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Projected effects of climate change on the carbon stocks of European beech (*Fagus sylvatica* L.) forests in Zala County, Hungary

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Abstract
Recent studies suggest that climate change will lead to the local extinction of many tree species from large areas during this century, affecting the functioning and ecosystem services of many forests. This study reports on projected carbon losses due to the assumed local climate change-driven extinction of European beech (*Fagus sylvatica* L.) from Zala County, South-Western Hungary, where the species grows at the xeric limit of its distribution. The losses were calculated as a difference between carbon stocks in climate change scenarios assuming an exponentially increasing forest decline over time, and those in a baseline scenario assuming no climate change. In the climate change scenarios, three different sets of forest management adaptation measures were studied: (1) only harvesting damaged stands, (2) additionally salvaging dead trees that died due to climate change, and (3) replacing, at an increasing rate over time, beech with sessile oak (*Quercus petraea* Matt. Lieb.) after final harvest. Projections were made using the open access carbon accounting model CASMOFOR based on modeling or assuming effects of climate change on mortality, tree growth, root-to-shoot ratio and decomposition rates. Results demonstrate that, if beech disappears from the region as projected by the end of the century, over 80% of above-ground biomass carbon, and over 60% of the carbon stocks of all pools (excluding soils) of the forests will be lost by 2100. Such emission rates on large areas may have a discernible positive feedback on climate change, and can only partially be offset by the forest management adaptation measures.

Key words: climate change; mortality; silviculture; forest carbon balance; European beech

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Introduction

Climate change has been repeatedly shown to be unequivocal, and it continues globally at unprecedented rates and with already observed widespread and consequential effects (IPCC 2013, 2014a). Evidence on observed impacts as well as projections of potential effects of climate change on terrestrial ecosystems are mounting (e.g., Allen et al. 2010; Lindner et al. 2010; Vayreda et al. 2012; IPCC 2014a; Lindner et al. 2014) suggesting significant vulnerability of forest ecosystems.

The effects of climate change at the species level are closely related to the fact that, compared with herbs and animals, the maximum speed of trees to move in order to follow climatic changes is much lower than the average climate change velocity (IPCC 2014a). Over time, this may lead to large-scale dieback (Thuiller et al. 2011; Hanewinkel 2012), and even species displacement (Millennium Ecosystem Assessment 2005). While changes in regional temperature and precipitation patterns may also create better growing conditions for forest ecosystems in large areas (Lindner et al. 2010, 2014), the local extinction of certain species from severely affected areas may lead to large losses of carbon and associated emissions of CO_2.

The issue of large-scale displacement of habitat suitability was recently analyzed by studies showing that climate change sensitivity of 38 European tree species in Europe is rather complex and has significantly different patterns over climatic (temperature and precipitation), geographic and temporal (e.g. winter vs summer) dimensions (Zimmermann et al. 2013a). The range of species such as beech (*Fagus sylvatica* L.) and Norway spruce (*Picea abies* L.) is likely to shrink, sometimes dramatically. More drought-tolerant species such as Sessile oak (*Quercus petraea* (Matt.) Liebl) can, however, be expected to become more abundant (at least at lower altitudes, Zimmermann et al. 2013b). Species displacement and local extinction may also depend on pre-climate change conditions, weather extremes and an increase in climate variability, resulting in even more severe extremes (Zimmermann et al. 2009). Such extremes have also been shown to affect the health status of forests (e.g. Jung 2009).

One of the countries of Europe where local extinction might become a serious problem is Hungary where, until recently, the mean annual temperature was about 10 – 11 °C and the mean annual precipitation was about 500 – 750 mm which already represented a limiting factor for many tree species. These species, including beech and sessile oak, occur at the xeric limit of their distribution (Mátyás et al. 2010; Czúcz et al. 2011). Additional climatic vulnerability due to climate change is expected to be more expressed in Hungary than in many parts of Europe as the increase of regional mean temperature is projected to be 1.4 °C relative to each 1 °C of global temperature increase, whereas precipitation is projected to considerably decrease in summer and increase in winter (Bartholy et al. 2007, 2014). The level of tree mortality has so far been low, mainly expressed by a density-related self-thinning, and extreme weather events such as ice-breaks,

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snow-breaks and windbreaks (Hirka 2013) and droughts (Jung 2009) were small scale only.

This situation may, however, be dramatically changed under recent assumptions concerning possible rates of global and regional warming. The results of the REMO regional climate model simulations (Jacob et al. 2007) suggest that mean temperature will increase by 3.7 °C by 2100 relative to the average of 1961–1990. If this temperature increase will take place, trees will have to stand repeated and increasing drought and heat stress. Beyond certain levels of warming, however, an increasingly occurring tree mortality, referred to below as extinction mortality, may affect the Hungarian forests, potentially leading to the extinction of several or many species from certain sites. Czúcz (2011), Móricz et al. (2013), Rasztovics (2014) and others have recently projected that beech forests will almost entirely disappear from Hungary by the end of the century, whereas sessile oak will only be found along the southwest border of the country and in higher mountain regions.

Extinction mortality may lead to carbon-dioxide emissions first from the biomass, and later also from other pools of the affected forests. Such large emissions due to various disturbances have already been shown to be a serious problems if it occurs on a large scale (e.g., Kurz et al. 2008; Seidl et al. 2014).This study is an attempt to project forest carbon stock changes due to climate change-induced mass mortality of trees. Based on the projected forest decline by 2100 by Móricz et al. (2013), extinction mortality, the expected main driver of future carbon emissions, was assumed to increase exponentially for the beech forests of the Zala County, South-Western Hungary, for the period from 2015 to 2100. Using models or assumptions, the effects of climate changes on tree growth, root-to-shoot ratio, and decay rates were estimated, too. Finally, the effect of forest management options such as species replacement and harvesting was also estimated in forest management adaptation scenarios. The projections were developed using the open access carbon accounting model CASMOFOR.

