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Introduction: Natural disturbances in Alpine forests

In alpine areas, mountain forests are an essential ecosystem as they provide various ecosystem services to people (De Leo and Levin, 1997; Baral et al., 2017). One of the fundamental objectives of long-term mountain forest management strategies is to apply adequate and cost-effective forest management approaches in regard to natural hazard mitigation while maintaining other ecosystem services (Kräuchi et al., 2000; Brang, 2001; Brang et al., 2001; Bebi et al., 2001). Mountain forests that are classified as protection forests can provide protection against natural hazards such as rockfalls, avalanches, debris flows, shallow landslides as well as surface erosion. They grow on steep slopes at high elevations and are crucial to, for example, stabilize slopes, reduce avalanche formation or stop falling rocks and, therefore, provide protection to people, settlements and infrastructure (e.g. Schönenberger, 2000; Brang et al., 2001; Brang et al., 2006; Sakals et al., 2006; Moos et al., 2017). Forests act as protection measures since they are able to absorb and dissipate kinetic energy from mass movement processes, and can influence their onset (likelihood of mass movement initiation/triggering/release), propagation (probability of spatial occurrence), and intensity (size and velocity of a mass movement; e.g. Perret et al., 2004; Dorren et al., 2004; Dorren and Berger, 2005; Dorren et al., 2005; Frehner et al., 2005; Brang et al., 2006; Dupire et al., 2016; Moos et al., 2018).

Important goals of mountain forest management are to maintain the integrity and stability of forest ecosystems, mainly in terms of preserving the ecosystem structure and functions over long time periods (Dorren et al., 2004). Due to the evolution of mountain forests, it is impossible that functions of a forest remain constant, especially during transition phases where the protective effect is at its lowest because of a non-optimal forest structure due to tree aging, breakdown of the initial structure, and the presence and abundance of pioneer species (Motta and Haudemand, 2000; Dorren et al., 2004; Dorren and Berger, 2006). The rate of transition is influenced by forest structure (which is in constant flux) (Dorren et al., 2004), and by natural disturbances (Peterson et al., 2000). Natural disturbances are defined as non-anthropogenic events that change structure, composition and function of an ecosystem (White and Picket, 1985; Attiwill, 1994; Frelich, 2002). Due to natural disturbances in protection forests, the protective effects of forests against natural hazards can fluctuate over time and space. Therefore, it is difficult to quantify it over long time periods (e.g. Wehrli et al., 2006). The protective effect of a forest is related to the ability of the forest stand to withstand disturbances without changing (resistance), and to its adaptive and regenerative capacity (resilience) (Moos et al., 2017). Resilience and resistance of forest stands against natural disturbances, and consequently the protective capability against natural hazards, are strongly related to stand parameters that describe the structure of the forest (e.g. Cordonnier et al., 2008). The protective effect of forest against natural hazards is primarily related to stand structure parameters such as tree density, species composition, gap size and diameter distribution (Wasser and Frehner, 1996; Bebi et al., 2001; Gauquelin et al., 2006; Berger et al., 2013).

Forest fires, windthrow, ice and snow breakage, drought, insects, pathogens, and other natural hazards can influence the structure, composition and function of protection forests (e.g. Holtmeier et al., 2009; Kulakowski et al., 2012; Bebi et al., 2017; Seidl et al., 2014b; Seidl et al., 2017). Natural disturbance regimes are described as two-way interactions (Bebi et al., 2009), where disturbances affect forest structure and composition, and in return, forest stand structure and composition also affect disturbance regimes. Forest cover and forest structure were identified as factors influencing frequency, severity and extent of natural disturbances (e.g. Klopčič et al., 2009; Seidl et al., 2011a; Kulakowski et al., 2011). The spatial scale of natural disturbances in forest ecosystems can be divided into small-scale events (generally high frequency), and large-scale events (generally low frequency) (Coates and Burton, 1997; Dale et al., 2001). Both small and large-scale events can be either low or high in intensity. Small-scale events (around 2 ha; e.g. Nagel and Diaci, 2006; Bartelt and Stöckli, 2001) create gaps or eliminate individual trees, and forest in these areas can recover quickly since small gaps can be overgrown by lateral in-growth of existing canopy trees, and other tree species and age classes can establish (e.g. Veblen et al., 1991; Schönenberger, 2002; Zeibig et al., 2005). On the other hand, large-scale events (> 10s of hectares; Nagel and Diaci, 2006) can eradicate thousands of hectares of forest (e.g. Bebi et al., 2017), altering recovery of the forest for several decades, and drastically changing the forest structure (Ulanova, 2000; Schneebeli and Bebi, 2004; Brang et al., 2006; Maringer et al., 2016a). Occurrence of large-scale disturbances is usually less common in alpine areas since large areas of continuous forest structure and composition are rare and site conditions vary over small spatial scales. Furthermore, large amounts of summer precipitation, cooler temperatures, and low fuel loads as well as infrequent large-scale and intense meteorological events (e.g. winter windstorms, ice storms, wet snowfall events, droughts) have limited the occurrence of such events (Brang et al., 2006). Therefore, large-scale and severe events are rare regardless of vegetation conditions or climate (Brang et al., 2006; Lausch et al., 2011; Vacchiano et al., 2016), and are mainly caused by windstorms, insects and avalanches (in that order) (Bebi et al., 2017; Kulakowski et al., 2017). However, this is expected to change due to changing climate conditions, i.e. an increase in forest damages caused by forest fires, bark beetles, and wind has been already observed over the last two decades and is predicted to continue in Europe (Seidl et al., 2009, 2014) as well as in North America (Bentz et al., 2010, 2016). Climate change is likely to influence the nature of disturbance regimes and their interactions in terms of both frequency and intensity (Lindner et at., 2010, 2014; Seidl et al., 2011a, 2011b, 2017; Thom and Seidl, 2016). It is expected that climate change will have various direct, indirect and mutual effects on natural disturbances and stand structure (e.g. Seidl et al., 2014a, 2017), which could affect the protective effects of forest against natural hazards (Schumacher and Bugmann, 2006). At the same time, driven by climate change, structure and composition of forest will also change as well as the dynamics of natural hazard events (e.g. Lexer et al., 2002; Bentz et al., 2010, 2016; Alpine strategy, 2013; IPCC, 2014; Seidl et al., 2011a; Berger et al., 2013; Castebrunet et al., 2014).

Interactions between natural disturbances can lead to “cascading” or “synergistic” effects (Dale et al., 2001), resulting in unexpected changes in forest structure (Buma, 2015). Positive feedbacks between natural disturbances (e.g. drought and wind) occur when one natural disturbance influences the occurrence of another disturbance or disturbances (Seidl et al., 2017), while in the case of negative feedbacks the susceptibility to subsequent disturbances is reduced (e.g. avalanches tracks that act as fire-breaks) (Veblen et al., 1994; Germain et al., 2006; Bebi et al., 2017). Any future changes in interactions between climate, forest structure and natural disturbances will possibly present one of the greatest challenges of mountain forest management to sustain forest as a sufficient mitigation measure against natural hazards.

The objectives of this report are a) to review the influences of abiotic and biotic natural disturbances on protection forests, b) to review the most relevant forest stand parameters that influence the protective effect of forest against avalanches and rockfalls, and c) to discuss the effects of individual natural disturbances on those forest stand parameters.

Materials and methods

The outcome of this study is based on a review of scientific literature and project reports, which dealt with natural disturbances, their influence on protection forest stand parameters, and the onset and propagation conditions of avalanches and rockfalls in alpine areas. First, we addressed all major natural disturbances. Then, we identified all relevant stand parameters that influence the onset and propagation probability of snow avalanches and rockfalls. Lastly, the influence of natural disturbances on forest stands parameters and protective effects against snow avalanches and rockfalls are discussed.

Natural disturbances in protection forest

Abiotic disturbances

Abiotic disturbances are the consequence of non-living factors, which can be caused by different events: a) meteorological (cyclones, storms, tornadoes, thunderstorms and lightning), b) climate (drought), c) hydrological (floods, avalanches, landslides and mudslides), d) geophysical (earthquakes), and e) anthropogenic (fire, air pollution) (Moore and Allard, 2011). In this report only the most influential abiotic disturbances on forests in the Alps were reviewed: forest fire, windthrow, ice break, snow break, avalanches and rockfalls.

