Technische Universität München
TUM School of Life Sciences

European forests and the land sector under climate change: Strategies for mitigation, adaptation, and ecosystem service provision

Konstantin Philip Michael Gregor

Vollständiger Abdruck der von der TUM School of Life Sciences der Technischen Universität München zur Erlangung des akademischen Grades eines

Doktors der Naturwissenschaften (Dr. rer. nat.)

genehmigten Dissertation.

Vorsitz:
Prof. Dr. Johannes Kollmann

Prüfende der Dissertation:
1. Prof. Dr. Anja Rammig
2. Prof. Dr. Thomas Knoke
3. Prof. Dr. Almut Arneth

Die Dissertation wurde am 22.01.2024 bei der Technischen Universität München eingereicht und durch die TUM School of Life Sciences am 10.06.2024 angenommen.

Acknowledgments

First and foremost, I would like to express my sincere gratitude to my supervisor Anja Rammig, for giving me the opportunity for this work, for the advice and encouragement throughout the years, for all the opportunities to explore my research interests, and for creating a research group that is founded on such a respectful and collaborative work environment. Going back into academia was a big leap for me, but I felt welcomed and comfortable from day one. I am very grateful for being part of this group and to be able to continue working here for the next few years. I would like to thank all members of the LSAI group for the great spirit and their support during the last years, especially Phillip for helping me set foot in this new field of work, and Andy for always being extremely helpful. Thanks to Ben, Christopher, and Tati for all the fun times in our office and all the other members of LSAI and UPE for making these past four years so incredibly enjoyable, both during the workday and in all our great social events like the absolute cooking madness at our retreat.

I would like to express my gratitude to the scientists who have helped me tremendously along the way, Christopher Reyer, Thomas Knoke, Mats Lindeskog, Stefan Olin, the entire LPJ-GUESS community, as well as my co-authors Andy Krause, Anne-Sofie Lansø, Anniki Mäkelä, Ben Meyer, Tom Nagel, Phillip Papastefanou, Ben Smith, and Susanne Suvanto. I would not have been able to finish this work without you. Special thanks to Andy, Ben, Peter, and Phillip, who helped finalize this thesis. A big thanks to Trevor Keenan and his group for the amazing opportunity to work with them for one summer and to my mentor Julia Pongratz for always being available for my questions. Thank you to everyone involved in the bayklif network, especially Anne, Jana, Max, Mona, Stefan, and Sven. I am hoping for some more great retreats in Iffeldorf!

I also want to express my gratitude to the entire scientific community of vegetation modeling and climate science. It is truly a blessing to work in a field where everyone is so passionate, friendly, and open-minded and where work events never actually feel like work. I can't wait for the next get-together at EGU or elsewhere.

Finally, this thesis would not have been possible without the support of many amazing people in my life. First and foremost, thanks to my family for their support and love throughout my life, for always being encouraging to all my endeavors, and for teaching me deep respect for this planet and all the life on it. I would like to thank my father for helping me with this thesis, for always answering my stupid questions about ecology, and for always going an extra mile to help with anything. I want to thank my mother for always being there, for her helpful advice on anything, and for still insisting on making sandwiches so her son would not get hungry on his way back to Munich. Thank you to my brother Kristof for always serving as a role model in so many ways, from when I was a little kid until today. And thank you to my sister Julia, for always showing me how to break out of conventions. I wish I had told her that in person.

I am also extremely grateful for everyone who played a significant role in my life and their support throughout the last years, Annika and Maria, and all my amazing friends: my former flatmates, my surf, skate, snowboard, and music buddies, simply all my dear friends from my times in Berlin, Heidelberg, Munich, and California. You know who you are and it's such a blessing to have you in my life. Thank you for being there, and thank you for still listening to my rants about climate change and ecology.

Abstract

Climate change poses a pressing challenge for humanity, and the land sector emerges as a key player in mitigation efforts. In that regard, forests play a crucial role, offering climate change mitigation through the forest carbon sink, the carbon sink in wood products, and substitution effects when wood replaces carbon-intensive non-wood products. Despite their significance, there remain large uncertainties regarding optimal strategies in forest-based and land-based mitigation, as well as regarding their absolute potentials. Moreover, forests are vulnerable to climate change, requiring adaptation measures. They also offer numerous important ecosystem services to humanity, ranging from timber provision and local climate regulation to water cycling. The approach of considering all these aspects from mitigation, adaptation, and ecosystem services is also termed "climate-smart forestry".

Global loss of biodiversity poses another major threat, impacting ecosystem resilience, food security, and human health. Addressing this issue requires the enhancement of land use strategies to address biodiversity habitat provision. In Europe, the multifaceted demands on the land sector are formalized by recent legislation, including targets for carbon sink enhancement, climate change mitigation through forest management, and biodiversity protection. This poses a complex challenge for the development of land use strategies. Rising demands for various wood products add to this complexity, raising the question how to reconcile these diverse and sometimes conflicting demands.

In this thesis, I address these research gaps in a European context by 1) quantifying the key uncertainties in forest-based mitigation, 2) developing a methodology to derive climate-smart forestry strategies under a broad range of climate scenarios, 3) assessing the impacts of various European Union (EU) targets on the land sector, and 4) estimating the realistic potentials of land-based climate change mitigation measures for Bavaria. My thesis is divided into four sections, containing three research papers and one section with additional results.

The first paper delves into forest-based carbon mitigation, a topic still associated with large uncertainties and sometimes conflicting literature recommendations. Such uncertainties arise from incomplete considerations of relevant factors and because of the substantial influence that a few assumptions can exert on the results. Here, I addressed these assumptions to answer the first research question: *What are the impacts of the main factors on the carbon mitigation potential of central European temperate forests and which recommendations for mitigation efforts can be drawn?*

In particular, I considered the factors forest age, forest type, climate change, disturbances, harvest intensities, wood usage patterns, salvage logging practices, and multiple decarbonization scenarios – from current policies to the net-zero targets of the EU. To achieve this, I enhanced the well-established dynamic global vegetation model LPJ-GUESS with a more detailed representation of wood usage and disturbance patterns. Subsequently, I conducted factorial simulation experiments to quantify the impacts of the aforementioned factors on mitigation potentials and derived recommendations for mitigation strategies. The main findings suggest that although disturbance scenarios and decarbonization pace are the most crucial sources of uncertainty, this uncertainty remains small enough until 2050 to still formulate recommendations. Namely, substitution effects have a critical impact at least until 2050 such that decreasing sustainable harvests is currently not recommendable from a mitigation point of view (neglecting potential benefits for other ecosystem services). The avoided emissions from using wood products instead of more carbon-intensive non-wood products have a higher impact than an increased forest carbon sink due to the decreased harvests. This also adds to the considerable mitigation impact of an increased material usage of wood, instead of using it as fuel. This impact remains past the time when substitution effects have ceased, because of the enhanced carbon sink in wood products. Assessments of substitution effects hinge on the realistic quantification of displacement factors which reveal how many emissions are avoided when using a wood product. Hence, their improved quantification should be a future key research priority. Mitigation strategies also need to be tailored to forest type and age due to their distinct product portfolios and growth patterns. Finally, for longer time horizons, the mentioned uncertainties become substantial. It is thus recommendable to apply robust methodologies and risk diversification strategies for mitigation planning beyond 2050.

In the second paper I extended the focus beyond mitigation towards climate-smart forestry. Alongside mitigation, this includes the adaptation of forests to future climate change, and the provision of ecosystem services including local climate regulation, water cycling, biodiversity habitat provision, and timber provision. I specifically addressed the following research questions: *How can we derive strategies for climate-smart forestry in Europe despite the large uncertainty surrounding future climate? What are the implications for the European forest landscape?*

Employing LPJ-GUESS and the modifications introduced in the first paper, I extended the model by implementing coppice management as well as the computation of indicators for the aforementioned ecosystem services. These were derived from both existing and newly introduced output variables of LPJ-GUESS.

To develop strategies that address these manifold objectives across a wide range of climate change scenarios, I combined robust optimization with multi-criteria decision making. This robust multi-criteria optimization was applied to compute optimized forest management portfolios that ensured a balanced provision of mitigation, adaptation, and other ecosystem services, regardless of the climate scenario. The application of this methodology on a European scale revealed substantial shifts towards more broad-leaved forests and unmanaged forests, as this was particularly beneficial for adaptation, biodiversity, and carbon storage. Moreover, this methodology yielded diverse portfolios, effectively spreading risks and thereby decreasing uncertainties. However, while all ecosystem services were provided in a balanced way in all climate scenarios, a substantial decrease in timber provision was projected. This highlights that intricate trade-offs need to be addressed when deriving strategies for climate-smart forestry in Europe.

In the third paper, I assessed the impacts of new EU strategies regarding the land sector, forests, and biodiversity. These strategies impose hard constraints on land use, including targets for an enhanced carbon sink, strict forest protection, and the promoted use of wood products for climate change mitigation. Building on the results of the second paper, I formulated the following research questions: *How can we align climate-smart forestry strategies to reconcile with new EU strategies and demands for wood products? And what are the associated implications?*

To address these questions, I extended the optimization framework established in the second paper by introducing hard constraints on strict protection areas and harvest levels. These constraints needed to be met on the European continent across all climate scenarios. The main findings were that while these constraints could theoretically be reconciled according to the model, they implied that highly productive forests, particularly in southern Scandinavia, were required to prioritize timber provision. Consequently, this jeopardized a fair distribution of protection areas, posing negative consequences for biodiversity habitat provision and other ecosystem services in those regions.

Furthermore, when considering these additional constraints, the portfolios were less diverse, potentially posing a large risk due to lower risk diversification. Consequently, my results highlight the need to alleviate the pressure on forests. Potential measures for this issue are discussed, including increased investments in renewable energies other than woody biomass, or addressing demands for single-use wood products such as packaging.

As a final step, I assessed land-based mitigation options within the German federal state of Bavaria, aiming to answer the research question: *What are the greenhouse gas mitigation potentials of different climate change mitigation strategies of forests and agriculture in Bavaria?* For this, I extended the scope from forests towards land use as a whole. Using LPJ-GUESS, with an improved parametrization of the bioenergy crop *Miscanthus*, I simulated five mitigation actions at realistic scales across the federal state, encompassing increased material usage of wood, cultivation of bioenergy crops with and without carbon capture and storage, reforestation, and reduced nitrous oxide emissions from decreased crop fertilization. The results indicate a modest yet significant impact of the land sector until 2100, equivalent to about six years of Bavarian greenhouse gas emissions. Furthermore, they show a declining impact of land-based mitigation due to the likely decreasing displacement factors. This highlights the necessity to implement these strategies fast to maximize their effectiveness. However, it is crucial to recognize the limitations, and that land-based mitigation can only be supplementary to rapidly and strongly reducing greenhouse gas emissions.

To conclude, my thesis addresses key issues regarding the role of forests and the land sector in terms of climate change mitigation, adaptation, biodiversity, and ecosystem service provision. It highlights the potential impact of mitigation strategies in central

Europe, providing recommendations to improve forest-based mitigation efforts. It also offers a methodology to derive strategies for climate-smart forestry in Europe, and indicates the need for transformative actions. However, my thesis also highlights the numerous strong demands on the land sector and underscores the need to alleviate the associated pressures. In that regard, land-based mitigation is important to address climate change. But its impact is limited, emphasizing the urgent need for rapid emission reductions. Furthermore, the rise in demand for wood products needs to be addressed. This will allow forests and agriculture to continue to play an important supplementary role in mitigation, while simultaneously enabling their adaptation to climate change, their sustained provision of ecosystem services, and the thriving of biodiversity.

Deutsche Zusammenfassung

Der Klimawandel stellt eine dringende Herausforderung für die Menschheit dar, und der Landsektor spielt eine Schlüsselrolle beim Klimaschutz. In dieser Hinsicht sind insbesondere Wälder von tragender Bedeutung. Ihr Beitrag zum Klimaschutz basiert auf drei fundamentalen Aspekten: der Kohlenstoffsenke in Wäldern selbst, der Kohlenstoffsenke in Holzprodukten und den Substitutionseffekten, wenn Holz kohlenstoff-intensive Nichtholzprodukte ersetzt. Trotz ihrer Relevanz bestehen noch immer große Unsicherheiten in Bezug auf optimale Strategien zum Klimaschutz durch Wälder und den Landsektor, sowie in Bezug auf das Gesamtpotenzial. Des Weiteren sind Wälder selbst anfällig für den Klimawandel und Anpassungsmaßnahmen sind nötig. Sie vollbringen außerdem zahlreiche wichtige Ökosystemleistungen, unter anderem Holzproduktion, lokale Klimaregulierung und die Speicherung und Filtration von Wasser. Forstwirtschaftsstrategien, welche all diese Aspekte berücksichtigen, werden auch mit dem Terminus "climate-smart forestry" bezeichnet.

Der weltweite Verlust von Biodiversität stellt eine weitere große Bedrohung dar und beeinträchtigt die Widerstandsfähigkeit von Ökosystemen, die Ernährungssicherheit und die menschliche Gesundheit. Um dieses Problem anzugehen, müssen auch die derzeitigen Landnutzungsstrategien verbessert werden, um eine Regeneration der biologischen Vielfalt zu gewährleisten. In der Europäischen Union (EU) werden die vielfältigen Anforderungen an den Landsektor durch neue Strategien formalisiert, die beispielweise Ziele für Kohlenstoffsenken, Klimaschutz durch Forstwirtschaft und die Förderung von Biodiversität vorsehen. Diese Ziele stellen eine komplexe Herausforderung für die Entwicklung von Landnutzungsstrategien dar. Die steigende Nachfrage nach Holzprodukten trägt zu dieser Komplexität bei und wirft die Frage auf, wie diese vielfältigen und manchmal gegensätzlichen Anforderungen miteinander in Einklang gebracht werden können.

In dieser Arbeit gehe ich diese Problematiken in einem europäischen Kontext an, indem ich 1) die wichtigsten Unsicherheiten im Bereich des waldbasierten Klimaschutzes quantifiziere, 2) eine Methodik zur Entwicklung von Strategien von climate-smart forestry vorstelle, welche für eine breites Spektrum von Klimaszenarien tragfähig sind, 3) die Auswirkungen verschiedener EU-Ziele auf den Landsektor bewerte und 4) die realistischen Potenziale landbasierter Klimaschutzmaßnahmen in Bayern abschätze. Meine Dissertation ist in vier Abschnitte unterteilt, die drei Forschungspapers sowie einen Abschnitt mit zusätzlichen Ergebnissen enthalten.

Das erste Paper befasst sich mit dem waldbasierten Klimaschutz, einem Thema, das noch immer mit großen Unsicherheiten und manchmal widersprüchlichen Aussagen in der wissenschaftlichen Literatur verbunden ist. Diese Unsicherheiten entstehen durch eine unvollständige Berücksichtigung relevanter Faktoren und durch den erheblichen Einfluss,

den einige wenige Annahmen auf die Ergebnisse haben können. Hier gehe ich auf diese Annahmen ein, um die erste Forschungsfrage zu beantworten: *Welche Auswirkungen haben die wichtigsten Faktoren des waldbasierten Klimaschutzes in mitteleuropäischen Wäldern auf dessen Potenziale und welche Empfehlungen für Maßnahmen können daraus abgeleitet werden?* Hierbei betrachte ich das Waldalter, die Waldarten, den Klimawandel, natürliche Störungen, die Durchforstungsintensität, die Holznutzung, Schadholzeinschläge und mehrere Dekarbonisierungsszenarien – von den derzeitigen Maßnahmen bis hin zu den Netto-Null-Zielen der EU.

Hierfür habe ich das dynamische globale Vegetationsmodell LPJ-GUESS um detailliertere Modellierungen von Holznutzung und Störungen erweitert. Anschließend führte ich faktorielle Simulationsexperimente durch, um die Auswirkungen der oben genannten Faktoren auf das Klimaschutzpotenzial zu quantifizieren und Empfehlungen für Klimaschutzstrategien abzuleiten. Die wichtigsten Ergebnisse deuten darauf hin, dass, obwohl die Störungsszenarien und das Dekarbonisierungstempo die wichtigsten Quellen der Unsicherheit sind, diese Unsicherheit bis 2050 klein genug bleibt, um Empfehlungen zu formulieren. Insbesondere haben Substitutionseffekte zumindest bis zu diesem Zeitpunkt einen entscheidenden Einfluss, sodass eine Verringerung von nachhaltiger Waldbewirtschaftung aus Sicht des Klimaschutzes derzeit nicht empfehlenswert ist (unter Vernachlässigung anderer Ökosystemleistungen). Die vermiedenen Emissionen durch die Verwendung von Holzprodukten haben eine größere Wirkung als eine erhöhte Kohlenstoffsenke des Waldes durch verminderte Durchforstung. Hinzu kommen beträchtliche positive Klimaauswirkungen einer verstärkten stofflichen (statt energetischen) Nutzung von Holz. Durch die erhöhte Kohlenstoffsenke in den Holzprodukten bleiben diese Auswirkungen auch bestehen wenn Substitutionseffekte abnehmen und schließlich verschwinden. Meine Ergebnisse zeigen auch, dass Aussagen bezüglich der Substitutionseffekte stark von der Höhe der verwendeten Substitutionsfaktoren abhängen. Deren verbesserte Quantifizierung sollte daher priorisiert werden. Klimaschutzstrategien müssen auch Waldtyp und -alter berücksichtigen, da diese einen Einfluss auf Wachstumsverhalten und Produktportfolios haben. Bei längeren Zeithorizonten werden die Unsicherheiten schließlich erheblich zunehmen. Für die Planung von Maßnahmen über das Jahr 2050 hinaus ist es daher empfehlenswert, robuste Methoden und Risikodiversifizierungsstrategien zu verwenden.

Im zweiten Paper habe ich den Schwerpunkt des Klimaschutzes auf den der climate-smart forestry erweitert. Diese enthält neben Klimaschutz auch die Anpassung der Wälder an den Klimawandel und die Berücksichtigung von Ökosystemleistungen wie lokaler Klimaregulierung, Wasserkreislauf, Holzproduktion, und die Schaffung oder Beibehaltung von Lebensräumen für die biologische Vielfalt. Ich habe mich speziell mit der folgenden Frage beschäftigt: *Wie können wir trotz der großen Unsicherheiten in Bezug auf die künftige Klimaentwicklung Strategien für climate-smart forestry in Europa entwickeln? Und was sind die Auswirkungen auf die europäische Waldlandschaft?* Unter Verwendung von LPJ-GUESS und den im ersten Paper vorgestellten Erweiterungen habe ich die Niederwaldbewirtschaftung in das Modell implementiert und außerdem Indikatoren für die oben genannten Ökosystemleistungen berechnet. Diese wurden sowohl aus bestehenden als auch aus neu eingeführten Outputvariablen von LPJ-GUESS bestimmt.

Um Strategien zu entwickeln, die diese vielfältigen Ziele unter einer Vielzahl von Klimawandelszenarien berücksichtigen, habe ich robuste Optimierung mit multi-kriterieller Entscheidungsanalyse kombiniert. Diese robuste multikriterielle Optimierung wurde angewandt, um optimierte Waldbewirtschaftungsportfolios zu berechnen, die unabhängig vom Klimaszenario eine ausgewogene Bereitstellung von Klimaschutz, Klimaanpassung und anderen Ökosystemleistungen gewährleisten. Die Anwendung dieser Methodik auf europäischer Ebene ergab erhebliche Veränderung der Waldzusammensetzung hin zu mehr Laubwäldern und unbewirtschafteten Wäldern, da beides besonders vorteilhaft für die Anpassung, die biologische Vielfalt und die Kohlenstoffspeicherung war. Darüber hinaus führte diese Methodik zu diversifizierten Portfolios, welche die Risiken effektiv streuen und dadurch Unsicherheiten verringern. Die berechneten Portfolios hatten allerdings auch einen Rückgang der Holzproduktion zur Folge. Dies macht deutlich, dass bei der Entwicklung von Strategien für climate-smart forestry in Europa Kompromisse eingegangen werden müssen.

Im dritten Paper habe ich die Auswirkungen der neuen EU-Strategien für den Landsektor, die Wälder und die biologische Vielfalt bewertet. Diese Strategien sehen Auflagen für die Landnutzung vor, darunter Ziele für eine verstärkte Kohlenstoffsenke und strengen Waldschutz, aber auch die Aufrechterhaltung der Holzproduktion und die vermehrte Nutzung von Holz für den Klimaschutz. Aufbauend auf den Ergebnissen des zweiten Papers habe ich mich mit den folgenden Forschungsfragen beschäftigt: *Wie können wir climate-smart forestry mit den neuen EU-Strategien in Einklang bringen? Und was sind die Auswirkungen?*

Um diese Fragen zu beantworten, habe ich die Optimierung aus dem zweiten Paper um strikte Nebenbedingungen erweitert, welche den geplanten strengen Schutz von Wäldern und die Erntemengen beschränken. Diese Beschränkungen mussten auf dem europäischen Kontinent in allen Klimaszenarien eingehalten werden. Die wichtigsten Ergebnisse waren, dass Beschränkungen ähnlich derer der EU-Strategien zwar theoretisch laut dem Modell miteinander vereinbar sind, aber dazu führen, dass hochproduktive Wälder, insbesondere in Südskandinavien, auf Holzproduktion reduziert werden müssen. Dies widerspricht einer gerechte Verteilung der Schutzgebiete und würde sich negativ auf die biologische Vielfalt und andere Ökosystemleistungen in diesen Regionen auswirken.

Des Weiteren führte die Berücksichtigung dieser zusätzlichen Einschränkungen zu weniger diversifizierten Portfolios, was wegen der Unsicherheiten bezüglich der Zukunft ein Risiko darstellt. Folglich unterstreichen meine Ergebnisse die Notwendigkeit, den Druck auf die Wälder zu mindern, beispielsweise durch verstärkte Investitionen in andere erneuerbare Energien als Holz oder die Verringerung der Nachfrage nach Einweg-Holzprodukten wie Verpackungen.

In einem letzten Schritt quantifizierte ich landbasierte Klimaschutzoptionen in Bayern, um die folgende Forschungsfrage zu beantworten: *Welches sind die Potenziale verschiede-*

ner wald- und landbasierter Klimaschutzmaßnahmen in Bayern? Unter Verwendung von LPJ-GUESS mit einer verbesserten Parametrisierung der Bioenergiepflanze *Miscanthus* habe ich fünf Klimaschutzmaßnahmen auf realistischen Skalen im gesamten Bundesland simuliert: erhöhte stoffliche Nutzung von Holz, Anbau von Bioenergiepflanzen mit und ohne CO_2-abscheidung und -speicherung (carbon capture and storage, CCS), Wiederaufforstung und reduzierte Düngung von Nutzpflanzen. Die Ergebnisse deuten darauf hin, dass der Landsektor in Bayern bis zum Jahr 2100 etwa sechs Jahre bayerischer Treibhausgasemissionen vermeiden bzw. kompensieren kann. Darüber hinaus zeigten die Ergebnisse eine abnehmende Relevanz von landbasierten Klimaschutzmaßnahmen aufgrund der wahrscheinlich abnehmenden Substitutionsfaktoren. Dies unterstreicht die Notwendigkeit, diese Strategien schnell umzusetzen, um ihre Wirkung zu maximieren. Es ist jedoch entscheidend, die physikalischen Grenzen solcher Strategien zu erkennen und sich bewusst zu machen, dass landbasierter Klimaschutz nur eine Ergänzung zu einer schnellen und starken Reduzierung von Treibhausgasemissionen sein kann.

Zusammengefasst befasst sich meine Arbeit mit zentralen Fragen zur Rolle des Landsektors, insbesondere der Wälder, im Hinblick auf den Klimaschutz und die Förderung von biologischer Vielfalt und Ökosystemleistungen. Sie zeigt die potenziellen Auswirkungen von Klimaschutzstrategien in Mitteleuropa auf und gibt Empfehlungen zur Verbesserung waldbasierter Klimaschutzmaßnahmen. Darüber hinaus bietet sie eine Methodik zur Entwicklung von Strategien für climate-smart forestry in Europa und zeigt auf, dass schnelle Handlungen hierfür erforderlich sind. Meine Arbeit hebt jedoch auch die Vielzahl von Anforderungen an den Landsektor hervor und unterstreicht die Notwendigkeit, den damit verbundenen Druck zu verringern. In dieser Hinsicht ist Klimaschutz durch Landnutzung zwar äußerst wichtig, seine Wirkung jedoch begrenzt. Dies macht deutlich, dass eine schnelle Reduzierung von Teibhausgasemissionen dringend nötig ist. Darüber hinaus muss die steigende Nachfrage nach verschiedensten Holzprodukten beschränkt werden. Auf diese Weise können Wälder weiterhin eine wichtige Rolle beim Klimaschutz spielen, während gleichzeitig ihre Klimaanpassung, Ökosystemleistungen und die biologische Vielfalt gefördert werden.

Contents

List of Figures

List of Tables

Glossary

BD broad-leaved deciduous

BE broad-leaved evergreen

BECCS bioenergy with carbon capture and storage

CCS carbon capture and storage

DBH diameter at breast height

DGVM dynamic global vegetation model

ES ecosystem service

ESI ecosystem service indicator

ESM Earth system model

EU European Union

FPC foliar projective cover

GHG greenhouse gas

hlp harvests for long-lived wood products

IPCC Intergovernmental Panel on Climate Change

LAI leaf area index

MCDM multi-criteria decision making

ND needle-leaved deciduous

NE needle-leaved evergreen

NPP net primary productivity

PAI plant area index

PES payments for ecosystem services

PFT plant functional type

RCP Representative Concentration Pathway

RCP2.6 Representative Concentration Pathway 2.6

SSP shared socio-economic pathway

List of Publications

Published and accepted publications included in this thesis:

- **Gregor, K.**, Knoke, T., Krause, A., Reyer, C. P. O., Lindeskog, M., Papastefanou, P., Smith, B., Lansø, A.-S., and Rammig, A. (2022). Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate. Earth's Future, 10 (9), 1–25. https://doi.org/10.1029/2022EF002796

- **Gregor, K.**, Krause, A., Reyer, C.P.O. et al. Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests. Carbon Balance and Management 19, 10 (2024). https://doi.org/10.1186/s13021-023-00247-9

Manuscripts included in this thesis that are not yet accepted for publication:

- **Gregor, K.**, Reyer, C. P. O., Nagel, T. A., Mäkelä, A., Krause, A., Knoke, T., and Rammig, A. (in review). Reconciling climate-smart forestry with EU forest, biodiversity and climate legislation.
  Note: the paper was accepted in the meantime and is available at https://doi.org/10.1111/gcb.17431

Additional publications published during the PhD period of the candidate that are not part of this thesis:

- Reichert, T., Rammig, A., Papastefanou, P., Lugli, L. F., Darela Filho, J. P., **Gregor, K.**, Fuchslueger, L., Quesada, C. A., and Fleischer, K. (2023). Modeling the carbon costs of plant phosphorus acquisition in Amazonian forests. *Ecological Modelling*, *485*, 110491. https://doi.org/10.1016/j.ecolmodel.2023.110491

- Tschumi, E., Lienert, S., Bastos, A., Ciais, P., **Gregor, K.**, Joos, F., Knauer, J., Papastefanou, P., Rammig, A., Williams, K., Xu, Y., Zaehle, S., and Zscheischler, J. (2023). Large Variability in Simulated Response of Vegetation Composition and Carbon Dynamics to Variations in Drought-Heat Occurrence. *Journal of Geophysical Research: Biogeosciences*, *128*(4), e2022JG007332. https://doi.org/10.1029/2022JG007332

- Krause, A., Papastefanou, P., **Gregor, K**., Layritz, L. S., Zang, C. S., Buras, A., Li, X., Xiao, J. and Rammig, A. (2022). Quantifying the impacts of land cover change on gross primary productivity globally. *Sci Rep* **12**, 18398. https://doi.org/10.1038/s41598-022-23120-0

Additional publications currently in review written during the PhD period of the candidate that are not part of this thesis:

- Meyer, B. F., Buras, A., **Gregor, K.**, Layritz, L. S., Principe, A., Kreyling, J., Rammig, A., and Zang, C. S. (in review). Frost matters: Incorporating late-spring frost in a dynamic vegetation model regulates regional productivity dynamics in European beech forests. *Biogeosciences*, https://doi.org/10.5194/bg-2023-139

- Bouriaud, O., Schulze, E.D., **Gregor, K.**, Boukhris, I., Högberg, P., Irslinger, R., Papastefanou, P., Pongratz, J., Rammig, A., Valentini, R., and Körner, C. (in review). Selective harvest regulates leaf area without affecting ecosystem fluxes. *Global Biogeochemical Cycles*.

1 Introduction

1.1 Climate change: a pressing issue for humanity

Global mean surface temperatures have risen by about 1.1 °C since the pre-industrial era (IPCC, 2023a). Surface temperature over land has warmed faster than that over the oceans, reaching warming levels of about 1.6 °C (IPCC, 2023a). Over recent years, the impacts of this change in climate, including sea level rise, more frequent extreme weather events, and species losses, have become increasingly evident (IPCC, 2023a; Nicholls & Cazenave, 2010). Furthermore, the occurrence of hot extremes has increased virtually everywhere across the globe since 1950 with high confidence in the human contribution to these observed changes (IPCC, 2023a), resulting in a rise of heat-related human mortality on every continent (Vicedo-Cabrera et al., 2021). In general, there has been a notable increase in the frequency and severity of extreme weather events, which has already exposed millions of people to acute water and food security situations. Currently, over three billion people live in regions highly vulnerable to climate change (IPCC, 2023a). These impacts will be exacerbated with increasing future warming which is currently estimated to reach 3.2 °C by 2100 (IPCC, 2023a).

Moreover, there are numerous so-called "feedback loops" in the climate system, many of which are not yet fully understood. Among these, some are negative feedback loops, meaning they mitigate the warming effect. One example is the CO_2 fertilization effect, in which rising atmospheric CO_2 concentrations enable plants to absorb more carbon from the atmosphere, thereby acting as a buffer against anthropogenic emissions (Haverd et al., 2020). Another example of a negative feedback loop relates to clouds: elevated temperatures amplify evapotranspiration, leading to increased cloud cover in the lower atmosphere, resulting in a cooling effect (Laguë et al., 2021). On the other hand, there are numerous positive, i.e., self-amplifying, feedback loops. One example is the ice-albedo feedback, where increased warming reduces ice cover on the planet, thereby lowering the planet's reflectivity (albedo), causing a greater absorption of shortwave solar radiation and consequently intensifying warming (Bonan, 2016). A similar process occurs with the tundra-taiga feedback. Higher temperatures cause the taiga's tree line to expand northward into the tundra, also reducing albedo (Harding et al., 2002). Elevated temperatures also trigger increased thawing of permafrost soils, releasing substantial emissions of CO_2 and methane, thereby amplifying the greenhouse effect (Schuur et al., 2022). There is a multitude of other positive feedbacks and substantial uncertainty surrounds their impact on global temperatures (IPCC, 2023a). Furthermore, every further increase in temperature also increases the possibility of tipping points in the Earth system (Armstrong McKay et al., 2022; Lenton et al., 2019).

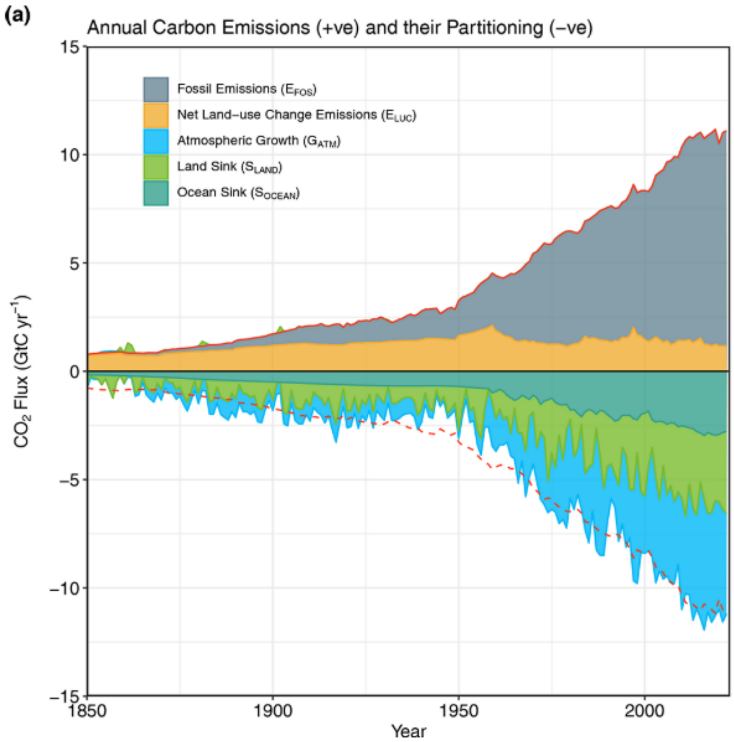

Figure 1.1: Carbon emissions since 1850. Figure taken from Friedlingstein et al. (2023).

The evidence is unequivocal that the observed warming is a direct consequence of increasing anthropogenic greenhouse gas (GHG) emissions, primarily CO_2 (IPCC, 2023a). While the majority of these emissions stem from the combustion of fossil fuels, a substantial portion (over 10%) currently arises from land use change, mainly due to deforestation (Friedlingstein et al., 2023). It is worth noting that this fraction used to be considerably higher, e.g., accounting for about 50% of total emissions in the 1950s (Fig 1.1).

Countries globally are starting to take action to combat climate change, with every nation worldwide having implemented at least one climate change law (Eskander & Fankhauser, 2020). In 2020, these laws included legislation to reduce GHG emissions in 56 countries, collectively covering over 50% of global emissions (IPCC, 2023b). This legislation has already demonstrated effectiveness: in 2016, an estimated 1.6 GtC of emissions were avoided because of existing legislation, a substantial reduction considering the roughly 10 GtC of CO_2 emissions of that year (Eskander & Fankhauser, 2020; Le Quéré et al., 2018). Nevertheless, total carbon emissions from fossil fuels and land use change have still continued to rise in the past decades, but have leveled around 11 GtC since 2010 (Fig 1.1).

Levels of ambition in climate laws and targets vary among countries. For instance, the European Union (EU) has established one of the most ambitious targets, aiming to achieve net zero emissions by 2050 (European Commission, 2019). This means that there will still remain some gross GHG emissions, but they will be offset by negative emission solutions. These solutions encompass technological approaches like direct air capture or natural solutions such as afforestation, changes in forest management, or ecosystem restoration (Griscom et al., 2017). Schreyer et al. (2020) recently estimated that achieving the EU's net-zero target requires that negative emission solutions offset about 20% of present-day emission values.

The current commitments of the world's countries are, however, not sufficient to meet the goal of the Paris Agreement to limit global warming to "well below 2 °C" above pre-industrial levels (IPCC, 2023a; United Nations, 2015). While some countries have already started to decrease their emissions, others have yet to reach their peak (refer to Friedlingstein et al. (2023) for CO_2, and Van Dingenen et al. (2018) for CH_4). As a result, the Intergovernmental Panel on Climate Change (IPCC) indicates that a wide range of climate change scenarios remains plausible. These scenarios are most often depicted by five shared socio-economic pathways (SSPs) which envision different developments of the world in terms of population, economy, emissions, and other factors (Popp et al., 2014). Depending on the SSP, challenges to mitigation and adaptation will vary, resulting in differing levels of radiative forcing and hence global warming.

Regardless of the scenario, adaptation to climate change will be necessary and is already underway. A variety of strategies are being employed for adaptation. Prominent examples in Europe include the conversion of forests into more resilient, climate-adapted mixed forests (e.g., BMEL, 2014), and implementing coastline protections such as the sea wall of Venice (Molinaroli et al., 2019). Other measures involve shoreline restorations and the promotion of green areas, creation of air corridors, and reduction of soil sealing in urban areas (Carter, 2011).

However, it is crucial to avoid the pitfalls of maladaptation when implementing adaptation strategies. Instances of maladaptation that have already occurred include the installation of costly irrigation systems in drought-prone areas which are projected to be insufficient for future conditions. Another example is the construction of seawalls that increase risks for surrounding unprotected areas, or exacerbate long-term exposure by trapping water masses within them (IPCC, 2023b; Piggott-McKellar et al., 2020).

Naturally, the extent of required adaptation measures diminishes as the degree of future climate change decreases, and it has been demonstrated that the advantages of mitigation surpass the associated costs (IPCC, 2023b).

1.2 Loss of biodiversity

Global loss of biodiversity poses another major threat to humanity, although it is sometimes overlooked in discussions of global change (Sage, 2020). One of the most apparent indicators of this threat is the alarming decline in populations of plant and animal species. For instance, in the past 50 years, the populations of terrestrial vertebrates have declined by 40% and currently, 25% of animals and plants are threatened (IPBES, 2019). These rates of extinction are unprecedented in millions of years and at least 10–100 times higher than the natural rate over the last 10 million years. Some experts even suggest that Earth is on the brink of a “sixth mass extinction” (Ceballos et al., 2015; IPBES, 2019; Sage, 2020).

There are two main reasons for the global loss of biodiversity. Firstly, human activities have altered up to 75% of all ice-free land (Ellis et al., 2021; IPBES, 2019; Luyssaert et al., 2014), thereby also decimating the habitats of many species. Land use and land use change have been the main drivers of biodiversity loss (Pereira et al., 2012; Vitousek, 1994). Secondly, climate change has another major impact, for example because rising temperatures force species out of their natural habitats (Diamond, 2018; Parmesan et al., 1999; Urban, 2015; Warren et al., 2018). Hence, it seems obvious that any climate change mitigation will also be beneficial for biodiversity. However, this is not always the case. Some mitigation efforts are undoubtedly helpful, but some mitigation pathways also lead to considerable land use change, posing an additional threat to biodiversity (Hof et al., 2018; Ohashi et al., 2019). For example, scenarios like the SSPs for the Representative Concentration Pathway 2.6 (RCP2.6) have a strong focus on bioenergy, requiring substantial amounts of land that could otherwise serve as habitat for various species.

Forests are particularly important ecosystems for biodiversity, because they provide habitat for 70–80% of the world’s amphibians, birds, and mammals (FAO, 2022c). Therefore, global deforestation has been and still is a major driver of biodiversity loss, with global forest cover now being at approximately 68% of pre-industrial levels (IPBES, 2019). It is not only deforestation but also the management of existing forests that impacts biodiversity. For instance, unsustainable forestry practices are increasing globally, while the fraction of intact forests decreases (IPBES, 2019). In Europe, there is a focus on forest management that emulates natural forest characteristics (Hengeveld et al., 2012), and large fractions of the forest are classified as semi-natural (Forest Europe, 2020). Still, the majority of assessed conservation statuses of habitats and species in Europe’s forests remain unfavorable (EEA, 2015).

Fortunately, efforts are being made to promote biodiversity in European forests, with an increasing designation of areas for conservation and the implementation of new strategies (European Commission, 2020; Forest Europe, 2020). Numerous forest management options exist to enhance biodiversity while maintaining timber provision. These include management practices that yield forests with natural characteristics, featuring diverse tree species and sizes, abundant deadwood (including large standing dead trees), and ample large living trees. Encouragingly, there is a positive correlation between forest biodiversity and productivity (Liang et al., 2016) suggesting that win-win strategies may be possible.

1.3 The importance of ecosystem services

For some, biodiversity may appear to be merely a desirable feature of ecosystems, but this falls far short from the truth. Biodiversity is not only about the variety and complexity of ecosystems, and goes beyond its vital role for other species’ survival; it is essential for human well-being. In fact, diverse and healthy ecosystems offer a variety of critical services to humanity, termed ecosystem services (ESs). To illustrate, 75% of global food crops rely on animal pollination, making their extinction rates a direct threat to human survival (IPBES, 2019). ESs can generally be defined as all benefits that people obtain from ecosystems (Millenium Ecosystem Assessment, 2005) including those that are instrumental in sustaining human life (Díaz et al., 2006). While the importance of animal pollination is universally recognized, the significance of other ESs will depend on the context, with different groups or societies valuing them differently (Díaz et al., 2006). For instance, the ES of wildlife population control by re-introduced wolves is seen positive by some stakeholders, but negative by others, for example due to potential attacks on livestock (Díaz et al., 2018).

ESs are typically grouped into four categories: *regulating services* such as the control of local climate or air quality, *provisioning services*, including the provision of food or wood, *cultural services* offering educational or recreational value, and *supporting services* like, for instance, soil formation, which do not provide a direct benefit to people, but facilitate the provision of other ESs (Millenium Ecosystem Assessment, 2005).

Alongside biodiversity, ecosystem functions and services are changing, and often deteriorating worldwide (IPBES, 2019). Nevertheless, they have gained increasing recognition through global initiatives like the Millenium Ecosystem Assessment (2005) and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2019). Also trends in online searches for the term "ecosystem services" show increasing interest (Fig 1.2). Simultaneously, there has been a "steady growth in the number of articles and reports on the monetary valuation of natural resources, ecosystem services and biodiversity" (de Groot et al., 2012). Apparently, awareness of human reliance on nature is at an all-time high (Guerry et al., 2015) and this growing awareness has led to the enactment of legislation to protect nature, such as the Natura2000 program (Sundseth, 2008). Most recently, the European Commission has introduced new strategies, including the *EU Biodiversity Strategy for 2030* and the *EU Forest Strategy for 2030*. Among the key objectives, they aim to protect 30% of the EU's land and sea area, with 10% strictly protected, and promote the development of ecological corridors (European Commission, 2020).

Improving the state of ESs necessitates their quantification, which is a complicated endeavor, especially for supporting or regulating ESs (Díaz et al., 2006). Humans often aim to quantify things in monetary values. For ESs, this could however yield in bad measurements as it tends to favor those ESs that are easily quantifiable (DeFries et al., 2005). On the other hand, while not ideal, assigning a monetary value may make the concept more comprehensible, raise awareness, and convey the significance to policy makers (de Groot et al., 2012). For example, knowing that over half of the global gross domestic product highly or moderately depends on nature (WEF, 2020) or understanding that the cost-to-benefit ratio of conserving nature is 1:100 (Balmford et al., 2002) makes it evident that increased investments in nature protection are imperative.

Other means of measurement may involve collecting expert opinions by local stakeholders or simply by assessing relative changes compared to some baseline. In any case, it is crucial to assess ESs and include them into the decision-making process (Martinez-Harms

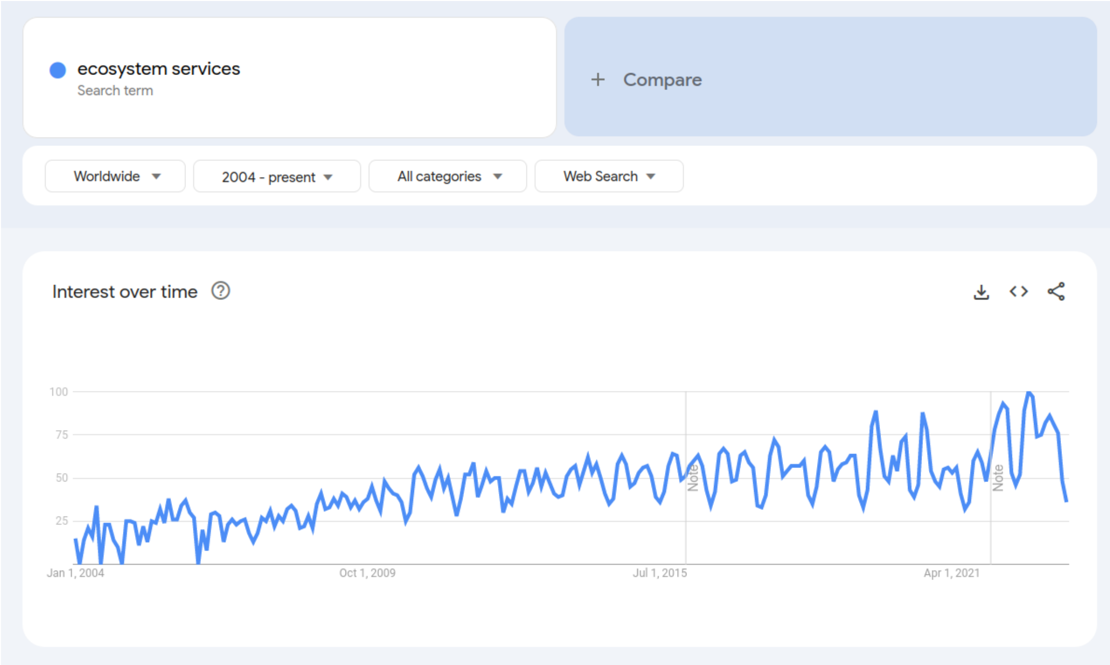

Figure 1.2: The search term "ecosystem services" has gained attention worldwide over the past 20 years. The y-axis shows the relative interest in the topic, with 100 reflecting the highest search volume (highest interest) of the considered period. © Google.

et al., 2015; Ruckelshaus et al., 2015). In fact, this has been emphasized as crucial for addressing the central challenge of sustaining the well-being of present and future generations (Guerry et al., 2015).

In this thesis, I quantify ES performance based on indicator functions and a normalization approach, as detailed in sections 2.3 and 2.5.2. This approach facilitates the comparison of ES provision against both the least and most favorable outcomes across management and climate scenarios. Moreover, I included the possibility to assign weights to different ESs, allowing for adjustments in their priority.

1.4 Land use and land use changes and their impact on biodiversity and ecosystem services

As previously mentioned, human activities have significantly altered substantial shares of the land area, for resource extraction, urban development, and the creation of agricultural fields and pastures. Such land uses, and the changes they undergo, have tremendous impacts on climate change, biodiversity, and ecosystem services. Nevertheless, land use also provides various ecosystem services as well as opportunities for the adaptation to, and mitigation of, climate change.

In this thesis, I assess these impacts and opportunities in conjunction with the provision of ecosystem services. The main focus lies on forests, but a secondary exploration of agriculture is also undertaken. Peatlands, another important land cover in terms of climate change mitigation (Loisel et al., 2021; Pan et al., 2011), are not within the scope of this thesis.

1.4.1 Forests and forest management

Global forest area currently amounts to around 4 billion hectares, equivalent to approximately 31% of the Earth's land surface (FAO, 2022c). In the absence of human influence, their share would be substantially higher, as they would be the natural vegetation in most parts of the world (Hengl et al., 2018). In Europe, after a long history of deforestation and some forest expansion in the past century or so, around 35% of the land area is covered with forests today. This percentage varies substantially among countries, from below 15% in Great Britain up to 75% in Scandinavia (Forest Europe, 2020; Kirby & Watkins, 1998; Nabuurs et al., 2001).

Forests offer important ESs to humanity. Most notably, through the process of photosynthesis, they provide us with oxygen. They play a pivotal role in the global water cycle, by regulating both quality and quantity of water in their region. Forests also offer recreational value, support pollination and aid in pest regulation (Binder et al., 2017; Brockerhoff et al., 2017). They are also critical for global biodiversity, serving as habitats for a wide range of species (Binder et al., 2017; Brockerhoff et al., 2017). In many regions, forests also act as a natural protection against soil erosion, landslides, and avalanches (Bugmann et al., 2017).

Managed forests provide another highly important ES: timber production. Wood harvested from these forests serves a variety of purposes, including construction, furniture, pulp and paper, and woody bioenergy. In 2015, these products contributed to approximately 0.7% of the European gross domestic product and supported 2.6 million jobs (Forest Europe, 2020). The demand for wood products has been on the rise over the past decades, a trend that is projected to continue (IEA, 2022). This global surge in demand stems from an increased need for packaging, sanitary paper, and construction wood (FAO, 2022c; IEA, 2022). In Europe, the demand is also driven by a growing demand for woody bioenergy (FAO, 2022c) which currently constitutes an important energy source in Europe, providing 60% of Europe's renewable energy (European Commission, 2021b), but see my discussion on this topic in section 4.2.2.

Finally, also non-wood forest products, such as honey, fruits, or game meat, contribute to the forest economy. These products were reported to have a market value of 4 billion Euros in 2015, representing a smaller but still noteworthy economic facet of European forests (Forest Europe, 2020).

Large fractions of the world's forests are managed for timber production. In Europe, about 75% of forests are available for wood supply, with higher proportions found in central-west Europe (Forest Europe, 2020). The concept of sustainable forest management plays a pivotal role in this context. Sustainable management involves ensuring that the volume of timber harvested is less than what naturally regenerates within one year. In Europe, this principle is generally adhered to: roughly 73% of the net annual increment (inside the forests available for wood supply) are harvested each year (Forest Europe, 2020). This practice contributes to the sustained growth of the forest's growing stock.

Sustainable forest management encompasses a variety of approaches. Its focus primarily lies on the sustainable removal of timber, with no direct implications for the sustainability of other variables. In Europe, the actual management regimes are highly diverse, including intensive even-aged forestry, selective logging, coppice management, short-rotation forestry, and close-to-nature approaches (Hengeveld et al., 2012; Schelhaas et al., 2018; Suvanto et al., in review). Even within similar management approaches, the actual applications vary widely, not least due to the fact that European forests are owned by millions of private owners and thousands of public entities (Schelhaas et al., 2018).

1.4.2 Agriculture

Large fractions of historically forested areas have been cleared globally for agricultural purposes, leading to substantial carbon emissions (Fig. 1.1, Friedlingstein et al., 2023; Kirby & Watkins, 1998). But these agricultural land use changes do not only exacerbate climate warming; they also stand as the primary driver behind the ongoing global biodiversity loss (IPBES, 2019).

Similar to the area of forests, agricultural land now also covers about one-third of the global land area (FAO, 2022b) with a somewhat higher fraction in the EU (39% in 2018, Eurostat (2021b)). Agriculture provides essential ESs to humanity, including the production of food, fodder, materials, and energy. However, these beneficial services come at a substantial environmental cost, particularly concerning their predominantly adverse impacts on climate, as I will elaborate on later.

In Europe, the forest area has been expanding in recent decades (Forest Europe, 2020). This forest expansion is primarily driven by the abandonment of agricultural land and is predicted to persist in the next decades, albeit with large uncertainties regarding both the extent and type of succession, e.g., natural regrowth or urban development (van der Zanden et al., 2017).

Nevertheless, farmland has the potential to be highly biodiverse. The decline in farmland biodiversity of the past 60 years was caused by the intensification of agricultural practices, such as increased pesticide and fertilizer usage, the cultivation of homogeneous crop types, and the elimination of non-cropped areas (Benton et al., 2003; Scheper et al., 2013). On the other hand, for instance, adequately managed pastures can be highly biodiverse (Klein et al., 2020; Nerlekar & Veldman, 2020; Seibold et al., 2019; Weisser et al., 2017) and offer cultural value and other ecosystem services (Bengtsson et al., 2019; Pellaton et al., 2022). As a consequence, projects have been initiated to promote biodiversity in agricultural landscapes, for instance through flower strips or organic farming (Scheper et al., 2013). Such measures can ensure the sustained provision of agricultural ESs like food production while allowing biodiversity to thrive.

1.5 The relationship between land use and climate change

1.5.1 Climate change mitigation through forestry

In recent years, one ES provided by forests has gained significant attention due to its role in mitigating climate change: carbon sequestration through photosynthesis. There are three primary ways in which this carbon uptake contributes to climate change mitigation:

1. The forest carbon sink: Forests store about 861 GtC globally in vegetation, soil, and deadwood (Pan et al., 2011). They currently are the main contributor to the terrestrial carbon sink which takes up about 25% of annual carbon emissions (Friedlingstein et al., 2023; Pan et al., 2011). About the same amount is taken up by the oceans and the rest by the atmosphere (Friedlingstein et al., 2023). Remarkably, even ancient forests, like the Amazon rainforest, which should theoretically reach carbon balance, continue to act as carbon sinks, most likely due to the CO_2-fertilization effect (Brienen et al., 2015; Haverd et al., 2020; Hubau et al., 2020; Walker et al., 2021), but see also McGrath et al. (2024). Nevertheless, this increased carbon uptake and plant growth may lead to shorter lifespans and, consequently, to a lagged effect, potentially neutralizing these carbon gains (Brienen et al., 2020; Bugmann & Bigler, 2011; Körner, 2017). Climate change can have varied effects on this carbon sink. In some regions of Europe, warmer temperatures and a prolonged growing season may enhance the sink (Linderholm, 2006). However, excessively high temperatures could in turn inhibit photosynthesis and decrease the sink (Duffy et al., 2021). Also other adverse climate change effects such as increased evaporative demand appear to reverse the positive trend in carbon uptake (Rahmati et al., 2023). Furthermore, natural and anthropogenic disturbances, such as wildfires and deforestation, significantly decrease the global forest carbon sink (Nabuurs et al., 2013).

2. The carbon sink in wood products: When wood is used for long-lived products such as furniture or construction materials, the entire pool of wood products can act as a carbon sink. Globally, this sink is estimated at around 91 MtC/yr (Johnston & Radeloff, 2019), constituting about 0.9% of global carbon emissions (Friedlingstein et al., 2023). Recent estimates in Europe suggest that the product carbon sink is approximately 11 MtC/yr, making up also approximately 1.33% of the total carbon emissions (Grassi et al., 2021). The magnitude of this sink depends heavily on the types of wood products produced from the timber, with larger sinks resulting from increased use in construction or furniture manufacturing rather than for pulp, paper, or fuel.

3. Substitution Effects: The carbon balance is also influenced by substitution effects that occur when wood products replace fossil fuels or carbon-intensive materials like concrete and steel, which collectively contribute to about 14% of global emissions (IEA, 2020; Lehne & Preston, 2018). Substitution effects are calculated using displacement factors (or substitution factors), measured in units of tC/tC, signifying the amount of emissions that one tonne of carbon in wood products can potentially avoid. Here, the product types, their lifetimes, and their end-of-life fate play crucial roles. In a comprehensive review, Sathre and O'Connor (2010) identified displacement factors to typically range between 1.0 to 3.0 tC/tC when wood is used as a material, though more recent studies often employ displacement factors around 1 tC/tC. When wood replaces fossil fuels, typical values range between 0.5 and 1.0 tC/tC.

Despite the increasing awareness of the carbon mitigation potential of forests, there is no definitive consensus in the literature on its amount and the best strategies to enhance it. This is due to the multitude of driving factors and the complexity of their interactions, which is why this is one key topic of my thesis (section 3.1). Firstly, the age and type of forests are pivotal factors, with some studies emphasizing the crucial role of the strong carbon sink in young forests for climate mitigation (Matsumoto et al., 2016; Schulte et al., 2022). Nevertheless, also old-growth forests have been found to still sequester significant amounts of carbon (Luyssaert et al., 2008; Luyssaert et al., 2021).

The debate extends to the realm of forest management. Some studies suggest that increasing rotation times or reducing harvest intensity is the most effective approach for climate mitigation (Dugan et al., 2018; Schulte et al., 2022; Skytt et al., 2021; Soimakallio et al., 2021). Others conversely find that increases in harvest have a higher positive impact on the carbon balance (Gustavsson et al., 2021; Petersson et al., 2022). In large parts, this discrepancy is due to widely different assumptions about substitution effects, addressing the types of products being replaced and the corresponding emissions being avoided. For instance, Gustavsson et al. (2021) indicated a greater effect when woody biomass replaced coal rather than gas, or when wood was used for modular timber construction. Petersson et al. (2022) demonstrated that the choice of the substitution factor substantially altered the outcomes, although this impact was weaker in Schulte et al. (2022).

The influence of climate change on the mitigation potentials requires further exploration. Some studies focus on a single climate scenario, and some only consider the positive effects such as prolonged growing seasons, overlooking negative effects such as droughts or increased forest disturbances (Gustavsson et al., 2021) or including them in a simplified manner only (Petersson et al., 2022). Other studies do not consider climatic change at all in their analyses (Dugan et al., 2018; Schulte et al., 2022; Skytt et al., 2021; Soimakallio et al., 2021).

Consequently, substantial uncertainties persist regarding the carbon mitigation potentials of forests, due to the unclear impact of various factors. To address these gaps in my thesis, I used a vegetation modeling approach and factorial simulation experiments to disentangle and quantify the impacts of different factors on the mitigation potential of temperate managed forests in Europe.

1.5.2 The influence of biogeophysical effects on the climate

Beyond the carbon cycle, land use also affects the climate through biogeophysical effects. Specifically, different land uses and management regimes substantially influence the energy balance of the land surface through albedo, surface roughness, and evapotranspiration (Bonan, 2016; Luyssaert et al., 2014).

Albedo is defined as the amount of shortwave solar radiation that is reflected back into space. Its values range from 0 to 1, representing the fraction of reflected incoming radiation, with higher reflectance resulting in cooling. Different land covers exhibit widely differing albedo values, making it an important aspect of land use changes. For instance, the albedo of forested land ranges between 0.05 and 0.2 whereas the albedo of cropland typically lies between 0.18 and 0.26 (Bonan, 2008). Within specific land covers, different species will also exert differing albedo characteristics. For instance, temperate evergreen trees have a lower albedo than deciduous ones (Boisier et al., 2013).

Another crucial aspect is the cooling effect of evapotranspiration, which results because the conversion of water from its liquid state to a gaseous form absorbs a substantial amount of energy from the surroundings. This process occurs through the evaporation of water from the soil and intercepted water from the canopy, or through plant transpiration. The type of forest not only influences evapotranspiration rates (e.g., no transpiration by deciduous trees after leaf shedding), but also the atmospheric transport of energy. Namely, a forest's structural features (also called "surface roughness") determine the atmospheric resistance of the land, affecting the transport of latent and sensible heat and thereby surface air temperature (Raupach, 1994).

Although the impact of CO_2 on climate is estimated to exceed that of biogeophysical effects globally (Pongratz et al., 2010), these biogeophysical factors have a significant regional impact (Winckler et al., 2019). It was found that boreal forests have a positive (i.e., warming) climate effect through their low albedo (especially in winter, with a highly reflective snow cover). In contrast, tropical forests have a negative (i.e., cooling) effect through their evaporative cooling and surface roughness (Bonan, 2008; Davin & de Noblet-Ducoudre, 2010).

The previously mentioned mitigation studies did not include these biogeophysical effects. Nonetheless, they should be considered when assessing multi-functional forests. In this thesis I have integrated the carbon mitigation assessment with other ecosystem services, including local climate regulation from biogeophysical effects.

1.5.3 Adaptation of forests to climate change

Forests are vulnerable to climate change. Rising temperatures force species to migrate out of their climatic niches (Diamond, 2018; Parmesan et al., 1999; Urban, 2015; Warren et al., 2018). By 2100, climatic zones in Europe could shift northwards by 272 to 645 km, depending on the climate scenario (Ohlemüller et al., 2006). Strong climate change scenarios suggest that around one-third of the potential natural vegetation of Europe could shift outside of its current range by 2085 (Hickler et al., 2012). In addition, by 2080, more than half of 1350 assessed European plant species could be vulnerable or threatened in their current location (Thuiller et al., 2005).

Already today, large fractions of forests in Europe are unhealthy due to various aspects of global change, with 19% of monitored sites reporting defoliation and deteriorating conditions (Forest Europe, 2020). In recent years, European forests have also particularly suffered from droughts and so-called "hotter droughts" in which low precipitation occurs in conjunction with high temperatures (Buras et al., 2020; Hammond et al., 2022; Thom et al., 2023; van der Wiel et al., 2023). The impacts of these droughts are clearly visible in the assessed vegetation greenness of the European forest condition monitor (Fig. 1.3).

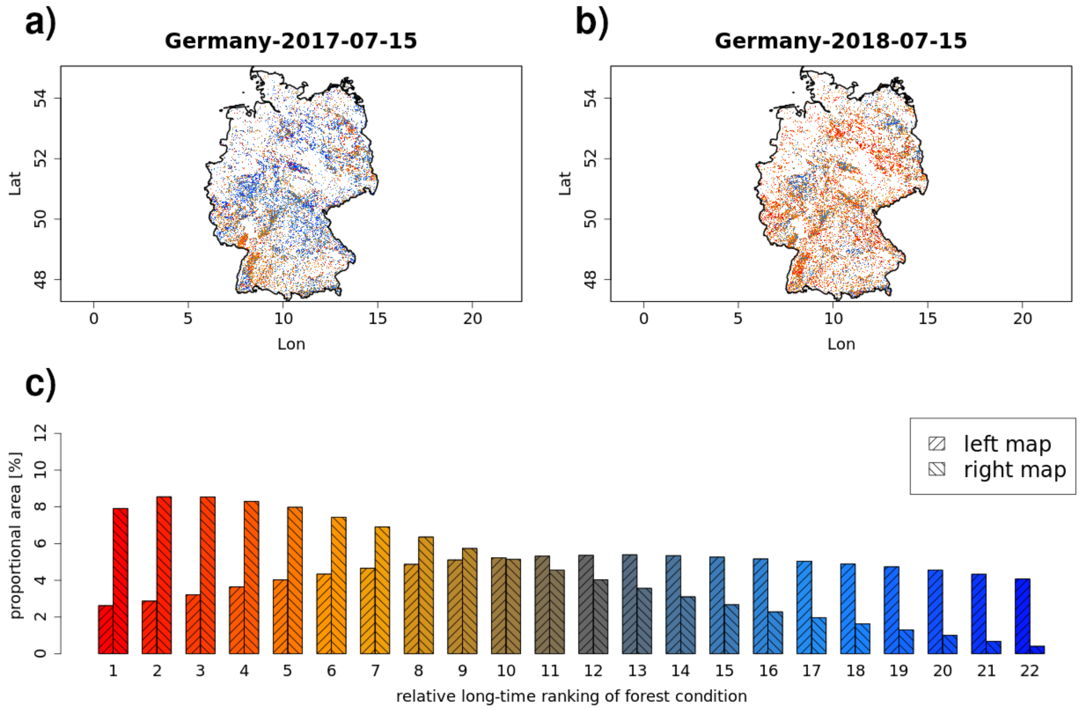

Figure 1.3: Canopy greenness measured via the Normalized Difference Vegetation Index in two consecutive years as derived by the Forest Condition Monitor (Buras et al., 2021). In summer months, this can be regarded as a proxy for forest health. Blue colors thus indicate healthy forests while red colors indicate unhealthy forests. The left map **(a)** shows a rather normal year with regard to climate. The quantile rankings of forest condition are rather uniformly distributed **(c)**. Some locations (around 2-3%) show the lowest canopy greenness of the given day-of-year over the entire observation period 2001-2023 (rank 1). Some locations on the other hand (around 4-5%) show the highest canopy greenness (rank 22) of the given day-of-year over the observation period 2001-2023. In contrast, in the drought year of 2018, a large fraction of pixels shows very low greenness and thus forest health (right map, **(b)**). Around one-third of pixels had a greenness in the lowest 4 quantiles (ranks) indicating very poor forest health **(c)**.

Unfortunately, droughts are expected to increase in frequency and severity across most parts of Europe (Spinoni et al., 2018).

Moreover, climate change serves as a catalyst for further forest disturbances, including windthrows, wildfires, and bark beetle infestations. These disturbances have exhibited increased frequencies in recent years (Patacca et al., 2023; Senf & Seidl, 2021a, 2021b). Consequently, although forests play a pivotal role in climate change mitigation, adaptation is equally important. The extent of required adaptation depends on the climate scenario. Similar to other systems, the stronger the increase in global warming, the higher the need for adaptation.

1.5.4 Agriculture and climate change

Climate change has already impacted global agriculture in multiple, primarily negative ways (IPCC, 2023a). Although the overall agricultural productivity has increased, climate change has slowed this increase over the past 50 years. This slowdown has been most pronounced in mid and low latitude regions due to negative impacts on crop yields, while some positive impacts have been observed in high latitude regions (IPCC, 2023b). These trends are also evident in Europe, where northern agricultural regions have benefited from prolonged growing seasons and increased crop yields, likely due to the CO_2 fertilization effect (Bindi & Olesen, 2011; Iglesias et al., 2012; Walker et al., 2021). Southern regions, on the other hand, have been and will be more prone to extreme weather events and drought (Bindi & Olesen, 2011; Iglesias et al., 2012; IPCC, 2023b). Hence, adaptation measures are necessary, including strategies related to water management, such as soil moisture conservation and irrigation, but also through changes in consumer behavior, including dietary choices (IPCC, 2023b).

Conversely, agricultural practices themselves contribute to climate change. In this context methane (CH_4) plays a major role. It is the second most important GHG, responsible for nearly a quarter of today's total additional radiative forcing (Saunois et al., 2020). Agricultural activities are the primary source of methane emissions, especially livestock farming (one-third of anthropogenic methane emissions) and rice cultivation, though the latter plays a smaller role in Europe (Saunois et al., 2020). In addition, the application of manure and synthetic fertilizers is the main source for nitrous oxide (N_2O) emissions, another important GHG (IPCC, 2021). The amounts of N_2O emissions are relatively small, but the global warming potential over 100 years of one molecule N_2O is equal to that of 265 molecules of CO_2 (Rivera et al., 2017).

Agriculture is also a key contributor in climate change mitigation efforts, mainly through the cultivation of bioenergy crops. Agricultural crops and by-products recently provided 27% of the EU's domestic biomass for energy, providing 2.7% of the EU's gross final energy consumption (Scarlat et al., 2019). Bioenergy with carbon capture and storage (BECCS) is considered a major strategy in climate change mitigation. Namely, all SSPs with strong mitigation efforts include a substantial increase in bioenergy production (Fig 1.4).

Despite its potential, bioenergy has faced criticism, including concerns about its competition with land use for food production and its direct environmental impacts such as the displacement of other ecosystems (Fajardy et al., 2019; Popp et al., 2014). In addition, overly relying on BECCS could delay actual decarbonization measures (Asayama, 2021; Xu et al., 2022). Even worse, BECCS depends on technologies that are far from been deployed at scale (Anderson & Peters, 2016; Fajardy et al., 2019; Jagu Schippers et al., 2022). If these technologies are ultimately unable to be deployed at large scales, this will invalidate a critical component of humanity's strategy towards climate change mitigation.

In sum, agriculture can both positively and negatively affect the climate. Regional mitigation strategies will need to take not only forests, but also agriculture into account. Here, research is necessary to understand the impact of different land use strategies

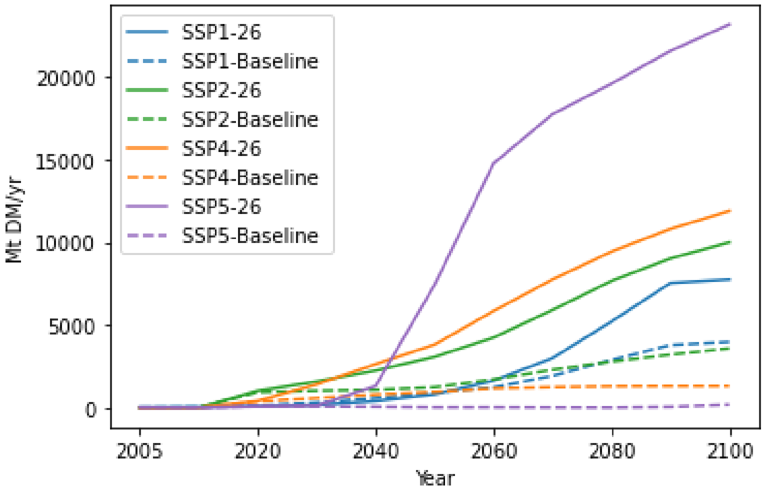

Figure 1.4: Bioenergy production in million tons dry mass per year, for the baseline marker scenarios of the SSPs, and the RCP2.6 mitigation scenario of the marker. Mitigation efforts highly depend on bioenergy, thus the amount of bioenergy crops produced are always higher in the RCP2.6-scenarios compared to the baseline scenarios without additional mitigation. SSP3 was omitted since there is no scenario in which SSP3 mitigation efforts reach RCP2.6. Data from the SSP Database (Gidden et al., 2019; Riahi et al., 2017; Rogelj et al., 2018).

on mitigation and other ecosystem services. In central Europe, for example, Krause et al. (2020) investigated the carbon mitigation potential of various land use strategies, including forest adaptation, reforestation, and BECCS. Changes in wood usage patterns and nitrogen fertilization of crops were however not yet considered. In this thesis, I will analyze the GHG impacts of reforestation, bioenergy, BECCS, increased material usage of wood, and nitrogen fertilization at realistic scales in Bavaria to achieve a reasonable estimate of land-based mitigation potentials in this region.

1.6 Deriving multi-functional land-use strategies under global change

It is evident that terrestrial ecosystems, especially forests, play a critical role in climate change mitigation. However, as discussed above, the contribution of various factors to the mitigation potential are not yet fully clear. It is thus of paramount importance to quantify the impacts of these factors on mitigation, in order to understand how to best include forests as one key element to climate change mitigation efforts. I focused on this task in paper 1 (section 3.1). Also the mitigation impact of other part of the land sector need to be quantified. I address this topic in an additional assessment (section 3.4).

But mitigation is just one facet of the broader picture. Also wood and non-wood products, local climate regulation, water provisioning, habitat preservation, and other ecosystem services need to be considered. Strategy development thus needs to take all these demands into account, which partly are inherently conflicting. Furthermore, these strategies need to be robust to the wide range of uncertainty stemming from future climate change. These strategies and implications are addressed in papers 2 and 3 (sections 3.2 and 3.3).

1.6.1 Modeling vegetation dynamics under future global change

Developing land-use strategies for the future requires understanding the impact of climate change and other factors on vegetation dynamics and vice versa. For this task, dynamic global vegetation models (DGVMs) can be employed. These are process-based models simulating plant geography, physiology, and biochemistry, as well as vegetation dynamics and biophysics (Prentice et al., 2007). They can simulate vegetation growth under future climate scenarios, serving as essential tools for quantifying the effects of climate change on global biogeochemical cycles, vegetation dynamics, composition, and structure.

DGVMs have been successfully applied on global and regional scales, for instance to assess the terrestrial sources and sinks of carbon (Friedlingstein et al., 2023) and methane (Saunois et al., 2020). In Europe, these models have elucidated the reasons for the severity of the 2018 drought (Bastos et al., 2020) and the effects of shifting climate conditions on potential natural vegetation (Hickler et al., 2012). Additionally, DGVMs have been instrumental in emphasizing the influence of CO_2 concentrations and forest age on the European carbon sink and understanding observed responses of forest productivity to climate variations (Bellassen et al., 2011; Levy et al., 2004; Zhang et al., 2018). DGVMs have also been employed to estimate the carbon mitigation potentials associated of various land use strategies (e.g., Krause et al., 2020).

1.6.2 Developing forest management strategies under global change: climate-smart forestry

Climate change requires a new paradigm of forest management to deal with the upcoming changes in temperature, precipitation, extreme events, and others (Jandl et al., 2019). In addition, the key role of forests in climate change mitigation and the provision of ecosystem services need to be included in forestry planning. This combination of mitigation, adaptation, forest resilience, timber provision and ecosystem services has been termed “climate-smart forestry” (e.g., Nabuurs et al., 2017; Verkerk et al., 2020). Originally, climate-smart forestry was conceived as a holistic framework that includes all climate-related aspects in forest management objectives. As such, the approach defined three main objectives: i) reducing GHG emissions, ii) adapting forests to climate change, and iii) promoting sustainable increases in productivity and incomes (Nabuurs et al., 2017).

In recent years, this definition has evolved, and in particular, objective (iii) has been improved to consider “all benefits that forests can provide” (Nabuurs et al., 2018) and “ensuring the sustainable provision of ESs” (Mathys et al., 2021). Still, the concept of climate-smart forestry remains a work in progress, requiring further refinements and additions particularly in bridging the existing gaps between the theoretical framework and its practical application (Cooper & MacFarlane, 2023; Shephard et al., 2022).

Climate-smart forestry can encompass numerous measures, including improved management, introduction of more resilient species, afforestation, and increasing the share of wood used for long-lived products (e.g., Nabuurs et al., 2018). Although climate-smart forestry has been explored and implemented in various regions, it has been most extensively studied in Europe. For instance, it has been estimated that climate-smart forestry can increase the current carbon impact of European forests of 569 $MtCO_2$/yr by another 441 $MtCO_2$/yr through measures like productivity enhancement, adaptation, improvement of wood usage, and reforestation (Nabuurs et al., 2017). Detailed case studies of climate-smart forestry have been conducted for various countries in Europe (Hetemäki et al., 2022; Nabuurs et al., 2018) as well as North and South America (Giongo et al., 2022; Shephard et al., 2022).

In this thesis, I define climate-smart forestry as all strategies aimed at simultaneously improving the impact of forestry on climate change mitigation (including wood products, substitution effects, and biogeophysical effects), adaptation, and the provision of ecosystem services. This definition can be easily extended to other areas, such as agriculture. Developing such strategies is a challenging task due to the multitude of involved objectives that are sometimes conflicting.

Multi-criteria decision making (MCDM) is one effective method for this task (Ishizaka & Nemery, 2013). It has frequently been applied in multi-functional land use planning and forestry. For instance, Bagdon et al. (2016) employed MCDM to propose the best forest management regime to provide eight ESs in forests in northern Arizona, including the provision of timber, biodiversity habitat, fire hazard reduction, and carbon storage. Their analysis revealed that the performance of all ESs could be improved, but the required treatments such as management measurements to address fire risk would need increased financial investments. Diaz-Balteiro et al. (2017) used MCDM in central Spain to determine the optimal management form to provide multiple ESs, including timber production, carbon storage, biodiversity, and recreation. Their study suggested that unmanaged forests were usually best fit to the task (however with obvious effects on timber provision).

1.6.3 Addressing future uncertainties

Dealing with complex ecosystems such as forests involves inherent uncertainties, particularly when developing strategies for the future. These uncertainties can originate from numerous sources, including underlying data, assumptions about timber price development, uncertainties within model processes and parametrizations, and in particular, uncertainty about future forest development, especially under climate change (Hamel & Bryant, 2017; Uhde et al., 2015). Addressing these uncertainties is paramount in developing successful strategies, but this was not yet considered in the previously mentioned studies on forestry for multiple ESs. To address this gap, Knoke et al. (2016) combined MCDM with robust optimization (Ben-Tal & Nemirovski, 2002; Gorissen et al., 2015) to account for uncertainties in various ES indicators arising from direct field measurements or stakeholder interviews. Their study of land-allocation of abandoned farmland in Ecuador illustrated that incorporating a diverse set of ESs into the analysis led to a landscape characterized by high diversity, effectively serving as a buffer against uncertainties. This underscores the value of adopting a portfolio-based approach to deal with uncertainties, aligning with the overarching concept that diversification is critical for successful risk management (Wagner & Lau, 1971). Building on this, Uhde et al. (2017) applied robust MCDM for forestry planning in Chile, highlighting that integrating native species into forest portfolios could improve ES provision. Furthermore, Knoke, Paul, et al. (2020) employed robust MCDM for landscape planning in the tropics, assessing the impact of landscape diversification on ES provision. Their study indicates that aiming for multi-functional landscapes may actually lead to increased deforestation, because of the inclusion of non-natural land uses to provide certain ESs.

However, it is important to acknowledge that the influence of future climate change on the indicators was not yet considered in these studies. This may have a significant impact on the results as the implementation of such land-use portfolios should be viable not only under current conditions but also under those anticipated for the coming decades. Recognizing the need to account for the suitability of land use portfolios in the face of future climate change, Knoke, Kindu, et al. (2020) included survival curves of tree species and the impact of climate change on survivability. Their findings suggested that the impact of climate was moderate, although it is essential to note that only one moderate climate change scenario was considered (RCP4.5).

In light of this, vegetation modeling emerges as a key instrument to assess the impacts of different climate change scenarios on land use portfolios. For example, Luyssaert et al. (2018) used the land surface model ORCHIDEE to compute optimized forest management portfolios for Europe under detailed climate change assumptions for three specific objectives. However, their study did not explicitly address multi-functionality, and the results were generated independently for two climate scenarios (RCP4.5 and RCP8.5). Although their results did not differ significantly between the two scenarios it should be noted that, in different contexts, projected ecosystem development (e.g., carbon fluxes or vegetation structure) may exhibit more substantial disparities between scenarios (e.g., Bonannella et al., 2023; Buras & Menzel, 2019; Koch et al., 2022; Krause et al., 2019; Nishina et al., 2015). This underscores the need for the simultaneous consideration of a wide range of climatic futures.

While some argue that certain scenarios, especially RCP8.5, are becoming increasingly unlikely (Hausfather & Peters, 2020), others argue that it remains crucial to include such low-probability, high-risk scenarios into future planning (Lawrence et al., 2020; Schwalm et al., 2020). Generally, using a wide spectrum of emission scenarios is recommendable (IPCC, 2021; Pedersen et al., 2020). In any case, it is essential to emphasize that strategies for the future must be formulated already today, even amidst substantial uncertainty surrounding future climate. The novelty of our approach is that it addresses this imperative by taking into account a multitude of ecosystem services and an extensive array of climate scenarios.

1.6.3.1 Addressing conflicting demands

Developing strategies for a large range of forest functions remains a highly challenging task. Although some synergies may emerge among these functions, conflicts will be inevitable. A significant conflict centers on balancing biodiversity conservation against timber production. Various studies have shed light on this conflict, underscoring its complexity (Başkent & Kašpar, 2023; Felton et al., 2016; Gutsch et al., 2018; Lessa Derci Augustynczik & Yousefpour, 2021; Verkerk et al., 2014). However, some research has also revealed potential synergies when diversity was included as a management goal (Biber et al., 2020). Furthermore, distinct regions across Europe may harbor diverse objectives and face varying limitations, often influenced by the changing climate. For instance, numerous European regions need to make water availability a key priority in forest and land management (e.g., Bredemeier, 2011).

Also the conflicts and trade-offs between new European laws for ecosystem protection (European Commission, 2020, 2021b) and a rising demand for timber (FAO, 2022a) need to be urgently reconciled. In this thesis, I extended the robust multi-criteria approach with hard constraints. This allowed to formulate restrictions on, e.g., harvest levels, or protected areas that had to be met under every climate scenario. With this extended framework, I investigated how climate-smart forestry could be achieved in Europe, whilst meeting these imposed constraints.

1.7 Goals of this thesis and research questions

To conclude, climate change poses a tremendous threat to humanity and strategies for mitigation need to be developed. While forests are anticipated to play a pivotal role in such strategies, a clear picture of the mitigation potential of forests and its drivers is still missing. At the same time, forests are vulnerable to climate change themselves, emphasizing the need to include adaptation measures. The potential loss of biodiversity and the diverse ecosystem services provided by forests are further aspects that need to be taken into account. All these considerations need to be addressed under the broad uncertainty of future climate change. Strategies must be designed to withstand this uncertainty spectrum rather than catering to specific scenarios. Furthermore, it needs to be evaluated how conflicting demands like sustained timber provision and newly mandated forest protection regulations can be reconciled. Additionally, recognizing the impact of agriculture on climate change and biodiversity, it is imperative to incorporate agriculture into future scenario analyses, exploring its impact on mitigation and ecosystem services.

In this thesis, and through three first-author papers (of which two are published/accepted) and supplemental unpublished work, I addressed the indicated arising open questions in a European context. Precisely, these open questions were:

Research question 1: What are the impacts of the main factors on the carbon mitigation potential of central European temperate forests and which recommendations for mitigation efforts can be drawn?

To answer this question, I analyzed the main factors that affect the climate change mitigation potential of central European temperate forests, namely forest age and type, climate change and nitrogen deposition, natural disturbances, management regimes, wood usage, salvage logging, and the decarbonization pace of other industries. I quantified their impacts on the mitigation potential, taking into account the forest carbon sink, the product carbon sink, and substitution effects. On the basis of this analysis, I derived recommendations for mitigation efforts, taking into account the uncertainties of the factors.

Research question 2: How can we derive strategies for climate-smart forestry in Europe despite the large uncertainty surrounding future climate? What are the implications for the European forest landscape?

In study 2, I focused on formulating climate-smart forestry strategies for Europe amidst the substantial uncertainty of future climate conditions. For this, I computed optimized forest management portfolios for Europe, taking into account climate change mitigation and adaptation, as well as various ecosystem services such as water provision, timber provision, local climate regulation, and biodiversity habitat provision. Since the extent of future climate change is highly uncertain, but decisions need to be made already today, I included a broad range of climate scenarios into the algorithm. The computed portfolios needed to be viable across the entire range of climate uncertainty. Furthermore, I analyzed the trade-offs and synergies between the provision of different ecosystem services and the impacts on forest composition.

Research question 3: How can we align climate-smart forestry strategies to reconcile with new EU strategies and demands for wood products? And what are the associated implications?

In this study, I investigated the compatibility of forest management portfolios, as derived from research question 2, with prevailing increases in wood demand and new EU strategies targeting the strict protection of ecosystems. I analyzed the regional implications of the protection goals, and how the burden of such additional constraints was shared across Europe. In particular, I assessed whether some regions needed to focus entirely on wood production to enable other regions to focus on other ecosystem services.

Research question 4: What are the greenhouse gas mitigation potentials of different climate change mitigation strategies of forests and agriculture in Bavaria?

To investigate the potential mitigation impact of land-based strategies against climate change, I conducted simulations for five of such land-based strategies on realistic scales for the German federal state of Bavaria. These strategies were namely reforestation, the cultivation of bioenergy crops with and without carbon capture and storage (CCS), the reduced fertilization of crops, and the increased material usage of wood.

2 Methods

In the studies of this thesis, I addressed carbon mitigation potentials of forests and land use, alongside the provision of ecosystem services (ESs) by forests, while taking into account the wide uncertainty of future climate change. To understand the future dynamics of forests and land use, I employed the dynamic global vegetation model (DGVM) LPJ-GUESS. This model is capable of simulating vegetation dynamics and land management scenarios under future climate change. It is forced with numerous datasets, including climate and soil data, nitrogen deposition, and land cover maps.

I enhanced the model to incorporate missing processes such as coppice management and detailed wood usage patterns. I also derived additional functions to assess the performance of various ESs based on modeled characteristics. To be able to confidently run projections into the future, numerous datasets were employed to evaluate the model's ability to replicate current conditions and historical trends.

Various model runs were conducted, applying different scenarios regarding climate change, land management, and other aspects. The simulation outputs were used to investigate the theoretical carbon mitigation potentials under various assumptions and management scenarios. Furthermore, the model was used to estimate carbon mitigation potentials of the federal state of Bavaria, Germany, and to develop strategies for climate-smart forestry in Europe. For the latter, robust optimization and multi-criteria optimization were combined to develop multi-functional forest strategies under uncertain future climate. Finally, the optimization was refined by introducing constraints to address various demands such as those related to wood provision or legislative demands on (strict) forest protection.

2.1 Using LPJ-GUESS to model vegetation dynamics under climate change

To explore the impacts of climate change and other factors on vegetation dynamics, I employed the well-established DGVM LPJ-GUESS (Sitch et al., 2003; Smith et al., 2014; Smith et al., 2001). LPJ-GUESS is part of the TRENDY model ensemble which is used by the Global Carbon Project to estimate the global land carbon sink (Friedlingstein et al., 2023).

In LPJ-GUESS, plant species are represented as plant functional types (PFTs), characterized by parameters determining their phenology, growth form, leaf physiognomy, shade and drought tolerance, bioclimatic limits, and other features. These PFTs have been parametrized globally (Smith et al., 2001) and specifically for European tree species (Hickler et al., 2012).

