

Sustainably managing forests for a resilient future: an evaluation of different thinning intensities

Duurzaam bosbeheer voor een veerkrachtige toekomst: een evaluatie van verschillende dunningsintensiteiten

Promotor:

Prof. Bart Muys

Departement Aard- en

Omgevingswetenschappen

Afdeling Bos, Natuur en Landschap

Masterproef voorgedragen

tot het behalen van het diploma van

Master of Science in de bio-ingenieurswetenschappen:

Landbeheer

Anais NEUFKENS

Juni 2024

Dit proefschrift is een examendocument dat na de verdediging niet meer werd gecorrigeerd voor eventueel vastgestelde fouten. In publicaties mag naar dit proefwerk verwezen worden mits schriftelijke toelating van de promotor, vermeld op de titelpagina.

Acknowledgements

With my studies coming to an end, which have been a quite significant and memorable chapter of my life, this thesis clearly marks the finish. Making this thesis was truly an enriching experience, where I not only learned more about a really interesting topic but also grew personally as well. A thesis is not written without some struggles now and then and a lot of trial and error, and although it can be quite frustrating sometimes, when you finally overcome these obstacles, it really feels like a true accomplishment. However, this would not have been possible without the opportunities I got and the people that helped and supported me along this thesis. Therefore, in the following lines, I want to express my gratitude to some people forming an important part of this achievement.

First of all, I want to thank my promotor prof. Bart Muys for making this thesis possible. I am glad I was able to write a thesis on a topic which combined multiple aspects that really interested me, being forests, stability, climate change and forest management. Next, I would like to say a huge thank you to my supervisor Dr. Ilié Storms who was present in multiple parts of my thesis, going from teaching me how to use iLand and helping me with unravelling parts of my methodology to proofreading my thesis and keeping me motivated. I also would like to thank Dr. Werner Rammer for answering my questions about the ABE module of iLand and even optimizing certain parts of the model for me. Next, I would also like to thank my friends and family for their enormous support. Specifically, I would like to thank my parents and my brother, Maxime, for always being there for me and to always listen to what I had to say about my thesis, even though they probably never quite understood what it was about. Secondly, I would also like to thank my boyfriend, Quinten Dhaenens, for supporting me almost every day and to also proofread this thesis. Finally, I would also like to mention my best friends (and fellow bio-engineers in spe), Marijke Geuskens, Jessica Flamen, Sarah van Lommel, Hannah Guyon, Lieselot Lauwers and Laura Raddoux, for bringing laughter and joy in every other day, and like this making this thesis and study journey complete and memorable.

Summary

Forests are essential to human society as they deliver different kinds of ecosystem services. However, nowadays, climate change is exposing our forests to changing environmental conditions, being higher temperatures, erratic rainfall, and changing disturbance regimes. To ensure the continuous delivery of ecosystem services, we should therefore manage our forests to increase their resistance and resilience against climate change, which means increasing their ability to resist change as well as to recover fast after a disturbance took place. This type of sustainable management is called Climate-Smart Forestry. In this regard, thinning has already been proposed as a management action that can reduce between-tree competition and increase drought tolerance, mitigating the effect of climate change. Therefore, we used the mechanistic individual-based landscape model, iLand, to evaluate how different thinning intensities could influence the stability of species composition and forest structure to climate change, in the Meerdaal and Heverlee forest complex. We expected that higher thinning intensities could maintain or improve forest stability. We programmed the management of this forest complex by using the forest management plan and suggestions from the forest managers and tested four different management scenarios, being no management and a removal of 5%, 10%, or 20% of the total basal area of the forest every eight years, under varying climatic conditions. Our results indicated that overall, increasing thinning intensity, shifted the species composition and the diameter class distribution more, hence reducing the resistance against climate change, although trends in the resistance of the diameter class distribution were less clear. The resilience of the basal area also dropped with increasing thinning intensity in all climate scenarios, apart from the most hot and dry climate, where no difference was observed between different thinning intensities, and where resilience was overall the lowest. We found that climate influenced both resistance and resilience of the forest, indicating the climate sensitivity of these stability metrics. Usually, the largest change was found under the most extreme climate, where the forest characteristics shifted towards smaller trees and an increased abundance of native oaks, Douglas fir and black pine. Our research explored the potential of using a mechanistic landscape model to simulate the effect of climate-smart forest management strategies for the first time in the Atlantic region and pinpoints some new research lines that could be addressed using this model.

Samenvatting

Bossen zijn essentieel voor de maatschappij omwille van de verschillende ecosysteemdiensten die ze ons leveren. Niettemin worden onze bossen tegenwoordig bedreigd door de klimaatsverandering, hetgeen gepaard gaat met hogere temperaturen, onregelmatige regenval en een veranderend verstoringspatroon. Om de ecosysteemdiensten die ze leveren te garanderen naar de toekomst toe, moeten we daarom ons bosbeheer richten op het verhogen van de weerstand en veerkrachtigheid van het bos tegen de klimaatsverandering. Dit betekent dat we zijn capaciteit om veranderingen te weerstaan en snel te herstellen na een verstoring moeten verhogen. Dit type van duurzaam bosbeheer wordt ook wel klimaatslim bosbeheer genoemd. Dunning werd reeds voorgesteld als een beheeringreep die de weerbaarheid van het bos tegen klimaatsverandering kan vergroten door de competitie tussen bomen te reduceren en de droogte weerstand te vergroten. Bijgevolg onderzochten we met het mechanistische landschapsmodel iLand hoe verschillende dunningsintensiteiten de stabiliteit van de soortensamenstelling en de bosstructuur tegen klimaatsverandering kunnen beïnvloeden voor het Meerdaalwoud en Heverleebos. Er werd verwacht dat een hogere dunningsintensiteit gepaard zou gaan met een hogere stabiliteit. We programmeerden het huidige bosbeheer van het bos op basis van het huidige bosbeheerplan en suggesties van de boswachters, en evalueerden vier beheersscenario's, met name een nulbeheer en een verwijdering van 5%, 10%, of 20% van het grondvlak van het bos om de acht jaar, onder variërende klimaatscenario's. Onze resultaten toonden aan dat een hogere dunningsintensiteit de soortensamenstelling en de bosstructuur sterker veranderden, alhoewel deze trend minder duidelijk was voor de bosstructuur. Het grondvlak van het bos herstelde minder goed bij een hogere dunningsintensiteit, wat het geval was voor alle klimaten behalve in het warm en droog klimaat waar de veerkrachtigheid van het bos het laagst was en niet verschillend tussen de dunningsscenario's. Daarenboven vonden we ook dat de stabiliteit van het bos beïnvloed werd door het klimaat. Het meest extreme klimaat werd gekenmerkt door de grootste veranderingen in de boskenmerken, waarbij het aantal kleine bomen toenam en soorten zoals inheemse eiken, douglasspar en Corsicaanse den algemener werden. Ons onderzoek verkende voor het eerst in de Atlantische regio het gebruik van een mechanistisch landschapsmodel om klimaatslimme beheerstrategieën te onderzoeken en geeft suggesties voor verder onderzoek gebruikmakend van dit model.

List of Abbreviations and Symbols

Abbreviation	Explanation
ABE	Agent-based model of forest management
ANOVA	Analysis of variance
CH ₄	Methane
CO ₂	Carbon dioxide
CSF	Climate-Smart Forestry
EURO-CORDEX	Coordinated downscaling experiment - European domain
FAO	Food and Agriculture Organization of the United Nations
FLM	Forest landscape models
HD relationship	Height-diameter relationship
HI	Higher intensity thinning
iLand	The individual-based forest landscape and disturbance model
INBO	Instituut voor Natuur- en Bosonderzoek
IPCC	Intergovernmental Panel on Climate Change
LI	Lower intensity thinning
NI	Normal intensity thinning
NM	No management
N ₂ O	Nitrous oxide
O ₃	Ozone
PERMANOVA	Permutational multivariate analysis of variance
RCP	Representative concentration pathway
S ₀	The value of the system variable farthest away from the initial conditions
S _R	Initial value of the system variable
SSP	Shared socio-economic pathway
STP	Stand treatment program
S _Y	The value of the system variable after a certain period following the disturbance
VPD	Vapor pressure deficit

List of Tables

Table 1: The median Bray-Curtis Dissimilarity between the species composition at the start and the species composition at the end of the simulation period (2020-2095) for the different climate and management scenarios.	37
Table 2: The median Bray-Curtis Dissimilarity between the diameter distribution at the start and the diameter distribution at the end of the simulation period (2020-2095) for the different climate and management scenarios.	41
Table 3: The median of the resilience over the thinning interventions for every management and climate scenario.	46
Table 4: Comparison of the resilience at the beginning (in the first three thinning interventions) with the resilience at the end (in the last three thinning interventions) for every climate and management scenario.	49
Table 5: The mean share of each species (%) in the last five years of the simulation period as presented in Figure 8 and Figure 9 for every climate and management scenario.	91
Table 6: The mean share of trees (%) in every diameter class (D) of the last five years of the simulation period as presented in Figure 10 and Figure 11 for every climate and management scenario.	92
Table 7: The Scheirer-Ray-Hare test showing the H-statistic, the degrees of freedom (df), and the <i>P</i> -value to determine the influence of management, climate, and their interaction on the resistance of the species composition.	93
Table 8: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the historical climate.	93
Table 9: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the moderate climate.	94
Table 10: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the hot climate.	94

Table 11: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the hot and dry climate.....	94
Table 12: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the no management scenario.	95
Table 13: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the low intensity scenario.	95
Table 14: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the normal intensity scenario.	96
Table 15: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the high intensity scenario.	96
Table 16: The Scheirer-Ray-Hare test showing the H-statistic, the degrees of freedom (df), and the <i>P</i> -value to determine the influence of management, climate, and their interaction on the resistance of the diameter class distribution.	97
Table 17: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the historical climate.	97
Table 18: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the moderate climate.	98
Table 19: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the hot climate.....	98
Table 20: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the hot and dry climate.....	98

Table 21: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the no management scenario.....	99
Table 22: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the low intensity scenario.	99
Table 23: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the normal intensity scenario.....	100
Table 24: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the high intensity scenario.	100
Table 25: The Scheirer-Ray-Hare test showing the H-statistic, the degrees of freedom (df), and the <i>P</i> -value to determine the influence of management, climate, and their interaction on the resilience of the basal area of the forest complex.	101
Table 26: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the historical climate.	101
Table 27: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the moderate climate.	102
Table 28: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the hot climate.....	102
Table 29: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the hot and dry climate.....	102
Table 30: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different climate scenarios in the low intensity scenario.	103

Table 31: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different climate scenarios in the normal intensity scenario.....	103
Table 32: The Z-statistic, unadjusted and adjusted <i>P</i> -values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different climate scenarios in the high intensity scenario.	103

List of Figures

Figure 1: Figure based on Van Meerbeek et al. (2021) showing the system variable (S) through time (t) in different disturbance scenarios (blue zone), with A) a pulse perturbation smaller than the system's tolerance, B) a pulse perturbation larger than the system's tolerance, C) a ramp perturbation, and D) a press perturbation.	11
Figure 2: Delineation of the different climate scenarios based on the average maximum temperature (°C) and the average precipitation sum (mm) in June, July, and August in the period 2081-2100.....	27
Figure 3: A) Division of the forest landscape into management units, which are further divided into stand units or forest stands, B) a managed stand in the forest complex, C) an unmanaged stand in the forest complex.	28
Figure 4: The basal area ($\text{m}^2.\text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the historical climate	34
Figure 5: The basal area ($\text{m}^2.\text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the moderate climate	34
Figure 6: The basal area ($\text{m}^2.\text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the hot climate.....	35
Figure 7: The basal area ($\text{m}^2.\text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the hot and dry climate.....	35
Figure 8: The basal area shares of the most abundant species at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate.	38
Figure 9: The basal area shares of the most abundant species at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity.....	39
Figure 10: The share of trees for each diameter class at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate.....	42
Figure 11: The share of trees for each diameter class at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity.....	43

Figure 12: The mean basal area ($\text{m}^2.\text{ha}^{-1}$) of the landscape through time for each climate and management scenario.	44
Figure 13: The median resilience index through time for each thinning intervention given per climate scenario, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate.	45
Figure 14: The distribution of the resilience for every management scenario with the significant differences denoted by different letters above the boxplots, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate.....	47
Figure 15: The distribution of the resilience for every climate scenario with the significant differences denoted by different letters above the boxplots, with A) lower intensity, B) normal intensity, and C) higher intensity.	48

Table of Contents

1	Introduction	1
2	Literature study.....	3
2.1	Forests.....	3
2.2	Climate change.....	5
2.3	Forests and climate change.....	6
2.4	Climate-Smart Forestry	9
2.5	Forest modelling	15
3	Methodology.....	20
3.1	Study site.....	20
3.2	Model description	23
3.3	Landscape, climate and management setup.....	26
3.4	Simulation design	30
3.5	Analysis.....	31
4	Results	33
4.1	Species dynamics	33
4.2	Resistance of the species composition to climate change	36
4.3	Resistance of the diameter class distribution to climate change.....	40
4.4	Basal area dynamics	44
4.5	Resilience of the basal area of the forest.....	45
5	Discussion.....	50
5.1	Species dynamics	50
5.2	Resistance of the species composition to climate change	52
5.3	Resistance of the diameter class distribution to climate change.....	53
5.4	Basal area dynamics	55
5.5	Resilience of the basal area of the forest.....	56

5.6	Limitations.....	58
5.7	Forest management implications.....	61
6	Conclusion.....	64
7	References	65
8	Appendices.....	88
	Appendix A: Detailed and numerical description of the programming of the ABE module in iLand.....	88
	Appendix B: The numerical details of the species composition and the diameter class distribution at the end of the simulation period.....	91
	Appendix C: the <i>P</i> -values from the Scheirer-Ray-Hare test and the Dunn's post hoc test for the pairwise comparison of the resistance of the species composition between management or climate scenarios.....	93
	Appendix D: the <i>P</i> -values from the Scheirer-Ray-Hare test and the Dunn's post hoc test for the pairwise comparison of the resistance of the diameter class distribution between management or climate scenarios.....	97
	Appendix E: the <i>P</i> -values from the Scheirer-Ray-Hare test and the Dunn's post hoc test for the pairwise comparison of the resilience of the basal area of the forest between management or climate scenarios.....	101
	Appendix F: Gebruik van Generatieve Artificiële Intelligentie (GenAI) - Formulier	104

1 Introduction

The intensification of the climate and changing disturbance regimes are putting pressure on ecosystems worldwide (IPCC, 2023a; Nolan et al., 2018; Patacca et al., 2023). Among those ecosystems forests have already experienced increased forest disturbances, and mortality as a consequence (McDowell et al., 2020; Patacca et al., 2023). Yet, towards the future forest dynamics remain uncertain, as rising temperatures and atmospheric CO₂ concentrations can increase forest productivity while disturbances reduce productivity (Lindner et al., 2010; Mina et al., 2022). Additionally, changing disturbance regimes have the ability to significantly change forest structure and composition, and push forests to a new state (Johnstone et al., 2016; Seidl, Rammer, & Spies, 2014). As forests are essential to human society because of the ecosystem services and goods they deliver, and their ability to mitigate climate change, which is reflected in their carbon sink potential (Harris et al., 2021; Winkel et al., 2022), we should aim at increasing the stability of forests against climate change. This can be done by incorporating Climate-Smart Forestry in our forests. This type of adaptive management aims at increasing the resistance and resilience of the forests against climate change, as well as increasing their mitigation potential (Bowditch et al., 2020). The former relates to an increased capacity of the forest to withstand change and a fast recovery to the initial conditions when a disturbance takes place (Van Meerbeek et al., 2021).

Within CSF, thinning is seen as a way to increase the stability of forests to climate change on the short term. Specifically, thinning can alleviate the competition between trees for resources, reducing tree stress consequently (Chase et al., 2016), and can reduce impacts from drought (Sohn, Saha, et al., 2016). Nevertheless, there still exists uncertainty about how to implement CSF (Sousa-Silva et al., 2016, 2018). Moreover, it is unsure whether a lower or higher thinning intensity would increase forest stability, as thinning can also make the microclimate more extreme by increasing the fluctuations in temperature and vapor pressure deficit within the stand, as well as decrease the stability of the stand against wind on the short term, although increasing it on the long term (Jiao-Jun et al., 2003; Rambo & North, 2009).

Therefore, in this thesis, we investigated the influence of different thinning intensities on the stability of forest composition and structure to climate change. To deal with the large spatial and temporal scales of our research question, forest modelling emerges as a useful

instrument to get a better understanding of the effect of management on ecosystem stability (He et al., 2011; Shifley et al., 2006). Especially process-based forest landscape models can help us look at forest dynamics under future climate while also accounting for spatial explicit forest processes such as management and disturbances (Landsberg, 2003; Shifley et al., 2017). Therefore, we used the individual-based forest landscape and disturbance model, iLand, which can model trees explicitly (Seidl, Rammer, et al., 2012). We investigated our research question for the case study of Meerdaal and Heverlee forest, situated in the south of Leuven, Belgium, for which iLand was already initialized. To study the influence of management on the forest's stability, we programmed the current management of the Meerdaal and Heverlee forest using the forest management plan and suggestions from the forest managers. We tested four different management scenarios being a no management scenario and management scenarios with a removal of 5%, 10%, or 20% from the basal area, under different climate scenarios. Finally, with this factorial experiment we could investigate the effect of management and climate on two stability components, being the resistance of the species composition and diameter class distribution against climate change and the resilience of the basal area of the forest under climate change. We hypothesized that increasing thinning intensity would increase the stability of the forest by increasing the available resources for the remaining trees (Chase et al., 2016), making trees more robust against droughts and other disturbances.

2 Literature study

2.1 Forests

Forests form an important part of the Earth's biosphere by occupying approximately 31% of the global land area. When looking at the global forest distribution, the tropical regions comprise the largest fraction of this total forest cover (45%), followed by boreal (27%), temperate (16%), and sub-tropical (11%) regions. One-fourth of this global forest area has been identified as primary forest, meaning it has never been disturbed by humans. On the other side of the spectrum, 3% of the total forest area are plantations, with nearly half of these plantations consisting of introduced species. This already gives a glimpse of the variety of forests in the world, ranging from forests that have been undisturbed by humans for extended periods up to intensively managed monofunctional forests that only serve for wood production. Furthermore, nearly half of the total global forest area is subjected to management plans, reflecting the high extent of human influence in general and the need to steer ecosystem services provisioning (FAO, 2020a).

When zooming in on Europe, 35% of its land area is covered by forests (Forest Europe, 2020). Six European countries, specifically Sweden, Finland, Spain, France, Norway, and Turkey, cover 70% of Europe's forest area. Central-Europe overall shows the largest forest productivity, in comparison to northern and southern Europe, where forest growth is limited by shorter growing seasons and lower water availability, respectively. About 95% of European forests are under some form of forest management with only 10% of this total forest cover under intensive management and 30% under multiple-use management. Additionally, forests are managed to achieve different goals ranging from intensive forestry with monocultures for bio-energy production to protected forests for conserving biodiversity. Forestry is quite an important economic activity in Europe as the majority of timber is exported, providing a large proportion of roundwood on the global market (European Environment Agency, 2016). In 2022, the total roundwood production amounted to 508 million m³ for Europe which is 12.8% of the worldwide production of roundwood (FAO, 2023).

In Belgium, forests cover 689,000 ha or 22.5% of the Belgian territory (FAO, 2020b), of which 140,279 ha are located in Flanders, a subregion of Belgium (Statistiek Vlaanderen, 2020). With a forest cover of 10% (Statistiek Vlaanderen, 2020), Flanders is one of the least forested

regions in Europe (Forest Europe, 2020). In Flanders, 38,060 ha of forests are under a management plan and 1602 ha are considered forest reserves (INBO, 2023). Over the last decades, forest management has become more sustainable and multifunctional in Flanders. This is reflected by observed changes in forest stand characteristics, such as a shift from coniferous forests to deciduous and mixed forests (Govaere, 2020). The share of species such as Scots pine (*Pinus sylvestris* L.), Corsican pine (*Pinus nigra* subsp. *laricio* (Poir.) Maire), and poplars (*Populus* spp.) decreased, while the share of most broadleaved species (except poplar and red oak (*Quercus rubra* L.)) increased. The mean standing volume per hectare expanded and the structural diversity increased. Furthermore, the amount of dead wood and old trees increased in the forest, enhancing the ecological importance of forests (Govaere, 2020).

There is no doubt that forests are essential to human society by providing us with a wide consortium of services, such as the provisioning of wood, regulating processes, and cultural functions. Additionally, forests will also play an important role in climate change mitigation (Winkel et al., 2022). The latter is reflected in the contribution of forests to the global carbon cycle, considering that over the last two decades, forests have been acting as a sink of greenhouse gasses (i.e. $-7.6 \pm 49 \text{ GtCO}_2\text{e yr}^{-1}$; Harris et al., 2021). Recently, however, our forests have been experiencing some major threats, affecting their carbon sink potentials. Specifically, during the period 1990 to 2020, 4% of the global forest cover was net lost (FAO, 2020a). This relates both to human and natural causes, with human causes mostly comprising urbanization, agricultural expansion, or timber harvesting leading to deforestation, while natural causes consist of droughts or storms which lead to unintentional forest loss (Keenan et al., 2015). For forests in Europe, the forested area actually expanded over the last decades. However, defoliation increased, indicating a decline in forest health (Forest Europe, 2020). Different factors such as habitat loss and degradation, invasive alien species, pollutants and nutrient overloads, and climate change are seen as prominent threats to European forests (European Environment Agency, 2016). This raises questions about how our forests will cope with a changing environment and evolve towards the future. Considering the high pressure that is placed on these ecosystems nowadays, we should focus on these questions as a decline in forest health can lead to the loss of ecosystem services. When this happens, the restoration of these services might take several decades (Trumbore et al., 2015).

2.2 Climate change

At present, climate change has become a prominent theme in many different aspects of our lives, impacting our societies and the environment where it has already caused a lot of damage. Human-induced greenhouse gas emissions are seen as the cause of the rise in global temperatures (IPCC, 2023a). These warmer conditions have already induced changes in different spheres of the Earth's system. Climate change has caused glaciers, ice sheets, and sea ice to melt (Garcia-Soto et al., 2021; IMBIE, 2018; Khan et al., 2015; Zemp et al., 2019). Moreover, the ocean has warmed up and acidified, and the sea level has risen (Garcia-Soto et al., 2021). A shift in weather patterns has emerged as shown by the intensification of climatic extremes, like heatwaves, droughts, and heavy precipitation events (Rummukainen, 2012). Globally, different kinds of ecosystems are affected by climate change and are put under stress, thereby losing their ability to deliver ecosystem services (Nolan et al., 2018). Furthermore, besides nature, society also experiences direct effects from climate change as it impacts economy, water, and food security, as well as physical and mental health (IPCC, 2023a).

Between the periods 1850-1900 (the reference period) and 2011-2020, temperature increased by 1.1 °C. This is partially explained by the rise in greenhouse gases over this period (IPCC, 2023a). The most important greenhouse gasses are CO₂ and CH₄, but also N₂O, tropospheric O₃, and halogenated gases can contribute to global warming, in contrast to anthropogenic aerosols which have a cooling effect on the planet (IPCC, 2023a; Wigley, 2001). These greenhouse gases originate mainly from the energy sector, the industry, and land use changes (IPCC, 2023a). It is expected that in the period 2081 to 2100 the temperature will have increased by a total of 1.4 or 4.4 °C compared to the reference period and depending on the socioeconomic global changes (IPCC, 2023b).

Reducing greenhouse gas emissions has the potential to dampen the temperature rise, even though temperature will still continue to increase in the following decades. Actually, if we would strongly and rapidly reduce the main greenhouse gasses (CO₂ and CH₄), the temperature increase could stay under the 1.5 or 2.0 °C threshold. This will require that efficient mitigation strategies are set in place. If not, further warming will happen in line with further greenhouse gas emissions, which can lead to irreversible changes (IPCC, 2023b). These different greenhouse gas emission scenarios are represented by the Representative

Concentration Pathways (RCPs). There are four primary scenarios, of which one represents a strong reduction in greenhouse gases (RCP 2.6), two represent intermediate situations (RCP 4.5 and RCP 6.0), and the last one represents a scenario where greenhouse gas emissions are very high (RCP 8.5). These different scenarios are useful to investigate their related impacts on the environment and society (IPCC, 2014). It should be noted that recently also new kinds of scenarios that are similar to the RCPs emerged, being emission scenarios based on the Shared Socio-economic Pathways (SSPs), but as we will use the RCPs, these scenarios are out of the scope of this thesis and will thus not be further explained (IPCC, 2023a).

As previously mentioned, droughts and heavy precipitation events have already become more intense and frequent over the last decades. Hot extremes like heatwaves increased in frequency and intensity in contrast to cold extremes which decreased in frequency and intensity (IPCC, 2023a; Rummukainen, 2012). Hot extremes will likely continue to manifest and even become more severe in the future as the Earth continues to warm up. Moreover, precipitation will become more erratic and intense in most regions. Droughts are also projected to increase in frequency and intensity (IPCC, 2023b). Specifically, for our study region, central Belgium, the precipitation is expected to decrease and increase in summer and winter, respectively. Meanwhile the potential evapotranspiration will increase in both seasons, as a result of the higher temperatures. In winter, the net rainfall resulting from this, i.e. the rainfall remaining after subtracting evapotranspiration, is still higher than in the historical records (1961-1990). In summer, these changes will push the situation towards even drier conditions (Baguis et al., 2010; Tabari et al., 2015).

2.3 Forests and climate change

Climate change will affect forests in different ways, both positively and negatively. Specifically, increasing temperatures and higher CO₂ levels in the atmosphere (climate change as such) could increase forest productivity, in contrast to disturbances which will have negative effects on forest growth and health (Lindner et al., 2010; Mina et al., 2022). There is some evidence that when looking at the combined effect of climate change as such and disturbances, both influencing productivity in an opposite way, the net result may be no net gain in productivity anymore or a further amplification of productivity loss in most cases. However, there were also some minor cases where climate change as such and disturbances together did have a positive influence on productivity. For example, low-impact fires can

reduce competition among trees and fertilize the soil, which promotes productivity. Hence, the combined effect of climate change as such and natural disturbances is context-specific (Reyer et al., 2017). Changing environmental circumstances are giving rise to changing forest dynamics. Hence, it is unclear how forests will behave in the future. As for now, climate change is leading to forest mortality and pushing forest stands to stay relatively young, hence, carbon storage is reduced. Nonetheless, these impacts may be partially buffered by adaptation, acclimation, and migration. At the other side, growth has increased by the fertilization effect of CO₂, although this effect only promotes growth in an early stage and when nutrients and water are not limited. Nevertheless, with the projected increase in mortality and variability in growth, forest canopy cover and biomass will most likely drop correspondingly (McDowell et al., 2020). In addition, Forzieri et al. (2022) showed that forests are losing their ability to withstand disturbances, which is likely linked to increasing incidents of drought and variability in climate.

Furthermore, disturbance regimes have been changing in Europe over the last decades, becoming more intense and increasingly affecting our forests (Patacca et al., 2023; Seidl et al., 2011). Wind caused the most damage to Europe's forests over the period 1950-2019 and is thus the most important disturbance agent, followed by fire and bark beetle. For the Sub-Atlantic zone, which comprises Belgium, fire and bark beetle disturbance have been observed to decrease and increase, respectively, with no clear trend observed for wind disturbance (Patacca et al., 2023). Forests have evolved to tolerate a natural disturbance regime (Seidl, Rammer, & Spies, 2014; Trumbore et al., 2015). A shift in disturbance regime may cause a mismatch between species traits, evolved in line with a certain history of disturbances (i.e. their information legacies), and disturbance characteristics. Besides information legacies, disturbances can create material legacies, such as nutrients, individuals, and seeds. These material legacies arise due to or persist from a perturbation, which make the transfer of species and their traits possible. Material legacies together with information legacies form what is called the ecological memory of a system. If this latter degrades due to changing disturbance regimes, resilience is reduced and the system might evolve to a new state (Johnstone et al., 2016). For example, we could think of certain tree species that can keep seeds stored for long periods of time, and only releases them after a fire event takes place (i.e. serotiny; Johnstone et al., 2016; Lamont & Enright, 2000). This is an information legacy

as this trait has evolved in line with a historical occurrence of fire events (Johnstone et al., 2016; Lamont et al., 2019). The material legacies are the seeds as they guarantee the survival of the species (Johnstone et al., 2016). In case fire frequency increases, regeneration will be hampered, and this system might evolve towards a deforested state (Buma et al., 2013).

In fact, forest composition and structure may undergo significant changes in response to an increasing disturbance frequency, in comparison with how it was under the historic disturbance regime (Seidl, Rammer, & Spies, 2014). Thom et al. (2022) found that climate-induced changes in disturbance regimes may actually have a stronger impact on forest dynamics than climate change itself. Moreover, legacy effects are important here as they can support the recovery to the pre-disturbance forest state, such as the persistence of tree patches after a fire event which can steer recovery to the pre-fire state (Seidl, Rammer, & Spies, 2014)

There is also a strong interconnection between climate, forests, and disturbances, causing different interactions and feedbacks to happen. Seidl et al. (2017) have shown that climate change influences forest disturbances such that drought, fires, and insect outbreaks are more likely to occur in a dry and warm climate, in contrast to pathogens and wind which will have a higher impact in a wet and warm climate. Interactions between disturbances most often increase their negative impact (e.g., drought increases the susceptibility of Norway spruce (*Picea abies* (L.) Karsten) to bark beetles (Temperli et al., 2013)), and these interactions may be altered by climate change (Seidl, Thom, et al., 2017). Moreover, interacting disturbances can have larger negative impacts on forests when climate changes. For example, the interaction between wind and bark beetle disturbances can affect larger forest areas in a scenario with elevated temperatures (Seidl & Rammer, 2017). Changing forest composition and structure, indirectly mediated by climate change and disturbance itself, may change the vulnerability of forests to different disturbances (Seidl, Thom, et al., 2017). In fact, disturbances may actually help forests to adapt faster to changing conditions. This relationship was found to be valid for disturbance severity and frequency, meaning that higher levels of these parameters gave rise to faster adaptation, but was not valid for disturbance size (the size of the affected area; Thom et al., 2017). Eventually, a change in forest composition and structure can also be effectuated by forest management as a way to strengthen forests against disturbances (Seidl et al., 2011, 2017). For the latter, we could for

example think of reducing the share of bark beetle-sensitive tree species, thereby reducing the vulnerability of the whole forest to bark beetle disturbances.