Forest fire

Historically, forest fires in the Alpine Space were mainly caused by human activity (e.g. new pasture land), which resulted in a shift in tree species composition and a decrease in forest cover (Valese et al., 2011; Schwörer et al., 2015; Bebi et al., 2017). Besides man-induced fire, meteorological conditions that favor the occurrence of lightning strikes are related to a high number of forest fires, especially in the summer (Conedera et al., 2006; Wastl et al., 2013; Bebi et al., 2017). In the Alps, fire regimes are highly heterogeneous, especially when comparing Northern to Southern Alps (Valese et al., 2011; Wastl et al., 2013; Bebi et al., 2017). In the Southern Alps, forest fires occur over large forest areas (Brang et al., 2006; Wastl et al., 2013; Bebi et al., 2017). Especially in Italy and France forest fires occur with greater frequency and larger spatial coverage, whereas in Austria and Germany forest fires rarely exceed 50 ha (Valese et al., 2011; Wastl et al., 2013; Bebi et al., 2017). The average area burned in the Alpine region is 9 ha (Valese et al., 2011).

Forest fires may greatly impact the extent of forests since they can be stand-replacing events and the top soil layer will be exposed, resulting in surface water runoff and soil erosion (Brang et al., 2006; Shakesby et al., 2006; Cerdà and Doerr, 2008; Shakesby et al., 2011; Sass et al., 2012; Holtmeier and Broll, 2018), and affecting different slope-related processes (e.g. avalanches, rockfalls) (Butler et al., 1992; Weir, 2002; Germain et al., 2006). Forest fire frequency may increase in the following decades due to lower fuel moistures under sustained dry conditions (Williams and Abatzoglou, 2016), increased ignitions due to lightning activity (Conedera et al., 2006), and prolonged droughts (Cook et al., 2014).

Windthrow

In the Alpine Space, wind is one of the most prevalent natural disturbance agents (e.g. Schelhaas et al., 2003; Krehan and Steyer, 2008; Seidl et al., 2011a, 2011b; Bebi et al., 2017). Storms are highly variable in terms of their damage to forests, because they can either have a thinning effect or destroy entire stands (Brang et al., 2006; Gardiner et al., 2013; Wohlgemuth et al., 2017) (Figure 1). One should distinguish between thunderstorms and synoptic weather events – cyclonal storms. Thunderstorms (also whirlwinds) are in general local small-scale events that last several hours, causing a variety of damages. Cyclonal storms are large-scale, can last several days and result in catastrophic destruction of the forest. In particular, winter cyclonic storms cause the most severe damages to forests in central and northwestern Europe (Usbeck et al., 2010b). For example, the large-scale synoptic storms Vivian (1990) and Lothar (1999) downed 19 million m^3 of Swiss mountain forests (Dobbertin, 2002), many of which were protection forests (Schönenberger, 2002). Windthrow is an especially important disturbance in Norway spruce

(*Picea abies* Karst.)-dominated stands with high growing stocks in the Northern Alps (Bebi et al., 2017). Although windthrow in the Alps can result in 1000+ ha of forest damage with high rates of tree mortality, such large-scale events are infrequent. The most common windthrow events are often periodic and small-scale (median from 0.1 to 0.5 ha), medium to high severity events (Splechtna et al., 2005; Nagel et al., 2007; Firm et al., 2009; Vacchiano et al., 2016; Bebi et al., 2017).

Windthrow is an important ecological disturbance factor (Mitchell, 2013). It changes the amount and quality of litter, influences temperature and water balances (Holtmeier and Broll, 2018), affects species composition (in favor of broadleaved species) (Brang et al., 2004), and changes soil properties (Schaetzl et al., 1989; Šamonil et al., 2010) and forest structure (Mitchell et al., 2008; Ulanova, 2000). Forest stands that are more susceptible to windthrow are generally older, slender (i.e. a large H/D ratio = height/diameter at breast height [dbh]) and taller (e.g. Dobbertin, 2002; Peltola, 2006; Klopčič et al., 2009; Schmidt et al., 2010; Đodan and Peric, 2019) compared to stands with trees of smaller H/D ratios and optimal vertical structure. Moreover, the presence of canopy gaps (especially along forest edges), young growth, wet shallow soil, Norway spruce-dominated stands, and forest fragmentation also increases the susceptibility to wind damage (Peltola et al., 1999; Ogris et al., 2004; Lanquaye and Mitchell, 2005; Klopčič et al., 2009; Gardiner et al., 2013). Uneven-aged and regeneration stands are often less susceptible to windthrow (Klopčič et al., 2009), because of small H/D ratios, better tree architecture, stand distribution, root patterns, enhanced vitality, and wind dampening roughness of the canopy layer (Hanewinkel et al., 2014; Diaci et al., 2017).

It is expected that climate change will have direct impacts on the occurrence of strong winds (Donat et al., 2011), their duration (Peltola et al., 2010), intensity (Usbeck et al., 2010a), and frequency (Bender et al., 2010; Usbeck et al., 2010a, 2011; Bebi et al., 2017). Future higher mean temperatures are expected to increase forest damage by also facilitating interactions between bark beetles and windthrows (Seidl and Rammer, 2017).

Figure 1: Protection forest in the Trenta valley after a storm. Fallen trees present obstacles for falling rocks, especially if they are positioned parallel to the slope. This enhanced protective effect against rockfall lasts until the downed wood starts to break down from decay processes.

Ice and snow break

In Europe in the last decades, an increase in forest damage as the result of snow and ice break has been reported (Schelhaas et al., 2003). In the Alps, snow and ice break is a widespread disturbance at different spatial scales, from very small (<0,1 ha) to substantial (e.g. 1.5 million m^3 were damaged by an event in the Slovenian Alps in 2014) (Nagel et al., 2016; Bebi et al., 2017). Damage patterns vary from broken branches of individual trees to major stand destruction (e.g. Nykänen et al., 1997; Duguay et al., 2001; Isaacs et al., 2014). Low intensity ice break events mainly cause crown damage whereas severe ice break events can also uproot and break trees, especially larger trees on steep slopes (Bragg et al., 2002, 2003; Nagel et al., 2016). Ice break-related mortality is highest in severely bent trees, or trees with extensive damage to crowns or roots (Bragg et al., 2002). Trees with foliage are in general more susceptible to ice/snow break due to a greater surface area (e.g. Peltola et al., 1999). Mid-elevation and broadleaved forests on exposed terrain can be more susceptible to snow/ice break damage (Hlasny et al., 2011; SFS, 2015; Nagel et al., 2016; Bebi et al., 2017), and severity of ice break can be higher at higher elevations (temperature-related ice formation) (Rhoads et al., 2002). Susceptibility to ice/snow break is also influenced by tree height and dbh (Jalkanen and Mattila, 2000; Peltola et al., 1999), where trees with lower dbh (median at 17.5 cm) were found to be more susceptible to snow break (Klopčič et al., 2009), trees with larger dbh or height (particular slender trees) are more resistant to uprooting or stem breakage (Peltola et al., 1999). Furthermore, numerous tree, site, stand, and meteorological characteristics influence ice-break damage; however, their effects on the susceptibility to ice/snow break are reported to be various (positive/negative) and often contradictory (Bragg et al., 2003).

Compared to other disturbance agents that have the capability to impair protective effects over large continuous areas, snow break often leaves surviving trees, which provide some protective effect (Brang et al., 2006). Any future long-term variation in snow/ice break disturbance can be mainly attributed to climate (Bebi et al., 2017), especially warmer and drier conditions, which will result in a decrease (Seidl et al., 2017). However, due to changes in the forest structure (large proportion of pole stage stands, larger H/D ratios) susceptibility of forest to ice/snow break disturbance is expected to increase in the future (Bebi et al., 2017).

