Assessing the effect of invasive tree species on rockfall risk – The case of Ailanthus altissima

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The invasive tree species *Ailanthus altissima* increasingly occupies forests in the southern parts of the Alps. Many of these forests grow on steep slopes and protect settlements and infrastructure from natural hazards, such as rockfall. It is feared that the mechanical properties of *Ailanthus* are less favourable for energy reduction of falling rocks due to lower wood strength as well as higher prevalence of heart rot. Therefore, the question has arisen whether the spread of *Ailanthus* will substantially change rockfall risk. The objective of this study is to analyse the influence of the spread of *Ailanthus* trees in forest stands on their protective effect against rockfall. We first quantify the block energy reduction capacity of single *Ailanthus* trees based on a block-tree impact model and compare it to that of other species. Subsequently, we analyse the effect of *Ailanthus* on rockfall risk for different forest scenarios with a varying proportion of *Ailanthus* at the stand scale. The capacity of Ailanthus to reduce the block energy was quantified using a model of the block impact on a tree based on the Discrete Element Method. We then integrated the obtained results in the rockfall trajectory model RockyFor3D. Based on rockfall simu-lations, we finally calculated rockfall risk for different forest scenarios representing current forest conditions and an increasing spread of *Ailanthus.* The energy reduction capacity of *Ailanthus* lies in the range of the species predominantly present in the study area, as well as other species that are typically found on rockfall slopes in the Alps. Rockfall risk for houses and roads does not increase with an increasing proportion of *Ailanthus* without change in the forest structure. Assuming a decrease in tree diameters with an increasing proportion of *Ailanthus* trees, rockfall risk, however, critically increased. Consequently, whether or not *Ailanthus* changes rockfall risk in the long term, strongly depends on its influence on the forest structure. To anticipate the evolution of protection forests invaded by *Ailanthus,* more long-term ecological data on growth and succession of *Ailanthus* at stand scale is required.

1. Introduction

Human activity, habitat alteration and climate change have led to an increased spread of tree species beyond their natural range. A few of them are exceptionally invasive and potentially cause substantial impact on ecosystems and human livelihoods (Richardson et al., 2014; Vitousek, 1997). An example is *Ailanthus altissima (Mill.)*, commonly known as tree of heaven, which is increasingly invading forests in the southern parts of the Alps (Knüsel et al., 2015). This deciduous tree native to northeast and central China and Taiwan grows rapidly and reaches heights of 15 m in 25 years. *Ailanthus* trees quickly colonise new gaps in forests caused by disturbances, such as fire, insect outbreaks and windthrow, and successfully suppress competition by inducing allelopathic compounds (Csiszár, 2009). They are tolerant to a wide range of ecological conditions and disperse fast due to high seed production and lateral shoot growth (Gómez-Aparicio and Canham, 2008; Sladonja et al., 2015).

Many of the forests in the Southern Alps invaded by *Ailanthus* grow on steep slopes and protect subjacent settlements and infrastructure from natural hazards, such as rockfall. Trees act as a barrier and can

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slow down, stop or deviate rocks (Dorren et al., 2007). In doing so, forests can reduce the frequency and intensity of falling blocks and, thus, rockfall risk at a given element at risk (e.g. house or road) (Moos et al., 2017). How effectively a forest reduces rockfall risk depends on the one hand on the size and energy of the falling blocks and, on the other hand, on the forest structure and condition (Bigot et al., 2009; Dupire et al., 2016). The key characteristics are tree density, diameter distribution, the spatial arrangement of trees, tree species, and the forested slope length from the release area to the element at risk (Dupire et al., 2016; Moos et al., 2017). Generally speaking, the denser a forest and the longer the forested slope length, the more evenly distributed the trees (i.e. no gaps), the larger their diameters and the higher their capacity to dissipate the energy of falling blocks, the more blocks will be stopped (Berger and Dorren, 2007; Cordonnier et al., 2008). In case a new species invades the forest, these characteristics can substantially be changed. On the one hand, the species may differ regarding its energy reduction capacity which may potentially influence forest effects on rockfall risk (Dorren et al., 2005). On the other hand, the species may influence the structure of the forest in terms of diameter distribution, tree density and spatial arrangement of trees.

The energy reduction capacity of trees of different species has been studied based on winching tests (Stokes et al., 2005), dynamic impact test (Bertrand et al., 2013) and in-situ rockfall experiments (Dorren et al., 2006; Jonsson, 2007; Lundström et al., 2009). Dorren and Berger (2005) found an exponential relationship between stem diameter and the maximum amount of block energy reduction based on experimental data. In general, more energy can be reduced by broadleaved compared to coniferous species (Dorren et al., 2007). Questions remain on the effect of the positions of the impact on a tree stem, the velocity of the rock and differences between tree species (Toe, 2016). To investigate these questions, Toe et al. (2017b) developed a tree impact model based on the discrete element method (DEM) integrating the response of the root system, the tree stem and the crown.

Due to the increased abundance of *Ailanthus* in protection forests in the Southern Alps, the question has arisen how effectively this species protects against natural hazards. In the case of rockfall, it is feared that the mechanical properties of *Ailanthus* trees are less favourable for energy reduction of falling rocks due to lower wood strength, fragile behaviour under static and dynamic loadings, as well as higher prevalence of heart rot (Knüsel et al., 2015). On the other hand, fast colonization of forest gaps (Maringer et al., 2012) may have a positive effect on the protective capacity of forests. Besides rockfall, this is also relevant for other natural hazards potentially occurring in these forests, such as shallow landslides or snow gliding. In addition, and depending on the natural hazard process, standing requirements for an effective protection may also differ (Sakals et al., 2006). In general, knowledge on its growth behaviour and on the evolution of invaded stands is scarce. Knüsel et al. (2015) found a lower sensitivity to drought for *Ailanthus* in comparison to *Castanea sativa*, which may promote the species with climate warming. Radtke et al. (2013) showed that coppice management, which is frequently practiced especially in chestnut forests in the Southern Alps, favours the spread of *Ailanthus*. In the future, an increasing invasion by *Ailanthus*, but also by other species, can thus be expected (Wunder et al., 2016).

