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MODELLING WIND AND BARK BEETLE DISTURBANCES AT FOREST STAND SCALE IN THE AUSTRIAN ALPS

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Preface

This thesis is based on three papers. They can be found in the Appendix (sections 9.1-9.3). There are differences in the formatting of the articles due to the requirements of the particular journals.

Sections 1-6 provide a framework and a summary of the rationale for each of the journal papers. The specific methods, results and their discussion can be found in a detailed form in the respective articles.

Please, quote as Pasztor F (2014) Modelling wind and bark beetle disturbance at forest stand scale in the Austrian Alps. Dissertation. University of Natural Resources and Life Sciences (BOKU), Vienna. p. 96 or refer to the individual papers.

List of papers

I) Pasztor F, Matulla C, Rammer W, Lexer MJ (2014) Drivers of the bark beetle disturbance regime in Alpine forests in Austria. For Ecol Manage. 318:349-358. doi: 10.1016/j.foreco.2014.01.044

II) Pasztor F, Matulla C, Zuvela-Aloise M, Rammer W, Lexer MJ (2014) Developing predictive models of wind damage in Austrian forests. Ann For Sci. (in revision)

III) Pasztor F, Rammer W, Lexer MJ (2014) Comparative analysis of bark beetle and wind disturbance models within a dynamic forest simulation model. Manuscript.

Abstract

Wind and bark beetle disturbances have a big impact on Austrian forests, leading to substantial loss in timber value and other ecosystem services. Intensifying disturbance regimes are expected in the future due to climate change. A better understanding of interrelationships of damage from disturbances and forest characteristics, management and climate is required to sustain the provisioning of demanded ecosystem services from forests and to allow targeted adaptation of silviculture and forest management to future conditions. The purpose of this work was to examine relationships between wind and bark beetle disturbances and forest stand, site and weather properties at forest stand scale. Two different approaches were employed in this thesis. First, statistical models were developed from an empirical database compiled of forest management plans, harvest records and weather data covering an area of more than 40,000 ha of forest land in various parts of the Eastern Alps in Austria. Second, the established empirical disturbance models were then integrated into a dynamic ecosystem model, and a simulation-based comparative analysis of model behaviour as affected by different disturbance sub-models was conducted.

Results of the study revealed that wind and bark beetle disturbance events at stand scale can be identified well by means of statistical modelling (generalized linear mixed modelling approach). However, the intensity of damage in case of a disturbance event could not be explained satisfactorily in the models. The comparative analysis of the empirical wind and bark beetle models integrated into a dynamic ecosystem model indicated plausible and consistent behaviour under current climatic conditions and revealed limitations of the empirical models when a changing climate was considered. These findings emphasized the need for generalized process-based disturbance modules in long-term ecosystem simulations.

Keywords: wind; bark beetle; disturbance; stand scale; Austrian Alps

Contents

1 Introduction

There is a growing awareness of the importance of both biotic and abiotic natural disturbances in European forests (Seidl et al. 2011). In Austria, wind and bark beetles are the two most detrimental disturbance agents (see Fig. 1). They can pose a threat to not only to the timber-based value chains in the forest sector but a wide range of ecosystem services like water protection (Emelko et al. 2011), landslide and avalanche protection (Brang et al. 2008) or carbon sequestration (Thürig et al. 2005, Seidl et al. 2008).

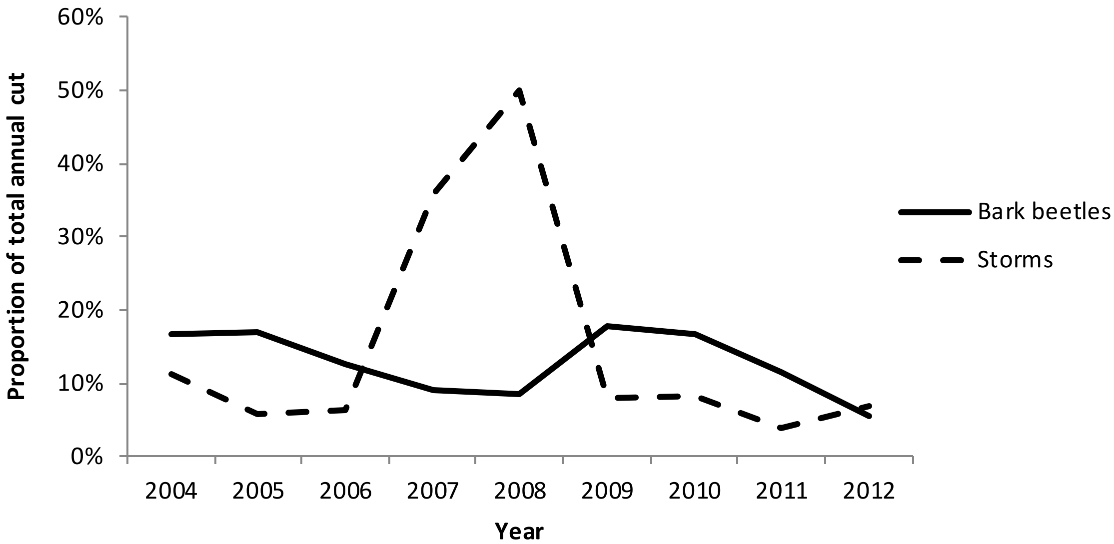

Fig. 1 Proportion of wind and bark beetle salvage in the annual national harvest records in Austria (Anonymous 2005-2013)

Storm damage can occur in the form of uprooting and stem breakage. Wind gusts, which are short, very fast air movements, occur frequently during storms and are a major factor in causing storm damage. They can have a cumulative effect on trees, as the first few gusts can cause structural damages in a tree (often not visible to human eye), and then even a lower-speed gust can make the tree windthrown (Schütz et al. 2006). There are differences among trees considering their vulnerability to wind. One important attribute is the surface and structure of the crown, which behaves as a catchment for the moving air. The other important attribute is the expansion of the root system, which works as a tether for the tree against the wind. That is why trees with big crowns and shallow root systems are more prone to uprooting (Schaetzl et al. 1989). On the other hand, if the roots have a solid grip, then the main stem will rather break. Trees generally have a critical wind speed above which they get uprooted (Gardiner et al. 2000). This is also true for stem breakage; critical wind speeds are higher in this case. However, at stand level the vulnerability of trees to wind also depends on the other trees surrounding them (crown contact increases stability; Quine et al. 1995). Windthrown trees in stands can also fall over other trees causing a domino-effect, and therefore more damage. In the mountainous topography and climate of Austria, wind storms are rather frequent and many of them cause tree damage at various extent. Norway spruce (Picea abies (L.) karst.) is considered as one of the most vulnerable tree species to wind in the Central European region (Hanewinkel et al. 2013).

Norway spruce trees are also highly vulnerable to the European spruce bark beetle (*Ips typographus* [L.]) that is responsible for most of the bark beetle damage in Austrian forests (Krehan and Steyrer 2004, 2005, 2006). Overall, *I. typographus* is the most destructive biotic threat to Austrian forests, which is partly due to the wide spatial range of its host (i.e. Norway spruce) in the country and also to its ability to reach gradation levels easily when environmental conditions are favourable. *I. typographus* uses Norway spruce trees under stress (inter alia drought, heat waves, damage caused by logging, wind, pollution) as habitat, but in case of a gradation, even healthy trees get infested (Mulock and Christiansen 1986). Large-scale windthrow events typically lead to wide availability of breeding habitat for bark beetles (Marini et al. 2013). Therefore, the interaction between storm and bark beetle disturbances is an important factor in the ecosystem dynamics of the Austrian forests (Thom et al. 2013).

There are various implications of disturbances for forest management. Gaps and stand edges are frequently formed in forests by timber harvesting and stand regeneration, and increase the vulnerability of forest stands to windthrow (Schütz et al. 2006). Also, in case of bark beetles, forest management can have impact on the disturbance regime, as there are several forest stand characteristics (e.g. species composition, open stand edges, tree vitality) which affect the vulnerability of stands and are controllable by forest management (Kautz et al. 2013; Thom et al. 2013). However, a big difference is that in case of bark beetles there are also possibilities to control the disturbance agent itself, by e.g. removing potential breeding habitat (i.e. vulnerable and infested trees) from the stands to prevent the spread (Schroeder and Lindelöw 2002).

As concerns about the potential effects of climate change on forest ecosystems grow (see Dale et al. 2001), a better understanding of the interrelationship of forest stand and site characteristics, forest management and climate-sensitive disturbances is required. In recent years, various scientific studies have been conducted on the wind and bark beetle disturbance regimes of Central Europe. Beside experimental work, the two main approaches were empirical modelling and process-based modelling. While empirical modelling of observational data can be used to discover potential driving factors of wind (e.g. Hanewinkel et al. 2008; Klopcic et al. 2009; Schindler et al. 2009; Schmidt et al. 2010; Thom et al. 2013) and bark beetle disturbances (e.g. Overbeck and Schmidt 2012; Marini et al. 2012; Stadelmann et al. 2013; Thom et al. 2013; Mezei et al. 2014), their use in scenario-based analysis of potential future forest development is based on the assumption that future disturbance events will occur under circumstances similar to those found for past events. On the other hand, process-based models mimic underlying ecological processes, and as these processes are universally defined, they can be used also under novel conditions (e.g. climate change). Nevertheless, there is still much uncertainty of underlying processes due to knowledge gaps and heterogeneity and spatio-temporal dynamics in forest landscapes. In recent years, process-based models of bark beetle disturbances in Central Europe have been developed (e.g. Baier et al. 2007; Seidl et al. 2007; Fahse and Heurich 2011; Temperli et al. 2013). On the other hand, available mechanistic wind damage models from boreal and atlantic regions in Europe (Peltola et al. 1999; Gardiner et al. 2000; Ancelin et al. 2004; Schelhaas et al. 2007) have not been evaluated for Central European conditions yet.

Despite the long history of wind and bark beetle damage of forests in the Eastern Alps of Austria, studies analysing the quantitative relationship of these disturbances and their potential driving factors in the region are scarce. Lexer (1995) developed a bark beetle hazard rating model for Norway spruce stands using detailed stand- and site-related data from a limited number of forest stands. This model served as a basis for subsequent bark beetle damage modelling in Lexer and Hönninger (1998) and Seidl et al. (2007), with more focus on the process-based modelling of bark beetle phenology. Thom et al. (2013) made a comprehensive analysis of bark beetle and wind damage in Austrian forests at regional scale, emphasizing the difference between slow, predisposing factors and fast, inciting factors. A comprehensive quantitative analysis based on stand level data of forest stand, site, weather and forest management, covering a large area with various environmental conditions has not been made so far.

Such an analysis could contribute to giving insight to individual disturbance events, their variability in time and space and the interactions among multiple disturbance events and agents and also the correlations with internal and external driving factors (Seidl et al. 2011).

2 Objectives

The aim of this thesis was to develop quantitative models of the relationship of the occurrence of wind and bark beetle disturbance events and their intensity in Austrian forests and the characteristics of the forests at stand level. As a prerequisite, the compilation of a database with large spatial and temporal coverage including stand and site attributes, harvest and weather data was required (see specific papers in Appendix, sections 9.1 and 9.2). Beyond the purpose of developing empirical disturbance models to discover the driving factors of the disturbance events, the focus was also on scrutinizing the implications of using such disturbance models in the frame of a dynamic ecosystem model (see specific paper in Appendix, section 9.3).

3 Material and methods

3.1 Data

The database used for the analysis was created from various data types (see Appendix, sections 9.1 and 9.2). The major source of the forest related data were management plans from forest management units of the Austrian federal Forests (AFF). These 10-year plans contain an inventory of the forest stands at the beginning of the planning period, and stand properties like species shares, age, yield class and timber stock volume of the stand and site attributes like altitude, slope steepness, aspect and site type (defined by the classification system of the Austrian Federal Forests – AFF) were derived from them. Data were provided for approximately 8,000 forest stands covering an area of approximately 40,000 ha in four forest management units (FMU). The four FMUs were chosen for the study in a way that the major prevailing site and stand types of Austrian production forests were present. The weather data set was prepared by the Austrian Central Institute for Meteorology and Geodynamics (ZAMG) and consisted of time series data of daily maximum and minimum temperatures, precipitation, vapour pressure deficit, global radiation and daily maximum ten-minute mean wind speeds. Among others, the potential number of bark beetle generations per year and frozen soil status were calculated from the climate variables to enhance the database. Finally, the database was expanded by harvest records, in which the annual timber removals in each stand and the reason for the harvest were provided (naming the disturbance agent in case of salvage). The completed database consisted of one record for each of the ten years of a forest stand, stating the actual stand, site and weather properties and the timber removals of that year arranged into wind, snow and bark beetle disturbance salvage and regular harvests. Also, the aggregated amount of removals in the previous four years was calculated and arranged accordingly.

3.2 Analysis of empirical data

Statistical models were fitted to the empirical data set to quantify the relationship of wind and bark beetle disturbance events and related damage intensities and forest stand characteristics and weather-related site attributes. Annual probabilities of disturbance events were modelled by logistic regression in a framework of a generalized linear mixed model (GLMM). Data records classified as “damage” were used as input for modelling annual damage intensities by linear regression in a linear mixed model framework (LMM). Both, event probabilities and intensities were modelled separately for wind and bark beetle disturbances. However, timber removals due to disturbances and regular harvests in preceding years (up to a period of four years prior to any given year) were considered in the analysis. For a detailed description of the models, see Appendix, sections 9.1 and 9.2.

A mixed model framework was used for model development because of the hierarchical structure of the data set. Spatial units (sub-compartments; stands) were nested within bigger units (compartments, districts, FMUs) and carried a 10-year time series data set. Thus, spatial and temporal autocorrelation of model residuals could lead to bias in model estimates (Pinheiro and Bates 2000). To overcome this, random effects were added to the model equations. The correlation structure of observations in the same data cluster (in our case sub-compartment, compartment and district) was considered, and thus the effects of confounding variables related to these clusters could be estimated (i.e. the random effects) and separated from the scrutinised explanatory variables (i.e. the fixed effects).

To find the set of explanatory variables that carried the most information regarding the response variable, an information theory (IT) approach was used. There are a growing number of ecological studies following this instead of the frequentist approach (Johnson and Omland 2004). Instead of using significance levels for the explanatory variables, it scrutinizes the amount of information that a set of explanatory variable holds with regard to the outcome. The most widely used score to describe this information content is the Akaike Information Criterion (AIC), which also penalizes for the number of explanatory variables in a model (Akaike 1973). This method of model selection is increasingly advocated for predictive models in Ecology (see e.g. Burnham et al. 2011; Hegyi and Garamszegi 2011). Models with different explanatory variables were compared by their AIC values, and the model with the lowest AIC was chosen for further analysis. The probability models were evaluated with sensitivity, specificity and the area under receiver operator characteristic curve (AUC) and residual analysis. The intensity models were evaluated with the coefficient of determination (R^2), the root mean squared error (RMSE) and residual analysis. Versions of the models that included the fixed effects only (the random effects were excluded) were also evaluated to assess how well the fixed effect variables could explain the disturbance events and intensities in a general context without using the local implicit information related to the four FMUs as represented by the random effects.

3.3 Comparative analysis of disturbance models

The empirical models created in the study were integrated into the dynamic ecosystem model PICUS v1.6, and a simulation experiment was designed in order to assess the mid- to long-term implications of different disturbance models for simulated ecosystem dynamics. For a detailed description of the model, see Lexer and Hönninger (2001) and Seidl et al. (2005). Overall, the comparative analysis involved (i) a model version without any disturbance models, (ii) the original spruce bark beetle disturbance model BBDM-1 (Lexer and Hönninger 1998, Seidl et al. 2007), (iii) the new bark beetle model BBDM-2 (see section 3.2) and (iv) the combined use of the new bark beetle (BBDM-2) and wind disturbance models (WDM) (see section 3.2). For details, see Appendix, section 9.3.

The different model versions were employed to run 100-year simulations for site and stand conditions (i) covering typical conditions in the Eastern Alps, (ii) including both pure Norway spruce and mixed species stands, (iii) considering an altitudinal gradient and (iv) considering potential effects of a transient climate change. The behaviour of the different model variants was compared using mean total productivity, standing timber stock and mean annual damage. Furthermore, simulated disturbance regimes were compared against an empirical database which had been compiled from the calibration dataset. An in-depth description of the simulation setup and details on the empirical data set are given in Appendix, section 9.3.

4 Results

4.1 Empirical models

Both the wind and the bark beetle disturbance probability model showed good results considering the AUC values of 0.84 and 0.88, respectively. When evaluating the fixed effect-only models, the bark beetle model retained an AUC score of 0.80. On the other hand, the AUC value of the wind model decreased slightly to 0.71. This is still considered acceptable (Hosmer et al. 2013); however it indicates the higher joint effect of the variables that were represented by the random effects in the model. To see how well the models could distinguish between the events (i.e. disturbances) and non-events (i.e. no disturbance occurred in a given year), a cut-off value analysis was done identifying the cut-off value where the number of events and the number of non-events of the model predictions were the closest to the number of events and the number of non-events in the dataset of the observations. Non-events could be predicted with good reliability (specificity values being 0.95), while sensitivity was much lower at values of 0.26 for wind and 0.29 for bark beetles (fixed effects only). It means that around one fourth of the disturbance events were identified correctly by the models in the original dataset used for model fitting.

For the intensity models, low R^2 and high RMSE values indicated a rather poor performance. In case of the fixed effect-only version of the models, values of R^2=0.09 for wind and R^2=0.13 for bark beetles showed that the explanatory power of the models were limited. Among the explanatory variables, timber stock volume and natural disturbances in the previous four years had the biggest impact on the predictions in general. For details on the effect sizes of the explanatory variables and a description of the disturbance regimes, see Appendix, sections 9.1 and 9.2.

4.2 Comparative analysis of disturbance models

Results of the simulation experiment yielded a contrasting behaviour of the different model variants. Differences in timber stock volumes between the simulations using different disturbance models were apparent. Compared to the model version without any disturbance model, consideration of disturbances led to substantial loss of timber in the simulations and subsequently lower standing stock, except when BBDM-2 was used as the only disturbance model in the simulations. The difference in this case was not significant. Under the baseline climate scenario, both the original (BBDM-1) and the new bark beetle model (BBDM-2) had similar results at intermediate and high altitudes, but differed substantially at low altitude, where BBDM-1 showed much higher damage due to bark beetles. Here, it is interesting to note that low-altitude sites had not been represented well in the calibration dataset of BBDM-2. When compared to a s et of stands from the empirical database (see section 3.1), all model versions provided plausible results compared to observations. However, differences between runs with different disturbance models were substantial when a t ransient climate change scenario was considered in the simulations. While the new wind (WDM) and bark beetle models (BBDM-2) produced results similar to those of the baseline climate scenario, the earlier bark beetle disturbance model (BBDM-1) predicted higher timber losses by the end of the 21st century, mainly at low altitude sites. For details of the analysis results, see Appendix section 9.3.

5 Discussion

5.1 Empirical modelling of disturbance regimes

Hierarchical data are a common feature of many ecological studies. The availability of mixed modelling techniques allows considering various issues that arise from spatial and temporal autocorrelation and to avoid or at least diminish unwanted effects (Zuur et al 2008; Bolker et al. 2009). The choice of GLMM and LMM for the current analysis proved to be successful. In case of logistic regression models, the area under the receiver operating characteristic curve (AUC) has become a widely used index to test classification performance. It can serve as basis for comparison between models, as its value is independent from any subjectively set threshold. However, it does not provide information in a separate form on how well events and non-events are predicted by a model, therefore it can be beneficial to use various cut-off values and look at the respective sensitivity and specificity values, as well. When a cut-off value that resulted in similar numbers of events and non-events in the predictions and observations was chosen, the probability models could identify approximately one fourth of the events in the observations while identifying almost all non-events correctly. These results imply that forest stands with low risk can be identified easily, which constitutes the major part of the forest landscape. Depending on the risk control policy, forest management can implement measures in those stands that were classified as “high-risk” (i.e. predicted to have disturbance-related damage). If these measures are financially less demanding than potential losses from damaged timber, then a higher share of false positive predictions may be acceptable. Of course, this decision will vary depending on the decision environment and the risk perception of decision makers (see Gardiner and Quine 2000). If it is deemed more important to discover all high-risk stands at an acceptable level of misclassified low-risk (i.e. non-event) stands, the cut-off value needs to be selected accordingly.

Developing models for the intensity of disturbance events was less successful. The big differences in R^2 (explained share in variation of the response variable) between the full models (fixed and random effects) and the fixed effects-only models indicated that the predictor variables used in the study explained only a small proportion of the variance in both bark beetle and wind salvage and that the major share of variation was accounted for by implicit relationships represented by the locality. Therefore, a comprehensive analysis of potential driving factors that were not considered in this study should be focused on in the future. However, this is a difficult task, as data describing these additional factors might be unavailable for large areas in the required high spatial and temporal resolution, or data collection may be too costly. Hence, potential means of acquiring such data also need to be discovered.

Stand attributes characterizing the forest structure were found to have an important effect on both wind and bark beetle damage probability and intensity (standing timber stock, age, share of Norway spruce). Standing timber stock was significantly correlated with damage probability and intensity in both disturbance models. Thom et al. (2013) also found that growing stock was a significant predictor variable for both wind and bark beetle damages. Netherer and Nopp-Mayr (2005) and Klopcic et al. (2009) consider growing stock as a relevant variable to explain the wind and bark beetle disturbance regime, as well. How can these findings be interpreted in terms of potential silvicultural mitigation measures? The conclusion, that reducing the growing stock in mature stands will reduce damage risk, may be too simplistic. It is rather the opposite effect which is to be expected, at least with regard to wind disturbance, as such measures will increase canopy roughness. The effect of standing timber stock in the model rather points at the fact that tending measures in the forest stands have not been implemented properly in the past, and thus tree attributes such as slenderness, crown geometry and tree vitality may have developed unfavourably.