Campioli et al. (2012) investigated the impact of climate change on forests specifically for Belgium, as such making a more detailed assessment at the level of a country by considering regional characteristics. As expected, their conclusions align with observations on the (sub)continental scale. For Belgium, they stated that rising CO₂ concentrations and atmospheric nitrogen deposition would increase forest productivity, but other factors such as droughts, climatic extremes, and the impact of fungi and insects would arise as key threats for our Belgian forests. Specifically, European (*Fagus sylvatica* L.) and Norway spruce would suffer from climate change in Belgium as a result of increasing aridity (Campioli et al., 2012). Climate change effects were also observed on the individual tree level in Belgium based on ring-width chronologies by Latte et al. (2016). They found reduced growth for beech in past decades with changing soil conditions, i.e. increased nitrogen content and soil compaction, as the dominant driver until 1980 and climate change becoming dominant in the recent decades (Latte et al., 2016). Nevertheless, questions still remain regarding the negative as well as positive effects of climate change. It is unclear whether climate change affects each tree species equally and which tree species will thrive or perish under the changing climatic conditions.

Projections towards the future can take different directions, as is illustrated by the findings of McDowell et al. (2020) who stated that forest change and shifts in dynamics will accelerate further towards the future, in contrast to Thom et al. (2022) who found that current forest change as caused by climate change may attenuate in the second part of the 21st century by negative feedback mechanisms. Nevertheless, climate change and related effects impact forests and if we want to maintain and ensure the provisioning of ecosystem services to meet societal demands, forest management should aim to increase forests' capacity to withstand changes in climate (Temperli et al., 2012). We call this adaptive forest management Climate-Smart Forestry.

2.4 Climate-Smart Forestry

Climate-Smart Forestry (CSF) is a new way of managing forests in times of climatic changes that recently emerged and comes from the more mature concept of sustainable forest

management (Bowditch et al., 2020). A clear definition of CSF was given by Bowditch et al. (2020):

“Climate-Smart Forestry is sustainable adaptive forest management and governance to protect and enhance the potential of forests to adapt to, and mitigate climate change. The aim is to sustain ecosystem integrity and functions and to ensure the continuous delivery of ecosystem goods and services, while minimising the impact of climate-induced changes on mountain forests on well-being and nature’s contribution to people.”

Although its focus is set on mountain forests, this definition remains applicable to European forestry in general. Adaptation, mitigation, and the social dimension stand central and summarize the goals of CSF. In full, CSF aims at adapting forests to the future climate by enhancing their resilience and resistance to climate change, and at the same time maintaining mitigation capacity which translates into sequestration, storage, and substitution of carbon. The latter refers to the replacement of fossil fuels or carbon-intensive materials by forestry-derived products. The social dimension involves a wide range of factors such as the communication between stakeholders or the transfer of knowledge and technology (Bowditch et al., 2020).

Stability, resilience, resistance, and other related concepts are ambiguous concepts with different definitions in different fields. Within ecology, the adoption of these terms from different fields led to confusion about their exact meaning. Specifically, two fields of study developed different definitions for the same stability concepts, being the study of systems staying close to their equilibrium and systems behaving in a non-equilibrium way, i.e. systems with multiple stable states (Van Meerbeek et al., 2021). Different attempts to unify these diverging definitions arose from this, with the paper of Van Meerbeek et al. (2021) being one of the most recent contributions to this field, synthesizing previous work that led to an overarching framework for stability research in ecology. In the next part of this paragraph, we will elaborate on this framework. A system’s stability consists of different components (Figure 1): resistance (*“the ability to withstand change following a perturbation”*), resilience (*“the rate at which a system returns to the reference after change”*), recovery (*“the ability to fully return to the reference”*), latitude (*“the maximum amount a system variable can be changed before losing its ability to recover”*), and tolerance (*“the ability of a system to tolerate*

perturbations”; Van Meerbeek et al., 2021). Resilience as defined here is also known as engineering resilience in the literature (Nikinmaa et al., 2020). Furthermore, system variables are defined as parts of an ecological system such as the biomass or vegetation cover, while system parameters are for example precipitation and temperature, which are able to affect the system variables (Justus, 2007 in Van Meerbeek et al., 2021). Within this framework, perturbations are seen as processes, forces, or agents able to alter the system’s variables and/or parameters (Van Meerbeek et al., 2021). They can be characterized as being a pulse, press, or ramp perturbation, meaning the perturbation happens in a limited time space, is continuous, or is gradually increasing in time, respectively. For example, the rising temperature can be identified as a ramp perturbation, while pulse or press perturbations could be a drought or a continuous low groundwater level, respectively (Donohue et al., 2016; Van Meerbeek et al., 2021).

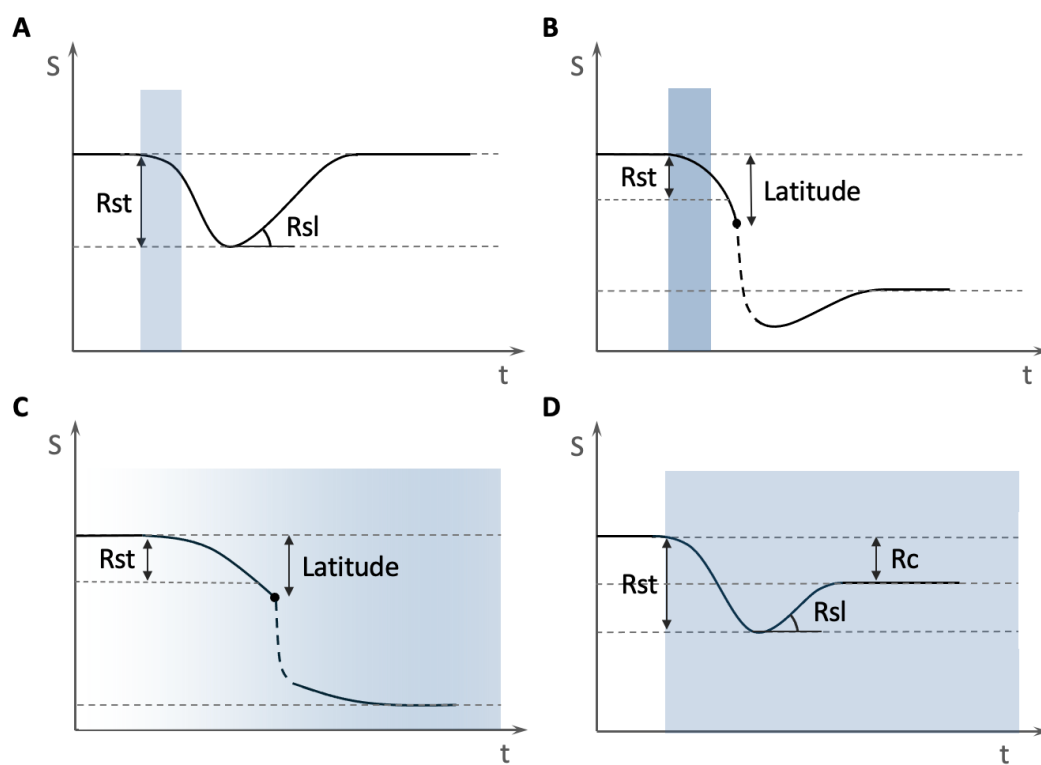


Figure 1: Figure based on Van Meerbeek et al. (2021) showing the system variable (S) through time (t) in different disturbance scenarios (blue zone), with A) a pulse perturbation smaller than the system’s tolerance, B) a pulse perturbation larger than the system’s tolerance, C) a ramp perturbation, and D) a press perturbation. R_{st} , R_{sl} , and R_c stands for resistance, resilience, and recovery, respectively.

Coming back to CSF, we can see that stability concepts are important in this management strategy to adapt forests to climate change. By adapting the forest, we aim to make the forest more resistant and resilient, meaning that we want to increase the forest's capacity to withstand perturbations and to recover fast in case the system deviates from its reference condition (Van Meerbeek et al., 2021). Seidl (2014) shows that given the uncertainty of the risk, i.e. climate change, focus should be set on fostering resilience because the predictability of the risk and knowledge to handle it is low. If we want to maintain ecosystem provisioning and mitigation by forests, we should make sure that forests can tolerate perturbations and return to their initial state or another favourable state when disturbed. With another favourable state we mean a state where the forest is still able to deliver the desired ecosystem services.

The importance of CSF to achieve mitigation goals, has been acknowledged by Verkerk et al. (2020), seeing CSF as “the missing link” on the way to get in accordance with the objectives as stated in the Paris Agreement. If CSF would be incorporated in Europe's forests, it would have a carbon mitigation potential of 441 Mt CO₂. year⁻¹ by 2050 in addition to what is currently captured by the forests of Europe in terms of CO₂, which amounted up to 569 Mt CO₂. year⁻¹ in the previous decade. The latter equals 13% of greenhouse gas emissions in Europe (Nabuurs et al., 2017). This illustrates that CSF has a large potential for contributing to the mitigation of climate change.

Thinning is a silvicultural intervention frequently applied in forests. It reduces the density of a stand to give the remaining trees more space and consequently, improve their growth (Gonçalves, 2021). Multiple parameters are important when executing a thinning, which are the thinning frequency (the time between two consecutive thinnings), the age of the stand at the first thinning, the thinning intensity (the number of trees or volume that is removed from the stand), and the type of thinning (Klingen et al., 2010). There are multiple types of thinning, with the most important ones being a thinning from below, a thinning from above, and a future tree thinning. A thinning from below will mainly focus on removing suppressed (usually smaller) trees in the stand, whereas a thinning from above will intervene in the crown layer by removing dominant or co-dominant trees. The management goals will determine the trees that are kept in the stand (e.g. better form, nature values, etc.). Future tree thinning can be

seen as a form of thinning from above, where the future trees to be kept are already chosen in an early stage of the rotation period (Klingen et al., 2010).

Thinning is seen as an important activity to directly react to climate change. More intensive thinnings have been reported as a way to cope with climate change, as it would increase stand stability and health, and reduce the competition for water (Keskitalo & Carina, 2011). In general, thinnings can alleviate the competition for resources, such as light, nutrients and soil moisture, amongst trees, which can therefore reduce tree stress. Trees that experience stress are more prone to insect infestations, diseases, and mortality (Chase et al., 2016). Furthermore, thinning can promote warmer soil temperatures and higher soil moisture (Chase et al., 2016; Ma et al., 2010), which can improve N mineralization and, thus, N availability in the soil (Chase et al., 2016; Powers, 1990). The content of available phosphorus and available potassium was also shown to be higher in thinned stands compared to unthinned stands (Xiangrong et al., 2018). Moreover, a meta-analysis by Sohn, Saha, et al. (2016) showed that thinning is an important silvicultural practice that can reduce the impact of drought on the growth of trees in a stand. This effect was valid for both conifers and broadleaved trees, but in different ways. For conifers this effect translated mainly into a better recovery to initial growth levels after drought, while for broadleaved trees the growth reduction was less during drought. The latter was explained by an increased soil water availability. Additionally, heavier thinnings could achieve these effects better. Following Bréda et al. (1995), this increased soil water availability is linked to a decreased transpiration and interception after thinning. These benefits of thinning are apparent for some conventional species of Europe. For Scots pine, trees were found to be more resistant during drought and recovered better relative to the growth reduction after drought in thinned stands compared to unthinned stands. The recovery in relation to the growth reduction was higher in more heavily thinned stands. Nevertheless, this effect diminished with time after the last thinning (Sohn, Hartig, et al., 2016). For beech, a very common species with low drought tolerance, thinning reduced growth reduction due to drought and increased the ability of the remaining trees to return to pre-disturbance growth levels after drought. Higher thinning intensity resulted in an increased drought tolerance (Diaconu et al., 2017). Moreover, sessile oak (*Quercus petraea* (Matt.) Liebl.) also experienced increased resistance to droughts following a thinning event. Thinned trees showed an increased ability to recover with

reference to the minimum growth level and to reach pre-disturbance growth levels (Schmitt et al., 2020). The positive effects of thinning have also been illustrated by other studies (Gradel et al., 2017; Manrique-Alba et al., 2022; Misson et al., 2003), however, more studies investigated conifer species than broadleaved species.

However, one could argue that more heavily thinned stands could also be less stable considering an altered microclimate due to thinning. Rambo & North (2009) found that in overstory-thinned stands the microclimate was more extreme. Summer and winter temperature as well as vapor pressure deficit (VPD) showed larger fluctuations during day- and nighttime in thinned stands compared to unthinned stands, especially closer to the forest floor. This was explained by more solar radiation entering the thinned stand during day, which led to higher maximum temperatures and VPD. During night, more longwave radiation was lost to the atmosphere and resulted in lower minimum temperatures and VPD (Rambo & North, 2009). Nonetheless, the effect of this altered microclimate on the trees itself remains uninvestigated, as most often research looks at the global effect of thinning on the trees, such as the studies mentioned in the previous paragraph. Possibly, the positive effects of thinning, i.e. the increased availability of resources, are higher and compensate for the negative effects, i.e. the changed microclimate. Secondly, as thinning can increase the openness of the stand, wind disturbances could also become more problematic. Specifically, in the short-term, the stand stability against wind is reduced after a thinning due to increased canopy roughness and the reduction in neighbouring trees which provided shelter (Jiao-Jun et al., 2003). Nevertheless, in the long-term, stand stability against wind is often enhanced by thinning, especially when executed in an early phase of the stand, due to effects on growth and adaptation to wind (Jiao-Jun et al., 2003; Picchio et al., 2020).

In Europe, most forest stakeholders are aware of the effect of climate change on their forests. Despite this, only half of them are convinced that adapting forest management is necessary to cope with climate change and only a third have already taken action to adapt their management, yet with differences in percentages between countries. This shows that management is adapting slowly, whereby the deficiency of knowledge both in practice and theory is identified as the main obstacle allowing for action to take place (Sousa-Silva et al., 2018). In our study region, Belgium, forest stakeholders have already experienced climate change in the form of heatwaves, droughts, and heavy wind and precipitation events.

Nevertheless, only a third of the stakeholders have altered their management. If adaptive measures were set into place, it mostly consisted of promoting mixed stands and planting better-adapted species and varieties. However, a lot of other measures were also reported, including the altering of thinning intensity. In Belgium, knowledge was also seen as the limiting factor to allow for forest adaptation (Sousa-Silva et al., 2016).

Different factors can hamper the achievement and preservation of the sustainable supply of ecosystem services by forests. These factors include not only climate change, but also land use change, management, economy, socio-culture, and governance. In the case of management, a conservative approach with a lack of adaptive measures is seen as a main obstacle to be able to adapt and increase resilience of forests to climate change, and, in this way, ensure the sustainable provisioning of ecosystem services. The main solution is to switch from this traditional management to climate-smart management as a novel way to deal with a changing world (Hernández-Morcillo et al., 2022). Nevertheless, it is clear that for this to happen, knowledge should increase about how to implement CSF. In this case, forest modelling can help to get a better mechanistic understanding of the interactions between management and forest dynamics, and aid in developing CSF strategies.

2.5 Forest modelling

Experimental setups are often inadequate to study the effects of processes such as climate change, disturbances, and forest management on large spatiotemporal scales. Here, forest modelling emerges as a useful instrument to make this possible (He et al., 2011; Shifley et al., 2006). The three most important types of forest models are empirical, mechanistic or process-based, and hybrid models. The first models that arose somewhere in the thirties, were empirical models such as yield tables and regression models to predict future timber yield. From here on, further development and progress resulted in other kinds of models. This development, where model complexity and performance increased, happened in line with improvements in different technological and practical aspects, which were the computing capacity of computers, the availability of data to calibrate and validate the model, the model's applicability in practice, and the capacity to contain social, biological, and economical agents to steer change in the model. As these limitations became less critical over the years, new kinds of models emerged such as the process-based models and later the hybrid models (Shifley et al., 2017).

Empirical models are models based on regression equations derived from large field inventory datasets to represent forest growth (Shifley et al., 2017; Twery & Weiskittel, 2013). The most widely adopted empirical models are yield tables, where site index and stand age are used to predict forest growth (Twery & Weiskittel, 2013). Empirical models are often more practical to use for forest management (e.g. yield tables; Landsberg, 2003; Twery & Weiskittel, 2013), but they have some significant downsides as well. First, the data that underlies the model should be of sufficient quality, as this will also determine the quality of the model. This data should comprise a broad range of observations to include the internal variability and extremes within these observations (Twery & Weiskittel, 2013). Additionally, their spatial applicability is limited as they are generally specific for a certain site and the spatial scale on which these models are used, is most often the forest stand. Another limitation of these models is that they are not able to make predictions when new conditions arise, as they are only valid for the conditions in which the data that underlies the model was collected. In other words, these models consider the climate to be stable and, therefore, are often used on smaller timescales (Landsberg, 2003; Twery & Weiskittel, 2013). Disturbances or management activities, such as thinning and fertilisation, which can impact growth, are not accounted for in these models. Additionally, they are both limited in input as output, as they are not able to use data sources such as remote sensing data to steer the model and usually also offer limited outputs (Landsberg, 2003). Lastly, site index, a parameter often used in empirical models, relies on assumptions which are not always valid (Skovsgaard & Vanclay, 2008).

Mechanistic or process-based models, on the contrary, rely on equations representing the underlying physiological processes steering forest growth such as photosynthesis and respiration (Landsberg, 2003; Twery & Weiskittel, 2013). In other words, they try to describe these processes, including the interaction with factors that influence them (e.g. water supply, CO₂ level, etc.), in mathematical equations to predict the outcome rather than measuring the outcome from which then statistical equations are derived at a higher level such as in empirical models (Landsberg, 2003; Landsberg & Sands, 2011a). Mechanistic models can overcome most of the disadvantages of the empirical models, but they are less commonly used in forest management planning because of their complexity. Specifically, they are more generic as they allow for simulating forest growth under changing conditions (e.g. climate change). Additionally, effects of different silvicultural measures or disturbances on forest

growth can often be assessed (Landsberg, 2003). At the other side, they often need a lot of parameters to be defined, for which the numerical value is not always known (Landsberg, 2003; Twery & Weiskittel, 2013). Important parameters to define are usually those which set species apart from each other in the model and are called species-specific parameters. In general, the values of the parameters will influence the output, although these influences are usually subtle. Nevertheless, this is not always the case. These parameters are usually found by the process of calibration or parametrisation. Calibration is the process of assigning values to parameters which were found by doing experiments, whereas parametrisation is the process of varying the values of the parameters in the model until model output and observed output matches best (Landsberg & Sands, 2011b).

Models are always simplifications of reality as ecosystems are very complex. Nevertheless, if we want to produce accurate predictions for these ecosystems, we should describe the underlying processes in enough detail in the model to approximate these. At the other side, too much detail is unwanted because then the model would become too complex and require too many parameters to be specified. Additionally, models will always contain uncertainties, which are linked to the description of the processes, the parameter values, and the assumptions made (Landsberg & Sands, 2011a). Specifically, error propagation should be considered when using models, where errors in the parameters or input data of the model will lead to errors in the model output (Crosetto & Tarantola, 2001), which can become more severe when simulating over longer time frames (Albrich, Rammer, Turner, et al., 2020).

There also exist different kinds of mechanical models such as gap (stand-scale) models and forest landscape models (FLMs; landscape-scale). Gap models model establishment, growth, and mortality of individual trees on a small area usually around 0.01 to 0.1 ha. They were initially not spatially explicit, however with time also spatially explicit gap models emerged able to model seed dispersal and recruitment (Perry & Millington, 2008). FLMs arose after the gap models as a consequence of the development of better technologies. In contrast to gap models, they can model entire forest landscapes instead of only the forest stand and, hence, also simulate the interaction between forest stands (Xi et al., 2009). FLMs are spatially explicit models which means that they can model spatially explicit processes such as seed dispersal, wind and fire disturbances, pathogen and insect spreading, and silvicultural interventions (He et al., 2011; Shifley et al., 2017). The incorporation of these processes into a forest model,

opened new possibilities for research (Shifley et al., 2017). Additionally, they have the advantage that they can use remote sensing and other spatial data as input data (Xi et al., 2009). An example of a process-based forest landscape model is iLand, which can model the resource competition between spatially explicit individual trees based on physiological processes (Seidl, Rammer, et al., 2012). iLand is discussed in more detail in the methodology section.

Finally, hybrid models combine both empirical and mechanistic components to combine the benefits of both model types (Landsberg, 2003; Twery & Weiskittel, 2013). In other words, they combine empirical equations to predict growth together with mathematical representations of the main physiological processes (Weiskittel et al., 2010). Their mechanistic side enables them to make projections for new situations, while their empirical side makes them more robust overall (Twery & Weiskittel, 2013). Hybrid models also often require less parameters to be defined in comparison to process-based models (Weiskittel et al., 2010). An example of a hybrid model is FORECAST, which can use yield as predicted by historical regression equations (the empirical side) and, consequently, this yield is then altered by simulating the light competition or the availability of nutrients through time (the mechanistic side; Kimmins et al., 1999).

As maintaining forest stability becomes a more important target of forest management with the recently changing climate and environment, models arise as an important research tool (Keane et al., 2018). Models have already been used to study the resilience or overall stability of forests against climate change and disturbances (e.g. Albrich et al., 2020; Cantarello et al., 2017), and in addition to this, the role of forest management herein has also been investigated. For example, Duveneck & Scheller (2016) used the LANDIS-II model to look at the interplay between climate change, forest stability, and management in the northern Great Lake forests of the USA. Forest resistance and resilience dropped in high emission scenarios when management was not considered. When management was accounted for, especially the planting of climate suitable species resulted in higher resilience after fire events. Another study also investigated the effect of different management strategies on the reduction of disturbances in Norway spruce dominated forests in Central-Europe with the forest model iLand (Dobor et al., 2020). They concluded that management promoting species diversity by planting site-adapted tree species led to less disturbances in contrast to management that

promoted spruce. Nevertheless, studies looking at the effect of management on the stability of forest landscapes against climate change remain scarce.

3 Methodology

3.1 Study site

Abiotic and biotic properties

The Meerdaal and Heverlee forests are situated in the south of Leuven, Belgium (N50.80°, E4.67°). The forest complex is an old forest making it highly biodiverse and ecologically valuable. The topography is relatively flat, with a maximal elevational difference of approximately 70 m. The soil is characterized by the presence of tertiary sandy layers which were covered with aeolian silt during the last ice age. This gave rise to a silt layer of varying thickness over the landscape, with locally exposed hill tops and slopes where the silt layer eroded away. Therefore, the soil texture can mostly be classified as silt in Meerdaal forest. In Heverlee forest most of the aeolian silt eroded away or has become thin, causing the soil texture to be mostly sandy silt (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

The forest complex is divided into 120 forest parcels (Dutch: “bospercelen”) which are subdivided into forest stands, resulting in 644 forest stands over the entire forest complex. A large part of the forest consists of only broadleaved species, with only a minor part containing only conifers and a third containing a mixture of broadleaved and conifer tree species. The presence of homogenous stands consisting of only one tree species is limited in the forest complex, with Meerdaal forest containing less homogenous stands than Heverlee forest. Most forest stands also have a large range in age classes. Analysing the distribution in age classes has shown that Heverlee forest had been almost fully deforested in World War II, together with the northeast part of Meerdaal forest, and afterwards reforested again (mostly with conifers). Consequently, Meerdaal forest consists of older stands than Heverlee forest. The forests are also characterized by a high canopy closure. Based on percentual share in basal area the largest part of Heverlee forest (excluding avenues and forest reserves) consists of red oak, followed by European beech, Scots pine, and larch (*Larix* spp.). In the same way, Meerdaal forest consists of mainly European beech, followed by Scots pine, native oak (pedunculate (*Quercus robur* L.) and sessile oak), and Corsican pine. The forest reserves in the Heverlee forest are mainly constituted of red oak followed by Scots pine and European beech, in contrast to the forest reserves of Meerdaal forest where European beech followed by sycamore maple and native oak are more common. The large share of red oak in Heverlee

forest is a consequence of the avenues which have been planted with red oak in the past, giving this species eventually the ability to spread further over the whole forest (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

Management

In the early 1600s the forests came under the ownership of the Duke of Arenberg, after they have been the property of several other noble families. From this moment on, the forests have been subjected to an active management whereby forest plunder was prohibited and hunting together with wood production became the main activities in the forests. Ever since, the forests were managed as coppice and coppice with standards producing mainly firewood, small wood products, and bigger wood products such as beams or planks. With the introduction of coal as main energy source for heating in households, the management transitioned to a high forest management at the end of the nineteenth century. Hereby, several parts of Meerdaal forest were planted with European beech. In this period also large deforestations occurred as a cause of the expanding agricultural land. After World War I the forests came under the confiscation of the Belgian government and in 1934 the first management plan was introduced, giving rise to the use of a Swiss group system to regenerate the forests. Thinnings favoured mainly European beech and oak. During World War I and II, large clearcuts took place, especially in Heverlee forest causing this forest to be nearly fully deforested after which it was replanted again with mainly conifers. From the 1970s and 80s, various foresters got to manage these forests and new insights in forest dynamics and management emerged, as well as other perspectives about the functions of a forest. Important additions to the management plan were the use of secondary species for maintaining site quality but also for wood production, the growing focus on dead wood in the forest, the enlargement of regeneration groups, the growing interest in the recreational function of the forests, the foundations of the future forest reserves, and the establishment of a connection between Meerdaal and Heverlee forest. New possibilities were given by natural regeneration as the forest had become older, containing mature trees which were able to produce seeds. Eventually, the current management plan (2006 – 2026) was born (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

Nowadays the forests are managed for multifunctionality and sustainability. This is achieved by applying the concept of zoning which aims at realizing a balance between the recreational, ecological, and economical function of the forests. The management aims at being complementary to the natural processes, e.g., by allowing for regeneration, aging, and succession to happen and consequently steering management based on these processes. Additional to these three main functions, the forests also fulfil two more, that is, protecting the environment by e.g., controlling erosion, and by serving as a study site for scientific research. The forest reserves occupy 240 ha or 12% of the forest complex and are different from the main part of the forests as here, the forests are mostly not managed and can evolve freely, and this allows for specific forest types to emerge or to be maintained. Some parts of the forest reserves are managed to get specific forest structures, species, or communities or to try different silvicultural systems for nature conservation. Usually, the forest reserves have interesting biotic and abiotic characteristics and are often also characterized by larger volumes of fallen and standing dead wood (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

Current targets in the management plan aim at increasing the ecological potential of Heverlee forest as this forest was mainly used for wood production and recreation in the past. In the case of Meerdaal forest multifunctionality is already nearly at its optimum (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006). In what follows next, we will illustrate the active management currently practiced in the forest complex as well as the main goals, which will form the basis for how we programmed the management in iLand, as explained in 3.3. Landscape, climate and management setup.

Following the forest management plan, most of the forest stands will be managed as a high forest to harvest wood, with only a minor part that will not be used for wood production. Future tree thinning is and will be the default thinning method used. The timeframe to perform a thinning is every 16 years (full rotation), but usually this happens every 8 years (half rotation) with even the possibility to lower it further to every 4 years (quarter rotation) if necessary. The forest complex will be expanded by the afforestation of adjacent land, targeting 153 ha of new additional forest in the future (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

In the scope of forest regeneration, one important consideration in the management plan is the lowering of the share of European beech in the forest. Not only is it important to have a wide variety of species to be able to also satisfy the wood market in the future, it is also relevant from an ecological point of view as European beech acidifies the soil and will potentially suffer under climatic changes because of its sensitivity to dry summers (Langenbruch et al., 2012; Leuschner, 2020). The share of oak and other native tree species will therefore be increased. There will mainly be chosen for sessile oak rather than pedunculate oak because of its higher tolerance to droughts which will most likely accompany climate change (Ducousso & Bordacs, 2004; Lévy et al., 1992). The management plan foresees only a small reduction in the share of Corsican pine, Scots pine, Douglas fir (*Pseudotsuga menziesii* (Mirbel) Franco) and larch in terms of regeneration due to economic reasons. The share of sweet chestnut will be maintained constant, and finally, a reduction of red oak will be effectuated. The regeneration is done primarily by using a group system with groups of 20 to 50 are. Natural regeneration is preferably used. Only if the latter is not possible artificial regeneration is used (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

Through forest conversion, mainly by decreasing the share of red oak and exotic conifers, the share of native and suitable species will be raised. Secondly, the management will aim at mixing native broadleaved species within homogenous exotic conifer and Scots pine stands. As European beech acidifies and degrades the soil (Langenbruch et al., 2012), some beech stands will also be converted, with the aim to realize an oak - hornbeam (*Carpinus betulus* L.) forest on some stands. Red oak will also be converted as this tree species has also the potential to degrade the soil and darken the forest (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006; Dyderski et al., 2020).

3.2 Model description

We used the process-based forest landscape model iLand, the individual-based forest landscape and disturbance model, to study the effect of disturbances and changing climate on forest dynamics. iLand considers individual trees as the primary entity which are modelled explicitly (Seidl, Rammer, et al., 2012). The competition between trees is founded on the ecological field theory (Walker et al., 1989; Wu et al., 1985). Each tree can, because of its height and crown shape, influence the light availability into its direct vicinity. Over the forest landscape these influences of the individual trees are combined into a continuous field of

interference, which will determine the light availability for each tree. Based on the light use efficiency approach of Landsberg & Waring (1997), the gross primary productivity is derived monthly on a 100 x 100 m grid from the intercepted radiation and taking into account the available soil water, air temperature, vapor pressure deficit, available nitrogen, and atmospheric CO₂ concentrations. Subsequently, the net primary production is computed as being a fraction of the gross primary production. Aging will reduce the net primary production based on age and height of the tree. The formed carbohydrates are allocated hierarchically to the different tree compartments based on allometric equations (Landsberg & Waring, 1997). The strategy of trees to invest more biomass into height or diameter increment based on its competitive status relative to other trees is also included and based on the approach of Bossel (1996). Seed dispersal is accounted for by the simulation of probabilistic dispersal kernels around mature trees and calculated for a 20 x 20 m grid. Next, establishment of seeds is influenced by environmental conditions (temperature, nutrients, and water) and the availability of light. Saplings are represented by a mean tree approach on a 2 x 2 m grid and only after reaching a height of 4 m are modelled as individual trees in iLand (Seidl, Spies, et al., 2012). Finally, mortality is divided into an inherent mortality and a stress-related mortality, based on the maximum attainable age and carbon starvation, respectively (Seidl, Rammer, et al., 2012). For more details about the model approach of iLand we refer to Seidl, Rammer, et al. (2012).

iLand can account for three different disturbances, being wind, fire, and bark beetle disturbances. Time series of wind events can be provided to the model and wind speed over the landscape is adjusted based on topography. The forest landscape is susceptible for wind disturbances where a difference of 10 m between the forest height of neighbouring grid cells is present. Additionally, during a storm event, the landscape is continuously updated, which means that newly formed edges because of a previous wind disturbance are considered in the next wind disturbance during the same storm. The shelter effect from neighbouring trees as well as upwind gap size is taken into account. Depending on the wind speed, trees can either be broken or uprooted (Seidl, Rammer, & Blennow, 2014). Secondly, bark beetles are able to infest Norway spruce trees in iLand. The model includes the development of the bark beetles, their dispersal and colonization of potential host trees, the ability of the tree to defend itself and partial mortality of the beetles during winter (Seidl & Rammer, 2017). Fire

is simulated explicitly and represented by different phases, being fire ignition, spread and extinction (Seidl, Rammer, & Spies, 2014). Lastly, drought is not included as an explicit disturbance in the model but is simulated through calculating the soil moisture content out of a daily water balance and then, relating this to the minimum soil water potential given for each species to model water stress (Seidl, Rammer, et al., 2012). Species-specific minimum soil water potentials were derived from the drought tolerance values from Niinemets & Valladares (2006).