To study the influence of *Ailanthus* on rockfall risk, we firstly assessed the energy reduction capacity of a single tree based on a block-tree impact model and compared it to that of other species (tree scale) and secondly, we calculated rockfall risk for different forest scenarios, with varying proportions of *Ailanthus* at the scale of an entire forest complex, based on rockfall simulations.

2. Materials and methods

2.1. Study site

The study site in Mendrisio (Fig. 1) is a south-east facing slope in the southern part of the Canton of Ticino (Switzerland) with an average length of approximately 150 m from below the release area and an average slope of approximately 35°. The total surface has a size of eleven hectare. The release area is a limestone cliff (*Calcare di Moltrasio*) of the tectonic unit of the Southern Alps and has an average height of 60 m. The slope is completely covered by forest, mainly consisting of *Ostrya* sp., *Tilia* sp. and *Carpinus* sp. In certain parts of the forest complex, a considerable percentage of *Ailanthus* with relatively small diameters (mean DBH ≈ 15 cm) can be found (see Table 2 and Fig. 2). Several rockfall nets at the bottom of the slope protect the houses and roads of the village Mendrisio. They were not considered in this study.

2.2. Protective capacity of single tree

We used a DEM model of the block impact on a tree developed by Toe et al. (2017a) to analyse the capacity to reduce the block energy of *Ailanthus*, of other species groups predominantly occurring in the study area (see Section 2.3.2; *Fraxinus*, *Carpinus*, *Tilia*, *Castanea*, *Ostrya*, *Acer*, *Quercus*), and of species most typically found in rockfall protection forests in the Alps (*Fagus*, *Abies*, *Picea*, *Pinus*). In the DEM model, the block is represented as a rigid spherical body and the tree as a deformable beam. When a block impacts the tree, forces are applied to the two bodies depending on their overlap and relative velocities (Bourrier et al., 2013).

The input parameters for the DEM model related to stem mechanical parameters were determined based on the literature (Niklas and Spatz, 2010; Table 1) and real-size rockfall impact experiments in the field on *Ailanthus* trees, for which literature values were missing. Additionally, we conducted three-point bending tests on 30 Ailanthus tree stems with a diameter ranging between 8.4 and 16.35 cm. These tests resulted in a mean value of the modulus of rupture (MOR) of 73 MPa (Table 1). The MOR is a measure for the wood strength before rupture (also referred to as bending strength). We only report MOR here, because it is the most sensitive parameter for energy reduction capacity. Since the root system stiffness and strength do mainly depend on the anchorage conditions and to a lesser extent on the tree species, the model parameters were set at mean values for all tree species according to data available in current literature (KR = $7.5*10^6$ Nmrad^{-1}, mRootMax = $4.75*10^5$ Nm; Lundström et al., 2009; Stokes et al., 2005; Toe et al., 2017b).

The DEM model was used to calculate the energy reduction of the considered tree species for varying block volumes (0–0.5 m^3), block velocities (5–40 ms^{-1}), tree diameters (0.05–0.6 m), and impact eccentricity (0–1).

The differences in the energy reduction of *Ailanthus* and the other tree species were compared with the *Wilcoxon rank sum test* and a level of significance of 5%. We further calculated the relative bias (i.e. relative difference) between each considered main species group and *Ailanthus* using a bootstrap simulation with 999 repetitions according to Eq. (1).

$$bias_i\% = \frac{(mean(Ered_{Species,i}) - Ered_{Ailanthus,i})}{mean(Ered_{Species,i})} * 100\% \tag{1}$$

With $Ered_{Species,i}$ the energy reduction of the respective species for the ith bootstrap simulation and $Ered_{Ailanthus,i}$ the energy reduction of *Ailanthus* for the ith bootstrap simulation.

2.3. Protective capacity of forest complex

The protective capacity of different forest scenarios was assessed using slope-scale rockfall simulations. The simulation results served as basis for the calculation of rockfall risk for houses and infrastructure at the bottom of the slope to evaluate the risk reducing effect of *Ailanthus*.