2. Methods

2.1. Study area

With its forest area of 114,602 ha, the hilly (200 – 400 m a.s.l) Zala County located in south-western Hungary (Fig. 1) has the highest forest cover (32%) in the country. The native beech is well adapted to both local pre-climate change site conditions (Table 1) and predominantly deep forest soils. However, considering the climate envelope of beech in Europe using long term (1950–2000) climatic average of annual precipitation and mean July temperature (Mátyás et al. 2010), this species is close to its xeric limits in the Zala County. The mass mortality event of 2003–2004 which followed the drought period from 2000 to 2004 in the region was also taken as an indication of the existence of these limits (Lakatos & Molnár 2009; Mátyás et al. 2010). Nevertheless, similar symptoms were also recorded in the eastern part of the Carpathian basin in the 1880s (Lakatos & Molnár 2009).

The area of all 6406 pure and beech dominated forest stands in the County, identified by the nation-wide stand-wise continuous forest inventory, shows relatively large uneven distributions by both age and yield class (Fig. 2). The yield class of a stand was taken from standard local yield tables (Mendlik 1983) based on the age and the measured mean stand height of the stands. These yield tables employ six yield classes of equal height differences where class 1 is

Table 1. Ranges of key climatic data in the Zala county (mean values for 1981–2010 based on data of the Hungarian Met. Office, Gálos & Vigh 2014).

Climate characteristics	Value for months				
	I–XII	I	IV–IX	VII	X–III
Mean temperature [°C]	9.8 – 11.0	−0.9 – 0.2	16.2 – 17.6	19.6 – 21.2	3.4 – 4.5
Total precipitation [mm]	611 – 770	27 – 34	363 – 472	75 – 94	248 – 301

Fig. 1. The forests of Zala County as situated in the county map of Hungary (Source: National Forestry Database).

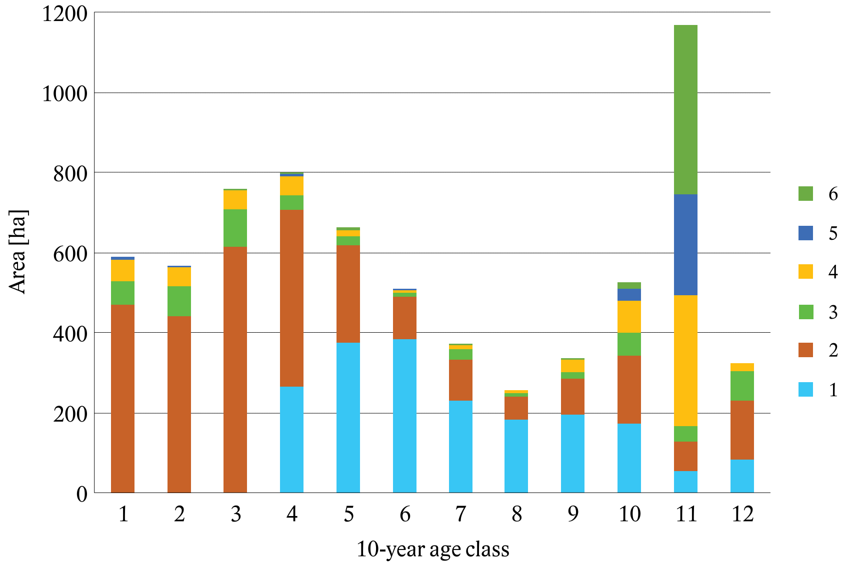

Fig. 2. The area of beech forests in the Zala County in 2012 in 10-year age classes and six yield classes. Yield class, shown on the right, ranges from 1 for the fastest growing stands to 6 for the slowest growing stands. (Source: National Forestry Database by the Forestry Directorate of the National Food Chain Safety Office, Budapest.)

assigned to the fastest and class 6 is assigned to the slowest growing stands, respectively. All forests in both the Zala County and the entire country have been rather intensively managed for several centuries.

2.2. Modelling framework

This study estimates using model CASMOFOR (see below) how much the carbon storage will change in the study area (i.e., the entire area that was covered by beech in Zala County in 2012) due to forest decline and other effects of climate change. Annual changes of carbon stocks were calculated as differences between carbon stocks projected for a baseline (BL) scenario assuming a stable climate (left box in Fig. 3) and those projected for climate change scenarios (right box in Fig. 3, see also below). For all climate change scenarios, extinction mortality rates based on the projection of Móricz et al. (2013) were used. The differences were calculated for the period 2015–2100.

Carbon stocks in each scenario were estimated using the open access forest carbon accounting model CASMOFOR (Somogyi 2010, www.scientia.hu/casmofor). This model is an MS Excel based system of carbon accounting functions which was mainly designed to estimate, by yield class and species, carbon stocks in annual steps as a function of forest