The model simulates physiological processes every day of the year, including photosynthesis and stomatal conductance based on BIOME3 (Haxeltine & Prentice, 1996a; Haxeltine & Prentice, 1996b), which relies on the Collatz-Farquhar photosynthesis model (Collatz et al., 1991; Farquhar et al., 1980). Further processes modeled on a daily basis include leaf phenology, plant and soil respiration, evapotranspiration (soil evaporation, interception, and transpiration), and soil processes. Soil and litter dynamics are based on the CENTURY model (Parton et al., 1993; Smith et al., 2014). Here, the decay of, and movement between, eleven carbon and nitrogen pools are simulated (including coarse and fine woody debris, surface humus, and soil organic carbon pools), based on temperature, soil moisture, and soil texture. Soil hydrology is implemented as a two-layer model, taking account rain water, snow melt, percolation, and runoff (Gerten et al., 2004).

At the end of each year, the model allocates the accumulated net primary productivity (NPP) to leaves, fine roots, and woody compartments, considering allometric constraints and PFT-specific parameters (Sitch et al., 2003). In addition, turnover of leaves and fine roots to litter and from sapwood to hardwood occurs. Population dynamics are computed on the last day of each year using one of three methodologies: individual, cohort, and population mode (Smith et al., 2001). In population mode, all individuals of a PFT are modeled as one average entity while in individual mode, every single individual is modeled. In this study, I employed the most common mode, cohort mode, in which each age cohort of each PFT is modeled by one average individual. This yields similar results to the individual mode, but is much less computationally intensive. In this mode, the plants compete for resources such as light, water, and nitrogen. Here, plant characteristics such as height, root depth or leaf area influence which individuals will be able to access the limited resources. Note that there is also competition within each cohort. Establishment of species is based on the available light at the forest floor and PFT-specific establishment rates. Mortality is simulated by multiple processes, for example if the individual's growth over the past five years is below a PFT-specific threshold, or when the climatic conditions are not suitable (anymore).

Replicate patches are used in the model to represent heterogeneity within a forest stand. The heterogeneity stems from different realizations of the stochastic processes in the model, i.e., plant mortality, establishment, and disturbances. The latter are implemented stochastically as patch-destroying disturbances, resembling large-scale windthrows or bark-beetle attacks. Throughout this thesis, I employed forest type specific disturbances rates (Pugh et al., 2019) and increasing disturbance probabilities based on Senf and Seidl (2021b). Fire, on the other hand, is modeled mechanistically and different types of fire modules exist (Rabin et al., 2017; Thonicke et al., 2001).

Input data of the model include climate data (temperature, precipitation, solar radiation, and potentially other climate data such as wind fields), global CO_2 concentrations, a soil map, and atmospheric nitrogen deposition. Furthermore, land use maps may be provided that specify the distribution of natural and managed forests, croplands, pastures, and barren land. More detailed land use maps can also be included, for example, defining forest and crop types (see section 2.2).

The spatial scales of a model run are determined by the gridded climate inputs with each grid cell being run independently. The model usually undergoes a spin-up phase of about 1000 years where the first 30 years of the climate input are recycled. This spin-up is needed to bring the soil carbon pools close to equilibrium. Climate data covers historical and future periods, typically extending until 2100.

Over the last decade, LPJ-GUESS has been successfully evaluated against numerous independent datasets, proving its capability to represent various aspects of vegetation development under different conditions. As such, it was shown that LPJ-GUESS can accurately represent environmental variables such as carbon and nitrogen fluxes (Smith et al., 2014), net biome productivity (Chang et al., 2017), gross primary productivity (Ito et al., 2017), the CO_2 fertilization effect (Haverd et al., 2020), the global land carbon sink (Friedlingstein et al., 2023), European biomass and wood harvests (Lindeskog et al., 2021), and other biogeophysical variables such as runoff and evapotranspiration in Europe (Gregor et al., 2022).

2.1.1 Land management in LPJ-GUESS

The integration of land use into LPJ-GUESS in 2013 based on LPJmL (Bondeau et al., 2007; Lindeskog et al., 2013) introduced croplands and pastures into the model. This incorporation featured eleven crop functional types such as winter and summer cereals, rice, millet, and soybeans. Pastures were populated with two competing grass PFTs (C3 and C4).

In the model, climatic factors influence sowing timing, crop development, organ allocation affecting yield and harvest index (the ratio of carbon in harvestable organs to that in the rest of the plant), senescence, and other variables and processes. Other implemented land management activities include irrigation and pasture grazing. The nitrogen cycle of LPJ-GUESS facilitated the implementation of crop fertilization (Olin et al., 2015) and nitrogen fixation in legumes (Ma et al., 2022). To account for second-generation bioenergy crops, I included a new PFT for *Miscanthus* in my simulations, adapted from Krause et al. (2020), but with a lower C:N root ratio and turnover rates (Poeplau et al., 2019).

Recently, a comprehensive representation of forest management was introduced to LPJ-GUESS (Lindeskog et al., 2021). This implementation allows for detailed specifications of silvicultural treatments, but also stand initialization through land use histories, and species and age distributions.

Wood harvest is implemented through thinnings and final harvest. Thinnings can follow an automated routine based on the rule of Reineke (1933), where thinning is executed to reduce competition and prevent self-thinning mortality. Furthermore, thinning can also be parametrized with detailed specifications of timing, strength, and targeted trees (e.g., depending on the age, type, or shade-tolerance of the trees). Final harvests can be automated based on desired stand densities, rotation times, or biomass targets. Tree plantings can be executed with defined species, timings, and seedling density.

Wood usage is implemented in a simple form, with one pool for long-lived products where every year 4% of the pool decays. The remaining harvest is either immediately

(a) (b)

Figure 2.1: Coppiced trees in Bavaria, a) in the year of a coppice event, b) a few years after a coppice event. Photo: own.

released into the atmosphere (e.g., as firewood) or left on-site as litter, following standard LPJ-GUESS processes of litter decay and soil dynamics. Parameters can be configured to determine the fractions of wood harvests allocated to the long-lived and short-lived pools or to litter. In this thesis, I implemented more detailed wood usage patterns, described in section 2.1.1.2.

2.1.1.1 Implementation of coppice forest management

To enhance the representation of forest management practices in Europe, I implemented coppice management into LPJ-GUESS. Coppice is a form of management where broad-leaved trees are cut near the stump, after which they will resprout with multiple shoots, from the stump or from roots (Fig. 2.1). Its advantage lies in rapid regrowth in the first years, most likely because resprouting trees do not need to grow a new root system and can rely on remaining food reserves in the roots (West, 2014).

Coppice management was historically very important in Europe until the 1950s (Albert & Ammer, 2012; Evans, 1984; McGrath et al., 2015). Its importance declined with the

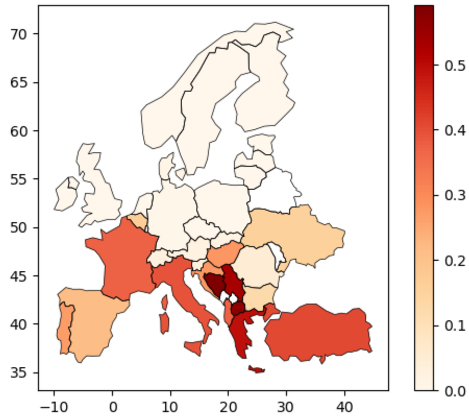

Figure 2.2: Share of coppice management in Europe in 2010 (share of total forest area). Note that this also contains short-rotation coppice. Graphic: own, Data: Maganotti et al. (2018).

substitution of fuelwood and construction wood from coppice forests by coal and metal, respectively (McGrath et al., 2015). Today, it continues to play a central role in Southern Europe (Fig. 2.2, Maganotti et al., 2018; McGrath et al., 2015) and is gaining renewed attention as short-rotation coppice for bioenergy purposes.

I implemented coppice management into LPJ-GUESS (Fig. 2.3) following a similar approach to the implementation in ORCHIDEE_CAN_r3069 (Luyssaert et al., 2018), where a fixed number of shoots (species-dependent) emerges upon cutting down a broad-leaved tree. For this, I assumed 22% of the total sapwood and heartwood to be coarse roots which stayed alive after the cutting (Lindeskog et al., 2021).

The exact number of emerging shoots after a coppice event did not play a significant role, as self-thinning would lead to the mortality of a considerable number of shoots after a few simulation years. This simulated behavior is also observed in practice (Johansson, 2008; Leonardsson & Götmark, 2015; Rydberg, 2000; Verlinden et al., 2015). In terms of management, I included anthropogenic thinning in coppices, reflecting a common practice (Nicolescu et al., 2017). Finally, shoots can be harvested based on a specified diameter, enabling the simulation of both “regular” coppice as well as short-rotation coppice plantations.

2.1.1.2 Implementation of more detailed wood usage, substitution effects and decarbonization

Although LPJ-GUESS contains a representation of wood products and their decay, its implementation is rather simple: All wood products go into the same pool, undergoing exponential decay at a rate of 4% annually. Here, I included a more detailed representation, consisting of four product pools: long-lived products such as construction wood, medium-lived products like furniture, short-lived products such as paper, and fuelwood. The

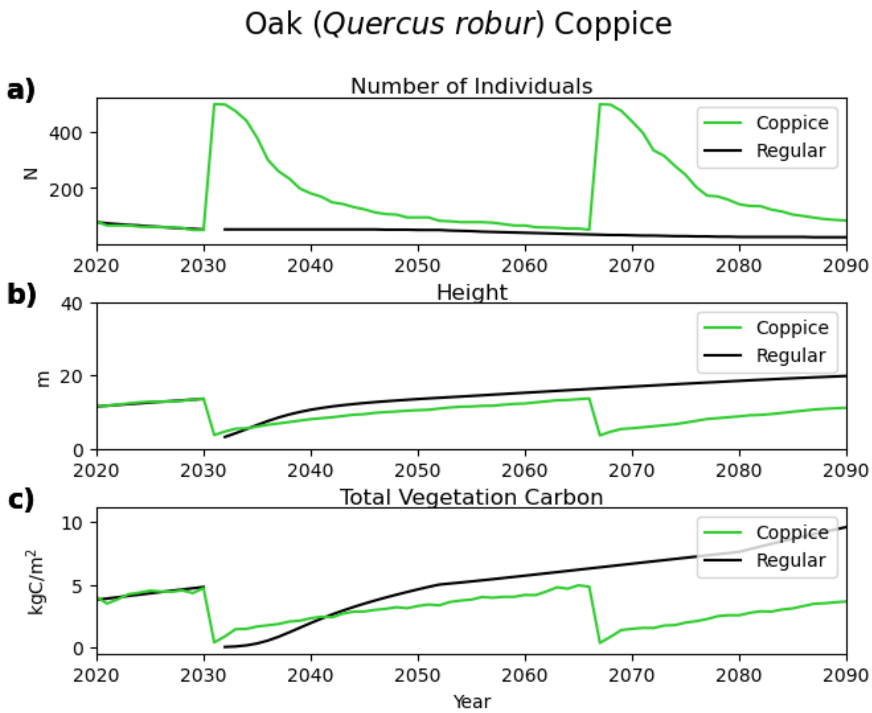

Figure 2.3: Depiction of the coppice implementation. When a specified diameter is reached, the broad-leaved trees are cut down (harvesting happened in 2030). New shoots resprout from the stump (**a**, green line), sharing the remaining roots. The black line shows what would happen if after a clearcut the same number of trees were to be planted in 2031 as existed before. Note that LPJ-GUESS only counts trees that survive the first year, which is why the black line only starts again in 2032. As in practice, a coppiced tree initially exhibits stronger growth than a planted tree (**c**) though this benefit will eventually be lost (West, 2014). Graphic from Gregor et al. (2022).

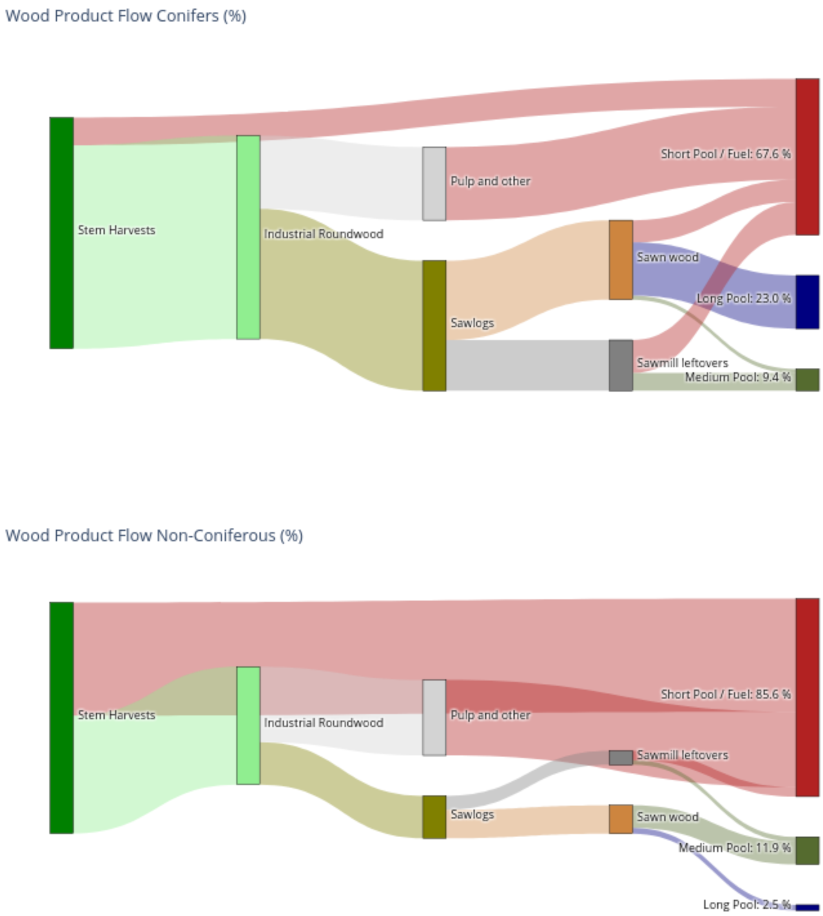

Figure 2.4: Distribution of harvests into the different product pools for the European applications in this thesis. Note that fuelwood and short-lived pool are combined in this graphic. Data from Eurostat (2021a), graphic from Gregor et al. (2022).

distribution of wood from broad-leaved and needle-leaved trees into these pools was based on data from Eurostat (2021a) for Europe (Fig. 2.4) and Klein et al. (2013) and Krause et al. (2020) for Bavaria.

Unlike the conventional LPJ-GUESS approach, I implemented the decay using Gamma functions (Fig. 2.5, Klein et al., 2013; Krause et al., 2020). This accounts for the age of products and for the fact that, e.g., when 100 wooden houses are built, those 100 houses will still exist in the initial years post-construction. This contrasts the default model where four of these houses would no longer be present one year after construction.

To quantify substitution effects, I used a displacement factor of 1.5 tC/tC for all material usages (Knauf et al., 2015). This factor implies that 1.5 tons of carbon emissions are avoided per ton of carbon in used wood products. This estimate is based on an analysis of 16 key products and their alternatives, making up 90% of the wood usage

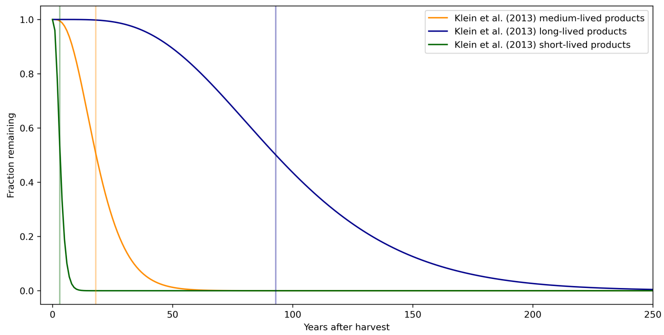

Figure 2.5: Decay rates for the wood product pools as assumed throughout this thesis for the LPJ-GUESS simulations. The vertical lines indicate the points in time where 50% of the products are still intact.

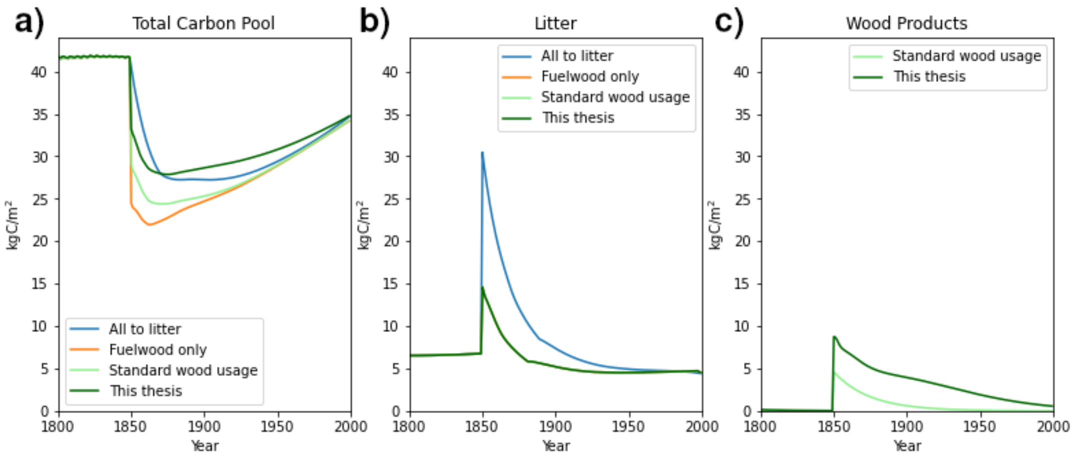

Figure 2.6: Example of different wood usage scenarios and implementations. A clearcut is executed in 1850, and the wood is either left on site ("All to litter") or used in three different ways. In the "All to litter"-scenario, the carbon mass decreases exponentially (**a**) as the biomass decays on site as litter (**b**). In the three wood usage scenarios, the carbon mass drops immediately (**a**) and the amount of litter is much lower (**b**, only twigs, leaves, etc.). The carbon mass drops most in the "Fuelwood only"-scenario when all wood is immediately burned. In the two other scenarios, the carbon mass does not drop as much, as a portion of the wood is converted to products (**c**). The "This thesis" scenario represents the situation with three wood product pools and decay rates based on Gamma functions as described in section 2.1.1.2.

spectrum (Knauf et al., 2015). This value of 1.5 tC/tC was originally estimated for Germany and is lower than the commonly used value of 2.1 (Sathre & O'Connor, 2010) which is a mean value over studies considering different applications and countries across the world and which does not incorporate an end-of-life assessment.

Depending on the modeled region, I included estimates for the amount of land-filled waste, e.g. 23% in Europe (Eurostat, 2023), and consequently lowered the displacement factor to 1.1 tC/tC to account for CO_2 and CH_4 emissions from landfills (Sathre & O'Connor, 2010). For the applications in Bavaria, landfilling was not included because wood is not allowed to be landfilled in Germany and landfilling rates are negligible (Eurostat, 2023).

For fuel wood, I used a displacement factor of 0.67 tC/tC. This value is based on a simple assessment of the possible energy provision of wood compared to light oil (Rüter, 2011). It is comparable to those of studies on other developed nations (Myllyviita et al., 2021) and well in the range of "less than 0.5 up to about 1.0" as suggested by Sathre and O'Connor (2010). I also applied this factor to those materials that were not land-filled at the end of their lifetime, assuming their use for energy generation.

Furthermore, I included discounting of the displacement factors to account for changes in the energy mix towards renewables or the development of less carbon-intensive construction materials apart from wood (Brunet-Navarro et al., 2021). For this, I decreased the displacement factors proportionally to the reduction in carbon emissions in the RCPs (paper 2 and 3, Fig. 2.7) or those from European emissions projections (paper 1 and additional results for Bavaria, Fig. 2.8). For instance, when emissions halved, the displacement factors in my studies were also halved. Note that for RCP8.5, I kept the displacement factors stable and did not increase them. The reason is that a wood product in RCP8.5 will likely still avoid the same amount of emissions, because the emissions increase will probably rather stem from increased usage and not from a more carbon-intensive production chain.

For the Bavarian applications (paper 1 and additional results), I used exponential fits to the European emissions projections to estimate effects of current policies, planned policies, and a pathway in between (Fig. 2.8).

2.2 Datasets used in this thesis

2.2.1 Forcing data to drive LPJ-GUESS

2.2.1.1 Climate change scenario data

DGVMs are forced with input data describing past and future environmental conditions, including climate data (temperature, precipitation, radiation, and others), CO_2 concentrations, and nitrogen deposition. When considering future values, climate input data for vegetation models are typically derived from simulations of Earth system models (ESMs). These ESMs model the atmosphere, land surface, vegetation, oceans, seas, and ice, based on data related to greenhouse gases, aerosols, other gases, and land use maps (e.g., Taylor et al., 2012). These models are particularly valuable to investigate the impact of future

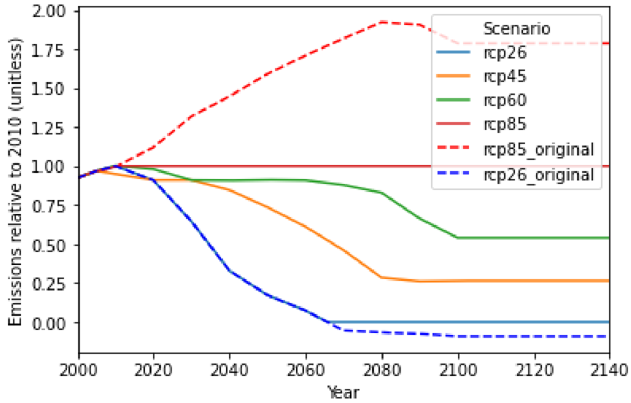

Figure 2.7: The discounting of the displacement factors based on the RCPs. In 2010, the default displacement factor was used, and in the years after that, it was multiplied by the factor representing the change in global emissions compared to 2010. In the RCP2.6 case, emissions become negative after 2060 (dashed line). Thus the displacement factor was set to 0 tC/tC (solid line). For RCP8.5, on the other hand, emissions increase to almost twice the 2010 values (dashed line). The substitution factor for RCP8.5 was kept stable (see main text).

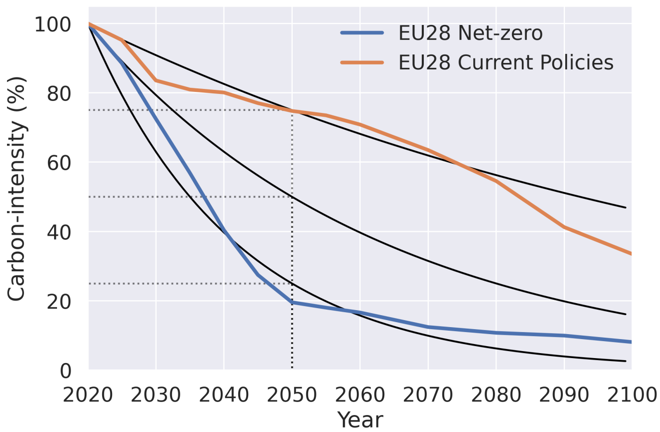

Figure 2.8: Decarbonization pathways used for the studies in this thesis that focus on Bavaria. Three pathways were considered, following an exponential decay reaching 25%, 50%, and 75% decarbonization by 2050. These pathways encompass European emission trajectories for current policies (orange) and net-zero targets (blue, Schreyer et al., 2020). Figure from Gregor et al. (2024).

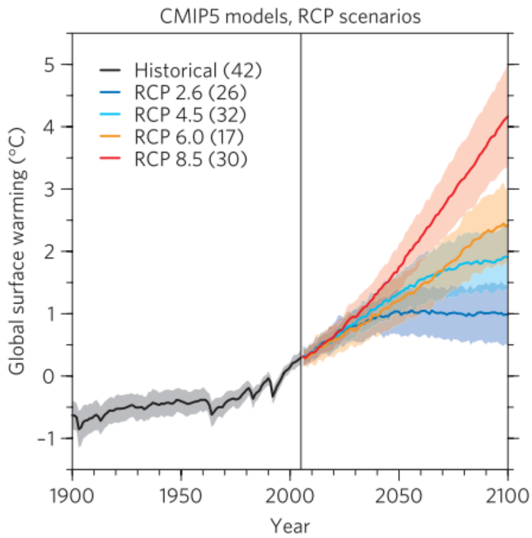

Figure 2.9: Global temperature increase for different RCPs according to the CMIP5 models (Knutti & Sedláček, 2013).

anthropogenic emissions and land use change on climate. The information derived from these models about future climate conditions is subsequently employed to analyze the detailed impacts of anticipated climate changes on terrestrial ecosystems.

Anticipated anthropogenic emissions are based on Representative Concentration Pathways (RCPs). These describe potential pathways of greenhouse gas (GHG) emissions leading to certain radiative forcings at the top of the atmosphere by 2100. For example, RCP2.6 describes a scenario that likely keeps global warming below 2 °C in 2100, compared to pre-industrial levels (Fig. 2.9). In this scenario, emissions decline from 2020 to 2100, resulting in a radiative forcing of 2.6 W/m^2 in 2100. Conversely, RCP8.5 outlines a scenario of continuously increasing GHG emissions and thus a strong temperature rise and a radiative forcing of 8.5 W/m^2 in 2100. This scenario is increasingly considered implausible (Hausfather & Peters, 2020) but remains widely used for worst-case scenarios, and to account for the fact that many climate feedbacks are not yet well understood and could have a stronger impact than thought.

For the European applications, I used climate data from the IPSL-CM5A-MR ESM from the Coupled Model Intercomparison Project CMIP5 (Dufresne et al., 2013; Taylor et al., 2012). This data was down-scaled bi-linearly from 2.5°×1.25° to 0.5°×0.5° spatial resolution. For the Bavarian applications, I used data from the Bavarian ministry of the environment (LfU). This data stems from the combined climate model EC-EARTH-RACMO (Hazeleger et al., 2012; Van Meijgaard et al., 2012). It is based on EURO-CORDEX regional climate data (Jacob et al., 2014) and was statistically downscaled to 5×5 km resolution (Bayerisches Landesamt für Umwelt [Hrsg.], 2020; ReKliEs, 2017). For all studies contained in this thesis, CO_2 concentrations and nitrogen depositions corresponding to the RCPs were taken from Meinshausen et al. (2011) and Lamarque et al. (2011), respectively.

When using climate data it is critical to apply bias correction against observations from the modeled area (Maraun, 2016). The IPSL-CM5A-MR data was bias-corrected against 1961–1990 observations from CRU-NCEP using a delta change method for temperature and relative anomalies and multiplication for precipitation and shortwave radiation (see Ahlström et al. (2012)). The LfU data was itself already bias-corrected against regional observations using a quantile delta mapping approach (Bayerisches Landesamt für Umwelt [Hrsg.], 2020).

2.2.2 Land use and land cover data for LPJ-GUESS simulations

2.2.2.1 Land cover data

The CORINE project of the European Environment Agency provides a land cover map of Europe at 100×100 m spatial resolution, containing various land cover classes (Büttner, 2014). For Bavarian applications I used the CORINE land cover map for 2018, aggregated to the LfU climate data resolution of 5×5 km, and mapped to LPJ-GUESS land cover classes as depicted in Table 2.1.

Table 2.1: CORINE land cover classes (Büttner, 2014) and the mapping to LPJ-GUESS land cover classes as used in this thesis.

Id	CORINE name	LPJ-GUESS land cover class
111-142	Artificial surfaces	Barren
211-223	Arable land, permanent crops	Cropland
231	Pastures	Pasture
241	Crops	Cropland
242	Complex cultivation	70% Cropland, 30% Pasture
243	Agriculture + natural veg.	50% Crops, 30% Mix. Forest, 20% Pasture
244	Agroforestry	Cropland
311	Broad-leaved forest	BD forest
312	Coniferous forest	NE forest
313	Mixed forest	Mixed forest
321	Natural grasslands	Pasture
322	Moors and heath	Pasture
323	Sclerophyllous vegetation	Mixed forest
324	Transitional woodland, shrubs	80% Mixed forest, 20% Pasture
333	Sparsely vegetated	70% Barren, 30% Pasture
411	Inland marshes	60% Barren, 40% Pasture
412	Peat bogs	60% Barren, 40% Pasture
other	other	Barren

2.2.2.2 Age and species distribution data for forest initialization

To obtain a realistic representation of European forests, I used age maps and tree species occurrence maps for the initialization of the simulations.

For the age distribution, I used the global forest age dataset of Poulter et al. (2018), which provides age data on a 0.5°×0.5° grid, derived from satellite data and forest inventories. It displays the age of four PFTs: needle-leaved evergreen (NE), needle-leaved deciduous (ND), broad-leaved evergreen (BE), and broad-leaved deciduous (BD) trees, categorized into 15 ten-year age classes based on the year 2010. In the model, I simulated the fellings of the natural forests and subsequent replanting starting in 1870 to obtain the age distribution of the age map in 2010 (Lindeskog et al., 2021). Those fractions that were older than 140 years according to the map were exempt from cutting.

For the tree species distribution, I used the dataset of Brus et al. (2012), which is a European tree species occurrence map with 1 × 1 km spatial resolution. This map was generated using forest inventory data from ICP Forests (6238 plots) and national forest inventories (over 300,000 plots). The data from these sites were extrapolated using a multinomial multiple logistic regression model with various predictors including soil and climate variables. The results were aggregated to 18 main species groups (e.g., *Pinus pinaster, Pinus sylvestris*, and *Pinus spp.*) and two miscellaneous groups ("other needle-leaved", "other broad-leaved"). To make the map of Brus et al. (2012) usable with LPJ-GUESS, Lindeskog et al. (2021) mapped these species onto the LPJ-GUESS European PFTs (Hickler et al., 2012) and aggregated the map to the most common species per 0.5°×0.5° grid cell (Lindeskog et al., 2021). In my simulations, I used this resulting dataset.

The overall forest area distribution in Europe was also provided by the maps of Poulter et al. (2018) and Brus et al. (2012).

2.2.2.3 Soil data

Finally, I used soil data directly from LPJ-GUESS, which provides a soil data map at 0.5°×0.5° resolution. This map specifies sand, silt, and clay fractions, as well as details on thermal and hydrological properties. This dataset was originally used in the development of the BIOME3 model (Haxeltine & Prentice, 1996b), and stems from a digitalization of the soils map of the Food and Agriculture Organization (FAO, 1974; Zobler, 1986).

2.2.3 Datasets used for model evaluation

LPJ-GUESS has been extensively evaluated using observational data (see section 2.1). However, depending on the research question and the spatial and temporal scales of the study and the modeled region, other datasets may need to be consulted. In this thesis, I used various data sources to evaluate the model's performance. Namely, I assessed runoff using the UNH-GRDC Composite Runoff Fields V1.0 which provides runoff data from a combination of observed river discharge and a water balance model (Fekete et al., 1999). To evaluate evapotranspiration I used GLEAM, which computes evapotranspiration and its components based on computations of the potential evapotranspiration of different land cover types, derived from MODIS data (Martens et al., 2017). Regarding European forest cover, I employed tree cover data from Hansen et al. (2013), based on Landsat satellite data at 30 m resolution. I also used numerous data sources to evaluate carbon and biomass stocks. These encompassed information from official country reporting (Forest Europe, 2015), data derived from country-level inventory plots (Pan et al., 2011), measurements of vegetation optical depth from microwave satellite sensors (Liu et al., 2015), data from satellite and synthetic aperture radar measurements (Santoro et al., 2021), data from wood industry companies (FAO, 2022a), and data based on inventories and detailed regional estimations of wood usage (Klein & Schulz, 2012).

2.3 Estimation of ecosystem service performance from LPJ-GUESS model runs

This section focuses on the estimation of ES performance using LPJ-GUESS model runs. Forests and ecosystems must not be limited to their ability to mitigate climate change. Therefore, in this thesis, I also included the focus on the provision of other ESs. To quantify changes in ES provision and develop strategies for the future, I employed indicator functions for each considered ES.

First of all, the carbon mitigation potential was estimated through three variables: the forest sink, the product sink, and substitution effects. The term sink here refers to the change in carbon stocks, with positive values indicating carbon uptake (i.e., a positive carbon sink) and negative values indicating carbon losses, where the forest acts as a carbon source to the atmosphere (i.e., a negative carbon sink). For the forest sink, the total sink consisting of above- and belowground biomass, deadwood, and soil carbon was considered. The product sink was calculated as described in section 2.1.1.2, comprising of the total carbon in the long-lived, medium-lived, and short-lived pools. Fuelwood was assumed to be burned within the year it was harvested, ignoring potential short-term storage effects for, e.g., drying. Substitution effects were assessed using displacement factors from the literature (section 2.1.1.2). Additionally, I included a decrease of substitution effects in the future due to, e.g., a greener energy mix or advancements in technologies for currently carbon-intensive materials (Brunet-Navarro et al., 2021).

Local climate regulation was assessed through changes in albedo, evapotranspiration, and surface roughness. Evapotranspiration, a direct model output from LPJ-GUESS, comprised soil evaporation, vegetation transpiration, and evaporation of rainfall interception. I computed surface roughness based on Raupach (1994) using canopy height and plant area index (PAI). For this, the canopy height of a stand was computed using the trees' foliar projective cover and height similar to Forrest et al. (2020). I computed PAI by combining leaf area index (LAI) with the estimated woody hemi-surface area of each tree. This area was computed based on the representation of stems in LPJ-GUESS as cylinders, and the fact that branches typically contribute to at least 50% of PAI (Kucharik et al., 1998).

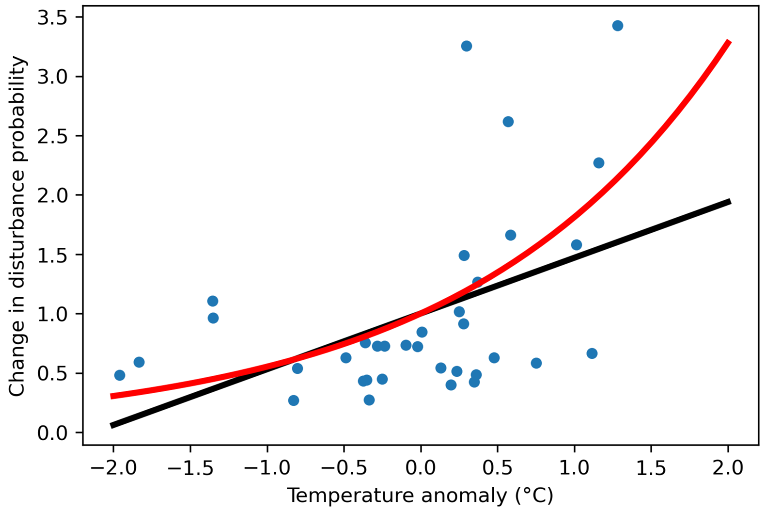

Figure 2.10: Yearly disturbance rate in Germany related to temperature anomaly compared to the 2000–2014 average temperature with a linear and an exponential fit as used in this work. The y-values represent a unitless factor, e.g., when temperatures are one degree warmer, disturbances become 1.47 times as likely as today in the linear case (1.8 times in the exponential case). Data from Senf and Seidl (2021a).

Albedo was determined by the share of species types (broad-leaved, needle-leaved, grass, or bare ground) and summer and winter albedo values depending on snow cover conditions (Boisier et al., 2013). I used the equation $\text{snowcover} = \frac{\text{snowdepth}}{0.01+\text{snowdepth}}$ to estimate the snow cover from the snow depth output of LPJ-GUESS (Wang & Zeng, 2010).

For timber provision I used two indicators: firstly, the total timber harvested, and secondly, the timber used for long-lived products, to account for different types of timber products. Species-specific conversion factors were used to estimate the m^3 wood from the given kgC from the model (Harja et al., 2019; Jenkins, 2004; Savill, 2019; Stimm et al., 2014).

As an estimate for water availability I used the lowest monthly soil water potential of each year based on the implementation of Hickler et al. (2006).

Finally, to assess biodiversity, I applied so-called linker functions to gauge the suitability of a forest to provide habitat for various species (Cordonnier et al., 2014). The evaluation included dead wood quantity, tree size diversity (Shannon index of tree sizes using 5 cm diameter at breast height (DBH) classes), and the amount of large trees (larger than 50 cm DBH).

2.4 Factorial simulation experiments

DGVMs provide a convenient way to assess the impact of climate and other factors on vegetation growth and related variables. Understanding the impact of individual factors on ecological variables is often crucial, and factorial simulation experiments offer a valuable approach for this (Kleijnen, 2005). In this context, a factorial simulation experiment involves running a baseline simulation with default settings and subsequently repeating it with one input variable (factor) altered.

Table 2.2: The considered values of the factors used in this study. All possible combinations were simulated, leading to $3 \times 3 \times 2 \times 2 \times 4 \times 2 \times 2 \times 2 \times 3 = 3456$ simulations.
($*$) Note that we used the exponential increase as the default in our analyses unless stated otherwise.

Factor	Values	Comment
Climate change and N dep.	RCPs 2.6, 4.5, 8.5	
Disturbance prob. change ($*$)	Constant, linear, exponential	Changes in disturbance frequency based on temperature anomaly (Fig. 2.10)
Forest age	Mature, young	Planted between 1921 and 1940, or between 1981 and 2000, respectively
Forest type	BD, NE	Broad-leaved deciduous, or needle-leaved evergreen forests
Harvest intensity	0%, 50%, 100%, 150%	Direct change in harvest intensity starting after 2020 compared to current values
Salvage logging	Yes, no	After every disturbance after 2020
Material wood usage	100%, 150%	The increase to 150% was implemented as a linear change from 2020 until 2050 at the expense of short-lived products and firewood
Cascade usage	100%, 150%	The change to 150% was implemented as a direct change of the lifetime of products created after 2020
Decarbonization in 2050	25%, 50%, 75%	Exponential decrease based on Schreyer et al. (2020), reaching the given percentage value in 2050 (Fig. 2.8)

2.4.1 Factorial simulation experiment to disentangle the impact of various factors on the carbon mitigation potential of forests

In this thesis, I used a factorial simulation approach to disentangle the impact of various factors on the carbon mitigation potential (defined above) of temperate central-European forests. I ran 3456 simulations for five sites in Bavaria, each time systematically varying one of the key factors affecting the carbon mitigation potential: climate scenario, forest type, forest age, harvest regime, wood usage patterns, the magnitude of substitution effects, the execution of salvage logging after a disturbance, and the decarbonization pace of other industries (Table 2.2).

In the table, 100% always refers to model parameters based on currently observed values. Harvest intensities reflecting present-day conditions were adopted from Suvanto et al. (in review) and wood usages were as described in section 2.1.1.2. I furthermore included species-specific disturbance probabilities (Pugh et al., 2019) and temperature-dependent changes in those disturbance probabilities, based on the data of Senf and Seidl (2021a), see Fig. 2.10. For this, I fitted both an exponential ($f(x) = \exp(0.59374x)$) and a linear function ($g(x) = 1 + 0.46951x$) to the response of the change in disturbance probability related to the temperature anomaly compared to the mean temperature of 2001–2014. Moreover, I repeated the analysis with constant disturbance rates.

2.5 Multi-criteria decision making to optimize multiple objectives

Finding forest management strategies that address multiple ecosystem services at the same time poses a challenging task. Such a decision problem, which involves optimizing multiple objectives simultaneously, is highly complex. Multi-criteria decision making (MCDM) is the general concept of tackling such decision problems with multiple optimization objectives and offers numerous methods and theories. Here, I employed linear programming and particularly, as I will describe in more detail below, I used a variant of a goal programming approach, where one minimizes the deviation of each criterion from a best attainable outcome.

2.5.1 A brief introduction to linear programming

Linear programming is an optimization method for problems with linear objectives and constraints, e.g.

$$\begin{aligned} &\min_x 5x_1 + 6x_2 \\ &\text{subject to } \; x_1 \leq 2 \\ &\qquad\qquad\quad x_2 \geq 2 \\ &\qquad\quad x_1 + x_2 = 5 \end{aligned} \tag{2.1}$$

Such an optimization problem spans a polyhedral solution space and can be solved using the Simplex algorithm (Dantzig, 1951) or other methods. In this thesis, I used the revised simplex algorithm (Bartels, 1971) from the Python package scipy (Virtanen et al., 2020). The optimal solution of the example program 2.1 is

$$(x_1, x_2) = (2, 3)$$

In general, linear programs are depicted in their *standard form*, i.e.,

$$\begin{aligned}
\max_x\ & c^T x \\
s.t.\ \ & Ax = b \\
& x \geq 0
\end{aligned}$$

To achieve this from an arbitrary linear program, equivalence transformations and slack variables are used. The above program 2.1 in standard form would thus be written as:

$$\begin{aligned}
\max_x\ \ & -5x_1 - 6x_2 \\
s.t.\ \ & x_1 + s_1 = 2 \\
& -x_2 + s_2 = -2 \\
& x_1 + x_2 = 5 \\
& x_1, x_2, s_1, s_2 \geq 0
\end{aligned}$$

The optimal solution is equivalent to that of the original program (2.1):

$$(x_1, x_2, s_1, s_2) = (2, 3, 0, 1)$$

The slack variable s_2 is 1, since x_2 did not fulfill its inequality constraint $x_2 \geq 2$ with equality, but with a difference of 1. The slack variable s_1 is 0, because x_1 fulfilled its inequality constraint with equality: $x_1 = 2$.

2.5.2 Using linear programming to create forest management portfolios for ecosystem service provision

To derive a forest management portfolio for the provision of multiple ESs, we can optimize an objective function using a portfolio-variable ω. This portfolio variable ω allocates the given area into distinct, non-negative portions of management options represented by $\omega_s \geq 0$ for all management options s, and ensuring that $\sum_s \omega_s = 1$. The objective function is some sort of "performance" of the grid cell with regard to all ESs. It depends on the how the area is split up between the management options and should be maximized. The linear program of such an optimization for each grid cell then looks like this:

$$\begin{aligned}
\max_\omega\ \ & \sum_s \omega_s \cdot \text{performance}(s) \\
& \omega_s \geq 0\ \forall\ s \\
& \sum_s \omega_s = 1
\end{aligned} \tag{2.2}$$

This optimization yields one portfolio ω that divides the grid cell into different management options, such that the total performance is maximized. The total performance is the proportionate sum of the performances of the individual management options.

The question remains how to measure the performance of each management option s. Here, I used the performance of different ESs, based on indicator functions as described in section 2.3. For each management option s, I computed the values for an ecosystem service indicator (ESI) and normalized them to the range $[0, 1]$. Thus, the best attainable value of all management options was set to 1 and the worst attainable value was set to 0, with the others lying in between. I denote this normalized value ("quality") for an ESI for a management option s as $q(esi, s)$.

The ideal situation would be to find a management option that provides all ESIs in the best possible way. But this is of course rarely the case. Thus, we need to compromise. A straightforward approach involves computing the average performance of a management option s in the objective function of the program 2.2:

$$\text{performance}(s) = \frac{1}{|\text{ESI}|} \sum_{esi} q(esi, s) \tag{2.3}$$

Here, |ESI| denotes the total number of considered ESIs and the objective function of program 2.2 becomes:

$$\max_{\omega} \frac{1}{|\text{ESI}|} \sum_{esi} \sum_{s} \omega_s \cdot q(esi, s) \tag{2.4}$$

This optimization would yield the best possible average provision of ESs. However, this approach has critical limitations, as illustrated by the following example (Diaz-Balteiro et al., 2018). Consider a scenario with two ESs and three management options with the following performances in the range $[0, 1]$:

$$(1, 0), (0, 1), (0.45, 0.45)$$

The first two management options both have average ESI performances of 0.5 while the third management option only has an average ESI performance of 0.45. Therefore, the first two options are the better choices when the average performance is maximized. However, from an ecological point of view, the first two management options are not ideal because each of them completely lacks the provision of one of the two ESs. The third option, providing both ESs, would likely be the better choice.

To achieve this, one can instead choose the management option that has the best worst-case ES performance. This would lead to the selection of management option 3, as its worst-case ES performance is 0.45 while both other options have a worst-case performance of zero.

To include this worst-case optimization into the linear program 2.2, we can adapt the objective function by using:

$$\text{performance}(s) = \min_{esi} q(esi, s) \tag{2.5}$$

This leads to a so-called *maximin* problem, because the objective function then reads

$$\max_{\omega} \min_{esi} \sum_{s} \omega_s \cdot q(esi, s) \tag{2.6}$$

Optimizing the worst case is a quite pessimistic approach, as it allows for no compensation of one ES by another. To introduce a level of compensation, a trade-off parameter $\lambda \in [0, 1]$ has been proposed, creating an objective function that combines the optimization of the average and the worst case (Diaz-Balteiro et al., 2018). The objective function is formulated as follows:

$$\max_{\omega} \left((1 - \lambda) \min_{esi} \sum_{s} \omega_s q(esi, s) + \lambda \frac{1}{|\mathrm{ESI}|} \sum_{esi} \sum_{s} \omega_s q(esi, s) \right) \tag{2.7}$$

Choosing $\lambda = 0$ means falling back entirely to optimizing the worst-case, while choosing $\lambda = 1$ means returning to optimizing the average.

Finally, it is worth considering whether all ESs are equally important. In drought-prone regions, for instance, the provision of water cycling could be considered more crucial than timber provision. To account for this, weights can be added to the ESIs. These could be derived from, e.g., stakeholder interviews, or other measurements. In this thesis, I only explore potential effects of changing the weights of ESIs, but do not go into detail on how to derive them. Including the weights yields the following objective function. Note that ensuring $\sum_{esi} W_{esi} = 1$ and $W_{esi} \geq 0$ makes the explicit fraction $1/|ESI|$ of eq. 2.7 obsolete.

$$\max_{\omega} \left((1 - \lambda) \min_{esi} \sum_{s} \omega_s W_{esi} q(esi, s) + \lambda \sum_{esi} \sum_{s} \omega_s W_{esi} q(esi, s) \right) \tag{2.8}$$

2.5.2.1 Converting a maximin problem to a linear program

Optimizing the average ES performance is indeed a linear program, since the objective, a weighted sum of ESI values, is a linear function. But optimizing the worst case is not a linear program since $\min_{esi} q(esi, s)$ is not a linear function. However, such a worst case (or "maximin") optimization can be converted to a linear program.

Consider the following maximin problem with the optimal solution $(x_1, x_2, x_3) = (4, 4, 4)$.

$$\begin{aligned} &\max_{x} \min(x_1, x_2, x_3) \\ &s.t. \;\; x_1 + x_2 + x_3 = 12 \end{aligned} \tag{2.9}$$

This program can be transformed into a linear program by adding an auxiliary variable z which can replace the min function with inequalities:

$$\begin{aligned}
\max z& \\
s.t. \quad x_1 &\geq z \\
x_2 &\geq z \\
x_3 &\geq z \\
x_1 + x_2 + x_3 &= 12
\end{aligned} \tag{2.10}$$

The resulting optimal solution is equivalent to that of problem 2.9: $(x_1, x_2, x_3, z) = (4, 4, 4, 4)$. Throughout the thesis, maximin problems were solved with this transformation without further description.

2.6 Robust optimization for addressing uncertainties in forest management strategies

Robust optimization is a conceptual approach to optimization problems where the available data is considered uncertain. It does not assume that probabilities of the realizations of the data are known. Instead, all data is assumed to lie within a defined uncertainty set. While robust optimization is not limited to linear programs, this thesis exclusively considers linear programs.

An uncertain linear program is defined as follows (Ben-Tal & Nemirovski, 2002):

$$\begin{aligned}
\max c^T x& \\
s.t. \quad Ax &= b \quad \forall\ (c, A, b) \in \mathbb{U} \\
x &\geq 0
\end{aligned}$$

Here, $\mathbb{U}$ represents the uncertainty space. Consequently, the goal is to optimize the objective with regard to the entire given range of uncertainty. Thus, it will provide a solution that will be viable for all realizations of uncertainty that are deemed possible. However, this usually involves a trade-off: a superior solution that is feasible for one specific realization may not be feasible for other instances of $\mathbb{U}$ and is therefore not a solution of the entire linear program.

The nature of the uncertainty can arise from many sources. In ecological applications it could stem for example from the spread of measurements or stakeholder answers (e.g., Knoke, Paul, et al., 2020). In this thesis, the uncertainty originates from the climate scenarios, where substantial uncertainty surrounds future climate conditions (Fig. 2.9). By incorporating the uncertainty of the future climate as the uncertainty set of a robust optimization, we can obtain solutions that are viable under all these considered climate scenarios. This gives confidence to implement such strategies today as they will be robust to the possible future climates.

To project the values of the ESs in the future, I used the DGVM LPJ-GUESS (section 2.1) to simulate different management options into the future, and then measured their performance on different ESs by the beginning of the next century (see also the methods summary of section 3.2).

2.7 Robust multi-criteria optimization for climate-smart forestry strategies under uncertain future climate

Finally, I combined all previous aspects into one optimization problem. Specifically, I adapted the objective function 2.8 by including the modeled outcomes of all ES performances across all RCPs. For each grid cell, the following linear program then computed one forest management portfolio that provides all ESs in the most balanced way when considering all RCPs. Notably, the normalization of ESIs was still done for each RCP separately. Therefore, a value of 1 always meant the best attainable performance under a given RCP. The final objective function reads as follows, where the trade-off parameter was consistently set to $\lambda = 0.2$ throughout this thesis, because it offered a reasonable trade-off between optimizing the average ES provision and the balanced ES provision.

$$\max_{\omega} \left((1-\lambda) \min_{esi,rcp} \sum_{s} \omega_s W_{esi} q(esi, s, rcp) + \lambda \sum_{esi,rcp} \sum_{s} \omega_s W_{esi} q(esi, s, rcp) \right) \tag{2.11}$$

It is crucial to incorporate forest adaptation in the problem formulation. In particular, only forest management options should be considered that guarantee forest cover by 2100 under all climate scenarios. This was achieved by adding a *hard constraint* (Gorissen et al., 2015). This means that the constraint needed to be fulfilled under any RCP:

$$\mathrm{fpc}(2100, s, rcp) \geq \min(0.1, \mathrm{fpc}(2010)) \quad \forall\ rcp \in \{\text{RCP2.6, RCP4.5, RCP6.0, RCP8.5}\} \tag{2.12}$$

In words, the constraint stipulates that for every RCP the foliar projective cover (FPC) in 2100 of the management option s needs to be at least the minimum of 10% or the FPC of the region in 2010. The 10% threshold is derived from the forest definition of the FAO (2020). This constraint thus guaranteed that only management options were considered that ensured forest cover in the future.

The complete optimization problem looks as follows and was independently optimized for each grid cell:

$$\begin{aligned} &\max_{\omega} \left((1-\lambda) \min_{esi,rcp} \sum_{s} \omega_s W_{esi} q(esi, s, rcp) + \lambda \sum_{esi,rcp} \sum_{s} \omega_s W_{esi} q(esi, s, rcp) \right) \\ \text{s.t.} \quad & \sum_{s \in S} \omega_s = 1 \\ & \omega_s \geq 0 \quad \forall\ s \in S \\ & \mathrm{fpc}(2100, s, rcp) \geq \min(0.1, \mathrm{fpc}(2010)) \quad \forall\ rcp \in \{\text{RCP2.6, RCP4.5, RCP6.0, RCP8.5}\} \end{aligned} \tag{2.13}$$

where $S = \{\text{base, toBd, toBe, toCoppice, toNe, unmanaged}\}$
ω_s : Share of management type s in the optimized portfolio
W_{esi} : Preference / weight for esi
$\text{fpc}(year, s, rcp)$: Foliar Projective Cover of the grid cell under management option s in RCP rcp in year $year$
$q(esi, s, rcp)$: Normalized quality of esi for management option s in rcp

2.7.1 Additional constraints for the optimization

To address legislative and societal demands on forests, I also explored additional hard constraints within the optimization problem. One of them was already mentioned above: the constraint on the FPC (eq. 2.12). Furthermore, I investigated additional constraints related to the quantities of future harvest, and the area of unmanaged forests. This approach was aimed to address demands on forests for both timber provision and forest conservation.

These constraints were applied not only on a per-gridcell basis, but also on a Europe-wide scale to assess the possibilities of achieving the targets as a joint effort. For instance, I included the constraint to increase harvest levels from present-day values, reaching 9% higher harvests by the beginning of the next century. This represents an average increase in harvest levels of 0.5 million m^3 per year. This is lower than recent trends (2 million m^3 per year), and corresponds to increasing harvests from the current three-fourths of the net annual increment to seven-eighths. On the per-gridcell basis, this required each grid cell to increase its current harvest levels by 9%. On the Europe-wide basis, it allowed for grid cells to compensate for others that did not meet the constraint. It was only important that the overall harvest levels of the continent achieved the target.

Per-grid cell constraints were simply added as an additional inequality to the original linear program. This program could still be run independently for each grid cell. For instance, ensuring that a grid cell had at least 50% unmanaged forests would be represented as:

$$\omega_{\text{unmanaged}} \geq 0.5$$

Adding Europe-wide constraints, however, required to adapt the entire optimization approach, as now grid cells were no longer independent entities. While the computation of the performance of a grid cell remained unchanged, the performances of all grid cells now needed to be combined into one objective function.

Similar to the question about the compensation of ESs (section 2.5.2), this led to the question of how the grid cells could compensate for one another. Consequently, I explored two objectives, akin to equations 2.3 and 2.5: maximizing the average of grid cell performances, or the worst case grid cell performance. The corresponding linear programs looked as follows (note that ES weights were omitted in this study).

For optimizing the average of grid cell performances ($|GC|$ is the number of grid cells):

$$\begin{aligned}
&\max_{\omega} \frac{1}{|GC|} \sum_{gc} \text{performance}(\omega^{(gc)}, gc) && (2.14)\\
\text{subject to} \quad &\sum_{s \in S} \omega_s^{(gc)} = 1 \;\; \forall \text{ grid cells } gc && (2.15)\\
&\omega_s^{(gc)} \geq 0 \quad \forall\, s \in S, \forall \text{ grid cells } gc && (2.16)\\
\text{fpc}^{(gc)}(2100, s, rcp) &\geq \min(0.1, \text{fpc}^{(gc)}(2010)) \;\; \forall\, gc \text{ and } rcp && (2.17)
\end{aligned}$$

And to optimize the worst-case grid cell performance, the objective function (eq. 2.14) is replaced with:

$$\max_{\omega} \min_{gc} \text{performance}(\omega^{(gc)}, gc)$$

In both cases, the grid cell performance is the same as from the original, grid-cell-wise optimization (eq. 2.11). As Europe-wide constraints I included a constraint on the share K of the total unmanaged forest area in Europe:

$$\sum_{gc} \text{area}(gc) \cdot \omega_{\text{unmanaged}}^{(gc)} \geq K \cdot \sum_{gc} \text{area}(gc)$$

Two other constraints were that Europe-wide harvests and harvests for long-lived wood products (hlp) needed to increase by 9% compared to present-day levels under every RCP:

$$\begin{aligned}
\sum_{gc} \sum_{s} \text{harvest}(gc, s, rcp, 2100) \cdot \omega_{\text{S}}^{(gc)} &\geq 1.09 \sum_{gc} \text{harvest}(gc, 2010)\\
&\forall\, rcp \in \{\text{RCP2.6, RCP4.5, RCP6.0, RCP8.5}\}\\
\sum_{gc} \sum_{s} \text{hlp}(gc, s, rcp, 2100) \cdot \omega_{\text{S}}^{(gc)} &\geq 1.09 \sum_{gc} \text{hlp}(gc, 2010)\\
&\forall\, rcp \in \{\text{RCP2.6, RCP4.5, RCP6.0, RCP8.5}\}
\end{aligned}$$

2.8 Generative artificial intelligence tools

For the purpose of streamlining the writing of my papers and this thesis, I employed ChatGPT3.5 (https://chat.openai.com). To be precise, after I had written the entire manuscripts by myself, I prompted ChatGPT to paraphrase selected sections with the sole outcome as to provide me with more precise wording and grammar. Furthermore, I employed the language tool DeepL (https://deepl.com) to help me with the German summary of the thesis.

3 Results

3.1 Paper 1: Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests

This chapter is a summary of the following accepted paper:

Gregor, K., Krause, A., Reyer, C. P. O., Knoke, T., Meyer, B. F., Suvanto, S., & Rammig, A. (2024). Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests. *Carbon Balance and Management*, forthcoming

Background summary

Forests are a major contributor to climate change mitigation. In particular, they offer carbon mitigation through three aspects: the in-situ forest carbon sink, the ex-situ carbon sink in wood products, and substitution effects when wood products replace carbon-intensive non-wood products. The scientific literature is actively debating the magnitude of each of these contributions, with various factors influencing a forest's carbon mitigation potential. The diverse characteristics and assumptions considered in different studies lead to varying calculated mitigation potentials and, consequently, differing recommendations for mitigation strategies.

This study aimed to address the disparities in outcomes observed in research on forest-based carbon mitigation. To achieve this, I identified key factors influencing the mitigation potential and quantified their impact through systematic changes. These factors were namely forest type, forest age, climate change and nitrogen deposition, natural disturbances, harvest intensities and wood usage patterns, salvage logging practices, the carbon-intensity of substituted products, and the considered time frame.

Finally, I assessed which of these factors play the largest roles and what conclusions and recommendations can be derived, taking into account all the uncertainties, contributing to a more comprehensive understanding of forest-based carbon mitigation.

Methods summary

For this study, the dynamic global vegetation model (DGVM) LPJ-GUESS was employed to simulate different forest characteristics and mitigation factors (section 2.1). In particular, I used a factorial simulation experiment (see section 2.4 for details) to disentangle the impact of all aforementioned factors (see Table 2.2). To be precise, 3456 model runs were conducted, with each run involving the alteration of a single factor.

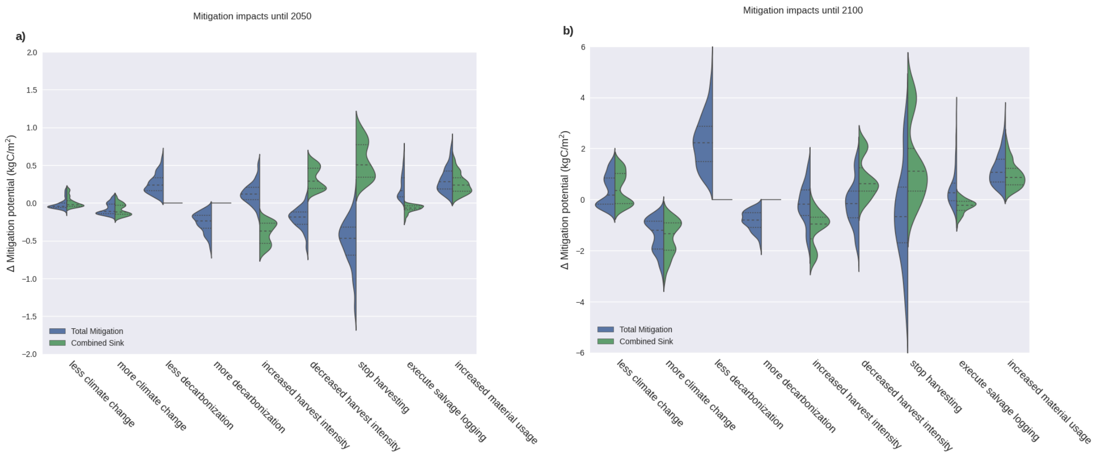

Figure 3.1: The densities depict the impact of a change in one of the given factors. Green densities refer to the changes in the combined carbon sink (forest and wood products) while blue densities additionally take into account the substitution effects. A positive value means that the change of the driver resulted in a positive effect on mitigation, i.e., a larger carbon sink, or a smaller carbon source. For example, an increased harvest intensity in panel a) has strictly negative effects on the carbon sink, but both positive and negative ones on the total mitigation potential. Panel a) shows the impacts until 2050 while panel b) shows those until 2100. For a detailed explanation of the derivation of the figure, please refer to section 2.4. Figure from Gregor et al. (2024).

I accounted for different climate change scenarios (RCP2.6, RCP4.5, and RCP8.5) and disturbance probabilities dependent on forest type and temperature (Fig. 2.10, Pugh et al., 2019; Senf & Seidl, 2021a). Both broad-leaved deciduous (BD) and needle-leaved evergreen (NE) forests were considered, and I distinguished between mature and young forests. For the strategies, I considered changes in harvest regimes (50% higher harvest intensity compared to present day values and 50% lower harvest intensities) and a complete cessation of management. Regarding wood use, I included a combined 50% increase of recycling and a 50% increase of the amount of fresh wood to be used in long-lived products as opposed to short-lived products and firewood. Furthermore, I investigated the effects of when salvage logging was executed after a disturbance or not. Finally, I assessed the impact of the pace of decarbonization of the economy on the mitigation potential. Namely, if products replaced by wood become less carbon-intensive, the substitution effect of wood will become lower as fewer emissions are offset. For this I used three decarbonization scenarios, ranging from current policies to the net-zero targets of the European Union (EU), see section 2.1.1.2.

Results summary

The main results of this study (Fig. 3.1) are summarized by the following eight key take-aways:

1) Decarbonization and climate change impacts are among the most crucial factors influencing mitigation potentials and should not be neglected in mitigation studies. They also introduce significant uncertainties, which until 2050, however, are

narrow enough to develop mitigation strategies today. Robust methods and risk diversification may help to address the larger uncertainties for strategies beyond 2050.

2) Forests are already under pressure from climate change. Any global action to address climate change will thus also diminish the threat on forests, thereby also promoting the additional local forest-based mitigation.

3) Disturbances are usually only rudimentarily represented in ecosystem models, but they critically affect the computed mitigation potentials. Improving their representation will increase the confidence in mitigation projections.

4) The type and age of forests have a substantial impact on the mitigation potentials. Consequently, mitigation strategies need to take these local conditions into account. Ongoing forest adaptation programs promoting BD species will change wood product portfolios towards more short-lived products and will consequently have a negative impact on the mitigation potential of a forest. Investing in innovative construction materials from hardwood should thus be promoted to avoid unintended negative mitigation impacts.

5) A careful quantification of displacement factors and substitution effects should be a key research priority because of the large impact of substitution effects on the total mitigation potential.

6) The mitigation impact of wood products is crucial, particularly until 2050 when substitution effects are still high. The negative effect of a reduction of harvests on mitigation would outweigh the positive effects on the forest sink. The larger growing stocks would also pose a larger threat to disturbances. While increases in harvest intensity may enhance the mitigation potential, this would come at the cost of other ecosystem services.

7) Increasing the share of long-lived products that are made from wood, and enhancing their lifetimes, has substantial positive impacts on mitigation under every climate scenario. The promotion of long-lived wood products should be included in mitigation strategies, but the economic impacts need to be taken into account.

8) Focusing on the decarbonization of the economy will heavily reduce the pressure on forests to provide mitigation and allow forest management to prioritize other ecosystem services.

3.2 Paper 2: Trade-offs for climate-smart forestry in Europe under uncertain future climate

This chapter is a summary of the following published paper:

Gregor, K., Knoke, T., Krause, A., Reyer, C. P. O., Lindeskog, M., Papastefanou, P., Smith, B., Lansø, A.-S., & Rammig, A. (2022). Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate. *Earth's Future*, *10*(9), 1–25. https://doi.org/10.1029/2022EF002796

Background summary

As demonstrated before, European forests are a major contributor to climate change mitigation. In the previous study (section 3.1), the carbon mitigation potential of forests was assessed. However, biophysical effects like albedo, evapotranspiration, and surface roughness, which play an important role, were not considered (Bonan, 2008). Furthermore, forests provide numerous other ecosystem services that need to be considered in forest planning, including timber provision, habitat for biodiversity, and water cycling. Finally, forests are vulnerable to future climate change and its consequences such as the anticipated increase in frequency and severity of disturbances and need to be adapted to future conditions. Forward-looking forest management strategies need to take all these aspects into account and are termed "climate-smart forestry". The resulting complexity is exacerbated by the large uncertainties surrounding the extent of future climate change (Fig. 2.9). In particular, a strategy for one climate scenario might not at all be useful if a different scenario materializes. Using dynamic vegetation modeling and robust multi-criteria optimization, I addressed these issues to compute forest management portfolios that provide climate-adapted, multi-functional forests across a wide range of RCPs.

Methods summary

To investigate the impact of different management options under future climate change, I conducted 24 simulations of forests in Europe until the year 2130 using the DGVM LPJ-GUESS (section 2.1). I simulated transitions to six simplified management options, each for RCP2.6, RCP4.5, RCP6.0, and RCP8.5 from the Earth system model (ESM) IPSL-CM5A-MR (Dufresne et al., 2013). The model was initialized with observed present-day (2010) forest age and species distributions (section 2.2.2.2).

Forest management was simulated with an automated thinning routine and final harvests, and my new implementation of coppice management (section 2.1.1). At the time of final harvest, the management was changed deterministically in each run according to one of the six management options (Fig. 3.2): *base* - execute final harvest and then continue the same management as before, *toBd* - execute final harvest and then plant broad-leaved deciduous species and continue the management with thinning and final harvest, *toBe* and *toNe* - same as *toBd* but plant broad-leaved evergreens and needle-

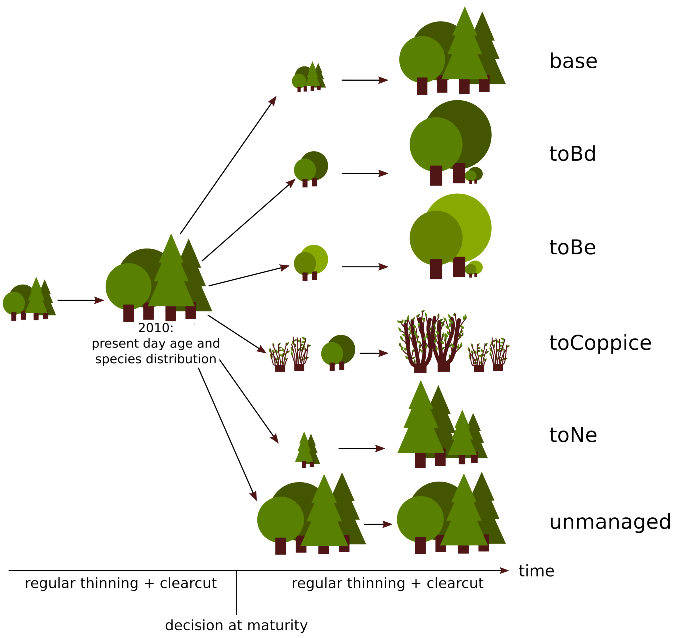

Figure 3.2: Illustration of the management scenarios, Figure from Gregor et al. (draft).

leaved evergreens, respectively, *toCoppice* - convert the forest to coppice, *unmanaged* - refrain from final harvest and leave the forest untouched from this point in time.

I furthermore implemented detailed wood usage patterns and mitigation assessments (section 2.1.1.2). To account for an expected increase in the frequency of disturbances, starting in 2010, I increased their frequency every year by 1% based on observations (Senf & Seidl, 2021a).

From the model outputs I computed indicator functions (section 2.3) to estimate the performance of ecosystem services (ESs) for each management option and RCP, encompassing climate change mitigation, timber provision, water availability, biodiversity habitat provision, and local climate regulation.

Robust multi-criteria optimization (section 2.7) was applied, resulting in one forest management portfolio per grid cell, balancing ES provision optimally under all RCPs (Fig. 3.3). Adaptation was implicitly considered through a constraint that ensured forest cover in the grid cell for each RCP.

Finally, I investigated changes in portfolios due to a priority on timber provision or climate change mitigation, imposing hard constraints on harvest levels, different assumptions on substitution effects, and the exclusion of certain climate scenarios.

Results summary

In the model, strong changes in European forest composition were necessary to make these forests climate-smart and provide the considered ecosystem services across all RCPs.

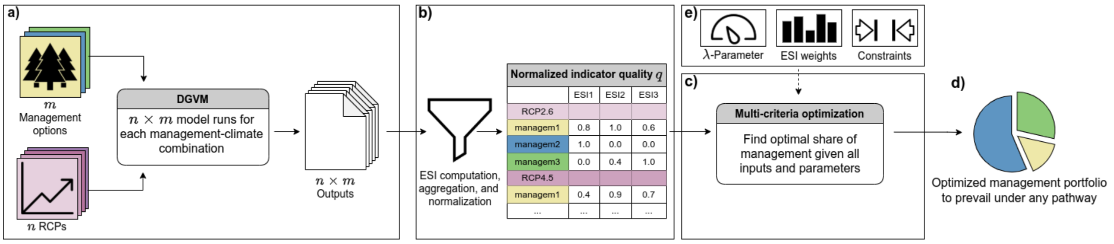

Figure 3.3: Illustration of the methodology: 24 simulations are run for six management options and four RCPs. These outputs are used in a robust optimization to obtain one forest management portfolio for each grid cell that is viable for all climate scenarios. Figure from Gregor et al. (2022).

The optimized portfolios consisted of a high share of broad-leaved forests, namely 55%, compared to the current share of 38% (Fig. 3.4), due to their lower vulnerability to future conditions such as increased aridity, their reduced susceptibility to disturbances, and their positive impact on biodiversity. Furthermore, large shares of unmanaged forests (over one-fourth of the entire forest area) were proposed by the optimization, because of their benefits in terms of carbon storage and habitat provision for biodiversity. Coppice, prevalent in southern Europe today, nearly vanished from that region, mainly due to its high water use, low harvests for long-lived products, and limited carbon sequestration.

These portfolios highlight a complex trade-off between demands. High shares of unmanaged forests will on the one hand be beneficial for many ecosystem services. But on the other hand, they will also lead to decreased wood production. This is highly problematic in times of rising demands for wood products. The proposed shift towards broad-leaved forests will exacerbate this because their wood is less commonly used for construction purposes and other long-lived products.

When excluding the discounting of substitution effects (thus assuming no change in the carbon-intensity of products like concrete or steel), the share of unmanaged forests dropped to 17%, emphasizing the potentially high impact of wood products for mitigation. Consequently, the importance of unmanaged forests diminished for mitigation, but continued to have a substantial share in the portfolios due to their other benefits, e.g., for habitat provision. A similar result was observed when doubling the importance of harvests in the optimization.

Optimizing exclusively for RCP2.6 mainly changed the share of needle-leaved forests. In that scenario, the negative climate-change impacts are not as pronounced and consequently those impacts on the less climate-adapted needle-leaved species are only of minor concern.

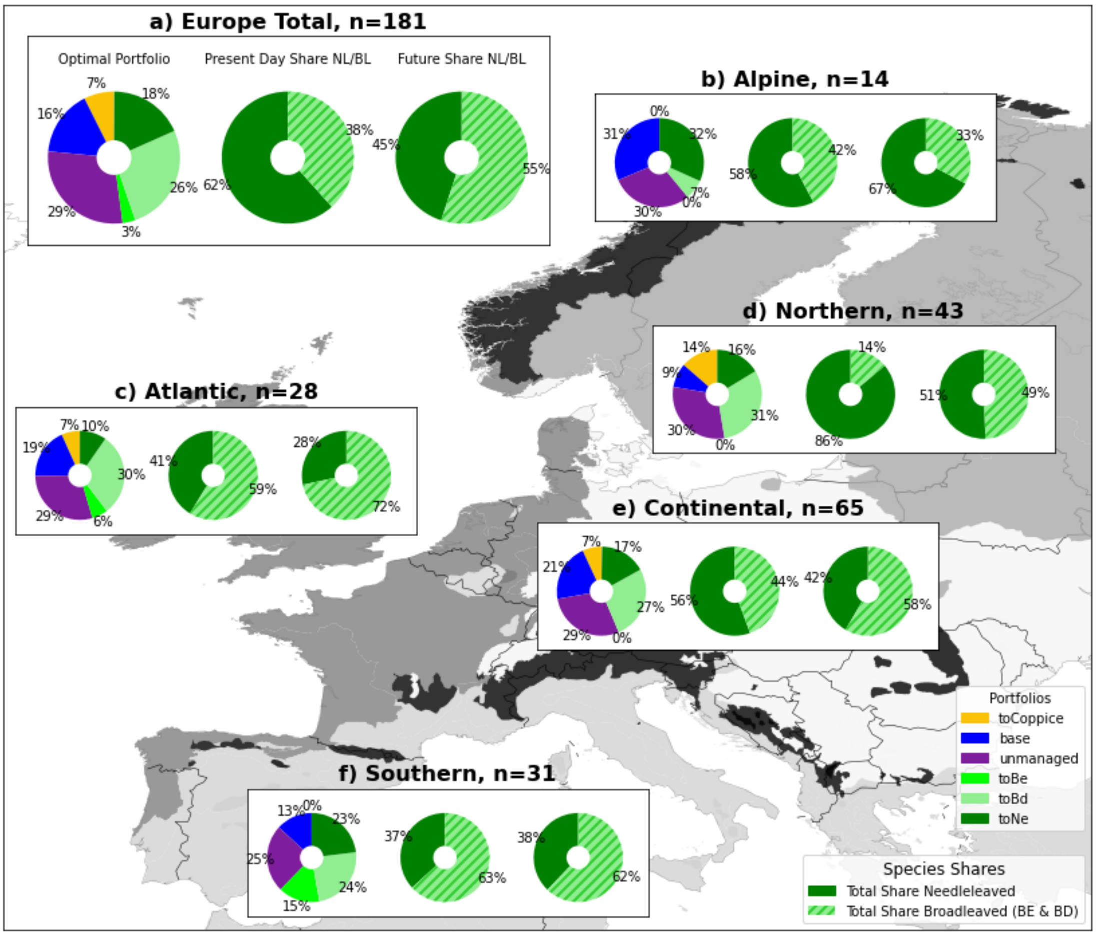

Figure 3.4: Forest management portfolios when the methodology is applied on the entire continent. Although the optimized portfolios differ in the different regions, a trend towards more broad-leaved species is visible, except for in the Alpine regions. Furthermore, large fractions of unmanaged forests are contained in the portfolios throughout the continent. Figure from Gregor et al. (2022).

3.3 Paper 3: Reconciling climate-smart forestry with EU forest, biodiversity and climate strategies

This chapter is a summary of the following paper which is ready for submission:

Gregor, K., Reyer, C. P. O., Nagel, T. A., Mäkelä, A., Krause, A., Knoke, T., & Rammig, A. (draft). Reconciling climate-smart forestry with EU forest, biodiversity and climate legislation

Background summary

The analyses of study 1 and 2 provide versatile frameworks to develop multi-faceted forest management strategies for climate change mitigation and ecosystem services under global change. Concrete strategies, however, need to also consider legal and social constraints. In Europe, for instance, the *EU Forest Strategy for 2030* and the *EU Biodiversity Strategy for 2030* (European Commission, 2020, 2021b) set targets for strict forest protection as well as for forest-based climate change mitigation. Moreover, global demand for wood products is on the rise and anticipated to further increase (FAO, 2022c; IEA, 2022). Such external constraints limit the set of feasible strategies. In this study, I explored the implications of strict forest protection goals and targets for increased timber production and addressed the central question whether and how these demands may be reconciled and what their impact on the forest landscape would be.

Methods summary

To tackle the research questions of this study, I adapted the robust multi-criteria optimization framework of paper 2 (section 2.7). In particular, I added additional constraints to the optimization that resembled the goals of the EU's strategies (section 2.7.1). These constraints were so-called *hard constraints* meaning they had to be fulfilled under each considered climate scenario (RCP2.6, RCP4.5, RCP6.0, and RCP8.5). Note that these constraints were imposed on the optimized portfolios, which were optimized for the beginning of the next century, whereas the targets of the EU strategies are formulated for 2030. This study therefore serves as a general assessment of such targets and is not a direct evaluation of the new EU strategies. The considered constraints were the following:

1. increase wood harvests by 9% from present-day levels (*min-harv*)
2. increase harvests for long-lived wood products (hlp) by 9% from present-day levels (*min-hlp*)
3. strictly protect 13.8% of the forest area available for wood supply (*strict-protection*)
4. strictly protect 13.8% of the forest area available for wood supply in each grid cell (*strict-protection-cell*)
5. apply constraints 1-4 at the same time (*all-constraints*)

The 13.8% strict protection area were derived from applying the 10% strict protection goal over the land cover types that are suitable for strict protection (see Gregor et al. (draft)). The 9% increase in harvests correspond to the intermediate value between reported present-day harvests and harvesting 100% of the net annual increment. It represents an average increase of 0.5 million m^3 per year, lower than current trends for Europe which are estimated at 2 million m^3 per year (Forest Europe, 2020).

Results summary

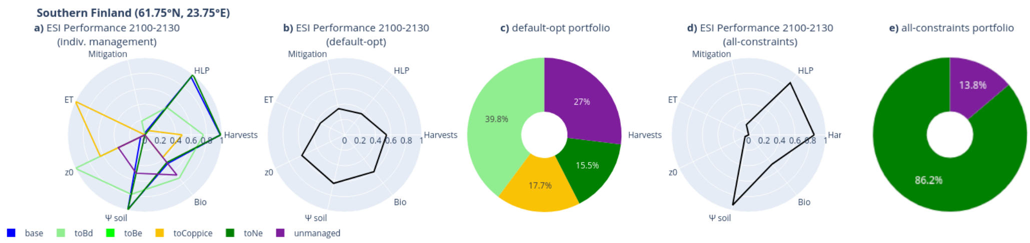

Figure 3.5: Ecosystem service performance of the different management options for a grid cell in Finland (a), the un-constrained optimized portfolio (b and c) and the portfolio when imposing all hard constraints (d and e). Note that the ecosystem service provision for the un-constrained portfolio (b) is much more balanced than that of the constrained one (d). Furthermore, the un-constrained portfolio (c) is much more diversified than the constrained one (e). Figure from Gregor et al. (draft).

According to the optimization, imposing all additional constraints on the portfolios was possible, but the amount of feasible solutions to the optimization problem was small. This means that there was not a lot of room to optimize other ecosystem services and some of them were provided at the worst possible level (Fig. 3.5d). In particular, productive regions of Europe had to focus on timber provision to compensate for the low timber provision of other regions. These productive regions thus did not have the opportunity to strictly protect large shares of their forests. In fact, when enforcing the *min-harv* or *min-hlp* constraints, the northern region of the continent contained only 6–8% strictly protected forests (Fig. 3.6d). When the strict protection constraint was instead applied within every grid cell, most of the northern grid cells contained the minimally allowed amount of strictly protected forests in their portfolios (Fig. 3.6d and Fig. 3.5e). Furthermore, when applying constraints, the forest management portfolios within grid cells were much less diversified compared to the un-constrained optimization (compare Fig. 3.5 c and e). This poses an additional threat because diversification is a key strategy to tackle future risks and uncertainties.

To conclude, the methodology suggests that reconciling all demands on European forests will be challenging if not impossible.

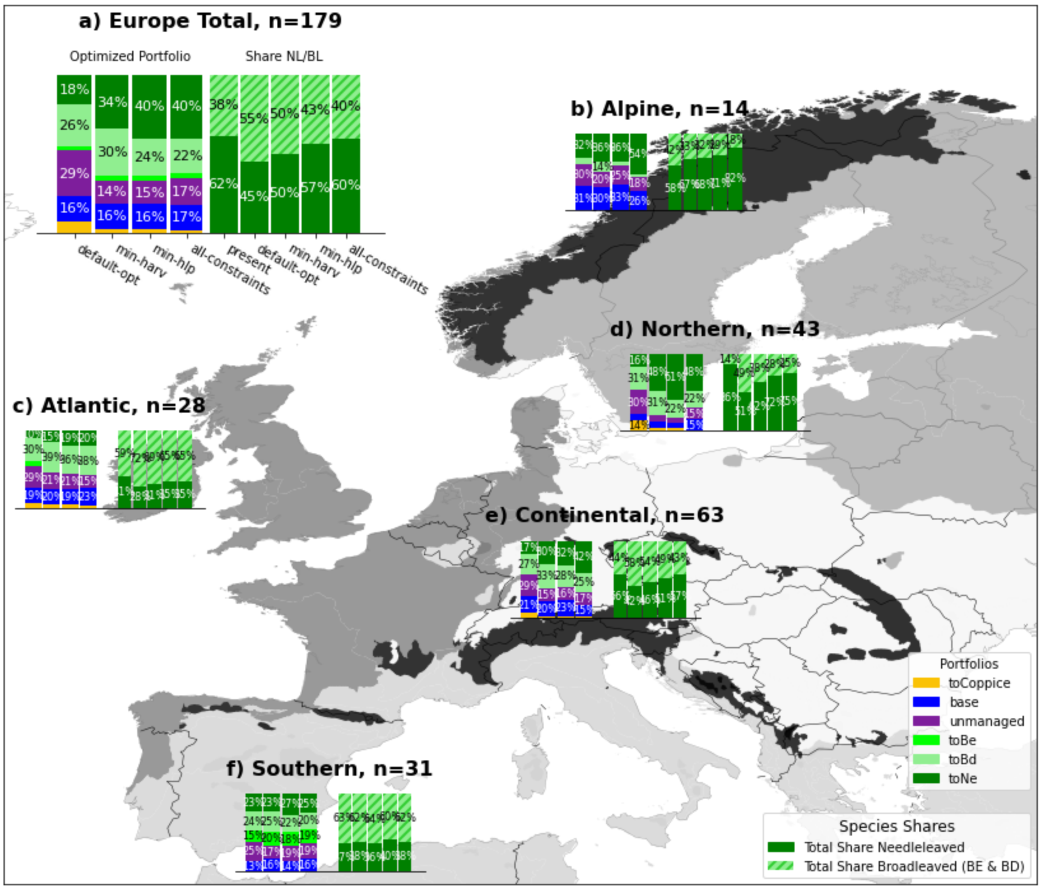

Figure 3.6: Forest management portfolios and species shares for the un-constrained optimization and the constraints *min-harv*, *min-hlp*, and *all-constraints*. Values in a) are aggregated to the whole of Europe whereas b-f) show the aggregated portfolios of the indicated regions. Figure from Gregor et al. (draft).

3.4 Additional results: Land-based climate change mitigation potentials in Bavaria

This study extends the focus from forest-based mitigation towards land-based mitigation by quantifying the impact of five climate change mitigation options in forests and agriculture in Bavaria.

Background summary

The German federal state of Bavaria emits about 95 $MtCO_{2e}$ of greenhouse gases (GHGs) each year (Statistische Ämter der Länder, 2022). Carbon dioxide makes up the lion's share of these emissions, with about 77 $MtCO_2$, or 81% of total emissions. About 16% of Bavaria's current annual GHG emissions are from nitrous oxide (N_2O) and methane (CH_4), of which 82% stem from agriculture, mainly fertilizer application and ruminants (Statistische Ämter der Länder, 2022). This adds to the importance of the land sector for climate change mitigation and offers numerous pathways to reduce emissions or improve carbon uptake. A state-wide calculation of mitigation measures for Bavaria has been previously conducted by Klein and Schulz (2012). However, their analysis was restricted to forests and did not consider changes in wood usage patterns, forest area, or carbon-intensity of substituted products which I have shown to have a crucial impact on the mitigation potentials (see section 3.1). Krause et al. (2020) included changes in forest area and bioenergy cultivation when investigating land-based mitigation potentials for Bavaria. However, changes in wood usage patterns were not considered and changes in substitution effects only for bioenergy, not for material usage. Furthermore, disturbances were either not considered in the mentioned studies, or without an anticipated increase in disturbance frequencies in the future. Here, I enhanced these assessments by including the missing components, to provide a more nuanced view on land-based mitigation potentials in Bavaria.

