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NATURAL AND ARTIFICIAL REGENERATION OF MONTANE *PICEA ABIES* FORESTS IN A CLEARED WINDTHROW AREA IN VITOSHA NATURE PARK

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Abstract

The regeneration of montane Norway spruce (*Picea abies* (L.) Karst.) forests is among the most challenging silvicultural issues due to the extreme climatic conditions at high altitudes, particularly after large-scale natural disturbances. The aim of the study was to compare the development of the natural and artificial regeneration in the period 2006–2014 in a cleared windthrow area of 14 ha, which resulted from a wind-disturbance in 2001. The study area was situated in Ofeliite site, Vitosha Mountain, West Bulgaria. The forest regeneration was studied in 16 permanent sample plots of size 10 m × 10 m (100 m^2). We found that the intensity of natural regeneration decreased significantly five years after the windthrow. Dominant species was *Picea abies*, which regenerated most frequently on decaying deadwood. The artificial regeneration had 5 times lower numbers than the natural one. This could be due to the low planting density and the higher mortality rate of planted spruce seedlings in comparison to the naturally established ones. The risk of mortality was highest among saplings lower than 0.5 m. The main reasons for the observed mortality were the withering and suppression by other plant species such as *Rubus ideaus* L. and *Calamagrostis arundinacea* (L.) Roth. Our results suggest that the natural regeneration was more important than the artificial planting for the restoration of the wind-disturbed spruce forest in Ofeliite site. In this respect, sustaining abundant seedling bank as well as sufficient quantities of decayed deadwood as regeneration substrate and microhabitat in the managed spruce forests can help adequate natural regeneration after large-scale disturbances.

Key words: montane Norway spruce, mortality, natural regeneration, plantation, regeneration substrate, wind disturbance.

Introduction

Wind disturbances form an integral part of the natural dynamics of Norway spruce (*Picea abies* (L.) Karst.) forests in Europe (Korpel 1995, Panayotov et al. 2011, Svoboda et al. 2010, 2012, 2014, Ulanova 2000, Wohlgemuth et al. 2002). Dependent on their magnitude, the impact of wind disturbances can vary from the creation of forest gaps to the emergence of larger windthrow areas. In recent decades an increasing number of large-scale windthrows in the Norway spruce forests was reported all over Europe (Schelhaas et al. 2003), which can be associated with the increased frequency of weather extremes such as wind storms and droughts (Dale et al. 2001, Schlyter et al. 2006). In Bulgaria the last large windthrow took place in Vitosha

Nature Park in 2001 affecting a total of 74 ha of Norway spruce forests (Dountchev et al. 2014). It was followed by a bark beetle outbreak which killed another 181 ha spruce forests in reserve non-intervention territories and 63 ha in managed forests, which were cleared and replanted.

After severe wind disturbances the most important question is how to reduce the calamity risk and to accelerate the forest restoration (Brang et al. 2006, Schönenberger 2002). Traditionally the silvicultural measures after natural disturbances include salvage logging and replanting of the cleared areas. In the high mountain zone the replanting is considered as a key measure for forest restoration since the tree regeneration processes in open areas can be slow as a result of the unfavourable ecological conditions and the effective suppression by the dense vegetation cover (Brang et al. 2006, Dimitrov 1999, Milev and Iliev 2000, Pavlov et al. 1996, Wohlgemuth et al. 2002). Unfavourable environmental factors such as climatic extremes, drought and suppression by grass or shrub vegetation can have negative impact also on the survival of planted trees (Frehner 2002, Jonsson 1999, Milev and Iliev 2000, Penev and Marinov 1964, Schönenberger 2002).

Recent studies showed, however, that the natural regeneration can play key role in the recovery of wind disturbed spruce forests (Dimitrov 1999, Jehl 2001, Jonášová et al. 2010, Wohlgemuth et al. 2002). A precondition for abundant spruce recruitment is the availability of sufficient advance regeneration, as well as the presence of appropriate regeneration substrates such as decayed deadwood and exposed mineral soil when seed sources are available (Brang et al. 2006, Holeksa et al. 2012, Kuuluvainen and Kalmari 1993, Penev and Marinov 1964, Zielonka 2006). Moreover, the natural regeneration is considered as more sustainable and economically advantageous option for forest restoration in comparison to artificial plantations (Schönenberger 2002).

In spite of the existing information about these advantages of natural over artificial regeneration, there are still some debates mostly concerning whether this advantage is valid also under the conditions of Vitosha Mountains (Naidenov 2006).

Therefore, our aim was to compare the development of the natural and the artificial regeneration in a cleared windthrow area and to make recommendations for enhancement of the effectiveness of the restoration measures after wind disturbances in montane *Picea abies* forests. For this purpose we studied the regeneration dynamics 5 years and 13 years after a wind disturbance in 2001 and examined the effect of the microsites on the establishment and growth of the *Picea* saplings.