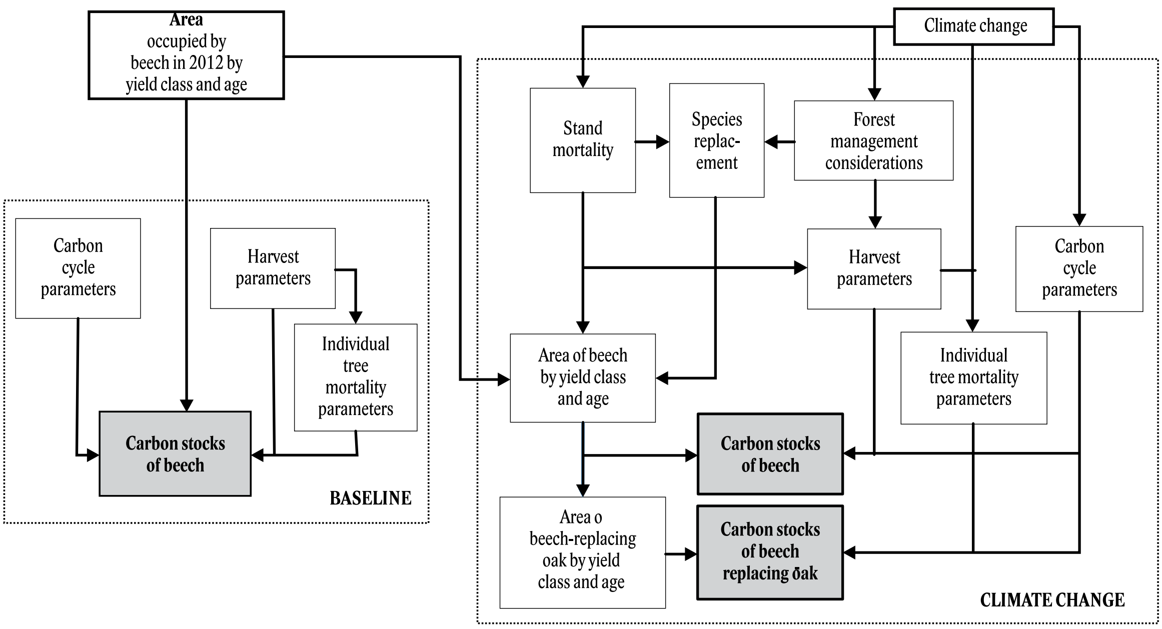

Fig. 3. The scheme of estimating changes of carbon stocks due to climate change as the difference between projected carbon stocks of beech in a baseline scenario (left box) and those of beech and oak in a climate change scenario (right box) for the area that was covered by beech in Zala County in 2012. “Parameters” are those applied in model CASMOFOR (see below).

area and age class structure. Estimates are developed for the forest carbon pools defined by IPCC (2006) based on their respective dynamics. For biomass, the gain-loss method by IPCC (2006) is used. Tree growth is modelled using standard yield tables, whereas thinning and density-dependent mortality (i.e., self-thinning) are modelled using country-level silvicultural models. The dead organic pools are modelled using exponential decay functions. The harvested wood products (HWP) pool is modeled based on the latest methodological guidance by IPCC (2014b). Soils are excluded from this analysis due to knowledge gaps. The model, together with its accounting functions (accessible at http://www.scientia.hu/casmofor/equationsE.php) as well as the parameters used for 19 species of the country (mainly age-dependent data in yield tables and silvicultural models, http://www.scientia.hu/casmofor/creditsE.php, and age-independent ones, (http://www.scientia.hu/casmofor/parametersE.php#flowchart), are described in detail at www.scientia.hu/casmofor.

2.3. Extinction mortality

Mortality (both density-dependent and density-independent) under the current ("no climate change") site conditions is included in the silvicultural models. To develop rates for the expected extinction mortality, the rather strong correlation between the current distribution of the tree species and climatic factors was considered. This correlation is modelled in the forestry practice in Hungary by applying four forest climatic types (Borhidi 1960; Járó 1966; Mátyás & Czimber 2004) that were identified by the dominant occurrence of respective indicator species. These climate types can also be characterized by historical temperature and precipitation data (Table 2), but the occurrence of climate types is also affected by local site factors such as soil type, aspect, hydrological conditions and others.

The above system of climate types indicates that, in general, and subject to variability due to other factors mentioned above, mean annual temperature differences of about 1 °C together with differences of precipitation of about 50 mm are enough in the long term for different tree species to become dominant in the different climate types and disappear from others. As the projected warming of the mean July temperature of about 3.7 °C during this century (Jacobs et al. 2007 is several times the difference between adjacent climate types, it is reasonable to expect this warming to cause a shift in the bioclimatic niche. This shift, in terms of temperature, from the current Beech climate type to the Turkey oak or Forest steppe type may even lead to the local extinction of beech from the Zala County especially if also reduced summer precipitation is considered.

Extinction mortality can take many forms and can occur either directly due to droughts or other weather-related effects such as extreme winds, floods etc., or indirectly due to exacerbating biotic agents, forest fires (due to mortality-driven fuel accumulation) and others. Currently, it is not possible to model these events, only assume the final outcome. Czúcz et al. (2011) projected that, as early as 2050, 56 – 99% of present-day beech forests might be outside their present bioclimatic niche, whereas Móricz et al. (2013) projected that most sites will become intolerable for beech by 2100. These projections were based on the results of the REMO regional climate model simulations by Jacob et al. (2001) and Jacob et al. (2007) assuming the A1B IPCC-SRES emission scenario.

Based on these projections, the main assumptions of this study are that (1) extinction mortality of beech will increasingly appear, (2) all beech stands will disappear from Zala country by around the end of the century, and (3) mortality starts and terminates earlier on sites of yield class 6, and later on sites of yield class 1 that have more potentials to support the trees with water. It was also assumed that, at lower intensities, extinction mortality only affects individual trees. This effect could be seen equivalent to a self-thinning so that, up to a certain level, dead trees can be harvested as part of regular thinnings. Stands affected this way do not need to be regenerated, and the remaining trees of these stands continue to grow at least until more severe mortality ensues. This mortality is referred to below as *individual tree mortality*. In contrast, higher rates of mortality that will result in the dieback of entire stands or their part(s) so that they need to be harvested and regenerated (if that is possible at all), will be referred to below as *stand mortality*. The occurrence of this mortality triggers CASMOFOR to simulate the harvesting and regenerating of the area.

Considering all the above, the rate of extinction mortality over time was modelled to increase along yield class-specific exponential curves for both individual mortality (Fig. 4a) and stand mortality (Fig. 4b). Extinction mortality is simulated to occur in each stand each year, irrespectively of the age of the stand, assuming that the causes of mortality may be mostly age-independent. For individual tree-level mortality, it is further assumed that, beyond a threshold of a cumulative mortality of 0.5 relative to the standing volume, individual tree mortality rate is not increased, and any mortality becomes part of stand mortality.

Table 2. Main characteristics of the forest climate types applied in Hungary based on data of the Hungarian Meteorological Office and the forest climatic types as defined by Borhidi (1960), Járó (1966) and Mátyás & Czimber (2004).