Avalanches

Avalanches are a rapid, gravity driven mass of snow, air and debris (Bebi et al., 2009). In the Alps, avalanches play a major role in shaping mountain ecosystems, especially subalpine forests (e.g. Holtmeier and Broll, 2018). Avalanches can either occur several times within a year in one path or once every century (Bebi et al., 2009). They have the capacity to disturb or destroy forested areas greater than 1000 ha (e.g. Vacchiano et al., 2016; Bebi et al., 2017). Avalanches with higher velocities and more compacted snow can be highly destructive (e.g. Holtmeier and Broll, 2018). Forest stands that are frequently disturbed by avalanches tend have characteristics that are different compared to undisturbed stands (Bebi et al., 2009). Disturbed stands are typically composed of trees with lower annual growth rates, smaller diameters, shorter stature, greater structural and vegetation diversity (Patten and Knight, 1994; Kulakowski et al., 2006; Rixen et al., 2007; Holtmeier and Broll, 2018), lower tree densities, and are dominated by shade intolerant tree species (e.g. Bebi et al., 2001) and Krummholz species (e.g. prostrate mountain pine [*Pinus mugo* Turra]) (Holtmeier and Broll, 2018). Avalanches form pathways due to the channelizing ability of gullies, where pioneer species usually find their ecological niche (Holtmeier and Broll, 2018).

In the following decades, changes to forest cover and composition due to climate change or land-use change will probably influence snow avalanche disturbance regimes (e.g. Bebi et al., 2009; Alpine strategy, 2013). Based on projections of snow conditions, a general decrease in avalanche activity is expected in spring months at low altitudes, and an increase of wet snow avalanche activity in winter months (Castebrunet et al., 2014).

Rockfalls

Rockfall can be defined as a “fragment of rock (a block) detaching from a release area and propagating downslope by bouncing, falling or rolling” (Whittow, 1984). Rockfall activity can disturb individual trees or eliminate entire stands (e.g. Brang et al., 2001). Continuous rockfall activity especially those with greater frequencies and magnitudes may prevent forests from reaching late successional stages (e.g. Holtmeier and Broll, 2018). On rockfall sites, under the limited availability of moisture and nutrients due to slow soil formation, vegetation usually remains in its initial stage over long time periods, especially in sun exposed locations (Holtmeier and Broll, 2018). Consequently, when site conditions improve, pioneer species such as birch (*Betula pubescens* Ehrh.), alder (*Alnus viridis* (Chaix) D.C.), and mountain pine establish

(Holtmeier and Broll, 2018). Forest growth may thus be limited to relatively low altitudes due to unstable slope debris, talus cones and steep rock walls (Holtmeier and Broll, 2010).

Biotic disturbances

Mass outbreaks of phyllophagous insects, bark beetles, pathogens, diseases and damage caused by herbivores (e.g. browsing) are considered biotic disturbances (e.g. Kautz et al., 2017; Holtmeier and Broll, 2018).

Insects

Most damaging insect outbreaks are caused by xylophagous and phyllophagous insects (Kautz et al., 2017). In mountain forests, bark beetles (*Ips typographus* L., *Pityogenes chalcographus* L.), are the main biotic disturbance agents in terms of the amount of forest damage (e.g. Schelhaas et al., 2003; Seidl et al., 2007; Overbeck and Schmidt, 2012; Thom et al., 2013). The damage in terms of timber volume has even increased in the past few years (Bebi et al., 2017). Additionally, bark beetle damage may increase further, if we account for continuation of climate change projections (Seidl et al., 2014a). Damage from bark beetles can be correlated to storm damage (Schelhaas et al., 2003; Bouget and Duelli, 2004; Temperli et al., 2013), drought (e.g. Jactel et al., 2012; Hart et al., 2014), forest fire (e.g. Amman, 1977; Bebi et al., 2003; Bigler et al., 2005), tree species (especially spruce), standing volume, and growing season temperature (Klopčič et al., 2009; Thom et al., 2013; Stadelmann et al., 2014). Interactions between wind and bark beetles were often observed (e.g. Temperli et al., 2013; Vacchini et al., 2016). Bark beetle outbreaks may be local and endemic or can lead to large-scale epidemic level attacks where an entire stand may become infested (Holtmeier and Broll, 2018). The effect of bark beetles on mountain forests are numerous, including changes in structure, function and composition of forest ecosystems (Jenkins et al., 2014; Holtmeier and Broll, 2018), triggering needle loss, which lowers the albedo of snow cover, thus altering snow accumulation, microstructural properties of subcanopy snowpack and melt processes (Pugh and Small, 2012), reduction in canopy interception, increases in light transmission, and increases in wind speeds (Jenkins et al., 2014).

Phyllophagous insects (defoliators) are another type of biotic disturbance that affects mountain forests (e.g. Vacchiano et al., 2016; Holtmeier and Broll, 2018). Defoliation is often times limited to smaller areas (e.g. mountain slopes) (Holtmeier 2009; Holtmeier, 2015); however, mass outbreaks of leaf-consuming insects can be found in boreal forests of northern Europe (Neuvonen et al., 2005; Holtmeier 2009; Holtmeier, 2015; Neuvonen and Viiri, 2017). In the Southern Alps, damage caused by defoliators may vary depending on outbreak duration and severity, although they are rarely stand replacing (Vacchiano et al., 2016). In the subalpine larch-cembran pine forest, growth of European larch (*Larix decidua* Mill.) can be affected by larch tortrix (*Zeiraphera griseana* Hübner), nun moth (*Lymantria monacha* L.) (Vacchiano et al., 2016), and larch bud moth (*Zeiraphera diniana* Gn.), which can cause up to 90% needle loss (Delucchi, 1982; Baltensweiler and Fischlin, 1988; Holteimer, 2015). North-exposed slopes are favored by nun moth almost exclusively (Vacchiano et al., 2016). However, subalpine larch stands in 'warm slope zones' (Holtmeier and Broll, 2018) and locations above 1600 m (Delluchi, 1982) are still susceptible (1800 m for larch tortrix – Vacchiano et al., 2016). At lower altitudes (1200 m – 1600 m), defoliation events occur less often, and at altitudes lower than 1000 m almost never (Baltensweiler and Fischlin, 1988; Holtmeier 2009; Holtmeier, 2015).

Pathogens

Fungi, bacteria and other microorganisms are considered as pathogens, and are often associated with an insect infestation and drought (e.g. Desprez-Lostau et al., 2006; Jactel el at., 2012). Pathogens do not necessarily kill affected trees immediately, but the capability of trees to resist other agents is typically lowered (Desprez-Lostau et al., 2006). Pathogens such as root rot fungi (*Heterobasidion annosum, H. parviporum, Armillari spp*. (Fr.) Staude) play important roles in alpine mountain pine forest development dynamics as they affect tree mortality, structure, density and species composition (Durrieu et al., 1985; Dobbertin et al., 2002; Bendel et al., 2006; Gonthier et al., 2012; Garbelotto and Gonthier, 2013). *Heterobasidion annosum* (Fr.) Bref. is one of the most widespread wood decay agents that leads to conifer mortality, especially in Norway spruce (Garbelotto and Gonthier, 2013). Larger trees of especially silver fir (*Abies alba* Karst.) and Norway spruce are more frequently attacked, although the opposite is true for larch (Thor et al., 2005; Gonthier et al., 2012). Mechanical resistance of rotten trees is weakened, leading to greater susceptibility to other natural disturbances such as windthrow (Figure 2).

At high altitudes, especially in wet vegetation periods with consecutive summer frost, scleroderris canker (*Gremmeniella abientina* Lagerb., M. Morelet) can develop on Austrian pine (*Pinus nigra* J.F. Arnold), Norway spruce, Swiss stone pine (*Pinus cembra* L.) and Scots pine (*Pinus sylvestris* L.) (La porta et al., 2008; Barbeito et al., 2012), and destroy hundreds of hectares of forest (Bernhold, 2006). Alongside other parasitic organisms, snow fungi (*Herpotrichia juniperi* (Duby) Petr.*, Herpotrichia coulteri* Peck.*, Phacidium infestans* P.Karst) are a problem for tree cover establishment, especially near treeline where afforestation initiatives are part of avalanche control programs (Schönenberger, 1978; Roll-Hansen, 1989; Donaubauer, 1984; Senn, 1999; Cunningham et al., 2006; Barbeito et al., 2013; Holtmeier and Broll, 2018). Snow fungi together with snow distribution have been found to be major factors that influence the position of the treeline (Barbeito et al., 2012; Barbeito et al., 2013). On a global scale, warmer and wetter conditions that can result from climate change are likely to amplify forest damage due to pathogens (Sturrock et al., 2011; Weed et al., 2013; Seidl et al., 2014a; Seidl et al., 2017).