Material and Methods

Study site

The study site is situated at an altitude of 1500–1700 m a.s.l. in the mountain valley of Vladaiska river in Vitosha Nature Park, South-Western Bulgaria (42°33' N; 23°18' E). The site is named “Ofeliite” (Fig. 1) and falls within a forest complex of 700 ha dominated by Norway spruce (*Picea abies* (L.) Karst.), to lesser extent European beach (*Fagus sylvatica* L.) and small share of *Salix caprea* L., *Sorbus aucuparia* L. and *Betula pendula* Roth. According to the formerly used Russian approach for classification (Aleksandrova 1978), the forest communities in the study area belong to the associations *Piceetum calamagrostosum, Piceetum myrtilloso-calamagrostosum* and *Piceetum Fageto herbosum* (Pavlov and Dimitrov 1995). No special phytosociological studies following Braun-Blanquet approach were done in Vitosha, but the Norway spruce communities in Rila were classified into alliance *Piceion excelsae*, subaliances *Eu-Piceenion* and *Abieti-Piceenion* (Roussakova and Dimitrov 2005). The mean age of the stands is 120 years. Having predominantly homogeneous structure, the forests are considered to be in the self-thinning phase. The bedrock is formed by syenite rocks and is covered by Dystric and Umbric Cambisols. In some areas the soils lie on large spherical syenite blocks and therefore the rooting depth is limited.

The average annual temperature at 1700 m a.s.l. is 4.0 °C, ranging from a mean monthly temperature of –5.2 °C in January to +12.9 °C in July–August. The annual precipitation is 1060 mm with a maximum in May–June and a minimum in August–September. The snow cover reaches up to 100 cm and lasts less than 150 days.

In 2001, nearly 14 ha of the Norway spruce forests in the area were blown down by a wind storm. The windthrow area (NW exposure) was subject to salvage logging. After 2003 the adjacent forest stands were clear cut as a sanitary measure against the bark beetle outbreak which followed the windthrow. The salvage logging was followed by artificial reforestation of the cleared area which took place in the period 2003–2008. The technological plans recommended planting of 2500–3000 2-year-old Norway spruce saplings per hectare in the cleared windthrow area. Information about the origin of the saplings was either not provided or limited to the name of the forest nursery – Godech, Pirdop or Samokov. The planting was prescribed to be carried out in the spring period, whereby the saplings should be planted without a planting scheme in 30-cm-deep hand-made holes. The cultivation measures included removal of the suppressive grass vegetation, as well as additional planting in areas with high saplings' mortality.

Data Collection and Analysis

In 2006, the 14-ha windthrow area at the study site was identified by GPS measurements and subsequently mapped in GIS using ArcGIS 10 software (ESRI Inc.). In the same year we established 16 permanent sample plots of size 10 m × 10 m (100 m^2) (Fig. 1). Tree recruitment, field vegetation and regeneration microsites

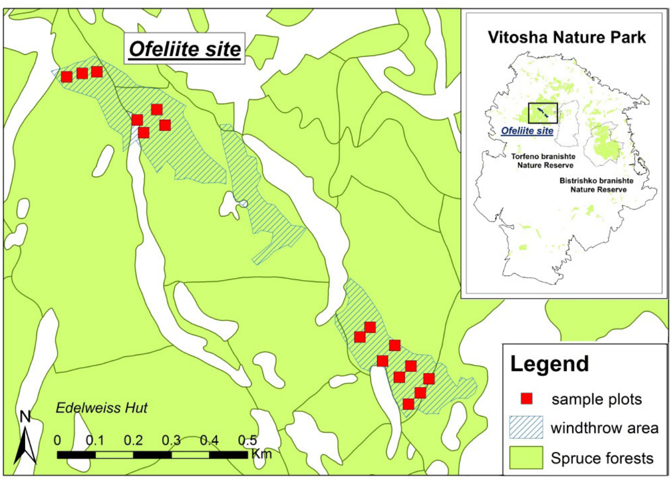


Fig. 1. Study site.

were scored in each sample plot in 2006 and 2014, following methodologies applied by Fischer et al. (2002), Jehl (2001) and Siegrist (2000). First, we made a record of the species composition, density, height and the type (i.e. natural or artificial) of the tree recruitment. The planted saplings were distinguished by the presence of planting holes. Further, the coverage of the main types of microsites (e.g. windthrow mounds, decayed deadwood, litter, bare ground, vegetation cover and rocks) were estimated on basis of subplots of 1 m^2 and the type of regeneration substrate of each *Picea abies* sapling was determined. The regeneration was classified into 6 height groups: <10 cm, 11–25 cm, 26–50 cm, 51–100 cm, 101–150 cm and >151 cm. The average mortality rate was estimated as a ratio of the number of saplings, which have not survived in the period 2006–2014 compared to all saplings registered in 2006.

The normality of data distribution was checked by Shapiro-Wilk test. The normality tests showed that all data had non-normal distribution requiring non-parametric tests to be applied for statistical analyses. Mann-Whitney rank sum test was used to check for significant differences between the natural and artificial regeneration. Wilcoxon Signed Rank tests were applied to check for changes in *Picea* regeneration during the period of the study. Chi-square test was used to check for significant differences in the mortality levels between natural and artificial regeneration.

Results

Regeneration dynamics

Thirteen years after the disturbance, the regeneration registered in the sample plots was formed by both naturally established and planted saplings. The natural regeneration consisted of 7 tree species. Most abundant were *Picea abies* (831 ha^{-1} or 56.8 %), *Sorbus aucuparia* (319 ha^{-1} or 21.8 %), *Salix caprea* (150 ha^{-1} or 10.3 %) and *Fagus sylvatica* (100 ha^{-1} or 6.8 %). Typical pioneer species such as *Betula pendula*, *Populus tremula* L. and *Pinus peuce* Griseb. had altogether nearly 4 % participation (Fig. 2). The artificial plantation consisted exclusively of *Picea abies* saplings (375 ha^{-1}).