Decreasing the share of Norway spruce or the rotation period (stand age) appears also as a viable option to positively affect slow, predisposing stand attributes. On the other hand, the importance of previous disturbances highlighted that the spatial structure of a stand after the salvage of trees may be just as crucial regarding future disturbance events. This confirmed the important interactions between disturbance agents in the region (Thom et al. 2013). Although effect sizes of weather-related variables such as number of bark beetle generations and maximum gust speed were rather low in the models, their informative nature ("significance" in the frequentist approach) indicated the importance of external drivers of the disturbance regimes (Raffa et al. 2008). For a detailed discussion of the limitations of the models and implications for forest management, see Appendix, sections 9.1 and 9.2.

5.2 Using empirical disturbance models in dynamic ecosystem simulations

The comparative analysis of the dynamic simulations involving different disturbance models highlighted the implications of the different functioning of the models when a system-level change (warmer climate) was considered in the simulations. In this regard, the climate-sensitive terms in the model equations were of great importance. The empirical bark beetle disturbance model did not show any significant difference in the estimated damage between climate scenarios despite the higher numbers of potential bark beetle generations in a warmer climate. This is contrary to the common expectation of rising bark beetle damage in warmer climatic conditions in the scientific literature (e.g. Seidl et al. 2009; Ogris and Jurc 2010; Marini et al. 2012; Stadelmann et al. 2013). A similar behaviour was revealed in response to an altitudinal gradient in the simulation setup. A crucial issue in bark beetle disturbance models is the effect of proactive forest protection measures which are usually not explicitly considered but have the potential to significantly affect the beetle disturbance regime. Considering wind damage in the simulations showed the importance of the role that additional disturbance agents can potentially have in shaping the structural characteristics of forests (Franklin et al. 2002) as primary and secondary cause. The empirical models BBDM-2 and WDM provided the unique opportunity to analyse the interaction effects of different disturbance agents. For detailed results and the discussion of the simulations, see Appendix section 9.3.

6 Conclusions

The importance of predictor variables describing vegetation structure indicated that forest management has possibilities to mitigate the effects of bark beetle as well as wind disturbances and also to promote measures to adapt to a future climate. However, this underpins the need for stand tending programmes, which will become effective on mid- to long term only. Another finding of the empirical disturbance modelling was that earlier preceding salvage cuts increase the probability of new damage. Although it is not a new finding, it confirms earlier studies consistently, and it is based on a large empirical database. Self-reinforcing processes can hence intensify the disturbance regimes, emphasizing the importance of stand stability and resilience in forest management.

It was shown that standard data from management plans can be used to identify stands at high risk of bark beetle or wind damage. However, expectations that salvage volumes at stand level can be accurately projected from such data seems to be too optimistic. There was a big difference in performance between models with fixed and random effects and models without random effects, which highlighted the importance of unknown attributes that need to be discovered in order to improve disturbance modelling. For instance, the limited improvement in model performance by including wind speed data points was a weak point in the modelling process; nevertheless it indicated that there is potential to further improve empirical wind disturbance models by improved matching of damage events and weather data.

Overall, it is concluded that scenario analysis of effects of climate change and adapted management on forest development and related ecosystem services must take into account disturbances when delivering useful information in forest management decision support is required. Because of the various feedback relationships in the simulated ecosystem, approaches to estimate risk or predisposition indices without explicit simulation of disturbance events are considered inappropriate.

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9 Appendix

9.1 Paper I

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Drivers of the bark beetle disturbance regime in Alpine forests in Austria

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ABSTRACT

Bark beetles are the major biotic disturbance factor threatening Norway spruce (*Picea abies* [L.] karst.) forests in Austria. The increase of bark beetle salvage after large storm damages is well known. However, over the recent two decades salvage from bark beetle damages in Austria has increased and varied between 0.6 and 3.0 million m^3, where at regional scale a complex interplay of initial beetle population density, forest conditions, weather phenomena such as drought periods, various other disturbance agents such as snow and storm and forest management have been hypothesized as major determinants. This points at the need to develop tools to assess the risks of damage from bark beetle disturbances at the operational scale of forest stands, so that adaptation measures can be developed and implemented in a targeted approach. In the current analysis, binomial generalized linear mixed models (GLMMs) were used to assess the effects of site, stand and climate conditions on the probability, and linear mixed models (LMMs) for the intensity of bark beetle disturbance events at forest stand level. The database used for model development combined 10-year forest management plans and related harvest records of four management units of the Austrian Federal Forests covering in total more than 40,000 ha of forest, and a gridded climate data set provided by the Austrian Central Institution for Meteorology and Geodynamics. In the models, timber stock volume and previous disturbances had the largest impacts on bark beetle damage. Potential bark beetle generations estimated from a beetle phenology model were also a useful predictor. While the model of disturbance probability correctly classified 90% of all cases in the data set (specificity 95%, sensitivity 29%), the model for damage intensity explained only low shares of the variation in the recorded damage data (full model R^2 = 0.45, fixed-effects-only model R^2 = 0.13; cross validation in the four forest management units yielded similar R^2 values). Benefits and limitations of the data set are critically discussed and conclusions for operational forest management are drawn.

1. Introduction

Between 1958 and 2001 an average of 2.9 million cubic metres of timber was damaged annually by bark beetles (Scolytidae) in Europe, with an average increase of 5.31% per year during that period (Schelhaas et al., 2003). European spruce bark beetle (*Ips typographus* (L.)) is the most destructive biotic threat to Norway spruce (*Picea abies* [L.] karst.) forests in Europe (Christiansen and Bakke, 1988). In Austria, salvage due to bark beetle damages fluctuated between approximately 0.6 and 3.0 million m^3 of timber per year in the period 1992–2012, which was approximately 4–18% of the total annual cut (e.g. Steyrer and Krehan, 2009; Anonymous, 2010–2013). This huge share of salvage from bark beetle infestations, mainly in Norway spruce forests, was correlated with dry and unusually warm climate and, depending on the region, also with storm damages, thus confirming the potential future risks related to climate change (Lindner et al., 2010). Salvage from bark beetle infestations was determined by the interplay of population levels of bark beetles, forest conditions, drought periods, occasional storm and snow damages and how focused forest management responded to the emerging regional disturbance dynamics with proactive protection measures and sanitation cuts (Steyrer and Krehan, 2009). Warmer climate will favour the development of insects such as bark beetles and allow more frequently the completion of two and even three insect life cycles per year leading to potentially rapid population build-up and subsequent damages in host trees (Baier et al., 2007; Jönsson and Bärring, 2011). Simultaneously, frequency and severity of drought periods may increase, as well, impacting negatively on tree vigour and consequently increase the vulnerability to bark beetle infestation (Wermelinger, 2004; Marini et al., 2013). While Norway spruce forests at low elevation sites naturally supporting mixed broadleaved stands have

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been a hotspot of bark beetle damages for decades, an intensified bark beetle disturbance regime is already observed at higher elevations in mountainous spruce forests, as well (Lausch et al., 2011; Marini et al., 2012; Mezei et al., 2014).

Empirical studies have confirmed earlier theoretical concepts (e.g. Berryman, 1976) and experimental work (e.g. Mulock and Christiansen, 1986) and have established general relationships between specific forest stand attributes and the susceptibility of the stands to bark beetles. Regarding the European spruce bark beetle, the most obvious structural feature is the availability of Norway spruce, i.e. the host tree, at larger dimensions (i.e. mature stands; Eriksson et al., 2005; Zolubas et al., 2009; Schroeder, 2010). The high susceptibility of pure spruce stands compared to spruce stands with admixed tree species is shown by numerous studies (e.g. Klopcic et al., 2009; Ogris and Jurc, 2010; Overbeck and Schmidt, 2012; Hlásny and Turčáni, 2013). Norway spruce trees under physiological stress such as from drought, fungal diseases or intense competition in overly dense stands are particularly suitable habitat for bark beetles (e.g. Wiener, 1988; Lexer, 1995). In experimental studies, it has been found that also the spatio-temporal availability of susceptible trees within stands may affect infestation risk and damage intensity (see Fettig et al., 2007; Becker and Schröter, 2000). Canopy closure is among the few stand structural attributes which are influential on bark beetle susceptibility (e.g. Netherer and Nopp-Mayr, 2005), indicating that (i) open stands with very low canopy closure are susceptible due to exposure of trees to direct radiation and related physiological stress, and (ii) higher bark temperatures favour the development of beetles compared to closed canopy stands.

Damages are not only an economic loss regarding timber value, but also lead to increases in harvesting costs and may cause further follow-up costs regarding planting, tending and other silvicultural measures. Furthermore, there may also be negative effects on other forest ecosystem services such as carbon sequestration (Seidl et al., 2008a) or the protection of infrastructure and settlements from rockfall and avalanches in mountainous landscapes (Brang et al., 2006).

Summarizing, intensifying disturbance regimes are a growing challenge for forest management when aiming at the sustainable provision of ecosystem services (Lindner et al., 2010). Hence, since the 1990s there are an emerging number of scientific studies which aim at the investigation of bark beetle disturbances and the interplay with other disturbance factors (see Seidl et al., 2011) to better understand the disturbance processes and to develop useful management tools which allow identifying high-risk conditions for a targeted management strategy.

Empirical studies are built on salvage records kept by forest enterprises (e.g. Hanewinkel et al., 2008; Klopcic et al., 2009; Overbeck and Schmidt, 2012), regional to national scale damage statistics (Stadelmann et al., 2013; Thom et al., 2013) or damage estimates derived with remote sensing (Hais and Kučera, 2008; Jakus et al., 2011; Lausch et al., 2011). However, there are limitations in all approaches. Standard management data do not place emphasis on stand and site variables, and consequently, variables that may explain the damages are scarce. However, a huge advantage of management records is that they report also small damages of a few m^3 of timber only and potentially provide large spatial coverage. Available remote sensing based damage data are usually confined to large-scale damage events and fail to inform about low-intensity damages. Moreover, the cause of damage frequently remains unclear. Statistics based on regional damage monitoring systems lack the operational scale of specific forest conditions and rely usually on some kind of qualitative damage assessment procedures (e.g. Steyrer et al., 2011). In contrast, detailed damage inventories, which have been implemented recently, provide accurate local data; however, spatial coverage is limited (e.g. Seidl et al., 2007).

There are also model-based studies which aim to explore the effects of stand structure and composition as driven by management or the impacts of climate change on bark beetle disturbance regimes. These studies usually link bark beetle phenology models driven by climate data and forest attributes either measured in the field or simulated by forest ecosystem models (e.g. Seidl et al., 2007; Fahse and Heurich, 2011; Jönsson et al., 2012; Temperli et al., 2013). These model-based approaches are valuable to improve understanding of potential future developments of disturbance regimes. However, to develop such disturbance models, empirical damage data are required, as well.

Therefore, our overall objective in the current contribution was to identify driving factors of the bark beetle disturbance regime in Norway spruce forests at stand scale, employing an empirical database considering stand and site characteristics, climate data, forest management and other disturbance agents. Emphasis was on the inclusion of climate sensitive process-based predictors of the bark beetle disturbance regime. Specific objectives were the development of statistical models to (i) explain the occurrence of bark beetle damage events, and (ii) to estimate the intensity of the damage.

2. Material and methods

2.1. Study area

In Austrian forests, Norway spruce (*P. abies* [L.] karst.) has an area share of approximately 70% (Schieler and Schadauer, 2011). Forests dominated by this species occur in all ecoregions of Austria from elevations as low as 400 m a.s.l. on sites, where naturally mixed broadleaved forests would prevail to mountain forests at 1800 m a.s.l, where Norway spruce is the main naturally dominating species. For the current analysis, data from four management units (FMU) of the Austrian Federal Forests (AFF) being responsible for the management of federal public forests in Austria were available. The four FMUs were Traun-Innviertel (TRV), Steyrtal (STT), Waldviertel-Voralpen (WVV) and Steiermark (STMK) (Fig. 1), and the area under consideration was approximately 40,790 hectares of forest.

Norway spruce is the most abundant tree species with varying shares of other conifers (*Abies alba* Mill., *Larix decidua* Mill., *Pinus sylvestris* [L.]) in the four FMUs. *Fagus sylvatica* [L.] is the main broadleaved species while other admixed broadleaves have just minor shares of up to 5%. Altitude varies among the FMUs and ranges from submontane (400–800 m a.s.l.) to high montane (approximately 1600 m a.s.l.) vegetation belts (Table 1). In all the FMUs of the study area, the age class distribution is fairly balanced; rotation lengths of the main species are between 100 and 140 years. Forest area of the AFF is divided into units for administration purposes at different spatial levels. The four regional FMUs consist of smaller units, i.e. districts that usually have an area of a few thousand hectares and can be seen as sub-regions in a geographical sense. A district contains several tens of compartments, an administrational unit that – in a mountainous environment – follows major topographic borders with similar site and climate properties inside its area. However, the smallest operational unit is the sub-compartment (i.e. stand), usually several hectares in area. Sub-compartments aim at homogeneous site and stand conditions and are the basic silvicultural planning and treatment unit. In Austria, 10-year management plans are prepared at district level.

All four FMUs had been hit by storm damages in 2007 and 2008 with subsequent increases in bark beetle salvage 2–3 years later. During the beetle gradation after storm damages regular harvests almost completely stopped or were at very low levels. The FMU Steyrtal (STT) suffered from above average storm damage also in

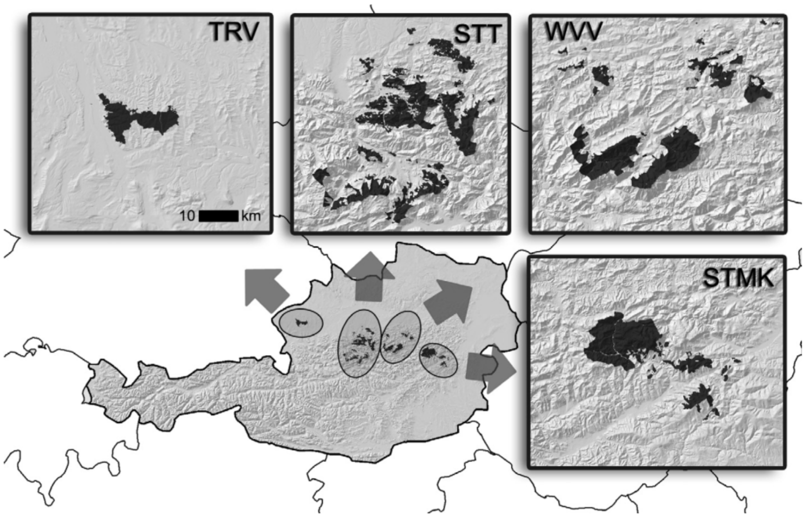

Fig. 1. Location of the four forest management units under study in Austria. TRV = Traun-Innviertel, STT = Steyrtal, WVV = Waldviertel-Voralpen, STMK = Steiermark.

Table 1
Characteristics of the districts from four AFF management units which were available for the analysis. Sub-compartments of age <20 years were excluded. TRV = FMU Traun-Innviertel, STT = FMU Steyrtal, WVV = FMU Waldviertel-Voralpen, STMK = FMU Steiermark. Species shares are based on volume.

Forest management unit	TRV	STT	WVV	STMK
Altitudinal range (m a.s.l.)	500–700	400–1,600	500–1,600	600–1,600
Bedrock	Acidic	Calcareous & flysch	Calcareous	Acidic & calcareous
Forest area (ha)	4,780.4	20,275.6	4,243.1	11,493.3
Number of districts	2	9	2	5
Number of compartments	126	721	145	388
Number of sub-compartments	929	4,062	943	2,238
Picea abies (%)	73	54	67	80
Fagus sylvatica (%)	16	30	17	7
Other conifers (%)	7	11	13	12
Other broadleaves (%)	4	5	3	1

2003. In all other FMUs under study, bark beetle salvage between 1992 and 2008 was at relatively low levels up to around 10% of annual cuts.

2.2. Database

In building the database, the stand and site description of the recently completed 10-year management plans of selected districts within the four FMUs, the related harvest records as well as a gridded climate data set covering the FMUs under study were combined.

2.2.1. Forest management plans

The 10-year management plans from 18 districts available for this study were all embedded in the period 1992–2010. The reason for this spread is that the AFF planning division requires several years to renew the district level plans in all FMUs. Management plans typically provide limited detail regarding forest stand conditions and include attributes such as yield class of the most abundant tree species in a stand, volume and age by species and eventually a qualitative description of the mixture type. No further reliable information on stand structure was available. Stands with an age of less than 20 years were not included in the analysis data set as such young stands were considered not to be vulnerable to *Ips typographus* infestations (e.g. Eriksson et al., 2005).

Since the management plans contain only the initial state of stands at the beginning of the respective 10-year planning period, the development of stand attributes (in our case timber stock volume) over time was projected using yield tables and the removals from the harvest records.

Regarding the site conditions, altitude, slope, aspect and site type from the AFF site classification system were available at sub-compartment level. Preliminary analysis with aspect as a multi-level factor (8 directions) showed high contrasts between direction groups of SE, S and SW as one group and all the remaining directions as the other. Hence, for a simpler inference in the analysis aspect was coded binomially. For each compartment an estimate for water holding capacity (WHC) was derived from the database collected by Seidl et al. (2008b) based on the available site descriptors.

2.2.2. Harvest records

Harvest records contained all timber harvests specifying the year of harvest, total extracted volume quantity (not species specific) and the reason for the removals. Harvested volume was multiplied by 1.2 when related to standing stock to account for harvest residues (Pretzsch, 2010). Harvest data distinguished regular harvests and salvage including the damaging agent. However, no indication of the spatial distribution of the fellings inside the sub-compartments was included in these records. Within a sub-compartment timber harvests for the same reason within a given year were aggregated. Salvage due to wind, snow and bark beetles and regular harvests were each cumulated for periods of up to four years prior to any year of the 10-year planning period. This shrinked the 10-year plans to 6-year analysis periods, excluding the first four years.

2.2.3. Climate data and related information

Climate data were available at a grid with 1 km resolution covering all four FMUs and were linked to forest compartment centroids using GIS software (ESRI, 2012). The data set included daily mean, maximum and minimum temperature, precipitation, vapour pressure deficit and global radiation. Sub-compartments within a compartment shared the same climate data.

Employing the phenology model as described in Seidl et al. (2007), the number of *I. typographus* generations in any given year was calculated based on temperature and daylength data. The core approach is based on the PHENIPS phenology model (Baier et al., 2007) and incorporates swarming in spring and the development of main and filial generations. Swarming starts when both a daylength and an air temperature threshold are met. Brood development depends on bark temperature sums; if the required heat sum of 557 °C (above a threshold of 8.3 °C) is reached, a new filial sister brood is started (Netherer and Pennerstorfer, 2001). The bark temperature is calculated using an empirical relationship taking air temperature, incoming global radiation and the relative radiation below the canopy into account. Beetle development stops in autumn when the day length drops below 14.5 h or when temperature requirements are no longer met. A more detailed account of the implemented phenology model is given in Seidl et al. (2007) and Baier et al. (2007).

To assess water availability to trees, a water balance at monthly resolution was calculated for each sub-compartment for any year of the 10-year data set, and the relation of actual evapotranspiration (AET) to potential evapotranspiration (PET) from May to August was taken as a proxy for soil water supply (compare Lexer and Hönninger, 2001; Seidl et al., 2007). All stand-level attributes available for the analysis are shown in Table 2.

2.3. General modelling approach

The modelling process consisted of two main steps. The initial step was to establish models for the probability of a bark beetle damage event by logistic regression. The second step was modelling the damage intensity, given that a damage event occurred, by linear regression analysis.

Before fitting the models, Pearson correlation coefficients among the continuous explanatory variables were calculated. YC was strongly correlated with several other variables (e.g. $r_P = 0.69$ for VOL, −0.54 for ALT), which could lead to multi-collinearity (see Zuur et al., 2008). YC was therefore not used together with such variables in model fitting.

For model development, a mixed model framework was used because of the hierarchical structure of the data set. Spatial units (sub-compartments) were nested within bigger units (compartments, districts, FMUs) and carried a 10-year time series data set, therefore spatial and temporal autocorrelation of model residuals could lead to bias in model estimates (Pinheiro and Bates, 2000). With the use of a mixed model framework, random effects were added to the model equations. These random effects induced a simple correlation structure for observations in the same data cluster (in our case sub-compartment, compartment and district), and therefore the effects of confounding variables related to these clusters could be estimated (i.e. the random effects) and separated from the actually studied predictor variables (i.e. the fixed effects) in the analysis.

All possible combinations of the candidate predictor variables were used in an automated procedure to fit model equations to the data set for both the probability and the intensity models. Resulting models were then compared regarding their Akaike Information Criterion (AIC) values (Akaike, 1973). This method not only rewards the goodness-of-fit of a model, but also penalizes for increasing number of predictors. Ultimately, the model with the lowest AIC value was chosen for further posterior analysis. Following Arnold (2010), the penalty for the inclusion of one additional parameter was 2 AIC units. If model deviance was not reduced by an amount sufficient to overcome the 2-unit penalty and, hence, the additional parameter provided no net reduction in AIC, that variable was excluded from the model. Simply put, the uninformative parameter did not explain enough variation to justify its inclusion in the model.