2.3.1. Rockfall simulations

Based on the results of the simulations with the DEM model for a

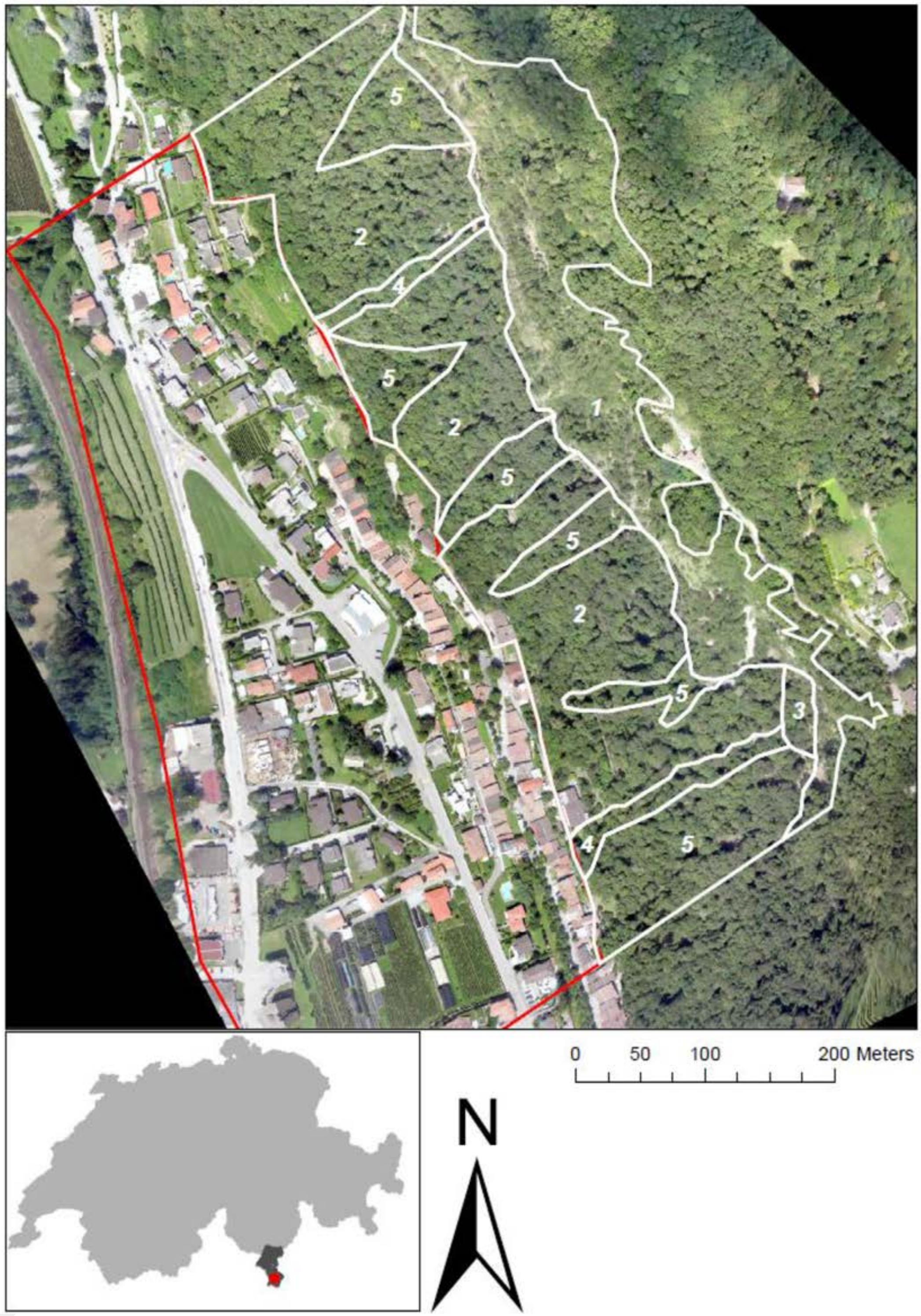

Fig. 1. Study area in Mendrisio (Ticino, Switzerland) with polygons with homogenous terrain characteristics (see Table 2).

variety of block volumes, block velocities, tree diameters and impact eccentricities, we derived a "tree factor" for the energy reduction capacity of each species. The tree factor represents the energy reduction capacity of a given tree species relative to *Abies alba* (Dorren and Berger, 2005). We determined the tree factors using linear regression between the energy reduction of the corresponding species and the energy reduction of *Abies alba* simulated with the tree impact model. The tree factors were then implemented in the rockfall trajectory model RockyFor3D (Dorren, 2015). This model calculates block trajectories in three dimensions based on deterministic algorithms with stochastic approaches. The modes of motion of the blocks represented in the model are flying, bouncing, and rolling, which is represented as a series of individual rebounds with a minimum displacement distance of twice the block radius in between two rebounds. The energy loss during the rebound is influenced by the slope surface roughness, the capacity of the surface material to dissipate energy as well as standing or lying trees. The underground is classified in seven different soil types predefining the normal coefficient of restitution (Rn), which is used to

Table 1
Modulus of rupture (MOR) for the considered main species based on literature values[1] (Niklas and Spatz, 2010) and real-size rockfall impact experiments[2] and bending tests[2].

Species	MOR [MPa]
ABIES[1]	43
PICEA[1]	39
PINUS[1]	41
LARIX[1]	53
ACER[1]	66
CARPINUS[1]	66
CASTANEA[1]	52
FAGUS[1]	65
FRAXINUS[1]	66
QUERCUS[1]	59
TILIA[1]	54
AILANTHUS[2]	55

Table 2
Homogenous polygons and their terrain characteristics: Rg10: mean obstacle height (MOH) in 10% of the surface; Rg20: MOH in 20% of the surface; Rg70: MOH in 70% of the surface. Soil types according to the classification of RockyFor3D (Dorren, 2015).

Polygon	Rg10 [m]	Rg20 [m]	Rg70 [m]	Soil type
1	0	0	0	5 (Bedrock with weathered material)
2	0.1	0	0	4 (Talus slope)
3	0	0	0	4
4	0.1	0	0.1	4
5	0.15	0	0.15	4

calculate its energy dissipative capacity. Surface roughness is represented by a mean obstacle height (MOH) representative for 70%, 20%, and 10% of the area of each homogeneous terrain unit. Trees are considered spatially explicitly, and each tree is characterized by its species and DBH. The deviation and energy loss after a tree impact is calculated depending on tree diameter, impact eccentricity, height and angle, as well as block volume and the species-specific tree factor.

2.3.2. Terrain characterisation and forest scenarios

The rockfall release area was determined based on high-resolution orthophotos, slope inclination calculated from digital elevation models and field surveys. In the simulations, blocks were released from a horizontal line (one pixel in width) in the vertical centre of the release area (Fig. 1). We assumed a rock density of 2700 kg m^{-3}, which is representative for the observed limestone lithology. Block shapes were classified as rectangular and block volumes were uniformly sampled within the volume range of the respective block volume scenarios (see Section 2.3.3). The volume of each simulated block was then again randomly varied by +-5%. Block fragmentation was not considered. We mapped homogenous terrain units (polygons) of terrain roughness and soil types following (Dorren, 2015) (Fig. 1; Table 2). Buildings are assumed to stop falling blocks immediately during the first impact.

We considered four different forest scenarios representing current forest conditions and an increasing spread of *Ailanthus* (Table 3). The current forest (scenario *today*) was characterized based on measurements in 10 square inventory plots of 20 × 20 m (planimetric) which cover all the representative forest stands on the slope (with homogenous tree density, species and tree diameter distributions). In each plot, we measured diameter at breast height (DBH) and species of all trees with a DBH > 5 cm. We considered 7 different "main species groups" (MSG), which have a proportion of ≥10% in at least one of the 10 homogeneous stands (*Ailanthus*, *Ostrya*, *Tilia*, *Fraxinus*, *Castanea*, *Acer*, *Quercus*). The remaining species were classified as "remaining broadleaved". Based on the stem number of each MSG, we generated random tree positions. Their DBH were determined based on a gamma distribution defined by a shape and scale parameter derived from the mean and the standard deviation of the DBH (Dorren, 2015).