The regeneration dynamics and the effectiveness of the planting activities were

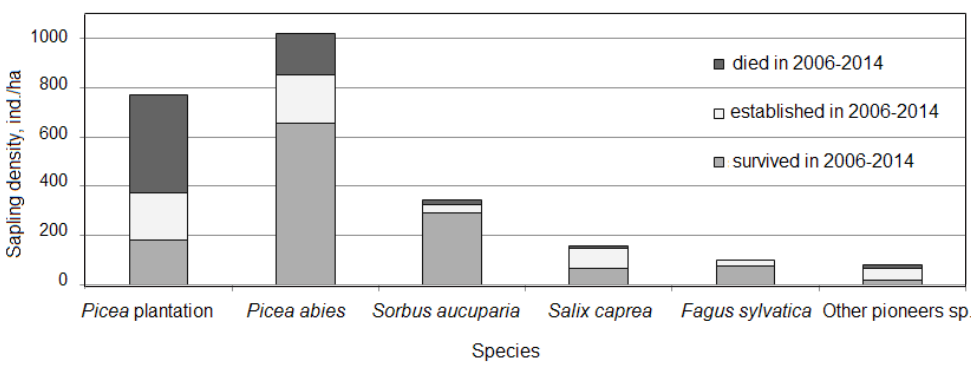


Fig. 2. Sapling density in the windthrow area in 2014.

assessed by comparison of the regeneration numbers in 2006 and 2014. In 2006 the total regeneration numbers in the sample plots were on average 1919 ha^{-1}. The natural regeneration accounted for 1344 saplings per 1 ha (of which 825 ha^{-1} were *Picea abies*), while the artificial plantation was significantly lower – 575 ha^{-1} (Mann Withney test, p=0.009). In 2014 the total regeneration numbers decreased slightly to 1863 saplings ha^{-1}, consisting of 1488 ha^{-1} natural regeneration (of which 831 ha^{-1} were *Picea abies*) and 375 ha^{-1} planted saplings. The increase in the natural regeneration from 1344 to 1488 ha^{-1} was not significant (Wilcoxon Signed Ranked test, p=0.6) and was attributed to the establishment of 369 saplings per 1 ha after 2006 – mainly *Picea abies* (200 ha^{-1}) and *Salix caprea* (81 ha^{-1}). Yet, we found that 213 saplings per 1 ha which had been registered in 2006 had not survived by 2014. This indicated an average mortality rate of 16 % for the study period, with highest values for *Betula pendula* (50 %) and *Picea abies* (20 %) (Fig. 2).

The comparison of the height structure of the natural regeneration in 2006 and 2014 (Fig. 3a, 3b) showed that during the study period all species exhibited a clear shift in the regeneration numbers from the lower height classes to the higher ones. For instance, the number of the *Picea* saplings in the first two height classes differed significantly between 2006 and 2014 – 362 ha^{-1} and resp. 50 ha^{-1} (Wilcoxon Signed Ranked test, p=0.002).

The number of registered planted saplings in the artificial plantation dropped

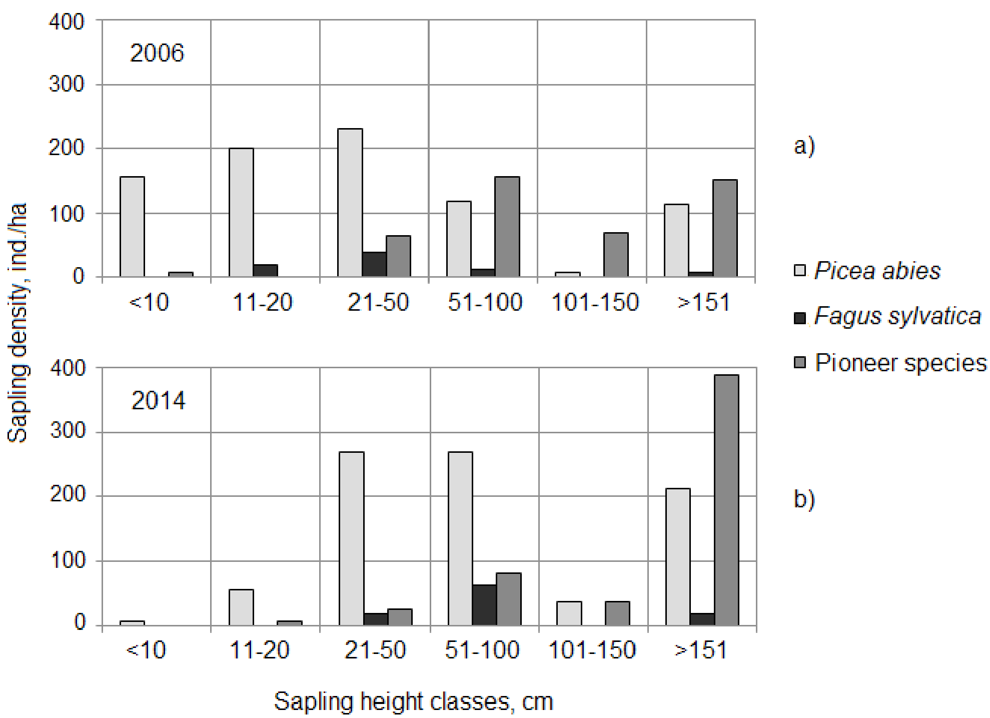


Fig. 3. Height structure of the natural regeneration in: a) 2006 and b) 2014.