Figure 2: Probability of stem breakage in windthrow areas is enhanced for trees with root rot. (Source: Domen Oven).

Other disturbance agents that can also influence protective effects of forests are wildlife herbivory, introduced species and anthropogenic disturbances such as logging (e.g. McClung, 2001; Germain et al., 2006; Holtmeier and Broll, 2018); however, they were not taken into account since this was not the focus of this report.

Most relevant stand parameters of forests that protect against snow avalanches and rockfalls

Protection forest against snow avalanches

Besides the characteristics of snow and topography, forest stand structure is one of the main factors that influences the occurrence of avalanches (Bebi et al., 2001; Holtmeier and Broll; 2018). Most relevant protection forest characteristics in terms of hazard components (onset probability, propagation probability and intensity) are: canopy coverage, species composition, surface roughness, tree height, stem density, forest gap size and dbh distribution (Meyer-Grass and Schneebeli, 1992; Rammig et al., 2006; Frehner et al., 2005; Berretti et al., 2006, Gauquelin et al., 2006, Bebi et al., 2009; Berger et al., 2013; Moos et al., 2017). Forests have the most important protective effect in the release areas where they stabilize the snow pack and intercept precipitation, while in avalanches tracks the effect of forests is limited to reduce the lateral spreading and to slow down smaller events (< 100 m^3) (Teich et al., 2012). In the case of large (> 1000 m^3), destructive events, the protective effect of forest is negligible (Viglietti et al., 2010).

Canopy cover influences the characteristics of the snowpack beneath it, so that the subcanopy snowpack is less prone to avalanche release. In forests, snow depth is lower compared to open (non-forested) areas and the density of snow is higher (Storck et al., 1999; Bründl et al., 1999; Mayer and Stöckli, 2006). Unloaded intercepted snow disturbs the (homogenous) layering of the snowpack and thus prevents the formation of continuous weak layers (Mayer and Stöckli, 2006). Snow interception by canopy cover is closely related to species composition (Bebi et al., 2009), and evergreen conifers are more efficient in intercepting snow than broadleaves resulting in lower snow depths that lead to smaller avalanche activity. However, the litter from silver fir and Norway spruce can enable sliding and possibly increase avalanche activity (Viglietti et al., 2010; Berger et al., 2013). In periods of lower snow depths, deciduous trees can be suitable since more sunlight reaches the canopy floor and melts the snow, therefore prevents snow gliding (Teich et al., 2012). In the case of long cold periods of large snow fall and extreme snow depths, the effects of canopy cover and tree species are reduced (Berger et al., 2013). In the propagation areas, conifers are more effective than broadleaves or larch trees (Bebi et al., 2009). In larch and broadleaved forest stands runout distances are significantly larger compared to evergreen coniferous stands and mixed forests (Teich et al., 2012). In contrast to evergreen trees, deciduous trees have smaller effective crown areas and are more likely to survive powder avalanche blasts (Feistl et al., 2014). Thus, larch, broadleaved trees and shrubs should be limited in release zones in favor of evergreen conifers (Newesely et al., 2000; Viglietti et al., 2010; Berger et al., 2013). In areas where both avalanches and rockfalls occur, mixed forest stands provide the most effective form of protection (Stokes, 2006).

Surface roughness in the avalanche path influences avalanche runout distances (Teich et al., 2012). High surface roughness (e.g. downed wood, logs, boulders) prevents also the release of full-depth glide avalanches since it provides stabilization and mechanical support of the snowpack, and hinders the formation of continuous weak layers (Veitinger, 2015). No avalanche events are reported in the literature where surface roughness elements were higher than 2 m (McClung et al. 2002; Veitinger, 2015). In the Alps, it has been observed that when farmers stopped cutting the grass on steep and open slopes, more avalanche events occurred due to the promotion of snow gliding conditions (Newesely et al., 2000; McClung and Schaerer, 2002). The presence of dead wood or staged terrain increased surface roughness and can prevent the gliding of the snow cover, which also protects young plants from being uprooted (Puttalaz, 2010; Teich et al., 2012; Feistl et al., 2013). In order to promote surface roughness in the propagation area, it is recommended that lying tree stems are left on the slope, and that high stumps (1.3 m) are left after cutting (Berger et al., 2013). With increasing snow accumulation, the surface roughness decreases, resulting in potentially larger release areas (Veitinger, 2015; Veitinger and Sovilla, 2016).

Stem density effects both frequency and magnitude of avalanche events. A high number of stems locally increases air temperature and consequently lowers the temperature gradient within the snowpack, which leads to the formation of rounded grains and fewer faceted crystals or depth hoar (which can form weak layers in the snowpack) compared to the snowpack in open areas (Freppaz et al., 2008; Viglietti et al., 2010). In forest release zones with high stem densities, the onset probability of avalanches is decreased, and will consequently have an influence on limiting their spatial extent. Recommended stem density in release areas of avalanches with low to moderate magnitude is 300 to 500 stems/ha in moderately steep slopes (30°), and 1000 to 2000 stems/ha in steeper slopes (40° or more) (Horvat and Zemljič, 1998). In order to reduce the release probability, the tree height in this area is recommended to be twice as high as the maximum snow depth. In the propagation area the height of the trees is recommended to be higher (Rudolf-Miklau et al., 2015). The diameter of a tree affects avalanche propagation, because trees with a greater dbh present greater mechanical obstacles; trees with dbh ≥ 10 cm present sufficient mechanical obstacles that limits propagation of an avalanche (Horvat and Zemljič, 1998). Trees with dbh in the range of 6–10 cm can only stabilize the snowpack marginally. However, avalanches that are released in the forest areas with larger mean dbh had longer runout distances (Teich et al., 2012). That is, high stem density in combination with small diameters (< 15 cm) had a significant effect on reducing avalanche runout distance (Teich et al., 2012), especially in the first 200 m of the avalanche track.

Forest gaps on slopes around 35° should not be wider than 50 m and longer than 40 m (Horvat and Zemljič, 1998). Avalanches may release in gaps longer than 30 m (in the direction of the slope), and 15 m in the horizontal direction (Imbeck, 1987). Gaps within the release and propagation areas are recommended to be < 15 m (Berger et al., 2013). The probability of avalanche release can be increased along forest edges, especially in the case of coniferous forests, where greater quantities of snow accumulates on the forest edge. Snow that accumulates on the forest edge changes slower and has different properties than snow under forest canopy. Therefore, snow avalanche release is more likely on forest edges (Horvat and Zemljič, 1998). Especially dangerous are areas where the transition from forest to meadow coincides with a break into steeper terrain (Pintar, 1968). Snow gliding is prevented by forest stands that are situated at the lower edge of gaps.

Protection forest against rockfalls

In the Alps, rockfalls most often occur as falling rocks with volumes between 0.5 and 5 m^3 (Berger et al., 2002; Dorren et al., 2005; Stoffel et al., 2005). Dorren et al. (2005) have shown that, if we express rockfall activity as the number of rocks that surpass an area, the total number of rocks will be 63% lower in forested areas than in areas without forests. Moreover, forested slopes also decrease bounce height (by 33%) and velocity (by 26%) of rocks. Most relevant forest characteristics in terms of hazard components (onset probability, propagation probability and intensity) are tree density, gap length, diameter distribution, species composition, the presence of trees in the release area, length of the forested part of the slope, and surface roughness (Dorren et al. 2005; Stokes et al. 2005; Frehner et al., 2005, Brang et al., 2006; Berretti et al., 2006, Stokes et al., 2006; Gauquelin et al., 2006, Bebi et al., 2009, Berger et al., 2013; Radtke et al., 2013; Dupire et al., 2016; Moos et al., 2017; Moos et al., 2018).