In the forest scenario *today_without*, we replaced all *Ailanthus* trees in the current forest with other species according to their share in the respective stand and assigned the DBH according to the DBH distribution of the new MSG. We further generated three hypothetic forest scenarios with a constant stand density, but a steadily increase of the proportion of *Ailanthus* by 10% of the rest of the trees starting from the current forest. In the forest scenario *increase_1*, we solely replaced the species, but not the DBH. In forest scenario *increase_2*, we replaced the species and assigned DBH according to the DBH distribution of current *Ailanthus* trees. In the forest scenario *increase_3*, we replaced the species and assigned DBH according to an adult *Ailanthus* stand (average age of *Ailanthus* trees ~55 yrs) in Northern Ticino (Claro TI), where data on DBH distribution was available from previous work (Knüsel et al., 2015).

2.3.3. Risk calculation

We define rockfall risk as the product of the probability of a damaging rockfall event and its consequences (e.g. Agliardi et al., 2009; Moos et al., 2018). The probability of the event will depend on the onset frequency ($F_{onset,j}$) of block volume j and the probability that the block will reach the element at risk i (*propagation probability*, $P_{prop,i,j}$). The consequences will depend on the element at risk, i.e. its value E_i, its *presence probability* $P_{presence,i}$ and its vulnerability $V(I)_{i,j}$. The vulnerability $V(I)_{i,j}$ expresses the capability of the element at risk to resist to or recover from the event depending on its intensity I (Cannon, 2006). The

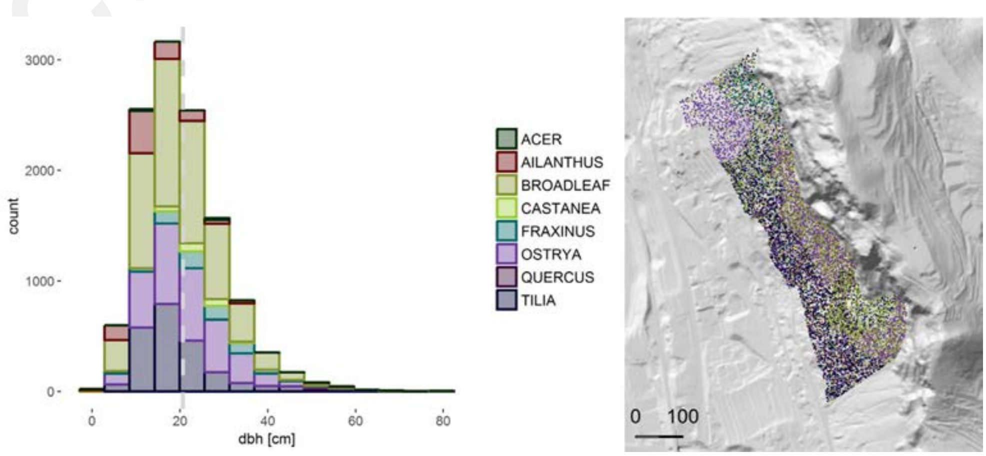

Fig. 2. DBH distribution (for the total area) and spatial arrangement of species of the current forest complex at Mendrisio (mean DBH = 20.7 cm).

Table 3
Description and characteristics of forest scenarios used in this study.

Forest Scenario	Description	Stem density [ha^{-1}]	Mean DBH [cm]	Tree species composition (MSG)
Today	Current forest structure based on measurements in the field	1090	BROADLEAVED: 20.5 OSTRYA: 21.7 TILIA: 17.7 AILANTHUS: 14.5 FRAXINUS: 26.1 CASTANEA: 24.0 ACER: 20.7 QUERCUS: 51.0	BROADLEAVED: 36.5% OSTRYA: 26.1% TILIA: 21.1% AILANTHUS: 10.1% FRAXINUS: 3.7% CASTANEA: 2.4% ACER: 0.09% QUERCUS: 0.04%
Today_without	Current forest structure without *Ailanthus*. Present *Ailanthus* trees were replaced with species of same stand according to their proportion.	1090	BROADLEAVED: 20.3 OSTRYA: 21.8 TILIA: 17.4 FRAXINUS: 26.2 CASTANEA: 24.1 ACER: 20.7 QUERCUS: 51.0	BROADLEAVED: 39.1% OSTRYA: 28.7% TILIA: 24.9% FRAXINUS: 3.7% CASTANEA: 2.9% ACER: 0.09% QUERCUS: 0.04%
Increase_1	Steadily increase of the proportion of *Ailanthus* by 10% of the rest of the trees; without changing DBH	1090	Total forest complex: 20.7	AILANTHUS: 10.1–99%
Increase_2	Steadily increase of the proportion of *Ailanthus* by 10% of the rest of the trees; with changing DBH (Ailanthus trees in stand)	1090	Total forest complex: Decreasing from 20.7 to 14.5	AILANTHUS: 10.1–99%
Increase_3	Steadily increase of the proportion of *Ailanthus* by 10% of the rest of the trees; with changing DBH (Ailanthus trees in adult stand in Claro, TI)	1090	Total forest complex: Decreasing from 20.7 to 20.2	AILANTHUS: 10.1–99%

Table 4
Catalogue of events recorded at the study site for the period between 1996 and 2017 (SFT, 2017). V_{max}: maximum observed volume; V_{tot}: total observed volume.

Year	V_{max} [m^3]	V_{tot} [m^3]	Number of blocks
2009	< 0.5	NA	< 10
2005	0.004	0.004	1
2003	0.3	1.0	4
2002	0.02	0.02	1
2001	0.3	NA	NA
2000	< 2	< 2	1
1999	0.032	0.032	1
1998	< 0.5	< 0.5	NA
1996	< 0.5	< 0.5	< 10
1996	0.0045	0.0045	1

Table 5
Block volume scenarios and respective return periods considered in the risk analysis at Mendrisio.