from 575 ha^{-1} in 2006 down to 375 ha^{-1} in 2014. Even if another 200 saplings per 1 ha were planted additionally in the period 2007–2014, the number of the saplings which died in this period was twice higher – 394 saplings per 1 ha. By 2014 the number of planted saplings in the sample plots varied between 0 and 2000 ha^{-1}.

Mortality of *Picea abies* saplings

Mortality of the dominant species *Picea abihes* was registered in saplings with height up to 1 m. It was highest for individuals lower than 20 cm in 2006 and varied per sample plot between 0 % and 86 % (Fig. 4a). Yet, the mortality rate of the natural *Picea* regeneration (<1 m height) was significantly lower than that of the artificially planted – 24 % vs. 69 % (Chi-square test p=0). The mortality rate of the planted saplings was highest among individuals that were lower than 50 cm in 2006 (Fig. 4b) and it varied per sample plot between 33 % and 100 %.

The highest mortality rates of the natural *Picea* regeneration were found among saplings that had been registered in 2006 on stones covered by thin organic layer (86 %) and on windthrow mounds (50 %), whereas lower mortality rates were regis-

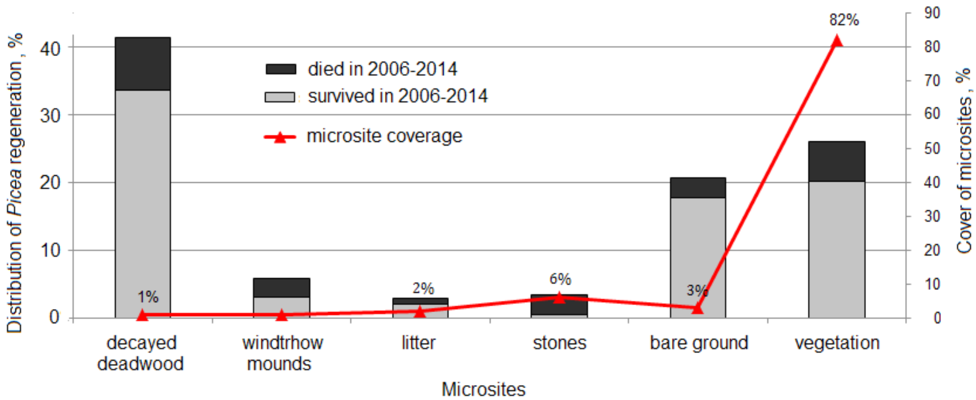
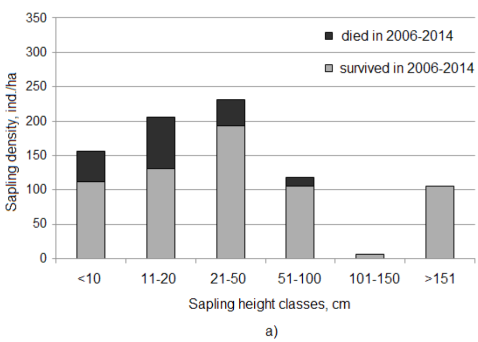


Fig. 4. Mortality of *Picea* regeneration per sapling height classes: a) natural regeneration; b) artificial regeneration.



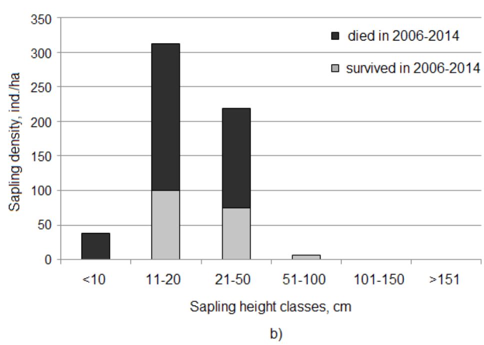


Fig. 5. Distribution of the natural *Picea* regeneration per substrate type.

tered on litter (30 %), vegetation covered ground (22 %), decayed deadwood (19 %) and bare ground (14 %) (Fig. 5). It was assessed that the critical factor for 67 % of the saplings, which have not survived in the period 2006–2014, was drought, for 8 % – erosion of the soil layer on mounds, and for the remaining 25 % the mortality factors were not identified.

All planted *Picea* saplings that did not survive in the period 2006–2014 were found in patches of grass and shrub vegetation dominated by *Calamagrostis arundinacea* or *Rubus ideaus*, and in areas with excessive soil moisture – *Juncus effusus* L. In 10 % of the mortality cases the planted saplings were missing, what was considered as indication for frost heaving. In the other 90 % of the mortality cases the saplings either showed signs of damages as result of careless cultivation or seemed to have withered.

