Temporal trends in the protective capacity of burnt beech forests (Fagus sylvatica L.) against rockfall

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Temporal trends in the protective capacity of burnt beech forests (*Fagus sylvatica* L.) against rockfall

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Abstract

Beech (*Fagus sylvatica* L.) forests covering relief rich terrain often provide direct protection for humans and their assets against rockfall. However, the efficacy in protecting against such hazards may abruptly and substantially change after disturbances such as fires, wind-throws, avalanches and insect outbreaks. To date, little knowledge exists on the mid-term protective capacity against rockfall of fire-injured beech stands. We selected 39 beech stands in the Southern European Alps that burnt with different severities over the last 40 years. We inventoried all living and dead trees in each stand and subsequently applied the rockfall model Rockfor.net to assess the protective capacity of fire-injured forests against falling rocks with volumes of 0.05, 0.2 and 1 m^3. We tested forested slopes with mean gradients of 27°, 30°, and 35° and lengths of 75 and 150 m.

Burnt beech forests hit by low severity fires provide nearly similar protective capacity as unburnt forests, because only thin fire-injured trees die while intermediate-sized and tall trees mostly survive. The protective capacity of moderate to high severity sites is significantly reduced, especially between 10 and 30 years after the fire. In those cases, silvicultural or technical measures may be necessary. Beside the installation of rockfall nets or dams, small-scale felling of dying trees and the placement of stems in oblique direction to the slope can mitigate the reduction of protection provided by the forest.

Keywords: forest fires, stand structure, burn severity, Rockfor.net

1 Introduction

In mountain regions forests often provide a direct protection for humans and for their assets against natural hazards (Dorren et al. 2005a; Brang et al. 2006). In comparison to man-built structures the protective effect of forests is naturally re-growing and relatively cost-efficient (Olschewski et al. 2012). In case of rockfall events, standing and lying trees act as barriers against falling rocks (Motta and Haudemand 2000) and the understory vegetation increases the surface uneveness that also may contribute to the energy dissipative capacity of a forest stand (Dorren et al. 2004b; Brauner et al. 2005). Whether the protection provided by a particular forest stand is effective or not is mainly determined by: (1) terrain characteristics and the total length of the forested part of a slope between the rockfall release area and the area to be protected, (2) the size and kinetic energy of the falling rock, and (3) the basal area of the forest and dendrometrical characteristics that reduce or adsorb the impact energy of falling rocks (Dorren et al. 2015).

Since forests are dynamic ecosystems, their protective capacity changes constantly. In particular, natural disturbances such as forest fires, wind-throws, insect and pest outbreaks and snow avalanches have the potential to abruptly and substantially reduce the protective capacity of the concerned stands. Their influence on the protective capacity highly depends on (1) the intensity and scale of the disturbance, (2) the resistance and resilience of the disturbed stand, and (3) on the post-disturbance management (Bebi et al. 2015). For instance, insect outbreaks or low intensity wind-throw causes dispersed tree damages that increases light- and nutrient availability to favour the pre-regeneration (Kupferschmid Albisetti 2003; Collet et al. 2008; Kramer et al. 2014). In case of an immediately and comprehensive loss of living trees after the disturbance event, remnant dead wood may significantly decrease terrain patency and may thus at least partly compensate for this loss. However, slow succession rates after a disturbance event and relatively fast decay of dead wood may result in a time window of temporarily reduced protection against natural hazards (Bebi et al. 2015).

Fire affects both the pre-fire regeneration and the dead wood structure (Wohlgemuth et al. 2010), which may additionally reduce the protective capacity of burnt forests with respect to wind-throw areas. Unfortunately, to date little is known about fire resistance and post-fire resilience of different forest types with potentially important protection functions. This is particularly true for European beech (*Fagus sylvatica* L.) forests, an often used tree species in the protection against rockfall (Perzl 2009; Schmidt 2005). In the Swiss Alps, beech forests hold a share of 16% on the overall protection forests against rockfall (Brändli and Huber 2015).

However, recent studies demonstrated that fire-injured beeches generally collapse within first 20 years post-fire due to a lack in protection from heating by its thin bark and subsequent infections by wood decaying fungi (Maringer et al. subm. a). Within the same period, seed germination and seedlings emergence is enhanced by progressive canopy opening and by the removal of thick litter layers (Ascoli et al. 2015; Maringer et al. subm.). Both processes highly depend on the fire severity (i.e. immediate effect of fire; cf. Morgan et al., 2014). In case of very severe fires, most beeches die within the first few post-fire seasons. Due to the immediate collapse of seed providing trees, seed production and seedlings emergence may be hindered. Additionally, fast growing early post-fire colonizers like shrubs and ferns tend to build dense layers inhibiting additionally seedlings emergence (Maringer et al. subm.). Contrasting, after low severe fires only a few individuals (and usually small trees) are critically injured with marginal consequences to the stand dynamic. Fires of intermediate severity cause a progressive dieback of the stand according to the proportion of the bole injured and the proliferation of decaying fungi (Conedera et al. 2007; Conedera et al. 2010; Maringer et al. subm. a). Here the probabilities of successful seed germination and seedlings emergence are highest, especially when a mast year immediately follows the fire event (Ascoli et al. 2015). Those post-fire processes in beech forests show that there might be a lack in the forest protective capacity; particularly in moderate and high fire severity stands. It is thus crucial for foresters to know about the post disturbance processes and their influence in order to prevent the related risks.

Based on the assumption that the energy release by moving rocks is compensated by either rock-soil contact (Zinggeler et al. 1991), rock-tree contact (Berger and Dorren 2007), or both, process orientated models are able to assess the protective capacity of a concerned stand. In the present study we employed the rockfall model Rockfor.net (Berger and Dorren 2007) for quantifying the protective capacity of burnt beech forests. The model was originally developed to quickly quantify the protective capacity of different structured forest stands and has been often applied in the European Alps (Berger and Dorren 2007; Wehrli et al. 2006; Kajdiž et al. 2015). We used a dataset of 39 burnt beech stands differing in terms of years post-fire (2 to 40 years) and burn severity (burn severity refers to the long-term fire effects; cf. Morgan et al. 2014). In particular, we evaluated the conditions (rock size, forested slope length, slope inclination, burn severity) and post-fire phases under which deficits may be expected in the protective capacity against rockfall.

2 Materials and methods

2.1 Study area

The study was conducted in the Southern European Alps across the neighboring regions of Canton Ticino (Switzerland) and Piedmont (Italy). The area is characterized by a marked elevation gradient ranging from the Lake Maggiore (197 m a.s.l.) to the Adula Peak (3402 m a.s.l.) in Ticino and to Punta Nordend (4609 m a.s.l.) in Piedmont, respectively. The geology is characterized by the tectonics of the Alps with granit and gneiss dominating the bedrock (Pfiffner 2015). Due to the relief rich terrain, rockfalls are one of the major natural hazards threatening mountain settlements and roads in both regions (Regione Autonoma Valle d'Aosta - Regione Piemonte 2011; Ambrosi and Thüring 2005).

The regional climate can be described as warm and humid showing a high annual precipitation gradient ranging from 778 mm in Piedmont (climate station Susa: 07°3`0``E, 45°08`0``N) to 1897 mm in Ticino (climate station Locarno Monti: 08°47`43``E, 46°10`12``N) (ARPA 2015; MeteoSwiss 2015). More than half of the annual precipitation falls during the transition seasons (April-May and September-November), and in winter (December-March) precipitation is particularly low (162 mm for Piedmont, 316 mm for Ticino). Winters are generally mild with mean January temperatures around 3.5°C, and summers are warm with mean July temperatures around 21.7°C. In summer, periods without rain may last up to thirty consecutive days (Isotta et al. 2014), whereas in winter a katabatic warm and dry wind from the northern Alps (*north foehn*) drops the relatively humidity below 20% in average on 40 days yr^{-1} (Spinedi and Isotta 2005). These north-foehn winds dry the fine fuel of the forest understory and increase the fire danger. Forest fires are mostly of human origin and consist of surface fires in the understory of the deciduous forests. Those fires usually start from the urban-forest interface (Conedera et al. 2015) and spread into the higher elevated beech belt (900-1500m a.s.l.) mostly during prolonged dry conditions (Pezzatti et al. 2009). Fire in the region of Piedmont yearly affects 1.7% of the beech protection forests (Regione Autonoma Valle d`Aosta – Regione Piemonte 2011).

2.2 Selection of fire sites and data collection

Fire perimeters with less than 40 years were selected from the forest fire databases of Switzerland (Pezzatti et al. 2010) and of the State Forestry Corps of Italy (Inventorio nazionale delle foreste e dei serbatoi di Carbonio (INFC 2005), Corpo Forestale dello stato – ispettorato generale). They were overlaid with local vegetation (Ceschi 2006; Camerano et al. 2004) and geological maps in a geographical information system (ArcGIS version 10.0; © ESRI) to identify fires in beech stands on crystalline bedrock. First field observation took place in 2011 to indicate potential study sites: (i) larger than 0.25 ha, (ii) with no signs of pre-fire pasture or post-fire artificial plantation, (iii) and dominated by beech (> 95%) before the fire event. From the initial 94 potential sites, 36 satisfied all of the selection criteria and were retained for the field survey in the years 2012 and 2013 (Appendix 1).