To enable the implementation of forest management into the virtual forest landscape of iLand, an agent-based model for forest management (ABE) was coupled to it by Rammer & Seidl (2015) and has, to our knowledge, only been used in studies focusing on forests in Central-Europe (e.g., Albrich et al., 2018; Dollinger et al., 2023; Thom et al., 2018). The ABE module requires that the forest landscape is divided into management units, which are further divided into stand units. An agent, i.e. the forest manager, manages one management unit and is able to make operational decisions on the stand level as well as strategic decisions on the management unit level. A strategic decision, which is related to the key parameters of a silvicultural program, is for example the choice of the rotation age, while an operational decision could be whether to harvest a stand or not, because it might have reached the given rotation age. Moreover, the management strategy for a management unit will work out differently for each stand comprised in this management unit, as the initial conditions of the stands are different. The strategy can influence the operational decisions as for example a different rotation age will change the timing of the final harvests of the stands. In the model there is a two-way communication between the agent and the ecosystem. The agent is supplied with information about the environment and, conversely, management decisions taken by the agent are imposed on the landscape. Consequently, in case conditions change in the environment, the agent is able to adapt its behaviour. When stand conditions change, the agent might change its operational management. Additionally, changes in ecosystem conditions might also change the management strategy (i.e. the key parameters) of the agent (Rammer & Seidl, 2015). For thinning specifically, different parameters can be specified, such as the thinning frequency, the thinning intensity and the target variable on which the thinning applies (e.g. basal area), The percentage to remove from each diameter class, thinning

constraints, etc. (Rammer & Seidl, 2015; see <https://iland-model.org/startpage> for more practical details).

3.3 Landscape, climate and management setup

The landscape of Meerdaal and Heverlee forest was already initialized by Storms et al. (2023). They initialized soil conditions, being the clay, silt, and sand content as well as the chemical properties, based on data from the Belgian soil map, soil profile database and Forsite database (De Vos, 2009; Van Orshoven et al., 1988). Information about forest characteristics was derived from a forest inventory and the forest management plan. Additionally, satellite and LiDAR data were used to derive clear-cut areas as well as forest gaps and individual tree positions, respectively. We refer to their paper for more details.

The climate data we used were the same datasets as created and used by Storms et al. (2023). The historical climate data for the period 1985 to 2019 was retrieved from a weather station in Beauvechain (N50.76°, E4.76°) from the Royal Meteorological Institute (Storms et al., 2023). To extend this historical climate until 2095, we randomly sampled observations from this dataset. Possible future climate scenarios were then represented by six scenarios in RCP 4.5 and seven in RCP 8.5, all ranging from 2006 until 2100. These were formed based on regional and global climate models from EURO-CORDEX with a 12.5 km resolution and, additionally, they were bias corrected using a N-dimensional probability density function by Storms et al. (2023) (Cannon, 2018; Dalelane et al., 2018; Jacob et al., 2014). We used these scenarios from the year 2020 until 2095 without random sampling. Later, these RCP scenarios were divided into three main climatic pathways based on their average maximal temperature and precipitation sum in the period June, July, and August in 2081-2100, following the approach of Storms et al. (2023). These climatic pathways were a moderate climate (max. temperature < 24°C and precipitation > 150 mm), a hot climate (max. temperature > 24°C and precipitation > 150 mm), and a hot and dry climate (max. temperature > 24°C and precipitation < 150 mm; Figure 2).

Finally, a dataset with damaging historical wind events from Storms et al. (2023) was utilized to construct future wind events by random sampling, as no clear upward trend is observed for wind events in the Sub-Atlantic zone, which contains Belgium (Patacca et al., 2023). Wind events in the former were seen as damaging if the average wind speed was higher than 10

m.s⁻¹. Between replicates wind events were distinct because of a different random sampling. The rise in atmospheric CO₂ concentrations towards the future was accounted for by providing the model with CO₂ concentrations time series from 2020 until 2095 for both RCP 4.5 and RCP 8.5 scenarios. Wind and bark beetle disturbances were enabled in the model.

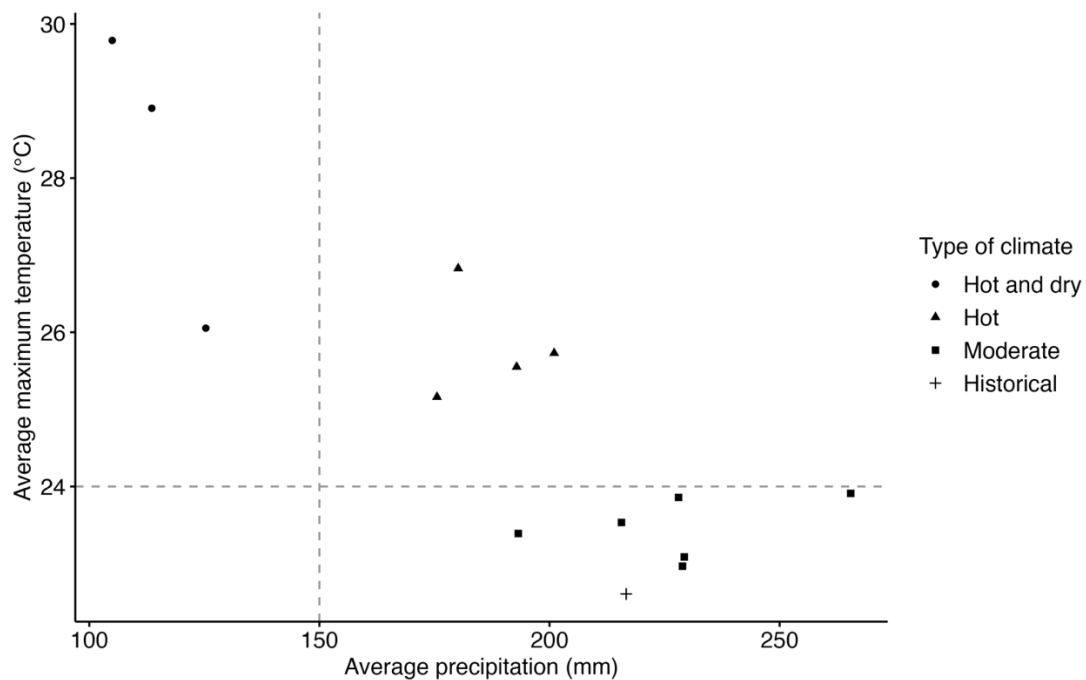


Figure 2: Delineation of the different climate scenarios based on the average maximum temperature (°C) and the average precipitation sum (mm) in June, July, and August in the period 2081-2100. The historical climate is the average of the temperature and precipitation in June, July, and August in the period 1985-2019.

For the implementation of the ABE module in practice, we divided the Meerdaal and Heverlee forest into management units, consisting of the forest stands that were already initialized in iLand. We did this based on the division in forest parts (Dutch: *bosplaatsen*) as stated in the management plan, as for each forest part clear management goals were defined (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006). We came up with four different management units, being the forest reserves, the arboretum, the connection area, and the main forest (Figure 3). These forest parts have different management goals; The forest reserves are managed for their ecological and scientific value, the arboretum for its scientific, educational, and recreational value, and the connection area for its ecological and recreational value. Lastly, the main forest distinguishes itself from the previous forest parts as here the economic

value becomes more important, besides the ecological and recreational value it also has (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006). The delineation of the management units was done with R (R Core Team, 2022), using the packages *sf* (Pebesma, 2018) and *raster* (Hijmans, 2023a).

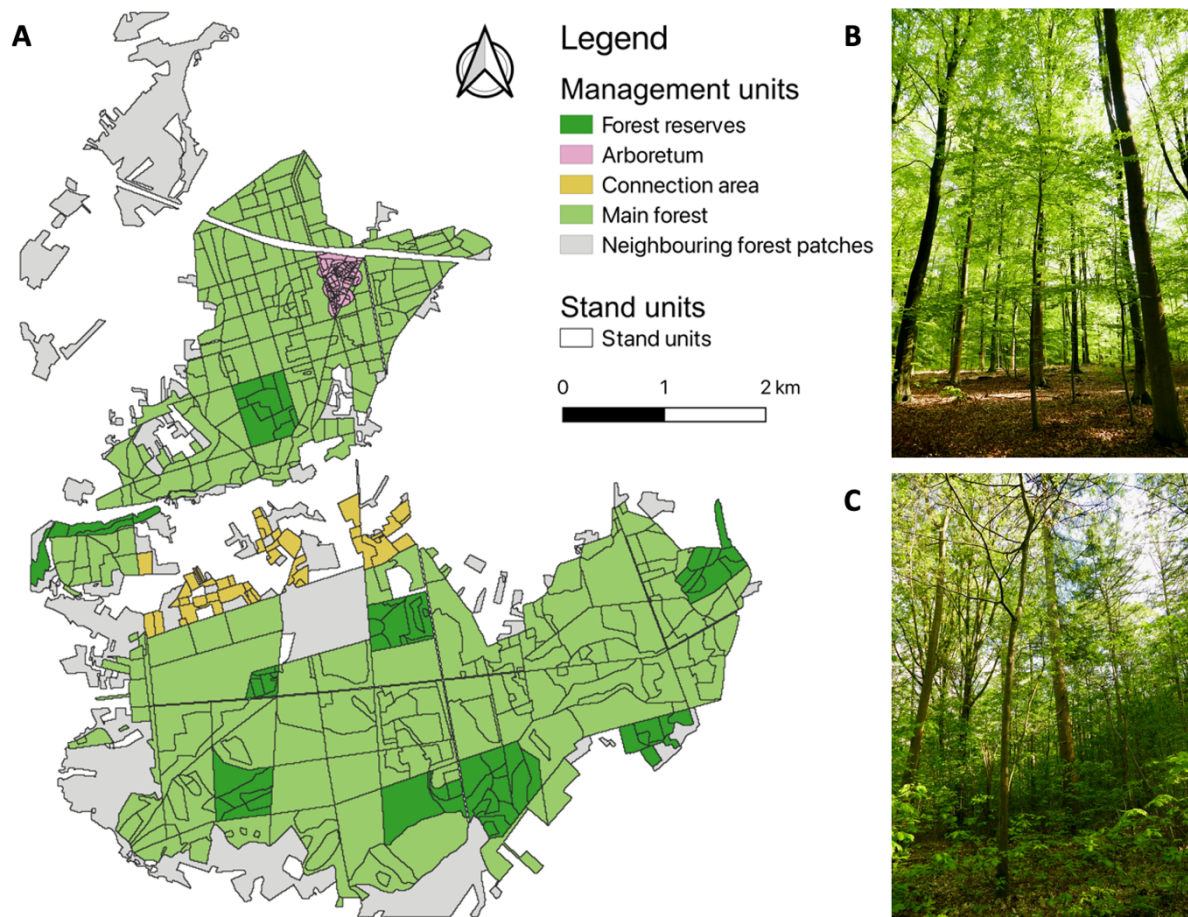


Figure 3: A) Division of the forest landscape into management units, which are further divided into stand units or forest stands, B) a managed stand in the forest complex, C) an unmanaged stand in the forest complex.

Next, as each management unit was set under management of an agent, we needed to define a stand treatment program (STP) for each agent. In ABE, a STP consists of a set of silvicultural activities, usually regeneration, thinning and harvesting activities, that will be executed by the agent in the management unit at the prescribed timing (Rammer & Seidl, 2015). We based ourselves on the management plan to construct these STPs (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006). For the forest reserves, management was mostly focused on the goal to allow for spontaneous forest dynamics to take place, thus the STP consisted of doing

nothing, i.e. zero management. Nevertheless, some parts of the forest reserves were managed actively for nature conservation, but as this concerned a minor part of some forest reserves, we neglected this and put the forest reserves as a whole under the zero management. For the arboretum and the connection area we used the same STP as for the forest reserves. For the arboretum this was decided based on its values as mentioned earlier, as well as for the connection area. Additionally, the management plan also envisaged a spontaneous development of the native species in the connection area, which supported our decision. It should be noted that these three management units could have equally been taken together to form one management unit as they have the same STP. Conversely, for the main forest, which agreed to the largest part of the forest landscape, we designed a STP consisting of the main silvicultural activities done in the forest complex. These were thinning, forest conversion, harvest, and regeneration. In the next paragraph we will go deeper into how these activities were implemented in the STP.

We based ourselves primarily on the management plan to define the activities in the main forest (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006). Thinnings were enforced on the landscape every 8 simulation years (thinning frequency) and removed a certain percentage of the basal area, which determined the thinning intensity (see 3.4 Simulation design for used thinning intensities). Most trees were thinned in the upper diameter classes, which approximated a thinning from above, as simulating a future tree thinning was too complex. No trees were cut in the lowest diameter class. An exception was made for homogenous stands consisting of red oak. These stands were not thinned to hamper regeneration of red oak (see forest conversion) as was suggested by the forest managers. A stand was seen as homogenous if the dominant tree species took up more than 80% of the total basal area.

Forest conversion was applied to lower the share of conifers and red oak in the forest complex, as well as make conifer stands more mixed. We did this as follows: in homogenous conifer stands (conifers included Norway spruce, silver fir (*Abies alba* Mill.), European larch (*Larix decidua* Mill.), Douglas fir, black pine (*Pinus nigra*), and Scots pine) we planted groups of young broadleaved trees. The species used were sessile oak and small-leaved lime (*Tilia cordata* Mill.) and represented the management strategy to achieve economic and ecological goals. Specifically, in the management plan oak as well as other native species are foreseen

to increase (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006). Multiple groups were planted in these stands twice in the simulation period (2020-2095), at the beginning and in the middle of this period, after a year with a thinning. Red oak conversion was organized differently. Here, we coupled the forest conversion to the group harvest, where we planted trees after every group harvest in stands containing more than 50% red oak in terms of basal area. This threshold was set low enough to allow for sufficient stands with red oak to be converted. This strategy is used by the forest managers to hamper red oak from regenerating. The planted tree species included also mainly sessile oak together with small-leaved lime, common hornbeam, and common hazel (*Corylus avellana* L.).

Finally, we programmed a group system harvest, where groups varied from 10 to 50 are in size and were cut in the landscape. The minimum size of the groups was set lower than was stated in the management plan (20 are), as was suggested by the forest managers considering climate change. Although, it should be noted that smaller groups can hinder the regeneration of more light-demanding species like oak. The groups were programmed to not overlap within a year and between years. The former was done to prevent too large groups to be cut, while the latter prevented that a group that was cut in the previous year was cut again in the years to come. Nevertheless, groups were able to expand over time if groups partly overlapped in different years. The number of groups per stand was taken in relation to the area of the stand. Group harvests were programmed four years after a thinning and every eight simulation years and unenforced, with a margin of two years before and after this timing, to allow the agent to be more flexible, making sure that a sustainable timber yield could be cut. This is based on the decision heuristic implemented in ABE to safeguard that harvests are in balance with the increment (Rammer & Seidl, 2015). The groups were taken differently between the replicates and were made with R (R Core Team, 2022), using the packages mentioned in Appendix A. For more numerical details about the programming of the management module, we refer also to Appendix A.

3.4 Simulation design

We tested four different thinning scenarios, being a scenario where no management (NM) was applied, meaning that the ABE module in iLand was not enabled, and a scenario with management having a lower (LI), normal (NI), or higher (HI) thinning intensity using the ABE module, all with a constant thinning frequency of eight years. Thinning intensity is defined as

the fraction of the total basal area that is removed (Gonçalves, 2021). The different thinning intensities in our study were set to 5%, 10%, and 20% of the basal area as was suggested by the forest managers. For each combination of thinning scenario and climate scenario 10 replicates were made to account for the intrinsic stochasticity of the model. Simulations were made from 2020 until 2095. This led to 4 thinning scenarios x 14 climate scenarios x 10 replicates, which was equal to 560 simulation runs or 42,000 simulation years.

3.5 Analysis

To study the stability of the forest, we mainly looked at species composition in terms of the basal area share of each species in the forest landscape and diameter class distribution in terms of tree number share in each diameter class. We analysed the landscape dynamics through time and studied both resistance and resilience, which were defined in the literature study. For resistance we looked at the total change in species composition as well as diameter class distribution by using the Bray-Curtis Dissimilarity, an approach already used in other stability studies (Albrich, Rammer, & Seidl, 2020; Duveneck & Scheller, 2016; Lucash et al., 2017). The Bray-Curtis Dissimilarity (Bray & Curtis, 1957) is used to quantify the difference in composition in space as well as in time and varies between 0 and 1. Zero reflects full similarity between the groups in space or time, whereas one indicates full dissimilarity. We calculated the Bray-Curtis Dissimilarity between the start of the simulation period and the end where the mean was taken of the species composition or diameter distribution of the last five years. Consequently, low resistance was reflected in a high Bray-Curtis Dissimilarity, and vice versa. Next, we statistically determined the influence of the climate and management scenario on the Bray-Curtis Dissimilarity by using a Scheirer-Ray-Hare test (Scheirer et al., 1976), as the assumptions for multi-factorial ANOVA were not met. The Scheirer-Ray-Hare test is an extension of the Kruskal-Wallis test, able to study the influence of more than one factor on the outcome in contrast to just one (Scheirer et al., 1976). Next, we used a Dunn's post hoc test to make a pairwise comparison between the levels of the studied factors and used a Benjamini-Hochberg adjustment of the *P*-values. It should be noted that probably a PERMANOVA analysis would have been better, as used by Albrich, Rammer, & Seidl (2020) as this test directly compares the composition of species or diameter classes between scenarios (rather than the Bray-Curtis Dissimilarity such as in our case). Nevertheless, this test was not usable because of the significant dispersion effects.

Further on, we also analysed the total basal area of the forest through time. We also examined the resilience of this parameter under different climate change scenarios (i.e., a ramp perturbation) against pulse perturbations, where we considered every thinning intervention to be these pulse disturbances. We used Equation 1 from Van Meerbeek et al. (2021) to calculate resilience:

$$Resilience = \frac{2 \times |S_R - S_0|}{|S_R - S_0| + |S_R - S_Y|} - 1, \quad (1)$$

Here, S_R is the reference value of the system variable, S_0 is the value of the system variable farthest away from the reference condition, and lastly S_Y is the value of the system variable after a standardized time interval after the disturbance (here, one year). The incorporation of $|S_R - S_0|$ or the maximal change of the system variable allows for standardization of this change caused by the disturbance. This resilience index ranges from -1 to 1, where 1 stands for full resilience, 0 for no recovery and -1 when the system variable even deviates further away from the reference (here, further than $|S_R - S_0|$ away; Van Meerbeek et al., 2021). We used a Scheirer-Ray-Hare test (as multi-factorial ANOVA assumptions were again not met) and, consequently, a Dunn's post hoc to determine the significant differences in resilience between the management scenarios, as well as between the climate scenarios. Finally, we also assessed if the resilience at the start of the simulation period (the resilience for the three first thinning interventions) was significantly different from the resilience at the end (the resilience for the last three thinning interventions). To compare these, we used a Wilcoxon signed rank test for paired observations. In this thesis, a significance level of 0.05 in all statistical tests was used. All analyses were done using R (R Core Team, 2022). The most important packages we used, were *DBI* (R Special Interest Group on Databases et al., 2022) and *RSQLite* (Müller et al., 2023) to work with SQLite Databases, *vegan* (Oksanen et al., 2022) to calculate the Bray-Curtis Dissimilarity, *rcompanion* (Salvatore, 2024) to use the Scheirer-Ray-Hare test, and *FSA* (Ogle et al., 2023) to use the Dunn's post hoc test.

4 Results

4.1 Species dynamics

The basal area ($\text{m}^2\cdot\text{ha}^{-1}$) occupied by each species in the forest landscape was analysed through time, and different dynamics were seen for different abundant species. In the historical climate beech seemed to become dominant when no management was set in place (+76.4%, Figure 4). Nevertheless, increasing thinning intensities seemed to favour Douglas fir (+526.0%, +545.5%, and +641.9%) over beech (+27.0%, +8.9, and -22.3%). This trend was also to be observed in the moderate (Figure 5) and hot climate (Figure 6). In the hot and dry climate beech was found to have a less strong increase in the NM scenario (+17.9%) and gradually disappeared more when management became more intensive (-19.5%, -31.8%, and -52.6%; Figure 7). Conversely, black pine (+64.2%, +35.8%, +21.5%, and +16.7% for the NM, LI, NI, and HI scenario, respectively) and Douglas fir (+253.5%, +284.8%, +281.8%, and +293.6% for the NM, LI, NI, and HI scenario, respectively) thrived in this climate, although this increase was more reduced for Douglas fir compared to the other climates. The basal area of native oak was found to be slightly higher at the end of the simulation period in management scenarios (1.94, 2.16, and 2.20 $\text{m}^2\cdot\text{ha}^{-1}$) compared to NM (1.36 $\text{m}^2\cdot\text{ha}^{-1}$) in the historical climate (Figure 4). The same trend was found in the moderate (Figure 5), hot (Figure 6), and hot and dry climate (Figure 7). The basal area of red oak seemed to stay constant throughout the simulation period ($\sim 5\text{-}6 \text{ m}^2\cdot\text{ha}^{-1}$; Figure 4, Figure 5, and Figure 6), except in the hot and dry climate where it dropped through time ($\sim 2\text{-}3 \text{ m}^2\cdot\text{ha}^{-1}$; Figure 7). Scots pine disappeared in every management and climate scenario, while management seemed to accelerate its disappearance (Figure 4-Figure 7).

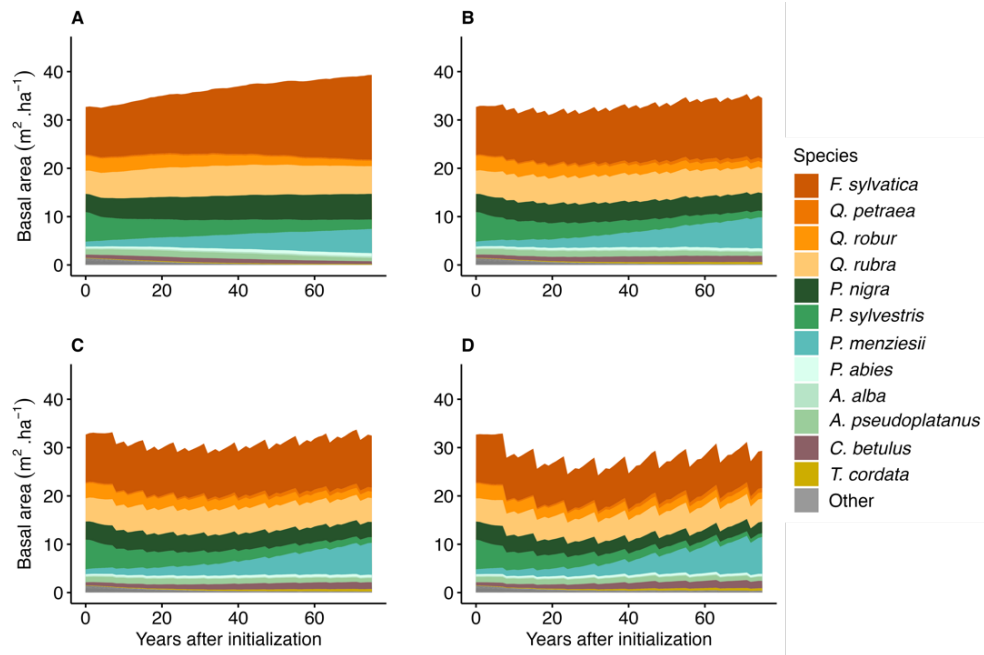


Figure 4: The basal area ($\text{m}^2.\text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the historical climate, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity.

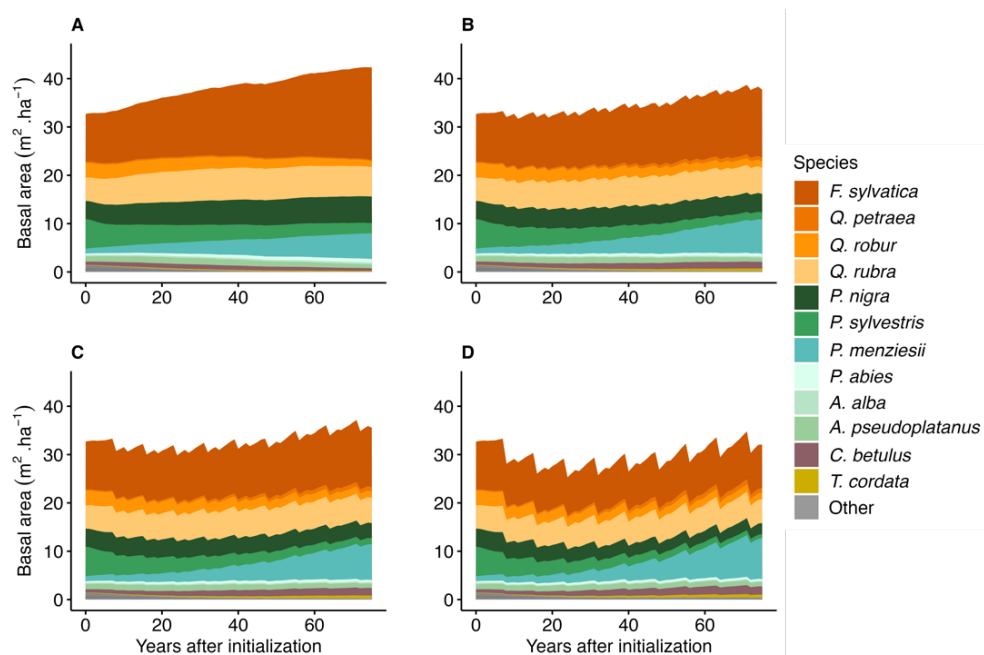


Figure 5: The basal area ($\text{m}^2.\text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the moderate climate, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity.

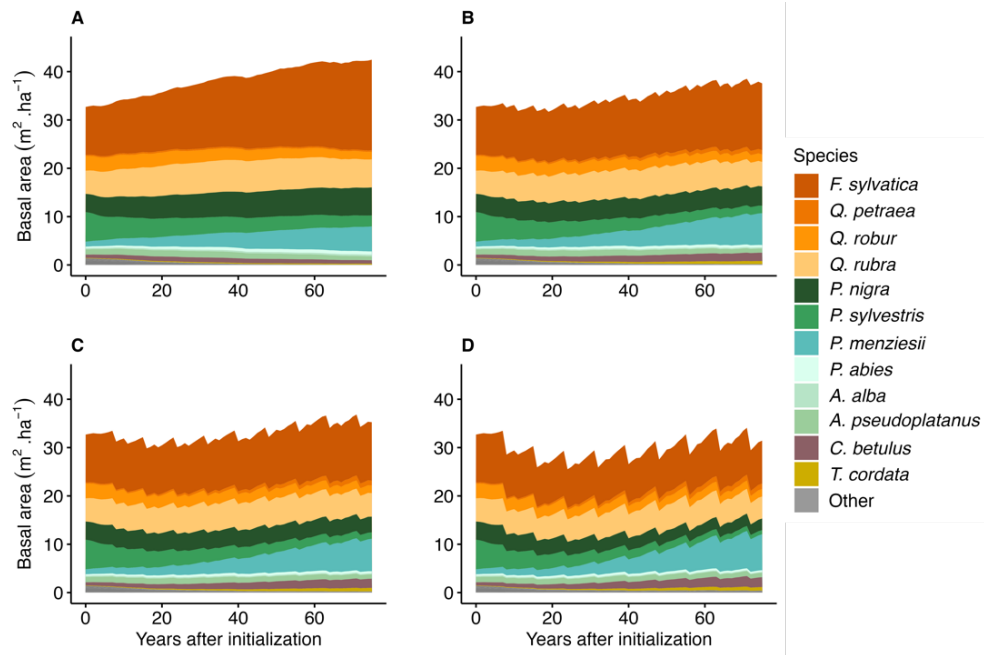


Figure 6: The basal area ($\text{m}^2 \cdot \text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the hot climate, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity.