Methods summary

For this assessment, I used the RCP2.6 scenario of EC-EARTH-RACMO combined with present-day land cover data encompassing detailed information on forest types and crops based on CORINE data (sections 2.2.1.1 and 2.2.2.1). I investigated the impact of five land-based mitigation measures on GHG emissions, compared to a business-as-usual scenario. Wood harvests and wood usage were modeled as described in section 2.1.1.2. I used a global warming potential of 298 to convert nitrous oxide emissions to carbon dioxide equivalents, CO_{2e}. Two decarbonization scenarios were assessed (see section 2.1.1.2): 25% and 75% decarbonization by 2050, which align with current policies, and EU's target net zero policies, respectively (Fig. 2.8). The considered measures were:

Fertilization: Nitrogen fertilization was decreased linearly starting in 2021, reaching a 20% lower fertilization by 2050 compared to present day values.

Bioenergy: From 2021 to 2050, 10% of the current cropland area were successively converted to bioenergy crop plantations with *Miscanthus* plants. The impact of bioenergy cultivation was assessed both with and without carbon capture and storage (CCS).

Carbon capture and storage: In the scenario with CCS, 75% of bioenergy emissions were assumed to be captured starting directly in 2021.

Reforestation: A gradual reforestation was implemented on cropland starting in 2021, such that the new forest area reached 5% of the present cropland area by 2050.

Wood usage: Material usage of wood was increased gradually until 2050, increasing the share of wood used for long-lived products from 27% to 40.5% for needle-leaved trees, and from 6% to 20% for broad-leaved trees. Furthermore, all harvest residues were used as fuelwood starting in 2021.

Results summary

Model evaluation

LPJ-GUESS was able to accurately represent fluxes and stocks of Bavarian forests and agriculture. Forest vegetation carbon was modeled as 313 MtC for 2002, within literature estimates of 305-325 MtC (Klein & Schulz, 2012). Forest gross and net primary production were simulated as 1590 $gC/m^2/yr$ and 670 $gC/m^2/yr$, respectively, for the period of 2000-2015 which is close to estimates based on satellite observations of 1444 $gC/m^2/yr$ and 687 $gC/m^2/yr$ (Li & Xiao, 2019; Running & Zhao, 2021). The long-lived wood product pool was modeled at 63 MtC, aligning with literature estimates of 58 MtC (Klein & Schulz, 2012). Finally, *Miscanthus* yields were modeled as 6.8 t dry matter / ha which is close to the median reported value for that region of 7.05 t dry matter / ha (Li et al., 2020), but note that these observed yields go up to 29 t dry matter / ha when heavily fertilized. Our values are thus rather conservative.

Climate change mitigation impacts of the considered land-based measures

The following paragraphs describe the additional mitigation effect of the given measures compared to a business-as-usual scenario with stable land cover areas and usage patterns.

Fertilization: The decreased fertilization of crops had a cumulative effect of 75 $MtCO_{2e}$ lower nitrous oxide emissions until 2100, while carbon stocks were 3 MtC (11 $MtCO_{2e}$) smaller compared to the business-as-usual simulation, driven by changes in soil carbon, leading to a total effect of 64 $MtCO_{2e}$. Corn yields were only marginally effected by the decrease in fertilization whereas wheat yields decreased by about 10%.

Bioenergy: Successively increasing the area of planted *Miscanthus* yielded cumulative emissions of 248 $MtCO_2$ from its combustion until 2100. Regarding substitution effects, this usage of bioenergy simultaneously avoided emissions from fossil fuels of 29-112 $MtCO_2$ until 2100, depending on the decarbonization scenario (Fig. 3.7). Furthermore, a small increase of 4 MtC in the carbon stocks of cropland occurred when replacing the currently grown maize and wheat with *Miscanthus*, representing a carbon sink of 14.67 $MtCO_2$, adding to a total mitigation potential of bioenergy of 44 to 127 $MtCO_2$.

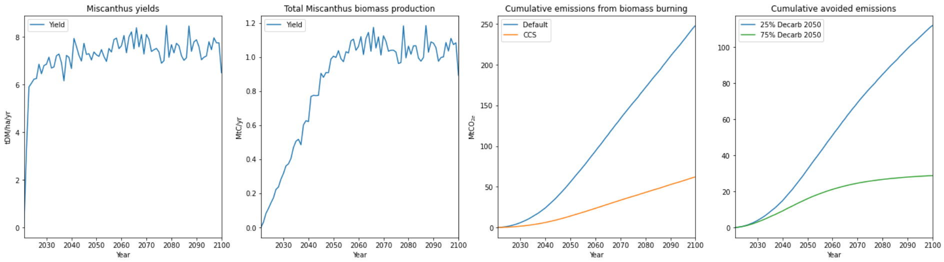

Figure 3.7: Yields, production, emissions, and avoided emissions from *Miscanthus* cultivation on 10% of the cropland area in Bavaria.

Carbon capture and storage: Assuming a capture rate of 75% starting in 2021, 186 $MtCO_2$ of the emissions from *Miscanthus* combustion could be avoided by CCS.

Reforestation: Successively planting forests on 5% of the cropland area led to increased carbon stocks of 21 MtC (77 $MtCO_2$) of which 14 MtC (51.3 $MtCO_2$) were increases in vegetation carbon, the rest stemmed from increases in deadwood and soil.

Wood usage: Changes in wood usage led to differences in the product carbon sink and substitution effects. The latter depended on the decarbonization scenario. The total impact of these measures were 48 MtC (176 $MtCO_2$) in the 25% decarbonization case, dominated by substitution effects, and 25 MtC (92 $MtCO_2$) in the 75% case (Fig. 3.8).

Total impact: Assuming that all measures could be implemented simultaneously, and ignoring impacts on other ecosystem services, the maximal achievable mitigation potential from the considered measures would be 64+127+186+77+176 = 630 $MtCO_{2e}$ until 2100. This represents about six years of present-day GHG emissions (about eight years of CO_2 emissions) in Bavaria. Should decarbonization pick up in pace and meet the EU's net-zero targets, the total mitigation potential would shrink to 64+44+186+77+92 = 463 $MtCO_{2e}$, almost five years of current GHG emissions. Finally, should CCS prove infeasible, the total mitigation potential would solely be 277 $MtCO_{2e}$, or about three years of GHG emissions.

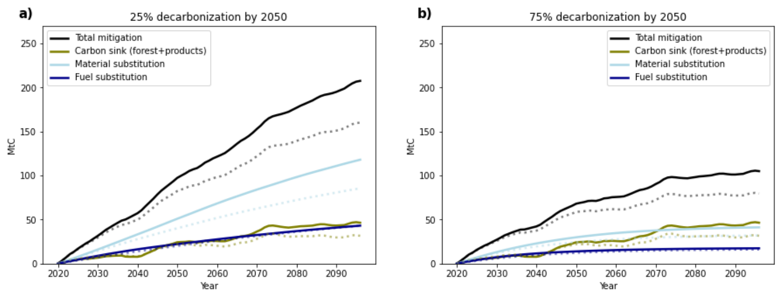

Figure 3.8: Impact on different aspects of mitigation by an increased material wood usage (continuous lines) compared to a business as usual scenario (dotted lines). Panel a) depicts the situation for the slow decarbonization scenario reaching 25% decarbonization in 2050, while panel b) shows the values for the scenario of fast decarbonization.

4 Discussion

4.1 Answers to the research questions

After having summarized the key methods and results of my research in chapter 3, I here briefly discuss the four research questions of this thesis based on these results. The detailed discussion of the results, their implications, and their contextual alignment with the scientific literature will follow in subsequent sections.

4.1.1 Research question 1: What are the impacts of the main factors on the carbon mitigation potential of central European temperate forests and which recommendations for mitigation efforts can be drawn?

First and foremost, I showed in study 1 (section 3.1) that substitution effects critically impact mitigation potentials. For instance, forest-based mitigation strategies that focused on enhancing the carbon sink by decreasing harvests often simultaneously decreased the overall mitigation potential because of reduced substitution effects.

My results further showed that climate change scenarios and associated disturbances heavily affect mitigation potentials. While warmer temperatures and prolonged growing seasons had a positive effect on simulated forest growth, this positive effect was counteracted by negative effects from increased disturbances. Which of these effect outweighed the other depended on the climate scenario, leading to high uncertainties. The same is true for the decarbonization scenarios, where a faster pace led to lower substitution effects. These uncertainties increased with time which highlights the need to link recommendations to specific time frames. However, before 2050, the simulated uncertainties stemming from climate change scenarios and decarbonization pathways were small enough to create mitigation strategies with confidence (Fig. 3.1a). For long-term strategies past 2050, robust methodologies, such as the one used in section 3.2, can help address the increasing uncertainties. The forest type (needle-leaved evergreen (NE) or broad-leaved deciduous (BD)) did not substantially influence the forest sink in my simulations, but notable differences existed in the product sinks. The forest age had an important impact, due to decelerating growth in older forests on the one hand and less suitable wood for long-lived products in younger forests on the other. Consequently, mitigation strategies must be tailored to specific sites, accounting for local conditions.

Concerning concrete measures, increasing the share of wood used for long-lived products provides substantial mitigation benefits. This strategy is robust to climate change and decarbonization scenarios and is particularly important given the ongoing conversion of needle-leaved forests to mixed forests, because broad-leaved forests provide much smaller shares of long-lived wood products. Decreased harvests or stopping management increased the forest carbon sink in my simulations. However, the influence on the total mitigation potential was mostly negative, especially until 2050 when substitution effects were still high (see Figure 3.1a). This suggests that a decrease of (sustainable) harvest intensities cannot be recommended from a mitigation point of view (ignoring potential benefits for other ecosystem service (ES)). Furthermore, it underscores the necessity to implement mitigation strategies based on wood products in a timely manner, to leverage the currently still high substitution effects.

4.1.2 Research question 2: How can we derive strategies for climate-smart forestry in Europe despite the large uncertainty surrounding future climate? What are the implications for the European forest landscape?

Combining the different aims of climate-smart forestry (mitigation, adaptation, and other ESs) is a complex task, exacerbated by the uncertainty surrounding future climate pathways. The results of study 2 (section 3.2) revealed that a management option can not only be beneficial for one ES and detrimental for another; it can even positively or negatively influence the same ES, depending on the climate scenario. For instance, unmanaged forests excel in mitigation under the low-emissions RCP2.6 scenario but perform less favorably under the high-emissions RCP8.5 scenario, where wood products and substitution effects remain critical for mitigation until the beginning of the next century (see Figure 4.1).

My research showed that robust multi-criteria decision making (MCDM) can address this issue and help diversify benefits, spread risks, and reduce the uncertainty of ES provision across climate scenarios. The proposed methodology implicitly incorporated adaptation by optimizing only those management options that ensured forest cover by 2100 (see section 2.7). This highlights that robust MCDM is a suitable tool to derive climate-smart forestry strategies under uncertainty.

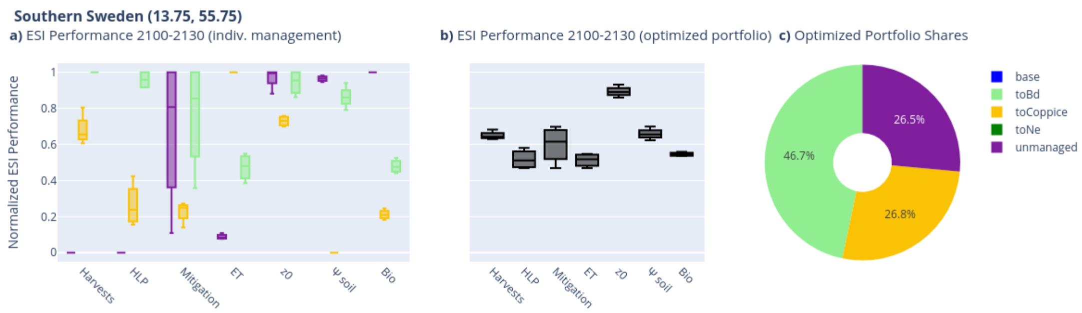

Figure 4.1: Results of robust multi-criteria optimization for a grid cell in Sweden. Panel a) illustrates the performance of different management options. The value 1 refers to the best attainable ES provision across all climate scenarios whereas 0 refers to the worst one. The y-axis spread of the box plots reflects the different performances for different climate scenarios. Panel b) presents the optimized portfolio's performance with reduced spread over climate scenarios. Panel c) depicts the composition of the optimized portfolio.

The optimization suggested a significant increase in the share of broad-leaved species (from 38% to 55%, Fig. 3.4). This shift was advantageous for adaptation and various ESs, including biodiversity habitat provision, aligning with findings from other studies (Astrup et al., 2018; Felton et al., 2010; Pugh et al., 2019; Schwaab et al., 2020; Seidl et al., 2017). Furthermore, the method proposed including 29% unmanaged forests. This had a positive impact on biodiversity habitat provision and water availability but entailed a critical trade-off with timber provision. Detailed implications are discussed below.

4.1.3 Research question 3: How can we align climate-smart forestry strategies to reconcile with new EU strategies and demands for wood products? And what are the associated implications?

According to the simulations and optimizations conducted for paper 3 (section 3.3), the conflicting demands on timber provision and forest protection could theoretically be reconciled. However, there were consequently limited opportunities to optimize the provision of other ESs. In particular, certain regions, especially in southern Scandinavia, had to focus on timber provision to meet the continent-wide constraint on harvest levels in the simulations (Fig. 3.6d). This came as a trade-off with other ESs in those regions, particularly the provision of habitat for biodiversity. If meeting the strict protection target was only enforced on a pan-European level (and not for each grid cell), those regions barely contributed to the Europe-wide targets for strict forest protection at all (Fig. 3.6d). Therefore, the types of protected forest ecosystems would be less geographically and ecologically diverse than desired. In addition, portfolios were much less diverse when applying constraints to the optimization (compare Fig. 3.6c and e). This conflicts with the aim of creating diversified portfolios to address future uncertainty. Nevertheless, numerous pathways exist to alleviate the pressure from forests, such as a focus away from woody bioenergy, promoting increased material wood usage, reforestation, and addressing rising demands for single-use wood products.

4.1.4 Research question 4: What are the greenhouse gas mitigation potentials of different climate change mitigation strategies of forests and agriculture in Bavaria?

Large-scale land-based climate change mitigation holds significant potentials. However, its impact decreases when it is implemented at scales that avoid interference with food production and other land uses. In section 3.4, I analyzed five land-based mitigation measures in Bavaria at realistic rates. The effects of decreased fertilizer application, increased material use of harvested wood, enhanced bioenergy crop utilization, carbon capture and storage (CCS), and reforestation each roughly fell in the order of 100 $MtCO_{2e}$. The combined greenhouse gas mitigation impact when applying all measures simultaneously and assuming a decrease in the carbon-intensity of non-wood products in line with current European Union (EU) policies was estimated at 630 $MtCO_{2e}$. This is equivalent to about six years of the state's current annual greenhouse gas emissions. However, these values are subject to decrease when decarbonization speeds up. Should the economies of the EU meet their net-zero targets, land-based mitigation potentials would decrease by 25%. This underscores the urgency of a rapid implementation of measures, as also discussed in the answer to research question 1. Furthermore, the impact of bioenergy may become marginal if decarbonization accelerates in line with the EU's net-zero targets and when CCS proves infeasible. In that case, the total mitigation potential would be a mere 277 $MtCO_{2e}$, or three years of emissions.

4.2 Implications of the results

4.2.1 The need for, and implications of, the adaptation of forests to climate change

Forest management today must consider potentially drastic changes in environmental conditions including rising temperatures, changing rainfall patterns, and more frequent disturbances. My simulations indicated that a forest management option that is sensible in one climate scenario may not be sensible in a different one (see Fig. 4.1). In addition, stronger climate change substantially decreased the carbon sink of European forests in my simulations, especially due to the related increased frequencies of disturbances (see Fig. 3.1). Consequently, forward-looking, adaptive management strategies are necessary, including measures like enhancing species diversity and structural diversity, and taking into account the potential range of changes in climate and disturbance regimes (see also Jandl et al., 2019).

The mitigation potentials of a forest and the provision of other ESs differed significantly depending on the climate and disturbance scenario in my simulations (see Figs. 4.1 and 3.1). Nevertheless, forest management strategies need to be developed today and adaptation measures take a long time to materialize, emphasizing the need for robust strategies. In paper 2 (section 3.2), I therefore introduced a methodology to derive forest management strategies suitable for a wide range of climate scenarios.

Consistent with recommendations by Lawrence et al. (2020) and Schwalm et al. (2020), my results suggest that adapting forests for a wide range of emission and climate scenarios is crucial. This is also endorsed by the IPCC, to avoid costly adaptation oversights for extreme cases (IPCC, 2021). The robust methodology presented here indicated that transitioning to more mixed and broad-leaved forests is beneficial for climate-adapted multi-functional forests in Europe (Fig. 3.4). This aligns with various studies emphasizing advantages of broad-leaved species, such as lower air temperatures in summer (Schwaab et al., 2020) and reduced fire risk (Astrup et al., 2018). Furthermore, similar to my results, biodiversity benefits have been found when replacing coniferous monocultures in Scandinavia with mixed and broad-leaved forests (Felton et al., 2010). In addition, positive effects on disturbance resistance and resilience, as well as reduced financial risks have been identified from admixing broad-leaved trees to coniferous stands (Knoke et al., 2008). Notably, such forest adaptation is already occurring in many countries, such as the "Waldumbau" in Germany (BMEL, 2014).

However, increasing the share of broad-leaved species will likely alter the wood product portfolio, potentially leading to a smaller product sink which is important for total mitigation impact (section 3.1). Therefore, this adaptation could have unintended negative consequences for the carbon balance and mitigation targets. Such measures should be accompanied by investments in innovative technologies for hardwood use in construction (Hassan & Eisele, 2015) to compensate the lower supply of softwood. Another strategy would be to further enhance the share of softwood that is used for long-lived products, but this has limited potential since these shares are already high (Eurostat, 2021a).

4.2.2 Insights on the climate change mitigation potentials of forests

Natural climate solutions, including forests and forest management, play a pivotal role in climate change mitigation strategies (Griscom et al., 2017). However, there is ongoing debate about the optimal approach for utilizing forests in mitigation efforts. Numerous studies advocate for a reduction in harvests to increase carbon uptake (Matsumoto et al., 2016; Schulte et al., 2022; Skytt et al., 2021). Nature conservation organizations have even suggested that leaving substantial portions of forests unmanaged would provide the largest climate benefits due to the enhanced carbon sink (Greenpeace, 2018). Reducing or refraining from harvesting can have numerous additional benefits, for example for biodiversity or water availability (Diaz-Balteiro et al., 2017; Gregor et al., 2022; Vuidot et al., 2011).

However, understanding the true impact of decreased harvest intensities on climate change mitigation requires an in-depth analysis. Firstly, it needs to be acknowledged that studies advocating for less intensive management to enhance the carbon sink often overlook the influence of historical and future global environmental changes on forest growth and mortality. For instance, the influence of nitrogen deposition and CO_2 fertilization on historical increases in forest growth are often ignored (Nabuurs et al., 2013; Walker et al., 2021). Moreover, the CO_2 fertilization effect has been observed to decline (Wang et al., 2020; Winkler et al., 2021) and the associated accelerated growth is thought to be linked to faster tree mortality (Brienen et al., 2020; Bugmann & Bigler, 2011; Körner, 2017). This may eventually offset the increased carbon uptake. Additionally, mitigation studies often overlook the projected decline in total nitrogen deposition across all RCPs (Lamarque et al., 2011) and the rising frequency of disturbances (Senf & Seidl, 2021a).

In the simulations from paper 1 (section 3.1), where I included all the aforementioned factors, stopping or decreasing harvest intensities led to an increased forest sink until 2050 (Fig. 3.1). Until 2100, however, stopping management exerted a negative impact on the carbon sink in some simulations, mainly driven by carbon losses from disturbances. More importantly, the total mitigation impact of decreased or stopped harvests was mostly negative until 2050. This was because the missing wood products needed to be replaced by other materials, usually with a negative climate impact (Knauf et al., 2015). My results emphasize the pivotal role of these substitution effects, especially until 2050. But the timing is crucial, as displacement factors are expected to decrease with ongoing decarbonization (Brunet-Navarro et al., 2021). This underscores the importance of implementing wood-based strategies in a timely manner, while displacement factors are still high.

Even with fast decarbonization, substitution effects remain high enough in the coming decades that wood products continue to offer a climate benefit over non-wood products. This result seemingly contradicts findings from the literature suggesting a mitigation benefit from reduced harvest intensities or prolonged rotation periods (Dugan et al., 2018; Matsumoto et al., 2016; Schulte et al., 2022). However, the displacement factors considered in these studies were lower than the ones used here, namely around 0.45–0.6 tC/tC whereas I employed displacement factors of 1.5 tC/tC obtained from regional analyses of wood and non-wood products (Knauf et al., 2015). This difference could have multiple reasons, including the underlying energy mix, product portfolios, and end-of-life handling in the considered regions.

Increasing harvest intensities, on the other hand, may provide climate benefits until 2050 according to my simulations, in line with Nabuurs et al. (2018). However, this positive effect depends on high substitution effects and diminishes with ongoing decarbonization. Furthermore, increasing harvests will likely have negative consequences for other ESs. For instance, a reduction in the abundance of large trees leads to a decreased number of micro-habitats (Vuidot et al., 2011).

Wood products offer various societal benefits and play a significant role in the carbon balance. The product sink in my simulations accounted for over 10% of the in-situ forest sink (section 3.1), aligning with other estimates in Europe (Grassi et al., 2021; Holmgren, 2020; Verkerk et al., 2022; Wolf et al., 2020). Increasing the share of material usage holds significant potential for augmenting both the product sink and substitution effects according to my results, consistent with other studies (Nabuurs et al., 2018; Smyth et al., 2014). This could, for example, be achieved by investing in wood-based construction methods (Churkina et al., 2020). However, this would entail the necessity for worker training, new building codes and addressing the public perception of wood in construction (Howard et al., 2021). Furthermore, large shares of wood from conifers are already used for long-lived products, the potential of increasing this share is thus limited. For hardwood, however, there is a higher potential, because large fractions are currently used for energy (Eurostat, 2022). There is also the potential to increase lifespans of wood products. These are frequently lower than they could be, often for aesthetic reasons (Hill et al., 2022).

A shift towards increased material usage would lead to a decreased provision of woody bioenergy, which currently provides about 60% of the EU's renewable energy (European Commission, 2021b). But woody bioenergy is not a particularly efficient energy source. Over the course of one year, the energy yield per unit area obtained from forest plantations is more than 100 times lower than that of solar panels and 10 times lower than that of wind turbines (Smil, 2015). The impacts of woody bioenergy on the carbon balance also remain heavily debated, due to its large emissions per unit energy, uncertainties regarding forest regrowth in the light of climate change and disturbances, and often inefficient combustion (Booth et al., 2020; Cherubini et al., 2011; Fajardy et al., 2019; Holtsmark, 2013; Law & Harmon, 2011). I therefore argue that focusing on a material usage of wood and at the same time investing into the deployment of working renewable technologies and energy storage is a more promising pathway.

Furthermore, reducing landfilling and enhancing recycling emerges as a key strategy in some regions to provide rapid and large climate benefits, given its crucial impact on displacement factors. For example, in the meta-analysis of Sathre and O'Connor (2010), mean displacement factors for wood products were 2.1 tC/tC and 1.1 tC/tC when excluding and including landfilling, respectively, highlighting its potential climate impacts (Sathre & O'Connor, 2010). In 2021, 23% of municipal waste in the EU was still landfilled, varying significantly among countries with shares ranging from 1% to 97% (Eurostat, 2023). Therefore, avoiding wood from being landfilled could have a substantial impact on greenhouse gas (GHG) emissions in some countries.

Finally, my results indicate that climate change mitigation strategies can critically depend on the assumed displacement factors (Gregor et al., 2024; Skytt et al., 2021). Their quantification depends on the analysis of wood usage statistics, cradle-to-cradle emissions, and determining functional equivalences between wood and non-wood products. To improve this quantification, it is necessary to consider a broader range of products and more detailed usage patterns, particularly regarding end-of-life usage and recycling. Furthermore, data on the carbon uptake resulting from the carbonization of cement-based materials need to be taken into account (Gustavsson et al., 2021), along with detailed insights into product lifetimes. Also, critical assumptions of substitution effects must be considered (van Kooten & Johnston, 2016). Key questions in this regard are whether the product would have been created anyway (additionality), whether the avoided material was not simply used otherwise (leakage), and what type of material of fuel was actually substituted (replaceability).

4.2.3 Effects of forest conservation and adaptation in Europe and abroad

As discussed before, reducing harvest intensities or completely stopping forest management might not be the most sensible solution for climate change mitigation due to the decreased substitution effects (section 4.2.2). However, such measures provide numerous other benefits beyond mitigation, including improved provision of biodiversity habitat (for instance through higher abundance of deadwood and large trees) and other ESs (Başkent & Kašpar, 2023; Felton et al., 2016; Gutsch et al., 2018; Lessa Derci Augustynczik & Yousefpour, 2021; Verkerk et al., 2014). Furthermore, it is critical that forests are not reduced to their climate change mitigation potentials, but to consider these numerous ESs, not least because human well-being depends on them (Díaz et al., 2006; Millenium Ecosystem Assessment, 2005). The methodology proposed in paper 2 created optimized forest management portfolios for the balanced provision of multiple ESs and revealed that setting aside portions of managed forests improves the provision of other ESs (section 3.2).

However, it is essential to acknowledge that significant portions of forests in Europe belong to private owners. And though their objectives are diverse, timber production is one of their key objectives (Hirsch & Schmithüsen, 2010). Therefore, reducing or even halting harvests to foster carbon sequestration and biodiversity would need to be compensated financially (Khanal et al., 2017). Fortunately, financial incentives for forest conservation have already proven effective in Europe. A pioneer of such successful programs is the *Metso programme* of Finland, a payments for ecosystem services (PES) scheme that offers payments to forest owners for conserving existing forests. The project has already achieved more than 90% of its target, protecting over 85,000 ha of forests today (Metso, 2023). Numerous other European countries have PES schemes or frameworks in place to enhance carbon sequestration and other ESs (Maier et al., 2021).

Nonetheless, decreasing harvests and conserving forests stand in conflict with increasing wood demands (FAO, 2022c; IEA, 2022). In addition, as mentioned before, forest adaptation towards more broad-leaved species is already underway. This leads to a change in product portfolios, reducing the supply of softwood, which is more suitable for construction purposes than hardwood. Therefore, without corresponding action, these measures pose the threat of exporting ecological pressure elsewhere (Berlik et al., 2002; Mayer et al., 2005). Indeed, studies have shown that the EU's New Green Deal leads to negative consequences in other parts of the world, including deforestation and an increased use of fertilizer, herbicides, and pesticides (Cerullo et al., 2023; Fuchs et al., 2020; Rosa et al., 2023). It is thus imperative that any measure undertaken in Europe will be accompanied by corresponding actions to safeguard ecosystems elsewhere.

My results indicated that large shares of unmanaged forests are specifically important for biodiversity habitat provision (section 3.2), posing a conflict to increasing wood demands. However, the simulations did not include detailed management schemes that realistically depict possible co-benefits between productivity and biodiversity. Research indicates that combining biodiversity protection and forest usage may be possible. Firstly, biodiversity depends more on stand and tree characteristics than whether the forest is managed or not (Vuidot et al., 2011). Secondly, biodiversity also fosters productivity (Liang et al., 2016). Therefore, according to these studies, forest management that also supports species diversity and fosters microhabitats can provide timber while simultaneously safeguarding biodiversity. The large shares of unmanaged forests in the optimized portfolios could thus potentially be partly replaced by forests where the management is in line with biodiversity-promoting characteristics.

4.2.4 Using robust MCDM to improve projects for afforestation, reforestation, and avoided deforestation

In contrast to other parts of the world, the forest area in Europe is increasing. In recent years, this has happened at a rate of approximately 388.000 ha/yr, mainly because of agricultural abandonment and reduced grazing pressure, but also due to reforestation projects (European Environment Agency, 2019; Forest Europe, 2020; Grassi et al., 2021; van der Zanden et al., 2017). The resulting forest regrowth has absorbed approximately 45 $MtCO_{2e}$/year, or 1.5% of the EU's 2021 CO_2 emissions, thereby posing a small, but important, contribution to the EU's net-zero target (Friedlingstein et al., 2023; Schreyer et al., 2020). The EU Biodiversity Strategy aims to sustain this trend by planting three billion trees (European Commission, 2020).

It is imperative that multiple goals are considered when planning reforestation projects, because simple tree planting schemes have been linked to numerous negative ecological and economic impacts, including the decimation of biodiverse local ecosystems or the displacement of cropland, which may lead to deforestation elsewhere (Bond, 2016; Bond et al., 2019; Gómez-González et al., 2020; Holl & Brancalion, 2020).

The methodologies presented in this thesis can be applied to inform and plan such reforestation projects. A sensible management of these new forests with regard to species and harvest intensity can maximize their mitigation impact. Furthermore, as presented in section 3.2, detailed modeling and robust multi-criteria optimization can help develop optimal reforestation projects, providing not only carbon sequestration but also various ESs under a wide range of climate change scenarios. In that regard, it is particularly important to include indicators for biogeophysical effects in these assessments, as afforestation projects may have unintended negative consequences on the climate, for instance through changes in albedo (Baldocchi & Penuelas, 2019; Betts, 2000; Bonan, 2008).

Similar to forest conservation, reforestation in Europe may lead to deforestation elsewhere. Deforestation, primarily driven by agricultural expansion and livestock management (dos Santos et al., 2021), significantly impacts climate change and biodiversity loss worldwide (Friedlingstein et al., 2023; Pereira et al., 2012; Vitousek, 1994). The responsibility for these clearings extends beyond the borders of the affected country, because a large share of the corresponding products are exported (Fuchs et al., 2020; Picoli et al., 2020). Economic motivations are the main reason for these clearings, highlighting the necessity for compensation strategies, for example by compensating landowners through payments for ecosystem services (Araya & Hofstad, 2016; Bruzzese et al., 2023; Wells et al., 2020; Wunder et al., 2018; Wunder et al., 2020).

Compensations for reforestation and avoided deforestation are now increasingly often paid for by companies offering carbon offsets for corporate emission goals and to soothe the conscience of individuals. There are numerous issues with such offsets and the underlying reforestation projects, including fraud and poor planning (Greenfield, 2023). Such projects can lead to negative consequences for the region, such as reduced water supply, the destruction of native ecosystems, or increased social inequality (Bond, 2016; Cao, 2008; Holl & Brancalion, 2020). Key issues are that ESs are not considered in such projects, and that simple monoculture plantations are planted, often also without subsequent monitoring or tending (Cao, 2008; Hua et al., 2022).

Including the assessment of ESs can make such projects a success for climate mitigation while also providing the local communities with income, wood, and clean water (Araya & Hofstad, 2016; Neary et al., 2009). Indeed, a large fraction of the population in the tropics (where these projects are usually conducted) depends on forests and nature for their basic needs, such as housing, water, energy, occupation, and food (Brandon, 2014; Fedele et al., 2021). Consequently, these services need to be included. Also economic indicators can help develop strategies for the financial benefit of local and indigenous communities, potentially reducing necessary financial compensation for clearings (e.g, Knoke et al., 2014; Knoke, Paul, et al., 2020). In that regard, the difficulty regarding the access to money (e.g., in terms of financial loans) should be addressed, for example, by including the payback times of investments into the planning (Knoke, Paul, et al., 2020). Furthermore, financial constraints on project costs can be incorporated into the robust optimization approach presented in this thesis. Indeed, previous applications of multi-criteria optimization for global ecosystem restoration have already shown that restoration can be optimized with regard to climate and biodiversity, while offering a 13-fold increase in cost-effectiveness compared to a baseline restoration approach (Strassburg et al., 2020).

Also the impacts of climate change and disturbances on forests need to be considered when planning and implementing reforestation projects, but this is commonly not the case (Lefebvre et al., 2021). Including robustness towards climate change scenarios, as done in this thesis, would be the sensible next step for such an assessment. Nevertheless, as discussed below, the climate change mitigation potentials of realistic afforestation and reforestation should not be overestimated and used as a reason to delay actual emission reductions.

4.2.5 Implications of regional considerations and stakeholder engagement for the development of climate-smart forestry strategies

There was generally no preference given to the ESs in my assessments and they were included into the optimization with equal weights. As an example, however, I have shown how increasing the weights for certain ESs changed the outcomes of the optimization. For instance, increasing the preference for wood products decreased the share of unmanaged forests in the optimized portfolios (see section 3.2). Given the strong trade-offs that arose when focusing on multiple ESs, it appears that a ranking or weighting of ESs will be necessary. Such a ranking will also depend on the region where the methodology is applied.

In Mediterranean regions, for instance, water retention will likely outweigh other considerations. In this region, decreased soil moisture and rainfall, and an increase in rainfall seasonality are expected in the future, alongside more hot extremes and agricultural and ecological drought (IPCC, 2021, 2023c). The effects of forestry on water have already been acknowledged in this arid region, making "water management through forest management" a key strategy (Bredemeier, 2011). This entails forestry measures improving the water holding capacity of soils and preventing erosion from heavy rainfall events, for instance by appropriate species selection and deadwood management (Bredemeier, 2011; Dunjó et al., 2004). In Scandinavia, on the other hand, the emphasis might instead lie on timber provision. As my results of paper 3 have shown, meeting increasing timber demands in Europe might require this region to focus on timber provision (see section 4.2.6).

Other regions will again have different objectives and likely other rankings of ESs. Obtaining such weights may involve consulting regional stakeholders and experts, for example through surveys. Nevertheless, it must be kept in mind that such weighting of ESs can have a significant influence on the results. Knoke, Paul, et al. (2020) for example have shown how increasing the weights on economic indicators may lead to substantial deforestation in a land allocation study in the tropics. In general, weighting ESs can introduce biases (Knoke, Paul, et al., 2020), therefore stakeholders must be selected carefully. The methodology of stakeholder involvement and education also plays a critical role in obtaining useful weightings. In that regard, iterative and interactive approaches are suggested to build trust and cohesion between scientists, stakeholders, and decision makers (Ruckelshaus et al., 2015). Different stakeholders may also have diverging preferences. One suggestion to address these could be to include the spread in weights as uncertainty in the robust optimization. This would result in solutions that are viable for a range of weightings, potentially finding strategies that satisfy various stakeholder groups.

4.2.6 Addressing additional demands on forests

Throughout this thesis, I have highlighted the multitude of demands that are placed on forests, including timber provision, climate change mitigation, provision of habitat for biodiversity, and other ESs. In the EU, new strategies formalize such demands. The EU Forest Strategy, for instance, aims at increasing the supply of long-lived wood products, fostering multi-functionality, and boosting EU forest ecotourism, while also promoting broad-leaved trees, deadwood, and the strict protection of large forest areas (European Commission, 2021b).

If such strategies are enacted into legislation, they pose hard constraints on the options to allocate forest management portfolios. Furthermore, total demand on wood products is rising and projected to further increase (FAO, 2022c; IEA, 2022). Meeting this demand poses another constraint. In paper 3 (section 3.3), I demonstrated how such constraints can be incorporated into the robust optimization, to optimize ES provision while respecting the given constraints across all climate scenarios. The results indicate that these constraints interfere with one another, leaving only limited room to further optimize the provision of other ESs. This limitation also led to the fact that forest portfolios were much less diversified compared to the original optimization without additional constraints (compare Fig. 3.5 c and e), posing a critical threat as diversification is a key strategy to minimize future risks.

To meet timber demands in Europe, the constrained optimization suggested that highly productive forests need to be predominantly used for timber provision to compensate for low timber provision elsewhere (Figs. 4.2 and 3.6). However, protection should target multiple ecosystems. And many red-listed insects depend on highly productive forests (Hämäläinen et al., 2018). Therefore, also strict protection of such areas are necessary and not only protection of low-productivity areas and regions that are hard to access (though note that protecting low-productivity ecosystems may on the other hand be particularly important for plant diversity, see Ellenberg (1986)).

Including such protection of productive forests will further decrease the options for ES optimization and possibly lead to an optimization problem without a feasible solution that satisfies all constraints. Nonetheless, these constraints are reasonable to ensure healthy forests, soils, fauna, and ecosystems in the future and thereby also ensure the livelihood of people on the continent. Therefore, the pressure on forests must somehow

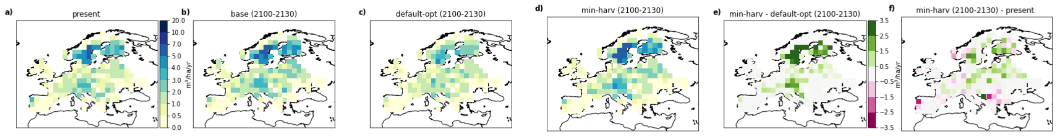

Figure 4.2: Harvest levels for different scenarios: present (a), base (business as usual) (b), un-constrained optimization (c), and optimization when imposing constant harvest rates across the continent (d). In productive regions, for example southern Scandinavia, the management portfolio needed to focus on timber production to ensure that the Europe-wide constraint on increased harvests was fulfilled. This depicted by much stronger wood harvests in that region compared to the default option (e) and somewhat stronger harvests than today (f).

be alleviated. In Gregor et al. (draft), I have discussed some ideas to ease this pressure, for instance by increasing the share of material usage of wood, enhancing the lifespan and recycling of products, focusing on other renewable energy sources, and promoting innovative usages of softwood. But also addressing the increasing per-capita floor sizes (Bierwirth & Thomas, 2015) and the rise in demand of single-use products like packaging and sanitary papers (FAO, 2022c) are possible pathways.

In conclusion, the desire of using forests for timber, bioenergy, mitigation, recreation, as habitat for biodiversity, and for local climate regulation is too much to demand of an ecosystem that faces environmental challenges. While strategies can be derived to optimize the provision of ESs, these can only be successful if we simultaneously address the demand side of the equation. Fortunately, win-win solutions may exist, such as simultaneously improving biodiversity and productivity, for instance by mimicking natural characteristics in managed forests (Brockerhoff et al., 2017; Liang et al., 2016; Vuidot et al., 2011).

4.2.7 The influence of agriculture and land management in Bavaria on climate change and biodiversity

Thus far, I have only considered forests, but agriculture also has a tremendous impact on climate change and biodiversity. Historical deforestation for agriculture and intensive agricultural practices have led to habitat loss and degradation, posing the most critical threat for biodiversity (Pereira et al., 2012). Forest clearing for agriculture has also contributed to the majority of the rather stable ∼1 GtC/yr emissions from land use change over the past century (Friedlingstein et al., 2023). Therefore, re- and afforestation are currently receiving high attention for climate mitigation, with papers indicating a large potential of such measures (e.g., Bastin et al., 2019; Mo et al., 2023; Walker et al., 2022). But some of these papers are disputed, see comments by Friedlingstein (2019), Lewis et al. (2019), and Veldman (2019). These comments note methodological inconsistencies and emphasize that we cannot plant our way out of the climate crisis, as the original papers were often interpreted by the media, politicians, and the general public. My estimations for realistic reforestation rates in Bavaria support this statement, in part also because unlike those papers, my simulations considered the necessary long timescales for forest growth, as well as the fact that larger carbon stocks pose a higher risk to future disturbances.

My simulations indicated that successively reforesting 5% of Bavaria's cropland area until 2050 could provide a small carbon uptake of about 77 MtCO_2 until 2100, equivalent to less than one year of Bavaria's annual GHG emissions. This aligns with estimates of Krause et al. (2020), indicating a potential of 1767 MtCO_2 when reforesting the entire cropland area of Bavaria immediately. Notably, the average carbon uptake of reforested land in my simulations reached 226 tCO_2/ha/yr, almost twice the rate that was recently estimated for Europe, at 116 tCO_2/ha/yr (Grassi et al., 2021). Potential reasons may include Bavaria's climate, its fertile soils and previous land use patterns (Krause et al., 2016).

Reforestation certainly is an important measure for mitigation, but its impact must not be overstated. It is also crucial that reforestation projects are carefully planned regarding their location, implementation, and continued management, to avoid unintended negative side effects (discussed in section 4.2.4).

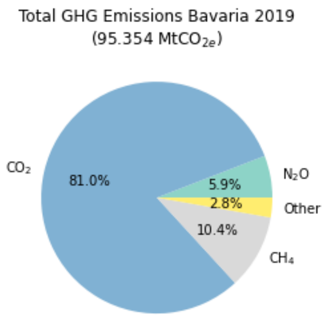

Figure 4.3: Total greenhouse gas emissions in Bavaria in CO_2-equivalents based on a 100-year time horizon. About 16.3% stem from methane (CH_4) and nitrous oxide (N_2O). These predominantly occur in agricultural practices (Statistische Ämter der Länder, 2022).

Also ongoing agricultural practices such as animal husbandry and fertilizer application lead to substantial emissions. These recently accounted for over 13% of Bavaria's annual GHG emissions (Fig. 4.3).

According to my results, a 20% decrease of fertilizer application in Bavaria can achieve a reduction of approximately 75 MtCO_{2e} of nitrous oxide emissions until 2100. Wheat yields decreased by 10% in response, while the model only showed a marginal decrease in maize yields. One reason for this is that maize in Germany is already heavily fertilized, partly even over-fertilized (Villwock et al., 2022). Model improvements for crops in LPJ-GUESS are still underway (Camargo-Alvarez et al., 2023), but my results already indicate that to a small extent, a reduction in fertilizer application may be a suitable means towards climate change mitigation. Furthermore, numerous new ways of fertilizer application have been shown to reduce emissions while keeping yields stable (Banger et al., 2020; Li et al., 2022; Sishodia et al., 2020).

Methane emissions, primarily from cows, contribute to over 10% of current Bavarian GHG emissions[1] and global food consumption alone could contribute to 1 °C of climate warming by 2100 (Ivanovich et al., 2023). Therefore, reductions in animal product consumption could achieve significant reductions in GHG emissions and land usage. Such reductions are potentially even necessary to meet EU climate targets (Alexander et al., 2017; Bryngelsson et al., 2016; Joyce et al., 2014; Treu et al., 2017). Notably, also food supplements for cows can substantially reduce their methane emissions (Bačėninaitė et al., 2022; Kolling et al., 2018).

Agriculture also plays an active role in combating climate change, primarily through bioenergy crops which avoid emissions from fossil fuels. In the future, bioenergy will ideally be combined with CCS, though this technology is still in very early stages. However, according to the scenarios of the Intergovernmental Panel on Climate Change (IPCC), bioenergy with carbon capture and storage (BECCS) is included in many pathways that are compliant with the 1.5 °C target (Rogelj et al., 2022).

My results have indicated a large hypothetical potential of BECCS (up to 312 $MtCO_2$), but a much lower potential of bioenergy without CCS, in line with estimates by Krause et al. (2020). If displacement factors declined in line with EU's net-zero targets, the cumulative avoided emissions of dedicated bioenergy crops without CCS in Bavaria (using 10% of the current cropland area) would only have a marginal impact of 29 $MtCO_2$ from avoided fossil emissions. Bioenergy and BECCS are heavily debated due to the lower energy density compared to other energy sources (Smil, 2015), and because bioenergy often replaces other renewable energy sources and not fossil fuels (Booth et al., 2020). Furthermore, CCS is far from ready for deployment at scale (Fajardy et al., 2019). Overemphasizing speculative technologies like BECCS carries substantial risks, as it diverts resources away from proven technologies. It may also provide a false sense of reassurance that allows for continued high emissions, assuming that CCS will address this issue in the future. Should BECCS ultimately prove ineffective, this commitment would bind us to a high trajectory of global warming. In addition, bioenergy requires large land areas which conflicts with biodiversity goals and food production (Hof et al., 2018; Ohashi et al., 2019). Finally, bioenergy targets partly rely on further yield increases of bioenergy crops despite recent declines in yield improvements, consequently requiring fertilization which was not considered here (Fajardy et al., 2019). Future analyses need to investigate the trade-off between carbon uptake and avoided fossil emissions, and nitrous oxide emissions from fertilization.

To conclude, the combination of decreased fertilizer usage, increased material usage of wood, reforestation, and BECCS in my study achieved a mitigation impact of up to six years of emissions (which could be enhanced by reduced methane emissions). This number refers to the impact of the additional measures, that needs to be added to the effect of already existing mitigation, e.g., from the forest sink of existing forests. This estimation underscores that although nature-based solutions are critical to meet climate targets (Griscom et al., 2017; Schreyer et al., 2020), their total impact should not be

[1] This considers a 100-year time horizon to convert methane emissions to CO_2-equivalents. Due to the short lifetime of methane in the atmosphere, however, its short-term climate impact is much higher.

overstated. Furthermore, the impact on other ecosystem services of such measures need to be considered (Krause et al., 2017). However, the faster the changes are implemented, the higher the impact, especially because substitution effects will likely decrease.

4.2.8 Developing land management strategies for numerous ecosystem services under uncertain future climate

The robust portfolio approach – providing management strategies for numerous ESs under a wide range of climate scenarios – can be extended to agriculture and to land use as a whole. A recent MCDM study has shown that land use optimization can lead to co-benefits between food production, water availability, and carbon storage (Bayer et al., 2023). Such an optimization may therefore be useful to derive concrete land use strategies. In future work, a multitude of ESs could be considered, such as mitigation, adaptation, biodiversity, water availability and quality, timber and food provision, fuel provision, and pollution such as eutrophication. An additional key field that should be included is soil health. In Europe, soil has started to gain increased attention by policy makers through the *EU Soil Strategy for 2030* (European Commission, 2021a). This strategy not only aims at improving soil health, but also formalizes a "no net land take by 2050" target. Similar to the previously included constraints for forestry (section 3.3), this target can be formalized as a hard constraint on a stable sealed land area in the optimization. Such a constraint requires that the available land needs to be used wisely. An optimization approach such as the one discussed here would be a reasonable tool, particularly because in some regions of Europe, the production efficiency of crops cannot be enhanced any further (Neumann et al., 2010). Thus, substituting cropland expansion with intensification is not possible. Further additional constraints could ensure certain levels of food provision (e.g., calories, proteins, and fats) for all strategies and all climate change scenarios.

As discussed above, land use likely conflicts with biodiversity. However, with careful planning, land management can be be executed in a way that minimizes its adverse effects on biodiversity. For instance, it should be considered whether land sparing (e.g., separating landscapes into production and conservation) or land sharing (e.g., providing production and biodiversity on the same land) is the suitable approach in a given region (Betts et al., 2021). For this, species density - yield curves need to be obtained to assess the optimal combination of production and biodiversity (Betts et al., 2021). For example, for some animals, land sparing is better because these species thrive only on either unmanaged forests, or only on farmland (Phalan et al., 2011). On the other hand, for some animals, land sharing is better because they need intermediate landscapes where land is used but not intensively managed (Phalan et al., 2011). The so-called Triad approach is a combination of the two strategies, incorporating intensive management, extensive management, and nature reserves (Betts et al., 2021). My portfolio approach implicitly applied land sparing by separating the landscape into multiple management forms. Further methodology improvements such as the inclusion of various intensities of management should be included to facilitate a sharing-sparing-Triad analysis.

Also in other regions, such as the tropics, land management and not only forest management strategies are important, especially in times of deforestation. Here, the proposed methodology may help derive land use strategies for the benefit of local communities, ecosystems, and the climate. However, recent applications of robust MCDM have found that striving for multi-functional landscapes may potentially clash with the goal of decreasing deforestation (Knoke, Paul, et al., 2020). This could be alleviated by compensation payments or by allowing sustainable harvesting practices for local communities (Araya & Hofstad, 2016).

4.3 Required further work to apply the methodologies into practice

The analyses I have carried out in this thesis offer a foundation to develop concrete land use strategies for mitigation, adaptation, and ecosystem service provision. However, apart from the enhancements mentioned above, additional efforts in terms of model and framework development are necessary for practical applications such as forest management planning or the development of reforestation projects. The key points are summarized below.

4.3.1 Model improvements

4.3.1.1 Implementation of realistic disturbances and extreme events

In my analyses, disturbances emerged as a key driver of forest dynamics and highly affected mitigation potentials and ES provision. In Europe, disturbances have also been attributed a pivotal role in the decreasing forest carbon sink (Nabuurs et al., 2013). Nevertheless, disturbances are currently inadequately represented in models and often neglected in mitigation studies. Throughout this thesis, I accounted for the different susceptibilities of forest types to disturbances and the anticipated increase in disturbance frequencies (Fig. 2.10). However, this may still be an underestimation because it neglected species-specific effects such as the high susceptibility of spruce monocultures to bark-beetle attacks (de Groot et al., 2019). Newly emerging threats, such as those affecting European beech in central Europe, underscore the evolving challenges (Buras et al., 2020; Schuldt et al., 2020). First efforts towards a more detailed representation of bark-beetle outbreaks and windthrow have already been made (Jönsson et al., 2012; Lagergren et al., 2012) but further refinements of these models are necessary.

In recent decades, drought and heat events have led to severe impacts on humans and ecosystems in Europe (Bastos et al., 2020; Buras et al., 2020; Ciais et al., 2005). Such extreme events are expected to increase in frequency and intensity both in Europe (Meehl & Tebaldi, 2004; Spinoni et al., 2018) and world-wide (Allen et al., 2010; Dai, 2013; IPCC, 2023b). In this thesis, drought effects on plants were modeled explicitly through the impact of water stress on photosynthesis and impaired re-establishment. Furthermore, soil water content was used as an indicator for water availability in the optimizations. However, drought influences many tree physiological processes such as the down-regulation of stomatal conductance and cavitation-induced mortality. These processes are currently not included in LPJ-GUESS but research indicates that they are critical in adequately capturing vegetation responses to drought (Papastefanou et al., in review). The implementation of these processes is a complex undertaking which is currently addressed within many modeling communities (Kennedy et al., 2019; Meyer et al., draft; Papastefanou et al., in review; Xu et al., 2023; Xu et al., 2016; Yao et al., 2022). Furthermore, by influencing forest structure, forest management can improve the resistance and resilience of a forest to disturbances and extreme events, but this intricate relationship still needs to be implemented in models (Brockerhoff et al., 2017; de Groot et al., 2019; Millar et al., 2007; Mitchell, 2013; Pretzsch et al., 2013; Senf et al., 2019).

LPJ-GUESS also does not include micro-climate and its response to disturbances. Micro-climate strongly affects forest ecology (Buras et al., 2018; De Frenne et al., 2021). Alongside other aspects such as deer browsing or grass cover, it can also strongly impair the regeneration after disturbance, i.e., the forest's resilience (see Fig. 4.4). For instance, daytime temperatures in a clearing can be several degrees higher than in a nearby forest, accompanied with increased dryness and radiation (Bonan, 2008). Consequently, a seed that would have thrived in the forest may not establish due to the significantly altered micro-climate resulting from a disturbance. Incorporating these aspects into models will greatly improve our confidence in model projections, especially considering that extreme events are expected to become more frequent in the future.

Figure 4.4: A disturbed forest in Črnivec, near Kamnik in Slovenia. This site was hit by a strong summer storm in 2008 and was salvage logged afterwards. By the time the photo was taken in 2023, natural regeneration was barely visible, due to the thick grass cover, deer browsing, and changed micro-climate (see also Fidej et al. (2018)). Photo: own.

4.3.1.2 Improvements in land use implementation, species parametrization, and ecosystem service indicators

Apart from the impact of forest management on disturbance susceptibility, the implementation of management in models needs to be improved. In particular, new paradigms such as closer-to-nature forestry and continuous cover forestry need to be implemented in greater detail to allow the development of concrete and applicable strategies towards climate-smart forestry. Similarly to disturbance impacts, the effect of clearcuts needs to be realistically assessed which requires the implementation of micro-climate and other local effects. These detailed management implementations will allow the investigation of land-sharing, land-sparing and triad approaches (Betts et al., 2021), because they will allow a realistic inclusion of extensive and intensive management practices into the simulations. This may also include using my coppice implementation to simulate short-rotation coppices. Furthermore, numerous new species including Douglas fir (*Pseudotsuga menziesii*) and black pine (*Pinus nigra*) are deemed important for climate-resilient forests in Europe (Spiecker et al., 2019; Vacek et al., 2023). Their parametrization into models is another key research alley.

Additional ecosystem service indicators need to be implemented to assess climate-smart forestry and land use, requiring model improvements. Biodiversity habitat provision, for example, was assessed here through tree size diversity, the amount of large trees, and the amount of deadwood. This assessment can be improved by implementing more detailed types of deadwood that are important micro habitats, for example standing dead trees (Vuidot et al., 2011). Furthermore, the representation of grasslands in models is often highly simplified (Wirth et al., 2023). However, these ecosystems can foster thriving biodiversity (Weisser et al., 2017). Real applications of robust MCDM for land use strategy development will require a more detailed implementation of grasslands and other non-forest ecosystems. Soil quality, critical for healthy ecosystems, may be assessed through additional indicators including pH, base saturation, or carbon and nitrogen mineralization (Knoke et al., 2014). Also protection services such as avalanche or landslide protection may be integrated, for instance by computing protection indices from stand characteristics (Cordonnier et al., 2014).

Finally, biophysical effects (albedo, surface roughness, and evapotranspiration) need to be addressed in more detail because of their important role in the climate system (Betts, 2000; Bonan, 2008). In sections 3.2 and 3.3, I have included indicators to assess the impact of forestry on these biophysical effects. However, this did not enable concrete assertions on surface temperature, because LPJ-GUESS was not coupled to a climate model. Luyssaert et al. (2018) have already optimized European forest management strategies to minimize surface air temperature using the coupled model ORCHIDEE-CAN. However, they did not consider the uncertainty of multiple climate scenarios and the multi-functionality of forests, but optimized the forest management portfolios for single objectives and Representative Concentration Pathways (RCPs). In my view, the crucial next step is to apply the methodologies proposed in this thesis to coupled climate models, using surface air temperature as one key variable in the objective function of the optimization.

4.3.2 Further enhancement of the robust optimization methodology to enable concrete forestry and land-use strategy development

Beyond the outlined model improvements and efforts to extend it from forestry to land use, the strategy development requires further enhancement. A first step involves including more climate models, because different models result in distinct trajectories for the same RCP (Ahlström et al., 2012; Warszawski et al., 2014). These climate model inputs could be integrated into the uncertainty set of the robust optimization, or individual portfolios could be created for each model and then combined, resulting in management portfolios with associated uncertainties. For the same reasons, multiple dynamic global vegetation models (DGVMs) should be used to simulate the impacts of climate change scenarios on vegetation to account for strengths and weaknesses of various models (e.g., Tschumi et al., 2023).

A critical limitation of the analyses in this thesis is the lack of spatial explicitness of the portfolios within grid cells. The portfolio optimization was done at a $0.5°\times0.5°$ resolution, but a much finer scale is necessary to allocate portfolios to concrete locations. Notably, LPJ-GUESS has been successfully applied at high resolutions, including 5 km, 2 km, and 1 km (Gregor et al., 2024; Krause et al., 2020; Su et al., 2022). The model's predefined patch size of 0.1 ha is a lower bound on the resolution. However, running it at 1 km resolution should suffice, especially given the lack of reliable climate projections at finer scales. Nevertheless, 86% of privately owned forests in Europe are 5 ha or smaller in size (Hirsch & Schmithüsen, 2010). Consequently, LPJ-GUESS can be used to derive recommendations for portfolios at local scales of a few kilometers, and finer models can be applied to allocate the portfolio to the specific locations within that area. This allocation process should also consider constraints on landscape fragmentation to enhance landscape diversity and support ecological corridors, in line with the *EU Biodiversity Strategy for 2030*. Simultaneously, factors such as accessibility and harvest adjacency could be included to enable cost-effective management operations while also avoiding large openings in the landscape.

Moreover, the timing of management changes should be addressed. Here, I considered abrupt changes in management, but strategies that follow a gradual conversion have been shown to be beneficial for forest adaptation (BMEL, 2014). In that regard, adaptive and iterative strategies that can react to changing conditions are crucial for forestry under climate change (Jandl et al., 2019) and should be incorporated into the portfolio development. For instance, management options that contain different rates of forest conversion could be simulated, as well as multiple simulations for the same forest type, but with different (and potentially temperature- and moisture-dependent) timings of thinnings and plantings.

5 Conclusion

The land sector plays a key role in combating climate change and global biodiversity loss. It provides a small, but substantial, potential for climate change mitigation, especially while global greenhouse gas emissions are still high. This potential can be further enhanced by informed mitigation strategies that are robust to the large uncertainties of the future. Furthermore, a rapid implementation of such measures is crucial to profit from high substitution effects when large amounts of emissions can still be avoided through the use of wood products.

However, it is imperative to broaden the focus away from climate change mitigation. Land use strategies must also encompass their impact on biodiversity and the wide array of ecosystem services that they may provide. In addition, terrestrial ecosystems – especially forests – are themselves vulnerable to current and future climate change and need to be adapted to these changing conditions. Without this adaptation, mitigation of climate change or sustained provision of ecosystem services is impossible.

To this end, this thesis shows how management portfolios can be developed to simultaneously provide mitigation, adaptation, and ecosystem services under uncertain future climate. These portfolios offer risk-diversified strategies that acknowledge future uncertainty, providing a framework for action today.

Nevertheless, my work also highlights that the vast amount of demands on forests and land pose a substantial threat to these vulnerable ecosystems. It is crucial to alleviate this pressure by addressing the various (often conflicting) demands, for example by accelerating investments into renewable energies like photovoltaic systems, to reduce the demand for bioenergy. Generally, a change of perspective is necessary, going from viewing forests and the land sector as a mere tool for climate change mitigation towards a holistic approach including also adaptation and ecosystem services.

Bibliography

Ahlström, A., Schurgers, G., Arneth, A., & Smith, B. (2012). Robustness and uncertainty in terrestrial ecosystem carbon response to CMIP5 climate change projections. *Environmental Research Letters*, *7*(4), 044008. https://doi.org/10.1088/1748-9326/7/4/044008

Albert, K., & Ammer, C. (2012). Biomasseproduktivität ausgewählter europäischer Mittel- und Niederwaldbestände - Ergebnisse einer vergleichenden Metaanalyse. *Allgemeine Forst- und Jagdzeitung*, *183*, 225–237.

Alexander, P., Brown, C., Arneth, A., Dias, C., Finnigan, J., Moran, D., & Rounsevell, M. D. (2017). Could consumption of insects, cultured meat or imitation meat reduce global agricultural land use? *Global Food Security*, *15*(March), 22–32. https://doi.org/10.1016/j.gfs.2017.04.001

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D. D., Hogg, E. H., 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

Anderson, K., & Peters, G. (2016). The trouble with negative emissions. *Science*, *354*(6309), 182–183. https://doi.org/10.1126/science.aah4567

Araya, M. M., & Hofstad, O. (2016). Monetary incentives to avoid deforestation under the Reducing emissions from deforestation and degradation (REDD)+ climate change mitigation scheme in Tanzania. *Mitigation and Adaptation Strategies for Global Change*, *21*(3), 421–443. https://doi.org/10.1007/s11027-014-9607-y

Armstrong McKay, D. I., Staal, A., Abrams, J. F., Winkelmann, R., Sakschewski, B., Loriani, S., Fetzer, I., Cornell, S. E., Rockström, J., & Lenton, T. M. (2022). Exceeding 1.5°C global warming could trigger multiple climate tipping points. *Science*, *377*(6611). https://doi.org/10.1126/science.abn7950

Asayama, S. (2021). The Oxymoron of Carbon Dioxide Removal: Escaping Carbon Lock-In and yet Perpetuating the Fossil Status Quo? *Frontiers in Climate*, *3*(July), 1–8. https://doi.org/10.3389/fclim.2021.673515

Astrup, R., Bernier, P. Y., Genet, H., Lutz, D. A., & Bright, R. M. (2018). A sensible climate solution for the boreal forest. *Nature Climate Change*, *8*(1), 11–12. https://doi.org/10.1038/s41558-017-0043-3

Bačėninaitė, D., Džermeikaitė, K., & Antanaitis, R. (2022). Global Warming and Dairy Cattle: How to Control and Reduce Methane Emission. *Animals*, *12*(19), 2687. https://doi.org/10.3390/ani12192687

Bagdon, B. A., Huang, C. H., & Dewhurst, S. (2016). Managing for ecosystem services in northern Arizona ponderosa pine forests using a novel simulation-to-optimization methodology. *Ecological Modelling*, *324*, 11–27. https://doi.org/10.1016/j.ecolmodel.2015.12.012

Baldocchi, D., & Penuelas, J. (2019). The physics and ecology of mining carbon dioxide from the atmosphere by ecosystems. *Global Change Biology*, *25*(4), 1191–1197. https://doi.org/10.1111/gcb.14559

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R. E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., & Turner, R. K. (2002). Economic Reasons for Conserving Wild Nature. *Science*, *297*(5583), 950–953. https://doi.org/10.1126/science.1073947

Banger, K., Wagner-Riddle, C., Grant, B. B., Smith, W. N., Drury, C., & Yang, J. (2020). Modifying fertilizer rate and application method reduces environmental nitrogen losses and increases corn yield in Ontario. *Science of The Total Environment*, *722*, 137851. https://doi.org/10.1016/j.scitotenv.2020.137851

Bartels, R. H. (1971). A stabilization of the simplex method. *Numerische Mathematik*, *16*(5), 414–434. https://doi.org/10.1007/BF02169151

Başkent, E. Z., & Kašpar, J. (2023). Exploring the effects of various rotation lengths on the ecosystem services within a multiple-use management framework. *Forest Ecology and Management*, *538*(April). https://doi.org/10.1016/j.foreco.2023.120974

Bastin, J. F., Finegold, Y., Garcia, C., Mollicone, D., Rezende, M., Routh, D., Zohner, C. M., & Crowther, T. W. (2019). The global tree restoration potential. *Science*, *364*(6448), 76–79. https://doi.org/10.1126/science.aax0848

Bastos, A., Ciais, P., Friedlingstein, P., Sitch, S., Pongratz, J., Fan, L., Wigneron, J. P., Weber, U., Reichstein, M., Fu, Z., Anthoni, P., Arneth, A., Haverd, V., Jain, A. K., Joetzjer, E., Knauer, J., Lienert, S., Loughran, T., McGuire, P. C., ... Zaehle, S. (2020). Direct and seasonal legacy effects of the 2018 heat wave and drought on European ecosystem productivity. *Science Advances*, *6*(24), eaba2724. https://doi.org/10.1126/sciadv.aba2724

Bayer, A. D., Lautenbach, S., & Arneth, A. (2023). Benefits and trade-offs of optimizing global land use for food, water, and carbon. *Proceedings of the National Academy of Sciences*, *120*(42), 1–8. https://doi.org/10.1073/pnas.2220371120

Bayerisches Landesamt für Umwelt [Hrsg.] (2020). Bayerische Klimadaten - Beobachtungsdaten, Klima- projektionsensemble und Klimakennwerte für Bayern.

Bellassen, V., Viovy, N., Luyssaert, S., Maire, G., Schelhaas, M.-J., & Ciais, P. (2011). Reconstruction and attribution of the carbon sink of European forests between 1950 and 2000. *Global Change Biology*, *17*(11), 3274–3292. https://doi.org/10.1111/j.1365-2486.2011.02476.x

Bengtsson, J., Bullock, J. M., Egoh, B., Everson, C., Everson, T., O'Connor, T., O'Farrell, P. J., Smith, H. G., & Lindborg, R. (2019). Grasslands-more important for ecosystem services than you might think. *Ecosphere*, *10*(2), e02582. https://doi.org/10.1002/ecs2.2582

Ben-Tal, A., & Nemirovski, A. (2002). Robust optimization - methodology and applications. *Mathematical Programming*, *92*(3), 453–480. https://doi.org/10.1007/s101070100286

Benton, T. G., Vickery, J. A., & Wilson, J. D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, *18*(4), 182–188. https://doi.org/10.1016/S0169-5347(03)00011-9

Berlik, M. M., Kittredge, D. B., & Foster, D. R. (2002). The illusion of preservation: a global environmental argument for the local production of natural resources. *Journal of Biogeography*, *29*(10-11), 1557–1568. https://doi.org/10.1046/j.1365-2699.2002.00768.x

Betts, M. G., Phalan, B. T., Wolf, C., Baker, S. C., Messier, C., Puettmann, K. J., Green, R., Harris, S. H., Edwards, D. P., Lindenmayer, D. B., & Balmford, A. (2021). Producing wood at least cost to biodiversity: integrating Triad and sharing–sparing approaches to inform forest landscape management. *Biological Reviews*, *96*(4), 1301–1317. https://doi.org/10.1111/brv.12703

Betts, R. A. (2000). Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. *Nature*, *408*(6809), 187–190. https://doi.org/10.1038/35041545

Biber, P., Felton, A., Nieuwenhuis, M., Lindbladh, M., Black, K., Bahýl', J., Bingöl, Ö., Borges, J. G., Botequim, B., Brukas, V., Bugalho, M. N., Corradini, G., Eriksson, L. O., Forsell, N., Hengeveld, G. M., Hoogstra-Klein, M. A., Kadıoğulları, A. I., Karahalil, U., Lodin, I., ... Tuček, J. (2020). Forest Biodiversity, Carbon Sequestration, and Wood Production: Modeling Synergies and Trade-Offs for Ten Forest Landscapes Across Europe. *Frontiers in Ecology and Evolution*, *8*(October). https://doi.org/10.3389/fevo.2020.547696

Bierwirth, A., & Thomas, S. (2015). Almost best friends : sufficiency and efficiency ; can sufficiency maximise efficiency gains in buildings? *ECEEE 2015 - European Council for an Energy Efficient Economy*, 71–82.

Binder, S., Haight, R. G., Polasky, S., Warziniack, T., Mockrin, M. H., Deal, R. L., & Arthaud, G. (2017). Assessment and valuation of forest ecosystem services: state of the science review.

Bindi, M., & Olesen, J. E. (2011). The responses of agriculture in Europe to climate change. *Regional Environmental Change*, *11*(S1), 151–158. https://doi.org/10.1007/s10113-010-0173-x

BMEL. (2014). The forest in Germany. Selected results of the third National Forest Inventory [Der Wald in Deutschland. Ausgewählte Ergebnisse der dritten Bundeswaldinventur.] *3. Ed.* 56.

Boisier, J. P., de Noblet-Ducoudré, N., & Ciais, P. (2013). Inferring past land use-induced changes in surface albedo from satellite observations: a useful tool to evaluate model simulations. *Biogeosciences*, *10*(3), 1501–1516. https://doi.org/10.5194/bg-10-1501-2013

Bonan, G. (2008). Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. *Science*, *320*(5882), 1444–1449. https://doi.org/10.1126/science.1155121

Bonan, G. (2016). *Ecological Climatology* (Vol. 8). Cambridge University Press. https://doi.org/10.1017/CBO9781107339200

Bonannella, C., Hengl, T., Parente, L., & de Bruin, S. (2023). Biomes of the world under climate change scenarios: increasing aridity and higher temperatures lead to significant shifts in natural vegetation. *PeerJ*, *11*, e15593. https://doi.org/10.7717/peerj.15593

Bond, W. J. (2016). Ancient grasslands at risk. *Science*, *351*(6269), 120–122. https://doi.org/10.1126/science.aad5132

Bond, W. J., Stevens, N., Midgley, G. F., & Lehmann, C. E. (2019). The Trouble with Trees: Afforestation Plans for Africa. *Trends in Ecology & Evolution*, *34*(11), 963–965. https://doi.org/10.1016/j.tree.2019.08.003

Bondeau, A., Smith, P. C., Zaehle, S., Schaphoff, S., Lucht, W., Cramer, W., Gerten, D., Lotze-campen, H., Müller, C., Reichstein, M., & Smith, B. (2007). Modelling the role of agriculture for the 20th century global terrestrial carbon balance. *Global Change Biology*, *13*(3), 679–706. https://doi.org/10.1111/j.1365-2486.2006.01305.x

Booth, M. S., Mackey, B., & Young, V. (2020). It's time to stop pretending burning forest biomass is carbon neutral. *GCB Bioenergy*, *12*(12), 1036–1037. https://doi.org/10.1111/gcbb.12716

Brandon, K. (2014). Ecosystem Services from Tropical Forests: Review of Current Science. *SSRN Electronic Journal*, (380), 85. https://doi.org/10.2139/ssrn.2622749

Bredemeier, M. (2011). Forest, climate and water issues in Europe. *Ecohydrology*, *4*(2), 159–167. https://doi.org/10.1002/eco.203

Brienen, R. J., Caldwell, L., Duchesne, L., Voelker, S., Barichivich, J., Baliva, M., Ceccantini, G., Di Filippo, A., Helama, S., Locosselli, G. M., Lopez, L., Piovesan, G., Schöngart, J., Villalba, R., & Gloor, E. (2020). Forest carbon sink neutralized by pervasive growth-lifespan trade-offs. *Nature Communications*, *11*(1), 1–10. https://doi.org/10.1038/s41467-020-17966-z

Brienen, R. J., Phillips, O. L., Feldpausch, T. R., Gloor, E., Baker, T. R., Lloyd, J., Lopez-Gonzalez, G., Monteagudo-Mendoza, A., Malhi, Y., Lewis, S. L., Vásquez Martinez, R., Alexiades, M., Álvarez Dávila, E., Alvarez-Loayza, P., Andrade, A., Aragaõ, L. E., Araujo-Murakami, A., Arets, E. J., Arroyo, L., … Zagt, R. J. (2015). Long-term decline of the Amazon carbon sink. *Nature*, *519*(7543), 344–348. https://doi.org/10.1038/nature14283

Brockerhoff, E. G., Barbaro, L., Castagneyrol, B., Forrester, D. I., Gardiner, B., González-Olabarria, J. R., Lyver, P. O., Meurisse, N., Oxbrough, A., Taki, H., Thompson, I. D., van der Plas, F., & Jactel, H. (2017). Forest biodiversity, ecosystem functioning and the provision of ecosystem services. *Biodiversity and Conservation*, *26*(13), 3005–3035. https://doi.org/10.1007/s10531-017-1453-2

Brunet-Navarro, P., Jochheim, H., Cardellini, G., Richter, K., & Muys, B. (2021). Climate mitigation by energy and material substitution of wood products has an expiry date. *Journal of Cleaner Production*, *303*, 127026. https://doi.org/10.1016/j.jclepro.2021.127026

Brus, D. J., Hengeveld, G. M., Walvoort, D. J., Goedhart, P. W., Heidema, A. H., Nabuurs, G. J., & Gunia, K. (2012). Statistical mapping of tree species over Europe. *European Journal of Forest Research*, *131*(1), 145–157. https://doi.org/10.1007/s10342-011-0513-5

Bruzzese, S., Blanc, S., Paletto, A., & Brun, F. (2023). A systematic review of markets for forest ecosystem services at an international level. *Canadian Journal of Forest Research*, *53*(7), 463–477. https://doi.org/10.1139/cjfr-2022-0230

Bryngelsson, D., Wirsenius, S., Hedenus, F., & Sonesson, U. (2016). How can the EU climate targets be met? A combined analysis of technological and demand-side changes in food and agriculture. *Food Policy*, *59*, 152–164. https://doi.org/10.1016/j.foodpol.2015.12.012

Bugmann, H., & Bigler, C. (2011). Will the CO2 fertilization effect in forests be offset by reduced tree longevity? *Oecologia*, *165*(2), 533–544. https://doi.org/10.1007/s00442-010-1837-4

Bugmann, H., Cordonnier, T., Truhetz, H., & Lexer, M. J. (2017). Impacts of business-as-usual management on ecosystem services in European mountain ranges under climate change. *Regional Environmental Change*, *17*(1), 3–16. https://doi.org/10.1007/s10113-016-1074-4

Buras, A., & Menzel, A. (2019). Projecting tree species composition changes of european forests for 2061–2090 under RCP 4.5 and RCP 8.5 scenarios. *Frontiers in Plant Science*, *9*(January), 1–13. https://doi.org/10.3389/fpls.2018.01986

Buras, A., Rammig, A., & Zang, C. S. (2020). Quantifying impacts of the 2018 drought on European ecosystems in comparison to 2003. *Biogeosciences*, *17*(6), 1655–1672. https://doi.org/10.5194/bg-17-1655-2020

Buras, A., Rammig, A., & Zang, C. S. (2021). The European Forest Condition Monitor: Using Remotely Sensed Forest Greenness to Identify Hot Spots of Forest Decline. *Frontiers in Plant Science*, *12*(December), 1–19. https://doi.org/10.3389/fpls.2021.689220

Buras, A., Schunk, C., Zeitrg, C., Herrmann, C., Kaiser, L., Lemme, H., Straub, C., Taeger, S., Gößwein, S., Klemmt, H. J., & Menzel, A. (2018). Are Scots pine forest edges particularly prone to drought-induced mortality? *Environmental Research Letters*, *13*(2). https://doi.org/10.1088/1748-9326/aaa0b4

Büttner, G. (2014). CORINE Land Cover and Land Cover Change Products. In I. Manakos & M. Braun (Eds.), *Land use and land cover mapping in europe: Practices & trends* (pp. 55–74). Springer Netherlands. https://doi.org/10.1007/978-94-007-7969-3_5

Camargo-Alvarez, H., Elliott, R. J. R., Olin, S., Wang, X., Wang, C., Ray, D. K., & Pugh, T. A. M. (2023). Modelling crop yield and harvest index: the role of carbon assimilation and allocation parameters. *Modeling Earth Systems and Environment*, *9*(2), 2617–2635. https://doi.org/10.1007/s40808-022-01625-x

Cao, S. (2008). Why Large-Scale Afforestation Efforts in China Have Failed To Solve the Desertification Problem. *Environmental Science & Technology*, *42*(6), 1826–1831. https://doi.org/10.1021/es0870597

Carter, J. G. (2011). Climate change adaptation in European cities. *Current Opinion in Environmental Sustainability*, *3*(3), 193–198. https://doi.org/10.1016/j.cosust.2010.12.015

Ceballos, G., Ehrlich, P. R., Barnosky, A. D., García, A., Pringle, R. M., & Palmer, T. M. (2015). Accelerated modern human–induced species losses: Entering the sixth mass extinction. *Science Advances*, *1*(5), 9–13. https://doi.org/10.1126/sciadv.1400253

Cerullo, G., Barlow, J., Betts, M., Edwards, D., Eyres, A., França, F., Garrett, R., Swinfield, T., Tew, E., White, T., & Balmford, A. (2023). The global impact of EU forest protection policies. *Science*, *381*(6659), 740–740. https://doi.org/10.1126/science.adj0728

Chang, J., Ciais, P., Wang, X., Piao, S., Asrar, G., Betts, R., Chevallier, F., Dury, M., François, L., Frieler, K., Ros, A. G. C., Henrot, A.-J., Hickler, T., Ito, A., Morfopoulos, C., Munhoven, G., Nishina, K., Ostberg, S., Pan, S., … Zhao, F. (2017). Benchmarking carbon fluxes of the ISIMIP2a biome models. *Environmental Research Letters*, *12*(4), 045002. https://doi.org/10.1088/1748-9326/aa63fa

Cherubini, F., Peters, G. P., Berntsen, T., Strømman, A. H., & Hertwich, E. (2011). CO2 emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming. *GCB Bioenergy*, *3*(5), 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x

Churkina, G., Organschi, A., Reyer, C. P., Ruff, A., Vinke, K., Liu, Z., Reck, B. K., Graedel, T. E., & Schellnhuber, H. J. (2020). Buildings as a global carbon sink. *Nature Sustainability*, *3*(4), 269–276. https://doi.org/10.1038/s41893-019-0462-4

Ciais, P., Reichstein, M., Viovy, N., Granier, A., Ogée, J., Allard, V., Aubinet, M., Buchmann, N., Bernhofer, C., Carrara, A., Chevallier, F., De Noblet, N., Friend, A. D., Friedlingstein, P., Grünwald, T., Heinesch, B., Keronen, P., Knohl, A., Krinner, G., … Valentini, R. (2005). Europe-wide reduction in primary productivity caused by the heat and drought in 2003. *Nature*, *437*(7058), 529–533. https://doi.org/10.1038/nature03972

Collatz, G., Ball, J., Grivet, C., & Berry, J. A. (1991). Physiological and environmental regulation of stomatal conductance, photosynthesis and transpiration: a model that includes a laminar boundary layer. *Agricultural and Forest Meteorology*, *54*(2-4), 107–136. https://doi.org/10.1016/0168-1923(91)90002-8

Cooper, L., & MacFarlane, D. (2023). Climate-Smart Forestry: Promise and risks for forests, society, and climate (K. Byrne, Ed.). *PLOS Climate*, *2*(6), e0000212. https://doi.org/10.1371/journal.pclm.0000212

Cordonnier, T., Berger, F., Elkin, C., Lämås, T., Martinez, M., Bugmann, H., Courbaud, B., Gosselin, F., & Volk, G. (2014). ARANGE D2.2 - Models and linker functions (indicators) for ecosystem services. (289437).

Dai, A. (2013). Increasing drought under global warming in observations and models. *Nature Climate Change*, *3*(1), 52–58. https://doi.org/10.1038/nclimate1633

Dantzig, G. B. (1951). Maximization of a linear function of variables subject to linear inequalities. *Activity analysis of production and allocation*, *13*, 339–347.

Davin, E. L., & de Noblet-Ducoudre, N. (2010). Climatic impact of global-scale Deforestation: Radiative versus nonradiative processes. *Journal of Climate*, *23*(1), 97–112. https://doi.org/10.1175/2009JCLI3102.1

De Frenne, P., Lenoir, J., Luoto, M., Scheffers, B. R., Zellweger, F., Aalto, J., Ashcroft, M. B., Christiansen, D. M., Decocq, G., De Pauw, K., Govaert, S., Greiser, C., Gril, E., Hampe, A., Jucker, T., Klinges, D. H., Koelemeijer, I. A., Lembrechts, J. J., Marrec, R., ... Hylander, K. (2021). Forest microclimates and climate change: Importance, drivers and future research agenda. *Global Change Biology*, *27*(11), 2279–2297. https://doi.org/10.1111/gcb.15569

DeFries, R., Pagiola, S., Adamowicz, W. L., Akcakaya, H. R., Arcenas, A., Babu, S., Balk, D., Cramer, W., Falconı, F., Fritz, S., Green, R., Gutie, E., Hamilton, K., Kane, R., Latham, J., Matthews, E., Ricketts, T., Yue, T. X., Ash, N., ... Supriatna, J. (2005). Analytical Approaches for Assessing Ecosystem Condition and Human Well-being. *Main*, *55*(7), 36–71. http://books.google.com/books?hl=en&lr=&id=UFVmiSAr-okC&oi=fnd&pg=PA37&dq=Analytical+Approaches+for+Assessing+Ecosystem+Condition+and+Human+Well-being&ots=l7CwPXWZSg&sig=J3EauVvpPXEiCyHwY_qbk44xFJ4

de Groot, M., Diaci, J., & Ogris, N. (2019). Forest management history is an important factor in bark beetle outbreaks: Lessons for the future. *Forest Ecology and Management*, *433*(November 2018), 467–474. https://doi.org/10.1016/j.foreco.2018.11.025

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L. C., ten Brink, P., & van Beukering, P. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services*, *1*(1), 50–61. https://doi.org/10.1016/j.ecoser.2012.07.005

Diamond, S. E. (2018). Contemporary climate-driven range shifts: Putting evolution back on the table (C. Fox, Ed.). *Functional Ecology*, *32*(7), 1652–1665. https://doi.org/10.1111/1365-2435.13095

Díaz, S., Fargione, J., Chapin, F. S., & Tilman, D. (2006). Biodiversity loss threatens human well-being. *PLoS Biology*, *4*(8), 1300–1305. https://doi.org/10.1371/journal.pbio.0040277

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., Hill, R., Chan, K. M. A., Baste, I. A., Brauman, K. A., Polasky, S., Church, A., Lonsdale, M., Larigauderie, A., Leadley, P. W., van Oudenhoven, A. P. E., van der Plaat, F., Schröter, M., Lavorel, S., ... Shirayama, Y. (2018). Assessing nature's contributions to people. *Science*, *359*(6373), 270–272. https://doi.org/10.1126/science.aap8826

Diaz-Balteiro, L., Alonso, R., Martínez-Jaúregui, M., & Pardos, M. (2017). Selecting the best forest management alternative by aggregating ecosystem services indicators over time: A case study in central Spain. *Ecological Indicators*, *72*, 322–329. https://doi.org/10.1016/j.ecolind.2016.06.025

Diaz-Balteiro, L., Belavenutti, P., Ezquerro, M., González-Pachón, J., Ribeiro Nobre, S., & Romero, C. (2018). Measuring the sustainability of a natural system by using multi-criteria distance function methods: Some critical issues. *Journal of Environmental Management*, *214*, 197–203. https://doi.org/10.1016/j.jenvman.2018.03.005

dos Santos, A. M., da Silva, C. F. A., Monteiro de Almeida Junior, P., Rudke, A. P., & de Melo, S. N. (2021). Deforestation drivers in the Brazilian Amazon: assessing new spatial predictors. *Journal of Environmental Management*, *294*(February), 113020. https://doi.org/10.1016/j.jenvman.2021.113020

Duffy, K. A., Schwalm, C. R., Arcus, V. L., Koch, G. W., Liang, L. L., & Schipper, L. A. (2021). How close are we to the temperature tipping point of the terrestrial biosphere? *Science Advances*, *7*(3), 1–9. https://doi.org/10.1126/sciadv.aay1052

Dufresne, J.-L., Foujols, M.-A., Denvil, S., Caubel, A., Marti, O., Aumont, O., Balkanski, Y., Bekki, S., Bellenger, H., Benshila, R., Bony, S., Bopp, L., Braconnot, P., Brockmann, P., Cadule, P., Cheruy, F., Codron, F., Cozic, A., Cugnet, D., ... Vuichard, N. (2013). Climate change projections using the IPSL-CM5 Earth System Model: from CMIP3 to CMIP5. *Climate Dynamics*, *40*(9-10), 2123–2165. https://doi.org/10.1007/s00382-012-1636-1

Dugan, A. J., Birdsey, R., Mascorro, V. S., Magnan, M., Smyth, C. E., Olguin, M., & Kurz, W. A. (2018). A systems approach to assess climate change mitigation options in landscapes of the United States forest sector. *Carbon Balance and Management*, *13*(1), 13. https://doi.org/10.1186/s13021-018-0100-x

Dunjó, G., Pardini, G., & Gispert, M. (2004). The role of land use–land cover on runoff generation and sediment yield at a microplot scale, in a small Mediterranean catchment. *Journal of Arid Environments*, *57*(2), 239–256. https://doi.org/10.1016/S0140-1963(03)00097-1

EEA. (2015). *The State of Nature in the European Union - Results from reporting under the nature directives 2007–2012* (tech. rep. No. 2). https://doi.org/10.2800/603862

Ellenberg, H. (1986). Veränderungen von Artenspektren unter dem Einflußvon düngenden Immissionen und ihre Folgen. *Allg. Forstz., München*, *41*, 466–467.