Depending on the area burnt, we placed one to three transects, spaced 50 m apart in elevation, from the unburnt to the burnt beech forests (Figure 1). Circular plots of 200 m^2 were placed regularly with 30 m distances in between starting in 10 m distance to the burn edge and following the contour lines. Whenever possible, a minimum of one control plot was placed in the unburnt beech forests at 20 m distance to the burn edge.

[**Fig. 1** Sampling design in a burnt and unburnt beech forests with regularly placed circular 200 m^2 plots placed 30 m apart along horizontal transects (figure left). Each plot is further characterized in terms of burn severity as a function of the portion of dead and living beeches (photographs)]

Data collection followed guidelines of the Swiss National Forest Inventory (NFI; Keller 2005) with specific focus on stand stability parameters (Herold and Ulmer 2001). Therefore, general plot characteristics were surveyed like slope [°], aspect, elevation [m a.s.l.], mircorelief (plane, convex, depression), as well as the cover of inhibitors for emerging regeneration such as common bracken (*Pteridium aquilinum* [L.] KUHN), common broom (*Cytisus scoparius* [L.] LINK), purple moor grass (*Molinia arundinacea* SCHANK), as well as the surface roughness in the form of deposited rocks (see Brauner et al. 2005). The coverages of common bracken, common broom and purple moor grass were summed up per plot (hereafter referred to as cover of early post-fire colonizers).

We inventoried all trees with diameter to breast height (DBH) ≥ 8 cm and omitted smaller trees because of their negligible role in the protective effectiveness (Wehrli et al. 2006). Each standing tree was identified down to the species level (Wagner et al. 2010) and the following characteristics were recorded: vitality, i.e., tree being alive or dead (snags and dead standing tree with crown portions but without visible green foliage, hereafter referred to as snags), DBH (at 1.30 m to the nearest cm), tree height (to the nearest meter), and the percentage of crown volume killed. The latter was visually estimated by the volumetric proportion of crown killed compared to the space occupied by the pre-fire crown volume (Hood et al., 2007). Data collection further included lying dead trees (hereafter referred to as logs) of which the average diameter and the length were recorded. For both snags and logs, the wood decay stage was recorded in four classes: (1) cambium still fresh, (2) knife penetrates low, cambium disappeared, (3) knife penetrates into the fiber direction, but not transversely or (4) knife penetrates in both directions. Lying branches and brushwood originated from falling crowns of dead trees with a decay stage below 4 were assessed after the method of Brown (1974). Pieces in the 200 m^2-plots were recorded in different diameter classes (1: 2.5-5 cm, 2: >5-7.5 cm, 3: >7.5-15 cm, 4: >15-30 cm) along the four cardinal directions. The obtained volume was then scaled up to standard hectare values (m^3 ha^{-1}).

In regions with such a relief rich terrain fires burn very heterogeneously. Therefore each plot was categorized in low, moderate and high burn severity. In accordance with a parallel study by Maringer et al. (subm.), we assessed burn severity by calculating the ratio of post-fire and pre-fire basal area of living trees. For fire sites older than 10 years, pre-fire conditions were assessed exclusively from the control plots, because of fast decaying dead wood. Whereas in burnt sites younger than 10 years, the number of visible dead trees in burnt plots determined the pre-fire stand characteristics. Based on this assumption, we defined low burn severity in plots with less than 5% crown volume loss and less than 20% basal area loss. High burn severity was indicated by extensive crown loss (> 50%) and basal area killed (> 60%), and all plots with intermediate losses— in terms of crown and basal area— were assigned to the moderate severity class.

2.3 Analysis methods

2.3.1 The Rockfor.net model

We employed the Rockfor.net model developed by Berger and Dorren (2007) for simulating the temporal trends in the protective capacity against rockfall in fire-injured beech stands.

The underlying idea of the model is to compare the theoretical basal area required for absorbing the kinetic energy of downhill moving rocks ($G_{required}$) and the available basal area of a particular forest stand ($G_{available}$). Therefore, the model regards all standing trees distributed in a forest as virtual tree lines parallel to the contour lines. All trees have the same species composition and diameters (weighting of the tree species see Dorren and Berger 2005), representing the mean values in the original forest stand. The model starts by calculating the total kinetic energy developed by a rock falling down the slope. Then it calculates the energy dissipative capacity of each tree line. The number of trees required to dissipate all kinetic energy are subsequently converted in a required basal area ($G_{required}$) using the mean DBH. In the last step the Rockfor.net model quantifies the protective effect of a forest stand by comparing the required theoretical $G_{required}$ with the available $G_{available}$ (see Berger and Dorren 2007 for more details).

In the Rockfor.net model we considered also the contribution of logs, assuming that their capacity of absorbing kinetic energy is proportional to the ratio between log-diameter and rock size. Olmedo-Manich (2015) demonstrated that deposited tree logs with rock/log diameter ratios between 0.8 and 1.55 favour optimal energy loss. In this study we assumed that energy dissipation efficiency is linearly related to the rock/log diameter ratio. The amount of lying logs was estimated in terms of volume (in $m^3\ ha^{-1}$) in the field. In our tool, this volume was converted into a total log length per hectare and finally into the number of potential logs impacts per hectare. Here we assumed that an efficient rock-log contact, meaning with a rock/log diameter ratio of 1 or smaller, is required every 10 m on a slope length of 100 m to stop 100% of the rocks by logs (see also Dorren et al 2015). The following equation was used to calculate the percentage of rocks stopped by logs ($\%R_{stopped}$):

$$\%R_{stopped} = Eff_{contact} \times Vol_{Log} \div (\pi \times \left(\frac{D_t}{2}^2\right)) \div 100m \div 10 \times 100\% \qquad \text{eq. (1)}$$

Where,

$Eff_{contact}$ = rock-log contact efficiency = min[1, D_t / Db]

D_t = tree diameter (in m)

D_b = rock diameter (in m)

Vol_{Log} = volume of lying logs (in m^3 ha^{-1})

The contribution of lying branches and brushwood to rockfall energy dissipation is hard to quantify in a model such as the Rockfor.net and was therefore neglected. Therefore, temporal changes of their volumes were only graphically visualized (see Figures 3-4).

In sum, the Rockfor.net model requires as input parameters both site and forest stand characteristics. Required site characteristics are cliff height (m), length of both the forested and unforested slope on the trajectory of a fallen rock, and mean slope inclination (°). Species composition, DBHs and densities of standing trees (including snags) as well as diameter and length of the logs (wood decomposition rate below 4) are required as stand characteristics.

2.3.2 Input data preparation and scenario specification

Data preparation followed the new rockfall protection guidelines of the "Sustainability and success monitoring in the protection forests of Switzerland (NaiS)" (see Frehner et al. 2005 and Dorren et al. 2015). Tree diameters were grouped in four DBH-classes (8-12 cm, 12-24 cm, 24-36 cm, and ≥ 36 cm) separately for living and dead standing trees and standardized to number of stems per hectare. Trees with large DBH values diameter most effectively dissipate the kinetic energy of falling rocks, especially those of large rocks, whereas small trees significantly increase the probability of rock—tree contacts due to the (generally) large stem densities. Therefore, the required basal area ($G_{required}$) to stop a falling rock within a specific forested slope is weighted for the DBH-classes according to the rock size

(Dorren et al. 2015). Moreover, to account for the differences in capacity of different tree types to dissipate the kinetic energy of falling rocks, Rockfor.net converted the proportions of the presence of 5 different tree 'types' in each stand into a mean energy dissipative capacity per study site. The following 5 tree 'types' were taken into account: beech, Norway spruce (*Picea abies* [L.] Karst.), silver fir (*Abies alba* Mill.), other broadleaves, and other conifers (cf. Dorren and Berger 2005).

Further we used standardized rock sizes, mean slope gradients, cliff heights and lengths of forested slopes. We defined standard rock volumes (0.05 m^3, 0.2 m^3, and 1 m^3, which corresponds to the rock diameters 0.37 m, 0.58 m and 1 m; Table 1) as traditionally used in NaiS (Frehner et al. 2005, Dorren et al. 2015). In order to simulate realistic field conditions, two options of horizontal distances (75 m, 150 m) have been defined in which a rock had to be stopped from the bottom of a cliff to the downslope forest edge. Finally, three different slope gradients were considered representing the 1st (27°) and 3rd quantile (35°), as well as at the mean (30°) of the slope distribution from the surveyed plots (Table 1). Slope inclination was standardized after testing the statistical non-significance between tree stem densities and slopes using a mixed effect model (Appendix 2).