Regeneration substrates of *Picea abies*

In 2006, 74 % of the naturally established *Picea* saplings with height <50 cm were observed on vegetation-free substrates such as decayed deadwood, bare ground, windthrow mounds and litter (Fig. 5). The highest regeneration densities were registered on decayed deadwood and on bare ground – 269 per 100 m^2. Despite its extremely low coverage of less than 1 % (e.g. <5–10 m^3/ha) this substrate hosted 41 % of all *Picea* saplings, often forming groups of individuals. On vegetation-free substrates such as bare ground and windthrow mounds we found 45 and 38 saplings per 100 m^2, respectively. In contrast, the regeneration density on vegetation covered ground was 2 saplings per 100 m^2, situated individually. In 2006 this microsite covered 82 % of the area of the sample plots but accounted for only 26 % of all *Picea* saplings. In 2014, the distribution of the *Picea* regeneration among the regeneration substrates did not change considerably, as the vegetation-free substrates hosted 68 % of all *Picea* saplings compared to 74 % in 2006.

Discussion

Regeneration dynamics

Our results showed that the wind disturbance and the subsequent salvage logging did not lead to considerable changes in the species composition of the disturbed Norway spruce forest. Most abundant was the tree recruitment of species such as *Picea abies* and *Sorbus aucuparia,* which are characteristic for this type of climax forest. This is not surprising since these species rely both on advance and post-disturbance regeneration in order to persist after natural disturbances such as windthrows and bark beetle outbreaks (Jehl 2001, Schönenberger 2002, Żywiec and Ledwoń 2008). Moreover, before the outbreak of the bark beetles in 2003, the cleared area had a maximal width of 150 m and was compact enough for the wind-dispersal of the *Picea abies* seeds, which is most intensive up to 100–200 m from the intact forest edge (Lässig et al. 1995).

However, the dominance of *Picea abies* and *Sorbus aucuparia* was not actually a result of high regeneration numbers, assessed at less than 1500 ha^{-1} in 2014. In contrast to the general knowledge about forest succession, we found that in the cleared winthrow area the recruitment of pioneer species was rather limited. The regeneration numbers of species such as *Pinus* sp., *Salix* sp., *Populus tremula* or *Betula pendula* did not exceed altogether 300 ha^{-1} in 2014. This was opposite to observations in windthrow areas situated at lower altitudes (Fischer et al. 2002, Ilisson et al. 2007, Moćalov and Lässig 2002, Pavlov et al. 1996, Siegrist 2000) where the milder climatic conditions favoured more abundant tree regeneration and higher participation of pioneer species. Moreover, except for *Salix caprea,* all other pioneer species were represented by a negligible number of seeding trees in the adjacent forest stands. This, in our view, limited their specific potential to populate disturbed areas including microsites with exposed mineral soil created by the uprooting of wind-felled trees and the harvesting (Ilisson et al. 2007, Jonášová et al. 2010, Schönenberger 2002, Ulanova 2000).

Further, we found that the intensity of the natural regeneration was highest shortly after the disturbance and the salvage logging (Dimitrov 1999, Wohlegemuth et al. 2002). On the one side, this was evident from the increase in the number of saplings in the height classes above 50 cm in the period 2006–2014. This increase resulted both from the release of the advance regeneration and the rapid grow-up of saplings which have established in the period 2001–2006. On the other side, the significant decrease in the numbers of individuals registered in the height classes lower than 20 cm in 2014 compared to 2006 revealed a clear drop in the regeneration numbers after 2006. Since the mortality of the natural regeneration was relatively low (i.e. 17 %), the regeneration drop resulted most probably from decline in the recruitment levels.

Whereas the surge in the regeneration process after 2001 was favoured by the improved light conditions after the removal of the forest cover, the observed drop in the regeneration intensity after 2006 indicated that the regeneration conditions have considerably worsen some years after the disturbance. As main factor for this negative trend was considered the rapid expansion of the grass and shrub vegetation, which is known to effectively suppress the regeneration of *Picea abies* and other tree species (Pavlov et al. 1996, Wohlgemuth et al. 2002). In particular, we found that five years after the removal of the forest cover the grass vegetation occupied more than 80 % of the disturbed area.

In terms of sapling density the artificial plantation played minor role compared to the natural regeneration for the restoration of the wind disturbed Norway spruce forest. This was shown by the significant differences between the number of naturally established saplings (mainly *Picea abies*) and the number of planted *Picea* saplings in the sample plots. Provided that the prescribed planting density was between 2500 and 3000 *Picea* saplings per hectare, the observed regeneration density of 375 ha^{-1} in 2014 could be explained by severe losses in the number of planted saplings. Indeed, the average mortality rate reported by the forest authorities for only two years in the period 2004–2005 was 48 %. In addition, we found that for the study period 2006–2014 the mortality rate was on average 69 %. In contrast, the observed lower mortality rates of species such as *Fagus sylvatica* and *Pinus* sp. indicates their potential for larger use by the artificial plantations in mountain forests (Dimitrov 1999).

Mortality of *Picea abies* saplings

The significantly lower mortality rate for the period 2006–2014 registered among the naturally established *Picea* saplings with height of <1 m in comparison to the planted ones (25 % vs. 69 %) suggests that the natural *Picea* regeneration was more resistant against the critical micro-environmental conditions than the artificial plantation. However, it should be noted that part of the natural regeneration can be damaged during the salvage activities in the disturbed forests (Wohlgemuth et al. 2002). For instance, we documented severe damages to the advance regeneration as result of controlled burning of the slash across the disturbed area.