Ellis, E. C., Gauthier, N., Klein Goldewijk, K., Bliege Bird, R., Boivin, N., Díaz, S., Fuller, D. Q., Gill, J. L., Kaplan, J. O., Kingston, N., Locke, H., McMichael, C. N. H., Ranco, D., Rick, T. C., Shaw, M. R., Stephens, L., Svenning, J.-C., & Watson, J. E. M. (2021). People have shaped most of terrestrial nature for at least 12,000 years. *Proceedings of the National Academy of Sciences*, *118*(17), e2023483118. https://doi.org/10.1073/pnas.2023483118

Eskander, S. M., & Fankhauser, S. (2020). Reduction in greenhouse gas emissions from national climate legislation. *Nature Climate Change*, *10*(8), 750–756. https://doi.org/10.1038/s41558-020-0831-z

European Commission. (2019). What is the European Green Deal. Retrieved April 29, 2022, from https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal/actions-being-taken-eu_en

European Commission. (2020). EU biodiversity strategy for 2030: Bringing nature back into our lives. *Communication for the Commission to the European Parliament,* *the Council, the European Economic and Social Committee and the Committee of the regions*. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A52020DC0380

European Commission. (2021a). *EU Soil Strategy for 2030 Reaping the benefits of healthy soils for people, food, nature and climate* (tech. rep.). Brussels. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52021DC0699

European Commission. (2021b). New EU Forest Strategy for 2030. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52021DC0572

European Environment Agency. (2019). *The European environment — state and outlook 2020. Knowledge for transition to a sustainable Europe*. https://doi.org/10.2800/96749

Eurostat. (2021a). Eurostat Forestry Database. Retrieved June 29, 2021, from https://ec.europa.eu/eurostat/web/forestry/data/database

Eurostat. (2021b). Land use statistics. Retrieved July 13, 2023, from https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Land_use_statistics#Land_use

Eurostat. (2022). Roundwood removals by type of wood and assortment (FOR_REMOV). Retrieved November 14, 2022, from https://ec.europa.eu/eurostat/databrowser/

Eurostat. (2023). Landfill rate of waste excluding major mineral wastes. Retrieved October 18, 2023, from https://ec.europa.eu/eurostat/databrowser/view/env_wasmun

Evans, J. (1984). Silviculture of broadleaved woodland (UK). *Forestry Commission Bulletin*, *62*.

Fajardy, M., Koberle, A., Mac Dowell, N., & Fantuzzi, A. (2019). BECCS deployment: a reality check. *Grantham Institute*, *28*(28), 1–14.

FAO. (1974). *Soil Map of the World: 1:5,000,000* (tech. rep.). UNESCO.

FAO. (2020). Food and Agriculture Organization of the United Nations: Global Forest Resources Assessment 2020: Terms and Definition FRA. http://www.fao.org/forestry/58864/en/

FAO. (2022a). FAOSTAT Database. Forestry Production and Trade. Retrieved January 14, 2022, from https://www.fao.org/faostat/en/#data/FO

FAO. (2022b). *Land use statistics and indicators. Global, regional and country trends – 2000-2020* (tech. rep.). Rome. https://doi.org/10.4060/cc0963en

FAO. (2022c). *The State of the World's Forests 2022. Forest pathways for green recovery and building inclusive, resilient and sustainable economies*. https://doi.org/10.4060/cb9360en

Farquhar, G. D., von Caemmerer, S., & Berry, J. A. (1980). A biochemical model of photosynthetic CO2 assimilation in leaves of C3 species. *Planta*, *149*(1), 78–90. https://doi.org/10.1007/BF00386231

Fedele, G., Donatti, C. I., Bornacelly, I., & Hole, D. G. (2021). Nature-dependent people: Mapping human direct use of nature for basic needs across the tropics. *Global Environmental Change*, *71*, 102368. https://doi.org/10.1016/j.gloenvcha.2021.102368

Fekete, B. M., Vörösmarty, C. J., & Grabs, W. (1999). *Global, composite runoff fields based on observed river discharge and simulated water balances*. Global Runoff Data Centre Koblenz.

Felton, A., Gustafsson, L., Roberge, J. M., Ranius, T., Hjältén, J., Rudolphi, J., Lindbladh, M., Weslien, J., Rist, L., Brunet, J., & Felton, A. M. (2016). How climate change adaptation and mitigation strategies can threaten or enhance the biodiversity of production forests: Insights from Sweden. *Biological Conservation*, *194*, 11–20. https://doi.org/10.1016/j.biocon.2015.11.030

Felton, A., Lindbladh, M., Brunet, J., & Fritz, Ö. (2010). Replacing coniferous monocultures with mixed-species production stands: An assessment of the potential benefits for forest biodiversity in northern Europe. *Forest Ecology and Management*, *260*(6), 939–947. https://doi.org/10.1016/j.foreco.2010.06.011

Fidej, G., Rozman, A., & Diaci, J. (2018). Drivers of regeneration dynamics following salvage logging and different silvicultural treatments in windthrow areas in Slovenia. *Forest Ecology and Management*, *409*(November 2017), 378–389. https://doi.org/10.1016/j.foreco.2017.11.046

Forest Europe. (2015). State of Europe's Forests 2015.

Forest Europe. (2020). State of Europe's Forests 2020.

Forrest, M., Tost, H., Lelieveld, J., & Hickler, T. (2020). Including vegetation dynamics in an atmospheric chemistry-enabled general circulation model: linking LPJ-GUESS (v4.0) with the EMAC modelling system (v2.53). *Geoscientific Model Development*, *13*(3), 1285–1309. https://doi.org/10.5194/gmd-13-1285-2020

Friedlingstein, P. (2019). Comment on "The global tree restoration potential". *Science*, *9*(3), 174–182. https://doi.org/10.1890/090179

Friedlingstein, P., O'Sullivan, M., Jones, M. W., Andrew, R. M., Bakker, D. C. E., Hauck, J., Landschützer, P., Le Quéré, C., Luijkx, I. T., Peters, G. P., Peters, W., Pongratz, J., Schwingshackl, C., Sitch, S., Canadell, J. G., Ciais, P., Jackson, R. B., Alin, S. R., Anthoni, P., … Zheng, B. (2023). Global Carbon Budget 2023. *Earth System Science Data*, *15*(12), 5301–5369. https://doi.org/10.5194/essd-15-5301-2023

Fuchs, R., Brown, C., & Rounsevell, M. (2020). Europe's Green Deal offshores environmental damage to other nations. *Nature*, *586*(7831), 671–673. https://doi.org/10.1038/d41586-020-02991-1

Gerten, D., Schaphoff, S., Haberlandt, U., Lucht, W., & Sitch, S. (2004). Terrestrial vegetation and water balance - Hydrological evaluation of a dynamic global vegetation model. *Journal of Hydrology*, *286*(1-4), 249–270. https://doi.org/10.1016/j.jhydrol.2003.09.029

Gidden, M. J., Riahi, K., Smith, S. J., Fujimori, S., Luderer, G., Kriegler, E., van Vuuren, D. P., van den Berg, M., Feng, L., Klein, D., Calvin, K., Doelman, J. C., Frank, S., Fricko, O., Harmsen, M., Hasegawa, T., Havlik, P., Hilaire, J., Hoesly, R., … Takahashi, K. (2019). Global emissions pathways under different socioeconomic scenarios for use in CMIP6: a dataset of harmonized emissions trajectories through the end of the century. *Geoscientific Model Development*, *12*(4), 1443–1475. https://doi.org/10.5194/gmd-12-1443-2019

Giongo, M., Santos, M. M., da Silva, D. B., Cachoeira, J. N., & Santopuoli, G. (2022). Climate-Smart Forestry in Brazil. In R. Tognetti, M. Smith, & P. Panzacchi (Eds.), *Climate-smart forestry in mountain regions* (pp. 545–570). https://doi.org/10.1007/978-3-030-80767-2_17

Gómez-González, S., Ochoa-Hueso, R., & Pausas, J. G. (2020). Afforestation falls short as a biodiversity strategy (J. Sills, Ed.). *Science*, *368*(6498), 1439–1439. https://doi.org/10.1126/science.abd3064

Gorissen, B. L., Yanikoğlu, I., & den Hertog, D. (2015). A practical guide to robust optimization. *Omega (United Kingdom)*, *53*, 124–137. https://doi.org/10.1016/j.omega.2014.12.006

Grassi, G., Fiorese, G., Pilli, R., Jonsson, K., Blujdea, V., Korosuo, A., & Vizzarri, M. (2021). Brief on the role of the forest-based bioeconomy in mitigating climate change through carbon storage and material substitution, 1–16. https://publications.jrc.ec.europa.eu/repository/handle/JRC124374

Greenfield, P. (2023). Revealed: more than 90% of rainforest carbon offsets by biggest certifier are worthless, analysis shows. https://www.theguardian.com/environment/2023/jan/18/revealed-forest-carbon-offsets-biggest-provider-worthless-verra-aoe

Greenpeace. (2018). *Wenn Wälder wieder wachsen* (tech. rep.). https://www.greenpeace.de/publikationen/s02061_greenpeace_studie_waldvision.pdf

Gregor, K., Knoke, T., Krause, A., Reyer, C. P. O., Lindeskog, M., Papastefanou, P., Smith, B., Lansø, A.-S., & Rammig, A. (2022). Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate. *Earth's Future*, *10*(9), 1–25. https://doi.org/10.1029/2022EF002796

Gregor, K., Krause, A., Reyer, C. P. O., Knoke, T., Meyer, B. F., Suvanto, S., & Rammig, A. (2024). Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests. *Carbon Balance and Management*, forthcoming.

Gregor, K., Reyer, C. P. O., Nagel, T. A., Mäkelä, A., Krause, A., Knoke, T., & Rammig, A. (draft). Reconciling climate-smart forestry with EU forest, biodiversity and climate legislation.

Griscom, B. W., Adams, J., Ellis, P. W., Houghton, R. A., Lomax, G., Miteva, D. A., Schlesinger, W. H., Shoch, D., Siikamäki, J. V., Smith, P., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R. T., Delgado, C., Elias, P., Gopalakrishna, T., Hamsik, M. R., … Fargione, J. (2017). Natural climate solutions. *Proceedings of the National Academy of Sciences of the United States of America*, *114*(44), 11645–11650. https://doi.org/10.1073/pnas.1710465114

Guerry, A. D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G. C., Griffin, R., Ruckelshaus, M., Bateman, I. J., Duraiappah, A., Elmqvist, T., Feldman, M. W., Folke, C., Hoekstra, J., Kareiva, P. M., Keeler, B. L., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., … Vira, B. (2015). Natural capital and ecosystem services informing decisions: From promise to practice. *Proceedings of the National*

Academy of Sciences of the United States of America, *112*(24), 7348–7355. https://doi.org/10.1073/pnas.1503751112

Gustavsson, L., Nguyen, T., Sathre, R., & Tettey, U. Y. (2021). Climate effects of forestry and substitution of concrete buildings and fossil energy. *Renewable and Sustainable Energy Reviews*, *136*(March 2020), 110435. https://doi.org/10.1016/j.rser.2020.110435

Gutsch, M., Lasch-Born, P., Kollas, C., Suckow, F., & Reyer, C. P. O. (2018). Balancing trade-offs between ecosystem services in Germany's forests under climate change. *Environmental Research Letters*, *13*(4), 045012. https://doi.org/10.1088/1748-9326/aab4e5

Hämäläinen, A., Strengbom, J., & Ranius, T. (2018). Conservation value of low-productivity forests measured as the amount and diversity of dead wood and saproxylic beetles. *Ecological Applications*, *28*(4), 1011–1019. https://doi.org/10.1002/eap.1705

Hamel, P., & Bryant, B. P. (2017). Uncertainty assessment in ecosystem services analyses: Seven challenges and practical responses. *Ecosystem Services*, *24*(December 2016), 1–15. https://doi.org/10.1016/j.ecoser.2016.12.008

Hammond, W. M., Williams, A. P., Abatzoglou, J. T., Adams, H. D., Klein, T., López, R., Sáenz-Romero, C., Hartmann, H., Breshears, D. D., & Allen, C. D. (2022). Global field observations of tree die-off reveal hotter-drought fingerprint for Earth's forests. *Nature Communications*, *13*(1), 1761. https://doi.org/10.1038/s41467-022-29289-2

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, *342*(6160), 850–853. https://doi.org/10.1126/science.1244693

Harding, R., Kuhry, P., Christensen, T. R., Sykes, M. T., Dankers, R., & Van der Linden, S. (2002). Climate feedbacks at the tundra-taiga interface. *Ambio*, *31*(SPEC. ISS. 12), 47–55.

Harja, D., Subekti, R., & Pambudi, S. (2019). Tree functional attributes and ecological database. Retrieved November 16, 2021, from http://db.worldagroforestry.org

Hassan, J., & Eisele, M. (2015). BauBuche - Der nachhaltige Hochleistungswerkstoff. *Bautechnik*, *92*(1), 40–45. https://doi.org/10.1002/bate.201400093

Hausfather, Z., & Peters, G. P. (2020). Emissions – the 'business as usual' story is misleading. *Nature*, *577*(7792), 618–620. https://doi.org/10.1038/d41586-020-00177-3

Haverd, V., Smith, B., Canadell, J. G., Cuntz, M., Mikaloff-Fletcher, S., Farquhar, G., Woodgate, W., Briggs, P. R., & Trudinger, C. M. (2020). Higher than expected CO 2 fertilization inferred from leaf to global observations. *Global Change Biology*, *26*(4), 2390–2402. https://doi.org/10.1111/gcb.14950

Haxeltine, A., & Prentice, I. C. (1996a). A General Model for the Light-Use Efficiency of Primary Production. *Functional Ecology*, *10*(5), 551. https://doi.org/10.2307/2390165

Haxeltine, A., & Prentice, I. C. (1996b). BIOME3: An equilibrium terrestrial biosphere model based on ecophysiological constraints, resource availability, and competition among plant functional types. *Global Biogeochemical Cycles*, *10*(4), 693–709. https://doi.org/10.1029/96GB02344

Hazeleger, W., Wang, X., Severijns, C., Ştefănescu, S., Bintanja, R., Sterl, A., Wyser, K., Semmler, T., Yang, S., van den Hurk, B., van Noije, T., van der Linden, E., & van der Wiel, K. (2012). EC-Earth V2.2: description and validation of a new seamless earth system prediction model. *Climate Dynamics*, *39*(11), 2611–2629. https://doi.org/10.1007/s00382-011-1228-5

Hengeveld, G. M., Nabuurs, G. J., Didion, M., van den Wyngaert, I., Clerkx, A. P., & Schelhaas, M. J. (2012). A forest management map of European forests. *Ecology and Society*, *17*(4). https://doi.org/10.5751/ES-05149-170453

Hengl, T., Walsh, M. G., Sanderman, J., Wheeler, I., Harrison, S. P., & Prentice, I. C. (2018). Global mapping of potential natural vegetation: An assessment of machine learning algorithms for estimating land potential. *PeerJ*, *2018*(8), 1–36. https://doi.org/10.7717/peerj.5457

Hetemäki, L., Kangas, J., & Peltola, H. (Eds.). (2022). *Forest Bioeconomy and Climate Change* (Vol. 42). Springer International Publishing. https://doi.org/10.1007/978-3-030-99206-4

Hickler, T., Prentice, I. C., Smith, B., Sykes, M. T., & Zaehle, S. (2006). Implementing plant hydraulic architecture within the LPJ Dynamic Global Vegetation Model. *Global Ecology and Biogeography*, *15*(6), 567–577. https://doi.org/10.1111/j.1466-8238.2006.00254.x

Hickler, T., Vohland, K., Feehan, J., Miller, P. A., Smith, B., Costa, L., Giesecke, T., Fronzek, S., Carter, T. R., Cramer, W., Kühn, I., & Sykes, M. T. (2012). Projecting the future distribution of European potential natural vegetation zones with a generalized, tree species-based dynamic vegetation model. *Global Ecology and Biogeography*, *21*(1), 50–63. https://doi.org/10.1111/j.1466-8238.2010.00613.x

Hill, C., Kymäläinen, M., & Rautkari, L. (2022). Review of the use of solid wood as an external cladding material in the built environment. *Journal of Materials Science*, *57*(20), 9031–9076. https://doi.org/10.1007/s10853-022-07211-x

Hirsch, F., & Schmithüsen, F. J. (2010). *Private forest ownership in Europe* (tech. rep.). United Nations. https://doi.org/10.3929/ethz-a-006311424

Hof, C., Voskamp, A., Biber, M. F., Böhning-Gaese, K., Engelhardt, E. K., Niamir, A., Willis, S. G., & Hickler, T. (2018). Bioenergy cropland expansion may offset positive effects of climate change mitigation for global vertebrate diversity. *Proceedings of the National Academy of Sciences*, *115*(52), 13294–13299. https://doi.org/10.1073/pnas.1807745115

Holl, K. D., & Brancalion, P. H. (2020). Tree planting is not a simple solution. *Science*, *368*(6491), 580–581. https://doi.org/10.1126/science.aba8232

Holmgren, P. (2020). Climate effects of the forest based sector in the European Union, 1–25.

Holtsmark, B. (2013). The outcome is in the assumptions: Analyzing the effects on atmospheric CO2 levels of increased use of bioenergy from forest biomass. *GCB Bioenergy*, *5*(4), 467–473. https://doi.org/10.1111/gcbb.12015

Howard, C., Dymond, C. C., Griess, V. C., Tolkien-Spurr, D., & van Kooten, G. C. (2021). Wood product carbon substitution benefits: a critical review of assumptions. *Carbon Balance and Management*, *16*(1), 1–11. https://doi.org/10.1186/s13021-021-00171-w

Hua, F., Bruijnzeel, L. A., Meli, P., Martin, P. A., Zhang, J., Nakagawa, S., Miao, X., Wang, W., McEvoy, C., Peña-Arancibia, J. L., Brancalion, P. H. S., Smith, P., Edwards, D. P., & Balmford, A. (2022). The biodiversity and ecosystem service contributions and trade-offs of forest restoration approaches. *Science*, *376*(6595), 839–844. https://doi.org/10.1126/science.abl4649

Hubau, W., Lewis, S. L., Phillips, O. L., Affum-Baffoe, K., Beeckman, H., Cuní-Sanchez, A., Daniels, A. K., Ewango, C. E., Fauset, S., Mukinzi, J. M., Sheil, D., Sonké, B., Sullivan, M. J., Sunderland, T. C., Taedoumg, H., Thomas, S. C., White, L. J., Abernethy, K. A., Adu-Bredu, S., . . . Zemagho, L. (2020). Asynchronous carbon sink saturation in African and Amazonian tropical forests. *Nature*, *579*(7797), 80–87. https://doi.org/10.1038/s41586-020-2035-0

IEA. (2020). *Iron and Steel Technology Roadmap* (tech. rep.). Paris. https://www.iea.org/reports/iron-and-steel-technology-roadmap

IEA. (2022). *Pulp and Paper* (tech. rep.). IEA. Paris. https://www.iea.org/reports/pulp-and-paper

Iglesias, A., Quiroga, S., Moneo, M., & Garrote, L. (2012). From climate change impacts to the development of adaptation strategies: Challenges for agriculture in Europe. *Climatic Change*, *112*(1), 143–168. https://doi.org/10.1007/s10584-011-0344-x

IPBES. (2019). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (S. Díaz, J. Settele, B. E.S., H. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. Brauman, S. Butchart, K. Chan, L. Garibaldi, K. Ichii, J. Liu, S. Subramanian, G. Midgley, P. Miloslavich, Z. Molnár, D. Obura, . . . C. Zayas, Eds.).

IPCC. (2021). AR6 - The Physical Science Basis - Summary for Policymakers. In *Climate change 2021 - the physical science basis. contribution of working group i to the sixth assessment report of the intergovernmental panel on climate change.*

IPCC. (2023a). Synthesis Report AR6 - Summary for Policymakers. *An Assessment of the Intergovernmental Panel on Climate Change.*

IPCC. (2023b). SYNTHESIS REPORT OF THE IPCC SIXTH ASSESSMENT REPORT (AR6) IPCC. Longer Report.

IPCC. (2023c). Water Cycle Changes. In *Climate change 2021: The physical science basis. contribution of working group i to the sixth assessment report of the intergovernmental panel on climate change* (pp. 1055–1210). Cambridge University Press. https://doi.org/10.1017/9781009157896.010

Ishizaka, A., & Nemery, P. (2013). *Multi-Criteria Decision Analysis*. John Wiley & Sons Ltd. https://doi.org/10.1002/9781118644898

Ito, A., Nishina, K., Reyer, C. P. O., François, L., Henrot, A.-J., Munhoven, G., Jacquemin, I., Tian, H., Yang, J., Pan, S., Morfopoulos, C., Betts, R., Hickler, T., Steinkamp, J., Ostberg, S., Schaphoff, S., Ciais, P., Chang, J., Rafique, R., … Zhao, F. (2017). Photosynthetic productivity and its efficiencies in ISIMIP2a biome models: benchmarking for impact assessment studies. *Environmental Research Letters*, *12*(8), 085001. https://doi.org/10.1088/1748-9326/aa7a19

Ivanovich, C. C., Sun, T., Gordon, D. R., & Ocko, I. B. (2023). Future warming from global food consumption. *Nature Climate Change*. https://doi.org/10.1038/s41558-023-01605-8

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

Jagu Schippers, E., Da Costa, P., & Massol, O. (2022). Coordinating the Deployment of Bioenergy with Carbon Capture and Storage (J.-P. Deflandre & F. Delprat-Jannaud, Eds.). *Science and Technology for Energy Transition*, *77*, 19. https://doi.org/10.2516/stet/2022018

Jandl, R., Spathelf, P., Bolte, A., & Prescott, C. E. (2019). Forest adaptation to climate change—is non-management an option? *Annals of Forest Science*, *76*(2), 1–13. https://doi.org/10.1007/s13595-019-0827-x

Jenkins, J. C. (2004). *Comprehensive database of diameter-based biomass regressions for North American tree species* (Vol. 319). United States Department of Agriculture, Forest Service, Northeastern Research Station.

Johansson, T. (2008). Sprouting ability and biomass production of downy and silver birch stumps of different diameters. *Biomass and Bioenergy*, *32*(10), 944–951. https://doi.org/10.1016/j.biombioe.2008.01.009

Johnston, C. M. T., & Radeloff, V. C. (2019). Global mitigation potential of carbon stored in harvested wood products. *Proceedings of the National Academy of Sciences*, *116*(29), 14526–14531. https://doi.org/10.1073/pnas.1904231116

Jönsson, A. M., Schroeder, L. M., Lagergren, F., Anderbrant, O., & Smith, B. (2012). Guess the impact of Ips typographus—An ecosystem modelling approach for simulating spruce bark beetle outbreaks. *Agricultural and Forest Meteorology*, *166-167*, 188–200. https://doi.org/10.1016/j.agrformet.2012.07.012

Joyce, A., Hallett, J., Hannelly, T., & Carey, G. (2014). The impact of nutritional choices on global warming and policy implications: examining the link between dietary choices and greenhouse gas emissions. *Energy and Emission Control Technologies*, 33. https://doi.org/10.2147/eect.s58518

Kennedy, D., Swenson, S., Oleson, K. W., Lawrence, D. M., Fisher, R., Lola da Costa, A. C., & Gentine, P. (2019). Implementing Plant Hydraulics in the Community

Land Model, Version 5. *Journal of Advances in Modeling Earth Systems*, *11*(2), 485–513. https://doi.org/10.1029/2018MS001500

Khanal, P. N., Grebner, D. L., Munn, I. A., Grado, S. C., Grala, R. K., & Henderson, J. E. (2017). Evaluating non-industrial private forest landowner willingness to manage for forest carbon sequestration in the southern United States. *Forest Policy and Economics*, *75*, 112–119. https://doi.org/10.1016/j.forpol.2016.07.004

Kirby, K., & Watkins, C. (1998). *Ecological History of European Forests*. CAB International.

Kleijnen, J. P. (2005). An overview of the design and analysis of simulation experiments for sensitivity analysis. *European Journal of Operational Research*, *164*(2), 287–300. https://doi.org/10.1016/j.ejor.2004.02.005

Klein, D., Höllerl, S., Blaschke, M., & Schulz, C. (2013). The Contribution of Managed and Unmanaged Forests to Climate Change Mitigation—A Model Approach at Stand Level for the Main Tree Species in Bavaria. *Forests*, *4*(1), 43–69. https://doi.org/10.3390/f4010043

Klein, D., & Schulz, C. (2012). *Die Kohlenstoffbilanz der Bayerischen Forst- und Holzwirtschaft* (tech. rep.). Bayerische Landesanstalt für Wald und Forstwirtschaft. https://www.lwf.bayern.de/mam/cms04/boden-klima/dateien/kohlenstoffbilanz_bayern_2012.pdf

Klein, N., Theux, C., Arlettaz, R., Jacot, A., & Pradervand, J.-N. (2020). Modeling the effects of grassland management intensity on biodiversity. *Ecology and Evolution*, *10*(23), 13518–13529. https://doi.org/10.1002/ece3.6957

Knauf, M., Köhl, M., Mues, V., Olschofsky, K., & Frühwald, A. (2015). Modeling the CO2-effects of forest management and wood usage on a regional basis. *Carbon Balance and Management*, *10*(1), 1–12. https://doi.org/10.1186/s13021-015-0024-7

Knoke, T., Ammer, C., Stimm, B., & Mosandl, R. (2008). Admixing broadleaved to coniferous tree species: a review on yield, ecological stability and economics. *European Journal of Forest Research*, *127*(2), 89–101. https://doi.org/10.1007/s10342-007-0186-2

Knoke, T., Bendix, J., Pohle, P., Hamer, U., Hildebrandt, P., Roos, K., Gerique, A., Sandoval, M. L., Breuer, L., Tischer, A., Silva, B., Calvas, B., Aguirre, N., Castro, L. M., Windhorst, D., Weber, M., Stimm, B., Günter, S., Palomeque, X., ... Beck, E. (2014). Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. *Nature Communications*, *5*. https://doi.org/10.1038/ncomms6612

Knoke, T., Kindu, M., Jarisch, I., Gosling, E., Friedrich, S., Bödeker, K., & Paul, C. (2020). How considering multiple criteria, uncertainty scenarios and biological interactions may influence the optimal silvicultural strategy for a mixed forest. *Forest Policy and Economics*, *118*(June), 102239. https://doi.org/10.1016/j.forpol.2020.102239

Knoke, T., Paul, C., Hildebrandt, P., Calvas, B., Castro, L. M., Hartl, F., Dollerer, M., Hamer, U., Windhorst, D., Wiersma, Y. F., Curatola Fernández, G. F., Obermeier, W. A., Adams, J., Breuer, L., Mosandl, R., Beck, E., Weber, M., Stimm, B., Haber, W., ... Bendix, J. (2016). Compositional diversity of rehabilitated tropical lands supports multiple ecosystem services and buffers uncertainties. *Nature Communications*, *7*(May). https://doi.org/10.1038/ncomms11877

Knoke, T., Paul, C., Rammig, A., Gosling, E., Hildebrandt, P., Härtl, F., Peters, T., Richter, M., Diertl, K. H., Castro, L. M., Calvas, B., Ochoa, S., Valle-Carrión, L. A., Hamer, U., Tischer, A., Potthast, K., Windhorst, D., Homeier, J., Wilcke, W., ... Bendix, J. (2020). Accounting for multiple ecosystem services in a simulation of land-use decisions: Does it reduce tropical deforestation? *Global Change Biology*, *26*(4), 2403–2420. https://doi.org/10.1111/gcb.15003

Knutti, R., & Sedláček, J. (2013). Robustness and uncertainties in the new CMIP5 climate model projections. *Nature Climate Change*, *3*(4), 369–373. https://doi.org/10.1038/nclimate1716

Koch, O., de Avila, A. L., Heinen, H., & Albrecht, A. T. (2022). Retreat of Major European Tree Species Distribution under Climate Change—Minor Natives to the Rescue? *Sustainability*, *14*(9), 5213. https://doi.org/10.3390/su14095213

Kolling, G., Stivanin, S., Gabbi, A., Machado, F., Ferreira, A., Campos, M., Tomich, T., Cunha, C., Dill, S., Pereira, L., & Fischer, V. (2018). Performance and methane emissions in dairy cows fed oregano and green tea extracts as feed additives. *Journal of Dairy Science*, *101*(5), 4221–4234. https://doi.org/10.3168/jds.2017-13841

Körner, C. (2017). A matter of tree longevity. *Science*, *355*(6321), 130–131. https://doi.org/10.1126/science.aal2449

Krause, A., Haverd, V., Poulter, B., Anthoni, P., Quesada, B., Rammig, A., & Arneth, A. (2019). Multimodel Analysis of Future Land Use and Climate Change Impacts on Ecosystem Functioning. *Earth's Future*, *7*(7), 833–851. https://doi.org/10.1029/2018EF001123

Krause, A., Knoke, T., & Rammig, A. (2020). A regional assessment of land-based carbon mitigation potentials: Bioenergy, BECCS, reforestation, and forest management. *GCB Bioenergy*, *12*(5), 346–360. https://doi.org/10.1111/gcbb.12675

Krause, A., Pugh, T. A. M., Bayer, A. D., Lindeskog, M., & Arneth, A. (2016). Impacts of land-use history on the recovery of ecosystems after agricultural abandonment. *Earth System Dynamics*, *7*(3), 745–766. https://doi.org/10.5194/esd-7-745-2016

Krause, A., Pughl, T. A., Bayer, A. D., Doelman, J. C., Humpenöder, F., Anthoni, P., Olin, S., Bodirsky, B. L., Popp, A., Stehfest, E., Arneth, A., Pugh, T. A., Bayer, A. D., Doelman, J. C., Humpenöder, F., Anthoni, P., Olin, S., Bodirsky, B. L., Popp, A., ... Arneth, A. (2017). Global consequences of afforestation and bioenergy cultivation on ecosystem service indicators. *Biogeosciences*, *14*(21), 4829–4850. https://doi.org/10.5194/bg-14-4829-2017

Kucharik, C. J., Norman, J. M., & Gower, S. T. (1998). Measurements of branch area and adjusting leaf area index indirect measurements. *Agricultural and Forest Meteorology*, *91*(1-2), 69–88. https://doi.org/10.1016/S0168-1923(98)00064-1

Lagergren, F., Jönsson, A. M., Blennow, K., & Smith, B. (2012). Implementing storm damage in a dynamic vegetation model for regional applications in Sweden. *Ecological Modelling*, *247*, 71–82. https://doi.org/10.1016/j.ecolmodel.2012.08.011

Laguë, M. M., Swann, A. L. S., & Boos, W. R. (2021). Radiative feedbacks on land surface change and associated tropical precipitation shifts. *Journal of Climate*, *34*(16), 1–63. https://doi.org/10.1175/JCLI-D-20-0883.1

Lamarque, J.-F., Kyle, G. P., Meinshausen, M., Riahi, K., Smith, S. J., van Vuuren, D. P., Conley, A. J., & Vitt, F. (2011). Global and regional evolution of short-lived radiatively-active gases and aerosols in the Representative Concentration Pathways. *Climatic Change*, *109*(1-2), 191–212. https://doi.org/10.1007/s10584-011-0155-0

Law, B. E., & Harmon, M. E. (2011). Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. *Carbon Management*, *2*(1), 73–84. https://doi.org/10.4155/cmt.10.40

Lawrence, J., Haasnoot, M., & Lempert, R. (2020). Climate change: making decisions in the face of deep uncertainty. *Nature*, *580*, 456.

Le Quéré, C., Andrew, R. M., Friedlingstein, P., Sitch, S., Pongratz, J., Manning, A. C., Korsbakken, J. I., Peters, G. P., Canadell, J. G., Jackson, R. B., Boden, T. A., Tans, P. P., Andrews, O. D., Arora, V. K., Bakker, D. C. E., Barbero, L., Becker, M., Betts, R. A., Bopp, L., . . . Zhu, D. (2018). Global Carbon Budget 2017. *Earth System Science Data*, *10*(1), 405–448. https://doi.org/10.5194/essd-10-405-2018

Lefebvre, D., Williams, A. G., Kirk, G. J., Paul, Burgess, J., Meersmans, J., Silman, M. R., Román-Dañobeytia, F., Farfan, J., & Smith, P. (2021). Assessing the carbon capture potential of a reforestation project. *Scientific Reports*, *11*(1), 2–11. https://doi.org/10.1038/s41598-021-99395-6

Lehne, J., & Preston, F. (2018). *Making Concrete Change; Innovation in Low-carbon Cement and Concrete* (tech. rep.).

Lenton, T. M., Rockström, J., Gaffney, O., Rahmstorf, S., Richardson, K., Steffen, W., & Schellnhuber, H. J. (2019). Climate tipping points — too risky to bet against. *Nature*, *575*(7784), 592–595. https://doi.org/10.1038/d41586-019-03595-0

Leonardsson, J., & Götmark, F. (2015). Differential survival and growth of stumps in 14 woody species after conservation thinning in mixed oak-rich temperate forests. *European Journal of Forest Research*, *134*(1), 199–209. https://doi.org/10.1007/s10342-014-0843-1

Lessa Derci Augustynczik, A., & Yousefpour, R. (2021). Assessing the synergistic value of ecosystem services in European beech forests. *Ecosystem Services*, *49*(October 2020), 101264. https://doi.org/10.1016/j.ecoser.2021.101264

Levy, P., Cannell, M., & Friend, A. (2004). Modelling the impact of future changes in climate, CO2 concentration and land use on natural ecosystems and the terrestrial carbon sink. *Global Environmental Change*, *14*(1), 21–30. https://doi.org/10.1016/j.gloenvcha.2003.10.005

Lewis, S. L., Wheeler, C. E., Mitchard, E. T., & Koch, A. (2019). Restoring natural forests is the best way to remove atmospheric carbon. *Nature*, *568*(7750), 25–28. https://doi.org/10.1038/d41586-019-01026-8

Li, J., Liu, Y., Tang, Y., Shao, J., Xu, T., Ma, R., Jiang, Y., & Cheng, D. (2022). Optimizing Fertilizer Management Based on Controlled-Release Fertilizer to

Improve Yield, Quality, and Reduce Fertilizer Application on Apples. *Journal of Soil Science and Plant Nutrition*, *22*(1), 393–405. https://doi.org/10.1007/s42729-021-00656-0

Li, W., Ciais, P., Stehfest, E., Van Vuuren, D., Popp, A., Arneth, A., Di Fulvio, F., Doelman, J., Humpenöder, F., B Harper, A., Park, T., Makowski, D., Havlik, P., Obersteiner, M., Wang, J., Krause, A., & Liu, W. (2020). Mapping the yields of lignocellulosic bioenergy crops from observations at the global scale. *Earth System Science Data*, *12*(2), 789–804. https://doi.org/10.5194/essd-12-789-2020

Li, X., & Xiao, J. (2019). A Global, 0.05-Degree Product of Solar-Induced Chlorophyll Fluorescence Derived from OCO-2, MODIS, and Reanalysis Data. *Remote Sensing*, *11*(5), 517. https://doi.org/10.3390/rs11050517

Liang, J., Crowther, T. W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E.-D., McGuire, A. D., Bozzato, F., Pretzsch, H., De-Miguel, S., Paquette, A., Hérault, B., Scherer-Lorenzen, M., Barrett, C. B., Glick, H. B., Hengeveld, G. M., Nabuurs, G.-J., Pfautsch, S., … Reich, P. B. (2016). Positive biodiversity-productivity relationship predominant in global forests. *Science*, *354*(6309). https://doi.org/10.1126/science.aaf8957

Linderholm, H. W. (2006). Growing season changes in the last century. *Agricultural and Forest Meteorology*, *137*(1-2), 1–14. https://doi.org/10.1016/j.agrformet.2006.03.006

Lindeskog, M., Arneth, A., Bondeau, A., Waha, K., Seaquist, J., Olin, S., & Smith, B. (2013). Implications of accounting for land use in simulations of ecosystem carbon cycling in Africa. *Earth System Dynamics*, *4*(2), 385–407. https://doi.org/10.5194/esd-4-385-2013

Lindeskog, M., Smith, B., Lagergren, F., Sycheva, E., Ficko, A., Pretzsch, H., & Rammig, A. (2021). Accounting for forest management in the estimation of forest carbon balance using the dynamic vegetation model LPJ-GUESS (v4.0, r9710): implementation and evaluation of simulations for Europe. *Geoscientific Model Development*, *14*(10), 6071–6112. https://doi.org/10.5194/gmd-14-6071-2021

Liu, Y. Y., Van Dijk, A. I., De Jeu, R. A., Canadell, J. G., McCabe, M. F., Evans, J. P., & Wang, G. (2015). Recent reversal in loss of global terrestrial biomass. *Nature Climate Change*, *5*(5), 470–474. https://doi.org/10.1038/nclimate2581

Loisel, J., Gallego-Sala, A. V., Amesbury, M. J., Magnan, G., Anshari, G., Beilman, D. W., Benavides, J. C., Blewett, J., Camill, P., Charman, D. J., Chawchai, S., Hedgpeth, A., Kleinen, T., Korhola, A., Large, D., Mansilla, C. A., Müller, J., van Bellen, S., West, J. B., … Wu, J. (2021). Expert assessment of future vulnerability of the global peatland carbon sink. *Nature Climate Change*, *11*(1), 70–77. https://doi.org/10.1038/s41558-020-00944-0

Luyssaert, S., Jammet, M., Stoy, P. C., Estel, S., Pongratz, J., Ceschia, E., Churkina, G., Don, A., Erb, K., Ferlicoq, M., Gielen, B., Grünwald, T., Houghton, R. A., Klumpp, K., Knohl, A., Kolb, T., Kuemmerle, T., Laurila, T., Lohila, A., … Dolman, A. J. (2014). Land management and land-cover change have impacts of similar magnitude on surface temperature. *Nature Climate Change*, *4*(5), 389–393. https://doi.org/10.1038/nclimate2196

Luyssaert, S., Marie, G., Valade, A., Chen, Y.-y. Y., Djomo, S. N., Ryder, J., Otto, J., Naudts, K., Lansø, A. S., Ghattas, J., & McGrath, M. J. (2018). Trade-offs in using European forests to meet climate objectives. *Nature*, *562*(7726), 259–262. https://doi.org/10.1038/s41586-018-0577-1

Luyssaert, S., Schulze, E. D., Börner, A., Knohl, A., Hessenmöller, D., Law, B. E., Ciais, P., & Grace, J. (2008). Old-growth forests as global carbon sinks. *Nature*, *455*(7210), 213–215. https://doi.org/10.1038/nature07276

Luyssaert, S., Schulze, E.-D., Knohl, A., Law, B. E., Ciais, P., & Grace, J. (2021). Reply to: Old-growth forest carbon sinks overestimated. *Nature*, *591*(7851), E24–E25. https://doi.org/10.1038/s41586-021-03267-y

Ma, J., Olin, S., Anthoni, P., Rabin, S. S., Bayer, A. D., Nyawira, S. S., & Arneth, A. (2022). Modeling symbiotic biological nitrogen fixation in grain legumes globally with LPJ-GUESS (v4.0, r10285). *Geoscientific Model Development*, *15*(2), 815–839. https://doi.org/10.5194/gmd-15-815-2022

Maganotti, N., Schweier, J., Spinelli, R., Tolosana, E., Jylhä, P., Sopushynskyy, I., Otepka, P., Nestorovski, L., Costa, M., Rodriguez, A., Rossney, D., Ruch, P., Tsioras, P., Stampfer, K., Mivelic, M., Krajnc, N., Mine, V., Mederski, P., & McEwan, A. (2018). *Coppice Forests in Europe*.

Maier, C., Hebermehl, W., Grossmann, C. M., Loft, L., Mann, C., & Hernández-Morcillo, M. (2021). Innovations for securing forest ecosystem service provision in Europe – A systematic literature review. *Ecosystem Services*, *52*(2021), 101374. https://doi.org/10.1016/j.ecoser.2021.101374

Maraun, D. (2016). Bias Correcting Climate Change Simulations - a Critical Review. *Current Climate Change Reports*, *2*(4), 211–220. https://doi.org/10.1007/s40641-016-0050-x

Martens, B., Miralles, D. G., Lievens, H., van der Schalie, R., de Jeu, R. A. M., Fernández-Prieto, D., Beck, H. E., Dorigo, W. A., & Verhoest, N. E. C. (2017). GLEAM v3: satellite-based land evaporation and root-zone soil moisture. *Geoscientific Model Development*, *10*(5), 1903–1925. https://doi.org/10.5194/gmd-10-1903-2017

Martinez-Harms, M. J., Bryan, B. A., Balvanera, P., Law, E. A., Rhodes, J. R., Possingham, H. P., & Wilson, K. A. (2015). Making decisions for managing ecosystem services. *Biological Conservation*, *184*, 229–238. https://doi.org/10.1016/j.biocon.2015.01.024

Mathys, A., Bottero, A., Stadelmann, G., Thürig, E., Ferretti, M., & Temperli, C. (2021). Presenting a climate-smart forestry evaluation framework based on national forest inventories. *Ecological Indicators*, *133*(September), 108459. https://doi.org/10.1016/j.ecolind.2021.108459

Matsumoto, M., Oka, H., Mitsuda, Y., Hashimoto, S., Kayo, C., Tsunetsugu, Y., & Tonosaki, M. (2016). Potential contributions of forestry and wood use to climate change mitigation in Japan. *Journal of Forest Research*, *21*(5), 211–222. https://doi.org/10.1007/s10310-016-0527-4

Mayer, A. L., Kauppi, P. E., Angelstam, P. K., Zhang, Y., & Tikka, P. M. (2005). Importing Timber, Exporting Ecological Impact. *Science*, *308*(5720), 359–360. https://doi.org/10.1126/science.1109476

McGrath, M. J., Luyssaert, S., Meyfroidt, P., Kaplan, J. O., Bürgi, M., Chen, Y., Erb, K., Gimmi, U., McInerney, D., Naudts, K., Otto, J., Pasztor, F., Ryder, J., Schelhaas, M.-J., & Valade, A. (2015). Reconstructing European forest management from 1600 to 2010. *Biogeosciences*, *12*(14), 4291–4316. https://doi.org/10.5194/bg-12-4291-2015

McGrath, M. J., Schulte-Frohlinde, A., & Luyssaert, S. (2024). New ways for (in)validating the forest carbon neutrality hypothesis. *Global Change Biology*, *30*(1), 1–11. https://doi.org/10.1111/gcb.16982

Meehl, G. A., & Tebaldi, C. (2004). More intense, more frequent, and longer lasting heat waves in the 21st century. *Science*, *305*(5686), 994–997. https://doi.org/10.1126/science.1098704

Meinshausen, M., Smith, S. J., Calvin, K., Daniel, J. S., Kainuma, M. L. T., Lamarque, J.-F., Matsumoto, K., Montzka, S. A., Raper, S. C. B., Riahi, K., Thomson, A., Velders, G. J. M., & van Vuuren, D. P. (2011). The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Climatic Change*, *109*(1-2), 213–241. https://doi.org/10.1007/s10584-011-0156-z

Metso. (2023). Miten METSO etenee? Retrieved October 18, 2023, from https://metsonpolku.fi/miten-metso-etenee

Meyer, B. F., Darela-Filho, J., Gu, Q., Gregor, K., Krause, A., Papastefanou, P., Buras, A., Hesse, B., Asuk, S. A., Liu, D., Grams, T. E. E., Zang, C. S., & Rammig, A. (draft). Disentangling parameter sensitivity in european-wide forest flux responses to drought in a dynamic vegetation model.

Millar, C. I., Stephenson, N. L., & Stephens, S. L. (2007). Climate change and forests of the future: Managing in the face of uncertainty. *Ecological Applications*, *17*(8), 2145–2151. https://doi.org/10.1890/06-1715.1

Millenium Ecosystem Assessment. (2005). *Ecosystems and Human Well-being: Synthesis* (tech. rep.). https://www.millenniumassessment.org/documents/document.356.aspx.pdf

Mitchell, S. J. (2013). Wind as a natural disturbance agent in forests: A synthesis. *Forestry*, *86*(2), 147–157. https://doi.org/10.1093/forestry/cps058

Mo, L., Zohner, C. M., Reich, P. B., Liang, J., de Miguel, S., Nabuurs, G.-J., Renner, S. S., van den Hoogen, J., Araza, A., Herold, M., Mirzagholi, L., Ma, H., Averill, C., Phillips, O. L., Gamarra, J. G. P., Hordijk, I., Routh, D., Abegg, M., Adou Yao, Y. C., … Crowther, T. W. (2023). Integrated global assessment of the natural forest carbon potential. *Nature*. https://doi.org/10.1038/s41586-023-06723-z

Molinaroli, E., Guerzoni, S., & Suman, D. (2019). Do the Adaptations of Venice and Miami to Sea Level Rise Offer Lessons for Other Vulnerable Coastal Cities? *Environmental Management*, *64*(4), 391–415. https://doi.org/10.1007/s00267-019-01198-z

Myllyviita, T., Soimakallio, S., Judl, J., & Seppälä, J. (2021). Wood substitution potential in greenhouse gas emission reduction–review on current state and application of displacement factors. *Forest Ecosystems*, *8*(1). https://doi.org/10.1186/s40663-021-00326-8

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

Nabuurs, G. J., Päivinen, R., & Schanz, H. (2001). Sustainable management regimes for Europe's forests — a projection with EFISCEN until 2050. *Forest Policy and Economics*, *3*(3-4), 155–173. https://doi.org/10.1016/S1389-9341(01)00058-2

Nabuurs, G.-J., Lindner, M., Verkerk, P. J., Gunia, K., Deda, P., Michalak, R., & Grassi, G. (2013). First signs of carbon sink saturation in European forest biomass. *Nature Climate Change*, *3*(9), 792–796. https://doi.org/10.1038/nclimate1853

Nabuurs, G.-J., Verkerk, Pieter Johannes Schelhaas, M.-J., González Olabarria, José Ramón Trasobares, A., & Cienciala, E. (2018). *Climate-Smart Forestry: mitigation impacts in three European regions*. https://doi.org/10.36333/fs06%20https://www.efi.int/sites/default/files/files/publication-bank/2018/efi_fstp_6_2018.pdf

Neary, D. G., Ice, G. G., & Jackson, C. R. (2009). Linkages between forest soils and water quality and quantity. *Forest Ecology and Management*, *258*(10), 2269–2281. https://doi.org/10.1016/j.foreco.2009.05.027

Nerlekar, A. N., & Veldman, J. W. (2020). High plant diversity and slow assembly of old-growth grasslands. *Proceedings of the National Academy of Sciences*, *117*(31), 18550–18556. https://doi.org/10.1073/pnas.1922266117

Neumann, K., Verburg, P. H., Stehfest, E., & Müller, C. (2010). The yield gap of global grain production: A spatial analysis. *Agricultural Systems*, *103*(5), 316–326. https://doi.org/10.1016/j.agsy.2010.02.004

Nicholls, R. J., & Cazenave, A. (2010). Sea-Level Rise and Its Impact on Coastal Zones. *Science*, *328*(5985), 1517–1520. https://doi.org/10.1126/science.1185782

Nicolescu, V.-N., Carvalho, J., Hochbichler, E., Bruckman, V., Piqué-nicolau, M., Hernea, C., Viana, H., Štochlová, P., Ertekin, M., Tijardovic, M., Dubravac, T., Vandekerkhove, K., Kofman, P. D., Rossney, D., & Unrau, A. (2017). *Silvicultural Guidelines for European Coppice Forests* (tech. rep.). Freiburg, Germany: Albert Ludwig University of Freiburg.

Nishina, K., Ito, A., Falloon, P., Friend, A. D., Beerling, D. J., Ciais, P., Clark, D. B., Kahana, R., Kato, E., Lucht, W., Lomas, M., Pavlick, R., Schaphoff, S., Warszawaski, L., & Yokohata, T. (2015). Decomposing uncertainties in the future terrestrial carbon budget associated with emission scenarios, climate projections, and ecosystem simulations using the ISI-MIP results. *Earth System Dynamics*, *6*(2), 435–445. https://doi.org/10.5194/esd-6-435-2015

Ohashi, H., Hasegawa, T., Hirata, A., Fujimori, S., Takahashi, K., Tsuyama, I., Nakao, K., Kominami, Y., Tanaka, N., Hijioka, Y., & Matsui, T. (2019). Biodiversity can benefit from climate stabilization despite adverse side effects of land-based mitigation. *Nature Communications*, *10*(1), 1–11. https://doi.org/10.1038/s41467-019-13241-y

Ohlemüller, R., Gritti, E. S., Sykes, M. T., & Thomas, C. D. (2006). Towards European climate risk surfaces: The extent and distribution of analogous and non-analogous climates 1931-2100. *Global Ecology and Biogeography*, *15*(4), 395–405. https://doi.org/10.1111/j.1466-822X.2006.00245.x

Olin, S., Schurgers, G., Lindeskog, M., Wårlind, D., Smith, B., Bodin, P., Holmér, J., & Arneth, A. (2015). Modelling the response of yields and tissue C : N to changes in atmospheric CO2 and N management in the main wheat regions of western Europe. *Biogeosciences*, *12*(8), 2489–2515. https://doi.org/10.5194/bg-12-2489-2015

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., Phillips, O. L., Shvidenko, A., Lewis, S. L., Canadell, J. G., Ciais, P., Jackson, R. B., Pacala, S. W., McGuire, A. D., Piao, S., Rautiainen, A., Sitch, S., & Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, *333*(6045), 988–993. https://doi.org/10.1126/science.1201609

Papastefanou, P., Pugh, T. A. M., Buras, A., Fleischer, K., Grams, T. E. E., Hickler, T., Lapola, D., Liu, D., Zang, C. S., & Rammig, A. (in review). Simulated sensitivity of the Amazon rainforest to extreme drought.

Parmesan, C., Ryrholm, N., Stefanescu, C., Hill, J. K., Thomas, C. D., Descimon, H., Huntley, B., Kaila, L., Kullberg, J., Tammaru, T., Tennent, W. J., Thomas, J. A., & Warren, M. (1999). Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature*, *399*(6736), 579–583. https://doi.org/10.1038/21181

Parton, W. J., Scurlock, J. M. O., Ojima, D. S., Gilmanov, T. G., Scholes, R. J., Schimel, D. S., Kirchner, T., Menaut, J.-C., Seastedt, T., Garcia Moya, E., Kamnalrut, A., & Kinyamario, J. I. (1993). Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Biogeochemical Cycles*, *7*(4), 785–809. https://doi.org/10.1029/93GB02042

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

Pedersen, J. S. T., van Vuuren, D. P., Aparício, B. A., Swart, R., Gupta, J., & Santos, F. D. (2020). Variability in historical emissions trends suggests a need for a wide range of global scenarios and regional analyses. *Communications Earth & Environment*, *1*(1), 1–7. https://doi.org/10.1038/s43247-020-00045-y

Pellaton, R., Lellei-Kovács, E., & Báldi, A. (2022). Cultural ecosystem services in European grasslands: A systematic review of threats. *Ambio*, *51*(12), 2462–2477. https://doi.org/10.1007/s13280-022-01755-7

Pereira, H. M., Navarro, L. M., & Martins, I. S. (2012). Global biodiversity change: The Bad, the good, and the unknown. *Annual Review of Environment and Resources*, *37*, 25–50. https://doi.org/10.1146/annurev-environ-042911-093511

Petersson, H., Ellison, D., Appiah Mensah, A., Berndes, G., Egnell, G., Lundblad, M., Lundmark, T., Lundström, A., Stendahl, J., & Wikberg, P.-E. (2022). On the role of forests and the forest sector for climate change mitigation in Sweden. *GCB Bioenergy*, *14*(7), 793–813. https://doi.org/10.1111/gcbb.12943

Phalan, B., Onial, M., Balmford, A., & Green, R. E. (2011). Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared. *Science*, *333*(6047), 1289–1291. https://doi.org/10.1126/science.1208742

Picoli, M. C., Rorato, A., Leitão, P., Camara, G., Maciel, A., Hostert, P., & Sanches, I. D. A. (2020). Impacts of public and private sector policies on soybean and pasture expansion in mato Grosso-Brazil from 2001 to 2017. *Land*, *9*(1), 1–15. https://doi.org/10.3390/land9010020

Piggott-McKellar, A. E., Nunn, P. D., McNamara, K. E., & Sekinini, S. T. (2020). Dam(n) Seawalls: A Case of Climate Change Maladaptation in Fiji. In W. Leal Filho (Ed.), *Managing climate change adaptation in the pacific region* (pp. 69–84). Springer International Publishing. https://doi.org/10.1007/978-3-030-40552-6_4

Poeplau, C., Germer, K., & Schwarz, K.-U. (2019). Seasonal dynamics and depth distribution of belowground biomass carbon and nitrogen of extensive grassland and a Miscanthus plantation. *Plant and Soil*, *440*(1-2), 119–133. https://doi.org/10.1007/s11104-019-04074-1

Pongratz, J., Reick, C. H., Raddatz, T., & Claussen, M. (2010). Biogeophysical versus biogeochemical climate response to historical anthropogenic land cover change. *Geophysical Research Letters*, *37*(8), 1–5. https://doi.org/10.1029/2010GL043010

Popp, J., Lakner, Z., Harangi-Rákos, M., & Fári, M. (2014). The effect of bioenergy expansion: Food, energy, and environment. *Renewable and Sustainable Energy Reviews*, *32*(February), 559–578. https://doi.org/10.1016/j.rser.2014.01.056

Poulter, B., Aragão, L., Andela, N., Bellassen, V., Ciais, P., Kato, T., Lin, X., Nachin, B., Luyssaert, S., Pederson, N., Peylin, P., Piao, S., Saatchi, S., Schepaschenko, D., Schelhaas, M., & Shivdenko, A. (2018). The global forest age dataset (GFADv1.0). https://doi.org/10.1594/PANGAEA.889943

Prentice, I. C., Bondeau, A., Cramer, W., Harrison, S. P., Hickler, T., Lucht, W., Sitch, S., Smith, B., & Sykes, M. T. (2007). Dynamic Global Vegetation Modeling: Quantifying Terrestrial Ecosystem Responses to Large-Scale Environmental Change. *Terrestrial Ecosystems in a Changing World*, 175–192. https://doi.org/10.1007/978-3-540-32730-1_15

Pretzsch, H., Schütze, G., & Uhl, E. (2013). Resistance of European tree species to drought stress in mixed versus pure forests: evidence of stress release by inter-specific facilitation. *Plant Biology*, *15*(3), 483–495. https://doi.org/10.1111/j.1438-8677.2012.00670.x

Pugh, T. A. M., Arneth, A., Kautz, M., Poulter, B., & Smith, B. (2019). Important role of forest disturbances in the global biomass turnover and carbon sinks. *Nature Geoscience*, *12*(9), 730–735. https://doi.org/10.1038/s41561-019-0427-2

Rabin, S. S., Melton, J. R., Lasslop, G., Bachelet, D., Forrest, M., Hantson, S., Kaplan, J. O., Li, F., Mangeon, S., Ward, D. S., Yue, C., Arora, V. K., Hickler, T., Kloster, S., Knorr, W., Nieradzik, L., Spessa, A., Folberth, G. A., Sheehan, T., ... Arneth, A. (2017). The Fire Modeling Intercomparison Project (FireMIP), phase 1: experimental and analytical protocols with detailed model descriptions. *Geoscientific Model Development*, *10*(3), 1175–1197. https://doi.org/10.5194/gmd-10-1175-2017

Rahmati, M., Graf, A., Poppe Terán, C., Amelung, W., Dorigo, W., Franssen, H.-j. H., Montzka, C., Or, D., Sprenger, M., Vanderborght, J., Verhoest, N. E. C., & Vereecken, H. (2023). Continuous increase in evaporative demand shortened the growing season of European ecosystems in the last decade. *Communications Earth & Environment*, *4*(1), 236. https://doi.org/10.1038/s43247-023-00890-7

Raupach, M. R. (1994). Simplified expressions for vegetation roughness length and zero-plane displacement as functions of canopy height and area index. *Boundary-Layer Meteorology*, *71*(1-2), 211–216. https://doi.org/10.1007/BF00709229

Reineke, L. H. (1933). Perfecting a stand-density index for even-aged forest. *Journal of Agricultural Research*, *46*, 627–638.

ReKliEs. (2017). Regionale Klimaprojektionen Ensemble für Deutschland. https://doi.org/10.2312/WDCC/ReKliEsDe_Ergebnisbericht

Riahi, K., van Vuuren, D. P., Kriegler, E., Edmonds, J., O'Neill, B. C., Fujimori, S., Bauer, N., Calvin, K., Dellink, R., Fricko, O., Lutz, W., Popp, A., Cuaresma, J. C., KC, S., Leimbach, M., Jiang, L., Kram, T., Rao, S., Emmerling, J., ... Tavoni, M. (2017). The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. *Global Environmental Change*, *42*, 153–168. https://doi.org/10.1016/j.gloenvcha.2016.05.009

Rivera, A., Bravo, C., & Buob, G. (2017). *IPCC: Climate Change and Land*. https://doi.org/10.1002/9781118786352.wbieg0538

Rogelj, J., Shindell, D., Jiang, K., Fifita, S., Forster, P., Ginzburg, V., Handa, C., Kheshgi, H., Kobayashi, S., Kriegler, E., Mundaca, L., Séférian, R., & Vilariño, M. (2022). Mitigation Pathways Compatible with 1.5°C in the Context of Sustainable Development. In *Global warming of 1.5°c* (pp. 93–174). Cambridge University Press. https://doi.org/10.1017/9781009157940.004

Rogelj, J., Popp, A., Calvin, K. V., Luderer, G., Emmerling, J., Gernaat, D., Fujimori, S., Strefler, J., Hasegawa, T., Marangoni, G., Krey, V., Kriegler, E., Riahi, K., van Vuuren, D. P., Doelman, J., Drouet, L., Edmonds, J., Fricko, O., Harmsen, M., ... Tavoni, M. (2018). Scenarios towards limiting global mean temperature increase below 1.5 °C. *Nature Climate Change*, *8*(4), 325–332. https://doi.org/10.1038/s41558-018-0091-3

Rosa, F., Di Fulvio, F., Lauri, P., Felton, A., Forsell, N., Pfister, S., & Hellweg, S. (2023). Can Forest Management Practices Counteract Species Loss Arising from

Increasing European Demand for Forest Biomass under Climate Mitigation Scenarios? *Environmental Science and Technology*, *57*(5), 2149–2161. https://doi.org/10.1021/acs.est.2c07867

Ruckelshaus, M., McKenzie, E., Tallis, H., Guerry, A., Daily, G., Kareiva, P., Polasky, S., Ricketts, T., Bhagabati, N., Wood, S. A., & Bernhardt, J. (2015). Notes from the field: Lessons learned from using ecosystem service approaches to inform real-world decisions. *Ecological Economics*, *115*, 11–21. https://doi.org/10.1016/j.ecolecon.2013.07.009

Running, S., & Zhao, M. (2021). MODIS/Terra Net Primary Production Gap-Filled Yearly L4 Global 500m SIN Grid V061 [Data set]. https://doi.org/10.5067/MODIS/MOD17A3HGF.061

Rüter, S. (2011). Welchen Beitrag leisten Holzprodukte zur CO2-Bilanz? *AFZ/Der Wald*, *15*, 15–18.

Rydberg, D. (2000). Initial sprouting, growth and mortality of European aspen and birch after selective coppicing in central Sweden. *Forest Ecology and Management*, *130*(1-3), 27–35. https://doi.org/10.1016/S0378-1127(99)00187-5

Sage, R. F. (2020). Global change biology: A primer. *Global Change Biology*, *26*(1), 3–30. https://doi.org/10.1111/gcb.14893

Santoro, M., Cartus, O., Carvalhais, N., Rozendaal, D. M. A., Avitabile, V., Araza, A., de Bruin, S., Herold, M., Quegan, S., Rodríguez-Veiga, P., Balzter, H., Carreiras, J., Schepaschenko, D., Korets, M., Shimada, M., Itoh, T., Moreno Martínez, Á., Cavlovic, J., Cazzolla Gatti, R., … Willcock, S. (2021). The global forest above-ground biomass pool for 2010 estimated from high-resolution satellite observations. *Earth System Science Data*, *13*(8), 3927–3950. https://doi.org/10.5194/essd-13-3927-2021

Sathre, R., & O'Connor, J. (2010). Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science & Policy*, *13*(2), 104–114. https://doi.org/10.1016/j.envsci.2009.12.005

Saunois, M., Stavert, A. R., Poulter, B., Bousquet, P., Canadell, J. G., Jackson, R. B., Raymond, P. A., Dlugokencky, E. J., Houweling, S., Patra, P. K., Ciais, P., Arora, V. K., Bastviken, D., Bergamaschi, P., Blake, D. R., Brailsford, G., Bruhwiler, L., Carlson, K. M., Carrol, M., … Zhuang, Q. (2020). The Global Methane Budget 2000–2017. *Earth System Science Data*, *12*(3), 1561–1623. https://doi.org/10.5194/essd-12-1561-2020

Savill, P. S. (Ed.). (2019). *The silviculture of trees used in British forestry*. CABI. https://doi.org/10.1079/9781786393920.0000

Scarlat, N., Dallemand, J.-F., Taylor, N., Banja, M., Sanchez Lopez, J., & Avraamides, M. (2019). Brief on biomass for energy in the European Union. *EC publication*, 1–8. https://doi.org/10.2760/546943

Schelhaas, M. J., Fridman, J., Hengeveld, G. M., Henttonen, H. M., Lehtonen, A., Kies, U., Krajnc, N., Lerink, B., Dhubháin, Á. N., Polley, H., Pugh, T. A., Redmond, J. J., Rohner, B., Temperli, C., Vayreda, J., & Nabuurs, G. J. (2018). Actual European forest management by region, tree species and owner based on

714,000 re-measured trees in national forest inventories. *PLoS ONE*, *13*(11), 1–23. https://doi.org/10.1371/journal.pone.0207151

Scheper, J., Holzschuh, A., Kuussaari, M., Potts, S. G., Rundlöf, M., Smith, H. G., & Kleijn, D. (2013). Environmental factors driving the effectiveness of European agri-environmental measures in mitigating pollinator loss - a meta-analysis (J. Gomez, Ed.). *Ecology Letters*, *16*(7), 912–920. https://doi.org/10.1111/ele.12128

Schreyer, F., Luderer, G., Rodrigues, R., Pietzcker, R. C., Baumstark, L., Sugiyama, M., Brecha, R. J., & Ueckerdt, F. (2020). Common but differentiated leadership: strategies and challenges for carbon neutrality by 2050 across industrialized economies. *Environmental Research Letters*, *15*(11), 114016. https://doi.org/10.1088/1748-9326/abb852

Schuldt, B., Buras, A., Arend, M., Vitasse, Y., Beierkuhnlein, C., Damm, A., Gharun, M., Grams, T. E., Hauck, M., Hajek, P., Hartmann, H., Hiltbrunner, E., Hoch, G., Holloway-Phillips, M., Körner, C., Larysch, E., Lübbe, T., Nelson, D. B., Rammig, A., . . . Kahmen, A. (2020). A first assessment of the impact of the extreme 2018 summer drought on Central European forests. *Basic and Applied Ecology*, *45*, 86–103. https://doi.org/10.1016/j.baae.2020.04.003

Schulte, M., Jonsson, R., Hammar, T., Stendahl, J., & Hansson, P.-A. (2022). Nordic forest management towards climate change mitigation: time dynamic temperature change impacts of wood product systems including substitution effects. *European Journal of Forest Research*, *141*(5), 845–863. https://doi.org/10.1007/s10342-022-01477-1

Schuur, E. A., Abbott, B. W., Commane, R., Ernakovich, J., Euskirchen, E., Hugelius, G., Grosse, G., Jones, M., Koven, C., Leshyk, V., Lawrence, D., Loranty, M. M., Mauritz, M., Olefeldt, D., Natali, S., Rodenhizer, H., Salmon, V., Schädel, C., Strauss, J., . . . Turetsky, M. (2022). Permafrost and Climate Change: Carbon Cycle Feedbacks From the Warming Arctic. *Annual Review of Environment and Resources*, *47*, 343–371. https://doi.org/10.1146/annurev-environ-012220-011847

Schwaab, J., Davin, E. L., Bebi, P., Duguay-Tetzlaff, A., Waser, L. T., Haeni, M., & Meier, R. (2020). Increasing the broad-leaved tree fraction in European forests mitigates hot temperature extremes. *Scientific Reports*, *10*(1), 1–9. https://doi.org/10.1038/s41598-020-71055-1

Schwalm, C. R., Glendon, S., & Duffy, P. B. (2020). Reply to hausfather and peters: RCP8.5 is neither problematic nor misleading. *Proceedings of the National Academy of Sciences of the United States of America*, *117*(45), 27793–27794. https://doi.org/10.1073/pnas.2018008117

Seibold, S., Gossner, M. M., Simons, N. K., Blüthgen, N., Müller, J., Ambarlı, D., Ammer, C., Bauhus, J., Fischer, M., Habel, J. C., Linsenmair, K. E., Nauss, T., Penone, C., Prati, D., Schall, P., Schulze, E.-D., Vogt, J., Wöllauer, S., & Weisser, W. W. (2019). Arthropod decline in grasslands and forests is associated with landscape-level drivers. *Nature*, *574*(7780), 671–674. https://doi.org/10.1038/s41586-019-1684-3

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. (2017). Forest disturbances under climate change. *Nature Climate Change*, *7*(6), 395–402. https://doi.org/10.1038/nclimate3303

Senf, C., Müller, J., & Seidl, R. (2019). Post-disturbance recovery of forest cover and tree height differ with management in Central Europe. *Landscape Ecology*, *34*(12), 2837–2850. https://doi.org/10.1007/s10980-019-00921-9

Senf, C., & Seidl, R. (2021a). Mapping the forest disturbance regimes of Europe. *Nature Sustainability*, *4*(1), 63–70. https://doi.org/10.1038/s41893-020-00609-y

Senf, C., & Seidl, R. (2021b). Storm and fire disturbances in Europe: Distribution and trends. *Global Change Biology*, (March), 3605–3619. https://doi.org/10.1111/gcb.15679

Shephard, N. T., Narine, L., Peng, Y., & Maggard, A. (2022). Climate Smart Forestry in the Southern United States. *Forests*, *13*(9), 1460. https://doi.org/10.3390/f13091460

Sishodia, R. P., Ray, R. L., & Singh, S. K. (2020). Applications of Remote Sensing in Precision Agriculture: A Review. *Remote Sensing*, *12*(19), 3136. https://doi.org/10.3390/rs12193136

Sitch, S., Smith, B., Prentice, I. C., Arneth, A., Bondeau, A., Cramer, W., Kaplan, J. O., Levis, S., Lucht, W., Sykes, M. T., Thonicke, K., & Venevsky, S. (2003). Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. *Global Change Biology*, *9*(2), 161–185. https://doi.org/10.1046/j.1365-2486.2003.00569.x

Skytt, T., Englund, G., & Jonsson, B.-g. (2021). Climate mitigation forestry—temporal trade-offs. *Environmental Research Letters*, *16*(11), 114037. https://doi.org/10.1088/1748-9326/ac30fa

Smil, V. (2015). *Power Density*. The MIT Press. https://doi.org/10.7551/mitpress/10046.001.0001

Smith, B., Wårlind, D., Arneth, A., Hickler, T., Leadley, P., Siltberg, J., & Zaehle, S. (2014). Implications of incorporating N cycling and N limitations on primary production in an individual-based dynamic vegetation model. *Biogeosciences*, *11*(7), 2027–2054. https://doi.org/10.5194/bg-11-2027-2014

Smith, B., Prentice, I. C., & Sykes, M. T. (2001). Representation of vegetation dynamics in the modelling of terrestrial ecosystems: comparing two contrasting approaches within European climate space. *Global Ecology and Biogeography*, *10*(6), 621–637. https://doi.org/10.1046/j.1466-822X.2001.t01-1-00256.x

Smyth, C. E., Stinson, G., Neilson, E., Lemprière, T. C., Hafer, M., Rampley, G. J., & Kurz, W. A. (2014). Quantifying the biophysical climate change mitigation potential of Canada's forest sector. *Biogeosciences*, *11*(13), 3515–3529. https://doi.org/10.5194/bg-11-3515-2014

Soimakallio, S., Kalliokoski, T., Lehtonen, A., & Salminen, O. (2021). On the trade-offs and synergies between forest carbon sequestration and substitution. *Mitigation and*

Adaptation Strategies for Global Change, *26*(1), 4. https://doi.org/10.1007/s11027-021-09942-9

Spiecker, H., Lindner, M., & Schuler, J. (2019). *Douglas-fir - an option for Europe. What Science Can Tell Us*. EFI.

Spinoni, J., Vogt, J. V., Naumann, G., Barbosa, P., & Dosio, A. (2018). Will drought events become more frequent and severe in Europe? *International Journal of Climatology*, *38*(4), 1718–1736. https://doi.org/10.1002/joc.5291

Statistische Ämter der Länder. (2022). *Umweltökonomische Gesamtrechnungen der Länder* (tech. rep.). https://www.statistikportal.de/de/ugrdl/veroeffentlichungen

Stimm, B., Roloff, A., Lang, U. M., & Weisgerber, H. (Eds.). (2014). *Enzyklopädie der Holzgewächse: Handbuch und Atlas der Dendrologie*. Wiley. https://doi.org/10.1002/9783527678518

Strassburg, B. B. N., Iribarrem, A., Beyer, H. L., Cordeiro, C. L., Crouzeilles, R., Jakovac, C. C., Braga Junqueira, A., Lacerda, E., Latawiec, A. E., Balmford, A., Brooks, T. M., Butchart, S. H. M., Chazdon, R. L., Erb, K.-H., Brancalion, P., Buchanan, G., Cooper, D., Díaz, S., Donald, P. F., … Visconti, P. (2020). Global priority areas for ecosystem restoration. *Nature*, *586*(7831), 724–729. https://doi.org/10.1038/s41586-020-2784-9

Su, Y., Zhang, J., Peng, S., & Ding, Y. (2022). Simulating Ecological Functions of Vegetation Using a Dynamic Vegetation Model. *Forests*, *13*(9), 1464. https://doi.org/10.3390/f13091464

Sundseth, K. (2008). Natura 2000: Protecting Europe's biodiversity. *53*(9), 1689–1699.

Suvanto, S., Muelbert, A. E., Schelhaas, M.-J., Astigarraga, J., Astrup, R., Cienciala, E., Fridman, J., Henttonen, H. M., Kunstler, G., Kändler, G., König, L. A., Ruiz-Benito, P., Senf, C., Stadelmann, G., Starcevic, A., Talarczyk, A., Zavala, M. A., & Pugh, T. A. (in review). Understanding Europe's forest harvesting regimes. https://doi.org/10.31223/X5910J

Taylor, K. E., Stouffer, R. J., & Meehl, G. A. (2012). An Overview of CMIP5 and the Experiment Design. *Bulletin of the American Meteorological Society*, *93*(4), 485–498. https://doi.org/10.1175/BAMS-D-11-00094.1

Thom, D., Buras, A., Heym, M., Klemmt, H. J., & Wauer, A. (2023). Varying growth response of Central European tree species to the extraordinary drought period of 2018 – 2020. *Agricultural and Forest Meteorology*, *338*(April). https://doi.org/10.1016/j.agrformet.2023.109506

Thonicke, K., Venevsky, S., Sitch, S., & Cramer, W. (2001). The role of fire disturbance for global vegetation dynamics: Coupling fire into a dynamic global vegetation model. *Global Ecology and Biogeography*, *10*(6), 661–677. https://doi.org/10.1046/j.1466-822X.2001.00175.x

Thuiller, W., Lavorel, S., Araújo, M. B., Sykes, M. T., & Prentice, I. C. (2005). Climate change threats to plant diversity in Europe. *Proceedings of the National Academy of Sciences of the United States of America*, *102*(23), 8245–8250. https://doi.org/10.1073/pnas.0409902102

Treu, H., Nordborg, M., Cederberg, C., Heuer, T., Claupein, E., Hoffmann, H., & Berndes, G. (2017). Carbon footprints and land use of conventional and organic diets in Germany. *Journal of Cleaner Production*, *161*, 127–142. https://doi.org/10.1016/j.jclepro.2017.05.041

Tschumi, E., Lienert, S., Bastos, A., Ciais, P., Gregor, K., Joos, F., Knauer, J., Papastefanou, P., Rammig, A., van der Wiel, K., Williams, K., Xu, Y., Zaehle, S., & Zscheischler, J. (2023). Large Variability in Simulated Response of Vegetation Composition and Carbon Dynamics to Variations in Drought-Heat Occurrence. *Journal of Geophysical Research: Biogeosciences*, *128*(4). https://doi.org/10.1029/2022JG007332

Uhde, B., Hahn, A., Griess, V. C., & Knoke, T. (2015). Hybrid MCDA Methods to Integrate Multiple Ecosystem Services in Forest Management Planning: A Critical Review. *Environmental Management*, *56*(2), 373–388. https://doi.org/10.1007/s00267-015-0503-3

Uhde, B., Heinrichs, S., Stiehl, C. R., Ammer, C., Müller-Using, B., & Knoke, T. (2017). Bringing ecosystem services into forest planning – Can we optimize the composition of Chilean forests based on expert knowledge? *Forest Ecology and Management*, *404*(March), 126–140. https://doi.org/10.1016/j.foreco.2017.08.021

United Nations. (2015). Summary of the Paris Agreement. Retrieved January 15, 2024, from https://unfccc.int/sites/default/files/english_paris_agreement.pdf

Urban, M. C. (2015). Accelerating extinction risk from climate change. *Science*, *348*(6234), 571–573. https://doi.org/10.1126/science.aaa4984

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 der Wiel, K., Batelaan, T. J., & Wanders, N. (2023). Large increases of multi-year droughts in north-western Europe in a warmer climate. *Climate Dynamics*, *60*(5-6), 1781–1800. https://doi.org/10.1007/s00382-022-06373-3

van der Zanden, E. H., Verburg, P. H., Schulp, C. J., & Verkerk, P. J. (2017). Trade-offs of European agricultural abandonment. *Land Use Policy*, *62*, 290–301. https://doi.org/10.1016/j.landusepol.2017.01.003

Van Dingenen, R., Crippa, M., J., Anssens-Maenhout, G., Guizzardi, D., & Dentener, F. (2018). *Global trends of methane emissions and their impacts on ozone concentrations* (Vol. EUR29394EN). https://doi.org/10.2760/820175

Van Meijgaard, E., Van Ulft, L. H., Lenderink, G., De Roode, S. R., Wipfler, E. L., Boers, R., & van Timmermans, R. M. A. (2012). *Refinement and application of a regional atmospheric model for climate scenario calculations of Western Europe*. KvR.

van Kooten, G. C., & Johnston, C. M. (2016). The Economics of Forest Carbon Offsets. *Annual Review of Resource Economics*, *8*(1), 227–246. https://doi.org/10.1146/annurev-resource-100815-095548

Veldman, J. (2019). Comment on “The global tree restoration potential”. *Science*, *366*(6469), 1–5. https://doi.org/10.1126/science.aaz0111

Verkerk, P. J., Costanza, R., Hetemäki, L., Kubiszewski, I., Leskinen, P., Nabuurs, G. J., Potočnik, J., & Palahí, M. (2020). Climate-Smart Forestry: the missing link. *Forest Policy and Economics*, *115*(April). https://doi.org/10.1016/j.forpol.2020.102164

Verkerk, P. J., Mavsar, R., Giergiczny, M., Lindner, M., Edwards, D., & Schelhaas, M. J. (2014). Assessing impacts of intensi fi ed biomass production and biodiversity protection on ecosystem services provided by European forests. *Ecosystem Services*, *9*, 155–165. https://doi.org/10.1016/j.ecoser.2014.06.004

Verkerk, P. J., Delacote, P., Hurmekoski, E., Kunttu, J., Matthews, R., Mäkipää, R., Mosley, F., Perugini, L., Reyer, C. P. O., Roe, S., & Trømborg, E. (2022). *Forest-based climate change mitigation and adaptation in Europe. From Science to Policy 14*. European Forest Institute. https://doi.org/10.36333/fs14

Verlinden, M. S., Broeckx, L. S., & Ceulemans, R. (2015). First vs. second rotation of a poplar short rotation coppice: Above-ground biomass productivity and shoot dynamics. *Biomass and Bioenergy*, *73*, 174–185. https://doi.org/10.1016/j.biombioe.2014.12.012

Vicedo-Cabrera, A. M., Scovronick, N., Sera, F., Royé, D., Schneider, R., Tobias, A., Astrom, C., Guo, Y., Honda, Y., Hondula, D. M., Abrutzky, R., Tong, S., Coelho, M. d. S. Z. S., Saldiva, P. H., Lavigne, E., Correa, P. M., Ortega, N. V., Kan, H., Osorio, S., ... Gasparrini, A. (2021). The burden of heat-related mortality attributable to recent human-induced climate change. *Nature Climate Change*, *11*(6), 492–500. https://doi.org/10.1038/s41558-021-01058-x

Villwock, D., Kurz, S., Hartung, J., & Müller-Lindenlauf, M. (2022). Effects of Stand Density and N Fertilization on the Performance of Maize (Zea mays L.) Intercropped with Climbing Beans (Phaseolus vulgaris L.) *Agriculture (Switzerland)*, *12*(7). https://doi.org/10.3390/agriculture12070967

Virtanen, P., Gommers, R., Oliphant, T. E., Haberland, M., Reddy, T., Cournapeau, D., Burovski, E., Peterson, P., Weckesser, W., Bright, J., van der Walt, S. J., Brett, M., Wilson, J., Millman, K. J., Mayorov, N., Nelson, A. R. J., Jones, E., Kern, R., Larson, E., ... Contributors, S. 1. (2020). SciPy 1.0: fundamental algorithms for scientific computing in Python. *Nature Methods*, *17*(3), 261–272. https://doi.org/10.1038/s41592-019-0686-2

Vitousek, P. M. (1994). Beyond Global Warming: Ecology and Global Change. *Ecology*, *75*(7), 1861–1876. https://doi.org/10.2307/1941591

Vuidot, A., Paillet, Y., Archaux, F., & Gosselin, F. (2011). Influence of tree characteristics and forest management on tree microhabitats. *Biological Conservation*, *144*(1), 441–450. https://doi.org/10.1016/j.biocon.2010.09.030

Wagner, W., & Lau, S. (1971). The Effect of Diversification on Risk. *Financial Analysts Journal*, *27*(6), 48–53. https://doi.org/10.2469/faj.v27.n6.48

Walker, A. P., De Kauwe, M. G., Bastos, A., Belmecheri, S., Georgiou, K., Keeling, R. F., McMahon, S. M., Medlyn, B. E., Moore, D. J. P., Norby, R. J., Zaehle, S., Anderson-Teixeira, K. J., Battipaglia, G., Brienen, R. J. W., Cabugao, K. G., Cailleret, M., Campbell, E., Canadell, J. G., Ciais, P., . . . Zuidema, P. A. (2021). Integrating the evidence for a terrestrial carbon sink caused by increasing atmospheric CO 2. *New Phytologist*, *229*(5), 2413–2445. https://doi.org/10.1111/nph.16866

Walker, W. S., Gorelik, S. R., Cook-Patton, S. C., Baccini, A., Farina, M. K., Solvik, K. K., Ellis, P. W., Sanderman, J., Houghton, R. A., Leavitt, S. M., Schwalm, C. R., & Griscom, B. W. (2022). The global potential for increased storage of carbon on land. *Proceedings of the National Academy of Sciences*, *119*(23), 1–12. https://doi.org/10.1073/pnas.2111312119

Wang, S., Zhang, Y., Ju, W., Chen, J. M., Ciais, P., Cescatti, A., Sardans, J., Janssens, I. A., Wu, M., Berry, J. A., Campbell, E., Fernández-Martínez, M., Alkama, R., Sitch, S., Friedlingstein, P., Smith, W. K., Yuan, W., He, W., Lombardozzi, D., . . . Peñuelas, J. (2020). Recent global decline of CO 2 fertilization effects on vegetation photosynthesis. *Science*, *370*(6522), 1295–1300. https://doi.org/10.1126/science.abb7772

Wang, Z., & Zeng, X. (2010). Evaluation of snow albedo in land models for weather and climate studies. *Journal of Applied Meteorology and Climatology*, *49*(3), 363–380. https://doi.org/10.1175/2009JAMC2134.1

Warren, R., Price, J., Graham, E., Forstenhaeusler, N., & VanDerWal, J. (2018). The projected effect on insects, vertebrates, and plants of limiting global warming to 1.5°C rather than 2°C. *Science*, *360*(6390), 791–795. https://doi.org/10.1126/science.aar3646

Warszawski, L., Frieler, K., Huber, V., Piontek, F., Serdeczny, O., & Schewe, J. (2014). The Inter-Sectoral Impact Model Intercomparison Project (ISI–MIP): Project framework. *Proceedings of the National Academy of Sciences*, *111*(9), 3228–3232. https://doi.org/10.1073/pnas.1312330110

WEF. (2020). Nature Risk Rising: Why the Crisis Engulfing Nature Matters for Business and the Economy. *New Nature Economy*, (January), 36. http://www3.weforum.org/docs/WEF_New_Nature_Economy_Report_2020.pdf

Weisser, W. W., Roscher, C., Meyer, S. T., Ebeling, A., Luo, G., Allan, E., Beßler, H., Barnard, R. L., Buchmann, N., Buscot, F., Engels, C., Fischer, C., Fischer, M., Gessler, A., Gleixner, G., Halle, S., Hildebrandt, A., Hillebrand, H., de Kroon, H., . . . Eisenhauer, N. (2017). Biodiversity effects on ecosystem functioning in a 15-year grassland experiment: Patterns, mechanisms, and open questions. *Basic and Applied Ecology*, *23*, 1–73. https://doi.org/10.1016/j.baae.2017.06.002

Wells, G., Ryan, C., Fisher, J., & Corbera, E. (2020). In defence of simplified PES designs. *Nature Sustainability*, *3*(6), 426–427. https://doi.org/10.1038/s41893-020-0544-3

West, P. W. (2014). *Growing Plantation Forests* (Vol. 9783319018). Springer International Publishing. https://doi.org/10.1007/978-3-319-01827-0

Winckler, J., Lejeune, Q., Reick, C. H., & Pongratz, J. (2019). Nonlocal Effects Dominate the Global Mean Surface Temperature Response to the Biogeophysical Effects of Deforestation. *Geophysical Research Letters*, *46*(2), 745–755. https://doi.org/10.1029/2018GL080211

Winkler, A. J., Myneni, R. B., Hannart, A., Sitch, S., Haverd, V., Lombardozzi, D., Arora, V. K., Pongratz, J., Nabel, J. E. M. S., Goll, D. S., Kato, E., Tian, H., Arneth, A., Friedlingstein, P., Jain, A. K., Zaehle, S., & Brovkin, V. (2021). Slowdown of the greening trend in natural vegetation with further rise in atmospheric CO2. *Biogeosciences*, *18*(17), 4985–5010. https://doi.org/10.5194/bg-18-4985-2021

Wirth, S. B., Poyda, A., Taube, F., Tietjen, B., Müller, C., Thonicke, K., Linstädter, A., Behn, K., & Rolinski, S. (2023). Connecting CSR theory and LPJmL 5.3 to assess the role of environmental conditions, management and functional diversity for grassland ecosystem functions. (April). https://doi.org/10.5194/bg-2023-55

Wolf, T., Untergutsch, A., Wensing, C., Dr. Mittelbach, H., Pagenkopf, F.-L., Kellenberger, D., & Kubowitz, P. (2020). *Potenziale von Bauen mit Holz* (tech. rep.).

