DISS. ETH. NO. 16358

Mountain forest dynamics and their impacts on the long-term protective effect against rockfall - A modelling approach

A dissertation submitted to the

SWISS FEDERAL INSTITUTE OF TECHNOLOGY ZURICH

for the degree of

DOCTOR OF SCIENCES

presented by

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Zürich, December 2005

“O beatissime lector, lava manus tuas et sic librum adprehende, leniter folia turna, longe a littera digito pone. Quia qui nescit sciscere, putat hoc esse nullum laborem. O quam gravis est scientia: oculos gravat, renes frangit, simul et omnia membra contristat. Decem digita scribunt, totus corpus laborat...“

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Summary

Many mountain forests effectively protect people and assets against natural hazards. Their protective effect is, however, not constant but depends on forest dynamics. These dynamics can be influenced by silvicultural measures, but the influence of many measures on the long-term protective effect of a forest is often uncertain. In order to optimize the management of protection forests in a long-term view, it would therefore be essential to gain more knowledge on the influence of such measures on the long-term protective effect. By doing so, the efficiency and efficacy of alternative silvicultural measures could be assessed.

Unfortunately, the effects of silvicultural measures and even of natural forest dynamics on the protective effect against natural hazards are difficult to study due to the long time periods involved in mountain forest dynamics. Given these difficulties, simulation models provide most valuable tools to investigate the protection forest system and to gain knowledge for optimizing the management of protection forests. A useful simulation tool for this purpose should thereby not only allow accurate predictions of structural forest patterns, but it should also be able to accurately assess the level of protection provided by the predicted stand structures. Such a combined tool does not exist up to date, but it could be designed by joining existing models of forest dynamics and natural hazards. Thus, the aim of this thesis was to develop a prototype of a new combined simulation tool (*CoST*), which allows investigating the effects of mountain forest dynamics on the long-term protective effect against rockfall. To achieve this aim, promising model candidates for a *CoST* were first evaluated. Detected model shortcomings were then improved before combining two models in a *CoST.*

Section I of this thesis was dedicated to the evaluation of two promising model candidates for a *CoST.* In *Paper I*, the forest patch model ForClim was shown to accurately predict structural patterns of mountain forest stands for several decades, if slight modifications of the establishment and mortality submodels were used. Thus, ForClim was found to be in principle suitable for a *simplistic CoST*, if two of the major shortcomings detected were improved. These shortcomings included (1) the simplistic representation of tree establishment, which led to unrealistically high numbers of young trees, as well as (2) the reproduction of the light competition in the model, which led to a strong overestimation of stress-induced mortality.

In *Paper II*, the process-based rockfall model Rockyfor was shown to allow accurate predictions of the spatial distribution of rockfall trajectories on three forested slopes with different slope and stand characteristics, based on input data with a resolution of at least 5 m x 5 m. However, Rockyfor underestimated mean impact heights observed on trees at those two sites where high- and medium-resolution input data were available, and it overestimated them at the site where input data with the lowest resolution data were used. Still, the protective effect of the stands could be assessed. Thus, Rockyfor was found to be a valuable tool for investigating the protective effect of different stands and, therefore, it can be used for a *spatially explicit, 3D CoST* in its present form.

Given the considerable differences in the performance of the two evaluated models, the suitability of ForClim and Rockyfor for a *CoST* is rather unlike. To be suitable for a combination with Rockyfor in a 3D *CoST*, not only the major shortcomings of ForClim

detected in *Paper I* need to be improved, but the model should additionally become more spatially explicit, i.e. horizontal relationships between individual patches should be taken into account. Since it was not possible to achieve all these modifications in the context of this thesis, a combination of ForClim and Rockyfor did not seem to be appropriate. Thus, instead of trying to combine these two models, I first adapted the ForClim model by enhancing the major shortcomings detected in *Paper I*. This was mandatory for the use of ForClim in a *CoST*. I then focused on developing a simplistic *CoST*, based on the enhanced ForClim version and the simplistic empirical rockfall model RockFor^{NET}.

In section II, the performance of ForClim was consequently first enhanced. This was done by improving the reproduction of light competition and by adapting the establishment submodel. The former was refined by implementing a simplistic dynamic crown structure, which allows accounting for self-pruning of tree crowns in real stands. The establishment submodel was adapted to the needs of a *CoST* by disaggregating the regeneration process into two stages, seedling establishment and sapling growth. In the latter, sapling growth is modelled explicitly by taking into account two important constraints to sapling growth in mountain forests, namely canopy shading and ungulate browsing. A comparison of the efficacy of the old (V2.9.3) and new (V2.9.4) ForClim model version revealed that the latter allowed a more accurate reproduction of mountain forest dynamics over a multi-decadal period. This, in turn, makes it more suitable for the use in a simplistic *CoST*.

To give an example of the use of the new model version, ForClim V2.9.4 was then applied to investigate the ability of a particular mountain forest to provide effective protection against rockfall during a period of 60 years. This forest, called *Stotzigwald*, is very steep with a slope gradient of approximately 45°. It protects one of the most heavily used traffic routes in in the Swiss Alps (46°45’ N, 08°39’ E) against rockfall. The current stand, which is dominated by *Picea abies* (83%) and *Abies alba* (13%), includes approx. 561 trees ha^{-1} > 4 cm DBH, and is thought to provide an almost optimal protective effect against rockfall in its present state. The current level of tree regeneration, however, is alarmingly low with approx. 2230 saplings ha^{-1}, most of them being smaller than 0.4 m in height. Therefore, the development of the long-term protective effect of the *Stotzigwald* is rather dubious. Two scenarios were simulated, i.e. (i) canopy shading but no browsing impact and (ii) canopy shading and high browsing impact. Under both scenarios, the initial sparse level of tree regeneration affected the long-term protective effect of the forest, which considerably declined during the first 40 years. Still, in the complete absence of browsing, the density of small trees was able to recover after 60 years. In the scenario including browsing, however, the density of small trees remained at very low levels.

In section III, a prototype of a simplistic *CoST* that allows investigating the effects of forest dynamics on the long-term protective effect against rockfall was finally developed. This was done based on ForClim V2.9.4 developed in section II and the empirical rockfall model RockFor^{NET}. The *CoST* was then applied to a case study to give an example of its use. Based on empirical data, the development of three mountain forests was simulated over a period of 60 years assuming several scenarios (e.g., different levels of tree regeneration). The protective effect of the simulated stands was then assessed by projecting those on the Stotzigwald site, from where terrain and rock characteristics were available.

The long-term protective effect of the three stands against rockfall was generally high for small rocks, but only limited for larger rocks (diameter d > 0.8 m), indicating the limit of the protective potential of stands on the relatively short and steep Stotzigwald slope (length: 325

m, gradient: 45°). Initial stand conditions, in particular a high initial stand density, as well as a relatively low mortality rate were found to be key factors for a high protective effect after 60 years. These findings, which are in agreement with current expert knowledge, suggest that silvicultural measures in protection forests such as the Stotzigwald should be moderate, since a decrease of tree density (e.g., due to thinning) is likely to reduce the protective effect of a stand over several years.

Additionally, a high density of tree regeneration in the initial stand was found to increase the long-term protective effect against small rocks (d = 0.2 m). This hints at the importance of tree regeneration for a high long-term protective effect, even if the initial level of tree regeneration was not found to significantly increase the protective effect against larger rocks (d > 0.2 m) after a period of 60 years. One reason for this finding can probably be found in the relatively short simulation period of 60 years, during which initial tree regeneration under shelter could not turn into very large trees. Therefore, their potential of dissipating energy is still limited, and consequently, the current regeneration cannot contribute to a significant reduction of the residual hazard after 60 years. With the present version of ForClim, however, accurate simulations over a longer period were not possible. Another reason why a high density of tree regeneration did not increase the long-term protective effect against larger rocks might be the rather vague representation of stand structure within RockFor^{NET}, which currently only uses one measure of location to describe the DBH distribution of a stand. Thus, the latter and also the *CoST* could benefit from a more detailed representation of the DBH distribution.

Whereas the approach of the *CoST* presented in this thesis is thought to be promising, the *CoST* is currently not ready to be used in practice due to the shortcomings of the underlying simulation models. Still, it allows a first estimate of the influence of different stand parameters on the long-term protective effect against rockfall. Moreover, it can be helpful for objectively assessing whether or not the protective effect of a given stand is likely to decrease in the coming decades. Once the underlying models in the *CoST* are improved, the latter could probably be used for several practical applications. When kept simplistic while increasing its accuracy, it could for instance become a decision-support tool for an effective long-term forest management by e.g. predicting accurate site-specific target values for tree regeneration, which are necessary for a high long-term protective effect.

Still, even if this thesis shows that simulation models such as Rockyfor, RockFor^{NET} and the *CoST* are useful tools for investigating the protection forest system and for optimizing its management, such models should always only be used complementary to expert knowledge.

André Wehrli: ***Mountain forest dynamics and their impacts on the long-term protective effect against rockfall – A modelling approach.*** Dissertation Nr. 16358, ETH Zürich.

Zusammenfassung

Viele Gebirgswälder schützen wirksam gegen Naturgefahren. Ihre Schutzwirkung wird dabei durch die vorhandenen Bestandesstrukturen bestimmt. Letztere werden bedingt durch die Walddynamik laufend verändert. Somit ist auch die Schutzwirkung eines Bestandes zeitlich nicht konstant. Die Dynamik eines Waldes und somit auch seine langfristige Schutzwirkung kann aber durch waldbauliche Massnahmen beeinflusst und in eine gewünschte Richtung geführt werden. Allerdings ist der Einfluss solcher Massnahmen auf die langfristige Schutzwirkung schwierig abzuschätzen, da viele Prozesse in Gebirgswäldern über sehr lange Zeiträume ablaufen.

Um die Schutzwaldpflege effizienter und effektiver zu gestalten, ist es daher äusserst wichtig, die Dynamik von Gebirgswäldern besser zu verstehen und deren Einfluss auf die langfristige Schutzwirkung gegen Naturgefahren abzuklären. Simulationsmodelle bieten dabei ein äusserst wertvolles Hilfsmittel, um das „Schutzwaldsystem“ unter Berücksichtigung der langen Zeiträume zu untersuchen. Ein geeignetes Modell muss dabei einerseits die Entwicklung von strukturellen Bestandeseigenschaften möglichst genau vorhersagen, andererseits sollte es eine präzise Abschätzung der Schutzwirkung der simulierten Bestandesstrukturen ermöglichen. Ein derartiges Instrument existiert bislang noch nicht, könnte aber basierend auf bestehenden Modellen entwickelt werden. Die Entwicklung eines solchen *„kombinierten Simulationsmodelles“* (*CoST*) war dann auch das Ziel dieser Arbeit.

Um dieses Ziel zu erreichen, wurden im ersten Teil der Arbeit zwei mögliche Kandidaten für ein solches *CoST* evaluiert. In *Artikel I* konnte gezeigt werden, dass das Waldsukzessionsmodell ForClim strukturelle Bestandeseigenschaften von Gebirgswäldern über mehrere Jahrzehnte präzise vorhersagen kann, wenn geringe Anpassungen am Verjüngungs- resp. Mortalitätssubmodel vorgenommen werden. Grundsätzlich wäre ForClim somit geeignet für ein *einfaches CoST*, vorausgesetzt, dass zwei bedeutende Mängel zuvor behoben werden, nämlich (1) die allzu vereinfachte Darstellung des Verjüngungsprozesses im Modell, welche zu einer unrealistisch hohen Anzahl von jungen Bäumen führte, sowie (2) die Wiedergabe der Verhältnisse der Lichtkonkurrenz im Modell, welche zur starken Überschätzung der wachstumsabhängigen Mortalitätsrate führte.

In *Artikel II* wurde das prozess-basierte Steinschlagmodell Rockyfor mit Daten aus drei verschiedenen Bergwäldern evaluiert. Basierend auf Eingabedaten mit einer Auflösung von mindestens 5 m x 5 m, konnte Rockyfor die räumliche Verteilung von Steinschlagtrajektorien jeweils genau vorhersagen. Hingegen unterschätzte das Modell die mittleren Einschlaghöhen von Steinen an Bäumen für die beiden Orte, wo Eingabedaten mit hoher und mittlerer Auflösung vorhanden waren, und überschätzte die mittleren Einschlaghöhen für den Ort mit der geringsten Auflösung der Eingabedaten. Trotzdem konnte die Schutzwirkung der verschiedenen Bestände abgeschätzt werden. Rockyfor erwies sich somit als geeignet zur Abschätzung der Schutzwirkung von verschiedenen Beständen und könnte daher in seiner jetzigen Form für ein *räumlich explizites, 3D CoST* benutzt werden.

Die Evaluation von ForClim und Rockyfor brachte eine unterschiedliche Eignung der beiden Modell für die Verwendung in einem *CoST* zu Tage: Rockyfor kann in seiner aktuellen Version für ein räumlich explizites *CoST* verwendet werden. ForClim benötigt indessen einige gewichtige Anpassungen, bevor es sich überhaupt für ein einfaches *CoST* eignet. Die Kopplung der beiden Modelle war dadurch im zeitlich beschränkten Rahmen dieser Arbeit nicht realisierbar. Vielmehr mussten vorderhand die nötigen Anpassungen von ForClim realisiert werden, welche für eine Verwendung des Modells in einem einfachen *CoST* unabdingbar waren. Schliesslich sollte basierend auf der angepassten Modellversion von ForClim und dem empirischen Steinschlagmodell RockFor^{NET} ein einfaches *CoST* entwickelt werden.

Im zweiten Teil dieser Arbeit wurde ForClim so angepasst, dass es in einem einfachen *CoST* verwendet werden konnte. Dazu wurde einerseits die Wiedergabe der

Lichtverhältnisse (Lichtkonkurrenz) verbessert und andererseits das Verjüngungssubmodel angepasst. Die Wiedergabe der Lichtverhältnisse wurde durch den Einbau einer einfachen, dynamischen Kronenstruktur verbessert, welche die Nachbildung der Kronenreduktion in realen Beständen erlaubt. Das Verjüngungssubmodel in ForClim wurde gegenüber der Standardversion verfeinert, indem der Verjüngungsprozess in die Teilprozesse Keimlingsetablierung und Verjüngungswachstum aufgeteilt wurde. Das Verjüngungswachstum wird dabei explizit modelliert unter Berücksichtigung zweier limitierender Faktoren, nämlich Beschattung durch die Oberschicht und Verbiss durch Huftiere. Nach dem Einbau wurde die Leistungsfähigkeit der alten (V2.9.3) und der angepassten (V2.9.4) Modellversion bezüglich der Vorhersage von strukturellen Bestandeseigenschaften von Gebirgswäldern verglichen. Dabei zeigte sich, dass ForClim V2.9.4 eine realistischere Vorhersage über mehrere Jahrzehnte ermöglichte, und somit besser geeignet ist für ein einfaches *CoST*.

Um die Zweckdienlichkeit der neuen Modellversion zu demonstrieren, wurde ForClim V2.9.4 schliesslich auf eine Fallstudie angewendet, wo die Schutzwirkung des Bestandes *Stotzigwald* (Schweizer Alpen, 46°45' N, 08°39' E) gegen Steinschlag über 60 Jahre abgeschätzt wurde. Der Stotzigwald ist sehr steil mit einer Hangneigung von ungefähr 45° und schützt eine der wichtigsten Verkehrsrouten Europas gegen Steinschlag (Gotthardautobahn). Er erstreckt sich von 650 bis etwa 1000 m ü.M., aber seine Schutzfunktion konzentriert sich auf eine ca. 7,5 ha grosse Steinschlagzone im unteren Teil des Waldes. Wie aus der Vielzahl von rezenten Schäden an Bäumen durch Steinschlag ersichtlich ist (79% der Bäume mit ≥ 1 Schaden), ist die Steinschlagaktivität in dieser Zone relativ hoch. Der aktuelle Bestand in dieser Zone wird von *Picea abies* (83%) und *Abies alba* (13%) dominiert und zählt ungefähr 561 Bäume ha^{-1} > 4 cm BHD. Damit scheint er momentan eine beinahe optimale Schutzwirkung zu gewährleisten. Die Verjüngungsdichte ist hingegen relativ tief mit ungefähr 2230 Bäumchen ha^{-1}, wobei die meisten kleiner als 0,4 m sind. Aus diesem Grund ist die Entwicklung der langfristigen Schutzwirkung im Stotzigwald unsicher. In der Fallstudie wurden zwei Szenarien simuliert: (i) Beschattung durch die Oberschicht und kein Verbiss durch Huftiere und (ii) Beschattung durch die Oberschicht und hoher Verbiss durch Huftiere. In beiden Szenarien führte die ursprünglich tiefe Verjüngungsdichte zu einer beträchtlichen Reduktion der Schutzwirkung in den ersten 40 Jahren. Während die Stangenholzdichte im Szenario ohne Verbiss durch Huftiere nach 60 Jahren langsam anstieg, blieb sie im Szenario mit Verbiss auf einem bescheidenen Niveau.

Im dritten Teil dieser Arbeit wurde schliesslich ein Prototyp eines einfachen *CoST* entwickelt, welcher es ermöglicht, die Einflüsse der Walddynamik auf die langfristige Schutzwirkung gegen Steinschlag zu untersuchen. Dazu wurde das im zweiten Teil der Arbeit angepasste Waldsukzessionsmodell ForClim V2.9.4 mit dem empirischen Steinschlagmodell $RockFor^{NET}$ gekoppelt. Um den Nutzen dieses neuen Modells zu demonstrieren, wurde das *CoST* schliesslich auf eine Fallstudie angewendet. Basierend auf empirischen Daten wurde die Entwicklung von drei verschiedenen Gebirgswäldern unter verschiedenen Szenarien über 60 Jahren simuliert. Die Schutzwirkung der simulierten Bestände wurde dann mit $RockFor^{NET}$ abgeschätzt, indem die Bestände virtuell auf den Standort Stotzigwald projiziert wurden.

Die langfristige Schutzwirkung der drei Bestände war generell sehr hoch für kleine Steine (Durchmesser d = 0,2 m), aber nur limitiert für grössere Steine (d = 0,8 m). Dies deutet auf das limitierte Schutzpotential von jeglichen Beständen auf dem relativ kurzen und steilen Hang im Stotzigwald hin (Länge: 325 m, Neigung: 45°). Der Zustand des Ausgangsbestandes, insbesondere eine hohe Dichte an Bäume > 8 cm BHD, wie auch eine relativ tiefe Mortalitätsrate wurden als wichtige Faktoren für eine hohe Schutzwirkung nach 60 Jahren identifiziert. Dieses Ergebnis, welches das aktuelle Expertenwissen bestätigt, legt Nahe, dass waldbauliche Eingriffe in Steinschlag-Schutzwäldern wie dem Stotzigwald in gemässigtem Ausmass stattfinden sollten, da sich eine Reduktion der Stammzahl (z.B. durch Durchforstung) über mehrere Jahre auf die Schutzwirkung eines Bestandes auswirken kann.

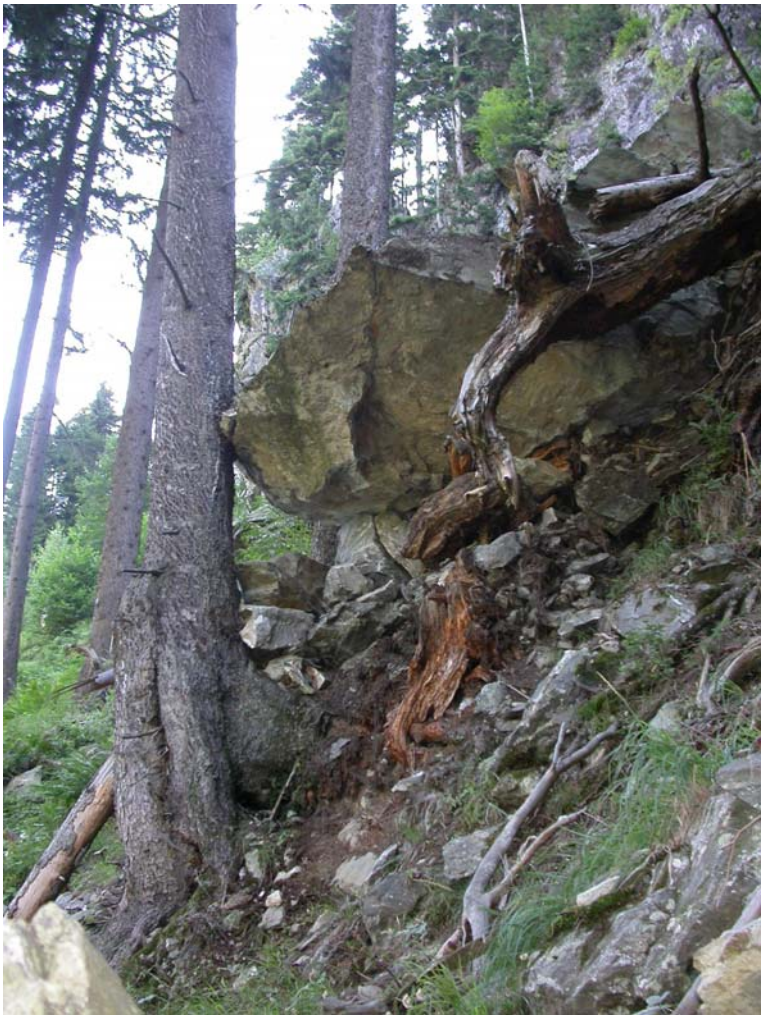
Zusätzlich zu den obengenannten Faktoren erhöhte eine hohe Verjüngungsdichte im Ausgangszustand die langfristige Schutzwirkung gegen kleine Steine (d = 0,2 m). Dies ist ein

Hinweis auf die generelle Bedeutung der Verjüngungsdichte für eine gute langfristige Schutzwirkung eines Bestandes, auch wenn nach einer Periode von 60 Jahren kein signifikanter Einfluss der Verjüngungsdichte auf die Schutzwirkung gegen grössere Steine (d > 0,2 m) festgestellt werden konnte. Ein Grund für diesen Befund dürfte in der relativ kurzen Simulationsperiode von 60 Jahren liegen, in welcher sich die ursprüngliche Verjüngung unter Schirm nicht zu stämmigen Bäumen entwickeln kann. Somit ist das Schutzpotenzial dieser Bäume gegen grössere Steine noch beschränkt, und sie können nach 60 Jahren nur begrenzt zur Schutzwirkung gegen grosse Steine beitragen. Die verwendete Modellversion von ForClim lässt präzise Vorhersagen von strukturellen Bestandeseigenschaften über längere Zeiträume jedoch nicht zu, weshalb der Zeitrahmen in dieser Studie vorderhand auf 60 Jahre beschränkt blieb. Ein weiterer Grund für den Befund, dass sich hohe Verjüngungsdichten nicht positiv auf die Schutzwirkung über 60 Jahre auswirkten, dürfte in der zu groben Abbildung der BHD-Verteilung eines Bestands in RockFor^{NET} liegen, welche bislang durch ein einziges Lagemass abgebildet wird. Eine präzisere Abbildung der Bestandesstruktur könnte daher sowohl die Genauigkeit von RockFor^{NET} wie auch des gesamten *CoST* verbessern.

Das in dieser Arbeit entwickelte kombinierte Simulationsmodell (*CoST*) bietet einen neuen, vielversprechenden Ansatz, um das Schutzwaldsystem zu untersuchen. Obwohl das *CoST* aufgrund gewisser Mängel der zugrunde liegenden Simulationsmodelle noch nicht reif ist für die Anwendung in der Praxis, erlaubt es eine erste Abschätzung des Einflusses von verschiedenen Bestandesparametern auf die langfristige Schutzwirkung gegen Steinschlag. Darüber hinaus ermöglicht es, objektiv einzuschätzen, ob die Schutzwirkung eines Bestandes über einige Jahrzehnte gesehen tendenziell abnimmt. Wenn die Genauigkeit der des *CoST* zugrunde liegenden Modelle verbessert werden kann, könnte das *CoST* vermutlich für verschiedene Anwendungen in der Praxis eingesetzt werden. Ein einfach bedienbares *CoST* könnte beispielsweise als Entscheidungshilfe oder zur Optimierung von Verjüngungssollwerten in Schutzwäldern eingesetzt werden.

Zu guter Letzt sollte man allerdings bedenken, dass Modelle stets nur als Ergänzung zu aktuellem Expertenwissen eingesetzt werden sollten. Dies gilt, auch wenn die Nützlichkeit von Simulationsmodellen wie Rockyfor, RockFor^{NET} und dem *CoST* zur Erforschung des Schutzwaldsystems und zur Optimierung der Schutzwaldpflege in verschiedenen Teilen dieser Arbeit deutlich gemacht wird.

General Introduction



General introduction

Importance of protection forests

Forests represent an important cover type in the Alpine landscape. In the Swiss Alps, they cover 23-43% of the landscape (Brassel & Brändli, 1999), depending on the region. At least 10 to 30% (Brassel & Brändli, 1999) of those forests must be considered as protection forests, the primary function of which is to protect people or assets against the impacts of natural hazards such as rockfall, snow avalanches, soil erosion, landslides, torrents, debris flows and floods (Brang et al., 2001). Many villages in the Alps depend on protection forests. They would become uninhabitable, or temporarily inaccessible, if the protection offered by forests was insufficient. In the Swiss National Forest Inventory, evidence of moving snow was recorded on 37% of the plots in mountain forests, evidence of rockfall on 31%, and evidence of erosion on 16% (Mahrer et al., 1988). In publicly owned French mountain forests, the dominant natural hazards were torrent erosion (65% of the area), snow avalanches (14%), rockfall (10.5%) and landslides (10.5%; cf. Sonnier,1991). In the Bavarian Alps of south-eastern Germany, 63% of the forests are estimated to provide protection against soil erosion and debris flows, 42% against snow avalanches and 64% against floods (Plochmann, 1985). Despite obvious differences in methodology, these figures clearly reveal the importance of protection forests in the Alps.

During the past decades, the importance of protection forests has even increased. Remote mountainous areas that were formerly avoided in winter time are now expected to be permanently accessible for tourists, settlements have been spreading into areas that were considered unsafe by our ancestors, and transports crossing the Alps (using roads, railways and power lines) have strongly increased (BUWAL, 2001). Hence, the damage potential in the mountainous regions has been raised, and therefore, large investments in protective measures have become necessary. For example, the Swiss government spent between 120 – 150 million Swiss Francs per year for protective measures in the forest including technical measures during the last decade ("Schutzaktivitäten im Waldbereich", cf. Schärer, 2004). Approximately 60% of this amount (i.e., 70 – 94 million Swiss Francs) was spent for measures to maintain or enhance the protective effect of the forests (Schärer, 2004). In spite of the increasing importance of protection forests in Switzerland, however, these annual

subsidies will be strongly reduced in the coming years (BUWAL and BHP Brugger & Partner, 2004, p. 59: 54 million Swiss Francs for protection forests per year). Therefore, the restricted financial means will have to be applied more efficiently to sustain and improve the protective effect of these forests. In order to apply the most efficient and effective measures, however, additional knowledge on the dynamics of mountain forests and their effects on their long-term protective effect is needed. In the following, the most important characteristics of the *protection forest system* are introduced and a short overview over the existing knowledge on the protective effect of forests is presented.

Main components of the protection forest system

The protection forest system consists of three main components, namely (1) the hazard potential (e.g., an instable rock cliff), (2) the damage potential (i.e. the value of the assets at risk), and (3) the forest in between (Brang et al., 2001). The latter is considered a protection forest if it is able to provide effective protection against the natural hazard at the site.

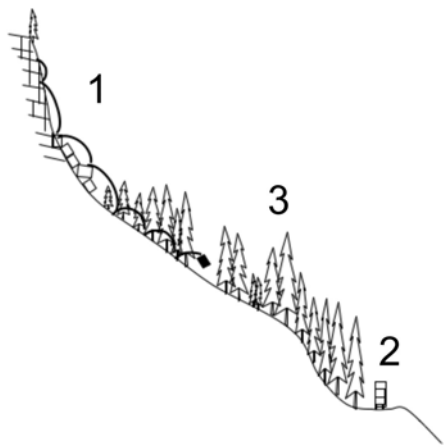


Figure 1. The protection forest system. Numbers are explained in the text.

The protective effect a forest can provide strongly depends on the hazard type and can have different aspects, i.e., prevention or mitigation of natural hazards. For snow avalanches, for instance, the protective effect of a forest is rather preventive than mitigating. This is due to the large volumes and high energies involved with

snow avalanches (Bartelt & Stöckli, 2001), which make an effective mitigation, i.e. the deceleration of snow avalanches, almost impossible. For rockfall events, however, the masses involved are generally small (i.e. < 5 m^3; Berger et al. 2002), allowing mountain forests to be effective in terms of hazard mitigation (Lafortune et al., 1997, Hétu & Gray, 2000, Dorren et al., 2005). The focus of this thesis is restricted to rockfall, i.e. only the protective effect of mountain forests against rockfall is addressed in detail in the following.

Protective effect of mountain forests against rockfall

The protective effect of a mountain forest against rockfall is mainly provided by the presence of trees. Thereby, dense stands with only small gaps parallel to the slope are thought to be most effective (Zinggeler et al., 1990, Gsteiger, 1993, Cattiau et al., 1995). The stems of living or dead trees can slow down or even stop falling rocks (Cattiau et al., 1995). Even dead tree trunks lying on the ground, rootplates or snags, may act as dams to down slope mass transfers (Mössmer et al., 1994, Schönenberger et al., 2005, Dorren et al., 2005). Since trees are crucial forest components in preventing and mitigating rockfall, their size, density, spatial distribution, species and condition are important factors influencing the protective effect of forests. Target values of stand parameters for effective protection have therefore been established in different countries (Chauvin et al., 1994, Wasser and Frehner, 1996, Frehner et al., 2005; cf. Tab. 1).

Table 1. Target values for the optimal stand structure against rockfall as defined by Frehner et al. (2005). DBH: diameter at breast height.

rock diameter [cm]	effective DBH [cm]	target values [trees with effective DBH ha^{-1}]
< 40	> 12	600
40-60	> 24	400
> 60	> 36	200

While forest managers can easily apply these target values, their scientific basis is currently unsatisfactory since they are mainly based on expert knowledge rather than on empirical data. Empirical data on the protective effect against rockfall are

sparse, and therefore, the protective effect of a stand on a given site can currently only be determined with substantial uncertainty. In order to assess the hazard and to design appropriate forest stands, it would therefore be desirable to improve these target values. To do so, dynamic rockfall models could be very useful, if they allow assessing the level of protection provided by different stand structures (Peng, 2000).

Nevertheless, such target values are useful for assessing the current protective effect of a forest against rockfall. For instance, Brändli and Herold (1999) determined the current protective effect of the rockfall protection forests in Switzerland using data from the second National Forest Inventory and a simple model based on a stand density index. Thereby, 20% were considered to provide a good protective effect, and 30% a sufficient protective effect, respectively. The rest, i.e. 50%, were considered to provide an insufficient protective effect. While being simple, this model gives a first impression of the current state of rockfall protection forests in Switzerland and could be used to roughly estimate the call for action.

Forest dynamics and their influence on the protective effect of a stand

As stand structures are continuously changing, the protective effect against rockfall cannot be constant. For instance, stands with high stem density, which are highly effective in preventing rockfall (Omura and Marumo, 1988, Cattiau et al., 1995), cannot be maintained in the long term. Such stands are usually susceptible to storm damage (Rottmann, 1985) and snow break (Rottmann, 1986, Oliver & Larsen, 1990) and they do not allow for sufficient tree regeneration. In protection forests, however, sufficient regeneration is crucial, since it ensures continuous forest cover in the long term, which in turn provides long-term protective effect. In mountain protection forests, a lack of renewal is particularly severe since, because of slow tree growth at high altitudes (Ott et al., 1997), it will impair the protective effect only after decades, and may therefore be noticed too late. Up to now, however, the influence of different levels of tree regeneration on the future stand structure of mountain forests, and by this, on its long-term protective effect, is not known.

Nevertheless, different approaches have been taken to determine target values for the required levels of tree regeneration in mountain forests. To do so, many authors use seedling or sapling densities (see Kupferschmid Albisetti, 2003 for an overview). Another approach was applied by Brang and Duc (2002), who used a

measure of regeneration cover, a static regeneration model and National Forest Inventory data to establish minimum levels of regeneration for Swiss mountain forests dominated by *Picea abies* (L.) Karst. Their model takes into account the maximum longevity of stands with effective protection, the time needed to pass the regeneration stage, the mortality in the regeneration stage, the increase of cover during the regeneration stage, and the proportion of microsites that are permanently free of trees. By means of this model, target values for the required coverage of tree regeneration for different site conditions were developed and included in the revised version of the Swiss management guidelines for protection forests (Frehner et al., 2005).

In addition, these target values were used to roughly estimate the current state of tree regeneration in Swiss mountain forests, i.e. to determine whether the level of tree regeneration is likely to be sufficient to provide a permanent forest cover or not. Based on calculations with their model, Brang and Duc (2002) concluded that, depending on the assumptions, there is a lack of regeneration in 30-60% of the forests that should provide protection or timber in the long term. The reasons for the sparse tree regeneration are thought to be manifold: Many stands are too dark for tree regeneration due to a lack of silvicultural treatment (Brang and Duc, 2002) since harvesting on steep slopes is difficult and often not cost-effective. During the last ten years, for instance, silvicultural treatments have only been executed on approximately 34% of the forested area in the northern Swiss Alps, and on 13% of the forested area in the southern Swiss Alps (cf. Brassel and Brändli, 1999, p. 187). Furthermore, ungulate browsers are abundant in many regions of Switzerland, leading to a high browsing impact on tree regeneration (Brändli, 1995). Finally, the stands have a considerable lack of coarse woody debris; decaying wood is on certain sites indispensable for the establishment of saplings (particularly *Picea abies*; cf. Holeksa, 1998, in Brang et al., 2004).

However, the impact of this sparse level of tree regeneration on the long-term protective effect of rockfall protection forests is difficult to quantify. Moreover, since the approach of Brang and Duc (2002) and most other approaches do not explicitly take into account several factors that are known to be important in mountain forest dynamics, e.g. browsing pressure by ungulates, the accuracy of these target values remains doubtful.

Management of protection forests

The dynamics of a mountain forest as well as its protective effect can be influenced by silvicultural measures (Schönenberger and Brang, 2004). However, as is evident from the preceding paragraphs, there is not only a certain lack of knowledge regarding the current protective effect of a stand against rockfall, but especially the impacts of natural or anthropogenically influenced (i.e. managed) forest dynamics on the long-term protective effect of a stand need further investigation. This is particularly important since the financial means for the management of protection forests is running short, while the importance of these forests increases. By improving the knowledge on the protection forest system, the influence of alternative management options could be assessed (e.g., regeneration cutting, reduction of browsing ungulates, harvest), which in turn could help to identify the most effective and most efficient silvicultural measures and to optimize the management of protection forests.

The protection forest system is, however, difficult to study since due to the long time periods involved in mountain forest dynamics, empirical data are sparse and field observations are difficult. To overcome this problem, simulation models could be used for investigating forest ecosystem dynamics (Johnson et al., 2001).

The role of models in investigating the protection forest system

Forest dynamics as well as rockfall processes have frequently been modelled, and therefore, many models exist for both processes. In order to investigate the effects of forest dynamics, and in particular the influence of different levels of tree regeneration on the long-term protective effect of mountain forests, two of these models could be joined in one combined simulation tool (*CoST*). By taking into account the temporal aspect of mountain forest dynamics, such a new approach could be most useful for investigating the complex protection forest system.

However, since simulation models are generally developed to answer specific questions and not for “object modelling” (Ch. Wissel, UFZ Leipzig, pers. comm.; Bugmann, 2005), model candidates for a *CoST* first have to be identified. Useful model candidates for a *CoST* should fulfil the following minimal requirements: The *model of forest dynamics* should accurately project the development of key stand characteristics that determine its protective effect over several decades. These key

characteristics include tree density, diameter distribution and species composition (Dorren et al., 2005). Additionally, the regeneration process should be included in sufficient detail to reflect the most important features of mountain forest regeneration, e.g. the long regeneration period at high altitudes or natural constraints such as browsing impact by ungulates.

The *rockfall model* in a *CoST* (or the model of natural hazard in general) should allow an accurate assessment of the protective effect of a stand. For rockfall, this means that the interaction of falling rocks and trees has to be reproduced with sufficient detail in the model, e.g., the dissipation of energy due to impacts against individual trees needs to be included in a realistic way.

In order to determine the most suitable model candidates for a *CoST*, a short overview of models of forest dynamics and rockfall is given in the following, and promising model candidates are identified.

Models of forest dynamics

While models of forest dynamics have been important forest management tools for a long time, many models for mixed-species forests have been developed during the past decades (Porté and Bartelink, 2002). They are designed to describe, quantify and reproduce forest ecosystem processes (Hasenauer et al., 2000).

According to the exhaustive review by Porté and Bartelink (2002) only a few types are suitable for a *CoST* (Fig. 2). For accurate predictions in complex systems like mixed (mountain) forests, tree-level models are necessary to account for competition effects (Porté and Bartelink, 2002). Thereby, two model types have been mainly applied for investigations on forest dynamics and succession: (i) distance-dependent tree models, and (ii) forest patch models (or ‘gap’ models, sensu Shugart, 1984; cf. Fig. 2). Thereby, apart from a few exceptions (cf. Porté and Bartelink, 2002, p. 166), only patch models include the effects of natural disturbances (e.g., fire occurrence) or natural constraints (e.g., wildlife impacts) in the description of forest dynamics. Nevertheless, both model types are potential candidates for the proposed *CoST*, and their main characteristics are therefore described below.

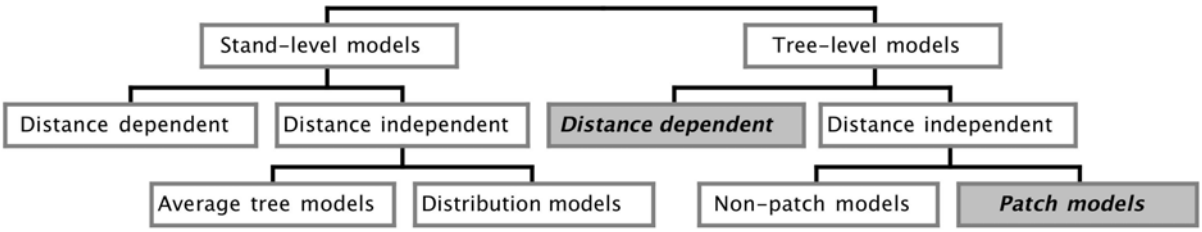


Figure 2. Classification of models of forest dynamics (modified after Porté and Bartelink, 2002). The model types that are mostly used to describe forest dynamics and succession are highlighted and written in bold italics.

Distance-dependent tree models (*DDTM*) are models that keep track of (1) individual tree characteristics (e.g., growth or mortality), *and* of (2) the exact location of these tree individuals in a certain area or a stand (Porté and Bartelink, 2002). Most models include establishment, growth, and mortality of individual trees, but particularly establishment is often neglected (e.g., Bartelink, 1998).

If included, *establishment* is modelled in terms of numbers, size and location of seedlings. In most models, each tree can produce a potential number of seeds, which will germinate in case of suitable environmental conditions. Seed dispersal thereby always requires the presence of adult trees nearby. Seedling position, i.e. the location of a seedling, results from the estimation of seed dispersal, and thus, seedling density decreases with increasing distance from parent trees (cf. Porté and Bartelink, 2002). Growth of established seedlings or saplings is not modelled explicitly in most *DDTM* (cf. Kupferschmid Albisetti, 2003).

Tree growth is usually modelled as DBH increment, and other relevant characteristics are estimated from allometric relationships (e.g., tree height is generally estimated using relationships to DBH, cf. Porté and Bartelink, 2002). Growth modelling is typically based on empirical relationships using multivariate regressions or on potential growth rates with reduction factors (Porté and Bartelink, 2002). Competition is based on distance-dependent competition indices derived from the spatial location of the trees. Thereby, competition for light is included in all models, but apart from very few exceptions (e.g., SILVA, cf. Kahn and Pretzsch, 1997), the competition for other resources, e.g., water, nutrients, is generally neglected. A few *DDTM* differ from the consensual empirical approach and use a more mechanistic reproduction of growth and competition based on the description of the processes of primary and secondary production (e.g., Williams, 1996, Chave, 1999, Bartelink, 2000; cf. Porté and Bartelink, 2002). In these models, competition is

based on a functional approach, e.g., a 3D representation of the stand, which is used to determine the available light for each tree (cf. Porté and Bartelink, 2002).

Mortality is modelled at a tree level using probabilities. *DDTM* usually include a maximum age limit per species, and mortality is either increased by using competition indexes depending on tree density or resource availability (e.g., Kahn and Pretzsch, 1997), or it is submitted to internal limitations such as insufficient growth rate (e.g., Bartelink, 1998; cf. Porté and Bartelink, 2002). Additionally, *DDTM* generally allow different thinning regimes.

DDTM are widely applied in forest science to project short- to mid-term forest growth, i.e. exact forecast of structural forest patterns over 10-20 years (cf. Hasenauer et al., 2000; for examples of *DDTM* see e.g., Ek and Monserud, 1974, Pacala et al., 1993, Courbaud et al., 1997, Pretzsch, 2001). Many *DDTM* have been developed for European conditions (e.g., Bartelink, 1998, Courbaud, 1997, Pretzsch et al., 2002), and hundreds of growth and yield plots, which have been observed for several decades, have enabled a sound calibration and validation of these models. In most of these models, however, the focus was on commercial timber species, and therefore, only few species are usually fully integrated and parameterized (e.g., 5 species included in SILVA 2.2, P. Biber, pers. comm.; cf. Pretzsch, 2001).

While being used mostly for growth and yield predictions, some models of this type have been applied to describe forest dynamics and succession (cf. Porté and Bartelink, 2002). However, natural constraints known to be important in mountain forest dynamics, e.g., wildlife impact due to browsing ungulates, are rarely considered (cf. Porté and Bartelink, 2002).

Forest patch models (*FPM*) are models that simulate establishment, growth and mortality of individual trees in a restricted area, a *patch* (Botkin et al., 1972, Shugart, 1984; cf. Bugmann, 2001, for a recent review). The exact location of a tree within a patch is thereby not known. In traditional *FPM*, patches of a stand are mutually independent, but recent *FPM* are more spatially explicit by taking into account the horizontal relationships between individual patches within a stand (Bugmann, 2001).