Forest climate type	Indicator species	Occurrence of beech	1961–1990 mean annual temperature [°C]	1961–1990 mean annual precipitation [mm]
Beech	European beech	Dominant	6.5 – 8	>700
Oak-hornbeam	Sessile oak and Hornbeam (*Carpinus betulus* L.)	Mixing	8 – 9	650 – 700
Turkey oak	Turkey oak (*Quercus cerris* L.)	Rare	9 – 10	600 – 650
Forest steppe	None	Practically extinct	10 – 11	<600

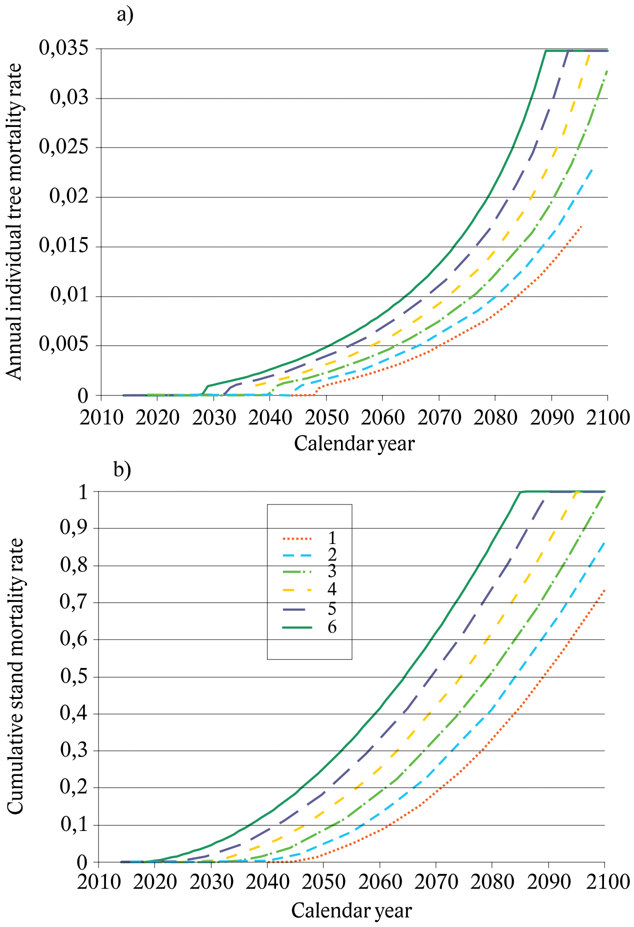

Fig. 4. Assumed (a) annual individual tree and (b) cumulative stand mortality rates in beech stands by yield class (1 is best, 6 is poorest).

2.4. Tree growth, root-to-shoot ratio, and decay of wood and litter

In the baseline scenario, the growth of beech was modelled using the area and yield class of the beech stands as estimated by the National Forest Database for 2012.

In order to estimate volume growth as accurately as possible, the country-level yield tables for beech (by Mendlik 1983) were adjusted by Veperdi (2013) using the local height growth curves over age that were developed from the most recent statistical forest inventory data. First, yield data of Mendlik (1983) was adjusted for each yield class using height values from these local height growth curves and the strong relationship between height and yield as represented in the country-level yield tables. Then, volume growth rates were calculated from the adjusted yield curves. Growth rates for ages not covered by the yield table (i.e., 1 – 10 years) were developed by linear interpolation.

For the estimation of annual volume growth over age, the annual height growth rate was used which was assumed to change over time due to climate change. The expected change was modelled using the evidence by Somogyi (2008a) who showed that, for sites of the same characteristics except for climate type, the mean height of beech stands in the "Beech" and "Turkey oak" climate types at the age of 80 years amounts to 25 m and 18 m (i.e., 0.31 and 0.23 m yr^{-1}), respectively. The difference, i.e., 7 m, is attributed to differences between the mean annual temperature of the two above climate types, which is about 2 °C. If such a temperature is to occur by 2100, it might induce a reduction of mean tree height (at the age of 80 years) of a bit over 0.1 m per year. The expected rate of temperature increase will be larger, however, a shift from the Beech climate type to the Turkey oak climate type would also involve a reduction of the annual precipitation of about 100 mm. Considering that no decrease of the total annual precipitation is projected (although summer precipitation is projected to decrease), a slower linear decrease of growth of 0.05 m per year was assumed.

For sessile oak, the country-level yield table by Béky (1981) was used for the entire projection period, assuming that the sites will not deteriorate for this species. As only the yield class of beech is known for forests currently under predominantly beech cover, the yield class of sessile oak (YC_{SO}) had to be estimated from the current yield class of beech (YC_B). This was done using the following regression equation:

$$YC_{SO} = p_1 + p_2 x \, YC_B \qquad [1]$$

The parameters of the equation were estimated using data of stands with each species having a species ratio of at least 33%.

Consistent with empirical studies (e.g., Usoltsev 2001), root-to-shoot ratios in CASMOFOR are larger by 50% for yield classes 5 and 6 than for yield classes 1 and 2 in the baseline scenario. In the climate change scenario, this relationship was used to gradually increase the root-to-shoot ratio of beech under the changing climate scenario as the yield class changes due to growth decline. For oak, consistent with the assumption that its growth will not decline, the ratio was kept constant.

For decay rates for both beech and sessile oak, it is assumed based on available (unpublished) local evidence that they will first gradually increase due to warming (until the middle of the century, by maximum 1.4 times relative to current rates), then gradually decrease by the same rate due to dry conditions.

2.5. Regeneration, preventive species replacement and forest area

Sustainable forest management requires that forest areas are maintained as far as possible by regenerating all areas that have lost their forest cover due to either harvest or natural disturbances. In Hungary and elsewhere in Europe, it has been a general practice for a long time to regenerate stands of indigenous species using the species of the mature stand that are adapted to local conditions. The expected large-scale dieback of beech may require to change this practice, and to use more drought-tolerant species (even despite some adverse implications like higher costs) to avoid climate change induced mortality in the regenerated beech stands. Therefore, as a preventive measure in the "maximum adaptation" climate change scenario (see below), beech is assumed to be gradually regenerated with sessile oak. The replacement was assumed to occur in the first few years if a stand is disturbed and must be harvested. Later, however, replacement is projected to increasingly become a practice at regular final harvests (Fig. 5). It was additionally assumed that sessile oak may not only be able to regenerate but also survive until 2100. Also assuming no afforestations in the region, the total area of all forests will thus assumed to remain constant during the simulations.