Wunder, S., Brouwer, R., Engel, S., Ezzine-De-Blas, D., Muradian, R., Pascual, U., & Pinto, R. (2018). From principles to practice in paying for nature's services. *Nature Sustainability*, *1*(3), 145–150. https://doi.org/10.1038/s41893-018-0036-x

Wunder, S., Brouwer, R., Engel, S., Ezzine-de-Blas, D., Muradian, R., Pascual, U., & Pinto, R. (2020). Reply to: In defence of simplified PES designs. *Nature Sustainability*, *3*(6), 428–429. https://doi.org/10.1038/s41893-020-0545-2

Xu, C., Christoffersen, B., Robbins, Z., Knox, R., Fisher, R. A., Chitra-Tarak, R., Slot, M., Solander, K., Kueppers, L., Koven, C., & McDowell, N. (2023). Quantification of hydraulic trait control on plant hydrodynamics and risk of hydraulic failure within a demographic structured vegetation model in a tropical forest (FATES–HYDRO V1.0). *Geoscientific Model Development*, *16*(21), 6267–6283. https://doi.org/10.5194/gmd-16-6267-2023

Xu, S., Wang, R., Gasser, T., Ciais, P., Peñuelas, J., Balkanski, Y., Boucher, O., Janssens, I. A., Sardans, J., Clark, J. H., Cao, J., Xing, X., Chen, J., Wang, L., Tang, X., & Zhang, R. (2022). Delayed use of bioenergy crops might threaten climate and food security. *Nature*, *609*(7926), 299–306. https://doi.org/10.1038/s41586-022-05055-8

Xu, X., Medvigy, D., Powers, J. S., Becknell, J. M., & Guan, K. (2016). Diversity in plant hydraulic traits explains seasonal and inter-annual variations of vegetation dynamics in seasonally dry tropical forests. *The New phytologist*, *212*(1), 80–95. https://doi.org/10.1111/nph.14009

Yao, Y., Joetzjer, E., Ciais, P., Viovy, N., Cresto Aleina, F., Chave, J., Sack, L., Bartlett, M., Meir, P., Fisher, R., & Luyssaert, S. (2022). Forest fluxes and mortality response to drought: model description (ORCHIDEE-CAN-NHA r7236) and evaluation at the Caxiuanã drought experiment. *Geoscientific Model Development*, *15*(20), 7809–7833. https://doi.org/10.5194/gmd-15-7809-2022

Zhang, Z., Babst, F., Bellassen, V., Frank, D., Launois, T., Tan, K., Ciais, P., & Poulter, B. (2018). Converging Climate Sensitivities of European Forests Between Observed Radial Tree Growth and Vegetation Models. *Ecosystems*, *21*(3), 410–425. https://doi.org/10.1007/s10021-017-0157-5

Zobler, L. (1986). *A world soil file for global climate modeling. NASA TM-87802* (tech. rep.). NASA. Washington, DC.

A Appendix

- Paper 1 (accepted): Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests
- Paper 2 (published): Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate
- Paper 3 (draft): Reconciling climate-smart forestry with EU forest, biodiversity and climate legislation

RESEARCH

Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests

Konstantin Gregor[1*], Andreas Krause[1], Christopher P. O. Reyer[2], Thomas Knoke[1], Benjamin F. Meyer[1], Susanne Suvanto[3,4,5] and Anja Rammig[1]

Correspondence:
konstantin.gregor@tum.de
[1] TUM School of Life Sciences, Technical University of Munich, Freising, Germany
Full list of author information is available at the end of the article

Abstract

Background: Forests mitigate climate change by reducing atmospheric CO_2-concentrations through the carbon sink in the forest and in wood products, and substitution effects when wood products replace carbon-intensive materials and fuels. Quantifying the carbon mitigation potential of forests is highly challenging due to the influence of multiple important factors such as forest age and type, climate change and associated natural disturbances, harvest intensities, wood usage patterns, salvage logging practices, and the carbon-intensity of substituted products. Here, we developed a framework to quantify the impact of these factors through factorial simulation experiments with an ecosystem model at the example of central European (Bavarian) forests.

Results: Our simulations showed higher mitigation potentials of young forests compared to mature forests, and similar ones in broad-leaved and needle-leaved forests. Long-lived wood products significantly contributed to mitigation, particularly in needle-leaved forests due to their wood product portfolio, and increased material usage of wood showed considerable climate benefits. Consequently, the ongoing conversion of needle-leaved to more broad-leaved forests should be accompanied by the promotion of long-lived products from broad-leaved species to maintain the product sink. Climate change (especially increasing disturbances) and decarbonization were among the most critical factors influencing mitigation potentials and introduced substantial uncertainty. Nevertheless, until 2050 this uncertainty was narrow enough to derive robust findings. For instance, reducing harvest intensities enhanced the carbon sink in our simulations, but diminished substitution effects, leading to a decreased total mitigation potential until 2050. However, when considering longer time horizons (i.e. until 2100), substitution effects became low enough in our simulations due to expected decarbonization such that decreasing harvests often seemed the more favorable solution.

Conclusion: Our results underscore the need to tailor mitigation strategies to the specific conditions of different forest sites. Furthermore, considering substitution effects, and thoroughly assessing the amount of avoided emissions by using wood products, is critical to determine mitigation potentials. While short-term recommendations are possible, we suggest risk diversification and methodologies like robust optimization to address increasing uncertainties from climate change and decarbonization paces past 2050. Finally, curbing emissions reduces the threat of climate change on forests, safeguarding their carbon sink and ecosystem services.

Keywords: Climate change; carbon mitigation; forest; substitution effect; displacement factor; decarbonization; disturbance; salvage logging; wood usage

Background

Forests are pivotal in the fight against global climate change due to their significant role in the global carbon cycle [1, 2]. Most obviously, forests mitigate climate change through the in situ forest carbon sink which sequestered 2.4 PgC yr^{-1} or roughly 22% of anthropogenic CO_2-emissions in recent decades [3, 4]. However, in places like Europe, where about 75% of forests are used for wood production provision [5], at least two other aspects are highly relevant. One is the wood product carbon sink which in Europe accounts for about 13% of the forest carbon sink, the other is the substitution of carbon-intensive fuels and materials with wood products [6]. In Europe, this currently has a similar magnitude as the combined carbon sink [7, but see our remarks on this in sections *Model evaluation* and *Decarbonization*].

The exact magnitudes of the forest and wood product carbon sinks and the substitution effects, however, depend on a multitude of factors. The forest sink depends on the forest type and age, forest management, climate change, and natural disturbances. For example, the European forest sink has been shown to be in decline due to ageing and increased disturbances [8, 9]. The wood product sink depends on the same factors, but in addition, the product types and their lifetimes play a critical role. Moreover, because of more frequent disturbances, an increasing amount of timber from salvage logging affects wood quality and availability [10]. Finally, substitution effects depend on the carbon intensity of the replaced products. Though highly important, studies have suggested that these effects may have previously been overestimated, partly because the replaced materials and fuels will likely become less carbon-intensive in the future, e.g., due to a different energy mix [11, 12]. In construction, for example, wood can replace concrete and steel which recently accounted for roughly 14% of global CO_2-emissions [13, 14]. But also other materials such as plastics or aluminum can be substituted with wood with a carbon benefit [15]. The decarbonization of these materials has already been initiated, for instance through increased efficiencies and recycling, but also through the adoption of existing technologies and investments in innovation [13, 14, 16]. The exact speed of this decarbonization, however, remains uncertain.

On the other hand, new technologies in the wood industry offer the possibility of enhanced wood use in the construction sector [17–19]. This calls for investigating the climate impact of such enhanced wood usage. Several studies have already found mitigation benefits of increased long-term usage of wood in various regions [20–23]. However, these studies did not yet consider in full detail to what degree the impact of forest structure, climatic change, disturbance regimes, and changes to the substitution dynamics might affect their results.

The complexity of determining the carbon mitigation potential of forests has resulted in a debate over the role of forests and wood products as a natural climate solution [24]. While some studies indicate a mitigation benefit of stable or increased harvest intensities [25, 26] other studies highlight the potential of decreased harvest intensities for increased carbon sequestration [9, 23, 27]. Verkerk et al. [9] also recen/tly highlighted how different forest-based mitigation measures might conflict with each other.

To address this complexity, we used a model-based factorial experiment with a well-established process-based ecosystem model to set up a framework for quantifying the impact of all previously introduced factors: forest age, forest type, climate change, nitrogen (N) deposition, disturbances, harvest intensity and salvage logging, wood usage patterns, and changes in the carbon-intensity of substituted products. Using forests in the state of Bavaria in central Europe as an example, we quantified the impact of each factor independently, as well as their interactions and uncertainties. We then contextualized the findings of other mitigation studies and discussed how our results may be used as groundwork for developing mitigation strategies.

Methods

Description of the process-based ecosystem model LPJ-GUESS

For our simulations, we used the process-based ecosystem model LPJ-GUESS v4.1. The model is driven by environmental conditions (temperature, precipitation, shortwave radiation, atmospheric CO_2, nitrogen deposition) and models a detailed forest structure via cohorts of different age classes. LPJ-GUESS simulates photosynthesis, allocation, growth, competition, nutrient limitation, establishment, and mortality of plant functional types [28, 29]. These are represented by parameters governing phenology, growth, drought and shade tolerance, bioclimatic limits for establishment and mortality, and others. For each forest location, a number of replicate patches (we used 100) are simulated. These depict random samples of forests at different stages after disturbances. Disturbances are modeled stochastically, killing the entire vegetation of a patch. These represent stand-replacing disturbances such as windthrows or insect infestations, the main disturbance agents in central Europe [30, 31]. The other mortality mechanisms such as growth efficiency mortality and age-related mortality kill fractions of the cohorts. Dead biomass is moved to various litter pools and the litter and soil carbon-nitrogen dynamics are simulated following the CENTURY model [29, 32].

LPJ-GUESS contains a forest management module that allows detailed representation of forestry, including thinnings and partial harvests, clearcuts, wood products and their decay, residue outtake, re-establishment and planting [33]. A full description of LPJ-GUESS can be found in Smith et al. [29].

Modeling protocol

We conducted a simulation experiment for the federal state of Bavaria, Germany, in central Europe. We selected this region because of the detailed data availability on forest structure, harvests, wood usage, and product pools [e.g., 34]. For computational reasons, we selected five grid cells, covering the differences in regional climate (Fig. 1). All of these grid cells contained both needle-leaved evergreen (NE) and broad-leaved deciduous (BD) forests in 2018 according to the CORINE land cover data [35]. We used the plant functional types of shade-tolerant broad-leaved summergreen trees and shade-tolerant needle-leaved evergreen trees as a representation of the most dominant tree species in Bavaria. We used the default parameters as in Smith et al. [29] but used C/N ratios of sapwood and fine roots of 32 (53) and 661 (373) for the needle-leaved (broad-leaved) species, respectively [36, 37]. This was necessary to make LPJ-GUESS, which is calibrated mostly for global or continental applications, capture the high productivity of forests in central Europe (see section *Model evaluation*).

For the climate input, we used daily values of temperature, precipitation, and shortwave radiation from the regional climate model RACMO driven by the global climate model EC-EARTH [38, 39] from the EURO-CORDEX project [40]. The data were bias-corrected using quantile mapping and statistically down-scaled from 12.5 to 5 km resolution [41]. The climate data was available from 1951 to 2100. We initialized the model with a 1200 year spin-up period by recycling climate data from 1951-1980 to bring the carbon pools close to equilibrium. Yearly CO_2 concentrations and decadal values for nitrogen deposition were taken from Lamarque et al. [42].

We considered forest-type-specific disturbance intervals, i.e. we set the disturbance interval to 300 years for needle-leaved and 1000 years for broad-leaved forests [43]. Additionally, we assumed three different scenarios of disturbances in the future (see below). Fire was not explicitly simulated in this study since it is contained in the disturbance intervals and also not an important disturbance agent in the studied region [10, 30].

For our simulation experiment, we varied the main factors that govern the forest and product carbon sink and substitution effects: forest age, forest type, harvest intensity, salvage logging, wood usage patterns, climate change and nitrogen deposition, disturbances, and decarbonization of substituted products, as described in detail below. We simulated all 3456 possible combinations of the factors (Table 1).

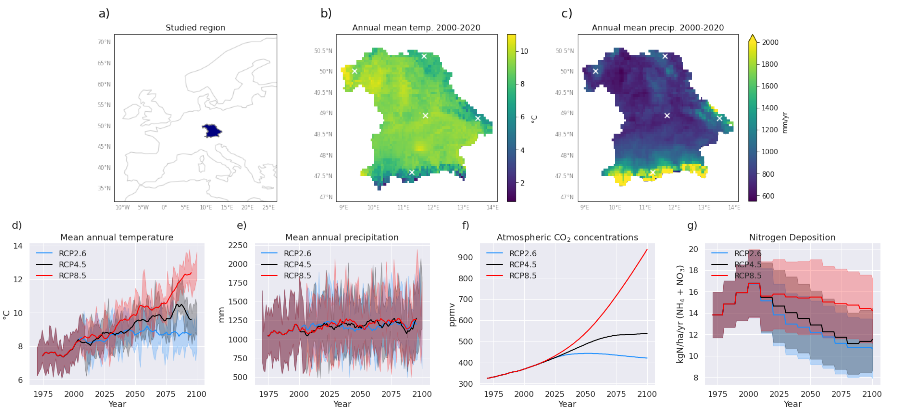

Figure 1 The study region is Bavaria, located in central Europe (a), where we considered five sites (white crosses) for our simulation experiment. Mean annual temperature and precipitation are shown in panels (b) and (c). The forcing data (d-g) used as input for LPJ-GUESS shows 10-year rolling means averaged over the five sites with standard deviation bands for three emission scenarios (RCPs).

Table 1 The considered values of the factors used in this study. All possible combinations were simulated, leading to $3 \times 3 \times 2 \times 2 \times 4 \times 2 \times 2 \times 2 \times 3 = 3456$ simulations.
($*$) Note that we used the exponential increase as the default in our analyses unless stated otherwise.

Factor	Values	Comment
Climate change and N deposition	RCP2.6, RCP4.5, RCP8.5	See Fig. 1
Disturbance probability change ($*$)	constant, linear, exponential	changes in disturbance frequency based on temperature anomaly (Fig. S1)
Forest age	mature, young	planted between 1921 and 1940, or between 1981 and 2000, respectively (Fig. S2)
Forest type	BD, NE	broad-leaved deciduous, needle-leaved evergreen forests
Harvest intensity	0%, 50%, 100%, 150%	direct change in harvest intensity starting after 2020 compared to current values
Salvage logging	yes, no	after every disturbance after 2020
Material wood usage	100%, 150%	the increase to 150% was implemented as a linear change from 2020 until 2050 at the expense of short-lived products and firewood
Cascade usage	100%, 150%	the change to 150% was implemented as a direct change of the lifetime of products created after 2020
Decarbonization in 2050	25%, 50%, 75%	exponential decrease based on Schreyer et al. [44], reaching the given percentage value in 2050 (Fig. S5)

Considered factors for the carbon mitigation potential of forests

Climate change

To consider the full range of potential climate pathways and to make our assessment as broad as possible, we ran the simulations for the low-warming scenario RCP2.6, the intermediate scenario RCP4.5, and the high-warming scenario RCP8.5. This encompasses the projections of climate change according to current trends (2.1 to 3.9 °C global warming by 2100, 90% confidence interval [45]). Assessing wide ranges of climate change scenarios in preparation for low likelihood outcomes remains important, particularly given the remaining uncertainty of positive climate feedbacks [46, 47].

Disturbances

We implemented three scenarios of future changes to disturbance probabilities. Throughout this paper, we focused on an exponential increase in disturbance rates with temperature, based on recent observations of increased disturbances in Germany [10, Fig. S1]. For an additional assessment, we included a linear increase and constant rates.

Forest age

We investigated the impact of "young" and "mature" forests. For this, forests were planted in each model run yearly over the 1981-2000 and the 1921-1940 period, respectively. The implemented harvesting (see below) allowed for the continuous establishment of young trees, resulting in structured, uneven-aged forest stands (Fig. S2).

Forest type

In the plantings of forests in 1921-1940 and 1981-2000, only the broad-leaved or the needle-leaved plant functional type were planted to model BD and NE forests separately, depending on the model run (Table 1).

Harvest intensity

We simulated partial timber harvests to occur on average every 20 (NE) and 25 (BD) years with an intensity of 24% for all age cohorts and species, i.e. 24% of the trees of each age cohort were harvested. These are average numbers derived from recent German forest inventory data [48]. This setup resulted in stable growing stocks and describes a forward-looking forest management strategy [49, 50]. For the alternative scenarios, harvest intensity was then either decreased or increased by 50% starting directly after 2020, while the harvest intervals remained constant. When increasing the harvest intensity, stable growing stocks could no longer be guaranteed. We furthermore included a scenario without wood harvests after 2020.

During harvest events, 65% of the sapwood and heartwood of a selected tree were considered stem material of which 90% ("harvest efficiency") were removed. For trees older than 20 years, the harvested stem wood was distributed to different product pools based on forestry statistics as described in section *Wood products and substitution effects*. The rest of the stem wood, all coarse roots and 60% of the branches (13% of the woody biomass are considered branches) were assumed to be left to decay on-site. 40% of branches were assumed to be removed and burned [33, 51]. Trees younger than 20 years were completely used as firewood [52].

Salvage logging

We ran simulations both with and without salvage logging after a disturbance, taking into account the increased difficulty of harvest. For this, we assumed that 20% of the affected trees were left on site [53] and lowered the harvest efficiency, i.e., only 75% of a harvested stem was removed from the forest in a salvaging operation. We used the same usage patterns for salvaged wood as for fresh wood, because German wood use statistics indicate no substantial change in such patterns (Fig. S3).

Wood products and substitution effects

For needle-leaved (broad-leaved) trees, 37% (6%) of harvested wood went into a long-lived product pool, 17% (34%) into a medium-lived pool, 36% (25%) into a short-lived pool; the rest was considered fuel wood and was returned to the atmosphere in the same year, following Klein et al. [54] and Krause, Knoke, and Rammig [26]. The decay of the products was modeled with Gamma-functions, such that 50% of the products have decayed after 3, 18, and 93 years for the short, medium, and long-lived pools, respectively [54]. To assess the effect of changes in material usage on the mitigation potential, we increased the fractions for medium- and long-lived products by 50% at the expense of short-lived products and firewood. This change in usage was implemented gradually from 2020 until 2050, based on current trends of construction wood usage in Germany [19]. To investigate the impact of more cascading (i.e. longer usage, more recycling), we also increased the residence time of all products by 50% by increasing the shape-parameter of the Gamma-functions (Fig. S4) for products created after 2020.

For the substitution effects of wood products, we assumed a displacement factor (DF, avoided emissions in relation to the mass of carbon in the wood product [55]) of 1.5 tC/tC for material substitution and 0.67 tC/tC for fuel substitution [15]. Note that this DF < 1 means that if 1 tC of wood is harvested and used as fuel completely, the emission to the atmosphere is 1 tC and with a different energy source 0.67 tC would be emitted. The material DF does not contain end-of-life handling of the products. Since in Germany only 2% of waste is landfilled and wood is not allowed to be landfilled, we assumed full energy recovery of all products using again the DF for fuel.

Decarbonization

To assess the impacts of different decarbonization paces of the substituted materials and fuels [12, 56], we used three exponential functions to gradually decrease the DFs by 25%, 50%, and 75% of today's values in 2050 (Fig. S5). For instance, in the 75% scenario, the substitution factor of 1.5 tC/tC became 0.375 tC/tC in 2050, and approached zero around 2100, meaning that at that time, the product had no more substitution effect at all. We used this approach because the 25% and 75% scenarios closely match recent projections for the EU's carbon intensity for current policies and net-zero targets, respectively [44].

It is important to note that throughout the text, we refer to the decarbonization of the *local* economy and viewed it as independent of the emissions of the rest of the world (defined here through the RCPs). For instance, RCP8.5 combined with

Table 2 Yearly mitigation values in gC/m^2/yr from this study and the literature. Literature values were converted from $MtCO_2e$ to gC per m^2 forest area. The values for Grassi et al. [6] represent those of forest remaining forest. Note that the substitution values are theoretical values only: A substitution effect was attributed to the entire wood production. Such a value can sensibly only be used in comparison to a baseline scenario. Values that are not comparable to this study due to differences in accounting were excluded.

Source	Bavaria 2020-2025 This study	Germany 2014 Wolf et al. [19]	Europe 2018 Holmgren [7]	Europe 2016-2018 Grassi et al. [6]	Bavaria 2003-2008 Klein and Schulz [34]
Forest Sink	138 (mature only: 104)	148	63	49	138 (veg. carbon only)
Product Sink	15 (16)	8	6	6	50
Fuel Substitution	27 (35)	92	25 (includes paper)		
Material Substitution	32 (43)	77	38 (including energy end-use)		
Total Substitution	59 (78)	169	63		130
Total Forest Mitigation	212 (198)	325	132		318

75% decarbonization can be viewed as an edge case where the considered region implements strong mitigation policies, but the rest of the world misses current global mitigation targets and increases their reliance on fossil fuels.

Assessment of the carbon mitigation potential

In our study, we investigated the impact of the aforementioned factors on the forests' carbon mitigation potential. For this, we computed the combined carbon sink, i.e., the change in carbon stored in the forest (live and dead biomass, and soil) and in products between 2020 and 2100. Furthermore, we computed the total carbon mitigation potential defined as the carbon sink plus the cumulative avoided emissions from substitution between 2020 and 2100. We averaged over the grid cells and focused on the mid- and long-term mitigation potential (years 2050 and 2100).

We based our main analysis on the case of an exponential increase in disturbance probability and then, to assess the impact of each factor on the mitigation potential, we computed pair-wise differences within the simulations. We repeated this experiment for the two other assumptions on disturbance frequencies. For instance, to quantify the impact of salvage logging, we subtracted the mitigation potential of each simulation where salvage logging was disabled from the mitigation potential of its "partner simulation" where salvage logging was enabled but all other settings were the same. For all factors with two possible values, this led to 576 comparisons (e.g., using the 1152 simulations with the exponential disturbance scenario, there were 576 simulations with salvage logging, 576 simulations without). For increased harvest intensity, it led to 288 comparisons between the 288 simulations with 150% harvest intensity and the 288 simulations with 100% harvest intensity. The 576 simulations with 0% and 50% harvest intensity were ignored in this particular assessment (but used to assess the implications of a decrease in harvest intensity). Accordingly, 384 comparisons were available for each of the three decarbonization paces. For increased material usage, we compared the simulations with both increased shares of long-lived products and increased cascading to those of the default values.

Our approach gives an insight into the potential impact of, e.g., a change in harvest intensity, while also considering the uncertainty stemming from all other factors, e.g., climate change. This helps to disentangle and compare all considered driving factors.

Results

Model evaluation

The simulations with default settings (current harvest intensities, wood usage, DFs, and salvage logging practices) on average (over forest types, ages, and RCPs) and in relative terms are in the range of current literature estimates (Table 2). Simulated forest productivity also resembled independent estimates of Bavarian forests: For 2000-2015, GPP and NPP in our study were on average simulated as 1492 and 714 $gC/m^2/yr$, respectively, close to satellite-based estimates of 1444 and 687 $gC/m^2/yr$, respectively [57–59]. Literature estimates of the present-day product sink for European regions range around 5 to 12% of the forest sink (vegetation, soil, and litter), but higher values have also been estimated for Bavaria, where forests are heavily managed and dominated by conifers providing long-lived products (Table 2). Our simulations resulted in a product sink of 11% of the forest sink. We simulated the total *theoretical* substitution effects to be 28% of the total mitigation potential (theoretical because they are computed without comparison to a baseline, see discussion section *Decarbonization*). Literature values range between 41 and 52%. Key reasons for the spread are the assumed substitution factors but also the forest age structure, changes in forest area, and wood usage patterns of the considered regions. The main reason for our low magnitude of substitution effects is that we considered only young and mature forests, with young forests having very low substitution effects due to their low volumes. For mature forests only, substitution effects accounted for 39% of the total carbon mitigation. Nevertheless, our setup resulted in comparable mitigation dynamics as estimated by these studies which increases confidence in our calculations of the mitigation potential of central European forests. The wide range of estimates in the literature, however, also underscores the relevance of our study.

Due to continuous cuttings and regeneration, the age structure of the mature forests was already quite diverse in 2020 with mean tree ages of 72 (NE) and 84 years (BD). Young forests were still quite homogeneous in 2020 with a mean tree age of 28 years (Fig. S2). The density of the BD forests was lower which was mostly driven by many small trees in the NE forests ($\leq$ 20cm).

Impacts of different factors on the mitigation potential

NE forests had slightly higher mitigation potentials than their BD counterparts, and young forests had a considerably higher mitigation potential than their mature counterparts (Fig. 2). Figure 3 shows a summary of the impacts of all considered factors on the carbon sink and total mitigation potential while Figures 4 and S8 show these impacts in more detail. In general, the impact on the carbon sink was often different from that on the total mitigation potential. The impact of the factors on the different forest types was rather similar, but in a few cases the effect had a different magnitude, e.g., changes in harvest intensities had a stronger effect on mature BD forests than on mature NE forests (Fig. S6). Also the time frame mattered: For instance, in the medium term (until 2050) decreasing or stopping harvests generally increased carbon storage but had a negative effect on the total mitigation potential. In the long term (until 2100), however, the total mitigation potential increased for a substantial proportion of simulations for stopping harvests. In general, the spread in mitigation potentials increased with time, due to increasing uncertainty about climate, disturbances, and decarbonization.

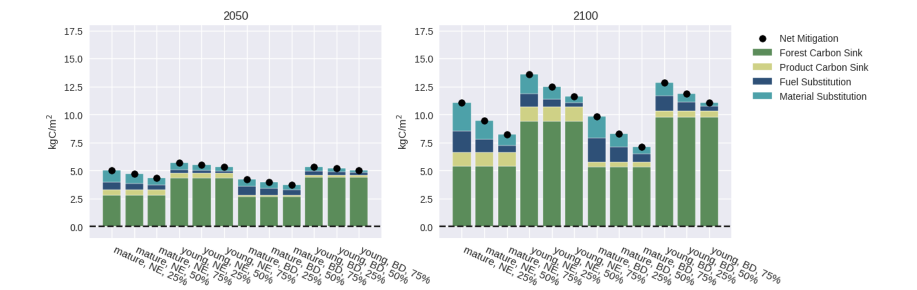

Figure 2 Contribution of each of the three aspects of mitigation to the total mitigation potential. Depicted is the situation for RCP4.5, without salvage logging and with harvest intensity, usage, and cascading at 100%. The different bars correspond to different combinations of forest age, forest type, and decarbonization pace. The dots represent total net mitigation and are the same values as the corresponding dots in Figures 4 and S8.

Forest type

The simulated carbon stocks and sinks were rather similar between NE and BD forests, ranging around 5.5 kgC/m^2 for mature forests until 2100 under RCP4.5 (Figs. 2b+S7). The product sink was much smaller for BD forests (0.4 kgC/m^2 compared to NE's 1.2 kgC/m^2 for mature forests until 2100, Fig. 2), because hardwood is rarely used for long-lived products, unlike softwood. Substitution effects were slightly higher for NE forests, but the contribution of fuel substitution was larger for BD forests. Furthermore, towards the end of the century, disturbances had a pronounced effect on NE forests, leading to lower vegetation carbon stocks. This was, however, counterbalanced by their larger wood product pools (Fig. S7).

Forest age

Forest age also affected the mitigation potential. Young forests in the simulations provided a larger carbon sink (about 9.7 kgC/m^2 until 2100) than mature ones (about 5.5 kgC/m^2 until 2100). However, they provided a different product portfolio, leading to a smaller share of long-lived products, particularly in NE forests. Until 2050, the product sink was simulated as 8% of the forest sink for young NE forests compared to 15% for mature ones, while this share was 4% and 5% for both young and mature BD forests, respectively. Substitution effects were also smaller in young forests (Fig. 2).

Climate change

The impacts of climate change, including the related exponential increase in disturbance probability based on temperature anomaly, did not fully materialize by 2050, leading to a narrow spread in the changes of total mitigation potentials (Fig. 3). More warming (RCP8.5 instead of RCP4.5) changed the total mitigation potentials by -0.1 to 0.1 kgC/m^2, while less warming (RCP2.6 instead of RCP4.5) changed them by -0.2 to 0.1 kgC/m^2.

In contrast, until 2100, the climate change scenarios exhibited a wider spread in the differences in mitigation potentials: less warming yielded -0.6 to +1.6 kgC/m^2 while more warming led to a decrease in mitigation potentials of 0.0 to -2.6 kgC/m^2. The interaction between climate change and forest types was significant, with a

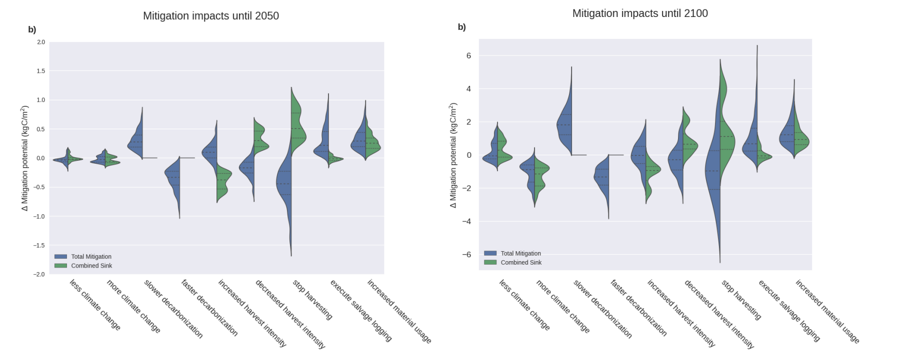

Figure 3 Differences in mitigation potential when changing each considered factor for the time until 2050 (a) and 2100 (b, note the different scales). Data is based on the assumption of exponential increases in disturbance probability. Each violin plot is created by computing pair-wise differences between simulations with a change in one factor, see section *Assessment of the carbon mitigation potential*. The split densities show the change in total mitigation potential including substitution effects (blue) and the change in the combined carbon sink only (forest and products). Positive values denote that a higher mitigation potential was simulated compared to its "partner simulation". In the carbon sink case, this means that it was either a larger carbon sink or a smaller carbon source compared to the "partner simulation". Note that a negative value does not mean that it was a carbon source. The carbon sink was only smaller than in the "partner simulation". "More climate change" means subtracting the values of RCP4.5 from the values of RCP8.5, while "less climate change" means subtracting the values of RCP4.5 from the values of RCP2.6. Similarly for decarbonization where the changes to the intermediate 50% decarbonization pace were computed.

stronger response in mitigation potential for NE forests compared to BD forests (e.g., compare Fig. S6 f and h).

Disturbances

The assumed temperature-related disturbance scenario had a substantial impact. In the default exponential case, a change from RCP4.5 to RCP8.5 led to a change in total mitigation potential by 2100 of -1.2 kgC/m^2 in the median (Fig. 3). In the linear case, however, the median change until 2100 was only -0.3 kgC/m^2, and in the constant case, -0.1 kgC/m^2 (Figs. S9 and S10). In the constant case, changing from RCP4.5 to RCP2.6 even had strictly negative impacts on the mitigation potential until 2100, indicating positive effects from climate change when no changes in disturbances are assumed.

When focusing on RCP4.5, our results showed a 31% higher carbon sink in mature NE forests by 2100 for simulations without increasing disturbances compared to those with an exponential increase (Fig. S11). The remaining results remained largely unaffected by the disturbance scenario except for the impacts of salvage logging practices, and minor changes in the effects of decreasing or increasing harvests.

Decarbonization

The pace of decarbonization had no effect on the carbon sinks. Consequently, the densities in Fig. 3 are single points, also because we ignored emissions from forestry operations and considered the local decarbonization independent of global emissions. However, it heavily affected the substitution effects: Until 2050, they contributed to 9%-34% of the total mitigation potential for the 100% harvest scenario, depending on the decarbonization pace and the forest type (Fig. 2). Until 2100, faster decarbonization (75% instead of 50% by 2050) decreased the total mitigation potential by 1.3 kgC/m^2 in the median. Similarly, slower decarbonization (25% instead of 50%) by far had the highest positive mitigation impact, with +1.8 kgC/m^2 in the median.

Finally it is notable that by 2100, the absolute impact of slower decarbonization exceeded that of faster decarbonization. This is because, by 2100, the carbon intensity in both the 50% and 75% decarbonization scenario was rather low (3% and 16% of today's value, respectively), while it remained high in the 25% scenario (47%, see Fig. S5).

Changes in management

Increasing the harvest intensity had strictly negative effects on the combined carbon sink (-0.2 to -0.7 kgC/m^2 until 2050 and -0.0 to -2.5 kgC/m^2 until 2100, Figs. 3 and 4) while decreases in harvest intensity had opposite outcomes. This effect was higher for BD forests than NE forests and for mature compared to young forests (Fig. S6). Changing the harvest intensity heavily affected the importance of the product sink. Higher harvests increased the importance of the product sink, contributing over 30% of the combined sink for mature NE forests in both time frames (Fig. S12).

The influence of such changes on the total mitigation potential, however, was different. Until 2050, the change in total mitigation potential driven by decreased harvests was mostly negative, ranging from -0.7 to 0.2 kgC/m^2. Until 2100 it spanned -2.4 to +1.8 kgC/m^2. This was driven by substitution effects, dependent on the decarbonization pace. For increased harvests, it was slightly more negative than positive: -1.9 to 1.6 kgC/m^2 until 2100, with the beneficial cases in the highest carbon-intensity scenario (i.e. 25% decarbonization). But until 2050, there were more instances where increased harvests were beneficial. Regardless of the climate change scenario, benefits of decreased harvests until 2100 almost exclusively occurred in the 75% decarbonization scenario and when there was no increased material usage (Fig. S13). Note that this result was largely independent from the disturbance assumption S14).

Stopping management completely in 2020 increased the combined sink by 0.2 kgC/m^2 to 1.0 kgC/m^2 until 2050, with BD forests showing a larger resulting carbon sink compared to NE forests (Fig. S6). Until 2100, the combined sink increased by 0.3 kgC/m^2 to 2.8 kgC/m^2. With stronger climate change and increasing time, this positive impact sink decreased, especially for NE forests. In a few cases with NE forests, strong climate change and otherwise high material usage, stopping harvests led to a negative impact on the carbon sink. The impact on the total mitigation potential was almost exclusively negative until 2050, but spread between -5.1 and 3.4 kgC/m^2 until 2100.

Salvage logging

Salvage logging after disturbances had small effects on the combined carbon sink (-0.1 to 0.1 kgC/m^2 until 2050, -0.7 to 1.1 kgC/m^2 until 2100, Fig. 3). The sign of the effects depended on future climatic conditions (temperature and moisture affecting how fast deadwood decays) and usage patterns (depending on whether the

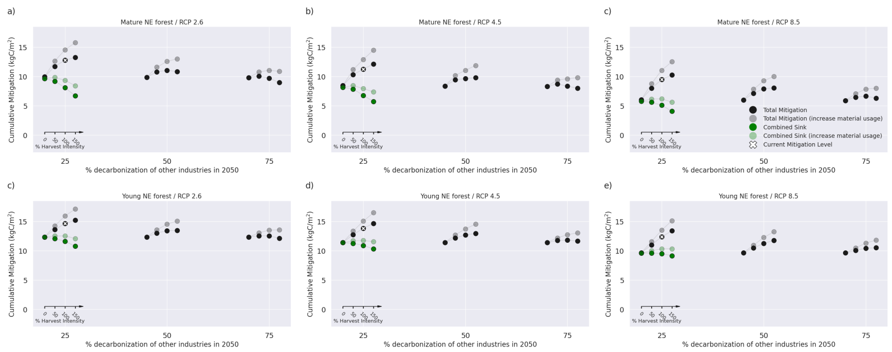

Figure 4 Carbon mitigation potential from 2020 to 2100 for needleleaved evergreen forests. The effects of harvest intensity, wood usage, and decarbonization of substituted materials on the carbon sink and total mitigation potential are displayed. Each group of dots represents one usage scenario and four harvest intensity scenarios (0%, 50%, 100%, and 150%). Black dots represent total mitigation, green dots represent the carbon sink (including products). The darker dots represent current wood usage patterns while the lighter dots refer to simulations with a 50% increase in both material and cascade usage. In low decarbonization scenarios, the substitution effect is so high that increased harvest intensity leads to higher mitigation (black dots). This is not the case when substituted materials become "greener". The combined carbon sink becomes lower with higher management (including the product sink, as depicted here in green). Shown are the simulations of the results without salvage logging.

residence time of products exceeded that of on-site deadwood). The effects on the total mitigation potential were consistently positive until 2050 and predominantly positive until 2100. Notably, assuming constant disturbance rates decreased the magnitude of the impact. but the generally positive impact remained, regardless of the disturbance scenario (Fig. S10).

Increased material usage and cascading

As expected, an increase in the material usage of wood (i.e., higher shares of long-lived products and more cascade usage) consistently had a positive impact on both carbon sink and total mitigation. The influence of the combination of the two measures on the combined carbon sink was between 0.1 and 0.6 $\mathrm{kgC/m^2}$ until 2050 and 0.3 and 2.8 $\mathrm{kgC/m^2}$ until 2100. The effect on the total mitigation potential was more pronounced, spanning 0.1 to 0.9 $\mathrm{kgC/m^2}$ until 2050 and 0.3 and 4.0 $\mathrm{kgC/m^2}$ until 2100 (Fig. 3). Thus, increased material usage exerted an important positive effect, regardless of the time frame or the uncertainty from climate and decarbonization scenarios. However, its effect was lower in younger forests compared to mature ones, at least until 2050 (Fig. S6). After that, the trees in young forests had become similarly large to those of the initially mature forests, making them similarly suitable for long-lived products.

Discussion

Numerous studies have investigated the carbon mitigation potential of temperate forests. However, these studies often diverge in their findings since they are restricted to a few key assumptions and neglect the corresponding uncertainties. Our study quantifies the impacts and uncertainties associated with the different factors, offering a range of insights for mitigation strategies.

Effects of external conditions

Forest type

Our model simulations indicated similar growth patterns for BD and NE forests. LPJ-GUESS models productivity and growth of trees based on individual characteristics such as their morphology and physiognomy, which affect factors like the amount of absorbed radiation. Also site conditions such as available water and nutrients play a role, which are affected by the composition of tree species at the given site [29]. This simulated similar growth of NE and BD forests might sound counter-intuitive since NE species are known for their rapid growth, but this growth refers to volume, not carbon. BD trees possess a substantially higher wood density than NE trees. For instance, the common NE species of central Europe, spruce and pine, have wood densities of approximately 0.4 to $0.5\,\mathrm{t\,m^{-3}}$, while the typical BD species, oak and beech, have wood densities of around $0.7\,\mathrm{t\,m^{-3}}$ [60]. The similar carbon assimilation of these species is manifested in yield tables and observations [61–63].

The difference in wood product portfolios provided by the different forest types is not unique to our study region but common across Europe [64]. Especially NE forests provide a considerable product sink (15% of the forest sink for mature NE forests until 2050) highlighting its importance for mitigation strategies. Consequently, adaptation efforts in Europe towards more BD species will alter the product portfolio and could thus decrease the product sink. One solution to address this are new technologies enabling the usage of BD wood in construction, such as the emerging construction timber products made from European beech [18]. Furthermore, our results indicate that substitution effects are of high importance, regardless of the forest type, because also medium-lived products can exert a large substitution effect [15]. Finally, the higher susceptibility of NE forests to disturbances increases the risk for strategies that decrease or halt harvests, compared to BD forests.

Forest age

Younger forests exhibited a greater forest sink but their product sink and substitution effects were smaller (Fig. 2). Age structure also influences resistance against disturbances and maintaining a diverse age structure is proposed as an adaptation measure [65]. In our simulations, we managed forests with a "closer-to-nature" approach, yielding a diverse age structure (Fig. S2), but future work should consider the likelihood of disturbances in relation to forest structure which we only represented by species-specific disturbance probabilities. Furthermore, different levels of forest productivity should be taken into account [66].

Climate change

Climate change can significantly impact mitigation potentials, but this factor is often excluded in mitigation studies [22, 23, 27, 66]. Positive effects on the carbon sink such as CO_2 fertilization and prolonged growing season stand in contrast to negative ones such as enhanced drought stress, and increased frequencies of disturbances. Also reductions in nitrogen deposition can play a role [8]. The considered time frame is critical in determining whether positive and adverse effects cancel each other out or not [see also 67]. Our simulations indicated that the differences between RCPs were minor until 2050 (Fig 3) because the disturbance frequencies, CO_2 fertilization, growing season length, etc. remained similar between scenarios.

However, the differences became significant when projecting until 2100. This long-term view remains important because of the generally long time frames in forestry, and because a permanence in carbon storage beyond 2050 is desired (Fig. 3). The effect size also hinged on forest type, with disturbances posing a particular threat to NE forests. In addition, the simulated NE trees were less adapted to warm temperatures, leading to impaired (re-)establishment. Furthermore, a larger carbon stock is at risk in mature compared to young forests. Consequently, the negative impacts of climate change were most pronounced in mature NE forests (Fig. S6). The fact that BD forests are better adapted to a changing climate is in line with the literature [43, 68, 69]. Therefore, ongoing forest adaptation efforts in Europe towards larger shares of BD species may positively affect mitigation by protecting the in-situ forest carbon sink. However, it is important to note that also broad-leaved trees have started to be impacted by, e.g., droughts [70, 71].

Disturbances

We based our results on disturbance frequencies that increased exponentially with temperature (Fig. S1). Altering this assumption introduced substantial variations in total mitigation potentials (Figs. 2 and S11). When assuming constant disturbance rates, there were positive effects when transitioning from the RCP2.6 to RCP4.5 scenarios, with a relatively minor difference between RCP4.5 and RCP8.5 (Fig. S10). Even until 2100, the difference in RCPs was much smaller compared to other factors. With linear increases in disturbance frequencies, slightly negative impacts on the mitigation potential emerged going from RCP4.5 to RCP8.5, albeit dampened by positive effects from increased temperatures and prolonged growing seasons (Fig. 1). Scenarios with an exponential increase in disturbance probabilities exhibited substantial negative effects when transitioning to higher RCPs due to disturbances, overshadowing other climate change effects.

It is important to note that LPJ-GUESS employs simplistic models of water stress. Generally, dynamic vegetation models exhibit varied responses to drought and heat [72] and new hydraulic models are necessary for a more comprehensive understanding [73]. Moreover, the anticipated increase in disturbances is a critical aspect that is often overlooked in models, highlighting the significance of accounting for this aspect, as emphasized by our results. Nevertheless, although the magnitude of impacts was affected by the assumption on the disturbance scenario, the qualitative conclusions remained largely unaffected.

Decarbonization

The potential impact of anticipated decarbonization is often overlooked in mitigation studies. Wood offers climate benefits when substituted for various materials, including concrete, steel, glass, plastics, and aluminum [15]. Plans to decarbonize these industries are already in motion, driven by enhancements in efficiency, recycling, and product lifetimes, and the promotion of state-of-the-art technologies and innovations [13, 14, 16]. The pace of this decarbonization, however, remains uncertain. Currently implemented EU policies are projected to reach a 25% reduction in gross emissions by 2050, whereas the target is a reduction of about 80% [Fig. S5, 44].

In this study, the decarbonization pace played a pivotal role, driving differences between the combined carbon sink and total mitigation potential, occasionally resulting in contrary outcomes between the two. As decarbonization accelerates, substitution effects become less significant (Fig. 2). But even in the most ambitious, net-zero-compliant scenario, decarbonization will take time. Within the next decades, wood usage thus remains an important lever for carbon mitigation. Decreased harvests and wood usage would lead to negative substitution effects, i.e., higher emissions. After 2050, the importance of wood products for mitigation will diminish in a fast-decarbonization world, offering the opportunity to focus on other ecosystem services. Conversely, with slow decarbonization, the pressure on forests to provide mitigation will persist beyong 2050. This is evident in the substantial differences in total mitigation impact by 2100 between slower and faster decarbonization. The 25% decarbonization pace is close to projections for current policies, emphasizing that a fast speed-up of decarbonization would remove the pressure from forests to provide mitigation via substitution effects.

Apart from its pace, also the general concept of substitution effects bears uncertainties [56, 74, 75]. Three main issues are additionality (would the wood product have been created anyway?), leakage (was the substituted material not simply used elsewhere?), and replaceability (what type of fuel was replaced by wood?). Here, we neglected these aspects and examined the *theoretical* maximal substitution potential, offering insight into the overall importance of substitution. For instance, if wood production ceased in the EU, an additional 410 Mt CO_{2e} could be emitted annually through alternative products, equivalent to 15% of the EU's 2022 emissions [4, 7]. Concrete applications of substitution effects require comparing a scenario against a baseline to grasp the true mitigation impact.

Changes in management

Decreasing harvest intensity enhanced the forest carbon sink. This effect was smaller in NE forests because the increased growing stock led to higher losses from disturbances, particularly until 2100. The effects on the total mitigation potential depended on the time frame. They were clearly negative until 2050, but diverging until 2100 (Fig. 3). In the slow decarbonization scenario (25% decarbonization by 2050), reduced substitution effects more than outweighed the enhanced sink (Figs. 4+S8). Conversely, with faster decarbonization, the increased carbon sink outweighed the decreased substitution effects in most cases. The almost consistently decreased mitigation potentials due to decreased harvests until 2050 indicate that decreasing harvest intensities in sustainably managed forests might be counterproductive from a mitigation point of view.

Halting forest management yielded similar outcomes. While it would have numerous benefits for biodiversity and ecosystem services, doing so in sustainably managed mature forests might not provide anticipated mitigation benefits. Forest growth is eventually reduced, leading to a loss in sink strength. Simultaneously, climate change and associated disturbances threaten forests with high growing stocks. Studies suggesting that decreased harvest levels or prolonged rotations are beneficial for mitigation only seemingly contradict our findings. Their regional DFs for material substitution (0.45-0.6 tC/tC, [22, 27, 76]) are considerably lower than ours (1.5 tC/tC). One important reason for this discrepancy is landfilling: in the meta-analysis by Sathre and O'Connor [77], average material DFs are 2.1 tC/tC and 1.1 tC/tC without and with landfilling (partially even including methane recovery), respectively. The aforementioned studies are thus more comparable to our results with 75% decarbonization where also in our simulations decreased harvests had a total mitigation benefit in most cases (Figs. 4 and S8). Consequently, it is critical that DFs are assessed regionally and thoroughly, including end-of-life treatment, because their magnitude heavily impacts the mitigation impact of different management intensities.

Many mitigation studies were conducted in Northern Europe, where the forest carbon sink has been particularly strong due to the prevalence of young stands [23, 27]. In our simulations, the contribution of the carbon sink to total mitigation was also higher in young stands (Fig. 2). Furthermore, in Scandinavia, larger fractions of wood are used for pulp and energy than in Bavaria, even for conifers [27]. Also, unlike here, negative impacts of climatic change were either not considered in the cited studies, or only simplified [78, 79], potentially leading to an overestimation of the forest sink's contribution to mitigation. In our simulations, for instance, removing the temperature-dependent increase in disturbance frequencies led to a 15-30% higher carbon sink in NE forests until 2100 (Fig. S11).

Increased harvests in our simulations had mostly positive effects on mitigation until 2050, aligning with findings in studies employing high DFs [80]. However, considering the likely decrease in substitution effects, this positive effect only persisted until 2100 for slow decarbonization paces, where high substitution effects outweighed the decreased forest sink. For faster decarbonization, increasing harvests had strictly negative effects, aligning with other studies, e.g. Seppälä et al. [81] who advocated against increasing harvests with DFs less than 1.1 tC/tC. Also Skytt, Englund, and Jonsson [66], using multiple DFs (e.g., 0.7-2.8 for sawn wood), arrived at a similar conclusion: with strongly decreasing substitution effects, less harvesting is beneficial.

The harvest routines implemented in this study are very simplified, but different management regimes and replanting schemes could potentially provide co-benefits for the carbon sink and harvest volumes. These were beyond the scope of this study but need to be assessed in future work. Furthermore, we did not consider clearcuts because they are not allowed in Germany on a large scale and would not be compatible with other demands on forests. LPJ-GUESS also does not model the important impacts on micro-climate, and thus cannot simulate the impaired reestablishment after clearcut [e.g., 82].

Salvage logging

Salvage logging is a controversial [53, 83–85], yet common and in some European regions mandatory practice [86, 87]. Here, we quantified its direct impacts on the carbon balance, excluding considerations related to preventing subsequent disturbances, micro-climate effects, or habitat provisioning.

Whether salvage logging is beneficial for the carbon balance depends chiefly on wood usage and the climate scenarios that determines how fast the wood decays if left in the forest. Our simulations generally indicated a modest and ambiguous impact of salvage logging on the carbon sink, in line with studies showing similar decay times of deadwood and wood product portfolios [11, 88–91]. In our simulated BD forests, however, salvaging had mostly negative impacts on the carbon sink, because BD wood is predominantly burned (Fig. 3).

In terms of total mitigation potential, salvaging was mostly beneficial. Its impacts were particularly high in NE forests under RCP8.5, driven by high disturbance frequencies, a faster decay of deadwood, and a product portfolio with long lifespans.

While we considered that not all wood undergoes salvage logging after disturbances [53], we did not adjust wood usage patterns. It is difficult to get estimates on the use of salvaged wood, but statistics from Germany indicate no substantial change to default patterns (Fig. S3). Depending on the preceding disturbance, salvaged wood may be as effectively utilized as fresh wood [92]. However, post-disturbance logging disrupts wood markets, usually increasing exports [93]. This poses another complexity in estimating mitigation benefits, as transport emissions rise, and displacement effects become more challenging to estimate.

Wood products

The advantages of increased material usage align with other studies [9, 20–23]. This benefit was relevant across all decarbonization scenarios because of the enhanced product sink (Fig. 4). However, the earlier measures are implemented to facilitate increased material usage, the greater their impact because of substitution effects. This key role of material substitution, especially in the coming decades, was also highlighted by Nabuurs et al. [80].

This importance of wood products seems to contradict Johnston and Radeloff [94] who suggested a limited significance of the global product sink. However, their and other studies, e.g., Skytt, Englund, and Jonsson [66], used IPCC's global estimate of 35 years as the maximum half-live of long-lived products. Here, we used much higher residence times from regional analyses. Even more importantly, we used decay rates based on Gamma functions. These, unlike the usual exponential functions, account for a lag in decay after product creation (Fig. S4). Even with these high estimates, our simulated product sink is in the order of 10% of the forest sink, aligning with other estimates [6, 7, 9, 19]. This underlines that wood products in regions with sustainable harvests and high material usage are important for the mitigation potential. This significance is further magnified when substitution effects are considered. Nevertheless, it is necessary to mention that we neglected local changes in the wood industry (e.g., scaling effects) and trade impacts that are relevant in the study region [5]. Both affect product portfolios and substitution effects, but their detailed assessment is beyond the scope of this study.

Feasibility of other harvest intensities and wood usage patterns

Unlike changes in wood usage, modifications of harvest intensity affect various ecosystem services such as habitat provision, recreation, or local climate regulation through biophysical effects. Changes in the latter could even offset positive impacts on atmospheric CO_2 concentrations [95–99]. It is crucial that forests are not only considered for their mitigation potential, but also for their manifold ecosystem services [51, 100]. We agree with Erb et al. [24], proposing that forest management strategies should consider a desired sink strength and then estimate how much wood can be safely removed from the forest.

However, wood demand has been rising and is projected to continue increasing [101]. A reduced timber supply could lead to negative substitution effects with highly carbon-intensive products replacing wood. While some additional wood could be directly used with present-day construction methods, some additional usage likely depends on technological advances [17, 18, 75]. These necessitate new building codes, worker training, and public acceptance [74]. However, we assumed a linear increase in usage, allowing these changes to occur at a similar pace as historically observed in the study region [19].

Increased harvest intensities lead to smaller trees and a different product portfolio. Our assumptions about increased wood use thus also rely on the belief that parts of the stem currently allocated for short-lived products can serve longer-term purposes. Cross-laminated timber, for instance, can be derived from low-value wood [102]. Moreover, while a significant portion of needle-leaved trees already serves construction purposes in Bavaria, achieving a 50% increase (from 37% to 55.5%) is challenging. For broad-leaved trees, however, there is greater potential, given their currently low utilization (6%).

Impact on fuel provision and short-lived products

Promoting long-lived products decreases the share of short-lived products and fuel wood. Heated debates revolve around the climate effects of wood fuels, which many governments currently consider carbon-neutral, assuming eventual forest regrowth [103, 104]. This assumption is questionable given increasing disturbances and eroding forest resilience. Additionally, wood fuels emit more CO_2 per unit of energy than fossil fuels (indicated by a DF < 1) and it is unclear which energy sources wood fuels replace. This makes the carbon impact of wood fuels debatable [103–108].

In the EU, about one-fourth of all roundwood harvests currently serve as fuel wood [109], but all combined direct and indirect wood supplies (i.e., including recycled wood) contribute only about 6% to total gross final energy consumption [110]. Since wind and solar can generate significantly higher amounts of energy per area than bioenergy [89, 111], it should be possible to compensate for a gradual reduction in wood fuel provision via alternative energy sources. Nevertheless, energy security has recently become a major societal concern again in Europe due to the Russian war in Ukraine. This increases the importance of locally available fuels like wood. There is also a growing global demand for pulp and paper products, largely driven by increased packaging and sanitary paper needs [112]. These developments directly conflict with the increased provision of long-lived products assumed in this study and may require reduced usage and enhanced recycling efforts.

Conclusion

Optimizing forestry for mitigation, while simultaneously considering other ecosystem services, is one of many important strategies to mitigate climate change and complements the urgent need to reduce emissions. However, the multitude of factors determining the mitigation potential and their interactions have often been excluded in previous studies, making it difficult to draw general conclusions. In this study, through factorial modeling experiments for Bavaria as an example of the central European domain, we assessed a wide range of such factors: forest age, forest type, climate change and nitrogen deposition, disturbances, harvest intensity, salvage logging, wood usage, and the carbon intensity of other industries. Our approach allows us to suggest eight recommendations for forest-based mitigation assessments.

1) Climate change impacts (especially disturbances) and decarbonization are among the most important yet uncertain factors influencing mitigation and must not be neglected. Our analysis indicates that until 2050 these uncertainties are narrow enough to confidently develop mitigation projections. Looking beyond 2050 (which is necessary due to the long time spans in forestry), we suggest utilizing robust methods and risk diversification to account for the large uncertainties.

2) Increasing climate change enhances pressure on forests, especially through disturbances. In that regard, global climate change mitigation offers co-benefits for forest health and local forest-based mitigation.

3) The substantial differences in mitigation potentials arising from assumptions about changes in disturbance frequencies highlight the necessity for further model improvements.

4) Mitigation strategies need to be tailored to local forest conditions. Forest age and type heavily influence mitigation potentials, e.g. through different growth dynamics or product portfolios. Adaptation efforts towards more BD species – crucial to foster resistance to climate change – should be accompanied by the promotion of technologies to use more hardwood for long-lived products, thereby maintaining the product sink and maximizing substitution effects.

5) Substitution effects and the magnitude of DFs are crucial factors determining the mitigation potential. A thorough quantification of DFs, including end-of-life management of wood products, should be a key research priority.

6) Our simulations suggest that decreasing or stopping harvests reduces the mitigation potential in the considered region, especially until 2050. The product sink and substitution effects are still high and dismissing them would outweigh the increased forest carbon sink. In the long-term, this may change, but increased growing stocks then are at higher risk to be affected by disturbances. Modest increases in harvest intensity could provide mitigation benefits until 2050 depending on forest characteristics and decarbonization pace, but likely at the cost of other ecosystem services.

7) Increased material usage has a clear climate benefit, regardless of the scenario. However, the trade of wood products and other economic aspects affecting the mitigation potential need to be addressed in future studies.

8) Delaying decarbonization puts long-term pressure on forests to provide mitigation and puts forest health at risk. Any speed-up in decarbonization will thus greatly lift the pressure off forests and allow forest management to focus on other ecosystem services.

Our study provides a foundation for evaluating the carbon mitigation potentials of managed central European forests by quantifying key factors and uncertainties. These need to be taken into account when developing forest-based mitigation strategies, all the while keeping in mind the broader value of forests in providing numerous ecosystem services.

Our study quantifies key factors and uncertainties of carbon mitigation potentials in managed, central European forests laying the groundwork for forest-based mitigation. These results provide an impetus to develop strategies aimed at maximizing carbon mitigation while not losing sight of the diverse array of ecosystem services forest provide.

Acknowledgements
The authors gratefully acknowledge the computational and data resources provided by the Leibniz Supercomputing Centre (`www.lrz.de`). We acknowledge using ChatGPT3.5 to help us with grammar and language editing.

Funding
We acknowledge funding from the Bavarian State Ministry of Science and the Arts in the context of the Bavarian Climate Research Network (bayklif) through its BLIZ project (Grant No. 7831-26625- 2017, `www.bayklif-bliz.de`). We further acknowledge funding from the European Forest Institute (EFI) Networking Fund FORMASAM. The contribution of SS has been funded by the European Union's Horizon 2020 Research and Innovation Programme under the Marie Sklodowska-Curie Grant Agreement No 895158. KG further acknowledges funding from the VELUX foundation (`www.veluxstiftung.ch`).

Abbreviations
BD - broad-leaved deciduous, DBH - diameter at breast height, DF - displacement factor, NE - needle-leaved evergreen, RCP - Representative Concentration Pathway, N - nitrogen

Availability of data and materials
All code and data to reproduce the results and figures are available at `https://github.com/k-gregor/carbon-mitigation`

Competing interests
The authors declare that they have no competing interests.

Authors' contributions
KG conceived of the study, implemented necessary changes into the model, conducted the model runs and data analysis, and drafted the manuscript. AK provided help for the model runs. AK, AR, TK, and CR helped in the design of the study. AR coordinated the work. SS provided inputs for the model runs. All authors provided considerable help in the interpretation of the results. All authors read, commented on, and approved the final manuscript.

Author details
[1]TUM School of Life Sciences, Technical University of Munich, Freising, Germany. [2]Potsdam Institute for Climate Impact Research, Member of the Leibniz Association, Potsdam, Germany. [3]Natural Resources Institute Finland (Luke), Helsinki, Finland. [4]School of Geography, Earth and Environmental Sciences, University of Birmingham, Birmingham, United Kingdom. [5]Birmingham Institute of Forest Research, University of Birmingham, Birmingham, United Kingdom.

References
1. United Nations. *Summary of the Paris Agreement*. 2015. URL: `https://unfccc.int/sites/default/files/english_paris_agreement.pdf` (visited on 01/15/2024).
2. IPCC. "Synthesis Report AR6 - Summary for Policymakers." In: *An Assessment of the Intergovernmental Panel on Climate Change* (2023). ISSN: 14685833.
3. Yude Pan et al. "A Large and Persistent Carbon Sink in the World's Forests". In: *Science* 333.6045 (Aug. 2011), pp. 988–993. ISSN: 0036-8075. DOI: `10.1126/science.1201609`. URL: `https://www.science.org/doi/10.1126/science.1201609`.
4. Pierre Friedlingstein et al. "Global Carbon Budget 2022". In: *Earth System Science Data* 14.11 (Nov. 2022), pp. 4811–4900. ISSN: 1866-3516. DOI: `10.5194/essd-14-4811-2022`. URL: `https://essd.copernicus.org/articles/14/4811/2022/`.
5. Forest Europe. "State of Europe's Forests 2020". In: (2020).
6. G. Grassi et al. "Brief on the role of the forest-based bioeconomy in mitigating climate change through carbon storage and material substitution". In: (2021), pp. 1–16. URL: `https://publications.jrc.ec.europa.eu/repository/handle/JRC124374`.
7. P. Holmgren. "Climate effects of the forest based sector in the European Union". In: (2020), pp. 1–25.
8. Gert-Jan Nabuurs et al. "First signs of carbon sink saturation in European forest biomass". In: *Nature Climate Change* 3.9 (Sept. 2013), pp. 792–796. ISSN: 1758-678X. DOI: `10.1038/nclimate1853`. URL: `http://www.nature.com/articles/nclimate1853`.
9. Pieter Johannes Verkerk et al. *Forest-based climate change mitigation and adaptation in Europe. From Science to Policy 14*. European Forest Institute, 2022. ISBN: 9789527426258. DOI: `10.36333/fs14`.
10. Cornelius Senf and Rupert Seidl. "Mapping the forest disturbance regimes of Europe". In: *Nature Sustainability* 4.1 (2021), pp. 63–70. ISSN: 23989629. DOI: `10.1038/s41893-020-00609-y`. URL: `http://dx.doi.org/10.1038/s41893-020-00609-y`.
11. Beverly Elizabeth Law and Mark E. Harmon. "Forest sector carbon management, measurement and verification, and discussion of policy related to climate change". In: *Carbon Management* 2.1 (2011), pp. 73–84. ISSN: 17583004. DOI: `10.4155/cmt.10.40`.

12. Pau Brunet-Navarro et al. "Climate mitigation by energy and material substitution of wood products has an expiry date". In: *Journal of Cleaner Production* 303 (2021), p. 127026. ISSN: 09596526. DOI: 10.1016/j.jclepro.2021.127026. URL: https://doi.org/10.1016/j.jclepro.2021.127026.
13. IEA. *Iron and Steel Technology Roadmap*. Tech. rep. Paris, 2020. URL: https://www.iea.org/reports/iron-and-steel-technology-roadmap.
14. Johanna Lehne and Felix Preston. *Making Concrete Change; Innovation in Low-carbon Cement and Concrete*. Tech. rep. 2018, pp. 1–122. URL: www.chathamhouse.org.
15. Marcus Knauf et al. "Modeling the CO2-effects of forest management and wood usage on a regional basis". In: *Carbon Balance and Management* 10.1 (2015), pp. 1–12. ISSN: 17500680. DOI: 10.1186/s13021-015-0024-7. URL: http://dx.doi.org/10.1186/s13021-015-0024-7.
16. Paul Fennell et al. "Cement and steel — nine steps to net zero". In: *Nature* 603.7902 (Mar. 2022), pp. 574–577. ISSN: 0028-0836. DOI: 10.1038/d41586-022-00758-4. URL: https://www.nature.com/articles/d41586-022-00758-4.
17. Galina Churkina et al. "Buildings as a global carbon sink". In: *Nature Sustainability* 3.4 (2020), pp. 269–276. ISSN: 23989629. DOI: 10.1038/s41893-019-0462-4. URL: http://dx.doi.org/10.1038/s41893-019-0462-4.
18. Jan Hassan and Matthias Eisele. "BauBuche - Der nachhaltige Hochleistungswerkstoff". In: *Bautechnik* 92.1 (Jan. 2015), pp. 40–45. ISSN: 09328351. DOI: 10.1002/bate.201400093. URL: https://onlinelibrary.wiley.com/doi/10.1002/bate.201400093.
19. Tobias Wolf et al. *Potenziale von Bauen mit Holz*. Tech. rep. 2020, p. 248.
20. C. E. Smyth et al. "Quantifying the biophysical climate change mitigation potential of Canada's forest sector". In: *Biogeosciences* 11.13 (July 2014), pp. 3515–3529. ISSN: 1726-4189. DOI: 10.5194/bg-11-3515-2014. URL: https://bg.copernicus.org/articles/11/3515/2014/.
21. C. E. Smyth et al. "Climate change mitigation in Canada's forest sector: a spatially explicit case study for two regions". In: *Carbon Balance and Management* 13.1 (Dec. 2018), p. 11. ISSN: 1750-0680. DOI: 10.1186/s13021-018-0099-z. URL: https://doi.org/10.1186/s13021-018-0099-z; https://cbmjournal.biomedcentral.com/articles/10.1186/s13021-018-0099-z.
22. Alexa J. Dugan et al. "A systems approach to assess climate change mitigation options in landscapes of the United States forest sector". In: *Carbon Balance and Management* 13.1 (Dec. 2018), p. 13. ISSN: 1750-0680. DOI: 10.1186/s13021-018-0100-x. URL: https://doi.org/10.1186/s13021-018-0100-x; https://cbmjournal.biomedcentral.com/articles/10.1186/s13021-018-0100-x.
23. Sampo Soimakallio et al. "On the trade-offs and synergies between forest carbon sequestration and substitution". In: *Mitigation and Adaptation Strategies for Global Change* 26.1 (Jan. 2021), p. 4. ISSN: 1381-2386. DOI: 10.1007/s11027-021-09942-9. URL: http://link.springer.com/10.1007/s11027-021-09942-9.
24. Karl-Heinz Erb et al. "Changes in perspective needed to forge 'no-regret' forest-based climate change mitigation strategies". In: *GCB Bioenergy* 14.3 (Mar. 2022), pp. 246–257. ISSN: 1757-1693. DOI: 10.1111/gcbb.12921. URL: https://onlinelibrary.wiley.com/doi/10.1111/gcbb.12921.
25. Tomas Lundmark et al. "Potential Roles of Swedish Forestry in the Context of Climate Change Mitigation". In: *Forests* 5.4 (Mar. 2014), pp. 557–578. ISSN: 1999-4907. DOI: 10.3390/f5040557. URL: http://www.mdpi.com/1999-4907/5/4/557.
26. Andreas Krause, Thomas Knoke, and Anja Rammig. "A regional assessment of land-based carbon mitigation potentials: Bioenergy, BECCS, reforestation, and forest management". In: *GCB Bioenergy* 12.5 (May 2020), pp. 346–360. ISSN: 1757-1693. DOI: 10.1111/gcbb.12675. URL: https://onlinelibrary.wiley.com/doi/abs/10.1111/gcbb.12675.
27. Maximilian Schulte et al. "Nordic forest management towards climate change mitigation: time dynamic temperature change impacts of wood product systems including substitution effects". In: *European Journal of Forest Research* 141.5 (Oct. 2022), pp. 845–863. ISSN: 1612-4669. DOI: 10.1007/s10342-022-01477-1. URL: https://doi.org/10.1007/s10342-022-01477-1; https://link.springer.com/10.1007/s10342-022-01477-1.
28. Benjamin Smith, I. Colin Prentice, and Martin T. Sykes. "Representation of vegetation dynamics in the modelling of terrestrial ecosystems: comparing two contrasting approaches within European climate space". In: *Global Ecology and Biogeography* 10.6 (Nov. 2001), pp. 621–637. ISSN: 1466822X. DOI: 10.1046/j.1466-822X.2001.t01-1-00256.x. URL: http://doi.wiley.com/10.1046/j.1466-822X.2001.t01-1-00256.x.
29. B. Smith et al. "Implications of incorporating N cycling and N limitations on primary production in an individual-based dynamic vegetation model". In: *Biogeosciences* 11.7 (Apr. 2014), pp. 2027–2054. ISSN: 1726-4189. DOI: 10.5194/bg-11-2027-2014. URL: https://bg.copernicus.org/articles/11/2027/2014/.
30. Rupert Seidl et al. "Increasing forest disturbances in Europe and their impact on carbon storage". In: *Nature Climate Change* 4.9 (Sept. 2014), pp. 806–810. ISSN: 1758-678X. DOI: 10.1038/nclimate2318. URL: http://www.nature.com/articles/nclimate2318.
31. Cornelius Senf and Rupert Seidl. "Storm and fire disturbances in Europe: Distribution and trends". In: *Global Change Biology* March (2021), pp. 3605–3619. ISSN: 13652486. DOI: 10.1111/gcb.15679.
32. W. J. Parton et al. "Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide". In: *Global Biogeochemical Cycles* 7.4 (Dec. 1993), pp. 785–809. ISSN: 08866236. DOI: 10.1029/93GB02042. URL: http://doi.wiley.com/10.1029/93GB02042.
33. Mats Lindeskog et al. "Accounting for forest management in the estimation of forest carbon balance using the dynamic vegetation model LPJ-GUESS (v4.0, r9710): implementation and evaluation of simulations for Europe". In: *Geoscientific Model Development* 14.10 (Oct. 2021), pp. 6071–6112. ISSN: 1991-9603. DOI: 10.5194/gmd-14-6071-2021. URL: https://gmd.copernicus.org/articles/14/6071/2021/.

34. Daniel Klein and Christoph Schulz. *Die Kohlenstoffbilanz der Bayerischen Forst- und Holzwirtschaft*. Tech. rep. Bayerische Landesanstalt für Wald und Forstwirtschaft, 2012. URL: `https://www.lwf.bayern.de/mam/cms04/boden-klima/dateien/kohlenstoffbilanz_bayern_2012.pdf`.
35. György Büttner. "CORINE Land Cover and Land Cover Change Products". In: *Land Use and Land Cover Mapping in Europe: Practices & Trends*. Ed. by Ioannis Manakos and Matthias Braun. Dordrecht: Springer Netherlands, 2014, pp. 55–74. ISBN: 978-94-007-7969-3. DOI: 10.1007/978-94-007-7969-3_5. URL: `https://doi.org/10.1007/978-94-007-7969-3_5; http://link.springer.com/10.1007/978-94-007-7969-3_5`.
36. Edward Ayres et al. "Tree physiological responses to above-ground herbivory directly modify below-ground processes of soil carbon and nitrogen cycling". In: *Ecology Letters* 7.6 (June 2004), pp. 469–479. ISSN: 1461-023X. DOI: 10.1111/j.1461-0248.2004.00604.x. URL: `https://onlinelibrary.wiley.com/doi/10.1111/j.1461-0248.2004.00604.x`.
37. A. Bantle et al. "Quantity and quality of dissolved organic carbon released from coarse woody debris of different tree species in the early phase of decomposition". In: *Forest Ecology and Management* 329 (2014), pp. 287–294. ISSN: 03781127. DOI: 10.1016/j.foreco.2014.06.035. URL: `http://dx.doi.org/10.1016/j.foreco.2014.06.035`.
38. E Van Meijgaard et al. *Refinement and application of a regional atmospheric model for climate scenario calculations of Western Europe*. KVR 054/12. KvR, 2012. ISBN: 978-90-8815-046-3.
39. W. Hazeleger et al. "EC-Earth V2.2: description and validation of a new seamless earth system prediction model". In: *Climate Dynamics* 39.11 (Dec. 2012), pp. 2611–2629. ISSN: 0930-7575. DOI: 10.1007/s00382-011-1228-5. URL: `http://link.springer.com/10.1007/s00382-011-1228-5`.
40. Daniela Jacob et al. "EURO-CORDEX: new high-resolution climate change projections for European impact research". In: *Regional Environmental Change* 14.2 (Apr. 2014), pp. 563–578. ISSN: 1436-3798. DOI: 10.1007/s10113-013-0499-2. URL: `http://link.springer.com/10.1007/s10113-013-0499-2`.
41. Bayerisches Landesamt für Umwelt [Hrsg.] "Bayerische Klimadaten - Beobachtungsdaten, Klima-projektionsensemble und Klimakennwerte für Bayern". In: (2020).
42. Jean-François Lamarque et al. "Global and regional evolution of short-lived radiatively-active gases and aerosols in the Representative Concentration Pathways". In: *Climatic Change* 109.1-2 (Nov. 2011), pp. 191–212. ISSN: 0165-0009. DOI: 10.1007/s10584-011-0155-0. URL: `http://link.springer.com/10.1007/s10584-011-0155-0`.
43. Thomas A. M. Pugh et al. "Important role of forest disturbances in the global biomass turnover and carbon sinks". In: *Nature Geoscience* 12.9 (Sept. 2019), pp. 730–735. ISSN: 1752-0894. DOI: 10.1038/s41561-019-0427-2. URL: `http://www.nature.com/articles/s41561-019-0427-2`.
44. Felix Schreyer et al. "Common but differentiated leadership: strategies and challenges for carbon neutrality by 2050 across industrialized economies". In: *Environmental Research Letters* 15.11 (Nov. 2020), p. 114016. ISSN: 1748-9326. DOI: 10.1088/1748-9326/abb852. URL: `https://iopscience.iop.org/article/10.1088/1748-9326/abb852`.
45. Peiran R. Liu and Adrian E. Raftery. "Country-based rate of emissions reductions should increase by 80% beyond nationally determined contributions to meet the 2 °C target". In: *Communications Earth and Environment* 2.1 (2021), pp. 1–10. ISSN: 26624435. DOI: 10.1038/s43247-021-00097-8. URL: `http://dx.doi.org/10.1038/s43247-021-00097-8`.
46. J. Lawrence, M. Haasnoot, and R Lempert. "Climate change: making decisions in the face of deep uncertainty". In: *Nature* 580 (2020), p. 456.
47. IPCC. "SYNTHESIS REPORT OF THE IPCC SIXTH ASSESSMENT REPORT (AR6) IPCC. Longer Report". In: (2023).
48. Susanne Suvanto et al. "Understanding Europe's forest harvesting regimes". In: (in review). DOI: 10.31223/X5910J.
49. Jørgen Bo Larsen et al. *Closer-to-Nature Forest Management. From Science to Policy 12*. European Forest Institute, 2022. ISBN: 9789527426197. DOI: 10.36333/fs12. URL: `https://efi.int/publications-bank/closer-nature-forest-management`.
50. Thomas Knoke et al. "Assessing the Economic Resilience of Different Management Systems to Severe Forest Disturbance". In: *Environmental and Resource Economics* 0123456789 (Sept. 2022). ISSN: 0924-6460. DOI: 10.1007/s10640-022-00719-5. URL: `https://doi.org/10.1007/s10640-022-00719-5; https://link.springer.com/10.1007/s10640-022-00719-5`.
51. Konstantin Gregor et al. "Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate". In: *Earth's Future* 10.9 (Sept. 2022), pp. 1–25. ISSN: 2328-4277. DOI: 10.1029/2022EF002796. URL: `https://onlinelibrary.wiley.com/doi/10.1029/2022EF002796`.
52. Bayerische Staatsforsten. "Richtlinie: Bewirtschaftung von Fichten- und Fichtenmischbeständen im Bayerischen Staatswald: Stabilität – Strukturreichtum – Klimaanpassung". In: *Waldbauhandbuch Bayerische Staatsforsten* (2009), pp. 1–83. URL: `https://www.baysf.de/fileadmin/user_upload/04-wald_verstehen/Publikationen/Fichtenrichtlinie.pdf`.
53. David B Lindenmayer, Philip J Burton, and Jerry F Franklin. *Salvage logging and its ecological consequences*. Island Press, 2012. ISBN: 9781597264020.
54. Daniel Klein et al. "The Contribution of Managed and Unmanaged Forests to Climate Change Mitigation—A Model Approach at Stand Level for the Main Tree Species in Bavaria". In: *Forests* 4.1 (Jan. 2013), pp. 43–69. ISSN: 1999-4907. DOI: 10.3390/f4010043. URL: `http://www.mdpi.com/1999-4907/4/1/43`.
55. Jay H. Arehart et al. "Carbon sequestration and storage in the built environment". In: *Sustainable Production and Consumption* 27 (July 2021), pp. 1047–1063. ISSN: 23525509. DOI: 10.1016/j.spc.2021.02.028. URL: `https://linkinghub.elsevier.com/retrieve/pii/S235255092100066X`.

56. Mark E. Harmon. "Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions". In: *Environmental Research Letters* 14.6 (June 2019), p. 065008. ISSN: 1748-9326. DOI: 10.1088/1748-9326/ab1e95. URL: https://iopscience.iop.org/article/10.1088/1748-9326/ab1e95.
57. Xing Li and Jingfeng Xiao. "A Global, 0.05-Degree Product of Solar-Induced Chlorophyll Fluorescence Derived from OCO-2, MODIS, and Reanalysis Data". In: *Remote Sensing* 11.5 (Mar. 2019), p. 517. ISSN: 2072-4292. DOI: 10.3390/rs11050517. URL: https://www.mdpi.com/2072-4292/11/5/517.
58. S. Running and M. Zhao. *MODIS/Terra Net Primary Production Gap-Filled Yearly L4 Global 500m SIN Grid V061 [Data set]*. 2021. DOI: 10.5067/MODIS/MOD17A3HGF.061. (Visited on 12/01/2021).
59. Andreas Krause et al. "Quantifying the impacts of land cover change on gross primary productivity globally". In: *Scientific Reports* 12.1 (Nov. 2022), p. 18398. ISSN: 2045-2322. DOI: 10.1038/s41598-022-23120-0. URL: https://doi.org/10.1038/s41598-022-23120-0; https://www.nature.com/articles/s41598-022-23120-0.
60. P. S. Savill, ed. *The silviculture of trees used in British forestry*. Wallingford: CABI, 2019. ISBN: 9781786393920. DOI: 10.1079/9781786393920.0000. URL: http://www.cabi.org/cabebooks/ebook/20193143675.
61. M. Albert et al. *Eine neue Generation von Ertragstafeln für Eiche, Buche, Fichte, Douglasie und Kiefer. Version 1.1*. 2021. DOI: 10.5281/zenodo.6343906. URL: https://doi.org/10.5281/zenodo.6343906 (visited on 11/14/2022).
62. Bayerische Staatsforsten. *Grundsätze für die Bewirtschaftung von Kiefern- und Kiefernmischbeständen im Bayerischen Staatswald*. Tech. rep. 2014, pp. 1–61. URL: https://www.baysf.de/fileadmin/user_upload/04-wald_verstehen/Publikationen/WNJF-RL-005_Kiefernrichtlinie.pdf.
63. S. Luyssaert et al. "The European carbon balance. Part 3: forests". In: *Global Change Biology* 16.5 (May 2010), pp. 1429–1450. ISSN: 13541013. DOI: 10.1111/j.1365-2486.2009.02056.x. URL: https://onlinelibrary.wiley.com/doi/10.1111/j.1365-2486.2009.02056.x.
64. Eurostat. *Eurostat Forestry Database*. 2021. URL: https://ec.europa.eu/eurostat/web/forestry/data/database (visited on 06/29/2021).
65. Kevin L. O'Hara and Benjamin S. Ramage. "Silviculture in an uncertain world: Utilizing multi-aged management systems to integrate disturbance". In: *Forestry* 86.4 (2013), pp. 401–410. ISSN: 0015752X. DOI: 10.1093/forestry/cpt012.
66. Torbjörn Skytt, Göran Englund, and Bengt-gunnar Jonsson. "Climate mitigation forestry—temporal trade-offs". In: *Environmental Research Letters* 16.11 (Nov. 2021), p. 114037. ISSN: 1748-9326. DOI: 10.1088/1748-9326/ac30fa. URL: https://iopscience.iop.org/article/10.1088/1748-9326/ac30fa.
67. Christopher P.O. Reyer et al. "Are forest disturbances amplifying or canceling out climate change-induced productivity changes in European forests?" In: *Environmental Research Letters* 12.3 (2017). ISSN: 17489326. DOI: 10.1088/1748-9326/aa5ef1.
68. Rasmus Astrup et al. "A sensible climate solution for the boreal forest". In: *Nature Climate Change* 8.1 (2018), pp. 11–12. ISSN: 17586798. DOI: 10.1038/s41558-017-0043-3.
69. Rupert Seidl et al. "Forest disturbances under climate change". In: *Nature Climate Change* 7.6 (2017), pp. 395–402. ISSN: 17586798. DOI: 10.1038/nclimate3303. URL: http://dx.doi.org/10.1038/nclimate3303.
70. Allan Buras, Anja Rammig, and Christian S. Zang. "Quantifying impacts of the 2018 drought on European ecosystems in comparison to 2003". In: *Biogeosciences* 17.6 (Mar. 2020), pp. 1655–1672. ISSN: 1726-4189. DOI: 10.5194/bg-17-1655-2020. URL: https://bg.copernicus.org/articles/17/1655/2020/.
71. Bernhard Schuldt et al. "A first assessment of the impact of the extreme 2018 summer drought on Central European forests". In: *Basic and Applied Ecology* 45 (June 2020), pp. 86–103. ISSN: 14391791. DOI: 10.1016/j.baae.2020.04.003. URL: https://linkinghub.elsevier.com/retrieve/pii/S1439179120300414.
72. Elisabeth Tschumi et al. "Large Variability in Simulated Response of Vegetation Composition and Carbon Dynamics to Variations in Drought-Heat Occurrence". In: *Journal of Geophysical Research: Biogeosciences* 128.4 (Apr. 2023). ISSN: 2169-8953. DOI: 10.1029/2022JG007332. URL: https://agupubs.onlinelibrary.wiley.com/doi/10.1029/2022JG007332.
73. Phillip Papastefanou et al. "A Dynamic Model for Strategies and Dynamics of Plant Water-Potential Regulation Under Drought Conditions". In: *Frontiers in Plant Science* 11.April (Apr. 2020), pp. 1–13. ISSN: 1664-462X. DOI: 10.3389/fpls.2020.00373. URL: https://www.frontiersin.org/article/10.3389/fpls.2020.00373/full.
74. Christina Howard et al. "Wood product carbon substitution benefits: a critical review of assumptions". In: *Carbon Balance and Management* 16.1 (2021), pp. 1–11. ISSN: 17500680. DOI: 10.1186/s13021-021-00171-w. URL: https://doi.org/10.1186/s13021-021-00171-w.
75. G. Cornelis van Kooten and Craig M.T. Johnston. "The Economics of Forest Carbon Offsets". In: *Annual Review of Resource Economics* 8.1 (Oct. 2016), pp. 227–246. ISSN: 1941-1340. DOI: 10.1146/annurev-resource-100815-095548. URL: http://www.annualreviews.org/doi/10.1146/annurev-resource-100815-095548.
76. Mitsuo Matsumoto et al. "Potential contributions of forestry and wood use to climate change mitigation in Japan". In: *Journal of Forest Research* 21.5 (Oct. 2016), pp. 211–222. ISSN: 1341-6979. DOI: 10.1007/s10310-016-0527-4. URL: https://www.tandfonline.com/doi/full/10.1007/s10310-016-0527-4.
77. Roger Sathre and Jennifer O'Connor. "Meta-analysis of greenhouse gas displacement factors of wood product substitution". In: *Environmental Science & Policy* 13.2 (Apr. 2010), pp. 104–114. ISSN: 14629011.

DOI: 10.1016/j.envsci.2009.12.005. URL: http://dx.doi.org/10.1016/j.envsci.2009.12.005; https://linkinghub.elsevier.com/retrieve/pii/S1462901109001804.

