. S10). Results represent conditions of low air humidity ($H = 0.2$) and uniformly dry soil ($\Psi = -1.25$ [MPa]).

Diversifying the landscape by introducing broadleaf-deciduous trees, which have a lower baseline fire probability P_1, decreases fire risk (Fig. 6). Replacing 30% of the conifers in each stand by broadleaf-deciduous species is comparable to reducing the stand size by an order of magnitude. When broadleaf-deciduous trees only appear in separate stands (compare orange with green lines), there is little interaction between stand size and stand type: there is a constant decrease in fire risk by roughly half due to 30% broadleaf-deciduous stands, regardless of the average stand size. This contrasts with having the same overall proportion of broadleaf-deciduous trees, but mixed throughout all the stands (gray line), which leads to dramatically lower fire risk when stand size is large (a decrease by a factor of 10 when stand size is 1000 [ha]).

Landscape augmentation, i.e. creating ditches to reduce soil moisture and thus both live and dead fuel moisture, leads to larger fires. The fire enhancement by ditching (compare red to blue lines in Fig. 7) is stronger when stand sizes are large (right side of each panel), in particular when air humidity is high (right panel), increasing fire risk by over a factor of 2 for a stand size larger than 500 [ha].

The three management options considered; A) planning smaller stands; B) avoiding ditching; C) including broadleaf-deciduous stands, can considerably lower fire risk in typical conditions (Fig. 8). All three options can limit the added fire risk caused by reduced air humidity (orange bars) — likely under climate change. Combinations of two or more options have compounded beneficial effects, and in all cases lead to lower fire risk even for reduced air humidity (blue bars).

4. Discussion

Fire spread in boreal forests is strongly influenced by external conditions (e.g. temperature, wind speed) that are beyond the control of forest managers. Soil moisture is also an important driver, in large part dependent on climatic conditions and soil texture, but also partially controllable via landscape ditching. Forest management decisions regarding stand size, species composition and ditching, can affect fire risk by a magnitude comparable with the variability in external conditions (Fig. 4, Fig. 6, Fig. 7, Fig. 8,). Mitigation actions to combat wildfires are thus possible. In particular, fire risk can be mitigated by increasing the spatial heterogeneity of the landscape to create natural firebreaks. While reserving parts of the landscape to act as fire breaks has often been enacted (Agee et al., 2000; Zong et al., 2021), our results show that forest management can similarly reduce fire risk *without* designating parts of the landscape for fire breaks. These management approaches are reducing stand size (Fig. 4), incorporating broadleaf-deciduous trees (Fig. 6), and minimizing landscape augmentation practices such as ditching (Fig. 7). By implementing these actions, fire risk can be reduced and the overall resilience of forest ecosystems enhanced to withstand the impacts of wildfires.

The first question we have explored is how fire risk is influenced by forest stand size. We find that decreasing forest stand size consistently reduces average fire risk, where a decrease of stand size from 100 [ha] to 1 [ha] leads to average fire risk reductions of up to a factor of five under very dry conditions (Fig. 4). Overall, the best strategy