Block volume scenario [m^3]	0.1–0.3	0.3–0.7	0.7–1.5	1.5
Return period [year]	10	30	100	300

risk of a specific element at risk i per block volume scenario j finally represents the expected consequences of all possible events $n_{i,j}$:

$$R_{i,j} = F_{onset,j} \times P_{prop,i,j} \times E_i \times P_{presence,i} \times \frac{1}{n_{i,j}} \sum_{n_{i,j}} V(I)_{i,j} \tag{2}$$

The total risk is the sum of the risks of all elements at risk e for all expected volume scenarios v:

$$R = \sum_{i=1}^{e} \sum_{j=1}^{v} R_{i,j} \tag{3}$$

We considered only direct hits of single blocks and calculated risk for four block volume scenarios with return periods of 10, 30, 100 and 300 years, which are typically considered in risk analysis according to the Swiss risk concept (Bründl et al., 2009). The respective block volumes (Table 5) were derived from a power law based magnitude-frequency relationship. Power law distributions have been proven to fit the release volume distribution of rockfalls (Dussauge-Peisser et al., 2002; Hantz et al., 2003). They have the general form:

$$F(V \geq V_j) = \alpha V_j^{-ß} \tag{4}$$

where $F(V \geq V_j)$ is the annual exceedance frequency of any specific volume V_j and α and ß are site-dependent parameters. In case there is a minimum volume defined, the annual exceedance frequency can be expressed as:

$$F(V \geq V_j) = A \times \left(\frac{V_j}{V_0}\right)^{-\beta} \tag{5}$$

where A is the average onset frequency of blocks with $V \geq V_0$. We estimated A to $1\,yr^{-1}$ for volumes larger than $V_0 = 0.02\,m^3$ based on a catalogue of rockfall events from 1996 to 2017 registered by the cantonal authorities (SFT, 2017; Table 4) and validated it by assessing rockfall deposits and tree damage below the rock cliff. We counted all visible tree damages classified as "fresh" (younger than approximately 1 year) in a strip of 5 m width 20 m below the cliff, based on the work of Trappmann and Stoffel (2013). We then used the "conditional impact probability" (CIP) concept (Moya et al., 2010) assuming rocks with a diameter > 10 cm to cause damage to a tree. This resulted in a yearly rockfall frequency of $0.95\,yr^{-1}$, which corresponds well to the estimated A. The ß parameter of 1.3 was determined based on literature values for similar geologies (Ruiz-Carulla et al., 2017) and the resulting volumes were validated with observations in the field.

The *propagation probability*, P_{prop}, of a falling block depends on its size and shape, the slope characteristics (e.g. damping capacities of the soil) and standing or lying trees. To determine $P_{prop,i,j}$, block trajectories were simulated with the rockfall model RockyFor3D (Dorren, 2015). $P_{propi,j}$ at the element at risk i and for a block volume j is the ratio of the potential rockfall trajectories of volume j through i ($n_{i,j}$) and the total number of simulated blocks (n_{tot}):

$$P_{prop,i,j} = \frac{n_{i,j}}{n_{tot}} \tag{6}$$

The intensity I corresponds to the energy of the possible events of a scenario j at the element at risk i and was classified as low (0–30 kJ), medium (30–300 kJ) and high (> 300 kJ) according the Swiss risk concept (Bründl et al., 2009) for each block reaching an element at risk i. The monetary value E_i of an element at risk i was calculated as the sum of object values, using standardized base values for different object categories, and the monetized values of persons present (given as 5 Mio CHF $person^{-1}$ in Switzerland; Bründl et al., 2015). The calculation requires the following parameters: building category, building volume,