78. Hans Petersson et al. "On the role of forests and the forest sector for climate change mitigation in Sweden". In: *GCB Bioenergy* 14.7 (July 2022), pp. 793–813. ISSN: 1757-1693. DOI: 10.1111/gcbb.12943. URL: https://onlinelibrary.wiley.com/doi/10.1111/gcbb.12943.
79. L. Gustavsson et al. "Climate effects of forestry and substitution of concrete buildings and fossil energy". In: *Renewable and Sustainable Energy Reviews* 136.March 2020 (2021), p. 110435. ISSN: 18790690. DOI: 10.1016/j.rser.2020.110435. URL: https://doi.org/10.1016/j.rser.2020.110435.
80. Gert-Jan Nabuurs et al. *Climate-Smart Forestry: mitigation impacts in three European regions*. April. 2018, p. 32. ISBN: 9789525980530. URL: https://doi.org/10.36333/fs06; https://www.efi.int/sites/default/files/files/publication-bank/2018/efi_fstp_6_2018.pdf.
81. Jyri Seppälä et al. "Effect of increased wood harvesting and utilization on required greenhouse gas displacement factors of wood-based products and fuels". In: *Journal of Environmental Management* 247.December 2018 (2019), pp. 580–587. ISSN: 10958630. DOI: 10.1016/j.jenvman.2019.06.031. URL: https://doi.org/10.1016/j.jenvman.2019.06.031.
82. Derek W. Carlson and Arthur Groot. "Microclimate of clear-cut, forest interior, and small openings in trembling aspen forest". In: *Agricultural and Forest Meteorology* 87.4 (Dec. 1997), pp. 313–329. ISSN: 01681923. DOI: 10.1016/S0168-1923(95)02305-4. URL: https://linkinghub.elsevier.com/retrieve/pii/S0168192395023054.
83. Thomas Knoke et al. "Economic losses from natural disturbances in Norway spruce forests – A quantification using Monte-Carlo simulations". In: *Ecological Economics* 185.January (2021), p. 107046. ISSN: 09218009. DOI: 10.1016/j.ecolecon.2021.107046. URL: https://doi.org/10.1016/j.ecolecon.2021.107046.
84. Laura Dobor et al. "Is salvage logging effectively dampening bark beetle outbreaks and preserving forest carbon stocks?" In: *Journal of Applied Ecology* 57.1 (2020), pp. 67–76. ISSN: 13652664. DOI: 10.1111/1365-2664.13518.
85. Cornelius Senf, Jörg Müller, and Rupert Seidl. "Post-disturbance recovery of forest cover and tree height differ with management in Central Europe". In: *Landscape Ecology* 34.12 (Dec. 2019), pp. 2837–2850. ISSN: 0921-2973. DOI: 10.1007/s10980-019-00921-9. URL: http://link.springer.com/10.1007/s10980-019-00921-9.
86. Simon Thorn et al. "Estimating retention benchmarks for salvage logging to protect biodiversity". In: *Nature Communications* 11.1 (2020), pp. 1–8. ISSN: 20411723. DOI: 10.1038/s41467-020-18612-4.
87. A. Camia et al. *The use of woody biomass for energy production in the EU and impacts on forests The use of woody biomass for energy production in the EU, EUR 30548 EN*. Publications Office of the European Union, Luxembourg, 2021. ISBN: 9789276278672. DOI: 10.2760/831621.
88. Ingolf Profft et al. "Forest management and carbon sequestration in wood products". In: *European Journal of Forest Research* 128.4 (2009), pp. 399–413. ISSN: 16124669. DOI: 10.1007/s10342-009-0283-5.
89. Ernst Detlef Schulze et al. "The climate change mitigation effect of bioenergy from sustainably managed forests in Central Europe". In: *GCB Bioenergy* 12.3 (Mar. 2020), pp. 186–197. ISSN: 1757-1693. DOI: 10.1111/gcbb.12672. URL: https://onlinelibrary.wiley.com/doi/10.1111/gcbb.12672.
90. Tiemo Kahl et al. "Wood decay rates of 13 temperate tree species in relation to wood properties, enzyme activities and organismic diversities". In: *Forest Ecology and Management* 391 (May 2017), pp. 86–95. ISSN: 03781127. DOI: 10.1016/j.foreco.2017.02.012. URL: http://dx.doi.org/10.1016/j.foreco.2017.02.012; https://linkinghub.elsevier.com/retrieve/pii/S0378112716307137.
91. S. Müller-Using and N. Bartsch. "Decay dynamic of coarse and fine woody debris of a beech (Fagus sylvatica L.) forest in Central Germany". In: *European Journal of Forest Research* 128.3 (2009), pp. 287–296. ISSN: 16124669. DOI: 10.1007/s10342-009-0264-8.
92. Dirk Berthold. *Wertschöpfende Verwendung von Kalamitätsholz*. 2022. URL: https://www.3-n.info/media/4_Downloads/pdf_NwsTrmn_3NVrnstltng_NHNTagung2022_Berthold.pdf (visited on 12/12/2022).
93. Destatis. *Exports of raw timber up 42.6% in 2020*. 2021. URL: https://www.destatis.de/EN/Press/2021/05/PE21_N031_51.html (visited on 02/10/2023).
94. Craig M. T. Johnston and Volker C. Radeloff. "Global mitigation potential of carbon stored in harvested wood products". In: *Proceedings of the National Academy of Sciences* 116.29 (July 2019), pp. 14526–14531. ISSN: 0027-8424. DOI: 10.1073/pnas.1904231116. URL: https://pnas.org/doi/full/10.1073/pnas.1904231116.
95. Richard A. Betts. "Offset of the potential carbon sink from boreal forestation by decreases in surface albedo". In: *Nature* 408.6809 (Nov. 2000), pp. 187–190. ISSN: 0028-0836. DOI: 10.1038/35041545. URL: http://www.nature.com/articles/35041545.
96. Gordon Bonan. "Forests and climate change: Forcings, feedbacks, and the climate benefits of forests". In: *Science* 320.5882 (2008), pp. 1444–1449. ISSN: 00368075. DOI: 10.1126/science.1155121.
97. Elizabeth Burakowski et al. "The role of surface roughness, albedo, and Bowen ratio on ecosystem energy balance in the Eastern United States". In: *Agricultural and Forest Meteorology* 249.December 2017 (2018), pp. 367–376. ISSN: 01681923. DOI: 10.1016/j.agrformet.2017.11.030.
98. Sara Alibakhshi et al. "Quantitative analysis of the links between forest structure and land surface albedo on a global scale". In: *Remote Sensing of Environment* 246.April (2020), p. 111854. ISSN: 00344257. DOI: 10.1016/j.rse.2020.111854. URL: https://doi.org/10.1016/j.rse.2020.111854.
99. Sebastiaan Luyssaert et al. "Trade-offs in using European forests to meet climate objectives". In: *Nature* 562.7726 (2018), pp. 259–262. ISSN: 14764687. DOI: 10.1038/s41586-018-0577-1. URL: http://dx.doi.org/10.1038/s41586-018-0577-1.

100. Luis Diaz-Balteiro et al. "Selecting the best forest management alternative by aggregating ecosystem services indicators over time: A case study in central Spain". In: *Ecological Indicators* 72 (2017), pp. 322–329. ISSN: 1470160X. DOI: 10.1016/j.ecolind.2016.06.025. URL: http://dx.doi.org/10.1016/j.ecolind.2016.06.025.

101. FAO. *The State of the World's Forests 2022. Forest pathways for green recovery and building inclusive, resilient and sustainable economies.* Rome: FAO, May 2022. ISBN: 978-92-5-135984-6. DOI: 10.4060/cb9360en. URL: http://www.fao.org/documents/card/en/c/cb9360en.

102. Omar Espinoza and Urs Buehlmann. "Cross-Laminated Timber in the USA: Opportunity for Hardwoods?" In: *Current Forestry Reports* 4.1 (Mar. 2018), pp. 1–12. ISSN: 2198-6436. DOI: 10.1007/s40725-018-0071-x. URL: http://link.springer.com/10.1007/s40725-018-0071-x.

103. John D Sterman, Lori Siegel, and Juliette N Rooney-Varga. "Does replacing coal with wood lower CO 2 emissions? Dynamic lifecycle analysis of wood bioenergy". In: *Environmental Research Letters* 13.1 (Jan. 2018), p. 015007. ISSN: 1748-9326. DOI: 10.1088/1748-9326/aaa512. URL: https://iopscience.iop.org/article/10.1088/1748-9326/aaa512.

104. Timothy Searchinger. "Fixing a Critical Climate Accounting Error". In: *Clean Air* 326.October (2009), pp. 527–528. ISSN: 0036-8075. DOI: 10.1126/science.1178797.

105. Mary S. Booth, Brendan Mackey, and Virginia Young. "It's time to stop pretending burning forest biomass is carbon neutral". In: *GCB Bioenergy* 12.12 (Dec. 2020), pp. 1036–1037. ISSN: 1757-1693. DOI: 10.1111/gcbb.12716. URL: https://onlinelibrary.wiley.com/doi/10.1111/gcbb.12716.

106. Bjart Holtsmark. "The outcome is in the assumptions: Analyzing the effects on atmospheric CO2 levels of increased use of bioenergy from forest biomass". In: *GCB Bioenergy* 5.4 (2013), pp. 467–473. ISSN: 17571707. DOI: 10.1111/gcbb.12015.

107. Francesco Cherubini et al. "CO2 emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming". In: *GCB Bioenergy* 3.5 (2011), pp. 413–426. ISSN: 17571707. DOI: 10.1111/j.1757-1707.2011.01102.x.

108. Philippe Leturcq. "GHG displacement factors of harvested wood products: the myth of substitution". In: *Scientific Reports* 10.1 (2020), pp. 1–9. ISSN: 20452322. DOI: 10.1038/s41598-020-77527-8. URL: https://doi.org/10.1038/s41598-020-77527-8.

109. Eurostat. *Roundwood removals by type of wood and assortment (FOR_REMOV)*. 2022. URL: https://ec.europa.eu/eurostat/databrowser/ (visited on 11/14/2022).

110. Nicolae Scarlat et al. "Brief on biomass for energy in the European Union". In: *EC publication* (2019), pp. 1–8. DOI: 10.2760/546943. URL: https://publications.jrc.ec.europa.eu/repository/handle/JRC109354.

111. T.J. Dijkman and R.M.J. Benders. "Comparison of renewable fuels based on their land use using energy densities". In: *Renewable and Sustainable Energy Reviews* 14.9 (Dec. 2010), pp. 3148–3155. ISSN: 13640321. DOI: 10.1016/j.rser.2010.07.029. URL: http://dx.doi.org/10.1016/j.rser.2010.07.029; https://linkinghub.elsevier.com/retrieve/pii/S1364032110002091.

112. IEA. *Pulp and Paper*. Tech. rep. Paris: IEA, 2022. URL: https://www.iea.org/reports/pulp-and-paper.

Supplementary Material: Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests

Konstantin Gregor[1,*], Andreas Krause[1], Christopher P O Reyer[2], Thomas Knoke[1], Benjamin F Meyer[1], Susanne Suvanto[3,4,5], and Anja Rammig[1]

[1]TUM School of Life Sciences, Technical University of Munich, Freising, Germany

[2]Potsdam Institute for Climate Impact Research, Member of the Leibniz Association, Potsdam, Germany

[3]Natural Resources Institute Finland (Luke), Helsinki, Finland

[4]School of Geography, Earth and Environmental Sciences, University of Birmingham, United Kingdom

[5]Birmingham Institute of Forest Research, University of Birmingham, United Kingdom

[*]Corresponding author, konstantin.gregor@tum.de

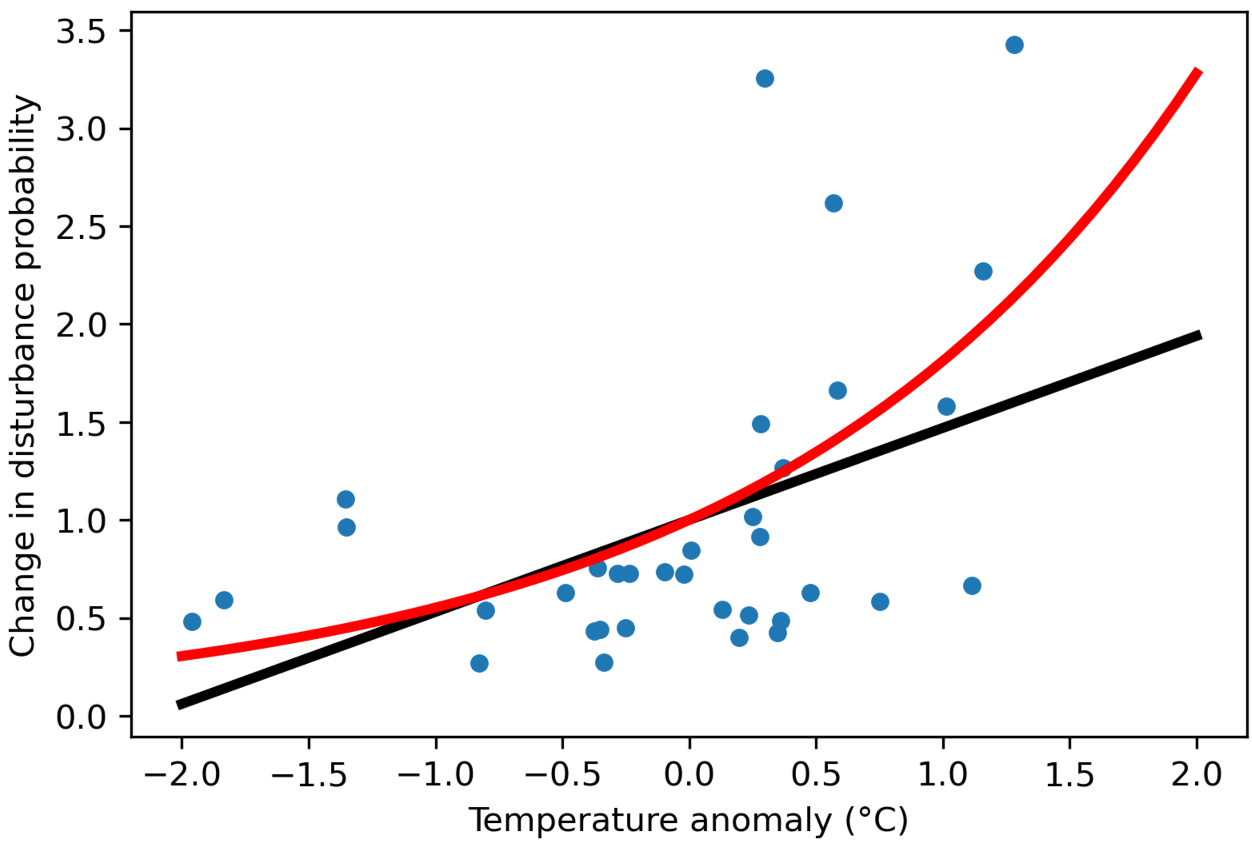

Figure S1: Fitted exponential (red) and linear (black) relationship between temperature and probability of a disturbance based on observed data of disturbances (blue) in Germany from Senf and Seidl (2021). The mean surface temperature of a grid cell of 2001-2014 was used as the baseline temperature as this was the time frame from which the disturbance return times of Pugh et al. (2019) were observed.

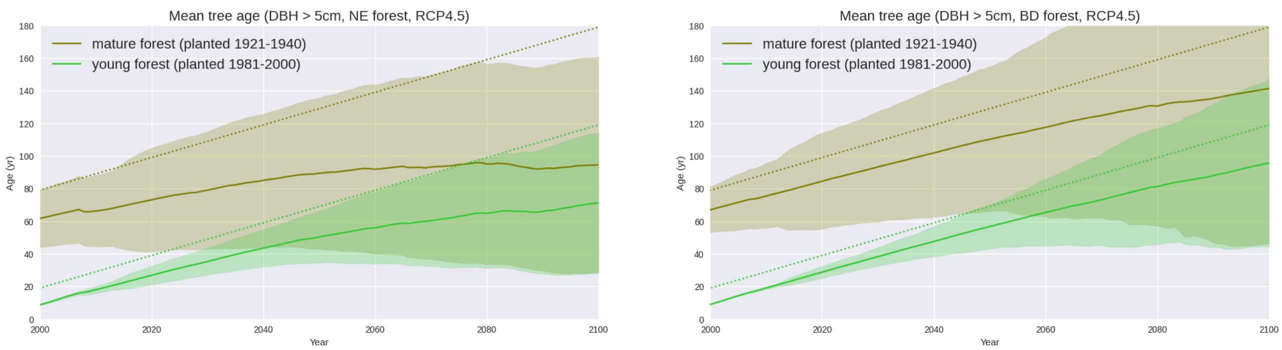

Figure S2: Example of the age structure of the forests for the 100% management scenario, and RCP4.5 with exponential increase in disturbance frequencies. The forests are planted and consecutively thinned, leaving room for new establishments. Depicted is the mean age with bands of one standard deviation. The dotted lines depict the maximum tree age inside the forest.

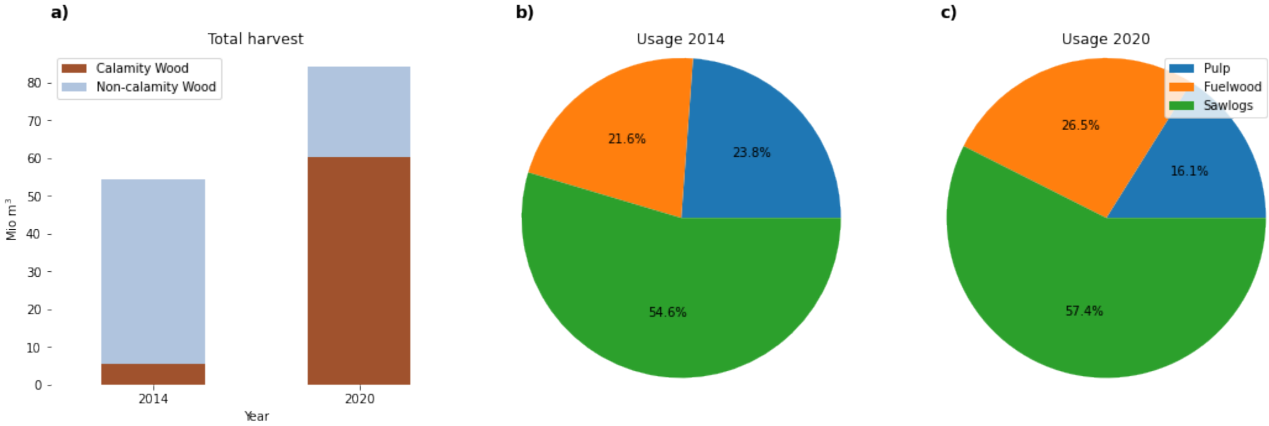

Figure S3: a) In Germany in 2020, about 72% of total wood harvests came from salvage logging, whereas this fraction was only 10% in 2014 (BMEL, 2021; Destatis, 2022). b+c) The resulting wood usage portfolio was not drastically different between the two years.

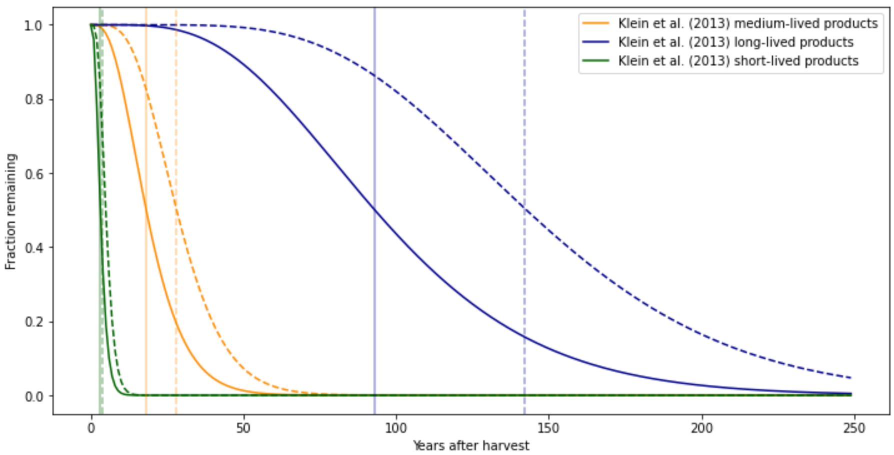

Figure S4: Decay functions for the different product pools. Vertical lines show the median residence time, dashed lines are for the simulation with 50% increased residence time, representing increased cascade usage.

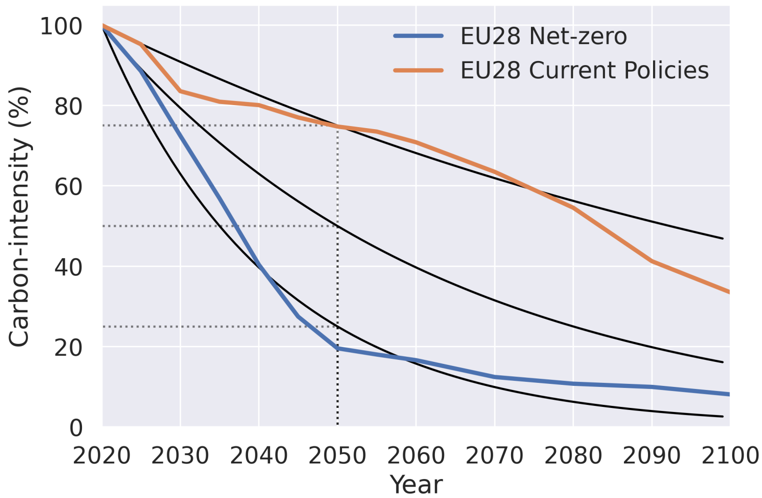

Figure S5: The three considered decarbonization scenarios. They reach 25%, 50%, and 75% decarbonization, respectively. The 25% and 75% scenarios are fitted to closely match the carbon-intensity of the EU with current policies and net-zero policies, respectively, based on data of Schreyer et al. (2020).

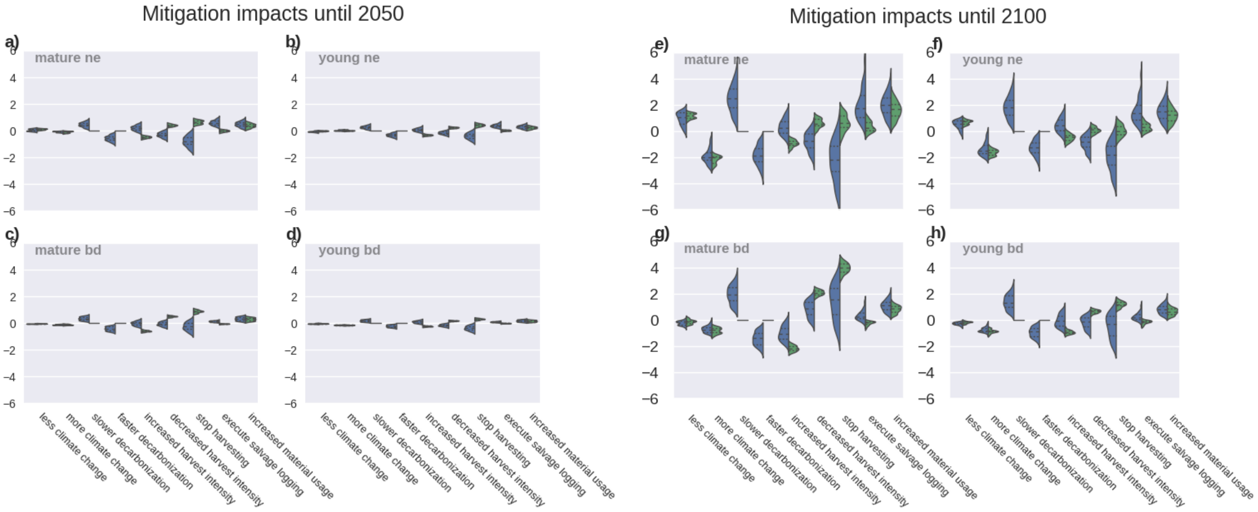

Figure S6: Same as figure 3 but split up into the forest types.

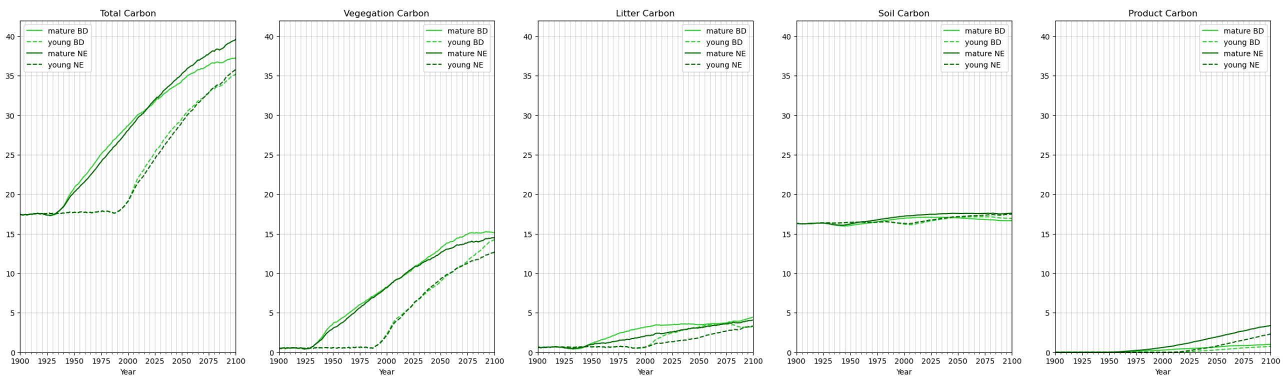

Figure S7: The different carbon pools for young and mature NE and BD forests under RCP4.5 and 100% management.

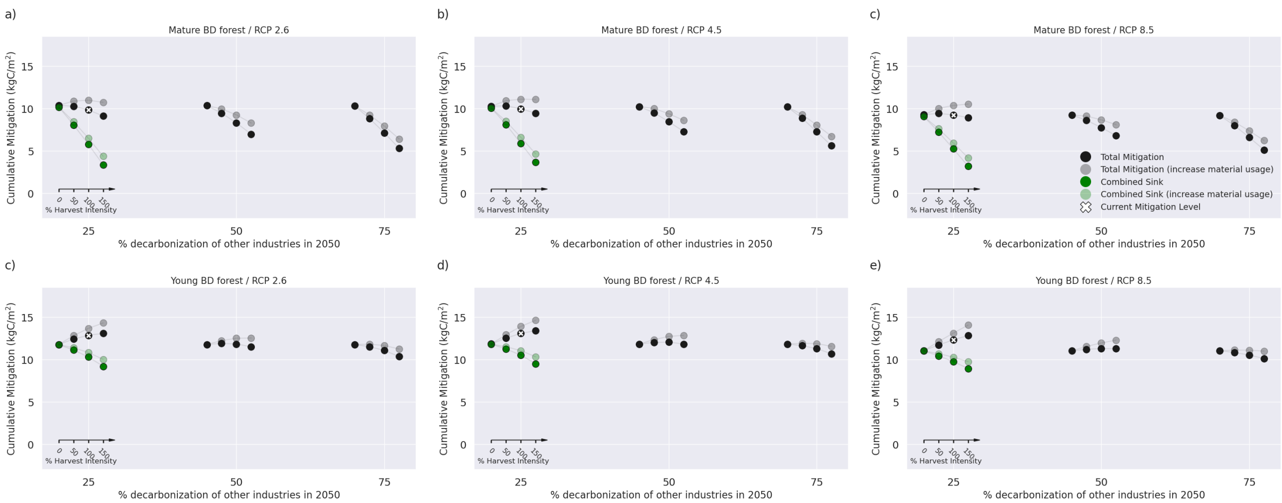

Figure S8: Same as Figure 4, but for BD forests.

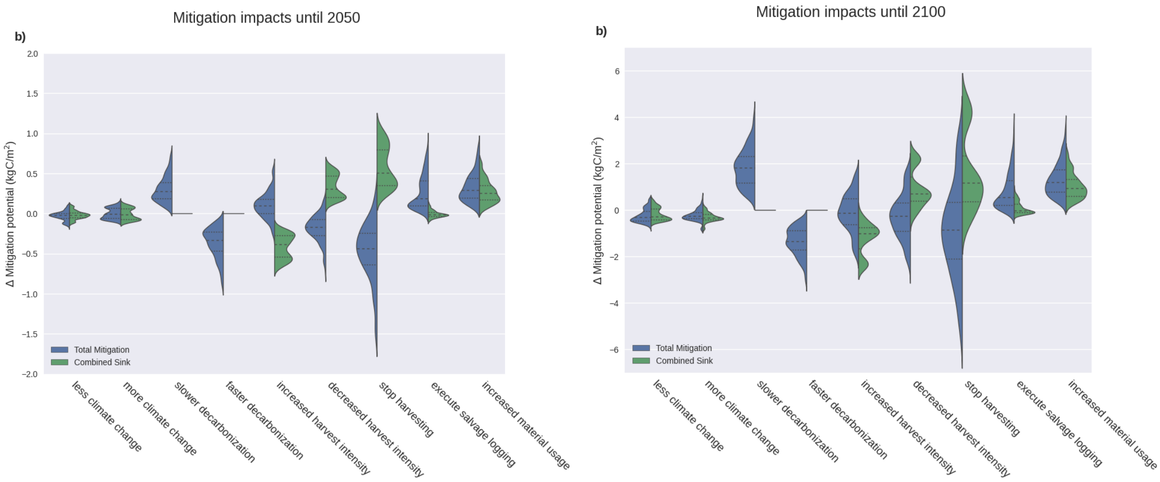

Figure S9: Same as Figure 3 but with a linear increase in disturbance probability based on temperature anomaly.

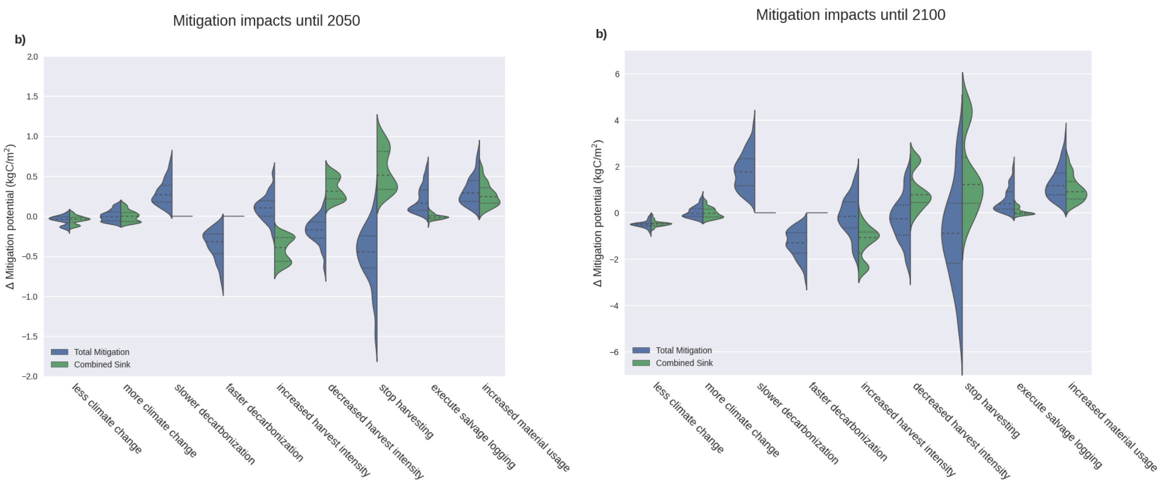

Figure S10: Same as Figure 3 but with constant disturbance rates

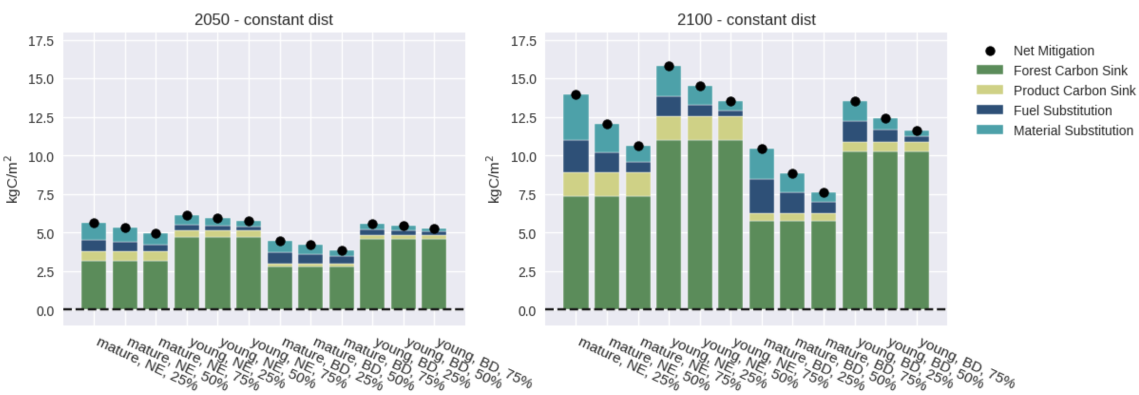

Figure S11: Same as Fig. 2 but without temperature-dependent increases in disturbance frequencies.

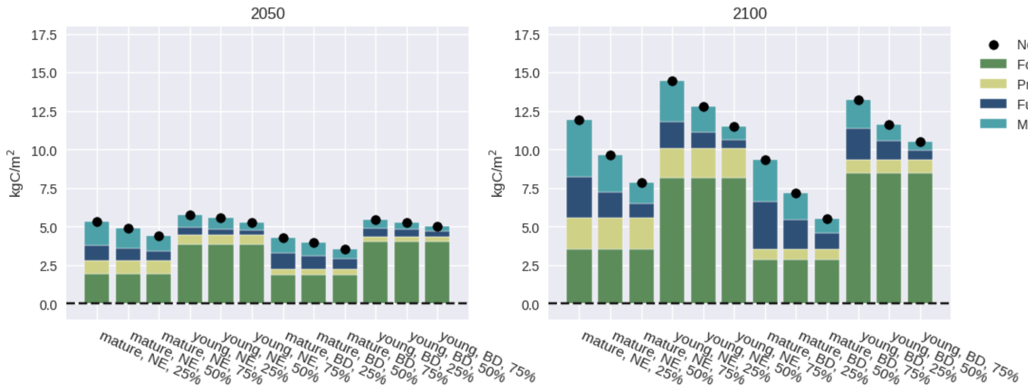

Figure S12: Same as Fig. 2 but with 150% harvest intensity.

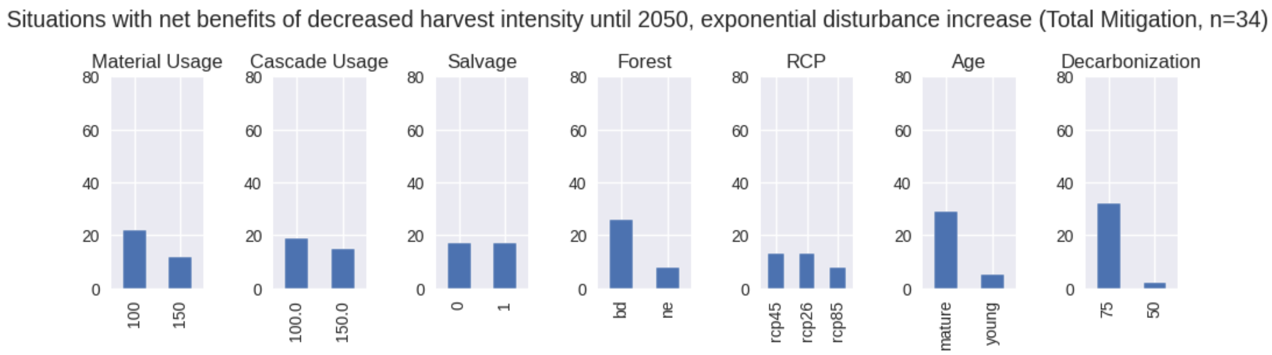

Figure S13: Situations where decreasing harvesting was beneficial for the carbon mitigation until 2050.

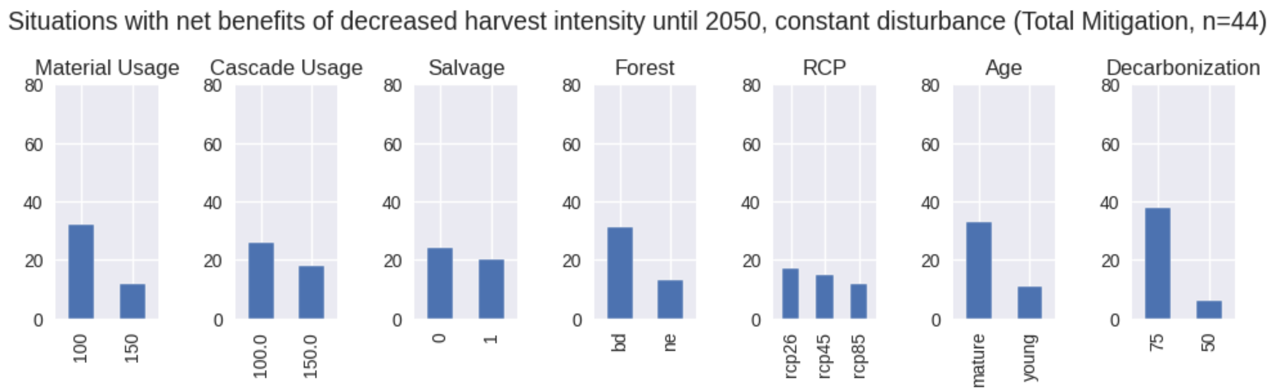

Figure S14: Situations where decreasing harvesting was beneficial for the carbon mitigation until 2050, assuming constant disturbance rates.

References

BMEL. (2021). *Holzmarktbericht 2020* (tech. rep.). Bundesministerium für Ernährung und Landwirtschaft (BMEL). https://www.bmel-statistik.de/fileadmin/daten/FHB-0120005-2020.pdf

Destatis. (2022). Durch Schäden bedingter Holzeinschlag. Retrieved November 9, 2022, from https://www.destatis.de/DE/Im-Fokus/Klima/_Grafik/_Interaktiv/schadholzeinschlag.html

Pugh, T. A. M., Arneth, A., Kautz, M., Poulter, B., & Smith, B. (2019). Important role of forest disturbances in the global biomass turnover and carbon sinks. *Nature Geoscience*, *12*(9), 730–735. https://doi.org/10.1038/s41561-019-0427-2

Schreyer, F., Luderer, G., Rodrigues, R., Pietzcker, R. C., Baumstark, L., Sugiyama, M., Brecha, R. J., & Ueckerdt, F. (2020). Common but differentiated leadership: strategies and challenges for carbon neutrality by 2050 across industrialized economies. *Environmental Research Letters*, *15*(11), 114016. https://doi.org/10.1088/1748-9326/abb852

Senf, C., & Seidl, R. (2021). Mapping the forest disturbance regimes of Europe. *Nature Sustainability*, *4*(1), 63–70. https://doi.org/10.1038/s41893-020-00609-y

Earth's Future

RESEARCH ARTICLE
10.1029/2022EF002796

Key Points:

- Strategies for climate-smart forestry under a range of climate scenarios always lead to trade-offs between different ecosystem services (ESs)
- Higher shares of unmanaged and broad-leaved forests are beneficial for numerous ESs, but lead to decreased timber provision
- The mitigation potential of forests strongly relies on substitution effects which depend on the carbon-intensity of the alternative products

Supporting Information:

Supporting Information may be found in the online version of this article.

Correspondence to:

K. Gregor,
konstantin.gregor@tum.de

Citation:

Gregor, K., Knoke, T., Krause, A., Reyer, C. P. O., Lindeskog, M., Papastefanou, P., et al. (2022). Trade-offs for climate-smart forestry in Europe under uncertain future climate. *Earth's Future*, *10*, e2022EF002796. https://doi.org/10.1029/2022EF002796

Received 23 MAR 2022
Accepted 24 AUG 2022

Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate

Konstantin Gregor[1], Thomas Knoke[1], Andreas Krause[1], Christopher P. O. Reyer[2], Mats Lindeskog[3], Phillip Papastefanou[1,3], Benjamin Smith[3,4], Anne-Sofie Lansø[5,6], and Anja Rammig[1]

[1]TUM School of Life Sciences, Technical University of Munich, Freising, Germany, [2]Potsdam Institute for Climate Impact Research, Member of the Leibniz Association, Potsdam, Germany, [3]Department of Physical Geography and Ecosystem Science, Lund University, Lund, Sweden, [4]Hawkesbury Institute for the Environment, Western Sydney University, Penrith, NSW, Australia, [5]Department of Environmental Science, Aarhus University, Aarhus, Denmark, [6]Laboratoire des Sciences du Climat et de l'Environnement (LSCE/IPSL) CEA-CNRS-UVSQ, Université Paris-Saclay, Gif-sur-Yvette, France

Abstract Forests mitigate climate change by storing carbon and reducing emissions via substitution effects of wood products. Additionally, they provide many other important ecosystem services (ESs), but are vulnerable to climate change; therefore, adaptation is necessary. Climate-smart forestry combines mitigation with adaptation, whilst facilitating the provision of many ESs. This is particularly challenging due to large uncertainties about future climate. Here, we combined ecosystem modeling with robust multi-criteria optimization to assess how the provision of various ESs (climate change mitigation, timber provision, local cooling, water availability, and biodiversity habitat) can be guaranteed under a broad range of climate futures across Europe. Our optimized portfolios contain 29% unmanaged forests, and implicate a successive conversion of 34% of coniferous to broad-leaved forests (11% vice versa). Coppices practically vanish from Southern Europe, mainly due to their high water requirement. We find the high shares of unmanaged forests necessary to keep European forests a carbon sink while broad-leaved and unmanaged forests contribute to local cooling through biogeophysical effects. Unmanaged forests also pose the largest benefit for biodiversity habitat. However, the increased shares of unmanaged and broad-leaved forests lead to reductions in harvests. This raises the question of how to meet increasing wood demands without transferring ecological impacts elsewhere or enhancing the dependence on more carbon-intensive industries. Furthermore, the mitigation potential of forests depends on assumptions about the decarbonization of other industries and is consequently crucially dependent on the emission scenario. Our findings highlight that trade-offs must be assessed when developing concrete strategies for climate-smart forestry.

Plain Language Summary Forests help mitigate climate change by storing carbon and via avoided emissions when wood products replace more carbon-intensive materials. At the same time, forests provide many other "ecosystem services (ESs)" to society. For example, they provide timber, habitat for various species, and they cool their surrounding regions. They are, however, also vulnerable to ongoing climate change. Forest management must consider all these aspects, which is particularly challenging considering the uncertainty about future climate. Here, we propose how this may be tackled by computing optimized forest management portfolios for Europe for a broad range of future climate pathways. Our results show that changes to forest composition are necessary. In particular, increased shares of unmanaged and broad-leaved forests are beneficial for numerous ESs. However, these increased shares also lead to decreases in harvest rates, posing a conflict between wood supply and demand. We further show that the mitigation potential of forests strongly depends on how carbon-intensive the replaced materials are. Consequently, should these materials become "greener" due to new technologies, the importance of wood products in terms of climate change mitigation decreases. Our study highlights that we cannot optimize every aspect, but that trade-offs between ESs need to be made.

1. Introduction

Forests contribute to climate change mitigation (Canadell & Raupach, 2008; Pan et al., 2011; Ramstein et al., 2019) through their significant role in the global carbon cycle. Annually, over 25% of global anthropogenic carbon emissions are absorbed by the terrestrial biosphere, primarily by forests (Friedlingstein et al., 2020; Harris et al., 2021; Pan et al., 2011). Consequently, Article 5 of the Paris Agreement states specifically that the world's

AGU ADVANCING EARTH AND SPACE SCIENCE

Earth's Future

10.1029/2022EF002796

existing forest carbon sinks should be conserved or enhanced (United Nations, 2015a). So far, this remains a challenging task since forest carbon sinks are vulnerable to climate change, mainly due to increased frequency and severity of disturbances (Dai, 2013; IPCC, 2014; Seidl et al., 2014; Spinoni et al., 2018).

European forests are estimated to take up 9% of Europe's emissions (Grassi et al., 2019). Since 90% of these forests are managed (IPCC, 2014), they additionally provide considerable amounts of wood products that store carbon and substitute carbon-intensive materials and fuels (Grassi et al., 2021; Harmon, 2019; Howard et al., 2021; van Kooten & Johnston, 2016). Furthermore, apart from climate change mitigation, forests provide numerous other ecosystem services (ESs) such as, for example, timber production, water regulation, local climate regulation, and recreation (Binder et al., 2017; Cordonnier et al., 2014; Mori et al., 2017). Therefore, future management strategies must also consider the continued provision of these manifold ESs (Brockerhoff et al., 2017; Díaz et al., 2006; Hua et al., 2022; Millenium Ecosystem Assessment, 2005), which is also reflected in the UN's sustainable development goals (United Nations, 2015b) and the declaration of sustainable forest management in Europe (Forest Europe, 2015a). This holistic approach of jointly considering adaptation, mitigation, and ESs in forest management has been coined "climate-smart forestry" (Kauppi et al., 2018; Nabuurs et al., 2017).

One approach to tackle the issue of assessing multiple goals in forest planning is using multi-criteria decision making (MCDM, Ishizaka and Nemery [2013], see also Uhde et al. [2015]). MCDM has been applied to find the best possible forest management regarding multiple ESs under different climate scenarios (e.g., in Spain, Diaz-Balteiro et al., 2017). Using models to develop and assess forest management portfolios—defined as the relative proportions of species and forest management alternatives—allows for a more nuanced assessment since these models integrate several ESs. In addition, portfolios can combine the benefits of multiple management options. This was recently demonstrated by Luyssaert et al. (2018) who created forest management portfolios from six simplified management options (e.g., species changes, conversion to coppice, or refraining from management) and applied these across Europe using the ORCHIDEE-CAN model. However, their optimization of three single-criterion objectives (maximize albedo, maximize carbon sink, minimize surface temperature) entailed trade-offs that must be made between the optimized objective and other ESs.

Management decisions in forestry are particularly complicated: The long life-spans of trees and changing societal demands require assumptions about the future resulting in large uncertainties. These uncertainties may be addressed with the concept of robust optimization (Ben-Tal & Nemirovski, 2002; Gorissen et al., 2015). One of the first studies that applied robust optimization to land-use portfolios was done by Knoke et al. (2015). Furthermore, several recent studies have shown how robust optimization and MCDM can be combined to optimize forest management portfolios for multiple ESs considering uncertainty (Knoke et al., 2016, 2020; Uhde et al., 2017).

Currently, one of the most uncertain aspects of forest planning is climate change and its local impacts (Lindner et al., 2014). For instance, it is challenging to find tree species that are well-suited for present and future conditions (Hickler et al., 2012). Such future uncertainty can be explored in modeling studies where various species can be assessed under different climate scenarios based on Representative Concentration Pathways (RCPs) covering several plausible trajectories of anthropogenic emissions and climates (Dufresne et al., 2013; IPCC, 2018; Meinshausen et al., 2011). However, recent studies on future forest management strategies do not embrace the full range of uncertainty; therefore ignoring the issue that a proposed forest management option deemed practical for one RCP might not be useful if another RCP materializes. Alternatively, it is crucial to broadly evaluate the full range of possible climates (Pedersen et al., 2020) and provide solutions that will be practical across all RCPs. This prevents costly adaptation if the world develops in unforeseen ways (Lawrence et al., 2020) and is also promoted by the IPCC which states that also low-likelihood outcomes need to be included in risk assessments (IPCC, 2021).

To address this, we considered RCP2.6, RCP4.5, RCP6.0, and RCP8.5, and determined how forest management portfolios may be constructed to provide all ESs across these RCPs. We combined the well-established dynamic vegetation model LPJ-GUESS (B. Smith et al., 2001; B. Smith et al., 2014; Lindeskog et al., 2021) with MCDM and robust optimization to consider the ESs climate change mitigation, timber provision, water availability, local climate regulation, and biodiversity habitat. Additionally, we covered climate change adaptation by evaluating which forest types can survive the different climate scenarios. Specifically, we focused on European forests, using six simplified forest management options similar to Luyssaert et al. (2018). Our approach allowed us to assess several ESs simultaneously under uncertain future climate development across Europe. We present a set of

Earth's Future 10.1029/2022EF002796

management portfolios for different European forest types that are viable across many future climate scenarios. We further investigated the potential trade-offs arising when implementing such portfolios, mainly by considering harvest rates and mitigation impacts. Lastly, we evaluated how different preferences and assumptions affected the portfolios.

2. Methods

We applied LPJ-GUESS for the European domain with a 0.5° × 0.5° spatial resolution and combined it with an optimization framework (Figure 1). We performed 24 simulations for $m = 6$ management options and $n = 4$ RCPs (RCP2.6, RCP4.5, RCP6.0, RCP8.5). We computed seven ES indicators (ESIs—representing the ESs climate change mitigation, local climate regulation, biodiversity habitat, timber provision, and water availability) to evaluate the different management options (Table 2). Finally, our optimization algorithm computed one management portfolio per grid cell which allowed an optimally balanced provision of all considered ESIs across all RCPs.

2.1. Dynamic Vegetation Model and Simulation Protocol

LPJ-GUESS (B. Smith et al., 2001; B. Smith et al., 2014) is a dynamic vegetation model that simulates establishment, growth, competition, management, and mortality of plant functional types (PFTs). Each PFT is represented by different parameters governing, for example, life-history strategy, phenology, growth form, drought tolerance, bioclimatic limits, and others (B. Smith et al., 2014). The modeled processes include photosynthesis and stomatal conductance based on BIOME3 (Haxeltine & Prentice, 1996), stochastic implementations of population dynamics (B. Smith et al., 2001), allocation of carbon to different compartments and allometric relationships for plant growth (Sitch et al., 2003). They also include the nitrogen cycle, soil, and litter processes (B. Smith et al., 2014). LPJ-GUESS has been designed to simulate the vegetation response under future climate change, including responses to climate extremes. As such it has been benchmarked to a wide variety of independent datasets (e.g., Chang et al., 2017; Haverd et al., 2020; Ito et al., 2017; Lindeskog et al., 2021; B. Smith et al., 2014).

Mortality is implemented as a stochastic process considering a tree's growth efficiency and age. The simulations in this study were carried out in "cohort-mode," where one average individual represents a cohort of the same age. Each stand was modeled via 25 replicate patches representing random samples of the same stand.

Disturbances in LPJ-GUESS are modeled stochastically as patch-destroying disturbances. Our study followed Pugh et al. (2019) and used different average return times for different stand types. We used their global median values of 1000 and 500 years for broad-leaved deciduous and evergreen stands, respectively, and 300 years for needle-leaved and mixed stands. Note that we used these values even though they contain management disturbances, since return times in Europe are partly much lower than these global median values (Pugh et al., 2019). Additionally, we simulated fire using the GlobFIRM fire model (Thonicke et al., 2001) in the historical and spin-up phase to retrieve a reasonable representation of present-day European forests (see Section 3.1.1); however,

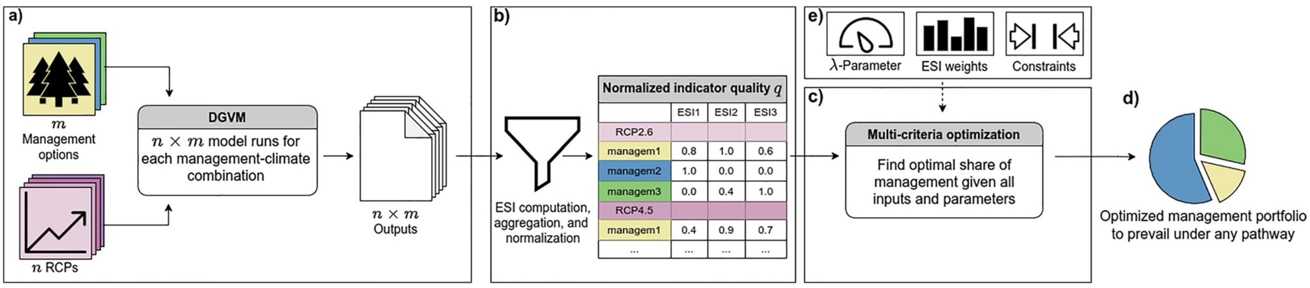

Figure 1. Visualization of the methodology, leading to one single portfolio per grid cell. Panel (a) $n \times m$ independent model simulations are run for the management options and Representative Concentration Pathways (RCPs). (b) Ecosystem service indicators (ESIs)—if not computed by LPJ-GUESS directly—are computed from the model outputs, then aggregated (e.g., 2100–2130 mean), and normalized to [0; 1], resulting in one table per grid cell containing the normalized values for all management types and RCPs. (c) For each grid cell, one optimization is run resulting in (d) one optimized portfolio per grid cell which for any RCP ensures a balanced provision of all ESIs. (e) Additional parameters may be passed into the process: A parameter $\lambda \in [0, 1]$ governs the focus on the balanced provision of the ESI. A low λ results in a more balanced provision of all ESIs whereas a high λ lays more focus on the average ESI values (and less balance) instead, see also Section 2.3. Weights can be added to put more focus on one or more ESI. Additional constraints can put bounds on the variables, for example, harvest needs to be at least at 2010 levels.

Earth's Future

10.1029/2022EF002796

we excluded fire for the future period as it was included in the disturbance intervals. Furthermore, starting in 2010, we increased the disturbance rates by 1% each year to account for expected increasing disturbance frequencies in Europe (Senf & Seidl, 2021a).

We used specific parametrizations for 18 European tree species and four more generic PFTs for three shrubs and grass (Table S1 in Supporting Information S1). We used the parameters of LPJ-GUESS v4.0 (based on the values proposed by Hickler et al. [2012]) with adaptations as specified in Lindeskog et al. (2021). However, we removed the maximum temperature of the coldest month bioclimatic limit for establishment ($tcmax_{est}$) of certain species (as in Hickler et al. [2012]) after observing that otherwise modeled present-day occurrences were too far off observations for these species (Mauri et al., 2017).

2.1.1. Representation of Forest Management in LPJ-GUESS

Several forest management options are implemented in LPJ-GUESS (Lindeskog et al., 2021). Here we applied the thinning scheme following the self-thinning rule of Reineke (1933), which was implemented similarly as in Bellassen et al. (2010) as described in Lindeskog et al. (2021). Apart from thinning, forests were clear-cut when a stand fell below a corresponding target density predefined for broad-leaved summer green and needle-leaved forests (Bellassen et al., 2010; Lindeskog et al., 2021). We defined an additional target density for broad-leaved evergreen forests (75 ind/ha) to yield comparable rotation times between the forest types to facilitate comparison of harvests between forests in the limited time period of interest (2100–2130). In achieving the correct age distribution by 2010, clear-cuts were omitted between stand establishment and 2010 (see Section 2.1.3). For stands that attained the target density before 2010, clear-cuts were spread out through 2010–2020 to avoid a significant anomaly in 2010 (see Figure S20 in Supporting Information S1). During harvesting, 65% of the total carbon of the sapwood and heartwood was considered stem material of which 90% ("harvest efficiency") were taken out. Harvests were distributed to different product pools as described in Section 2.1.4. Small fractions of leaves and branches were assumed to be taken out and burned, and the rest was left to decay on-site together with the stem residues and coarse roots (see Lindeskog et al., 2021). We also implemented salvage logging, described in detail in Section S1.1 of Supporting Information S1.

2.1.2. Coppice Management Implementation in LPJ-GUESS

Coppice is a forest type that historically was very important in Europe (Albert & Ammer, 2012; Evans, 1984), still accounting for large forest shares in Southern European countries (Maganotti et al., 2018). Therefore, we implemented a simple form of coppice management into LPJ-GUESS similar to ORCHIDEE_CAN_r3069 (Luyssaert et al., 2018). If a broad-leaved tree is cut down, its coarse roots (assumed to be 22% of the total carbon mass) stay alive, and new shoots emerge from the stump (assumption: 10 per stump for all species). Choosing the initial number of shoots after a coppice event showed no significant difference after a few years as the mortality routines of LPJ-GUESS consequently resulted in strong self-thinning in cases with a higher number of shoots (see Figure S4 in Supporting Information S1). Besides this expected self-thinning (Johansson, 2008; Leonardsson & Götmark, 2015; Rydberg, 2000; Verlinden et al., 2015) we also applied anthropogenic thinning in coppices (Nicolescu et al., 2017). The shoots were harvested individually once they reached a diameter of 20 cm, our coppice forests were thus representative of traditional coppice forests, not short-rotation coppice plantations.

2.1.3. Simulation Protocol for LPJ-GUESS Simulations

For our simulations we followed the setup of Lindeskog et al. (2021) by using monthly temperature, radiation, and precipitation data (including the number of wet days) from CMIP5 (Taylor et al., 2012) simulations of the general circulation model (GCM) IPSL-CM5A-MR (Dufresne et al., 2013). We selected this GCM to compare our results with Luyssaert et al. (2018).

The climate data was interpolated bi-linearly from $2.5° \times 1.25°$ to $0.5° \times 0.5°$ spatial resolution and bias-corrected against 1961–1990 observations from CRU-NCEP (see Ahlström et al., 2012). For each climatic variable, this data covered a transient "historical" (1850–2005) and a future period (2006–2100) per RCP.

After a 1200-year spin-up period to bring soil carbon pools close to equilibrium, the simulations were run from 1800 until 2130. Before 1850, the 1850–1879 climate was detrended and recycled. It was similarly performed with each scenario's 2071–2100 data for the time after 2100. Our optimization considered the period 2100–2130. By then, most simulated forests had undergone a full rotation (Figure S22 in Supporting Information S1).

Earth's Future
10.1029/2022EF002796

CO_2 concentrations of the RCPs were taken from Meinshausen et al. (2011), and a constant pre-industrial (year 1765) atmospheric CO_2 concentration of 278 ppmv was applied for the spin-up. Decadal values for Nitrogen deposition from 1850 to 2100 were used, depending on the grid cell and RCP (Lamarque et al., 2011).

To commence projections from a realistic representation of forests in Europe today, forest stands established during spin-up were successively clear-cut and replaced by stands matching the species distribution map from Brus et al. (2012). The timing of these clear-cuts was predefined to reproduce the observed 2010 age distribution from the global forest age data set (GFADv1.0, Poulter et al., 2018). This is why we turned off disturbances and clear-cuts in such stands between their establishment and 2010.

Old-growth forests (i.e., older than 140 years in 2010 according to GFAD) were not subject to any management before and after 2010. We kept the forest area constant at 2010 levels for future projections and focused on "forests remaining forests." Large-scale deforestation is unlikely due to the Paris Agreement (United Nations, 2015a) and national laws. Forest area increase however has recently been observed (Forest Europe, 2020). This trend might continue if land abandonment outweighs effects of increasing demand for agricultural products (Luyssaert et al., 2018; Perpiña Castillo et al., 2021; P. Smith et al., 2010). This competition is not part of our study.

We performed one simulation until 2010 for all 3124 grid cells in Europe to evaluate the present-day model outputs against observations and other models (Section 3.1.1). To create forest management portfolios, we implemented six simplified forest management options (Table 1) based on Luyssaert et al. (2018). We performed one future projection simulation (2010–2130) for each RCP and management option. For example, in the toBd ("to broad-leaved deciduous") simulation, all stands were converted to a broad-leaved deciduous forest after clear-cut. When changing a non-broad-leaved stand to coppice, the stand was initially converted to a broad-leaved deciduous forest and then managed as coppice.

We sampled grid cells every 2° latitude and longitude for future runs to reduce the computational load resulting in simulations for 193 grid cells across Europe. In the optimization, we combined the management options to create a portfolio for each grid cell since the stands within a grid cell were independent from one another. We excluded cells from the portfolio analysis where the average harvests between 1990 and 2010 were below 0.1 $gC/m^2/yr$ or less than 25% of the forest area was converted by 2100. The reason for this is that with such a low conversion, the forests looked very similar at the end of the simulation period and the difference in ESIs were only marginal, not allowing for a sensible comparison of management options. This resulted in a final set of 181 grid cells.

2.1.4. Contributions of Forests to Climate Change Mitigation

Apart from ecosystem carbon storage, the effect of forests in mitigating climate change depends on the life cycle of wood products, attributed to their ability to substitute carbon-intensive materials and fuels (Grassi et al., 2021; Harmon, 2019; Howard et al., 2021; van Kooten & Johnston, 2016). To measure this, we followed Krause et al. (2020) by using two wood product pools for medium and long-lived products, respectively, with corresponding decay rates described by two Gamma-functions (Klein et al., 2013, Figure S24 in Supporting Information S1).

Table 1
The Six Simplified Management Options

Name	Explanation
base	After clearcut, plant the same species as before, same management as before (thinning and clearcut)
toBd	After clearcut, convert to broad-leaved deciduous forest by planting the most common broad-leaved deciduous species in the stand. Continue with thinning and clearcut
toBe	Like toBd, but planting *Q. ilex*, the LPJ-GUESS broad-leaved evergreen species
toCoppice	Broad-leaved forests are clearcut and left to regrow from the stumps, needle-leaved forests are clearcut and replaced with broad-leaved species. The forest is managed as coppice from then on
toNe	Like toBd, but planting the most common needle-leaved evergreen species in the stand
unmanaged	Refrain from the clearcut and also refrain from any thinnings after this point in time, leaving the forests completely untouched

Note. A management decision was taken for each stand after 2010 as soon as it reached maturity (i.e., a target density). The conversion was implemented by planting the most common species of each forest type for that grid cell.

Earth's Future

10.1029/2022EF002796

These functions account for the fact that the entire amount of wood entering a product pool will typically remain there for some time before it starts to be returned to the atmosphere.

We based product flows on Klein et al. (2013) and Eurostat (2021a), where for conifers (non-conifers), 23% (2.5%) of stem mass entered the long-lived pool, and 9.4% (11.9%) entered the medium pool. Meanwhile, the rest (plus some harvest residues) was converted to fuel wood or short-lived products and transferred to the atmosphere within one year (Figure S19 in Supporting Information S1). For coppice, we assumed that 2% entered the medium product pool. The rest was turned over immediately, because most wood from coppice forests is used for energy, heating, and paper, and only very small portions for, for example, furniture (Maganotti et al., 2018). For the medium and long-lived pools the median residence times were 18 and 93 years, respectively, that is, 50% of products remained in that pool after this time (Figure S24 in Supporting Information S1).

To assess the emission reductions, we used displacement factors (defined as avoided emissions in relation to the mass of carbon in the wood product, Arehart et al., 2021) for fuels and materials, respectively. We used 0.67 tC/tC as the displacement factor for fuels (Knauf et al., 2015). It is based on a simple assessment of the possible energy provision of wood compared to light oil (Rüter, 2011). It is comparable to those of studies of other developed nations (Myllyviita et al., 2021) and well in the range of "less than 0.5 up to about 1.0" as suggested by Sathre and O'Connor (2010). The displacement factor for material substitution is based on an analysis of 16 key products and their alternatives, making up 90% of the wood usage spectrum (Knauf et al., 2015). Its value of 1.5 tC/tC was originally estimated for Germany and is lower than the commonly used value of 2.1 (Sathre & O'Connor, 2010) which is a mean value over studies considering different applications and countries across the world. It does not contain end-of-life assessment. We used this displacement factor for 77% of wood products and assumed energy recovery at the end of their lifetime. In Europe, 23% of all waste ends up in landfills (Eurostat, 2021b). Consequently, for the remaining 23% of products we used a lower displacement factor of 1.1 tC/tC to also account for emissions from landfills (Sathre & O'Connor, 2010).

Wood fuels emit more CO_2 per unit of energy than fossil fuels, and it requires a long time until a forest has regrown and absorbed the initially emitted carbon ("carbon debt," Cherubini et al., 2011). Consequently, wood-based fuels are not carbon-neutral, at least on short and medium time-scales (Booth et al., 2020; Cherubini et al., 2011; Holtsmark, 2012; Leturcq, 2020). It is crucial to take this into account, which is done here by considering the total carbon effect including stocks in forests and products, similarly as performed by Knauf et al. (2016). This approach quantifies the actual emissions from burning the wood.

The actual number of avoided emissions heavily depends on the energy mix at a given time or the type of materials that are replaced and is expected to decrease in the future (Harmon, 2019). We accounted for this by gradually discounting the displacement factors based on the RCPs, similar to Brunet-Navarro et al. (2021, Figure S23 in Supporting Information S1). Additionally, we evaluated our approach without discounting (see Section 4.3).

Finally, it is crucial to take into account additionality. Any measurement of mitigation from forests needs to be compared to a reasonable baseline, because only an additional measure would contribute to climate change mitigation. In our case this is done implicitly by including a baseline simulation ("base"). Any change in forestry can thus be compared to this baseline.

2.2. Ecosystem Services and Their Indicators

Our study investigated the potential changes in seven ESIs (Table 2 and Section S1.2 in Supporting Information S1). As indicators for the provision of timber we used both the simulated total harvest values ($\mathrm{t\,ha^{-1}\,yr^{-1}}$ dry mass) and harvests used only for long-lived wood products (HLP).

We used one combined indicator for a forest's (global) climate change mitigation potential by combining the total simulated carbon pool (vegetation + soil + litter + products) with cumulative avoided emissions through material and energy substitution. For local climate regulation we included evapotranspiration (ET) and surface roughness. Since the *local* effects of albedo have previously been found to be small compared to ET and surface roughness (Winckler et al., 2019), we included albedo only in an additional experiment. Although we acknowledge that biogeophysical variables also possess non-local and global effects (Pongratz et al., 2010; Winckler et al., 2017), these might be small compared to the biogeochemical effects (Pongratz et al., 2010). Therefore, we solely evaluated increases in these variables as an indication for local cooling, omitting possible feedback on global

Earth's Future

10.1029/2022EF002796

Table 2
The Ecosystem Service Indicators Used in the Main Part of This Study

Variable Name	Ecosystem service indicator (ESI)	Explanation
Harvests	Total harvests	Total wood provision (including products like firewood, pulp etc.)
HLP	Harvests for long-lived wood products	Timber provision for long-lived products (e.g. construction)
Mitigation	Carbon storage plus material and energy substitution effects	Total carbon in vegetation, soil, litter, and products, plus avoided emissions from substitution with wood products
z_0	Surface roughness	Indicator for atmospheric conductance, influencing heat fluxes. Higher roughness results in higher fluxes
ET	Total evapotranspiration	Indicator for latent heat fluxes. More ET means more local cooling
Ψ_{soil}	Soil water potential	Yearly minimum of monthly values, indicator of water availability and drought stress
Bio	Combined indicator (RCP-normalized mean) of amount of coarse woody debris, Shannon entropy of 5 cm diameter-at-breast-height (DBH) classes and number of trees with DBH >50 cm	Coarse woody debris, large trees, and an abundance of various tree sizes provide high numbers of habitats and resources (Cordonnier et al., 2014)

Note. Albedo was included in an additional experiment. See Section S1.2 in Supporting Information S1 for technical details.

climate. The yearly minimum of monthly soil water potential Ψ_{soil} was used as an indicator for water availability (Rajasekaran et al., 2018), where a decrease in Ψ_{soil} indicates less water available for plants. We implemented a combined indicator for biodiversity based on tree sizes and dead wood, adapted from Cordonnier et al. (2014). Due to the model setup (i.e., planting the most common species of the grid cell), it was not sensible to include an indicator for the diversity of tree species. Finally, to assess the changes in ESI performances, we compared their values at the beginning of the next century (mean of 2100–2130) to present-day values (mean of 2000–2010).

2.3. Robust Multi-Criteria Optimization

To compute portfolios that provide various ESs of different units and magnitudes, we used MCDM (e.g., Ishizaka & Nemery, 2013; Marler & Arora, 2004). To account for uncertainty in the inputs stemming from the RCPs, we combined this with robust optimization (Ben-Tal & Nemirovski, 2002). The solution of such an optimization guarantees feasibility for all realizations of the uncertainty set and can be written as a maxi-min linear program (Gorissen et al., 2015). To enable comparison, the computed ESIs from the simulations were first normalized to [0, 1]. This was done for each ESI and RCP separately, since ESI values of different RCPs do not belong to the same environmental conditions and we argue that they cannot be compared to an ESI from an entirely different climatic future. Our normalization thus yields a better interpretability of the worst case results: When the worst case ESI is, say, 0.4, this means that in every RCP we will achieve 40% of the achievable outcome in this emission pathway. For each grid cell this resulted in one linear program creating a single portfolio for all RCPs:

$$\max_{\omega} \left((1-\lambda) \min_{esi,rcp} \sum_{s} \omega_s W_{esi} q(esi, s, rcp) + \lambda \sum_{esi,rcp} \sum_{s} \omega_s W_{esi} q(esi, s, rcp) \right) \tag{1}$$

$$\text{subject to} \qquad \sum_{s \in S} \omega_s = 1 \tag{2}$$

$$\omega_s \geq 0 \qquad \forall \ s \in S \tag{3}$$

$$\sum_{esi} W_{esi} = 1 \tag{4}$$

AGU ADVANCING EARTH AND SPACE SCIENCE

Earth's Future

10.1029/2022EF002796

$$W_{esi} \geq 0 \qquad \forall\ esi \tag{5}$$

$$\text{fpc}(2100, s, rcp) \geq \min(0.1, \text{fpc}(2010)) \tag{6}$$

where $S = \{\text{base}, \text{toBd}, \text{toBe}, \text{toCoppice}, \text{toNe}, \text{unmanaged}\}$

ω_s : Share of management type s in the optimized portfolio

W_{esi} : Preference / weight for esi

$\text{fpc}(year, s, rcp)$: Foliar Projective Cover of the grid cell under management option s in RCP rcp in year $year$

$q(esi, s, rcp)$: Normalized quality of esi for management option s in rcp

$\sum_{s \in S} \omega_s q(esi, s, rcp)$: Quality of esi for the whole grid cell for a portfolio ω (without ESI weights)

The objective function, Equation 1, expresses that we wanted to find the portfolio ω that results in optimally balanced ESIs over all RCPs. It is a weighted combination of the worst case ESI and the average ESI. Optimizing the worst case would lead to the most balanced solution but is quite pessimistic, since a bad performance of one ESI cannot be compensated with the excellent performance of another ESI. Meanwhile, optimizing the average ESI, results in a much less balanced solution but could improve some ESIs. In the theory of sustainability, the former approach termed "strong sustainability" (Ruiz et al., 2011) is preferred (Diaz-Balteiro et al., 2018). We combined the two approaches (Diaz-Balteiro et al., 2018) by adding a trade-off parameter $\lambda = 0.2$ that resulted in balanced ESI performances and robustness to small changes in ESIs, thus obtaining more similar results in adjacent grid cells. We argue that a higher λ-value is unreasonable since it would invalidate the approach of treating all RCPs and ESIs equally. Also, mathematically, a higher λ would decrease the relevance of the worst cases (left summand) due to their naturally lower value than the averages (right summand).

Equations 2 and 3 ensure that our portfolio contained non-negative fractions for all strategies and allotted the whole managed forest area of the grid cell. Additionally, weights can be given to each ESI (Equations 4 and 5). Throughout this study we applied equal preferences W_{esi} unless otherwise noted. Finally, Equation 6 ensures the maintenance of forest cover at the beginning of the next century, using a value of foliar projective cover of 10% based on FAO's definition of forests (FAO, 2020), or the 2010 foliar projective cover if lower than 10%. Notably, violation of this equation led to a management option not to be considered for a grid cell, since such a low foliar projective cover indicates that this type of forest does not grow well under the given conditions. This also led to the exclusion of this management option from the normalization process and the q-values for the other management options. The normalization was excluded from the equations above for brevity, and the optimization was implemented using SciPy (Virtanen et al., 2020).

3. Results

3.1. Simulation Results

3.1.1. Model Evaluation

We found that LPJ-GUESS reproduced present-day data from observations and models (Table 3, Figures S6–S8 in Supporting Information S1). For example, modeled vegetation measures such as tree cover, biomass, net and gross primary productivity (NPP and GPP), ET, and runoff were close to estimates from the literature, on average over the continent and more regionally. Total carbon was overestimated. Additionally, simulated fellings were above values from country reporting. We found a historical increase in GPP of 30% in response to changing environmental conditions such as increased CO_2-concentrations from 1900 to 2010 (Figure S1 in Supporting Information S1).

3.1.2. Projected Changes in ESIs Until 2100–2130

In our simulations, a European forest landscape with solely unmanaged forest by 2100–2130 resulted in the highest carbon stocks (including products) with 12.1–12.4 kgC/m^2 (Tables S3–S6 in Supporting Information S1) over the 181 analyzed grid cells which is on average 19% above present-day values and 24% above the base scenario. Other management options generally resulted in carbon stocks below present-day values in all RCPs.

Earth's Future

10.1029/2022EF002796

Table 3
Comparison of Our Present-Day (2000–2010 Average) Simulation Outputs to Results From Other Studies and Publicly Available Datasets

	This study	Literature	Data source[a]	Reference
Vegetation C (PgC)	13.7	13	I	Pan et al. (2011)
		11.7[b]	C	Forest Europe (2015b)
		11.6[c]	S	Liu et al. (2015)
Total C (PgC)	59.8	40.9[d]	I	Pan et al. (2011)
Forest-GPP (gC/yr/m^2)	1175	1107–1199	E, F	Luyssaert et al. (2010)
Forest-NPP (gC/yr/m^2)	426	447	I	Luyssaert et al. (2010)
Fellings (10^6 m^3 yr^{-1})	663	562	C	Forest Europe (2015b)
Total Tree Cover (%)	24.9	26.8[e]	S	GFC v1.7 Map (Hansen et al., 2013)
ET (mm/yr)	459[f]	490	S, M	GLEAM (Martens et al., 2017)
Runoff (mm/yr)	286[f]	297	G, M	UNH-GRDC (Fekete et al., 1999)

Note. The considered region is the European continent excluding Russia, Iceland, Cyprus, and Malta, but including Kaliningrad region and European part of Turkey, resulting in 3124 grid cells for our simulation. Values are representative only for the forest area of this region, except for tree cover, ET, and runoff, which are averaged over the entire region.

[a]Data source meanings: S, satellite-derived; I, inventory data; C, country reporting; M, modeled; G, Gauge-observations; F, flux measurements from ecological site; E, ecological site studies. [b]Using aboveground biomass = 79% of vegetation biomass. [c]Includes Turkey. [d]Only upper 100 cm of soil considered in their study. Our value is for the upper 150cm. [e]Tree cover here refers to canopy closure for all vegetation taller than 5 m (GFC) and LPJ-GUESS simulated crown cover of all simulated forests. GFC contains tree cover also for non-forest areas. [f]Assuming C3 grass growing everywhere outside of forest areas.

The values for simulated total harvest increased with higher-emission RCPs. The toNe management option showed higher harvests (2.9–3.5 m^3 ha^{-1} yr^{-1}) than toBd (2.1–3.3 m^3 ha^{-1} yr^{-1}, Figure S20 in Supporting Information S1), whereas converting to coppice caused strong reductions in harvests. Harvests for long-lived products (HLP) were much higher in the toNe management option than in toBd and lowest in coppice forests (Figure S21 in Supporting Information S1). For the unmanaged option harvests converged to zero by the end of the simulation period, as stands were successively left untouched.

Climate mitigation strongly depended on the RCP. For RCP2.6, the unmanaged forest had a higher cumulative mitigation potential than the base management (1.6 vs. 0.8 kgC/m^2), the situation was opposite for RCP8.5 (1.9 vs. 2.2 kgC/m^2).

The broad-leaved deciduous forests showed the highest roughness values, among other reasons due to the low plant area index in winter after senescence and a higher forest density than the needle-leaved evergreen forests. Also, the unmanaged forests showed higher surface roughness, majorly attributed to their tall trees.

Evapotranspiration was projected to be highest in coppices in our simulations (505 mm yr^{-1} on average) but showed higher variation between the RCPs (Figure S5 in Supporting Information S1). Additionally, the management options revealed small differences in ET when averaging over the continent, with much larger differences locally (not shown).

The simulations also showed that soil water potential varied between RCPs due to different projected patterns in rainfall and CO_2 concentrations (Figure S10, Tables S3–S6 in Supporting Information S1). Soil water potentials of the base management option were projected on average 10% lower than present-day values. Different vegetation and management affected the soil water potential, with the lowest values in coppice (around −2.1 MPa).

Higher biodiversity habitat provision was observed in the unmanaged stands compared to other forest types. It was also higher in the toBd option compared to toNe, which had higher values than coppice.

3.2. Optimized Management Portfolios

3.2.1. Optimized Portfolios and Species Shares in European Regions

The portfolios were optimized for the 2100–2130 period but it is important to keep in mind that the different management transformations occurred gradually (Figure S22 in Supporting Information S1). Across Europe we

ADVANCING EARTH AND SPACE SCIENCE
Earth's Future
10.1029/2022EF002796

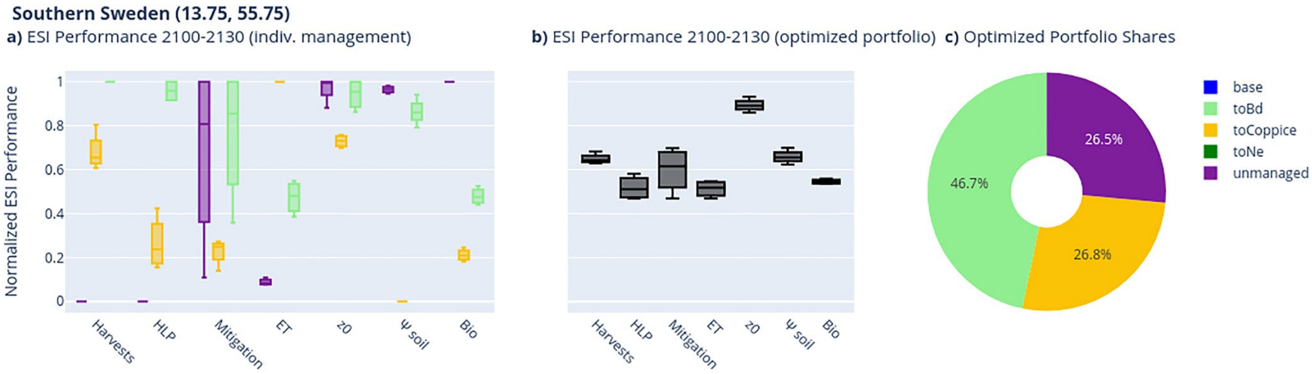

Figure 2. Example of the multi-criteria robust optimization approach for one grid cell. (a) Each box plot shows the spread of the normalized performance of the ecosystem service indicators (ESIs) over the four Representative Concentration Pathways (RCPs) when converting the forest of the entire grid cell to broad-leaved deciduous (toBd, green), to coppice (toCoppice, yellow), or to unmanaged forest (purple). ESIs for the other management options are omitted for clarity. Panel (b) is similar to (a), but for the optimized portfolio. Panel (c) shows the distribution of management options in the optimized portfolio for this grid cell. The figure illustrates how balanced the optimized portfolio is (b): regardless of the RCP, every ESI will be provided at a reasonable level (i.e., no ES performance will deteriorate in the future), whereas any other single management option would lead to very different performance of different ESIs. A value of 1 here means that this is the best attainable performance of an indicator given all management options. For example, in (a), in this grid cell, ET is highest for coppice for all RCPs, which is why the box plot is reduced to a single point at 1. This is not the case for, for example, unmanaged forest regarding mitigation: The box plots spans from 0.1 to 1 meaning that for some RCPs, mitigation is best for unmanaged, but for other RCPs it is quite bad (though never the worst, since the box does not go down to 0).

generally found that the optimized portfolios were quite diversified, comprised a large area of newly unmanaged forests, net increases in broad-leaved species shares, relevance of broad-leaved evergreen taxa in the South, and a divided importance of coppice (Figure 3).

Also, continually leaving a high share of the currently managed forest unattended played an important role in the optimized portfolios to maintain balanced ESIs. This share, 29% (Figure 3a), was similarly high across the different regions in Europe.

According to our optimization, the optimized species distribution generally shifted toward more broad-leaved species, especially in Northern Europe (Figure 3d) where the current high share of conifers decreased from 86% to 51%. Transitions in species proportion were visible in all regions with a net shift to more broad-leaved taxa from 38% to 55% by 2100–2130 across Europe (Figure 3a). However, for some grid cells the optimization caused a shift from broad-leaved species to needle-leaved species corresponding to 11% of current broad-leaved forests (mainly in Southern Europe). In comparison, roughly 34% of current needle-leaved forests were converted to broad-leaved ones by the next century (Figure 4). This transition included an increased extent of broad-leaved evergreen species (represented in this simulation by *Quercus ilex*) in Southern Europe with 22% of forests in that region consisting of broad-leaved evergreens (in the toBe, base, and unmanaged fractions of the portfolios).

According to our optimization, large regional differences were observed for coppice in the future forest management with no forests in the Southern and Alpine regions managed as coppice. At the same time, it had higher importance in Atlantic, Northern, and Continental grid cells (7%, 14%, and 7%, respectively, Figures 3c–3e).

3.2.2. Optimized Portfolio Compared to Present-Day Values and Base Management

According to our optimized portfolios (row "optimized" in Table 4 and Tables S3–S6 in Supporting Information S1), total carbon stocks were 1%–5% above present-day levels by the beginning of the next century. On the other hand, maintaining the base management (present-day species composition) in all regions across Europe decreased stocks by 3%–7%. The base management resulted in many regions becoming carbon sources, particularly in Central Europe and Southern Scandinavia (see Figure 5b, for visualization of RCP 4.5). Instead, the optimized portfolio provided more carbon storage in almost all grid cells than the base scenario (Figure 5d).

The mitigation ESI was higher in the optimized portfolio than base management in RCPs 2.6 and 4.5 whereas it was about equal in RCPs 6.0 and 8.5. When considering additionality, this means that in the lower RCPs, the optimized portfolio offered a positive net impact on climate change mitigation compared to base management, whereas in higher RCPs, there was no net effect on global climate.