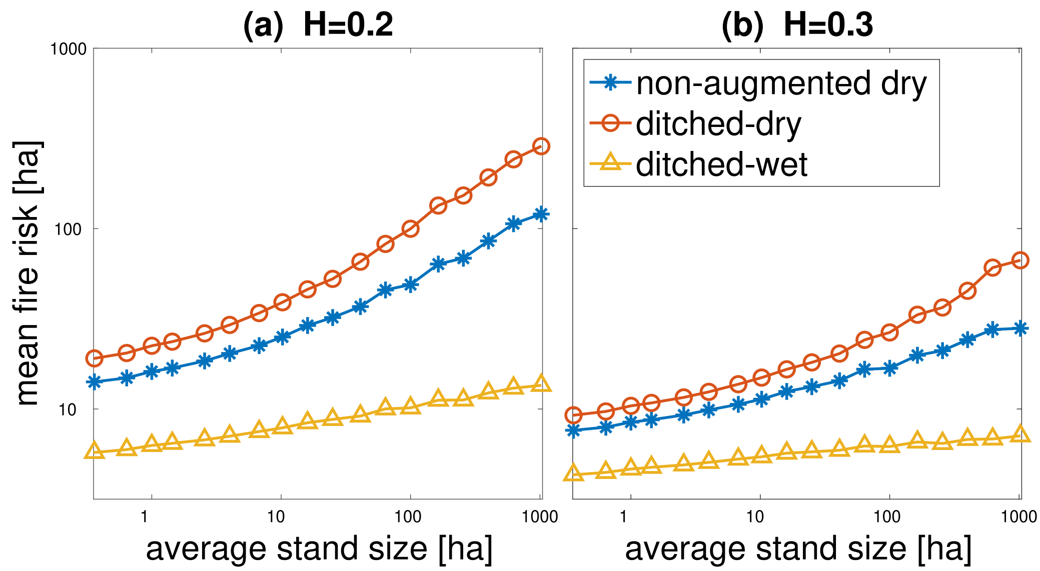

Fig. 7. Effect of landscape augmentation via ditching on fire dynamics. Axes, in logarithmic scale, correspond to the average stand size, and the fire risk (average fire size for a given landscape). Results are calculated over 500 landscapes with a given average stand area, and 500 fire simulations for each landscape. Blue asterisks line shows the fire spread when the landscape has not been ditched (soil water potential varying in range $-0.05 > \Psi > -1.25$ [MPa]), so that its center has the highest soil moisture leading to a higher dead and live fuel moisture. Red circles line shows the result of ditching, which keeps the soil moisture uniformly low throughout the landscape ($\Psi = -1.25$ [MPa]). Yellow triangles line is shown for reference, if the whole landscape has a uniformly high moisture ($\Psi = -0.05$ [MPa]). Panels correspond to relative humidity of $H = 0.2$ (left) and $H = 0.3$ (right).

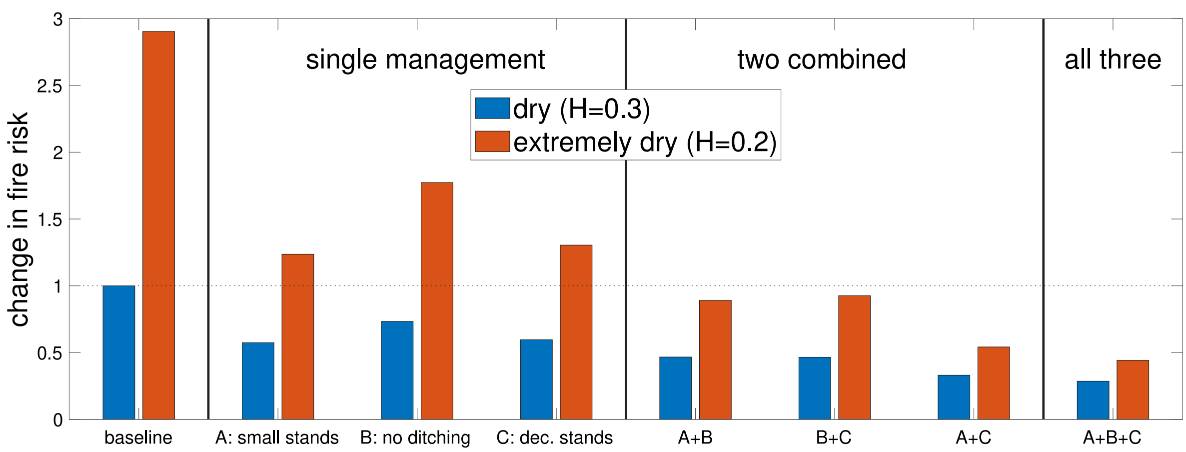

Fig. 8. Comparison of how management options and environmental conditions affect fire risk. Bars show the fire risk (average fire area burnt) for different conditions relative to baseline conditions of: low humidity (H=0.3), large forest stands (size of 25 [ha]), soil drained by ditching (uniform soil water potential of −1.25 [MPa]), and no broadleaf-deciduous trees (all conifers). Three management choices are: A) smaller stands (size of 1 [ha]), B) no ditching of soil (leading to gradient of soil water potential between −1.25 and −0.05 [MPa]) and C) planting some stands with broadleaf-deciduous trees (30% of stands). Red and blue bars correspond to low and very low values of air humidity (H=0.3 and H=0.2, respectively), with the very low values demonstrating potential effect of climate change.

from a fire mitigation perspective is to have large variability in stand attributes (age, density, etc.) across the landscape, i.e. small mean stand size. There are, however, inherent trade-offs in implementing these management strategies. A landscape consisting of small stands is generally harder and likely more costly to manage (DeLong, 2002). Similarly, refraining from ditching the landscape could prevent larger fires, but also cause slower tree growth if large parts of the landscape are often waterlogged. These potential costs could be balanced by other positive impact besides fire mitigation, such as higher biodiversity and resilience to pest outbreaks (Felton et al., 2023).