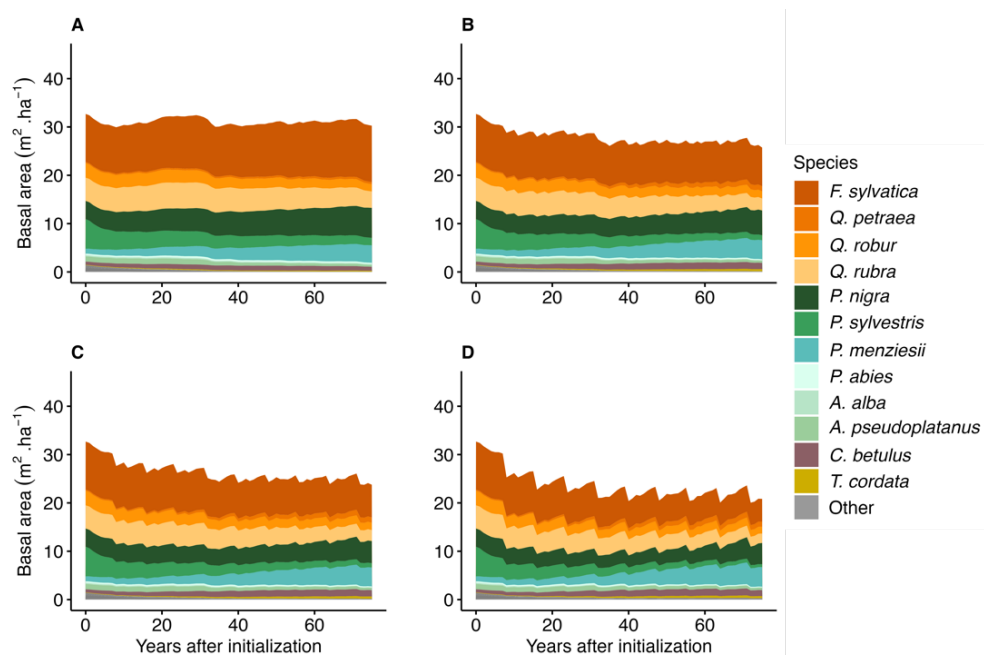


Figure 7: The basal area ($\text{m}^2 \cdot \text{ha}^{-1}$) evolution through time of the most abundant species for each management scenario in the hot and dry climate, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity.

4.2 Resistance of the species composition to climate change

Most of the previous trends, could also be derived by plotting the share in basal area of each species at the end of the simulation period (Figure 8 and Figure 9; Appendix B for numerical details). Beech became less dominant with higher thinning intensity (42.8%, 35.4%, 32.7%, and 26.6% averaged over climates; Figure 8), and this was more pronounced in the hot and dry climate (38.2%, 31.1%, 28.6%, 22.9%). Pedunculate oak together with sessile oak became more dominant when management became more intensive (4.4%, 6.9%, 7.8%, and 8.8% averaged over climates). Additionally, their share in the landscape was higher in the hot and dry climate (6.7%, 10.0%, 11.1%, and 12.0%). The share of red oak stayed constant in most climate scenarios (~13-16%) but dropped in the hot and dry climate (~9-11%). Black pine became less abundant with increasing thinning intensity in most climate scenarios (averaged over historical, moderate, and hot: 13.4%, 10.6%, 9.3%, and 7.6%) but flourished in the hot and dry climate (19.9%, 19.2%, 18.3%, and 19.8%). Scots pine almost disappeared in all scenarios (from 18.7% to 2.7%-5.5%) and Douglas fir thrived (from 3.1% to 11.7%-25.5%). The latter being more pronounced with higher thinning intensity (12.1%, 17.0%, 18.6%, and 23.2% averaged over climates). The share of hornbeam and small-leaved lime slightly increased in the landscape and was also positively correlated with a higher thinning. We observed no large differences in species composition between the historical, moderate, and hot climate, while the hot and dry climate did show a large difference with the former climates. The hot and dry climate is mainly characterized by a higher share of black pine and Douglas fir, and a lower share of Scots pine and red oak in relation to the start situation and across all management scenarios. This indicates that mainly the reduction in precipitation contributed to a shift in species composition, whereas higher temperatures (hot climate) did not have this effect.

Both climate ($P < 0.001$) and management ($P < 0.001$) had a significant influence on the Bray-Curtis Dissimilarity and thus, resistance against climate change (Appendix C). The interaction term was not significant ($P = 0.19$), meaning that climate would not affect the effect of management on the resistance and vice versa. This meant that between climate scenarios management had the same kind of influence on resistance and vice versa.

Generally, the resistance against climate change seemed to decrease when thinning intensity increased and was usually a lot lower in the HI scenario (Table 1). A lower resistance meant that the species composition shifted more strongly, which is indicated by a higher Bray-Curtis

dissimilarity. When comparing management scenarios statistically (Figure 8), the former trend was confirmed; the resistance became significantly lower going to a higher management intensity in the moderate, hot, and hot and dry climate. In the historical climate the NM, LI and NI scenarios showed no significant difference in resistance, whereas the HI scenario had a significantly lower resistance compared with the other management scenarios.

Trends in relation to the climate were less clear, except that in the hot and dry climate the resistance was always the lowest (highest Bray-Curtis Dissimilarity) compared to all other climates (Table 1). This trend was significant for all management scenarios apart from the NM scenario where there was no difference between the historical and hot and dry climate (Figure 9). Furthermore, the resistance was always significantly higher in the hot climate compared with the moderate and hot and dry climate. The moderate climate was always situated between the hot and hot and dry climate in terms of resistance which was also confirmed statistically. The historical climate showed less clear trends, being not significantly different from the moderate (NM and HI), hot (LI, NI, and HI) or hot and dry climate (NM). In conclusion, trends remained vague in terms of differences between the climate scenarios.

Table 1: The median Bray-Curtis Dissimilarity between the species composition at the start and the species composition at the end of the simulation period (2020-2095) for the different climate and management scenarios. The species composition at the end is equal to the mean species composition of the last five years of the simulation period. The 0.25 and 0.75 quantile values are given between square brackets.

	No management	Lower intensity	Normal intensity	Higher intensity
Historical	0.264 [0.261-0.266]	0.261 [0.257-0.264]	0.265 [0.261-0.267]	0.320 [0.315-0.321]
Moderate	0.262 [0.257-0.268]	0.269 [0.264-0.272]	0.276 [0.273-0.280]	0.317 [0.303-0.329]
Hot	0.252 [0.247-0.259]	0.261 [0.258-0.263]	0.269 [0.263-0.275]	0.306 [0.293-0.317]
Hot and dry	0.272 [0.256-0.274]	0.284 [0.270-0.287]	0.296 [0.284-0.302]	0.355 [0.345-0.361]

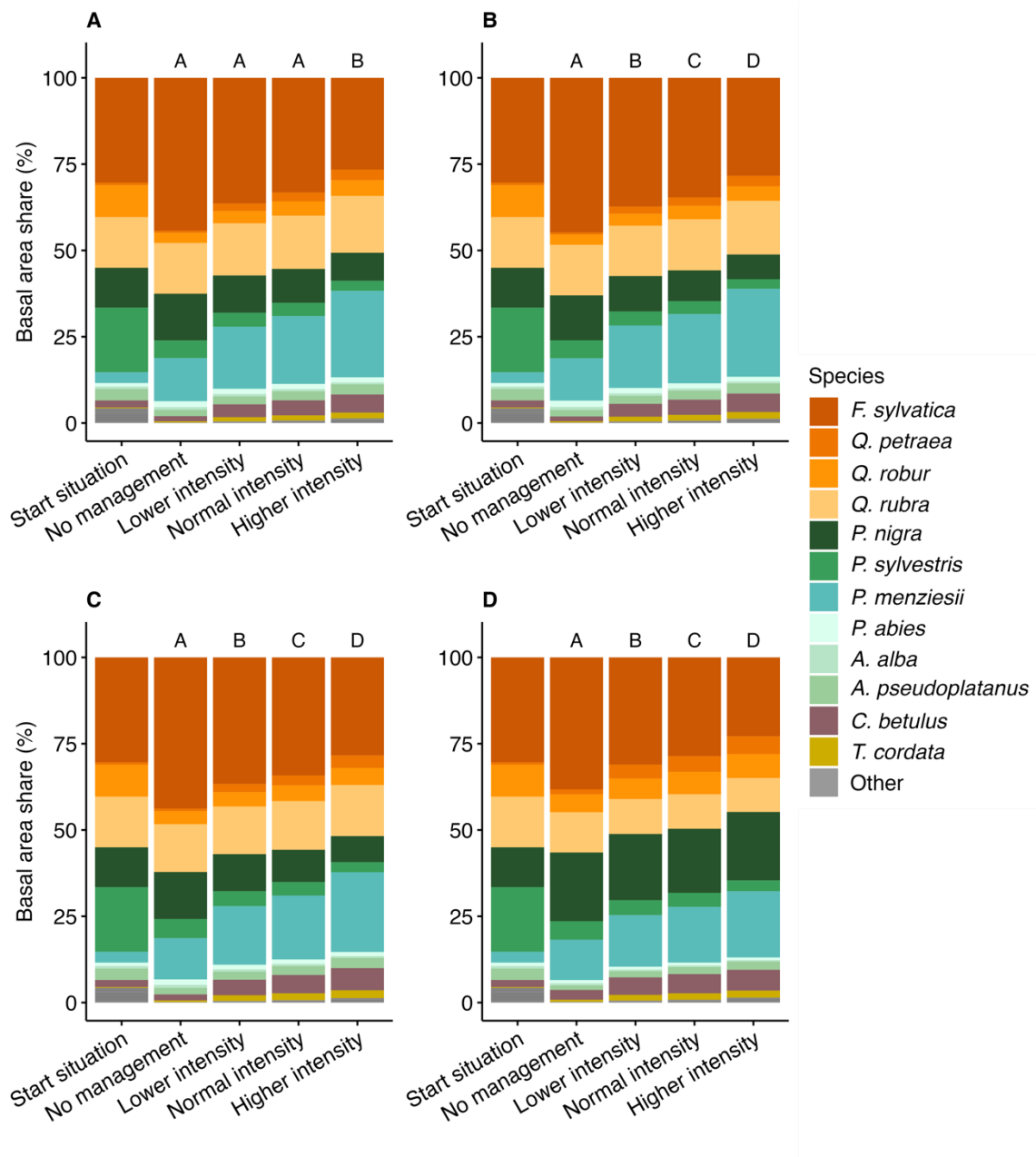


Figure 8: The basal area shares of the most abundant species at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate. The species composition at the end is the mean species composition of the last five years of the simulation period. The letters above the bars show the significant differences in the Bray-Curtis Dissimilarity values from Table 1 between the different management scenarios within a certain climate. Management scenarios show significant differences in the Bray-Curtis Dissimilarity when they do not share the same letter. *P*-values can be found in appendix C.

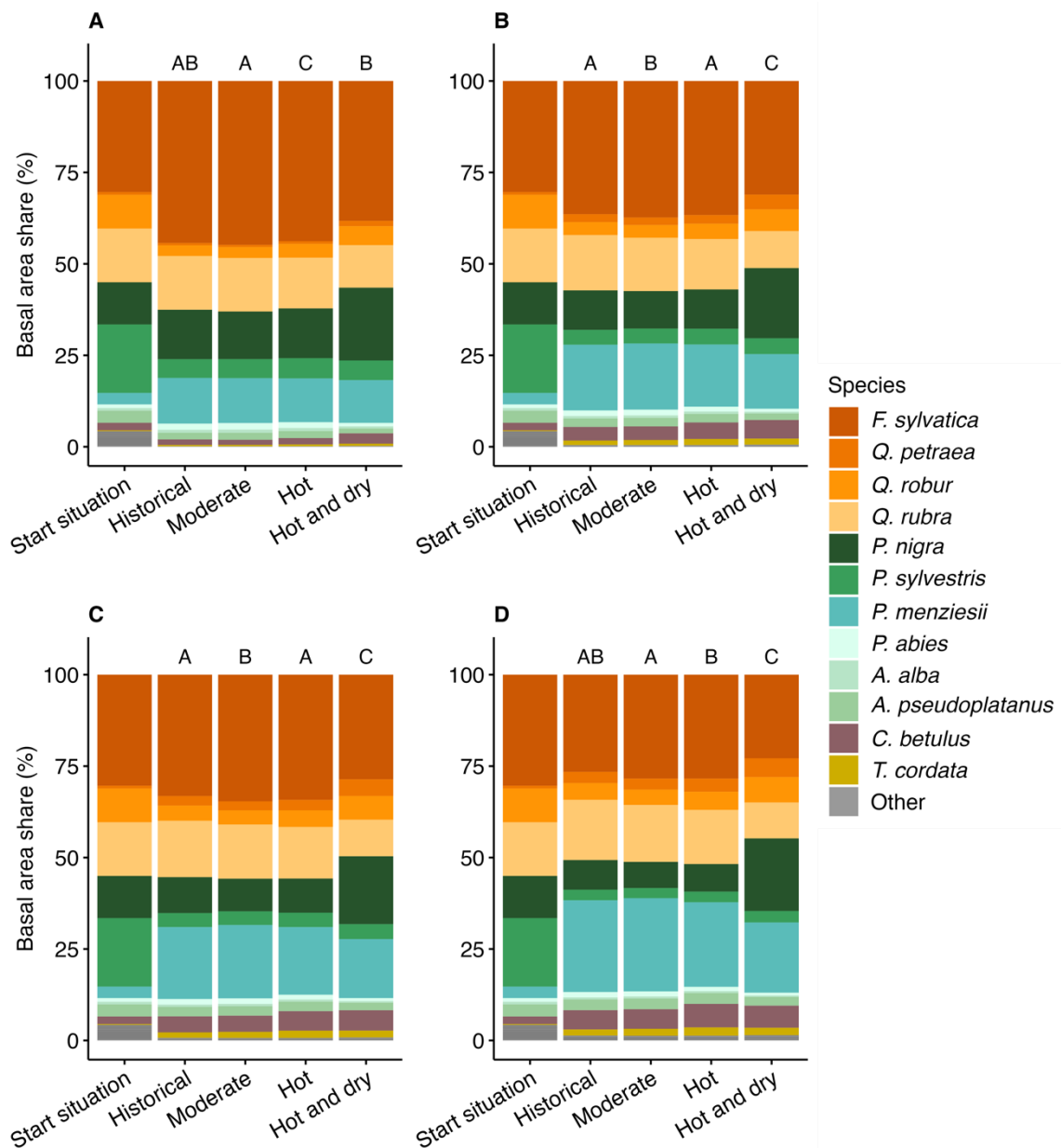


Figure 9: The basal area shares of the most abundant species at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity. The species composition at the end is the mean species composition of the last five years of the simulation period. The letters above the bars show the significant differences in the Bray-Curtis Dissimilarity values from Table 1 between the different climate scenarios within a certain management regime. Climate scenarios show significant differences in the Bray-Curtis Dissimilarity when they do not share the same letter. *P*-values can be found in appendix C.

4.3 Resistance of the diameter class distribution to climate change

The diameter class distribution evolved differently for every management and climate scenario. The NM scenario showed a higher proportion of trees with a diameter greater than 10 cm (47.7%) compared to managed scenarios in all climates (39.7%; Figure 10; Appendix B for numerical details). This proportion decreased with increasing thinning intensity (respectively, 47.7%, 40.9%, 39.2%, and 38.0% averaged over climates), which is a consequence of removing more big trees when thinning intensity was higher. When comparing between climates, older trees (>10 cm diameter) seemed to be more present in the moderate (44.0%) and hot climate (44.9%) compared to the historical (39.8%) and hot and dry climate (37.2%; Figure 11). In the hot and dry climate older trees seemed to be less abundant compared with the other climates.

Next, we observed that the change over time in the diameter class distribution was overall a lot smaller compared with the Bray-Curtis Dissimilarities for the species composition (Table 1 and Table 2), which is probably linked to the higher number of species classes than diameter classes. Both climate ($P < 0.001$) and management ($P < 0.001$) as well as the interaction between these two variables ($P < 0.001$) had a significant effect on the resistance (or Bray-Curtis Dissimilarity) of the diameter class distribution against climate change (Appendix D). The latter meant that climate could change the effect of management on the resistance and vice versa.

We observed partly the same trends as seen with the species composition, but with some more anomalies. First, the resistance against climate change seemed to also decrease with thinning intensity, except for the moderate and hot climate where the NM scenario also had a low resistance or a high Bray-Curtis Dissimilarity (Table 2). This was also confirmed statistically (Figure 10); in the historical climate we saw a subtle significant decrease in resistance with management intensity. In the moderate climate the resistance decreased significantly with thinning intensity, although the NM scenario is not found to be different from the NI scenario. In the hot climate only LI had a significantly higher resistance than all the other management scenarios. Finally, in the hot and dry climate there was again a decrease in resistance with management intensity although the NI and HI scenario were not significantly different from each other.

When comparing climates within the management scenarios, the hot and dry climate seemed to always have the lowest resistance (highest Bray-Curtis Dissimilarity) as was also observed for species composition (Table 2). This was almost statistically valid (Figure 11). The hot and dry climate had a significantly lower resistance compared to the moderate and hot climate in all management scenarios except in the NM scenario. Nevertheless, when considering the historical climate, the resistance between this climate and the hot and dry climate was almost never significantly different, except in the NM scenario. The former meant that there was a larger shift in the diameter class distribution in the historical and hot and dry scenario compared to the moderate and hot climates. This contradiction in trend between the NM and other management scenarios could be linked to the significant interaction effect or in other words, management seemed to have a different effect on the resistance trend between climates.

Table 2: The median Bray-Curtis Dissimilarity between the diameter distribution at the start and the diameter distribution at the end of the simulation period (2020-2095) for the different climate and management scenarios. The diameter distribution at the end is equal to the mean diameter distribution of the last five years of the simulation period. The 0.25 and 0.75 quantile values are given between square brackets.

	No management	Lower intensity	Normal intensity	Higher intensity
Historical	0.045 [0.032-0.049]	0.080 [0.076-0.083]	0.099 [0.095-0.104]	0.111 [0.107-0.114]
Moderate	0.076 [0.065-0.090]	0.057 [0.051-0.065]	0.074 [0.066-0.081]	0.084 [0.080-0.093]
Hot	0.081 [0.069-0.101]	0.070 [0.068-0.071]	0.075 [0.073-0.079]	0.085 [0.077-0.091]
Hot and dry	0.090 [0.069-0.092]	0.130 [0.120-0.132]	0.136 [0.129-0.139]	0.136 [0.130-0.137]

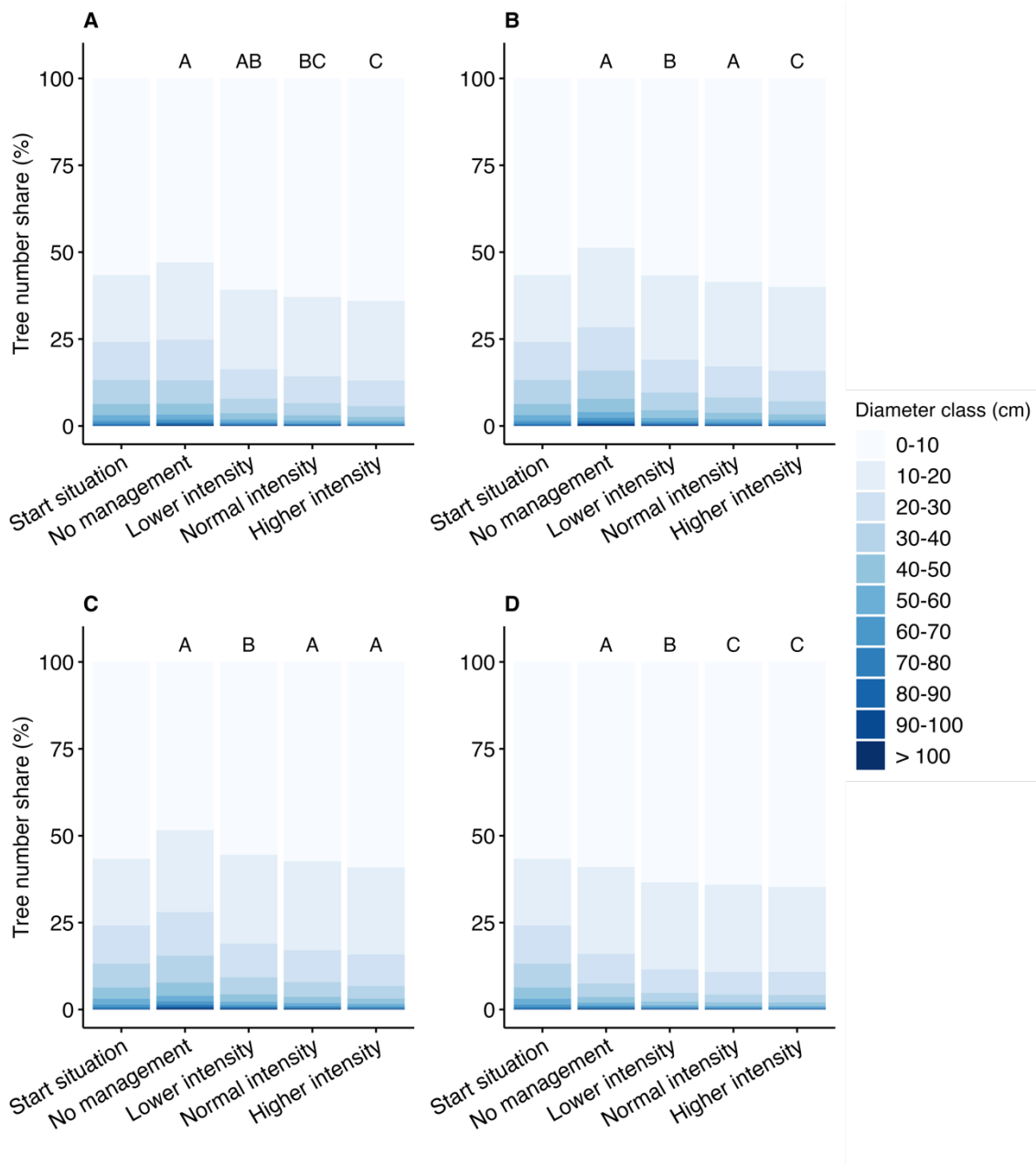


Figure 10: The share of trees for each diameter class at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate. The tree number share at the end is the mean tree number share of the last five years of the simulation period. The letters above the bars show the significant differences in the Bray-Curtis Dissimilarity values from Table 2 between the different management scenarios within a certain climate. Management scenarios show significant differences in the Bray-Curtis Dissimilarity when they do not share the same letter. *P*-values can be found in appendix D.

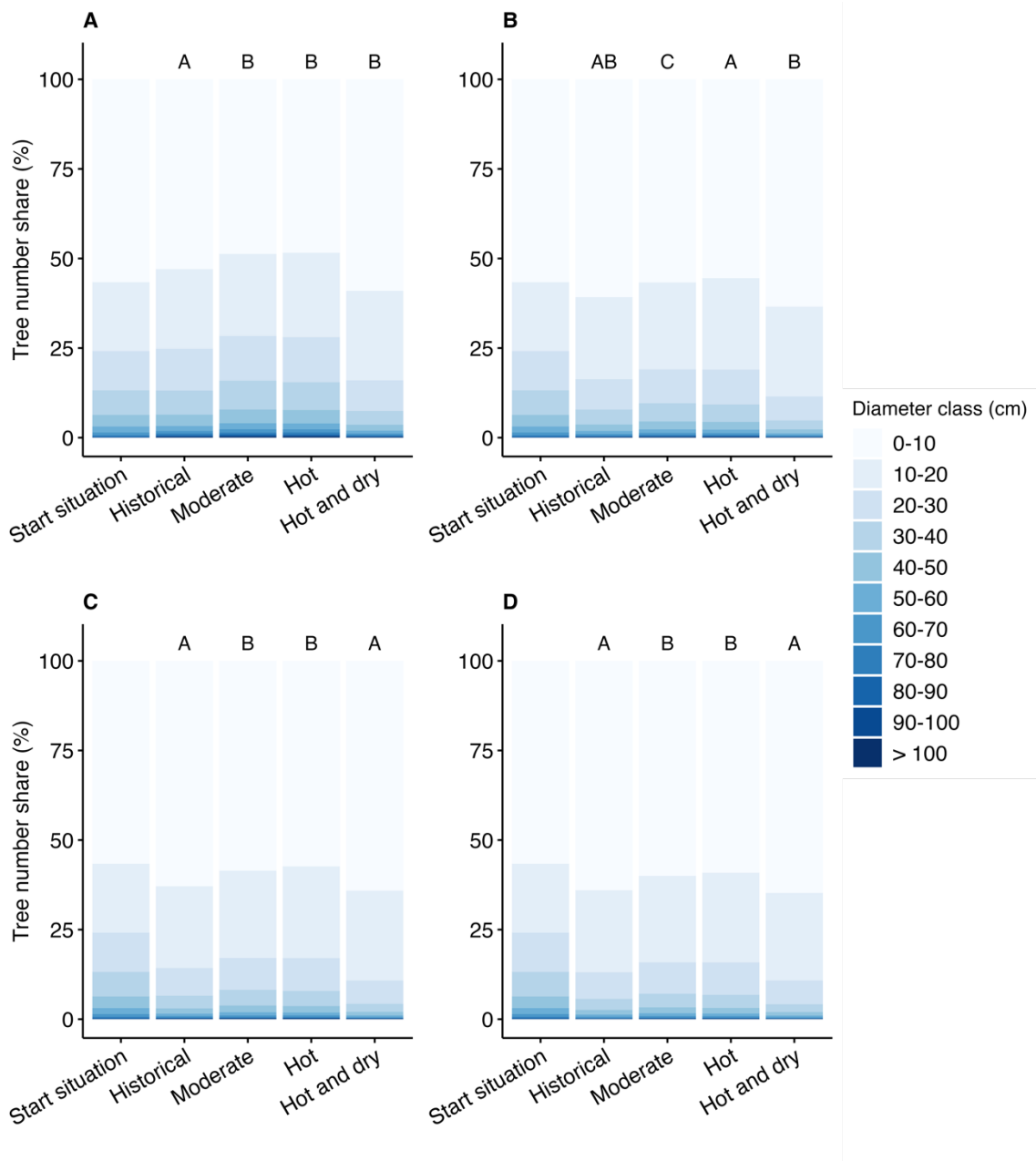


Figure 11: The share of trees for each diameter class at the start and at the end of the simulation period (2020-2095) for each climate and management scenario, with A) no management, B) lower intensity, C) normal intensity, and D) higher intensity. The tree number share at the end is the mean tree number share of the last five years of the simulation period. The letters above the bars show the significant differences in the Bray-Curtis Dissimilarity values from Table 2 between the different climate scenarios within a certain management regime. Climate scenarios show significant differences in the Bray-Curtis Dissimilarity when they do not share the same letter. *P*-values can be found in appendix D.

4.4 Basal area dynamics

In the NM scenario the basal area of the forest ($\text{m}^2.\text{ha}^{-1}$) usually increased continuously through time, except in the hot and dry climate where it stayed approximately constant ($\sim 30\text{--}33 \text{ m}^2.\text{ha}^{-1}$; Figure 12). The basal area increased more rapidly in the moderate and hot climate (reaching 42.3 and $42.5 \text{ m}^2.\text{ha}^{-1}$, respectively) compared with the historical climate ($39.3 \text{ m}^2.\text{ha}^{-1}$). Within the historical, moderate, and hot climate the same trend was to be observed for the different thinning scenarios; The higher the thinning intensity, the higher the fall in basal area in the first 20 to 30 years of the simulation period. Afterwards the basal area recovered and started to increase again in all thinning scenarios. In other words, the timber yield only became sustainable after the first decades. Conversely, in the hot and dry climate, when management was put in place, the basal area of the forest started to drop, with the decline being more severe when the thinning intensity increased (reaching 25.7 , 23.8 , and $20.8 \text{ m}^2.\text{ha}^{-1}$, respectively). Management was not sustainable, as the timber yield was probably higher than the growth of the forest, although the basal area seemed to drop less towards the end.

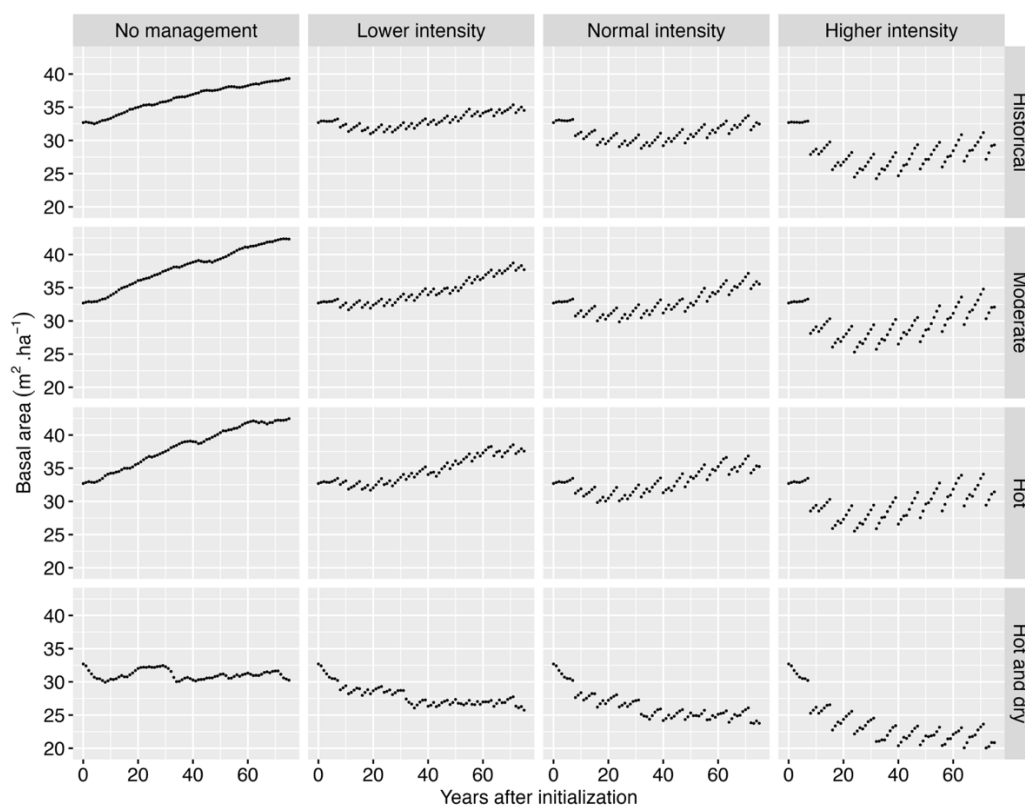


Figure 12: The mean basal area ($\text{m}^2.\text{ha}^{-1}$) of the landscape through time for each climate and management scenario.

4.5 Resilience of the basal area of the forest

Lastly, we calculated the resilience of the basal area of the forest against the thinning interventions under climate change by considering every thinning intervention as a disruption. This resulted in nine points in time where resilience could be computed for every thinning scenario (Figure 13). When looking at the trends, we observed that the variation in resilience was always higher for the LI scenario. In almost all climate scenarios the resilience seemed to slightly increase over the simulation period, except in the hot and dry scenario where the resilience seemed to be very variable for every thinning intensity.