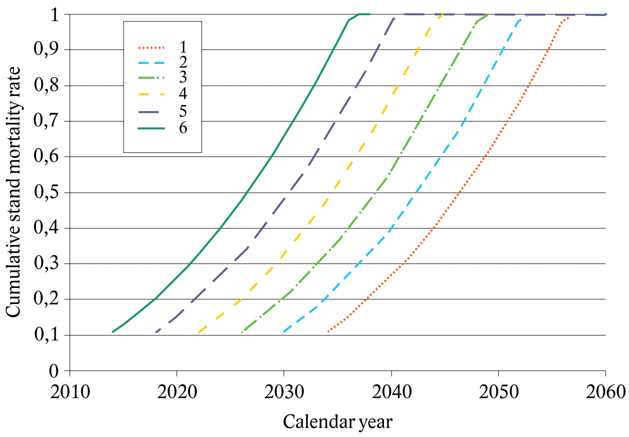

Fig. 5. The assumed fraction of beech that may be replaced by sessile oak in preventive regenerations (applied in the "maximum adaptation" scenario, see below) at the time of final harvest by yield class (1 is best, 6 is poorest).

2.6. Silviculture

For the mostly pure stands of the country, CASMOFOR's built in silvicultural model assumes a species-specific and yield-class dependent system of timing and intensity of thinnings (3 – 5 times over the rotation period, depending on yield class) and final harvest (at the age of 120 years). This silvicultural system, which is used in the baseline scenario without modification, assumes that density-dependent self-thinning may also occur but only at very low rates in the year before thinnings.

For the climate change scenarios that involve salvage cutting (see below), it is assumed that additional thinnings will be necessary between or in the years of standard thinnings whenever the rate of mortality exceeds the level of a light thinning, i.e., 15% of standing volume. One reason to conduct salvage cuttings is to avoid subsequent disturbances such as fire due to fuel accumulation. Small trees (i.e., those below 5 cm of DBH) and trees in stands with a mortality rate below the above threshold are, however, assumed to be left in the forest so that their dead organic matter decomposes.

When stand mortality occurs (at any age), all wood in the stand is harvested, and then the stand is artificially regenerated. Wood utilization rates (i.e., the amount of timber utilized in thinnings and final harvests relative to the amount of wood harvested) as set in CASMOFOR are assumed to be constant over time, and are age-dependent.

Finally, the age of final harvesting may be reduced in practice, for example in order to avoid mortality and any loss of valuable timber in the future. However, for practical modeling reasons, such reduction is not modelled in this study, and the same rotation period (i.e., 120 years) is used in both the baseline scenario and in the climate change scenarios unless stand mortality occurs and harvest and regeneration need to be applied.

2.7. Summary of forest management adaptation scenarios

In the baseline scenario ("BL"), CASMOFOR was run using the time-independent default parameter set of the model. This includes parameters of the standard silvicultural model, the assumptions that all stands are harvested at the age of 120 years, and that harvested stands are regenerated with beech. In the climate change scenarios, model parameters of individual and stand mortality, growth rate, root-to-shoot ratio and decay rates were modified by climate change as described above. To compare the effect of possible forest management adaptation measures, three management adaptation scenarios were identified. In the non-action ("NA") scenario, which is unlikely to happen, it was assumed that thinnings are done according to the standard model, but climate change affects the natural processes, final cuttings are done at standard rotation age or when stand mortality makes it necessary, and regenerations are only done using beech. In the "salvage cutting" scenario ("SC") the dead wood that appears due to the extinction mortality, is assumed to be salvaged in additional thinnings. Finally, the maximum adaptation scenario ("MA"), which is expected to be the best approximation of what is going to take place in practice, involves salvage cutting but also assumes that the preventive replacement of beech with sessile oak will also take place as described above (Table 3).

3. Results

The adjustments of the standard country-level beech yield tables to the conditions of Zala County resulted in an increase of the yield of about 18 – 20%, depending on yield class.

Table 3. Parameters of biophysical processes and forest management adaptation measures used in the various scenarios. Abbreviations: BL – baseline scenario, NA – no-action scenario, SC – salvage cutting scenario, MA – maximum adaptation scenario.

Scenario	Tree growth, root-to-shoot ratio, decay of wood and litter	Thinnings	Age of final harvest	Preventive species replacement
BL	Standard	Standard	120 years	No
NA	Modified by climate change	Standard	120 years or whenever necessary due to stand mortality	No
SC		Salvage cutting is enabled		No
MA				Yes

The estimated parameters of the Equation 1 were p_1 = 1.2418 and p_2 = 0.4155 with statistics $R^2 = 0.2037$, N = 394, $p << 0.05$. As a consequence, replacing beech with oak, and considering differences in their growth, yield and wood density, will affect the area-specific mean annual biomass carbon increment at age 80 years: it will decrease by about 1.77 $tCha^{-1}yr^{-1}$ (from 7.6 $tCha^{-1}yr^{-1}$) in beech yield class 1 and increase by about 1.77 $tCha^{-1}yr^{-1}$ (from 2.3 $tCha^{-1}yr^{-1}$) in beech yield class 6.

The total carbon stocks projected for BL demonstrate variation over time around a long-term average (Fig. 6a). The variation is due to the combined effect of the age-specific yield and harvest dynamics and the dynamics of the distribution of the area of the individual stands by yield class. Because all model parameters have fixed values and the length of the rotation period is the same (i.e., 120 years) for each yield class, the total carbon stocks vary in a cycle whose length is equal to that of the rotation period.