In the model fitting process, annual salvage due to bark beetle damage on a per hectare basis was the response variable. Variables W_t, B_t, S_t and R_t were natural log transformed. With regard to these variables, as a preliminary analysis we manually fitted the models

Table 2
Available stand-level attributes for statistical modelling of bark beetle disturbances in the study area. Median and standard deviation (SD) values derived for the full data set. Median and SD not provided for time series attributes.

Variable	Unit	Median	SD	Description
ALT	M	900	257	Altitude above sea level
SL	°	25	9	Slope steepness
ASP	[0,1]	–	–	Aspect [1: SE, S, SW; 0: W, NW, N, NE, E]
SITE	Nominal	–	–	21 Site types from the site classification system of the Austrian Federal Forests; used in different groupings according to bedrock (calcareous, acidic, flysch), water and nutrient status
YC	$m^3\ ha^{-1}\ year^{-1}$	8	2.4	Yield class of the main species of the stand, mean volume production per ha and year over a period of 100 years
PA	%	70	30.3	Share of Norway spruce (*Picea abies* [L.] Karst.)
AGE	Years	90	42.1	Mean stand age
VOL	$m^3\ ha^{-1}$	310.6	151.8	Timber stock volume before harvests of actual year
W_t	$m^3\ ha^{-1}\ (t)\ years^{-1}$	–	–	Wind damage in previous (t) years, $t = [1–4]$
B_t	$m^3\ ha^{-1}\ (t)\ years^{-1}$	–	–	Bark beetle damage in previous (t) years, $t = [1–4]$
S_t	$m^3\ ha^{-1}\ (t)\ years^{-1}$	–	–	Snow damage in previous (t) years, $t = [1–4]$
R_t	$m^3\ ha^{-1}\ (t)\ years^{-1}$	–	–	Regular harvests in previous (t) years, $t = [1–4]$
$BGEN_t$	$N\ year^{-1}$	–	–	Potential annual bark beetle generations in the previous (t) years (including filial generations), $t = [1–4]$
TAV_m	°C	–	–	Monthly mean temperature
MAT	°C	7.7	1.5	Mean annual temperature
SMI	[0–1]	0.01	0.006	Soil moisture index (May–August); share of potential stand water demand which can be satisfied by available soil water

with $t = 4$ first, then did the same with the 1-, 2- and 3-year versions. In general, extending the number of preceding years improved the fit of the models. Considering a 4-year legacy period proved to be the best compromise between increasing model fit on one hand and decreasing the number of years available for model development. Moreover, when using 4 years to accumulate the damage history of each stand, 6 years remain for model fitting, and thus meets the requirements regarding the minimum level of random effects (here years within the random effect sub-compartment). In case of fewer levels, the variance of the random effect cannot be estimated correctly (Crawley, 2002).

The same method was used for the potential number of bark beetle generations, where $BGEN_1$ improved model fit compared to longer aggregation periods. However, when salvaged trees had been infested as well as the exact dates of salvage cuts within a given year were unknown, so whether the lag effect was due to data properties or real natural mechanisms could not be shown. $BGEN_1$ was also tested against combinations of monthly mean temperature (TAV_m) and mean annual temperature (MAT), and in most cases $BGEN_1$ model variants performed better. Due to its process-based nature and the higher information content, $BGEN_1$ was used for final model fitting.

Preliminary analysis was also done with different groupings of site type (SITE), which aimed at combining sites with similar soil moisture and nutrient supply. However, none of these approaches improved model fit. Similarly, several versions (aggregating different seasons of the year and of previous year) of predictor variable SMI were created and tested in the analysis. Finally, SMI was aggregated for the growing season from May to August and used in further analysis steps.

The impact of the predictor variables in the models were assessed and compared with partial effects plots (also called marginal effects or least square means). Partial effects measure the change in the expected value of the dependent variable as a result of a change in a certain explanatory variable while keeping all the other covariates at a fixed value. We used medians calculated from the available database as that fixed value (see Table 2).

To test the robustness of the models, 10-fold cross-validation was used, i.e. the data set was randomly broken into ten partitions, then models were fitted to data consisting all but one partition, which served as the test group. This procedure was repeated ten times using a different test group each time, then goodness-of-fit was evaluated (Mosteller and Tukey, 1968). Cross-validation tests were also implemented with FMUs being used as the partitions to see how the models perform in the four regionally different FMUs.

For posterior tests, we also used versions of the models that included fixed effects only. This was done by multiplying the design matrices of the models by the fixed effects calculated in the model fitting process. This is an assessment of how well the fixed effect variables could explain bark beetle damage in a general context without using the local implicit information related to the four FMUs.

All statistical modelling was done with the software package R (R Development Core Team, 2012). Automated fitting of models for model selection was done by the dredge function from package MuMIn (Barton, 2013). For partial effect plots, the plotLMER.fnc function of the R package languageR was used (Baayen, 2011).

2.4. Modelling the probability of disturbance events

Damage probabilities were modelled by logistic regression in a framework of a generalized linear mixed model (GLMM). Stands with bark beetle salvage values >1% of standing stock were considered as damaged. When the salvage rate dropped below that threshold, it was considered a registration error, and the data record was excluded from the analysis (see Klopcic et al., 2009; Overbeck and Schmidt, 2012). To translate the linear predictor of the model to probabilities Eq. (1) was used

$$\pi_i = (\exp^{\alpha+\beta\times X_i+\gamma_i})/(1+\exp^{\alpha+\beta\times X_i+\gamma_i}), \quad (1)$$

where π_i is the expected annual probability of the occurrence of a damage event in the ith row of the design matrix of the model. α is the intercept, β is the vector of fixed effect parameters, X_i is a row from the design matrix of the model and γ_i is the sum of random intercepts that account for the spatial and temporal cluster effect in the observed damages related to forest stand, compartment and district level. GLMMs were fitted with the lmer function of the package lme4 (Bates et al., 2012).

Model adequacy with regard to the linear relationship between the damage events and the explanatory variables was analysed by plotting the partial residuals of the model (see Zuur et al., 2008) and by fitting smoothed curves using the loess function of the basic R package. Goodness-of-fit of the models was assessed by classification tables (i.e. confusion matrix) and derived parameters such as sensitivity, specificity and the area under receiver operator characteristic curve (AUC). Sensitivity is the proportion of the true positive and the sum of true positive and false negative predictions (power to identify positives). Specificity is the proportion of the true negative and the sum of true negative and false positive predictions (power to identify negatives). Sensitivity and specificity values were plotted against a range of cut-off points (i.e. those π value above which an outcome is classified as damage; see Lalkhen and McCluskey, 2008). Choosing a cut-off value to decide whether a predicted probability value means an event or non-event depends highly on the intended use of the model (e.g. avoidance of future damages or avoidance of management costs).

In a receiver operator characteristic (ROC) curve, sensitivity was plotted against the false positive rate (1-specificity) for different cut-off points (Swets and Pickett, 1982). The area under the ROC curve (AUC) is the probability that a randomly selected observed positive event has a higher predicted probability value than a randomly selected observed negative event (Fawcett, 2006). AUC is also called the concordance-index, and can range from 0.5 (no predictive ability) to 1 (perfect discrimination) and is independent from cut-off values. The latter test was done also without the random effects (design matrix of model multiplied by the fixed effects only) to inform about the adequacy of the model for independent predictions. This approach was used for the ten-fold cross-validation and for the classification tables, as well. AUC values were calculated with the somers2 function of the R package Hmisc (Harrell, 2012).

2.5. Modelling the intensity of damage events

Data records classified as "damage" (threshold of 1% salvage rate; see previous section) were used as input for modelling damage intensities by linear regression in a linear mixed model framework (LMM). The response variable was natural log transformed to improve normality and homogeneity.

The general equation of the model was

$$\pi_i = \alpha + \beta \times X_i + \gamma_i, \quad (2)$$

where π_i is the expected annual intensity of a damage event in the ith row of the design matrix of the model. α is the intercept, β is the vector of fixed effect parameters, X_i is a row from the design matrix of the model and γ_i is the sum of random intercepts that account for the spatial and temporal cluster effect in the observed damages related to forest stand, compartment and district. LMMs were fitted with the lmer function of the package lme4 (Bates et al., 2012).

Goodness-of-fit was evaluated by the coefficient of determination (R^2). It indicates how much variation in the data is explained by the fitted model. Root mean squared error (RMSE) was also calculated to see how close predictions were to observed values. Test indices were calculated for predictions with fixed effects only, as well. Residuals were plotted against predicted values to check for normality and homogeneity. Normality of residuals was assessed by histograms. Residuals plotted against explanatory variables were inspected for linearity.

3. Results

3.1. The disturbance regime

During the six-year periods for 8172 stands that were available for the analysis, the ratio of events to non-events (i.e. years with a damage versus years without a damage) was approximately 1:14. Considering the damage intensities, 45% of disturbance events caused a damage under 10 $m^3\,ha^{-1}\,year^{-1}$, 46% between 10 and 50 $m^3\,ha^{-1}\,year^{-1}$ and 9% above 50 $m^3\,ha^{-1}\,year^{-1}$. The mean intensity of the disturbance events was 20.82 $m^3\,ha^{-1}\,year^{-1}$. In total 9.6% of the stands (788 out of 8172) suffered from bark beetle disturbance more than once. How intense the disturbance regime (including planned harvests) in the four FMUs was, is shown in Table 3. It is apparent that most bark beetle disturbance events were preceded by some other disturbance within the previous 4 years. The mean values of W_4, B_4, S_4 and R_4 were 3.2, 4.2, 0.5 and 29.3 $m^3\,ha^{-1}\,year^{-1}$ respectively, indicating that overall planned harvests were by far the most relevant disturbance.

3.2. Probability of disturbance events

In the probability model, all candidate explanatory variables were found to be informative, except SITE and SMI. Adding interaction terms to the models did not improve the fit. All signs of model coefficients were positive, except for SL. Increases in some of the predictor variables increased probability of damage substantially, mainly B_4 and VOL (Fig. 2).

AUC indicating accuracy of predictions had a value of 0.88. When used with fixed effects only, the AUC was 0.80, which is considered good in statistics literature (e.g. Obuchowski, 2003). Partial residual plots of all candidate predictor variables showed that assumption of linearity was valid (not shown here). During the 10-fold cross-validation of the selected model AUC values varied between 0.78 and 0.81, thus indicating good stability. When the four FMUs were used separately as test data, AUC values were 0.79, 0.78, 0.81 and 0.76 for TRV, STT, WVV and STMK, respectively.

The cut-off point analysis for the fixed effects-only model showed a proportion of 72.2% correctly classified cases for both the disturbance events and non-events at a cut-off value of 0.045 (Table 4). It may be more practical however, to take a look at these values at a cut-off value where the numbers of predicted events and non-events are closest to the observed ones, i.e. disturbance frequencies are similar. This cut-off value was 0.15, and the related sensitivity and specificity were 0.29 and 0.95 respectively; the share of overall correct classifications was 90.5% (Table 5). It means that non-events were predicted with good reliability, and in case of damage events the models could hit around one third of what was in the data set of observations.

Table 3
Share of bark beetle disturbance events with at least one salvage cut (wind, bark beetles or snow) or regular harvest in the same stand in the four preceding years. TRV = FMU Traun-Innviertel, STT = FMU Steyrtal, WVW = FMU Waldviertel-Voralpen, STMK = FMU Steiermark.

Disturbance factor	TRV (%)	STT (%)	WVV (%)	STMK (%)
Wind	29.1	42.8	39.7	59.6
Bark beetles	78.5	62.9	48.9	60.3
Snow	0.6	3.6	2.0	7.8
Regular harvests	64.3	57.1	38.4	41.3

3.3. Intensity of disturbance events

In case of bark beetle disturbance intensity, many of the available explanatory variables showed no or just very low predictive power and were not used in the models. ALT, PA, AGE, VOL, B_4 and $BGEN_1$ were found informative (Table 6). VOL showed slightly higher partial effects than the other variables (Fig. 3). The linear relationship of predictor variables with the response variable was confirmed by the residual plots. Normality and homogeneity improved a lot by ln-transforming the response variable. Residuals were normally distributed (not shown here). With regard to goodness-of-fit, there was a huge difference between R^2 values of the full model and the one refitted with fixed effects only (Table 7), with R^2 values of 0.45 and 0.13 respectively. RMSE values were 25.57 $m^3\,ha^{-1}\,year^{-1}$ and 28.69 $m^3\,ha^{-1}\,year^{-1}$, respectively.

R^2 values of the ten-fold cross-validation of the fixed effect-only model were between 0.05 and 0.13. When the four FMUs were used separately as test data, R^2 values were 0.05, 0.17, 0.09 and 0.13 for TRV, STT, WVV and STMK, respectively.

4. Discussion

4.1. Limitations of database and study design

Forest stand data in the current study were taken from forest management plans. It is important to note that the underlying data had not been compiled originally for scientific purposes and may carry some subjectivity, which is due to the underlying data collection procedures in practical forest management planning. This can lead to errors, which, together with measurement or recording errors of harvests, can lead to inconsistent conditions when linked together. Careful screening of the data is mandatory to prevent unjustified variation in the data. This argument holds also for the harvest records, where the assignment of removals to one of several causes is also a crucial issue. In the current analysis, we have used a low threshold of 1% removal rate to accept an observation being classified as damage event. Despite these potential limitations, we found the data on forest stand conditions and harvests to be reasonable and well suited for our study after excluding obvious erroneous records.

Another limitation was the impossibility to connect consecutive planning periods due to changes in spatial arrangements. Many stand polygons changed from one 10-year planning period to the next, for instance because of unplanned management activities due to disturbances. This problem could be circumvented by using compartments as the study unit instead, which remained mostly unchanged over time. However, a lot of information would have been lost due to aggregation of stand and harvest data over all stands within compartments. Ultimately, four years from the 10-year data record were used to build predictor variables characterizing harvest and damage history of stands. Nevertheless, these legacy effects proved to be important in explaining damage events in any given year. In a recent study of the disturbance regime in Austrian forests, also Thom et al. (2013) found that salvaged damages from the preceding two years were useful as explanatory variable. With the help of the mixed model framework, it was possible to keep annual records as separate observations, and therefore an ample amount of data points was available for model fitting.

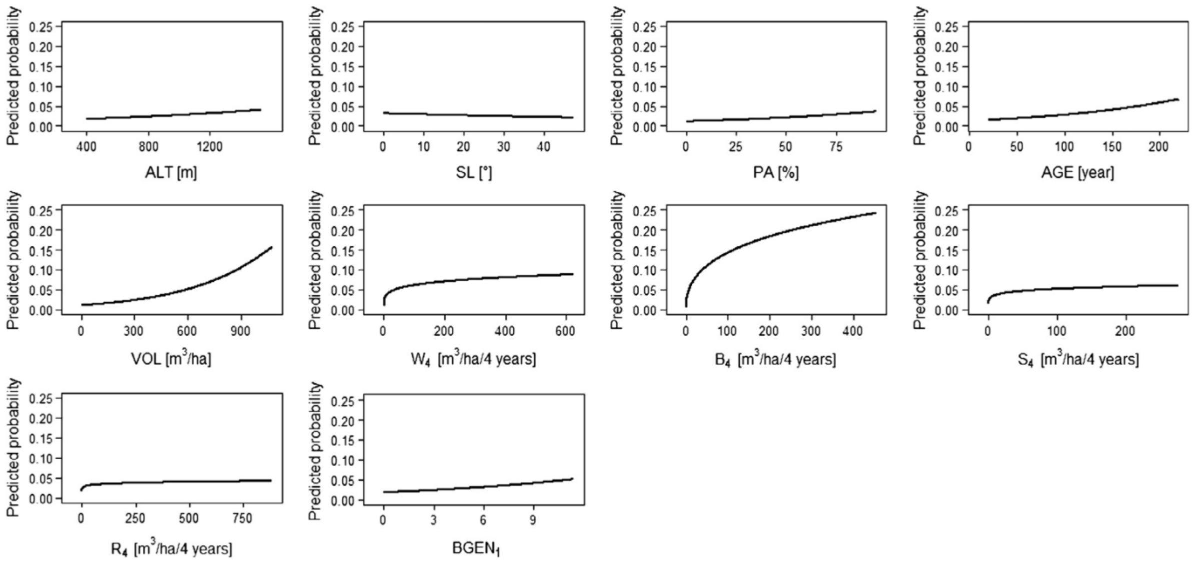

Fig. 2. Partial effects in the bark beetle damage probability model. *Y*-axis denotes the annual probability of a bark beetle disturbance event. *X*-axis denotes individual predictor variables. For a description of variables see Table 2.

Table 4
Classification table for the fixed effects-only probability models for the disturbance events. Cut-off value used in classification = 0.045.

		Observed	
		Event	Non-event
Predicted	Event	2,378	892
	Non-event	12,096	31,390

Table 5
Classification table for the fixed effects-only probability models for the disturbance events. Cut-off value used in classification = 0.15.

		Observed	
		Event	Non-event
Predicted	Event	948	2,322
	Non-event	2,126	41,360

Table 6
Estimated model coefficients of the generalized linear mixed model for the probability of occurrence of bark beetle damages and the linear mixed model for the intensity (m^3 ha^{-1} $year^{-1}$) of the disturbance events. For a description of variables see Table 2.

Explanatory variable	Probability model		Intensity model	
	Estimate	Std. error	Estimate	Std. error
(Intercept)	−7.2532	0.2711	1.4985	0.1340
ALT	0.0008	0.0002	−0.0003	0.0001
SL	−0.0088	0.0034		
ASP (1)	0.1539	0.0511		
PA	0.0125	0.0011	0.0046	0.0007
AGE	0.0077	0.0008	0.0033	0.0005
VOL	0.0026	0.0002	0.0012	0.0001
W_4	0.2138	0.0191		
B_4	0.4293	0.0154	0.0390	0.0095
S_4	0.1618	0.0509		
R_4	0.0952	0.0112		
$BGEN_1$	0.0938	0.0136	0.0294	0.0078

The silvicultural regimes in the four FMUs available for the current analysis are characterized mainly by shelterwood approaches with relatively short regeneration periods and strip-wise clear cut systems and can be considered as fairly typical for Austrian and Central European forestry. Also, the ecological conditions comprise of a broad range of sites and stands. However, uneven-aged management regimes were not covered by the current database. Population densities of bark beetles in general were high during the analysis period, which may have masked the relationship of predisposing factors and damage events (Raffa et al., 2008). Inclusion of bark beetle density monitoring data would be an interesting enhancement of the database. However, despite the potential value in explaining actual damages the spatial resolution of such monitoring data is usually low, and the combination with stand level data in complex terrain is not straightforward.

Salvage practices and pro-active forest protection measures may vary among regions, thus affecting the relation of stand and site conditions and management activities on one hand, and the damages from bark beetles on the other (Schroeder, 2001; Grodzki et al., 2006). It is interesting to note that cross-validation among FMUs in the current analysis indicated an obvious consistent management strategy including salvage practices in all four units of the AFF.

Spatial extent or spread of bark beetle disturbance events inside a stand, an important aspect of the disturbance regime, could not be considered. Making the database spatially explicit would require substantial effort and additional data mainly deducible from aerial photographs. Effects of damaged neighbour stands were not included as fixed effects in our models for similar reasons. However, it would have been also contrary to our aim to explore how well predictions could be made from stand level information only.

In general, the use of operational management data proved to be a valuable and promising approach. Extending the analysis as outlined above may add further relevant findings on bark beetle disturbances in managed forests.

4.2. Disturbance drivers and model quality

Goodness-of-fit tests and cross-validation (among FMUs) showed that the bark beetle damage probability model performed

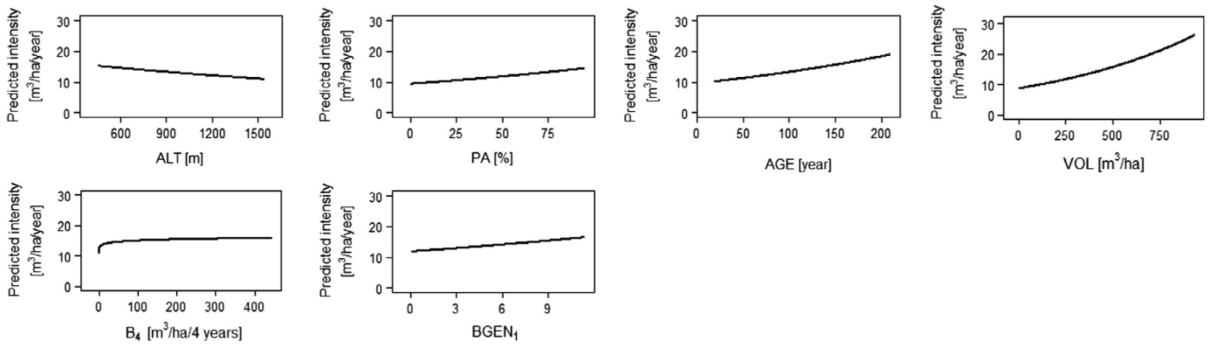

Fig. 3. Partial effects in the bark beetle disturbance intensity model. *Y*-axis denotes the annual intensity of a bark beetle disturbance event. *X*-axis denotes individual predictor variables. For a description of variables see Table 2.

Table 7
Indices for the evaluation of the performance of the generalized linear mixed model for the probability of occurrence of bark beetle damages and the linear mixed model for the intensity (m^3 ha^{-1} $year^{-1}$) of disturbance events.