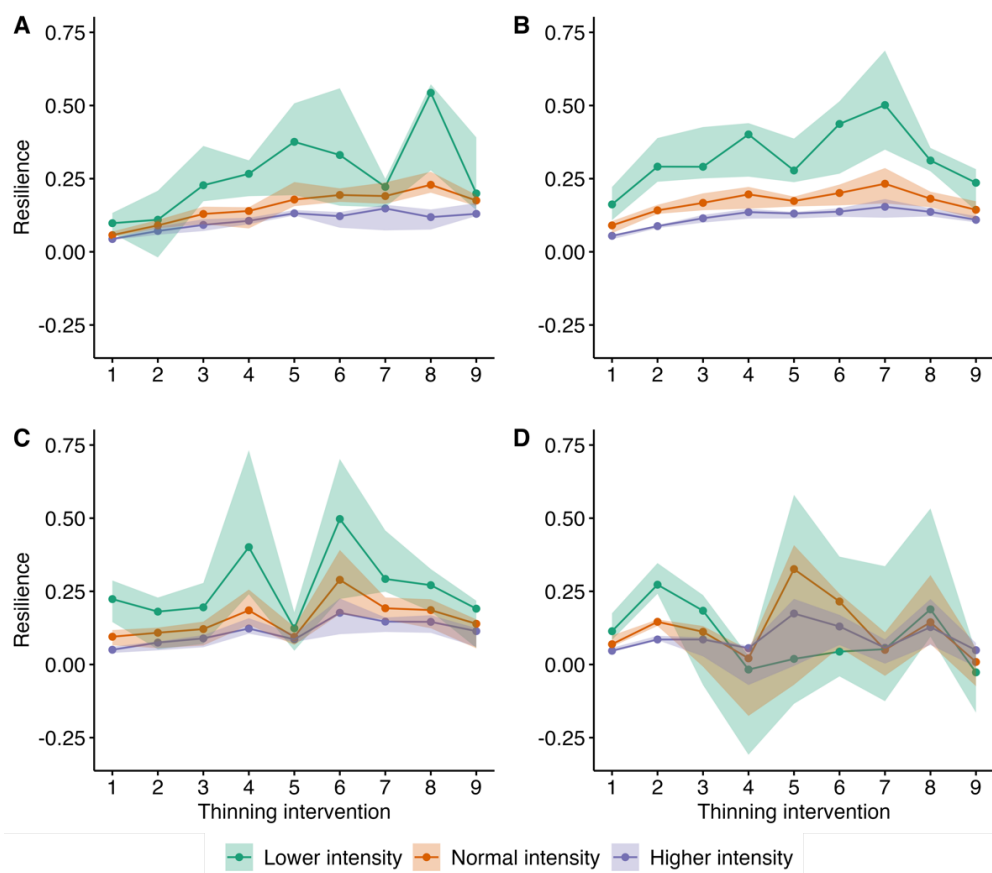


Figure 13: The median resilience index through time for each thinning intervention given per climate scenario, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate. The colored area around each line delineates the 0.25 and 0.75 quantile values.

The higher the thinning intensity, the lower the resilience appeared to be for every moment in time for the historical, moderate, and hot climate (Figure 13). In the hot and dry climate this trend was only observable until the third thinning intervention. Afterwards, the resilience

did not show a clear difference anymore between the thinning scenarios, and the uncertainty intervals started to overlap strongly. Both climate ($P<0.001$) and management ($P<0.001$) were found to have an important effect on the resilience index, as well as the interaction term ($P<0.001$; Appendix E).

The medians of the resilience over all thinning interventions were all statistically different between the management scenarios for the historical, moderate, and hot climate (Figure 14). Here, the LI scenario had the highest resilience, followed by the NI and HI scenario (Table 3). A high resilience meant that the forest landscape could recover faster after a disturbance (or thinning) took place. Conversely, in the hot and dry climate the resilience between all management scenarios was not significantly different. This could probably be linked to the significant interaction effect, where climate influences the effect of management on the resilience index.

Table 3: The median of the resilience over the thinning interventions for every management and climate scenario. The 0.25 and 0.75 quantile values are given between square brackets.

	Lower intensity	Normal intensity	Higher intensity
Historical	0.226 [0.134-0.392]	0.156 [0.088-0.199]	0.104 [0.064-0.132]
Moderate	0.288 [0.221-0.422]	0.161 [0.129-0.201]	0.118 [0.091-0.139]
Hot	0.235 [0.137-0.351]	0.133 [0.086-0.199]	0.098 [0.068-0.147]
Hot and dry	0.090 [-0.041-0.245]	0.080 [0.021-0.149]	0.069 [0.043-0.099]

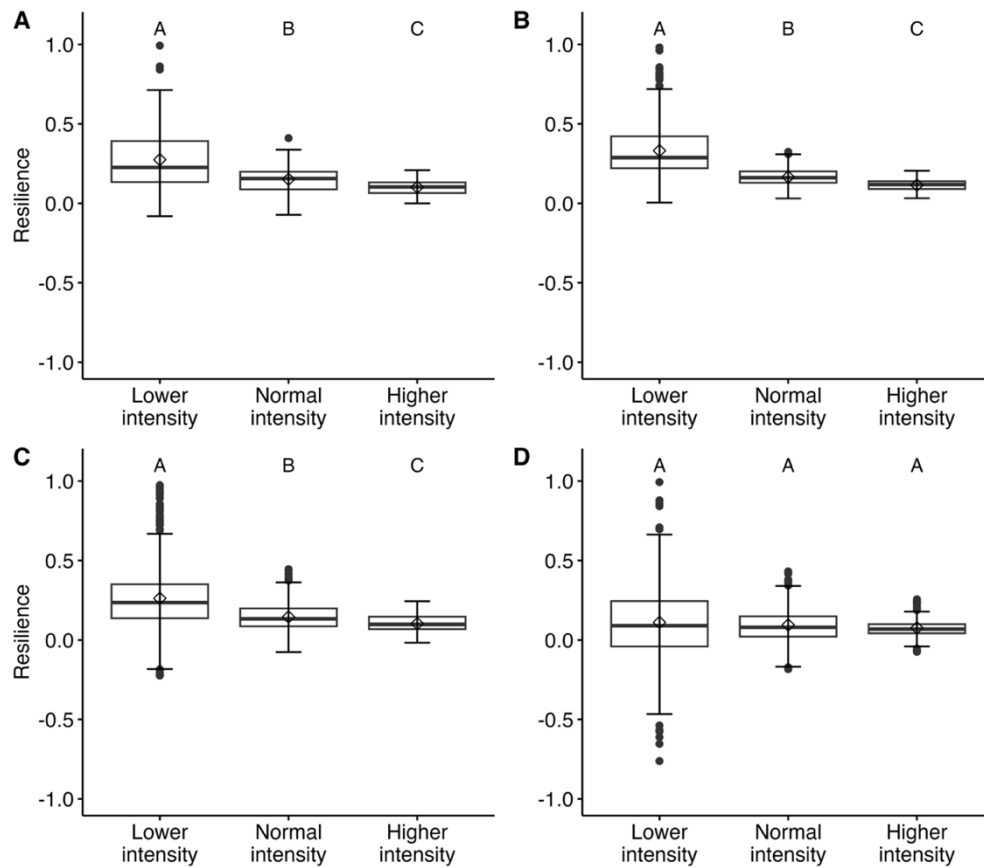


Figure 14: The distribution of the resilience for every management scenario with the significant differences denoted by different letters above the boxplots, with A) historical climate, B) moderate climate, C) hot climate, and D) hot and dry climate. Management scenarios show significant differences in resilience when they do not share the same letter. *P*-values can be found in appendix E.

It was clear that in every management scenario the resilience was significantly lower in the hot and dry climate compared to the other climates (Table 3 and Figure 15). In the LI and HI scenario also the moderate climate was significantly different from the other climates, showing a higher value for the resilience. This was not the case for the NI scenario, where the moderate climate had a significantly higher resilience than the hot and hot and dry climate but not than the historical climate.

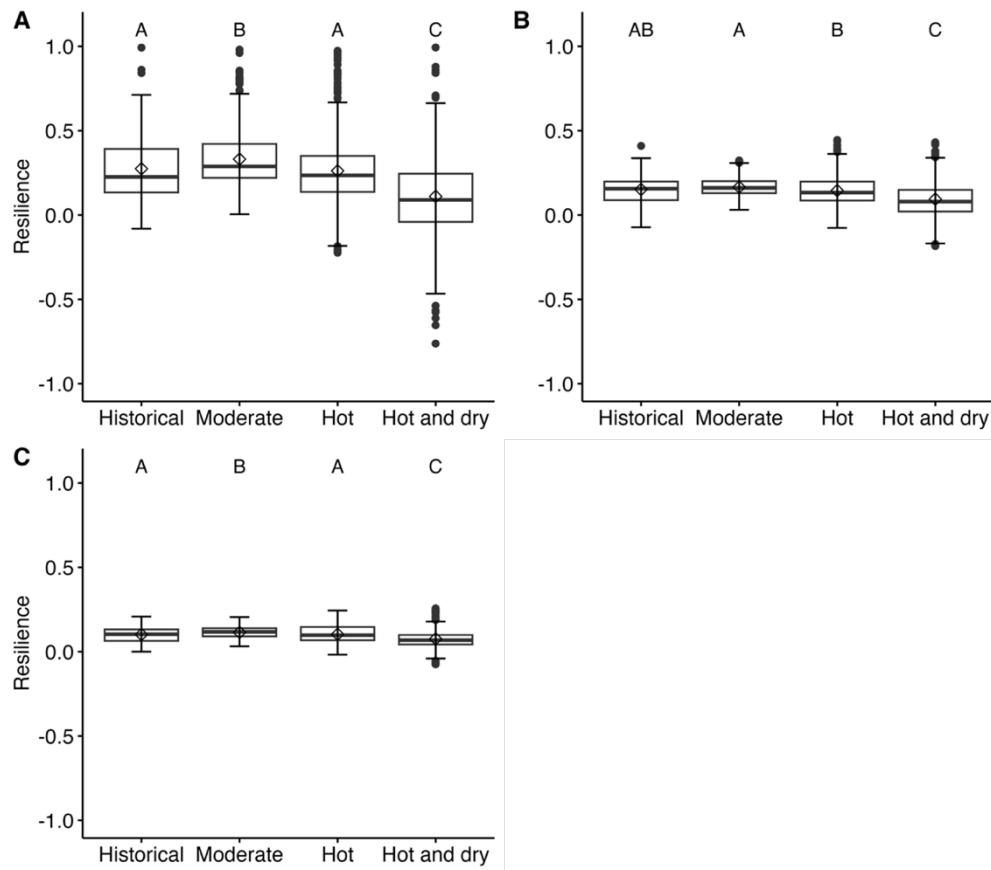


Figure 15: The distribution of the resilience for every climate scenario with the significant differences denoted by different letters above the boxplots, with A) lower intensity, B) normal intensity, and C) higher intensity. Climate scenarios show significant differences in resilience when they do not share the same letter. *P*-values can be found in appendix E.

Lastly, we observed that all resilience indices at the end of the simulation period were significantly different from these at the start, except for the HI scenario in the hot and dry climate (Table 4). In the historical, moderate, and hot climate the resilience index was always higher at the end in contrast to the hot and dry climate where it was always lower at the end for all thinning intensities.

Table 4: Comparison of the resilience at the beginning (in the first three thinning interventions) with the resilience at the end (in the last three thinning interventions) for every climate and management scenario. The 0.25 and 0.75 quantile values are reported between square brackets and the *P*-value indicates significant differences between the medians of these two time periods.

Climate	Management	Resilience start	Resilience end	P-value
Historical	Lower	0.148[0.058-0.235]	0.245[0.173-0.497]	0.02
	Normal	0.090[0.065-0.122]	0.195[0.158-0.237]	<0.001
	Higher	0.065[0.046-0.088]	0.132[0.092-0.153]	<0.001
Moderate	Lower	0.251[0.152-0.354]	0.311[0.223-0.443]	<0.001
	Normal	0.131[0.086-0.161]	0.178[0.139-0.216]	<0.001
	Higher	0.083[0.055-0.100]	0.128[0.110-0.150]	<0.001
Hot	Lower	0.200[0.122-0.251]	0.253[0.181-0.310]	<0.001
	Normal	0.109[0.065-0.130]	0.163[0.130-0.214]	<0.001
	Higher	0.067[0.040-0.085]	0.125[0.091-0.156]	<0.001
Hot and dry	Lower	0.188[0.108-0.256]	0.078[-0.116-0.189]	0.002
	Normal	0.112[0.068-0.135]	0.065[-0.037-0.144]	0.007
	Higher	0.082[0.045-0.087]	0.066[0.011-0.115]	0.85

5 Discussion

5.1 Species dynamics

Beech, a dominant species in the Meerdaal and Heverlee forest, experienced a decline in abundance with increasing management intensity in all climates, especially in the hot and dry climate projection (Figure 4-Figure 7). As beech is a shade-tolerant species (Packham et al., 2012), it can easily dominate over less shade-tolerant species (Gravel et al., 2010). When management or disturbances are introduced, canopy gaps are created and can promote co-existence with other (less shade-tolerant) species (Gravel et al., 2010). Moreover, in Flanders, pedunculate oak, a light-demanding species, was found to experience a reduction in growth when mixed with more shade-tolerant species, such as beech and red oak (De Groote et al., 2018; Niinemets & Valladares, 2006). This was confirmed by Vannoppen et al. (2019) who found a decreased and increased radial growth for oak and beech, respectively, in mixed stands in Flanders in relation to their shade tolerance, explaining why oak became more abundant when thinning intensity became higher and hence, competition lower (Figure 4-Figure 7). Beech suffered more in the hot and dry climate because it is a drought-sensitive species (Leuschner, 2020). Hence, the competitive power of beech reduced, and less drought-sensitive species like oak or black pine became more dominant (Figure 7; Niinemets & Valladares, 2006; Scharnweber et al., 2011; Vacek et al., 2023). Studies in Flanders also showed that in the 20th century growth of pedunculate oak increased, mainly due to higher growing season temperatures, whereas the growth of beech decreased in the last decades, which was mainly attributed to drought (Aertsens et al., 2014; Kint et al., 2012). Additionally, these trends are expected to continue towards the future (Kint et al., 2012), which corroborates with our results in the hot and dry climate.

In the hot and dry climate black pine thrived (Figure 7), which can be explained by its high resistance to droughts and heat waves (Lévesque et al., 2014; Vacek et al., 2023). Douglas fir became also more abundant in this climate. Specifically, Niinemets & Valladares (2006) reported black pine and Douglas fir to be more drought-tolerant than beech. Moreover, Lévesque et al. (2014) and Eilmann & Rigling (2012) found black pine and Douglas fir to be more tolerant against drought and variability in climate compared to Scots pine, larch, and Norway spruce in Central-Europe, although other studies have reported that Douglas fir may be sensitive to intense droughts (Sergent et al., 2014). This could explain why Douglas fir still

increased in terms of basal area in the hot and dry climate, although this increase was less than in the other climates (Figure 7). Thinning seemed to promote Douglas fir over beech (Figure 4-Figure 7), probably as Douglas fir is less shade-tolerant (Niinemets & Valladares, 2006). Additionally, the high increase in basal area of Douglas fir can relate to its high growth rates and ability to effectively suppress other trees by shading (Thomas et al., 2022). Nevertheless, Douglas fir became very abundant in every management and climate scenario, which could also be contributed to its parametrization in the model, as this species had high parameter values for the maximum height and fecundity. Scots pine seemed to disappear quickly in every climate and management scenario (Figure 4-Figure 7). This can be linked to the low competitive power of Scots pine on fertile sites, meaning it gets easily outcompeted by other species (Mátyás et al., 2004). Finally, reductions in basal area of some conifers with thinning intensity can also be attributed to the programmed management, which aimed to make homogenous conifer forests more mixed.

The share of red oak stayed relatively constant in all climate and management scenarios (Figure 4, Figure 5, and Figure 6), except in the hot and dry climate where it slightly decreased (Figure 7). Red oak is reported to be more drought-resistant than the native oak species (Dyderski et al., 2020), nonetheless native oaks are doing better than red oak in the hot and dry climate relative to the start situation and compared with other climate scenarios. When comparing the managed with non-managed scenarios, this could be a consequence of the management which suppressed red oak and promoted native oak by the planting of sessile oak, as well as the expansion of black pine in this climate.

We observed little difference between the moderate and hot climate in terms of the species composition as well as the diameter class distribution at the end of the simulation period and the basal area evolution of the forest. This can be explained based on how the model accounts for the temperature response in trees. Specifically, Seidl, Rammer, et al. (2012) adopted their approach to model the temperature response from Mäkelä et al. (2008), which assumes no growth decline when temperature becomes high, in contrast to a parabolic response function such as in Sands & Landsberg (2002). Consequently, the temperature might not have had large negative effects.

5.2 Resistance of the species composition to climate change

Increasing thinning intensity lowered the resistance of the species composition to climate change (Table 1 and Figure 8). This conflicts with our hypothesis, stating that a higher thinning intensity would increase stability, and thus resistance, of the forest against climate change, as between-tree competition would become lower (Chase et al., 2016) and drought tolerance would become higher (Sohn, Saha, et al., 2016). Therefore, we expected that different species, mostly the drought-sensitive species such as beech, would better maintain their niche in the landscape. Nevertheless, if we consider thinning interventions as a form of forest disturbances, our results corroborate with Thom et al. (2017), who found that increasing disturbance severity (here, linked to thinning intensity) could accelerate the adaptation of species composition to climate change. Specifically, an equilibrium between the species composition and the prevailing climate was reached faster when disturbances were introduced. Disturbances also promoted early successional species more (Thom et al., 2017). In our study, we observed that thinning promoted species which are often more light-demanding and better adapted to the prevailing climate, such as oaks at the cost of beech (Figure 8; Aertsens et al., 2014; Kint et al., 2012). The share of Douglas fir, being a light-demanding species with rapid height growth and being more drought tolerant than beech, also increased with increasing thinning intensity (Niinemets & Valladares, 2006; Thomas et al., 2022). Furthermore, In the hot and dry climate, black pine became an important species, mostly because it is better adapted to droughts (Vacek et al., 2023), although its share stayed approximately the same over management scenarios, which is probably due to the conifer conversion management. We should note that our shifts in species composition as a consequence of thinning, could be an overestimation compared to what would happen in reality, as here thinnings were imposed on the whole landscape at specific moments in time whereas in reality thinnings are executed on the stand level.

The resistance of the species composition against climate change was the highest in the hot scenario, followed by the moderate and hot and dry scenario (Table 1 and Figure 9). This held for every management scenario. However, the resistance in the historical climate showed different behaviour across the management scenarios, being not significantly different from the moderate (NM and HI), hot (LI, NI, and HI) or hot and dry climate (NM). As the hot and dry climate is the most different in terms of temperature and precipitation from the current

climate, it is not surprising that here the resistance of the species composition was lowest. Some species will probably be disadvantaged (such as the drought-sensitive beech; Leuschner, 2020), whereas other species will see a chance for expansion (such as the drought-tolerant black pine; Vacek et al., 2023). These species dynamics under different climate change scenarios were also observed by Storms et al. (2023). Moreover, Duveneck & Scheller (2016) also found a lower resistance in species composition in the most extreme climate change scenario for one of two forests they studied in the USA. The other forest showed higher resistance in this climate, which was explained by a delay in succession caused by climate change (Duveneck & Scheller, 2016). Lastly, we did not expect the hot climate to have a higher resistance than the moderate climate given the higher temperatures. This could again be linked to temperature having no directly negative effects on growth in iLand (Seidl, Rammer, et al., 2012), as well as the wetter conditions in the moderate climate (Figure 2). Additionally, higher temperatures can lengthen the growing season (Jeong et al., 2011), which can have positive effects on the growth of different species and maybe promote resistance. Furthermore, Lucash et al. (2017) found greater change in species composition under climate change compared to under historical climate for a forest in Minnesota. Our findings are not in accordance with this, as we observed the resistance of the historical climate to be similar to the moderate, hot, or hot and dry climate depending on the management scenario. This may indicate that the current species composition in the Meerdaal and Heverlee forest is not in equilibrium. Storms et al. (2023) found that in this forest under historical climate and without management beech would become very dominant, and an equilibrium in species composition would only be reached after approximately 500 years.

5.3 Resistance of the diameter class distribution to climate change

Our hypothesis stating that thinning intensity could increase the resistance of the diameter distribution to climate change could not be confirmed, as in every climate scenario higher thinning intensities decreased the resistance (Table 2 and Figure 10), although less clear compared to its effect on the resistance of the species composition, indicating that thinning shifts the forest structure rather than maintaining it. The NM scenario showed the highest resistance in the historical and hot and dry climate but was not different from the NI or HI scenario in the moderate and hot climate (Table 2 and Figure 10). Moreover, the hot and dry climate was found to most often have a significantly lower resistance compared to the

moderate and hot climate, but not compared with the historical climate, except when the forest was not managed (Figure 11). As we did not study the variables influencing the shift in diameter class distribution in detail, we can only describe some possible contributing factors. We expect that resistance could have been most likely influenced by the combination of management, disturbances (e.g. drought), the CO₂ fertilization effect, and species shifts.

The resistance as measured by the Bray-Curtis dissimilarity does not consider the direction of change, only the magnitude of change, which can explain some anomalies in the observed resistance trend, mainly when taking the non-managed scenarios into account. The larger share of trees with a diameter greater than 10 cm in the moderate and hot climate compared to the historical and hot and dry climate could be a consequence of the CO₂ fertilization effect which promotes growth (Norby et al., 1999). Furthermore, as height-diameter (HD) relationship thresholds are different between species in iLand (Seidl, Rammer, et al., 2012), and growth rates may differ between species, a shift in species composition could possibly also underly changes in diameter class distributions, but as the species composition at the end of the simulation period was visually very similar between the moderate and hot climate (Figure 9), this effect would probably be minimal here, in contrast to maybe the hot and dry climate (Figure 9). Moreover, although studies have found an influence of temperature and precipitation on the HD relationships of different species (Fortin et al., 2019; Hulshof et al., 2015), iLand does not take these influences into account and only accounts for the influence of light availability and plant age (or diameter) on a tree's HD relationship (Niklas, 1995; Seidl, Rammer, et al., 2012). Nevertheless, this means that thinning can impact these HD relationships by reducing the between-tree light competition, leading to trees becoming thicker when competition is reduced (del Río et al., 2019; Niklas, 1995), although this effect was not to be seen in the diameter class distribution as management counteracted this effect by removing trees in higher diameter classes, leading to less big trees (>10cm diameter) when thinning intensity increased (Figure 10).

The share of trees with a diameter greater than 10 cm was usually lower in the hot and dry climate compared with the historical climate (Figure 11), which is in line with literature. Specifically, Albrich, Rammer, & Seidl (2020) found that climate warming increased the number of smaller trees (with a diameter < 30 cm) in the Alps, with lower rainfall accelerating this transition. Although we looked at trees over 10 cm diameter, looking at the proportion

of trees over 30 cm diameter followed the same trends (Appendix B), with this proportion being greater in the moderate and hot climate and lower in the hot and dry climate relative to historical conditions. In this regard, our results from the hot and dry climate are in accordance with their findings, whereas this is not the case for the hot climate, which could relate to the absence of rising CO₂ concentrations in their study (Albrich, Rammer, & Seidl, 2020). Moreover, Taccoen et al. (2019) found that climate change contributed to the background mortality (besides stand conditions and the tree's social status) for half of the temperate tree species they studied, whereby increasing temperature and decreasing precipitation led to higher mortality. Moreover, review studies confirm that forest mortality as caused by climate change, mainly droughts and heat, is increasing worldwide and pushing stands to stay relatively young (Allen et al., 2010; McDowell et al., 2020). Even though CO₂ was still at elevated levels in this climate scenario, Obermeier et al. (2017) found that drought and/or heat can reduce the CO₂ fertilization effect. Dominant species that seemed to suffer under this climate, were mainly beech, red oak and Scots pine as their basal area mostly dropped through time, whereas black pine, Douglas fir, and native oak thrived (Figure 7). Although Douglas fir and black pine are fast-growing species (Enescu et al., 2016; Thomas et al., 2022), they did not lead to a higher share of big trees in the forest.

5.4 Basal area dynamics

When the forest complex was not managed, the basal area of the landscape increased continuously over time in the historical, moderate, and hot climate, whereas in the hot and dry climate it stayed constant (Figure 12). The CO₂ fertilization effect could explain the stronger increase in the moderate and hot climate, while in the hot and dry climate mortality could have led to a lower basal area. Specifically, it has been found that the CO₂ fertilization effect can increase the water-use-efficiency of the plant, due to a reduced stomatal conductance and hence transpiration (Ainsworth & Long, 2005; Drake et al., 1997), an effect also included in iLand (Seidl, Rammer, et al., 2012). Consequently, the plant will be more resistant to drought (Birami et al., 2020). However, prolonged heat and drought stress can eventually lead to a closure of the stomata and hence, reducing photosynthesis and leading to mortality at the end (Birami et al., 2020; Choat et al., 2018). Uncertainty remains about to what extent the positive effect of elevated CO₂ on the water use efficiency can compensate for drought (Xiao et al., 2021).

We concluded that the droughts in the hot and dry climate were likely too harsh and dominated over the positive effects of the CO₂ fertilization effect. Additionally, the dominant tree species in the landscape, beech, is also drought-sensitive (Leuschner, 2020). As mentioned before, empirical studies have found that extreme climate conditions, i.e. droughts and heat, can diminish the CO₂ fertilization effect (Obermeier et al., 2017). Moreover, the CO₂ fertilization effect can be limited by nutrient constraints, such as nitrogen limitation (also included in iLand (Seidl, Rammer, et al., 2012)) (Norby et al., 2010; Reich & Hobbie, 2013). Although the soils of Flanders are currently rich in nitrogen (VMM, 2018), we cannot exclude the influence of this effect in this study, and, hence, the persistence of the CO₂ fertilization effect towards the future must be questioned. Peñuelas et al. (2017) stated that towards the future the positive effects of CO₂ fertilization (as well as nitrogen fertilization) may be outweighed by the negative effects of changing climatic conditions and disturbance regimes as well as nutrient limitations, leading to ecosystems with a reduced carbon sink potential.

Finally, when managed, the basal area dropped in the first 20-30 years and started to increase afterwards in the historical, moderate, and hot climate (Figure 12). This drop was more pronounced when thinning intensity was higher. The increase could probably be linked to the group harvests becoming less pronounced already after the first decade. This is possible as the group harvests were flexibly planned in the STP, as they were planned unenforced with a margin of four years to be executed. Conversely, the thinnings were fixed in time and enforced. The ABE module of iLand contains a decision heuristic which ensures a sustainable harvest, and where the sustainable harvest is recalculated every 10 years (Rammer & Seidl, 2015), explaining the reduced group harvests. In the hot and dry climate, mortality may have been higher because of the more extreme climate as mentioned above, and even though the group harvests became also less pronounced over time, harvest was not sustainable anymore, leading to a continuous decrease of the basal area of the forest. Increasing the thinning frequency (e.g., 8 to 16 years) or applying no thinning at all could possibly prevent this drop in basal area.

5.5 Resilience of the basal area of the forest

The resilience was found to be significantly higher for the lower intensity thinning, followed by the normal and then higher intensity in the historical, moderate, and hot climate (Figure

13 and Figure 14). This conflicts with our hypothesis stating that a higher thinning intensity would increase the resilience of the forest, leading to a faster recovery of the basal area of the forest. Empirical studies investigating the resilience of the basal area growth on the individual tree level against drought, most often found increased resilience with increasing thinning intensity (Diaconu et al., 2017; Kohler et al., 2010; Sohn, Hartig, et al., 2016). This could indicate that resilience of the basal area growth can be scale-dependent, as we found the opposite trend on the landscape level, although we assessed resilience against pulse-perturbations being thinning "disturbances" and not drought events, highlighting the importance of the questions resilience "of what?" and "to what?" sensu Carpenter et al. (2001). Furthermore, a modelling study of Seidl, Vigl, et al. (2017) studied the resilience of the total stem volume (and leaf area index (LAI)) to thinning interventions (also seen as disturbances) of different intensity under different climate change scenarios. Although they studied even-aged Norway spruce forests, our findings were in line with theirs, as they found that increasing thinning (or disturbance) severity reduced resilience for a given age and basal area. However, they found that stand characteristics also influenced resilience. Specifically, increasing age and basal area negatively affected resilience. In our study, the basal area developed quite differently for every thinning intensity (Figure 12), but as the resilience trend $LI > NI > HI$ seemed to stay valid over the studied time period, the effect of disturbance severity could have been more important even though the basal area of the forest was lower for every moment in time for the HI scenario, followed by the NI and LI scenarios (Figure 12). Nevertheless, to understand the underlying drivers of resilience better, possible contributing variables should have been studied in more detail, although this was out of the scope of this thesis.

Under the hot and dry climate, we did not observe a significant difference between the different thinning intensities and overall, resilience was low (Figure 14), suggesting that the effects of extreme climatic changes can exceed those from management, as confirmed by Duveneck & Scheller (2016). This could indicate that under extreme future climate conditions, one adaptive measure, such as changing thinning intensity, may not be enough to make forests more resilient to climate change, and that probably additional climate-smart measures or more active management would be necessary (Duveneck & Scheller, 2016). Moreover, the resilience was found to be significantly lower for the hot and dry climate

compared to all other climates (Figure 15). This finding is in accordance with findings of Duveneck & Scheller (2016) who studied the resilience of two forest landscapes in the USA after wildfire disturbances under different climate scenarios. Although they looked at the resilience of aboveground biomass and species composition simultaneously, they found lower resilience under a high emissions climate scenario in comparison to historical and low emissions climate scenarios. Moreover, Lucash et al. (2017) found that resilience, as calculated similarly to Duveneck & Scheller (2016), was lower under climate change compared to historic climate. In our case, this was only valid when comparing the hot and dry climate (and not always for the moderate and hot climate) with the historical climate, where the former had a consistent lower resilience than the latter (Figure 15). Often, the moderate climate was found to be the most resilient, which could maybe be a consequence of the positive effects of the CO₂ fertilization effect and less disturbances compared to the hot and hot and dry climate. Moreover, Seidl, Vigl, et al. (2017) found that resilience does not per se decrease under climate change. Whether it increases or decreases under future climate is related to the site conditions as well as the change in precipitation and/or temperature.