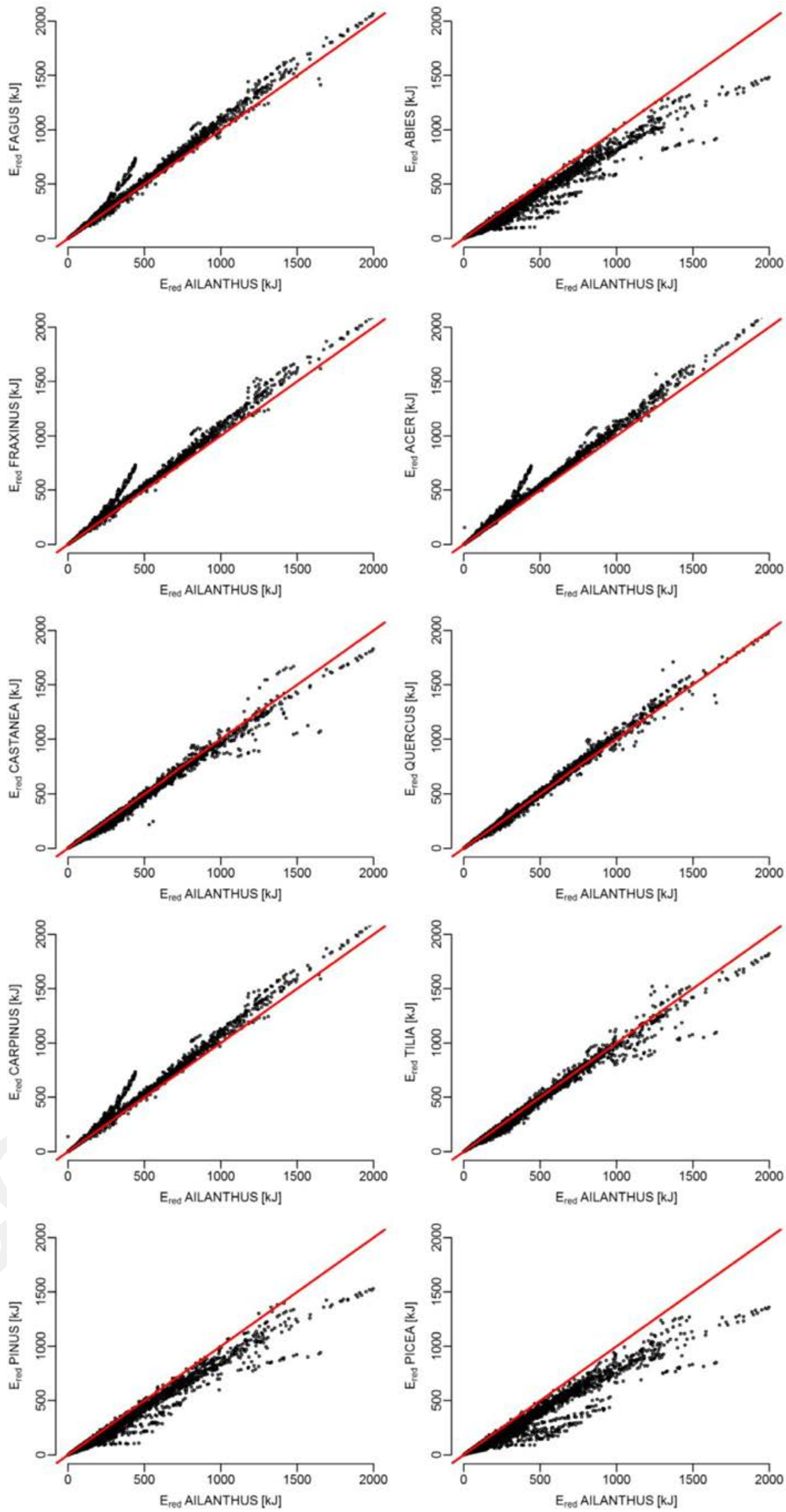

Fig. 3. Energy reduction values derived from simulations with the DEM model for the considered main species groups (y-axis) and *Ailanthus* (x-axis). The red line has a slope of 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

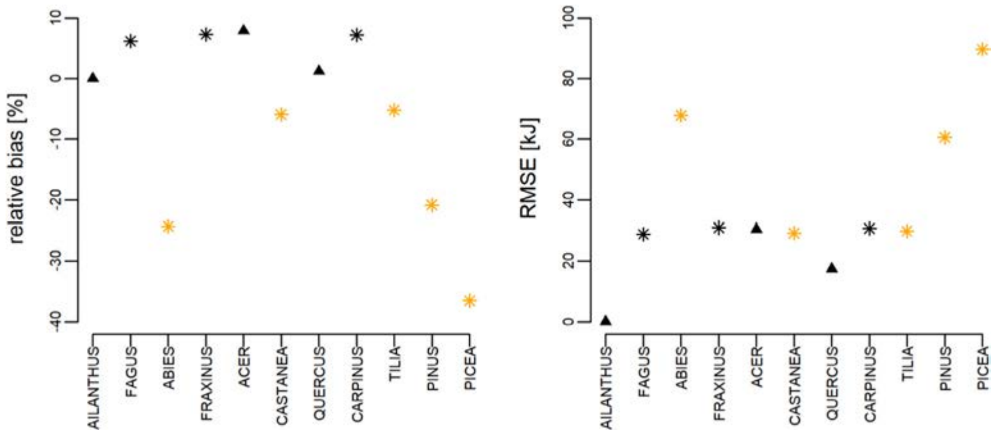

Fig. 4. Relative bias [%] calculated in a repeated bootstrap simulation (left) and RMSE (right) between *Ailanthus* and the other main species groups. *Stars:* Difference in energy reduction capacity is significant according to the Wilcoxon-rank sum test. *Orange*: Ailanthus has higher energy reduction capacity. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 6
Species-specific tree factor based on linear regression between the energy reduction of the considered main species groups and *Abies alba* (with a tree factor of 1).

Main species group	Tree factor	Adjusted R^2 of linear regression
ABIES	1.0	
FAGUS	1.3	0.97
AILANTHUS	1.2	0.97
FRAXINUS	1.3	0.97
ACER	1.3	0.97
CASTANEA	1.1	0.98
QUERCUS	1.2	0.98
TILIA	1.1	0.98
CARPINUS	1.3	0.97
BROADLEAVED	1.3	0.99
PINUS	1.0	0.99
PICEA	0.9	0.99

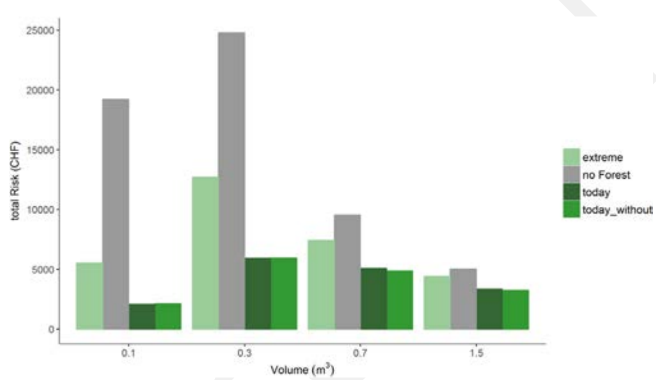

Fig. 5. Total risk per block volume for the current forest (today), the current forest without *Ailanthus* (*today_without*), an extreme scenario with 99% *Ailanthus* with the diameter distribution of the currently present *Ailanthus* trees (*extreme*) and the risk without forest (*no Forest*).

total number of persons in residential buildings, average presence probability ($P_{presence}$), road category and mean daily traffic (MDT). We estimated them based on freely available geodata (Swisstopo, 2017), census data from municipal and cantonal authorities as well as information from orthophotos and Google Maps and Street View. We considered several residential and industry buildings as well as two main roads and several residential roads as elements at risk (see Appendix, Table 1). The MDT of the two main roads was determined based on cantonal traffic census and upon consultation by the communal authorities and set to 4000 and 15,000 vehicles/day, respectively (see Appendix, Table 2). For the residential roads, it was estimated based on the number of houses that is reached assuming one car per house and two drives per day. In this study, we considered only physical vulnerability, which depends on the type of objects and the exposure of persons as well as the rockfall intensity I. We used vulnerability values for objects and persons for the intensity classes according to Bründl et al. (2009) (see Appendix, Table 3).

3. Results

3.1. Protective effect of a single tree

The energy reduction capacity of *Ailanthus* lies in the range of the species predominantly present in the study area, as well as other species that are typically found on rockfall slopes in the Alps. This is evidenced by the relatively low bias in the range of -10 to 10% and the RMSE in the range of 20–40 kJ for most main species groups except for the coniferous species *Abies*, *Pinus* and *Picea*, which yield a distinctly higher bias ($< -20\%$) and RMSE (> 60 kJ; Fig. 4). The energy reduction capacity of *Ailanthus* is significantly lower than the energy reduction capacity of the following main species groups: *Fagus*, *Fraxinus*, and *Carpinus* (Fig. 3 and Fig. 4). On the other hand, *Ailanthus* has a significantly higher energy reduction capacity than *Abies*, *Castanea*, *Tilia*, *Pinus*. and *Picea*. No significant differences can be found compared to *Acer* and *Quercus*. Based on linear regression between the energy reduction of *Ailanthus* and *Abies alba*, we derived a tree factor of 1.2 for *Ailanthus*, which lies in the range of the tree factor of other species (1.0–1.3; Table 6).