The estimation of the protective effect as calculated by the Rockfor.net model represents the probability of a rock to be stopped in the stand, which is expressed in the following categories: ≥ 90% very good protection, 75 - 90% good protection, 50 - 75% adequate protection, 25 - 50% moderate protection, and < 25% inadequate protection. Whether or not the level of protection provided by a forest stand is sufficient, can only be determined by means of a risk analysis in which the effective risk reduction of the forest is quantified and is therefore out of the scope of the present paper.

Table 1: Scenario specification for the Rockfor.net model

Input parameters	Scenario specification					
Cliff height (m)	20					
NFS[1] (m)	0					
Rock density (kg m^{-3})	2800					
Forested slope length (m)	75			150		
Mean slope inclination (°)	27	30	35	27	30	35
Mean rock volume (m^3)	0.05	0.05	0.05	0.05	0.05	0.05
	0.2	0.2	0.2	0.2	0.2	0.2
	1	1	1	1	1	1

[1] NFS: Non forested slope length between the foot of the cliff and the upper forest limit

2.3.3 Analysis of the modeled results

The protective capacity for each scenario was given as the sum of rocks stopped by standing trees (living and dead) and for logs at the plot-level. The result was set to 100% in case the sum exceeded the 100% mark. In order to assess the temporal post-fire evolution of the protective capacity, the results were plotted against the time since burning and visualized using standard loess-smoothing curves (Chambers and Hastie 1992) separately for low, moderate and high severity sites. The corresponding unburnt forests served as reference. Significant temporal trends in those smoothing curves were detected by employing linear regression models with protective capacity as response variable and the number of post-fire years as explanatory variable. Since the protective capacity is expressed as percentage (probability), the data was log-transformed ($y` = \log(\frac{y}{1-y})$) and the numbers of post-fire years were included as linear and quadratic term. Additionally, Mann-Whitney-Wilcox tests were applied in each of the calculated scenario for detecting significant differences in distributions of the forest protective capacity in different burn severity sites and the corresponding unburnt forests.

All analyses of the modeled results and the regression models were performed using R, the free software environment for statistical computing (R Development Core Team 2014). Negative binomial logistic regression models were fitted and validated using the glmmADMB package (Bolker et al.). Graphical outputs are mainly based on packages lattice (Deepayan 2008) and ggplot2 (Wickham and Chang 2015).

3 Results

3.1 Forest characteristics and development after fire

We assessed a total number of 189 plots in burnt and 27 plots in unburnt (control plots) beech dominated forests. Most of the burnt plots were classified as moderate (44.2%) and high (40.3%) (burn) severity sites, whereas only the remaining 15.5% were considered as low burn severity sites. Elevation of the fire sites and the corresponding unburnt forests ranged from 700 to 1486 m a.s.l. with mean slope inclinations of 30 ± 0.34°.

Beech grew frequently in the burnt forests, with percentages ranging from 20.75% to 100% (Appendix 1). The overall average tree height was 10.3 ± 0.11 m, and approximately 2 m higher when referring to living trees only. Average tree density was 227.6 ± 14.4 stems ha^{-1} with a decreasing tendency from low (360.5 stems ha^{-1}) to high (235.7 stems ha^{-1}) burn severity sites. Temporal patterns in tree densities (DBH ≥ 8 cm) followed a parabolic course, showing denser stands in early and late post-fire stages and a minimum between 10 and 20 years post-fire (Figure 2).

In low severity sites younger than 15 years post-fire, tree densities were only slightly lower than in the unburnt forests (Figure 2). Whereas the average basal area at the minimum (around 16-20 years post-fire) was only 1.5-times less than the ones recorded at the early (≤ 9 years) and late (> 32 years) post-fire stages. Only few thin (DBH < 12 cm) trees died, and densities of intermediate-sized (DBH 12-36 cm) and large (DBH > 36 cm) trees remained constant throughout the post-fire period of 40 years.

In moderate severity sites, tree densities decreased by half of the densities recorded for early and late post-fire stages, whereas the basal area depression lasted for 20 years between 10 and 32 years post-fire (Figure 2). Intermediate-sized trees dominated within the first decade post-fire while their densities rapidly decreased with a minimum by about 10 to 32 years post-fire. Thin and intermediate-sized trees increased in densities 32 years post-fire, and tall trees were present throughout the whole observation period.

In comparison to low and moderate burn severity, tree densities in high severity sites rapidly decreased throughout all DBH classes within the first decade post-fire and dropped by a factor of 2.3 from 10 to 20 years post-fire (Figure 2). After 20 years post-fire, the new regeneration characterized by thin (DBH < 24 cm in particular) trees increased and their densities doubled with each post-fire age class, peaking after 32 years post-fire.

[**Fig. 2** Tree densities (DBH ≥ 8 cm) for living and dead (shaded bars) trees in different DBH-classes (grey color gradient) for low, moderate and high (burn) severity sites and the corresponding unburnt beech forests, grouped by years post-fire]

3.2 Surface unevenness

Most burnt plots were located on a plane (46%) surface followed by small depressions

(31%), and convex (23%) microrelief. The average coverage of rocks in a burnt plot was 2%, ranging from zero to maximum 30%. Early post-fire colonizers grew frequently after fires of moderate and high burn severity. They reached average coverages of 28% in moderate and 56% in high severity sites (Figure 3). Over the years post-fire, they increased in coverage within the first decade post-fire and peaked (~30%) by around 20 years post-fire in moderate severity sites. In high severity sites they reached a maximum coverage (~ 60%) after 30 years post-fire. This contrasts to plots burnt of low burn severity, where early post-fire colonizers never exceeded 25%. There was no clear temporal tendency, which was similar to the pattern of early post-fire colonizers in the unburnt plots. Here coverages tended to be close to zero.

[**Fig. 3** Temporal trends for the cover of early post-fire colonizers (sum of *Pteridium aquilinum*, *Cytisus scoparius*, *Molinia arundinacea*) visualized by loess-smoothing curves (black dotted lines) including confidence intervals (grey) for the different burn severity classes and the corresponding unburnt forests]

Pattern in the volume of lying dead branches and brushwood were similar in the different burn severity sites with peaks at around 15 years post-fire (Figure 4). Afterwards volumes steadily decreased reaching similar values recorded for the unburnt forests. When considering different burn severities, the volume of lying branches and brushwood scored highest average values (106 m^3) in high severity sites; here it was 1.5-times higher than in moderate (75 m^3) and low (60 m^3) severity sites, respectively. Contrastingly, no clear temporal trend was detected in the unburnt forests where volumes of lying branches and brushwood never exceeded 25 m^3 ha^{-1}.

[**Fig. 4** Temporal trends in the volumes [m^3 ha^{-1}] of lying dead branches and brushwood visualized by loess-smoothing curves (black dotted lines) including confidence intervals (grey) for the different burn severity classes and the corresponding unburnt forests]

3.3 Temporal trends in the protective capacity of forests

The Rockfor.net model results highlight the mid-term (first 40 years post-fire) evolution of the protective capacity of burnt beech stands as a function of different burn severities, rock sizes, forested slope lengths, and slope inclinations. The average protective capacity aggregated over the years post-fire decreased with increasing rock size, slope inclination, and shortness of the forested slope length (Table 2). The protective capacity of low severity sites did not significantly differ from the unburnt forests for most of the scenarios. However, for moderate and high burn severity sites the protective capacity significantly differed from the unburnt forests in more than half (67%) of the scenarios (Table 2).

Table 2: Mean protection capacity [%] for the different scenario specifications grouped by low, moderate and high burn severity and the corresponding unburnt forests. Similarities (Mann-Whitney-Wilcox tests) in the protection capacity between unburnt and burnt forests of different severities are shown in the superscript.