Finally, we observed an increase in resilience through time in the historical, moderate, and hot climates, whereas the hot and dry climate most often showed a decline in resilience (Table 4). The latter could be explained by the climate becoming more extreme (warmer and drier) towards the end of the simulation period (Duveneck & Scheller, 2016; Lucash et al., 2017). As stand and site characteristics can influence resilience (Seidl, Vigl, et al., 2017), and species composition also changes over the simulation period, dynamics in these variables might explain the resilience increase in the historical, moderate, and hot climate. Again, more detailed research should be needed to look at the underlying drivers. Additionally, it is important to note that these resilience trends were observed for a period of 75 years. Simulating for longer periods could potentially show other trends. Specifically, we observed a slightly downward trend of the resilience in the moderate and hot climate at the end of the simulation period (Figure 13), but as the resilience index could be somewhat erratic, we should make longer simulations to better observe this trend.

5.6 Limitations

Lastly, we should acknowledge that our study has its limitations. Although models are very useful to investigate forest dynamics under unprecedented conditions, they approximate

reality and contain process descriptions, assumptions, and parameter values that can introduce uncertainty into the model outputs, which should be interpreted carefully (Landsberg & Sands, 2011a). Secondly, although multiple climatic pathways were used in this study (six scenarios in RCP 4.5 and seven in RCP 8.5), there is still uncertainty about the future climate and disturbance regimes. Specifically, our future greenhouse gas emissions, which will depend on our mitigation actions, will largely determine future warming and consequently, the future climate (IPCC, 2023b). Next, our modelling approach does not consider genetic adaptation and phenotypic plasticity of species, which could enhance adaptability both in the short and long term and thus, alleviate changes in species composition because of climate change (Fady et al., 2016; Richter et al., 2012). Finally, herbaceous understory vegetation was not included in this study. However, it has been found that this vegetation layer can significantly influence forest dynamics by competing for light with the tree regeneration layer (Thrippleton et al., 2016). Consequently, we encourage future research to take this effect into account.

Additionally, some aspects of the methodology of this study are subject to optimisation. We recognize that different stands of the Meerdaal and Heverlee forest are in different stages of the rotation cycle which means that thinnings happen at different moments in time for each stand. Due to the way the Meerdaal and Heverlee forest was initialized in iLand, leading to forest stands largely surpassing the rotation age, we could not plan the management activities relative to the rotation age. Consequently, thinnings were imposed on the whole forest complex at once at certain simulation times, which could underly the unsustainable harvest, rapid species dynamics, and resilience trends. Furthermore, the calculation of the resilience index was not optimal. Going back to Equation 1 S_R , depicting the initial value of the system variable, should ideally be the basal area in case the thinning did not happen at the moment of the thinning (Van Meerbeek et al., 2021) and should have been estimated by modelling the growth trend of the basal area. In this study, we used the basal area in the year before the thinning happened as S_R . We expected this method would be acceptable.

Additionally, we recognize that the variables used to assess the resistance and resilience of the forest could impact these metrics differently. Here, the question *resilience of what?* is of importance (Carpenter et al., 2001). Specifically, we looked at the resilience of the basal area of the forest, which was considered a meaningful system variable as it relates to the structure

of the forest as well as to a range of ecosystem services (e.g. timber harvest, microclimate, carbon storage, etc.). Nevertheless, looking at the resilience of more than one system variable (another variable could have been the LAI for example, as done in Seidl, Vigl, et al. (2017)) could have made our findings and conclusions more robust. Therefore, we encourage future research to look at more than one system variable when assessing resilience as well as resistance of a system. The same could apply on the disturbance to which resilience is assessed (*resilience to what?*, Carpenter et al., 2001), as here we looked at resilience after thinning interventions, but resilience could also be assessed after drought events for example.

Furthermore, looking at the species composition and diameter class distribution might not be ideal to assess resistance on, although this was based on previous studies assessing resistance of the forest against climate change (Duveneck & Scheller, 2016; Lucash et al., 2017). Specifically, it would be better to look at the resistance of variables linked to the functionality of the forest instead. Actually, species community and forest structure shifts may be inevitable in face of climate change and even necessary to maintain ecosystem services provisioning. A modelling study of Buma & Wessman (2013) found that for a forest in the USA no action as well as management aiming at maintaining a specific forest by the planting of the species currently present in the forest failed under climate change to preserve the forest, whereas management planting climate-suitable species preserved the forest and, thus, ecosystem services associated to the presence of a forest. Adaptive forest management may be necessary to avoid shifts to a non-forest, although it might imply the loss of a specific forest. Therefore, other variables better to assess resistance on, would be variables that are more desirable to keep at current conditions, such as the LAI (Seidl, Vigl, et al., 2017) or carbon stock (Buma & Wessman, 2013), but also the basal area, diameter or species diversity, growth rate, etc. might be suitable variables, and would correlate more with the functionality of the forest (delivering ecosystem services) than the identity of the forest (the species composition). For example, when looking at the basal area dynamics through time (Figure 12), the basal area is better preserved in the historical, moderate, and hot climate, whereas it drops in the hot and dry climate in every thinning scenario, which would indicate reduced resistance in the hot and dry climate and an increased potential to shift to non-forest. Actively planting climate-suitable species in the forest could mitigate the negative effect of this

extreme climate and better preserve the basal area, although this reduces the resistance of the species composition.

Lastly, we did not study the underlying variables influencing the resistance and resilience of the forest against climate change, as this was out of the scope of this thesis. Nevertheless, looking more in-depth to the drivers of resilience and resistance, especially for forests with a high species and structural diversity (in contrast to Seidl, Vigl, et al. (2017) for even-aged Norway spruce forests), could increase our understanding of how thinning can influence these stability metrics, and is therefore recommended towards future research.

5.7 Forest management implications

Increasing thinning intensities lowered the resistance of the species composition and the diameter class distribution against climate change. However, this shouldn't be viewed as an argument against higher thinning intensities. Much like the results of Thom et al. (2017), who found that disturbances can promote forest adaptation to climate change, higher thinning intensities can have similar results. Hence, thinnings could promote the conversion of a forest dominated by climate-sensitive species to climate-adapted species. This can be useful when considering the low rates of tree migration in relation to the rate of climatic changes (McLachlan et al., 2005). Especially in flat areas, where the velocity to stay at the same temperature is much higher than in mountainous areas (Loarie et al., 2009). The gradual conversion of species by thinning could also overcome the climate-induced tree species bottleneck, which is a recent finding suggesting that many species are not adapted to both current and future climate conditions, which narrows the species choice in forest management when considering the long lifetime of tree species (Wessely et al., 2024). Furthermore, Puettmann (2011) states that in face of global change, including climate change, and the large uncertainty associated with it, forests should be managed as complex adaptive systems where concepts such as adaptability and resilience stand central, rather than managing for resistance. Hence, in this regard, thinnings could be seen as a valuable management tool to promote species adaptation.

Nonetheless, there is a trade-off where this approach, leading to species adaptation, could lead to a lower productivity in terms of ecosystem services and goods. Although, at the other side, their delivery would be better assured towards the future as the forest is better adapted

to the prevailing climate (Puettmann, 2011). This is, for example, observable in the hot and dry climate where black pine became very abundant. Black pine is better adapted to droughts (Vacek et al., 2023), and consequently less prone to mortality in face of climate change. Nonetheless, this species may deliver less desirable goods and services. In Catalonia, García-Valdés et al. (2021) found, for example, that with climate change the species composition shifted towards more drought-tolerant species. However, this shift came along with a reduced forest productivity because of the lower growth of drought-tolerant species (García-Valdés et al., 2021), illustrating that this trade-off should therefore be kept in mind in forest management.

Furthermore, resilience of the basal area declined with increasing thinning intensity, meaning that the basal area recovered faster in the lower thinning intensity scenario. Towards forest management, this could indicate to keep thinning intensities low, although we should note that the thinnings might have been very intense in our landscape since they were imposed on the whole landscape at once. Consequently, the effect of intensity might have compensated for the effect of other variables influencing resilience, such as the age and basal area of the forest (Seidl, Vigl, et al., 2017). Therefore, we suggest that the relation between resilience and different thinning intensities should be further researched, as well as other thinning intensities besides the three thinning intensities studied here. In conclusion, even though higher thinning intensities led to an overall lower stability of the forest, it allowed for a faster conversion to drought-adapted species. We therefore recommend forest managers to apply a higher thinning intensity on a smaller scale, allowing drought-adapted species to establish while trying to maintain the overall stability of the forest.

A clear effect of climate on the resilience was identified and confirms the climate sensitivity of this stability metric (Seidl, Vigl, et al., 2017). This could be of interest for forest management, as climate might outweigh the effects of management on resilience, such as was observed in our most extreme climate change scenario. In the hot and dry climate, we observed no difference anymore between the thinning intensities, and resilience was overall low. Towards management this could highlight the importance of using more than one climate-smart measure to maintain resilience under extreme climate change (Duveneck & Scheller, 2016). These measures could include, for example, increasing species and functional diversity, as this measure was also found to increase resilience (Messier et al., 2022). Another

management technique could be to make rotation lengths shorter, which would lead to more generations of species and hence, an increased chance for genetic adaptation to the changing climate (Duveneck & Scheller, 2016). In this regard, to provide clear management recommendations, future research should also investigate the combination of multiple climate-smart measures on the stability of the forest under climate change.

6 Conclusion

In this modelling study we looked at the effect of thinning intensity on the resistance of the species composition and forest structure against climate change as well as the resilience under climate change. Therefore, we applied the mechanistic landscape and forest growth model, iLand, in combination with an agent-based management module for the first time in an Atlantic forest. We hypothesized that a higher thinning intensity would increase the stability of the forest against climate change, because it lowers between-tree competition and increases drought tolerance. However, we could not confirm this hypothesis. We found that increasing thinning intensity reduced both the resistance of the species composition as well as the diameter class distribution against climate change. Although the trend in the latter was less clear. Specifically, we observed an increase in the share of mainly oaks and Douglas fir in the landscape at the expense of beech with increasing thinning intensity. Secondly, we found that the climate could influence the resistance of the species composition as well as the diameter class distribution significantly, and consequently shift forest characteristics. Most often the hot and dry climate showed the lowest resistance, with a shift to a forest with more smaller trees and a larger abundance of black pine, Douglas fir, and native oaks. However, trends were less clear for the diameter class distribution, where the historical climate did not always differ in terms of resistance from the hot and dry climate. Lastly, the resilience of the basal area against the thinning events under climate change was found to become lower when thinning intensity increased. This could indicate that higher thinning intensities led to more difficulties for the basal area to recover rapidly from the intervention afterwards. However, climate significantly influenced resilience, and a hot and dry climate led mostly to an overall low resilience with no difference anymore between the thinning intensities, suggesting that more than one climate-smart measure could be necessary to increase resilience under extreme climatic conditions. Although our study could not fully address the questions related to higher thinning intensities, we were able to successfully initialize the mechanistic landscape model iLand in Flanders, allowing future studies to evaluate climate-smart forest management strategies more in depth. Specifically, we advise future studies to focus on (i) the underlying drivers of resistance and resilience, (ii) evaluate more dynamic planting and thinning strategies and rotation times and (iii) address the potential of the combination of climate-smart actions and more localised forest management.

7 References

- Aertsen, W., Janssen, E., Kint, V., Bontemps, J. D., Van Orshoven, J., & Muys, B. (2014). Long-term growth changes of common beech (*Fagus sylvatica* L.) are less pronounced on highly productive sites. *Forest Ecology and Management*, 312, 252–259. <https://doi.org/10.1016/j.foreco.2013.09.034>
- Agentschap voor Natuur en Bos Houtvesterij Leuven. (2006). *Bosbeheerplan: Meerdaalwoud - Heverleebos - Egenhovenbos*. https://www.natuurenbos.be/sites/default/files/beheerplan_heverleebos_meerdaalwoud.pdf
- Ainsworth, E. A., & Long, S. P. (2005). What have we learned from 15 years of free-air CO₂ enrichment (FACE)? A meta-analytic review of the responses of photosynthesis, canopy properties and plant production to rising CO₂. *New Phytologist*, 165(2), 351–372. <https://doi.org/10.1111/j.1469-8137.2004.01224.x>
- Albrich, K., Rammer, W., & Seidl, R. (2020). Climate change causes critical transitions and irreversible alterations of mountain forests. *Global Change Biology*, 26(7), 4013–4027. <https://doi.org/10.1111/gcb.15118>
- Albrich, K., Rammer, W., Thom, D., & Seidl, R. (2018). Trade-offs between temporal stability and level of forest ecosystem services provisioning under climate change. *Ecological Applications*, 28(7), 1884–1896. <https://doi.org/10.1002/eap.1785>
- Albrich, K., Rammer, W., Turner, M. G., Ratajczak, Z., Braziunas, K. H., Hansen, W. D., & Seidl, R. (2020). Simulating forest resilience: A review. *Global Ecology and Biogeography*, 29(12), 2082–2096. <https://doi.org/10.1111/geb.13197>
- Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D. D., Hogg, E. H. (Ted), Gonzalez, P., Fensham, R., Zhang, Z., Castro, J., Demidova, N., Lim, J. H., Allard, G., Running, S. W., Semerci, A., & Cobb, N. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, 259(4), 660–684. <https://doi.org/10.1016/j.foreco.2009.09.001>

- Baddeley, A., Rubak, E., & Turner, R. (2015). *Spatial Point Patterns: Methodology and Applications with R*. Chapman and Hall/CRC Press. <https://www.routledge.com/Spatial-Point-Patterns-Methodology-and-Applications-with-R/Baddeley-Rubak-Turner/p/book/9781482210200/>
- Baguis, P., Roulin, E., Willems, P., & Ntegeka, V. (2010). Climate change scenarios for precipitation and potential evapotranspiration over central Belgium. *Theoretical and Applied Climatology*, 99(3–4), 273–286. <https://doi.org/10.1007/s00704-009-0146-5>
- Bengtsson, H. (2023). *matrixStats: Functions that Apply to Rows and Columns of Matrices (and to Vectors)* (R package version 1.0.0). <https://CRAN.R-project.org/package=matrixStats>
- Birami, B., Nägele, T., Gattmann, M., Preisler, Y., Gast, A., Arneth, A., & Ruehr, N. K. (2020). Hot drought reduces the effects of elevated CO₂ on tree water-use efficiency and carbon metabolism. *New Phytologist*, 226(6), 1607–1621. <https://doi.org/10.1111/nph.16471>
- Bivand, R. S., Pebesma, E. J., & Gomez-Rubio, V. (2013). *Applied spatial data analysis with R* (2nd ed.). Springer. <https://asdar-book.org/>
- Bossel, H. (1996). TREEDYN3 forest simulation model. *Ecological Modelling*, 90(3), 187–227. [https://doi.org/10.1016/0304-3800\(95\)00139-5](https://doi.org/10.1016/0304-3800(95)00139-5)
- Bowditch, E., Santopuoli, G., Binder, F., del Río, M., La Porta, N., Kluvankova, T., Lesinski, J., Motta, R., Pach, M., Panzacchi, P., Pretzsch, H., Temperli, C., Tonon, G., Smith, M., Velikova, V., Weatherall, A., & Tognetti, R. (2020). What is Climate-Smart Forestry? A definition from a multinational collaborative process focused on mountain regions of Europe. *Ecosystem Services*, 43, 1–13. <https://doi.org/10.1016/j.ecoser.2020.101113>
- Bray, J. R., & Curtis, J. T. (1957). An Ordination of the Upland Forest Communities of Southern Wisconsin. *Ecological Monographs*, 27(4), 325–349. <https://doi.org/10.2307/1942268>

- Bréda, N., Granier, A., & Aussenac, G. (1995). Effects of thinning on soil and tree water relations, transpiration and growth in an oak forest (*Quercus petraea* (Matt.) Liebl.). *Tree Physiology*, 15(5), 295–306. <https://doi.org/10.1093/treephys/15.5.295>
- Buma, B., Brown, C. D., Donato, D. C., Fontaine, J. B., & Johnstone, J. F. (2013). The impacts of changing disturbance regimes on serotinous plant populations and communities. *BioScience*, 63(11), 866–876. <https://doi.org/10.1525/bio.2013.63.11.5>
- Buma, B., & Wessman, C. A. (2013). Forest resilience, climate change, and opportunities for adaptation: A specific case of a general problem. *Forest Ecology and Management*, 306, 216–225. <https://doi.org/10.1016/j.foreco.2013.06.044>
- Campioli, M., Vincke, C., Jonard, M., Kint, V., Demarée, G., & Ponette, Q. (2012). Current status and predicted impact of climate change on forest production and biogeochemistry in the temperate oceanic European zone: Review and prospects for Belgium as a case study. *Journal of Forest Research*, 17(1), 1–18. <https://doi.org/10.1007/s10310-011-0255-8>
- Cannon, A. J. (2018). Multivariate quantile mapping bias correction: an N-dimensional probability density function transform for climate model simulations of multiple variables. *Climate Dynamics*, 50(1–2), 31–49. <https://doi.org/10.1007/s00382-017-3580-6>
- Cantarello, E., Newton, A. C., Martin, P. A., Evans, P. M., Gosal, A., & Lucash, M. S. (2017). Quantifying resilience of multiple ecosystem services and biodiversity in a temperate forest landscape. *Ecology and Evolution*, 7(22), 9661–9675. <https://doi.org/10.1002/ece3.3491>
- Carpenter, S., Walker, B., Anderies, J. M., & Abel, N. (2001). From Metaphor to Measurement: Resilience of What to What? *Ecosystems*, 4(8), 765–781. <https://doi.org/10.1007/s10021-001-0045-9>
- Chase, C. W., Kimsey, M. J., Shaw, T. M., & Coleman, M. D. (2016). The response of light, water, and nutrient availability to pre-commercial thinning in dry inland Douglas-fir

- forests. *Forest Ecology and Management*, 363, 98–109.
<https://doi.org/10.1016/j.foreco.2015.12.014>
- Choat, B., Brodribb, T. J., Brodersen, C. R., Duursma, R. A., López, R., & Medlyn, B. E. (2018). Triggers of tree mortality under drought. *Nature*, 558(7711), 531–539.
<https://doi.org/10.1038/s41586-018-0240-x>
- Crosetto, M., & Tarantola, S. (2001). Uncertainty and sensitivity analysis: Tools for GIS-based model implementation. *International Journal of Geographical Information Science*, 15(5), 415–437. <https://doi.org/10.1080/13658810110053125>
- Dalelane, C., Früh, B., Steger, C., & Walter, A. (2018). A pragmatic approach to build a reduced regional climate projection ensemble for Germany using the EURO-CORDEX 8.5 ensemble. *Journal of Applied Meteorology and Climatology*, 57(3), 477–491.
<https://doi.org/10.1175/JAMC-D-17-0141.1>
- De Groote, S. R. E., Vanhellemont, M., Baeten, L., Van den Bulcke, J., Martel, A., Bonte, D., Lens, L., & Verheyen, K. (2018). Competition, tree age and size drive the productivity of mixed forests of pedunculate oak, beech and red oak. *Forest Ecology and Management*, 430, 609–617. <https://doi.org/10.1016/j.foreco.2018.08.050>
- De Vos, B. (2009). *Uncertainties of forest soil carbon stock assessment in Flanders (Unpublished doctoral thesis)*. KU Leuven.
- del Río, M., Bravo-Oviedo, A., Ruiz-Peinado, R., & Condés, S. (2019). Tree allometry variation in response to intra- and inter-specific competitions. *Trees*, 33(1), 121–138.
<https://doi.org/10.1007/s00468-018-1763-3>
- Diaconu, D., Kahle, H. P., & Spiecker, H. (2017). Thinning increases drought tolerance of European beech: a case study on two forested slopes on opposite sides of a valley. *European Journal of Forest Research*, 136(2), 319–328.
<https://doi.org/10.1007/s10342-017-1033-8>
- Dobor, L., Hlásny, T., & Zimová, S. (2020). Contrasting vulnerability of monospecific and species-diverse forests to wind and bark beetle disturbance: The role of management. *Ecology and Evolution*, 10(21), 12233–12245. <https://doi.org/10.1002/ece3.6854>

- Dollinger, C., Rammer, W., & Seidl, R. (2023). Climate change accelerates ecosystem restoration in the mountain forests of Central Europe. *Journal of Applied Ecology*, 60(12), 2665–2675. <https://doi.org/10.1111/1365-2664.14520>
- Donohue, I., Hillebrand, H., Montoya, J. M., Petchey, O. L., Pimm, S. L., Fowler, M. S., Healy, K., Jackson, A. L., Lurgi, M., McClean, D., O'Connor, N. E., O'Gorman, E. J., & Yang, Q. (2016). Navigating the complexity of ecological stability. *Ecology Letters*, 19(9), 1172–1185. <https://doi.org/10.1111/ele.12648>
- Drake, B. G., González-Meler, M. A., & Long, S. P. (1997). MORE EFFICIENT PLANTS: A Consequence of Rising Atmospheric CO₂? *Annual Review of Plant Biology*, 48, 609–648. <https://doi.org/10.1146/annurev.arplant.48.1.609>
- Ducousso, A., & Bordacs, S. (2004). *EUFORGEN Technical guidelines for genetic conservation and use for Pedunculate and sessile oaks (Quercus robur and Q. petraea)*. International Plant Genetic Resources Institute. https://www.euforgen.org/fileadmin//templates/euforgen.org/upload/Publications/Technical_guidelines/Technical_guidelines_Quercus_robur-petraea.pdf
- Duveneck, M. J., & Scheller, R. M. (2016). Measuring and managing resistance and resilience under climate change in northern Great Lake forests (USA). *Landscape Ecology*, 31(3), 669–686. <https://doi.org/10.1007/s10980-015-0273-6>
- Dyderski, M. K., Chmura, D., Dylewski, Ł., Horodecki, P., Jagodziński, A. M., Pietras, M., Robakowski, P., & Woziwoda, B. (2020). Biological Flora of the British Isles: *Quercus rubra*. *Journal of Ecology*, 108(3), 1199–1225. <https://doi.org/10.1111/1365-2745.13375>
- Eilmann, B., & Rigling, A. (2012). Tree-growth analyses to estimate tree species' drought tolerance. *Tree Physiology*, 32(2), 178–187. <https://doi.org/10.1093/treephys/tps004>
- Enescu, C. M., De Rigo, D., Caudullo, G., Mauri, A., & Houston Durrant, T. (2016). *Pinus nigra* in Europe: distribution, habitat, usage and threats. In J. San-Miguel-Ayanz, D. de Rigo, G. Caudullo, T. Houston Durrant, & A. Mauri (Eds.), *European Atlas of Forest Tree Species* (pp. 126–127). Publ. Off. www.euforgen.org.

- European Environment Agency. (2016). *European forest ecosystems: State and trends* (Report No. 5/2016). Publications Office of the European Union.
<https://doi.org/10.2800/964893>
- Fady, B., Aravanopoulos, F. A., Alizoti, P., Mátyás, C., von Wühlisch, G., Westergren, M., Belletti, P., Cvjetkovic, B., Ducci, F., Huber, G., Kelleher, C. T., Khaldi, A., Kharrat, M. B. D., Kraigher, H., Kramer, K., Mühlethaler, U., Peric, S., Perry, A., Rousi, M., ... Zlatanov, T. (2016). Evolution-based approach needed for the conservation and silviculture of peripheral forest tree populations. *Forest Ecology and Management*, 375, 66–75.
<https://doi.org/10.1016/j.foreco.2016.05.015>
- FAO. (2020a). *Global Forest Resources Assessment 2020 - Key findings*.
<https://doi.org/10.4060/ca8753en>
- FAO. (2020b). *Global Forest Resources Assessment 2020: Main report*.
<https://doi.org/10.4060/ca9825en>
- FAO. (2023). *Forestry Production and Trade*.
<https://www.fao.org/faostat/en/#data/FO/visualize>
- Forest Europe. (2020). *State of Europe's Forests 2020*. www.foresteurope.org
- Fortin, M., Van Couwenberghe, R., Perez, V., & Piedallu, C. (2019). Evidence of climate effects on the height-diameter relationships of tree species. *Annals of Forest Science*, 76(1), 1–20. <https://doi.org/10.1007/s13595-018-0784-9>
- Forzieri, G., Dakos, V., McDowell, N. G., Ramdane, A., & Cescatti, A. (2022). Emerging signals of declining forest resilience under climate change. *Nature*, 608(7923), 534–539.
<https://doi.org/10.1038/s41586-022-04959-9>
- Garcia-Soto, C., Cheng, L., Caesar, L., Schmidtke, S., Jewett, E. B., Cheripka, A., Rigor, I., Caballero, A., Chiba, S., Báez, J. C., Zielinski, T., & Abraham, J. P. (2021). An Overview of Ocean Climate Change Indicators: Sea Surface Temperature, Ocean Heat Content, Ocean pH, Dissolved Oxygen Concentration, Arctic Sea Ice Extent, Thickness and Volume, Sea Level and Strength of the AMOC (Atlantic Meridional Overturning

- Circulation). *Frontiers in Marine Science*, 8, 1–24.
<https://doi.org/10.3389/fmars.2021.642372>
- García-Valdés, R., Vayreda, J., Retana, J., & Martínez-Vilalta, J. (2021). Low forest productivity associated with increasing drought-tolerant species is compensated by an increase in drought-tolerance richness. *Global Change Biology*, 27(10), 2113–2127.
<https://doi.org/10.1111/gcb.15529>
- Giraud, T., & Commenges, H. (2022). *potential: Implementation of the Potential Model* (R package version 0.2.0). <https://CRAN.R-project.org/package=potential>
- Gonçalves, A. C. (2021). Thinning: An Overview. In A. C. Gonçalves (Ed.), *Silviculture* (pp. 41–58). IntechOpen. <https://doi.org/10.5772/intechopen.93436>
- Govaere, L. (2020). Een blik op de kenmerken van bos in Vlaanderen – eerste resultaten van twee opeenvolgende Vlaamse bosinventarisaties. *Bosrevue*, 83a, 1–14.
https://bosplus.be/wp-content/uploads/2022/02/bosrevue83aVlaamse_Bosinventarisaties_3.pdf
- Gradel, A., Ammer, C., Ganbaatar, B., Nadaldorj, O., Dovdondemberel, B., & Wagner, S. (2017). On the effect of thinning on tree growth and stand structure of white birch (*Betula platyphylla* Sukaczew.) and Siberian larch (*Larix sibirica* Ledeb.) in Mongolia. *Forests*, 8(4), 1–23. <https://doi.org/10.3390/f8040105>
- Gravel, D., Canham, C. D., Beaudet, M., & Messier, C. (2010). Shade tolerance, canopy gaps and mechanisms of coexistence of forest trees. *Oikos*, 119(3), 475–484.
<https://doi.org/10.1111/j.1600-0706.2009.17441.x>
- Guillemot, J., Klein, E. K., Davi, H., & Courbet, F. (2015). The effects of thinning intensity and tree size on the growth response to annual climate in *Cedrus atlantica*: a linear mixed modeling approach. *Annals of Forest Science*, 72(5), 651–663.
<https://doi.org/10.1007/s13595-015-0464-y>
- Harris, N. L., Gibbs, D. A., Baccini, A., Birdsey, R. A., de Bruin, S., Farina, M., Fatoyinbo, L., Hansen, M. C., Herold, M., Houghton, R. A., Potapov, P. V., Suarez, D. R., Roman-Cuesta, R. M., Saatchi, S. S., Slay, C. M., Turubanova, S. A., & Tyukavina, A. (2021).