AGU ADVANCING EARTH AND SPACE SCIENCE
Earth's Future
10.1029/2022EF002796

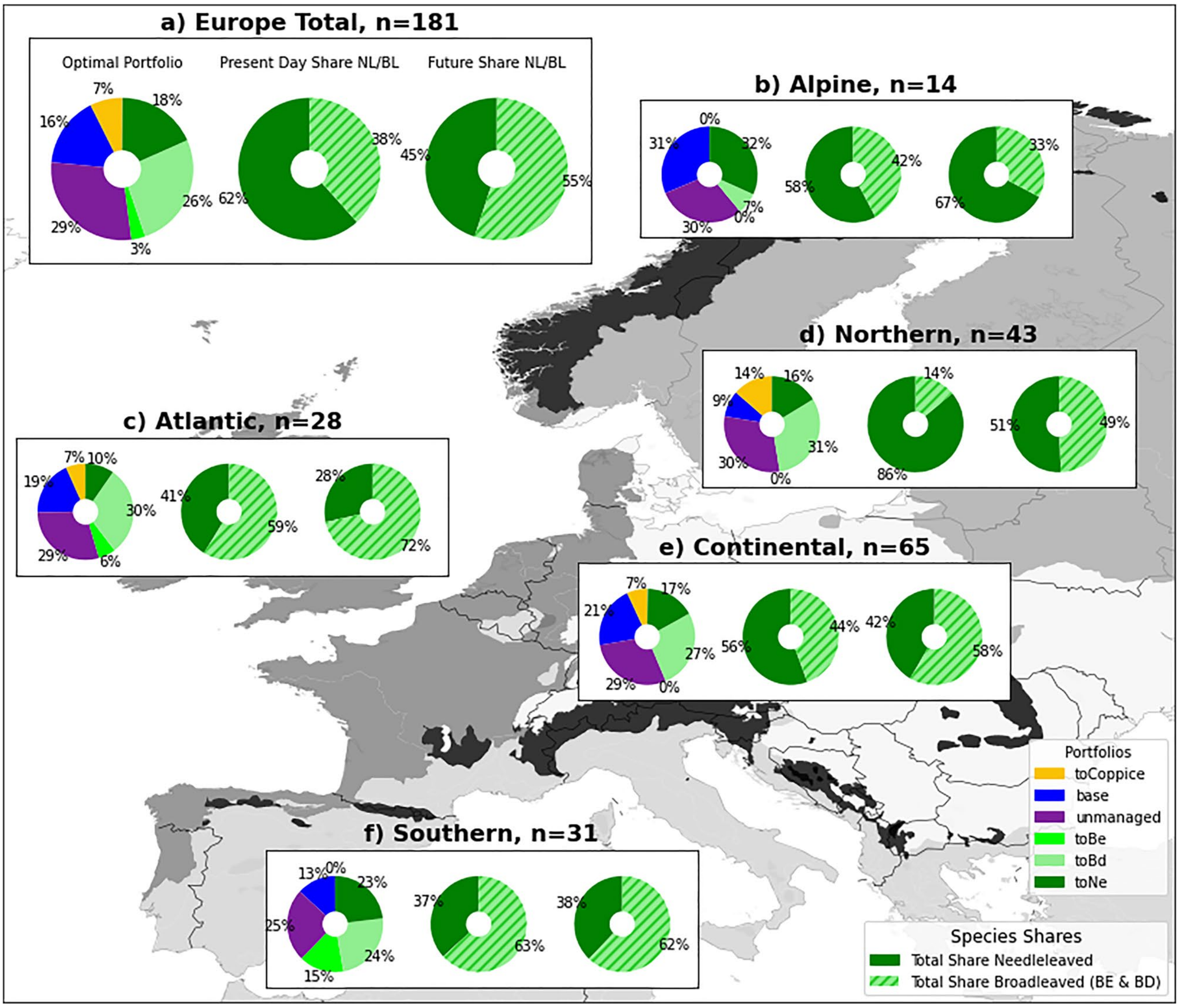

Figure 3. Proposed management portfolios in Europe, aggregated over all simulated grid cells (a) and for different regions (b–f). The five regions (indicated by different shades of gray) are based on the climatic zones of Metzger et al. (2005) but aggregated as in the IPCC AR5 (Kovats et al., 2015). The leftmost chart in each box always shows the optimized share of management options in the entire region. The middle and right charts show the current and future shares of needle-leaved and broad-leaved tree species, respectively. The depiction of broad-leaved shares is shaded as it may contain both deciduous and evergreen species. Note that portfolios and future species shares differ between cells of a region. The computation is done for each grid cell independently (Figure S3 in Supporting Information S1). Here we show the forest-area-weighted aggregated shares.

The increase in carbon storage came at the cost of lower harvests compared to the base management option. Harvests for the optimized portfolio were simulated to be 4%–29% lower than present-day values, whereas for base they increased by 2%–33% depending on the RCP (Tables S3–S6 in Supporting Information S1). Additionally, Figure 6 shows the spatial patterns of these changes in the harvest volumes. Harvests were highest in Scandinavia, both today and in the future; however, in the optimized portfolio they decreased in various parts of Europe (Figure 6c).

Regarding the biogeophysical effects, roughness length was 17%–26% higher for the optimized portfolio than in present-day and 18%–21% higher than the future base values. These increases occurred in most regions except for a few cells (Figure S17c in Supporting Information S1). The optimized portfolio showed spatially different changes in ET with increases in Central, Eastern, and Northern Europe; however, they decreased in Atlantic and Southern regions (Figure S16c in Supporting Information S1). The effect of optimization regarding ET was

Earth's Future 10.1029/2022EF002796

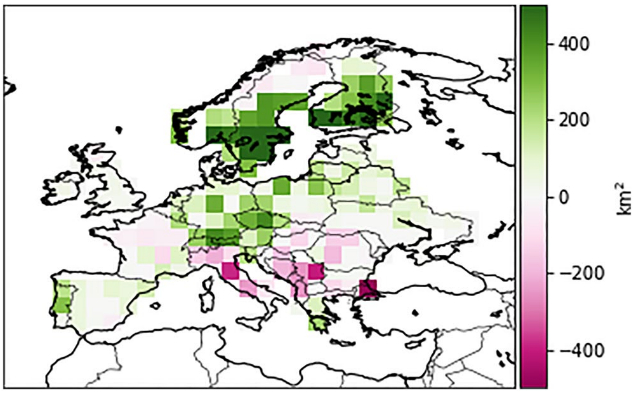

Figure 4. Net forest conversion from needle-leaved to broad-leaved forests in km². Negative values indicate a conversion from broad-leaved to needle-leaved forests.

small compared with the base management according to our simulations (Figure S16d in Supporting Information S1).

3.2.3. Optimized Portfolios With Alternative Assumptions

The previous results were based on the standard settings equally valuing all ESIs. However, decision-makers will likely encounter specific constraints or have regional preferences for some ESIs. Therefore, we briefly express our results when adding weights or constraints to the optimization.

First we found that counteracting the decrease in harvests by adding a constraint to keep total harvest at present-day levels in every grid cell is an unsuitable strategy: In about a fourth of the analyzed grid cells, such harvest levels could not be maintained until 2100–2130 for all RCPs since many stands did not reach the density and size conditions for a clear-cut. This made the optimization infeasible for many grid cells as the harvest constraint could not be met.

However, when valuing harvests twice as important as all other ESIs ("double harv"), the unmanaged fraction of the optimized portfolio was reduced to 18% (Figure S11 in Supporting Information S1). Harvests were higher than in the default optimization, ranging around the present-day levels when aggregating over the continent, nonetheless, they depended highly on the RCP (−16% to +12%, Table 4 and Tables S3–S6 in Supporting Information S1), with lower values for HLP.

Doubling the importance of climate mitigation ("double mit") resulted in a higher share of unmanaged forest such that 41% of currently managed European forests were proposed to be successively left untouched (Figure S12 in Supporting Information S1). This resulted in carbon stocks 5%–9% higher than present-day values, with a 4% higher mitigation potential than the standard optimization. There were also benefits for other ESIs compared with the standard optimization, but with even lower harvests than in the standard optimization portfolio. Additionally, coppice virtually vanished from the continent. Adding summer and winter albedo as ESIs did not change the overall portfolios much (Figure S14 in Supporting Information S1).

Furthermore, optimizing for RCP2.6 only led to slightly larger fractions of unmanaged forest (31% vs. 29%) and lower shares of coppice forests (3% vs. 7%). Also the total share of broad-leaved forests decreased from 55% to 49%

Table 4
Ecosystem Service Indicator Mean Performances if One Management Were Applied to the 181 Grid Cells

	C-Pool	Mitigation	Harvests	HLP	ET	$\Psi soil$	$z0$	Bio
Unit	kgC/m²	kgC/m²	m³/ha/yr	m³/ha/yr	mm/yr	MPa	m	unitless
base	9.8(±0.2)	1.7(±0.6)	2.99(±0.36)	0.77(±0.09)	495(±22)	−1.55(±0.10)	0.53(±0.01)	0.47(±0.01)
toBd	10.1(±0.1)	1.7(±0.7)	2.77(±0.52)	0.43(±0.07)	498(±17)	−1.66(±0.12)	0.75(±0.03)	0.55(±0.02)
toBe[a]	3.0(±0.1)	0.5(±0.2)	0.78(±0.08)	0.12(±0.01)	457(±47)	−2.76(±0.08)	0.41(±0.02)	0.08(±0.00)
toCoppice	9.9(±0.1)	1.3(±0.5)	1.77(±0.35)	0.11(±0.01)	505(±22)	−2.12(±0.10)	0.60(±0.03)	0.21(±0.01)
toNe	9.7(±0.2)	1.7(±0.6)	3.20(±0.29)	0.98(±0.10)	490(±25)	−1.44(±0.07)	0.40(±0.01)	0.42(±0.02)
unmanaged	12.2(±0.2)	1.8(±0.2)	0.08(±0.01)	0.02(±0.00)	492(±23)	−1.53(±0.07)	0.66(±0.04)	0.77(±0.04)
Present	10.3	0.0	2.48	0.65	501	−1.41	0.52	–[b]
Optimized	10.7(±0.1)	1.7(±0.5)	2.10(±0.29)	0.44(±0.05)	498(±22)	−1.64(±0.09)	0.63(±0.02)	0.57(±0.01)
Double harv	10.4(±0.2)	1.7(±0.6)	2.47(±0.32)	0.55(±0.06)	498(±22)	−1.62(±0.09)	0.62(±0.02)	0.54(±0.01)
Double mit	11.0(±0.1)	1.8(±0.5)	1.75(±0.26)	0.34(±0.04)	495(±21)	−1.58(±0.09)	0.67(±0.02)	0.64(±0.00)

Note. The values are the area-weighted means averaged over the 4 RCPs, values in brackets are standard deviation and indicate the spread over the RCPs. The lower rows of the table correspond to the optimized portfolio in the standard setting ("optimized") as well as for the experiments with different weights and constraints as described in the text.

[a]toBe management was only feasible in few regions in Europe, the reported values for this case are only for those regions ($n = 38$). In the optimized portfolio of a grid cell, only management was considered that ensured forest cover by the end of the century. [b]Computation of the Bio indicator is not sensible for present day, as it is solely a relative value between management options.

Earth's Future 10.1029/2022EF002796

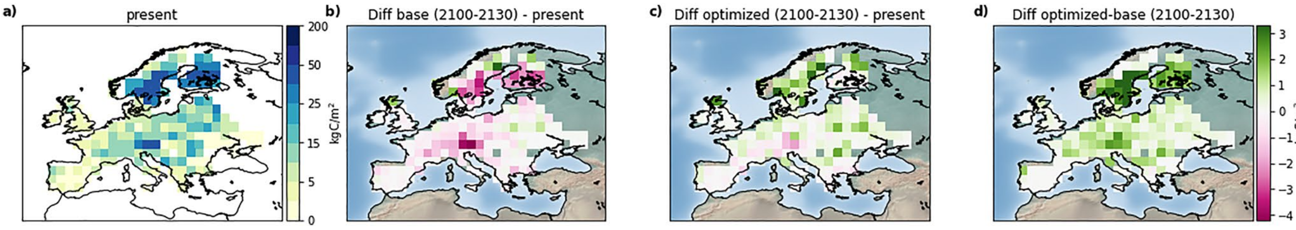

Figure 5. (a) Simulated present-day and (b–d) future carbon storage (above and below ground in kgC/m^2) in European forests for RCP4.5. Applying the base management (keeping species shares at present-day proportions, see Table 1) mainly led to decreases in carbon stock across Europe (b), the optimized management portfolio led to stable or increased carbon stocks in most parts of Europe (c), particularly in the East and North. By 2100–2130, the optimized portfolio had higher carbon stocks throughout Europe compared to the base scenario (d).

(Figure S15 in Supporting Information S1). Keeping displacement factors constant resulted in lower fractions of unmanaged forests (17%; in the Northern and Atlantic region around 10%), but a stronger focus on broad-leaved trees and coppice, compared to the standard optimization (Figure S13 in Supporting Information S1).

4. Discussion

4.1. Evaluation of LPJ-GUESS Simulation Results

LPJ-GUESS was able to reproduce present-day forest vegetation characteristics (Table 3 and Figures S6–S8 in Supporting Information S1) on which we based our management scenarios. However, soil carbon (and consequently total carbon) in our simulations was larger than estimates. This could result from the fact that we did not account for forest management prior to 1871. Since wood was never removed from the forest, trees that died simply decomposed on-site (Lindeskog et al., 2021). However, the estimate of Pan et al. (2011) also only considered the first 100 cm of the soil layer. A crucial point in projections via DGVMs is the response to changing environmental conditions, including elevated CO_2-concentrations. In our runs, the increase in GPP between 1900 and 2010 was around 30% as observations also suggest (Figure S1 in Supporting Information S1, see Campbell et al., 2017). Regarding historical changes in the land sink, LPJ-GUESS shows an intermediate response compared to other models used in the Global Carbon Budget (Figure S2 in Supporting Information S1, see Friedlingstein et al., 2020).

4.2. Species Composition of Future Forests

According to our simulations and optimizations, implementing climate-smart forestry and adapting European forests to be productive under uncertain future climate conditions may imply strong changes in their composition, as well as trade-offs that need to be addressed.

First of all, throughout the continent a shift toward more broad-leaved species is proposed by our optimization. This result aligns with the reduce-air-temperature-portfolio of Luyssaert et al. (2018). However, it is in stark contrast to their maximize-carbon-sink-portfolio, proposing higher shares of conifers in the future. Since they optimized single objectives, their maximization of carbon sink focused mostly on coniferous forests due to their overall higher volumes and avoided emissions from substitution effects. On the other hand, our optimization

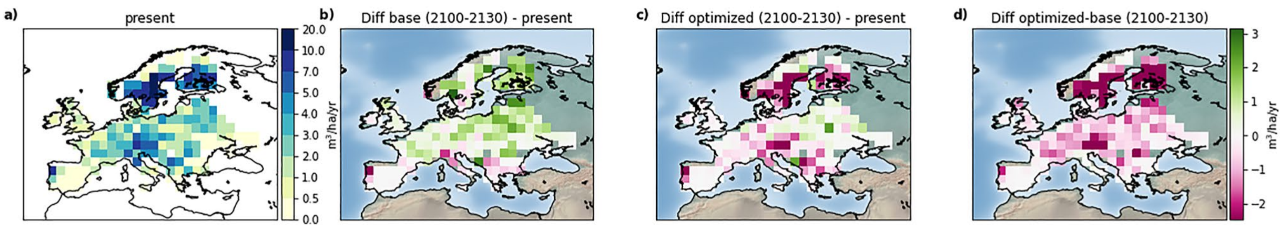

Figure 6. (a) Simulated present-day and ([b–d] future harvest; in m^3 ha^{-1} yr^{-1} dry biomass) in European forests for RCP4.5. Harvests decreased in most regions in Europe in the optimized portfolios (c) but mostly increased if present-day management was extended into the future (b), leading to lower harvests for the optimized scenario compared to base (d). The main reason for this is the high share of unmanaged forests throughout Europe in the optimized portfolios.

Earth's Future

10.1029/2022EF002796

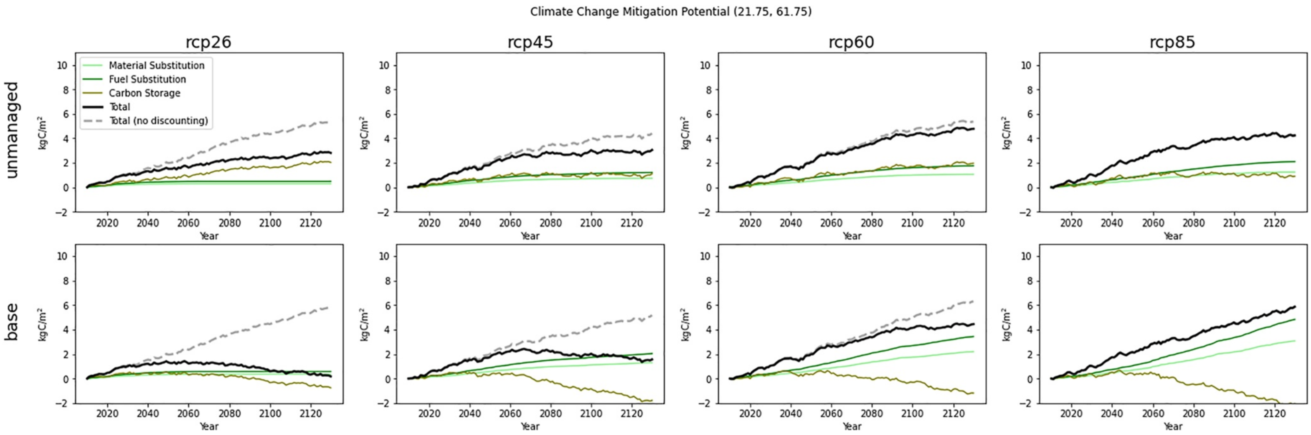

Figure 7. Climate change mitigation of forests highly depends on the Representative Concentration Pathway (RCP): In this grid cell in Finland (21.75, 61.75), continuing the base management led to higher mitigation in RCP 8.5 compared to successively leaving the forest unmanaged (same species). This was different in low-RCP scenarios where the substitution of carbon-intensive fuels and materials through the usage of wood products was assumed to abate rather quickly, making the carbon storage in forests and products the main driver for mitigation potential. The reason for there still being harvests in the unmanaged option is that the change to unmanaged forest only happened gradually, since only once each stand reached maturity a decision about future management was made. See also Figures S20 and S22 in Supporting Information S1. The dashed lines show the mitigation if constant discounting factors are assumed, where in all RCPs, wood products keep avoiding the same amount of emissions today and in the future.

considered multiple ESIs, dampening the importance of conifers due to their negative performance in terms of other ESIs. Additionally, we assumed a higher risk of disturbance for needle-leaved compared to broad-leaved species (Figure S25 in Supporting Information S1, Pugh et al., 2019; Seidl et al., 2017), another reason for the lower needle-leaved shares in our portfolios. Since multiple factors govern air temperature, the reduce-air-temperature portfolio of Luyssaert et al. (2018) implicitly optimized for more than one variable. Therefore, its similarity to ours, in terms of species composition, is not surprising.

Propositions for more broad-leaved forests in Europe have previously been made regarding lower summer temperatures (Schwaab et al., 2020) and lower fire risk (Astrup et al., 2018). Felton et al. (2010) indicated similar potential benefits for biodiversity as our approach, particularly for replacing of coniferous monocultures in Scandinavia. Nevertheless, it must be noted that in our study, those broad-leaved forests were modeled using stands of one or two species. An actual implementation should naturally include more diverse forests.

Second, our simulations suggested a large (but gradual) shift toward more unmanaged forest attributed to its benefits for biodiversity, water availability, and surface roughness (see, e.g., Figure 2). These benefits of unmanaged forests regarding multiple ESs were also found in an MCDM case-study by Diaz-Balteiro et al. (2017). Also, we found that the role of unmanaged forests was divided regarding global climate change mitigation, strongly dependent on the assumptions about the decarbonization of the construction sector (Section 4.3, Figure 7). Luyssaert et al. (2018) similarly showed a 30% fraction of unmanaged forest in their maximize-carbon-sink-portfolio (note that their values reflect the entire European forest whereas we considered only the currently managed part). On the other hand, their reduce-air-temperature-portfolio only contained 19% of unmanaged forest. This matches our optimization results when we kept displacement factors constant as they did (see Section 3.2.3 and Figure S13 in Supporting Information S1). This emphasizes the importance of a thorough analysis of wood substitution effects and the carbon-intensity of other sectors (more detail in Section 4.3).

Third, coppice, currently an abundant forest type in Southern Europe with areas well above one third of total forest area in some countries (Maganotti et al., 2018), is proposed to vanish from Southern European forests by our optimization. The optimization penalized its disadvantages such as the missing provision of long-lived wood products, low surface roughness and water availability (Figure 8), the latter agreeing with findings by Drake et al. (2009), see also Hartwich et al. (2014). This computed decrease in coppice forests due to ES provision complements an observed and proposed transition of coppice forests toward other management forms in some regions of Southern Europe, in part due to economical aspects (e.g., Stajic et al., 2009; Vacchiano et al., 2017). In other areas where coppice is currently less relevant (but potentially used to be), its role could increase as the high LAI of dense coppice forests leads to high transpiration rates and consequently local cooling.

ADVANCING EARTH AND SPACE SCIENCE

Earth's Future

10.1029/2022EF002796

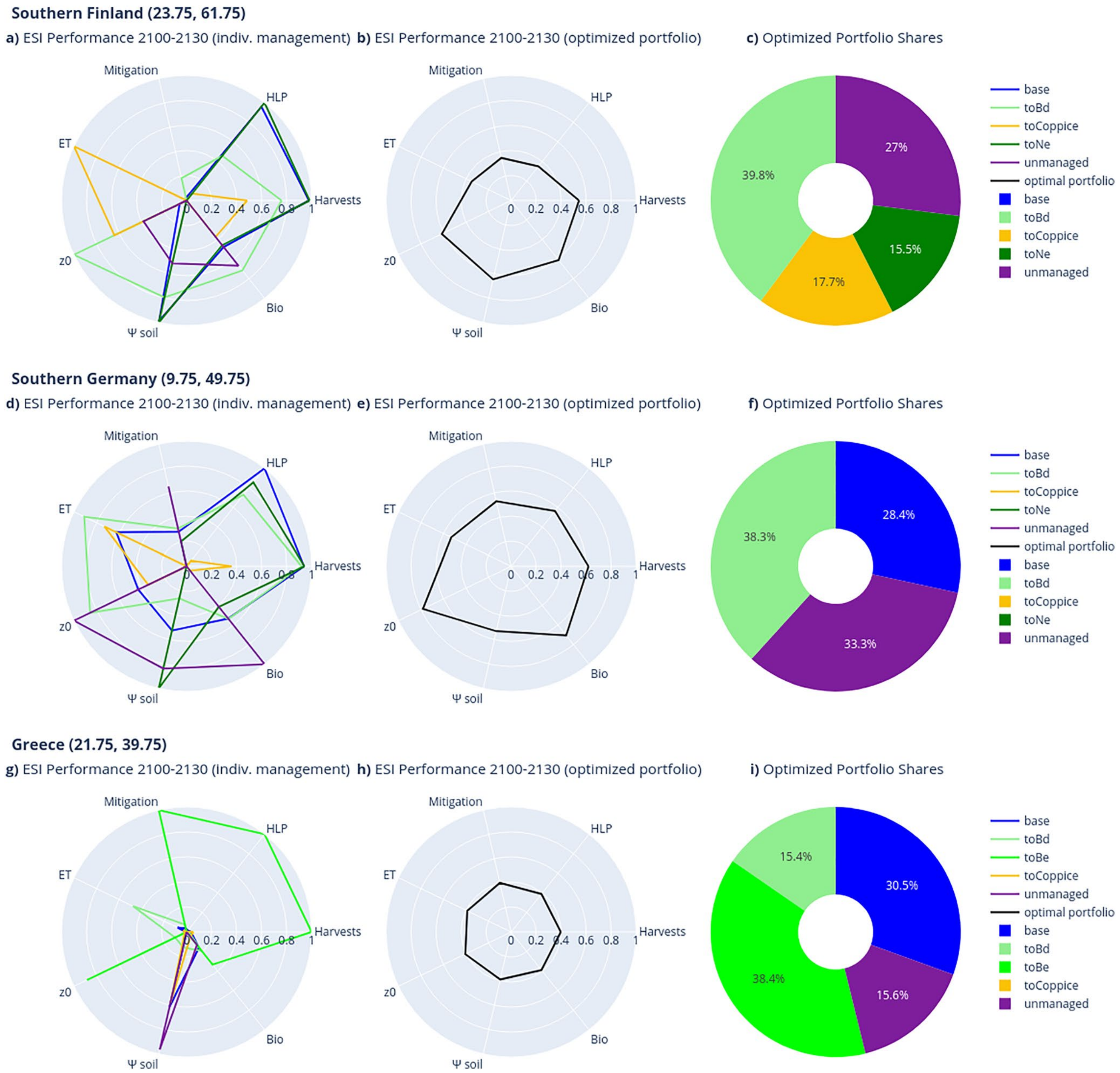

Figure 8. Examples of the optimization across a North-South gradient. Plots (a, d, and g) show the *worst cases* over the Representative Concentration Pathways (RCPs): a value of 1 in these radar charts thus indicates that this management had the best performance for this Ecosystem service indicator (ESI) in any given RCP. A value of 0 consequently does not necessarily mean that this management was always the worst for this ESI, it only means that this management had the worst performance *in at least one RCP*. The optimized portfolio of the grid cells contained multiple forest types (c, f, and i) because each type had its own advantages and disadvantages (a, d, and g). Coppice was mixed into the portfolio for the Finnish cell (c) but not into that of the cell in Germany (f) because in the latter, the merits of coppice (i.e., ET) were not high enough to outweigh its disadvantages ([d] compare to [a]). The grid cell in Greece was dominated by broad-leaved evergreen forests (i) because it had many advantages. It contained a much smaller fraction of unmanaged forests because other than Ψ_{soil}, they performed poorly compared to the other options (g). Panels (b, e, and h) also show how the proposed portfolio provided the ESIs in a very balanced fashion: In the worst cases, all ESIs were provided at a balanced level (compare Figure 2). Note that we used $\lambda = 0.2$, thus not only optimizing for the worst case, but partly also for the average ESI performance which explains some peculiarities, for example, the high share of base management in (i).

Fourth, our simulations indicated a trend to more drought-tolerant species especially in Southern Europe attributed to the low availability of water (Figure S10 in Supporting Information S1). Also, in our simulations, drought-induced higher mortality was intensified by lower regeneration due to the additional warmer winter temperatures that led to decreased natural reproduction. Consequently, species that are more drought-tolerant and do not rely on winter chilling such as *Q. ilex* and *P. halepensis* were indicated to dominate in those regions, a result that is supported by the literature (López-Tirado & Hidalgo, 2018).

4.3. Forests and Climate Change Mitigation

The European forest carbon sink has been stable in recent decades (Grassi et al., 2019; Pan et al., 2011) but changes to forest management strategies already have been proposed necessary to sustain it (Nabuurs et al., 2013). One means of sustaining or increasing the sink could be to increase the forest area which happened in the past decades (Forest Europe, 2020) and could continue to do so (Perpiña Castillo et al., 2021). This however is not part of our study as we focused solely on "forests remaining forests." According to our simulations, keeping the area of managed forest constant will transform the sink into a source (Table 4, Tables S3–S6 in Supporting Information S1); however, leaving some forests unmanaged could sustain a small sink until the beginning of the next century (1%–5% increase in carbon stocks including products).

Nevertheless, climate change mitigation is not only about carbon storage. Substitution of fossil fuels and carbon-intensive products plays another significant role. Although many studies previously found that substitution effects are crucial for mitigation highlighting the importance of managed forests, recent research has indicated that the benefits of substitutions will likely decrease, depending on the RCP (Brunet-Navarro et al., 2021; Harmon, 2019). This trend is caused by an increased share of renewables in the energy mix and technological improvements in the construction sector (see, e.g., IEA, 2020; Lehne & Preston, 2018). Therefore, the importance of harvests for our mitigation ESI decreased since we explicitly considered such decreasing substitution effects (Figure S23 in Supporting Information S1) and optimized for multiple RCPs. Hence the high proposed share of unmanaged forests. This share even increased when only looking at RCP2.6 where substitution effects diminished very quickly (Figure S15 in Supporting Information S1). The opposite was true when keeping substitution effects constant for all RCPs (Figure S13 in Supporting Information S1).

It is important to note that our proposed shift to more unmanaged forests was gradual, that is, forests were successively taken out of management. Consequently, our approach still strongly utilized the substitution effects in the shorter term when they are still important in low emission scenarios. The total benefit of management or non-management in terms of mitigation depended on the time frame and RCP; hence our portfolios offer a diversified mitigation strategy.

Additionally, our assessment is somewhat unbalanced: The production of steel and concrete combined account for 14% of global carbon emissions, and their improvements in carbon-intensity partly depend on breakthrough technologies (IEA, 2020; Lehne & Preston, 2018). Although we accounted for such improvements, we ignored potential innovations in wood usage. Many new technologies will likely facilitate higher amounts of wood usage in the construction sector, including strategies to use wood from broad-leaved trees and increase the building materials' carbon content (Churkina et al., 2020). Considering this, the increased usage of wood as construction material (as opposed to burning) and longer product lifetimes could drastically affect the mitigation potential. Future studies should focus on this change which is currently rarely explicitly assessed (e.g., Dugan et al., 2018; Smyth et al., 2014, 2018). Another aspect that is important but out of scope is the impact of different materials and energy sources on other environmental and social dimensions (see, e.g., Blankendaal et al., 2014; Hertwich et al., 2016; Santangeli et al., 2016).

4.4. Management and Disturbances

Future management must consider increases in disturbances (Seidl et al., 2014, 2017; Senf & Seidl, 2021a, 2021b). A common strategy is to diversify risks against these disturbances (Jandl et al., 2019; Mitchell, 2013), which to some extent is implicitly included in our propositions: Since we assessed the provision of multiple ESIs under increasing disturbances, our optimization automatically resulted in diverse forest compositions. Additionally, our portfolios contained an increased share of broad-leaved forests which are less prone to disturbances (Figure S25 in Supporting Information S1, Astrup et al., 2018; Pugh et al., 2019). Due to the uncertainties and complexity related to disturbances (e.g., Ahlström et al., 2015; Reyer et al., 2017; Seidl et al., 2011), we chose a rather simple representation of the change of disturbance regimes. While we used species-specific disturbance intervals, we refrained from making their increases species-specific, too, because research has shown that also species that were previously thought to be rather resilient have experienced heavy impacts of disturbances in recent years, for example, European beech in Central Europe (Buras et al., 2020; Schuldt et al., 2020).

Earth's Future

10.1029/2022EF002796

The direct effect of management regarding disturbances was only partly contained in our study, namely through salvage logging in managed stands and because in managed stands, some trees are already harvested and thus the impact of a disturbance is lower (Figure S25 in Supporting Information S1). The impact of salvage logging on mitigation was rather small since disturbed wood also remains in the forest for many years before it decays back to the atmosphere (Suzuki et al., 2019), and because we assumed low fractions of salvaged wood to be used in products. However, an increased material usage of salvaged wood and measures to dampen disturbance impacts will foster the relevance of forest management for mitigation. Examples are harvesting trees that are more likely to be affected by disturbance, adapting the age structure (O'Hara & Ramage, 2013), or active fire suppression (Agee et al., 2000; Fernandes et al., 2013). This should be addressed in more detail in future studies.

4.5. Harvest Reductions

The large fractions of successively unmanaged forest in the portfolios naturally resulted in a gradual reduction in harvest volumes in favor of other ESIs. Future harvests of the optimized portfolios in low-emission RCPs were simulated to be 20%–29% lower than present-day values, but closer to present-day values in the high-emission RCPs, likely due to simulated CO_2-fertilization (Tables S3–S6 in Supporting Information S1). Furthermore, long-lived wood products provision would be further reduced in the optimized portfolio as it includes broad-leaved forests and coppice, which provide lower shares of wood being useable for those products (especially coppice, almost completely used for firewood).

From a purely financial perspective, a decrease in harvests for mitigating climate change is unremarkable. Previous reports indicated that financial incentives are probably necessary to implement forest management measures for mitigation (e.g., Khanal et al., 2017). However, a decrease in harvests in Europe will lead to increased wood imports, possibly stemming from unsustainable sources leading to dislocation of emissions and increased ecological pressure in other regions (Berlik et al., 2002; Mayer et al., 2005).

The increasing demand for wood in various industries (FAO, 2022; Nabuurs et al., 2007) and a proposed timber usage rise for construction (Churkina et al., 2020) combined with increasing floor areas per capita (e.g., Bierwirth & Thomas, 2015) will cause great pressure on forests in terms of wood production. Therefore, this stands in direct conflict with the various other ESs offered by forests. Countermeasures could be to redirect harvests for fuel toward long-lived products (Churkina et al., 2020), and societal changes in the amount of wood required per capita. Other means could be forest expansion, changes in harvest intensity, and inclusion of fast-growing species such as Douglas fir (Thomas et al., 2022). Sensible intensities of such measures and their impact on ESs could also be assessed by our framework.

4.6. Provision of Ecosystem Services and Trade-Offs

Our study was aimed to assess the possibility of achieving climate-smart forestry by combining mitigation, adaptation, and the provision of other ESs, under the uncertainty of future climate pathways. Above we mainly focused on forest adaptation (e.g., some species are no longer suitable in certain regions, see Section 4.2) and the issues arising for mitigation and harvests. But our portfolios can help enabling the continued provision of other ESs.

Roughness length was estimated to be about 22% higher than present-day values. It is mostly governed by plant area index and canopy height (Moene & van Dam, 2014; Raupach, 1994). Higher roughness decreases aerodynamic resistance and consequently increases heat fluxes. The effect of canopy height explains the high simulated roughness values of unmanaged forests. Also, the change to more deciduous species increased the average roughness, since due to senescence their winter plant area index was quite low leading to lower resistance (Figure S9 in Supporting Information S1).

Future ET values heavily depended on the RCP, especially in the high-emission scenario, where we found strong ET reductions in all management options (Table S6, Figure S5 in Supporting Information S1). With elevated atmospheric CO_2 concentrations, plants do not need to open their stomata as much, hence losing less water (Keenan et al., 2013). Consequently, trees are more drought-resistant but the local cooling reduces, an implicit trade-off between adaptation and local climate regulation. However, in low-emission RCPs, this effect was not large enough to compensate for increased precipitation and temperature, causing a net increase in ET. Note that

Earth's Future 10.1029/2022EF002796

vegetation-climate feedbacks will likely amplify temperature changes through ET, increasing warming in Southern Europe, and reducing warming in Central Europe (Wramneby et al., 2010).

Although our simulations indicated positive effects of elevated CO_2 levels on productivity through higher water use efficiency, soil water potentials decreased until the beginning of the next century for all RCPs except RCP2.6 (Figure S10 in Supporting Information S1). Furthermore, our water availability indicator depicting the minimal monthly soil water potential decreased for almost all management options and RCPs compared with the present-day values, indicating higher drought risk in the future. Needle-leaved forest showed the highest water availability—for RCP 2.6 even slightly higher than today. In contrast, coppice showed the lowest, due to its high LAI and water usage.

In terms of biodiversity, our approach favors unmanaged forests over managed ones due to the higher abundance of different age classes, dead biomass, and large trees. This result is confirmed by Vuidot et al. (2011) who argue that the number of different habitats makes unmanaged forests preferable in terms of biodiversity. However, more important than the "management versus conservation antagonism" is the diversity of landscapes (e.g., Schall et al., 2021) which our portfolio-based approach implicitly offers. Additionally, making management mimic features of unmanaged forests and especially avoiding clear-cuts with species changes could further improve the suitability of managed forests in terms of biodiversity (Paillet et al., 2010; Vuidot et al., 2011). Although diversity is often evaluated at a within-stand level, concentrating on landscape-scales similar to this study can be a reasonable approach (Schall et al., 2018). Importantly, using biodiversity indicators such as those used in this study is encouraged, though still speculative (Davies et al., 2008) and insufficient biodiversity assessment. Also, simulations with more heterogeneous species are necessary, for a larger variety of species and microhabitats (Vuidot et al., 2011).

4.7. Different Experiments and Applicability of the Methodology

In a few experiments (Section 3.2.3) we showed the implications of changing the optimization focus. Interestingly, a stronger focus on mitigation resulted in a near-complete disappearance of coppice management from the continent (Figure S12 in Supporting Information S1) due to low material substitution and low carbon stocks. Its apparent benefit for fuel substitution is also decreased since other management options obtain similar fuel substitutions because large fractions of wood products can displace fossil fuels at the end of their lifetime (Knauf et al., 2015). In the considered time frame, the fuel substitution of coppice thus did not have a significant impact. We consequently suggest that more research is necessary to assess the currently rising interest in coppice due to its apparent potential for bioenergy (Maganotti et al., 2018). Note that we mean traditional coppice here, not short-rotation plantations.

Inclusion of albedo only changed portfolios marginally (Figure S14 in Supporting Information S1) because of the strong focus on worst cases. Since we already assessed seven ESIs, chances were high that the "crucial" ESI was already included. Nevertheless, in some cells, including albedo led to changes, often toward more unmanaged forest, partly because these were simulated to be less dense, that is, some highly reflective grass shone through (not shown).

Increasing the importance of harvests in the optimization disclosed that harvest levels can be increased compared with the standard optimization while still adequately representing other ESs (Section 4.5), with harvests spread around present-day values, depending on the RCP. Adding constraints to the optimization to keep harvests at present-day levels in every grid cell could not provide a mathematically feasible solution for many grid cells. This was because harvest rates in these grid cells inevitably decreased until the end of the century, demonstrating that collaboration between countries will be necessary to meet future wood demands and climate objectives.

Our weighting was done equally for the entire continent, but the importance of ESs actually depends on the geographic location and local socioeconomic circumstances. Furthermore, our weights were independent of the RCP although different RCPs could yield different ESs to be most important. While technically possible, including ES-weights that depend on the RCP is a major task of its own and was excluded from this study. We would like to note however, that adaptation was accounted for as RCP-dependent in our study, since Equation 6 guarantees a healthy forest cover under all RCPs.

Considering all RCPs simultaneously is a very risk-averse strategy, especially considering the decreasing likelihood of RCP8.5 (Hausfather & Peters, 2020). However, including the full range of uncertainty could avoid costly adaptation (Lawrence et al., 2020) and is also suggested by the IPCC (IPCC, 2021). We however also applied our methodology to RCP2.6 only where at least the shift to broad-leaved forests is not as pronounced as when considering all RCPs (Figure S15 in Supporting Information S1).

Finally, our results showed management options and potential impacts on a very coarse $0.5° \times 0.5°$ resolution. In practice, an application of our methodology must be made on a fine scale taking into account site-specific characteristics, ideally with more detailed regional assessments of the inputs. Our general results can serve as a guideline for such an application, in which also ES preferences should be included, obtained for example by conducting regional stakeholder surveys (see, e.g., Knoke et al., 2020). Some ESs could also be ignored by setting weights to zero. This could for example, be the case when the mitigation potential of a particular forest is rather small and other ESs are more important. Additionally, harvest adjacency and avoidance of fragmentation (Baskent & Keles, 2005; Millar et al., 2007) could be directly included as constraints in our optimization, but lead to increased (computational) complexity.

4.8. Conclusion

The goal of our study was to obtain insights on possible climate-smart forest management portfolios across Europe allowing for a balanced provision of many ESs. The resulting portfolios with high shares of broad-leaved forests and unmanaged forests revealed a consequent trade-off between better performance of a few ESs against a balanced provision of all ESs with potentially lower performances.

The main trade-off shown in this study lies in harvest reductions in favor of other ESs, such as increased biodiversity, mitigation, and local cooling. Such harvest reductions and potentially increasing wood demand will put additional pressure on forest ecosystems, especially when forest areas remain constant as assumed here. This might also entail further ecological issues when wood is imported from unsustainable sources. Possible countermeasures might be increasing the fraction of wood harvests used for long-lived products (Churkina et al., 2020) and changing societal behavior. We also indicated how adding a preference to wood harvests in the optimization could alleviate the harvesting decreases to some extent; however, at the cost of other ESs.

Furthermore, our approach revealed another trade-off related to climate uncertainty. Portfolios optimized for a various climatic futures show weaker performance in some ESs than optimizing for a certain future, a cost of deriving robust solutions. However, concurrently, this is an important benefit of our approach: the possibility to make valid propositions for a wide range of climatic futures. We found this uncertainty-related trade-off to be particularly pronounced when assessing climate change mitigation. Mitigation depends on the forest carbon sink as well as wood products that substitute carbon-intensive materials and fuels in various economic sectors. However, in low-emission scenarios, the potential future decarbonization of these sectors may cause a decreased importance of substitution effects compared to the carbon sink, whereas the opposite is true in high-emission scenarios. Our portfolios offer an intermediate mitigation strategy, based on both, enhancing the carbon sink and continued material and fuel substitution. In any case, a crucial means for enhancing mitigation is increasing the fraction of harvested wood used for long-lived products. Further work should focus on this aspect and more detailed assessments of management options and ESIs. Also, the uncertainty space should be further explored by including forcing data from multiple GCMs and multiple process formulations in other DGVMs.

Conclusively, we applied a combination of forest management simulations with a multi-criteria robust optimization framework. Our study gives recommendations on climate-smart forest management options capable of providing many ESs in the future under a broad range of future climate scenarios. Our results provide insights on a general direction of European climate-smart forestry that may be used as a baseline when developing regional forest management strategies in practice. However, we also revealed that such a holistic approach does not eliminate all trade-offs. Although we acknowledge that further considerations are required, this study lays the groundwork for future research that examines trade-offs of forest management strategies in terms of ES provisioning and especially climate change mitigation under a highly uncertain future.

AGU ADVANCING EARTH AND SPACE SCIENCE

Earth's Future

10.1029/2022EF002796

Data Availability Statement

The code to compute the ESIs, analyze them, and run the optimization is publicly available at https://doi.org/10.5281/zenodo.6667489 together with data for the four grid cells used for the illustrations. The entire data set to reproduce all optimizations for the European continent is available at https://doi.org/10.5281/zenodo.6612953.

Acknowledgments

The authors gratefully acknowledge the computational and data resources provided by the Leibniz Supercomputing Centre (www.lrz.de). We would also like to thank Heinz-Detlef Gregor, Wolfgang Falk, and Thomas Hickler for fruitful discussions. We also acknowledge funding from the ERA-Net "Sumforest" project "FOREXCLIM" (Grant No. 2816ERA01S), and the Bavarian State Ministry of Science and the Arts in the context of the Bavarian Climate Research Network (bayklif) through its BLIZ project (Grant No. 7831-26625-2017, www.bayklif-bliz.de). We further acknowledge funding from the European Forest Institute (EFI) Networking Fund FORMASAM. Open Access funding enabled and organized by Projekt DEAL.

References

Agee, J. K., Bahro, B., Finney, M. A., Omi, P. N., Sapsis, D. B., Skinner, C. N., et al. (2000). The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management*, *127*(1–3), 55–66. https://doi.org/10.1016/S0378-1127(99)00116-4

Ahlström, A., Schurgers, G., Arneth, A., & Smith, B. (2012). Robustness and uncertainty in terrestrial ecosystem carbon response to CMIP5 climate change projections. *Environmental Research Letters*, *7*(4), 044008. https://doi.org/10.1088/1748-9326/7/4/044008

Ahlström, A., Xia, J., Arneth, A., Luo, Y., & Smith, B. (2015). Importance of vegetation dynamics for future terrestrial carbon cycling. *Environmental Research Letters*, *10*(5), 054019. https://doi.org/10.1088/1748-9326/10/5/054019

Albert, K., & Ammer, C. (2012). Biomasseproduktivität ausgewählter europäischer Mittel- und Niederwaldbestände - Ergebnisse einer vergleichenden Metaanalyse. *Allgemeine Forst und Jagdzeitung*, *183*, 225–237.

Arehart, J. H., Hart, J., Pomponi, F., & D'Amico, B. (2021). Carbon sequestration and storage in the built environment. *Sustainable Production and Consumption*, *27*, 1047–1063. https://doi.org/10.1016/j.spc.2021.02.028

Astrup, R., Bernier, P. Y., Genet, H., Lutz, D. A., & Bright, R. M. (2018). A sensible climate solution for the boreal forest. *Nature Climate Change*, *8*(1), 11–12. https://doi.org/10.1038/s41558-017-0043-3

Baskent, E. Z., & Keles, S. (2005). Spatial forest planning: A review. *Ecological Modelling*, *188*(2–4), 145–173. https://doi.org/10.1016/j.ecolmodel.2005.01.059

Bellassen, V., Le Maire, G., Dhôte, J. F., Ciais, P., & Viovy, N. (2010). Modelling forest management within a global vegetation model—Part 1: Model structure and general behaviour. *Ecological Modelling*, *221*(20), 2458–2474. https://doi.org/10.1016/j.ecolmodel.2010.07.008

Ben-Tal, A., & Nemirovski, A. (2002). Robust optimization—Methodology and applications. *Mathematical Programming*, *92*(3), 453–480. https://doi.org/10.1007/s101070100286

Berlik, M. M., Kittredge, D. B., & Foster, D. R. (2002). The illusion of preservation: A global environmental argument for the local production of natural resources. *Journal of Biogeography*, *29*(10–11), 1557–1568. https://doi.org/10.1046/j.1365-2699.2002.00768.x

Bierwirth, A., & Thomas, S. (2015). *Almost best friends: Sufficiency and efficiency; can sufficiency maximise efficiency gains in buildings? ECEEE 2015 - European Council for an Energy Efficient Economy* (pp. 71–82).

Binder, S., Haight, R. G., Polasky, S., Warziniack, T., Mockrin, M. H., Deal, R. L., & Arthaud, G. (2017). Assessment and valuation of forest ecosystem services: State of the science review (Vol. *170*). Retrieved from https://www.fs.fed.us/nrs/pubs/gtr/gtr_nrs170.pdf

Blankendaal, T., Schuur, P., & Voordijk, H. (2014). Reducing the environmental impact of concrete and asphalt: A scenario approach. *Journal of Cleaner Production*, *66*, 27–36. https://doi.org/10.1016/j.jclepro.2013.10.012

Booth, M. S., Mackey, B., & Young, V. (2020). It's time to stop pretending burning forest biomass is carbon neutral. *GCB Bioenergy*, *12*(12), 1036–1037. https://doi.org/10.1111/gcbb.12716

Brockerhoff, E. G., Barbaro, L., Castagneyrol, B., Forrester, D. I., Gardiner, B., González-Olabarria, J. R., et al. (2017). Forest biodiversity, ecosystem functioning and the provision of ecosystem services. *Biodiversity & Conservation*, *26*(13), 3005–3035. https://doi.org/10.1007/s10531-017-1453-2

Brunet-Navarro, P., Jochheim, H., Cardellini, G., Richter, K., & Muys, B. (2021). Climate mitigation by energy and material substitution of wood products has an expiry date. *Journal of Cleaner Production*, *303*, 127026. https://doi.org/10.1016/j.jclepro.2021.127026

Brus, D. J., Hengeveld, G. M., Walvoort, D. J., Goedhart, P. W., Heidema, A. H., Nabuurs, G. J., & Gunia, K. (2012). Statistical mapping of tree species over Europe. *European Journal of Forest Research*, *131*(1), 145–157. https://doi.org/10.1007/s10342-011-0513-5

Buras, A., Rammig, A., & Zang, C. S. (2020). Quantifying impacts of the 2018 drought on European ecosystems in comparison to 2003. *Biogeosciences*, *17*(6), 1655–1672. https://doi.org/10.5194/bg-17-1655-2020

Campbell, J., Berry, J., Seibt, U., Smith, S. J., Montzka, S. A., Launois, T. et al. (2017). Large historical growth in global terrestrial gross primary production. *Nature 544*, 84–87. https://doi.org/10.1038/nature22030

Canadell, J. G., & Raupach, M. R. (2008). Managing forests for climate change mitigation. *Science*, *320*(5882), 1456–1457. https://doi.org/10.1126/science.1155458

Chang, J., Ciais, P., Wang, X., Piao, S., Asrar, G., Betts, R., et al. (2017). Benchmarking carbon fluxes of the ISIMIP2a biome models. *Environmental Research Letters*, *12*(4), 045002. https://doi.org/10.1088/1748-9326/aa63fa

Cherubini, F., Peters, G. P., Berntsen, T., Strømman, A. H., & Hertwich, E. (2011). CO_2 emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming. *GCB Bioenergy*, *3*(5), 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x

Churkina, G., Organschi, A., Reyer, C. P., Ruff, A., Vinke, K., Liu, Z., et al. (2020). Buildings as a global carbon sink. *Nature Sustainability*, *3*(4), 269–276. https://doi.org/10.1038/s41893-019-0462-4

Cordonnier, T., Berger, F., Elkin, C., Lämås, T., Martinez, M., Bugmann, H., et al. (2014). ARANGE D2.2 - Models and linker functions (indicators) for ecosystem services. (289437).

Dai, A. (2013). Increasing drought under global warming in observations and models. *Nature Climate Change*, *3*(1), 52–58. https://doi.org/10.1038/nclimate1633

Davies, Z. G., Tyler, C., Stewart, G. B., & Pullin, A. S. (2008). Are current management recommendations for saproxylic invertebrates effective? A systematic review. *Biodiversity & Conservation*, *17*(1), 209–234. https://doi.org/10.1007/s10531-007-9242-y

Diaz-Balteiro, L., Alonso, R., Martínez-Jaúregui, M., & Pardos, M. (2017). Selecting the best forest management alternative by aggregating ecosystem services indicators over time: A case study in central Spain. *Ecological Indicators*, *72*, 322–329. https://doi.org/10.1016/j.ecolind.2016.06.025

Diaz-Balteiro, L., Belavenutti, P., Ezquerro, M., González-Pachón, J., Ribeiro Nobre, S., & Romero, C. (2018). Measuring the sustainability of a natural system by using multi-criteria distance function methods: Some critical issues. *Journal of Environmental Management*, *214*, 197–203. https://doi.org/10.1016/j.jenvman.2018.03.005

Drake, P. L., Mendham, D. S., White, D. A., & Ogden, G. N. (2009). A comparison of growth, photosynthetic capacity and water stress in *Eucalyptus globulus* coppice regrowth and seedlings during early development. *Tree Physiology*, *29*(5), 663–674. https://doi.org/10.1093/treephys/tpp006

Earth's Future 10.1029/2022EF002796

Dufresne, J.-L., Foujols, M.-A., Denvil, S., Caubel, A., Marti, O., Aumont, O., et al. (2013). Climate change projections using the IPSL-CM5 Earth System Model: From CMIP3 to CMIP5. *Climate Dynamics*, *40*(9–10), 2123–2165. https://doi.org/10.1007/s00382-012-1636-1

Dugan, A. J., Birdsey, R., Mascorro, V. S., Magnan, M., Smyth, C. E., Olguin, M., & Kurz, W. A. (2018). A systems approach to assess climate change mitigation options in landscapes of the United States forest sector. *Carbon Balance and Management*, *13*(1), 13. https://doi.org/10.1186/s13021-018-0100-x

Díaz, S., Fargione, J., Chapin, F. S., & Tilman, D. (2006). Biodiversity loss threatens human well-being. *PLoS Biology*, *4*(8), 1300–1305. https://doi.org/10.1371/journal.pbio.0040277

Eurostat. (2021a). Eurostat forestry database. Retrieved from https://ec.europa.eu/eurostat/web/forestry/data/database

Eurostat. (2021b). Landfill rate of waste excluding major mineral wastes. Retrieved from https://ec.europa.eu/eurostat/databrowser/view/t2020_rt110/default/table?lang=en

Evans, J. (1984). Silviculture of broadleaved woodland. *Forestry Commission Bulletin*, *62*.

FAO. (2020). Food and Agriculture Organization of the United Nations: Global forest resources assessment 2020: Terms and definition FRA. Retrieved from http://www.fao.org/forestry/58864/en/

FAO. (2022). FAOSTAT database. *Forestry production and trade*. Retrieved from https://www.fao.org/faostat/en/%23data/FO

Fekete, B. M., Vörösmarty, C. J., & Grabs, W. (1999). *Global, composite runoff fields based on observed river discharge and simulated water balances*. Global Runoff Data Centre Koblenz.

Felton, A., Lindbladh, M., Brunet, J., & Fritz, Ö. (2010). Replacing coniferous monocultures with mixed-species production stands: An assessment of the potential benefits for forest biodiversity in northern Europe. *Forest Ecology and Management*, *260*(6), 939–947. https://doi.org/10.1016/j.foreco.2010.06.011

Fernandes, P. M., Davies, G. M., Ascoli, D., Fernández, C., Moreira, F., Rigolot, E., et al. (2013). Prescribed burning in southern Europe: Developing fire management in a dynamic landscape. *Frontiers in Ecology and the Environment*, *11*(S1). https://doi.org/10.1890/120298

Forest Europe. (2015a). Seventh Ministerial Conference on the Protection of Forest in Europe (pp. 1–8). Retrieved from https://foresteurope.org/wp%2Dcontent/uploads/2016/11/III.%2DELM%5F7MC%5F2%5F2015%5FMinisterialDeclaration%5Fadopted%2D2.pdf%23page%3D5

Forest Europe. (2015b). State of Europe's forests 2015.

Forest Europe. (2020). State of Europe's forests 2020.

Friedlingstein, P., O'Sullivan, M., Jones, M. W., Andrew, R. M., Hauck, J., Olsen, A., et al. (2020). Global carbon budget 2020. *Earth System Science Data*, *12*(4), 3269–3340. https://doi.org/10.5194/essd-12-3269-2020

Gorissen, B. L., Yanikoğlu, I., & den Hertog, D. (2015). A practical guide to robust optimization. *Omega*, *53*, 124–137. https://doi.org/10.1016/j.omega.2014.12.006

Grassi, G., Cescatti, A., Matthews, R., Duveiller, G., Camia, A., Federici, S., et al. (2019). On the realistic contribution of European forests to reach climate objectives. *Carbon Balance and Management*, *14*(1), 1–5. https://doi.org/10.1186/s13021-019-0123-y

Grassi, G., Fiorese, G., Pilli, R., Jonsson, K., Blujdea, V., Korosuo, A., & Vizzarri, M. (2021). Brief on the role of the forest-based bioeconomy in mitigating climate change through carbon storage and material substitution (pp. 1–16). Retrieved from https://publications.jrc.ec.europa.eu/repository/handle/JRC124374

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., et al. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, *342*(6160), 850–853. https://doi.org/10.1126/science.1244693

Harmon, M. E. (2019). Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. *Environmental Research Letters*, *14*(6), 065008. https://doi.org/10.1088/1748-9326/ab1e95

Harris, N. L., Gibbs, D. A., Baccini, A., Birdsey, R. A., de Bruin, S., Farina, M., et al. (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

Hartwich, J., Bölscher, J., & Schulte, A. (2014). Impact of short-rotation coppice on water and land resources. *Water International*, *39*(6), 813–825. https://doi.org/10.1080/02508060.2014.959870

Hausfather, Z., & Peters, G. P. (2020). Emissions – The 'business as usual' story is misleading. *Nature*, *577*(7792), 618–620. https://doi.org/10.1038/d41586-020-00177-3

Haverd, V., Smith, B., Canadell, J. G., Cuntz, M., Mikaloff-Fletcher, S., Farquhar, G., et al. (2020). Higher than expected CO_2 fertilization inferred from leaf to global observations. *Global Change Biology*, *26*(4), 2390–2402. https://doi.org/10.1111/gcb.14950

Haxeltine, A., & Prentice, I. C. (1996). BIOME3: An equilibrium terrestrial biosphere model based on ecophysiological constraints, resource availability, and competition among plant functional types. *Global Biogeochemical Cycles*, *10*(4), 693–709. https://doi.org/10.1029/96GB02344

Hertwich, E., de Larderel, J. A., Arvesen, A., Bayer, P., Bergesen, J., Bouman, E., et al. (2016). *Green Energy Choices: The benefits, risks, and trade-offs of low-carbon technologies for electricity production* (Technical Report). UNEP.

Hickler, T., Vohland, K., Feehan, J., Miller, P. A., Smith, B., Costa, L., et al. (2012). Projecting the future distribution of European potential natural vegetation zones with a generalized, tree species-based dynamic vegetation model. *Global Ecology and Biogeography*, *21*(1), 50–63. https://doi.org/10.1111/j.1466-8238.2010.00613.x

Holtsmark, B. (2012). Harvesting in boreal forests and the biofuel carbon debt. *Climatic Change*, *112*(2), 415–428. https://doi.org/10.1007/s10584-011-0222-6

Howard, C., Dymond, C. C., Griess, V. C., Tolkien-Spurr, D., & van Kooten, G. C. (2021). Wood product carbon substitution benefits: A critical review of assumptions. *Carbon Balance and Management*, *16*(1), 1–11. https://doi.org/10.1186/s13021-021-00171-w

Hua, F., Bruijnzeel, L. A., Meli, P., Martin, P. A., Zhang, J., Nakagawa, S., et al. (2022). The biodiversity and ecosystem service contributions and trade-offs of forest restoration approaches. *Science*, *376*(6595), 839–844. https://doi.org/10.1126/science.abl4649

IEA. (2020). Iron and steel technology roadmap (Technical Report). Retrieved from https://www.iea.org/reports/iron-and-steel-technology-roadmap

IPCC. (2014). Summary for policymakers. In *Climate change 2013—The physical science basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1–30). Cambridge University Press. https://doi.org/10.1017/CBO9781107415324.004

IPCC. (2018). Proposed outline of the special report in 2018 on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change. *IPCC - Sr15*, *2*, 17–20.

IPCC. (2021). *Summary for policymakers*. In *Climate change 2021—The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*.

Ishizaka, A., & Nemery, P. (2013). *Multi-criteria decision analysis*. John Wiley & Sons Ltd. https://doi.org/10.1002/9781118644898

Ito, A., Nishina, K., Reyer, C. P. O., François, L., Henrot, A.-J., Munhoven, G., et al. (2017). Photosynthetic productivity and its efficiencies in ISIMIP2a biome models: Benchmarking for impact assessment studies. *Environmental Research Letters*, *12*(8), 085001. https://doi.org/10.1088/1748-9326/aa7a19

Jandl, R., Spathelf, P., Bolte, A., & Prescott, C. E. (2019). Forest adaptation to climate change—Is non-management an option? *Annals of Forest Science*, *76*(2), 1–13. https://doi.org/10.1007/s13595-019-0827-x

Johansson, T. (2008). Sprouting ability and biomass production of downy and silver birch stumps of different diameters. *Biomass and Bioenergy*, *32*(10), 944–951. https://doi.org/10.1016/j.biombioe.2008.01.009

Kauppi, P., Hanewinkel, M., Lundmark, T., Nabuurs, G.-J., Peltola, H., Trasobares, A., & Hetemäki, L. (2018). *Climate smart forestry in Europe*. European Forest Institute.

Keenan, T. F., Hollinger, D. Y., Bohrer, G., Dragoni, D., Munger, J. W., Schmid, H. P., & Richardson, A. D. (2013). Increase in forest water-use efficiency as atmospheric carbon dioxide concentrations rise. *Nature*, *499*(7458), 324–327. https://doi.org/10.1038/nature12291

Khanal, P. N., Grebner, D. L., Munn, I. A., Grado, S. C., Grala, R. K., & Henderson, J. E. (2017). Evaluating non-industrial private forest landowner willingness to manage for forest carbon sequestration in the southern United States. *Forest Policy and Economics*, *75*, 112–119. https://doi.org/10.1016/j.forpol.2016.07.004

Klein, D., Höllerl, S., Blaschke, M., & Schulz, C. (2013). The contribution of managed and unmanaged forests to climate change mitigation—A model approach at stand level for the main tree species in Bavaria. *Forests*, *4*(1), 43–69. https://doi.org/10.3390/f4010043

Knauf, M., Joosten, R., & Frühwald, A. (2016). Assessing fossil fuel substitution through wood use based on long-term simulations. *Carbon Management*, *7*(1–2), 67–77. https://doi.org/10.1080/17583004.2016.1166427

Knauf, M., Köhl, M., Mues, V., Olschofsky, K., & Frühwald, A. (2015). Modeling the CO_2-effects of forest management and wood usage on a regional basis. *Carbon Balance and Management*, *10*(1), 1–12. https://doi.org/10.1186/s13021-015-0024-7

Knoke, T., Paul, C., Hildebrandt, P., Calvas, B., Castro, L. M., Hartl, F., et al. (2016). Compositional diversity of rehabilitated tropical lands supports multiple ecosystem services and buffers uncertainties. *Nature Communications*, *7*(1), 11877. https://doi.org/10.1038/ncomms11877

Knoke, T., Paul, C., Härtl, F., Castro, L. M., Calvas, B., & Hildebrandt, P. (2015). Optimizing agricultural land-use portfolios with scarce data—A non-stochastic model. *Ecological Economics*, *120*, 250–259. https://doi.org/10.1016/j.ecolecon.2015.10.021

Knoke, T., Paul, C., Rammig, A., Gosling, E., Hildebrandt, P., Härtl, F., et al. (2020). Accounting for multiple ecosystem services in a simulation of land-use decisions: Does it reduce tropical deforestation? *Global Change Biology*, *26*(4), 2403–2420. https://doi.org/10.1111/gcb.15003

Kovats, R. S., Valentini, R., Bouwer, L. M., Georgopoulou, E., Jacob, D., Martin, E., et al. (2015). Europe. In V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, & K. J. Mach (Eds.), *Climate change 2014: Impacts, adaptation and vulnerability* (pp. 1267–1326). Cambridge University Press. https://doi.org/10.1017/CBO9781107415386.003

Krause, A., Knoke, T., & Rammig, A. (2020). A regional assessment of land-based carbon mitigation potentials: Bioenergy, BECCS, reforestation, and forest management. *GCB Bioenergy*, *12*(5), 346–360. https://doi.org/10.1111/gcbb.12675

Lamarque, J. F., Kyle, P. P., Meinshausen, M., Riahi, K., Smith, S. J., van Vuuren, D. P., et al. (2011). Global and regional evolution of short-lived radiatively-active gases and aerosols in the Representative Concentration Pathways. *Climatic Change*, *109*(1), 191–212. https://doi.org/10.1007/s10584-011-0155-0

Lawrence, J., Haasnoot, M., & Lempert, R. (2020). Climate change: Making decisions in the face of deep uncertainty. *Nature*, *580*(7804), 456. https://doi.org/10.1038/d41586-020-01147-5

Lehne, J., & Preston, F. (2018). Making concrete change; innovation in low-carbon Cement and concrete (Technical Report). Retrieved from www.chathamhouse.org

Leonardsson, J., & Götmark, F. (2015). Differential survival and growth of stumps in 14 woody species after conservation thinning in mixed oak-rich temperate forests. *European Journal of Forest Research*, *134*(1), 199–209. https://doi.org/10.1007/s10342-014-0843-1

Leturcq, P. (2020). GHG displacement factors of harvested wood products: The myth of substitution. *Scientific Reports*, *10*(1), 1–9. https://doi.org/10.1038/s41598-020-77527-8

Lindeskog, M., Smith, B., Lagergren, F., Sycheva, E., Ficko, A., Pretzsch, H., & Rammig, A. (2021). Accounting for forest management in the estimation of forest carbon balance using the dynamic vegetation model LPJ-GUESS (v4.0, r9710): Implementation and evaluation of simulations for Europe. *Geoscientific Model Development*, *14*(10), 6071–6112. https://doi.org/10.5194/gmd-14-6071-2021

Lindner, M., Fitzgerald, J. B., Zimmermann, N. E., Reyer, C., Delzon, S., van der Maaten, E., et al. (2014). Climate change and European forests: What do we know, what are the uncertainties, and what are the implications for forest management? *Journal of Environmental Management*, *146*, 69–83. https://doi.org/10.1016/j.jenvman.2014.07.030

Liu, Y. Y., Van Dijk, A. I., De Jeu, R. A., Canadell, J. G., McCabe, M. F., Evans, J. P., & Wang, G. (2015). Recent reversal in loss of global terrestrial biomass. *Nature Climate Change*, *5*(5), 470–474. https://doi.org/10.1038/nclimate2581

Luyssaert, S., Ciais, P., Piao, S. L., Schulze, E.-d., Jung, M., Zaehle, S., et al. (2010). The European carbon balance. Part 3: Forests. *Global Change Biology*, *16*(5), 1429–1450. https://doi.org/10.1111/j.1365-2486.2009.02056.x

Luyssaert, S., Marie, G., Valade, A., Chen, Y.-Y., Djomo, S. N., Ryder, J., et al. (2018). Trade-offs in using European forests to meet climate objectives. *Nature*, *562*(7726), 259–262. https://doi.org/10.1038/s41586-018-0577-1

López-Tirado, J., & Hidalgo, P. J. (2018). Predicting suitability of forest dynamics to future climatic conditions: The likely dominance of Holm oak [Quercus ilex subsp. ballota (Desf.) Samp.] and Aleppo pine (Pinus halepensis Mill.). *Annals of Forest Science*, *75*(1), 19. https://doi.org/10.1007/s13595-018-0702-1

Maganotti, N., Schweier, J., Spinelli, R., Tolosana, E., Jylhä, P., Sopushynskyy, I., et al. (2018). Coppice forests in Europe.

Marler, R. T., & Arora, J. S. (2004). Survey of multi-objective optimization methods for engineering. *Structural and Multidisciplinary Optimization*, *26*(6), 369–395. https://doi.org/10.1007/s00158-003-0368-6

Martens, B., Miralles, D. G., Lievens, H., van der Schalie, R., de Jeu, R. A. M., Fernández-Prieto, D., et al. (2017). GLEAM v3: Satellite-based land evaporation and root-zone soil moisture. *Geoscientific Model Development*, *10*(5), 1903–1925. https://doi.org/10.5194/gmd-10-1903-2017

Mauri, A., Strona, G., & San-Miguel-Ayanz, J. (2017). EU-Forest, a high-resolution tree occurrence dataset for Europe. *Scientific Data*, *4*(1), 160123. https://doi.org/10.1038/sdata.2016.123

Mayer, A. L., Kauppi, P. E., Angelstam, P. K., Zhang, Y., & Tikka, P. M. (2005). Importing timber, exporting ecological impact. *Science*, *308*(5720), 359–360. https://doi.org/10.1126/science.1109476

Meinshausen, M., Smith, S. J., Calvin, K., Daniel, J. S., Kainuma, M. L. T., Lamarque, J.-F., et al. (2011). The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Climatic Change*, *109*(1–2), 213–241. https://doi.org/10.1007/s10584-011-0156-z

Metzger, M. J., Bunce, R. G. H., Jongman, R. H. G., Mücher, C. A., & Watkins, J. W. (2005). A climatic stratification of the environment of Europe. *Global Ecology and Biogeography*, *14*(6), 549–563. https://doi.org/10.1111/j.1466-822X.2005.00190.x

Millar, C. I., Stephenson, N. L., & Stephens, S. L. (2007). Climate change and forests of the future: Managing in the face of uncertainty. *Ecological Applications*, *17*(8), 2145–2151. https://doi.org/10.1890/06-1715.1

Millenium Ecosystem Assessment. (2005). Ecosystems and human well-being: Synthesis (Technical Report). Retrieved from https://www.millenniumassessment.org/documents/document.356.aspx.pdf

Mitchell, S. J. (2013). Wind as a natural disturbance agent in forests: A synthesis. *Forestry*, *86*(2), 147–157. https://doi.org/10.1093/forestry/cps058

Moene, A. F., & van Dam, J. C. (2014). *Transport in the atmosphere-vegetation-soil continuum*. Cambridge University Press. https://doi.org/10.1017/CBO9781139043137

Mori, A. S., Lertzman, K. P., & Gustafsson, L. (2017). Biodiversity and ecosystem services in forest ecosystems: A research agenda for applied forest ecology. *Journal of Applied Ecology*, *54*(1), 12–27. https://doi.org/10.1111/1365-2664.12669

Myllyviita, T., Soimakallio, S., Judl, J., & Seppälä, J. (2021). Wood substitution potential in greenhouse gas emission reduction–review on current state and application of displacement factors. *Forest Ecosystems*, *8*(1), 42. https://doi.org/10.1186/s40663-021-00326-8

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

Nabuurs, G. J., Lindner, M., Verkerk, P. J., Gunia, K., Deda, P., Michalak, R., & Grassi, G. (2013). First signs of carbon sink saturation in European forest biomass. *Nature Climate Change*, *3*(9), 792–796. https://doi.org/10.1038/nclimate1853

Nabuurs, G. J., Pussinen, A., van Brusselen, J., & Schelhaas, M. J. (2007). Future harvesting pressure on European forests. *European Journal of Forest Research*, *126*(3), 391–400. https://doi.org/10.1007/s10342-006-0158-y

Nicolescu, V.-N., Carvalho, J., Hochbichler, E., Bruckman, V., Piqué-nicolau, M., Hernea, C., et al. (2017). *Silvicultural guidelines for European coppice forests* (Technical Report). Albert Ludwig University of Freiburg.

O'Hara, K. L., & Ramage, B. S. (2013). Silviculture in an uncertain world: Utilizing multi-aged management systems to integrate disturbance. *Forestry*, *86*(4), 401–410. https://doi.org/10.1093/forestry/cpt012

Paillet, Y., Bergès, L., Hjältén, J., Ódor, P., Avon, C., Bernhardt-Römermann, M., et al. (2010). Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. *Conservation Biology*, *24*(1), 101–112. https://doi.org/10.1111/j.1523-1739.2009.01399.x

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., et al. (2011). A large and persistent carbon sink in the world's forests. *Science*, *333*(6045), 988–993. https://doi.org/10.1126/science.1201609

Pedersen, J. S. T., van Vuuren, D. P., Aparício, B. A., Swart, R., Gupta, J., & Santos, F. D. (2020). Variability in historical emissions trends suggests a need for a wide range of global scenarios and regional analyses. *Communications Earth & Environment*, *1*(1), 1–7. https://doi.org/10.1038/s43247-020-00045-y

Perpiña Castillo, C., Jacobs-Crisioni, C., Diogo, V., & Lavalle, C. (2021). Modelling agricultural land abandonment in a fine spatial resolution multi-level land-use model: An application for the EU. *Environmental Modelling & Software*, *136*, 104946. https://doi.org/10.1016/j.envsoft.2020.104946

Pongratz, J., Reick, C. H., Raddatz, T., & Claussen, M. (2010). Biogeophysical versus biogeochemical climate response to historical anthropogenic land cover change. *Geophysical Research Letters*, *37*(8), 1–5. https://doi.org/10.1029/2010GL043010

Poulter, B., Aragão, L., Andela, N., Bellassen, V., Ciais, P., Kato, T., et al. (2018). The global forest age dataset (GFADv1.0) [data set]. PANGAEA. https://doi.org/10.1594/PANGAEA.889943

Pugh, T. A. M., Arneth, A., Kautz, M., Poulter, B., & Smith, B. (2019). Important role of forest disturbances in the global biomass turnover and carbon sinks. *Nature Geoscience*, *12*(9), 730–735. https://doi.org/10.1038/s41561-019-0427-2

Rajasekaran, E., Das, N., Poulsen, C., Behrangi, A., Swigart, J., Svoboda, M., et al. (2018). SMAP soil moisture change as an indicator of drought conditions. *Remote Sensing*, *10*(5), 788. https://doi.org/10.3390/rs10050788

Ramstein, C., Dominioni, G., Ettehad, S., Lam, L., Quant, M., Zhang, J., et al. (2019). *State and trends of carbon pricing 2019*. The World Bank. https://doi.org/10.1596/978-1-4648-1435-8

Raupach, M. R. (1994). Simplified expressions for vegetation roughness length and zero-plane displacement as functions of canopy height and area index. *Boundary-Layer Meteorology*, *71*(1–2), 211–216. https://doi.org/10.1007/BF00709229

Reineke, L. H. (1933). Perfecting a stand-density index for even-aged forest. *Journal of Agricultural Research*, *46*, 627–638.

Reyer, C. P., Bathgate, S., Blennow, K., Borges, J. G., Bugmann, H., Delzon, S., et al. (2017). Are forest disturbances amplifying or canceling out climate change-induced productivity changes in European forests? *Environmental Research Letters*, *12*(3), 034027. https://doi.org/10.1088/1748-9326/aa5ef1

Ruiz, F., Cabello, J. M., & Luque, M. (2011). An application of reference point techniques to the calculation of synthetic sustainability indicators. *Journal of the Operational Research Society*, *62*(1), 189–197. https://doi.org/10.1057/jors.2009.187

Rydberg, D. (2000). Initial sprouting, growth and mortality of European aspen and birch after selective coppicing in central Sweden. *Forest Ecology and Management*, *130*(1–3), 27–35. https://doi.org/10.1016/S0378-1127(99)00187-5

Rüter, S. (2011). Welchen Beitrag leisten Holzprodukte zur CO_2-Bilanz? *AFZ/Der Wald*, *15*, 15–18.

Santangeli, A., Toivonen, T., Pouzols, F. M., Pogson, M., Hastings, A., Smith, P., & Moilanen, A. (2016). Global change synergies and trade-offs between renewable energy and biodiversity. *GCB Bioenergy*, *8*(5), 941–951. https://doi.org/10.1111/gcbb.12299

Sathre, R., & O'Connor, J. (2010). Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science & Policy*, *13*(2), 104–114. https://doi.org/10.1016/j.envsci.2009.12.005

Schall, P., Gossner, M. M., Heinrichs, S., Fischer, M., Boch, S., Prati, D., et al. (2018). The impact of even-aged and uneven-aged forest management on regional biodiversity of multiple taxa in European beech forests. *Journal of Applied Ecology*, *55*(1), 267–278. https://doi.org/10.1111/1365-2664.12950

Schall, P., Heinrichs, S., Ammer, C., Ayasse, M., Boch, S., Buscot, F., et al. (2021). Among stand heterogeneity is key for biodiversity in managed beech forests but does not question the value of unmanaged forests: Response to Bruun and Heilmann-Clausen (2021). *Journal of Applied Ecology*, *58*(9), 1817–1826. https://doi.org/10.1111/1365-2664.13959

Schuldt, B., Buras, A., Arend, M., Vitasse, Y., Beierkuhnlein, C., Damm, A., et al. (2020). A first assessment of the impact of the extreme 2018 summer drought on Central European forests. *Basic and Applied Ecology*, *45*, 86–103. https://doi.org/10.1016/j.baae.2020.04.003

Schwaab, J., Davin, E. L., Bebi, P., Duguay-Tetzlaff, A., Waser, L. T., Haeni, M., & Meier, R. (2020). Increasing the broad-leaved tree fraction in European forests mitigates hot temperature extremes. *Scientific Reports*, *10*(1), 1–9. https://doi.org/10.1038/s41598-020-71055-1

Seidl, R., Rammer, W., & Lexer, M. J. (2011). Adaptation options to reduce climate change vulnerability of sustainable forest management in the Austrian Alps. *Canadian Journal of Forest Research*, *41*(4), 694–706. https://doi.org/10.1139/x10-235

Seidl, R., Schelhaas, M.-J., Rammer, W., & Verkerk, P. J. (2014). Increasing forest disturbances in Europe and their impact on carbon storage. *Nature Climate Change*, *4*(9), 806–810. https://doi.org/10.1038/nclimate2318

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., et al. (2017). Forest disturbances under climate change. *Nature Climate Change*, *7*(6), 395–402. https://doi.org/10.1038/nclimate3303

Senf, C., & Seidl, R. (2021a). Mapping the forest disturbance regimes of Europe. *Nature Sustainability*, *4*(1), 63–70. https://doi.org/10.1038/s41893-020-00609-y

Earth's Future 10.1029/2022EF002796

Senf, C., & Seidl, R. (2021b). Storm and fire disturbances in Europe: Distribution and trends. *Global Change Biology*, *27*(15), 3605–3619. https://doi.org/10.1111/gcb.15679

Sitch, S., Smith, B., Prentice, I. C., Arneth, A., Bondeau, A., Cramer, W., et al. (2003). Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. *Global Change Biology*, *9*(2), 161–185. https://doi.org/10.1046/j.1365-2486.2003.00569.x

Smith, B., Prentice, I. C., & Sykes, M. T. (2001). *Global Ecology & Biogeography*, *10*, 621–637. http://dx.doi.org/10.1046/j.1466-822X.2001.t01-1-00256.x

Smith, B., Wårlind, D., Arneth, A., Hickler, T., Leadley, P., Siltberg, J., & Zaehle, S. (2014). Implications of incorporating N cycling and N limitations on primary production in an individual-based dynamic vegetation model. *Biogeosciences*, *11*(7), 2027–2054. https://doi.org/10.5194/bg-11-2027-2014

Smith, P., Gregory, P. J., van Vuuren, D., Obersteiner, M., Havlík, P., Rounsevell, M., et al. (2010). Competition for land. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1554), 2941–2957. https://doi.org/10.1098/rstb.2010.0127

Smyth, C. E., Smiley, B. P., Magnan, M., Birdsey, R., Dugan, A. J., Olguin, M., et al. (2018). Climate change mitigation in Canada's forest sector: A spatially explicit case study for two regions. *Carbon Balance and Management*, *13*(1), 11. https://doi.org/10.1186/s13021-018-0099-z

Smyth, C. E., Stinson, G., Neilson, E., Lemprière, T. C., Hafer, M., Rampley, G. J., & Kurz, W. A. (2014). Quantifying the biophysical climate change mitigation potential of Canada's forest sector. *Biogeosciences*, *11*(13), 3515–3529. https://doi.org/10.5194/bg-11-3515-2014

Spinoni, J., Vogt, J. V., Naumann, G., Barbosa, P., & Dosio, A. (2018). Will drought events become more frequent and severe in Europe? *International Journal of Climatology*, *38*(4), 1718–1736. https://doi.org/10.1002/joc.5291

Stajic, B., Zlatanov, T., Velichkov, I., Dubravac, T., & Trajkov, P. (2009). Past and recent coppice forest management in some regions of South Eastern Europe. *Silva Balcanica*, *10*(10), 9–19.

Suzuki, S. N., Tsunoda, T., Nishimura, N., Morimoto, J., & Suzuki, J. I. (2019). Dead wood offsets the reduced live wood carbon stock in forests over 50 years after a stand-replacing wind disturbance. *Forest Ecology and Management*, *432*, 94–101. https://doi.org/10.1016/j.foreco.2018.08.054

Taylor, K. E., Stouffer, R. J., & Meehl, G. A. (2012). An overview of CMIP5 and the experiment design. *Bulletin of the American Meteorological Society*, *93*(4), 485–498. https://doi.org/10.1175/BAMS-D-11-00094.1

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*, 119956. https://doi.org/10.1016/j.foreco.2021.119956

Thonicke, K., Venevsky, S., Sitch, S., & Cramer, W. (2001). The role of fire disturbance for global vegetation dynamics: Coupling fire into a dynamic global vegetation model. *Global Ecology and Biogeography*, *10*(6), 661–677. https://doi.org/10.1046/j.1466-822X.2001.00175.x

Uhde, B., Hahn, A., Griess, V. C., & Knoke, T. (2015). Hybrid MCDA methods to integrate multiple ecosystem services in forest management planning: A critical review. *Environmental Management*, *56*(2), 373–388. https://doi.org/10.1007/s00267-015-0503-3

Uhde, B., Heinrichs, S., Stiehl, C. R., Ammer, C., Müller-Using, B., & Knoke, T. (2017). Bringing ecosystem services into forest planning – Can we optimize the composition of Chilean forests based on expert knowledge? *Forest Ecology and Management*, *404*, 126–140. https://doi.org/10.1016/j.foreco.2017.08.021

United Nations. (2015a). Summary of the Paris Agreement. Retrieved from http://bigpicture.unfccc.int/%23content%2Dthe%2Dparis%2Dagreemen

United Nations. (2015b). Transforming our world: The 2030 agenda for sustainable development. Retrieved from https://sdgs.un.org/2030agenda

Vacchiano, G., Garbarino, M., Lingua, E., & Motta, R. (2017). Forest dynamics and disturbance regimes in the Italian Apennines. *Forest Ecology and Management*, *388*, 57–66. https://doi.org/10.1016/j.foreco.2016.10.033

van Kooten, G. C., & Johnston, C. M. (2016). The economics of forest carbon offsets. *Annual Review of Resource Economics*, *8*(1), 227–246. https://doi.org/10.1146/annurev-resource-100815-095548

Verlinden, M. S., Broeckx, L. S., & Ceulemans, R. (2015). First vs. second rotation of a poplar short rotation coppice: Above-ground biomass productivity and shoot dynamics. *Biomass and Bioenergy*, *73*, 174–185. https://doi.org/10.1016/j.biombioe.2014.12.012

Virtanen, P., Gommers, R., Oliphant, T. E., Haberland, M., Reddy, T., Cournapeau, D., et al. (2020). SciPy 1.0: Fundamental algorithms for scientific computing in Python. *Nature Methods*, *17*(3), 261–272. https://doi.org/10.1038/s41592-019-0686-2

Vuidot, A., Paillet, Y., Archaux, F., & Gosselin, F. (2011). Influence of tree characteristics and forest management on tree microhabitats. *Biological Conservation*, *144*(1), 441–450. https://doi.org/10.1016/j.biocon.2010.09.030

Winckler, J., Reick, C. H., & Pongratz, J. (2017). Robust identification of local biogeophysical effects of land-cover change in a global climate model. *Journal of Climate*, *30*(3), 1159–1176. https://doi.org/10.1175/JCLI-D-16-0067.1

Winckler, J., Reick, C. H., Bright, R. M., & Pongratz, J. (2019). Importance of surface roughness for the local biogeophysical effects of deforestation. *Journal of Geophysical Research: Atmospheres*, *124*(15), 8605–8618. https://doi.org/10.1029/2018JD030127

Wramneby, A., Smith, B., & Samuelsson, P. (2010). Hot spots of vegetation-climate feedbacks under future greenhouse forcing in Europe. *Journal of Geophysical Research*, *115*(D21), D21119. https://doi.org/10.1029/2010JD014307

References From the Supporting Information

Boisier, J. P., De Noblet-Ducoudré, N., & Ciais, P. (2013). Inferring past land use-induced changes in surface albedo from satellite observations: A useful tool to evaluate model simulations. *Biogeosciences*, *10*(3), 1501–1516. https://doi.org/10.5194/bg-10-1501-2013

Clarke, L., Edmonds, J., Jacoby, H., Pitcher, H., Reilly, J., & Richels, R. (2007). *Scenarios of greenhouse gas emissions and atmospheric concentrations. Sub-report 2.1a of Synthesis and Assessment Product 2.1 by the US Climate Change Science Program and the Subcommittee on Global Change Research*. Department of Energy, Office of Biological & Environmental Research.

Dobor, L., Hlásny, T., Rammer, W., Zimová, S., Barka, I., & Seidl, R. (2020). Is salvage logging effectively dampening bark beetle outbreaks and preserving forest carbon stocks? *Journal of Applied Ecology*, *57*(1), 67–76. https://doi.org/10.1111/1365-2664.13518

Forrest, M., Tost, H., Lelieveld, J., & Hickler, T. (2020). Including vegetation dynamics in an atmospheric chemistry-enabled general circulation model: Linking LPJ-GUESS (v4.0) with the EMAC modelling system (v2.53). *Geoscientific Model Development*, *13*(3), 1285–1309. https://doi.org/10.5194/gmd-13-1285-2020

Fujino, J., Nair, R., Kainuma, M., Masui, T., & Matsuoka, Y. (2006). Multi-gas mitigation analysis on stabilization scenarios using aim global model. *The Energy Journal*, *27*(01), 343–353. https://doi.org/10.5547/ISSN0195-6574-EJ-VolSI2006-NoSI3-17

Harja, D., Subekti, R., & Pambudi, S. (2019). Tree functional attributes and ecological database. Retrieved from http://db.worldagroforestry.org

Hickler, T., Prentice, I. C., Smith, B., Sykes, M. T., & Zaehle, S. (2006). Implementing plant hydraulic architecture within the LPJ dynamic global vegetation model. *Global Ecology and Biogeography*, *15*(6), 567–577. https://doi.org/10.1111/j.1466-8238.2006.00254.x

Earth's Future

10.1029/2022EF002796

Hijioka, Y., Matsuoka, Y., & Nishimoto, H. (2008). Global GHG emission scenarios under GIC; concentration stabilization targets. *Journal of Global Environmental Engineering*, *13*, 97–108.

Jenkins, J. C. (2004). *Comprehensive database of diameter-based biomass regressions for North American tree species* (Vol. 319). United States Department of Agriculture, Forest Service.