Forested slope length			75 m			150 m	
Mean slope gradient		27°	30°	35°	27°	30°	35°
rock size	burn severity		Mean protective capacity [%]				
$0.05\ m^3$	unburnt	97	95	91	95	95	95
	low	96 $^{(ns)}$	92 $^{(ns)}$	87 $^{(ns)}$	92 $^{(ns)}$	92 $^{(ns)}$	92 $^{(ns)}$
	moderate	89 $^{(ns)}$	85 $^{(*)}$	76 $^{(**)}$	88 $^{(*)}$	87 $^{(*)}$	87 $^{(*)}$
	high	73 $^{(*)}$	68 $^{(**)}$	61 $^{(**)}$	74 $^{(*)}$	73 $^{(**)}$	69 $^{(**)}$
$0.2\ m^3$	unburnt	94	84	69	95	94	89
	low	87 $^{(*)}$	83 $^{(ns)}$	71 $^{(ns)}$	94 $^{(ns)}$	91 $^{(ns)}$	84 $^{(ns)}$
	moderate	77 $^{(**)}$	66 $^{(ns)}$	57 $^{(*)}$	89 $^{(ns)}$	85 $^{(*)}$	71 $^{(*)}$
	high	55 $^{(***)}$	49 $^{(**)}$	40 $^{(***)}$	73 $^{(*)}$	67 $^{(**)}$	53 $^{(*)}$
$1\ m^3$	unburnt	62	48	30	94	75	58
	low	61 $^{(ns)}$	54 $^{(ns)}$	37 $^{(ns)}$	93 $^{(ns)}$	76 $^{(ns)}$	56 $^{(ns)}$
	moderate	47 $^{(**)}$	37 $^{(ns)}$	28 $^{(ns)}$	86 $^{(ns)}$	59 $^{(*)}$	39 $^{(*)}$
	high	33 $^{(***)}$	28 $^{(**)}$	23 $^{(ns)}$	65 $^{(**)}$	41 $^{(***)}$	29 $^{(**)}$

Signif. codes: '***' 0.001 '**' 0.01 '*' 0.05 '•' 0.1 'ns' 1

Low and moderate severity sites yielded a protective capacity above 50% (more than adequate) for small and intermediate-sized rocks regardless of the forested slope length (Figure 5-6). Only in scenarios with rocks of $0.2\ m^3$, slope inclination ≥ 30° and forested slopes length shorter than 75 m the protective capacity decreased below 50%, mostly between 20 and 30 years post-fire (Figure 6 a). In similar scenarios, the protective capacity in high severity sites ranged between ~10% (inadequate) and 45%, and was at a minimum in scenarios combining intermediate-sized rocks with steep and short forested slopes (Figure 6 a).

For scenarios with rocks of $1\ m^3$ and 150 m forested slopes, the protective capacity of the forests was above 50% (adequate protection) for the unburnt and low severe burnt forests without any clear temporal trend (Figure 7 b). In case of shorter forested slopes, the protective capacity of those forest types ranged only between 25% (satisfying) and 75% (adequate) (Figure 7 a). Contrastingly, the protective capacity in moderate and high severity sites younger than 15 years post-fire rapidly decreased below 50%, reaching its minimum (~10% that is inadequate) around 20 years post-fire.

The linear regression models applied to detect temporal trends in the protective capacity of the burnt and unburnt forests showed significant correlations between the protective effect and the linear and quadratic term of the number of post-fire years for most of the moderate and high burn severity scenarios. Such a significant correlation was missing for low severity sites and the unburnt forests (Appendix 3).

[**Fig. 5** Temporal trends in the protective effect [%] of beech stands hit by low, moderate and high burn severity and the corresponding unburnt beech forests against small rocks [0.05 m^3], 75 m (a) and 150 m (b) forested slopes

Fig. 6 Temporal trends in the protective effect [%] of beech stands hit by low, moderate and high burn severity and the corresponding unburnt beech forests against intermediate-sized rocks [0.2 m^3], 75 m (a) and 150 m (b) forested slopes

Fig. 7 Temporal trends in the protective effect [%] of beech stands hit by low, moderate and high burn severity and the corresponding unburnt beech forests against large rocks [1 m^3], 75 m (a) and 150 m (b) forested slopes]

4 Discussion

The protective effect of forest stands against rockfall highly depends on species composition, stand structure, and sustainability of the forest regeneration capacity (Motta and Haudemand 2000; Dorren et al. 2004a; Dorren and Berger 2005). Disturbances such as forest fires abruptly and substantially change the forest structures, which may temporarily affect the protective capacity of the concerned forest stand (e.g. Dorren et al. 2004a).

Our results show that in beech dominated stands, episodic surface fires cause little changes in the tree species composition. Beech directly re-grows (Maringer et al. subm.) after single fire events, resulting in stable and locally adapted forests on the long-term (Dorren et al. 2004a; Rigling and Schaffer 2015).

However, the post-fire vertical and horizontal stand structures, as well as the amount and timing of regeneration, depends strongly on the burn severity. The forest structure in low severity sites is mostly comparable to those of the unburnt forests (Keyser et al. 2008). The small, fire related changes in tree density, canopy layer, and regeneration dynamics do no seem to affect the overall protective effect. This contrasts to moderate and high severity sites, where significant structural changes occur after fire, what may cause failures in the protective effectiveness against rockfall depending on the forested slope length, the mean slope gradient and the rock size. Structural changes in moderate severity sites are mostly due to the dieback of small and intermediate-sized trees, which goes in line with post-fire observations in conifer stands (Keyser et al. 2008). Surviving tall beeches maintain to some extent the protective capacity (Volkwein et al. 2011) and provide at the same time seeds for new regeneration (Ascoli et al. 2015). The gradual canopy opening of the dominant tree layer leads to emerging beech regeneration, so that the forest protective effect increases again after 20 years post-fire. In the long-term, the mixture of surviving tall and emerging small and intermediate-sized trees results in a multi-layer stand structure that may better meet the protective function standards than mono-layered stands (Dorren et al. 2005b; O`Hara 2006). Nevertheless, the temporary deficit in the protective effectiveness of the forests seem to occur between 10 and 35 years post-fire, especially in case of forested slopes limited in length.

Tree mortality in high severity sites happens immediately and concerns all tree sizes. This is similar to crown fires in conifer stands (Keyser et al. 2008; Brown et al. 2013) and to wind-throw areas, where most trees die immediately after the disturbance event. In those areas, standing and lying dead trees mostly maintain the forest protective effect (Frey and Thee 2002; Schönenberger et al. 2005; Bebi et al. 2015), although their resistance decreases with time, as shown by tensile tests (Frey and

Thee 2002; Bebi et al. 2012; Bebi et al. 2015). The dead wood quantity and quality might be also lower in fire sites than in wind-throw areas (Wohlgemuth et al. 2010; Priewasser et al. 2013), especially in case of tree species such as beech displaying a rapid decaying wood (Maringer et al. subm. a). As shown by our results, the amount of dead wood consistently decreases from 15 years post-fire on, contributing little in the long-term to the forest protective capacity (Frey and Thee 2002). Such a loss in protective capacity has to be compensated by the upcoming regeneration, which might be delayed due to a lack of seed providing trees and/or a thick layer of competing, fast growing early post-fire colonizers. The latter are able to prevent immediate post-fire beech regeneration (Herranz et al. 1996; Ascoli et al. 2013; Maringer et al. subm.), inhibiting the forest re-growth for several decades (Koop and Hilgen 1987). At the same time our results indicate significantly increase in the coverage of early post-fire colonizer and lying dead branches, which may contribute to some extent to the protective capacity against falling rocks with volumes smaller than 0.2 m^3 in the first 20 years post-fire. However, to date their effective contribution is hard to quantify in process-orientated models.

5 Conclusion and practical consequences for forest managers

In this paper we analyze the temporal trends in the forest protection capacity against rockfall of burnt beech stands in the Southern Alps. Based on our results, standing or lying dead trees should in general be left at the burnt site because they contribute temporally to the forest protective effect and provide shade, moisture and nutrients to the emerging tree regeneration (Maringer et al. subm.). In particular, burnt beech forests hit by low severity fires provide nearly similar protective effects as unburnt forests. Hence, silvicultural measures are generally not necessary, whereby the protective capacity has to be assessed on an individual basis.

In case of moderate to high severe fires stands may experience a temporal deficit in their protective capacity between 10 to 30 years post-fire depending on the effective burn severity, the rock sizes, the length of the forested slopes and the mean slope gradient. The cumulative effect of dieback of pre-fire trees and slow re-growth of the regeneration may drop the protective capacity below 50%, especially in case of large falling rocks on steep slopes. Consequently, silvicultural and/or technical measures may be necessary in such critical scenarios depending on the risk for humans and their assets in relation to the cost-benefit ratio. Beside the installation of rockfall nets or walls, small-scale felling of standing dying trees and obliquely positioning of the resulting logs offers a possibility to mitigate the loss in protective capacity. However, directional felling has to be conducted within a particular time frame, because (i) the time-lag between salvage logging and a beech mast year affects the regeneration process, and (ii) beech wood decays relatively fast with progressive time (Ascoli et al. 2013; Maringer et al., submit. a). As mentioned by Ascoli et al. (2013; 2015), salvage logging should be carried out the following winter after a beech mast year—because the success of beech regeneration highly depends on quantitative seed input—, and within the first five year post-fire to protect established beech saplings. Moreover, weed control combined with artificial beech seed dispersal could reduce the inter-species competition and may accelerate the establishment of a new beech generation.

We were not able to quantify the contribution of brushwood and coverage of early-post-fire colonizers in the rockfall modeling. Hence further research is needed in order to quantify the dissipative energy of dense shrub vegetation and their implementation in process-based models.

Appendix

Appendix 1: Investigated fire sites sorted by the date of fire. Further listed: slope [°], elevation (elev. [m a.sl.]), number of plots, mean stem density [stems ha^{-1}], mean basal area [m^2 ha^{-1}], species (F.s.= *Fagus sylvatica*, Broad. = other broadleaf species, P.a. = *Picea abies*, Conif. =other conifer species, species proportion of living trees [%], number of plots in the corresponding unburned forest (control).