Several results shed light on our second question, of how other landscape features interact with stand size to affect fire dynamics. The interaction between the two forms of landscape management, stand size and ditching, is sublinear. Specifically, when the effects of small stand size are substantial, the additional effect of ditching is minimal (Fig. 7). This contrasts with the interaction of stand size and tree species (by having stands of pure broadleaf-deciduous trees), which does not show a similar sublinear interaction (Fig. 6, green vs. orange lines). However, planting broadleaf-deciduous trees throughout the forest can have a more substantial effect when stand size is large, possibly because it increases spatial heterogeneity also at very small scales (gray line). The exact effect of mixing tree type depends on the baseline fire spread probability P_1 for mixed stands (see Section 2.4.1), which remains uncharacterized. It was assumed for simplicity to be linear with forest composition, which is consistent with general trends previously reported (Ouarmim et al., 2016; Hart et al., 2019; Boulanger and Pascual Puigdevall, 2021). Despite this uncertainty, our results hint at the potential use of management at different spatial scales as a useful tool.

Fires are disturbances that interact with the environment in which they occur. This leads to critical behavior — small changes in conditions, e.g. drier fuel, can lead to exponentially larger fires (note logarithmic scale for fire size in Figs. 4–7). Ultimately counter-intuitive results emerge, such as the different response of average and median fire risk to average stand size (Fig. 4), due to the wide distributions of fire size (Fig. 5). Such diverging response can have practical implications for decision making. Which metric is most important when considering fire risk – the average, the median, or the maximum – depends on the management goals, the associated risk coping attitude, and the spatial and temporal scales considered by the management strategy. For instance, at the country scale, which comprises many landscapes such as the ones modeled here, the average fire size is likely the most important metric, because it takes into account the overall cost fires will have on a multitude of forest landscapes (and owners). In contrast, a stakeholder concerned with a specific landscape is likely more interested in the median fire risk, in particular if he/she is more risk averse and therefore not likely to accept the worst case scenario.

The last question we set out to answer was how effectively forest management practices can mitigate fire risk, and how these effects compare to changes in environmental conditions. Our results demonstrate that increasing the heterogeneity of managed forests, by managing for smaller stands (Fig. 4), refraining from soil–water manipulation via ditching (Fig. 7), and planting more broadleaf-deciduous trees (Fig. 6), can substantially mitigate the risks posed by fires. Combining two or more of these management strategies can further decrease fire risk (Fig. 8). In particular, such combinations can lower fire risk even if environmental conditions change drastically, for example if air humidity decreases, as expected due to climate change (orange bars in Fig. 8). While the potential for fire mitigation by diversifying tree species is well established (Schelhaas et al., 2010; Girardin and Terrier, 2015; Triviño et al., 2023), other possibilities, such as changing stand size or avoiding ditching to improve soil moisture conditions, have rarely been considered or analyzed. Our results highlight the potential for using a wider range of management actions for mitigating fire risk, and in particular combining several actions at once.

Both environmental conditions and societal norms can change much faster than the forest rotation cycle, making adaptive management difficult to implement effectively. Models can help us test such strategies, identify the most appropriate management strategy in the face of projected conditions, and explore inherent trade-offs. Our modeling approach combines a cellular-automata (CA) fire spread model with semi-mechanistic descriptions of the effects of external environmental conditions (weather), landscape and stand attributes, and their combined effects on fire spread. This framework is useful as it connects external conditions and management choices and fire risk in a coherent way. In particular, our approach is novel in that it includes several generally overlooked ecohydrological and eco-physiological processes in the fire spread model. Specifically, we explicitly linked dead fuel moisture to soil moisture, included a process based description of live fuel moisture, and considered the role of wind speed inside the canopy as opposed to above canopy.

Many fire modeling studies focus on precise predictions of fire spread and risk (Pais et al., 2021; Jones et al., 2025), for instance by using detailed mechanisms like elliptical fire spread (Pais et al., 2021) or relying only on easily available data for predictions but lacking a clear connection to environmental processes (Jones et al., 2025). While these models are a useful tool to predict fire behavior, their complexity and broader focus make them less suited for exploring the interactions between fire dynamics and management. For instance, while numerous studies show that more broadleaf-deciduous trees limits fire spread (Rupasinghe and Chow-Fraser, 2021; Jones et al., 2025), our study directly addresses this issue as a landscape management question, by comparing planting such trees to other possible management practices (e.g. smaller stands) and to environmental factors that managers cannot control (e.g. relative humidity). At the same time, we focus on management practices that are not solely implemented for fire prevention, such as fire breaks strategies (Zong et al., 2021). Similarly, while empirical work has shown that wet soils can limit fire spread (Hellberg et al., 2004; Forkel et al., 2012), modeling efforts have neither taken a mechanistic approach to this issue (Fang et al., 2018), nor addressed this as a question of management.