In terms of forest structure, stands with high stem densities, similar age and diameter of trees may have maximum effects on reducing travel distances of rocks (Perret et al., 2004). Yet, it is difficult to maintain this optimum stage of forest stands (Dorren et al., 2005). Realistic upper limits of stem density in rockfall protection forests are 350 trees/ha with a mean dbh of 35 cm (Nais, 2003; Perret et al., 2004; Gauquelin et al., 2006; Berger et al., 2013); however, this strongly depends on tree species and site characteristics. Although trees with a large dbh can dissipate higher kinetic energy of rocks, the density of a forest seems to be more important in reducing rockfall propagation area and length than dbh itself (Dorren et al., 2004; Dorren et al., 2005; Frehner et al., 2005; Brang et al., 2006; Berretti et al., 2006; Berger et al., 2013), especially for rocks of smaller diameters (from 13 to 45 cm) (Jahn, 1988). Although not all simulation results agree with these findings (Radtke et al., 2014). Jancke et al. (2009) even suggest that a density of trees per hectare between 5000 and 10000 (where stands are younger than 30 years old) can be sufficient to provide the efficient protection against rocks with diameters > 20 cm. Only extreme stem density (~ > 7000 stems ha^{-1}) provides acceptable protection against very small rocks (<= 0.25 m^3) (Jancke et al., 2009; Radtke et al., 2014). Research by Radtke et al. (2014) showed that in coppice stands basal area and dbh are more important factors in the case of small (0.25 - 0,5 m^3) and bigger rocks than stem density. They recommend a heterogeneous dbh distribution in coppice forests. Required stem density and mean stem diameter can be calculated based on mean diameter of falling rocks, their mean kinetic energy, the maximum length of the stopping zone and the tree species (Dorren et al., 2005; Stokes et al. 2005; Brang et al., 2006). Basal area should be maintained high at the foot of a release area (Radtke et al., 2014; Dupire et al., 2016; Moos et al., 2017). With trees that have dbh ≥ 15 cm, basal area is recommended to be ≥ 25 m^2/ha in the rockfall propagation area, and ≥ 20 m^2/ha in the rockfall deposit area (Bebi et al., 2009; Berger et al., 2013).

Gap size in rockfall propagation areas should be as limited as possible. The maximum gap size should be around 1.5 times the dominant height of the surrounding forest stand (high forest < 40 m, coppice < 20 m) (Ancelin et al., 2006; Berger et al., 2013). Corridors in rockfall propagation zones are areas with high rockfall activity, which inhibits forest regeneration. In rockfall corridors forest can be artificially arranged in a way that it directs falling rocks towards ‘channels’ (e.g. Kupferschmid Albisetti et al., 2003; Dorren et al., 2005; Berger et al., 2013). In this case, on either side of the corridor, a forest band of 25 m in width of high stem density should be located (Berger et al., 2013). Distribution of trees in the rockfall propagation area should be random, while in deposition area, coppice stands can also be effective in stopping rocks (Berger et al., 2013). The distance between the potential rockfall release areas and forest stands should be limited so that the kinetic energy of rocks is lower and they can be stopped by the forest (Dorren et al., 2004; Berger et al., 2013). In propagation areas, rockfall protection forest should be at least 200 m long in order to effectively stop rolling rocks (Berger et al., 2013).

Both in rockfall release and propagation areas broadleaved species are preferred as they are more resistant to the rockfall impacts than conifers (Stokes et al., 2006). At least 30 % of the thickest trees in the forest stand should thus be broadleaves (Stokes, 2006; Berger et al., 2013). The regeneration of the following tree species should be promoted in rockfall protection forests (Berger et al., 2013): sessile oak (*Quercus petraea* L.), European beech (*Fagus sylvatica* L.) and sycamore maple (*Acer pseudoplatanus* L.) since they are most resistant to rockfall impacts (Dorren et al., 2005; Dorren and Berger, 2006). Forest stands should also be multi-layered in order to provide long-term sustainable risk mitigation. Unstable trees in rockfall release areas can potentially increase rockfall probability due to the effect of wind and roots, which can loosen cliffs and outcrops (Dorren et al., 2005).

High surface roughness reduces kinetic energies of rocks and can change the paths of rockfalls. It influences contact angles of rocks and changes rock movement from falling to rolling and sliding (Wang and Lee, 2010). Surface roughness presents the micro topography of the slope and obstacles on the slope for the falling rocks (Dorren, 2016). In order to increase surface roughness in propagation areas in rockfall protection forests, it is recommended to promote and leave dead wood and stumps up to (1.3 m), and that logged trees are positioned perpendicular to the slope (Berger et al., 2013).

Influence of natural disturbances on forest stand parameters

Forest fire

In general, the mortality of trees is high in case of high-severity fires (i.e. crown fires), and trees with dbh < 35 cm are less resilient in the case of medium-severity fires (Maringer et al., 2016b). Compared to pedunculate oak (*Quercus robur* L.), sessile oak, sweet chestnut (*Castanea sativa* L.), and European beech are considered to be more susceptible to fire due to their thin bark and poor re-sprouting capabilities (Conedera et al., 2010; Maringer et al., 2016a; Dupire et al., 2019). European larch is highly resilient to mixed-severity (low, moderate, high) forest fires due to strong recruitment after fire (Moris et al., 2017). Silver fir, Norway spruce, mountain pine and Swiss stone pine are fire sensitive species, whereas Austrian pine and Scots pine can survive several surface fires of low to moderate severities (Dupire et al., 2019). In forest stands with dense canopies, low-severity fires can transition into a high-severity crown fires leading to an overall decreased canopy cover. This type of fire can kill large numbers of trees and will decrease the stem density (Graham et al., 2003; Kashian et al., 2005). When the majority of trees and understory vegetation is burned, surface roughness will decrease, and, if there is any forest remaining, it can be expected that the size of gaps will increase. European beech forests of the Southern Alps that were affected by low-severity fires had almost the same protective effect against rockfalls as unburned forests, whereas moderate- to high-severity fires greatly reduced their protective effect for the next 10 to 30 years (Dupire et al., 2016). Due to the abundant growth of post-fire colonizers and scarcity of seed-producing trees, poor regeneration of European beech can postpone the reestablishment of protection forests for a few decades (Ascoli et al., 2013; Maringer et al., 2016a, 2016b, 2016c). After forest fire, protective effects of forest against avalanches will decrease since: 1) there will be lower interception of snow leading to higher snow depths and increased snow gliding, 2) stand density will decrease once tree fall rates rise post fire, so that forests' abilities to reduce avalanche formation and to stop avalanches will decrease, and 3) gap sizes within forest stands will increase resulting in new potential release areas. Similar conclusions can be made for rockfall protection forest: 1) as tree density will decrease potentially in both release and propagation areas, fewer rocks will be stopped by the trees, 2) increased gap lengths lead to higher kinetic energies of rocks that will not be stopped by trees, and 3) surface roughness will decrease leading to fewer obstacles that could stop the rocks.

Windthrow, ice and snow break

Windthrow, ice and snow break can result in breakage of branches, tree tops, stems, and also in tree uprooting (Nykänen et al., 1997; Bragg et al., 2003). In general, survival rates of trees with low to moderate or even severe damage is high (Irland, 1998; Coons, 1999), although post-event disturbance agents such as insect outbreaks negatively influence survival rates (Bragg et al., 2003; Köster et al., 2012). Large-scale events can demolish whole forest stands, so that the protection forest can be completely lost (Schönenberger, 2002). Within the forest stands, the gap sizes will increase resulting in new potential avalanche release areas (Coates and Burton, 1997), and decreased length of forested slopes where rockfall deposit areas can increase. In case of dispersed damage and low-intensity windthrows, ice and snow breaks, the protective effect against avalanches and rockfalls can improve in even-aged forests (Frey and Thee, 2002), due to the increased surface roughness (downed logs, stumps) (Figure 3), and the potential shifts in structure caused by increases in light and nutrient availability, which favors pre-regeneration (e.g. Collet et al., 2008; Kramer et al., 2014). In the first 10 to 30 years after a disturbance, downed logs and stumps can act as barricades for avalanches (Frey and Thee, 2002; Kupferschmid

Albisetti et al., 2003) and rockfalls (Gerber, 1998; Wohlgemuth et al., 2017). Harvesting trees leads to less forest cover and tree density, resulting in reduced protective effects of forests (Brang et al., 2006; Teich et al., 2019). Compared to forest fires, windthrow, ice and snow break usually leave an intact tree regeneration layer and abundant logs (Franklin et al., 2002), which is favorable in terms of protective effects (Maringer et al., 2016a). Susceptibility of tree species to uprooting or stem breakage varies. Species (e.g. Norway spruce) with shallow roots are prone to wind damage as well as stands with lower stem densities, and large H/D ratios (Meunier et al., 2002; Quine and Gardiner, 2007; Klopčič et al., 2009; Schmidt et al., 2010; Albrecht et al., 2012; Pukkala et al., 2016; Díaz-Yáñez et al., 2019).