Probability model		Intensity model	
AUC (fixed and random effects)	0.88	R^2 (fixed and random effects)	0.45
AUC (fixed effects only)	0.80	R^2 (fixed effects only)	0.13
Sensitivity (fixed effects only; cut-off value = 0.15)	0.29	RMSE (fixed and random effects)	25.57 m^3 ha^{-1} $year^{-1}$
Specificity (fixed effects only; cut-off value = 0.15)	0.95	RMSE (fixed effects only)	28.69 m^3 ha^{-1} $year^{-1}$

well also in case of a "fixed effects-only scenario", i.e. the model could be used for prediction purposes outside of the study area. This is an improvement compared to other similar studies, where model results were valid for the study area only (see Bolker et al., 2009).

Effect signs and magnitude of predictor variables clearly indicated higher probabilities of bark beetle disturbance events for old, highly stocked stands dominated by Norway spruce, after years with high number of potential bark beetle generations. This is in line with other studies (e.g. Schroeder, 2001; Stadelmann et al., 2013; Thom et al., 2013), with the exception for stand age, whose predisposing effect in some studies (e.g. Overbeck and Schmidt, 2012) was found to level off beyond 90–100 years. Previous disturbances and regular harvests also had positive effects on bark beetle damage probability, which is in line with findings by Klopcic et al. (2009). Previous bark beetle damages particularly increased the risk of follow-up damages, thus highlighting the importance of careful inspection of stands in the forthcoming years after infested trees have been found and pro-active forest protection measures (Marini et al., 2012). The important interaction of wind and bark beetle disturbances was also verified by our study (compare Schroeder, 2001). In general, all types of stem removal increased bark beetle disturbance probability, most likely due to more breeding habitat and freshly opened vulnerable stand edges (Kautz et al., 2013). Surprisingly, predictors describing the forest site showed the smallest effect on damage probability. This may be explained by the rather low representativeness of site descriptors for stand polygons due to substantial small-scale variation of site and soil conditions in mountain forests. Moreover, the spatial variability of site attributes was apparently much larger compared to bark beetle damages, reducing their explanatory power when being aggregated at stand or even compartment level.

The positive effect sign of predictor ALT in explaining the probability of a bark beetle damage event calls for some attention. A possible reason for this is that the maximum elevation in our study was 1600 m a.s.l., where spruce is the dominant species, and thermal conditions for bark beetle reproduction have become more favourable since the 1990s. Wind storms are also more frequent at higher elevations, providing more dead wood as breeding habitat, which is also more difficult to remove from stands less accessible than stands at lower elevations. For similar findings, see Lausch et al. (2011) and Mezei et al. (2014). It should be noted that in the models $BGEN_1$ indicates decreasing damage risk with cooler temperatures. This clearly shows that ALT represents indirectly other drivers than temperature.

SL was the only predictor with a negative-signed coefficient, i.e. stands on flat ground had a higher predicted probability of disturbance in our study. ASP showed a very small effect on probability, indicating a slightly higher risk of bark beetle disturbance events on South-facing slopes. The uninformative nature of the water supply proxy SMI was most probably due to the limited available site information required to estimate stand specific soil water storage capacity (see Seidl et al., 2008b), which in turn did not allow identifying stands with water shortage during the growing season. However, including soil parameters in defining water availability increases the spatial resolution compared to using precipitation values only and thus enhance the stand specific information available to filter out vulnerable stands. Using $BGEN_1$ in the model as a predictor variable proved to be an efficient way of incorporating climatic drivers, and thus making the model climate sensitive. Moreover, this predictor has a higher explanatory value compared to simple temperature parameters. High propagation rates due to sister broods were explicitly considered in $BGEN_1$ (e.g. Anderbrant, 1989), just like the photoperiodic induction of diapause in estimating the potential numbers of insect generations. Due to the integrated bark beetle development processes, the predictive power of this variable may also be valid under future climatic conditions.

In case of the intensity model, the big difference of R^2 between the full model (fixed and random effects) and the fixed effects-only model indicated that the variables used in the study explained only a small proportion of the variance in bark beetle salvage. With regard to confounding variables, the position of admixture trees may be important as their volatiles can interfere with bark beetle pheromone communication (Zhang et al., 1999). The spatial position of resistant and susceptible spruce trees in the stand is also a relevant characteristic (Becker and Schröter, 2000). However, a priori identification of such tree categories may be possible in a detailed research study at small scale, but not from standard operational management data. The practised forest protection measures may also be important (i.e. treatment of harvest residues; timely extraction of trees infested by bark beetles or damaged by other disturbance agents). Unfortunately, beside general information at regional scale, no specific data on such management practices were available for the study FMUs (Grodzki et al., 2006). The potential relevance of such information is confirmed by Thom et al. (2013), who found that forest stewardship related attributes contributed significantly in explaining the variation in damage data at the level of administrative districts. The abundance of natural enemies of bark beetles (predators and pathogens) can also have an effect on the disturbance intensity (Weslien and Regnander, 1992). Such hypothesized influences were not known directly from the available data but might be indirectly indicated by proxies such as share of admixed tree species.

4.3. Implications for forest management

The importance of predictor variables describing vegetation structure (PA, AGE, VOL) indicated that forest management has possibilities to mitigate the effects of bark beetle disturbances and also to promote measures to adapt to a future climate. Highly stocked spruce stands face higher risks of bark beetle damage, but those that also suffered from either wind or bark beetle damage previously should be paid high attention. This is not a new finding but confirms consistently earlier studies, based on a huge empirical database. Model results also gave hints about the worrying fact that in these stands probability of disturbance will increase in the future, as thermal breeding conditions for bark beetles are expected to improve in a warmer climate (Jönsson and Bärring, 2011). Since other driving factors are not controllable by forest management, decreasing the share of spruce, the rotation length and timber stocks in stands prone to bark beetle disturbance are the only viable options to mitigate the risk (beside direct population control of bark beetles with pro-active forest protection measures). However, the relevance of these "slow drivers" (compare Thom et al., 2013) indicate substantial time lags before adaptation measures become effective, and emphasize the need for timely adaptation. Although not based on findings from the current study, silvicultural measures should avoid sudden opening of the canopy and creation of sun-exposed stand edges. This, in turn, may further constrain the implementation of adaptation strategies. Real-time monitoring of sites with phenology models such as PHENIPS (Baier et al., 2007) appear as promising risk assessment tools. However, such monitoring systems based on a combination of simulation and remote sensing techniques must be linked with proactive protection measures on the ground when beetle population densities and insect development indicate high risk conditions. For a comprehensive review on management options, we refer to Wermelinger (2004). Overall, management costs in Norway spruce forests are expected to increase under climate change.

The bark beetle damage probability model developed in this study can be used to identify stands with high bark beetle infestation risk (i.e. susceptibility, predisposition) from data which are either available from standard management plans or can be generated with relatively low efforts. However, the experiences from analysing the damage intensities also demonstrates that the expectation that salvage volumes at stand level can be projected from simple standard data may be too optimistic.

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9.2 Paper II

Developing predictive models of wind damage in Austrian forests

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Abstract

- Context: Among natural disturbances wind storms cause the highest damage in forests in Austria.
- Aim: To quantify the effects of site, stand and meteorological attributes on the wind disturbance regime at operational scale of forest stands.
- Methods: We used binomial generalized linear mixed models (GLMMs) to quantify the probability of damage events and linear mixed models (LMMs) to explain the damage intensity at forest stand level in four management units with a total forest area of approximately 28800 ha.
- Results: Timber stock volume, stand age, altitude, previous disturbances, gust wind speed and frozen state of soil contributed in explaining probability of wind damage. While the model of disturbance probability correctly classified 90% of

all cases in the data set (specificity 95%, sensitivity 26%), the model for damage intensity explained only low percentages of the variation in the observed damage data (full model R^2 =0.38, fixed effects-only model R^2 = 0.09; cross validation in the four forest management units yielded similar R^2 values).

- Conclusion: The developed models indicate that decreasing the share of spruce, the age and the timber stock in stands exposed to wind disturbance can mitigate the risk and the expected damage intensity. Self-reinforcing processes may lead to increasing disturbance probability in the future, emphasizing the importance of stand stability and resilience in forest management.

Key-words: storm; disturbance; windthrow; forest management; stand scale

1 Introduction

In Europe during the period between 1950 and 2000, an average of 18.7 mill. m^3 of timber were damaged by wind annually (Schelhaas et al. 2003). This makes storms leading to uprooting and stem breakage of trees the most detrimental natural threat to European forests (i.e. approximately 66% of total damage from wind, fire, bark beetles and snow). In Austria, post-windthrow salvage logging fluctuated between approximately 1 and 11 mill. m^3 of timber per year in the period 1990-2012, which corresponds to shares of 4 to 50% of the annual cut (Prem and Beer 2012, Anonymous 2013). Peak years due to large-scale stand-replacing events mainly during the winter season were 1990 (7 mill. m^3), 2007 (9 mill. m^3) and 2008 (10 mill. m^3). Beside these severe storm events, a high proportion of the timber salvage was due to small scale or low intensity disturbance events. However, also these less intense damage events accumulate to substantial losses in timber value and cause additional costs for harvesting and further follow-up costs regarding planting, tending and other silvicultural measures. Also, management plans become obsolete and need to be updated. Beside the adverse economic consequences in timber production, windthrow can negatively affect other forest ecosystem services like protection against rockfall and avalanches (Brang et al. 2006), drinking water preservation (Weis et al. 2006) or in situ carbon sequestration (Thürig et al. 2005).

The wind disturbance regime is driven by the interplay of forest characteristics and weather (Dale et al. 2000). Species composition, stand height, stand edges, canopy roughness and tree attributes like crown length and slenderness correlate with wind damage (e.g. Valinger and Fridman 1999; Mitchell et al. 2001; Olofsson and Blennow 2005; Sellier and Fourcaud 2009). Beyond a wind speed of 45 m/s, stand-replacing damage is almost certain, regardless of stand condition (Gardiner et al. 2010). With decreasing wind speed, the effects of tree and stand characteristics on damage intensity become more apparent (Xi and Peet 2011).

It is expected that frequency of storm events in Central Europe may increase in a warmer climate (Lindner and Rummukainen 2013) and that an intensifying wind disturbance regime may exert a positive feedback on bark beetle disturbances through the provision of abundant breeding habitat (e.g. Marini et al. 2013). Additionally, a warmer climate will also benefit bark beetles, which may then complete two or even three life cycles per year (Jönsson and Bärring 2011). Other disturbance agents like snow breakage and regular harvests can also modify the structure of forest stands and increase their susceptibility to wind disturbance. Because of the magnitude and economic relevance, interest has been growing to identify stand and site attributes which explain the variation in damage and to develop quantitative models to assess the vulnerability of forests to wind damage as a prerequisite for targeted risk management. The literature of storm damage in Central European forests is extensive, and many studies scrutinized various driving factors of large-scale storm damage (e.g. Dobbertin 2002; Schütz et al. 2006; Schindler et al. 2009; Schmidt et al. 2010). However, intermediate and small-scale endemic wind disturbances are less widely researched, although their cumulative effect can be significant on forest ecosystem services (e.g. Nagel and Diaci 2005; Klopcic et al. 2009).

Major data sources for such studies include salvage records kept by forest enterprises (e.g. Hanewinkel et al. 2008; Klopcic et al. 2009), regional to national scale damage statistics either based on large-scale forest inventories (e.g. Jalkanen and Mattila 2000) or semi-quantitative salvage reporting schemes on administrative district or province level (e.g. Thom et al. 2013). Recently, the use of damage estimates derived from remote sensing information (e.g. aerial photographs and satellite images) has attracted much attention (e.g. Lanquaye-Opoku et al. 2005; Usbeck et al. 2012). However, each of these approaches has some limitations. Standard management records provide local operational context and report damage also of a few m^3 of timber only; however detailed information on stand and site variables is generally missing, and as a consequence, variables that can explain the damage are scarce. Spatial coverage is usually restricted, as book keeping rules vary greatly among forest enterprises. On the other hand, a great advantage of large-scale forest inventories is large spatial coverage. Operational context of individual inventory plots is missing however, similarly to regional damage monitoring systems that rely on some kind of qualitative damage assessment in a highly aggregated form. So far, remote sensing has been used mainly to assess damage extent after large-scale events (e.g. Schindler et al. 2012). Beside these observational approaches, there has been experimental work on tree pulling (e.g. Nicoll et al. 2006) and mechanistic modelling to determine critical wind speed for either uprooting or stem breakage and then calculating the probability of the occurrence of such wind speed by assessment of the local wind climate attributes (e.g. Peltola et al. 1999; Gardiner et al. 2000). While the latter approaches provide a clearly defined link to weather phenomena and are thus potentially applicable for climate change impact assessments, most empirical studies contain only local relationships without general transferability to other regions or conditions.

In this study, our objective is to develop quantitative statistical models to estimate (i) the probability for wind damage events, and (ii) the intensity of the damage in Eastern Alpine mountain forests. We employ a large empirical database considering forest and site characteristics, weather data, forest management and other disturbance agents. The focus will be at stand scale due to its importance for operational forest management.

In particular we hypothesized that

(a) by utilizing data with huge spatial coverage, established empirical relationships are robust over a wide range of conditions,
(b) the use of weather-related predictor variables improve model performance and reliability.

2 Material and methods

2.1 Study area

Data from four management units (FMU) of the Austrian Federal Forests (AFF) were available for the current analysis. AFF is responsible for the management of 588000 ha of forest in Austria (i.e. 15% of total forest area). The four FMUs were Traun-Innviertel, Steyrtal, Waldviertel-Voralpen and Steiermark (Fig. 1); for the current study, 28870 ha of forest were considered.

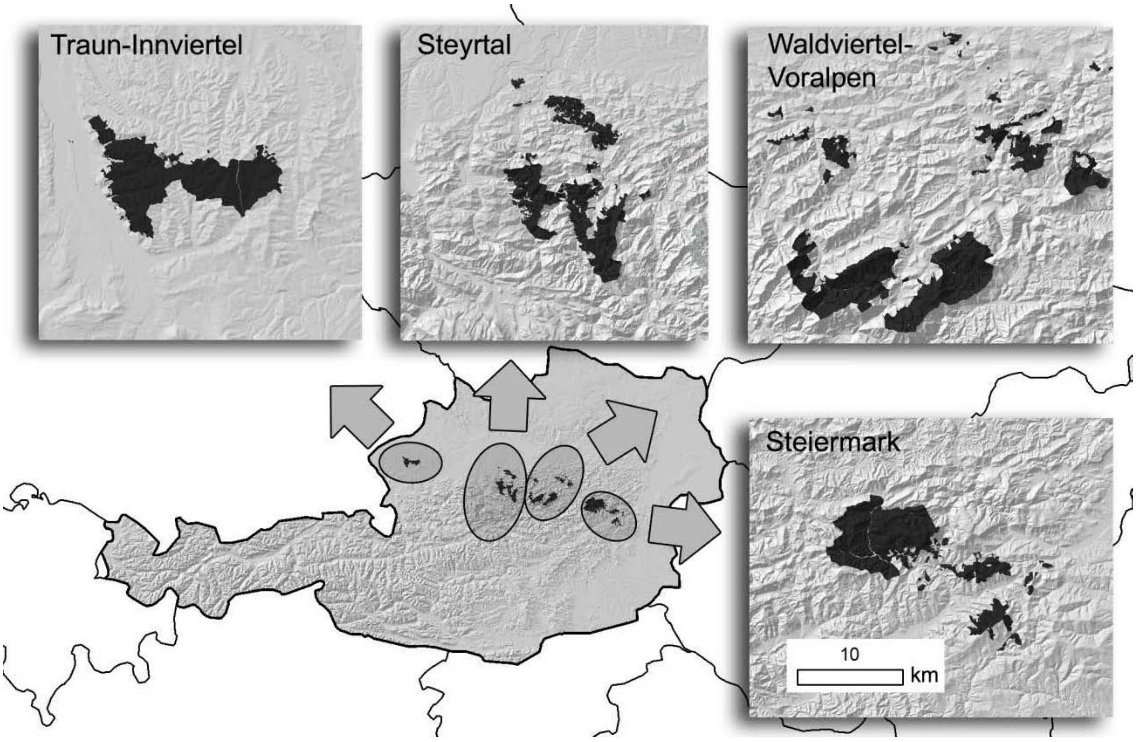

Fig. 1 Location of the four forest management units under study in Austria

Altitude ranges from submontane (400-800 m a.s.l.) to high montane (approximately 1600 m a.s.l.) vegetation belts (Table 1) in all the FMUs except in Traun-Innviertel, which is exclusively located at altitudes up to 700 m. Norway spruce is the most abundant tree species in these AFF management units, a tree species which is considered to be one of the most vulnerable to wind damage in the region (Hanewinkel et al. 2013). Other conifers are present in the area, but with a much smaller share (*Abies alba* Mill., *Larix decidua* Mill., *Pinus sylvestris* [L.]). The main broadleaved species is *Fagus sylvatica* [L.]; other broadleaves have just minor shares of up to 4% of basal area. All four FMUs have a uniform age class distribution up until the usual rotation length of the main species (100-140 years). Age classes above that show a strongly decreasing trend, with the oldest stands being approximately 220 years old.

Forest area within the FMUs is structured into several administrative levels. Districts usually have an area of a few thousand hectares. A district contains several tens of compartments, an administrational unit that – in a mountainous environment – confines an area with similar site properties within major topographic borders (27.1 ha on average in the study area). The smallest operational unit is the sub-compartment (i.e. the forest stand), which usually has an area of several hectares (6.6 ha on average in the study area). Sub-compartments have quite homogeneous site and stand conditions and are the basic silvicultural planning and treatment unit. In Austria, 10-year management plans include operational silvicultural prescriptions at stand level and yield regulation at district level (i.e. determination of allowable annual cut).

All four FMUs suffered great losses of timber due to storms in 2007 and 2008, with proportion of stands being damaged as high as 28% (FMU Traun-Innviertel in 2008). FMU Steyrtal suffered also high storm damage in 2003. See Fig. 2 for a detailed comparison of wind salvage and total timber removals throughout the years of the study period.

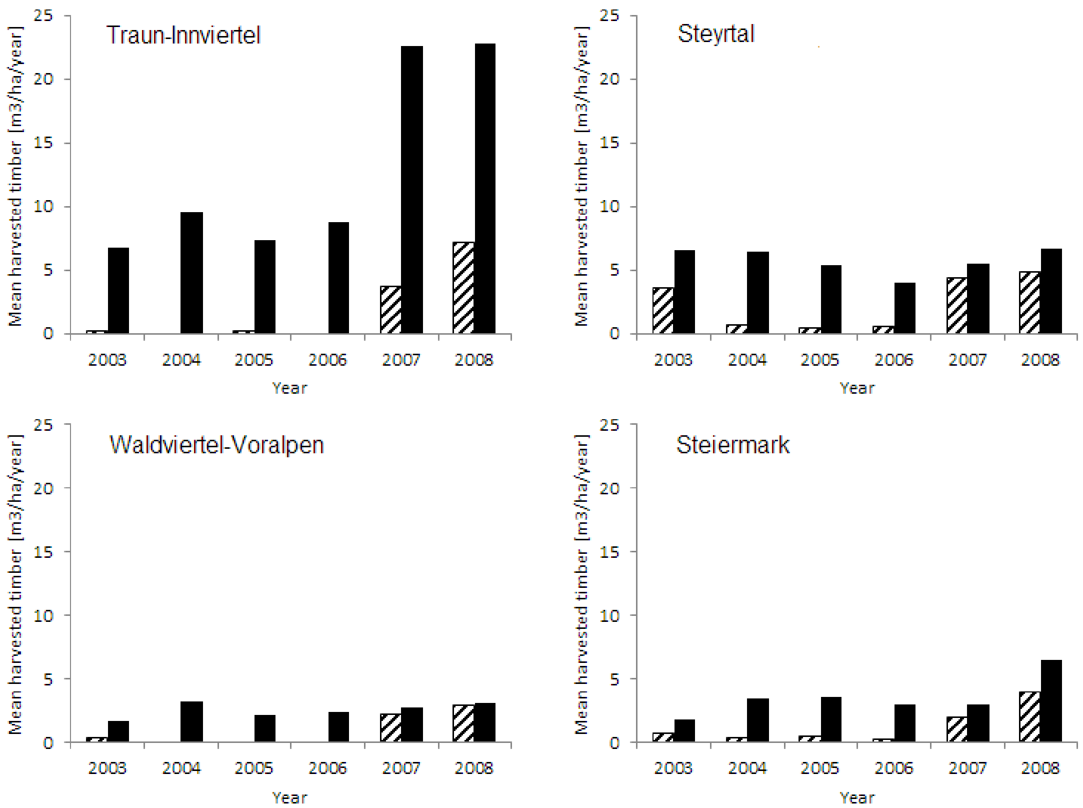

Fig. 2 Average harvested timber volumes over all stands in the four forest management units of the study area (black column: total timber removals including salvage; striped column: wind salvage)

2.2 Database

To build the database for the analysis, stand and site data of management plans of 15 districts within the four FMUs were combined with the related harvest records and a gridded weather data set covering the FMUs under study (Table 1).