3.2. Protective effect of the forest complex

Risk for subjacent houses and roads is not substantially lower for the current forest without *Ailanthus* (total risk = 16,200 CHFyr^{-1}; Fig. 5) compared to the current forest (total risk = 16,400 CHFyr^{-1}; Fig. 5). Both scenarios reduce rockfall risk by about 72% (1490 CHF/ha forest and year) compared to the situation without forest. With a steady increase in the proportion *Ailanthus* trees, risk does not change if the diameter distribution remains constant (Fig. 6). In case the diameters of the trees are also changed according to the current diameter distribution of *Ailanthus* (increase_2), a steady increase of rockfall risk can be observed, with an almost doubled risk of 30,100 CHFyr^{-1} for the extreme scenario (99% *Ailanthus* with diameter reduction). The increase is particularly pronounced for volumes of 0.1 and 0.3 m^3. If diameters are changed according to an older *Ailanthus* stand (increase_3), risk increases only slightly with increasing proportion of *Ailanthus* (maximum risk = 18,800 CHFyr^{-1}). The change in risk is mainly due to an increase in the number of blocks reaching the elements at risk (Fig. 7). The energy of the block is only slightly enhanced with an increasing difference between the scenarios *increase_1* and *increase_3*, respectively, and *increase_2* with increasing volume.

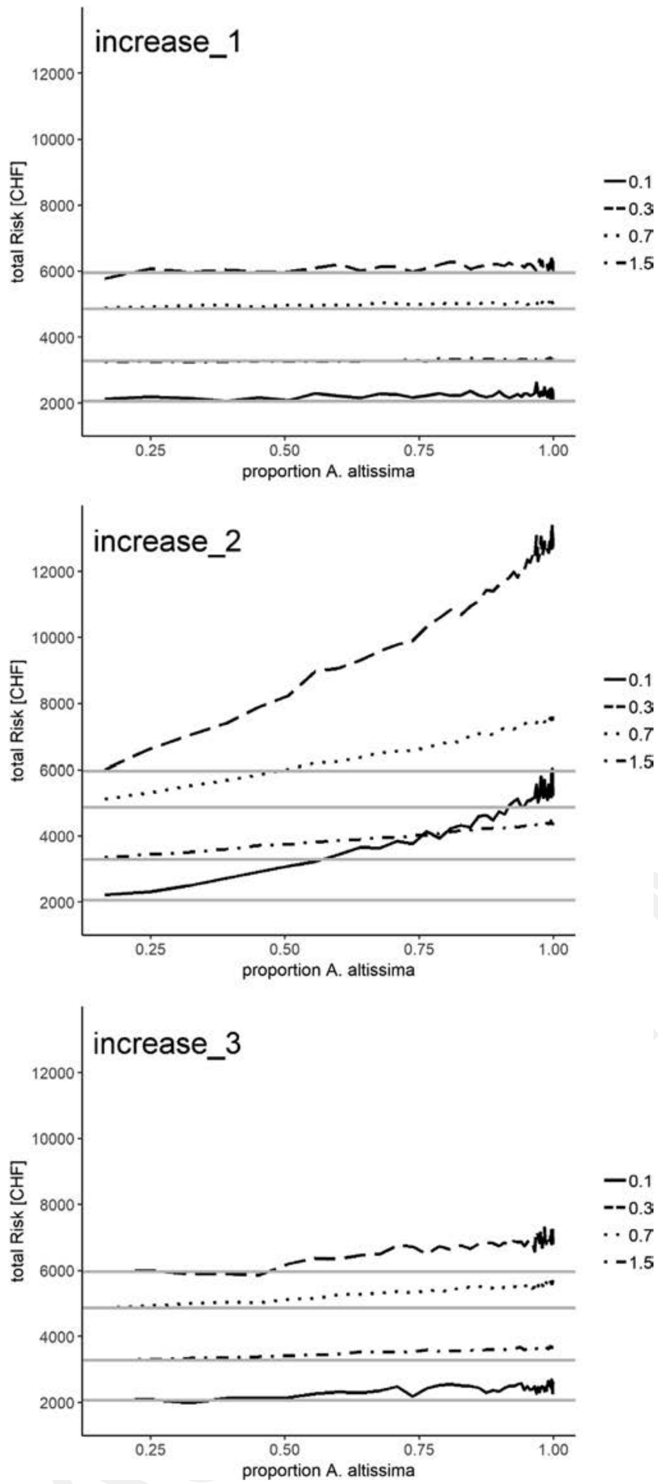

Fig. 6. Total rockfall risk (y-axis) depending on the proportion of *Ailanthus* (0 = no *Ailanthus*; 1.0: only *Ailanthus*) with constant diameter distribution (*increase_1*), changing diameter distribution according to the diameter distribution of current *Ailanthus* trees in the stand (*increase_2*), and changing diameter distribution according to an older *Ailanthus* stand in the region (Claro, TI) (*increase_3*).