Figure 3: Uprooted trees after windthrow present obstacles for falling rocks. (Source: Domen Oven).

Snow avalanches

Smaller avalanches that flow through forests can break, uproot and overturn trees, while large avalanches can destroy large parts of mountain forest (Bartelt and Stöckli, 2001; Takeuchi et al., 2011; Feistl et al., 2014; Casteller et al., 2018). The damaging potential of avalanches that carry larger amounts of tree debris is higher due to the increase in high-density avalanche debris. However, the presence of dead wood will also increase the surface roughness and will prevent snow gliding (Putallaz et al., 2010; Teich et al., 2012; Feistl et al., 2013). Stand density and tree height in protection forests will be reduced due to avalanche activity (Patten et al., 1994;

Kulakowski et al., 2006). Avalanche magnitude and tree strength are the factors that influence the degree of forest destruction (Feistl et al., 2014). The breakage of stems is influenced by tree size, i.e. smaller trees tend to bend under snow pressure, while larger trees easily break (Johnson, 1987). Species composition may change after the avalanche event, resulting in a change from coniferous to mixed forest, which will result in lower interception of snow, higher snow accumulation and higher onset probability of snow avalanches (Veblen et al., 1994; Bebi et al., 2009). In avalanche tracks, small short-lived trees and shrubs (maple, willow, birch, alder) are often established (Holtmeier and Broll, 2018), because of their higher stem flexibility (Johnson, 1987). In the case of powder avalanches, higher trees are more susceptible to avalanche damage (Bebi et al., 2009). Due to the decreased canopy cover, tree size and stem density, the interception of snow is decreased leading to a higher probability of avalanche release, and the inability of forests to stop small avalanche events (Newesely et al., 2000). Increased avalanche activity can also be caused by the increased gap sizes and new non-forested areas. Subsequently, the rockfall activity in these areas might increase as well (Wasser and Frehner, 1996; Feistl et al., 2014).

Rockfalls

Small-scale rockfalls will damage individual trees, while larger events can demolish larger forest areas. The main types of tree damage due to rockfalls in forests are stem wounds, uprooting, partial fracture of the stem, explosion of tree stem into wood pieces, stem breakage and tree top breakage (Dorren et al., 2004). As a consequence of tree uprooting, rockfall activity might lead to the formation of rockfall paths, which follow the slope direction. In these areas, avalanche activity might increase as well (avalanches within the forest). This leads to the channelization of rockfall activity, greater frequencies and consequently larger impacts on trees. The velocities of rocks in non-forested parts will increase leading to larger impacts on trees, and to the state where trees cannot stop them due to higher kinetic energies. Injuries due to impacts will eventually result in tree death, which can lead to lower stand densities, increased gap sizes and a lowered length of the forested part of the slope. Since broadleaved trees are more resistant compared to coniferous trees, it is more likely that a reduction of coniferous trees can occur on rockfall slopes (Stokes et al., 2005, 2006). Surface roughness can increase in cases of uprooted trees, fallen trunks or tree tops (Schönenberger et al., 2002), which are additional barriers in the forest that can stop rocks or reduce their kinetic energies.

Insects and pathogens

Bark beetle outbreaks can change composition and structure of forest stands, and can alter the protective effects against avalanches and rockfalls. Large trees are more prone to mortality across a wide range of forest types and topographic conditions, since bark beetles target larger diameter trees. High stand densities are also often associated with increased tree mortality following insect infestations, especially if the stand is even-aged (Raffa et al., 2009; O'Brien et al., 2017; Pile et al., 2019). Bark beetles cause a decrease in canopy bulk density. This needle loss reduces canopy interception, increases light transmission and wind speeds, alters snow accumulation and melting, and changes the microstructural properties of subcanopy snowpack (Teich et al., 2019 and references therein), which could lead to a higher avalanche activity. However, a recent study from Teich et al. (2019) showed that even standing dead spruce trees could provide some avalanche protection. Species composition in avalanche protection forests might change drastically due to a high mortality of coniferous species and lead to a shift to more mixed and deciduous forests (Heurich, 2001; Stokes, 2002). Eventually, high tree mortality will lead to a decreased stand density, resulting in new and or larger forest gaps (Maroschek et al., 2015). This can create new avalanche release areas, and can lead to decreased protective effects of forest in the avalanche propagation areas. In case of rockfalls, decreased forest stand density and length of forested area will also reduce protective effects of forests by increasing the rockfall runout lengths, with new gaps potentially creating new release areas. Surface roughness does not change immediately after bark beetle infestation; however, dead trees eventually fall, resulting in an increase in surface roughness (Wohlgemuth et al., 2017). Leaving downed trees after other natural disturbance events (e.g. windthrow, ice and snow break, avalanche events) creates conditions which raise the risk for bark beetle infestations, which usually kill the remaining trees (Wermelinger, 2004), and can cause the spreading of the outbreak into the undisturbed parts of the protection forest.

Disease can kill trees or predispose them to mechanical failure (Franklin et al., 1987). Mortality of trees due to root rot disease in protection forest results in larger and longer gaps in the forest canopy, a decline in tree cover and the elimination of larger trees, which potentially leads to higher onset probabilities and greater spatial impact of avalanches and rockfalls (Newesely et al., 2000). Trees that are affected by root rot or other fungi are also more susceptible to windthrow (Papaik et al., 2005; Gonthier, 2012; Garbelotto and Gonthier, 2013), which can further expand gap sizes or lengths. Mainly conifer trees are affected by root rot, which shifts species composition towards broadleaves, leading to greater onset probability of avalanches; however, this can be in favor to preventing propagation of rockfalls. Snow fungi increase the mortality of tree especially near the treeline, lowering forest coverage and promoting snow gliding. Due to the absence of vegetation and lowered surface roughness, avalanches can develop greater velocities in longer paths without obstacles, resulting in bigger avalanches (Schneebeli and Bebi, 2004) (Figure 4).

Figure 4: Reduction in forest cover after bark beetle outbreak and salvage logging above the Dovje settlement (NW Slovenia) could lead to new snow avalanche release areas. Risk assessment after a natural disturbance event is crucial in such cases.

Overview of the influences of natural disturbances on protection forest stand parameters against avalanches and rockfalls

After a natural disturbance event, protective effects of forests against natural hazards are altered. That is, a natural disturbance event can have positive or negative consequences, no effect or an unclear effect. The scale of a natural disturbance event is a crucial factor that will influence the changes in the protective effects; while small-scale natural disturbance events might not be as important in altering the protective effects of forest, large-scale events can have devastating consequences. Here, the changes in protective effects are therefore described for the case of large-scale, high-severity disturbance events through the changes of individual forest stand parameters that are the most important ones of forests protecting against avalanches and rockfalls. Based on the little or no available literature of direct influences of natural disturbances on protective effects of forests against avalanches or rockfall, our best findings are summarized in Tables 1 and 2.

Table 1: Influence of natural disturbances on forest stand parameters of protection forest against avalanches. The symbols present different effects on protective effects of forest: + positive effect (increase), - negative effect (reduction), +- negative or positive effect, 0 no effect, ? effect unclear.