Establishment is generally modelled as a stochastic process, triggered by different environmental conditions. In traditional *FPM*, all species are supposed to dispose of an infinite seed source on a patch, but recent *FPM* have included alternatives to the assumption of unlimited seed availability (cf. Bugmann, 2001, p.

286 for details). In case of suitable environmental conditions, tall saplings with an initial DBH between 0.5 – 1.5 cm rather than seedlings are usually established on a patch (cf. Kupferschmid Albisetti, 2003). This means, that neither seedling nor sapling growth are modelled explicitly in most *FPM* (cf. Kupferschmid Albisetti, 2003).

Tree growth is usually calculated in terms of DBH increment using a maximum, site-independent growth potential derived from empirical data. As in *DDTM*, other relevant tree characteristics are then estimated from allometric relationships (e.g., Bugmann, 1994, p. 47). The actual DBH increment is obtained by reducing the maximum growth potential by environmental constraints, i.e., light availability, temperature, soil moisture and nutrients (cf. Bugmann, 2001). Among these constraints, light competition is commonly a major factor (e.g., Bugmann, 1996). A few models differ from the consensual approach by estimating growth mechanistically (cf. Porté and Bartelink, 2002). For instance, Jorritsma et al. (1999) simulated tree growth as biomass increment of single trees based on the amount of absorbed light and the radiation use efficiency concept, including the effects of water and nutrient limitations.

Mortality is assumed to be a consequence of several factors. In most patch models, it is simulated as a combination of an age-related and a stress-induced mortality rate (e.g., Botkin et al. 1972, Bugmann, 1994; cf. Shugart, 1984, Keane et al. 2001), giving rise to high mortality in small trees due to strong competition for light, and in old trees due to low vigour. In some models, additional mortality due to natural constraints (e.g., browsing, cf. Jorritsma et al., 1999) or due to episodic large-scale disturbances such as fire, or windthrow is considered (cf. Porté and Bartelink, 2002). In contrast to *DDTM*, *FPM* usually do not include managment submodels (Hasenauer et al., 2000).

FPM have been widely used in ecological applications for more than three decades (Shugart 1998, Hasenauer et al. 2000). They focus on the simulation of forest development and long-term succession in natural forest stands under different climatic conditions and in different climatic regions (e.g., Botkin et al. 1972, Bugmann 1994, Desanker 1996, Talkkari and Hypen 1996, Lexer and Hönniger 2001, Shao et al. 2001). Therefore, they generally include much more species than *DDTM*, e.g. 30 species parameterized in ForClim (Bugmann, 1994), or 29 species parameterized in PICUS (Lexer and Hönninger, 2001). Moreover, the effects of natural constraints such as wildlife impact (e.g., Jorritsma et al, 1999), or natural disturbances such as

fire occurrence (e.g., Kienast and Kuhn, 1989) on the dynamics of forests have often been integrated in *FPM*.

Nevertheless, *FPM* have also been relatively successful in reproducing average compositional and structural forest patterns such as tree size distributions in Africa, America, Asia and Australia (cf. Shugart 1998, Huth and Ditzer 2000). In European forests, however, these models have rarely been used to predict structural forest patterns such as diameter distributions, even if different *FPM* have been developed for European forests and even European mountain forests (e.g., Kienast, 1987, Leemans, 1989, Bugmann, 1994, Lexer and Hönninger, 2001). Therefore, their accuracy in simulating such patterns is unknown.

In spite of this important drawback, *FPM* seem to be more suitable for a *CoST* than *DDTM*, even if both model types currently cannot completely fulfil the requirements for a *CoST* due to the coarse reproduction of tree regeneration (see above). Compared to *FPM*, *DDTM* have several serious shortcomings, which makes their adjustment for a *CoST* more complex and time consuming:

(1) *DDTM* usually only include a few tree species, focusing on commercial timber species. Thus, the dynamics of mountain forests in terms of species succession can only be reproduced in an insufficient manner. Additional parameterization of species known to be important in mountain forests, e.g. *Sorbus aucuparia (*L.) would therefore be necessary. This is, however, rather difficult due to a lack of extensive data sets for such species with sufficient quality for a parameterization (P. Biber, pers. comm.).

(2) *DDTM* merely project short- to mid-term forest growth (cf. above and Hasenauer et al., 2000) on productive sites, which are rather in lowland areas. Therefore, their suitability for an accurate projection (i) over longer periods (e.g., 50 years), and (ii) in mountain forests is unknown. Thus, their accuracy for these special conditions is not necessarily higher than the accuracy of *FPM*.

(3) *DDTM* rarely consider natural constraints known to be important in mountain forest dynamics, e.g., wildlife impact. Moreover, some models even neglect different processes of forest dynamics (e.g. establishment, see above). It is to say, however, that the reproduction of different processes is currently also insufficient in many *FPM* (see above and cf. Keane et al., 2001, Price et al., 2001). Still, it is probably easier to adapt *FPM* since the source code of these

models is normally available. This is not always the case for *DDTM* since many of them are commercial products (e.g., SILVA, cf. Pretzsch, 2001).

(4) *DDTM* need the exact coordinates of each tree for calculating the competition index. This makes the collection of data rather laborious for large study sites. The required coordinates can, however, also be virtually generated by a simple modelling tool (e.g., Strugen in SILVA 2.2, cf. Pretzsch, 2001, p. 212 ff.).

Rockfall simulation models

Many rockfall simulation models have been developed during the last decades. In a recent review, Dorren (2003) distinguishes three main types of rockfall models: (1) empirical models, (2) process-based models, and (3) GIS-based models. Out of these three model types, only *empirical* and *process-based rockfall models* are suitable for a *CoST* since they allow including the interaction of falling rocks and trees. In contrast, most GIS-based models are used for analyses on a regional scale rather than on a local scale, which makes them unsuitable for a *CoST*. Thus, only empirical and process-based models are described in the following.

Empirical rockfall models *(ERM,* or statistical models sensu Keylock and Domaas, 1999) are models that calculate the energy balance of falling rocks on a slope based on empirical data. Thus, they are generally based on empirical relationships between topographical factors and the length of the run-out zone of one or more rockfall events (Dorren, 2003). Therefore, *ERM* are often successfully used to calculate a first approximation of the length of rockfall run-out zones (e.g., Tianchi, 1983, Toppe, 1987, Evans and Hungr, 1993). Recent models, however, not only calculate rockfall run-out zones, but they can also be used to assess the protective effect of a forest stand on a slope. Based on data from real-size experiments, such models explicitly consider the interaction between rocks and trees, which was neglected in former models of this type. For instance, the RockFor^{NET} model developed by Berger and Dorren (in review) includes stand structure as a virtual sequence of tree curtains consisting of a line of trees perpendicular to the direction of the slope. To assess the protective effect of a stand, RockFor^{NET} calculates the energy balance of a falling rock on a forested slope, i.e. it calculates the energy a falling rock can develop on a given slope, and sets it off against the energy that can

be dissipated by the stand on this slope. The model needs a few input parameters on *forest stand* (species composition, stand density, and a representative DBH, i.e. a measure of location that is representative for the DBH distribution), *terrain* (cliff height, slope length between the foot of the cliff and the top of the forested slope, slope length of the forested slope, mean slope gradient) and *rock characteristics* (mean rock diameter, rock density). These data are used to calculate (i) the number of probable impacts in the existing stand, and (ii) the energy loss per impact. By doing so, the residual rockfall hazard, which is defined as the percentage of rocks passing a forested slope, is determined. A similar tool has recently been developed by Brauner et al. (2005).

ERM are usually thoroughly validated for the sites where they have been developed. The RockFor^{NET} model, for instance, has been validated for several sites, and it has been shown to allow realistic assessments of the protective effect of different stand structures (Berger and Dorren, in review). Therefore, RockFor^{NET} as well as other *ERM* are frequently applied in practice to assess rockfall run-out zones or the protective effect of stands (L.K.A. Dorren, Cemagref Grenoble, pers. comm.).

Process-based rockfall models (PBRM) are models that simulate the modes of motion of falling rocks over slope surfaces in two or three dimensions based on a mechanistic calculation of the velocity of a falling rock on a slope. Therefore, these models are usually more suitable for applications in areas other than where they were developed (Dorren, 2003). Many *PBRM* have been developed in the two last decades (e.g., Descoeudres and Zimmermann, 1987, Bozzolo et al. 1988, Hungr and Evans, 1988, Pfeiffer and Bowen, 1989). Thereby, two-dimensional (2D) *PBRM* have been shown to accurately predict run-out zones, velocities, bounce heights and energies of falling rocks (e.g., Azzoni et al., 1995). Based on these models, different process-based three-dimensional (3D) models have been developed and successfully applied (e.g., Descoeudres & Zimmermann, 1987, Krummenacher, 1995, Guzzetti et al., 2002, Agliardi & Crosta, 2003, Dorren and Seijmonsbergen, 2003). However, most of the former *PBRM* neglected the effects of trees and stands on rockfall or only included them in a simplified way, i.e. by integrating stands in coefficients of restitution (e.g., Gsteiger 1989, 1993, Krummenacher 1995, Cattiau et al. 1995, cf. ROCKFOR 1999). These coefficients determine the efficiency of collisions of falling rocks (Chau et al., 2002, Dorren, 2002).

The recently developed model Rockyfor (Dorren, 2002, Dorren et al., 2004a) is one of the rare exceptions that explicitly simulates the impact of falling rocks on trees. Rockyfor simulates trajectories of falling, bouncing and rolling rocks and boulders within single raster cells, taking into account impacts on trees. It was shown to realistically predict different rockfall patterns e.g. mean velocity of rocks, run-out zone, impacts against trees, jump heights on two forested slopes in the Alps (cf. Dorren et al. 2004a, b). However, the model usually operates with a support of highly resolved input data (2.5 m x 2.5 m), which at the moment are still difficult to obtain for many areas in the Alps, and the minimum resolution of the input data that is required to obtain realistic modelling results for the assessment of the protective effect of mountain forests against rockfall hazards is not known yet (Dorren & Heuvelink 2004). Thus, the suitability of Rockyfor for practical applications and in particular for the proposed *CoST*, which as a management tool should rather operate on low resolution input data, has first to be assessed.

As is evident from this short overview, both rockfall model types are successful candidates for a *CoST* since they generally seem to fulfil the minimal requirements for a *CoST*. Thus, the choice of the rockfall model depends mainly on the spatial resolution needed in the *CoST*. For management purposes, a spatially explicit 3D *CoST* could be advantageous. Therefore, a *PBRM* like Rockyfor would be the first choice, since it allows predicting different patterns of rockfall processes, such as the spatial envelope of the rockfall trajectories. For a purely scientific *CoST*, however, an *ERM* like RockFor^{NET} that is less spatially explicit is probably sufficient.

Aim of the study and research questions

The present study aims at developing a combined simulation tool (*CoST)* that allows investigating the effects of mountain forest dynamics on the long-term protective effect against rockfall, with a particular focus on the influence of different levels of tree regeneration on the long-term protective effect. As stated above in the overview on the models of forest dynamics and rockfall, promising candidates for a *CoST* do exist. To model forest dynamics, a forest patch model seems to be an optimal candidate, since adapting it for a *CoST* would probably be less time

consuming. As for the rockfall simulation model, both options, an empirical or a process-based rockfall model, are possible for a *CoST* (see above).

Before the promising model candidates are combined in a *CoST* however, their suitability for a *CoST* first has to be assessed and potential model shortcomings have to be improved. The main objectives of this thesis are therefore:

1. To evaluate the performance of simulation models, which are promising candidates for a *CoST*, and to identify shortcomings of these models with regard to their suitability for a *CoST*.
2. To improve the shortcomings of the model candidates if necessary, and to adapt them for the use in a *CoST.*
3. To develop a prototype of a *CoST* that allows investigating the effects of forest dynamics on the long-term protective effect against rockfall.

The objectives are dealt with separately in three sections, followed by a synthesis over the whole thesis. An overview of the different chapters contained within each section is given in the following.

Section I: Evaluating models that are candidates for a combined simulation tool (*CoST*)

In section I, the suitability of the forest patch model ForClim (Bugmann, 1994; *Paper I*) and of the process-based rockfall model Rockyfor (Dorren 2002; *Paper II*) for a *CoST* are assessed based on empirical data from different mountain forests. Section I therefore deals with the following main questions:

Paper I

(i) Does the forest patch model ForClim allow accurate predictions of structural forest patterns in different mountain forests over several decades?

(ii) What model features have to be improved to enhance model accuracy?

Paper II

(iii) Does the process-based rockfall model Rockyfor allow accurate predictions of rockfall processes on forested slopes with different stand and slope characteristics?

(iv) What resolution of the input data is needed to obtain accurate simulation results?

Section I ends with a section summary and a short conclusion, which at the same time is an introduction to the following sections.

Section II: Adapting the forest patch model ForClim for the use in a *CoST*

Based on the findings of section I, the most relevant model shortcomings of the forest patch model ForClim are improved, and the model is adapted for the use in a *CoST*. Thus, section II deals with the following main questions:

Paper III

(v) How can the performance of the forest patch model be improved in order to more accurately reproduce mountain forest dynamics?

(vi) What is the influence of sparse tree regeneration on the protective effect of a mountain forest during a period of 60 years?

Section II ends with a section summary.

Section III: Developing a prototype of a *CoST* to investigate the effects of forest dynamics on the long-term protective effect against rockfall

In the last section, a prototype of a *CoST* is developed and applied to several scenarios. This section therefore deals with the following main questions:

Paper IV

(vii) What is the protective effect of different stands after a period of 60 years?

(viii) Which factors are important for a high protective effect of a stand after a period of 60 years?

Section III also ends with a section summary. It is followed by a synthesis of the whole thesis.

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Section I

Evaluating models that are candidates for a combined simulation tool (*CoST*)

Paper I

Using a forest patch model to predict the dynamics of stand structure in Swiss mountain forests

Paper II

Assessing the protective effect of mountain forests against rockfall using a 3D simulation model

Section summary

Paper I

Using a forest patch model to predict the dynamics of stand structure in Swiss mountain forests

Based on:

Wehrli, A., A. Zingg, H. Bugmann, and A. Huth. 2005. Using a forest patch model to predict the dynamics of stand structure in Swiss mountain forests. Forest Ecology and Management (205): 149-167.

Abstract – *Forest patch models have been applied to simulate forest development and long-term forest succession in many studies. The main focus of these simulations has been on species composition and biomass in natural forests, but these models could also become useful for the prediction of other structural forest patterns such as size distributions. Up to now, most of these models have been validated by approaches such as comparison of simulation results with potential natural vegetation (PNV), or national forest inventory data. While these approaches may be appropriate to validate the simulated species composition, they are not sufficient in testing the prediction of other structural patterns. Thus, little is known about the accuracy of forest patch models in simulating structural forest patterns such as size distribution of different forest types.*

For this reason, we tested the forest patch model ForClim against empirical data from three Swiss mountain forests. The objectives of this study were (i) to investigate the performance of ForClim in simulating structural forest patterns and (ii) to assess the influence of the establishment, growth and mortality submodels of ForClim on the simulation results.

Several shortcomings of the model were identified and quantified. In particular, the stress-induced mortality implemented in ForClim was found to overestimate the actual mortality rates. The excessive mortality was most likely caused by an inaccurate growth function or an overestimation of light competition. Once the stress-induced mortality was reduced, ForClim was able to reproduce structural forest patterns in an accurate manner. Based on these encouraging results, we suggest that ForClim as well as other forest patch models could become important tools for further applications in forest research.

Keywords: Forest patch model, ForClim, model test, mountain forest, diameter distributions, forest structure

Introduction

Forest patch models (or gap models cf. Shugart, 1984) have been widely used in ecological applications for more than three decades (Shugart, 1998, Hasenauer et al., 2000). Since the creation of the first forest patch model JABOWA (Botkin et al., 1972), many similar models have been developed for a broad range of forest types and other ecosystems, such as grassland and savannas (cf. Shugart, 1998). A recent review of forest patch models has been published by Bugmann (2001a).

The main focus of these models has been on the simulation of forest development and long-term succession (species composition and biomass) in natural forest stands under different climatic conditions and in different climatic regions (e.g., Botkin et al., 1972, Bugmann, 1994, Desanker, 1996, Lexer and Hönniger, 2001, Shao et al., 2001, Talkkari and Hypen, 1996). Therefore, most of the models do not include forest management submodels (Hasenauer et al., 2000). Nevertheless, patch models have not only been used to predict vegetation patterns under different conditions, but they have also been relatively successful in reproducing average compositional and structural forest patterns such as tree size distributions in Africa, America, Asia and Australia (cf. Shugart, 1998, Huth and Ditzer, 2000). Therefore, they have the potential to become an important tool for further applications in forest research in these regions.

In Europe, however, the application as well as the validation of patch models has been strongly constrained by the fact that the majority of the European forests are either still managed strongly, or they have at least been modified by humans (e.g., forest pasture, timber harvesting; cf. Badeck et al., 2001). Due to the lack of data on unmanaged forests, only very few models have been validated by comparing model results with empirical forest stand data. For example, Lindner et al. (1997) presented a test with data from a managed beech stand in Germany. Instead, different approaches have been used to test patch models. These approaches include (i) comparisons with the potential natural vegetation (PNV sensu Tüxen, 1956; e.g., Lexer, 2000), (ii) comparisons with national forest inventory data (e.g., Löffler and Lischke, 2001), and (iii) the use of pollen records to evaluate predictions of long-term

vegetation dynamics (e.g., Lischke et al., 1998). While these approaches may be appropriate to validate the simulation of tree species composition and biomass, they are not sufficient for testing the prediction of more detailed structural forest patterns. Thus, little is known about the accuracy of patch models in simulating structural forest patterns such as size distribution of different European forest types. Therefore, their application to other research questions in Europe has been limited to date.

Forest patch models simulate the establishment, diameter growth and mortality of each tree in a given area (a *patch*). Individual tree growth is thereby calculated using a maximum growth potential derived from empirical data, which is reduced by environmental constraints, i.e. light, temperature, soil moisture and nutrients (Bugmann, 2001a). Among these constraints, light competition is commonly a major factor. Therefore, the accuracy of the growth function strongly depends on the capability of the model to accurately predict stem numbers. Since stem numbers and species composition also depend on the performance of the regeneration and mortality algorithms included in a model (Lindner et al., 1997), these two processes have to be considered as well. The integration of these processes in current patch models has recently been reviewed critically by Keane et al. (2001) and Price et al. (2001). They stated that tree regeneration and tree mortality caused by natural or anthropogenic disturbances (i.e. silvicultural operations) are not satisfactorily integrated in patch models yet. However, the influence of these shortcomings on the accuracy of model predictions is not clear.

The objective of this study was (i) to test the capability of the forest patch model ForClim (Bugmann, 1994, 1996) to predict structural forest patterns in different European mountain forests, and (ii) to assess the performance of the establishment, mortality, and growth submodels included in ForClim. For this reason, we compared simulation results to empirical data from permanent plots in Swiss mountain forests. Since no time series on unmanaged forests were available, the test was performed with data from managed stands.

Methods

The patch model ForClim

ForClim is based on the FORECE model (Kienast, 1987) and was originally developed to assess the impacts of climatic changes on tree species composition and biomass of forests in the Swiss Alps (Bugmann, 1994). During its construction, special emphasis was placed on developing a model with a minimum number of ecological assumptions (i.e., maximum simplicity; Bugmann, 1996).

The applicability of ForClim was successfully extended from the Swiss Alps to other forests of central Europe (Bugmann and Cramer, 1998), to forests of eastern North America (Bugmann and Solomon, 1995), the Pacific Northwest of the United States (Bugmann and Solomon, 2000), the Rocky Mountains (Bugmann, 2001b) and to the Northeast of China (Shao et al., 2001) through different model modifications.

ForClim consists of three modular submodels, one for soil carbon and nitrogen turnover (ForClim-S), one for tree population dynamics (ForClim-P), and one for the abiotic environment (ForClim-E), respectively. For the present study, however, only the model variant ForClim-E/P was used, since precise data on soil properties were lacking. Thereby, the field capacity was assumed to be 20 cm and the available nitrogen for ForClim-E/P was assumed to be 100kg/ha (cf. Bugmann, 1996). In this way, potential water and nitrogen limitations were minimized. The use of ForClim-E/P as standard model set-up corresponds to most other investigations performed with the ForClim model, since the model variant ForClim-E/P/S did not improve model performance to a considerable degree (Bugmann, 1994, 1996).

In ForClim-P, no spatial interactions between the simulated patches are considered, i.e. the simulated patches are completely independent. *Tree establishment* in ForClim-P is determined by four limiting factors (i.e., light availability at the forest floor, browsing intensity, soil moisture and absolute winter minimum temperature). The response to these factors is species-specific and the regeneration process is modelled in a simplistic manner by applying these limiting factors as environmental filters to determine the establishment of saplings with an initial diameter at breast height (DBH) of

1.27 cm (cf. Bugmann, 1994, Bugmann, 1996, Price et al., 2001). *Tree growth* is derived from a simple carbon budget approach (Moore 1989). Four factors are used to take into account suboptimal conditions for tree growth: light availability, growing season temperature, soil moisture and soil nitrogen availability (Bugmann, 1994, 1996). To derive an overall growth reduction, these four factors are combined using a modified geometric mean (Bugmann, 1994). *Tree mortality* is simulated as a combination of an age-related and a stress-induced mortality rate (Botkin et al., 1972, Bugmann, 1994, Keane et al., 2001), giving rise to high mortality of small trees due to strong competition for light, and high mortality of old trees due to low vigour (Bugmann, 1994). Thus, trees that grow slowly due to adverse environmental conditions are subject to a higher stress-induced mortality rate.

The abiotic environment submodel (ForClim-E) is driven by the monthly mean temperatures and the monthly precipitation sums of every simulation year. The input for these variables can either be derived from measured time series data, or it can be generated by a weather generator based on long-term statistical data.

ForClim has not been calibrated against large-scale measured data, and its parameters were estimated from the ecological literature (Bugmann, 1994, Bugmann and Fischlin, 1996). It has, however, been validated thoroughly by systematically simulating the equilibrium species composition in different climate spaces (Bugmann, 1996).

The model version used in the present study is ForClim V2.9.2. It differs from the version described by Bugmann and Solomon (2000) by implementing a stress-induced mortality rate based on an approach similar to the one in FIRE-BGC (Keane et al., 1996). Furthermore, the light response of the tree establishment (kLy) for *Abies alba* and *Fagus sylvatica* has been slightly adapted for this study according to Ellenberg (1996, p. 119).

Site descriptions and weather data

The model test was performed for three forest stands at different sites in the Swiss Alps, where long-term data series from permanent research plots were available (cf. Tab 1).

The stand at Rougemont belongs to *Aceri-Fagetum rumicetosum* and – *prenanthetosum* (EK21, cf. Ellenberg and Klötzli, 1972) and to *Asplenio-Piceetum* (EK48, Ellenberg and Klötzli, 1972). It is managed as uneven-aged selection forest (plentering according to Schütz, 2001) and consists merely of *Picea abies* and *Abies alba*. The plot at Rougemont has been surveyed 9 times since 1937.

At Sigriswil, the stand belongs to *Sphagno-Piceeetum calamagrostietosum villosae* (*Asplenio-Piceetum*; EK 57, Ellenberg and Klötzli, 1972). It is an uneven-aged selection forest consisting of nearly 100% *Picea abies*. The plot has been surveyed 10 times since 1925, but only the data from 1930-1997 could be considered since no weather data were available for the earlier years at this site.

The stand at Hospental belongs to *Piceo-Adenostyletum stellarietosum* (EK60, Ellenberg and Klötzli, 1972). It is a 120-year old afforestation, which has been managed by thinning from above, and consists of *Picea abies*, *Larix decidua*, and *Pinus cembra*. The research plot has been surveyed 14 times since 1898, but only the data from 1958-1994 could be considered since no weather records were available for the period before 1958.

Figure 1 gives an overview of the diameter distribution at the sites at the start of the simulation. Since the diameter threshold at each site was at 7.5 cm, the first DBH class (0 to 10 cm) is not shown. This DBH class was, however, included for all the simulations.

Table 1. Site characteristics of the three different stands.

Site (area)	Altitude (m asl)	Stand characteristics (main tree species and total volume at beginning/end of simulation period)	Simulation period	Weather station (approximate distance to site)
Rougemont (1.5 ha)	1294	*Picea abies, Abies alba* 1937: 371 m^3 / 1995: 385 m^3	1937-1995 (58 years)	Château-d'Oex, 985m asl (6 km)
Sigriswil (1.5 ha)	1370	*Picea abies* 1930: 355 m^3 / 1997: 387 m^3	1930-1997 (67 years)	Interlaken, 574m asl (15 km)
Hospental (0.4 ha)	1475	*Picea abies, Larix decidua, Pinus cembra* 1958: 444 m^3 / 1995: 431 m^3	1958-1995 (37 years)	Andermatt, 1442m asl (4 km)

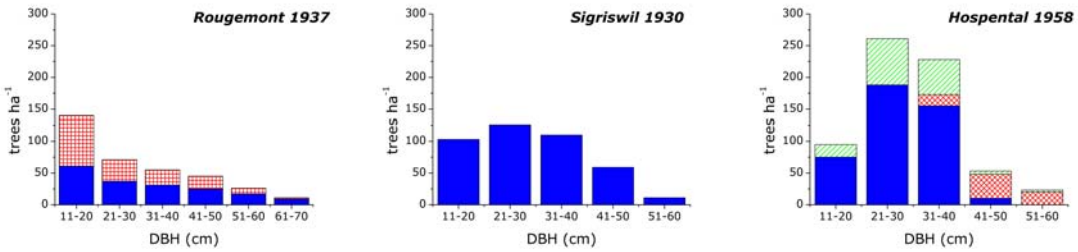

Fig. 1. Diameter distribution of the main tree species at the three sites at start of simulation. Species: *Picea abies* ( ), *Abies alba* ( ), *Larix decidua* ( ), *Pinus cembra* ( ).

Weather Data

The data for the abiotic environment submodel (ForClim-E) were derived from time series of monthly precipitation sums and monthly mean temperatures from weather stations close to the stands (Tab. 1). Thereby, the difference in altitude between the weather station and the forest stand was taken into account by modifying the temperature values by an adiabatic correction factor of -0.6° C per 100 m altitude (Gottfried et al., 1999). Precipitation values were not changed.

Weather variables for the simulations were based on a deterministic weather file, i.e. the stochastic weather generator included in ForClim-E was not used. In this way, the patch-specific variability, which is normally generated by the weather generator was excluded, and homogenous weather conditions were assumed for each stand.

Initialisation of ForClim with stand data

The empirical data did not contain the exact location of each single tree. Thus, the real spatial relations within the stands could not be included in the simulations, but the stand data had to be generated in a simplified manner. This was done by splitting the stand area into patches of equal size (833 m^2). Then, the trees were allocated randomly to the different patches, assuming that the trees were equally distributed over the whole forest stand (i.e. each patch has the same number of trees). This randomization was conducted once per site, since the import of a forest stand in different randomized replicates did not affect the simulation results (Kruskal-Wallis test, p=0.01).

Simulation set-up

Simulation experiments

In the investigations on tree density and species composition, all tree species which had been recorded at least once during the time series at one site were included in the simulations for that particular site. The establishment rates of all other species parameterized in ForClim were set to zero.

The investigations on mortality rate and diameter distribution were performed exclusively for the dominant tree species (i.e. those that count at least for 5% of total tree number). In these investigations, no tree regeneration was considered in these investigations, i.e. tree establishment was excluded from the empirical data set and the establishment rates of all species in ForClim were set to zero. In this way, only those trees present in the initial empirical forest stand were tracked and compared to the simulation results.

Management regime

ForClim does not include a management unit yet. Therefore, the management regimes at the different sites had to be addressed in a simplified way. This was done by harvesting the empirically prescribed number of tree individuals per species and size class manually after each time step, i.e. in each year of survey. This prescribed thinning regime was not included in the first simulation experiments on tree density, species composition and regeneration.

Number of simulation runs

Bugmann et al. (1996) suggest to perform simulations with n = 200 patches in order to reduce the stochastic noise in the simulation results. For the present study, however, the number of simulated patches was reduced since (i) the forest simulation did not start from bare ground, but was initialized with empirical stand data, and (ii) the weather data were not generated by the weather generator, but derived from weather records. In this way, several stochastic components of ForClim were eliminated, which in turn legitimated a reduction of the simulation runs.

Test runs with different numbers of simulated patches for the Rougemont site showed that simulations with 36 to 54 patches led to reproducible results.

Therefore, all the simulations were made with 36 to 50 patches (Rougemont: 36, Sigriswil: 36, Hospental: 50), corresponding to 3 ha and 4 ha, respectively.

Comparison of model results to empirical data

We compared simulated tree density, species composition, mortality rate, and diameter distribution (diameter at breast height, DBH) graphically. Furthermore, the relative model bias s% was calculated as the difference between simulated and empirical tree numbers per year of survey for the experiments on tree density, species composition and mortality rate as:

$$s\% = \left(n_{sim,t} - n_{emp,t}\right) / n_{emp,t}$$

where $n_{sim,t}$ and $n_{emp,\,t}$ are the numbers of trees in a given year of survey t (cf. Pretzsch and Dursky, 2001). The relative model bias was averaged over the time series, which resulted in a single value s%-mean, which gives a good characterization of the correspondence of simulation and empirical data over the whole time series. Thereby, a low value indicates a high correspondence between simulated and empirical data.

Moreover, empirical and simulated diameter distributions were tested for significant differences using a X^2 - test for goodness of fit (Sokal and Rohlf, 1995), with

$$\mathrm{X}^2 = \sum^{a} \frac{(f_i - \hat{f}_i)^2}{\hat{f}_i}$$

where f_i and $\hat{f}_i$ are the observed and expected frequency in a diameter class, respectively. Thereby, small observed significance levels indicate that the model does not fit well.

Results and Discussion

Tree density and species composition

The simulation experiments conducted with the default parameter values of ForClim showed a rather high correspondence between simulated and empirical tree density for the Rougemont site. For the two other sites, however, the correspondence was low, which is reflected in the high s%-means for total tree density on these sites (cf. Tab. 2 and Fig. 2). Moreover,

the prediction of the species composition was not satisfyingly accurate, as indicated by the high s%-means for all species (Tab. 2).

Table 2. Relative model bias s% (mean ± standard deviation over time series) for tree density and species composition.

	Rougemont	Sigriswil	Hospental
Tree density	-0.09 ± 0.04	-0.24 ± 0.12	-0.45 ± 0.08
Picea abies	***-0.15 ± 0.08***	***-0.44 ± 0.11***	***-0.36 ± 0.09***
Abies alba	***-0.42 ± 0.14***	+17.94 ± 12.43	-
Larix decidua	-	-	***+0.45 ± 0.38***
Pinus cembra	-	-	***-0.99 ± 0.01***
Fagus sylvatica	+2.04 ± 2.1	-	-
Acer pseudoplatanus	+6.8±2.64	+43.42 ± 40.1	-

Notes: Simulations were conducted with the default parameter values and without thinning regime. Signs indicate underestimation (negative) or overestimation (positive) compared to the empirical data. Values in bold-italics denote main tree species per site. **-** species does not occur on the site.

At the Rougemont and Sigriswil sites, the simulations led to a change in species composition compared to the real forest stand (Rougemont: increase of *Acer pseudoplatanus*, decrease of *Abies alba*; Sigriswil: increase of *Acer pseudoplatanus* and *Abies alba*, decrease of *Picea abies*, cf. Fig 2). At the Hospental site, all species but *Larix decidua* strongly decreased in number in the simulation (cf. Fig. 2). Especially the low accuracy in the reproduction of several main tree species (e.g., *Pinus cembra* at the site Hospental, Tab.2) was remarkable, since we had expected that the simulation of the dominating species would yield acceptable results (cf. Bugmann, 1996).

For all sites, there was a significant negative model bias for the total tree density as well as for most of the dominating tree species per site, indicating that the simulated mortality rates could be too high. This is also reflected in the rapid decrease of certain species in Fig. 2 (e.g., *Pinus cembra* at the site Hospental).

In contrast to the main tree species, the other species as well as *Larix decidua* at Hospental show a large positive model bias, which led to the mentioned change in species composition (e.g., increase of *Acer pseudoplatanus* at the site Rougemont). The reason behind this positive bias could be an insufficient reproduction of the regeneration, i.e. unrealistic establishment rates.

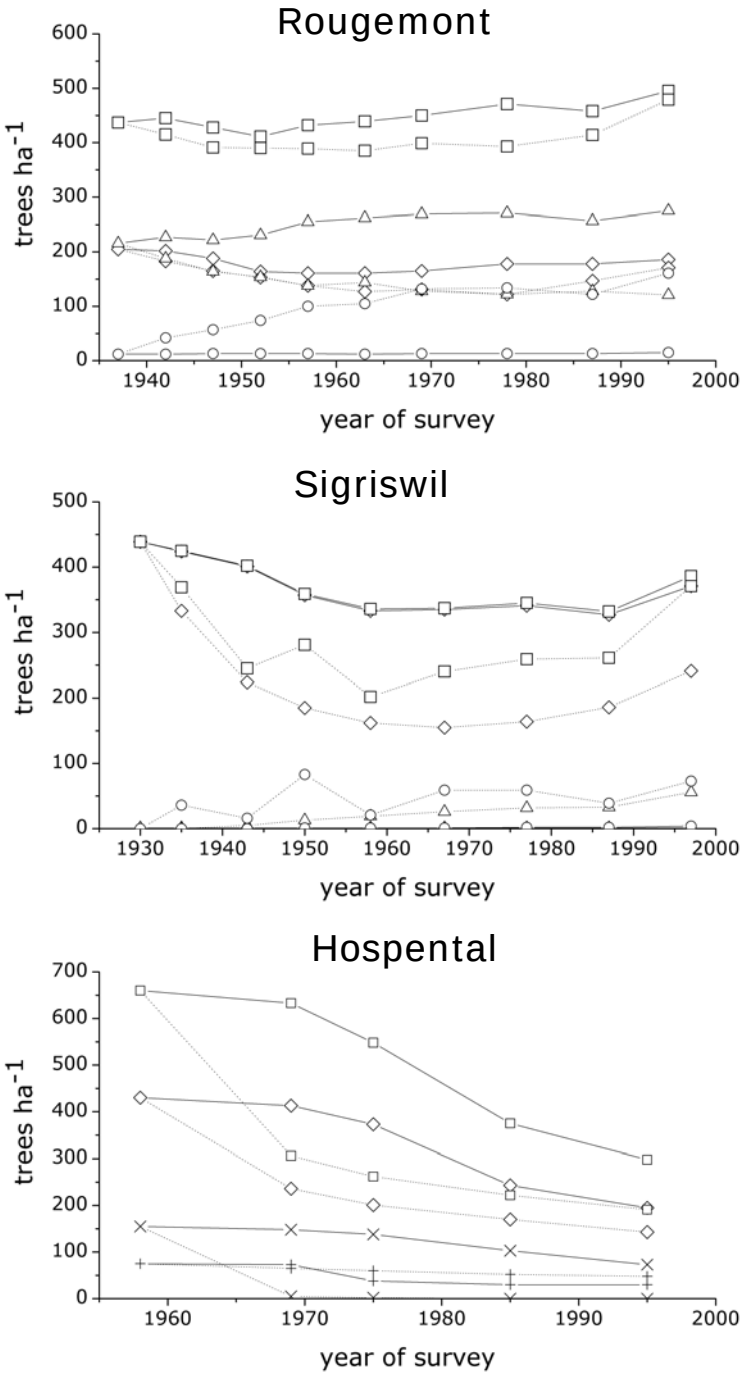

Figure 2. Empirical (solid line) and simulated (dotted line) tree density and species composition. Simulations were conducted with the default parameter values and without thinning regime. Symbols: (□) total tree density, (◊) *Picea abies*, (Δ) *Abies alba*, (O) *Acer pseudoplatanus*, (x) *Pinus cembra*, (+) *Larix decidua.*

Notes*: For Sigriswil, empirical tree density and density of Picea abies are almost equal, since the stand consists of nearly 100% Picea abies (see description of study site in text).*

The combination of the presumed unrealistic mortality and regeneration rates probably led to the peculiar pattern of the simulated tree numbers at the Sigriswil sites, where the simulated tree density drops very quickly (i.e. overestimation of mortality), but recovers during the simulation period, and almost equals the empirical tree density at the end of the simulation period (i.e. unrealistic establishment rate). Thus, the inaccuracy in the simulation experiments with default parameter values was very likely due to a combination of an inaccurate establishment and mortality submodel of ForClim. Therefore, both submodels were investigated further.

Regeneration

In a first step, we assessed the possible influence of unrealistic establishment rates on the prediction of species composition. Since no regeneration was recorded in the Hospental plot from 1958 until 1995, these investigations were performed exclusively for the Rougemont and Sigriswil sites.

The simulated increase of *Acer pseudoplatanus* at Rougemont and *Acer pseudoplatanus* and *Abies alba* at Sigriswil was addressed by a simple approach. Thereby, the default establishment probabilities included in ForClim, which are not species-specific, were replaced by species-specific values. The new establishment probabilities were derived from the empirical stand data, considering the abundance of each species in the initial stand (i.e. 1930 for Sigriswil and 1937 for Rougemont), and the frequency of its mast years. Only species with mature canopy trees were allowed to reproduce. Since the main focus of this test was on stand development over short periods (e.g., 50-100 years) rather than on long-term succession, and since the species composition was quite stable during the period of survey, these parameter adaptations seemed to be reasonable.

The simulations conducted with the new establishment probabilities yielded a more realistic prediction of species composition at both sites, which was reflected in the reduced s%-means of all tree species (Tab. 3 and Fig. 3). Yet, total tree density was reproduced with a similar accuracy as in the experiments with the default values (cf. Tab. 3).

Table 3. Relative model bias s % (mean ± standard deviation over time series) for total density and species composition.

	Rougemont	Sigriswil
Tree density	-0.07 ± 0.09	-0.27 ± 0.21
Picea abies	***-0.08 ± 0.15***	***-0.27± 0.21***
Abies alba	***-0.15 ± 0.07***	-0.25 ± 0.78
Fagus sylvatica	-0.75 ± 0.27	-
Acer pseudoplatanus	2.02 ± 1.28	-1±0

Notes: Simulations were conducted with the adapted establishment probabilities and without thinning regime. Signs indicate underestimation (negative) or overestimation (positive) compared to the empirical data. Values in bold-italics denote main tree species. - species does not occur on the site.

Thus, the use of species-specific establishment probabilities increased the accuracy of the predicted species composition. There was, however, still a considerable divergence between the empirical and simulated numbers of *Picea abies* at site Sigriswil for some years (Fig. 3). This divergence was addressed in the experiments on the mortality rate (see below).

Our findings indicate that the derivation of species-specific establishment probabilities from empirical data can improve the performance of ForClim in short-term simulations.

Still, we think that the simplistic regeneration process included in ForClim and other patch models deserves further investigations (see also Huth, 1999). Above all, it would be desirable to address some of the shortcomings discussed by Price et al. (2001) and to model explicitly some features in the establishment submodel, such as seed dispersal, migration, or browsing impact. We propose that this would improve the reliability of the establishment submodel of ForClim.

Mortality

The considerable divergence between empirical and simulated numbers of *Picea abies* at the Sigriswil site (Fig. 3) and the rapid decrease of the simulated tree density at the Hospental site (Fig. 2) indicated that the simulated mortality rate in ForClim was too high. The performance of the mortality submodel was therefore analyzed in some detail. In this analysis, only trees present in the initial empirical stand were tracked and compared to the simulation results, i.e. regeneration was set to zero.

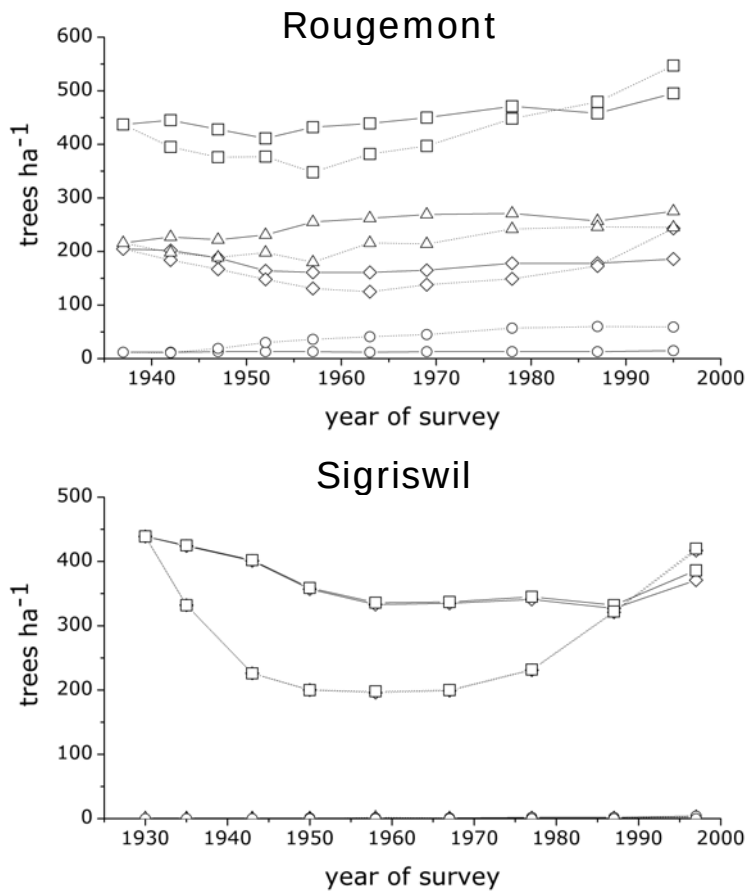

Figure 3. Empirical (solid line) and simulated (dashed line) tree density and species composition. Simulations were conducted with species-specific establishment rates and without thinning regime. Symbols: (□) total tree density, (◊) *Picea abies*, (Δ) *Abies alba*, (O) *Acer pseudoplatanus*, (x) *Pinus cembra*, (+) *Larix decidua.*
Notes*: For Sigriswil, empirical tree density and density of Picea abies are almost equal, since the stand consists of nearly 100% Picea abies (see description of study site in text).*

First, we tried to assess the influence of the age-related and the stress-induced mortality rates on the simulation results. Preliminary simulations conducted with the default mortality parameter values and *without* the prescribed thinning regime (i.e., no management considered in the simulations) led to a large overestimation of the mortality rates compared to the empirical data. As can be seen in Figure 4, the simulated tree numbers were lower than the empirical tree numbers for all species except *Larix decidua*, even if the prescribed thinning regime was not considered in the simulations. Including the thinning regime under these circumstances would have led to an even stronger divergence between simulated and empirical data.