Further, our results confirmed the finding of Milev and Iliev (2000) that *Picea* saplings lower than 100 cm have lower resistance against the unfavourable ecological factors. Particularly critical for the survival of the young trees was the period of growth to a height of 20–50 cm. In this regard, Schönenberger and Wasem (1997) reported that the mortality of planted saplings was highest in the first three years as result of the planting shock. These findings can be explained by the importance of the well-developed root system (Frehner 2002) for the resistance of the plants against the unfavourable micro-environmental conditions and the ability of the larger plants to out-compete the suppressive grass and shrub vegetation.

The potential mortality reasons by all saplings were assessed indirectly. As the most critical ecological factor for the mortality of the natural *Picea* saplings was considered the poor moisture retaining ability of particular regeneration substrates. In this way was explained the observed withering of saplings established on windthrow mounds, on sun-exposed deadwood and particularly on stones covered by thin organic layer. Soil slides on mounds during heavy rains also may lead to loss of individuals (Ilisson et al. 2007, Holeksa et al. 2012, Ulanova 2000). Further, we assume that the suppression of saplings growth due to grass vegetation as well as frost could have also negatively impacted the survival of individual saplings.

Most of the dead planted saplings (often 2-years-old) were found in patches of *Rubus ideaus* and *Calamagrostis arundinacea*, as well as in areas with excessive moisture covered by *Juncus effusus*. Provided that the mortality was highest by saplings lower than 50 cm, we suppose that the suppression by the grass and shrub vegetation may have aggravated the impact of other ecological factors such as frost or drought (Milev and Iliev 2000, Schönenberger and Wasem 1997).

Yet, our observations on the mortality patterns of the planted saplings suggest that the implementation of the artificial planting and the cultivation technology should be improved. In the first place, the risk of frost heaving or drought can be reduced by selecting planting microsites without excessive soil moisture, respectively by selecting moderately elevated planting microsites close to standing trees, shrubs or lying logs which can provide shelter to the planted saplings. Since in the montane and the sub-alpine zones the initial growth of the spruce saplings is rather slow, higher resilience of the planted saplings to critical ecological factors such as the suppressive vegetation, the frost or the drought is observed when well-developed plants, i.e. older than 4–5 years, are used (Milev and Iliev 2000). Particularly potted plants show higher survival rates (Schönenberger and Wasem 1997). Moreover, the higher survival rates of the naturally established regeneration of *Fagus sylvatica* and most of the pioneer species in Ofeliite site indicates the potential of these species to be effectively used by the replanting of the cleared windthrow area.

It is also expected that the usage of planting material, which is better adapted to the site-specific ecological conditions, can reduce the planting shock. For instance, the planting material in Ofeliite site was procured in forest nurseries situated at much lower altitude, while the *Picea* seeds presumably originated from spruce forests, where the ecological conditions considerably differed from the ones on the study site.

In the second place, the survival of the planted saplings can be increased if the removal of the competing vegetation during the cultivation of the *Picea* plantations is carried out until the saplings grow higher than 50 cm, i.e. for a period of not less than 3–5 years. Besides this, the mowing of the grass vegetation should be carried out in a way that no damages to the planted saplings are caused. As observed in the study site, the damaged coniferous saplings are the first to be suppressed and killed by the competing vegetation.

Regeneration substrates of *Picea abies*

Our results show that vegetation-free substrates such as decayed deadwood, bare ground, windthrow mounds and litter were of key importance for the natural regeneration of the dominant species *Picea abies*, despite their limited coverage in the windthrow area (less than 18 %). Shortly after the disturbance and the salvage logging these substrates provided the *Picea* seedlings a safe place to develop free of competing vegetation (Ulanova 2000, Schönenberger 2002, Kuuluvainen and Kalmari 2003). This means that even saplings, which have been registered in patches of grass and shrub vegetation had most probably germinated on bare ground before the expansion of the vegetation.

Most favourable for the regeneration of *Picea abies* proved to be the substrate formed by decayed logs and stumps, which exhibited the highest sapling density both in 2006 and in 2014. When available in sufficient quantities this substrate type is considered as critical for the regeneration of high mountain *Picea abies* forests (Hofgaard 1993, Holeksa et al. 2012, Jehl 2001, Svoboda et al. 2010). Since it takes at least 2 decades before the deadwood is sufficiently decayed to allow establishment of the *Picea* recruitment (Holeksa et al. 2012, Zielonka 2006), we assume that the decayed deadwood was present in the study site before the disturbance of 2001.

In terms of saplings density high importance for the regeneration of *Picea abies* had also the windthrow mounds, which resulted from the up-rooting of mature trees. Similarly to deadwood, this elevated microsite provides the seedlings better protection against the frost resulting from the stagnation of cold air in the shallow areas (Penev and Marinov 1964, Ilisson et al. 2007). Moreover, the windthrow mounds are mainly formed by exposed mineral soil, which is considered as optimal substrate for the regeneration of as well *Picea abies* as most of the pioneer species.