Table 1 Characteristics of four Austrian Federal Forests management units which provided data for the analysis. Species shares are based on volume

Forest management unit	Traun-Innviertel	Steyrtal	Waldviertel-Voralpen	Steiermark
Altitudinal range [m a.s.l.]	500-700	400-1,600	500-1,600	600-1,600
Bedrock	Acidic	Calcareous & flysch	Calcareous	Acidic & calcareous
Forest area [ha]	4,780.4	8,554.4	4,243.1	11,289.7
Number of districts	2	6	2	5
Number of compartments	126	408	145	385
Number of sub-compartments	929	1,920	943	2,223
Picea abies [%]	73	62	67	80
Fagus sylvatica [%]	16	26	17	7
Other conifers [%]	7	9	13	12
Other broadleaves [%]	4	3	3	1

2.2.1 Forest management plans

The management plans covered the decade from 1999 to 2008. For all stands, such plans described attributes like yield class of the most abundant tree species in a stand, volume and age by species, and provide a qualitative description of the mixture type, but miss details on compositional and structural features. Since management plans describe the initial state of stands at the beginning of the respective 10-year planning period only, the annual development of stand attributes (in our case timber stock volume at tree species level) over time was projected by means of yield tables (Marschall 1975) and removals reported in the harvest records. Stands younger than

20 years of age were not included in the analysis database, as such young stands were considered not to be vulnerable to wind damage.

2.2.2 Harvest records

Harvest records contained all timber removals specifying the year of harvest (without exact date), total extracted volume and the reason for the removals distinguishing regular harvests and salvage due to various damaging agents. However, no indication of the spatial distribution of the fellings inside the sub-compartments was included in these records. Regular harvests and salvage due to wind (no differentiation between uprooting and stem breakage in the records), snow and bark beetles were each cumulated for periods of up to four years prior to any year in the 10-year planning period to account for damage history in the stands. This reduced the length of the 10-year time series of damage data to six years. The reported harvested volume was multiplied by 1.2 when related to standing stock to account for standard practices with regard to treatment of harvest residues (Pretzsch 2010).

2.2.3 Weather data and related information

The assessment of the relationship between damage and weather data including storms necessitates local time series of weather variables at the level of the investigated forest stands at a temporal resolution that allows the identification of the driving weather stimuli. For the current analysis, air temperature and wind speed data were provided for all four FMUs. Daily time series of air temperature (minimum, mean, maximum), on a mesh with a width of 100 m over the FMUs were generated. The data were interpolated from the network of weather stations of the Austrian weather service (ZAMG; http://www.zamg.ac.at/cms/de/klima/messnetze/wetterstationen). The entire network consists of more than 200 automated stations spread all over Austria. A second order polynomial fit based on the four seasons that tracks vertical temperature gradients was applied to capture the behaviour of the air temperature field within the complex Alpine topography. The interpolation routine distinguished between three regions covering the study FMUs and depended on longitude, latitude and elevation. As no gap filling was applied to the observations, the interpolation relied on the original measurements. The number of used weather stations varied between 15 and 20 depending on year and FMU. Based on Paul et al. (2004), daily air temperature was used to calculate whether the soil was frozen at any day of the year. If the uppermost 10-cm layer of the soil was calculated to be frozen, then it was assumed to have a stabilizing effect on the trees against windthrow. Calculating the soil temperature was based on mean annual and summer air temperature and the minimum and maximum air temperature of the current day. Leaf area index, understory vegetation and litter mass of the soil was also taken into account, for which average values of the study area were used defined by expert knowledge. For more details on the soil temperature model, see Paul et al. (2004).

Wind is perhaps the most difficult weather parameter to be generated on a grid, especially in a complex orography as the European Alps. One obvious reason is that wind measurements are representative for only a very small area within which the measurements are taken. The highly discontinuous propagation of the wind field in space makes it almost impossible to homogenize observed time series data by comparing them to other series, farther away. Another inherent problem is that wind measurements carried out at one station are inhomogeneous in time as any change in the roughness length of the surrounding topography which may be caused by a growing tree has an impact on the measurements. So, wind observations are fraught with problems, and hence it is difficult to interpolate measurements in space and time.

In the current study, we used INCA (Integrated Nowcasting through Comprehensive Analysis) to provide wind data for the study FMUs (Haiden et al. 2011). INCA uses digital elevation data of 1x1 km grid size. In the case of wind, the nowcast starts with a three-dimensional analysis based on a first guess obtained from a NWP (Numerical Weather Prediction) model output that is enhanced by the consideration of further observations at weather stations. The wind fields are calculated by transforming 10-m wind observations to the NWP model level-wind using an elevation dependent factor and by applying an inverse distance squared interpolation routine on the observed corrections. Additionally, an iterative relaxation algorithm is enforced to warrant mass-consistent fields. Wind vectors at grid points near to stations are kept at the observed values during the relaxation procedure. Thus, the INCA data set has been designed to match the observed values, except from regions that are not covered by the ZAMG station network. The spatial resolution of the INCA data set might be too coarse to capture the details of the wind field within the forest area. However, no in-situ meteorological measurements in the forest management units were available. The

INCA system has not been intended for climatological purposes but for operational forecasts; therefore, the data set covered recent years only. In this study, maximum daily wind speeds were computed from 24-hourly wind speeds (10-minute wind speeds at full hours) covering the period 2003-2008. 10-minute wind speeds were transformed to 2-second gust speeds (VMAX) using multiplication by a gust factor of 1.65 (Cvitan 2003). Such short-term gusts are commonly considered as major determinant of wind damage in forests (Mayer 1987). To visualize the spatial heterogeneity of the wind speed data, the number of days with a 2-second gust speed above 30 ms^{-1} is shown for the study FMUs in Fig. 3.

Weather-related data were linked to forest compartment centroids using GIS software (ESRI 2012). All stands within a forest compartment were assumed to have the same weather attributes. All stand-level attributes available for the analysis are shown in Table 2.

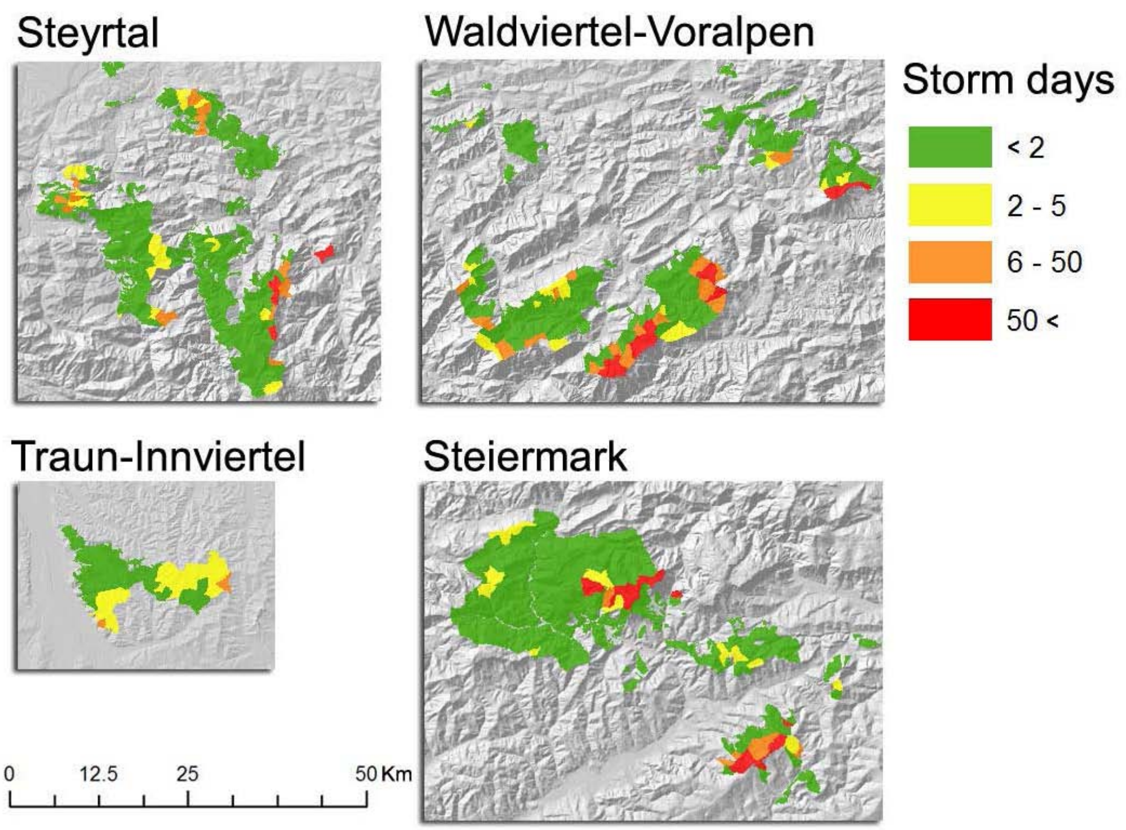

Fig. 3 Number of days with daily maximum 2-second gust speed [ms^{-1}] exceeding 30 ms^{-1} during the period 2003-2008 for the four forest management units

Table 2 Available stand-level attributes for modelling wind damage at stand level in the study area

Variable	Unit	Median	Description
ALT	m	900	Altitude above sea level
SL	°	25	Slope steepness
ASP	Nominal	-	Aspect [N, NE, E, SE, S, SW, W, NW]
SITE	Nominal	-	21 site types from the site classification system of the Austrian Federal Forests; used in different groupings according to bedrock (calcareous, acidic, flysch), water and nutrient status
YC	$m^3\ ha^{-1} year^{-1}$	8	Yield class of the main species of the stand; mean volume production per ha and year over a period of 100 years
PA	%	80	Share of Norway spruce (*Picea abies* [L.] Karst.)
AGE	years	90	Mean stand age
VOL	$m^3\ ha^{-1}$	290.51	Timber stock volume before removals of actual year
W_t	$m^3\ ha^{-1}$ (t) $years^{-1}$	-	Wind damage in previous (t) years, t = [1-4]
B_t	$m^3\ ha^{-1}$ (t) $years^{-1}$	-	Bark beetle damage in previous (t) years, t = [1-4]
S_t	$m^3\ ha^{-1}$ (t) $years^{-1}$	-	Snow damage in previous (t) years, t = [1-4]
R_t	$m^3\ ha^{-1}$ (t) $years^{-1}$	-	Regular harvests in previous (t) years, t = [1-4]
VMAX	ms^{-1}	14.44	Highest daily 2-sec gust speed per year
SF	[0,1]	-	Soil state on the day with the highest 2-sec gust speed (1: frozen; 0: not frozen)

2.3 General modelling approach

The modelling process was structured in two main steps. First, modelling the probability of a wind damage event, then as the second step, the damage intensity given that a damage event had occurred in the stand. A mixed model framework was used because of the hierarchical structure of the data set. Spatial units (sub-compartments) were nested within bigger units (compartments, districts, FMUs), and carried a ten-year time series data set, hence spatial and temporal autocorrelation of model residuals could lead to bias in model estimates (Pinheiro and Bates 2000). With the use of a mixed model framework, random effects were added to the model equations in the model fitting process. These random effects induced a simple correlation structure for observations in the same data cluster (in our case sub-compartment, compartment and district), and therefore the effects of confounding variables related to these clusters could be estimated (i.e. the random effects) and separated from the actually studied predictor variables (i.e. the fixed effects) during the analysis.

As an exploratory step, Pearson correlation coefficients among the continuous stand level variables were calculated to attain insight on the interrelationships of variables in the data set. Candidate predictor variables were used in an automated procedure with all possible combinations to fit model equations to the data set for both the probability and the intensity models. Afterwards, Akaike Information Criterion (AIC) values (Akaike 1973) of the resulting models were compared. This is a method that rewards the goodness-of-fit of a model and penalizes for the increasing number of predictors at the same time. The penalty for the inclusion of one additional parameter was 2 AIC units following Arnold (2010). If further posterior analysis (see below) did not reveal inadequacies, the model with the lowest AIC value was chosen (Burnham and Anderson 2002). Partial effects plots (also called marginal effects or least square means) were used to assess the effect of the predictor variables in the models. Partial effects measure the change in the expected value of the response variable as a result of a change in a certain predictor variable while keeping all the other covariates fixed at the median values of the respective variables in the data base (see Table 2).

The robustness of the models was tested by 10-fold cross-validation, in which the data set was randomly broken into ten partitions, and models were fitted to data consisting of all but one partition that served as the test group. This procedure was repeated ten times with a different test group each time, then goodness-of-fit was evaluated (Mosteller and Tukey 1968). Cross-validation tests were also implemented with the four FMUs being used as partitions.

Versions of the models that included fixed effects only, were also used for posterior tests. We did this by multiplying the design matrices of the models by the fixed effects calculated in the model fitting process. This served as an assessment of how well the fixed effect variables could explain wind damage in a general context without using the local implicit information related to the four FMUs.

In the model fitting process, predictor variables describing accumulated salvage and regular harvests in the previous four years (W_t, B_t, S_t and R_t) were all natural log transformed. Considering these variables, we manually fitted the models with different time periods (*t* = 1-4 years). Extending the number of the preceding years improved the fit of the models. The four-year period proved to be the best compromise between increasing model fit and decreasing the number of years available for model development at the same time. Besides, when using four years to accumulate the disturbance history of each stand, six years remained for model fitting, which met the requirement regarding the minimum level of a random effect, in our case years within the random effect "sub-compartment". The variance of a random effect cannot be estimated correctly in case of fewer than six levels (Crawley 2002).

In the analysis, different groupings of site type (SITE) were also tested. This aimed at combining sites with similar soil moisture and nutrient supply (as defined in the site classification system of the AFF; Weinfurter 2004). However, none of these groupings improved model fit.

The software package R was used for the statistical modelling (R Core Team 2013). The automated fitting of models for model selection was done with the dredge function from the R package MuMIn (Barton 2013). The plotLMER.fnc function of the R package languageR was used for the partial effect plots (Baayen 2011).

2.4 Modelling the probability of disturbance events

Wind damage probabilities were modelled by logistic regression. This was done in a framework of a generalized linear mixed model (GLMM). Salvage values below 1% of standing stock were considered registration errors, and the respective data record as a non-event (see Klopcic et al. 2009; Overbeck and Schmidt 2012). To translate the linear predictor of the model to probabilities equation (1) was used

[1] $$\pi_i = \left(\exp^{\alpha+\beta\times X_i+\gamma_i}\right) / \left(1 + \exp^{\alpha+\beta\times X_i+\gamma_i}\right)$$

where π_i is the expected annual probability of the occurrence of a damage event in the i^{th} row of the design matrix of the model, α is the intercept, β is the vector of fixed effect parameters, X_i is a row from the design matrix of the model and γ_i is the sum of random intercepts that account for the spatial and temporal cluster effects in the observed damage related to forest stand, compartment and district level. GLMMs were fitted with the lmer function of the package lme4 (Bates et al. 2012).

Linearity of the relationship between damage events and the explanatory variables was assessed by plotting the partial residuals of the model (see Zuur et al. 2008) and fitting smoothed curves using the loess function of the basic R package. Data transformation was used to account for eventual non-linearities in the data. Classification table (i.e. confusion matrix) and derived parameters such as sensitivity, specificity and the area under receiver operator characteristic curve (AUC) were used for assessing goodness-of-fit of the models. Sensitivity is the ratio of the true positive and the sum of true positive and false negative predictions (i.e. the power to identify positives). Specificity is the ratio of the true negative and the sum of true negative and false positive predictions (i.e. the power to identify negatives). To decide whether a predicted probability value means an event or non-event one has to choose a cut-off value (see Lalkhen and McCluskey 2008) that serves his or her intentions (e.g. avoidance of future damage or avoidance of management costs). Sensitivity and specificity values were plotted against a range of cut-off points.

The AUC shows the probability that a randomly selected observed positive event has a higher predicted probability value than a randomly selected observed negative event (Fawcett 2006). The AUC is also called the concordance-index, and can range from 0.5 (no predictive ability) to 1 (perfect discrimination) and is independent from cut-off values. The somers2 function of the R package Hmisc was used for the calculation of AUC values (Harrell 2012).

2.5 Modelling the intensity of damage events

For modelling damage intensities [$m^3ha^{-1}year^{-1}$] by linear regression, data records classified as "damage" (threshold of 1% salvage rate; see previous section) were used as input in a linear mixed model framework (LMM). The response variable was natural log transformed to improve normality and homogeneity.

The general equation of the model was

[2] $$\pi_i = \alpha + \beta \times X_i + \gamma_i$$

where π_i is the expected annual intensity [$m^3ha^{-1}year^{-1}$] of a damage event in the i^{th} row of the design matrix of the model, α is the intercept, β is the vector of fixed effect parameters, X_i is a row from the design matrix of the model and γ_i is the sum of random intercepts that account for the spatial and temporal cluster effects in the observed damage related to forest stand, compartment and district. LMMs were fitted with the lmer function of the package lme4 (Bates et al. 2012).

Goodness-of-fit was evaluated by the coefficient of determination (R^2). It indicates how much variation in the data is explained by the fitted model. We calculated root mean squared error (RMSE) to see how close predictions were to observed values. Normality and homogeneity were tested by plotting residuals against predicted values. Histograms were used to assess normality of residuals. Residuals plotted against explanatory variables were inspected for linear relationships. Test indices were also calculated for models with fixed effects only.

3 Results

3.1 The disturbance regime

For the 6015 stands that were available for the analysis during the six-year period, the ratio of events to non-events (i.e. years with a wind damage versus years without a damage) was approximately 1:16. With regard to the wind damage intensities, 43% of the disturbance events caused damage smaller than 10 m^3 ha^{-1}, 46% between 10 and 50 m^3 ha^{-1} and 11% more than 50 m^3 ha^{-1}. The mean intensity of the wind damage events was 25.9 m^3 ha^{-1}. Table 3 shows the proportion of wind disturbance events which were preceded by any other disturbance or regular harvest in the previous four years.

Table 3 Share of wind disturbance events in the period 2003-2008 with at least one salvage cut (wind, bark beetles or snow) or regular harvest occurring in the same stand in the four preceding years

Preceding disturbance	Traun-Innviertel	Steyrtal	Waldviertel-Voralpen	Steiermark
Wind	36.4%	42.0%	39.7%	71.4%
Bark beetles	79.5%	38.8%	38.9%	59.2%
Snow	2.3%	1.1%	3.8%	4.9%
Regular harvests	75.0%	51.6%	34.9%	34.5%

3.2 Probability of disturbance events

The final model included predictor variables ALT (altitude above sea level), AGE (mean stand age), VOL (timber stock volume before harvests of actual year), W4, B4, S4, VMAX and SF (soil state on the day with the highest 2-sec gust speed; 1: frozen, 0: not frozen); all other variables were found uninformative. YC (yield class of the main species of the stand) was excluded from the analysis to avoid multi-collinearity (see Zuur et al. 2008), as it strongly correlated with several other explanatory variables (e.g. Pearson correlation was 0.70 with VOL, -0.56 with ALT). Adding interaction terms to the models did not improve the fit.

The signs of parameter estimates of the predictor variables were all positive, except for SF (Table 4). Fig. 4 presents the effects of individual predictors on the estimated probability of a wind damage. The AUC value of the model was 0.84. When used with the fixed effects only, the AUC was 0.71, which is considered “acceptable” in statistics literature (Hosmer et al. 2013). Partial residual plots showed that the assumption of linearity was valid for all the candidate predictor variables (not shown here). In the 10-fold cross-validation of the selected model, AUC values varied between 0.68 and 0.76, thus indicating good stability. When the four FMUs were used separately as test data, AUC values were 0.75, 0.72, 0.70 and 0.80 for Traun-Innviertel, Steyrtal, Waldviertel-Voralpen and Steiermark, respectively. AUC values in the current study were in line with other studies of storm damage in Central Europe in which this index was used to evaluate the classification into damaged and undamaged stands (AUC=[0.78-0.79] in

Schindler et al. 2009; AUC=0.76 in Klaus et al. 2011; AUC=[0.73-0.74] in Schindler et al. 2012).

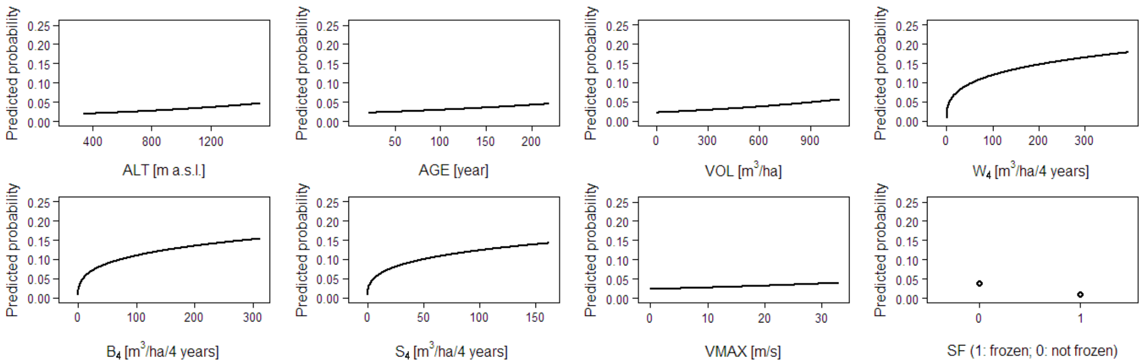

Fig. 4 Partial effects in the wind damage probability model. The Y-axis denotes the annual probability of a wind damage event. The X-axis shows the predictor variables. The solid lines represent the partial effects. For a description of variables, see Table 2

The cut-off point analysis (Fig. 5) for the fixed effects-only model showed a proportion of 65% correctly classified cases for both the disturbance events and non-events at a cut-off value of 0.04, i.e. both sensitivity and specificity had a value of 0.65 at this cut-off. Nevertheless, it may be more practical to take a look at these values at a cut-off value, where the numbers of predicted events and non-events are closest to the observed ones, i.e. disturbance frequencies in the observations and predictions are similar. This cut-off value was 0.13, and the related sensitivity and specificity were 0.26 and 0.95, respectively (the related share of overall correct classifications was 90%). This means that non-events were predicted with good reliability, and in case of damage events the model identified correctly one fourth of the observed events.