The model also has several limitations. We assume that fire spread is governed by the average tree and landscape attributes within each 20 [m] grid cell, that the attributes within a stand (comprising several cells) are constant, and that conditions throughout a fire are stable over time. While this allows us to model large-scale heterogeneity and its interactions with average soil and climate conditions, it does not capture small-scale fire dynamics (1–50 [m]), nor does it account for temporal changes in conditions like temperature and wind. Addressing these limitations is important when aiming at making precise predictions for specific landscapes and weather conditions, especially in areas with heterogeneity at scales smaller than the grid size. Furthermore, if fires are large and spread over long periods (e.g. several days), changes in conditions become important, and our simulations are not well suited to capture these dynamics — a limitation in regions where large fires are dominant, such as North America. We have also assumed that there is a unique relationship between tree age and stand density. These simplifications will not hold if a mixed-aged management is actively pursued. The same would apply to non-managed forests, such as are common in North America, where the interplay of tree mortality and natural regeneration can lead to high variability of tree age and size even within small distances, and tree species mixtures may be common (James et al., 2007; Kuuluvainen et al., 2021). Current knowledge on whether mixed-aged forests and mixed conifer-deciduous stands, are more or less susceptible to wild fire is very limited (Felton et al., 2023), making also its modeling highly uncertain.

A recurring issue when accounting for landscape structure and environmental conditions in fire dynamics is the difficulty of finding consistent relationships or semi-mechanistic models that capture their effects on fire spread. For example, the dependence of fire spread on tree density and age (Fig. 2a,b) is based on anecdotal data, where processes leading to the observed patterns are hard to tease apart, and the data is insufficient for precise parameter estimation. Even frequently used relationships, such as fire spread dependence on fuel moisture (most notability, the classic function by Rothermel (Rothermel, 1972); red curve in Fig. 2e) are based on a limited dataset, with little clarity on how small-scale experiments scale up to patterns observed in the landscape.

Moreover, there is still limited understanding of how environmental conditions, such as soil moisture, affect fire spread via dead vs. live fuel moisture. In general, it remains unclear how dead and live fuel moisture together determine fire spread (Rossa and Fernandes, 2017). Here we opted to use a single parameter (σ) to describe their relative contribution. Our qualitative results are not much impacted by our choice of σ (Fig. S12, S14), although lower σ values are associated with much higher fire risk, given that live fuels have much higher moisture than dead fuels. This clearly demonstrates that proper quantitative predictions require a robust parameterization of σ (Fig. S13) and hence understanding the relative role of dead and live fuel. The effect of ditching on fire dynamics is also largely absent in the literature, with only minimal work in this direction focusing on peatlands and the potential of dry peat to burn (Lohmus et al., 2015). We have included this effect in a simplified way as altered spatial patterns of soil moisture, with the purpose of quantifying the potential role of drainage on fire spread. Further empirical and modeling studies should focus on quantifying how and to which extent dead and live fuel moisture depends on local soil moisture affected by landscape topography (Seibert et al., 2007) and ditching (Rothwell et al., 1996). It is also necessary to test fire model results against data of how fires spread over landscapes with significant soil moisture heterogeneity (Fig. S11). Given that soil moisture levels are projected to decline under future climate scenarios (Wang et al., 2014), understanding these links becomes even more critical for anticipating changes in fire behavior and informing management strategies.

Finally, many of the parameters used, which influence fire behavior and risk, are difficult to estimate and remain uncertain. The effect of stand properties on fire spread (e.g. ρ_a and ρ_d) is not well characterized, and it remains unclear how important live fuel is in determining fire spread (i.e. how to estimate σ). Further, even often-used concepts in wildfire research, such as the typical drying time for dead fuel (τ), are difficult to scale up to landscape models, where different fuel classes coexists, and environmental conditions (e.g. shading and wind speed) can play a role. Fully considering these aspects would require incorporating the relevant physical processes. However, sensitivity analyses (Figs. S12, S13, S14, S15, S17) suggest that our results are robust to key parameter choices, even if quantitative predictions can be less precise.

Our approach can help understand how future environmental conditions and their variability could impact fire spread patterns over time, and identify which mechanisms controlling fire risk are most dominant under different conditions. For fire management agencies, our modeling approach offers a tool for predicting fire behavior and devising efficient strategies for fire suppression and containment. Simulating different fire scenarios in realistic landscapes would enable agencies to better prepare for potential fire events and allocate resources effectively. Insurance companies can benefit by using them to assess and quantify fire risks in specific regions under different climatic scenarios. Overall, the versatility of our modeling approach can support decision-support tools in addressing fire-related challenges and enhancing preparedness and resilience in the face of a changing climate.

CRediT authorship contribution statement

Yuval R. Zelnik: Writing – original draft, Software, Methodology, Conceptualization. **Samuli Launiainen:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Giulia Vico:** Writing – review & editing, Software, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare no competing interests.

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Appendix A. Supplementary data

Supplementary material related to this article can be found online at https://doi.org/10.1016/j.ecolmodel.2025.111222.

Data availability

All script files are accessible at the open repository: https://zenodo.org/records/15585543.

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