In contrast, the lowest importance for the regeneration of *Picea abies* had the vegetation covered ground exhibiting the lowest regeneration densities (less than 2 saplings per 100 m^2). The grass and shrub vegetation was mainly formed by dense patches of the species *Rubus ideaus* and *Calamagrostis arundinacea*, which are known to effectively suppress the germination and survival of *Picea* seedlings (Pavlov et al. 1996, Wohlgemuth et al. 2002). Since these species sustain an abundant seed bank in the soil and emerge immediately after increase of the light levels (Fischer et al. 2002), the germination of *Picea* seeds on bare ground after a disturbance may be successful in a very short time lapse.

Conclusion and implications for forest management

Our results show that the natural regeneration compared to the artificial planting can be a more sustainable and cost effective option for the recovery of wind disturbed spruce forests. Before taking a decision on the type of restoration measures needed it is advisable to observe the potential of the natural regeneration for a period of up to 5 years after the disturbance. When adequate advance regeneration is available, we suggest that the salvage logging as a traditional sanitary measure against potential outbreak of bark beetles is replaced by debarking of the wind-blown trees, as practiced in protected areas and hardly accessible territories in Central Europe (Dountchev et al. 2014). In general, the resilience of montane *Picea abies* forests against large-scale disturbances can be preliminary enhanced by sustaining abundant seedling bank, leaving sufficient quantities of deadwood as regeneration substrate and tolerating pioneer species as seeding trees (Brang et. al 2006).

When indispensable, the artificial reforestation should be implemented by strict application of the planting technology, usage of planting material adapted to the site-specific ecological conditions and complementation of the *Picea* plantations with saplings of pioneer species or *Fagus sylvatica*. The spruce plantations should be created by using of well-developed and preferably potted saplings with an age of at least 4 years (Frehner 2002, Milev and Iliev 2000) and with special preference to grass-free substrates and elevated microsites which are well-drained and airy. These measures are particularly important in areas with extreme climatic conditions, where the slowly growing spruce saplings most often die out as result of frost heaving or suppression by the high grass and shrub vegetation.

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References

Aleksandrova V.D. 1978. Russian approaches to classification. In: Whittaker R.H. (Ed.) Classification of Plant Communities. Dr. W. Junk bv. Publishers, The Hague, The Netherlands: 167–200.

Brang P., Schönenberger W., Frehner M., Schwitter R., Thormann J.J., Wasser B. 2006. Management of protection forests in the European Alps: An overview. Forest, Snow and Landscape Research 80: 23–44

Dale V.H., Joyce L.A., McNulty S., Neilson R.P., Ayres M.P., Flannigan M.D., Hanson P.J., Irland L.C., Lugo A.E., Peterson C.J., Simberloff D., Swanson F.J., Stocks B.J., Wotton B.M. 2001. Climate change and forest disturbances. BioScience 51: 723–734.

Dimitrov M. 1999. Successional changes of the vegetation in a windthrow area in the association Piceeto Abietum luzulosum. Forestry ideas 18(1): 13–27 (in Bulgarian).

Dountchev A., Tsvetanov N., Zhelev P., Panayotov M. 2014. Challenges before the conservation of the Norway spruce forests in Vitosha Nature Park after large-scale disturbances. Ecologica Balcanica, Special edition 5: 61–69.

Fischer A., Lindner M., Abs C., Lasch P. 2002. Vegetation dynamics in Central

European forest ecosystems (near-natural as well as managed) after storm events. Folia Geobotanica 37: 17–32.

Frehner M. 2002. Untersuchungen über den Einfluss unterschiedlicher Kleinstandorte und der Pflanztechnik auf Fichtenpflanzungen in subalpinen Lawinenschutzwäldern. PhD thesis, ETH Zurich, Schweizerische Zeitschrift für Forstwesen, Beiheft No 92: 279 p.

Hofgaard A. 1993. Structure and regeneration patterns of a virgin *Picea abies* forest in northern Sweden. Journal of Vegetation Science 4: 601–608.

Holeksa J., Zielonka T., Żywiec M. 2012. Norway spruce *Picea abies* regeneration and canopy disturbance in a Carpathian subalpine forest. In: Nowak K.I., Strybell H.F. (Eds.) Spruce: Ecology, Management and Conservation. New York, USA: 1–40.

Ilisson T., Köster K., Vodde F., Jogiste K. 2007. Regeneration development 4–5 years after a storm in Norway spruce dominated forests, Estonia. Forest Ecology and Management 250: 17–24.

Jehl H. 2001. Die Waldentwicklung nach Windwurf in den Hochlagen des Nationalparks Bayerischer Wald. In: Waldentwicklung im Bergwald nach Windwurf und Borkenkäferbefall. Herausgegeben Nationalpark Bayerischer Wald, Wissenschaftlische Reihe 14: 49–98.

Jonášová M., Vávrová E., Cudlin P. 2010. Western Carpathian mountain pruce forest after a windthrow: Natural regeneration in cleared and uncleared areas. Forest Ecology and Management 259: 1127–1134.

Jonsson B. 1999. Stand establishment and early growth of planted *Pinus sylvestris* and *Picea abies* related to microsite conditions. Scandinavian Journal of Forest Research 14: 425–440.