- Global maps of twenty-first century forest carbon fluxes. *Nature Climate Change*, 11(3), 234–240. <https://doi.org/10.1038/s41558-020-00976-6>
- He, H. S., Yang, J., Shifley, S. R., & Thompson, F. R. (2011). Challenges of forest landscape modeling-Simulating large landscapes and validating results. *Landscape and Urban Planning*, 100(4), 400–402. <https://doi.org/10.1016/j.landurbplan.2011.02.019>
- Hernández-Morcillo, M., Torralba, M., Baiges, T., Bernasconi, A., Bottaro, G., Brogaard, S., Bussola, F., Díaz-Varela, E., Geneletti, D., Grossmann, C. M., Kister, J., Klingler, M., Loft, L., Lovric, M., Mann, C., Pipart, N., Roces-Díaz, J. V., Sorge, S., Tiebel, M., ... Plieninger, T. (2022). Scanning the solutions for the sustainable supply of forest ecosystem services in Europe. *Sustainability Science*, 17(5), 2013–2029. <https://doi.org/10.1007/s11625-022-01111-4>
- Hijmans, R. J. (2023a). *raster: Geographic Data Analysis and Modeling* (R package version 3.6-14). <https://CRAN.R-project.org/package=raster>
- Hijmans, R. J. (2023b). *terra: Spatial Data Analysis* (R package version 1.7-3). <https://CRAN.R-project.org/package=terra>
- Hulshof, C. M., Swenson, N. G., & Weiser, M. D. (2015). Tree height-diameter allometry across the United States. *Ecology and Evolution*, 5(6), 1193–1204. <https://doi.org/10.1002/ece3.1328>
- IMBIE. (2018). Mass balance of the Antarctic Ice Sheet from 1992 to 2017. *Nature*, 558(7709), 219–222. <https://doi.org/10.1038/s41586-018-0179-y>
- INBO. (2023). *Oppervlakte met effectief natuurbeheer*. <https://www.vlaanderen.be/inbo/indicatoren/oppervlakte-met-effectief-natuurbeheer>
- IPCC. (2014). *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (Core Writing Team, R. K. Pachauri, & L. A. Meyer, Eds.). IPCC.
- IPCC. (2023a). Section 2: Current Status and Trends. In Core Writing Team, H. Lee, & J. Romero (Eds.), *Climate Change 2023: Synthesis Report. Contribution of Working Groups*

I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change (pp. 35–115). <https://doi.org/10.59327/IPCC/AR6-9789291691647>

IPCC. (2023b). Section 3: Long-Term Climate and Development Futures. In Core Writing Team, H. Lee, & J. Romero (Eds.), *Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 35–115). IPCC. <https://doi.org/10.59327/IPCC/AR6-9789291691647>

Jacob, D., Petersen, J., Eggert, B., Alias, A., Christensen, O. B., Bouwer, L. M., Braun, A., Colette, A., Déqué, M., Georgievski, G., Georgopoulou, E., Gobiet, A., Menut, L., Nikulin, G., Haensler, A., Hempelmann, N., Jones, C., Keuler, K., Kovats, S., ... Yiou, P. (2014). EURO-CORDEX: New high-resolution climate change projections for European impact research. *Regional Environmental Change*, 14(2), 563–578. <https://doi.org/10.1007/s10113-013-0499-2>

Jeong, S.-J., Ho, C.-H., Gim, H.-J., & Brown, M. E. (2011). Phenology shifts at start vs. end of growing season in temperate vegetation over the Northern Hemisphere for the period 1982-2008. *Global Change Biology*, 17(7), 2385–2399. <https://doi.org/10.1111/j.1365-2486.2011.02397.x>

Jiao-Jun, Z., Feng-Qin, L. I., Yutaka, G., Matsuzaki, T., & Yamamoto, M. (2003). Effects of thinning on wind damage in *Pinus thunbergii* plantation-Based on theoretical derivation of risk-ratios for assessing wind damage. *Journal of Forestry Research*, 14(1), 1–8. <https://doi.org/10.1007/BF02856755>

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G. L. W., Schoennagel, T., & Turner, M. G. (2016). Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment*, 14(7), 369–378. <https://doi.org/10.1002/fee.1311>

Justus, J. (2007). Complexity, diversity, and stability. In S. Sahotra & A. Plutynski (Eds.), *A Companion to the Philosophy of Biology* (pp. 321–350). Blackwell Publishing. <https://doi.org/10.1002/9780470696590.ch18>

- Keane, R. E., Loehman, R. A., Holsinger, L. M., Falk, D. A., Higuera, P., Hood, S. M., & Hessburg, P. F. (2018). Use of landscape simulation modeling to quantify resilience for ecological applications. *Ecosphere*, 9(9), 1–19. <https://doi.org/10.1002/ecs2.2414>
- Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, 352, 9–20. <https://doi.org/10.1016/j.foreco.2015.06.014>
- Keskitalo, H., & Carina, E. (2011). How can forest management adapt to climate change? Possibilities in different forestry systems. *Forests*, 2(1), 415–430. <https://doi.org/10.3390/f2010415>
- Khan, S. A., Aschwanden, A., Bjørk, A. A., Wahr, J., Kjeldsen, K. K., & Kjær, K. H. (2015). Greenland ice sheet mass balance: A review. *Reports on Progress in Physics*, 78(4), 1–26. <https://doi.org/10.1088/0034-4885/78/4/046801>
- Kimmins, J. P., Mailly, D., & Seely, B. (1999). Modelling forest ecosystem net primary production: the hybrid simulation approach used in FORECAST. *Ecological Modelling*, 122(3), 195–224. [https://doi.org/10.1016/S0304-3800\(99\)00138-6](https://doi.org/10.1016/S0304-3800(99)00138-6)
- Kint, V., Aertsens, W., Campioli, M., Vansteenkiste, D., Delcloo, A., & Muys, B. (2012). Radial growth change of temperate tree species in response to altered regional climate and air quality in the period 1901–2008. *Climatic Change*, 115(2), 343–363. <https://doi.org/10.1007/s10584-012-0465-x>
- Klingen, S., Mohren, F., Geudens, G., & Den Ouden, J. (2010). Dunning. In J. den Ouden, B. Muys, F. Mohren, & K. Verheyen (Eds.), *Bosecologie en bosbeheer* (pp. 375–387). Acco.
- Kohler, M., Sohn, J., Nägele, G., & Bauhus, J. (2010). Can drought tolerance of Norway spruce (*Picea abies* (L.) Karst.) be increased through thinning? *European Journal of Forest Research*, 129(6), 1109–1118. <https://doi.org/10.1007/s10342-010-0397-9>
- Lamont, B. B., & Enright, N. J. (2000). Adaptive advantages of aerial seed banks. *Plant Species Biology*, 15(2), 157–166. <https://doi.org/10.1046/j.1442-1984.2000.00036.x>

- Lamont, B. B., He, T., & Yan, Z. (2019). Evolutionary history of fire-stimulated resprouting, flowering, seed release and germination. *Biological Reviews*, 94(3), 903–928.
<https://doi.org/10.1111/brv.12483>
- Landsberg, J. (2003). Physiology in forest models: History and the future. *FBMIS*, 1, 49–63.
http://www.fbmis.info/A/3_1_LandsbergJ_1
- Landsberg, J., & Sands, P. (2011a). Introduction. In *Physiological ecology of forest production: principles, processes and models* (1st ed., pp. 2–12). Elsevier.
[https://doi.org/10.1016/S1936-7961\(11\)04001-2](https://doi.org/10.1016/S1936-7961(11)04001-2)
- Landsberg, J., & Sands, P. (2011b). Modelling Tree Growth: Concepts and Review. In *Physiological ecology of forest production: principles, processes and models* (1st ed., pp. 222–239). Elsevier. [https://doi.org/10.1016/S1936-7961\(11\)04008-5](https://doi.org/10.1016/S1936-7961(11)04008-5)
- Landsberg, J., & Waring, R. H. (1997). A generalised model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. *Forest Ecology and Management*, 95(3), 209–228. [https://doi.org/10.1016/S0378-1127\(97\)00026-1](https://doi.org/10.1016/S0378-1127(97)00026-1)
- Langenbruch, C., Helfrich, M., & Flessa, H. (2012). Effects of beech (*Fagus sylvatica*), ash (*Fraxinus excelsior*) and lime (*Tilia spec.*) on soil chemical properties in a mixed deciduous forest. *Plant and Soil*, 352(1–2), 389–403. <https://doi.org/10.1007/s11104-011-1004-7>
- Latte, N., Perin, J., Kint, V., Lebourgeois, F., & Claessens, H. (2016). Major changes in growth rate and growth variability of beech (*Fagus sylvatica* L.) related to soil alteration and climate change in Belgium. *Forests*, 7(8), 1–15. <https://doi.org/10.3390/f7080174>
- Leuschner, C. (2020). Drought response of European beech (*Fagus sylvatica* L.)—A review. *Perspectives in Plant Ecology, Evolution and Systematics*, 47, 1–22.
<https://doi.org/10.1016/j.ppees.2020.125576>
- Lévesque, M., Rigling, A., Bugmann, H., Weber, P., & Brang, P. (2014). Growth response of five co-occurring conifers to drought across a wide climatic gradient in Central Europe.

Agricultural and Forest Meteorology, 197, 1–12.

<https://doi.org/10.1016/j.agrformet.2014.06.001>

Lévy, G., Becker, M., & Duhamel, D. (1992). A comparison of the ecology of pedunculate and sessile oaks: radial growth in the centre and northwest of France. *Forest Ecology and Management*, 55(1), 51–63. [https://doi.org/10.1016/0378-1127\(92\)90091-M](https://doi.org/10.1016/0378-1127(92)90091-M)

Lindner, M., Maroschek, M., Netherer, S., Kremer, A., Barbati, A., Garcia-Gonzalo, J., Seidl, R., Delzon, S., Corona, P., Kolström, M., Lexer, M. J., & Marchetti, M. (2010). Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. *Forest Ecology and Management*, 259(4), 698–709. <https://doi.org/10.1016/j.foreco.2009.09.023>

Loarie, S. R., Duffy, P. B., Hamilton, H., Asner, G. P., Field, C. B., & Ackerly, D. D. (2009). The velocity of climate change. *Nature*, 462(7276), 1052–1055. <https://doi.org/10.1038/nature08649>

Lucash, M. S., Scheller, R. M., J. Gustafson, E., & R. Sturtevant, B. (2017). Spatial resilience of forested landscapes under climate change and management. *Landscape Ecology*, 32(5), 953–969. <https://doi.org/10.1007/s10980-017-0501-3>

Ma, S., Concilio, A., Oakley, B., North, M., & Chen, J. (2010). Spatial variability in microclimate in a mixed-conifer forest before and after thinning and burning treatments. *Forest Ecology and Management*, 259(5), 904–915. <https://doi.org/10.1016/j.foreco.2009.11.030>

Mäkelä, A., Pulkkinen, M., Kolari, P., Lagergren, F., Berbigier, P., Lindroth, A., Loustau, D., Nikinmaa, E., Vesala, T., & Hari, P. (2008). Developing an empirical model of stand GPP with the LUE approach: Analysis of eddy covariance data at five contrasting conifer sites in Europe. *Global Change Biology*, 14(1), 92–108. <https://doi.org/10.1111/j.1365-2486.2007.01463.x>

Manrique-Alba, À., Beguería, S., & Camarero, J. J. (2022). Long-term effects of forest management on post-drought growth resilience: An analytical framework. *Science of the Total Environment*, 810, 1–13. <https://doi.org/10.1016/j.scitotenv.2021.152374>

- Mátyás, C., Ackzell, L., & Samuel, C. J. A. (2004). *EUFORGEN Technical guidelines for genetic conservation and use for Scots pine (Pinus sylvestris)*. International Plant Genetic Resources Institute.
https://www.euforgen.org/fileadmin/templates/euforgen.org/upload/Publications/Technical_guidelines/Technical_guidelines_Pinus_sylvestris.pdf
- McDowell, N. G., Allen, C. D., Anderson-Teixeira, K., Aukema, B. H., Bond-Lamberty, B., Chini, L., Clark, J. S., Dietze, M., Grossiord, C., Hanbury-Brown, A., Hurtt, G. C., Jackson, R. B., Johnson, D. J., Kueppers, L., Lichstein, J. W., Ogle, K., Poulter, B., Pugh, T. A. M., Seidl, R., ... Xu, C. (2020). Pervasive shifts in forest dynamics in a changing world. *Science*, 368(6494), 1–10. <https://doi.org/10.1126/science.aaz9463>
- McLachlan, J. S., Clark, J. S., & Manos, P. S. (2005). Molecular indicators of tree migration capacity under rapid climate change. *Ecology*, 86(8), 2088–2098.
<https://doi.org/10.1890/04-1036>
- Messier, C., Bauhus, J., Sousa-Silva, R., Auge, H., Baeten, L., Barsoum, N., Bruelheide, H., Caldwell, B., Cavender-Bares, J., Dhiedt, E., Eisenhauer, N., Ganade, G., Gravel, D., Guillemot, J., Hall, J. S., Hector, A., Hérault, B., Jactel, H., Koricheva, J., ... Zemp, D. C. (2022). For the sake of resilience and multifunctionality, let's diversify planted forests! *Conservation Letters*, 15(1), 1–8. <https://doi.org/10.1111/conl.12829>
- Mina, M., Messier, C., Duveneck, M. J., Fortin, M. J., & Aquilué, N. (2022). Managing for the unexpected: Building resilient forest landscapes to cope with global change. *Global Change Biology*, 28(14), 4323–4341. <https://doi.org/10.1111/gcb.16197>
- Misson, L., Vincke, C., & Devillez, F. (2003). Frequency responses of radial growth series after different thinning intensities in Norway spruce (*Picea abies* (L.) Karst.) stands. *Forest Ecology and Management*, 177(1), 51–63. [https://doi.org/10.1016/S0378-1127\(02\)00324-9](https://doi.org/10.1016/S0378-1127(02)00324-9)
- Müller, K., Wickham, H., James, D., & Falcon, S. (2023). *RSQLite: SQLite Interface for R* (R package version 2.3.1). <https://CRAN.R-project.org/package=RSQLite>

- Nabuurs, G. J., Delacote, P., Ellison, D., Hanewinkel, M., Hetemäki, L., Lindner, M., & Ollikainen, M. (2017). By 2050 the mitigation effects of EU forests could nearly double through climate smart forestry. *Forests*, 8(12), 1–14. <https://doi.org/10.3390/f8120484>
- Niinemets, Ü., & Valladares, F. (2006). Tolerance to shade, drought, and waterlogging of temperate northern hemisphere trees and shrubs. *Ecological Monographs*, 76(4), 521–547. [https://doi.org/10.1890/0012-9615\(2006\)076\[0521:TTSDAW\]2.0.CO;2](https://doi.org/10.1890/0012-9615(2006)076[0521:TTSDAW]2.0.CO;2)
- Nikinmaa, L., Lindner, M., Cantarello, E., Jump, A. S., Seidl, R., Winkel, G., & Muys, B. (2020). Reviewing the Use of Resilience Concepts in Forest Sciences. *Current Forestry Reports*, 6(2), 61–80. <https://doi.org/10.1007/s40725-020-00110-x>
- Niklas, K. J. (1995). Size-dependent Allometry of Tree Height, Diameter and Trunk-taper. *Annals of Botany*, 75(3), 217–227. <https://doi.org/10.1006/anbo.1995.1015>
- Nolan, C., Overpeck, J. T., Allen, J. R. M., Anderson, P. M., Betancourt, J. L., Binney, H. A., Brewer, S., Bush, M. B., Chase, B. M., Cheddadi, R., Djamali, M., Dodson, J., Edwards, M. E., Gosling, W. D., Haberle, S., Hotchkiss, S. C., Huntley, B., Ivory, S. J., Kershaw, A. P., ... Jackson, S. T. (2018). Past and future global transformation of terrestrial ecosystems under climate change. *Science*, 361(6405), 920–923. <https://doi.org/10.1126/science.aan5360>
- Norby, R. J., Warren, J. M., Iversen, C. M., Medlyn, B. E., & McMurtrie, R. E. (2010). CO₂ enhancement of forest productivity constrained by limited nitrogen availability. *Proceedings of the National Academy of Sciences of the United States of America*, 107(45), 19368–19373. <https://doi.org/10.1073/pnas.1006463107>
- Norby, R. J., Wullschleger, S. D., Gunderson, C. A., Johnson, D. W., & Ceulemans, R. (1999). Tree responses to rising CO₂ in field experiments: Implications for the future forest. *Plant, Cell and Environment*, 22(6), 683–714. <https://doi.org/10.1046/j.1365-3040.1999.00391.x>
- Obermeier, W. A., Lehnert, L. W., Kammann, C. I., Müller, C., Grünhage, L., Luterbacher, J., Erbs, M., Moser, G., Seibert, R., Yuan, N., & Bendix, J. (2017). Reduced CO₂ fertilization

- effect in temperate C3 grasslands under more extreme weather conditions. *Nature Climate Change*, 7(2), 137–141. <https://doi.org/10.1038/nclimate3191>
- Ogle, D. H., Doll, J. C., Wheeler, A. P., & Dinno, A. (2023). *FSA: Simple Fisheries Stock Assessment Methods* (R package version 0.9.4). <https://CRAN.R-project.org/package=FSA>
- Oksanen, J., Simpson, G., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., O'Hara, R., Solymos, P., Stevens, M., Szoecs, E., Wagner, H., Barbour, M., Bedward, M., Bolker, B., Borcard, D., Carvalho, G., Chirico, M., De Caceres, M., Durand, S., ... Weedon, J. (2022). *vegan: Community Ecology Package* (R package version 2.6-4). <https://CRAN.R-project.org/package=vegan>
- Packham, J. R., Thomas, P. A., Atkinson, M. D., & Degen, T. (2012). Biological Flora of the British Isles: *Fagus sylvatica*. *Journal of Ecology*, 100(6), 1557–1608. <https://doi.org/10.1111/j.1365-2745.2012.02017.x>
- Patacca, M., Lindner, M., Lucas-Borja, M. E., Cordonnier, T., Fidej, G., Gardiner, B., Hauf, Y., Jasinevičius, G., Labonne, S., Linkevičius, E., Mahnken, M., Milanovic, S., Nabuurs, G. J., Nagel, T. A., Nikinmaa, L., Panyatov, M., Bercak, R., Seidl, R., Ostrogović Sever, M. Z., ... Schelhaas, M. J. (2023). Significant increase in natural disturbance impacts on European forests since 1950. *Global Change Biology*, 29(5), 1359–1376. <https://doi.org/10.1111/gcb.16531>
- Pebesma, E. J. (2018). Simple Features for R: Standardized Support for Spatial Vector Data. *The R Journal*, 10(1), 439–446. <https://doi.org/10.32614/RJ-2018-009>
- Pebesma, E. J., & Bivand, R. S. (2005). Classes and methods for spatial data in R. *R News*, 5(2). <https://cran.r-project.org/doc/Rnews/>
- Peñuelas, J., Ciais, P., Canadell, J. G., Janssens, I. A., Fernández-Martínez, M., Carnicer, J., Obersteiner, M., Piao, S., Vautard, R., & Sardans, J. (2017). Shifting from a fertilization-dominated to a warming-dominated period. *Nature Ecology and Evolution*, 1(10), 1438–1445. <https://doi.org/10.1038/s41559-017-0274-8>

- Perry, G. L. W., & Millington, J. D. A. (2008). Spatial modelling of succession-disturbance dynamics in forest ecosystems: Concepts and examples. *Perspectives in Plant Ecology, Evolution and Systematics*, 9(3–4), 191–210.
<https://doi.org/10.1016/j.ppees.2007.07.001>
- Picchio, R., Tavankar, F., Latterini, F., Jourgholami, M., Marian, B. K., & Venanzi, R. (2020). Influence of different thinning treatments on stand resistance to snow and wind in loblolly pine (*Pinus taeda* L.) coastal plantations of northern Iran. *Forests*, 11(10), 1–14.
<https://doi.org/10.3390/f11101034>
- Powers, R. F. (1990). Nitrogen Mineralization Along an Altitudinal Gradient: Interactions of Soil Temperature, Moisture, and Substrate Quality. *Forest Ecology and Management*, 30(1), 19–29. [https://doi.org/10.1016/0378-1127\(90\)90123-S](https://doi.org/10.1016/0378-1127(90)90123-S)
- Puettmann, K. J. (2011). Silvicultural Challenges and Options in the Context of Global Change: “Simple” Fixes and Opportunities for New Management Approaches. *Journal of Forestry*, 109(6), 321–331. <https://doi.org/10.1093/jof/109.6.321>
- R Core Team. (2022). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. <https://www.R-project.org/>
- R Special Interest Group on Databases (R-SIG-DB), Wickham, H., & Müller, K. (2022). *DBI: R Database Interface* (R package version 1.1.3). <https://CRAN.R-project.org/package=DBI>
- Rambo, T. R., & North, M. P. (2009). Canopy microclimate response to pattern and density of thinning in a Sierra Nevada forest. *Forest Ecology and Management*, 257(2), 435–442. <https://doi.org/10.1016/j.foreco.2008.09.029>
- Rammer, W., & Seidl, R. (2015). Coupling human and natural systems: Simulating adaptive management agents in dynamically changing forest landscapes. *Global Environmental Change*, 35, 475–485. <https://doi.org/10.1016/j.gloenvcha.2015.10.003>
- Reich, P. B., & Hobbie, S. E. (2013). Decade-long soil nitrogen constraint on the CO₂ fertilization of plant biomass. *Nature Climate Change*, 3(3), 278–282.
<https://doi.org/10.1038/nclimate1694>

- Reyer, C. P. O., Bathgate, S., Blennow, K., Borges, J. G., Bugmann, H., Delzon, S., Faias, S. P., Garcia-Gonzalo, J., Gardiner, B., Gonzalez-Olabarria, J. R., Gracia, C., Hernández, J. G., Kellomäki, S., Kramer, K., Lexer, M. J., Lindner, M., Van Der Maaten, E., Maroschek, M., Muys, B., ... Hanewinkel, M. (2017). Are forest disturbances amplifying or canceling out climate change-induced productivity changes in European forests? *Environmental Research Letters*, 12(3), 1–12. <https://doi.org/10.1088/1748-9326/aa5ef1>
- Richter, S., Kipfer, T., Wohlgemuth, T., Guerrero, C. C., Ghazoul, J., & Moser, B. (2012). Phenotypic plasticity facilitates resistance to climate change in a highly variable environment. *Oecologia*, 169(1), 269–279. <https://doi.org/10.1007/s00442-011-2191-x>
- Rummukainen, M. (2012). Changes in climate and weather extremes in the 21st century. *Wiley Interdisciplinary Reviews: Climate Change*, 3(2), 115–129. <https://doi.org/10.1002/wcc.160>
- Salvatore, S. M. (2024). *rcompanion: Functions to Support Extension Education Program Evaluation* (version 2.4.35). Rutgers Cooperative Extension. <https://CRAN.R-project.org/package=rcompanion>
- Sands, P. J., & Landsberg, J. J. (2002). Parameterisation of 3-PG for plantation grown *Eucalyptus globulus*. *Forest Ecology and Management*, 163(1), 273–292. [https://doi.org/10.1016/S0378-1127\(01\)00586-2](https://doi.org/10.1016/S0378-1127(01)00586-2)
- Scharnweber, T., Manthey, M., Criegee, C., Bauwe, A., Schröder, C., & Wilmking, M. (2011). Drought matters - Declining precipitation influences growth of *Fagus sylvatica* L. and *Quercus robur* L. in north-eastern Germany. *Forest Ecology and Management*, 262(6), 947–961. <https://doi.org/10.1016/j.foreco.2011.05.026>
- Scheirer, C. J., Ray, W. S., & Hare, N. (1976). The Analysis of Ranked Data Derived from Completely Randomized Factorial Designs. *Biometrics*, 32(2), 429–434. <https://doi.org/10.2307/2529511>
- Schmitt, A., Trouvé, R., Seynave, I., & Lebourgeois, F. (2020). Decreasing stand density favors resistance, resilience, and recovery of *Quercus petraea* trees to a severe drought,

- particularly on dry sites. *Annals of Forest Science*, 77(2), 1–21.
<https://doi.org/10.1007/s13595-020-00959-9>
- Seidl, R. (2014). The shape of ecosystem management to come: Anticipating risks and fostering resilience. *BioScience*, 64(12), 1159–1169.
<https://doi.org/10.1093/biosci/biu172>
- Seidl, R., & Rammer, W. (2017). Climate change amplifies the interactions between wind and bark beetle disturbances in forest landscapes. *Landscape Ecology*, 32(7), 1485–1498. <https://doi.org/10.1007/s10980-016-0396-4>
- Seidl, R., Rammer, W., & Blennow, K. (2014). Simulating wind disturbance impacts on forest landscapes: Tree-level heterogeneity matters. *Environmental Modelling and Software*, 51, 1–11. <https://doi.org/10.1016/j.envsoft.2013.09.018>
- Seidl, R., Rammer, W., Scheller, R. M., & Spies, T. A. (2012). An individual-based process model to simulate landscape-scale forest ecosystem dynamics. *Ecological Modelling*, 231, 87–100. <https://doi.org/10.1016/j.ecolmodel.2012.02.015>
- Seidl, R., Rammer, W., & Spies, T. A. (2014). Disturbance legacies increase the resilience of forest ecosystem structure, composition, and functioning. *Ecological Applications*, 24(8), 2063–2077. <https://doi.org/10.1890/14-0255.1>
- Seidl, R., Schelhaas, M. J., & Lexer, M. J. (2011). Unraveling the drivers of intensifying forest disturbance regimes in Europe. *Global Change Biology*, 17(9), 2842–2852.
<https://doi.org/10.1111/j.1365-2486.2011.02452.x>
- Seidl, R., Spies, T. A., Rammer, W., Steel, E. A., Pabst, R. J., & Olsen, K. (2012). Multi-scale Drivers of Spatial Variation in Old-Growth Forest Carbon Density Disentangled with Lidar and an Individual-Based Landscape Model. *Ecosystems*, 15(8), 1321–1335.
<https://doi.org/10.1007/s10021-012-9587-2>
- Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M. J., Trotsiuk, V., Mairota, P., Svoboda, M., Fabrika, M., Nagel, T. A., & Reyer, C. P. O. (2017). Forest disturbances under climate change. *Nature Climate Change*, 7(6), 395–402. <https://doi.org/10.1038/nclimate3303>

- Seidl, R., Vigl, F., Rössler, G., Neumann, M., & Rammer, W. (2017). Assessing the resilience of Norway spruce forests through a model-based reanalysis of thinning trials. *Forest Ecology and Management*, 388, 3–12. <https://doi.org/10.1016/j.foreco.2016.11.030>
- Sergent, A. S., Rozenberg, P., & Bréda, N. (2014). Douglas-fir is vulnerable to exceptional and recurrent drought episodes and recovers less well on less fertile sites. *Annals of Forest Science*, 71(6), 697–708. <https://doi.org/10.1007/s13595-012-0220-5>
- Shifley, S. R., He, H. S., Lischke, H., Wang, W. J., Jin, W., Gustafson, E. J., Thompson, J. R., Thompson, F. R., Dijak, W. D., & Yang, J. (2017). The past and future of modeling forest dynamics: from growth and yield curves to forest landscape models. *Landscape Ecology*, 32(7), 1307–1325. <https://doi.org/10.1007/s10980-017-0540-9>
- Shifley, S. R., Thompson, F. R., Dijak, W. D., Larson, M. A., & Millsbaugh, J. J. (2006). Simulated effects of forest management alternatives on landscape structure and habitat suitability in the Midwestern United States. *Forest Ecology and Management*, 229(1–3), 361–377. <https://doi.org/10.1016/j.foreco.2006.04.030>
- Skovsgaard, J. P., & Vanclay, J. K. (2008). Forest site productivity: A review of the evolution of dendrometric concepts for even-aged stands. *Forestry*, 81(1), 13–31. <https://doi.org/10.1093/forestry/cpm041>
- Sohn, J. A., Hartig, F., Kohler, M., Huss, J., & Bauhus, J. (2016). Heavy and frequent thinning promotes drought adaptation in *Pinus sylvestris* forests. *Ecological Applications*, 26(7), 2190–2205. <https://doi.org/10.1002/eap.1373>
- Sohn, J. A., Saha, S., & Bauhus, J. (2016). Potential of forest thinning to mitigate drought stress: A meta-analysis. *Forest Ecology and Management*, 380, 261–273. <https://doi.org/10.1016/j.foreco.2016.07.046>
- Sousa-Silva, R., Ponette, Q., Verheyen, K., Van Herzele, A., & Muys, B. (2016). Adaptation of forest management to climate change as perceived by forest owners and managers in Belgium. *Forest Ecosystems*, 3(1), 1–11. <https://doi.org/10.1186/s40663-016-0082-7>
- Sousa-Silva, R., Verbist, B., Lomba, Â., Valent, P., Suškevičs, M., Picard, O., Hoogstra-Klein, M. A., Cosofret, V. C., Bouriaud, L., Ponette, Q., Verheyen, K., & Muys, B. (2018).