A detailed analysis of the DBH distribution at the end of the time series (1995 for Rougemont and Hospental, 1997 for Sigriswil) revealed that the mortality of trees in the lower DBH-classes (i.e. suppressed trees having a small DBH) was much too high. Therefore, the stress-induced mortality was suspected to contribute most to the overestimation. This assumption was corroborated by a simple experiment: In the simulations conducted with the default mortality parameter values, the number of simulated trees always decreased strongly during the first four years compared to the number of trees in the empirical data. When the number of stress years that are required to accumulate in the model before the onset of the stress-induced mortality was changed from 2 years (default value) to 10 years, the same strong decrease of simulated tree numbers occurred, but with a delay of 8 years. Therefore, we concluded that the increase of the number of tolerated stress years led to the same overestimation of the mortality rate with a time lag of 8 years. Furthermore, the record of the number of stress years for all species indicated that small trees were stressed for the entire simulation period and therefore unable to grow sufficiently.

A pragmatic solution to overcome this overestimation of the mortality rate was to switch off the stress-induced mortality (i.e. setting it to zero), since any reduction of the parameter values did not improve the simulation results. By applying this modification, the simulated mortality rate was considerably lower compared to the empirical data, as could be expected in these managed stands, where thinning accounts for a large portion of the mortality (mean over time series: 33%-75% of the total mortality). At Hospental, where no thinning was made in 1969, and at Sigriswil, where only little thinning was made in 1935, simulated and empirical tree numbers were almost equal for these years (Fig. 5).

The inclusion of a simple thinning routine in combination with the modification of the mortality rate finally led to close matches between simulated and empirical data (Tab. 4 and Fig. 6). A better correspondence between simulated and empirical data is unlikely to be achieved, since the real stands are influenced by events, which are irreproducible in simulation experiments. For example, at Hospental, natural mortality was extremely elevated in 1985 and 1995 due to unknown reasons, resulting in a loss of up

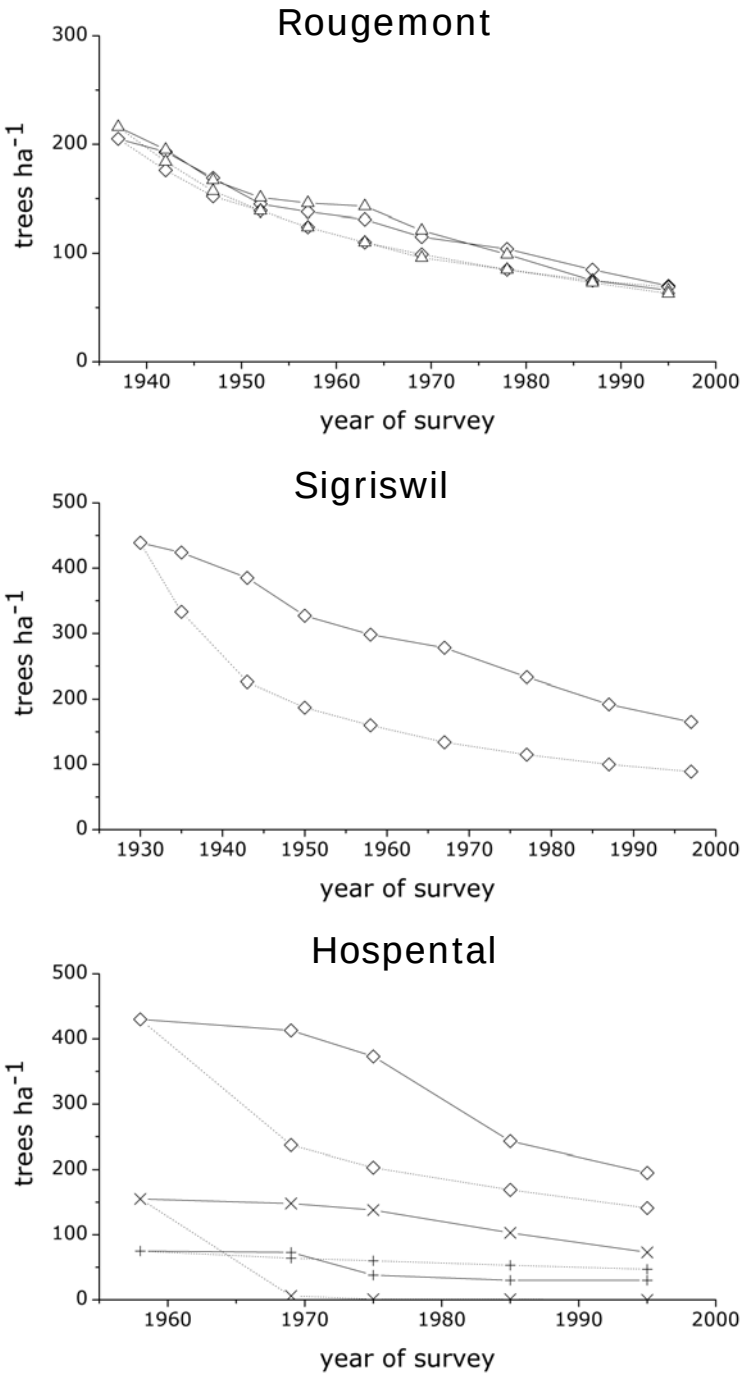

Figure 4. Empirical (solid line) and simulated (dashed line) number of surviving trees per species. Simulations were conducted with the default parameter values for mortality and without thinning regime. Symbols: (◊) *Picea abies*, (∆) *Abies alba*, (x) *Pinus cembra*, (+) *Larix decidua.*

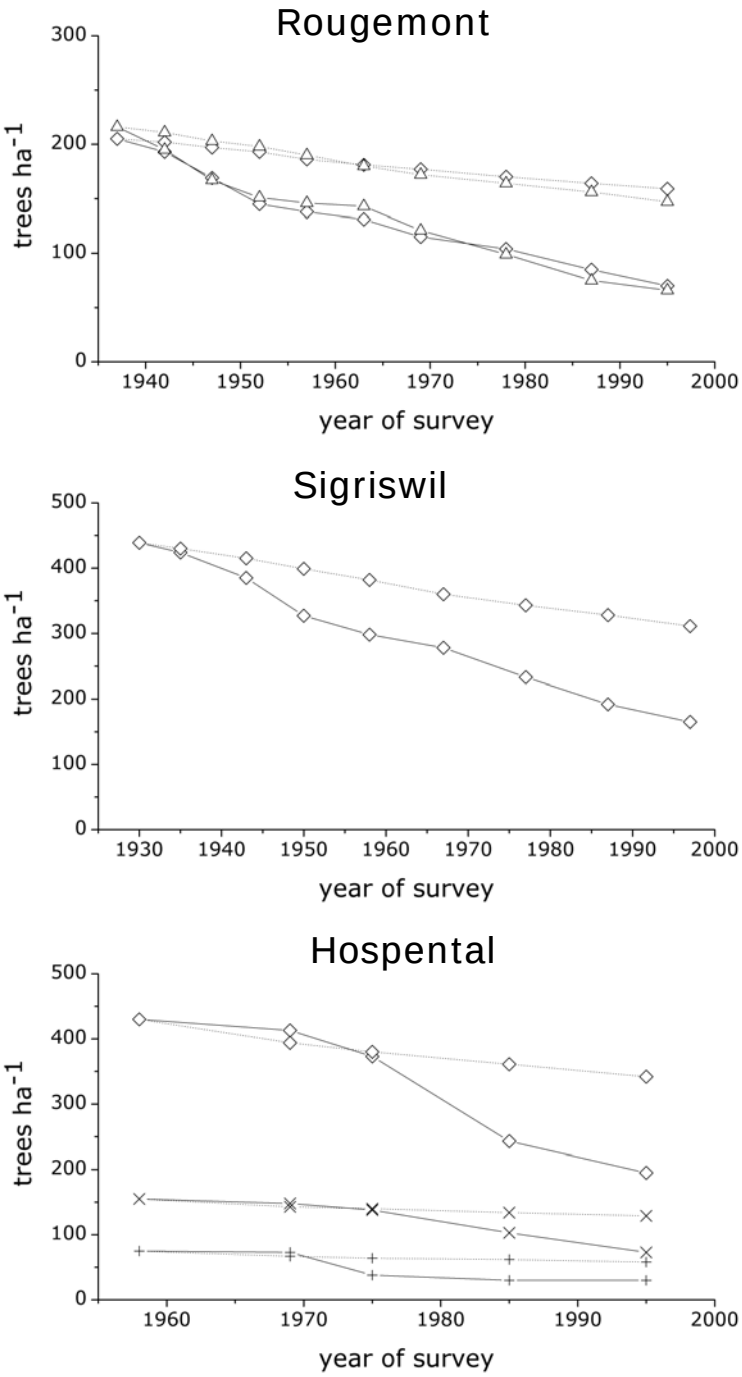

Figure 5. Empirical (solid line) and simulated (dashed line) number of surviving trees per species. Simulations were conducted with the reduced mortality rates, but without thinning regime. Symbols: (◊) *Picea abies*, (∆) *Abies alba*, (x) *Pinus cembra*, (+) *Larix decidua.*

to 21% of the total tree number. In contrast, mortality due to thinning was higher-than-average at Rougemont from 1969-1987 and at Sigriswil in 1950 and 1977, which yields that the natural mortality in the stands was probably lower than in the preceding years. These facts from stand history probably explain the simulated overestimation of mortality rates in the corresponding years as shown in Figure 6.

Table 4. Relative model bias s% (mean ± standard deviation over time series) for the main tree species.

	Rougemont	Sigriswil	Hospental
Picea abies	-0.01 ± 0.02	-0.12 ± 0.08	0.13 ± 0.22
Abies alba	-0.13 ± 0.11	-	-
Larix decidua	-	-	-0.09 ± 0.04
Pinus cembra	-	-	0.15 ± 0.27

Notes: Simulations were conducted with reduced mortality parameter values and prescribed thinning routine. Signs indicate under- (negative) or overestimation (positive) compared to the empirical data. **-** species does not occur on the site.

Our results demonstrate that the mortality submodel included in ForClim V2.9.2 overestimates natural mortality rates. Particularly, the mortality of suppressed trees was much too high, which corresponds to the findings of Lindner et al. (1997). This phenomenon is either caused by an inappropriate model of the stress-induced mortality (cf. Bigler & Bugmann, 2004), an inaccurate growth function in the lower DBH-classes, or an overestimation of the light competition on a patch. The latter two effects lead to a reduced DBH increase for suppressed trees and thus to an increased mortality rate of these trees due to stress-induced mortality. Hence, the inaccurate description of growth may lead to an inaccurate prediction of mortality – a fact that has already been criticized by Keane et al. (2001).

The mortality of trees still is a relatively poor known aspect of their ecology (Shugart, 1998), and mortality algorithms have been mostly limited to general relationships because of sparse data on the causal mechanisms of mortality (Keane et al., 2001). For this reason, Keane et al. (2001) suggested combining age-related and growth-related (i.e., stress-induced) mortality to keep patch models simple and consistent, if the explanatory power of models cannot be improved by other approaches (Loehle and LeBlanc, 1996).

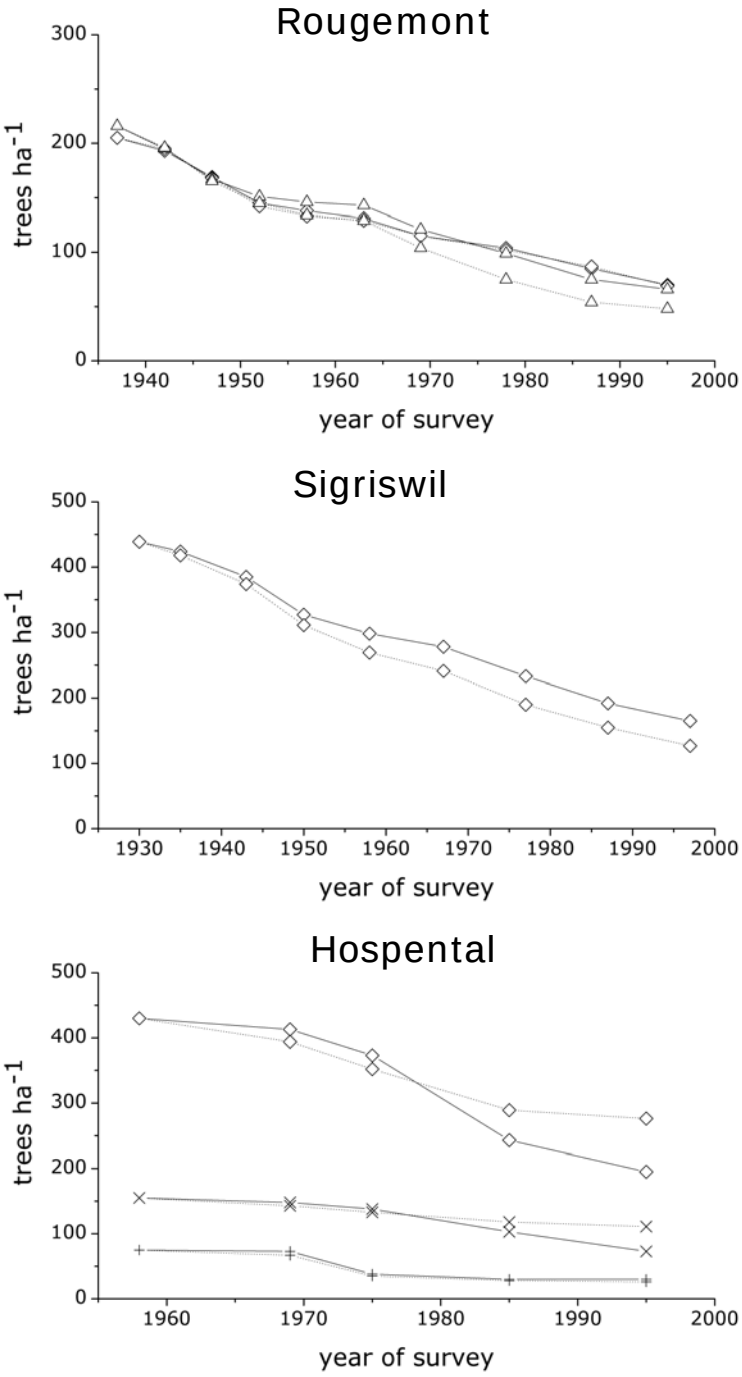

Figure 6. Empirical (solid line) and simulated (dashed line) number of surviving trees per species. Simulations were conducted with reduced mortality rates and prescribed thinning routine. Symbols: (◊) *Picea abies*, (∆) *Abies alba*, (x) *Pinus cembra*, (+) *Larix decidua.*

Thus, we suggest that our pragmatic modification of mortality parameters which led to the close match between simulated and empirical mortality rates could be included in simulation experiments in managed Swiss mountain forests until a better understanding of tree mortality in nature is available. For simulation experiments in unmanaged stands, however, the suggested exclusion of the stress-induced mortality would probably result in an underestimation of the actual mortality rates. This could, however, be compensated by increasing the age-related mortality rate in the model. In this way, mortality would not depend on the accuracy of the included growth function.

We are aware that the proposed modifications do not solve the problem of an inaccurate growth function in the lower DBH-classes or an overestimation of the light competition on a patch. These features need further investigation.

Diameter distribution

Stand density has a strong influence on tree growth and tree size. Therefore, an accurate simulation of diameter distributions can only be expected if the densities of observed and simulated trees are equal. This is due to the fact that tree growth in ForClim is mainly driven by light competition, which in turn is depending on the number of trees on a patch.

Using the modified model set-up described above (i.e. reduced mortality rate and prescribed thinning routine), ForClim was able to predict the diameter distribution reasonably well for about one decade for the Hospental site (Fig. 7) and for about three decades for the Rougemont site (Fig. 8), as indicated by the low X^2-values in Table 5. Since small *Abies alba* with a DBH < 10 cm were uncommonly frequent in the initial empirical stand at Rougemont, and since the accuracy of the tree growth submodel of ForClim is presumed to be low in these lower DBH-classes (see above), only data *Abies alba* with a DBH > 10 cm were included in Table 5. The inclusion of all DBH-classes for *Abies alba* led to an inferior fit. For the Sigriswil site (Fig. 9), the goodness of fit was already moderate after 13 years (Tab. 5). For simulations over longer periods, the accuracy of the prediction decreased for all sites.

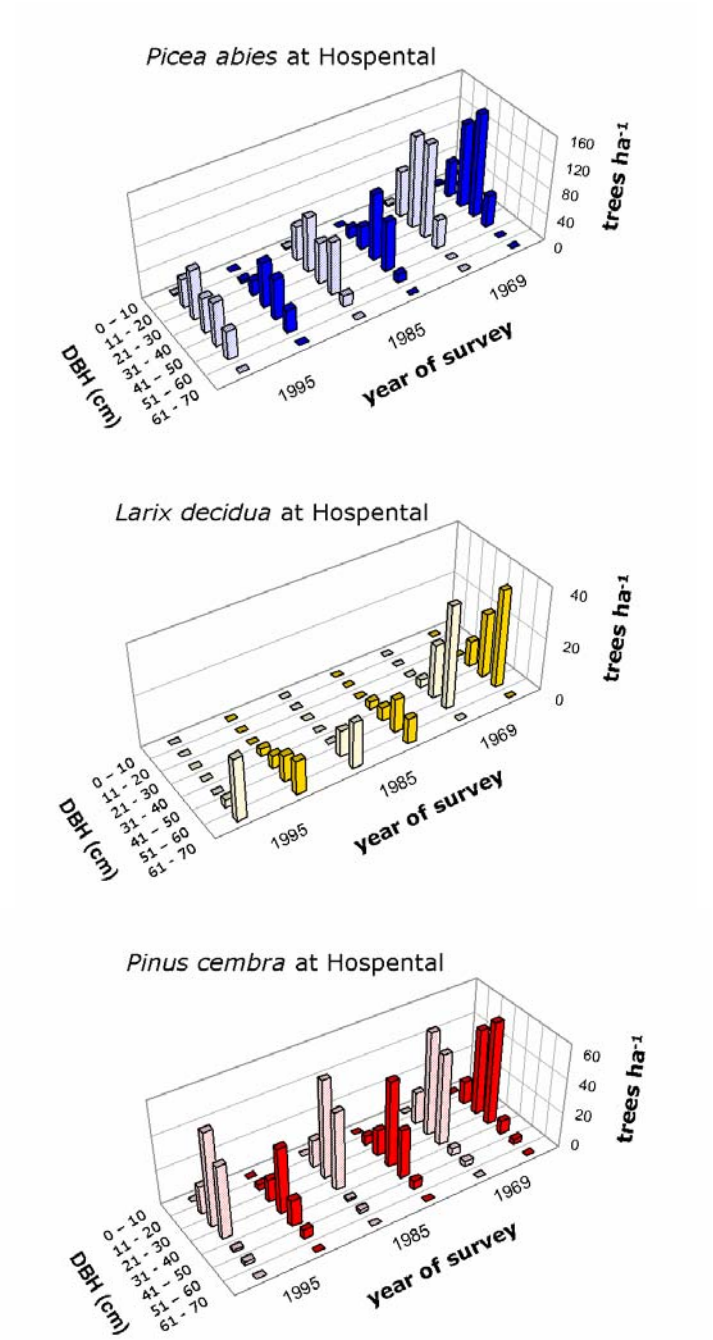

Figure 7. Empirical (solid columns) and simulated (dashed columns) diameter distribution of *Picea abies*, *Larix decidua* and *Pinus cembra* at Hospental 11, 27 and 37 years after start of the simulation. Simulations were conducted with reduced mortality rates, prescribed thinning regime and without regeneration.

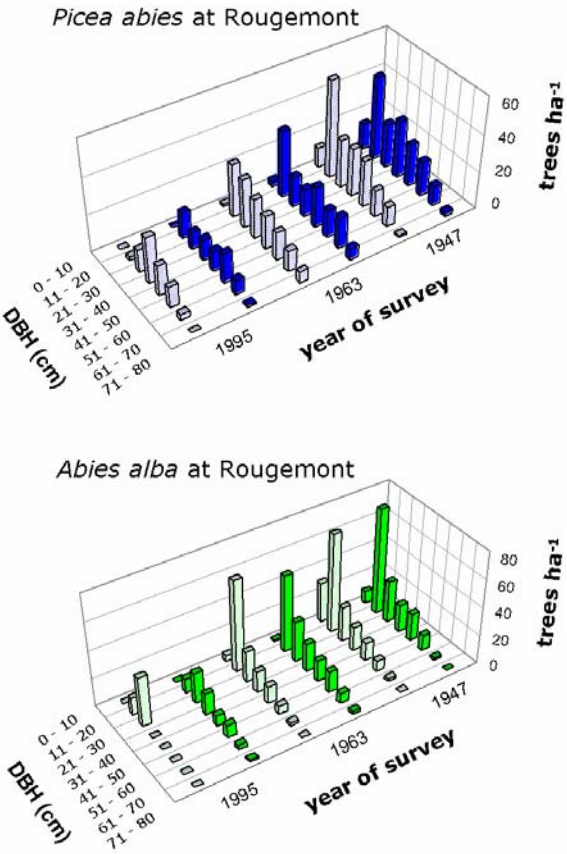

Figure 8. Empirical (solid columns) and simulated (dashed columns) diameter distribution of *Picea abies* and *Abies alba* at Rougemont 10, 26 and 58 years after start of the simulation. Simulations were conducted with reduced mortality rates, prescribed thinning regime and without regeneration.

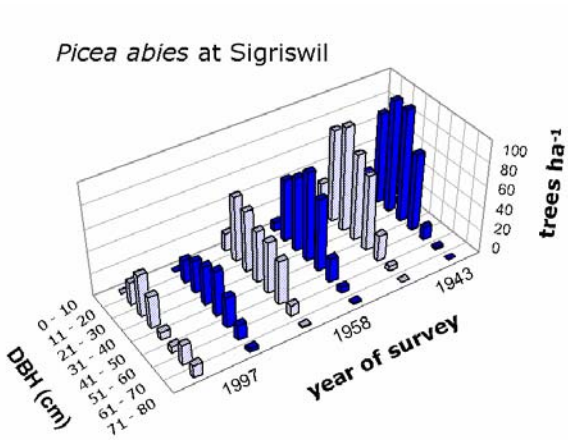

Figure 9. Empirical (solid columns) and simulated (dashed columns) diameter distribution of *Picea abies* at Sigriswil 13, 28 and 67 years after start of the simulation. Simulations were conducted with reduced mortality rates, prescribed thinning regime and without regeneration.

Table 5. Goodness of fit between empirical and simulated diameter distribution of main tree species.

	Rougemont			Sigriswil			Hospental		
	1947	1963	1995	1943	1958	1997	1969	1985	1995
Picea abies	5.06	14.45	37.59 *	21.35 *	54.33 *	80.81 *	11.94	166.96 *	294.28 *
Abies alba	2.26[1]	16.18[1]	52.90[1] *	-	-	-	-	-	-
Larix decidua	-	-	-	-	-	-	7.83	16.39 *	24.91 *
Pinus cembra	-	-	-	-	-	-	19.22 *	142.15 *	114.68 *

Notes: Simulations were conducted with reduced mortality parameter values and prescribed thinning routine. Values per species and survey year denote X^2-square between empirical and simulated DBH-distribution in 10 cm classes.
*: X^2-square values with $p < 0.01$, indicating that the model does not fit well. - species does not occur on the site.
[1]: as indicated in the text, goodness of fit for *Abies alba* was made without the trees with a DBH ≤ 10 cm.

The reasons for the moderate accuracy of the prediction at the Sigriswil site and the decreasing accuracy over time can be manifold. On the one hand, the results of the mortality experiments provide evidence for a model-intrinsic factor, i.e. an inaccurate growth function or an inexact integration of light competition in ForClim. This certainly affects the accuracy of the predicted diameter distribution and could be the reason for the surplus of simulated small *Abies alba* at site Rougemont (Fig. 8), and of simulated small *Picea abies* at sites Sigriswil in 1985 (Fig. 9) and Hospental in 1995 (Fig. 7), respectively. Therefore, the growth function should be investigated further, with a special focus on the growth of small trees. Furthermore, the integration of light competition should be analysed in detail. In this context, the integration of a 3-dimensional crown structure or the implementation of a light model for both direct and diffuse radiation as in the patch model Picus (Lexer and Hönninger 2001) could be considered, since these features are known to influence tree growth in mountain forests.

On the other hand, however, there are several model-independent factors, which can have an effect on the simulation results. First, the real spatial relationships and therefore the real light competition in the stand could not be included due to a lack of spatial explicitness in the empirical data. Since tree growth in ForClim is mainly driven by light competition, this factor can influence the performance of ForClim to a considerable degree.

Second, the weather data used for the abiotic environment submodel (ForClim-E) were not recorded on the sites, but at the closest weather stations, i.e. at a distance of several kilometres. Therefore, the weather input files derived from these “regional” weather records probably do not reflect the

local weather conditions or any microclimatic features of the forest stand to a sufficient degree. A low quality of weather input can affect the model performance indirectly, since tree regeneration and growth in ForClim are depending on temperature- and precipitation-derived indices (growing season temperature, soil moisture, absolute minimum winter temperature).

The weather input files for the Rougemont and Hospental sites probably reflect the weather conditions in the forest stands quite well. The weather input for Sigriswil, however, is probably only of moderate quality, due to the large horizontal and vertical distance to the closest weather station (cf. Tab. 1). Therefore, it could be of prime responsibility for the poor fit between empirical and simulated diameter distribution at the Sigriswil site.

A third model-independent factor that affected the accuracy of the prediction can be found in the application of the simple thinning routine. Since simulated and empirical tree growth were not absolutely equal due to the reasons mentioned above, the numbers of simulated trees in some diameter classes were lower than the prescribed number for harvesting in this class towards the end of a time series. In these cases, trees from the closest DBH-class had to be harvested as substitutes in order to follow the thinning regime as closely as possible. Thus, the manual harvesting of the empirically prescribed number of individual trees per diameter class led to an amplification of the differences between simulated and empirical diameter distribution over time. The bimodal diameter distribution of simulated *Picea abies* at Sigriswil in 1997 (Fig. 9) is very likely a result of this fact.

Other model-independent factors such as the small size of the study sites (0.4-1.5 ha) with the corresponding relatively low number of trees per DBH class and species may have influenced the prediction of the diameter distribution as well, but the three factors mentioned above are probably of prime importance.

The decreasing accuracy of the predicted diameter distribution over time is very likely due to a combination of several of the model-intrinsic and model-independent factors mentioned above. With the data available from the permanent plots, however, it was impossible to quantify the effects of each factor on the simulation results. Nevertheless, we think that besides the

mortality function, especially the growth function and the algorithm of the light competition included in ForClim need further investigation and improvement.

Conclusions

The aim of this study was first to test the capability of the forest patch model ForClim in predicting different structural forest patterns of European mountain forests. The second aim was to assess the performance of the regeneration, mortality, and growth algorithms included in ForClim. The model test was performed using empirical data from permanent plots, which is considered to be a powerful approach for testing forest patch models (Bugmann, 2001a).

Different shortcomings of ForClim were detected and quantified in the test. In particular, the mortality submodel included in ForClim V2.9.2 was found to overestimate natural mortality in the case of our forest sites, which may be due to an inaccurate growth function. However, after a slight modification of the establishment and mortality submodels, ForClim delivered results which were closer to the empirical patterns for short-term simulations. For longer simulation periods (i.e. more than 30 years), the accuracy of the prediction of the diameter distribution decreased, which may be due to several model-intrinsic and model-independent factors.

Different authors (e.g., Shugart and Prentice, 1992, Leemans, 1992, Lindner et al., 1997) state that forest patch models do not have the same precision in predicting structural forest patterns as empirical forest stand simulation models such as e.g., SILVA (Pretzsch, 2001, Pretzsch et al., 2002). This is mainly due to the fact that forest patch models are more general models which try to achieve a plausible performance under a wide range of environmental conditions and over long-term simulation periods, whereas empirical forest stand simulation models are designed to accurately predict the increase in timber volume over a relatively short period (10-20 years, cf. Hasenauer et al., 2000).

However, our study reveals that ForClim is able to predict structural patterns of mountain forest stands in an accurate manner for two to three

decades, if a slightly modified model set-up is used. Therefore, we conclude that ForClim as well as other forest patch models could become an important tool for further applications in forest research in the future. Not only could the forest patch models be used as a tool to support decision making in forest management under global change, as pointed out by Lindner et al. (2000), but the models could also be used to predict stand structures which in turn could be analyzed for their susceptibility to disturbances (Lindner et al., 1997, Huth and Ditzer, 2001) or for their long-term protective effect against natural hazards (cf. Wehrli et al., 2003).

Even if we are convinced that ForClim can be used to address further research questions, the results of our test give evidence of shortcomings of the model and provide starting points for future model enhancements. First of all, the growth function included in ForClim should be further investigated in order to increase the accuracy of the predicted diameter distribution over longer periods. An improved growth function would probably remove or at least reduce the detected shortcoming of the mortality submodel. Furthermore, an improvement of the establishment submodel would be desirable. This could be done by introducing several features of the regeneration process (e.g., seed dispersal, explicit browsing impact) as postulated by Price et al. (2001).

Acknowledgements

This study was carried out within the framework of the National Research Programme 48 (Landscapes and Habitats of the Alps). The modelling part was mainly performed during a Marie Curie Fellowship at the department of Ecological Modelling at the Centre for Environmental Research at Leipzig. The study was funded by the Swiss National Science Foundation (grant 4048-064409) and the Federal Office for Education and Science (Marie Curie Fellowship; grant No. 02.0392).

We would like to express our gratitude to the SMA (MeteoSwiss) for providing us with the long-term weather records. We very much appreciate comments on earlier drafts of this paper by Andrea D. Kupferschmid Albisetti, Walter Schönenberger and Peter Brang and two anonymous reviewers.

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Paper II

Assessing the protective effect of mountain forests against rockfall using a 3D simulation model

Based on:

Stoffel, M., A. Wehrli, R. Kühne, L.K.A. Dorren, S. Perret, and H. Kienholz. 2006. Assessing the protective effect of mountain forests against rockfall using a 3D simulation model. Forest Ecology and Management. In press.

Abstract – *We used one of the few rockfall models explicitly taking trees into account and compared the results obtained with the 3D simulation model Rockyfor with empirical data on tree impacts at three mountain forests in Switzerland. Even though we used model input data with different resolutions at the study sites, Rockyfor accurately predicted the spatial distribution of trajectory frequencies at all sites. In contrast, Rockyfor underestimated mean impact heights observed on trees at the two sites where high- and medium-resolution input data were available and overestimated them at the site where input data with the lowest resolution data were used. By comparing the results of the simulation scenarios "current forest cover" and "non-forested slope", we assessed the protective effect of the current stands at all three sites. The number of rocks reaching the bottom parts of the study sites would, on average, almost triple if the "current forest cover" were absent.*

We therefore conclude that Rockyfor is able to predict the spatial distribution of rockfall trajectories on forested slopes accurately, based on input data with a resolution of at least 5 m x 5 m. With the increasing availability of high-resolution data, it provides a useful tool for assessing the protective effect of mountain forests against rockfall.

Keywords: Protection forest; natural hazards; Rockyfor; dendrogeomorphology; rockfall; 3D simulation model; Swiss Alps

Introduction

Many mountain forests effectively protect people and their assets against natural hazards such as rockfall, snow avalanches, landslides, debris flows, soil erosion and floods (Brang et al., 2001). As a consequence, numerous settlements and transportation corridors in alpine regions directly depend on the protective effect of these forests and would – at least temporarily – become uninhabitable or inaccessible if this protection were to disappear or become inadequate (Bloetzer and Stoffel, 1998; Agliardi and Crosta, 2003).

In the Swiss Alps, rockfall and snow avalanches comprise the most common hazards, with evidence of rockfall observed in 31% and moving snow recorded in 37% of the National Forest Inventory (NFI) plots in mountain forests (Mahrer et al., 1988). While the large volumes and high energies occurring with snow avalanches often limit the protective effect of stands (Bartelt and Stöckli, 2001), the small masses that are generally involved in single rockfall events (< 5 m^3; Berger et al., 2002) allow mountain forests to absorb falling rocks (Leibundgut, 1986; Lafortune et al., 1997; Hétu and Gray, 2000). On forested slopes, both living and dead trees can stop falling rocks (Cattiau et al., 1995), whereas stems lying on the ground or root plates may act as barriers to rocks moving downslope (Mössmer et al., 1994; Schönenberger et al., 2005). Taking advantage of these effects on falling rocks, forest managers repeatedly tried to optimize the protective effect of their forests by applying target values for stand parameters such as tree density, spatial tree distribution, species composition, tree conditions, diameter distributions, and basal area (Chauvin et al., 1994; Wasser and Frehner, 1996; Frehner et al., 2005). While these target values undoubtedly provided a valuable tool for forest managers, they currently remain unsatisfactory, since those values are predominantly based on expert knowledge rather than on empirical data. Empirical data are very sparse, which is why the protective effect of a stand on a given site with a given damage potential could, up to now, only be assessed with considerable uncertainty (Dorren et al., 2005a). Given this lack of extensive empirical data on rockfall in mountain forests, dynamic modelling could provide a valuable

tool for investigating the protective effect of different stand structures against rockfall and for improving target values for forest management.

To be useful, a model should accurately predict different patterns of rockfall processes such as the spatial envelope of rockfall trajectories, impact heights of rocks, and runout zones. Furthermore, it should be applicable under various site and stand conditions and should consider the interaction of falling rocks with trees – as investigated by Jahn (1988), Zinggeler et al. (1991), Gsteiger (1993), Krummenacher and Keusen (1996), Berger and Lievois (1999) or Dorren et al. (2005a) – in sufficient detail.

The recently developed 3D rockfall model Rockyfor has accurately predicted different rockfall patterns for several forested and non-forested sites in mountainous terrain (Dorren et al., 2005b). The model operates with high-resolution input data (2.5 m x 2.5 m) so as to obtain sound results at the forest stand level. Such data hardly exist for many areas of the Alps. The model has also been shown to predict maximal runout zones with reasonable accuracy, even if based on low-resolution input data, i.e. 25 m x 25 m (Dorren and Heuvelink, 2004). In contrast, the minimum resolution of input data required to obtain realistic simulation results for other important rockfall features characterizing the protective effect of a stand (e.g. envelope of rockfall trajectories, mean impact height, mean velocity of rocks) is not yet known. Furthermore, even if Rockyfor accurately predicted rockfall runout zones and velocities at several sites in France and in Austria, it remains unknown whether it will reliably produce comprehensive results for other sites (Dorren et al., 2005b).

In this study, Rockyfor is applied and evaluated on three different sites in the Swiss Alps with different slope and stand characteristics as well as with data sets of different qualities. The simulated rockfall patterns were then compared with empirical data obtained from the study sites. Finally, we used Rockyfor to assess the protective effect of the investigated forest stands by comparing the results of simulation scenarios with and without the current forest cover.

Methods

The Rockyfor model

Rockyfor is a process-based rockfall simulation model that was originally developed with data obtained from field investigations in the Austrian Alps (Dorren et al., 2004). The model has since been improved and validated with data from 218 real-size rockfall experiments on forested and non-forested slopes in the French Alps (Le Hir et al., 2004; Dorren et al., 2005a).

Rockyfor uses raster maps as input files and simulates trajectories of falling, bouncing and rolling rocks (Ø < 0.5 m) and boulders (Ø > 0.5 m) within single raster cells. Moreover, it explicitly simulates the number of rockfall impacts against individual trees and sums them finally per raster cell. The model consists of three main modules. The first module calculates the rockfall trajectory, based on the topography of a site, which is represented by a Digital Elevation Model (DEM). At every step in the simulation, the fall direction of a rock can be towards one of the downslope cells from the cell where the rock is located during that simulation step. Hence, the model produces diverging rockfall trajectories.

The second main module calculates the energy loss due to impacts against single trees. As a result, the exact position of a falling rock and its current energy are modelled. If an impact against a tree takes place, the rock dissipates energy as a function of the relative position between rock and tree center and the stem diameter of the corresponding tree as follows:

$$\Delta E = -0.046 + \frac{0.98 + 0.046}{1 + 10^{(0.58-((Pi-CTA)/0.5*DBH))*-8.007}} \quad (1)$$

where *ΔE* = percentage of maximum amount of energy that can be dissipated by the tree [%]; *Pi–CTA* = horizontal distance between the position of the impact and the vertical central axis of the tree as seen from the impact direction [m]; *DBH* = stem diameter at breast height [m]. Thereby, the maximum amount of energy that can be dissipated by a tree (*max. E. diss.*) is a function of its DBH [m] as follows:

$$\max.E.diss. = FE\_ratio * 38.7 * DBH^{2.31} \quad (2)$$

where, *max.E.diss*. = maximum amount of energy that can be dissipated by a tree [kJ], *FE_ratio* = fracture energy ratio of a given tree species to *Abies alba*

(Mill.) as described by Dorren and Berger (2006) and DBH = stem diameter at breast height [cm].

The third main module calculates the velocity of the falling rock after a rebound on the slope surface (for details see Dorren et al., 2004). Here, the decrease of velocity after a rebound is mainly dependent on the tangential coefficient of restitution (r_t), which is determined by the composition and size of the material covering the surface as well as the radius of the falling rock itself (Kirkby and Statham, 1975). The coefficient is calculated as a function of the radius of the rock and the mean radius of the material on the ground as follows:

$$r_t = \frac{1}{1 + (D_{mean} / D_{rock})} \quad (3)$$

where D_{mean} = the mean diameter of the material on the slope surface [m]; D_{rock} = the diameter of the falling rock [m]. The calculated r_t is uniform randomly varied with 10% in order to take into account (i) the enormous local variation in the size of material covering rockfall slopes, as well as (ii) the geometry of the falling rocks. Furthermore, its value is limited to the range [0.1, 0.99] as to avoid unrealistic energy loss (Dorren et al., 2004, 2005b).

Study sites

Rockyfor was applied to three mountain forest sites in different areas of Switzerland. Figure 1 gives an overview of the sites, while a summary of relevant site characteristics is presented in Table 1.

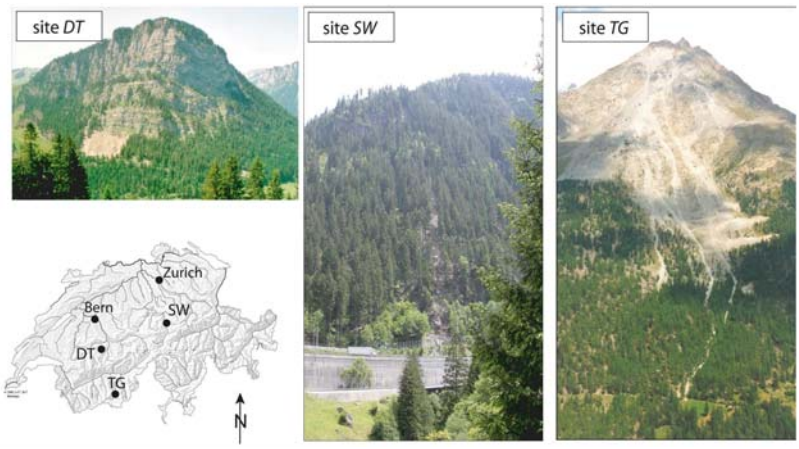

Figure 1. Localization of the study sites Diemtigtal (site DT), Stotzigwald (SW) and Täschgufer (TG).

Table 1. Characteristics of the three study sites.

Study site*(area)*	Altitudinal range [m a.s.l.]	Mean slope	Stand characteristics *(main tree species, tree density, mean DBH)*	*Predominating size of rocks (mean diameter)*
Diemtigtal (DT) 0.3 ha	1210 – 1280	40°	*Picea abies* 520 trees ha^{-1}, 21 cm	*0.2 m*
Stotzigwald (SW) 7.5 ha	650 – 1000	45°	*Picea abies*, *Abies alba* 561 trees ha^{-1}, 38 cm	*0.7 m*
Täschgufer (TG) 26 ha	1430 – 3214	20° – 48°	*Larix decidua* 150 trees ha^{-1}, 30 cm	*0.9 m*

The first site is a forest stand in the Diemtigtal (site DT) in the Swiss Prealps. The site lies at the foot of an approximately 400 m high limestone cliff (Fig. 1) on a southeast exposed talus slope with a mean slope gradient of 40°. The stand is dominated by *Picea abies* (L.) Karst. (77%), but other species such as *Sorbus aria* (L.) Crantz, *Sorbus aucuparia* L. or *Acer pseudoplatanus* L. occur as well (23%). The study site covers 0.3 ha, located between 1210 and 1280 m a.s.l. in the uppermost part of the talus slope. This area is in the transit zone of frequent, but mainly small falling rocks (Ø ~ 0.2 m). Below the study site, hiking trails and forest roads, which could be endangered by rockfall, traverse the slope.

The second site is the Stotzigwald (site SW), a stand in the central Swiss Alps covering a steep slope with a mean slope gradient of 45° and some interspersed cliffs (Fig. 1). This forest protects one of the most heavily used traffic routes connecting Germany and Italy. The elevation of the forest ranges from 650 to 1650 m a.s.l., but rockfall activity is mainly restricted to a zone of approximately 7.5 ha in the lower part of the forest, i.e. up to approximately 1000 m a.s.l. The stand within this zone mainly consists of *Picea abies* (83 %) and *Abies alba* (13 %). The slope is covered with rocks, boulders and morainic material. Bedrock consists of heavily weathered granite and gneiss. As a result, rockfall frequently occurs and rocks regularly reach the highway.

The third site is Täschgufer (site TG), which is located in the southern Swiss Alps (Fig. 1). Here, rockfall is frequently triggered from the heavily disintegrated paragneissic rockwalls below the Leiterspitzen summit (3214 m a.s.l.). In the upper part of the site, which covers 26 ha, mean slope gradients reach 48° and gradually decrease to 20° near the valley floor (1430 m a.s.l.).

In the central area affected by rockfall, continuous forest cover reaches 1780 m a.s.l., whereas the upper part of the slope remains mostly free of vegetation. The stand predominantly consists of *Larix decidua* Mill. (95%), accompanied by single *Picea abies* and *Pinus cembra ssp. sibirica*. In the recent past, rockfall regularly reached the valley floor, causing damage to roads, hiking trails and agricultural buildings.