AVALANCHE PROTECTION FOREST							
stand parameter → / natural disturbance ↓	canopy cover	species composition	surface roughness	tree size relative to snow depth	stem density	gap size	dbh distribution
forest fire	-	-	-	-	-	-	-
windthrow	-	-	+	-	-	-	-
ice and snow break	-	-	+	-	-	-	-
avalanches	-	-	+	-	-	-	-
rockfalls	o, -	-	+	-	-	-	-
insects	-	-	o, +	-	-	-	-
pathogens	-	-	?, o	-	-	-	-

Table 2: Influence of natural disturbances on forest stand parameters of protection forest against rockfalls. The symbols present different effects on protective effects of forest: + positive effect (increase), - negative effect (reduction), +- negative or positive effect, 0 no effect, ? effect unclear.

ROCKFALL PROTECTION FOREST							
stand parameter → / natural disturbance ↓	trees in release areas	species composition	surface roughness	length of the forested part of slope	stem density	gap length	dbh distribution
forest fire	-, ?	-/+	-	-	-	-	-
windthrow	-, ?	-/+	+	-	-	-	-
ice and snow break	-, ?	-/+	+	-	-	-	-
avalanches	-, ?	-/+	+	-	-	-	-
rockfalls	-, ?	-/+	+	-	-	-	-
insects	-, ?	-/+	o, +	-	-	-	-
pathogens	-, ?	-o, -	o, -	-	-	-	-

Discussion and conclusions

In this report, we discussed the influence of natural disturbances on protective effects of forests against avalanches and rockfall. The main findings are summarized in Tables 1 and 2, where the influences of particular disturbances on important forest stand parameters in relation to avalanche and rockfall protection are shown. Due to lack of literature on the direct effects of the discussed disturbances on avalanche and rockfall protection, a few caveats should be considered:

The results are presented individually, while synergistic effects between parameters, i.e. where one stand parameter influences another, are not discussed. In addition, the findings present immediate effects of the disturbance event on stand parameters. The characteristics of each presented stand parameter should remain fairly constant for several years after the event, at least to the point when decay begins to break down deadwood, regeneration is established, or new disturbance events occur. Factors influencing seedling establishment (e.g. site conditions, competing vegetation) after disturbance events seem to be especially important (Kramer et al., 2014) since they can delay regeneration for a few decades (Wohlgemuth et al., 2017), and prevent the establishment of an adequate forest structure and its associated protective capacity. Therefore, post-disturbance management influences the quantity of deadwood and the regeneration capacity of the stands, which further affects the recovery of protective effects (Wohlgemuth et al., 2017).

The influences of disturbances on forests' protective effects presented in Tables 1 and 2 are generalized. That is, the impact of a particular disturbance event on individual forest stand parameters may be different in a specific case, and we did not account for synergistic or cascading effects between stand parameters or natural disturbances. The vulnerability of a stand is strongly related to the stand parameters. For example, the change in surface roughness after windthrow is likely different in spruce-dominated stands compared to beech-dominated forests, due to their different vulnerability/susceptibility to windthrow. In addition, other site or weather characteristics may have an influence as well. For example, after a windthrow event, surface roughness can be relatively high due to broken branches and trunks; however, freshly fallen snow can cover the majority of obstacles, lowering surface roughness and leading to greater onset susceptibility of avalanches. Snow fall depth of between 30 and 50 cm can be critical for the initiation of moderate size avalanches (Schweizer et al., 2003).

The influence of natural disturbances on forest parameters was only discussed for high-severity events. Resistance of forests to low-severity events is usually high. For example, forests' protective effects remained the same after low-severity forest fire (Maringer et al, 2016a). Low to moderate severity snow or ice breaks can increase surface roughness due to deadwood, but the majority of trees survive, and the protective effect of such stands can be even greater than before the event. Furthermore, low-severity forest disturbances are often part of the natural cycle and, therefore, beneficial due to pre-regeneration capabilities (Kramer et al., 2014), which can produce more diverse stands in the future (Wohlgemuth et al., 2017). Diverse forest stands are especially desired in protection forest management because their resistance and resilience are greater compared to mono-species, even-aged stands (Brang, 2001; Frehner et al., 2005; Jactel et al., 2017).

In this report, the influence of abiotic and biotic disturbances on avalanche and rockfall protection forests is presented mainly indirectly through the resulting changes in forest stand structure (e.g. gaps length, stem density) and forest cover. The influence of each individual disturbance on ecosystems may be relatively well understood, nonetheless understanding the interactions and cascading effects between onset and magnitude of different disturbance events of different origin (abiotic or biotic) remains understudied (with possible exception of bark beetle outbreaks after windthrow – e.g. Kulakowski et al., 2017). Both negative and positive feedbacks between disturbances will become important factors to consider in the future (Buma et al., 2015; Bebi et al., 2017). Studying cascading effects between disturbances could be done by integrating data on natural disturbances into risk analysis, and coupling forest dynamics models with natural hazard models in order to better understand the protective effects of forests in the face of disturbances and climate change (Maroschek et al., 2015; Moos et al., 2017).

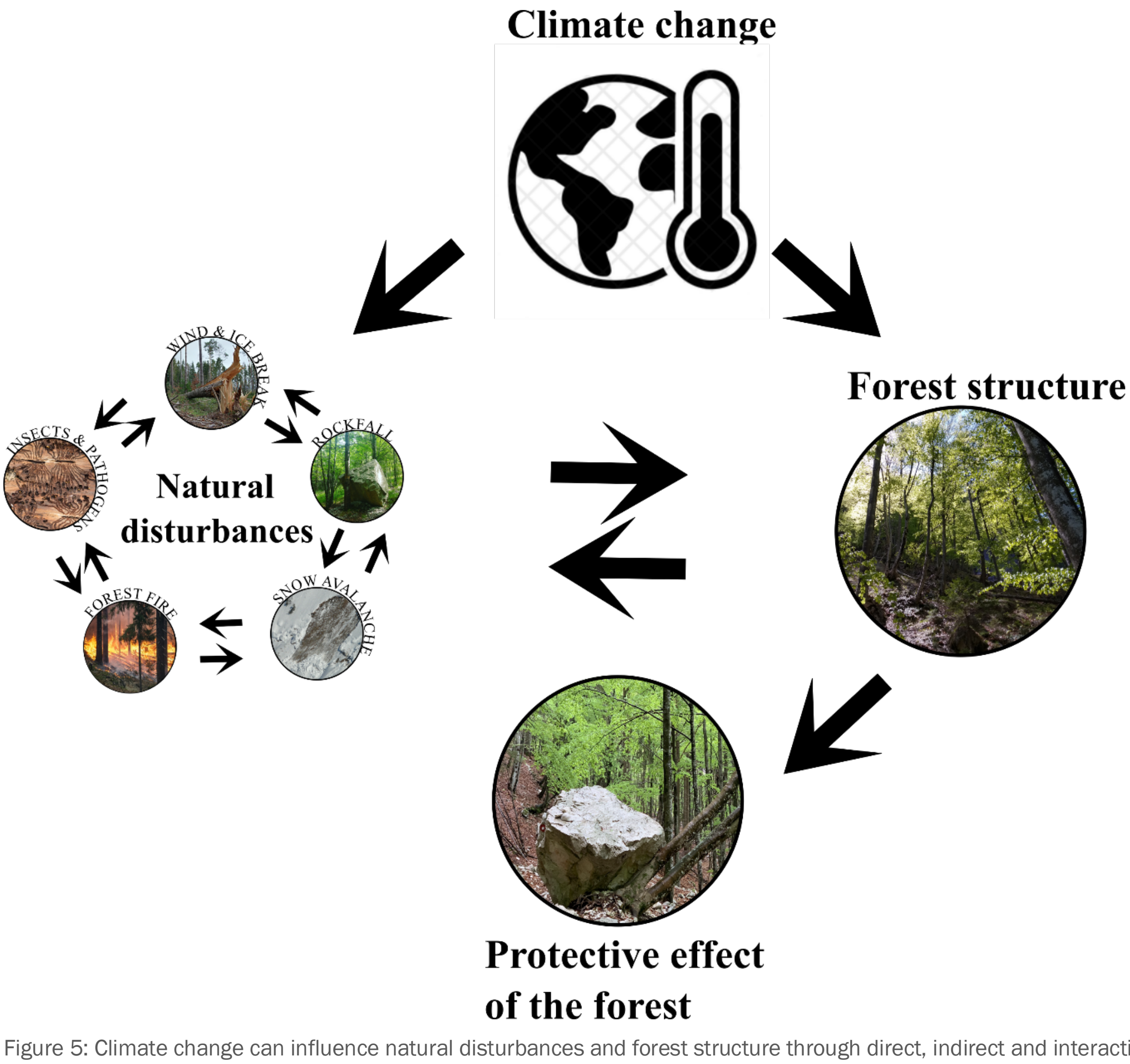

Figure 5: Climate change can influence natural disturbances and forest structure through direct, indirect and interactive effects. Natural disturbances interact between each other and can change forest structure and composition. In return, forest structure and composition influence natural disturbance regimes. Climate change influences, whether direct or indirect, forest structure and composition, which directly influences protective effects of forests.