Location	Site characteristics				burnt forests characteristics				control
Municipal	Date of fire	Slope	Elev.	Nr. plots	Mean stem density	Mean basal area	Species	Species proportion	Nr. plots
Gordevio	09.03.73	24	1460	1	1900	10	F.s.	97	0
							Broad.	3	
Moghegno	27.11.73	40	1100	3	883	38	F.s.	50	0
							Broad.	50	
Arbedo	20.03.76	31	1300	13	912	36	F.s.	76	1
							Broad.	22	
							P.a.	3	
Sparone	28.12.80	22	1100	16	753	27	F.s.	62	1
							Broad.	37	
							Conif.	1	
Astano	01.01.81	22	1050	2	750	35	F.s.	70	0
							Broad.	30	
Indemini	01.01.81	31	1200	12	613	13	F.s.	71	1
							Broad.	29	
Intragna	04.01.87	27	1150	3	583	18	F.s.	100	0
Aurigeno	01.08.89	35	900	2	1500	25	F.s.	84	1
							Broad.	16	
Corio	15.02.90	19	1080	10	295	26	F.s.	60	2
							Broad.	40	
Mugena	23.03.90	19	900	6	108	29	F.s.	100	1
Novaggio	10.03.90	35	1300	2	225	8	Broad.	38	1
							F.s.	62	
Rosazza	19.01.90	40	1000	5	460	49	F.s.	91	0
							Broad.	9	
Pollegio	09.04.95	22	1200	3	117	22	F.s.	56	2
							Broad.	44	
Tenero	21.04.96	37	950	3	200	15	Broad.	18	0
							F.s.	82	
Arola	04.06.97	40	800	13	646	37	F.s.	66	0
							Broad.	34	
Magadino	15.04.97	33	1200	24	427	28	F.s.	72	3
							Conif.	2	
							Broad.	26	
Ronco s. A.	15.03.97	22	1300	6	417	23	F.s.	100	1
Sonvico	03.04.97	24	1000	5	380	13	F.s.	49	2
							Broad.	51	
Indimini	19.12.98	33	1300	1	100	30	F.s.	50	1
Gordevio	24.04.02	24	1400	5	490	31	F.s.	100	4
Maggia	11.03.98	14	1380	3	617	32	F.s.	100	1
Bodio	17.03.99	33	1050	3	167	48	F.s.	62	1
							Broad.	38	
Dissimo	05.04.99	40	1000	3	900	27	F.s.	97	1
							Broad.	3	
Someo	05.08.99	27	1450	3	433	35	F.s.	100	1
Varallo	10.08.99	29	1300	11	323	26	F.s.	96	1
							Broad.	4	
Villadossola	15.03.01	37	1250	11	1009	27	F.s.	79	1
							Broad.	21	
Cugnasco	02.04.02	22	700	4	575	21	Broad.	53	1
							F.s.	47	

Ronco s.A.	22.04.03	3	1300	2	350	35	F.s.	100	1
Condove	01.03.08	19	1100	11	573	50	F.s.	98	1
							Broad.	2	
Drugno	26.03.12	29	1100	12	963	20	F.s.	90	1
							Broad.	10	
Giaglione	03.03.12	39	1300	8	994	44	F.s.	77	1
							Conif.	21	

Appendix 2: Estimates and standard error of the mixed-effect model for stem densities modeled against slope inclination.

Variable	Estimate	Standard error
Intercept	**5.9**	**<0.0001**
Slope	**0.009**	**0.25**
random intercept	**Variance**	**StdDev.**
	0.33	**0.6**

Slopes of the plots were measured in degree and implemented as explanatory variable in a mixed effect model with negative binomial distribution (Bolker et al., 2013). Stem densities served as response variable, and because of the high intra-class correlation fire sites were implemented as random effect in the model. The result shows that slope inclination was not significant at the 0.05-level (Appendix 3), and thus it was possible to use standardized slope inclination in the Rockfor.net tool. Against this background, the 1st (26.7°) and 3rd quantile (35°) as well as the mean (29.7°) was used as standardized slope inclinations.

Appendix 3: Linear regression models for temporal trends in the years post-fire (AGE) of the protective capacity of burned beech stands differing in burn severity (low, moderate, high) and the corresponding unburned forests. Models were separately conducted for scenarios differing in rocks size (0.05 m^3, 0.2 m^3, 1 m^3), forested slope length (75 m, 150 m) and slope inclination (27°, 30°, 35°). The sign and significance level of the predictor are displayed.

Scenario						
Rock size [m^3]	Forested slope length [m]	Slope inclination [°]	Burn severity	Intercept	AGE	AGE^2
0.05	75	27	Unburned	(+)***	ns	ns
			Low	(+)**	ns	ns
			Moderate	(+) •	ns	ns
			High	(+)***	(-)**	(+)**
0.05	150	27	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)•	(+)•
			High	(+)***	(-)***	(+)***
0.05	75	30	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)•	(+)•
			High	(+)***	(-)***	(+)***
0.05	150	30	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)•	(+)•
			High	(+)***	(-)***	(+)***
0.05	75	35	Unburned	(+)**	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)*	(+)•
			High	(+)***	(-)***	(+)***
0.05	150	35	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)•	(+)•
			High	(+)***	(-)***	(+)***
0.2	75	27	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)**	(+)**
			High	(+)***	(-)***	(+)***
0.2	150	27	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)*	(+)*
			High	(+)***	(-)***	(+)**
0.2	75	30	Unburned	(+)**	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)***	(+)***
			High	(+)***	(-)***	(+)***
0.2	150	30	Unburned	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)*	(+)*
			High	(+)***	(-)***	(+)***
0.2	75	35	Unburned	(+)***	(-)**	(+)**
			Low	(+)*	ns	ns
			Moderate	(+)***	(-)**	(+)*
			High	(+)***	(-)***	(+)***
0.2	150	35	Unburned	(+)**	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(-)***	(+)**

			High	(+)***	(-)***	(+)***
1	75	27	Unburned	ns	ns	ns
			Low	ns	ns	ns
			Moderate	(+)**	(-)*	(+)•
			High	(+)***	(-)***	(+)***
1	150	27	Unburned	(+)***	ns	(-)*
			Low	(+)***	•	ns
			Moderate	(+)***	(-)*	(+)*
			High	(+)***	(-)***	(+)***
1	75	30	Unburned	ns	ns	ns
			Low	ns	ns	ns
			Moderate	(+)*	(-)**	•
			High	(+)***	(-)***	(+)***
1	150	30	Unburned	(+)***	(-)**	(+)**
			Low	(+)*	ns	ns
			Moderate	(+)***	(-)**	(+)**
			High	(+)***	(-)***	(+)***
1	75	35	Unburned	ns	(-)•	ns
			Low	(+)**	ns	ns
			Moderate	(+)*	(-)**	(+)•
			High	(+)**	(-)***	(+)***
1	150	35	Unburned	ns	ns	ns
			Low	ns	ns	ns
			Moderate	(+)*	(-)**	(+)•
			High	(+)***	(-)***	(+)***