Model input data

For all sites, extensive data on stand structure, geomorphological characteristics and rockfall patterns were available from earlier field studies. In addition, dendrogeomorphological data on century-long fluctuations in rockfall activity exist for *site TG*. A summary of all input data used for the simulation with Rockyfor is presented in Table 2.

In addition to the pre-existing data, complementary data on different site characteristics were gathered in the field (Kühne, 2005): Firstly, potential rockfall source areas were mapped, integrated into a Geographical Information System (GIS) and converted to raster maps. Secondly, terrain and vegetation parameters were mapped and polygons with homogeneous terrain characteristics described. From this data, a raster map was created for the normal coefficient of restitution (r_n; cf. Dorren and Seijmonsbergen, 2003) and for the mean diameter of the material covering the slope surface. The latter was required to calculate the tangential coefficient of restitution (r_t).

In a third step, complementary data on stand structure and empirical rockfall patterns were gathered on validation plots of approximately 225 m^2 (15 m $\times$ 15 m) at sites *SW* and *TG* so as to provide (i) data on the tree diameter distribution and (ii) validation data for the simulation experiments. On these validation plots, DBH was therefore measured for all individual trees with a DBH ≥ 8 cm. Rock impacts were assessed on the stem surface of trees, and the mean and maximum impact heights measured on every single tree. In order to derive forest stand maps as needed by Rockyfor, field data were coupled with a tree distribution map obtained from photogrammetric analyses, where every individual tree crown was mapped on orthophotos (scale 1:9’000). For site *DT*, no complementary data on stand and rockfall

patterns were gathered, since data for every single tree on the site were available.

Table 2. Data used for the simulation experiments.

Feature class	Available data	Sampling method	Source	Derived model input
Stand structure	• Species composition	Inventory sampling procedure on stand plots	Perret et al., 2004; Wehrli et al., in review; Stoffel et al., 2005a, b	Stand raster map
	• Diameter distribution	Inventory sampling procedure on stand plots	Kühne, 2005; Perret et al., 2004; Wehrli et al., in review; Schneuwly, 2003	
	• Tree density	Analysis of aerial photographs	Kühne, 2005; Perret et al., 2004	
Surface roughness	• Granular composition of surface material	estimation in 5 classes (< 0.2 m, 0.2-0.5 m, 0.5-1 m, 1-2 m, > 2 m)	Kühne, 2005	r_t-raster map
	• Vegetation cover	estimation of proportion of bushes and shrubs	Kühne, 2005	
Subsurface damping	• Damping properties of subsurface	estimation in 6 classes (bedrock, scree/talus, stony soil, dry forest soil, fine humid soil)	Kühne, 2005	r_n-raster map
DEM	• Laser scan data (site *DT*)	Interpolation to raster using Spatial Analyst (ESRI 2005)	DTM-AV © 2004 *Swisstopo* (DV033531)	DEM
	• Aerial photographs (sites *SW* & *TG*)	Deriving contour lines from aerial photographs and interpolation to raster using Spatial Analyst (ESRI 2005)	DEM © 2004 WSL, P. Thee (site *SW*) DEM © 2005 GIUB, R. Kühne (site *TG*)	
Rock properties	• Rock size	Estimation of rock size Description of recent accumulation (3 granular classes)	Kühne, 2005	rock size
Validation data	• Number of tree impacts due to rockfall	Count of tree injuries due to rockfall in test plots	Kühne, 2005; Perret et al., 2004	Validation raster map
	• Mean and max. impact heights	Assessment of H_{max}, H_{mean}	Kühne, 2005; Perret et al., 2004; Schneuwly, 2003; Stoffel et al., 2005a, b	

Finally, high resolution Digital Elevation Models (DEM) of 1m x 1m (site *DT*) and 5m x 5m (sites *SW* and *TG*) were produced. For site DT, we created a DEM by interpolating LIDAR (LIgth Detection And Ranging - Laser Scanning) point data delivered by the Swiss Topographical Service (Swisstopo) that

represent the ground surface. The applied interpolation method was Ordinary Kriging using on average 12 points per raster cell. For site SW, the DEM was derived from contour lines (equidistance: 12.5 m) created on the basis of photogrammetric analyses realized with high quality aerial photographs (scale 1:9'000). At site TG, in contrast, surface points and breakline features were generated from orthophotos (scale 1:9'000) and coupled with 10 m contour lines digitized from a topographic map in a scale of 1:10'000.

Simulation set-up

Since Rockyfor models rockfall on the basis of various stochastic algorithms, at least one hundred simulation runs from each potential rockfall source cell were needed to obtain sufficiently stable results (Dorren and Heuvelink, 2004). In the first simulation run, one rock was released from each defined rockfall source cell, one after another, which means that the trajectory of each rock has been calculated individually. This process was then repeated 99 times. Thus from a cliff that consists of 200 rockfall source cells, 20'000 rocks and consequently different trajectories and velocities will be simulated.

To account for the varying size of falling rocks and boulders at the three sites, we simulated varying numbers of rocks for each size class (cf. Table 4). The selected number of simulations per size class was derived from estimates in the field and represents the diameter size distribution of rocks at the three sites.

The initial fall height of the rocks for the simulation experiments was set to 30 m at site *DT*, to 5 m at site *SW* and to 3 m at site *TG*. These values were determined by the morphology of the rockfall source areas and they correspond to the mean height of vertical cliff faces in the source areas of the three sites. Simulation runs were first realized on the slopes with the "current forest cover", before trees were removed in the "non-forested" scenario and simulation runs repeated.

Assessment of model accuracy

Simulated and empirical rockfall patterns were compared on the basis of (i) the spatial distribution of rockfall impacts on trees, which is an indicator for the spatial distribution of the rockfall trajectories, and (ii) mean impact heights,

which are indicators for bounce heights of rocks. To compare the simulation results with empirical values, we used the arithmetic mean per cell of the calculated variable taking into account all the simulation runs, since one simulation run cannot reproduce the data gathered in the terrain. At site *DT*, the accuracy of the model was assessed at the level of single trees, since detailed data were available (Perret et al., 2005, 2006). At sites *SW* and *TG*, however, the analysis was performed on validation plots of 225 m^2 (see above). At site *TG*, simulation results were also compared with data on spatio-temporal variations in rockfall activity derived from dendro-geomorphological analysis of 129 living trees for the last 400 years (Stoffel et al., 2005b).

The number of rockfall impacts per tree was directly assessed from the empirical data for site *DT*. For sites *SW* and *TG*, a Tree Impact Coefficient *(TIC*) was calculated as:

$$TIC = \frac{TreeHits_j}{n_{Trees,\,j}} \qquad (4)$$

where $TreeHits_j$ = sum of tree impacts per validation plot *j*, and $n_{Trees,\ j}$ = number of trees in validation plot *j*. For standardization purposes, both empirical and simulated data were expressed as proportions relative to the summed values over all trees (*site DT*), and over all validation plots (*sites SW* and *TG*), respectively.

The mean impact height was calculated for every single tree (site DT) and for every validation plot (sites SW and TG). At site TG, impact heights were also integrated from century-old *Larix decidua* trees analyzed with dendrogeomorphological methods (Stoffel et al., 2005b). While tree-ring analysis yielded data on 786 rockfall impacts since 1394 AD, scars remained recognizable on the stem surface in less than 10% of all cases (Stoffel and Perret, in review), mainly representing relatively recent or unusually large evidence of past rockfall events.

In a subsequent step, mean (*ME*) and root mean-squared errors (*RMSE*) between the predicted and the observed number of impacts and mean impact heights were calculated as follows:

$$ME = \frac{1}{n}\sum_{i=1}^{n}(P_i - O_i) \qquad (5)$$

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(P_i - O_i)^2} \qquad (6)$$

where n = number of trees (site *DT*) or validation plots (sites *SW* and *TG*); P_i = predicted (i.e. simulated) and O_i = observed rock impacts at tree *i* (site *DT*) and validation plot *i* (sites *SW* and *TG)*.

Furthermore, the proportional difference between the predicted and the observed number of impacts was calculated for each tree (site *DT*) and each validation plot (sites *SW* and *TG*), and then illustrated for deviations ≥ 2.5% and ≥ 5%. Thus, the simulated trajectories of all size classes of rocks were summed and compared with the empirical patterns.

Assessment of the protective effect of the different stands

The protective effect of the stands was then assessed at the three sites by quantifying changes in the frequency of simulated rockfall trajectories between the scenarios "current forest cover" and "non-forested slope". Differences were assessed in "evaluation zones" at the foot of every test slope where high damage potential exists (i.e. roads or buildings). We defined these zones in a vector map, which was subsequently transformed into a raster map. We then compared the number of rocks that entered the "evaluation zones" in the simulated raster maps containing the stopping positions of the rocks as produced by the two modeling scenarios. This has been done for all the simulated diameter classes. The difference between the two scenarios has been quantified by the following ratio:

$$RF\_ratio_i = \frac{PR_{non-forested\_slope,i}}{PR_{forested\_slope,i}} \qquad (7)$$

where $PR_{non\text{-}forested\_slope,\ i}$ and $PR_{forested\_slope,\ i}$ = number of rocks per diameter class *i* passing the evaluation zone on the "non-forested" and forested slope. Furthermore, the protective effect of the two scenarios was compared graphically after summing the simulated trajectories over all diameter classes of rocks.

Results

Model accuracy

In general, the simulation experiments yielded very close matches between simulated and empirical spatial distributions of tree impacts on all three sites. However, for mean impact heights, the correspondence between simulated and observed data varied considerably.

At site DT, the simulated number of tree hits corresponds well with the empirical data, as indicated by the RMSE of 0.9% (Table 3). Figure 2a shows that differences predominantly occur in the uppermost sector of the study site, where the model overestimates the number of hits in nine trees by +2.5 and +5%. On the southwestern edge of the study site, the model, in contrast, underestimates the number of impacts in one tree by more than –5%. For the remaining 128 trees (93%), differences between the predicted and the observed number of tree hits remain between ±2.5%. On the other hand, differences can be seen with impact heights, where Rockyfor underestimates the mean impact height by –0.21 m (ME) and 0.46 m (RMSE). As can be seen from Table 3, deviations from the observed mean impact height (0.85 m), therefore, account for 26% (ME) and 57% (RMSE).

Table 3. Mean (*ME*) and Root Mean-Squared Errors (*RMSE*) between observed number of tree hits and impact heights (h_{mean}) and model results obtained at study sites *DT*, *SW*, *TG* dataset 1 and *TG* dataset 2 (for explanation see text).

Site	Number of Trees	Number of Tree Hits or TIC		Mean Impact Height		
		ME [%]	*RMSE* [%]	h_{mean} observed [m]	*ME* [m (%)]	*RMSE* [m (%)]
DT	138	0	0.9	0.8	-0.2 (-26)	0.4 (57)
SW	23	0	3.6	1.1	2.6 (230)	3.5 (310)
TG dataset 1	46	0	4.4	1.1	-0.6 (-59)	1.1 (96)
TG dataset 2	129 / 38	0	3.6	1.7	-1.4 (-85)	1.7 (101)

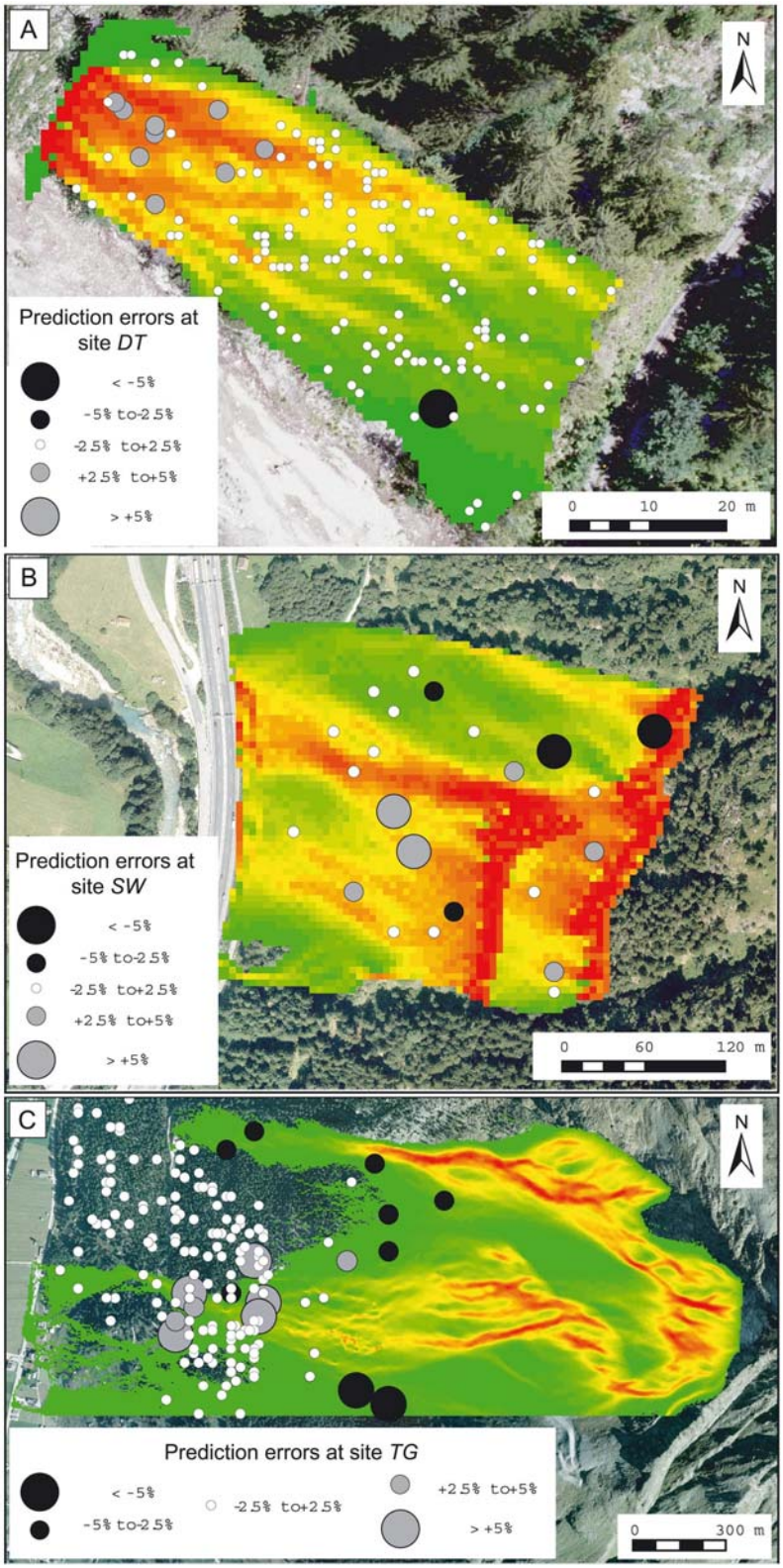

Figure 2. Differences between simulated and observed number of rock impacts on trees or validation plots at (a) *DT*, (b) *SW* and (c) *TG*. Gray circles indicate an overestimation by the model, whereas black circles show underestimation. White circles represent trees or validation plots with a very similar number of rock impacts for simulations and observations (±2.5%) (Orthophoto sources: site *DT*: © Baumgartner (2002); site *SW*: © Kanton Uri; site *TG*: © 2005 swisstopo (BA056895).

At site *SW*, the predicted *Tree Impact Coefficient (TIC)* pattern matches the empirical data from the validation plots, with a *RMSE* of 3.6% (Table 3). As illustrated in Figure 2b, the model more commonly overestimates the number of tree hits, but underestimates occur as well. Differences exceeding ±5% are identified in four validation plots (17%) and prediction errors ranging from 2.5 to 5% are present in six plots (26%).

In the other 13 validation plots (i.e. 57%), the difference between the predicted and the observed number of tree hits remains within a range of ±2.5%. As shown in Table 3, Rockyfor overestimates, however, mean impact heights with a *ME* of +2.6 m and a *RMSE* of 3.5 m. Compared to the mean impact height of 1.1 m observed in the field, this corresponds to an overestimation of 210% (*ME*) and 310% (*RMSE*), respectively.

At *site TG*, simulated rockfall data are compared with data gathered on 46 validation plots in the field (*TG dataset 1*) as well as with results from dendrogeomorphological reconstructions of past rockfall activity (*TG dataset 2*). As indicated in Table 3, the model again accurately predicts the empirical *TIC* pattern with a *RMSE* of 4.4% for *TG dataset* 1 and 3.6% for *TG dataset 2*. For *TG dataset 1*, underestimation occurs in eight validation plots (17%) with differences between the observed and the predicted number of tree impacts primarily remaining between –2.5 and –5%. For *TG dataset 2*, in contrast, overestimation can be observed in seven out of 129 validation plots (5%), mostly exceeding +5%, whereas underestimation can only be found in one plot (1%). Interestingly, the overestimated validation plots are concentrated along the upper fringe of the continuous forest stand and near the rockfall channel, as shown in Figure 2c. Results also indicate that in 80% (*TG dataset 1*) and 94% (*TG dataset 2*) of the validation plots, differences between simulated and predicted *TIC* patterns remain within a range of ±2.5%.

However, we observed a difference between the empirical and simulated impact heights. For *TG dataset 1*, the model underestimates the mean impact height observed in the field (1.1 m) with a *ME* of –0.6 m and a *RMSE* of 1.1 m. As can be seen from Table 3, this corresponds to a relative underestimation of 59% and 96%, respectively. For TG *dataset* 2, the observed mean impact height (1.7 m) is underestimated with a *ME* of –1.4 m (85%) and a *RMSE* of 1.7 m (101%).

Protective effect

The comparison between the two simulation scenarios "current forest cover" and "non-forested slope" yielded significant differences for all three sites. Table 4 first of all shows that the number of rocks and boulders passing the evaluation zones was considerably higher in the "non-forested" scenario.

Table 4. Assessment of the protective effect of the investigated stands. The number of rocks (RF) triggered per diameter class and start cell is given as *n*. The percentage of rocks and boulders passing the evaluation zones is given for the scenarios "forested slope" and "non-forested slope" and the differences between the two scenarios expressed with a *RF_ratio* (for explanation see text).

site start cells	forest cover	diameter of RF [m]	0.1	0.2	0.3	0.4	0.6	0.8	1	1.2	1.4	1.6	1.8	2
site *DT* start cells 12		RF triggered	1500	2000	1500	-	-	-	-	-	-	-	-	-
	forested:	passing RF [%]	0.7	5.2	13									
	non-forested:	passing RF [%]	6	18.1	32.4									
		RF_ratio	*8.5*	*3.5*	*2.5*									
site *SW* start cells 331		RF triggered	-	1200	-	1500	1200	1000	600	500	200	200	100	100
	forested:	passing RF [%]		1.5		3	5.2	5.7	6.7	7.9	8.4	8.8	8.7	8.7
	non-forested:	passing RF [%]		3.9		6.8	8.8	9.5	15.8	9.9	9.9	9.9	9.9	9.9
		RF_ratio		*2.5*		*2.3*	*1.7*	*1.7*	*2.4*	*1.3*	*1.2*	*1.1*	*1.1*	*1.1*
site *TG* start cells 7692		RF triggered	-	1200	-	1500	1200	1100	1000	800	700	600	500	400
	forested:	passing RF [%]		0		0	0	0.001	0.002	0.003	0.006	0.011	0.035	0.057
	non-forested:	passing RF [%]		0		0	0	0.004	0.011	0.018	0.026	0.043	0.076	0.107
		RF_ratio		*0*		*0*	*0*	*4.5*	*6.7*	*5.5*	*4.1*	*4.1*	*2.2*	*1.9*

At site *DT*, the complete removal of the forest stand would result in more rocks passing the evaluation zone at the foot of the study site, as indicated by the high *RF_ratio*s between the two scenarios. As can be seen from Table 4, this is particularly true for the smallest rock diameter class (0.1 m) used in the simulation runs, for which the *RF_ratio* increased by a factor of 8.5, as compared to the scenario with the "current forest cover". Figure 3 gives a qualitative impression of the differences emerging between the two scenarios, indicating that the increase in the transit of rocks is most obvious in the northeastern and central parts of the study site. In contrast, negative effects

appear to be less drastic in the southern half, where frequencies only slightly increased in the “non-forested” scenario.

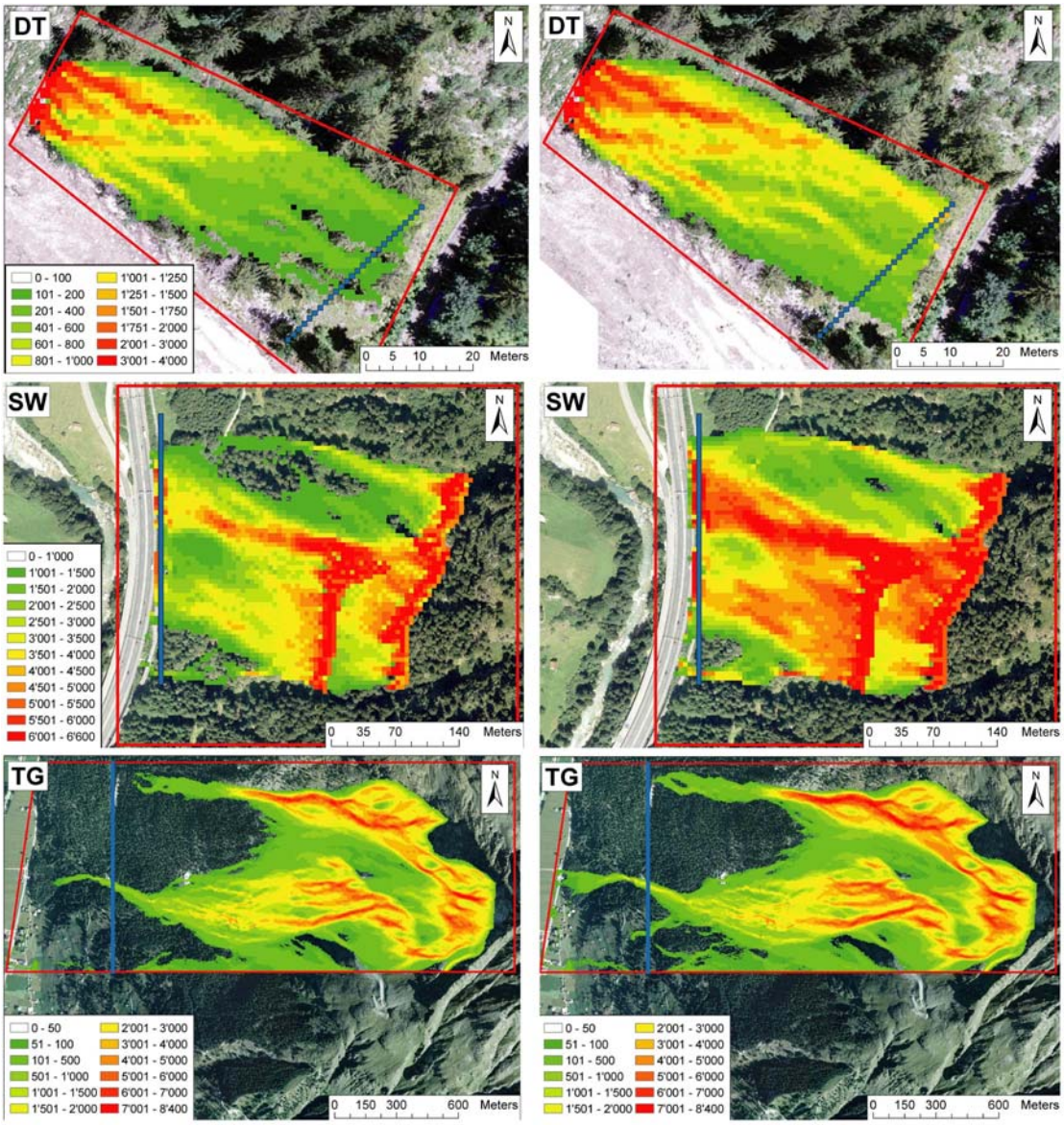

Figure 3. Comparison of simulated rockfall trajectories for the “current forest cover” (left) and for the “non-forested slope” (right) at (a) site *DT*, (b) site *SW*, and (c) site *TG*. Evaluation zones are indicated with blue lines.

At *site SW*, the number of rocks passing the evaluation zone is rather high in both scenarios. As shown in Table 4, differences between the two scenarios are most obvious for rocks and small boulders of up to 1 m in diameter, where *RF_ratio*s varied between a factor of 1.7 and 2.5. Figure 3 illustrates that in the “non-forested” scenario, a considerable increase in the

number of rockfall trajectories can be observed below the subvertical cliff in the central part of the study site. Results also indicate that in the scenario with the "current forest cover", rocks originating from this cliff would be partly stopped through the presence of trees, whereas the absence of trees would allow most rocks and boulders to travel down the slope and reach the adjacent highway. A similar protective effect of the stand is evident for rocks originating from the uppermost cliff area.

In contrast to sites *DT* and *SW*, Table 4 indicates that the differences in the number of rocks and boulders passing through the evaluation zone at site *TG* are considerably smaller for most diameter classes. Nonetheless, significant differences arise between the two scenarios for boulders with diameters > 0.8 m, as indicated by the high *RF_ratio*s in Table 4. The qualitative comparison of the scenarios in Figure 3 indicates a minor increase in the number of rocks passing down the slope in the rockfall channel located in the northwestern part of the study site, where the number of deposited rocks and boulders doubles in some locations. In the central part of the study site, rockfall activity increases as well, meaning that boulders would more frequently reach the main road in the valley floor (Fig. 3).

Discussion

Model accuracy

In the study we report here, the 3D rockfall simulation model Rockyfor was tested and its capability to accurately predict rockfall patterns assessed at three forested sites in the Swiss Alps. Overall, the comparison of observed with simulated rockfall patterns yielded a high correspondence for the spatial distribution of tree impacts and a low correspondence for the mean impact heights.

The closest match between empirical and simulated distributions of tree impacts was obtained at site *DT*, where a highly resolved DEM (1 m × 1 m) allowed very accurate modelling of rockfall trajectories. At sites *SW* and *TG*, the simulation based on a 5 m × 5 m DEM derived from contour lines with an equidistance of 12.5 m (site *SW*) and 10 m (site *TG*) still yielded close matches between the empirical and predicted distribution of tree impacts with

RMSE ≤ 4.4%. As stochastic elements are involved in rockfall processes, a complete agreement of empirical and simulated trajectories is unlikely, which is why we believe that for an accurate prediction of the spatial envelope of rockfall trajectories, a DEM with a resolution of 5 m × 5 m – as used at sites *SW* and *TG* – is largely sufficient.

In contrast to the spatial distribution of rockfall trajectories, the prediction of mean impact heights was less accurate. Even at site *DT*, predicted mean impact heights were considerably lower than the values observed in the field. This low correspondence is particularly surprising, since Rockyfor produced close matches between empirical and simulated mean impact heights for a site in the French Alps, based on a DEM with a resolution of 2.5 m × 2.5m (Dorren et al., 2005b). The underestimation of mean impact heights further increased at site *TG*, where the model produced a negative bias for both *TG dataset* 1 and *TG dataset* 2. In contrast, Rockyfor largely overestimated mean impact heights at site *SW*.

The reasons for the poor correspondence between the predicted and the observed mean impact heights may be manifold. Since Rockyfor was calibrated on the basis of more than 200 real-size experiments (Dorren et al., 2005a) and different rockfall patterns, including mean impact heights accurately predicted before, we believe the main reasons for the rather low agreement between empirical and simulated mean impact heights to be model-independent rather than model-intrinsic.

A first factor that might have affected the accuracy of the simulation results is the DEM, i.e. its spatial resolution. This is particularly true for sites *SW* and *TG*, where DEMs were derived from contour lines with relatively low resolution. Consequently, micro-topographical structures are neglected in the DEM, which in turn can influence the velocity of falling rocks. For instance, huge boulders from ancient rockslide deposits could not be included in the DEM at site *TG*. These boulders may, however, cause rocks to bounce and therefore produce higher impacts than suggested by the model. At site *SW*, the rather coarse DEM was probably the main reason for the large overestimation of the mean impact height, since on this steep slope with an average inclination of 45°, the frequently occurring, abrupt changes in the

slope gradient could probably not be reproduced in the DEM with sufficient accuracy. These observations are in agreement with Agliardi and Crosta (2003), who report a decrease in bounce height in the flatter parts of the slope and an increase in bounce height on steeper slopes as soon as the resolution of support data decreases. When using a 5 m × 5 m DEM, it would, therefore, be preferable if the underlying data had a support of at least five meters as well (e.g., 1–5 m contour lines or LIDAR data), so as to take essential terrain features into account.

A second factor influencing the mean impact height in the model can be identified in the uncertainty related to the delineation of rockfall source areas and initial fall heights. Within the present study, we determined rockfall source areas based on observations and geological advisory opinions. A large number of rocks were triggered from these start cells with a given initial fall height, which again was determined based on qualitative field observations. It is, however, clear that these observations can only be seen as an approximation to reality, since precise determination of rockfall source areas and initial fall heights was rendered impossible by complex terrain features such as the 400 m high limestone cliff at site *DT*, or subvertical cliffs at site *SW*. Nonetheless, a more precise assessment of the rockfall source areas and the initial fall heights seem to be decisive for a better prediction of bounce heights of rocks and, consequently, impact heights on trees.

A third factor affecting modelled impact heights is represented by the validation datasets, which were mainly based on the assessment of impact scars visible on the stem surface of trees. As previously shown by Stoffel (2005, 2006), scars as evidence of past rockfall events may become completely blurred with time and are, as a consequence, no longer visible on the stem surface. On the other hand, it is also conceivable that large scars caused by high-energy impacts at unusually high positions may persist for a long time on the stem surface and therefore lead to an overestimation of rare impact heights. Nonetheless and as half of the scars caused through the action of large rocks (Ø 0.8 m) prove to be no longer visible on the stem surface of *Larix decidua* after as little as 20 years (Stoffel and Perret, in review), we believe that our field observations of impact heights and trajectory

frequencies accurately illustrate the recent rockfall activity occurring in the current forest stands. Furthermore, the accuracy of the validation datasets was also influenced by the tree species and the age of single trees. At site *TG*, for instance, the overestimated plots illustrated in Figure 2c largely occur in areas where juvenile trees are recolonizing the slope. In contrast to their older neighbours, these trees only show a comparably low number of scars in the field, as there has not been sufficient time for scarring.

Other factors such as the small number of test plots for site *SW* or minor inaccuracies in the r_t/r_n-maps may have influenced the prediction of mean impact heights as well, but the three factors mentioned above are probably of prime importance. Nevertheless, model results clearly indicate that the 3D process-based model Rockyfor is able to accurately predict the spatial envelope of rockfall trajectories based on input data with a resolution of 5 m x 5 m.

Protective effect

The second aim of this study was to assess the protective effect of the investigated stands against rockfall through a comparison of simulation scenarios with the “current forest cover” and on a “non-forested slope”. This allowed the quantification of the protective effect of the forest stands at the three sites, as indicated by the *RF_ratio*s between the scenarios.

The protective effect of the stands is highest for rocks and small boulders (diameter ≤ 1 m) at sites *DT* and *SW* (cf. Table 4). Here, the number of rocks passing through the evaluation zones is between 1.7 and 8.5 times higher in the “non-forested” scenario, indicating an effective protective function of the current stands. At site *TG*, the stand seems, in contrast, to protect objects at risk from (large) boulders ranging from 0.8 to 2 m in diameter rather than from smaller rocks. Even though rocks passing through the evaluation zone appear to occur much less frequently here, the protective effect of the stand should not be discounted.

We therefore believe that at all three sites, the hazard potential would increase strongly without the current stands: At site *DT*, rocks would more frequently reach the forest road and endanger the nearby hiking trails, whereas at sites *SW* and *TG*, important infrastructure would be endangered

(site *SW*: highway, site *TG*: main road, buildings). Given the high damage potential at sites *SW* and *TG*, several countermeasures have been taken in the recent past. At site *SW*, restraining nets have been constructed along the highway, whereas at site *TG*, seven dams have been built on the slope. As evident from our simulation results, these countermeasures are more than justified.

Conclusions

This study clearly showed that, based on input data with a resolution of at least 5 m x 5 m, Rockyfor is able to accurately predict the spatial distribution of rockfall trajectories on forested sites with different slope and stand characteristics. In contrast, we were unable to confirm an accurate prediction of mean impact heights in the present. We believe that high-resolution input data including e.g., a laser scan based DEM, a better knowledge of rockfall source areas, and data on initial fall heights would considerably improve the quality of the predicted impact heights. In addition, the use of dendro-geomorphological analyses increases the amount and quality of validation data and should be used more systematically in rockfall forest research.

Due to its ability to accurately predict the spatial envelope of rockfall trajectories, the present version of Rockyfor also provides a valuable research tool for investigating the protective effect offered by different stand structures.

Acknowledgements

We are grateful to the European Commission for the Marie Curie Fellowship (QLK5-CT-2002-51705). Additional funding was provided by the Swiss National Science Foundation (grant 4048-064409) and the Swiss Forest Agency. We kindly acknowledge Marc Baumgartner, Catherine Berger, Monika Fässler, Connie Hett, Dominique Schneuwly and Mathias Zesiger for assistance in the field. Igor Lièvre and Patrick Thee are acknowledged for generating contour lines for sites *TG* and *SW*. We are grateful to Walter Schönenberger for his valuable comments on a former draft and Heather Murray for improving the English of the paper. Finally, we would like to thank Swisstopo for providing the aerial pictures.

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Section summary

The aim of section I was to assess the suitability of the forest patch model ForClim (Bugmann, 1994) and the process-based rockfall model Rockyfor (Dorren, 2002) for a combined simulation tool (*CoST*), which can be used to investigate the forest protection system. Moreover, shortcomings of these two models with regard to their use in a *CoST* were identified.

In *Paper I*, ForClim was shown to be able to predict structural patterns of mountain forests stands in an accurate manner for two to three decades, if slight modifications of the establishment and mortality submodels were used. For longer simulation periods, the accuracy of the predictions decreased, which was probably due to several model-intrinsic (e.g., an inaccurate growth function or an inexact integration of light competition) and model-independent factors (e.g., a lack of spatial information in stand input data, inaccurate weather data, a simplistic thinning routine, cf. *Paper I*). Thus, ForClim is in principle suitable for a simplistic *CoST*, if the major shortcomings detected in *Paper I* are improved. These major shortcomings include (1) the establishment submodel, which led to unrealistically high numbers of young trees and (2) the reproduction of the light competition in the model, which led to a strong overestimation of stress-induced mortality.

In *Paper II*, the process-based rockfall model Rockyfor was shown to allow accurate predictions of the spatial distribution of rockfall trajectories on three forested slopes with different slope and stand characteristics, based on input data with a resolution of at least 5 m x 5 m. However, Rockyfor underestimated mean impact heights observed on trees at those two sites where high- and medium-resolution input data were available, and it overestimated them at the site where input data with the lowest resolution data were used. Still, the protective effect of different stands could be assessed and was considerably high on all sites: The number of rocks reaching the bottom parts of the study sites would, on average, almost triple if the current forest cover were absent. Thus, the present version of Rockyfor model was found to be a valuable tool for investigating the protective effect of different stands and, therefore, it can be used for a spatially explicit, 3D *CoST* in its present form.

Section conclusion

As evidenced by the findings of *Paper I* and *II*, the performance of the two models, and thus their suitability for a *CoST*, is rather different. The current ForClim model needs some major improvements to be just suitable for a simplistic *CoST*. The Rockyfor model, in contrast, could be used for a 3D *CoST* in its current version.

Given these differences, a combination of ForClim and Rockyfor does not seem to be appropriate at the moment: To be suitable for a combination with Rockyfor in a 3D *CoST*, not only the major shortcomings of ForClim detected in *Paper I* should be improved, but the model should additionally become more spatially explicit, i.e. horizontal relationships between individual patches should be taken into account. This, however, could probably only be achieved with rather extensive and time-consuming modifications, which clearly go beyond the scope of this PhD thesis. Moreover, such modifications are not of prime importance since significant characteristics of the protection forest system could be investigated with a simplistic *CoST* that neglects the spatial dimension (e.g., the influence of different levels of tree regeneration on the long-term protective effect). In contrast, the improvement of the major shortcomings of ForClim detected in *Paper I* is mandatory for including the model in a *CoST*.

Thus, instead of trying to develop a spatially explicit version of ForClim that would be appropriate to be combined with Rockyfor in a 3D *CoST*, I will in the following focus on developing a simplistic *CoST*, based on ForClim and a simplistic rockfall model (see General Introduction). For this purpose, the performance of ForClim will first be enhanced in section II. This will be done by improving the reproduction of light competition and by adapting the establishment submodel (*Paper III*).

In section III, the new ForClim model version will finally be joined with the simplistic empirical rockfall model RockFor^{NET} (Berger and Dorren, in review) to a simplistic *CoST*. This simplistic *CoST* will then be applied to a case study to give an example of its use for investigating the protection forest system *(Paper IV*).

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Section II

Adapting the forest patch model ForClim for the use in a *CoST*

Paper III

Improving the establishment submodel of a forest patch model to assess the long-term protective effect of mountain forests

Section summary

Paper III

Improving the establishment submodel of a forest patch model to assess the long-term protective effect of mountain forests

Based on:

Wehrli., A., P.J. Weisberg, W. Schönenberger, P. Brang, and H. Bugmann. In review. Improving the establishment submodel of a forest patch model to assess the long-term protective effect of mountain forests. In review with the European Journal of Forest Research.

Abstract *– Simulation models such as forest patch models can be used to forecast the development of forest structural attributes over time. However, predictions of such models with respect to the impact of forest dynamics on the long-term protective effect of mountain forests may be of limited accuracy where tree regeneration is simulated with little detail. For this reason, we improved the establishment submodel of the ForClim forest patch model by implementing a more detailed representation of tree regeneration. Our refined submodel included canopy shading and ungulate browsing, two important constraints to sapling growth in mountain forests. To compare the old and the new establishment submodel of ForClim, we simulated the successional dynamics of the Stotzigwald protection forest in the Swiss Alps, over a 60-year period. This forest provides slope stability for an important traffic route, but currently contains an alarmingly low density of tree regeneration. The comparison yielded a significantly delayed regeneration period for the new model version, bringing the simulations into closer agreement with the known slow stand dynamics of mountain forests.*

In addition, the new model version was applied to forecast the future ability of the Stotzigwald forest to buffer the valley below from rockfall disturbance. Two scenarios were simulated: (I) canopy shading but no browsing impact, and (II) canopy shading and high browsing impact. Under both scenarios, the initial sparse level of tree regeneration affected the long-term protective effect of the forest, which considerably declined during the first 40 years. In the complete absence of browsing, the density of small trees was slightly increased after 60 years, raising hope for an eventual recovery of the protective effect. In the scenario that included browsing, however, the density of small trees remained at very low levels.

With our improved establishment submodel, we provide an enhanced tool for studying the impacts of structural dynamics on the long-term protective effect of mountain forests. For certain purposes, it is important that predictive models of forest dynamics adequately represent critical processes for tree regeneration, such as sapling responses to low light levels and high browsing pressure.

Keywords: mountain forest, protection forest, tree regeneration, ungulate browsing, forest patch model, ForClim, stand density

Introduction

Many mountain forests effectively protect people and their assets against natural hazards such as snow avalanches and rockfall (Brang et al., 2001). If the current protective effect of a forest is provided mainly by the adult trees, effective long-term protection is only possible if permanent tree cover is ensured by sufficient rates of renewal. A lack of regeneration is difficult to counter because trees grow slowly at high altitudes (Ott et al., 1997), and established tree seedlings require up to several decades before they contribute to the protective effect.

In many Swiss mountain forests, tree regeneration currently seems to be sparse. Based on inventory data, several authors concluded that the level of tree regeneration in mountain forests dominated by *Picea abies* (L.) Karst., which represent a large fraction of the protection forests in Switzerland, is quite low (Brändli and Herold, 1999, Brang and Duc, 2002, Zinggeler et al., 1999). The main reasons for the low level of tree regeneration are thought to be (i) recent silvicultural practices which avoided regeneration harvests, resulting in dense and even-aged stands with low light availability for the understory, and (ii) an increased browsing pressure due to high ungulate population densities (Duc and Brang, 2003). Canopy shading and browsing pressure interact in a complex causal relationship (Reimoser and Gossow, 1996, Bugmann and Weisberg, 2003), often leading to reduced sapling growth and prolonging the regeneration phase of forest development (Eiberle, 1975, Guler, 2004). This may further result in the loss of ecologically valuable tree species, such as *Abies alba* Mill. (Ammer, 1996a, Guler, 2004, Wasem and Senn, 2000). In this way, the protective effect of a forest might be reduced in the long term.

The importance of browsing and shading influences is difficult to determine, due to the scarcity of long-term data sets for mountain forest regeneration under different conditions of browsing and shading (but see e.g., Mosandl and El Kateb, 1988). Given this data gap, dynamic modelling may be a useful tool for investigating the structural dynamics of mountain forests and forecasting long-term changes in their ability to provide protection from rockfall and snow avalanches.

A useful model for this purpose should accurately project the development of structural forest patterns that characterize the protective effect of a forest, such as its diameter distribution. Additionally, the regeneration process needs to be included in sufficient detail to reflect the most important features of mountain forest tree regeneration.

Forest patch models have been relatively successful in reproducing structural forest patterns (cf. Huth and Ditzer, 2000, Lindner et al., 1997, Shugart, 1998, Risch et al., 2005, Wehrli et al., 2005). The regeneration process, however, is normally simulated without great detail (cf. Price et al., 2001). For example, sapling growth is seldom simulated explicitly, and the interactions between herbivores and tree regeneration, thought to be crucial in mountain forests (Mayer and Ott, 1991, p. 483 ff.; Ott et al., 1997), are rarely considered (but see Jorritsma et al., 1999, or Seagle and Liang, 2001). Predictions of how forest structure changes in response to herbivory may therefore be inaccurate. Such models, lacking a detailed representation of tree regeneration, might be insensitive to the impact of insufficient tree regeneration on the long-term protective effect of mountain forests.

The main objective of this study was to assess the efficacy of including a more detailed tree regeneration submodel within a forest patch model, toward the goal of conducting reasonable management scenarios for the protection function of a forest over a multi-decadal period. We first disaggregated the original establishment submodel included in the ForClim patch model (Bugmann, 1994, 1996) into a seedling establishment and a sapling growth module. After comparing the old and new model versions using data from a case study, we then applied the new model to give an example of its use for scenario modelling. The potential structural forest development and the long-term protective effect of a Swiss mountain forest were investigated under two alternative scenarios.