Relative to the BL, significant differences are projected in the "MA" scenario in areas that remain occupied by beech (Fig. 6b): the biomass and the litter pools are projected to lose 86% (Fig. 7a) and 70% of their carbon stock, respectively, whereas the deadwood pool is projected to gain 58% carbon by 2100. In total, the forest carbon pools (excluding soils, and again, relative to the baseline) are projected to lose 64% of their carbon stocks (Fig. 6b). In areas where sessile oak is expected to replace beech, all pools are projected to slowly but steadily increase after regeneration (Fig. 6c). Note that Figures 6b and 6c demonstrate annual changes in carbon stocks due to both changes in area covered by the respective species and specific processes (i.e., mortality, tree growth, harvest etc.) in any given year in the areas covered by these species.

Concerning the effect natural processes (i.e., the "NA" scenario) on the carbon balance (relative to the BL scenario, Fig. 7a–b), individual tree mortality seems to have a very small effect. The changes of root-to-shoot ratio and decay rates first have a slightly increasing combined effect, but then lead to a small carbon loss. The decline of volume growth is more expressed and leads to steadily increasing losses. Stand mortality is not a concern for decades, but then becomes by far the most important factor leading to large carbon losses. The effect of all natural processes and harvest (i.e., the "SC" scenario) was also found to be very small (to the point that it was not practical to include it in the figures). A replacement of beech with sessile oak (i.e., the "MA" scenario), however, moderately improved the situation concerning the total carbon stocks (Fig. 7a) and offset a bit more than the half of the loss from the above-ground biomass (Fig. 7b).

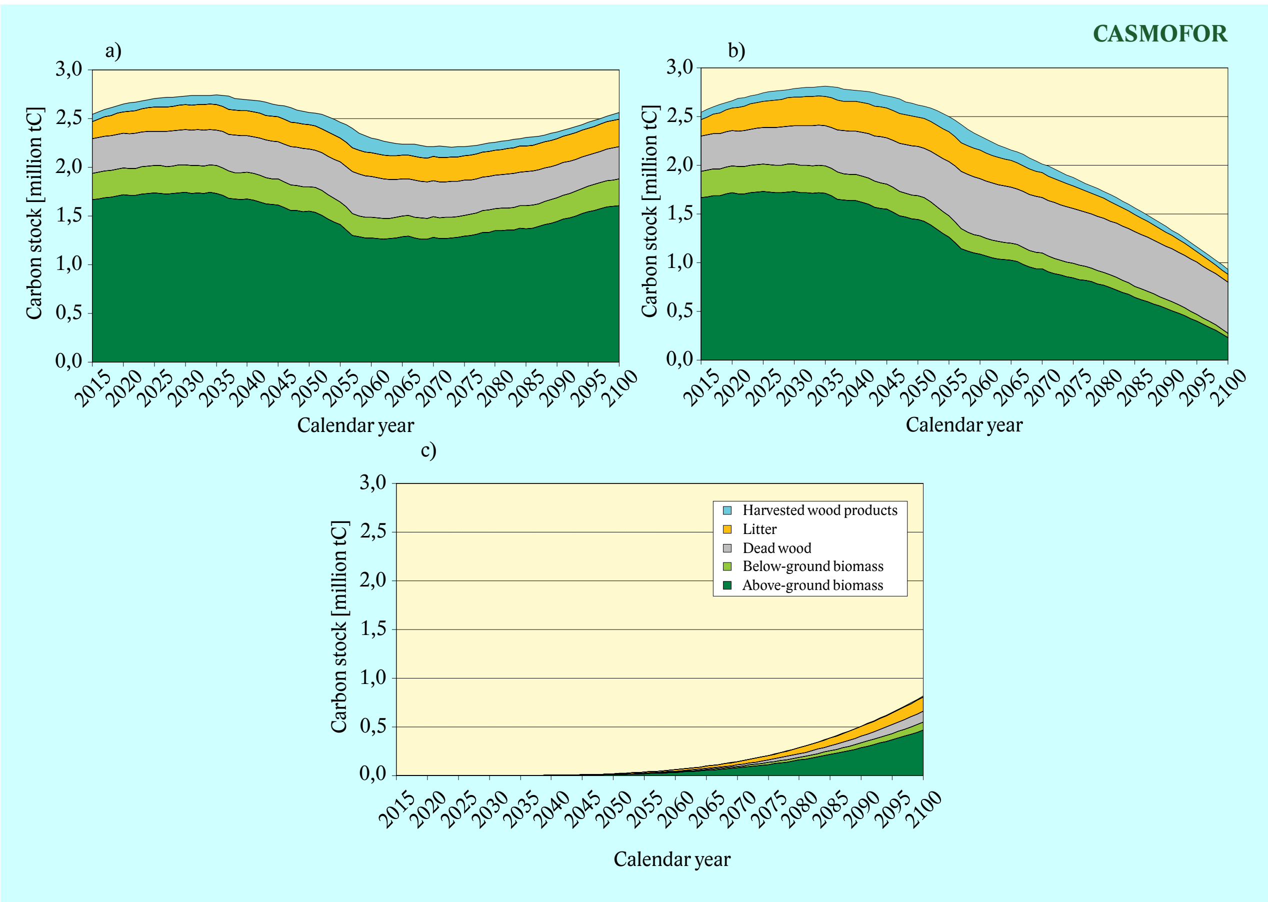

Fig. 6. The simulated evolution of carbon stocks in Zala County in the area currently occupied by beech by carbon pool (excluding soils): (a) for the BL scenario; and for the "MA" scenario: (b) for the area that at any time point in future remains covered by beech and (c) for the area where sessile oak replaces beech. The graphs represent direct outputs from CASMOFOR. The same scale is applied on the y-axis on all graphs.