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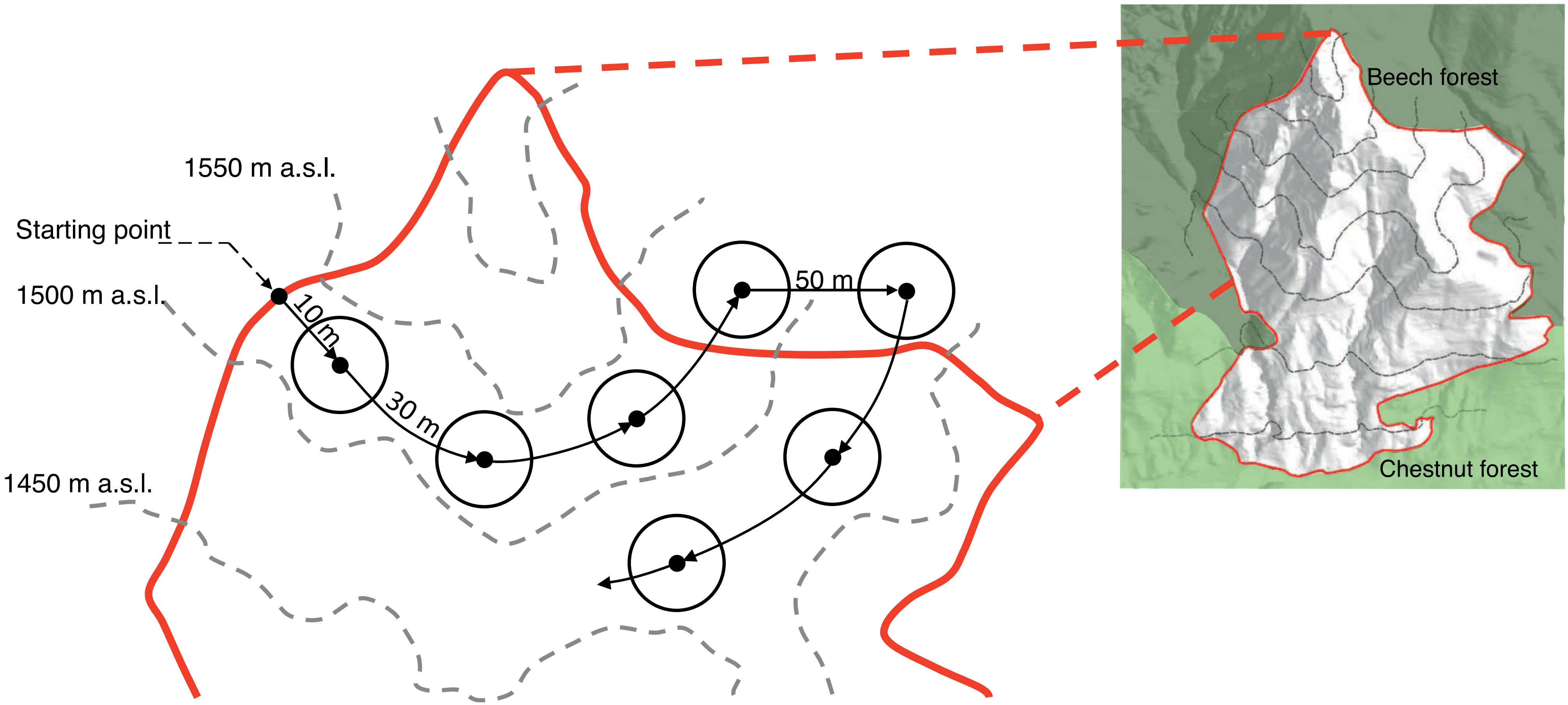
Figure