Korpel S. 1995. Die Urwälder der Westkarpaten. Gustav Fischer, Stuttgart, Jena, New York, 310 p.

Kuuluvainen T., Kalmari R. 2003. Regeneration microsites of *Picea abies* seedlings in a windthrow area of a boreal old-growth forest in southern Finland. Annales Botanici Fennici 40: 401–413.

Lässig R., Egli S., Odermatt O., Schönenberger W., Stöckli B., Wohlgemuth T. 1995. Beginn der Wiederbewaldung auf Windwurfflächen. Schweizerische Zeitschrift für Forstwesen 146(11): 893–911.

Naidenov J. 2006. The invasion of the barkbeetles in park Vitosha – calamity or regularity. Bulgarska gora 2(6): 1, 8 (in Bulgarian).

Milev M., Iliev N. 2000. Artificial regeneration of high mountain spruce forests in DL “Beglika”. Anniversary collection of scientific reports, University of Forestry – Sofia: 272–281.

Moćalov S.A., Lässig R. 2002. Development of two boreal forests after large-scale windthrow in the Central Urals. Forest, Snow and Landscape Research 77(1–2): 171–186.

Panayotov M., Kulakowski D., Larnjeiro Dos Santos L., Bebi P. 2011. Wind disturbances shape old Norway spruce-dominated forest in Bulgaria. Forest Ecology and Management 262: 470–481.

Pavlov D., Dimitrov M. 1995. Syntaxonomic analysis of spruce forest types in the water catchment area of Vladaiska River. Jubilee scientific session “70 years higher forestry education in Bulgaria”. Sofia, Higher Institute of Forestry Publishing House, vol.1: 61–75 (in Bulgarian).

Pavlov D., Tashev A., Kitin P., Dimitrov M. 1996. Secondary succession of the vegetation after a windthrow of the forest stand in the association Piceetum calamagrostoso-myrtillosum. Forestry ideas 6(1): 3–17 (in Bulgarian).

Penev N., Marinov M. 1964. Regeneration of coniferous stands by forest types. Zemizdat, Sofia. 150 p. (in Bulgarian).

Roussakova V., Dimitrov M. 2005. Caractéristiques phytosociologiques des forêts à *Picea abies* et *Abies alba* dans la montagne de Rila (Bulgarie). Acta Botanica Gallica 152: 563–571.

Schelhaas M.J., Nabuurs G.J., Schuck A. 2003. Natural disturbances in the European forests in the 19th and 20th centuries. Global Change Biology 9: 1620–1633.

Schlyter P., Stjernquist I., Barring L., Jonsson A.M., Nilsson C. 2006. Assessment of the impacts of climate change and weather extremes on boreal forests in northern Europe, focusing on Norway spruce. Climate Research 31: 75–84.

Schönenberger W. 2002. Post windthrow stand regeneration in Swiss mountain forests: the first ten years after the 1990 storm Vivian. Forest, Snow and Landscape Research 77: 61–80.

Schönenberger W., Wasem U. 1997. Wiederbewaldung einer Waldbrandfläche in der subalpinen Stufe bei Müstair. Schweizerische Zeitschrift für Forstwesen 148: 405–424.

Siegrist J. 2000. Natürliche Wiederbewaldung und Struktur eines ungeräumten Fichten-Windwurfs auf potentiellem Bergmischwaldstandort im Nationalpark Berchtesgaden. In: Nationalparkverwaltung Berchtesgaden (Hrsg.) Waldentwicklung im Nationalpark Berchtesgaden von 1983 bis 1997. Nationalpark Berchtesgaden Forschungsbericht 43: 93–146.

Svoboda M., Fraver S., Janda P., Bace R., Zenáhliíková J. 2010. Natural development and regeneration of a central European montane spruce forest. Forest Ecology and Management 260: 707–714.

Svoboda M., Janda P., Nagel T., Fraver S., Rejzek J., Bače R. 2012. Disturbance history of an old-growth sub-alpine *Picea abies* stand in the Bohemian Forest, Czech Republic. *Journal of Vegetation Science* 23: 86–97.

Svoboda M., Janda P., Bače R., Fraver S., Nagel T., Rejzek J., Mikoláš M., Douda J., Boublík K., Šamonil P., Čada V., Trotsiuk V., Teodosiu M., Bouriaud O., Biriş A., Sýkora O., Uzel P., Zelenka J., Sedlák V., Lehejček J. 2014. Landscape-level variability in historical disturbance in primary *Picea abies* mountain forests of the Eastern Carpathians, Romania. Journal of Vegetation Science 25: 386–401.

Ulanova N.G. 2000. The effects of windthrow on forest at different spatial scales: a review. Forest Ecology and Management 135: 155–167.

Wohlgemuth T., Kull P., Wüthrich H. 2002. Disturbance of microsites and early tree regeneration after windthrow in Swiss mountain forests due to the winter storm Vivian 1990. Forest, Snow and Landscape Research 77(1–2): 17–47.

Zielonka T. 2006. When does dead wood turn into a substrate for spruce replacement? Journal of Vegetation Science 17: 739–746.

Żywiec M., Ledwoń M. 2008. Spatial and temporal patterns of rowan (*Sorbus aucuparia* L.) regeneration in West Carpathian subalpine spruce forest. Plant Ecology 194: 283–291.