To assess the importance of weather related predictor variables in the models, we also fitted the final model in a version where these were excluded. When omitting VMAX and SF from the probability model, AUC values decreased slightly, to 0.83 in case of fixed and random effect model and to 0.70 in case of the fixed effect-only model. Cross-validation indicated only a small decrease in robustness (not shown here).

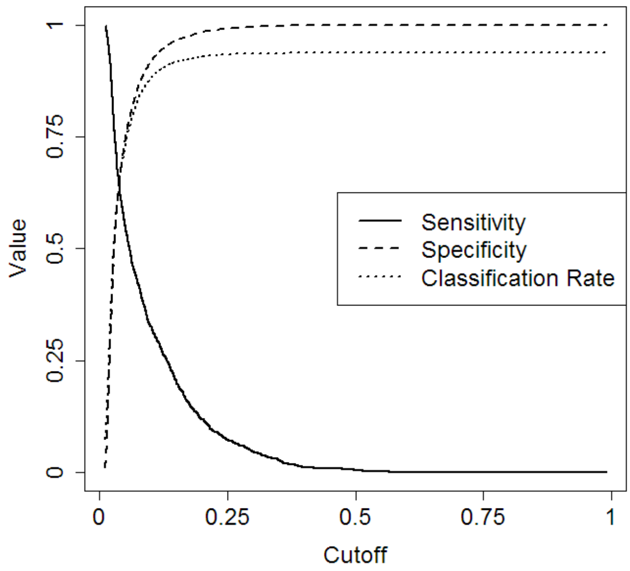

Fig. 5 Cut-off values and the related sensitivity, specificity and classification rate values for the probability model of wind disturbance events (predictions made without random effects)

Table 4 Estimated model coefficients of the generalized linear mixed model for the probability of occurrence of wind damage and the linear mixed model for damage intensity [m^3 ha^{-1} $year^{-1}$]. For a description of variables, see Table 2

	Probability model		Intensity model	
Explanatory variable	Estimate	Std. error	Estimate	Std. error
(Intercept)	-5.2165	0.3597	1.8212	0.1599
ALT	0.0008	0.0002		
PA			0.0025	0.0011
AGE	0.0037	0.0010		
VOL	0.0009	0.0003	0.0017	0.0002
W4	0.3429	0.0282	0.0527	0.0195
B4	0.3394	0.0282		
S4	0.3390	0.0753		
VMAX	0.0166	0.0064		
SF (1)	-0.7026	0.1759		

3.3 Intensity of disturbance events

When modelling wind disturbance intensity, many of the predictor variables were found uninformative in the model selection process, except for PA (share of Norway spruce), VOL and W 4 (Table 4). VOL showed higher partial effect than the other predictor variables (Fig. 6). The linear relationship between the predictor variables and the response variable was confirmed by the residual plots. Residuals were normally distributed. Normality and hom ogeneity improved substantially by In-transforming the response variable. Considering goodness-of-fit, there was a big difference between R^2 values of the full model and the one r efitted with fixed effects only (Table 5) with R^2 values of 0.38 and 0. 09, respectively. Related RMSE values were 38.49 m^3 ha^{-1} $year^{-1}$ and 43.77 m^3 ha^{-1} $year^{-1}$, respectively.

In the ten-fold cross-validation, the fixed effect-only model had R^2 values between 0.02 and 0.20. In case the four FMUs were used separately as test data, R^2 values were 0.08, 0.09, 0.08 and 0. 05 for Traun-Innviertel, Steyrtal, Waldviertel-Voralpen and Steiermark, respectively.

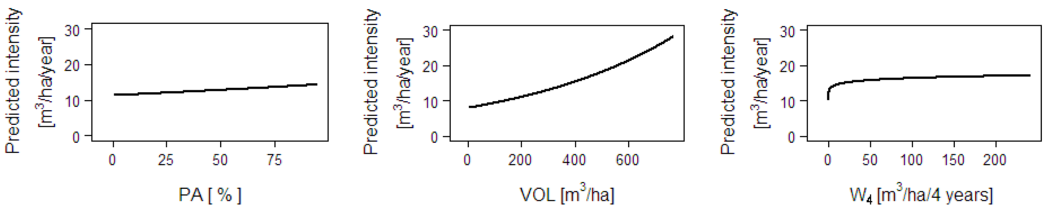

Fig. 6 Partial effects in the wind damage intensity model. The Y-axis denotes the intensity of a wind damage event. The X-axis denotes the individual predictor variables. The solid lines show the partial effects. For a description of variables, see Table 2

Table 5 Performance statistics of the generalized linear mixed model for the probability of occurrence of wind damage and the linear mixed model for the damage intensity [m^3 ha^{-1} $year^{-1}$]

Probability model		Intensity model	
AUC (fixed and random effects)	0.84	R^2 (fixed and random effects)	0.38
AUC (fixed effects only)	0.71	R^2 (fixed effects only)	0.09
Sensitivity (fixed effects only; cut-off value = 0.15)	0.26	RMSE (fixed and random effects)	38.49 m^3 ha^{-1} $year^{-1}$
Specificity (fixed effects only; cut-off value = 0.15)	0.95	RMSE (fixed effects only)	43.77 m^3 ha^{-1} $year^{-1}$

4 Discussion

4.1 Disturbance drivers and model quality

Classification performance tests showed that the wind damage probability model performed moderately well also in case of a “fixed effects-only” version. This indicates that the model could be used for prediction purposes outside of the study area (see Bolker et al. 2009). Model results clearly indicated a higher probability of wind disturbance for old, highly stocked forests at higher altitudes in years following high intensity disturbances (see Fig. 4). The inclusion of AGE may represent the positive effect of increasing tree height on damage probability, as the correlation between AGE and VOL was fairly small. ALT may capture the combined effects of increasing wind speed with increasing elevation as well as increased exposure to winds at higher altitudes due to reduced shielding effects of surrounding topography.

The strong effect of preceding disturbance events may be due to the newly created stand edges and increased canopy roughness (Rochelle et al. 1999; Schütz et al. 2006) and is in line with findings of other studies (e.g. Klopcic et al. 2009; Thom et al. 2013). W_4, B_4 and S_4 in our analysis had very similar effect sizes in case of the probability model (see Table 4), which means that changes in stand structure induced by salvage of wind, bark beetle and snow damage are apparently similar and that they are interchangeable (they also have the same measurement unit) in the model. Therefore, they could also be used in an aggregated form when estimating wind damage probability. Intensity of individual salvage cuts was in most cases below 20%

of standing volume and thus at levels similar to regular thinnings and regeneration cuts. The fact that regular harvests (R_4) were not found important in explaining wind damage indicates a certain difference between the effects of regular harvests and salvage cuts. Whether this is the result of careful forest management or some other factor, could not be tested in the current study.

The four FMUs in the current analysis represent a broad range of sites and stands in Austrian commercial forests. Salvage practices may vary among regions, which affects the relationship of stand and site conditions and management activities on one hand, and damage from wind on the other. The robustness of the developed models as revealed by the cross-validation among FMUs indicated a similar management strategy in all four units of the AFF under study.

Overall, standard predictors describing forest site yielded only small effects on damage probability (only ALT was found to be informative). This may be explained by a rather low representativeness of site descriptors for entire stand polygons due to substantial small-scale variation of site and soil conditions in mountain forests. Moreover, the spatial variability of site attributes was apparently much larger than that of wind damage, reducing their explanatory power. In the light of these arguments, the inclusion of SF (indicating frozen soil) in the model highlights the importance of the interplay of soil and weather conditions in determining the susceptibility to wind damage. However, SF also highlights the challenge of including attributes in model development which are subject to substantial uncertainty, regarding both space and time dimensions. With SF, this is particularly true to the required match of VMAX and unfrozen soil status. Related uncertainty would have even increased in case of including a stand specific soil water balance calculation which requires spatially accurate estimates of water holding capacity of the soil.

The small effect size of VMAX in the probability model and its exclusion from the intensity model confirmed the hypothesis that incorporating wind speed as a driving factor in our models was a challenge. Analysing whether the relatively coarse spatial resolution of the wind speed data (1x1 km) or the inherent variability in storm damage events was the major reason for the weak effect of VMAX in the probability and intensity models was beyond the scope of the current study. An alternative to using wind speed data is describing the local topographic exposure of forest stands to wind. This method is widely employed in storm damage research and uses indices like topex,

topex-to-distance or other indices that include information on the aspect and slope of stands (see e.g. Quine and White 1998).

There are other empirical studies of storm damage that link wind field characteristics to wind damage (e.g. Schütz et al. 2006; Schindler et al. 2009, 2012). However, they do this for large-scale singular storm events, where the exact timing of a storm is known, and linking the gust speed estimate to the damage event is less problematic. In our case, excluding the weather-related variables (VMAX and SF) from the probability models caused only a small decrease in explanatory power. Nevertheless, the sign of both VMAX and SF, and the effect size of SF in the probability model clearly indicated conceptual consistency and the relevance of including explicit weather-related attributes in wind disturbance models.

In case of the intensity model, the big difference of R^2 between the full model (fixed and random effects) and the fixed effects-only model indicated that the variables used in the study explained only a small proportion of the variance in wind salvage and that implicit local effects could not be revealed with the available data base. This may be due to features of the local wind climate or stand conditions which are not accounted for by the available data. Local adaptation of trees to higher wind speeds can decrease wind disturbance susceptibility (Nicoll et al. 2008). Thom et al. (2013) found that forest stewardship-related attributes (such as ownership or road density) contributed significantly to explaining the variation in damage data at the level of administrative districts.

4.2 Limitations of database and study design

The current study used data from forest management plans. It is important to note that originally, the underlying data had not been compiled for scientific purposes, and therefore just the typical standard set of site and stand characteristics was available for the analysis. Recording errors of timber removals can lead to inconsistent data sets. For instance, the assignment of removals to one of several causes is such a crucial issue. The huge advantage of the database is its spatial coverage and that the same data collection procedures have been used throughout all the studied FMUs. However, linking a disturbance event to a highly stochastic predictor such as gust speed may impose several problems. For the analysis, exact dates of harvests were not available. In case of windthrows, it is possible that a winter storm damages trees in November or

December, but the event only gets registered and the timber salvaged during the next year. This can lead to cases in the model matrix when high gust speeds (>30 ms^{-1}) are linked to no damage (compare Nilsson et al. 2007) in the stand, or high amounts of damaged timber to very low gust speeds (<5 ms^{-1}).

The topography used by INCA is given on a grid with a resolution of 1 km. As such, the terrain model may fail to approximate highly complex structured topography sufficiently well. However, since there is a lack of observational data, the INCA data set provides the nearest approximation of the wind conditions in the regions over a longer period. Further wind downscaling techniques combined with in-situ measurements could improve the analysis of the actual wind conditions within the forest areas. Despite these limitations, we intended to include VMAX in the analysis to test if it contributes to explaining damage from wind disturbances. The relevance of developing predictive models sensitive to changes in wind related drivers becomes evident when analysing climate change impacts on forests and related ecosystem services (e.g. Lindner et al. 2010). The effect size of VMAX in the probability model was relatively small, and it was not even included in the intensity model. However, with a higher spatial resolution of accurate wind speeds and an exact recording of the dates of wind salvage, empirical models for wind damage prediction using wind speed as a predictor could very likely be improved significantly.

Another issue related to the INCA wind speed data was the availability of data from 2003 onward whereas the disturbance damage data was available for the period 1999-2008. However, no straightforward approach was available to fill this gap in the wind speed data. In addition, four years from the 10-year data record were used to build predictor variables characterizing harvest and damage history of stands. Nevertheless, these predictor variables proved to be important in explaining damage events, similar to the findings of Thom et al. (2013), who used salvaged damage from the preceding two years as explanatory variable in a recent study of the wind and bark beetle disturbance regime at landscape scale.

An important aspect of the disturbance regime, the spatial extent of disturbance events inside a stand, or at higher aggregation level inside a compartment, could not be considered. Substantial efforts and additional data (mainly deducible from remote sensing sources) would have been needed to make the database spatially explicit. In our models, effects of damaged neighbour stands were not included as fixed effects for similar reasons. However, the latter practice would have been also contrary to our aim to explore how well predictions could be made from stand level information only.

5 Conclusions

The developed models indicate that decreasing share of spruce, age and timber stock in stands exposed to wind disturbance are options to mitigate the risk and the expected damage intensity. Furthermore, results clearly indicated that previous disturbances from wind, bark beetles and snow increase the risk of wind damage. Intensification of disturbance regimes may therefore be due to self-reinforcing processes, emphasizing the importance of stand stability and resilience in forest management.

It could be shown that standard data from management plans can be used to identify stands at high risk of wind damage. However, expectations that salvage volumes at stand level can be accurately projected from standard data appears as too optimistic.

The big difference in performance between models with fixed and random effects and models without random effects highlighted the importance of unknown attributes that are yet to discover in order to improve wind disturbance modelling. The limited improvement in model performance by including wind speed data points was a weakness on one hand, but indicated that there is potential to further improve empirical wind disturbance models by improved matching of damage events and weather data.

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9.3 Paper III

Comparative analysis of bark beetle and wind disturbance models within a dynamic forest simulation model

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Abstract

Natural disturbances are among the major drivers of forest ecosystem dynamics. It is expected that under climate change conditions disturbance regimes may intensify. As a consequence, disturbance modelling has attracted much attention in recent years and a number of disturbance modules have been developed and integrated into forest ecosystem models. Parameter and structural uncertainty in such modules are huge, and very limited knowledge is available on implications of different model formulations. We analysed two different bark beetle disturbance modules and a wind disturbance module for system behaviour as simulated with a dynamic forest simulation model. Results indicated that bark beetle and wind models having been developed from an extensive empirical database yielded plausible disturbance regimes under current climatic conditions over a wide gradient of stand and site conditions. However, long-term predictions under changing climate did not reveal intensification of the disturbance regime, contrary to a process-based bark beetle disturbance module and expectations in scientific literature. The process-based and the empirical bark beetle models provided substantially different results mostly at low altitudes, highlighting the importance of how process-based terms such as bark beetle phenology are considered in model equations. Limitations of the disturbance models and future development needs are discussed.

Key-words: bark beetle, wind, disturbance, climate change, modelling, PICUS

1 Introduction

Forest ecosystem dynamics are highly affected by natural disturbances, and awareness of consequential problems in providing ecosystem services has been growing in recent years leading to substantial research effort in that field (Lindner et al. 2010). Empirical studies discovering drivers of disturbance regimes highlighted importance of climate, and therefore probable future increases in damage of forest ecosystems in case of changing climatic conditions (Dale et al. 2001). Various models have been created to make analysis of potential future changes of disturbance regimes within ecosystem model simulations possible (see Seidl et al. 2011a). However, mechanistic representation of disturbance processes in models is a difficult task due to heterogeneity and spatio-temporal dynamics of forests. This leads to big differences in model formulations and parameterisation approaches of these models, and implies a need for comprehensive evaluation studies to compare the implications of disturbance models in controlled simulation experiments along various ecological gradients.

In Europe, European spruce bark beetle (*Ips typographus* (L.)) is the most deteriorating biotic disturbance agent to Norway spruce (*Picea abies* [L.] karst.) forests (Christiansen and Bakke 1988). Damage usually correlates with drought periods and warm weather (Stadelmann et al. 2013). Warmer climate will favour the development of two or even three life cycles of bark beetles per year in Central Europe (Jönsson and Bärring 2011), leading to potentially rapid population build-up and subsequent damage in host trees. At the same time, frequency and severity of drought periods may also increase, having a negative effect on tree vigour, and consequently, increasing the vulnerability of such trees to insect infestation (Wermelinger 2004; Marini et al. 2013). Consequences of bark beetle disturbances are not only economic; they may also lead to negative effects on other forest ecosystem services such as carbon sequestration (Seidl et al. 2008a) or the protection of infrastructure from rockfall and avalanches in mountainous landscapes (Brang et al. 2006). The importance of windthrown timber as bark beetle habitat, and hence the interaction between the two disturbance agents has been shown by various studies (e.g. Schroeder 2001, Eriksson et al. 2005, Marini et al. 2013). Beside this interaction, wind disturbance in itself leads to substantial losses in timber and is the most important abiotic disturbance agent in European forests (Schelhaas et al. 2003). These findings indicate the potential for intensifying disturbance regimes under climate change conditions (Seidl et al. 2009; Ogris and Jurc 2010; Hlásny and Turčáni 2013; Temperli et al. 2013)

In modelling the bark beetle disturbance regime, bark beetle phenology models driven by climate data and forest attributes either measured in the field or simulated by forest ecosystem models are frequently used (e.g. Lexer and Hönninger 1998, Seidl et al. 2007a; Jönsson et al. 2012; Temperli et al. 2013). In case of wind, a common approach is that critical wind speeds for uprooting and stem breakage are calculated using stand and site characteristics, and a distribution of wind speeds measured through time is used to calculate event probabilities (e.g. Peltola et al. 1999; Gardiner et al. 2000). These model-based approaches contribute to an improved understanding of the potential future developments of the bark beetle and wind disturbance regime, and therefore can support related decision making in forest management. However, there is a lack of such models that take both bark beetle and wind disturbances with their interactions into account.

A frequently used forest ecosystem model for temperate European forests is PICUS (Lexer and Hönninger 2001; Lexer 2001; Seidl et al. 2005). It is a hybrid forest patch model that contains a bark beetle disturbance sub-model, which has been evolving since the development of its first version (Lexer and Hönninger 1998). Recently, an empirical bark beetle disturbance model (Pasztor et al. 2014a) and an empirical wind disturbance model (Pasztor et al. 2014b) have also been developed and integrated into PICUS.

In this paper, we present a comparative analysis of simulated forest development along various ecological gradients as affected by the use of different disturbance modules for wind and bark beetle disturbances within the PICUS ecosystem model. Specifically, we aimed at (i) the analysis of model behaviour along a gradient of stand and site conditions and (ii) under current climate and climate change conditions, (iii) exploring the interaction of bark beetle and wind disturbances and stand development, and (iv) comparing model results to empirical disturbance data from Austria.

2 Material and methods

2.1 Model description

2.1.1 PICUS v1.6

Development of stands under current climate and transient climate change conditions in the study was simulated with the ecosystem model PICUS v1.6. It is a hybrid forest patch model, which integrates elements of a 3D patch model (Lexer and Hönninger 2001) and a simplified process-based model (Landsberg and Waring 1997). It focuses on ecological realism (e.g. PICUS simulates forest dynamics based on individual trees which are arranged on a grid of 10x10 m patches). Further components of PICUS include a sub-model for the simulation of forest management interventions based on management scripts allowing for high flexibility in terms of spatially and structurally explicit harvesting and planting operations. To keep track of belowground carbon and nitrogen processes and to dynamically update site nutrition status, a biogeochemical process model of carbon and nitrogen fluxes in forest soils (Currie et al. 1999) has been incorporated. It has been successfully applied in simulating inter alia forest C stocks including soil C in a case study at the forest management unit level (Seidl et al. 2007b). The model has been successfully evaluated considering various aspects of forest ecosystem dynamics (e.g. Seidl et al. 2005, 2008b; Didion et al. 2009).

2.1.2 The original bark beetle disturbance module (BBDM-1)

A first bark beetle sub-model for PICUS had been developed by Lexer and Hönninger (1998). It was a two-stage stand risk model based on an earlier hazard rating model for Norway spruce stands by Lexer (1995). The model separated the estimation of damage probability and damage intensity. Later, Seidl et al. (2007a) presented a refined version of the bark beetle module. It incorporates bark beetle phenology (Baier et al. 2007) and a site and stand related predisposition algorithm (Netherer and Nopp-Mayr 2005). The phenology model incorporates swarming and the development of main and filial generations. Swarming starts in the model when both a daylength and an air temperature threshold are met. Brood development depends on bark temperature sums; a new filial sister brood is started when the required heat sum of 557°C (above a threshold of 8.3°C) is reached (Netherer and Pennerstorfer 2001). Bark temperature is calculated using an empirical relationship that takes into account air temperature, incoming global radiation and relative radiation below the canopy. Beetle development stops when daylength drops below 14.5 hours or temperature requirements are no longer met. For more details on the implemented phenology model, see Seidl et al. (2007) and Baier et al. (2007).

The parameterisation of the sub-model was based on a reassessment of the stand and soil data set from Lexer (1995). Weather data (air temperature, precipitation and radiation) were also used, for modelling the beetle phenology; therefore a wide range of ecological driving factors of the bark beetle disturbance regime were taken into account. Estimation of disturbance probability is based on a non-linear relationship (Eq. 1):

$$p = 1 - e^{\left(-1.51 \cdot PI^{1.65}\right)^{GEN}} \qquad (1)$$

where p is annual probability of damage; PI is a stand predisposition index [0-1]; GEN is a relative scoring for completed generations (zero completed generations = 0; one completed generation = 0.1; one generation and one sister brood = 0.2; two generations = 0.6; more than two generations = 1; see Netherer and Nopp-Mayr, 2005). Stand predisposition is defined by share of spruce, stand age, stand density and drought days (see Netherer and Nopp-Mayr 2005). Forest management interventions are not taken into account explicitly but through the change in stand attributes in the predisposition index.

Estimation of damage intensity is based on a logistic regression (Eq. 2):

$$d = \frac{1}{1+e^{3.9725-2.9673SHI_{yr}}}, \qquad (2)$$

where d is annually damaged relative stem number [0-1]; $SHIyr$ is a stand hazard index (annual) [0-1]. Stand hazard is defined by a stand edge index representing the length of exposed open stand edges oriented south and west, share of spruce and a soil moisture index (SMI) over the growing season.