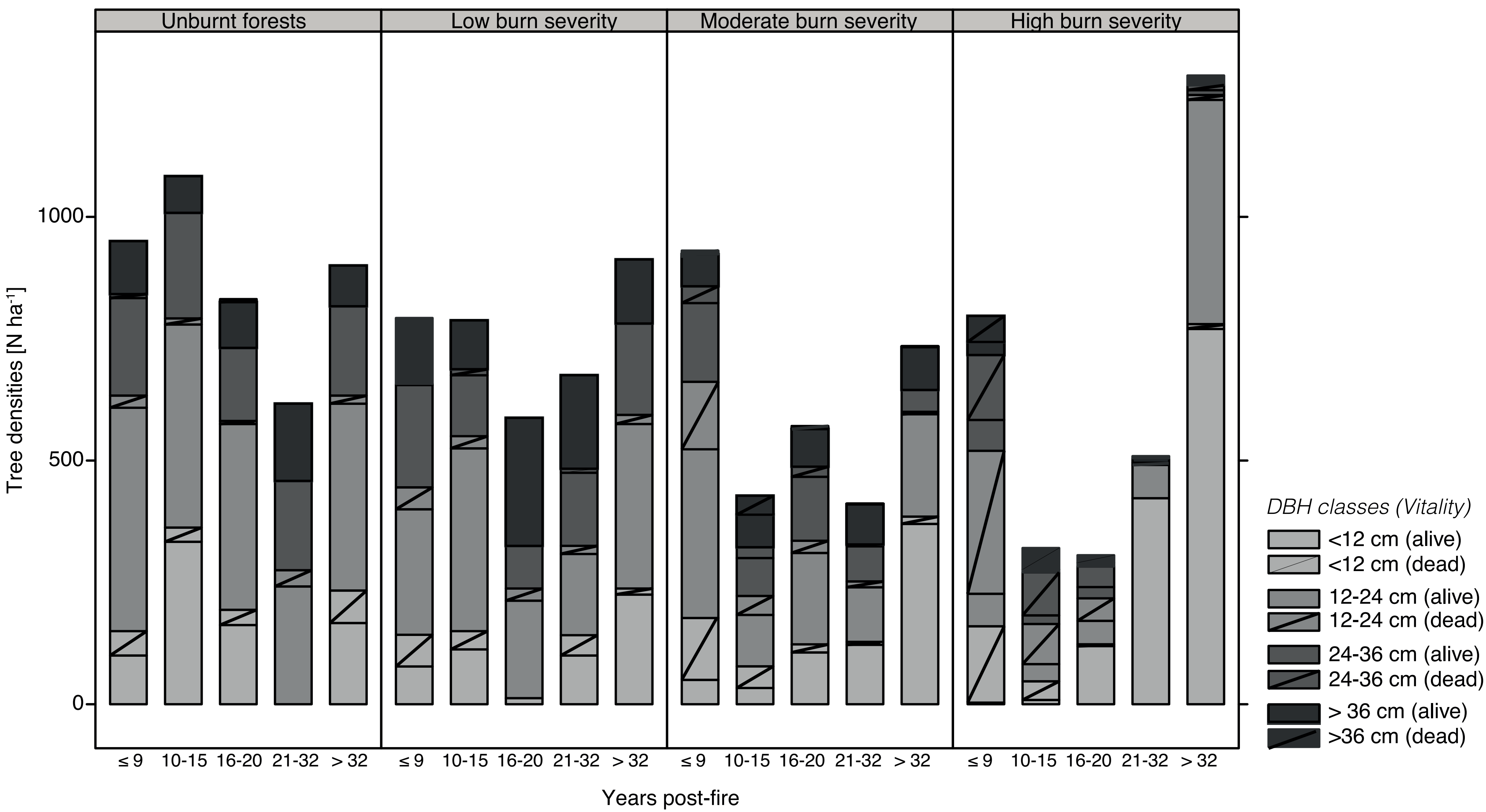


Low fire severity

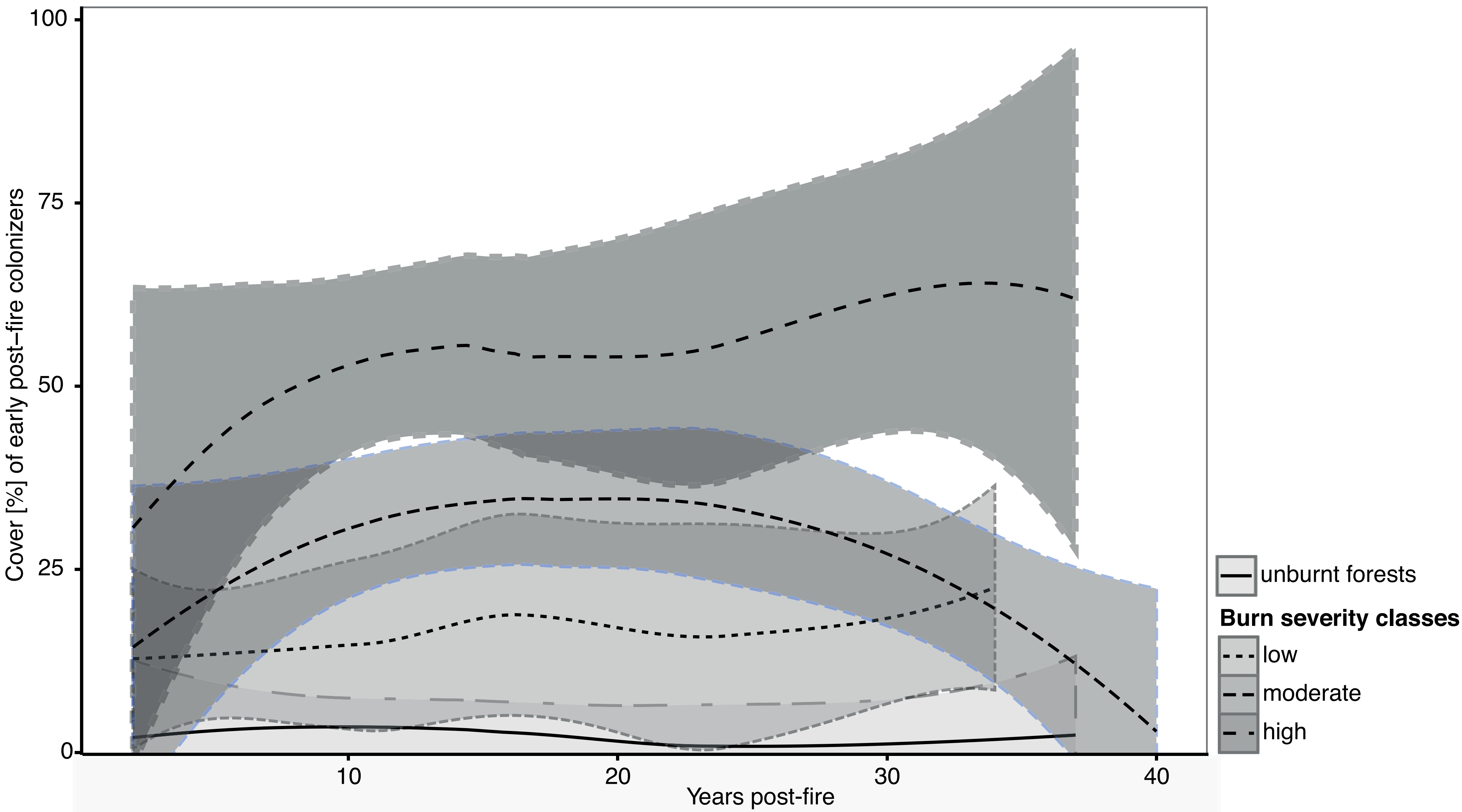
Moderate fire severity

High fire severity

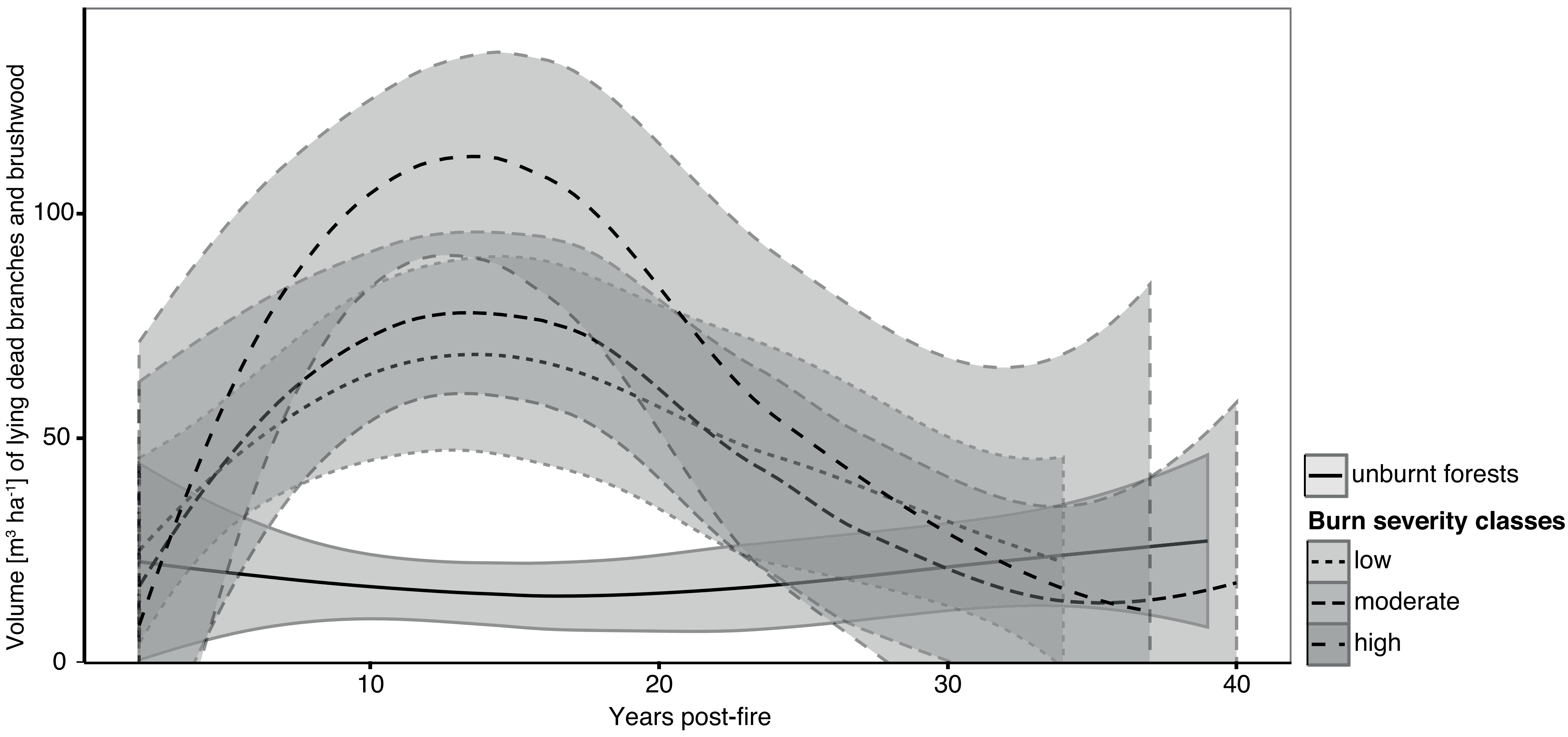
Figure


Unburnt forests
Low burn severity
Moderate burn severity
High burn severity
Tree densities [N ha-1]
1000
500
0
≤ 9
10-15
16-20
21-32
> 32
Years post-fire
DBH classes (Vitality)
<12 cm (alive)
<12 cm (dead)
12-24 cm (alive)
12-24 cm (dead)
24-36 cm (alive)
24-36 cm (dead)
> 36 cm (alive)
>36 cm (dead)

Figure

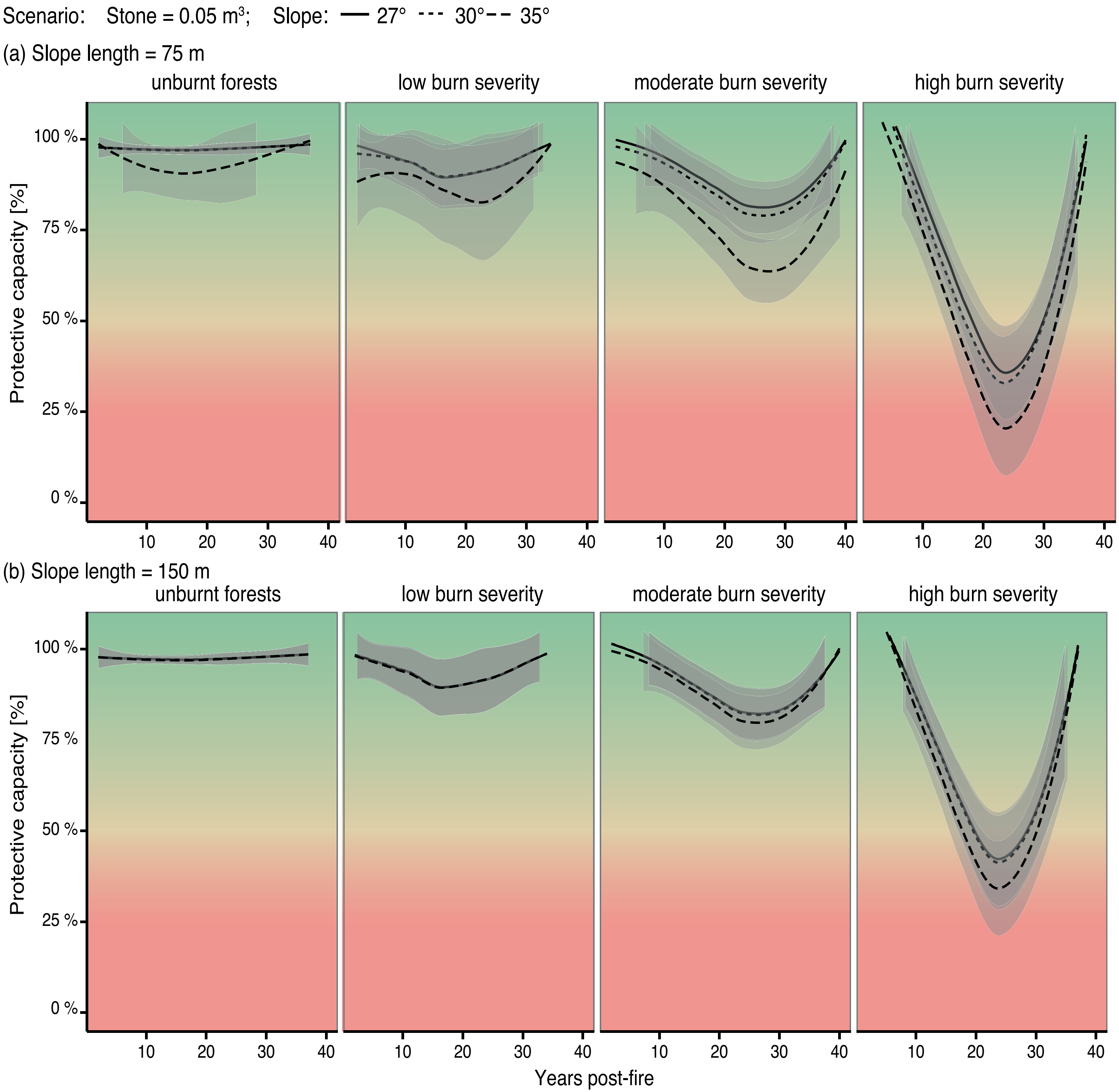

Cover [%] of early post−fire colonizers
100
75
50
25
0
10
20
30
40
Years post-fire
unburnt forests
Burn severity classes
low
moderate
high