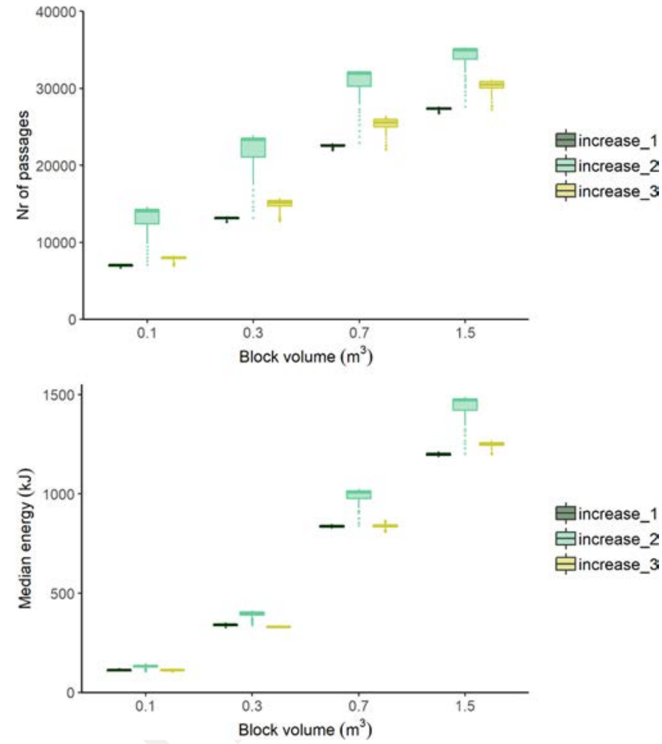

Fig. 7. Distribution of the number of passages and the block energies (median of all registered energies) at the bottom of the slope for all proportions of *Ailanthus* in the *increase_1*, *increase_2* and *increase_3* scenario.

4. Discussion and conclusions

In this study, the effect of the invasive neophyte *Ailanthus altissima* on rockfall risk was quantified for a forest complex based on detailed species-specific data at the tree scale. With a block-tree impact model, we assessed the energy reduction capacity of *Ailanthus* trees and compared it to other common species. This served as basis to quantify the effect of *Ailanthus* on rockfall risk, i.e. the yearly expected damage to human live and infrastructure, for different realistic and extreme forest scenarios at a case study site.

The energy reduction capacity of *Ailanthus* does not critically differ from other tree species typically found on rockfall slopes in the Alps and from the species currently present at the study site in Mendrisio. Accordingly, rockfall risk for subjacent houses and roads does not significantly increase with increasing abundance of *Ailanthus* under the assumption that there is no change in the current forest structure. Assuming a decrease in tree diameters with increasing abundance of *Ailanthus* trees – based on the diameter distribution of currently present *Ailanthus* trees – rockfall risk critically increases. The risk for this extreme scenario (99% *Ailanthus* with reduced diameters) is twice the risk of the current forest. This is, however, a mere diameter, and not a species effect. Compared to the non-forested situation, rockfall risk is still reduced by 50% in the extreme scenario. The comparison with an adult *Ailanthus* stand (on average 55 years according to Knüsel et al. (2015); the age of trees in Mendrisio is not known) with larger diameters revealed only a slight increase in risk. Consequently, whether *Ailanthus* changes rockfall risk in the long term, strongly depends on its influence on the local forest structure regarding stem density and diameter distribution. This supports recent studies showing that forest structure predominantly affects the protective capacity of a forest against rockfall compared to species effects (Dupire et al., 2016; Toe et al., 2017b).

How temperate forests invaded by *Ailanthus* will evolve with proceeding succession is still highly uncertain (Knüsel et al., 2015). *Ailanthus* is a typical pioneer species mainly invading forest gaps where much light for regeneration is available (Call and Nilsen, 2003). Maringer et al. (2012) found increased abundance of *Ailanthus* in burnt forests in Southern Switzerland. Radtke et al. (2013) showed that coppice management of forests favours the invasion of *Ailanthus* due to a colonization of clear-cuts. Knüsel et al. (2017) found, however, that

Ailanthus has a higher shade tolerance than previously stated and that it is also able to survive and grow in low light conditions. More long-term ecological data on growth and succession of *Ailanthus* at stand scale is required. Forest succession modelling may then provide new insights regarding the future evolution of protection forests invaded by *Ailanthus*. The fast colonization of disturbed and bare areas can also be advantageous with regards to protection from rockfall, because gaps are rapidly closed, and the protection capacity of the forest is only reduced for a short period.

There are studies indicating high susceptibility of *Ailanthus* trees to heart rot (Kasson et al., 2013). Consequently, a potential decrease of the energy reduction capacity of *Ailanthus* trees is probable (Stokes et al., 2005). However, Knüsel et al. (2015) did not found significantly more decay in *Ailanthus* trees compared to *Castanea* in forests of Southern Switzerland, with high susceptibility of both species. *Castanea* is originally also an introduced species in the area (in ∼0 AD). To assess the effect of heart rot on the protective capacity of forests, we need, firstly, mechanical tests on both, healthy and rotten trees, to quantify the difference in energy reduction. Secondly, the susceptibility to heart rot has to be quantified on stand scale. Open questions also remain on the energy that can be dissipated by the root system. The anchoring of trees and, consequently, the prevalence of uprooting in case of a rock impact strongly depends on the interaction between the root system and the local soil conditions. The required data related to this topic could be acquired using tree winching experiments in a range of different soil types.

Invasive species do not only potentially alter the protective function provided by forests, but also other ecosystem services. Biological invasions are classified as one of the major causes for biodiversity loss worldwide (Chapin et al., 2000; McGeoch et al., 2010). They spread into ecosystems and can alter basic ecosystem processes, such as nutrient cycling, and, thus, displace indigenous species. Furthermore, wood of *Ailanthus* is generally of poorer quality compared to most species present in typical forests of Southern Switzerland, potentially resulting in losses in wood sells (Sladonja et al., 2015).

The control of the invasion by *Ailanthus* proved to be a difficult task (Sladonja et al., 2015). Only chemical treatment of cut stumps showed long-term success, whereas mechanical removal is often contra-productive due to the extreme re-sprouting ability of *Ailanthus* (Constán-Nava et al., 2010; Wunder et al., 2016). Whether or not a targeted control of *Ailanthus* in protection forests is reasonable in the long term strongly depends on the current forest structure and species present (Wunder et al., 2018), and, therefore, the relative effect of *Ailanthus* on rockfall risk. Since the results of this study suggest that *Ailanthus* does not have a significantly lower energy reduction capacity compared to the native species on active rockfall slopes in the Alps the high costs of invasion control cannot be justified if the forest structure is not significantly altered over time due to the invasion of *Ailanthus*. This especially applies at sites with a low rockfall risk, i.e. a low damage potential or low frequency and intensity of rockfall events.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2019.03.001.

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