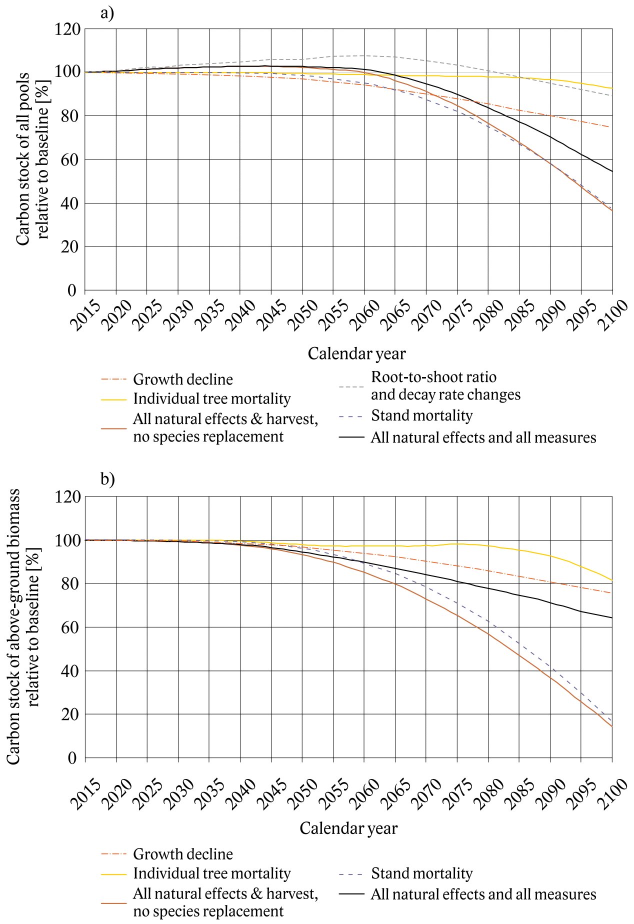

Fig. 7. The carbon stock of (a) all pools (excluding soil) and (b) the above-ground biomass pool over time, relative (%) to the BL, reflecting the partial and combined contribution of various natural and/or human-induced effects under climate change.

4. Discussion

Forest carbon balance has so far been projected under rather different conditions: mainly without assuming climate change (e.g., Zang & Xu 2003; Schmid et al. 2006; Chen et al. 2010; Krankina et al. 2012; Pilli et al. 2013); only considering some biophysical effects (such as change of growth rate) of climate change but excluding possible mass mortality (e.g., Matala et al. 2005; Morales et al. 2007; Eggers et al. 2008; Jansson et al. 2008; Rötzer et al. 2009; Tatarinov & Cienciala 2009; Wamelink et al. 2009; Hurteau et al. 2014; Smyth et al. 2014), assuming a low level tree mortality (Hlásny et al. 2014a), and/or the possible and necessary forestry measures (e.g., Birdsey et al. 1993). Also, projections differ with respect to scale from the stand level (Hlásny et al. 2014a) to large areas (Kurz et al. 2008).

This study reports on a comprehensive integrated assessment of the impact of some major possible effects of climate change, including an assumed mass mortality, for a large forest area. The assessment includes effects on tree growth, root-to-shoot ratio and decomposition rate, but excludes the effects of CO_2 fertilization and N-deposition. Non-CO_2 emissions are not considered, either. The results show that, if tree mortality will take place as projected, rather high CO_2 emissions from forests can be expected due to losses of forest carbon stocks. If upscaled to the country level using a simple ratio of total forest area in the country (1,933,604 ha) and the beech forest area in Zala County in 2012 (i.e., assuming the same average rate of forest decline all over the country, subject to large regional variation), these emissions may reach a level in the second half of the century that is about 50% of the current (2012) total emissions of the country (i.e., 57.6 million tCO_2 equivalent, NIR Hungary, 2014). These emissions might offset a significant portion or even all of the positive effect of future mitigation efforts, thus risking a positive discernible climatic feedback.

Therefore, it is important to analyses to what extent, if at all, forestry can prevent or mitigate these emissions. Of all theoretical (Yousefpour et al. 2013) and more realistic (Susaeta et al. 2014; Nabuurs et al. 2013; Smyth et al. 2014) forest management options, this study focuses on two practical ones.

Concerning salvage cutting, the results of the study demonstrate that it may have rather small effects on net emissions. This result confirms the conclusions by Nabuurs et al. (2007) and Smyth et al. (2014) that the forest management strategy with the largest sustained mitigation benefit is the one that maintains or increases forest carbon stocks.

A much more promising adaptation measure (Koltsröm et al. 2011), although with slow long-term effects, is to significantly speed up artificial species replacement where necessary. Species replacement has been suggested as a potentially successful adaptation measure (e.g., Lindner et al. 2010, 2014; Hlásny et al. 2014b). It has been proven to be possible in Zala County and elsewhere in Hungary at small scales for decades with the aim to improve stand quality in many damaged or degraded forests (Koloszár 2010). Replacing beech with oak can successfully be achieved through the promotion of natural regeneration of oak in mixed stands, or seeding or planting oak seedlings to replace pure beech stands. Favoring oak over beech during thinnings might also be used to replace beech with oak.

However, the availability of propagation material, and competition between tree species and even between trees, herbs and bushes might make such replacements difficult in some places. Also, large areas of the Forest Steppe climate type, or even the Turkey oak climate type in Hungary may become completely unsuitable to support trees as early as the end of this century. The amount of carbon fixed by any forest-replacing ecosystem will likely be very small in any event, only offsetting a fraction of the likely emissions. Finally, although artificial species replacements with the above techniques have been successfully conducted in Hungary from both domestic forestry budget and subsidies from the European Union, such interventions may be prohibitively costly if conducted on large scales (Lindner et al. 2010).

Because all of the above, the assumption that all beech stands will be replaced with oak by 2100 can be considered a “technical potential” scenario. More analysis is needed to explore the optimal speed, method and extent of re-structuring forests at reasonable costs to avoid forest loss as much as possible. Future analyses should also explore more accurate ways to estimate the effects of climate change and the various forest management options. This study was conducted using an open source model with a fully described methodology and database which ensures that calculations can be checked, and that the simulations can be reproduced for any tree species with appropriate data. CASMOFOR ranked highest in a recent comparison of carbon accounting models as it produced the smallest mean square error of biomass estimates among the models compared (Ndalowa 2014). CASMOFOR's prediction of average net annual biomass carbon removals in 2008–2012 by afforestations 1990–2012 of 1.1 $MtCO_2yr^{-1}$ (Somogyi 2006) was also validated by the 1.2 $MtCO_2yr^{-1}$ estimate in 2014 (NIR Hungary, 2014).