Methods

Stotzigwald case study

Study site

The Stotzigwald is a steep forest in the Swiss Alps (46°45’ N and 08°39’ E) with a mean slope of more than 40°, and some interspersed cliffs (Thali, 1997). It protects one of the most heavily used traffic routes in Switzerland against rockfall. The elevation of the protection forest ranges from 650 to approximately 1000 meters above sea level (m asl), but its protective function is mainly restricted to a rockfall zone of approximately 7.5 ha in the lower part of the forest. Rockfall is frequent in this zone, with the majority (79%) of canopy trees showing traces of recent damage. The forest within this zone consists mainly of *Picea abies* and *Abies alba*.

In the Stotzigwald, only minor timber harvesting took place from 1950 until 1998 (< 0.6 m^3 ha^{-1} a^{-1}; cf. Thali, 1997). Its current ungulate population, which consists of chamois (*Rupicapra rupicapra*), roe deer (*Capreolus capreolus*) and red deer (*Cervus elaphus*), is rather high since the site is partly located within a wildlife sanctuary (Odermatt, 1997). A dense forest resulting from a lack of recent silvicultural treatments, combined with a high ungulate population density, has resulted in a conspicuously low density of tree regeneration (Thali, 1997).

Stand description and current protective effect

The current stand structure of the Stotzigwald was surveyed from June to August 2003, to provide initial conditions for the simulation experiments. Based on a stratified sampling design in the lower part of the Stotzigwald (approx. 34 ha), stand and regeneration characteristics were sampled on approximately 100 concentric circular plots of 100 m^2 and 50 m^2 for overstory and regeneration data, respectively. To account for the steepness of the terrain, a slope correction factor was applied (Köhl and Brassel, 2001). For all trees ≥ 4 cm diameter at breast height (DBH) on the larger plots, DBH was measured and rockfall traces were counted. Tree height was measured for approximately one third of all trees. On the regeneration plots, all seedlings and saplings < 4 cm DBH were tallied by species in six different height classes (<0.10 m, 0.10-0.39 m, 0.40-0.69 m, 0.70-0.99 m, 1.00-1.3m, >1.3 m up to 3.9 cm DBH). For each individual seedling or sapling, browsing on lateral and terminal shoots was recorded. In addition, the light regime on each regeneration plot was recorded with five hemispherical photographs. Stand and regeneration characteristics were directly extrapolated from sampled plots within the rockfall zone to the entire rockfall zone.

The current structure of tree regeneration within the rockfall zone shows a rather low sapling density with approximately 2230 saplings ha^{-1}. *Abies alba* and *Picea abies* are most frequent, with 44% and 20%, respectively. Together with *Sorbus aucuparia* L. (6%) and *Betula pendula* Roth (14%) they account for 84% of the saplings. As can be seen from the current height distribution in Fig. 1a, most of the saplings (85 %) are smaller than 0.4 m. *Abies alba* saplings are particularly small, with 94% of the saplings <0.1 m, and 99% <0.4 m. A clear assessment of the browsing impact was not possible, due to the lack of taller individuals of palatable tree species (particularly *Abies alba*, but also *Sorbus aucuparia*). However,

qualitative observations indicate a high impact by ungulate browsers (Odermatt, 1997).

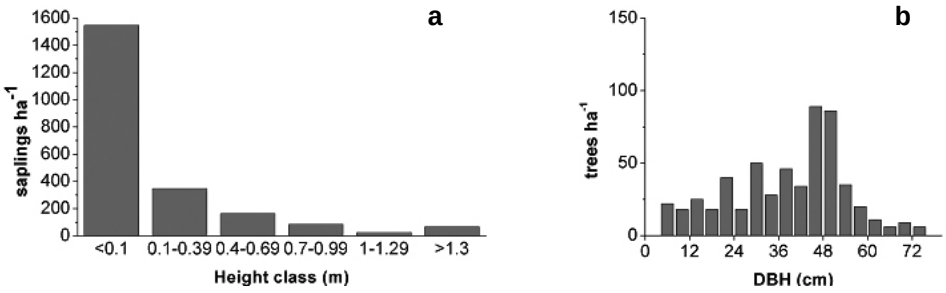

Figure 1. Current height distribution of tree regeneration (a) and diameter distribution of the stand (b) at the rockfall zone of the Stotzigwald.

The current stand includes approximately 561 trees ha^{-1} > 4 cm DBH, and is dominated by *Picea abies* (83%) and *Abies alba* (13%). The diameter distribution shows a maximum around 44 to 52 cm DBH with a rather low density of trees < 12 cm DBH (Fig. 1b).

Altogether, the present forest in the rockfall zone of the Stotzigwald exhibits a low intensity of tree regeneration between a height > 0.4 m up to a DBH of 12 cm. Nevertheless, it provides an almost optimal protective effect against rockfall when compared to the target values as defined in the revised version of the Swiss management guidelines for protection forests (Frehner et al., 2005). These guidelines define target stand structures (stem numbers, species composition) for different natural hazards and site conditions. As can be seen from Tab. 1, the recommended stem numbers for rockfall protection forests are reached in the current stand with the exception of the stem number for trees with DBH > 12 cm. Since rockfall events with smaller (i.e. < 40 cm diameter) and middle-sized rocks (40 – 60 cm diameter) currently are more frequent at the Stotzigwald (pers. observation), the low density of smaller trees is particularly troublesome. Given the sparse tree regeneration between a height > 0.4 m up to a DBH of 12 cm (see above), the protective effect of the forest may decrease in the long term.

Table 1. Comparison of the current stem number in the Stotzigwald study area with target values for the optimal stand structure against rockfall as defined in the Swiss management guidelines for protection forests (Frehner et al., 2005).

rock diameter (cm)	effective DBH (cm)	target values (trees ha^{-1} with effective DBH)	Stotzigwald (trees ha^{-1})
< 40	> 12	600	521
40 - 60	> 24	400	438
> 60	> 36	200	342

The ForClim model

The ForClim patch model was originally developed to assess the impacts of climatic changes on tree species composition and biomass of forests in the Swiss Alps (Bugmann, 1994). During its construction, special emphasis was placed on developing a model with a minimum number of ecological assumptions (Bugmann, 1996). The applicability of ForClim was successfully extended from the Swiss Alps to other climatic regions through several model refinements (Bugmann and Cramer, 1998, Bugmann and Solomon, 1995, Bugmann and Solomon, 2000, Bugmann, 2001a, Shao et al., 2001). A detailed description of ForClim can be found in Bugmann (1996), and the latest version of ForClim, V.2.9.3, is documented in Risch et al. (2005). Even though ForClim was not originally designed to simulate structural forest patterns such as diameter distributions, it has since been shown to accurately reproduce such patterns in simulations over periods of several decades (Risch et al., 2005, Wehrli et al., 2005).

However, Wehrli et al. (2005) reported that the stress-induced mortality implemented in ForClim strongly overestimated mature tree mortality rates in dense stands, leading to an unrealistically drastic reduction of simulated tree numbers in short-term simulations. This artefactual effect arose from an underestimation of the calculated light availability on a patch, which in the current model is a function of the leaf area index (LAI) of idealized trees, i.e. trees with long crowns. However, many mountain forests such as the Stotzigwald are even-aged and dense, and trees with short crowns predominate (Ott et al., 1997). For such forests, the LAI calculated for long-crowned trees is too high, and fails to account for the self-pruning of crowns.

For the present study, we implemented a simple dynamic crown structure to correct this problem. ForClim calculates LAI using a static, allometric relationship between foliage weight and DBH (Bugmann, 1996). We instead use a light-dependent, dynamic relationship, where the foliar mass and LAI of a tree with a given

DBH depend on the current light conditions on a patch, and can vary between a species-specific maximum and minimum. Our implementation of the new, dynamic crown structure was based on the original relationship between foliage fresh weight $gFolW$ and DBH from Bugmann (1994):

$$gFolW = kA_1 * dbh^{kA_2} \quad (1)$$

where kA_1 and kA_2 are species-specific parameters estimated from empirical data (cf. Bugmann 1994, p. 202). To make the relationship dynamic, we replaced the original parameter kA_1 that describes a single, fixed rate of change by a LAI-dependent parameter gkA_1 , which is calculated as follows:

$$gkA_1 = kA_{1,\max} - \Delta kA_1 * gkLAI \quad (2)$$

with

$$\Delta kA_1 = kA_{1,\max} - kA_{1,\min} \quad (3)$$

where $kA_{1,\,max}$ and $kA_{1,\,min}$ are the maximum and minimum envelope of the relationship between foliage fresh weight and DBH, respectively. The factor $gkLAI$ ranges from 0 (no canopy shading) to 1 (full canopy shading), and is calculated as follows:

$$gkLAI = \min\left(\frac{curLAI_{patch}}{\max LAI_{patch}}, 1\right) \quad (4)$$

where $curLAI_{patch}$ is the current LAI on the patch as calculated by ForClim and $\max LAI_{patch}$ is the maximal value for LAI on a patch. Thus, without canopy shading, i.e. without trees on a patch, gkA_1 is set to $kA_{1,\,max}$. Consequently, new trees on such a patch are initialized with a very high DBH-leaf area relationship. With increasing canopy shading, gkA_1 and thus the DBH-leaf area relationship decreases until it finally reaches $kA_{1,\,min}$, reflecting the maximum self-pruning of crowns. In contrast to the original, static crown structure, stress-induced mortality is assumed to initiate only if the DBH-leaf area relationship of a tree has already been reduced to the minimum. Following the death of one to several trees on a patch, the model does not allow gkA_1 of the remaining trees to increase in the following time steps, even though the available light on the patch increases.

While being simple and linear, we expected this approach to be promising, because it captures some salient features of the dynamics of tree crowns without undue reliance on data for parameterisation that are not readily available.

The values for $kA_{1,\,max}$ and $kA_{1,\,min}$ were estimated from species-specific tree data (Burger, 1947, 1948, 1950a, b, 1951, 1952, 1953) using quantile regression (Cade and Noon 2003, cf. Fig. 2, Tab. 2).

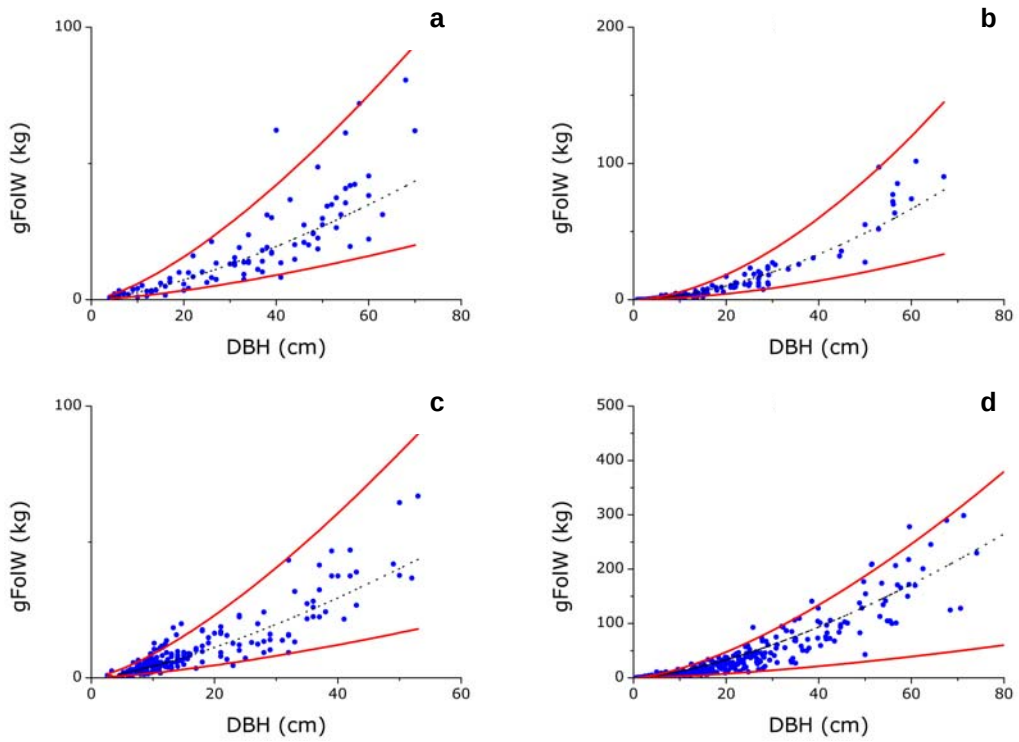

Figure 2. Relationship between foliage fresh weight (gFolW) and DBH for several species groups as derived from the data in Burger (1945-53). Dashed lines represent the original relationship included in ForClim, solid lines represent the minimum and maximum envelope based on a quantile regression (95%). Species grouping follows Bugmann (1994, p. 201ff). Group a: *Larix decidua*, group b: *Fagus silvatica*, *Quercus spp*., group c: *Pinus silvestris*, group d: *Abies alba, Picea abies, Pinus cembra, Pinus montana.*

The accuracy of the new dynamic crown implementation was corroborated by comparing simulated values of available light at the forest floor with values derived from hemispherical pictures from the study site. The data sets were compared graphically by plotting observed (y) versus simulated (x) data directly, with the line of perfect fit (y=x) marked, as suggested by Mayer and Butler (1993).

Table 2. Parameter estimations for the relationship between foliage fresh weight (gFolW) and DBH for several species groups as derived from the data in Burger (1945-53). $kA_{1,\ orig}$ denotes the original estimation of kA_1 as described by Bugmann (1994, p. 202), $kA_{1,\ min}$ and $kA_{1,\ max}$ represent the lower and upper boundary of the quantile regression. n denotes sample size.

Species group		$kA_{1,\ orig}$	$kA_{1,\ min}$	$kA_{1,\ max}$	kA_2	n
a	*Larix decidua*	*0.1*	*0.048*	*0.221*	*1.4*	*99*
b	*Fagus silvatica, Quercus spp.*	*0.06*	*0.025*	*0.105*	*1.7*	*144*
c	*Pinus silvestris*	*0.17*	*0.071*	*0.346*	*1.4*	*210*
d	*Picea abies, Abies alba*	*0.23*	*0.090*	*0.530*	*1.5*	*355*

In addition, a linear regression analysis was performed and the modelling efficiency statistic (EF; cf. Mayer and Butler, 1993) was calculated. The latter is very similar to the commonly used R^2 statistic, but indicates the proportion of variation explained by the line of perfect fit instead of a fitted regression line. As can be seen in Fig. 3, the fit between empirical and simulated values is only moderate. Moreover, there is a great deal of „noise“ as evidenced by a low EF statistic (EF = 0.11). However, the fit is much higher compared to the one with the old crown implementation (EF = -3.28). Thus, considering the simplicity of the approach and the many factors influencing real light availability at the forest floor that are neglected in the model (e.g., microtopography), we consider our model refinement to be a substantial improvement.

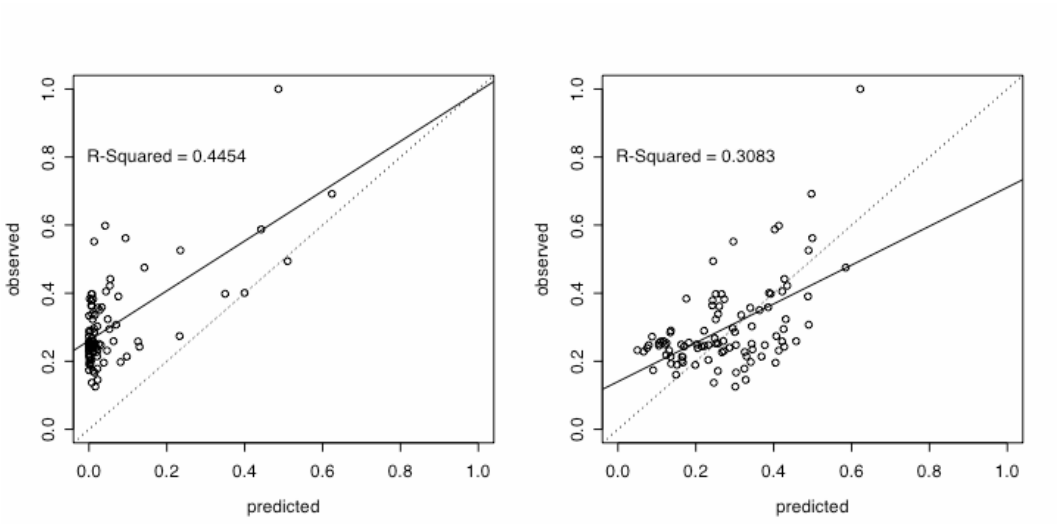

Figure 3. Corroboration of the new crown structure for the Stotzigwald site, from where hemispherical pictures and data on stand structure were available from 96 field plots. Observed refers to the available light at forest floor derived from hemispherical pictures. Predicted denotes the available light at forest floor as simulated by ForClim with the old (left) and the new (right) crown implementation. The diagonal line represents the line of perfect fit (y=x). Also shown is the regression line for observed vs. predicted values of the proportion of available light at forest floor.

To test the influence of the dynamic crown structure on the simulated tree mortality rates, model comparisons were conducted for the Stotzigwald and for another site from which empirical data were available from a preceding study (site Sigriswil, cf. Wehrli et al., 2005). The results of these test runs showed that the mortality rates obtained with the dynamic crown structure were significantly lower than the rates obtained with ForClim V2.9.3, and very similar to the empirical mortality rates at the Sigriswil site (Tab. 3). Therefore, subsequent simulation experiments utilize the modified, dynamic relationship between foliar mass and DBH.

Table 3. Mortality rates simulated with ForClim V2.9.3 (FC 2.9.3) and ForClim V2.9.4 (FC 2.9.4) for the sites Stotzigwald and Sigriswil (cf. Wehrli et al., 2005 for a detailed description of the Sigriswil site). For Sigriswil, additional empirical data were available allowing a comparison with real mortality rates. Since for simulations of real stands, the overestimation by the stress-induced mortality is highest in the first years (cf. Wehrli et al., 2005, p. 157), the test runs were only performed over short periods. na: no data available.

	Stotzigwald			Sigriswil		
	2005 (trees ha^{-1})	2015 (trees ha^{-1})	mortality rate	1930 (trees ha^{-1})	1943 (trees ha^{-1})	mortality rate
FC 2.9.3	561	231	0.59	425	220	0.48
FC 2.9.4	561	511	0.09	425	405	0.05
Empirical data	561	na	na	425	397	0.07

Improvement of the establishment submodel

In ForClim V2.9.3, tree establishment is determined by four limiting factors: light availability at the forest floor, browsing intensity, soil moisture and absolute winter minimum temperature. The response to these factors is species-specific, and tree regeneration for a given patch is modelled by applying these limiting factors as environmental “filters” to determine the establishment rate of trees with an initial DBH of 1.27 cm (cf. Bugmann, 1994, 1996, Price et al., 2001). This immediate establishment of trees with a DBH of 1.27 cm may be an over-simplification for mountain forests, where sapling growth is known to be very slow. In subalpine forests, for example, it can take up to 50 years for *Picea abies* to reach breast height (Brang and Duc, 2002, Ott et al., 1997). Neglecting delayed regeneration periods in the models may lead to overestimation of the abundance of small trees. A realistic assessment of the long-term protective effect of a forest such as the Stotzigwald may therefore not be achievable using a standard forest patch model such as ForClim V2.9.3. Therefore, we decided to more explicitly model the tree establishment process in our ForClim model refinement.

To maintain ForClim’s overall approach as a model with a minimum number of ecological assumptions, we decided to implement a simple sapling growth function that permits the explicit simulation of the regeneration time lag, rather than to develop a mechanistic regeneration model, which could prove difficult to parameterise. Following Price et al. (2001), we first disaggregated the regeneration process into two stages, resulting in two new modules, one for seedling establishment and one for sapling growth. In the seedling establishment module, seedlings establish under optimal conditions with an initial height of 0.05 m. They are then transferred to the sapling growth module, in which they grow to a height of 2 m before recruiting as

trees with an initial DBH of 1.27 cm. In the sapling growth module, individual saplings are modelled explicitly by taking into account suboptimal conditions due to canopy shading and ungulate browsing (see below). By doing so, the new model version allows us to take into account two major “delaying factors” of the long regeneration period of mountain forests. Thus, with the modified model, it should be possible to more accurately estimate the long-term protective effect of a mountain forest. The structure of the new establishment submodel of ForClim V2.9.4 is shown in Fig. 4. The two new modules are described in detail below.

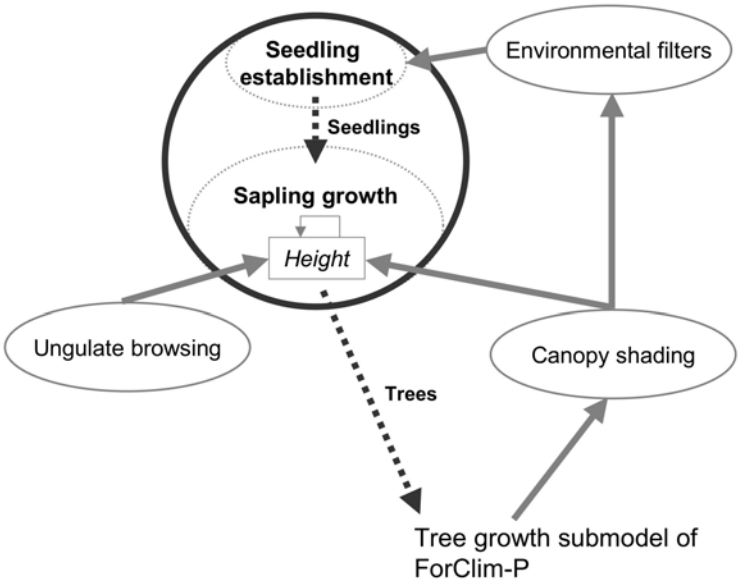

Figure 4. Structure of the new establishment submodel in ForClim V2.9.4. The square box denotes the state variable (height increment); ellipsoid boxes represent the different factors that influence the regeneration process.

Seedling establishment module

Seedling establishment in ForClim V2.9.4 is based on the original tree establishment submodel in ForClim V2.9.3 as described by Bugmann (1994, p. 58-61), i.e. the original limiting factors are used to determine the establishment of seedlings. Even if this approach has been used in many patch models for the establishment of trees, we suspected that further investigations, e.g. on the population dynamics of seedlings under different canopy structures would be

necessary before this module could be used in a study such as the present one (cf. Price et al. 2001). Since data for a reliable parameterisation and corroboration were not available, the seedling establishment module was not used for the present study. Thus, only the current tree regeneration at the Stotzigwald site was considered in the simulations, i.e. the number and height of measured saplings was directly imported into the sapling growth module. As evident from a comparison of preliminary simulation runs conducted without and with additional seedling establishment, the omission of additional seedling establishment did not have a significant influence on the stand structure after 60 years, i.e. the stand structure after 60 years did not depend on additional establishment of seedlings. This is not surprising since tree growth is slow in high-altitudes environments, and seedlings require several decades before they markedly influence stand structure (see above).

Sapling growth module

Similar to the tree growth module of ForClim, individual sapling growth is calculated using an optimal growth potential derived from empirical data, which is then reduced by limiting environmental constraints. Sapling height growth is modelled instead of diameter growth, since the former is ecologically more relevant and more sensitive to environmental constraints. Therefore, we introduced a new height growth function based on the growth equation of Bertalanffy (Bertalanffy, 1957, Rammig et al., 2005). The appeal of this equation lies in its simplicity and biological plausibility (Zeide, 1993, Pretzsch, 2001). The equation includes only two parameters, one for the asymptote (i.e. maximum tree height) and one for the slope (i.e. growth rate). Thus, the function is less flexible than equations with more parameters, but at the same time, it is more robust in parameter estimation procedures (Pretzsch, 2001). The optimum species-specific height increase $h_{inc,\,opt}$ of a sapling is calculated as:

$$h_{inc,opt} = 3 * H_{\max} * kG_{Sap} * e^{(-kG_{Sap} * age_{Sap})} * (1 - e^{(-kG_{Sap} * age_{Sap})})^2 \qquad (5)$$

where H_{max} is maximum tree height in cm, and kG_{sap} is the species-specific growth rate. These two parameters are estimated from empirical data. The *sapling age parameter* is calculated as:

$$age_{Sap} = Ln\left(1 - \left(h_{curr} / H_{\max}\right)^{(1/3)}\right) / (-kG_{Sap}) \qquad (6)$$

where h_{curr} is the current sapling height.

Parameterisation of the growth function and corroboration of the parameter estimations

The two species-specific parameters needed for the Bertalanffy equation were estimated for the four main tree species at the study site, i.e. *Picea abies*, *Abies alba*, *Betula pendula* and *Sorbus aucuparia*. Maximum tree height H_{max} was parameterised using data collected at the Stotzigwald site. While keeping H_{max} constant, the growth rate kG_{sap} was estimated using data from the literature (Commarmot, 1995, Schönenberger, 2002). The Bertalanffy equation was fit to data from similar sites where saplings had grown under optimal conditions, i.e. no canopy shading and no browsing, using nonlinear curve fitting (cf. Rammig et al., 2005). In this way, the estimated growth factors were intended to reflect a site-specific optimum height growth, but not the absolute maximum height growth. Estimates of the species-specific parameters are shown in Tab. 4.

Table 4. Model parameters for the sapling growth function. *Hmax* denotes the maximum tree height and was derived from field data from the Stotzigwald. *kGSap* stands for the sapling growth parameter, which was estimated for each species from empirical data with a nonlinear curvefit based on data from Schönenberger (2002, site Schwanden) and from Commarmot (1995, site Bourrignon). t_{200} indicates the time a sapling needs to pass the threshold of the sapling growth submodel of 200 cm.

	H_{max} (cm)	kG_{Sap}	Std. Error	n	t_{200} (years)	Data source
Abies alba	3500	0.017	0.0002	3354	24	Commarmot 1995
Picea abies	3500	0.020	0.002	53	21	Schönenberger 2002
Betula pendula	2500	0.090	0.023	13	7	Schönenberger 2002
Sorbus aucuparia	1500	0.055	0.004	320	13	Schönenberger 2002

Note: The parameter estimation for *Betula pendula* shows a relatively large standard error. However, since a variation of kG_{Sap} ± Std. Error has only a marginal influence on the t_{200} (± 1 year), the value of kG_{Sap} was set to 0.090.

By comparing calculated sapling height to field data from our study site, the estimated values of kG_{sap} were corroborated. For that purpose, only those saplings were used that had grown under near-optimal conditions at the study site (i.e. neither canopy shading nor browsing). Unfortunately, data for corroboration were rather sparse due to the low level of tree regeneration at the study site. Therefore, additional data from a nearby site were used for *Sorbus aucuparia* (Truninger and Bucher, 1994). For the other species, no additional data were available. The data sets were compared graphically and by means of the EF statistic and a linear regression (Mayer and Butler, 1993; cf. above). As can be seen from Fig. 5, the fit between observed and simulated sapling height is reasonably good. This is

confirmed by the EF statistics: While being only moderate for *Picea abies* (EF = 0.14), it is high for the other species (*Abies alba*: 0.86, *Sorbus aucuparia*: 0.87, *Betula pendula*: 0.89).

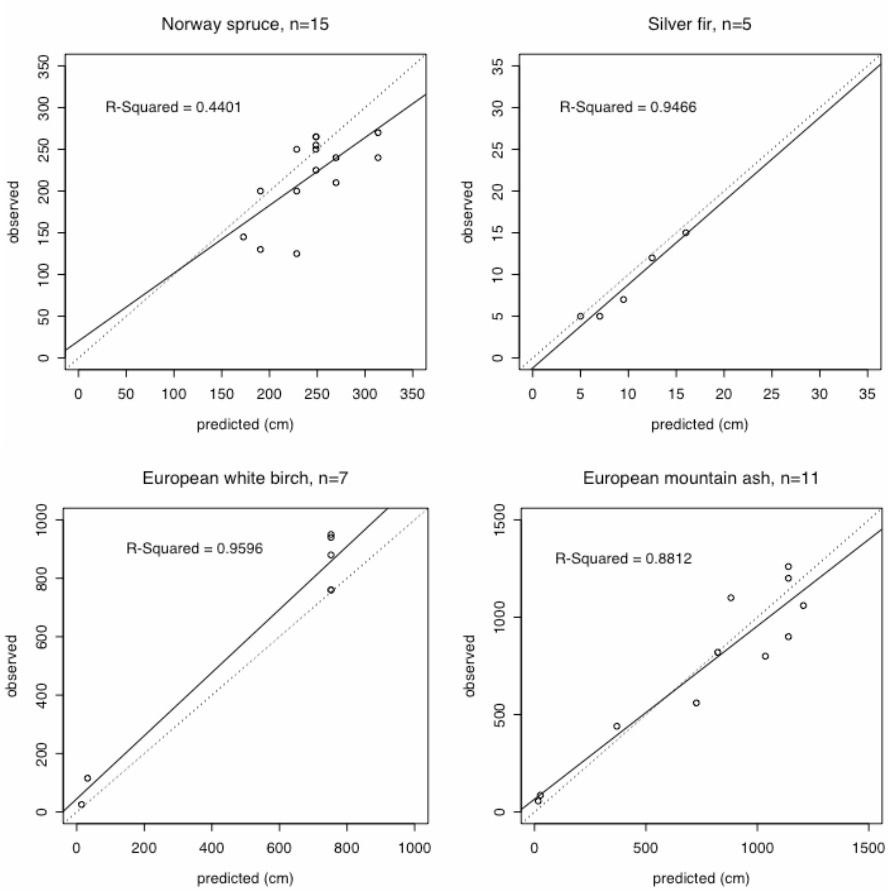

Figure 5: Corroboration of the sapling growth function for the Stotzigwald site. Observed refers to sapling heights from field data (*Picea abies [Norway spruce]*, *Abies alba [silver fir]*, *Betula pendula [European white birch]*) and to data from a nearby site (*Sorbus aucuparia [European mountain ash]*; cf. Truninger and Bucher, 1994), respectively. Predicted denotes the sapling height simulated by ForClim V2.9.4. n denotes sample size.

Since corroborative data for *Abies alba* and *Betula pendula* were particularly sparse, the parameter estimates for these species were additionally compared with literature data. For *Abies alba*, several authors (e.g., Ammer, 1996b, Schütz, 1969) report a similar sapling growth rate as for *Picea abies*, which is in agreement with our parameter estimation resulting in very close grow-out times for the two species (t_{200}, see Tab. 4). For *Betula pendula*, the parameterised growth function shows a similar trajectory during the early growth period as that used by Kupferschmid Albisetti (2003), which is based on empirical data from a similar site.

Growth reduction by environmental constraints

In the model, maximum height growth is reduced by a species-specific response to canopy shading and ungulate browsing. Sapling growth thus responds dynamically to the prevailing environmental conditions. The calculation of canopy shading is calculated as available light on the forest floor using Beer's extinction law as a function of leaf area index (Bugmann 1964, p. 63).

Simulated ungulate browsing affects saplings throughout their growth period, from 0.05 m to 2.0 m in height, taking into account the different ungulate species that occur in the Stotzigwald. Since more palatable species are subject to a higher browsing impact, a species-specific factor *kBrS* was introduced that reflects the known species-specific susceptibility to ungulate browsing (e.g., Ammer, 1996a, Kindermann and Hasenauer, 2003, Motta, 2003, Senn and Suter, 2003). The values for *kBrS* for the four main tree species at the Stotzigwald were derived from the literature (Ammer, 1996a, Brändli, 1996, Brassel and Brändli, 1999, Kindermann and Hasenauer, 2003, Motta, 1996, 2003, Rüegg and Schwitter, 2002; Tab. 5).

Table 5. Species-specific susceptibility to browsing according to various authors. For standardisation, all values are scaled to the range [0....1], where 0 = least susceptible to browsers and 1 = most susceptible to browsers. Mean susceptibility relates to the averaged susceptibility as derived from the different sources. kBrS denotes the parameter included in ForClim V 2.9.4. na: no data available.

species	Ammer 1996a $	Brändli 1996 *	Brassel & Brändli 1999 $	Kindermann & Hasenauer 2003 $	Motta 1996°	Motta 2003 §	Rüegg & Schwitter 2002 *	Mean susceptibility	**kBrS**
Picea abies	0.35	0.11	0.12	0.27	0.60	0.34	0.15	0.23	**0.25**
Abies alba	1	0.6	1	0.55	1	na	1	0.76	**0.75**
Betula pendula	na	0.38	na	0.64	na	na	na	0.51	**0.50**
Sorbus aucuparia	na	1	na	1	na	1	0.88	1	**1**

dimension for measuring species-specific susceptibility: * browsing intensity; ° species-specific damage; § browsing incident in terminal shoot
$ percentage browsed (Verbissprozent)

We also introduced a site-specific browsing intensity factor (*kBrI*), scaled from 0 (no browsing) to 4 (very high browsing for the least sensitive species, i.e. *Picea abies*). The species-specific browsing impact is calculated by multiplying *kBrS* with *kBrI*.

Data on the long-term development of tree regeneration in mountain forests under different shading and browsing regimes are rare. Therefore, we calculated the integrated response of sapling growth to these factors using simulation experiments with the mechanistic tree regeneration model HUNGER (Weisberg et al., 2005). HUNGER simulates sapling growth and the response to light availability and ungulate

browsing based on a well-established formulation for plant nutrient transport and conversion processes (Weisberg et al., 2005). It was calibrated with empirical data from mountain forests of eastern Switzerland. In a test with independent data, HUNGER was applied to *Picea abies*, and it accurately represented sapling response to browsing intensity and relative light availability. It slightly over-predicted sapling height, while no significant differences were found between simulated and observed basal diameter, total biomass or leaf biomass (Weisberg et al., 2005). We used HUNGER to derive a response surface for *Picea abies* saplings with respect to the browsing-light interaction. From this surface, a simplified response function was derived by approximating and discretising the values obtained by HUNGER (cf. Fig. 6).

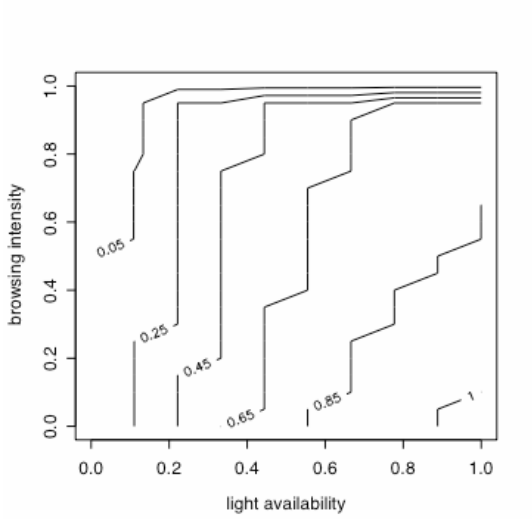

Figure 6. Contour plot of the browsing-light interaction. Browsing intensity ranges from 0.0 to 1.0 and refers to the annual probability of a sapling to be browsed. The light availability similarly ranges from 0.0 to 1.0 and refers to the proportion of light that is available to the sapling layer. The labeled contour lines represent the reduction factors included in the sapling growth module (see text).

This response function was used in the sapling growth module as a multiplier to reduce optimum species-specific height increase $h_{inc,\ opt}$ to yield current height increase h_{inc} as follows:

$$h_{inc} = h_{inc,opt} * f(br,l) \tag{7}$$

where $f(br,l)$ denotes the response function (Fig. 6).

Due to a lack of data, HUNGER could not be applied for the other species. Therefore, the response function for growth rate reduction due to browsing and shading was assumed to be equal for all species.

Mortality in the new establishment submodel

A solid theoretical framework for tree mortality remains elusive (Shugart, 1998), and therefore mortality algorithms have been mostly limited to general relationships (Keane et al., 2001). This is especially true for the mortality of seedlings and saplings. Many patch models therefore include seedling mortality only implicitly, by reducing the species-specific establishment probability (e.g., Bugmann, 1994, p. 61). While it is true that mortality in the seedling stage accounts for the largest loss of tree regeneration (Brang, 1998, Hillgarter, 1971, Imbeck and Ott, 1987, Stern, 1972), this approach neglects additional mortality at later juvenile stages (e.g., saplings). Evidence from the literature suggests that sapling mortality is influenced by various environmental constraints such as ungulate browsing (Ammer, 1996a, Eiberle and Nigg, 1983, Eiberle, 1989).

Our refined establishment submodel therefore includes a more explicit algorithm for sapling mortality, where mortality rate is based on the widely used approach of growth-dependent mortality (cf. Keane et al., 2001). Thereby, the probability of growth-dependent mortality is increased any time the response function $f(br,l)$ (see eq. 7) falls below a threshold value. Since no data for parameterisation were available, we used values from the literature: The probability of mortality was set to 0.368 for saplings that did not reach 10% of their optimum growth in any given year (e.g., Bugmann, 1994; cf. Shugart, 1984, Keane et al., 2001).

The new sapling mortality module was tested extensively, and test runs with and without the mortality module were compared. These comparisons, however, revealed no significant differences in stand structure after 60 years. This is probably due to the relatively short simulation period during which saplings with strongly reduced growth are not able to recruit as trees anyway, no matter if they just grow slowly or if they die after a certain period of slow growth. Therefore, the inclusion of a sapling mortality module for saplings does not seem to be mandatory for relatively short simulation periods such as those upon we are focusing in this study. Moreover, long-term data sets where the survival of individual saplings has been tracked under variable growing conditions are rare. Therefore, the effects of environmental constraints on sapling mortality are difficult to determine. This caused us to omit the sapling mortality module for the present study. In this way, the model is kept simple and consistent, as suggested by Keane et al. (2001).

Comparison of the model versions and preliminary application of the improved model

In order to compare the old and the new establishment submodel of ForClim, we first simulated the potential development of the stand for 60 years under current climatic conditions with both model versions. The simulation experiment with the new model version was performed using maximum sapling growth, considering neither canopy shading, nor browsing constraints. Although this scenario is rather hypothetical for the present stand at the Stotzigwald, it allows evaluating the differences in behaviour between the two model versions.

As a first application of the new model, we then investigated the effect of sparse tree regeneration on the long-term protective effect of the Stotzigwald. This was done by simulating the potential development of the stand for 60 years under current climate conditions, given two scenarios. The first scenario "*canopy shading and no browsing impact" (CS-scenario)* only considered canopy shading, by setting browsing intensity to zero. The second scenario *"canopy shading and high browsing impact" (CSBI-scenario)* considered both factors, using empirical data for browsing intensity at the Stotzigwald (Odermatt, 1997). Relative browsing intensity (*kBrI*) was set to 1, resulting in a species-specific browsing impact of 1 for *Sorbus aucuparia*, 0.75 for *Abies alba*, 0.5 for *Betula pendula* and 0.25 for *Picea abies*.

Initialisation of ForClim with stand and weather data

In ForClim, simulations are performed for small patches that are mutually independent (i.e. no horizontal spatial effects). Therefore, model initialisation with our field data (see section 2.1) was necessary at the scale of individual patches. ForClim patch size was thereby set to 225 m^2 (15 m x 15 m), and stand and regeneration data were extrapolated from single plots to this patch size, assuming a homogenous spatial distribution of trees and regeneration. To reduce stochastic "noise" in the results, simulation experiments were performed with numerous repetitions (n = 250 patches, cf. Bugmann, 1996, Pretzsch and Dursky, 2001).

The input for the weather generator in the abiotic environment submodel of ForClim was derived from monthly precipitation sums and mean temperatures from the weather station at Gurtnellen (739 m asl), located approximately 2.5 km from the Stotzigwald.

Results

Comparison of model versions

Simulating the structural development of the Stotzigwald with ForClim V2.9.3, i.e. without the improved establishment submodel, yielded a high rate of recruitment of all four species after 20 years (Fig. 7a), resulting in an overall tree density of 1874 trees ha^{-1} (Tab. 6). Since no species-specific time lag is considered in ForClim V2.9.3, the species composition after 20 years was strongly influenced by the current species composition of tree regeneration at the Stotzigwald (see above). This resulted in a large increase in the relative proportion of *Abies alba*, *Betula pendula* and *Sorbus aucuparia* (Tab. 6), leading to co-domination by *Abies alba* and *Picea abies*. After 40 years, the overall tree density decreased to 1435 trees ha^{-1} (Tab. 6), which was due to the ongoing competition between small trees, resulting in a high stress-induced mortality. The density of shade-intolerant species decreased faster than that of the shade-tolerant *Abies alba* (Tab. 6). Still, the frequency histogram for tree diameter showed a strong peak at 4-16 cm DBH (Fig. 7b). Twenty years later, the overall tree density had decreased again, and the peak of young trees was found at a higher DBH (8-20 cm DBH, Fig. 7c). Thus, after 60 years most of the regeneration had already reached pole stage. Compared to the situation after 20 years, the species composition remained relatively constant, dominated jointly by *Picea abies* and *Abies alba* (Tab. 6).

The simulation experiment with the new model version and maximum sapling growth yielded a strong recruitment of *Sorbus aucuparia* and *Betula pendula* after 20 years (Fig. 7d), which altered forest composition and resulted in an overall tree density of 725 trees ha^{-1} (Tab. 6).

After 40 years, the density of small trees increased to a higher level (Fig. 7e), leading to an overall tree density of 1429 ha^{-1} (Tab. 6). Thus, tree density almost doubled between years 20 and 40, primarily due to abundant *Abies alba* recruitment, whereas *Sorbus aucuparia* and *Betula pendula* densities had decreased (Tab. 6). Overall tree density finally decreased to 1241 ha^{-1} after 60 years (Tab. 6), with many young trees of 4-16 cm DBH (Fig. 7f). Again, as for the experiment with ForClim V2.9.3, the decrease after 60 years was mainly due to density-dependent self-thinning, represented in the model as an increase in stress-induced mortality.

Table 6. Overall tree density (> 4 cm DBH) and species composition of the potential stand after Δt = 20, 40 and 60 years. Simulations were conducted with ForClim V2.9.3 and ForClim V2.9.4, respectively.