For the selection of infested and killed trees in course of a simulation, the simulated patches are grouped into five equally large classes according to the aboveground live spruce biomass they hold. Proportional to the estimated relative damage, a specific number of patches are randomly selected from the class with the highest biomass to serve as starting point for a bark beetle infestation cluster. The first infestation spot is picked, and trees (with dbh > 10 cm) are killed in that patch. Then, trees in the surrounding eight patches are killed patch by patch (Fig. 1) until the accumulated volume of these killed trees equals the estimated damage. Selecting which of the adjacent patches are affected by the damage is a random process. If all the trees (with dbh > 10 cm) are removed from the infestation spot and all the surrounding patches, and the total timber removal up until this point does not reach the estimated damage intensity value, then the next infestation spot is picked and the same process starts again.

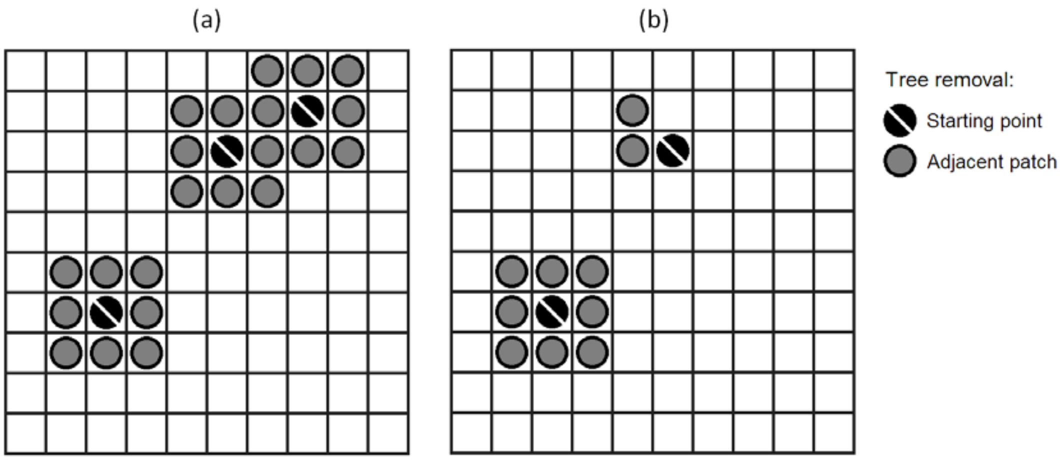

Fig 1 Schematic representation of the selection process for bark beetle infested trees in a hypothetically simulated forest of 10x10 patches. Starting spots are selected randomly from among the patches with the highest Norway spruce biomass proportional to the relative damage estimate (a). Then all spruce trees are killed in patches randomly selected from the ring patches surrounding the center spot (b). An additional starting point is only used when spruce trees in the patches around the previous starting point have all been killed. In the example, after 12 patches the volume of the estimated damage intensity (relative damaged stem number) was reached

This disturbance module (named BBDM-1 in this study) has been evaluated (Seidl et al. 2007a) and used in several studies (e.g. Seidl et al. 2008a, 2011b).

2.1.3 A new empirical bark beetle disturbance sub-model (BBDM-2)

The possible advantages of further development of modelling bark beetle disturbances in PICUS were due to a few remaining but potentially important issues of BBDM-1. Such issues were the fairly small empirical data set the algorithms were calibrated to and the lack of other disturbance factors considered. A new bark beetle disturbance sub-model (here called BBDM-2) was developed based on work by Pasztor et al. (2014a). BBDM-2 is based on statistical modelling; models were fitted to a large database comprising of forest stand, site and weather data and harvest records. The study area covered four regional forest management units of approx. 40,000 ha forest (approx. 8,000 forest stands) of the Austrian Federal Forests (AFF) representing a wide range of stand and site types of the Austrian Alps. The two-step approach, i.e. modelling the probability and the intensity of damage, was retained. Both probability and intensity are modelled annually at the stand level. Model variables are shown in Table 1. Weather data are used to estimate the potential number of bark beetle generations in a given year, i.e. the biotic pressure on the host trees. The number of potential annual bark beetle generations in the previous year (including filial generations; $BGEN_1$) is derived from the phenology model as described in Seidl et al. (2007a).

Table 1 Variables used in the bark beetle disturbance sub-model (BBDM-2). The probability model uses all the variables from the list. The intensity model uses the variables marked with *

Variable	Unit	Description
ALT *	M	Altitude above sea level
SL	°	Slope steepness
ASP	[0,1]	Aspect [1: SE, S, SW; 0: W, NW, N, NE, E]
PA *	%	Share of Norway spruce (*Picea abies* [L.] Karst.)
AGE *	Years	Mean stand age
VOL *	$m^3\ ha^{-1}$	Timber stock volume before harvests of actual year
W_4	$m^3\ ha^{-1}\ 4\ years^{-1}$	Wind damage in the previous 4 years
B_4 *	$m^3\ ha^{-1}\ 4\ years^{-1}$	Bark beetle damage in the previous 4 years
S_4	$m^3\ ha^{-1}\ 4\ years^{-1}$	Snow damage in the previous 4 years
R_4	$m^3\ ha^{-1}\ 4\ years^{-1}$	Regular harvests in the previous 4 years
$BGEN_1$ *	$N\ year^{-1}$	Potential annual bark beetle generations in the previous year (including filial generations)

The equation used for modelling the annual probability of a bark beetle disturbance event is:

$$p = (\exp^{\alpha+\beta\cdot X}) \,/\, (1 + \exp^{\alpha+\beta\cdot X}), \qquad (3)$$

where *p* is the annual probability of damage; α is an intercept, β is a vector of the coefficients of the predictor variables; *X* is a vector of the predictor variables (see Table 1).

The intensity of a bark beetle disturbance event is defined by Eq. (4).

$$d = \alpha + \beta \cdot X, \qquad (4)$$

where *p* is the intensity of damage [m^3 ha^{-1} $year^{-1}$]; α is an intercept, β is a vector of the coefficients of the predictor variables; *X* is a vector of the predictor variables (see Table 1).

In simulating damage intensities, we used a method frequently employed in fire damage modelling (Shpilberg 1977), where the mean damage value from the empirical data is used to define a lognormal distribution of damage intensities. In the simulations, damage estimates are randomly drawn from this distribution. For the disturbance module BBDM-2, we have taken up this approach, but we shift the mean damage from the empirical database used in model development (see Pasztor et al. 2014a) depending on the actual estimate from Eq. (4). With this modification, the distribution of the random draws fitted the distribution of the observed damage intensities very well also in case of extreme values (not shown here).

Selecting which trees get removed in the PICUS simulation when a bark beetle disturbance event is predicted follows the procedure of BBDM-1.

2.1.4 The wind disturbance sub-model (WDM)

An empirical model for wind disturbance was developed by Pasztor et al. (2014b), and integrated into PICUS (WDM). The modelling approach was identical to BBDM-2, and with some exception, the same database was used for fitting the models. Stand and site variables remained the same as in BBDM-2 (Table 2). Regarding weather-related variables, 2-second maximum daily gust speed (VMAX) is computed from 10-min maximum daily wind speed at full hours using a gust factor of 1.65 (Cvitan, 2003). Daily air temperatures are used to calculate if the soil was frozen at the time of the highest gust speed of the year (SF [0,1]). If the uppermost 10-cm layer of the soil is estimated to be frozen according to Paul et al. (2004), then it is assumed to have a stabilizing effect against windthrow. Calculating the soil temperature is based on the mean annual and summer air temperature and the minimum and maximum air temperature of the current day and also on leaf area index. For more details on the soil temperature model, see Paul et al. (2004) and Pasztor et al. (2014b).

Table 2 Variables used in the wind disturbance sub-model (WDM). The probability model uses all the variables from the list. The intensity model uses the variables marked with *

Variable	Unit	Description
ALT	M	Altitude above sea level
PA *	%	Share of Norway spruce (*Picea abies* [L.] Karst.)
AGE	Years	Mean stand age
VOL *	$m^3\ ha^{-1}$	Timber stock volume before harvests of actual year
W_4 *	$m^3\ ha^{-1}$ 4 $years^{-1}$	Wind damage in the previous 4 years
B_4	$m^3\ ha^{-1}$ 4 $years^{-1}$	Bark beetle damage in the previous 4 years
S_4	$m^3\ ha^{-1}$ 4 $years^{-1}$	Snow damage in the previous 4 years
VMAX	$m\ s^{-1}$	Highest daily 2-sec gust speed 10 m above the soil surface
SF	[0,1]	Soil state on the day with the highest 2-sec gust speed (1: frozen; 0: not frozen)

The parameterised equation for modelling the annual probability of a wind disturbance event is as Eq. (3). Predictor variables are shown in Table 2.

The intensity [$m^3\ ha^{-1}\ year^{-1}$] of a wind disturbance event is defined as in Eq. (4). Predictor variables are shown in Table 2. In simulating the individual damage intensities, the same approach as for BBDM-2 is used.

To determine the damaged trees in a wind disturbance event, the algorithm of the sub-model selects (i) the patches with the highest trees, and (ii) those patches that are close to the former group. The first step is to screen the top height per patch, i.e. the height of the tallest individual in a patch. Then from among those patches a random selection determines the starting points of damage. The number of starting points is proportional to the intensity of the disturbance. The second step is to disperse the rest of the damage in patches that are close to the initial kernel patches from step one. Damage spreads from the starting points as follows (see Fig. 2): the chance of tree damage is 66% for patches of the first adjacent ring (eight patches), and 33% for patches on the following ring (16 patches); all other patches have zero probability of being damaged. The resulting probability map is then processed in a top-down approach from patches with high to patches with low probabilities. All trees with a height above 10 m are killed in each patch until the prescribed damaged volume [m³ ha^{-1} year^{-1}] is reached. If the initially determined starting points are not sufficient to cover the estimated damage intensity, additional patches are selected as centre of a damage. The approach followed here was meant to mimic the pattern of windthrow events, i.e. the damage starts at those trees that are exposed to the highest wind loadings and can spread from those patches due to the newly created edges and the contact with falling trees (Seidl et al. 2014).

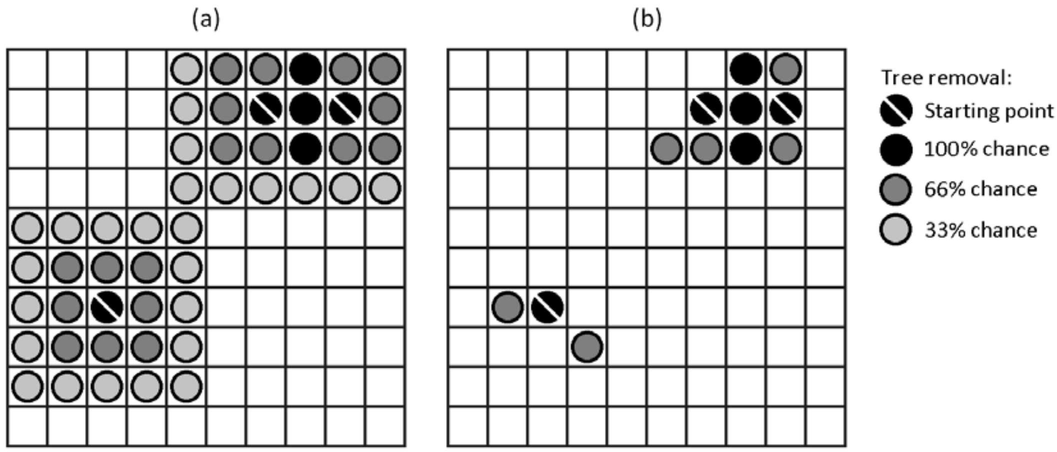

Fig. 2 Example of the selection process of removed trees in case of a wind disturbance event on a hy pothetical map of patches. The algorithm starts form the white-striped black dots in (a). Chances of tree removal are increased in surrounding patches (a) indicated by brighter dots. Note that increased risk is assigned to patches in the vicinity of several starting points. The finally affected patches are shown in (b): In this example, after 12 patches the predicted damaged volume was reached

2.1.5 Integrating the disturbance sub-models in PICUS

The disturbance modules can be used independently during the simulations. In each year of the simulation, a damage probability is calculated (Eq. 1) taking into account the current stand and weather conditions as well as the simulated damage and harvests of the previous years (Tables 1 and 2), thus the interactions follow the dynamics of the forest stand. The calculated probabilities are compared to random numbers between zero and one, and if the random number is below the calculated probability, then a disturbance event occurs in that year of the simulation. In case of both new modules (BBDM-2 and WDM) being enabled, the wind disturbance module is run first and the bark beetle module afterwards. Note, that in this case there is no direct interaction of WDM and BBDM-2, i.e. the wind damage of the current year is not taken into account by BBDM-2. However, the interaction manifests in the legacy effects of disturbances of previous years, i.e. via the predictor variables W_4 and B_4, the cumulated damage of the last four years by wind and bark beetle, respectively (see Tables 1 & 2). If a regular harvest occurs in the actual year of the simulation, it is implemented before calculating the disturbance probabilities and intensities thus the disturbance modules take into account the lower timber stock volume. Table 3 presents a comparative description of the disturbance models.

Table 3 Description of the disturbance models (BBDM-1, BBDM-2, WDM) used in the study. Spruce stands for *Picea abies* [L.] Karst, bark beetle for *Ips Typographus* [L.]

	BBDM-1	BBDM-2	WDM
Objectives	Modelling bark beetle disturbance probability and intensity	Modelling bark beetle disturbance probability and intensity	Modelling wind disturbance probability and intensity
Spatial resolution	Forest stand	Forest stand	Forest stand
Timestep	Year	Year	Year
Stand variables	Share of spruce, stand age, stand density	Share of spruce, stand age, stand volume	Share of spruce, stand age, stand volume
Site variables	Soil moisture index (SMI) over the growing season, stand edge index	Altitude, slope steepness, aspect	Altitude
Climate-related variables	Potential number of bark beetle generations in the actual year	Potential number of bark beetle generations in the previous year (with sister broods)	Highest daily 2-sec gust speed 10 m above the soil surface, soil state on the day with the highest 2-sec gust speed (1: frozen; 0: not frozen)
Disturbance-related variables		Wind salvage in the previous four years, bark beetle salvage in the previous four years, snow salvage in the previous four years, regular harvests in the previous four years	Wind salvage in the previous four years, bark beetle salvage in the previous four years, snow salvage in the previous four years
Evaluation	Seidl et al. 2007a	Pasztor et al. 2014a	Pasztor et al. 2014b

2.2 Simulation experiments

2.2.1 Simulation design

In designing the simulation setup, requirements were (i) to cover typical stand and site conditions in the Eastern Alps, (ii) to include both pure spruce and mixed stands, (iii) to consider an altitudinal site gradient and (iv) to include also possible transient effects of climate change. Therefore, two site types (differing in water holding capacity; WHC) and three altitudes (A1: 400 m , A2: 900 m , A3: 1400 m ; see also Table 4) were selected. In addition, for each altitude species mixture types were defined (Table 5). Other input variables were kept constant; namely slope steepness (20°), soil pH (5) and soil N (62.5 kg/ha/year). The de-trended baseline climate scenario (BL) was produced by randomly sampling years from the measurement period 1960-1990 in the region Steiermark assigned to the specific altitudes set in the simulation design. The climate change scenario (CC) used in the simulation experiment was based on the SRES A1B greenhouse gas emission scenario (Nakicenovic and Swart 2000) simulated by the global climate model (GCM) ECHAM5 (Roeckner et al. 2003) and the regional climate model (RCM) REMO (Jacob and P odzun 1997). Mean annual air temperature and precipitation for BL were 8.2° C and 1093 mm at A1, 6.3° C and 1158 mm at A2, 4.3° C and 1208 mm at A3, respectively. For CC, they were 11.8° C and 1085 mm at A1, 9.8° C and 1214 mm at A2, 7.9° C and 1266 mm at A3, respectively in the last 30 years of the simulation period. To represent the wind climate, measurements from the period 2003-2010 were randomly sampled from the database used for the development of WDM (Pasztor et al. 2014b). The same wind speed time series was used for all the climate scenarios and altitudes in the simulations. Soil frost state (SF) however was calculated in each year according to the air temperature values provided by the climate scenarios.

Table 4 Description of the variables used in the simulations with more than one predefined value. Spruce stands for *Picea abies* [L.] Karst, beech for *Fagus sylvatica* [L.], larch for *Larix decidua* [L.]

Variable	Abbreviation	Description
Altitude	A1	400 m a.s.l.
	A2	900 m a.s.l.
	A3	1400 m a.s.l.
Site type	ST1	Water holding capacity of 100 mm
	ST2	Water holding capacity of 200 mm
Species mixture type	S	100% spruce
	SB1	80% spruce, 20% beech
	SB2	50% spruce, 50% beech
	SL1	80% spruce, 20% larch
	SL2	50% spruce, 50% larch

Table 5 Design matrix of the comparative analysis of the disturbance sub-models setups (ND, BBDM-1, BBDM-2, BBDM-2&WDM) used in the simulations run with PICUS v1.6. The same simulation design was used for the baseline climate (BL) and the climate change scenarios (CC) and the managed forest (MF) and unmanaged forest (UMF) scenarios. See Table 4 for a description of the abbreviations. See Table A1 in the Appendix for a description of forest management

		S	SB1	SB2	SL1	SL2
A1	ST1	X	X	X		
	ST2	X	X	X		
A2	ST1	X	X	X		
	ST2	X	X	X		
A3	ST1	X			X	X
	ST2	X			X	X

Each simulation was started with 4000 saplings (height class of 10-30 cm) in the year 2000 on an ar ea of 1 ha. In case of the managed forest scenario (MF), forest management was simulated in a s imple form with a total of four management interventions in the 100 years of the simulation period (Table A1 in the Appendix). A final cut was not part of the simulation. Natural regeneration was limited to originating from seed trees within the simulation. For benchmark purposes, an unmanaged forest scenario (UMF) was also simulated. The same simulation design described above was used for the simulations run with different disturbance sub-models enabled. Beside the no disturbance-option (i.e. all the disturbance sub-models disabled; ND) the following options were tested: (i) only BBDM-1 used; (ii) only BBDM-2 used; (iii) BBDM-2&WDM used. ND provided the basis for evaluation of general growth patterns.

2.2.2 Comparison of sub-models

For the analysis, we divided the simulated period into three 30-year periods between 2010 and 2099 (P1, P2, P3). Due to the probabilistic nature of the disturbance modules, multiple replicated simulations were run to yield robust results. The number of required replicates was investigated in a preliminary analysis, and it was found that 10 simulation replicates for BBDM-1 and hundr ed replicates for BBDM-2 and W DM provided stable outcomes. The different number of required replications between the old model and t he new models is due t o the different way of incorporating stochastic elements in the models. Mean total production (MTP), mean volume of standing timber stock (MSTS) and mean annual damage (MAD) were the output variables used for the analysis. For the major part of the analysis, we focused on t he last 30-year period of the simulations (P3), i.e. on f orest stands that are between the age of 71 and 100 years, as this period showed the highest contrasts between different scenarios. To test the significance of the variables used in the simulations and the first order interactions between them, a generalized linear model (GLM; with Gaussian error distribution) was fitted to the simulation matrix with one dependent variable (MSTS, TP, MAD) at a time, and then an analysis of variance (ANOVA) was conducted. To compare the differences between the various levels of the significant factor variables, Tukey's HSD (honest significant difference) test was employed. Dependent variables were natural log-transformed to improve homogeneity of variance.

Beside comparing the different sub-model setups, the simulation output of managed forests under the baseline climate scenario was also evaluated against empirical data of the forest management unit Steiermark from the database used in Pasztor et al. (2014a, b). Empirical data were binned in a way that the resulting database for model evaluation was consistent with the simulation setup (Table 6). Stands with an age of

70-100 years in the empirical database were compared to the last simulation period (P3) under BL climate. For simulations and empirical data, mean annual damage (MAD), the number of disturbance events in the 30-year period and the corresponding damage intensities were selected to analyse the functioning of the models and the emerging simulated disturbance regimes. Since empirical data were only available for a 10-year period (Pasztor et al. 2014a, b), the number of events were multiplied by three to match the length of the simulation period. Several categories in the empirical database contained a fairly low number of forest stands. Therefore, only Se (100% spruce), SB1e (70-90% spruce, 10-30% beech) at A2e (altitude of 800-1000 m a.s.l.; number of stands were 72 and 15, respectively) and Se, SL1e (70-90% spruce, 10-30% larch) at A3e (altitude of 1300-1500 m a.s.l.; number of stands were 63 and 26, respectively) were used for the evaluation. For the comparison of the simulations and the empirical data, we did not differentiate between site types, as reliable information on that specific characteristic was not available in the empirical database.

Table 6 Categories of the binned empirical data (Pasztor et al. 2014a, b) of stands with an age of 70-100 years used for an evaluation of the disturbance models BBDM-1, BBDM-2 and WDM. Spruce stands for *Picea abies* [L.] Karst, beech for *Fagus sylvatica* [L.], larch for *Larix decidua* [L.]

Variable	Abbreviation	Description
Altitude	A1e	300-500 m a.s.l.
	A2e	800-1000 m a.s.l.
	A3e	1300-1500 m a.s.l.
Species mixture type	Se	100% spruce
	SB1e	70-90% spruce, 10-30% beech
	SB2e	40-60% spruce, 40-60% beech
	SL1e	70-90% spruce, 10-30% larch
	SL2e	40-60% spruce, 40-60% larch

3 Results

3.1 Simulated mean standing timber stock

All the main effects and first order interactions were significant ($p<0.05$) when mean standing timber stock (MSTS) in P3 was used as dependent variable. The natural log transformation of MSTS improved homogeneity considerably (not shown here).