Interactions between natural disturbances are expected to be further influenced by climate change, which will affect frequency and intensity of disturbances (Lindner et at., 2010, 2014; Seidl et al., 2011, 2017; Thom and Seidl, 2016). Climate change is expected to have a number of direct, indirect and interactive effects on natural disturbances and stand structure (e.g. Seidl et al., 2017), which could affect the protective effects of forests against natural hazards (Schumacher and Bugmann, 2006) (Figure 5).

Disturbance regimes that are influenced by temperature-related variables will have the highest importance in higher altitudes and boreal zones, and in coniferous forests (Seidl et al., 2017). Disturbance events that are most likely to increase in frequency and magnitude are windthrow, insect and pathogen outbreaks (Sturrock et al., 2011; Weed et al., 2013; Seidl et al., 2014a; Seidl et al., 2017), especially due to interaction between temperature increase and insect biology (Rouault et al., 2006; Battisti et al., 2005; Netherer and Schopf, 2010; Evangelista et al., 2011; Temperli et al., 2013; Maroschek et al., 2015). In addition, climate change might remove or relocate the barriers that limit present species ranges (Robinet and Roques, 2010), and influence their distribution ranges (Volney and Fleming, 2000; Lange et al., 2006; Netherer and Schopf, 2010) as well as the onset and propagation of natural hazards mainly through changes in forest structure, temperature, precipitation, freeze-thaw cycles, and snowpack characteristics (e.g. Lindner et al., 2010; Seidl et al., 2011; Berger et al., 2013; Castebrunet et al., 2014). Shifting natural disturbance regimes will most likely influence structure and dynamics of protection forests, especially forest cover extent, species composition, and gap size will be affected. This will consequently affect the protective effect and protective function of mountain forest ecosystems and presents a need for protection forest managers to re-evaluate the risk in altered ecosystems.

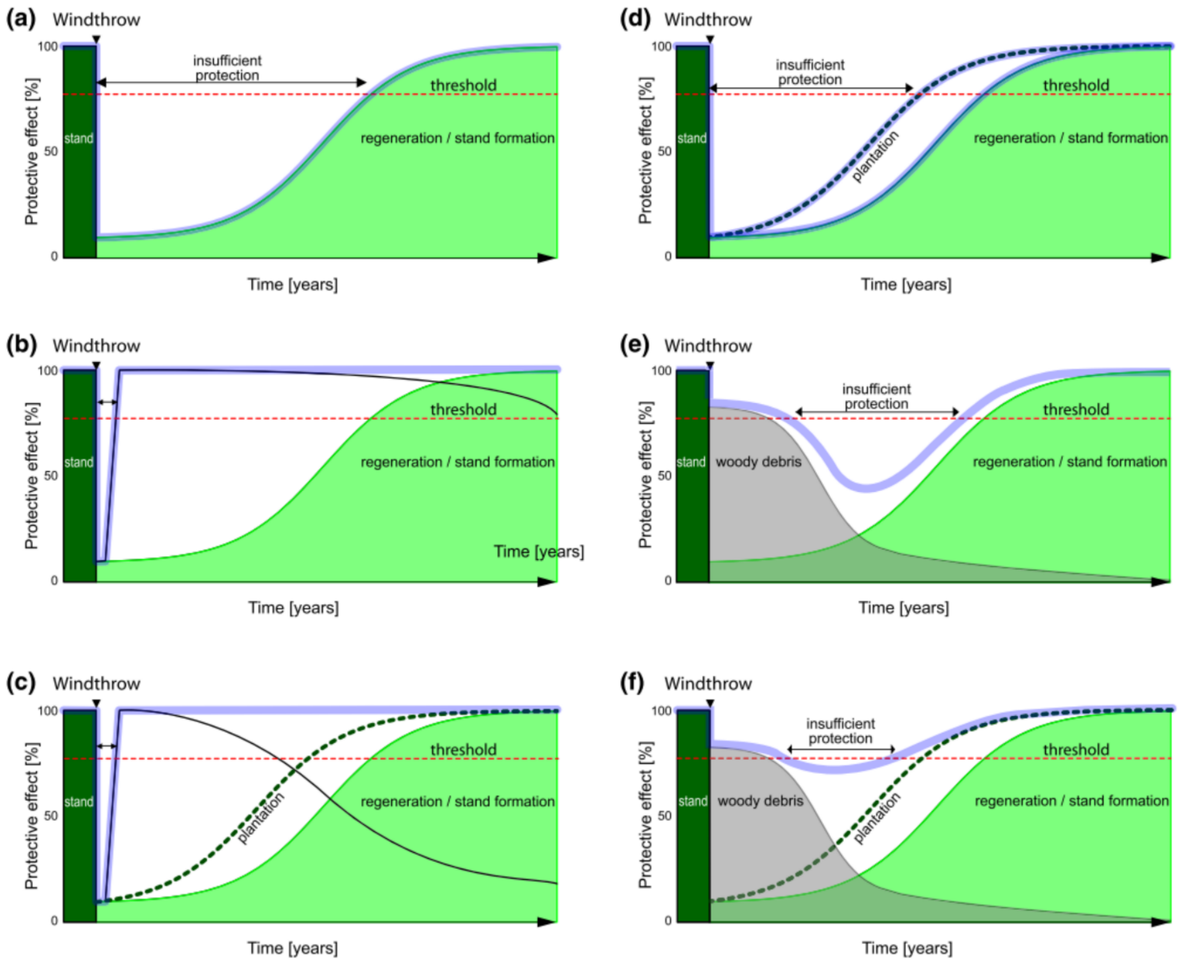

Figure 6: Protective effect of the forest (blue line) is influenced by post-disturbance management. After windthrow event, protective effect development differs under different scenarios: a) Salvage logging, natural regeneration, b) Salvage logging, permanent steel construction, c) Salvage logging, temporary wood construction and plantation, d) Salvage logging, plantation, e) No salvage logging, f) No salvage logging, plantation. (Wohlgemuth et al., 2017)

In conclusion, the protective effects of a forest stand against avalanches and rockfalls depend on 1) the spatial scale and intensity/severity of the natural disturbance, 2) the resistance and the resilience of a stand, and 3) the post-disturbance management (Bebi et al., 2015) (Figure 6). In Alpine mountain forests, few studies examined the risk after a disturbance event (e.g. Moos et al., 2017). Therefore, after a disturbance event a new assessment of the protective effects of the forest should be performed, and in cases where forest cover was completely lost or degraded, afforestation and/or technical solutions that offer protection against natural hazard has to be considered (Schönenberger and Wasem 1997; Schönenberger, 2002; Maringer et al., 2016a; Wohlgemuth et al., 2017). In order to sustain high protective effects in the face of natural disturbances, management of protection forests should increase forest resistance, resilience and elasticity with favoring species and structural diversity (e.g. mixed forests), adequate regeneration, and the presence of coarse woody debris (Brang and Lässig 2000; Kräuchi et al., 2000; Brang, 2001; Brang et al., 2001; Jactel et al., 2017). Changing natural disturbance regimes will be one of the future challenges in mountain forest management in the Alpine Space in regard to mitigation of natural hazards while maintaining other ecosystem services.

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