	Stotzigwald current stand	*ForClim V 2.9.3*			*ForClim V 2.9.4 (maximum sapling growth)*		
		Δt = 20 years	Δt = 40 years	Δt = 60 years	Δt = 20 years	Δt = 40 years	Δt = 60 years
Overall tree density (Trees > 4 cm DBH ha^{-1})	561	1874	1435	1033	725	1429	1241
Picea abies	466 (83%)	753 (40%)	589 (41%)	478 (46%)	405 (56%)	655 (46%)	541 (44%)
Abies alba	73 (13%)	766 (41%)	679 (47%)	470 (46%)	68 (9%)	605 (42%)	632 (51%)
Betula pendula	7 (1%)	104 (6%)	60 (4%)	35 (3%)	98 (14%)	60 (4%)	28 (2%)
Sorbus aucuparia	0 (0%)	242 (13%)	100 (7%)	45 (4%)	142 (20%)	101 (7%)	35 (3%)
others	15 (3%)	7 (0%)	5 (0%)	4 (0%)	12 (1%)	8 (1%)	5 (0%)

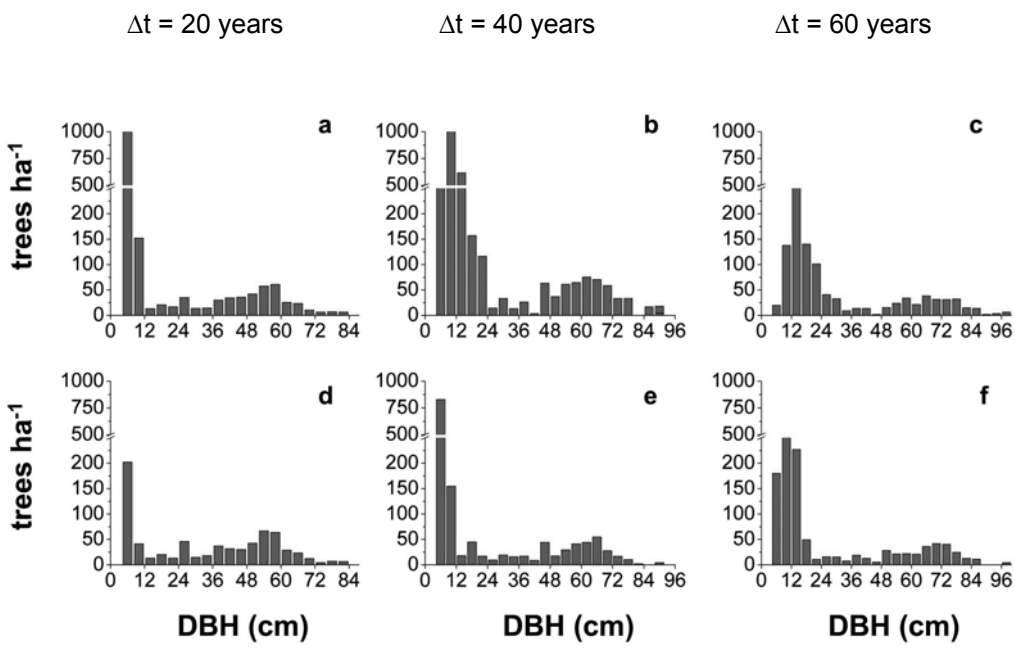

Figure 7. Simulated diameter distribution of trees > 4 cm DBH at the Stotzigwald site after Δt = 20 (a, d), 40 (b, e) and 60 (c, f) years. Simulations were conducted with ForClim V2.9.3 (a, b, c) and with ForClim V2.9.4 (d, e, f), respectively.

Preliminary application of the improved model

Stand dynamics

The simulation runs conducted with ForClim V2.9.4 under the CS-scenario (*"canopy shading and no browsing impact"*) yielded a gradual recruitment of small trees. As is evident from Figs. 8a-c, the density of small trees reached a high level only after 60 years. Overall tree density increased moderately from a density of 619 trees > 4 cm DBH ha^{-1} after 20 years to 755 and 944 after 40 and 60 years, respectively (Tab. 7). The increase after 20 years was mainly due to the recruitment of broad-leaved trees (*Sorbus aucuparia* and *Betula pendula*), whereas after 40 years it can be attributed to *Sorbus aucuparia*, *Picea abies* and particularly *Abies alba* (Tab. 7). The final increase after 60 years was due to a massive recruitment of *Abies alba*, which considerably changed species composition (Tab. 7).

The introduction of browsing impacts (*"canopy shading and high browsing impact",* CSBI-scenario) yielded an even slower recruitment of small trees (< 12 cm DBH; Fig. 8d-f). Overall tree density dropped below the initial density of 561 trees ha^{-1}, and exceeded the initial density only after 60 years, due to a substantial recruitment of *Abies alba* (Tab. 7). Compared to the CS-scenario, *Abies alba* recruitment was considerably lower and delayed by 20 years, as can be expected under conditions of high browsing pressure. The highly palatable *Sorbus aucuparia* was prevented from reaching the canopy layer.

Table 7. Overall tree density (> 4 cm DBH) and species composition of the potential stand under the two scenarios after Δt = 20, 40 and 60 years, respectively. Simulations were conducted with ForClim V2.9.4.

	Stotzigwald	*Canopy shading and no browsing*			*Canopy shading and high browsing*		
	current stand	Δt = 20 years	Δt = 40 years	Δt = 60 years	Δt = 20 years	Δt = 40 years	Δt = 60 years
Overall tree density (Trees > 4 cm DBH ha^{-1})	561	619	755	944	538	542	672
Picea abies	466 (83%)	388 (62%)	431 (57%)	485 (51%)	383 (71%)	398 (73%)	436 (65%)
Abies alba	73 (13%)	65 (11%)	142 (19%)	373 (39%)	61 (11%)	64 (12%)	187 (28%)
Betula pendula	7 (1%)	84 (14%)	53 (7%)	33 (4%)	81 (15%)	69 (13%)	43 (6%)
Sorbus aucuparia	0 (0%)	70 (11%)	120 (16%)	44 (5%)	0 (0%)	0 (0%)	0 (0%)
others	15 (3%)	13 (2%)	9 (1%)	8 (1%)	13 (3%)	11 (2%)	6 (1%)

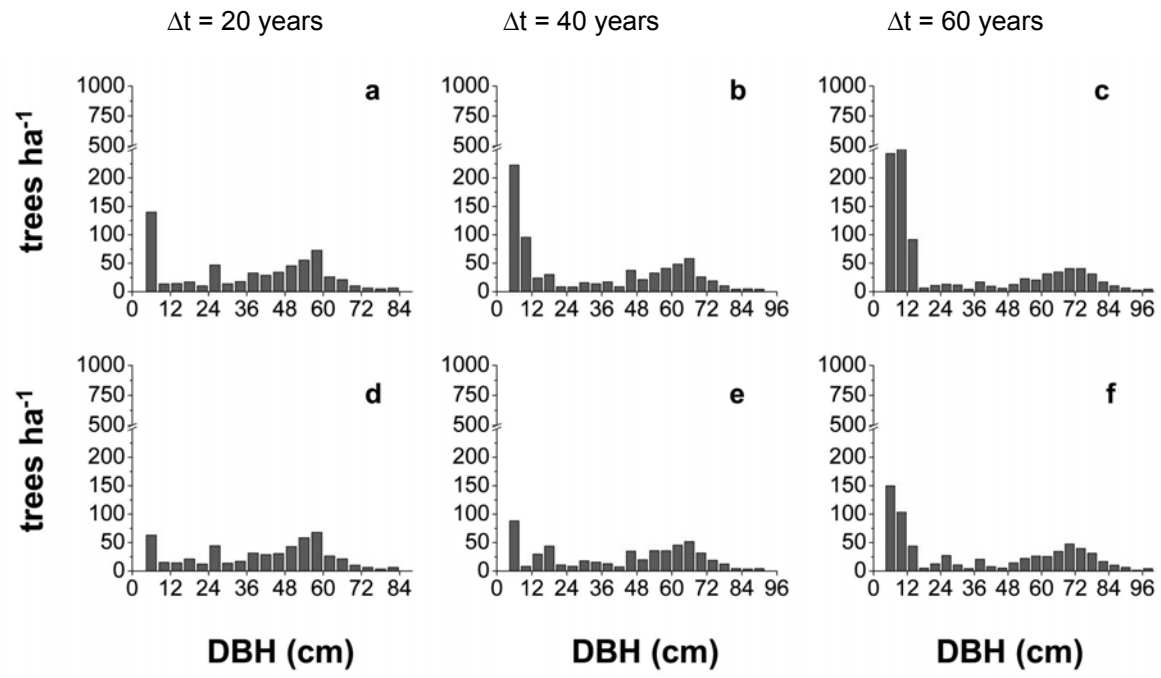

Fig. 8. Simulated diameter distribution of trees > 4 cm DBH at the Stotzigwald site after Δt = 20 (a, d), 40 (b, e) and 60 (c, f) years. Simulations were conducted with ForClim V2.9.4 under the scenarios *"canopy shading and no browsing"* (a, b, c) and *"canopy shading and high browsing"* (d, e, f), respectively.

Protective effect

As indicated by the simulated stem densities across size classes, the protective effect in the CS-scenario was equal to the initial protective effect of the Stotzigwald during the first 20 years (Tab. 8). After 40 years, the density of trees > 24 cm DBH fell below the target value (Tab. 8). Even though the target value for trees > 12 cm DBH was not reached after 60 years, the density of 478 trees > 12 cm DBH ha^{-1} at that time could indicate buoyancy, in that the target value might be reached in the coming decades.

Table 8. Comparison of target values for effective protection against rockfall with the potential stand structure under the two scenarios after Δt = 20, 40 and 60 years, respectively. Simulations were conducted with ForClim V2.9.4.

		Stotzigwald	*Canopy shading and no browsing*			*Canopy shading and high browsing*		
effective DBH	target values	current stand	Δt = 20 years	Δt = 40 years	Δt = 60 years	Δt = 20 years	Δt = 40 years	Δt = 60 years
trees ha^{-1} >12 cm	600	521	467	441	478	460	446	443
trees ha^{-1} >24 cm	400	438	424	373	341	415	361	357
trees ha^{-1} >36 cm	200	342	345	335	310	335	319	316

The protective effect in the CSBI-scenario was equal to that of the first scenario after the first 40 years. After 60 years, however, the density of trees > 12 cm DBH did not show an upward tendency, whereas the density of trees > 24 cm DBH stayed at a low level (Tab. 8).

Discussion

The main objective of this study was to improve the establishment submodel of the ForClim forest patch model to better understand the impacts of structural dynamics on the long-term protective effect of mountain forests.

We then applied our refined ForClim model (V2.9.4) to a preliminary investigation of the impact of low levels of tree regeneration on the long-term protective effect of one particular mountain forest, which has a stand structure that is typical of many Swiss mountain forests dominated by *Picea abies* (see Brändli and Herold, 1999).

Comparison of model versions

The simulation with both model versions suggests a rather optimistic stand development with a large recruitment of young trees during the first 40 years (Fig. 7). This is particularly true for the simulations conducted with ForClim V2.9.3, which yielded a very high tree density > 4 cm DBH after 20 years (Tab. 6). This extreme increase is due to fact that the initially present saplings of all species directly establish as young trees with an initial DBH of 1.27 cm after the first year in ForClim V2.9.3.

In Forclim V2.9.4, however, saplings first grow to a height of 2 m before establishing as trees. This is the reason why even under unrealistic maximum sapling growth, as used in this comparison, the recruitment of young trees was significantly lower and delayed compared to the simulations with ForClim V2.9.3.

The integration of the new establishment submodel had two additional effects on the simulated stand development. First, the species composition, and in particular the proportion of the two dominant species (*Picea abies* and *Abies alba*) changed more slowly than in the simulations with the old model version (cf. Tab. 6). Second, the onset of density-dependent self-thinning was significantly delayed (Tab. 6).

The most important aspect for the assessment of a forest's long-term protective function is, however, the delayed regeneration period. Whereas after 60 years in the

simulations with ForClim V2.9.3 most of the regeneration had already reached the pole stage, a large proportion of the young trees was still < 12 cm DBH at this time when simulated with ForClim V2.9.4 and maximum sapling growth. Thus, by simulating the stand without the new establishment submodel, the impact of a sparse level of tree regeneration on a forest's long-term protective function is likely to remain undetected.

Preliminary application of the improved model

Evaluation of stand dynamics

Stand development was quite gradual under both scenarios, as evidenced by the moderate increase of overall tree density (Tab. 7). This is due to the combination of a high mortality rate and a modest recruitment rate.

During the first 20 years in both scenarios, the number of trees decreased drastically (Tab. 7). This was especially true for densities of *Picea abies*, which decreased by 17-18% and 13-15% for the > 4 cm and > 8 cm DBH classes, respectively. However, given the current size distribution at the Stotzigwald site (Fig. 1b) with its maximum around 44 to 52 cm DBH, the simulated mortality rate is unlikely to be greatly overestimated. For the three sites reported in Wehrli et al. (2005), the empirical natural mortality rates for trees > 8 cm DBH during a period of 20 years were between 10-12%, with peaks up to 21%. Thus, our simulated mortality rates are comparable with values observed in other mountain forests of the region.

In contrast to the mortality rates, the recruitment rates of young trees differed greatly between the two scenarios. The high browsing level included in the CSBI-scenario led to a considerable delay of the regeneration period. *Sorbus aucuparia* was prevented completely from growing into the canopy layer, and the establishment of other species was prolonged. These findings are in agreement with the literature, which reports a delayed regeneration period (Eiberle, 1975, Guler, 2004) and eventual loss of overstory species (Ammer, 1996a, Guler, 2004, Wasem and Senn, 2000) as a consequence of severe ungulate browsing. Such delayed regeneration periods can be expected to have a severe impact on the long-term protective effect of a mountain forest.

Evaluation of the protective effect

The results of both scenarios CS and CSBI projected a similar decrease of the protective effect during the first 40 years, with densities of trees > 12 cm DBH and > 24 cm DBH falling below the target values (cf. Tab. 8). After 60 years, the density of trees > 12 cm DBH increased only slightly (CS-scenario) or even decreased (CSBI-scenario; Tab. 8). Thus, in the absence of browsing, the projected decline in the protective effect of the forest may be only temporary, and the high density of trees < 12 cm DBH after 60 years (Fig. 8c) raises hope for a recovery.

Given the simulated stand development under the current high browsing pressure (CSBI-scenario), however, the long-term prospects for an effective protection forest are rather limited, due to the moderate density of trees < 12 cm DBH (Fig. 8f). The high browsing pressure influences both species composition and stand structure, by preventing *Sorbus aucuparia* from growing into the canopy layer, and generally prolonging the regeneration period.

The two scenarios are not intended to be realistic, but represent two points along a complex gradient of environmental constraints in the context of which stand development is likely to occur. For example, browsing pressure fluctuates over time, rather than being constant at a high level. Windows of relatively low browsing intensity, even if infrequent, may prove to be vital for allowing tree saplings to grow beyond the reach of ungulate browsing. Nevertheless, our simulation results of the two scenarios indicate that the protective effect of the stand is likely to decrease in the coming decades, confirming the expectations of Thali (1997).

Additionally, the site-specific requirements as postulated in the Swiss management guidelines for protection forests (Frehner et al., 2005) can hardly be achieved during the coming decades (Tab. 7). Even if the proportion of *Abies alba* increases to 39% (CS-scenario) and 28% (CSBI-scenario) after 60 years, a higher proportion of *Abies alba* would be desirable for this site. For protection effects, *Abies alba* has several advantages over other species, in particular *Picea abies*. It is less light demanding and thus able to regenerate on smaller patches, minimizing the opening of the canopy and in this way maintaining the protective effect. It is also less susceptible to competing ground vegetation, and much less susceptible to biotic disturbances (Ott et al., 1997).

Simulation approach

The new model version of ForClim contains a number of simplifications that could be improved in future studies.

First, we only included canopy shading and ungulate browsing as limiting factors for sapling growth. There are, however, several other highly relevant factors. In particular, microsite types and competition with understory vegetation are known to be important for the establishment of tree regeneration in mountain forests (Brang, 1998, Diaci et al., 2000, Frehner, 2001, Kupferschmid, 2005). However, in the current dense stands of the Stotzigwald, understory vegetation is sparse due to low light availability at the forest floor. Therefore, we assumed its impact on sapling growth to be small and did not include it in the present study. Yet, with increasing light availability in the future, the impact of competing understory vegetation is likely to increase (Ammer and Weber, 1999).

Using a relatively coarse representation of the light regime to drive sapling and tree growth is a second simplification. As in many other patch models, tree crowns are not modelled in three dimensions in ForClim, but the leaves of each idealized tree are assumed to be located in an indefinitely thin layer at the top of the stem (Bugmann, 2001b). To improve the representation of the light regime, we integrated a simplistic dynamic crown structure for the present study. While our approach allows us to account for self-pruning of tree crowns in real stands, the representation of the light regime in ForClim would benefit from further investigation and corroboration.

A third simplification, the exclusion of sapling mortality, has already been discussed. This simplification has only a small influence on the simulated stand structure (see above). Explicit consideration of sapling mortality would be essential for simulations over longer periods than used in this study. However, before including a sapling mortality module, additional long-term data on the survival of saplings under different growing conditions are necessary. Thereby, particularly the impact of browsing on sapling mortality, which is still subject to debate (Ammer, 1996a, Eiberle, 1989, Kindermann and Hasenauer, 2003, Motta, 1996, Parks et al., 1998, Rüegg and Schwitter, 2002, Senn and Suter, 2003, Wasem and Senn, 2000), needs further investigation.

In spite of these simplifying assumptions, we think that the improved model version can be used to investigate the structural dynamics of mountain forests over periods of multiple decades.

Conclusions

Dynamic models that include browsing could prove useful for exploring the structural dynamics of mountain forests under the impact of ungulates. With our improved establishment submodel, the ForClim model provides an enhanced tool for investigating the forest-ungulate system and the impacts of structural dynamics on the long-term protective effects of mountain forests. When coupled with models of natural hazards (Wehrli et al., 2003), this tool could become most useful for investigations on the exposure of mountain forests to natural hazards, and could potentially be of value for risk assessment in mountainous areas.

Acknowledgements

This study was carried out within the framework of the National Research Programme 48 (Landscapes and Habitats of the Alps) and funded by the Swiss National Science Foundation (grant 4048-064409) and the Swiss Forest Agency. We would like to express our gratitude to the SMA (MeteoSwiss) for providing us with the long-term weather records and to Brigitte Commarmot for providing us with data for model parameterisation. We wish to thank Andrea Walther and Markus Tschopp for their assistance in the field and Urs Thali for providing us with additional data on the Stotzigwald. We thank Heike Lischke for her help with programming. Two anonymous reviewers significantly improved the focus and quality of the original manuscript.

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Section summary

The aim of section II was to improve the shortcomings of the forest patch model ForClim (Bugmann, 1994) detected in section I, and to adapt ForClim for the use in a combined simulation tool (*CoST*). This was done by improving the reproduction of light competition and by adapting the establishment submodel (Paper III). The former was refined by implementing a simplistic dynamic crown structure, which allows accounting for self-pruning of tree crowns in real stands. Moreover, the establishment submodel was adapted to the needs of a CoST by disaggregating the regeneration process into two stages, seedling establishment and sapling growth. In the latter, sapling growth is modelled explicitly by taking into account two important constraints to sapling growth in mountain forests, namely canopy shading and ungulate browsing. A comparison of the efficacy of the old (ForClim V2.9.3) and new (ForClim V2.9.4) model versions revealed that the latter allowed a more accurate reproduction of mountain forest dynamics over a multi-decadal period. Therefore ForClim V2.9.4 seems ready for the use in a simplistic CoST.

To give an example of the use of the new model version, ForClim V2.9.4 was applied to investigate the ability of a particular mountain forest to provide effective protection against rockfall during a period of 60 years. This forest currently shows sparse tree regeneration, and therefore, its long-term protective effect is dubious. Two scenarios were simulated, i.e. (i) canopy shading but no browsing impact and (ii) canopy shading and high browsing impact. The simulated stand structures were then compared to stand structure targets for rockfall protection, in order to assess their long-term protective effects. Under both scenarios, the initial sparse level of tree regeneration affected the long-term protective effect of the forest, which considerably declined during the first 40 years. Still, in the complete absence of browsing, the density of small trees was able to recover after 60 years. In the scenario including browsing, however, the density of small trees remained at very low levels.

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Section III

Developing a prototype of a *CoST* to investigate the effects of forest dynamics on the long-term protective effect against rockfall

Paper IV

Modelling long-term effects of forest dynamics on the protective effect against rockfall

Section summary

Paper IV

Modelling long-term effects of forest dynamics on the protective effect against rockfall

Based on:

Wehrli., A., L.K.A. Dorren, F. Berger, A. Zingg, W. Schönenberger, and P. Brang. In review. Modelling long-term effects of forest dynamics on the protective effect against rockfall. In review with Forest, Snow and Landscape Research.

Abstract *– The long-term impact of forest dynamics on the protective effect of mountain forests against natural hazards is difficult to study due to the long time periods involved in mountain forest dynamics. To investigate the protection forest system, we combined the forest patch model ForClim with the rockfall model Rockfor^{NET}, and applied the combined simulation tool to a case study to give an example of its use. Based on empirical data, we simulated the development of three mountain forests assuming different scenarios over a period of 60 years. The protective effect of the simulated stands was then assessed by projecting those on one particular site in the Swiss Alps, from where terrain and rock characteristics were available. By doing so, factors that are important for a high long-term protective effect of a mountain forest were determined. Furthermore, shortcomings of the combined simulation tool and the underlying models were identified.*

The long-term protective effect of the three stands against rockfall was generally high for small rocks, but only limited for larger rocks (diameter > 0.8 m), indicating the limit of the protective potential of stands on the slope.

Initial stand conditions, in particular a high initial stand density, as well as a relatively low mortality rate were found to be key factors for a high protective effect over 60 years. Additionally, a high density of tree regeneration in the initial stand was found to increase the long-term protective effect against small rocks, but not against larger rocks. The reasons for these findings as well as model shortcomings and potential improvements are discussed.

Keywords: mountain forest, protection forest, stand structure, mortality, ForClim, RockFor^{NET}

Introduction

Many mountain forests effectively protect people and their assets against natural hazards such as rockfall, snow avalanches, landslides, debris flow, soil erosion and floods (Brang et al., 2001). Thus, many settlements in mountainous areas rely on the protective effect of these forests. The protective effect is thereby mainly provided by the presence of trees in a forest stand. Stand structures, however, are depending on forest dynamics and thus continuously changing. Therefore, the protective effect of a stand is not constant. In the case of rockfall, which is in focus in this study, stands with high stem density and thick trees provide a high protective effect at the moment (Omura and Marumo, 1988, Cattiau et al., 1995). These stands are, however, not the best solution for effective long-term protection. They are usually susceptible to storm damage (Rottmann, 1985) and snow break (Rottmann, 1986, Oliver & Larsen, 1990) and they do not allow for sufficient tree regeneration. The latter however, is crucial, since it ensures continuous forest cover in the long-term, which in turn provides long-term protective effect. In mountain forests with a protective effect, a lack of renewal is particularly severe since, because of slow tree growth at high altitudes (Ott et al., 1997), it will impair the protective effect only after decades, and may therefore be noticed too late. Up to now, however, the influence of different levels of tree regeneration on the future protective effect of mountain forests has not been studied.

The dynamics of a mountain forest as well as its protective effect can be influenced by silvicultural measures (Schönenberger and Brang, 2004). Therefore, it is not only important to gain knowledge on the current protective effect of different stand structures, but it is also essential to investigate the dynamics of mountain forests and their impacts on the long-term protective effect against natural hazards. By doing so, the long-term efficiency of different silvicultural measures can be assessed. Given the importance of protection forests, it would therefore be desirable to gain additional knowledge on the impact of forest dynamics and in particular of low levels of tree regeneration on the long-term protective effect of mountain forests against natural hazards.

The protection forest system is a dynamic system, where many processes involve long time periods. Therefore, it is difficult to study, and empirical data on the influence of different silvicultural measures on the long-term protective effect against natural hazards are sparse. The same even accounts for data on the impact of natural forest dynamics. Thus, the efficiency and efficacy of silvicultural measures

can only be estimated with uncertainty up to now. To overcome this problem and to investigate the protection forest system, simulation models could be used for (1) investigating forest dynamics (Johnson et al., 2001) as well as for (2) assessing the level of protection provided by different stand structures (Peng, 2000). Finally, two of these models could be combined in one simulation tool.

Useful model candidates for such a combined simulation tool should fulfil some minimal requirements: The *model of forest dynamics* should accurately project the development of key stand characteristics that determine its protective effect over several decades. These key characteristics include tree density, diameter distribution and species composition (Dorren, et al., 2005). Additionally, the regeneration process should be included in sufficient detail to reflect the most important features of mountain forest regeneration, e.g. the long regeneration period at high altitudes or constraints such as browsing impact by ungulates. The *rockfall model* (or the model of natural hazard in general) in a *CoST* should allow an accurate assessment of the protective effect of a stand. For rockfall as natural hazard, this means that the interaction of falling rocks and trees has to be reproduced with sufficient detail in the model, e.g., the dissipation of energy due to impacts against individual trees needs to be included in a realistic way.

Both processes, forest dynamics and rockfall, have frequently been modelled. Forest dynamics in terms of the structural forest patterns described above, have successfully been reproduced by forest patch models (cf. Lindner et al., 1997, Shugart, 1998, Huth and Ditzer, 2000, Risch et al., 2005, Wehrli et al., 2005). Moreover, the regeneration process, which is normally simulated without great detail in standard patch models (cf. Price et al., 2001), has recently been improved in the patch model ForClim (Wehrli et al., in review), and important features of tree regeneration in mountain forests have been included (e.g. individual sapling growth, impact by browsing ungulates). The protective effect a stand can provide against rockfall can be assessed with sufficient accuracy by recently developed rockfall models, which include the interactions of falling rocks and trees (e.g., Dorren et al., 2004, Brauner et al. 2005, Berger and Dorren, in review, Stoffel et al., 2006; cf. Dorren 2003 for an overview of rockfall models). Thus, there are several promising models that fulfil the minimal requirements for a combined simulation tool to a sufficient degree. Such candidates have, however, not been combined up to now.

The aim of this study was therefore to join two models in a combined simulation tool, in order to investigate the impact of forest dynamics on the long-term protective effect of mountain forests. We therefore combined the forest patch model ForClim (Bugmann, 1994, 1996) with the rockfall model RockFor^{NET} (Berger and Dorren, in review), and applied the combined simulation tool to a case study to give an example of its use for investigating the protection forest system. Based on data from three different mountain forests in the Swiss Alps, we first simulated the potential development of three initial stands under different scenarios over a period of 60 years. The protective effect of the simulated stands against rockfall was then assessed by projecting those on one particular site in the Swiss Alps, from where data on terrain and rock characteristics were available. Finally, factors that are important for a high long-term protective effect of a mountain forest were identified.

Methods

Description of the simulation models and model combination

The forest patch model ForClim

ForClim was originally developed to assess the impacts of climatic changes on tree species composition and biomass of forests in the Swiss Alps (Bugmann, 1994). The applicability of ForClim was successfully extended from the Swiss Alps to other climatic regions through several modifications (Bugmann and Solomon, 1995, Bugmann and Cramer, 1998, Bugmann and Solomon, 2000, Bugmann, 2001, Shao et al., 2001). Even though ForClim was not originally designed to simulate structural forest patterns such as diameter distributions, it has been shown to accurately reproduce such patterns in simulations over periods of several decades (Risch et al., 2005, Wehrli et al., 2005).

ForClim originally consists of three modular submodels, one for the abiotic environment (ForClim-E), one for tree population dynamics (ForClim-P), and one for soil carbon and nitrogen turnover (ForClim-S), respectively. However, most investigations with ForClim are performed without the soil submodel since the model variant ForClim-E/P/S did not improve model performance to a considerable degree (Bugmann, 1994, 1996).

The abiotic environment submodel (ForClim-E) is driven by monthly mean temperatures and monthly precipitation sums of every simulation year. The input for

these variables can either be derived from measured time series data, or it can be generated by a weather generator based on long-term statistical data.

In the population dynamics submodel (ForClim-P), no spatial interactions between the simulated patches are considered, i.e. the simulated patches are completely independent. The representation of *tree establishment*, i.e. the establishment process of young trees, has recently been improved in the model (Wehrli et al., in review). It is modelled in two different modules, one for seedling establishment and one for sapling growth. In the first module, establishment of seedlings is determined by four limiting factors, i.e., light availability at the forest floor, browsing intensity, soil moisture and absolute winter minimum temperature. The response to these factors is species-specific and the limiting factors are applied as environmental filters to determine the establishment of seedlings with an initial height of 5 cm (cf. Wehrli et al., in review). Established seedlings are then transferred to the sapling growth module, in which they grow up to a height of 2 m before they recruit as trees with an initial diameter at breast height (DBH) of 1.27 cm. In this module, individual sapling growth is modelled explicitly by taking into account suboptimal growth conditions due to canopy shading and ungulate browsing (cf. Wehrli et al., in review).

Tree growth is derived from a simple carbon budget approach (Moore, 1989). Four factors are used to take into account suboptimal conditions for tree growth: light availability, growing season temperature, soil moisture, and soil nitrogen availability (Bugmann, 1994, 1996). To derive an overall growth reduction, these four factors are combined using a modified geometric mean (Bugmann, 1994).

Tree mortality is simulated as a combination of an age-related and a stress-induced mortality rate (Botkin, et al. 1972, Bugmann, 1994, Keane et al., 2001), giving rise to high mortality of small trees due to strong competition for light, and high mortality of old trees due to low vigour (Bugmann, 1994). Thus, trees that grow slowly due to adverse environmental conditions are subject to a higher stress-induced mortality rate.

A detailed description of ForClim can be found in Bugmann (1996). The model version used in this study, ForClim V2.9.4, is documented in detail in Wehrli et al. (in review).

The rockfall model RockFor^{NET}

The RockFor^{NET} model was recently developed by Berger and Dorren (in review) to provide a tool for the assessment of the probable residual hazard at the foot of a slope covered by a protection forest. The residual hazard is thereby a measure of the protective effect of a stand. It is defined as the percentage of rocks passing a forested slope, i.e. the percentage that cannot be stopped by the forest stand.

The model is based on knowledge and results gained from more than 100 real-size rockfall experiments on a forested mountain slope in the French Alps (Berger and Dorren, in review). In these experiments, real rockfall trajectories including impacts against trees were recorded by high-speed video cameras and analyzed in detail (Dorren et al., 2005).

RockFor^{NET} has been validated for several sites, and it has been shown to allow realistic assessments of the protective effect of different stand structures (Berger and Dorren, in review). Furthermore, it has been frequently applied in practice to assess the residual hazard for different sites in the French, Swiss and Austrian Alps.

To assess the residual hazard, RockFor^{NET} calculates the energy balance of a falling rock on a forested slope, i.e. it calculates the energy a falling rock can develop on a given slope, and sets it off against the energy that can be dissipated by the stand on this slope. To do so, the model needs a few input parameters on *forest stand* (species composition, stand density, and a representative DBH, i.e. a measure of location that is representative for the DBH distribution), *terrain* (cliff height, slope length between the foot of the cliff and the foot of the forested slope, slope length of the forested slope, mean slope gradient) and *rock characteristics* (mean rock diameter, rock density).

The terrain and rock input parameters are needed to determine the energy a falling rock can develop on a given slope. In RockFor^{NET}, this calculation is based on the approach of the energy line angle, i.e. the angle of the straight line between the starting point and the maximum stopping point (Heim, 1932, Toppe, 1987, Gerber, 1994). From the height difference between the energy line and the slope surface at the foot of the forested slope, the kinetic energy of the falling rock at the foot of the slope Ek_{foot} is calculated.

The forest stand is considered as spatially distributed "*rockfall curtains*" (Fig. 1), i.e. the stand input parameters are used to generate virtual rows of trees standing next to each other. Based on the findings from the real-size rockfall experiments, the

distance between two trees in a row is thereby set to 0.9 x mean rock diameter, and the distance between two tree rows along the slope is set to 33 m (Fig. 1; an explanation of this value is given in Dorren et al., 2005). All the trees in a row have a diameter equal to the given representative DBH, which in combination with the tree species determines the efficacy of a tree row regarding energy dissipation during a rockfall impact. The relationship between DBH and the amount of energy that can be dissipated per tree species is thereby derived from the data gained from the real-size rockfall experiments described above (Fig. 2; cf. Dorren and Berger, 2006).

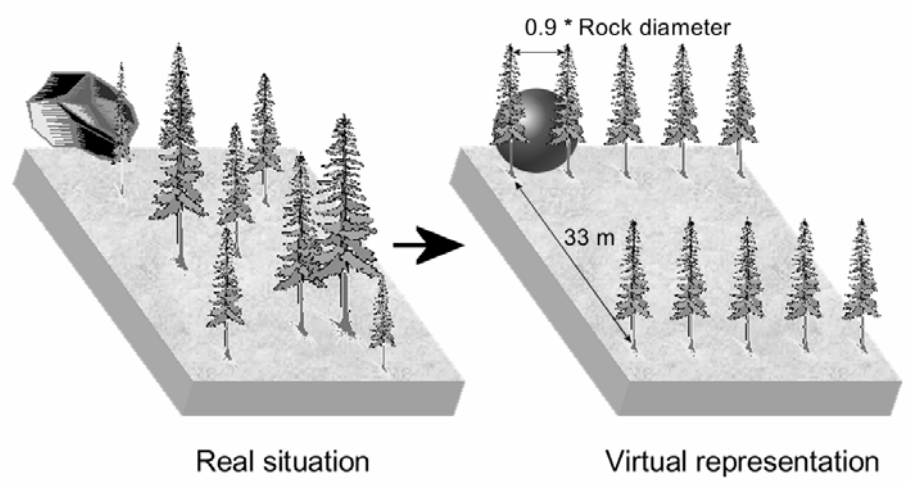

Figure 1. Explanation of the principle of RockFor^{NET}.

In order to calculate the probable residual hazard, the model determines the number of tree rows required for a full protection on the foot of the slope, i.e. 100% of the rocks stopped. Thereby, the total amount of energy that has to be dissipated by the stand is assumed to be 2.8 times the value of Ek_{foot}. This assumption is again based on the findings of the real-size rockfall experiments, in which the stand dissipated on average 2.8 times as much as energy as the Ek_{foot} calculated with the energy line (Berger and Dorren, in review).

Finally, RockFor^{NET} compares the required number of trees with the existing number of trees in the stand and translates the difference between the two into a probable residual hazard, ranging from 0-100%. A detailed description of RockFor^{NET} can be found in Berger and Dorren (in review).

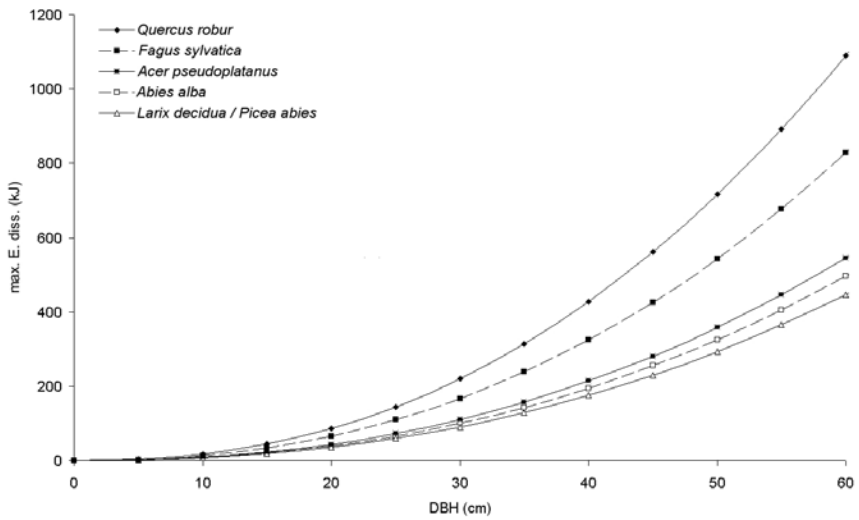

Figure 2. Relationship between DBH and amount of energy that can be dissipated for several tree species (from: Dorren and Berger, 2006).

Model combination

For the present study, the two models were not physically combined, but joined together by file exchange. Thus, the stand input data needed for the RockFor^{NET} model was derived from the projected stand structures simulated with ForClim.

Model input data and modelling scenarios

Stand and regeneration data

For our case study, we modelled several scenarios based on initial stand and regeneration data that were derived from empirical data from three mountain forests in the Swiss Alps, which are all dominated by *Picea abies* (L.) Karst. and *Abies alba* Mill. (cf. Tab. 1) and situated between 700 and 1000 m above sea level. Figure 3 and Table 1 give an overview of the current regeneration and stand structure of the three stands.

The first initial stand is rather even-aged and therefore referred to as EA-stand (even-aged stand). The stand consists of 561 trees ha^{-1} > 4 cm DBH, and only minor timber harvesting took place within the stand in the last decades. Tree regeneration is rather scarce with a sapling density of approximately 2230 saplings ha^{-1} (Tab. 1)

The second initial stand is currently converted into a selection forest (plentering according to Schütz, 2001). Thus, it is referred to as C-stand (stand under conversion). The stand consists of 430 trees ha^{-1} > 4 cm DBH, and there is ample tree regeneration with more than 30’000 saplings ha^{-1} (Tab. 1).

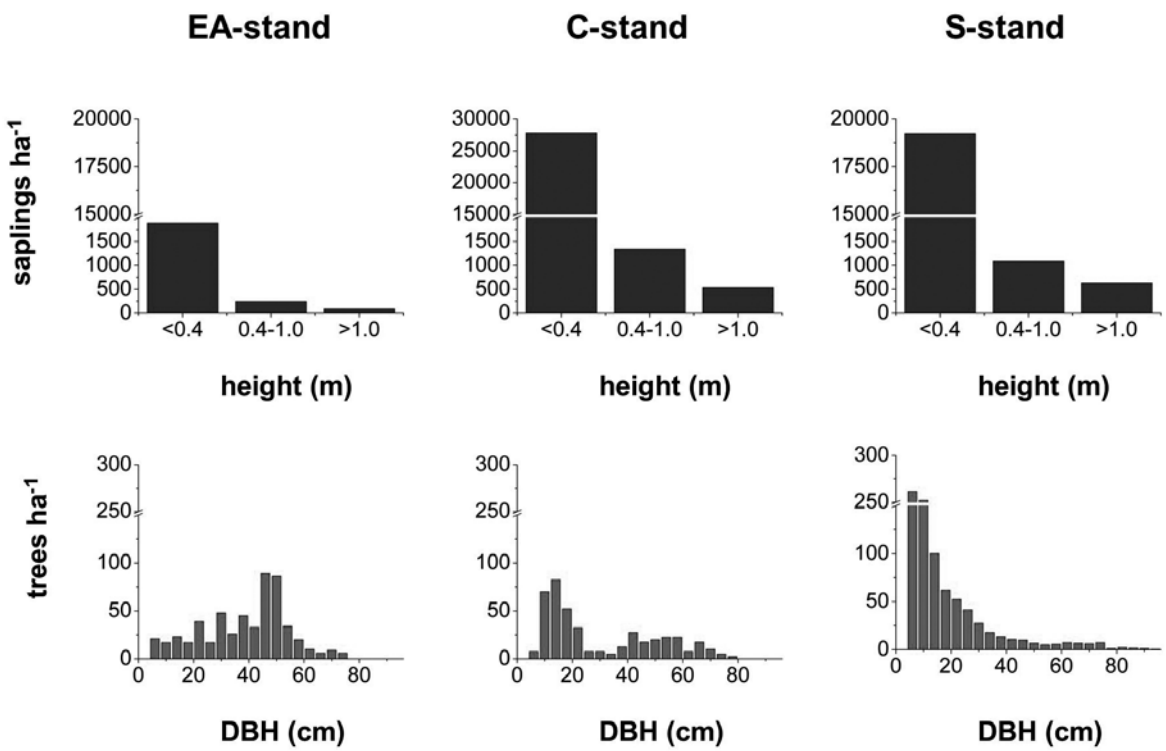

Figure 3. Size distributions of tree regeneration (above) and DBH (below) for the three initial stands.

The third initial stand has been managed as uneven-aged selection forest for several decades. It is referred to as S-stand (selection stand). The stand consists of 901 trees ha^{-1} > 4 cm DBH, and there is much tree regeneration with more than 20'000 saplings ha^{-1} (Tab. 1).

Table 1. Stand and regeneration characteristics of the three initial stands for the simulation of forest dynamics.

stand	Stand characteristics (main tree species, stand basal area (BA), median DBH and tree density at beginning of simulation [trees > 4 cm DBH])	Regeneration characteristics (main tree species, and sapling density up to 4 cm DBH at beginning of simulation)
EA-stand (even-aged stand)	*Picea abies* (83%), *Abies alba* (13%) BA = 73.8 m^2 ha^{-1}, DBH$_{med}$ = 42.3 cm, 561 trees ha^{-1}	*Picea abies* (20%), *Abies alba* (44%) 2'230 saplings ha^{-1}
C-stand (stand under conversion)	*Picea abies* (32%), *Abies alba* (62%) BA = 44.5 m^2 ha^{-1}, DBH$_{med}$ = 36.6 cm, 430 trees ha^{-1}	*Picea abies* (42%), *Abies alba* (57%) 31'090 saplings ha^{-1}
S-stand (selection stand)	*Picea abies* (43%), *Abies alba* (52%) BA = 35.1 m^2 ha^{-1}, DBH$_{med}$ = 26.4 cm, 901 trees ha^{-1}	*Picea abies* (49%), *Abies alba* (51%) 22'170 saplings ha^{-1}

Scenarios for the simulation of forest dynamics

The scenarios used for simulating forest dynamics were based on (i) different initial stand and regeneration structures (variation of input data), and on (ii) different model parameters (variation of growing constraints). An overview on the different scenarios for the simulation of forest dynamics is given in Tab. 2.

Table 2. Scenarios for the simulation of forest dynamics for each initial stand.

	Species composition of tree regeneration		
	Initial composition	100% *Picea abies*	100% *Abies alba*
Levels of sapling density	3	3	3
Levels of browsing impact	5	2	2
Levels of mortality rate	2	2	2
Total scenarios per initial stand	*30*	*12*	*12*

For the variation of input data, we used the three initial stand and regeneration structures described above. Additional scenarios for tree regeneration were included by varying sapling density and species composition in the regeneration input data. Sapling density was thereby varied on three levels, low, medium and high (Tab. 3), whereas the variation of species composition included the initial composition, 100% *Picea abies* regeneration, and 100% *Abies alba* regeneration, respectively.

Table 3. Variation of tree regeneration. Bold figures denote the actual initial level of tree regeneration. Regeneration levels were varied with factors 5 (medium) and 10 (high) for the EA-stand, and with factors 0.5 (medium) and 0.1 (low) for the C- and the S-stand, respectively.