- Adapting forest management to climate change in Europe: Linking perceptions to adaptive responses. *Forest Policy and Economics*, 90, 22–30.
<https://doi.org/10.1016/j.forpol.2018.01.004>
- Statistiek Vlaanderen. (2020, June 25). *Oppervlakte bos: 140.000 hectare bos in Vlaanderen*.
<https://www.vlaanderen.be/statistiek-vlaanderen/milieu-en-natuur/oppervlakte-bos>
- Storms, I., Thom, D., Verbist, B., Van Meerbeek, K., Gobin, A., Van Orshoven, J., & Muys, B. (2023). Vegetation dynamics under climate change. In *Managing forest ecosystem services under climate change* (pp. 21–44). Ilié Storms.
- Tabari, H., Taye, M. T., & Willems, P. (2015). Water availability change in central Belgium for the late 21st century. *Global and Planetary Change*, 131, 115–123.
<https://doi.org/10.1016/j.gloplacha.2015.05.012>
- Taccoen, A., Piedallu, C., Seynave, I., Perez, V., Gégout-Petit, A., Nageleisen, L. M., Bontemps, J. D., & Gégout, J. C. (2019). Background mortality drivers of European tree species: Climate change matters. *Proceedings of the Royal Society B: Biological Sciences*, 286(1900), 1–10. <https://doi.org/10.1098/rspb.2019.0386>
- Temperli, C., Bugmann, H., & Elkin, C. (2012). Adaptive management for competing forest goods and services under climate change. *Ecological Applications*, 22(8), 2065–2077.
<https://doi.org/10.1890/12-0210.1>
- Temperli, C., Bugmann, H., & Elkin, C. (2013). Cross-scale interactions among bark beetles, climate change, and wind disturbances: A landscape modeling approach. *Ecological Monographs*, 83(3), 383–402. <https://doi.org/10.1890/12-1503.1>
- Thom, D., Rammer, W., Garstenauer, R., & Seidl, R. (2018). Legacies of past land use have a stronger effect on forest carbon exchange than future climate change in a temperate forest landscape. *Biogeosciences*, 15(18), 5699–5713. <https://doi.org/10.5194/bg-15-5699-2018>
- Thom, D., Rammer, W., Laux, P., Smiatek, G., Kunstmann, H., Seibold, S., & Seidl, R. (2022). Will forest dynamics continue to accelerate throughout the 21st century in the

- Northern Alps? *Global Change Biology*, 28(10), 3260–3274.
<https://doi.org/10.1111/gcb.16133>
- Thom, D., Rammer, W., & Seidl, R. (2017). Disturbances catalyze the adaptation of forest ecosystems to changing climate conditions. *Global Change Biology*, 23(1), 269–282.
<https://doi.org/10.1111/gcb.13506>
- Thomas, F. M., Rzepecki, A., & Werner, W. (2022). Non-native Douglas fir (*Pseudotsuga menziesii*) in Central Europe: Ecology, performance and nature conservation. *Forest Ecology and Management*, 506, 1–17. <https://doi.org/10.1016/j.foreco.2021.119956>
- Thomassen, E., Buysse, W., Wijdeven, S., Winnock, M., Seynaeve, J., Sauren, P., Van Lommel, H., Schmitz, P., & Nyssen, B. (2020). *Boomgericht bosbeheer in de praktijk*. Eco2eco. <https://bosgroepen.be/wp-content/uploads/2020/10/Boomgericht-bosbeheer-in-de-praktijk.pdf>
- Thrippleton, T., Bugmann, H., Kramer-Priewasser, K., & Snell, R. S. (2016). Herbaceous Understorey: An Overlooked Player in Forest Landscape Dynamics? *Ecosystems*, 19(7), 1240–1254. <https://doi.org/10.1007/s10021-016-9999-5>
- Trumbore, S., Brando, P., & Hartmann, H. (2015). Forest health and global change. *Science*, 349(6250), 814–818. <https://doi.org/10.1126/science.aac6759>
- Twery, M. J., & Weiskittel, A. R. (2013). Forest-Management Modelling. In J. Wainwright & M. Mulligan (Eds.), *Environmental Modelling: Finding Simplicity in Complexity* (2nd ed., pp. 379–398). John Wiley and Sons. <https://doi.org/10.1002/9781118351475.ch23>
- Vacek, Z., Cukor, J., Vacek, S., Gallo, J., Bažant, V., & Zeidler, A. (2023). Role of black pine (*Pinus nigra* J. F. Arnold) in European forests modified by climate change. *European Journal of Forest Research*, 142(6), 1239–1258. <https://doi.org/10.1007/s10342-023-01605-5>
- Van Meerbeek, K., Jucker, T., & Svenning, J. C. (2021). Unifying the concepts of stability and resilience in ecology. *Journal of Ecology*, 109(9), 3114–3132.
<https://doi.org/10.1111/1365-2745.13651>

- Van Orshoven, J., Maes, J., Vereecken, H., Feyen, J., & Dudal, R. (1988). A structured database of Belgian soil profile data. *Pedologie*, 38(2), 191–206.
- Vannoppen, A., Kint, V., Ponette, Q., Verheyen, K., & Muys, B. (2019). Tree species diversity impacts average radial growth of beech and oak trees in Belgium, not their long-term growth trend. *Forest Ecosystems*, 6(1), 1–12. <https://doi.org/10.1186/s40663-019-0169-z>
- Verkerk, P. J., Costanza, R., Hetemäki, L., Kubiszewski, I., Leskinen, P., Nabuurs, G. J., Potočník, J., & Palahí, M. (2020). Climate-Smart Forestry: the missing link. *Forest Policy and Economics*, 115, 1–4. <https://doi.org/10.1016/j.forpol.2020.102164>
- VMM. (2018). *Milieuverkenning 2018. Oplossingen voor een duurzame toekomst. Milieurapport Vlaanderen*. Vlaamse Milieumaatschappij. www.vmm.be
- Walker, J., Sharpe, P. J. H., Penridge, L. K., & Wu, H. (1989). Ecological field theory: the concept and field tests. *Vegetatio*, 83, 81–95. <https://doi.org/10.1007/BF00031682>
- Weiskittel, A. R., Maguire, D. A., Monserud, R. A., & Johnson, G. P. (2010). A hybrid model for intensively managed Douglas-fir plantations in the Pacific Northwest, USA. *European Journal of Forest Research*, 129(3), 325–338. <https://doi.org/10.1007/s10342-009-0339-6>
- Wessely, J., Essl, F., Fiedler, K., Gattringer, A., Hülber, B., Ignateva, O., Moser, D., Rammer, W., Dullinger, S., & Seidl, R. (2024). A climate-induced tree species bottleneck for forest management in Europe. *Nature Ecology & Evolution*. <https://doi.org/10.1038/s41559-024-02406-8>
- Wickham, H., François, R., Henry, L., Müller, K., & Vaughan, D. (2023). *dplyr: A Grammar of Data Manipulation* (R package version 1.1.0). <https://dplyr.tidyverse.org>
- Wigley, T. M. L. (2001). The Science of Climate Change. In E. Claussen (Ed.), *Climate Change: Science, Strategies, & Solutions* (pp. 6–24). BRILL. https://doi.org/10.1163/9789004475748_005

- Winkel, G., Lovrić, M., Muys, B., Katila, P., Lundhede, T., Pecurul, M., Pettenella, D., Pipart, N., Plieninger, T., Prokofieva, I., Parra, C., Pölzl, H., Roitsch, D., Roux, J. L., Thorsen, B. J., Tyrväinen, L., Torralba, M., Vacik, H., Weiss, G., & Wunder, S. (2022). Governing Europe's forests for multiple ecosystem services: Opportunities, challenges, and policy options. *Forest Policy and Economics*, 145, 1–15.
<https://doi.org/10.1016/j.forpol.2022.102849>
- Wu, H., Sharpe, P. J. H., Walker, J., & Penridge, L. K. (1985). Ecological field theory: a spatial analysis of resource interference among plants. *Ecological Modelling*, 29, 215–243.
[https://doi.org/10.1016/0304-3800\(85\)90054-7](https://doi.org/10.1016/0304-3800(85)90054-7)
- Xi, W., Coulson, R. N., Birt, A. G., Shang, Z.-B., Waldron, J. D., Lafon, C. W., Cairns, D. M., Tchakerian, M. D., & Klepzig, K. D. (2009). Review of forest landscape models: Types, methods, development and applications. *Acta Ecologica Sinica*, 29(1), 69–78.
<https://doi.org/10.1016/j.chnaes.2009.01.001>
- Xiangrong, C., Mukui, Y., & Zhengcai, L. (2018). Short term effects of thinning on soil organic carbon fractions, soil properties, and forest floor in *Cunninghamia lanceolata* plantations. *Journal of Soil Science and Environmental Management*, 9(2), 21–29.
<https://doi.org/10.5897/jsssem2017.0661>
- Xiao, J.-L., Zeng, F., He, Q.-L., Yao, Y.-X., Han, X., & Shi, W.-Y. (2021). Responses of forest carbon cycle to drought and elevated CO₂. *Atmosphere*, 12(2), 1–13.
<https://doi.org/10.3390/atmos12020212>
- Zemp, M., Huss, M., Thibert, E., Eckert, N., McNabb, R., Huber, J., Barandun, M., Machguth, H., Nussbaumer, S. U., Gärtner-Roer, I., Thomson, L., Paul, F., Maussion, F., Kutuzov, S., & Cogley, J. G. (2019). Global glacier mass changes and their contributions to sea-level rise from 1961 to 2016. *Nature*, 568(7752), 382–386. <https://doi.org/10.1038/s41586-019-1071-0>

8 Appendices

Appendix A: Detailed and numerical description of the programming of the ABE module in iLand.

For the main forest an STP was programmed containing the main silvicultural activities executed in the forests. First, a rotation age needed to be specified, which was set equal to 100 years (with minimal and maximal value equal to 99 and 101, respectively). This rotation age was the mean rotation age as mentioned in the management plan (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006), but was not important as we eventually worked in simulation years to plan activities rather than planning the activities relative to the rotation age. We did this because most stands had already passed the rotation age due to how iLand was initialized and consequently, activities could not be planned relative to the rotation age. Additionally, defining one rotation age for the whole landscape is also not optimal as different species have different rotation ages, such as stated in the management plan. Nevertheless, it should be noted that the current approach was also not ideal. Planning the activities in simulation years does not allow to take into account the individual stand characteristics such as age, and plan management accordingly. Instead, one activity, e.g. a thinning, is imposed on the whole landscape at once. This is of course far from realistic but was necessary as a workaround for the initialization problem.

Thinnings were planned every 8 simulation years until the end of the simulation period, and forced, meaning that the activity would definitely be executed. Homogenous red oak stands (more than 80% in basal area share) were not thinned as suggested by the forest managers. A thinning removed a certain percentage of the basal area. This percentage was either 5%, 10% or 20% depending on the thinning scenario (low, normal, or high thinning intensity, respectively). A thinning from above was programmed by thinning in five evenly sized diameter classes, with 0%, 10%, 20%, 40%, 30% of removal, respectively. Some trees are removed from the lower diameter classes (10% and 20%) to increase the availability of resources and space for the remaining trees.

Conversion of conifer forest stands was programmed by executing a grouped planting in the 9th and 41st simulation year. This was done in homogenous conifer stands, or hence, when one of the species Norway spruce, silver fir, European larch, Douglas fir, black pine, or Scots

pine occupied more than 80% of the basal area. The number of conifers would be reduced (mainly because of thinning and harvesting) and these stands would gradually become more mixed with broadleaved species. The species planted were sessile oak and small-leaved lime. They were planted in a *circle5* pattern with a spacing of 40 m between the centres and an offset of 5 and 25 m for sessile oak and small-leaved lime, respectively (numbers based on Thomassen et al., 2020). The *circle5* pattern is a group of 5 pixels in diameter, which matches an area of 68 m². The offset only made sure the plantings were not overlapping. The saplings were given an initial height of 1.35 m (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006) and the stand was not cleared from saplings before planting. Less focus was placed on mixed stands containing conifer and broadleaved species as conifers are not to be fully excluded. They still are appreciated by recreationists and are not regenerating very easily in contrast to red oak (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

A group harvest was simulated for the regeneration of the forest. These were planned 4 years after each thinning with a margin of two years before and after this date to provide flexibility to the agent to ensure a more sustainable harvest. This activity was of the general type in iLand and was programmed manually by providing a grid defining the groups to be cut in the landscape. Trees situated in these groups were consequently filtered and harvested in iLand. The grids defining the groups were made using R (R Core Team, 2022). Here, randomly spaced points were generated in a stand. The number of points per stand was linked to the area of the stand, considering 1 point per 10 ha. The random spacing was only subjected to the constraint that points needed to be at least 90 m apart. This was done to prevent group cuts to become too big due to points laying close to each other. Next, buffers varying in radius from 17.8 m to 39.9 m were drawn around the points, creating groups of sizes varying between 10 to 50 are. This procedure was repeated nine times (for every harvest planned in the STP) and groups generated in a previous step were continuously subtracted from the next groups to prevent groups overlapping in time when used in iLand. To do this, we used the packages *sf* (Pebesma, 2018) for working with simple features, *terra* (Hijmans, 2023b), *raster* (Hijmans, 2023a), *sp* (Bivand et al., 2013; Pebesma & Bivand, 2005), and *spatstat* (Baddeley et al., 2015) for spatial data analysis, *potential* (Giraud & Commenges, 2022) for handling spatial interactions, *matrixStats* (Bengtsson, 2023) for summarizing matrices, and *dplyr* (Wickham et al., 2023) for data manipulation.

After every group harvest, plantings were executed in stands containing mostly red oak (more than 50% in basal area share). This was done to prevent red oak from regenerating, a technique used by the forest managers. As it was difficult to exactly plant in the group cuts, the whole stand was planted, assuming that most likely only the trees in open spaces from the group harvests would grow and the remaining trees would die due to competition with the mature trees. The following broadleaved trees were planted, as suggested by the forest managers, and the fraction of the stand that they would cover is given; sessile oak (40%), small-leaved lime (20%), common hornbeam (20%) and common hazel (20%). More sessile oak was planted because of its economic value and other species were considered for diversity. The naturally present regeneration was not cleared, as this would not be done in reality either. The saplings had again an initial height of 1.35 m (Agentschap voor Natuur en Bos Houtvesterij Leuven, 2006).

Appendix B: The numerical details of the species composition and the diameter class distribution at the end of the simulation period.

Table 5: The mean share of each species (%) in the last five years of the simulation period as presented in Figure 8 and Figure 9 for every climate and management scenario. The management scenarios are NM (no management), LI (low intensity), NI (normal intensity), and HI (high intensity).

	Historical				Moderate			
	NM	LI	NI	HI	NM	LI	NI	HI
<i>F. sylvatica</i>	44.28	36.41	33.18	26.57	44.78	37.31	34.62	28.40
<i>Q. petraea</i>	0.65	2.17	2.61	3.06	0.61	2.07	2.47	3.01
<i>Q. robur</i>	2.92	3.55	4.14	4.59	2.98	3.47	3.87	4.21
<i>Q. rubra</i>	14.69	15.12	15.41	16.45	14.63	14.57	14.80	15.55
<i>P. nigra</i>	13.52	10.80	9.82	8.12	13.02	10.25	8.93	7.16
<i>P. sylvestris</i>	5.13	4.06	3.84	2.89	5.20	4.07	3.74	2.77
<i>P. menziesii</i>	12.48	17.96	19.66	25.06	12.27	18.09	20.05	25.49
<i>P. abies</i>	1.69	1.53	1.51	1.48	1.80	1.56	1.49	1.38
<i>A. alba</i>	0.82	0.67	0.63	0.56	0.89	0.69	0.65	0.56
<i>A. pseudoplatanus</i>	1.82	2.31	2.62	2.94	1.90	2.34	2.60	2.94
<i>C. betulus</i>	1.51	3.72	4.37	5.26	1.39	3.71	4.41	5.34
<i>T. cordata</i>	0.34	1.22	1.48	1.66	0.37	1.38	1.66	1.91

	Hot				Hot and dry			
	NM	LI	NI	HI	NM	LI	NI	HI
<i>F. sylvatica</i>	43.79	36.65	34.23	28.40	38.24	31.05	28.60	22.87
<i>Q. petraea</i>	0.74	2.37	2.86	3.60	1.46	4.05	4.57	5.14
<i>Q. robur</i>	3.79	4.19	4.55	4.97	5.17	5.93	6.48	6.93
<i>Q. rubra</i>	13.85	13.78	14.09	14.79	11.63	10.11	9.98	9.83
<i>P. nigra</i>	13.64	10.74	9.34	7.56	19.90	19.19	18.56	19.83
<i>P. sylvestris</i>	5.51	4.31	3.93	2.90	5.38	4.32	4.08	3.14
<i>P. menziesii</i>	11.96	16.99	18.49	23.11	11.69	14.93	16.13	19.19
<i>P. abies</i>	1.59	1.34	1.29	1.17	0.94	0.84	0.83	0.77
<i>A. alba</i>	0.85	0.65	0.61	0.53	0.60	0.47	0.45	0.41
<i>A. pseudoplatanus</i>	1.91	2.34	2.60	2.96	1.31	1.78	2.05	2.39
<i>C. betulus</i>	1.73	4.51	5.32	6.43	2.84	5.08	5.56	6.03
<i>T. cordata</i>	0.47	1.64	1.97	2.26	0.59	1.64	1.85	2.02

Table 6: The mean share of trees (%) in every diameter class (D) of the last five years of the simulation period as presented in Figure 10 and Figure 11 for every climate and management scenario. The management scenarios are NM (no management), LI (low intensity), NI (normal intensity), and HI (high intensity).

Historical				Moderate				
D (cm)	NM	LI	NI	HI	NM	LI	NI	HI
0-10	52.98	60.80	62.91	64.03	48.74	56.71	58.56	59.99
10-20	22.21	22.89	22.82	22.88	22.89	24.25	24.34	24.15
20-30	11.67	8.46	7.70	7.40	12.47	9.43	8.88	8.75
30-40	6.71	4.18	3.54	3.08	8.04	5.08	4.41	3.78
40-50	3.16	1.81	1.48	1.29	3.84	2.21	1.86	1.65
50-60	1.37	0.80	0.67	0.61	1.66	0.98	0.84	0.77
60-70	0.76	0.43	0.36	0.32	0.94	0.54	0.46	0.41
70-80	0.45	0.24	0.20	0.17	0.55	0.31	0.26	0.22
80-90	0.29	0.15	0.12	0.10	0.35	0.18	0.15	0.12
90-100	0.18	0.10	0.08	0.05	0.23	0.12	0.09	0.07
>100	0.21	0.14	0.11	0.07	0.29	0.18	0.14	0.09

Hot				Hot and dry				
D (cm)	NM	LI	NI	HI	NM	LI	NI	HI
0-10	48.41	55.49	57.36	59.12	59.03	63.44	64.14	64.77
10-20	23.54	25.52	25.58	25.03	24.97	25.04	25.09	24.42
20-30	12.57	9.73	9.19	9.05	8.54	6.72	6.49	6.63
30-40	7.76	4.88	4.22	3.63	3.85	2.49	2.22	2.18
40-50	3.75	2.14	1.78	1.55	1.66	1.09	0.98	0.98
50-60	1.61	0.94	0.80	0.72	0.78	0.49	0.45	0.46
60-70	0.91	0.52	0.44	0.39	0.45	0.27	0.24	0.24
70-80	0.54	0.30	0.25	0.21	0.27	0.16	0.14	0.13
80-90	0.35	0.18	0.15	0.12	0.18	0.10	0.09	0.07
90-100	0.23	0.12	0.09	0.07	0.11	0.07	0.06	0.05
>100	0.32	0.19	0.15	0.10	0.16	0.12	0.10	0.08

Appendix C: the *P*-values from the Scheirer-Ray-Hare test and the Dunn's post hoc test for the pairwise comparison of the resistance of the species composition between management or climate scenarios.

The Scheirer-Ray-Hare test depicts the influences of management, climate, and their interaction on the resistance of the species composition against climate change (Table 7). Next, a Dunn's post hoc test compares the resistance of the species composition against climate change between management scenarios in the historical (Table 8), moderate (Table 9), hot (Table 10), and hot and dry climate (Table 11).

Table 7: The Scheirer-Ray-Hare test showing the H-statistic, the degrees of freedom (df), and the *P*-value to determine the influence of management, climate, and their interaction on the resistance of the species composition.

	H	df	<i>P</i> -value
Management	376.29	3	<0.001
Climate	69.14	3	<0.001
Management x climate	12.40	9	0.19

Table 8: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the historical climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	4.46	<0.001	<0.001
Higher intensity - No management	3.60	<0.001	<0.001
Lower intensity - No management	-0.86	0.39	0.47
Higher intensity - Normal intensity	3.42	<0.001	0.001
Lower intensity - Normal intensity	-1.03	0.30	0.45
No management - Normal intensity	-0.17	0.86	0.86

Table 9: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the moderate climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	10.19	<0.001	<0.001
Higher intensity - No management	12.81	<0.001	<0.001
Lower intensity - No management	2.63	0.009	0.009
Higher intensity - Normal intensity	5.37	<0.001	<0.001
Lower intensity - Normal intensity	-4.81	<0.001	<0.001
No management - Normal intensity	-7.44	<0.001	<0.001

Table 10: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the hot climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	7.74	<0.001	<0.001
Higher intensity - No management	10.37	<0.001	<0.001
Lower intensity - No management	2.63	0.009	0.009
Higher intensity - Normal intensity	5.03	<0.001	<0.001
Lower intensity - Normal intensity	-2.71	0.007	0.008
No management - Normal intensity	-5.34	<0.001	<0.001

Table 11: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different management scenarios in the hot and dry climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	6.68	<0.001	<0.001
Higher intensity - No management	9.25	<0.001	<0.001
Lower intensity - No management	2.57	0.01	0.01
Higher intensity - Normal intensity	4.12	<0.001	<0.001
Lower intensity - Normal intensity	-2.56	0.01	0.01
No management - Normal intensity	-5.13	<0.001	<0.001

A Dunn's post hoc test compares the resistance of the species composition against climate change between climate scenarios in the no management (Table 12), low intensity (Table 13), normal intensity (Table 14), and high intensity scenario (Table 15).

Table 12: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the no management scenario.

	Z	P.unadj	P.adj
Historical - Hot	2.77	0.006	0.01
Historical - Hot and dry	-1.40	0.16	0.19
Hot - Hot and dry	-6.17	<0.001	<0.001
Historical - Moderate	-0.03	0.98	0.98
Hot - Moderate	-4.84	<0.001	<0.001
Hot and dry - Moderate	2.24	0.02	0.04

Table 13: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the low intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	-0.41	0.68	0.68
Historical - Hot and dry	-5.39	<0.001	<0.001
Hot - Hot and dry	-7.55	<0.001	<0.001
Historical - Moderate	-3.13	0.002	0.002
Hot - Moderate	-4.52	<0.001	<0.001
Hot and dry - Moderate	4.02	<0.001	<0.001

Table 14: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the normal intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	-1.32	0.19	0.19
Historical - Hot and dry	-6.28	<0.001	<0.001
Hot - Hot and dry	-7.56	<0.001	<0.001
Historical - Moderate	-3.78	<0.001	<0.001
Hot - Moderate	-4.05	<0.001	<0.001
Hot and dry - Moderate	4.47	<0.001	<0.001

Table 15: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in species composition between the different climate scenarios in the high intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	1.61	0.11	0.13
Historical - Hot and dry	-3.94	<0.001	<0.001
Hot - Hot and dry	-8.32	<0.001	<0.001
Historical - Moderate	0.22	0.83	0.83
Hot - Moderate	-2.43	0.02	0.02
Hot and dry - Moderate	6.77	<0.001	<0.001

Appendix D: the *P*-values from the Scheirer-Ray-Hare test and the Dunn's post hoc test for the pairwise comparison of the resistance of the diameter class distribution between management or climate scenarios.

The Scheirer-Ray-Hare test depicts the influences of management, climate, and their interaction on the resistance of the diameter class distribution against climate change (Table 16). Next, a Dunn's post hoc test compares the resistance of the diameter class distribution against climate change between management scenarios in the historical (Table 17), moderate (Table 18), hot (Table 19), and hot and dry climate (Table 20).

Table 16: The Scheirer-Ray-Hare test showing the H-statistic, the degrees of freedom (df), and the *P*-value to determine the influence of management, climate, and their interaction on the resistance of the diameter class distribution.

	H	df	<i>P</i> -value
Management	70.83	3	<0.001
Climate	201.54	3	<0.001
Management x climate	97.00	9	<0.001

Table 17: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the historical climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	3.71	<0.001	<0.001
Higher intensity - No management	5.62	<0.001	<0.001
Lower intensity - No management	1.91	0.06	0.07
Higher intensity - Normal intensity	1.68	0.09	0.09
Lower intensity - Normal intensity	-2.03	0.04	0.06
No management - Normal intensity	-3.94	<0.001	<0.001

Table 18: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the moderate climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	9.21	<0.001	<0.001
Higher intensity - No management	2.36	0.02	0.02
Lower intensity - No management	-6.85	<0.001	<0.001
Higher intensity - Normal intensity	3.74	<0.001	<0.001
Lower intensity - Normal intensity	-5.47	<0.001	<0.001
No management - Normal intensity	1.38	0.17	0.17

Table 19: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the hot climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	5.26	<0.001	<0.001
Higher intensity - No management	0.50	0.62	0.62
Lower intensity - No management	-4.76	<0.001	<0.001
Higher intensity - Normal intensity	1.58	0.11	0.17
Lower intensity - Normal intensity	-3.68	<0.001	<0.001
No management - Normal intensity	1.08	0.28	0.34

Table 20: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different management scenarios in the hot and dry climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	2.66	0.008	0.009
Higher intensity - No management	7.41	<0.001	<0.001
Lower intensity - No management	4.75	<0.001	<0.001
Higher intensity - Normal intensity	-0.47	0.64	0.64
Lower intensity - Normal intensity	-3.13	0.002	0.003
No management - Normal intensity	-7.88	<0.001	<0.001

A Dunn's post hoc test compares the resistance of the diameter class distribution against climate change between climate scenarios in the no management (Table 21), low intensity (Table 22), normal intensity (Table 23), and high intensity scenario (Table 24).

Table 21: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the no management scenario.

	Z	P.unadj	P.adj
Historical - Hot	-4.97	<0.001	<0.001
Historical - Hot and dry	-5.07	<0.001	<0.001
Hot - Hot and dry	-0.40	0.69	0.69
Historical - Moderate	-4.47	<0.001	<0.001
Hot - Moderate	1.13	0.26	0.31
Hot and dry - Moderate	1.46	0.15	0.22

Table 22: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the low intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	1.90	0.06	0.07
Historical - Hot and dry	-1.44	0.15	0.15
Hot - Hot and dry	-4.96	<0.001	<0.001
Historical - Moderate	5.12	<0.001	<0.001
Hot - Moderate	5.28	<0.001	<0.001
Hot and dry - Moderate	10.18	<0.001	<0.001

Table 23: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the normal intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	3.48	<0.001	<0.001
Historical - Hot and dry	-1.36	0.17	0.21
Hot - Hot and dry	-7.15	<0.001	<0.001
Historical - Moderate	4.20	<0.001	<0.001
Hot - Moderate	1.00	0.32	0.32
Hot and dry - Moderate	8.63	<0.001	<0.001

Table 24: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resistance in diameter class distribution between the different climate scenarios in the high intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	3.66	<0.001	<0.001
Historical - Hot and dry	-1.35	0.18	0.21
Hot - Hot and dry	-7.40	<0.001	<0.001
Historical - Moderate	4.09	<0.001	<0.001
Hot - Moderate	0.51	0.61	0.61
Hot and dry - Moderate	8.46	<0.001	<0.001

Appendix E: the *P*-values from the Scheirer-Ray-Hare test and the Dunn's post hoc test for the pairwise comparison of the resilience of the basal area of the forest between management or climate scenarios.

The Scheirer-Ray-Hare test depicts the influences of management, climate, and their interaction on the resilience of the basal area of the forest under climate change (Table 25). Next, a Dunn's post hoc test compares the resilience of the basal area of the forest under climate change between thinning scenarios in the historical (Table 26), moderate (Table 27), hot (Table 28), and hot and dry climate (Table 29).

Table 25: The Scheirer-Ray-Hare test showing the H-statistic, the degrees of freedom (df), and the *P*-value to determine the influence of management, climate, and their interaction on the resilience of the basal area of the forest complex.

	H	df	<i>P</i> -value
Management	700.12	2	<0.001
Climate	366.50	3	<0.001
Management x climate	72.71	6	<0.001

Table 26: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the historical climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	-7.43	<0.001	<0.001
Higher intensity - Normal intensity	-4.00	<0.001	<0.001
Lower intensity - Normal intensity	3.43	<0.001	<0.001

Table 27: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the moderate climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	-26.68	<0.001	<0.001
Higher intensity - Normal intensity	-11.69	<0.001	<0.001
Lower intensity - Normal intensity	15.00	<0.001	<0.001

Table 28: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the hot climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	-13.57	<0.001	<0.001
Higher intensity - Normal intensity	-5.71	<0.001	<0.001
Lower intensity - Normal intensity	7.85	<0.001	<0.001

Table 29: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different thinning scenarios in the hot and dry climate.

	Z	P.unadj	P.adj
Higher intensity - Lower intensity	-1.72	0.09	0.26
Higher intensity - Normal intensity	-1.48	0.14	0.21
Lower intensity - Normal intensity	0.24	0.81	0.81

A Dunn's post hoc test compares the resilience of the basal area of the forest under climate change between climate scenarios in the low intensity (Table 30), normal intensity (Table 31), and high intensity scenario (Table 32).

Table 30: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different climate scenarios in the low intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	0.19	0.85	0.85
Historical - Hot and dry	5.04	<0.001	<0.001
Hot - Hot and dry	7.34	<0.001	<0.001
Historical - Moderate	-3.31	<0.001	0.001
Hot - Moderate	-5.87	<0.001	<0.001
Hot and dry - Moderate	-13.29	<0.001	<0.001

Table 31: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different climate scenarios in the normal intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	1.07	0.29	0.29
Historical - Hot and dry	4.91	<0.001	<0.001
Hot - Hot and dry	5.85	<0.001	<0.001
Historical - Moderate	-1.80	0.07	0.09
Hot - Moderate	-4.86	<0.001	<0.001
Hot and dry - Moderate	-10.76	<0.001	<0.001

Table 32: The Z-statistic, unadjusted and adjusted *P*-values of the Dunn's post hoc test pairwise comparison of the resilience of the basal area of the forest between the different climate scenarios in the high intensity scenario.

	Z	P.unadj	P.adj
Historical - Hot	-0.18	0.85	0.85
Historical - Hot and dry	3.81	<0.001	<0.001
Hot - Hot and dry	6.04	<0.001	<0.001
Historical - Moderate	-2.59	0.01	0.01
Hot - Moderate	-4.01	<0.001	<0.001
Hot and dry - Moderate	-10.18	<0.001	<0.001

Appendix F: Gebruik van Generatieve Artificiële Intelligentie (GenAI) - Formulier

Faculteit Bio-ingenieurswetenschappen, KU Leuven

(2023-2024)

Gebruik van Generatieve Artificiële Intelligentie (GenAI) - In te vullen formulier

Naam student: ... Neufkens Anais

Studentennummer: ... r0790040

Geef met "X" aan of het gaat om een cursusopdracht, een BIG-project of een masterproef:

☒ Dit formulier heeft betrekking op mijn masterproef.

Titel masterproef: ...Sustainably managing forests for a resilient future: an evaluation of different thinning intensities

Promotor: ... Muys Bart

☐ Dit formulier heeft betrekking op een BIG-project.

Titel BIG-project: ...

Promotor: ...

☐ Dit formulier heeft betrekking op een cursusopdracht.

Cursusnaam: ...

Cursusnummer: ...

Geef met "X" aan:

☒ Ik heb geen GenAI tool gebruikt.

☐ Ik heb wel een Gen AI tool gebruikt. Geef in dat geval aan welke (bv. ChatGPT/GPT-4/...): ...

Geef met "X" aan (mogelijk meerdere keren) op welke manier je de Gen AI tool hebt gebruikt:

- ☐ Als taalassistent voor het herwerken of verbeteren van zelf geschreven teksten zonder nieuwe inhoud toe te voegen. (Dit gebruik is gelijk aan spelling- en grammaticacheckers en hoeft je niet expliciet te vermelden.)
- ☐ Als zoekmachine om eerste informatie te krijgen over een onderwerp of voor een eerste aanzet om op zoek te gaan naar literatuur. (Deze toepassing is gelijkaardig aan het gebruik van een gewone zoekmachine bij de uitwerking van een persoonlijk werkstuk. Als student ben je verantwoordelijk voor het controleren en verifiëren van de afwezigheid of correctheid van referenties. Ga dus vervolgens zelf op zoek naar wetenschappelijke referenties en voer zelf een analyse van de brondocumenten. Interpreteer, analyseer en verwerk de verkregen informatie; kopieer ze niet zomaar. Indien je vervolgens eigenhandig een tekst opstelt, hoeft je het gebruik niet te vermelden.)
- ☐ Om tekstblokken te genereren. (Wanneer je tekstblokken van GenAI output letterlijk overneemt, dan vermeld je de GenAI bronnen en citeer je, m.a.w. je meldt duidelijk dat het item via GenAI is aangemaakt d.m.v. citatie/referentie.)
- ☐ Om grafieken of figuren te genereren. (Wanneer je grafieken/figuren van GenAI output letterlijk overneemt, dan vermeld je de GenAI bronnen en citeer je, m.a.w. je meldt duidelijk dat het item via GenAI is aangemaakt d.m.v. citatie/referentie.)
- ☐ Om code te laten genereren als deelaspect binnen een grotere opdracht. (Let op, dit mag enkel als de docent hier EXPLICIET toestemming voor heeft gegeven.)
- ☐ Anders (Neem contact op met de docent van de cursus of de promotor van de thesis of het BIG-project. Leg uit hoe je voldoet aan artikel 84 van het examenreglement. Leg uit wat het nut of de toegevoegde waarde van het gebruik van GenAI.)

Bijkomende richtlijnen en opmerkingen

De faculteit volgt het beleid van KU Leuven om GenAI verantwoord in te zetten. Daarbij is transparantie over het gebruik van GenAI door de student essentieel, waarvoor dit formulier hulp biedt. Onverantwoord en niet-transparant gebruik van GenAI kan beschouwd worden als een onregelmatigheid en gesanctioneerd worden. Studenten die overwegen om GenAI te gebruiken, moeten zich verder informeren via de universitaire website met bijkomende richtlijnen (Hoe GenAI correct te citeren en naar te refereren? Wat is (niet) toegelaten? Tips en aandachtspunten voor een verantwoord gebruik):

<https://www.kuleuven.be/onderwijs/student/onderwijs-tools/artificiele-intelligentie>