Figure

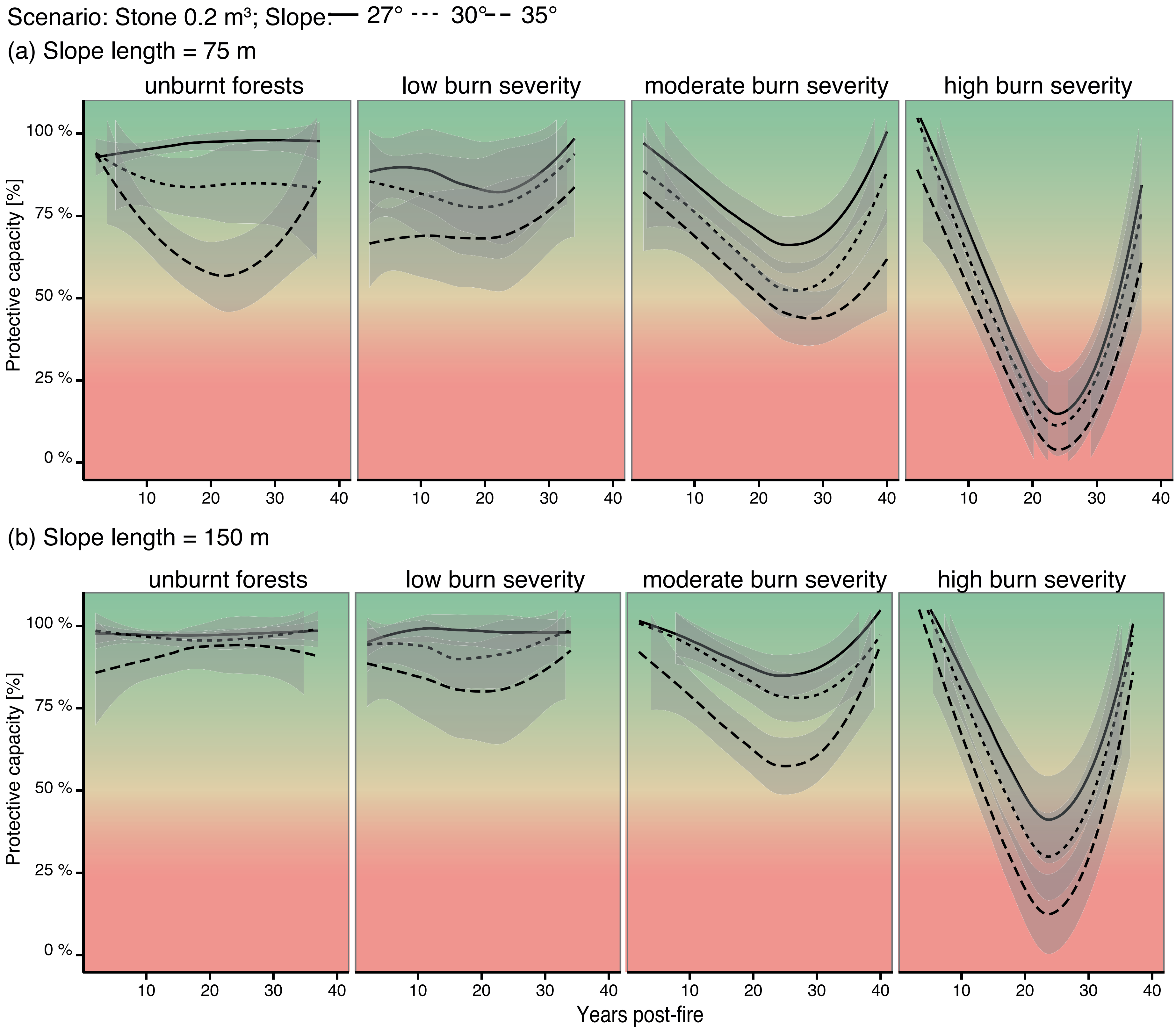


Volume [m³ ha⁻¹] of lying dead branches and brushwood
0
50
100
10
20
30
40
Years post-fire
unburnt forests
Burn severity classes
low
moderate
high

Figure

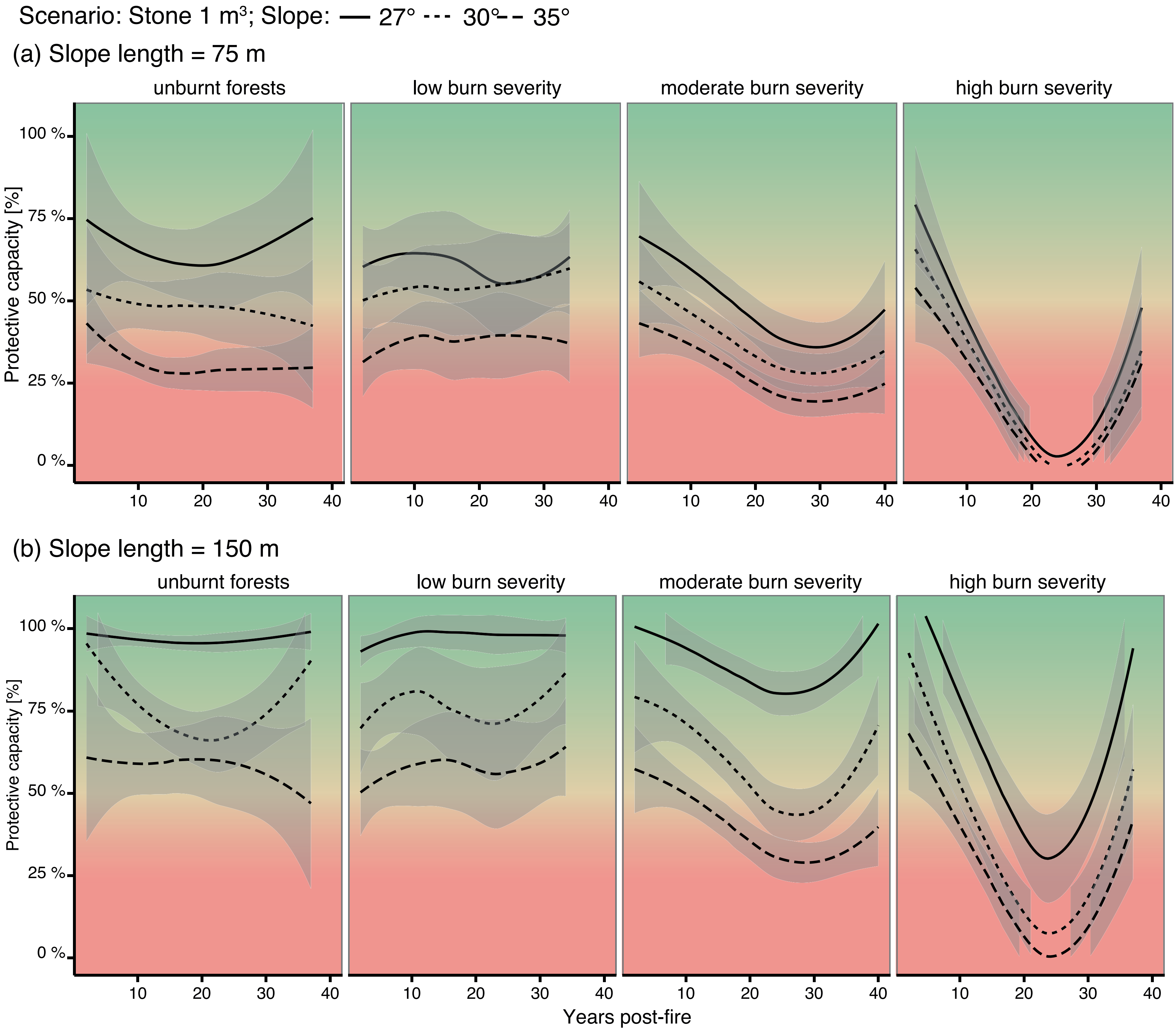

Scenario: Stone = 0.05 m³; Slope: 27° 30° 35°
(a) Slope length = 75 m
unburnt forests
low burn severity
moderate burn severity
high burn severity
Protective capacity [%]
100 %
75 %
50 %
25 %
0 %
10
20
30
40
(b) Slope length = 150 m
Years post-fire

Figure



Scenario: Stone 0.2 m³; Slope: 27° 30° 35°
(a) Slope length = 75 m
unburnt forests
low burn severity
moderate burn severity
high burn severity
Protective capacity [%]
100 %
75 %
50 %
25 %
0 %
10
20
30
40
(b) Slope length = 150 m
Years post-fire

Figure



Scenario: Stone 1 m³; Slope: 27° 30° 35°
(a) Slope length = 75 m
unburnt forests
low burn severity
moderate burn severity
high burn severity
Protective capacity [%]
100 %
75 %
50 %
25 %
0 %
10
20
30
40
(b) Slope length = 150 m
Years post-fire