Regeneration level	EA-stand	C-stand	S-stand
low	**2230**	3109	2217
medium	11150	15545	11085
high	22300	**31090**	**22170**

The variation of growing constraints was on the one hand based on the variation of input data, since particularly stand structure has a significant influence on stand dynamics. On the other hand, additional growing constraints were included by variation of two model parameters, one influencing the browsing impact on saplings and the other representing the mortality rate of trees. *Browsing impact* was varied in terms of browsing intensity, which led to a species-specific reduction of sapling growth (Tab. 4; for details see Wehrli et al., in review). *Tree mortality* was varied between a standard mortality rate and an increased mortality rate. The standard mortality rate is the standard value of age-related tree mortality in patch models, and it implies that 1% of all established saplings survive to their species-specific

maximum (Shugart 1984, Bugmann, 1994). In contrast, the increased mortality rate allows depicting processes that are not explicitly modelled such as a coarse reproduction of a thinning regime or increased tree mortality due to injuries by falling rocks. The mortality rate was thereby increased to reflect the assumption that only 1‰ of the established saplings reach their maximum age. The same increased mortality rate was used in the study by Risch et al. (2005), who stated that this increased mortality rate might be more realistic under the prevailing conditions of their study site. Thus, the increased mortality rate used in this study seems to be moderate.

Table 4. Variation of browsing impact for the main tree species. Browsing impact is determined by multiplying the species-specific browsing susceptibility included in ForClim (*Picea abies* = 0.25, *Abies alba* = 0.75) with browsing intensity. Together with canopy shading, browsing impact then determines the species-specific reduction of sapling growth (reduction factor for sapling growth within a range of 0-1, whereby 0 denotes optimum height growth and 1 denotes no more height growth; for details see Wehrli et al., in review).

	Browsing intensity				
Species	0	0.67	1	1.33	2
Abies alba	0	0.5	0.75	1	1
Picea abies	0	0.166	0.25	0.333	0.5

Terrain and rock characteristics

The terrain and rock input data needed for RockFor^{NET} were derived from empirical terrain and rock characteristics from the site with the EA-stand (see above) and are shown in Tab. 5. This site, called Stotzigwald (meaning „steep forest“), is a steep forested slope in the Swiss Alps (46°45’ N and 08°39’ E) with a mean slope gradient of approximately 45° and multiple interspersed cliffs (Thali, 1997). The stand at this site provides slope stability for one of the most heavily used traffic routes in Switzerland connecting northern and southern Europe.

Scenarios for the assessment of the protective effect

The protective effect of the simulated stand structures against rockfall at the Stotzigwald site was assessed for six different rock size classes S1-S6, ranging from a rock diameter of 0.2 m (S1) up to 1.2 m (S6; Tab. 5). Thus, for each rock size class, the residual hazard of each simulated stand structure was calculated.

Table 5. Terrain and rock characteristics for the assessment of the protective effect derived from Stotzigwald site.

	Site Stotzigwald
Cliff height	40 m
Slope lenght between cliff and forested slope	0 m
Slope lenght of the forested slope	325 m
Mean slope gradient	45°
Mean rock diameters	0.2, 0.4, 0.6, 0.8, 1.0, 1.2 m
Rock density	2700 kg / m^2

Simulation set-up

Simulation of forest dynamics

Starting from one of the three initial stands, each forest scenario was simulated for a period of 60 years. Thereby, the recruitment module allowing for additional seedling regeneration during the simulation period was not used for the simulations, since data for a reliable parameterisation and corroboration were not available. Thus, only the initial tree regeneration was considered in the simulations, i.e. the number of saplings was directly imported into the sapling growth module. As evident from the study by Wehrli et al. (in review), this simplification did not have a significant influence on the simulated stand structure for a period of 60 years, i.e. the stand structure after 60 years that is relevant for the protective effect against rockfall did not depend on additional seedling recruitment.

For the simulations based on the C- and the S-stand, slight modifications in the parameterization of the tree growth module were necessary, since preliminary simulation runs with the standard parameterizations yielded an overestimated tree growth compared to empirical growth data from long-term data series from these stands (Tab. 6). This overestimation is very probably due to the parameterization of the dynamic crown structure implemented in ForClim V2.9.4, which is mainly based on data from rather even-aged stands (cf. Wehrli et al., in review). Therefore, we introduced a correction factor in terms of a multiplier for the leaf area index for the C- (correction factor 1.2) and the S-stand (correction factor 1.3), which led to more realistic growth rates (Tab. 6).

Table 6. Growth rates simulated with ForClim with and without CF for the C- and S-stand, and comparison with empirical data from long-term data series for both stands. CF denotes the correction factor included in ForClim to modify the leaf area index (see text). Growth rates are measured in terms of increment of basal area ha^{-1} over 60 years. na: not available.

	empirical growth	No CF	CF 1.2	CF 1.3	CF 1.4
C-stand	42.0	50.1	42.5	na	na
S-stand	47.7	64.6	50.9	47.0	42.0

The initialisation of ForClim with stand and regeneration data was performed at the scale of individual patches, whereby the patch size was set to 225 m^2 (15 m x 15 m) for the present study. To reduce the stochastic "noise" in the simulation results, the simulation experiments were performed with numerous repetitions (n = 237-320 patches, depending on the area covered by the initial stand; cf. Bugmann, 1996, Pretzsch and Dursky, 2001).

The input for the weather generator in the abiotic environment submodel of ForClim was derived from time series of monthly precipitation sums and monthly mean temperatures from the weather station at Gurtnellen (739 m asl), which is located approximately 2.5 km from the site with the EA-stand (Stotzigwald).

Assessment of the protective effect

For the assessment of the protective effect of the simulated stands, we only considered trees with a DBH > 8 cm, as the exact behaviour of smaller trees regarding rockfall protection is unknown yet. Species composition and stand density were directly derived from the output file of ForClim and included in RockFor^{NET}. The measure of location for the DBH needed in RockFor^{NET} to calculate the energy that can be dissipated by a tree row was set to the median, since most of the simulated DBH distributions were rather asymmetric. For such distributions, the median is more representative than other measures of locations (Sokal and Rohlf, 1995), and therefore, the use of the median was thought to deliver more accurate results. This assumption was confirmed by preliminary tests with other measures of location such as the mean quadratic DBH. The latter yielded poor results for most of the simulation scenarios, with most of the scenarios for S1-S4 having no the residual hazards at all. This, however, seems very unlikely on the Stotzigwald site.

Analysis of simulation results

Assessment of the residual hazard under the different simulation scenarios

The level of the protective effect provided under the different scenarios was compared graphically for all stands and for each initial stand by boxplots, displaying median, outliers (1.5-3 box lengths from the end of the box) and extremes (> 3 box lengths from the end of the box; Sokal and Rohlf, 1995). The length of the box representing the interquartile range was computed from Tukey's hinges (SPSS Inc., 2001). Moreover, the range of the residual hazards over all scenarios and the mean residual hazard over all scenarios were calculated per rock size class. Additionally, the parametric (Pearson correlation, cf. Stahel, 2000) and non-parametric (Spearman correlation, cf. Stahel, 2000) correlations between the simulated stand structures and the residual hazards were determined.

Derivation of indicators for a high long-term protective effect

To determine the most important factors for a high long-term protective effect per rock size class, logistic regression models were applied to the data sets. The different levels of residual hazards per rock size class were thereby divided into two classes representing a high and a low protective effect, respectively, to provide the necessary binary target variable for the logistic models. The limit for the allocation into these classes was fixed per rock size class, based on qualitative observations of the frequency of rockfall events per size class at the Stotzigwald site (Tab. 7). Since for smaller rocks, these events are rather frequent, the limits for those rock size classes were set to low levels. In contrast, the limit for S3-S6 was set to considerably higher values, taking into account the lower frequency of these events.

Table 7. Limits of the binary target variable in the logistic regression models. Stands that delivered a residual hazard up to the limit values are considered to have a high protective effect, the others are allocated to the class with a low protective effect.

	Rock size class					
	S1 (d = 0.2 m)	S2 (d = 0.4 m)	S3 (d = 0.6 m)	S4 (d = 0.8 m)	S5 (d = 1.0 m)	S6 (d = 1.2 m)
Limit	0.1%	10%	30%	50%	50%	50%

The starting point for each logistic regression was a model including all explanatory variables. Thereby, the measurement variables (Sokal and Rohlf, 1995, p.11) were first transformed following the Tukey first aid transformations (cf. Stahel, 2000) and then included in the logistic model, whereas the categorical variables, i.e.

browsing impact and mortality rate, were included as categorical co-variables based on the indicator contrast method (SPSS Inc., 2001). The most important variables per rock size class were then determined with a backward selection method based on Wald-values (SPSS Inc., 2001, Sokal and Rohlf, 1995), i.e. the variables with the lowest Wald-values were excluded from the models. To verify the model assumptions, the residuals were finally examined (Stahel, 2000).

In addition to the logistic models, explorative graphical analyses by boxplots were performed and the parametric and non-parametric correlations between all variables were determined to investigate the relationships between the variables (details see above).

Results

Residual hazards of the different stands

Residual hazard of the initial stands

The different initial stands markedly differed in their protective effect against the six rock size classes as can be seen by the different levels of the initial residual hazard (Tab. 8). Whereas the EA-stand, which corresponds to the stand currently present at the Stotzigwald site, yielded an acceptable residual hazard for all rock size classes, the performance of the C- and in particular of the S-stand were very limited for rocks with a diameter > 0.2 m. From size class S4 on, the residual hazard for these two stands was > 90%, i.e. the protective effect was only marginal.

Table 8. Range of residual hazards and mean residual hazard per rock size class for all simulated stands as well as mean residual hazard and initial residual hazard per initial stand. RH: Residual hazard in %

	All sites		**EA-stand**		**C-stand**		**S-stand**	
Rock size class	mean RH	range of RH	initial RH	mean RH	initial RH	mean RH	initial RH	mean RH
S1 (d = 0.2 m)	1	0-24	0	1	4	1	2	0
S2 (d = 0.4 m)	24	0-76	0	24	17	37	51	10
S3 (d = 0.6 m)	61	0-97	0	54	60	82	90	47
S4 (d = 0.8 m)	78	0-99	6	67	91	95	98	74
S5 (d = 1.0 m)	89	2-100	36	76	97	98	99	91
S6 (d = 1.2 m)	93	31-100	51	84	99	99	100	97

Residual hazards under the different simulated stands

The assessment of the protective effect of the different simulated stands yielded a low residual hazard for rocks with a diameter of 0.2 m (rock size class S1, cf. Fig. 4). In most of the scenarios, the residual hazard was thereby around zero, but a few scenarios led to residual hazards up to 24%, i.e. 24% of the rocks still passed the simulated stand on the slope (Tab. 8). With increasing rock diameter, the mean residual hazard over all scenarios rapidly increased, and for S5 and S6, it was around 90% (Tab. 8, Fig. 4): For rock size class S2 (d = 0.4 m), the residual hazard was generally low as indicated by an average of 24% (Tab. 8, Fig. 4), whereas for size class S3 (d = 0.6 m) and S4 (d = 0.8 m), it strongly increased, as can be seen by the mean residual hazard of 61% and 78%, respectively (Tab. 8, Fig. 4). The protective effect against the largest rocks (size classes S5, d = 1.0 m, and S6, d = 1.2 m, cf. Tab. 8) was rather limited as evidenced by the high average residual hazard. Still, the range of the residual hazards for these classes indicated that under a few scenarios the simulated stands were able to yield a residual hazard < 50% (Fig. 4). Those scenarios were exclusively based on the EA-stand, and generally included the standard mortality rate and a relatively low regeneration density.

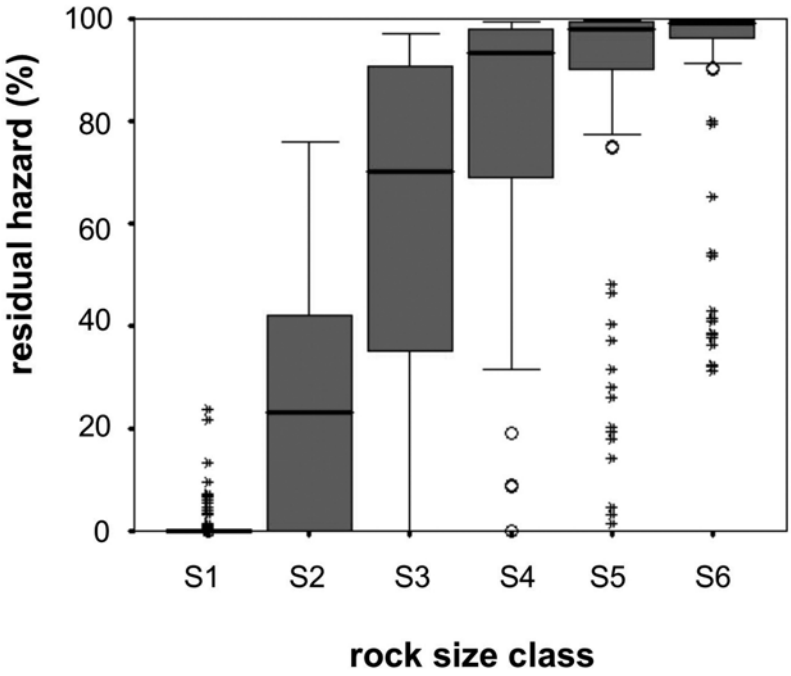

Figure 4. Residual hazards over all scenarios per rock size class. Circles denote outliers (1.5-3 box lengths from the end of the box), asterisks mark extremes (> 3 box lengths from the end of the box).

As can be seen from Fig. 5, the residual hazard under the different scenarios strongly depended on the initial stand. The EA-stand and the S-stand thereby performed much better than the C-stand, i.e. the simulated stands based on the EA- and S- stands generally showed a higher protective effect under the different scenarios than the ones based on the C-stand, which can be seen from the mean residual hazards per initial stand (Tab. 8). The latter only provided a low residual hazard for rock size class S1 (d = 0.2 m), and a rather low residual hazard for rock size class S2 (d = 0.4 m, mean residual hazard: 37%, cf. Fig. 5 and Tab. 8).

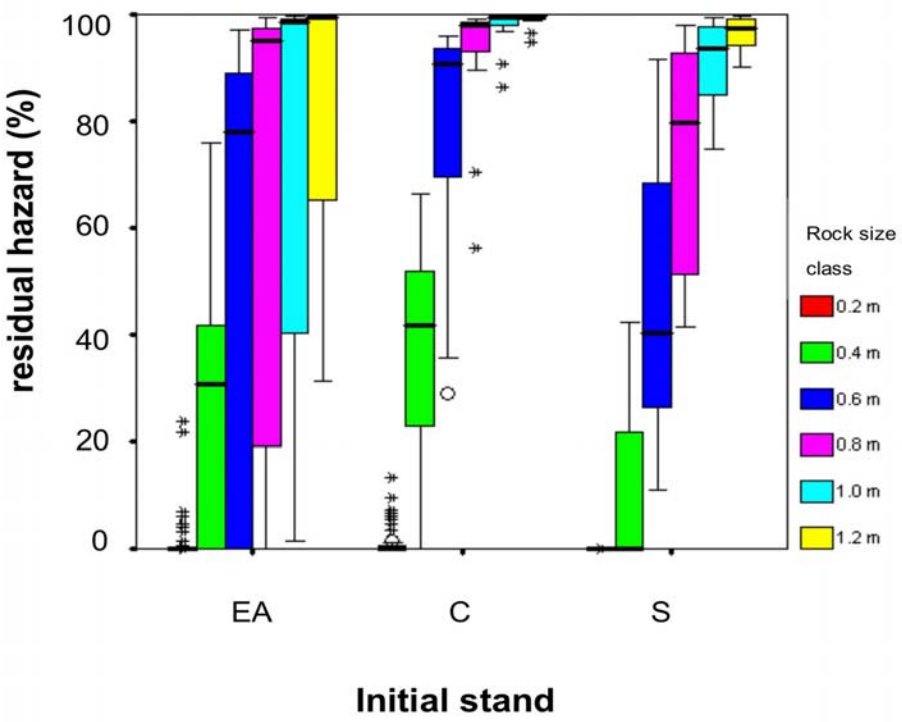

Figure 5. Residual hazards over all scenarios per rock size class and initial stand. Box plot symbols are explained in the legend to Fig. 4.

Development of the residual hazards over the simulation period

The EA-stand, which in its initial stand provides a relatively high protective effect (cf. Tab. 8), showed an increase of the residual hazard under several simulation scenarios: For the smallest rock size class S1, 13 scenarios yielded an increase of the residual hazard when compared to the initial stand. For rock size classes S2-S6, however, the residual hazards increased in 32 (S2) – 44 (S4-S6) of the 54 scenarios. Most of the scenarios leading to an increased residual hazard thereby included an increased mortality rate.

The scenarios based on the C-stand, which for its initial stand showed rather high levels of residual hazards (cf. Tab. 8), yielded an even more pessimistic development than the EA-stand: For S2-S6, almost all scenarios led to an increase of the residual hazard compared to the initial stand. For S1, however, the residual hazard only increased under seven scenarios with increased mortality rates.

In contrast to the C-stand, the S-stand, which similarly showed relatively high levels of residual hazards for its initial stand (cf. Tab. 8), provided a more optimistic development of the protective effect with decreasing residual hazards compared to the initial residual hazards under all simulation scenarios.

Key factors for the assessment of the residual hazard with RockFor^{NET}

The *simulated stand density* and for larger rocks particularly the *simulated median DBH* showed the highest correlations with the residual hazard for each rock size class, and can thus be seen as key factors for assessing the residual hazard with RockFor^{NET} (Tab. 9). The two key factors were highly negatively correlated (parametric: -0.66, non-parametric: -0.52, $p < 0.01$).

Table 9. Important correlations between simulated stand structure and residual hazard. Values denote parametric (Pearson) and non-parametric correlations (Spearman, in brackets). Negative signs indicate a negative correlation. All correlations are significant at the 0.01 level (2-tailed).

	All sites	
Rock size class	simulated median DBH	simulated density
S1 (d = 0.2 m)	-0.20 (-0.37)	-0.24 (- 0.3)
S2 (d = 0.4 m)	-0.69 (-0.95)	0.46 (0.45)
S3 (d = 0.6 m)	-0.88 (-0.98)	0.62 (0.55)
S4 (d = 0.8 m)	-0.96 (-0.98)	0.61 (0.55)
S5 (d = 1.0 m)	-0.96 (-0.98)	0.56 (0.55)
S6 (d = 1.2 m)	-0.93 (-0.98)	0.52 (0.55)

The *simulated median DBH* and the residual hazard were negatively correlated for all rock size classes with very high correlations for S3-S6 (cf. Tab. 9). In contrast, the *simulated stand density* (stem number) and the residual hazard were only negatively correlated for rock size class S1. For all larger rocks, the correlations were found to be positive and on a lower level than the correlations between *simulated median DBH* and residual hazard.

Important factors for a high long-term protective effect

By use of logistic regression models per rock size class, the most important factors that are correlated with a high long-term protective effect were determined for the rock size classes S1-4. Rock size classes S5 and S6 were excluded from these analyses since the protective effect against these rock sizes classes was found to be very limited (see above). An overview over the four logistic regression models is presented in Tab. 10. In those models, variables with a negative B-value as indicated by the negative sign denote factors that helped to reduce the residual hazard. Thus, in terms of long-term protection, a negative variable in the logistic model contributed to an increased protective effect.

Table 10. Logistic regression models for the rock size classes S1-S4. Negative signs indicate variable that lead to a reduced residual hazard.

		Multiple logistic regression model	
Rock size class	Parameter	B-value	Wald
S1 (d = 0.2 m)	Regeneration density 2005	-0.023	16.3
	Initial stand density 2005	-0.821	25.4
	Standard mortality rate	-1.605	10.0
	Constant	20.034	26.3
S2 (d = 0.4 m)	Regeneration density 2005	0.022	15.0
	Initial stand density 2005	-0.824	30.7
	Standard mortality rate	-2.644	25.3
	Constant	18.751	28.9
S3 (d = 0.6 m)	Regeneration density 2005	0.029	17.9
	Initial stand density 2005	-1.114	10.6
	Initial median DBH 2005	-0.166	7.2
	Standard mortality rate	-2.284	15.6
	Constant	32.030	10.2
S4 (d = 0.8 m)	Regeneration density 2005	0.028	17.2
	Initial stand density 2005	-0.453	9.5
	Standard mortality rate	-2.076	13.1
	Constant	10.868	9.6

All variables significant at the $p < 0.01$ level. n = 162 scenarios (54 per initial stand)

For the rock size class S1 (d = 0.2 m), the factors *initial stand density*, *regeneration density* and *standard mortality rate* were found to have a significant negative influence on the residual hazard (Tab. 10). These findings were confirmed by boxplots for the three factors over all simulation scenarios (Fig. 6), although, due to the overall low residual hazard for S1, the tendencies are barely visible.

For the rock size classes S2-S4, ranging from 0.4 – 0.8 m in diameter, the same factors had a significant influence. The *initial stand density* and the *standard mortality rate* were again found to have a significant negative influence on the residual hazard (Tab. 9). The influence of the *regeneration density*, however, was found to be slightly positive for all three rock size classes. Moreover, for rock size class S3, the *initial*

median DBH was found to have a negative influence on the residual hazard (Tab. 9). Again, these findings were confirmed in the boxplots (Fig. 6).

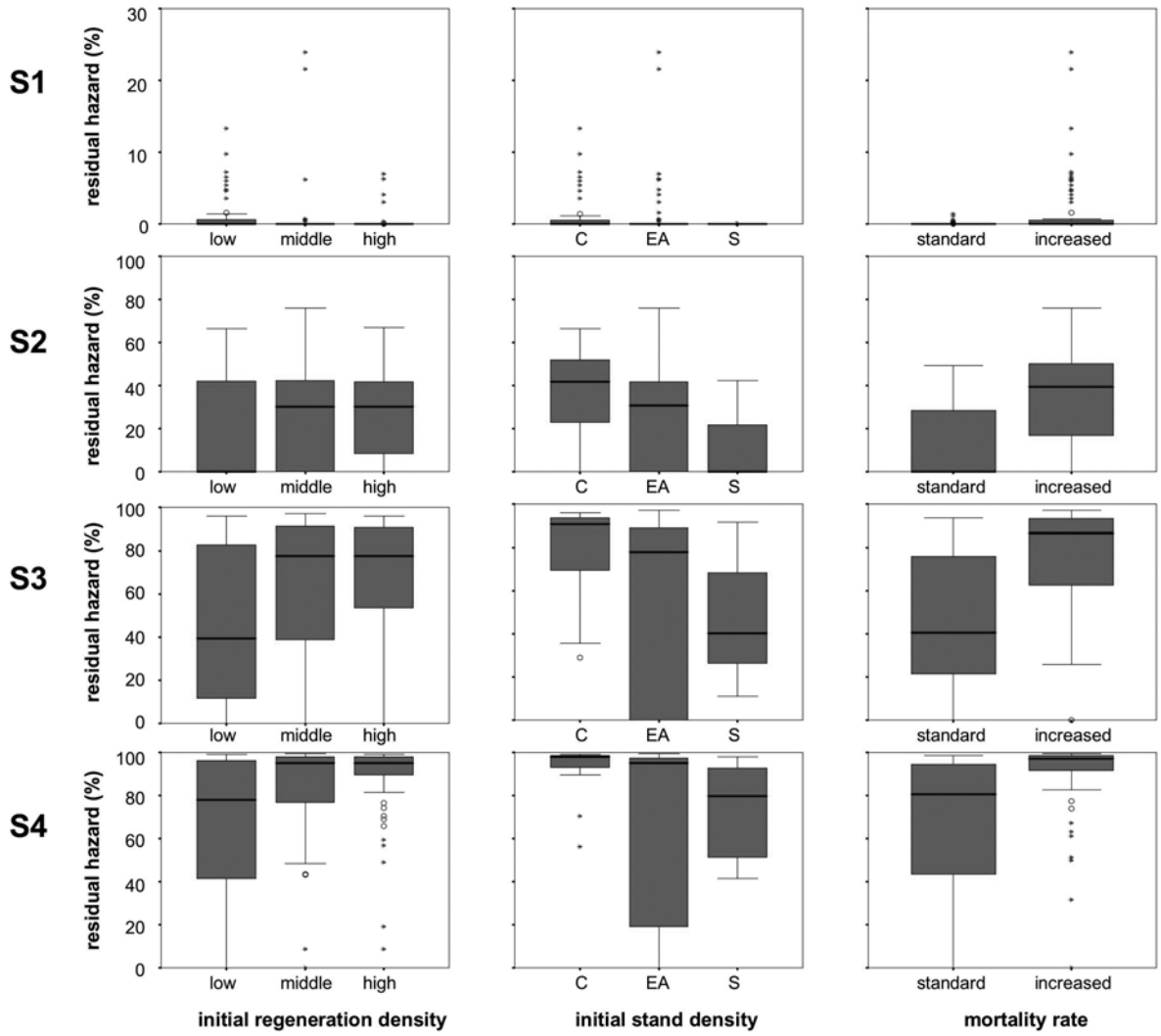

Figure 6. Relationship between residual hazard and the variables initial regeneration density (first column), initial stand density (second column) and mortality rate (third column) for rock size class S1 (first row), S2 (second row), S3 (third row), and S4 (fourth row). Box plot symbols are explained in the legend to Fig. 4.

Discussion

Residual hazards of the different stands

The residual hazards under the different simulation scenarios were largely different for the six rock size classes. For the smallest rocks included in this study, i.e. S1 with d = 0.2 m, all initial stands and almost all simulated stands showed low residual hazards and by this, a high protective effect. Additional assessments of the protective effect against rocks smaller than S1, i.e. with d < 0.2 m, always yielded a residual hazard of zero. Thus, we think that at least on the present slope if not in general, a rock diameter of about 0.2 m represents the lower boundary of the applicability of the RockFor^{NET} model. This is, however, not likely to be a model artefact, but it simply reflects the simplistic approach of the RockFor^{NET} model, which only takes into account the forest stand for the calculation of the energy balance of a rock. For smaller rocks, however, other factors are likely to highly affect the energy balance of falling rocks, such as surface roughness or dampening effects of the surface. In particular surface roughness, e.g. woody debris, snags, branches, bush vegetation, or stopped rocks can thereby effectively slow down or even stop falling rocks (Jahn, 1988, Meissl, 1998, Dorren, 2002). The impact of these additional factors on the energy balance of falling rocks is, however, very difficult to determine, and therefore, those factors are not included in the simplistic RockFor^{NET} model.

For falling rocks with d > 0.8 m, our results indicate the limit of the different stands on the present, relatively short slope (325 m). On this slope and under the given terrain characteristics, larger rocks can develop energy values, which cannot be dissipated by most of the simulated stand structures. This finding is in agreement with Rickli et al. (2004), who state that the mitigating effect of stands on rockfall is limited by mass, velocity and thus energy of the falling rocks. As a consequence, technical countermeasures against large rocks are needed to reduce the residual hazard for the highway at the Stotzigwald site. Such countermeasures have indeed been realized in terms of restraining nets, which can dissipate up to 500 kJ (Frei, 2003).

For all rock size classes, the initial stand conditions had a considerable impact on the long-term protective effect. This was confirmed by the four logistic regression models (see above). Thereby, the C-stand, which is currently converted into a selection forest, seems to be the most limited of the three initial stands in terms of a long-term protective effect. This stand only shows acceptable low residual hazards

for S1 and partially for S2 (Fig. 5). The reason for this low performance can very probably be found in the initial bimodal DBH distribution, which shows (i) a low tree density and (ii) a nadir around a DBH of 24-36 cm (Fig. 3). After 60 years, this initial nadir leads to a lack of larger trees by moving to the right side of the DBH distribution. Without large trees in a sufficient number, however, the large energy amounts developed by rocks in size classes S3-S6 cannot be dissipated sufficiently.

The S-stand, which together with the C-stand only showed a very limited protective effect in its initial condition (see above), yields a better performance in the long term, as is evident from the development of the residual hazards compared to the initial residual hazard. This is very probably due to the fact that the initial S-stand shows a considerable tree density with a DBH < 36 cm (Fig. 3), leading to a higher number of large trees after 60 years compared to the C-stand.

The best protective effect after 60 years, however, is generally still provided by the EA-stand, which in its initial conditions shows a high number of larger trees (both > and < 36 cm DBH). In its present, initial state, the EA-stand probably provides an almost optimal protective effect for the Stotziwald site. Over a period of 60 years, however, the protective effect could slightly decrease as indicated in the increase of residual hazards under several simulation scenarios for all rock size classes (see above). Over periods longer than 60 years, the protective effect of the EA-stand is likely to decrease further, since its initial DBH distribution shows a rather low density of small trees (Fig. 3). In combination with the rather sparse level of current tree regeneration in the EA-stand (Tab. 1), the latter will very probably lead to a lack of larger trees in the long term, which in turn will induce an increase of the residual hazard (see above). Thus, when looking at a time period > 60 years, the S-stand with its abundant tree regeneration and its high stand density, particularly for trees with a DBH < 36 cm, is likely to perform better than the EA-stand. This is already indicated in the mean residual hazards for the S-stand after 60 years, which for S1-S4 are either already lower or at least very close to the ones for the EA-stand. However, this assumptions cannot reliably be verified with the present simulation tool, since the accuracy of the projected stand structures obtained with the present ForClim version declines after several decades (cf. Wehrli et al., 2005). Therefore, accurate simulations with ForClim over periods longer than 60 years were not possible in this study.

Key factors for the assessment of the residual hazard with RockFor^{NET}

As can be seen from the strong correlations, the assessment of the residual hazard on the given site with RockFor^{NET} highly depends on two key factors, the *simulated stand density* and the *simulated median DBH*. The influence of these two factors on the residual hazard is thereby depending on the rock size, i.e. it changes with increasing rock size. For small rocks, both factors have a negative influence on the residual hazard, whereby the influence of both factors seems to be of similar importance. This is in agreement with e.g., Jahn (1988), who reports a significant influence of both, DBH and stand density, on the protective effect against small rocks.

For larger rocks, however, the negative influence of the median DBH increases to very high values, whereas the simulated stand density yields a positive correlation with the residual hazard (i.e. the more trees, the higher the hazard). We think, however, that this positive correlation is not causal but mainly due to the strong negative correlation between simulated median DBH and simulated stand density (parameteric correlation: -0.66, non-parametric correlation: -0.52): A high simulated stand density is mainly caused by the recruitment of a large number of young trees, which in turn leads to a decrease of the median DBH. Thus, this finding indicates a weak point of the current RockFor^{NET} version: Up to now, the model uses one single key indicator to represent the DBH distribution of a stand. This indicator, in turn determines the energy that can be dissipated by a tree row (see above). Therefore, the model outcome and by this, the “performance of the stand” is very sensitive to this indicator.

Still, our simulation results are in agreement with the findings from empirical field studies, which report a similar change of importance of the key factors determining the protective effect of a stand against rockfall from tree number (stand density) to the tree size (DBH distribution) from small to larger rocks (Dorren et al., 2005, Kalberer et al., 2005).

Important factors for a high long-term protective effect

In all logistic regression models, *initial stand density*, *regeneration density* and *standard mortality rate* were found to have a significant influence on the long-term protective effect. For S3, an additional factor, the *initial median DBH*, also showed a significant influence.

The influence of *initial stand density* and *standard mortality rate* on the residual hazard was negative in all models, indicating that high initial stand densities as well as a relatively low mortality rate led to a high long-term protective effect. Thus, in contrast to the *simulated stand density*, which only had a negative influence on the residual hazard for the smallest rock size class, the *initial stand density* seemed to have a reducing effect for the residual hazard over all rock size classes. This is not astonishing, however, since after a period of 60 years, the number of surviving large trees is mainly dependent on the *initial stand density*. The density of large trees, however, is a key factor for the assessment of the protective effect of a stand in reality (large trees in the DBH distribution) as well as in RockFor^{NET} (see above). A low tree mortality is thereby essential for a high long-term protective effect, and its influence is even higher for larger rocks, i.e. rock size classes S2-S4, as indicated by the higher B-values for S2-S4 compared to S1 (Tab. 10). A stand can only develop a sustainable long-term protective effect, if few trees are injured by falling rocks, which can lead to an increased mortality (Rickli et al., 2004). Moreover, tree mortality should only moderately be increased by thinning regimes, and large clear cuts should certainly be avoided. Thinning regimes in protection forests should thereby follow specific guidelines, such as the Swiss management guidelines for protection forests (Frehner et al., 2005).

Still, even if a low mortality rate is important for the long-term protective effect, the influence of an increased tree mortality rate can be reduced by several measures. For instance, forest managers can use cut trees as effective rockfall barriers by leaving them lying diagonally on the slope to decelerate and redirect rockfall (Dorren et al., 2005, Frehner et al., 2005). Additionally, dead trees (e.g., snags, cf. Kufperschmid Albisetti, 2003), and even stumps of trees felled at a height of 1.3 m or higher can at least temporary reduce the residual hazard on a site (Dorren et al., 2005, Frehner et al., 2005). Such measures are, however, not taken into account in the simplistic combined simulation tool, and dead trees do not contribute to the protective effect of a stand. Therefore, the influence of the mortality rate on the future protective effect might be slightly overestimated.

In contrast to initial stand density and standard mortality rate, the influence of *the initial regeneration density* was different in the four logistic models: While being negative for S1 as could be expected (higher initial regeneration density leads to a higher stand density and thus, to a lower residual hazard), the influence was found to

be positive for the other rock size classes. This positive influence is rather astonishing and needs some further explanation.

One reason can be found in the relatively short simulation period of 60 years, which is thought to be the temporal limit for accurate simulations with the current ForClim version (cf. Wehrli et al., 2005). Within such a short time period, initial tree regeneration cannot turn into very large trees that can effectively dissipate energy from large falling rocks. Therefore, the potential of these trees to significantly reduce the residual hazard is still limited, i.e., we could expect tree regeneration to have no significant influence at all on the residual hazard for larger rocks after 60 years.

Still, tree regeneration was found to have a positive influence for S2-S4, indicating that the residual hazard increases with increasing initial regeneration density. The reason for these findings is likely to be the same as for the positive correlation found between simulated stand density and residual hazard (see above). The simulated median DBH shows a significant negative correlation with the initial regeneration density (parametric: -0.39, non-parametric: -0.44, $p < 0.01$). This means that an increase of the initial tree regeneration density leads to the recruitment of more young trees and by this, it decreases the simulated median DBH, which in turn increases the residual hazard assessed by RockFor^{NET}. The median of the DBH distribution, which in the present study represents the representative DBH measure needed in RockFor^{NET}, is thereby relatively sensitive to the recruitment of a large number of young trees, and other measures of location would probably show a lower influence. The use of another measure of location is, however, not likely to improve the performance of the model, since it would hardly allow a better representation of a skewed DBH distribution. Thus, the positive influence found in the logistic model is probably an artefact, and it once more points at the weak point of the current RockFor^{NET} version described above.

Still, our findings indicate that particularly the initial stand conditions in terms of initial stand density and a relatively low mortality rate are two key factors for a high protective effect for a given rock size over a period of 60 years. In contrast to this, other variables such as browsing impact or species composition of tree regeneration did not seem to significantly influence the protective effect over the relatively short simulation period. For longer simulation periods, however, these variables as well as the regeneration density of the initial stand could become more important.

Modelling approach

The combined simulation tools used in this model as well as the underlying simulation models include a number of simplifications that need further investigations and could be improved in future studies.

The *model of forest dynamics* should be improved to allow more accurate predictions of different stands for periods longer than 60 years. Therefore, additional data for the parameterization of the recruitment module would be necessary in order to use this module for a continuous supply of tree regeneration over the whole simulation period. Moreover, the representation of the light regime and of tree mortality would probably benefit from further investigations and improvements. The improved model version should finally be intensively validated with long-term data sets. These are, however, difficult to obtain.

The *rockfall model* needs a refinement of its underlying principle, i.e. the virtual representation of stands as distributed "*rockfall curtains"*. Thereby, the representation of the DBH distribution in the model as well as the relationship between the energy, which can be dissipated by the real stand and by virtual tree curtains should be included with more detail. By doing so, RockFor^{NET} will certainly become a interesting tool for forest managers since it already now allows accurate reproductions of important features of rockfall processes and of the interaction with protection forests for certain stand structures.

Finally, the improved models for forest dynamics and rockfall could be combined physically in one single, simple simulation tool, which could be applied by forest managers. Thereby, the effects of falling rocks on trees, e.g. injuries leading to higher mortality rates or stem breakage could additionally be included in the tool as suggested by Dorren (2002).

Conclusions

A combination of dynamic models for forest dynamics and rockfall can prove useful for investigating the effect of forest dynamics on the long-term protective effect of mountain forests against rockfall. Mountain forest dynamics are known to be slow, and therefore, the management of these forests, and by this the control of their protective effect are long-lasting processes. The results of this study showed, that for

investigations on the long-term effects of forest dynamics on the protective effect against rockfall, a time period of 60 years is probably too short.

Still, our findings indicate that over a period of 60 years, the initial stand conditions in terms of initial stand density and a relatively low mortality rate are particularly important for a high protective effect. Therefore, silvicultural measures (e.g. thinning regimes or regeneration harvests) in protection forests should be moderate. To reduce the residual hazard in stands with a rather low tree density such as the C-stand, cut trees could for instance be left on the slope, diagonally to the slope direction (Dorren et al., 2005, Frehner et al., 2005).

Even though we are convinced that the approach presented in this study is promising, it will need further investigations and improvements in future studies. With such an improved combined simulation tool, it should become possible to determine factors that are relevant for a long-term protective effect over a period of more than 100 years. Moreover, target values for these factors could eventually be delivered, which in turn could be used for a more effective and efficient management of mountain protection forests.

Acknowledgements

This study was carried out within the framework of the National Research Programme 48 (Landscapes and Habitats of the Alps) and funded by the Swiss National Science Foundation (grant 4048-064409) and the Swiss Forest Agency. We gratefully thank the European Commission for the Marie Curie Fellowship (QLK5-CT-2002-51705) and for funding the ROCKFOR project (QLK5-CT-2000-01302). Further, we would like to express our gratitude to MeteoSchweiz for providing us with the long-term weather records.

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Section summary

The aim of section III was to develop a prototype of a simplistic, spatially non-explicit combined simulation tool (*CoST*) that allows investigating the effects of forest dynamics on the long-term protective effect against rockfall. This aim was achieved by joining the new ForClim version V2.9.4 (see section II) and the empirical rockfall model RockFor^{NET} (Berger and Dorren, in review). The *CoST* was then applied to a case study to give an example of its use. Based on empirical data, the development of three mountain forests was simulated over a period of 60 years assuming different scenarios. The protective effect of the simulated stands was then assessed by projecting those on a particular site in the Swiss Alps, from where terrain and rock characteristics were available.

The long-term protective effect of the three stands against rockfall was generally high for small rocks, but only limited for larger rocks (diameter d > 0.8 m), indicating the limit of the protective potential of stands on this relatively short (325 m) and steep (45°) slope.

Initial stand conditions, in particular a high initial stand density, as well as a relatively low mortality rate were found to be key factors for a high protective effect over 60 years. Additionally, a high density of tree regeneration in the initial stand was found to increase the long-term protective effect against small rocks (d = 0.2 m). This hints at the importance of tree regeneration for a high long-term protective effect, even if the initial level of tree regeneration was not found to significantly increase the protective effect against larger rocks (d > 0.2 m) after a period of 60 years. One reason for this finding can probably be found in the relatively short simulation period of 60 years, during which initial tree regeneration under shelter could not turn into very large trees. Therefore, its potential of dissipating energy is still limited, and thus, it cannot contribute to a significant reduction of the residual hazard after 60 years. Another reason why a high density of tree regeneration did not increase the long-term protective effect against larger rocks might be the rather vague representation of stand structure within RockFor^{NET}, which currently only uses one measure of location to describe the DBH distribution of a stand. Thus, the latter and also the *CoST* could benefit from a more detailed representation of the DBH distribution.

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Synthesis

Synthesis

Introduction

The aim of this thesis was to develop a combined simulation tool (*CoST*) that allows investigating the effects of mountain forest dynamics on the long-term protective effect against rockfall, with a particular focus on the influence of different levels of tree regeneration on the long-term protective effect. In this synthesis, I first want to present the most important results from the different preceding chapters. I then critically discuss the suitability of the different models for a *CoST* and point out potential improvements of these models. The synthesis ends with some reflective thoughts on the practical relevance of simulation models in investigating and managing the protection forest system, including the findings of this thesis.

Main findings of the present thesis

In section I, the suitability of two promising model candidates for a *CoST* was evaluated. The forest patch model ForClim (Bugmann, 1994; *Paper I*) was thereby shown to allow accurate predictions of important structural forest patterns over several decades, if slight modifications of the establishment and mortality submodels were used. Thus, ForClim was found to be in principle suitable for a simplistic *CoST*, if two of the major shortcomings detected are improved. These shortcomings included (1) the simplistic representation of tree establishment, which led to unrealistically high numbers of young trees, as well as (2) the reproduction of the light competition in the model, which led to a strong overestimation of stress-induced mortality.

In *Paper II*, the process-based rockfall model Rockyfor (Dorren, 2002) was shown to allow accurate predictions of the spatial distribution of rockfall trajectories on three forested slopes with different slope and stand characteristics, based on input data with a resolution of at least 5 m x 5 m. However, Rockyfor underestimated mean impact heights observed on trees at those two sites where high- and medium-resolution input data were available and overestimated them at one site where input data with the lowest resolution data were used. Still, the protective effect of different stands could be assessed and was considerably high on all sites: The number of

rocks reaching the bottom parts of the study sites would, on average, almost triple if the current forest cover were absent. Thus, the present version of Rockyfor is a valuable tool for investigating the protective effect of different stands and, therefore, it can be used for a spatially explicit, 3D *CoST* in its present form.