Scrutinizing the mean differences with Tukey's HSD test revealed significant differences between the levels of the independent factor variables (Table 7). Employing different disturbance models during the simulations had a clear significant effect on the outcomes. The only non-significant difference was between BBDM-2 and ND. Considering the species mixture types, pure spruce stands differed significantly (lower MSTS) from stands with admixed tree species ($p<0.001$). Differences between species mixture types with admixture trees were all non-significant (note that initial species shares remained relatively stable during the simulations), except for the higher MSTS of SB2 compared to SB1 ($p<0.01$). Differences of the distinct levels of each other model variable not described here were found significant ($p<0.001$); for the signs of differences see Table 7.

To visualize the effect of using different disturbance models, the mean standing stocks were plotted for a typical forest stand in Eastern Alpine conditions (Fig. 3) across the three simulation periods for both the BL and the CC scenario at site A2 (900 m a.s.l.). When none of the disturbance models was used, timber stock values were almost identical in the two climate scenarios. Also, mean values and variances of BBDM-2 and BBDM-2&WDM in CC remained very similar (a difference of several m^3 only) to that of the baseline climate. On the other hand, using BBDM-1 in the simulations led to a large difference in standing stock, mostly in P3 (513.6 m^3/ha for BL and 414.7 m^3/ha for CC). This difference was even more apparent at low altitude (A1; other variables fixed at the same level as for Fig. 3), where mean standing stock were 424.0 m^3/ha and 219.2 m^3/ha, respectively. The latter comparison revealed a much smaller difference when BBDM-2 was used (487.6 m^3/ha for BL and 390.6 m^3/ha for CC). At high altitude (A3), differences between BL and CC were small for all the disturbance models in general (up to a few tens of m^3).

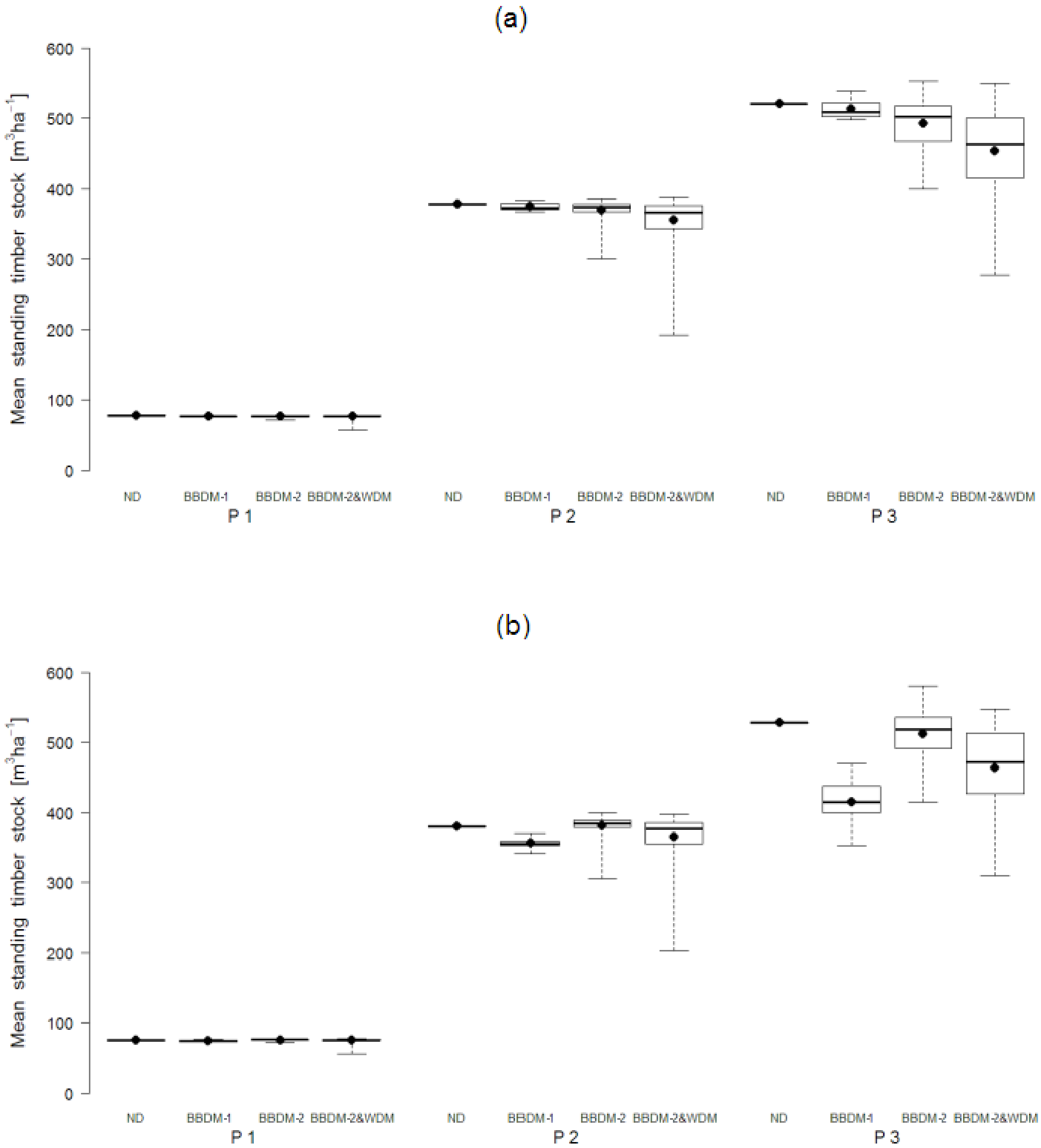

Fig 3 Mean volume of standing timber stock in the simulation periods 2010-2039 (P1), 2040-2069 (P2) and 2070-2099 (P3) in managed pure Norway spruce (*Picea abies* [L.] Karst) forest stands at medium altitude (900 m; A2) with water holding capacity of 200 mm under baseline climate (a) and a transient climate change scenario (b). X-axis labels denote which disturbance sub-models were used in the simulations of PICUS v1.6: ND – none of the sub-models used (*n*=1); BBDM-1 = earlier bark beetle disturbance sub-model used (*n*=10); BBDM-2= new bark beetle disturbance sub-model used (*n*=100); BBDM-2&WDM = new bark beetle disturbance sub-model and new wind disturbance sub-model used (*n*=100). Whiskers extend to the most extreme data point. Black dots denote mean values. *n* denotes the number of replicated simulations

Table 7 Tukey's HSD (honest significant difference) test of levels of factor variables for the simulation period 2070-2099. MSTS: mean standing timber stock [m^3/ha] in the simulation period 2070-2099; TP: mean total production [m^3/ha/year] in the full simulation period; MAD: mean annual damage caused by bark beetle disturbances [m^3/ha/year] in the simulation period 2070-2099. Significance level: * <0.05; ** <0.01; *** <0.001

		MSTS		MTP		MAD	
Variable	Contrast	Sign	p-value	Sign	p-value	Sign	p-value
Disturbance model	BBDM-1 - ND	-	<1e-04 ***	-	0.123	+	< 1e-06 ***
	BBDM-2 - ND	-	0.225	-	0.791	+	< 1e-06 ***
	BBDM-2&WDM - ND	-	<1e-04 ***	-	0.009 **	+	7.35e-06 ***
	BBDM-2 - BBDM1	+	<1e-04 ***	+	0.067	-	< 1e-06 ***
	BBDM-2&WDM - BBDM-1	+	<1e-04 ***	-	0.038 *	-	< 1e-06 ***
	BBDM-2&WDM - BBDM-2	-	<1e-04 ***	-	<0.001 ***	-	3.92e-05 ***
Climate scenario	CC-BL	-	<2e-16 ***	-	<2e-16 ***	+	0.037 *
Altitude	A1-A2	-	<2e-16 ***	-	<2e-16 ***	-	<1e-10 ***
	A3-A2	-	<2e-16 ***	-	<2e-16 ***	-	<1e-10 ***
	A3-A1	-	<2e-16 ***	-	<2e-16 ***	-	<1e-10 ***
Management	UMF-MF	+	<2e-16 ***	+	<2e-16 ***	+	<2e-16 ***
Species mixture	SB1-S	+	<0.001 ***	+	0.344	-	<1e-05 ***
	SB2-S	+	<0.001 ***	-	< 1e-04 ***	-	<1e-05 ***
	SL1-S	+	<0.001 ***	+	0.693	-	<1e-05 ***
	SL2-S	+	<0.001 ***	-	0.004 **	-	<1e-05 ***
	SB2-SB1	+	0.003 **	-	< 1e-04 ***	-	<1e-05 ***
	SL1-SB1	+	0.269	-	1.000	+	1.25e-05 ***
	SL2-SB1	+	0.118	-	<0.001 ***	+	0.999
	SL1-SB2	-	0.999	+	<0.001 ***	+	<1e-05 ***
	SL2-SB2	+	0.999	+	1.000	+	<1e-05 ***
	SL2-SL1	+	0.988	-	< 1e-04 ***	-	<1e-05 ***
Site type	ST2-ST1	+	<2e-16 ***	+	<2e-16 ***	+	<2e-16 ***

3.2 Simulated mean total production

All the main effects and first order interactions were significant ($p<0.001$) when mean total production (MTP) in the full simulation period was used as dependent variable. The natural log transformation of MTP improved homogeneity considerably (not shown here).

Tukey's HSD test revealed significant differences between the levels of each two-level factor variable and also of the different altitudes (Table 7). Higher shares of admixed tree species (SB2, SL2) led to significantly lower MTP, following the pattern of the different production potentials of the tree species ($p<0.01$). When WDM was used in the simulations, it led to significantly lower MTP compared to simulations without wind disturbance ($p<0.05$).

Mean total production at intermediate and high altitudes remained similar to baseline values or slightly increased (a few tens of m^3 ha^{-1} $year^{-1}$) in case of CC, depending on the other model variables. Nevertheless, there was a large difference at low altitude (a decrease from 9.81 m^3 ha^{-1} $year^{-1}$ to 7.90 m^3 ha^{-1} $year^{-1}$ in average across all levels of factor variables between BL and CC; this difference also remained approx. the same when none of the disturbance modules was used).

3.3 Simulated mean annual damage caused by bark beetle disturbances

All the main effects and first order interactions were significant ($p<0.05$) when mean annual damage (MAD) caused by bark beetle disturbances in P3 was used as dependent variable. The natural log transformation improved homogeneity considerably (not shown here).

Tukey's HSD test showed significant differences between each level of each variable ($p<0.05$), except for the difference between SL2 and SB1: higher MAD for SL2 was found not significant (Table 7).

MAD caused by bark beetle and wind disturbances is shown in Fig. 4. In the BL scenario, mean annual bark beetle damage of the different models were very similar (approx. 1 m^3/ha/year). In the transient climate change scenario, BBDM-2 (both with and without WDM) simulated almost identical mean annual damage to the baseline scenario. However, BBDM-1 predicted a mean damage approx. four times higher than in BL.

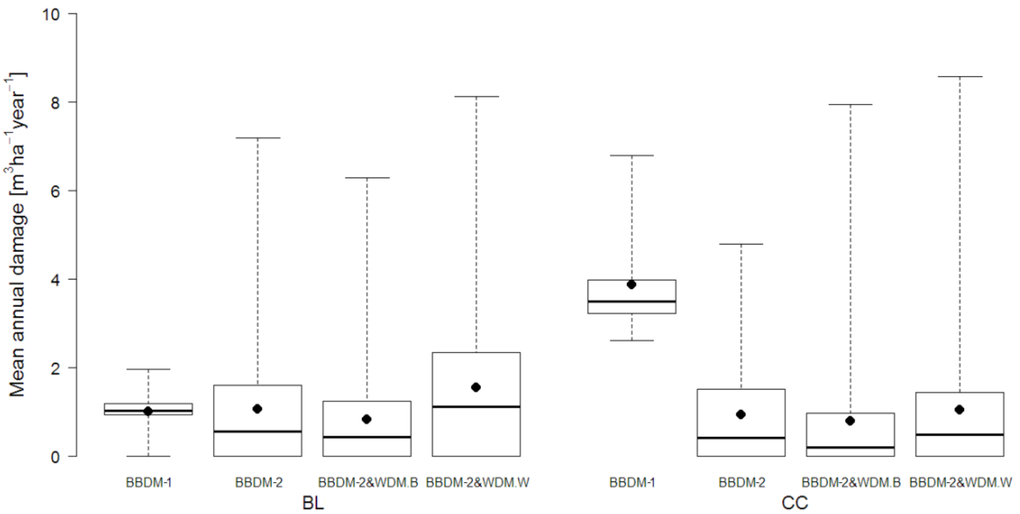

Fig 4 Mean annual damage [$m^3ha^{-1}year^{-1}$] caused by simulated disturbances in the simulation period 2070-2099 in managed pure Norway spruce (*Picea abies* [L.] Karst) forest stands at medium altitude (900 m; A2) with water holding capacity of 200 mm under baseline climate (BL) and a transient climate change scenario (CC). X-axis labels denote which disturbance sub-models were used in the simulations of PICUS v1.6: BBDM-1= earlier bark beetle disturbance sub-model used (*n*=10); BBDM-2 = new bark beetle disturbance sub-model used (*n*=100); BBDM-2&WDM.B = new bark beetle disturbance sub-model and new wind disturbance sub-model used (bark beetle damage shown; *n*=100); BBDM-2&WDM.W = new bark beetle disturbance sub-model and new wind disturbance sub-model used (wind damage shown; *n*=100). Whiskers extend to the most extreme data point. Black dots denote mean values. *n* denotes the number of replicated simulations

3.3 Comparison of the simulated and observed disturbance regime

BBDM-1 slightly overestimated the number of bark beetle disturbance events for A2 and underestimated for A3, while the results with BBDM-2 and WDM better matched the observations (Table 8). With regard to intensities of disturbance events, all the models predicted values relatively close to the observed ones. BBDM-1 had very small variance in the predictions, while the new sub-models could mimic the observed pattern closely, although overestimating values in general. When looking at mean annual damage, it is apparent that BBDM-2 predictions are close to the observed values. It is also true for BBDM-1 at A2, but it estimated very low values at A3. WDM estimated mean annual damage of wind disturbance close to observations at A3, but overestimated at A2.

Table 8 Number of disturbance events, intensity of disturbance events [$m^3 ha^{-1} year^{-1}$] and mean annual damage [$m^3 ha^{-1} year^{-1}$] (MAD) due to disturbances in the simulation period 2070-2099 (P3) compared to empirical data used for model development in Pasztor et al. 2014a, b. For a description of the abbreviations, see Table 3. Numbers in the table denote mean values. Numbers in brackets in the table denote standard deviation values. See Table 6 for characteristics of the observation data set. Number of available observation data points (forest stands): *n*(A2,S) = 72; *n*(A2,SB1) = 15; *n*(A3,S) = 63; *n*(A3,SL1) = 26. Numbers of disturbance events from the 10-year observation period were multiplied by three to match the length of P3

	BBDM-1				BBDM-2				BBDM-2&WDM				Observed			
	A2		A3		A2		A3		A2		A3		A2e		A3e	
	S	SB	S	SL	S	SB	S	SL	S	SB	S	SL	Se	SBe	Se	SLe
Events (bark beetles)	3	2.75	0.25	0.2	1.18	0.94	0.92	0.77	1.42	0.82	0.82	0.77	1.96	1.25	1.45	1.22
	(1.81)	(1.37)	(0.44)	(0.41)	(1.27)	(1.17)	(1.11)	(0.99)	(1.44)	(1.08)	(0.95)	(0.98)	(3.75)	(2.5)	(3.08)	(2.61)
Intensity (bark beetles)	9.59	8.69	4.93	4.39	22.8	19.77	13.12	13.08	20.63	17.74	10.48	11.2	13.51	8.79	8.93	9.97
	(1.31)	(0.96)	(1.33)	(0.34)	(15.65)	(13.89)	(8.08)	(10.52)	(13.33)	(12.5)	(5.9)	(7.7)	(18.34)	(6.32)	(5.83)	(9.2)
MAD (bark beetles)	0.98	0.78	0.04	0.03	0.97	0.65	0.42	0.33	1.02	0.49	0.29	0.29	0.97	0.41	0.46	0.44
	(0.59)	(0.36)	(0.08)	(0.06)	(1.36)	(1)	(0.67)	(0.53)	(1.34)	(0.78)	(0.41)	(0.44)	(5.99)	(2.27)	(2.38)	(2.75)
Events (wind)									1.31	1.09	1.24	1.2	1.22	0.88	1.71	1.94
									(1.36)	(1.26)	(1.3)	(1.21)	(3.15)	(2.69)	(2.95)	(3.65)
Intensity (wind)									29.43	28.43	23.05	20.45	9.72	8.54	15.46	15.23
									(22.86)	(23.75)	(16.89)	(13.97)	(8.06)	(6.13)	(31.55)	(27.14)
MAD (wind)									1.31	1.09	1.24	1.2	0.4	0.25	0.88	0.98
									(1.36)	(1.26)	(1.3)	(1.21)	(2.51)	(1.75)	(8.26)	(7.7)

4 Discussion and Conclusions

The study showed that considering natural disturbances in simulations of ecosystem models leads to significant differences in structural characteristics of forests, as shown by scenarios ND versus runs with different disturbance modules enabled. This is confirmed by a large number of other studies (e.g. Franklin et al. 2002; He et al. 2002; Littell et al. 2011). However, the results of the simulations with the different bark beetle disturbance models under climate change conditions indicated also substantial differences between runs with different disturbance models despite their close agreement under current climatic conditions. In particular, BBDM-1 estimated a large increase in damaged timber due to bark beetle-related mortality under the cc scenario.

This behaviour is related to how the models incorporate climatic attributes. BBDM-1 and BBDM-2 both use the number of bark beetle generations in a given year to take into account the effects of the weather on bark beetle phenology. However, the way this term is included in the model equations (Eqs. 1 & 3) is different. Therefore, an increase in the number of generations results in a differing increase in the estimated probabilities in the two models. When the number of generations is two or above, BBDM-1 estimates a rather high probability of damage. This leads to frequent disturbance events in the simulations under transient climate change in this study. On the other hand, the corresponding damage intensities remained close to the values estimated under the baseline climate. In summary, BBDM-1 predicted a large number of small-scale events in a warmer climate. Mean annual damage (MAD) is in fact the joint result of the number of disturbance events and the corresponding intensities. As MAD in the simulations with BBDM-1 showed good agreement with observed values in the current analysis (in an area outside of the one that was used for its model parameterization), we conclude that its use for simulations is recommended, with mean annual damage as the output describing bark beetle-related tree mortality.

When BBDM-2 was used in the simulations under the transient climate change conditionsthe effect of bark beetle generations BGEN increased both probabilities and intensities estimated by the model. However, at the same time decreasing standing timber stock (variable VOL) tended to decrease the disturbance estimates, which ultimately resulted in similar estimates of MAD under current climate and climate change conditions. The biological realism of this model behaviour is questionable, as many studies confirm that higher bark beetle population densities (due to the warmer climate in this case) may lead to increased damage (e.g. Seidl et al. 2009; Ogris and Jurc 2010; Jakus et al. 2011; Marini et al. 2012; Stadelmann et al. 2013). Despite the large and diverse database that was used for fitting the empirical model BBDM-2, its obvious inability to respond to climatic changes confirmed the general view of empirical models as being hardly capable of reliable long-term future predictions under a changing climate (Korzukhin et al. 1996).

The inclusion of wind disturbances in the simulations induced significant differences in the simulated outcomes. This highlights the potential importance of interdependencies among different disturbance agents (Paine et al. 1998). However, their inclusion in ecosystem simulation imposes various challenges. First, there is substantial lack of understanding regarding many disturbance agents which induces additional model uncertainty; and second, empirical data to develop, calibrate and evaluate such coupled and interrelated multi-agent disturbance models are scarce. Robust, consistent and plausible behaviour under a diverse set of conditions may be considered as more important than high accuracy under a limited set of conditions.

A particular challenge in disturbance modelling is the consideration of effects of forest protection measures such as proactive fellings of vulnerable or already infested trees in a forest. In most cases, this information is not available for empirical data records. One solution for this dilemma may be the inclusion of qualitative expert knowledge in model development.

Overall, it is concluded that scenario analysis of how climate change and adapted management affect forest development and related ecosystem services must consider disturbances in order to deliver useful information in forest management decision support. Approaches that estimate risk or predisposition indices without explicitly simulating disturbance events are considered inappropriate due to the manifold feedback relationships in the simulated ecosystem.

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Appendix

Table A1 Characterisation of the stand treatments in the managed forest (MF) scenario. Removals are expressed in removed tree stem number for the first two treatments and percent volume removed for five relative diameter classes (RDC; class-width = largest dbh minus smallest dbh divided by five; RDC1 = smallest diameter class) for the last two. Trees originating from regeneration (i.e. not present at the initialisation phase of the simulations) were not considered in calculating RDCs

Description of intervention	Stand age (years)	Removal (% volume of standing stock) RDC1	RDC2	RDC3	RDC4	RDC5
pre-commercial thinning (reduction to 1500 stems/ha at random)	20	-	-	-	-	-
pre-commercial thinning (reduction to 1000 stems/ha at random)	30	-	-	-	-	-
selection thinning (reduction to approx. 700 stems/ha)	60	0	15	35	45	0
selection thinning (reduction to approx. 400 stems/ha)	80	0	15	35	45	0

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