Nevertheless, the results of the modelling are subject to uncertainties due to a number of factors. The parameters of the model that can be used under constant environmental conditions have moderate uncertainties, some of which were used as part of a Monte Carlo uncertainty analysis of the model (see NIR Hungary, 2012). Parameters to estimate the effect of climate change (e.g., the change of height growth rate) have higher uncertainties which, in general, could not be estimated. Finally, the results may also be sensitive to the assumptions used.

The single most important assumptions with high uncertainties concern the rate and timing of beech mortality. Projecting future forest decline is rather challenging (Rasztovics et al. 2012). As the findings of both this and several other studies (e.g., Crookston et al. 2010; Hlásny et al. 2014a) confirm, the modeling results are much more sensitive to projections of mortality than to those of tree growth, and that emissions from mass mortality can indeed be large (Kurz et al. 2008).

The assumed mortality in this study is based on climate change scenarios that are consistent with those of IPCC (2013a). However, the projected increase of the temperature by the end of the century (3.7 °C) is much more than the current temperature range in the Zala county (1.5 °C) or the difference between the various adjacent climate types applied in Hungary (about 1 °C), but is by 0.3 – 1 – 7 °C lower than what Bartholy et al. (2014) predicted. Also, European beech is a species favoring cool and humid Atlantic climate (Fang 2006; Mátyás et al. 2010) and extreme droughts may be an increasingly serious limiting factor due to high summer temperature and low precipitation (Rasztovics et al. 2014). Recent beech decline events in the region (Lakatos & Molnar 2009; Mátyás et al. 2010) and other projections (Czucz et al. 2011) also suggest possible large beech decline in the future.

This possibility is further supported by the fact that the projected decline of the amount of water available for the trees during the growing season can be very significant in the light of evidence by Elkin et al. (2013) that a significant decline of forest biomass may occur even due to relatively small climatic shifts at initially warm-dry lower elevations due to the limitation of growth by precipitation, which is clearly the case in Zala County. Crookston et al. (2010) also concur that if climatic shifts go beyond the current climatic range for areas where a species occurs currently, then mortality rates will increase, eventually resulting in the local extinction of the species. The above evidence and assumptions are also in line with the assumption applied in developing the mortality scenarios that mortality will first happen on the poorer sites and start later on better sites.

An important limitation of the assumed mortality scenarios is that, because of their rare and unpredictable nature and the many knowledge gaps reported by Sommers et al. (2014) and others recently, the effects of extreme natural disturbances such as forest fires, droughts, insects and others that produce non-CO_2 emissions could not be fully, and separately, modelled. Also, the rate of disturbances may be overestimated as the current range limits are not always constrained by climate (Lindner et al. 2014).

Considering all the above, the dramatic forest decline assumptions that were developed based on Móricz et al. (2013) and Zimmermann et al. (2013a) and that were the basis for the emission estimation in this study, should be taken as a serious possibility.

Contrary to assumptions in this study, tree growth might, at least in the short run, increase due to increasing temperature (Crookston et al. 2010). However, a temperature increase of only 2 °C was assumed, and the calculated growth reduction rate was further decreased to be conservative, i.e., to underestimate potential emissions. Bošel'a et al. (2014) found that the growth of trees was positively affected at higher altitudes where low temperature is the limiting factor, but it was negatively affected by summer temperatures at lower altitudes where precipitation is the limiting factor. The Zala County is such a low-elevation area for which the above findings are consistent with the assumptions of this study. Hlásny et al. (2011) also reported a drop of about 30% in the growth of beech at elevations similar to those in the Zala Country.

Carbon sequestration may also change after mortality or mortality-induced harvest due to the increased available light through the canopy. However, the average annual rate of ecosystem carbon sequestration over a longer period may not change much, and it was indeed found to be similar in harvested and un-harvested forests (Davis et al. 2009). Battles et al. (2008) reported that tree growth declined under all climate scenarios and management regimes under the conditions of their analysis.

Assuming acclimation to CO_2 effects, Reyer et al. (2013) also estimated lower net primary productivity for the Zala region. However, Reyer et al. (2013) also estimated higher net primary productivity for the Zala region and elsewhere with persistent CO_2 effects. In lack of proper data, the effects of CO_2 or nitrogen fertilization on tree growth could not be included in our analysis. Also, only unverified data and expert judgment could be applied to model changes in less important parameters such as root-to-shoot ratio and decay rates. The allocation of NPP to slow and fast-turnover biomass pools can differ significantly in stands of different species (Konôpka et al., 2013), which indicates that this allocation may change as climate changes. However, for reasons mentioned above, uncertainties in the non-biomass estimation modules are of limited importance with respect to the conclusions of the study.

The modeling of the HWP stocks currently involves fixed parameters, ignoring their sensitivity to climate change or socio-economic processes. Thus, the HWP stock estimates are only indicative. The combination of the assumed half-life time of the various wood products (that are the same as the default values in IPCC 2014b) with their (fixed) utilization rates results in a combined half-life time for the HWP pool that is roughly equal to that applied for the dead wood pool. This implies that salvaging of trees that died due to extinction mortality has only an insignificant effect on reducing net carbon losses.

The current silvicultural model reflects historical practices at the country-level, but such practices will most probably change under increasing rates of mortality. This is partly modelled by including new thinnings and final harvests that become necessary above a given threshold of extinction mortality. However, reducing the age of final harvesting can have a larger effect and should be modeled in more details in future studies.

Finally, the uncertainty analysis shows that estimates of carbon stock changes involve significantly larger uncertainties than those of carbon stocks, especially for the deadwood pool. Clearly, additional evidence is needed to better model the effects of climate change on relevant natural processes (also considering that the effect of the various processes may be different if simulated in isolation or in combination with other processes). In conclusion, none of these uncertainties invalidate the conclusion of the analysis, which is that a discernible positive feedback from very large net emissions might occur from the forests of the Zala County, and maybe also from other forests, in case large-scale mortality occurs as suggested by recent studies.

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