Given the considerable differences in the performance of the two evaluated models, the suitability of ForClim and Rockyfor for a *CoST* is rather unlike. Therefore, a combination of ForClim and Rockyfor does not seem to be appropriate at the moment: To be suitable for a combination with Rockyfor in a 3D *CoST*, not only the major shortcomings of ForClim detected in *Paper I* should be improved, but the model should additionally become more spatially explicit, i.e. horizontal relationships between individual patches should be taken into account. This, however, could only be achieved with rather extensive and time-consuming modifications, which clearly go beyond the scope of this PhD thesis. Moreover, such modifications are not of prime importance since important characteristics of the protection forest system could be investigated with a simplistic *CoST* that neglects the spatial dimension (e.g., the influence of different levels of tree regeneration on the long-term protective effect). In contrast, the improvement of the major shortcomings of ForClim detected in *Paper I* is mandatory for including the model in a *CoST*. Thus, instead of trying to develop a spatially explicit version of ForClim that could be combined with Rockyfor in a 3D *CoST*, I focused on developing a simplistic *CoST*, based on an improved ForClim version and a simplistic, spatially less explicit rockfall model.

For this purpose, the performance of ForClim was first enhanced in section II (*Paper III*). This was done by improving the reproduction of light competition and by adapting the establishment submodel. Light competition was refined by implementing a simplistic dynamic crown structure, which allows accounting for self-pruning of tree crowns in real stands. The establishment submodel was adapted to the needs of a *CoST* by disaggregating the regeneration process into two stages, seedling establishment and sapling growth. In the latter, sapling growth is modelled explicitly by taking into account two important constraints to sapling growth in mountain forests, namely canopy shading and ungulate browsing. A comparison of the efficacy of the old (ForClim V2.9.3) and new (ForClim V2.9.4) model version revealed that the latter allowed a more realistic reproduction of structural forest dynamics over a multi-decadal period. This, in turn, makes it more appropriate for applications over several

decades (e.g., 60 years), and therefore, the new model version is more suitable for the use in a simplistic *CoST*.

To give an example of the use of the new model version, ForClim V2.9.4 was applied to investigate the ability of a particular mountain forest to provide effective protection against rockfall during a period of 60 years. This forest, called *Stotzigwald,* is located in the Swiss Alps (46°45' N, 08°39' E), and it is very steep with a slope gradient of approximately 45°. It protects one of the most heavily used traffic routes in Switzerland against rockfall. The elevation of the forest ranges from 650 to approx. 1000 m a.s.l., but its protective function is mainly restricted to a zone of approx. 7.5 ha in the lower part of the forest. Rockfall is frequent in this zone, with the majority (79%) of canopy trees showing traces of recent damage. The current stand, which is dominated by *Picea abies* (83%) and *Abies alba* (13%), includes approx. 561 trees ha^{-1} > 4 cm diameter at breast height (DBH), and is thought to provide an almost optimal protective effect against rockfall in its present state. The current level of tree regeneration, however, is alarmingly low with approx. 2230 saplings ha^{-1}, most of them being smaller than 0.4 m in height. Therefore, the development of the long-term protective effect of the *Stotzigwald* is rather dubious. Two scenarios were simulated, namely (i) canopy shading but no browsing impact and (ii) canopy shading and high browsing impact. Under both scenarios, the initial sparse level of tree regeneration affected the long-term protective effect of the forest, which considerably declined during the first 40 years. Still, in the complete absence of browsing, the density of small trees was able to recover after 60 years. In the scenario including browsing, however, the density of small trees remained at very low levels.

In section III, a prototype of a simplistic *CoST* that allows investigating the effects of forest dynamics on the long-term protective effect against rockfall was finally developed. This was done based on the new ForClim version V2.9.4 developed in section II and the empirical rockfall model $RockFor^{NET}$ (Berger and Dorren, in review). The *CoST* was then applied to a case study to give an example of its use. Based on empirical data, the development of three mountain forests was simulated over a period of 60 years assuming different scenarios. The protective effect of the simulated stands was then assessed by projecting those on the Stotzigwald site in the Swiss Alps, from where terrain and rock characteristics were available.

The long-term protective effect of the three stands against rockfall was generally high for small rocks, but only limited for larger rocks (diameter d > 0.8 m), indicating the limit of the protective potential of stands on this relatively short and steep slope (length: 325 m, gradient: 45°). This limited protective potential for larger rock was also confirmed by the findings in *Paper II* (cf. Tab. 4 in *Paper II*).

Initial stand density as well as a relatively low mortality rate were found to be key factors for a high protective effect after 60 years on the Stotzigwald site. These findings, which are in agreement with current expert knowledge, suggest that silvicultural measures in a protection forest such as the Stotzigwald should be moderate (e.g., less than 20% of the trees > 8 cm DBH, cf. indications on mortality rates in *Paper III*). This is essential since a decrease of tree density (e.g., due to thinning) is likely to reduce the protective effect of a stand over several years.

Additionally, a high density of tree regeneration in the initial stand was found to increase the long-term protective effect against small rocks (d = 0.2 m). This hints at the importance of tree regeneration for a high long-term protective effect, even if the initial level of tree regeneration was not found to significantly increase the protective effect against larger rocks (d > 0.2 m) after a period of 60 years. One reason for this finding can probably be found in the relatively short simulation period of 60 years, during which initial tree regeneration under shelter could not turn into large trees. Therefore, their potential of dissipating energy is still limited, and thus, they cannot contribute to a significant reduction of the residual hazard after 60 years. Over a longer period, however, the influence of the initial tree regeneration on the protective effect will steadily increase, until it will finally make the principle contribution to the protective effect of a stand.

Another reason why a high density of tree regeneration did not increase the long-term protective effect against larger rocks might be the vague representation of stand structure within RockFor^{NET}, which currently only uses one measure of location to describe the DBH distribution of a stand. Thus, the latter and also the *CoST* could benefit from a more detailed representation of the DBH distribution.

In contrast to the factors described in the two preceding paragraphs, other variables such as browsing impact or species composition of tree regeneration did not seem to significantly influence the protective effect over the relatively short simulation period. Nevertheless, for longer simulation periods and with an increasing

influence of tree regeneration on the protective effect, these variables are likely to become more important.

The *CoST* presented in *Paper IV* was found to be a promising tool, but it still needs further investigations and improvements in future studies, before being applicable in practice. In the following, I therefore critically discuss the suitability of the different models used for a *CoST*, and I try to point out potential improvements of these models as well as of the whole approach for a *CoST*. Last but not least, I will demonstrate the practical relevance of this thesis, and of simulation models in general, for optimizing the management of forests that protect against rockfall.

Suitability of ForClim for a *CoST* and potential model improvements

As stated in the General Introduction, two model types were candidates for a *CoST*, distance-dependent tree models (*DDTM*) and forest patch models (*FPM*). However, none did completely fulfil the requirements for a *CoST*. In this thesis, I chose a *FPM* since those models seem to be more suitable for a *CoST* (see General Introduction). This is especially true, if one is interested in the long-term dynamics of mixed stands, where the performance of *DDTM* is limited (Bugmann, 2005).

However, the chosen *FPM* ForClim proved not to be the ideal solution. While it is true that ForClim allowed quite accurate predictions of structural forest patterns over several decades after a slight modification of the regeneration and mortality submodels, the model test performed in *Paper I* revealed several serious shortcomings for the use in a *CoST*. Two of these shortcomings have been improved within the context of this thesis in order to make the model suitable for a simplistic *CoST* (*cf. Paper III*). Yet, even the model version V2.9.4 presented in *Paper III* could benefit from further improvements. This could enhance the accuracy of the predictions of structural forest patterns, and enable forecasts of structural dynamics over longer periods (e.g., > 100 years), which would be highly desirable for mountain forests (cf. *Paper IV*). In the following, I propose features of ForClim in need for improvement. The model structure can be found in Fig. 1, and an overview on current research projects on different features of the ForClim model is shown in Tab. 1.

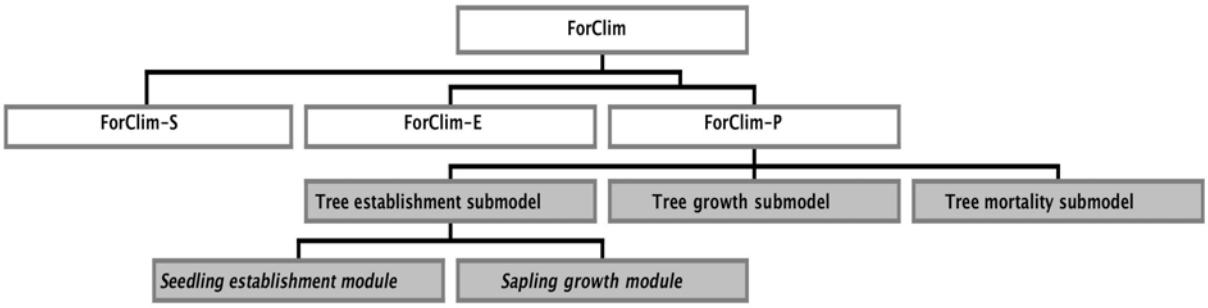

Fig. 1. Model structure of the ForClim model. The submodels of the plant population dynamics submodel ForClim-P are highlighted. Modules, which were introduced in this thesis, are written in italics. ForClim-E: Abiotic environment submodel; ForClim-S: Soil organic matter dynamics submodel (not used in this thesis, cf. *Paper I*).

Tree establishment submodel

Even if the establishment submodel introduced in ForClim V2.9.4 in *Paper III* led to a significant enhancement of the reproduction of tree establishment in the model, it still can be considerably improved. First, the seedling establishment module introduced in the new establishment submodel was neither applied in *Paper III* nor in *Paper IV* since data for a reliable parameterisation and corroboration were not available. While this simplification did not significantly influence the simulated stand structure for a period of 60 years (see *Paper III*), it certainly does so for longer simulation periods. Therefore, this module should be included in simulations over longer periods for a continuous and realistic supply of tree regeneration during the whole simulation period. This, however, is only possible if additional data for its parameterisation will become available. In addition to this, the seedling establishment module should be improved by integrating a more detailed reproduction of seed dispersal. This could be done by taking into account the horizontal relationships between individual patches, as has been done in several recent *FPM* (e.g. Lexer and Hönninger 1998, Jorritsma et al., 1999; cf. Bugmann, 2001 for an overview).

Second, the sapling growth module implemented in ForClim V2.9.4 (*Paper III*) only includes canopy shading and ungulate browsing as limiting factors for sapling growth. There are, however, several other relevant factors such as microsite types and competition with understorey vegetation. The former has already been successfully integrated in other modelling approaches (e.g., Kupferschmid Albisetti, 2003, Kupferschmid et al., 2005, Rammig, 2005), and therefore, an introduction in

FPM seems feasible. As for the competition of saplings with understorey vegetation, Weisberg et al. (2003) introduced a simple approach to include forest understorey dynamics based on empirical relationships between understorey patterns and overstorey diameter structure. The understorey vegetation could then in turn be included as an environmental constraint to sapling growth. However, when including additional constraints, the complexity of the establishment submodel will strongly increase.

Third, sapling mortality due to environmental constraints needs probably to be included if the model is intended to be used for simulations over longer periods (i.e. > 60 years). Before including a sapling mortality module, however, additional long-term data on the survival of saplings under different growing conditions are necessary. This is particularly true for the impact of browsing on sapling mortality, which is still subject to debate (cf. *Paper III*). Therefore, A. Kupferschmid (Forest Ecology, ETH Zurich, pers. comm.) is currently performing an extensive literature review to identify the most relevant factors of the complex causal relationship between browsing pressure and sapling mortality. She then tries to integrate a more realistic reproduction of browsing impacts on saplings in forest patch models (Bugmann, 2005).

Tree growth submodel

The introduction of a dynamic crown structure in *Paper III* is a significant improvement of ForClim, since it allows accounting for self-pruning of tree crowns in real stands. However, as is evident from *Paper IV*, where a correction factor for the dynamic crown structure had to be integrated to allow more realistic growth rates for two stands, the representation of the light regime in ForClim V2.9.4 needs further investigation and corroboration.

Moreover, the accuracy of the growth function under the new dynamic crown should be thoroughly studied. In order to improve the performance of the tree growth submodel in ForClim, C. Heiri (Forest Ecology, ETH Zurich) currently tries to identify the key processes in need of improvement within the growth submodel in her PhD thesis (Heiri, 2005). Thereby, even the inclusion of a 3-dimensional crown structure in ForClim is taken into consideration. In order to find the optimal balance between ForClim's overall approach as a model with a minimum number of ecological assumptions and an accurate reproduction of structural forest patterns over long

periods, M. Didion (Forest Ecology, ETH Zurich) will try to determine the level of complexity needed to accurately simulate tree growth by comparing three different *FPM* in his PhD thesis (ForClim, a *FPM* with a 3D crown and an eco-physiological *FPM*; A. Kupferschmid, Forest Ecology, ETH Zurich, pers. comm.; cf. Didion, 2005).

Tree mortality submodel

The mortality of trees still is a relatively poor known aspect of their ecology (Shugart, 1998, Bigler, 2003), and mortality algorithms in *FPM* have been mostly limited to general relationships because of sparse data on the causal mechanisms of mortality (Keane et al. 2001). However, the performance of these algorithms seems limited (e.g., Bigler, 2003, Risch et al., 2005). This was confirmed by the findings of *Paper I*, where particularly the stress-induced mortality implemented in ForClim was found to largely overestimate the actual mortality rates of the real stands. In *Papers I* and *III,* I therefore introduced two different pragmatic approaches to reduce the stress-induced mortality. By disconnecting tree mortality from tree growth to a certain degree, the accuracy of the tree mortality rate predicted by ForClim was enhanced as evidenced in *Paper I*. Still, the actual reproduction of tree mortality in ForClim did not become more realistic, and tree mortality in ForClim V2.9.4 now mainly depends on the age-related tree mortality. The latter is a mortality routine that serves as a proxy for several causes of death that can happen at any time in the lifetime of a tree (Keane et al., 2001). Thereby, chance plays a major role. As most other *FPM*, ForClim handles this chance by assuming a constant probability of death throughout the lifetime of a tree, ending with 1% of all trees of a species surviving to their maximum age (Bugmann, 1994; cf. Shugart, 1984, Keane et al., 2001). Whether or not such an approach copes with reality is difficult to determine, but the close match between simulated and empirical mortality rates in *Paper I* indicates the use of this simplistic approach.

To achieve a more realistic reproduction of tree mortality in ForClim, additional empirical data on the ecological process of tree death are necessary. For certain tree species, such data have been provided by Bigler (2003) who investigated growth-dependent tree mortality in his PhD thesis. Based on the work by Bigler (2003) and on complementary inventory data, J. Wunder (Forest Ecology, ETH Zurich) intends to implement a more realistic reproduction of tree mortality in ForClim in his PhD thesis

(cf. Wunder, 2004). His thesis should thus contribute to more accurate predictions of structural forest dynamics.

When improving the tree mortality submodel, additional mortality due to episodic events such as natural disturbances (e.g., bark beetle or windthrow) or management (e.g., thinning) should also be taken into account (cf. Shugart, 1984, Keane et al., 2001). ForClim currently allows depicting such processes that are not explicitly modelled by activating a disturbance-related mortality, which kills all trees on a patch if a disturbance occurs (cf. Bugmann, 1984, p.51), or by increasing the age-related mortality rate (cf. *Paper IV*). Nevertheless, the model would certainly benefit from a more detailed representation of these processes.

Spatial explicitness

In addition to the improvements of the population dynamic submodel (ForClim-P), an increase of the spatial explicitness of ForClim would be desirable for a spatially explicit version of a *CoST*. First, the horizontal interactions between patches should be included. Second, the influence of the patch size used in ForClim on the predictions of structural forest dynamics should be further investigated. ForClim originally operates on a patch size of 833 m^2 (1/12 ha, diameter: 32.5 m) without any spatial explicitness within a patch, i.e. the location of individual trees within a patch is unknown. For a spatially explicit *CoST*, this resolution is likely to be insufficient since in the case of uneven tree distributions, the protective effect of such a large patch is difficult to assess. In *Papers III* and *IV*, patch size was therefore reduced to 225 m^2 (patch of 15 m x 15 m) since preliminary studies did not yield significant influences of this reduction of patch size on the predictions of structural forest dynamics. Yet, the relationship between patch size and the predictions of structural forest patterns with ForClim needs further investigations. In his PhD thesis, M. Didion (Forest Ecology, ETH Zurich) therefore tries to determine the influence of patch size on the performance of ForClim (cf. Didion, 2005). Moreover, he intends to assess the significance of explicit accounting for spatial interactions between patches on the model performance (Didion, 2005).

As is evident from this overview (see also Tab.1), the performance of ForClim will be improved in the foreseeable future. Therefore, the model's suitability for a *CoST* is likely to increase. The improved model will, however, first have to be tested in extensive validation tests, similar to those carried out in *Paper I*.

Table 1. Overview on current research projects on different features of the ForClim model (including the results of this thesis). Bold: new feature, italic: improved feature.

Submodel	Model Feature	Reference
Tree establishment (ForClim-P)	*seedling establishment module*	*Paper III*
Tree establishment (ForClim-P)	**sapling growth module**	*Paper III*
Tree establishment (ForClim-P)	**sapling mortality due to browsing**	A. Kupferschmid, ETH Zürich
Tree growth (ForClim-P)	*reproduction of the light competition (dynamic crown)*	*Paper III*
Tree growth (ForClim-P)	*reproduction of tree growth*	C. Heiri, M. Didion, ETH Zürich
Tree mortality (ForClim-P)	*mortality algorithm*	Risch et al., 2005
Tree mortality (ForClim-P)	*reproduction of tree mortality*	J. Wunder, ETH Zürich
ForClim (overall model)	*spatial explicitness: patch size*	M. Didion, ETH Zürich
ForClim (overall model)	**spatial explicitness: horizontal interactions**	M. Didion, ETH Zürich

Alternative models

Instead of using an improved version of ForClim, the use of another existing, but more detailed *FPM* could provide a valuable alternative for a *CoST*. A most promising candidate is the PICUS model developed by Lexer and Hönninger (1998). However, since PICUS and its source code were not available for this thesis, I was not able to assess its suitability for a *CoST*.

In PICUS, the base elements of the simulated forest are cells of 10 m x 10 m x 5 m that contain all information on the distribution of tree biomass in space (Lexer, 2000). PICUS considers interactions among neighbouring patches as well as the effect of slope and orientation on incoming radiation and the shielding effects of the surrounding topography (Lexer, 2000). Together with the 3D crown structure and the implementation of a light model for both direct and diffuse radiation, PICUS could allow more accurate predictions of structural forest patterns as the rather simplistic ForClim model.

Like in the standard version of ForClim, however, tree regeneration is currently only included with little detail. While seed dispersal is explicitly modelled in PICUS 1.2, new trees are initialised with a DBH of approximately 1.0 cm (Lexer and Hönninger, 2001), neglecting the delayed regeneration period in mountain forests.

Thus, in order to be suitable for a *CoST*, the regeneration submodel of PICUS would need similar improvements as the ones included in ForClim (see *Paper III*).

Applicability of ForClim to other sites

In this thesis, the main focus was on mountain forests dominated by *Picea abies* since these represent a large fraction of the protection forests in Switzerland, and since many of these currently have a low level of tree regeneration (Brändli and Herold, 1999, Brang and Duc, 2002, Zinggeler et al., 1999). When the user is aware of certain shortcomings in the current model version (e.g., limited simulation duration, correction factor for the dynamic crown structure for certain stands, no explicit representation of large-scale disturbances and management; see above and *Papers III* and *IV*), ForClim could probably be applied to other such forests dominated by *Picea abies* for the time being.

On condition that (i) the necessary data for the parameterization of the establishment submodel could be obtained and (ii) the improvements described above would lead to more accurate predictions of structural forest patterns over longer time periods, an application to other mountain forest types seems possible. ForClim was originally developed for investigating successional patterns in mountain forests under different climatic scenarios (Bugmann, 1994), and therefore, it is not likely to be restricted to forests dominated by *Picea abies*.

Suitability of the chosen rockfall models for a *CoST* and potential improvements

In contrast to the model of forest dynamics, the two rockfall models used in this thesis can be used for a *CoST* in their present form (Rockyfor, see *Paper II*) or with moderate modifications (RockFor^{NET}, see *Paper IV*). This is probably why both models are currently used in practice, i.e. to assess the protective effect of forested slopes (L.K.A. Dorren, Cemagref Grenoble, pers. comm.). Nevertheless, both models could be optimised.

The process-based rockfall model Rockyfor

An improvement of the accuracy of Rockyfor is likely to be achieved by the improvement of the accuracy of the input data. As stated in *Paper II*, high-resolution input data, e.g. a digital elevation model (DEM) derived from laser scanning, as well as a better knowledge of active rockfall source areas and on initial fall heights would probably lead to a considerable improvement of the accuracy of model predictions.

A further improvement of the model is probably only feasible with excessive additional data on soil and terrain characteristics and their influence on bouncing rocks (i.e. more detailed data on the coefficient of restitution, cf. *Paper II*). These data are, however difficult to obtain, and therefore, it may not be worth trying to integrate a more detailed reproduction of soil and terrain characteristics in Rockyfor. Still, the model could probably be improved by (i) explicitly integrating dead tree trunks and stumps (but this could be done by means of the laser scan DEM) as well as (ii) by including a more detailed reproduction of the relation between DBH and effective tree height, i.e. the height where a tree with a given DBH can still effectively dissipate a certain amount of energy. This would allow taking into account stumps of felled trees. However, such improvements will need a certain amount of additional data, and therefore, the gain in model accuracy could be rather low in relation to the effort needed.

The empirical rockfall model RockFor^{NET}

As stated in *Paper IV*, RockFor^{NET} would benefit from a more detailed reproduction of the stand structure within the model. Up to now, all trees in a row have a diameter equal to the given representative DBH (i.e. a measure of location describing the DBH distribution), which in combination with the tree species determines the efficacy of a tree row regarding energy dissipation during a rockfall impact (see *Paper IV*). The representation of stand structure could be improved in a simplistic manner by (i) describing the DBH distribution by more than one measure of location and by (ii) randomly determining the DBH of each single tree in a tree row. While increasing the model accuracy, such simple improvements would not unnecessarily complicate the simplistic approach of RockFor^{NET}. By keeping RockFor^{NET} simple, even an improved, more accurate version could still be used in practice.

Applicability of the rockfall models to other sites

As is evident from *Paper II*, Rockyfor can be easily applied to different sites without any problems, as long as highly resolved input data can be provided. With the increasing availability of laser scan data, this requirement will certainly be fulfilled. Once improved, RockFor^{NET}, should also be applicable to any site. This has, however, first to be tested once the model has been improved.

Nevertheless, the applicability of both models is likely to be limited for *very small rocks* (e.g., rocks with a diameter of 0.1 m or even smaller; cf. *Paper IV*). This is not astonishing since for an effective protection against such small rocks, other factors that are only coarsely represented or even neglected in the models such as terrain characteristics (e.g., surface roughness) or other vegetation types (e.g., bushes) are probably of prime importance. The present knowledge on the influence of these factors on rockfall processes is still rather limited. Therefore, such features have not been sufficiently integrated in rockfall models, which, in turn imposes limits to the accurate modelling of very small rocks.

Still, it is questionable if additional studies on these features should be carried out since such studies would be rather laborious due to the extensive data sets needed on terrain characteristics (see above). Moreover, the gain in knowledge would probably not be decisive for the utility of the models in practical applications since the residual hazard for very small rocks can easily be mitigated by relatively low-dimensional technical measures.

Applicability of the presented *CoST* to other sites

The restrictions of the underlying models described above limit the applicability of the *CoST* presented in this thesis. Thus, when the user is aware of the described shortcomings (see above and *Papers III* and *IV*), the current *CoST* can in principle be applied to mountain forests dominated by *Picea abies* and *Abies alba* for roughly assessing the development of the protective effect over a period of approximately 60 years. To apply the *CoST*, the following input data are needed:

1. data on stand and regeneration (information for each individual tree and sapling regarding size [DBH, height] and species, if possible spatially explicit. Additional long-term data on stand development could help to parameterize the correction factor for the dynamic crown structure, if needed; cf. Tab. 6 in *Paper IV*)

2. data on terrain and rock characteristics (cliff height, slope length between cliff and forested slope, slope length of forested slope, mean slope gradient, mean rock diameter, rock density; cf. Tab. 5 in *Paper IV*).

As is evident, the data needed for the *CoST* is not excessive. Therefore, the data availability should not be too restrictive. However, since ForClim and RockFor^{NET} were not physically combined in the present *CoST*, the application of the latter is rather laborious and time-consuming. It would therefore be desirable to physically combine the two underlying simulation models in a single simulation tool.

Moreover, the application of the *CoST* for management use is to some extent limited by the simplicity of the underlying models, which neglect some features that could be important for forest managers. For instance, dead trees are simply removed from the model. By doing so, neither the potential protective effect of tree stumps nor of lying trees is taken into account when assessing the residual hazard (see *Paper IV* and below). Thus, before using the *CoST* as a management tool, different features of the underlying models should first be improved or completed as indicated above. Thereby, it would also be desirable to include the effects of falling rocks on trees. For instance, injuries leading to higher mortality rates or stem breakage could additionally be included in the tool as suggested by Dorren (2002). For the former, however, further investigations on the relationship between tree impacts by falling rocks and tree mortality would be necessary.

Whether or not the spatial explicitness of the *CoST* should be increased by e.g., combining ForClim with Rockyfor, depends on the intended use. For most purposes, a simplistic tool, which only needs a few input parameters, would probably be sufficient. However, if the tool would be used to evaluate management options at specific sites, a version that is spatially more explicit could be advantageous. To do so, however, ForClim would first need to become more spatially explicit (see above).

The use of simulation models in investigating and managing the protection forest system

„Every model is wrong, but some models are useful“ (Box, 1979)

The protection forest system is very difficult to study let alone to manage since the progressive forces (e.g., tree regeneration, tree growth) of mountain forests dynamics are very slow, and the destructive forces (e.g., snow avalanches, windstorm) sometimes sudden and violent. Therefore, the exact influence of many silvicultural measures on the long-term protective effect of a stand is unknown or at least uncertain, which in turn makes a successful management very difficult (Brang et al., 2004). Given these difficulties, simulation models provide most valuable tools to investigate this system and to gain knowledge for optimizing the management of protection forests (Dorren et al., 2004). Models always represent a strong simplification of reality, which in turn makes them most useful, since it allows identifying the most important factors within a system (Bugmann, 2005). Once these key factors are identified, the relationships between them can be investigated in detail. Moreover, the knowledge gained on these factors as well as on the whole system can be used to improve the management of the system, i.e. for increasing the efficiency and efficacy of silvicultural measures in protection forests.

In the following, I will recapitulate the current knowledge on the protective effect of forests against rockfall with a particular focus on the interaction of the different factors involved in rockfall processes on forested slopes. Thereby, I will give some examples of how models can be used to investigate the influence of different factors on the protective effect of a stand against rockfall. Additionally, I will present some thoughts on how the management of protection forests can be improved by the use of simulation models.

The influence of different factors on the protective effect of a stand

Rockfall on forested slopes is a highly stochastic process, which involves different factors such as size, spatial distribution and density of trees, slope length and gradient, size and shape of falling rocks, and many more. These factors can be grouped in three fields, (i) forest stand, (ii) terrain characteristics, and (iii) rock characteristics, and they interact in a highly complex manner. This makes it difficult to

determine let alone to quantify the protective effect of a stand. Since empirical data on rockfall on forested slopes are difficult to obtain (but see, e.g. Perret et al., 2004 or Stoffel et al., 2005 for indirect sampling methods), modelling approaches are helpful for gaining additional knowledge on the interactions of these different factors. This not only helps to understand the dynamics of rockfall on forested slopes, but it also allows assessing the protective effect of a stand in an accurate manner.

Several authors state that forest stands generally have a decelerating and mitigating effect on rockfall (e.g., Jahn, 1988, Rickli et al., 2004, Dorren et al. 2005). For instance, Dorren et al. (2005) found a significant reduction of bounce height and velocity in their extensive comparison of real-size rockfall experiments on a non-forested and a forested site on the same slope. While the mitigating effect of trees on the distribution, frequency and magnitude of rockfall events is evident (Jahn, 1988, Dorren et al. 2004), the protective effect of a stand in terms of a reduction of the hazard strongly depends on the interaction of the three factor fields mentioned above, and differs from site to site. For example, Dorren et al. (2005) report a relatively high protective effect of the stand at their study site in the French Alps. On their slope (slope gradient approx. 38°), the residual rockfall hazard, i.e. the number of rocks that surpass a certain zone, decreased by 63% compared to the non-forested site. However, under different *terrain characteristics*, such as a steeper or a shorter slope, the same stand might only have a modest protective effect, leading to an unacceptably high residual hazard. In the following, the influence of different factors on the protective effect of a stand against rockfall is discussed.

Terrain characteristics

The influence of terrain characteristics on the protective effect of a stand was very evident on the relatively short and steep forested slope at the Stotzigwald site (length of the forested slope ca. 325 m, mean slope gradient 45°, cf. *Paper IV*). For this site, the simulation results in *Paper IV* indicated that due to the high energy that can be developed by larger rocks (diameter > 0.8 m), the protective effect of any stand against such rocks is likely to be limited. However, if the length of the forested slope would for instance be twice as long as it is, the protective effect of some stands would certainly be considerably higher. The same is true if the slope would be less steep. In general, the protective effect is rather limited in case of a positive energy balance of a falling rock, i.e. if the energy a falling rock can develop on a slope is not

likely to be dissipated by the stand on this slope. Thereby, simplistic rockfall models such as RockFor^{NET}, which include major terrain characteristics (e.g., slope length, slope gradient), provide useful tools for assessing the potential protective effect of stands under different terrains conditions.

Other *terrain characteristics* that are known to significantly influence rockfall processes include (i) surface roughness (structural features that influence the velocity of falling rocks, e.g. lying rocks, but also other vegetation types, e.g. bushes) and (ii) surface dampening (ground or soil characteristics, which determine the amount of energy that can be absorbed during a bounce of a rock; e.g., thick needle cover on the ground). *Forest roads* provide an impressive example of how such characteristics can influence rockfall processes. For instance, Dorren et al. (2005) reported that forest roads effectively stopped 13-15% of the falling rocks on slopes with 35.5-38°. A similar effect was observed at the Stotzigwald site in December 2002, when a large rockslide was stopped on a forest road and left a crater of approximately 1 m depth (Markus Tschopp, Canton Uri, pers. comm.). However, even if such factors are known to sometimes significantly reduce the residual rockfall hazard, their influence on rockfall processes is very difficult to assess. Therefore, they are often only roughly included (e.g., Rockyfor, cf. *Paper II*) or even neglected (e.g., RockFor^{NET}, cf. *Paper IV*) in rockfall simulation models (see above).

Rock characteristics

In addition to terrain characteristics, *rock characteristics* (e.g., rock size or shape) have a most significant effect on the protective effect of any stand. Whereas *rock shape* is thought to have a certain influence on the rockfall process on forested slopes (cf. Frehner et al., 2005), *rock size* is generally the most important factor for determining the protective effect of a stand: For small rocks, tree density rather than tree size (DBH) is important, since many trees are required to increase the impact probability (Jahn, 1988, Dorren et al., 2005; see also *Paper IV*). The larger the rocks are, however, the larger the required mean tree diameter is. As a rule of thumb, Schwitter (1998) suggested that the mean DBH of trees in a stand should be around 1/3 of the decisive size of the falling rocks. This rule of thumb has been confirmed by the findings of Dorren et al. (2005) for their site. In their real-size experiments, they additionally observed that the number of impacts against trees was generally more important than the efficacy of the impact expressed in the amount of dissipated

energy. Therefore, they concluded, that for a high protective effect, a *"large number of trees is more important than having thick trees only"* (Dorren et al., 2005).

Rock size can additionally impose a limit on the protective effect of a stand. Rickli et al. (2004) assume that an effective mitigation of rockfall processes by forest stands is limited to a rock mass of approximately 10'000 kg and a velocity of approximately 20 m s^{-1}. Unfortunately, they do neither mention a slope gradient nor a length of the forested slope as a frame for their estimation. Thus, whereas there is no doubt that such an upper limit (i.e. a *maximum rock size*) exists, the values given by Rickli et al. (2004) do not necessarily represent absolute limits. The latter, however, could easily be assessed for each stand by the use of simplistic models such as $RockFor^{NET}$.

Rock / tree-interaction

The extensive data set on the interaction of falling rocks and trees that was generated by the experiments of Dorren et al. (2005) led to a significant improvement of the understanding of the rock / tree-interaction during rockfall (cf. Dorren and Berger, 2006). Their results and the findings of the tree stability project at the Swiss Federal Institute WSL/SLF (cf. Lundström, 2003, Lundström et al., 2005), where the energy absorption process during rock impacts for single trees was investigated, nowadays allow a realistic assessment of the amount of energy that can be dissipated by living trees during rockfall impacts (cf. Dorren et al., 2005, Lundström et al., 2005, Dorren and Berger, 2006). This knowledge has been used to improve the 3D simulation model Rockyfor (Dorren et al., 2005; cf. *Paper II*), and to develop the empirical rockfall model $RockFor^{NET}$ (Berger and Dorren, in review; cf. *Paper IV*).

Using models for improving the management of rockfall protection forests

Mountain forests are dynamic systems that are subject to change, and therefore, they cannot be maintained in a certain "desired" condition over a long period (Brang, 2001). The management of protection forests is therefore often a trade-off between improving and maintaining the current protective effect and increasing the protective effect in a long term. Thereby, forest managers always have to keep in mind the long-term perspective ("*how does the stand, and by this the protective effect, develop over e.g., 10 years, 50 years, 100 years?*"). This is particularly important, since the management of mountain forests is a long-lasting process, and the measures that we take today are likely to influence the protective effect of mountain forests for more

than 100 years. For instance, the fact that tree regeneration growing under shelter needs a very long time to provide effective protection against larger rocks (e.g. more than 60 years for the stands simulated in *Paper IV*), indicates that silvicultural measures, which promote regeneration, should be initiated in due time.

Recent management guidelines for protection forests (e.g., Frehner et al., 2005) take both aspects, i.e. the short- and the long-term protective effect, into account by suggesting (i) requirement profiles for natural hazards (e.g. minimal tree densities needed, maximal gap sizes, etc.), as well as (ii) requirement profiles for a long-term perspective (e.g., target values for tree regeneration, which are thought to be sufficient for a long-term effective protection). Therefore, such guidelines can support forest managers to determine optimal silvicultural measures, even if in some parts, they are rather based on expert knowledge than on scientific facts. This drawback, however, can be gradually reduced by periodically including the current scientific state-of-the-art on the protection forest system in such guidelines. In order to close existing gaps in knowledge and to improve these guidelines, simulation tools can be very useful. In the following, I will present a few examples of how simulation models can help to improve these guidelines or how they could be used as a completion to them.

Using models to determine the call for action

The call for action in a protection forest is often difficult to determine. Some studies indicate that effective protection is possible without any management measures for a certain period and under certain conditions (e.g., Frey and Thee, 2002, Kupferschmid Albisetti, 2003, 2004, Schönenberger et al., 2005). Such results can let raise questions on the legitimacy of the management of protection forests. In my eyes, however, much additional data on the development of the protective effect of such forests over longer periods is necessary in order to derive general rules. As long as these data do not exist, active management of protection forests following specific management guidelines might be more appropriate than renunciation of any management. This is particularly true for Swiss mountain forests dominated by *Picea abies*, which currently are rather dense with a sparse level of tree regeneration (cf. Brang and Duc, 2002). The development of the protective effect of these forests is rather dubious and a *“no management strategy”* would be likely to increase this uncertainty.

In order to determine the call for action in a given stand, simulation models can be used complementary to current management guidelines. For instance, models such as Rockyfor or RockFor^{NET} allow a more accurate, site-specific assessment of the current protective effect of a stand than management guidelines, since they take into account important features such as terrain characteristics. By applying such a model, a forest manager can easily assess the residual rockfall hazard on a site and estimate, if the current stand can provide effective protection or if additional silvicultural or technical measures are necessary. With the target values included in current management guidelines (e.g. Frehner et al., 2005), however, such an assessment is rather difficult since those do not take into account terrain characteristics.

Using models to optimize existing management guidelines

Simulation models can be used to improve the requirement profiles for natural hazards included in management guidelines. The use of models generally allows a more flexible application of the potential protective effect of a stand than management guidelines. For rockfall, for instance, the latter often only take into account trees with a certain “effective DBH”, i.e. a pre-defined DBH threshold above which trees are thought to provide effective protection (e.g. Frehner et al., 2005: only target values for trees > 12 cm DBH are considered). This might lead to an underestimation of the real protective effect of a stand since trees can provide a certain protection even before they reach such a DBH threshold. In simulation models, however, the protective effect of smaller trees can be considered, which in turn allows a more accurate assessment of the protective effect. By accurately determining the protective effect of a stand, simulation models allow “designing” optimal stand structures for given terrain and rock characteristics. This, in turn, could be used as site-specific target values for stand structures.

Furthermore, simulation tools such as the *CoST* developed in this thesis, which take into account the temporal dimension, could be used for a coarse estimate of the influence of alternative management options. For instance, the *CoST* could be applied to minimize the trade-off between maintaining the current protective effect of a stand and increasing the protective effect in the long term. This could be done by simulating stand development under several mortality rates in order to reproduce different thinning regimes (cf. *Paper IV*). By doing so, it should be possible to

estimate whether or not a planned measure is likely to (i) strongly decrease the protective effect of the stand and (ii) if it is sufficient to effectively promote tree regeneration. In case the trade-off between (i) and (ii) is thought to become too large, alternative measures, which are known to increase the protective effect of a stand could be taken into account. For instance, cut trees could be used as effective rockfall barriers by leaving them lying diagonally on the slope to decelerate and redirect rockfall (Dorren et al., 2005, Frehner et al., 2005). Additionally, stumps of trees felled at a height of 1.3 m or higher could be used for a temporary reduction of the residual hazard on a site (cf. Dorren et al., 2005, Frehner et al., 2005). As mentioned above, however, the current *CoST* does not allow taking such measures into account due to the complete removal of dead trees.

Once the underlying models in the *CoST* are improved, the latter could finally be used to derive more accurate site-specific target values for tree regeneration as the ones included in the current version of the Swiss management guidelines for protection forests (Frehner et al., 2005).

Using models for risk assessment and decision-support

Simulation models allow a more effective and efficient risk management. For instance, simulation models such as Rockyfor are most valuable to determine optimal combinations of technical and silvicultural measures for a given site. By doing so, they allow determining locations for technical measures where these relatively expensive measures are most effective or absolutely necessary. Furthermore, an accurate assessment of the protective effect of a stand by rockfall models could allow reducing the dimensions of technical measures at some sites (Dorren et al., 2005), which in turn can lower the costs.

Moreover, a model such as Rockyfor could be used as a decision-support tool. For example, if thinning is performed to promote regeneration, the protective effect is likely to decrease temporary and locally. The application of an accurate spatially explicit rockfall model thereby allows determining the optimal location for such an intervention, i.e. the location with the least increase of the residual hazard.

The use of current models as a "*risk assessment tool"* is, however, restricted up to date, since such models generally neglect alternative measures that can be taken by forest managers to reduce the residual hazard (e.g., lying tress, cf. above). Therefore, the models are likely to underestimate the real protective effect of a stand

in case a manager takes such measures. In terms of safety aspects, this drawback does not need to be problematic since the protective effect is rather underestimated. Still, an improvement of the models would be helpful to optimize their use as a decision-support tool, which allows determining cost efficient and effective measures. Whereas possible improvements in this respect have already been mentioned above for Rockyfor, an improvement of the *CoST* could for instance include a simple routine determining if a dead tree is taken out of a stand and replaced by a stump or if it is substituted by a lying trunk. Such a routine, however, seems to be more functional in a 3D *CoST*.

Final remarks: Future contribution of *CoST*s to research and management

As is evident from the few examples presented above, simulation models can be very useful for studying the protection forest system, as well as for improving the management of protection forests. This thesis presents a small step on the long way heading for an optimal management of forests that protect against rockfall. The scientific prototype of a combined simulation tool (*CoST*) offers a first possibility to investigate the protection forest system on a long-term perspective. Once the underlying models and by this the *CoST* are improved, the latter could become ready to be used for practical applications, i.e. it could be used as an accurate decision-support tool for an effective long-term forest management. In this way, the restricted financial means for managing protection forests could probably be applied more effectively and efficiently.

However, an optimal management of protection forests is still far away and much additional knowledge on the protection forest system is necessary to reach this aim. Thereby, it could be very useful to include additional factors, which currently are not included in the *CoST*, such as economic or technical aspects as well as other natural hazards (e.g., snow avalanches) or disturbances (e.g., windthrow, bark beetle). By doing so, a comparative evaluation of different management strategies could finally become possible. The comprehensive approach introduced by Brang et al. (2004, in review) could be a promising way to enable such evaluations, and by this, to further improve the management of protection forests.

Last but not least, one should always remember that due to the underlying simplification of reality, simulation tools will hardly ever replace empirical studies and

knowledge. Therefore, simulation tools such as a *CoST* or the model developed by Brang et al. (2004, in review) should always only be used complementary to expert knowledge.

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MERCI !!!

This thesis would not have been possible without the help of many persons. Thus, I would like to express my gratitude to

- Prof. Dr. H.R. Heinimann for his support and interest in my work.
- Dr. W. Schönenberger and Dr. P. Brang for the supervision of my thesis, the freedom they gave me to carry out the study and for many fruitful discussions.
- Prof. Dr. H. Bugmann, Dr. habil. A. Huth, and Dr. L. Fahse for their “back-up modelling support” and for a great deal of input in this thesis.
- my franco-dutch connection, Dr. L. Dorren and Dr. F. Berger for the great collaboration we had on rockfall (modelling).
- A. Zingg and H. Bachofen for providing me with data on different forests.
- Ch. Angst, A. Bürgi, B. Commarmot, M. Giamboni, B. Höwecke, H. Häsler, C. Heiri, M. Keller, D. Köchli, A. Kupferschmid, R. Kühne, N. Oesterling, S. Perret, B. Reineking, M. Stoffel, P. Thee, E. Thürig, M. Tschopp, A. Walther, J. Wunder, and many other members of the ETH Zürich, the WSL, and the F+D.
- the UFZ Leipzig for their warm support during my stay there, in particular to W. Eckart, M. and U. Müller, G. Nagel, S. Kramer, G. Wasserberg, Ch. Wissel, and T. Wissmann.
- the Swiss National Foundation (National Research Programme 48; grant 4048-064409), the Swiss Forest Agency (F+D), and the Swiss Agency for Education and Science (Marie Curie Fellowship, grant no. 02.0392) for funding my research.

Very special thanks go to my parents, who made my studies possible.

Finally, my heart goes out to the Giver of all blessings and to my family for their unending patience. Thank you for your endurance...

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