Jones, H. G. (2013). *Plants and microclimate* (Vol. 22, No. 2). Cambridge University Press. https://doi.org/10.1017/CBO9780511845727

Kärhä, K., Anttonen, T., Poikela, A., Palander, T., Laurén, A., Peltola, H., & Nuutinen, Y. (2018). Evaluation of salvage logging productivity and costs in windthrown Norway spruce-dominated forests. *Forests*, *9*(5), 280. https://doi.org/10.3390/f9050280

Knoke, T., Gosling, E., Thom, D., Chreptun, C., Rammig, A., & Seidl, R. (2021). Economic losses from natural disturbances in Norway spruce forests – A quantification using Monte-Carlo simulations. *Ecological Economics*, *185*, 107046. https://doi.org/10.1016/j.ecolecon.2021.107046

Kucharik, C. J., Norman, J. M., & Gower, S. T. (1998). Measurements of branch area and adjusting leaf area index indirect measurements. *Agricultural and Forest Meteorology*, *91*(1–2), 69–88. https://doi.org/10.1016/S0168-1923(98)00064-1

Lindenmayer, D., & Noss, R. (2006). Salvage logging, ecosystem processes, and biodiversity conservation. *Conservation Biology*, *20*(4), 949–958. https://doi.org/10.1111/j.1523-1739.2006.00497.x

Lindenmayer, D. B., Burton, P. J., & Franklin, J. F. (2012). *Salvage logging and its ecological consequences*. Island Press.

McGroddy, M. E., Daufresne, T., & Hedin, L. O. (2004). Scaling of C:N:P stoichiometry in forests worldwide: Implications of terrestrial redfield-type ratios. *Ecology*, *85*(9), 2390–2401. https://doi.org/10.1890/03-0351

Ministry of Agriculture and Forestry Finland. (2013). Forest Damages Prevention Act (1087/2013). Retrieved from http://www.finlex.fi/en/laki/kaannokset/2013/en20131087.pdf

Möllmann, T. B., & Möhring, B. (2017). A practical way to integrate risk in forest management decisions. *Annals of Forest Science*, *74*(4), 75. https://doi.org/10.1007/s13595-017-0670-x

RCP Database. (2021). RCP database (version 2.0). Retrieved from https://tntcat.iiasa.ac.at/RcpDb

Riahi, K., Grübler, A., & Nakicenovic, N. (2007). Scenarios of long-term socio-economic and environmental development under climate stabilization. *Technological Forecasting and Social Change*, *74*(7), 887–935. https://doi.org/10.1016/j.techfore.2006.05.026

Savill, P. S. (Ed.). (2019). *The silviculture of trees used in British forestry*. CABI. https://doi.org/10.1079/9781786393920.0000

Smith, S. J., & Wigley, T. (2006). Multi-gas forcing stabilization with minicam. *The Energy Journal*, *SI2006*(01), 373–391. https://doi.org/10.5547/ISSN0195-6574-EJ-VolSI2006-NoSI3-19

Stimm, B., Roloff, A., Lang, U. M., & Weisgerber, H. (Eds.). (2014). *Enzyklopädie der Holzgewächse: Handbuch und Atlas der Dendrologie*. Wiley. https://doi.org/10.1002/9783527678518

Thorn, S., Chao, A., Georgiev, K. B., Müller, J., Bässler, C., Campbell, J. L., et al. (2020). Estimating retention benchmarks for salvage logging to protect biodiversity. *Nature Communications*, *11*(1), 1–8. https://doi.org/10.1038/s41467-020-18612-4

van Vuuren, D. P., den Elzen, M. G. J., Lucas, P. L., Eickhout, B., Strengers, B. J., van Ruijven, B., et al. (2007). Stabilizing greenhouse gas concentrations at low levels: An assessment of reduction strategies and costs. *Climatic Change*, *81*(2), 119–159. https://doi.org/10.1007/s10584-006-9172-9

Wang, Z., & Zeng, X. (2010). Evaluation of snow albedo in land models for weather and climate studies. *Journal of Applied Meteorology and Climatology*, *49*(3), 363–380. https://doi.org/10.1175/2009JAMC2134.1

West, P. W. (2014). *Growing plantation forests* (Vol. 9783319018). Springer International Publishing. https://doi.org/10.1007/978-3-319-01827-0

Wise, M., Calvin, K., Thomson, A., Clarke, L., Bond-Lamberty, B., Sands, R., et al. (2009). Implications of limiting CO_2 concentrations for land use and energy. *Science*, *324*(5931), 1183–1186. https://doi.org/10.1126/science.1168475

Trade-offs for climate-smart forestry in Europe under uncertain future climate

Konstantin Gregor[1,*], Thomas Knoke[1], Andreas Krause[1], Christopher P.O. Reyer[2], Mats Lindeskog[3], Phillip Papastefanou[1,3], Anne-Sofie Lansø[4,5], Benjamin Smith[3,6], and Anja Rammig[1]

[1]TUM School of Life Sciences, Technical University of Munich, Freising, Germany

[2]Potsdam Institute for Climate Impact Research, Member of the Leibniz Association, Potsdam, Germany

[3]Department of Physical Geography and Ecosystem Science, Lund University, Lund, Sweden

[4]Laboratoire des Sciences du Climat et de l'Environnement (LSCE/IPSL) CEA-CNRS-UVSQ, Université Paris-Saclay, Gif-sur-Yvette, France

[5]Department of Environmental Science, Aarhus University, Denmark

[6]Hawkesbury Institute for the Environment, Western Sydney University, Penrith NSW, Australia

[*]Corresponding author, konstantin.gregor@tum.de

S1 Supplementary Material

S1.1 Salvage Logging

Salvage logging is the process of harvesting after a disturbance. Although its consequences in terms of economics, saving carbon stocks, dampening of biotic outbreaks, and impact on the ecosystem are controversial (Dobor et al., 2020; Knoke et al., 2021; Lindenmayer et al., 2012; Lindenmayer & Noss, 2006), it is a common practice in Europe (e.g., Thorn et al., 2020). Unfortunately, concise numbers are only seldomly available (e.g., Ministry of Agriculture and Forestry Finland, 2013). For our study, we assumed that after a patch-destroying disturbance 80% of the affected wood is harvested and the rest is left to decay on site. This is in line with Lindenmayer et al. (2012), who estimated this number to be at least 78%. Since salvage logging is much more difficult than harvesting in an undamaged stand (Kärhä et al., 2018), we additionally lowered the harvest efficiency from 90% to 75%. Finally, the harvested wood after a disturbance will have lower quality (Möllmann & Möhring, 2017). Since this should result in lower fractions of wood going into the medium- or long-lived pools, we assumed those fractions be reduced by 1/3.

S1.2 Deriving Ecosystem Service Indicators (ESIs) from LPJ-GUESS simulations

Some ESIs such as carbon stocks are direct outputs of LPJ-GUESS, while others needed to be specifically derived from other measures. For evapotranspiration (ET) we took the sum of evaporation from soil, interception, and transpiration. In LPJ-GUESS the latter is based mainly on water supply and demand, which in turn depends on canopy conductance and the given climate. We took the yearly total of all three values as an indicator for the local cooling effect from ET. Sensible heat fluxes are also not a direct output of LPJ-GUESS. Instead, we computed the roughness length z_0 as an indicator for heat fluxes, as increases in roughness lead to higher aerodynamic conductance and thus higher heat fluxes all other factors being equal (see, e.g., Jones, 2013). z_0 can be estimated from plant area index and canopy height (Raupach, 1994). We computed the canopy height of each stand using tree height and foliar projective cover (FPC) similar to Forrest et al. (2020). Additionally we computed the plant area index (PAI) by combining the current LAI together with estimates of the woody hemi-surface area of each tree, using the cylindrical representation of stems in LPJ-GUESS as well as the insight that branches are typically at least 50% of woody hemi-surface area (Kucharik et al., 1998). Albedo was computed by aggregating to species types (e.g., needle-leaved evergreens) and converting LAI to forest cover via the relation $\text{fcover} = 1 - \exp(-0.5\text{LAI})$. This resulted in an estimate of how much of each grid cell was covered by which species type. In addition, the fractional snow cover of each grid cell was estimated from snow depth, using the relation $\text{snowcover} = \frac{\text{snowdepth}}{0.01+\text{snowdepth}}$ (Wang & Zeng, 2010). Finally, the albedo values for different forest types with or without snow cover by Boisier et al. (2013) were used.

We calculated Ψ_{soil} based on Hickler et al. (2006) and averaged over the two soil layers in LPJ-GUESS. For the harvest-related indicators, we assumed the carbon fraction of the dry biomass to be 0.47 (McGroddy et al., 2004). Carbon values were converted to m^3 wood using species-specific values for wood density from the literature (Table S2).

Table S1: Selection of PFT parameters used in this study, adapted from Lindeskog et al. (2021). For the full list of parameters, refer to Lindeskog et al. (2021). Values in bold text are updated compared to Hickler et al. (2012), values for bioclimatic limits in red are reverted back to Hickler et al. (2012), see Methods.

Species/PFT	Phen.	Geo. range[1]	Shade tolerance[1]	Form[1]	Tcmin	Tcmax	Twmin	GDD_5
Abies alba	EG	temperate	tolerant	tree	**-6.5(-7.5)**	**2**	6	**1600**
Betula pendula	SG	temperate	intolerant	tree	-30	-	5	700
Betula pubescens	SG	boreal	intolerant	tree	-30	-	5	350
Carpinus betulus	SG	temperate	intermediate	tree	-8	-	5	1200
Corylus avellana	SG	temperate	intermediate	tree	**-11**	-	5	800
Fagus sylvatica	SG	temperate	tolerant	tree	**-6**	-	5	1500
Fraxinus excelsior	SG	temperate	intermediate	tree	-16	-	5	1100
Juniperus oxy.	EG	temperate	intolerant	**tree**	1(0)	-	-	2200
Larix decidua	SG	boreal	intermediate	tree	-30	-2	5	300
Picea abies	EG	boreal	tolerant	tree	-30	-1.5	5	600
Pinus halepensis	EG	temperate	intolerant	tree	3	-	21	3000
Pinus sylvestris	EG	boreal	intermediate	tree	-30	-1	5	500
Populus tremula	SG	temperate	intolerant	tree	-30(-31)	6	-	500
Quercus coccifera	EG	temperate	intermediate	shrub	0	11	21	2200
Quercus ilex	EG	temperate	intolerant	tree	3	-	5	1800
Quercus pub.	SG	temperate	intermediate	tree	-5	-	-	1900
Quercus robur	SG	temperate	intermediate	tree	**-9(-10)**	-	5	1100
Tilia cordata	SG	temperate	intermediate	tree	-11(-12)	-	5	**1100**
Ulmus glabra	SG	temperate	intermediate	tree	-9.5(-10.5)	-	5	850
Boreal EG shrub	EG	boreal	intolerant*	shrub	-	-1	-	**200**
Medit. RG shrub	RG	temperate	intolerant	shrub	1(0)	-	-	2200
C3 grass	SG/RG	temp-boreal	-	herb	-	-	-	-

[1]See group parameter table A2 in Lindeskog et al. (2021); Phenology: evergreen(EG); summergreen(SG), raingreen(RG); Tcmin, Tcmax = minimum and maximum temperature of the coldest month for establishment, value in brackets are minimum temperature for survival, if different from value for establishment; Twmin = minimum warmest month mean temperature for establishment; GDD5 = minimum degree-day sum above 5°C for establishment.

Table S2: Wood densities per species used to estimate harvest volumes from simulated harvested carbon. Values are for air dry wood, i.e. 12 or 15% water content

Species	Conversion factor ($t\,m^{-3}$)	Source
Abies alba	0.4	Savill (2019)
Betula pendula	0.67	Savill (2019)
Betula pubescens	0.67	Savill (2019)
Carpinus betulus	0.77	Savill (2019)
Corylus avellana	0.6	Stimm et al. (2014)
Fagus sylvatica	0.72	Savill (2019)
Fraxinus excelsior	0.71	Savill (2019)
Juniperus spp	0.4	Harja et al. (2019), Jenkins (2004)
Larix Decidua	0.59	Savill (2019)
Picea abies	0.405	Savill (2019)
Pinus halepensis	0.75	Stimm et al. (2014)
Pinus sylvestris	0.51	Savill (2019)
Populus Tremula	0.43	Savill (2019)
Quercus coccifera	0.86	Stimm et al. (2014)
Quercus ilex	0.88	Savill (2019)
Quercus pubescens	0.81	Stimm et al. (2014)
Quercus robur	0.72	Savill (2019)
Tilia Cordata	0.56	Savill (2019)
Ulmus glabra	0.69	Savill (2019)

Table S3: ESI RCP 2.6 performances if one management were applied to the 181 grid cells. The values are the area-weighted means. The lower rows of the table correspond to the optimized portfolio in the standard setting ("optimized") as well as for the experiments with different weights and constraints as described in the text.

unit	C-Pool kgC/m^2	Mitigation kgC/m^2	Harvests $m^3/ha/yr$	HLP $m^3/ha/yr$	ET mm/yr	$\Psi soil$ MPa	$z0$ m	Bio unitless
base	10.0	0.8	2.54	0.66	514	-1.41	0.52	0.49
toBd	10.2	0.9	2.13	0.33	510	-1.48	0.71	0.57
toBe*	3.1	0.3	0.68	0.11	513	-2.66	0.41	0.07
toCoppice	10.0	0.6	1.35	0.10	523	-1.97	0.56	0.22
toNe	9.9	0.8	2.86	0.87	513	-1.34	0.40	0.43
unmanaged	12.3	1.6	0.08	0.02	514	-1.43	0.62	0.72
present	10.3	0.0	2.48	0.65	501	-1.41	0.52	–**
optimized	10.7	1.1	1.76	0.38	517	-1.50	0.61	0.57
double harv	10.5	1.0	2.08	0.47	517	-1.48	0.59	0.55
double mit	11.1	1.2	1.43	0.29	513	-1.45	0.64	0.64

* toBe management was only feasible in few regions in Europe, the reported values for this case are only for those regions (n=38). In the optimized portfolio of a grid cell, only management was considered that ensured forest cover by the end of the century.
** Computation of the Bio indicator is not sensible for present day, as it is solely a relative value between management options.

Table S4: ESI RCP 4.5 performances if one management were applied to the 181 grid cells. The values are the area-weighted means. The lower rows of the table correspond to the optimized portfolio in the standard setting ("optimized") as well as for the experiments with different weights and constraints as described in the text.

unit	C-Pool kgC/m^2	Mitigation kgC/m^2	Harvests $m^3/ha/yr$	HLP $m^3/ha/yr$	ET mm/yr	$\Psi soil$ MPa	$z0$ m	Bio unitless
base	9.8	1.5	2.84	0.73	504	-1.57	0.53	0.48
toBd	10.1	1.5	2.60	0.40	507	-1.68	0.75	0.56
toBe*	3.0	0.5	0.78	0.12	469	-2.76	0.41	0.08
toCoppice	9.9	1.1	1.64	0.11	515	-2.18	0.60	0.21
toNe	9.7	1.5	3.06	0.93	500	-1.47	0.40	0.43
unmanaged	12.1	1.8	0.07	0.02	502	-1.54	0.65	0.75
present	10.3	0.0	2.48	0.65	501	-1.41	0.52	–**
optimized	10.6	1.6	1.98	0.42	508	-1.67	0.63	0.57
double harv	10.4	1.6	2.33	0.52	508	-1.65	0.61	0.55
double mit	11.0	1.7	1.64	0.32	505	-1.60	0.67	0.64

* toBe management was only feasible in few regions in Europe, the reported values for this case are only for those regions (n=38). In the optimized portfolio of a grid cell, only management was considered that ensured forest cover by the end of the century.
** Computation of the Bio indicator is not sensible for present day, as it is solely a relative value between management options.

Table S5: ESI RCP 6.0 performances if one management were applied to the 181 grid cells. The values are the area-weighted means. The lower rows of the table correspond to the optimized portfolio in the standard setting ("optimized") as well as for the experiments with different weights and constraints as described in the text.

unit	C-Pool kgC/m^2	Mitigation kgC/m^2	Harvests m^3/ha/yr	HLP m^3/ha/yr	ET mm/yr	$\Psi soil$ MPa	$z0$ m	Bio unitless
base	9.9	2.1	3.25	0.84	497	-1.62	0.55	0.47
toBd	10.2	2.1	3.05	0.47	501	-1.74	0.78	0.54
toBe*	3.1	0.7	0.88	0.14	448	-2.86	0.43	0.08
toCoppice	10.0	1.6	1.94	0.12	507	-2.19	0.63	0.20
toNe	9.8	2.1	3.47	1.07	492	-1.51	0.41	0.43
unmanaged	12.4	2.0	0.08	0.02	494	-1.56	0.69	0.78
present	10.3	0.0	2.48	0.65	501	-1.41	0.52	–**
optimized	10.8	2.1	2.29	0.48	501	-1.71	0.65	0.57
double harv	10.5	2.1	2.69	0.60	500	-1.69	0.63	0.54
double mit	11.2	2.1	1.91	0.38	497	-1.64	0.69	0.64

* toBe management was only feasible in few regions in Europe, the reported values for this case are only for those regions (n=38). In the optimized portfolio of a grid cell, only management was considered that ensured forest cover by the end of the century.
** Computation of the Bio indicator is not sensible for present day, as it is solely a relative value between management options.

Table S6: ESI RCP 8.5 performances if one management were applied to the 181 grid cells. The values are the area-weighted means. The lower rows of the table correspond to the optimized portfolio in the standard setting ("optimized") as well as for the experiments with different weights and constraints as described in the text.

unit	C-Pool kgC/m^2	Mitigation kgC/m^2	Harvests m^3/ha/yr	HLP m^3/ha/yr	ET mm/yr	$\Psi soil$ MPa	$z0$ m	Bio unitless
base	9.6	2.2	3.31	0.86	464	-1.60	0.53	0.46
toBd	10.0	2.3	3.31	0.50	474	-1.74	0.77	0.52
toBe*	2.9	0.6	0.79	0.12	399	-2.76	0.37	0.08
toCoppice	9.8	1.7	2.15	0.11	474	-2.15	0.62	0.19
toNe	9.5	2.2	3.41	1.05	454	-1.43	0.38	0.39
unmanaged	12.1	1.9	0.06	0.02	460	-1.58	0.70	0.83
present	10.3	0.0	2.48	0.65	501	-1.41	0.52	–**
optimized	10.5	2.2	2.38	0.48	467	-1.68	0.64	0.55
double harv	10.2	2.2	2.77	0.60	467	-1.66	0.62	0.52
double mit	10.9	2.2	2.00	0.38	466	-1.64	0.69	0.63

* toBe management was only feasible in few regions in Europe, the reported values for this case are only for those regions (n=38). In the optimized portfolio of a grid cell, only management was considered that ensured forest cover by the end of the century.
** Computation of the Bio indicator is not sensible for present day, as it is solely a relative value between management options.

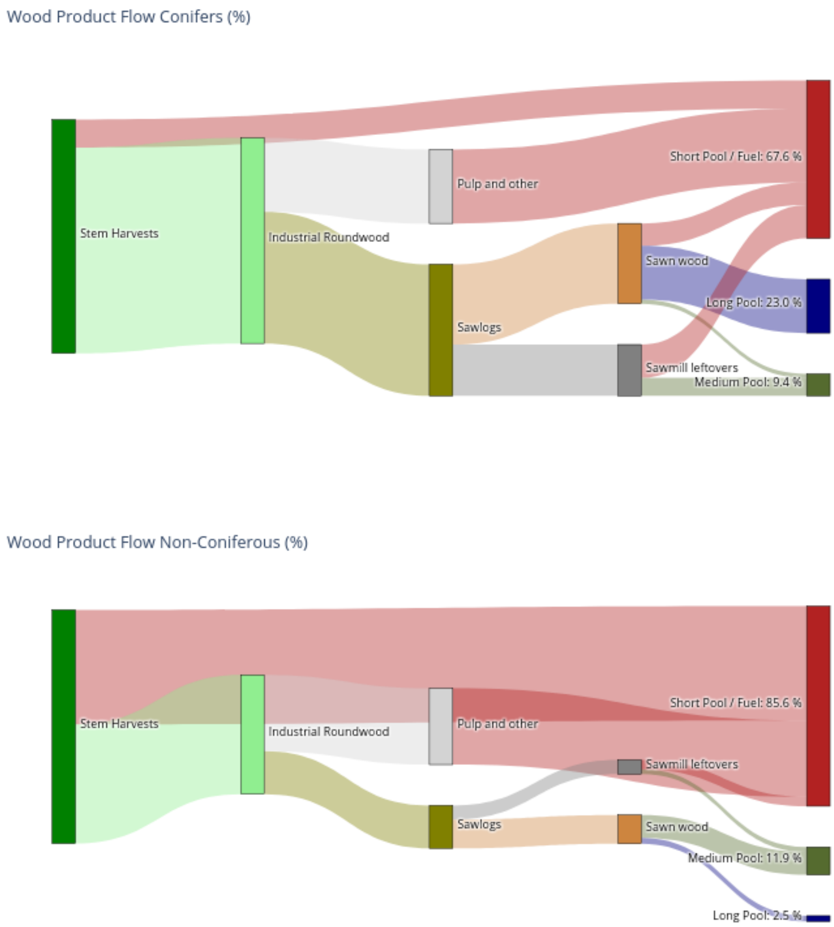

Figure S1: Visualization of the flows of stem wood derived from Eurostat (2021) and Klein et al. (2013)

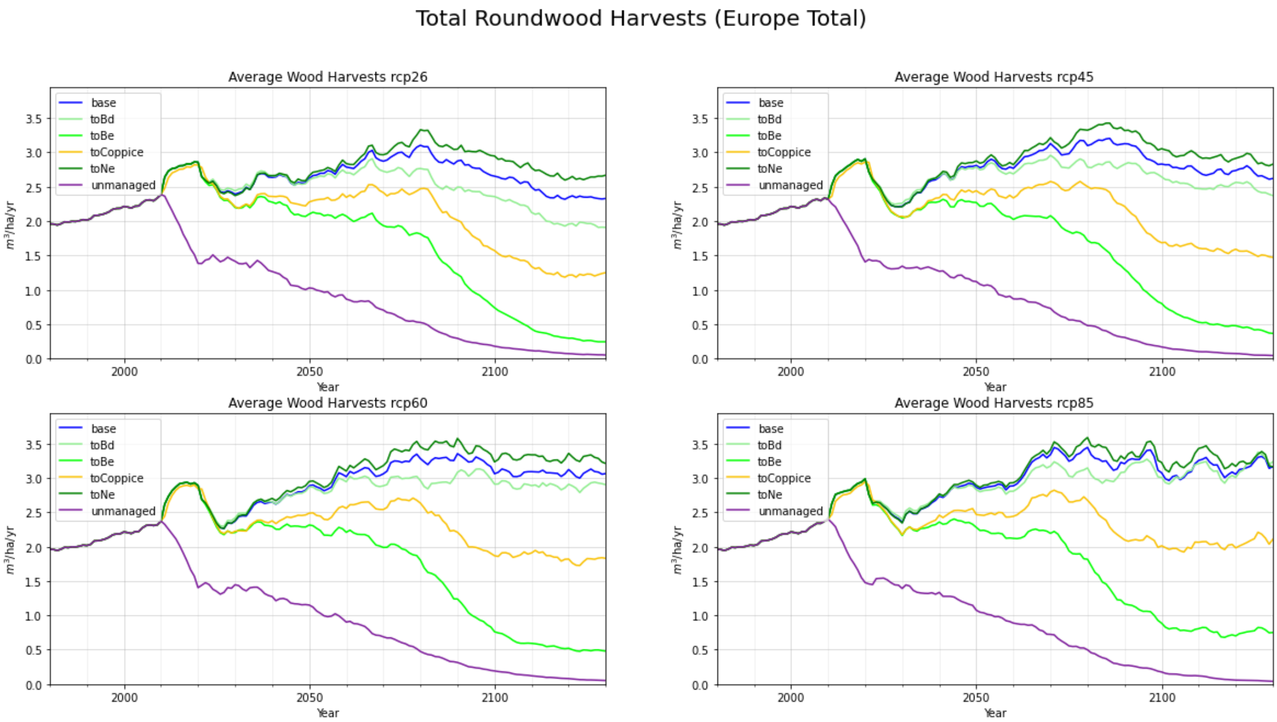

Figure S2: Time-series of harvests (10-year rolling average) in Europe for the 4 RCPs under different management options. Higher RCPs resulted in higher harvests due to the CO_2-fertilization effect. The spike after 2010 happened because some stands have reached maturity before that year and we scheduled their clearcut randomly in 2010-2020. As the forests reached maturity at different points in time, all conversions happened only gradually, hence harvests also only declined gradually in the unmanaged case. The reason for the decrease of harvests in the toBE case is simply because BE trees did not grow in all grid cells (but grew in more grid cells in higher RCPs).

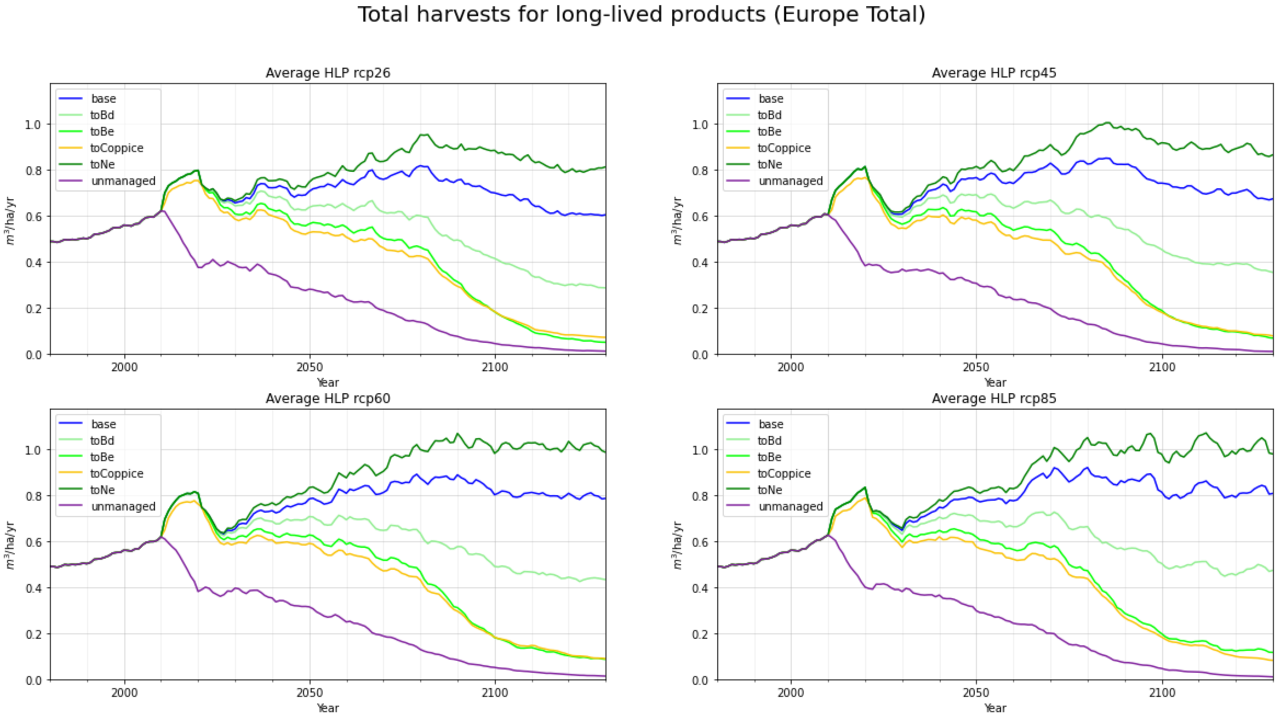

Figure S3: Like S2, but for harvests for long-lived products.

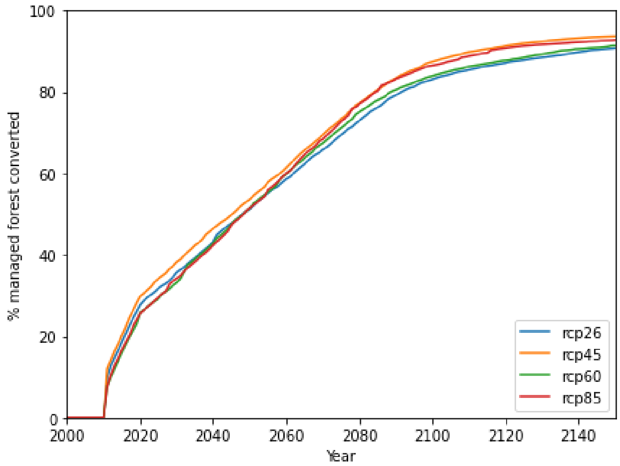

Figure S4: Process of conversion to optimized portfolios. Once a stand reached maturity, it was converted. Since stands had various ages in 2010, maturity was reached at different points in time, hence the gradual conversion.

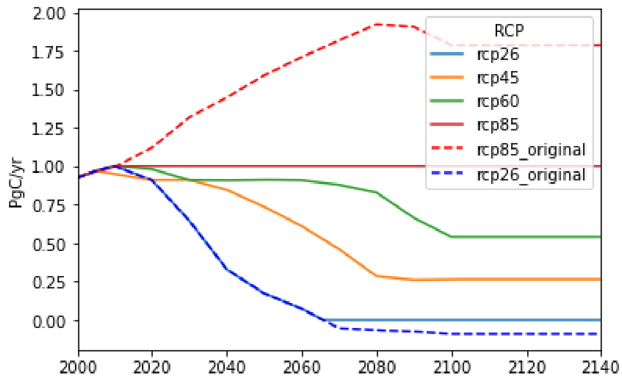

Figure S5: Substitution depends on carbon intensity of the industry. Similar to Brunet-Navarro et al. (2021) we used discounting factors for the different RCPs to account for decreasing amounts of substitution in the future when, e.g., the share of fossil fuels in the energy mix is lower. The substitution factors are based on the CO_2-emissions of the OECD countries, normalized to 2005 values. We additionally capped and floored those values at 1 and 0. Solid lines show the factor with which the substitution factors are multiplied in a given year. For reference, the dashed lines show the original (normalized) CO_2-emissions. Data retrieved from "RCP Database" (2021) which shows the RCP data (Clarke et al., 2007; Fujino et al., 2006; Hijioka et al., 2008; Riahi et al., 2007; Smith & Wigley, 2006; van Vuuren et al., 2007; Wise et al., 2009).

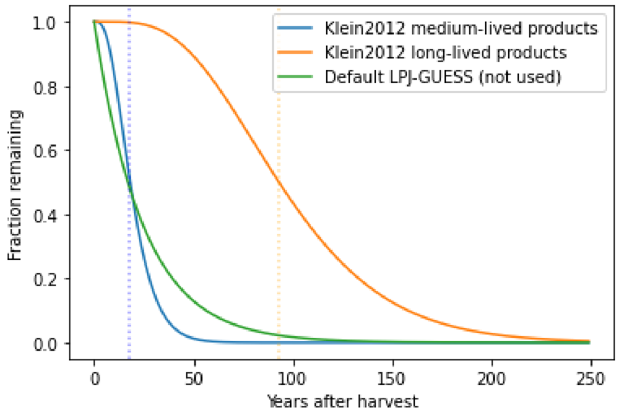

Figure S6: Decay functions based on cumulative density functions of the Gamma distributions as in Krause et al. (2020). Shape and scale parameters for the medium-lived (long-lived) pool are 3.68 and 5.52 (5.15 and 19.3), respectively. For comparison we show the default decay function of LPJ-GUESS. The Gamma functions account for the fact that wood products usually will not be turned over in the first years after they entered these pools, and that some products remain in usage for many decades or even centuries. The dashed vertical lines indicate the median residence times, i.e., when 50% of the products have decayed.

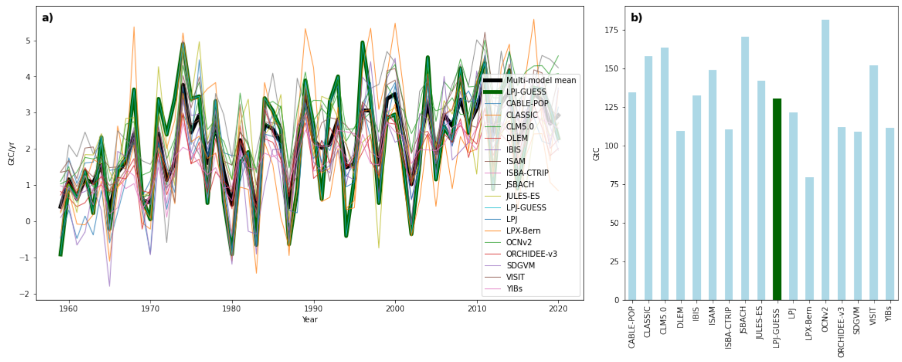

Figure S7: Response of LPJ-GUESS to historial increases in atmospheric CO_2-concentrations compared to other models of the Global Carbon Project (Friedlingstein et al., 2020). Plot a) shows the simulated terrestrial sink over time, while plot b) shows the aggregated C sink.

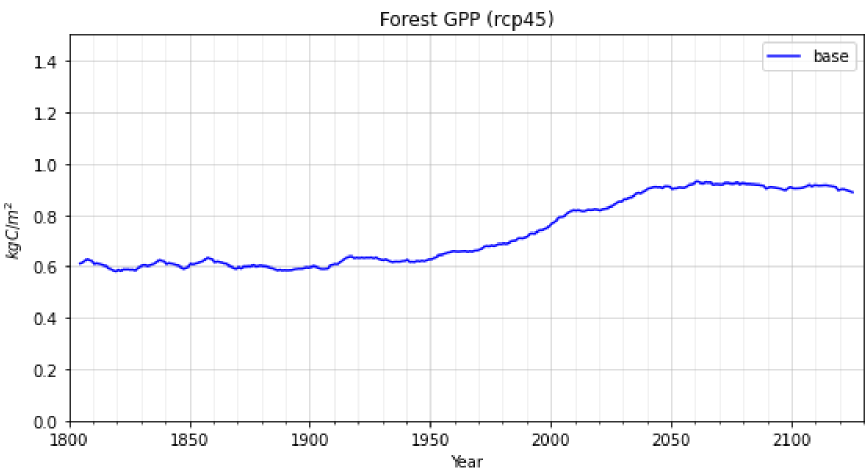

Figure S8: Change in GPP over time. Shown is modeled GPP by LPJ-GUESS for RCP 4.5 for the base management option.

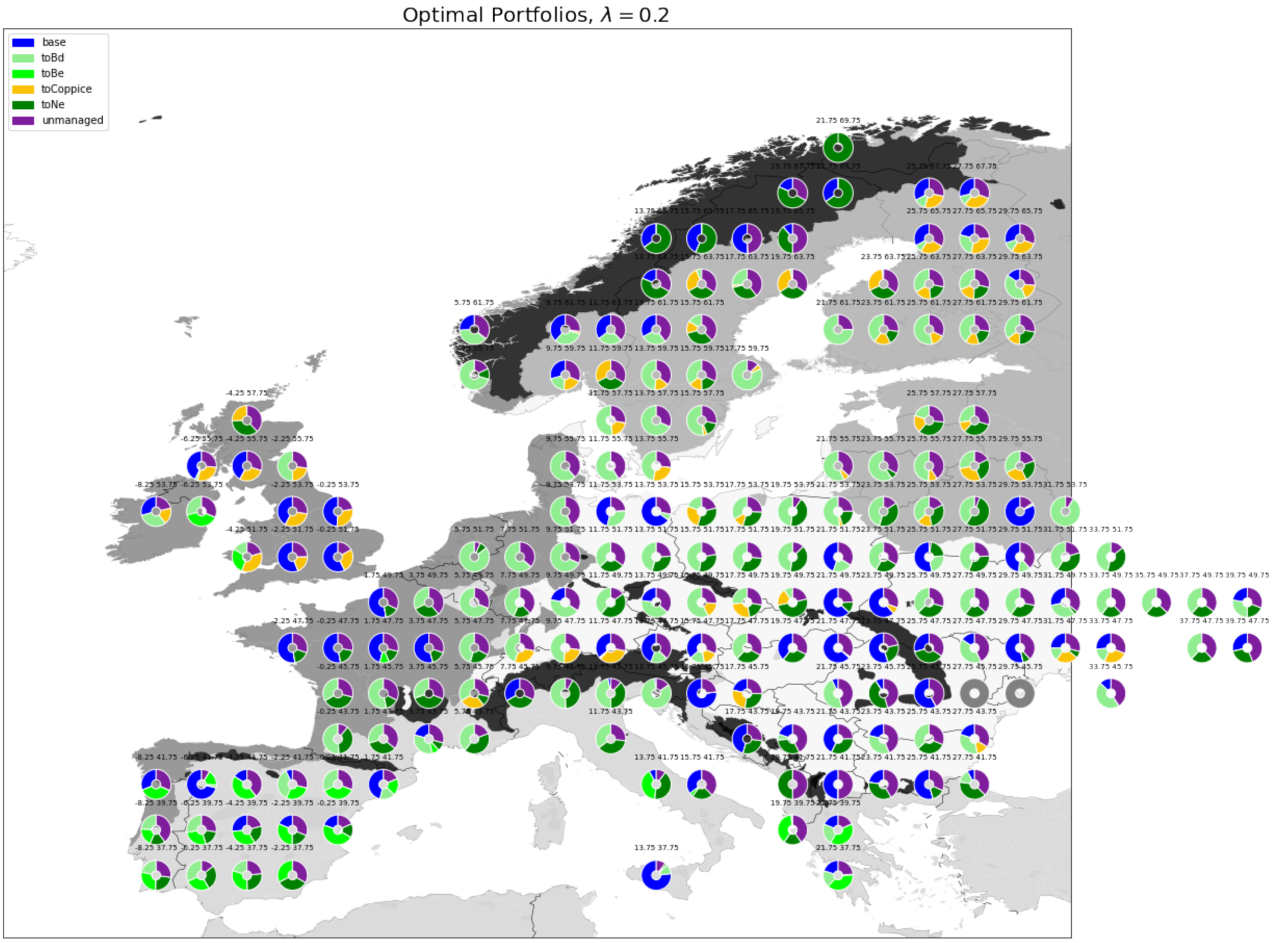

Figure S9: Visualization of the proposed management portfolios in Europe. The five regions are based on the climatic zones of Metzger et al. (2005) but aggregated as in the IPCC AR5 (Kovats et al., 2015). Grey circles mean that there was no management that could ensure forest cover in any RCP.

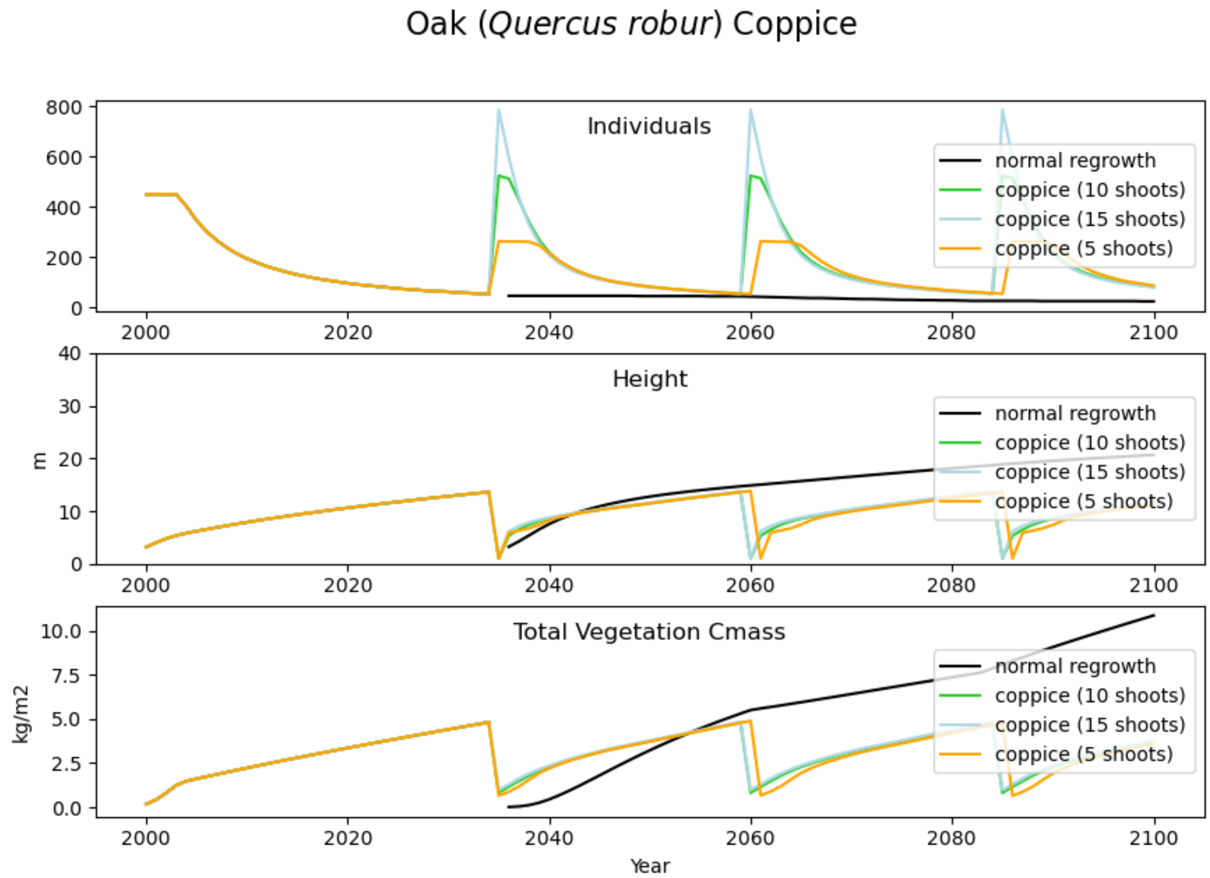

Figure S10: Time-series of tree regrowth using the implemented coppice mechanism. As soon as the trees reach a certain height/diameter, they are cut down. One year later they resprout, implemented as new individuals that share the same roots. Initially the growth is higher compared to a re-planted tree (black) but eventually, this initial benefit will be lost as is typical for coppice (West, 2014). The number of initial shoots per stool only has a marginal effect on the behavior of the coppiced tree.

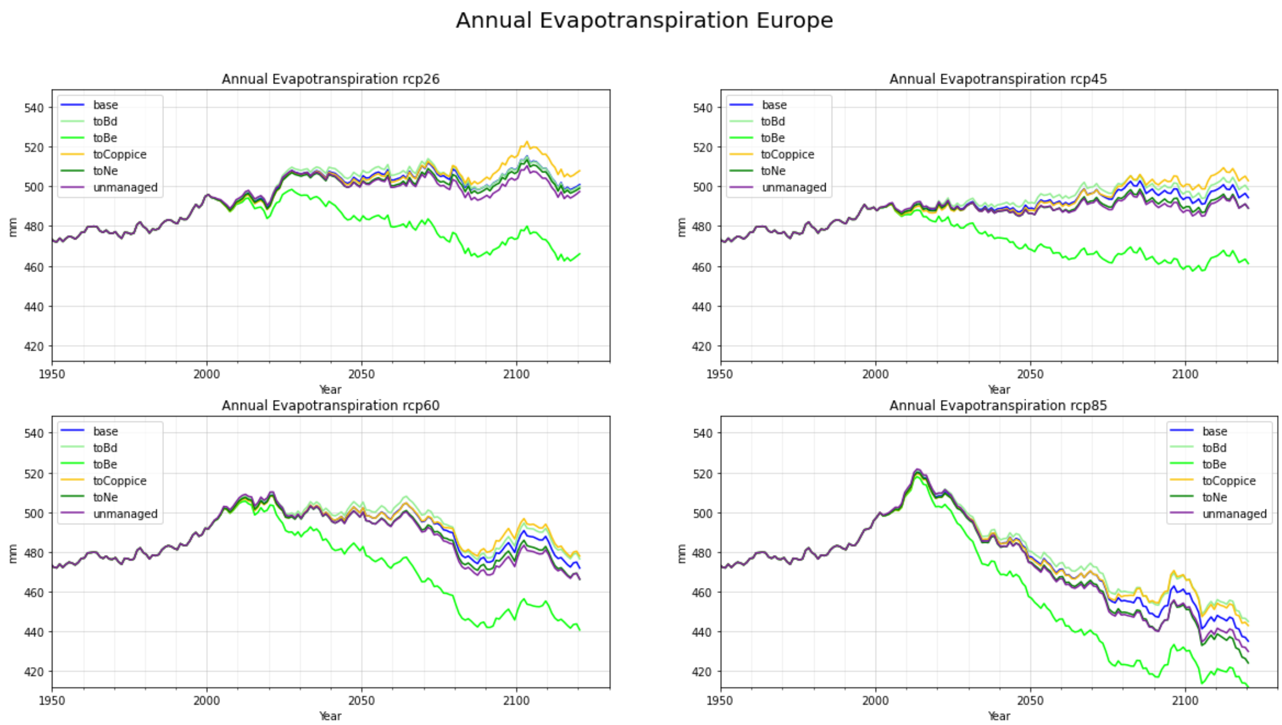

Figure S11: Projected average ET over Europe for different management options and RCPs. Shown is the 20-year rolling mean. Decreases in ET in the high RCP scenarios stemmed from increased water use efficiency.

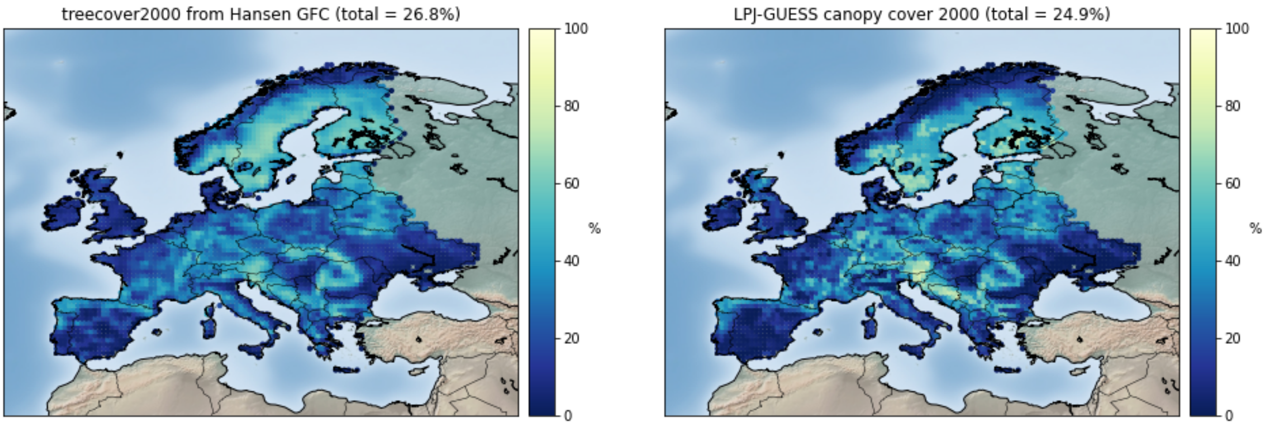

Figure S12: Comparison of observed and simulated tree cover in 2000. Hansen values were upsampled bilinearly to match our modeled 0.5°×0.5° degree grid cells. Hansen values also contain tree cover from patches not defined as forests.

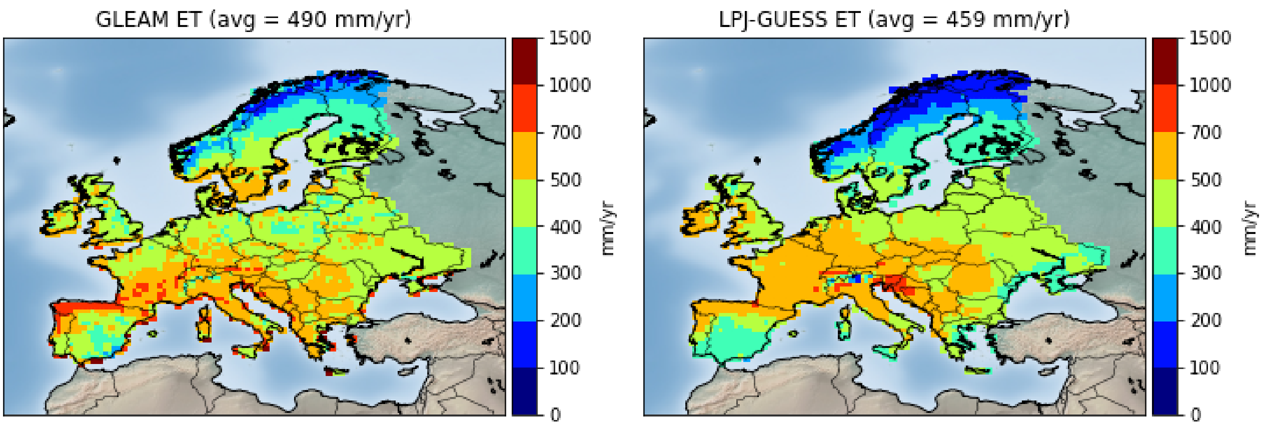

Figure S13: Average ET over 1980-2020 as modeled by GLEAM and by our study using LPJ-GUESS, assuming C3 grass growing in all areas that are not covered by forests.

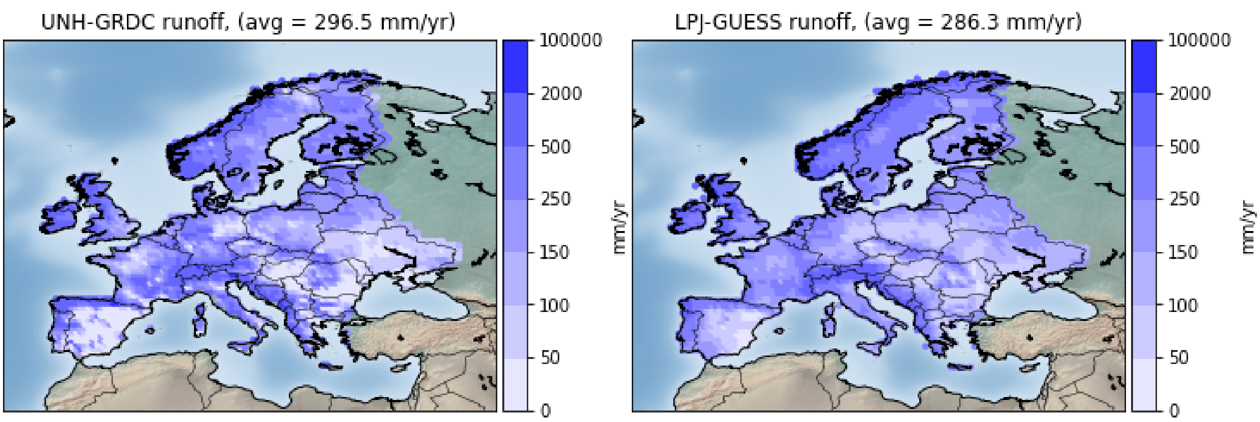

Figure S14: Comparison of UNH-GRDC vs. LPJ-GUESS runoff. UNH-GRDC is based on catchment data from various years of the past century and can be considered as a long-term average. LPJ-GUESS runoff was averaged over the 1950-2000 period. Total values are the area-weighted means. Assuming C3 grass growing in all areas that are not covered by forests.

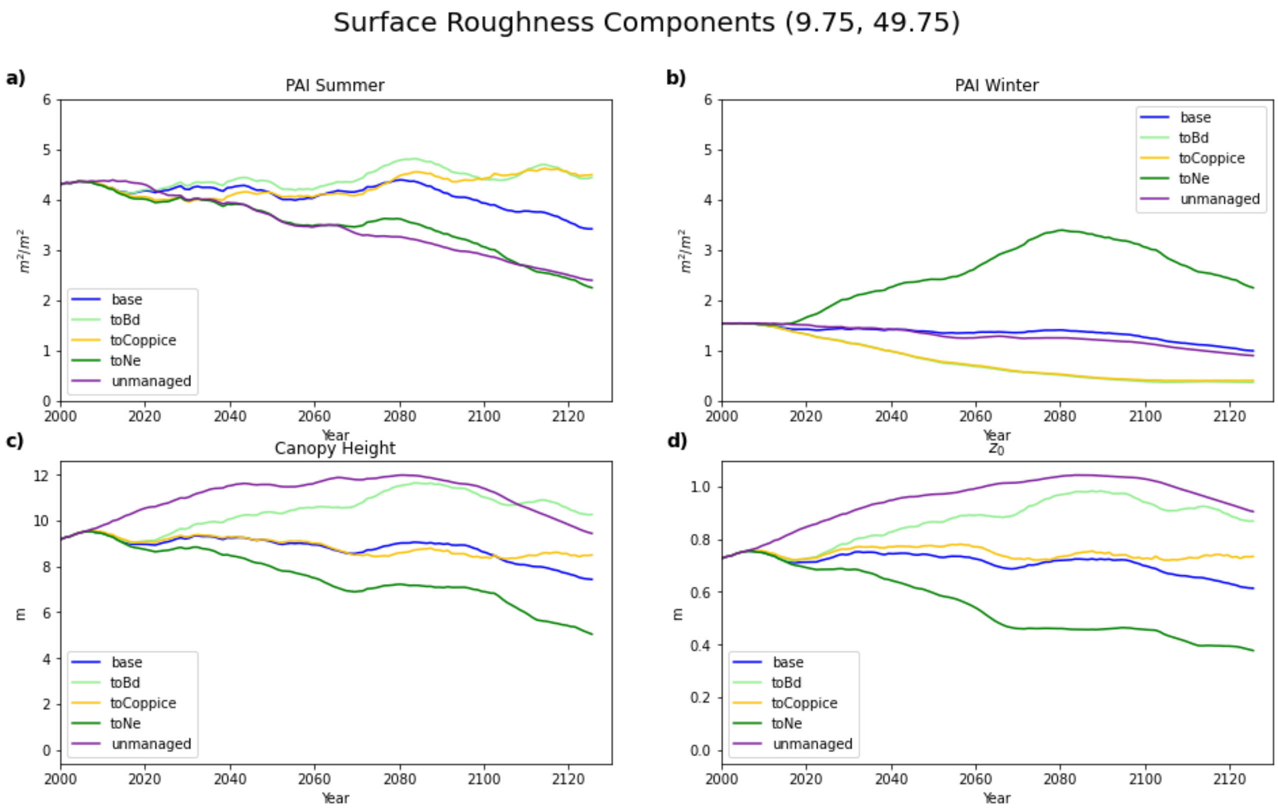

Figure S15: Components for surface roughness in a grid cell in Southern Germany (9.75, 49.75). **a)** and **b)** show the PAI values for summer and winter, respectively for the different management options. Winter-PAI for NE management converged to summer values once all forests are converted. Values for base and unmanaged lie in between NE and BD values as they contain both evergreen and deciduous forests. **c)** shows the canopy height as computed after Forrest et al. (2020) and **d)** shows the average yearly surface roughness using PAI and canopy height with the formulas of Raupach (1994).

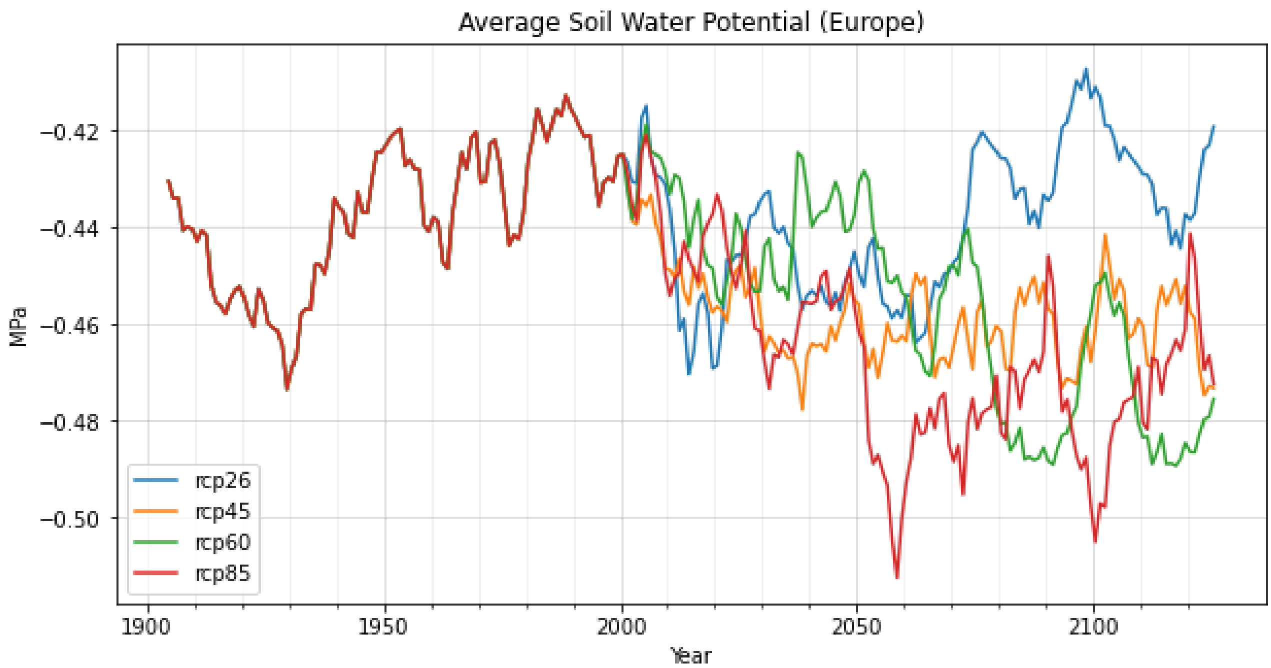

Figure S16: Different trends of soil water potentials for the RCPs. The graphs show the yearly average soil water potential over the entire continent for the base management. After the strong decrease until 2060, values for RCP8.5 started to increase again due to higher water use efficiency stemming from elevated CO_2-concentrations and due to an increase in precipitation towards the end of the century.

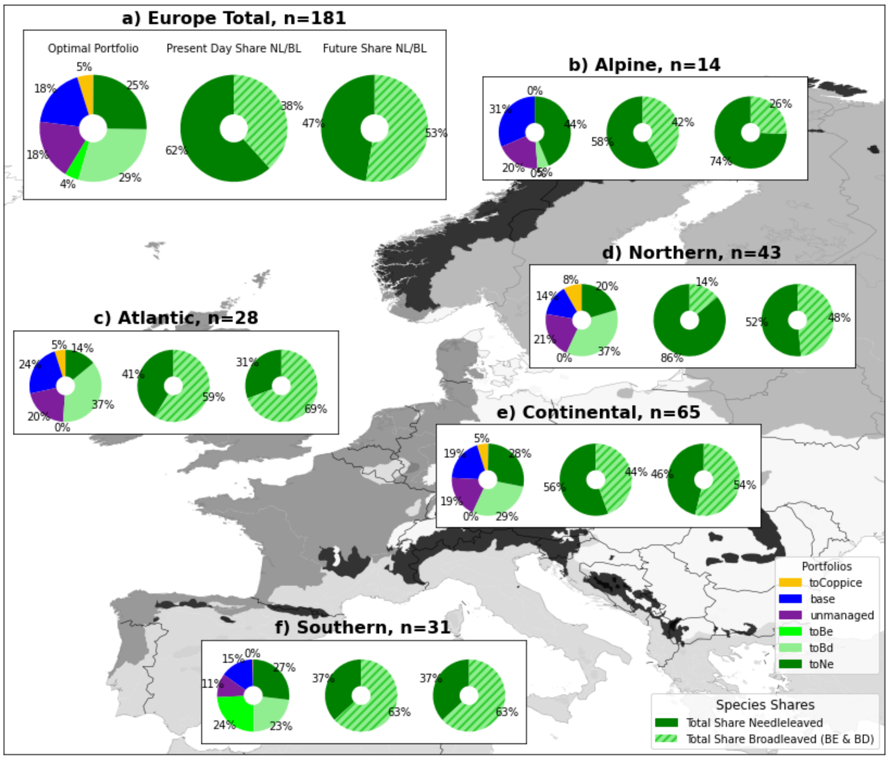

Figure S17: Doubling the importance of harvests via the weights in the optimization led to different portfolios compared to the standard optimization, including much lower shares of unmanaged forests.

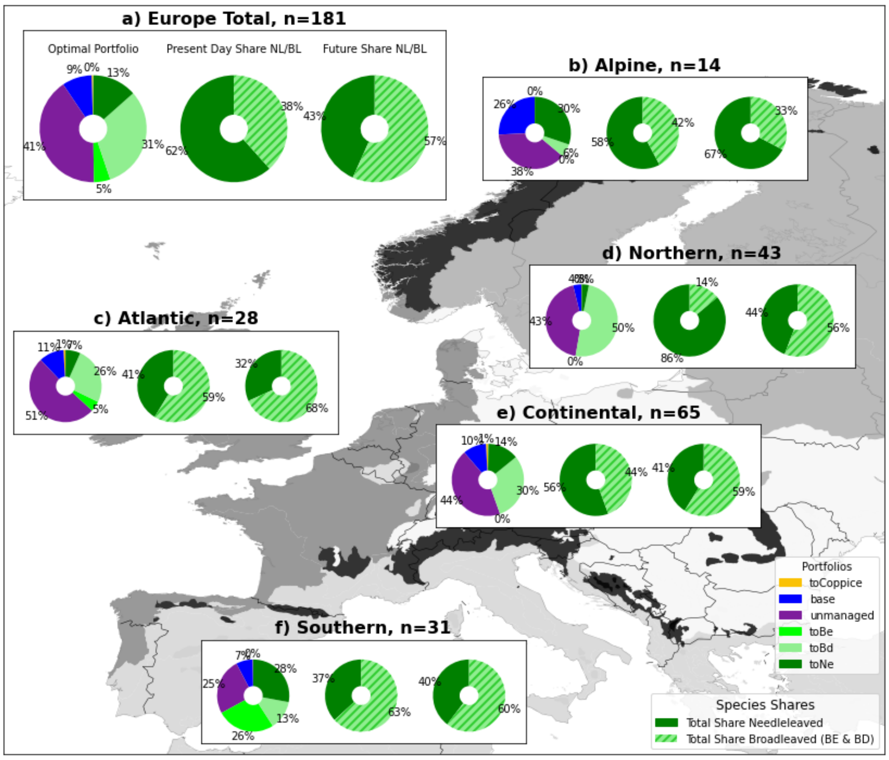

Figure S18: Doubling the importance of climate mitigation reduced the share of coppice and increased the share of unmanaged forest.

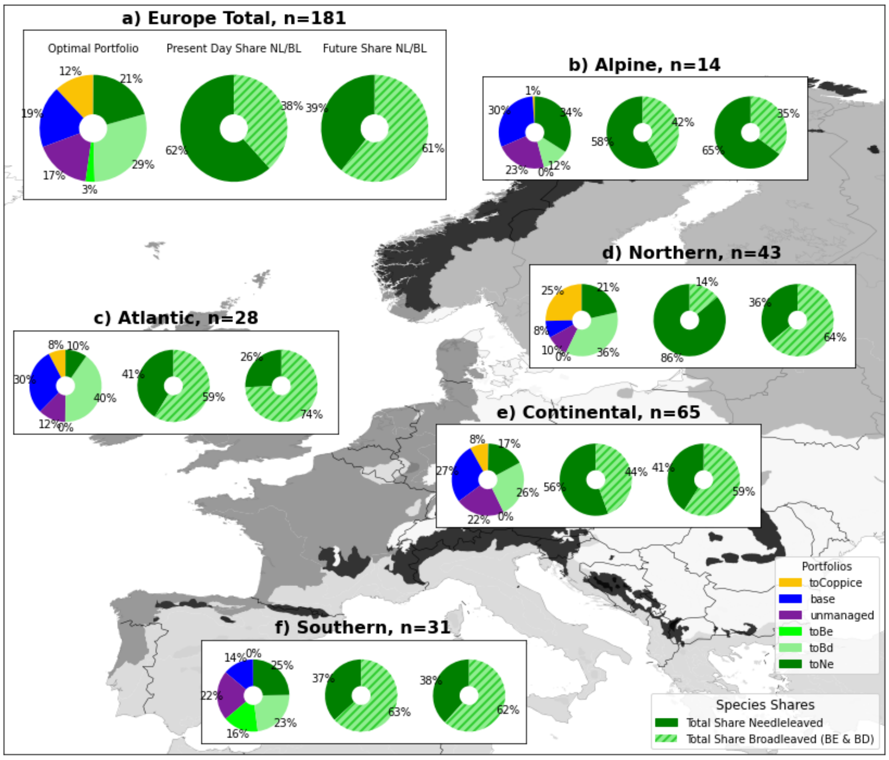

Figure S19: Portfolios for optimization without discounting of substitution effects. When displacement factors are kept constant in the future, the substitution effects outweigh the carbon sink from unmanaged forests.

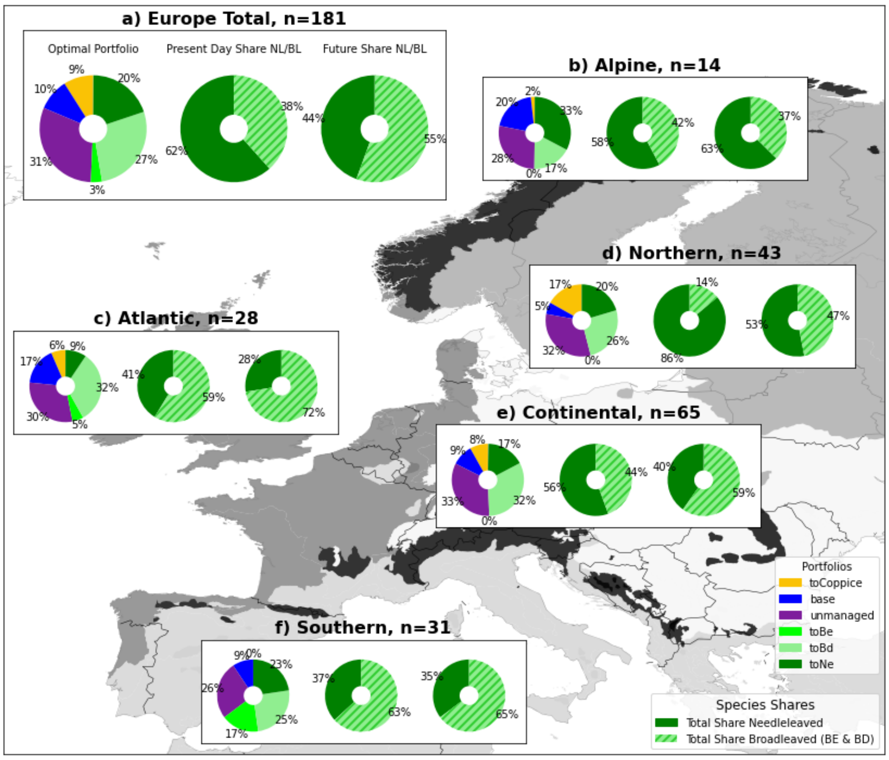

Figure S20: Portfolios for optimization including albedo.

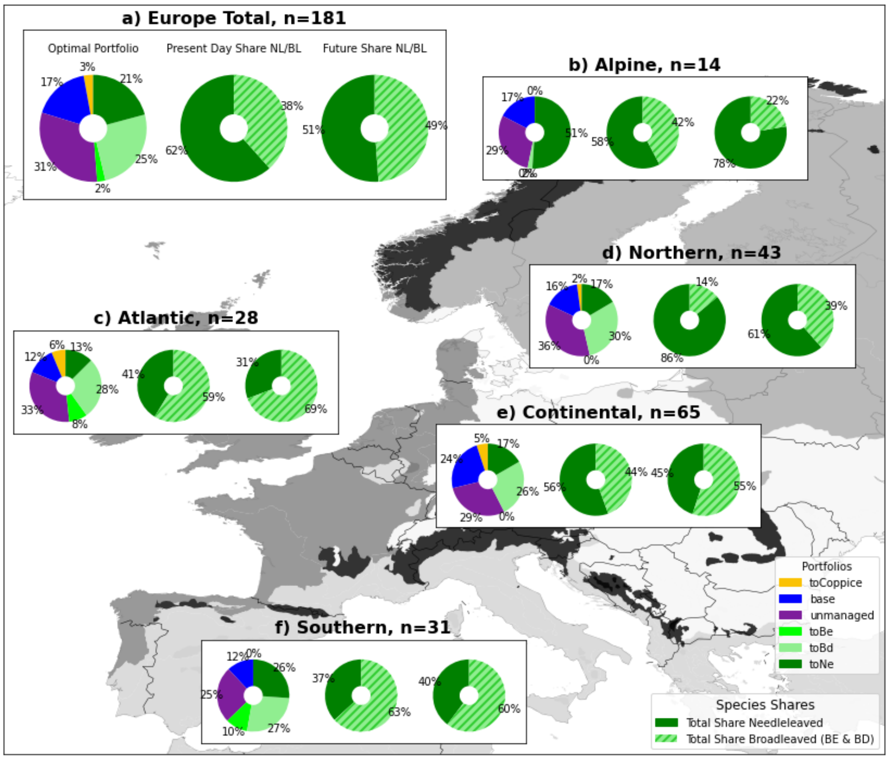

Figure S21: Portfolios when only optimizing for RCP2.6 only.

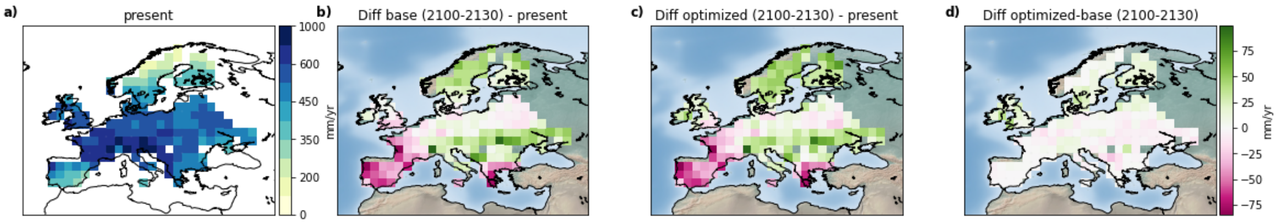

Figure S22: Projected changes in ET for RCP4.5 in Europe showed increases in Northern and Eastern Europe and decreases in Southern and Western Europe for the baseline management option. The benefit of the optimized portfolios compared to this was only marginal **(d)**.

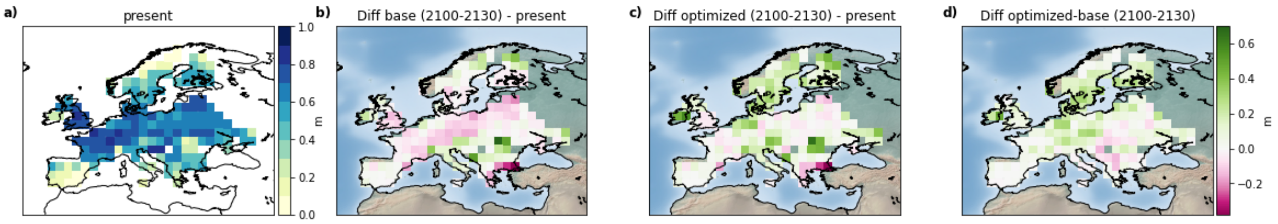

Figure S23: Surface roughness increased for most areas in Europe with the optimized portfolio, both compared to present-day and future base values. Depicted is RCP4.5.

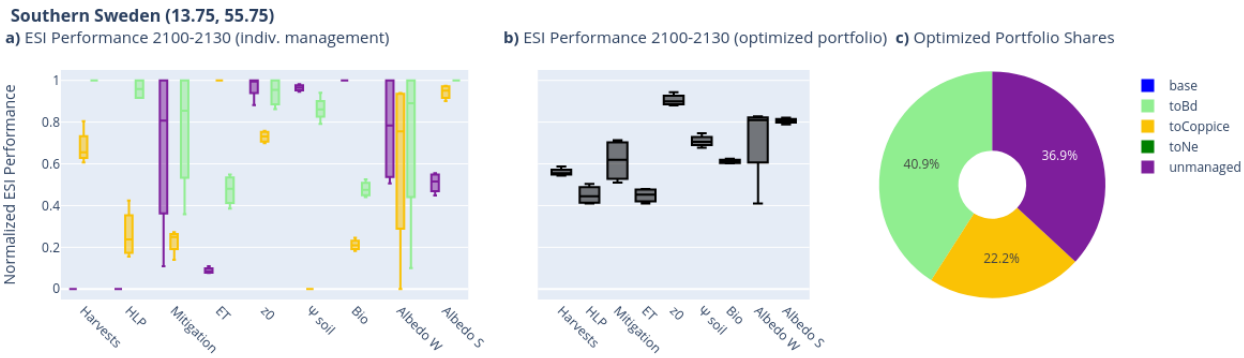

Figure S24: Results of the optimization of one grid cell when also including ESIs of summer and winter albedo.

References

Boisier, J. P., De Noblet-Ducoudré, N., & Ciais, P. (2013). Inferring past land use-induced changes in surface albedo from satellite observations: A useful tool to evaluate model simulations. *Biogeosciences*, *10*(3), 1501–1516. https://doi.org/10.5194/bg-10-1501-2013

Brunet-Navarro, P., Jochheim, H., Cardellini, G., Richter, K., & Muys, B. (2021). Climate mitigation by energy and material substitution of wood products has an expiry date. *Journal of Cleaner Production*, *303*, 127026. https://doi.org/10.1016/j.jclepro.2021.127026

Clarke, L., Edmonds, J., Jacoby, H., Pitcher, H., Reilly, J., & Richels, R. (2007). *Scenarios of greenhouse gas emissions and atmospheric concentrations. Sub-report 2.1 A of synthesis and assessment product 2.1 by the US Climate Change Science Program and the Subcommittee on Global Change Research.* Department of Energy, Office of Biological & Environmental Research, Washington DC., USA.

Dobor, L., Hlásny, T., Rammer, W., Zimová, S., Barka, I., & Seidl, R. (2020). Is salvage logging effectively dampening bark beetle outbreaks and preserving forest carbon stocks? *Journal of Applied Ecology*, *57*(1), 67–76. https://doi.org/10.1111/1365-2664.13518

Eurostat. (2021). Eurostat Forestry Database. Retrieved June 29, 2021, from https://ec.europa.eu/eurostat/web/forestry/data/database

Forrest, M., Tost, H., Lelieveld, J., & Hickler, T. (2020). Including vegetation dynamics in an atmospheric chemistry-enabled general circulation model: linking LPJ-GUESS (v4.0) with the EMAC modelling system (v2.53). *Geoscientific Model Development*, *13*(3), 1285–1309. https://doi.org/10.5194/gmd-13-1285-2020

Friedlingstein, P., O'Sullivan, M., Jones, M. W., Andrew, R. M., Hauck, J., Olsen, A., Peters, G. P., Peters, W., Pongratz, J., Sitch, S., Le Quéré, C., Canadell, J. G., Ciais, P., Jackson, R. B., Alin, S., Aragão, L. E. O. C., Arneth, A., Arora, V., Bates, N. R., ... Zaehle, S. (2020). Global Carbon Budget 2020. *Earth System Science Data*, *12*(4), 3269–3340. https://doi.org/10.5194/essd-12-3269-2020

Fujino, J., Nair, R., Kainuma, M., Masui, T., & Matsuoka, Y. (2006). Multi-gas Mitigation Analysis on Stabilization Scenarios Using Aim Global Model. *The Energy Journal*, *SI2006*(01), 343–353. https://doi.org/10.5547/ISSN0195-6574-EJ-VolSI2006-NoSI3-17

Harja, D., Subekti, R., & Pambudi, S. (2019). Tree functional attributes and ecological database. Retrieved November 16, 2021, from http://db.worldagroforestry.org

Hickler, T., Prentice, I. C., Smith, B., Sykes, M. T., & Zaehle, S. (2006). Implementing plant hydraulic architecture within the LPJ Dynamic Global Vegetation Model. *Global Ecology and Biogeography*, *15*(6), 567–577. https://doi.org/10.1111/j.1466-8238.2006.00254.x

Hickler, T., Vohland, K., Feehan, J., Miller, P. A., Smith, B., Costa, L., Giesecke, T., Fronzek, S., Carter, T. R., Cramer, W., Kühn, I., & Sykes, M. T. (2012). Projecting the future distribution of European potential natural vegetation zones with a generalized, tree species-based dynamic vegetation model. *Global Ecology and Biogeography*, *21*(1), 50–63. https://doi.org/10.1111/j.1466-8238.2010.00613.x

Hijioka, Y., Matsuoka, Y., & Nishimoto, H. (2008). Global GHG emission scenarios under GIC; concentration stabilization targets. *J. Glob. Environ. En*, *13*, 97–108.

Jenkins, J. C. (2004). *Comprehensive database of diameter-based biomass regressions for North American tree species* (Vol. 319). United States Department of Agriculture, Forest Service, Northeastern Research Station.

Jones, H. G. (2013). *Plants and Microclimate* (Vol. 22). Cambridge University Press. https://doi.org/10.1017/CBO9780511845727

Kärhä, K., Anttonen, T., Poikela, A., Palander, T., Laurén, A., Peltola, H., & Nuutinen, Y. (2018). Evaluation of salvage logging productivity and costs in windthrown Norway spruce-dominated forests. *Forests*, *9*(5). https://doi.org/10.3390/f9050280

Klein, D., Höllerl, S., Blaschke, M., & Schulz, C. (2013). The Contribution of Managed and Unmanaged Forests to Climate Change Mitigation—A Model Approach at Stand Level for the Main Tree Species in Bavaria. *Forests*, *4*(1), 43–69. https://doi.org/10.3390/f4010043

Knoke, T., Gosling, E., Thom, D., Chreptun, C., Rammig, A., & Seidl, R. (2021). Economic losses from natural disturbances in Norway spruce forests – A quantification using Monte-Carlo simulations. *Ecological Economics*, *185*(January), 107046. https://doi.org/10.1016/j.ecolecon.2021.107046

Kovats, R. S., Valentini, R., Bouwer, L. M., Georgopoulou, E., Jacob, D., Martin, E., Rounsevell, M., & Soussana, J. F. (2015). Europe. In V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, & K. J. Mach (Eds.), *Climate change 2014: Impacts, adaptation and vulnerability* (pp. 1267–1326). Cambridge University Press. https://doi.org/10.1017/CBO9781107415386.003

Krause, A., Knoke, T., & Rammig, A. (2020). A regional assessment of land-based carbon mitigation potentials: Bioenergy, BECCS, reforestation, and forest management. *GCB Bioenergy*, *12*(5), 346–360. https://doi.org/10.1111/gcbb.12675

Kucharik, C. J., Norman, J. M., & Gower, S. T. (1998). Measurements of branch area and adjusting leaf area index indirect measurements. *Agricultural and Forest Meteorology*, *91*(1-2), 69–88. https://doi.org/10.1016/S0168-1923(98)00064-1

Lindenmayer, D. B., Burton, P. J., & Franklin, J. F. (2012). *Salvage logging and its ecological consequences*. Island Press.

Lindenmayer, D., & Noss, R. (2006). Salvage Logging, Ecosystem Processes, and Biodiversity Conservation. *Conservation Biology*, *20*(4), 949–958. https://doi.org/10.1111/j.1523-1739.2006.00497.x

Lindeskog, M., Smith, B., Lagergren, F., Sycheva, E., Ficko, A., Pretzsch, H., & Rammig, A. (2021). Accounting for forest management in the estimation of forest carbon balance using the dynamic vegetation model LPJ-GUESS (v4.0, r9710): implementation and evaluation of simulations for Europe. *Geoscientific Model Development*, *14*(10), 6071–6112. https://doi.org/10.5194/gmd-14-6071-2021

McGroddy, M. E., Daufresne, T., & Hedin, L. O. (2004). SCALING OF C:N:P STOICHIOMETRY IN FORESTS WORLDWIDE: IMPLICATIONS OF TERRESTRIAL REDFIELD-TYPE RATIOS. *Ecology*, *85*(9), 2390–2401. https://doi.org/10.1890/03-0351

Metzger, M. J., Bunce, R. G. H., Jongman, R. H. G., Mücher, C. A., & Watkins, J. W. (2005). A climatic stratification of the environment of Europe. *Global Ecology and Biogeography*, *14*(6), 549–563. https://doi.org/10.1111/j.1466-822X.2005.00190.x

Ministry of Agriculture and Forestry Finland. (2013). Forest Damages Prevention Act (1087/ 2013). Retrieved July 5, 2021, from http://www.finlex.fi/en/laki/kaannokset/2013/en20131087.pdf

Möllmann, T. B., & Möhring, B. (2017). A practical way to integrate risk in forest management decisions. *Annals of Forest Science*, *74*(4). https://doi.org/10.1007/s13595-017-0670-x

Raupach, M. R. (1994). Simplified expressions for vegetation roughness length and zero-plane displacement as functions of canopy height and area index. *Boundary-Layer Meteorology*, *71*(1-2), 211–216. https://doi.org/10.1007/BF00709229

RCP Database. (2021). Retrieved July 16, 2021, from https://tntcat.iiasa.ac.at/RcpDb

Riahi, K., Grübler, A., & Nakicenovic, N. (2007). Scenarios of long-term socio-economic and environmental development under climate stabilization. *Technological Forecasting and Social Change*, *74*(7), 887–935. https://doi.org/10.1016/j.techfore.2006.05.026

Savill, P. S. (Ed.). (2019). *The silviculture of trees used in British forestry*. CABI. https://doi.org/10.1079/9781786393920.0000

Smith, S. J., & Wigley, T. (2006). Multi-Gas Forcing Stabilization with Minicam. *The Energy Journal*, *SI2006*(01), 373–391. https://doi.org/10.5547/ISSN0195-6574-EJ-VolSI2006-NoSI3-19

Stimm, B., Roloff, A., Lang, U. M., & Weisgerber, H. (Eds.). (2014). *Enzyklopädie der Holzgewächse: Handbuch und Atlas der Dendrologie*. Wiley. https://doi.org/10.1002/9783527678518

Thorn, S., Chao, A., Georgiev, K. B., Müller, J., Bässler, C., Campbell, J. L., Castro, J., Chen, Y. H., Choi, C. Y., Cobb, T. P., Donato, D. C., Durska, E., Macdonald, E., Feldhaar, H., Fontaine, J. B., Fornwalt, P. J., Hernández, R. M. H., Hutto, R. L., Koivula, M., ... Leverkus, A. B. (2020). Estimating retention benchmarks for salvage logging to protect biodiversity. *Nature Communications*, *11*(1), 1–8. https://doi.org/10.1038/s41467-020-18612-4

van Vuuren, D. P., den Elzen, M. G. J., Lucas, P. L., Eickhout, B., Strengers, B. J., van Ruijven, B., Wonink, S., & van Houdt, R. (2007). Stabilizing greenhouse gas concentrations at low levels: an assessment of reduction strategies and costs. *Climatic Change*, *81*(2), 119–159. https://doi.org/10.1007/s10584-006-9172-9

Wang, Z., & Zeng, X. (2010). Evaluation of snow albedo in land models for weather and climate studies. *Journal of Applied Meteorology and Climatology*, *49*(3), 363–380. https://doi.org/10.1175/2009JAMC2134.1

West, P. W. (2014). *Growing Plantation Forests* (Vol. 9783319018). Springer International Publishing. https://doi.org/10.1007/978-3-319-01827-0

Wise, M., Calvin, K., Thomson, A., Clarke, L., Bond-Lamberty, B., Sands, R., Smith, S. J., Janetos, A., & Edmonds, J. (2009). Implications of Limiting CO2 Concentrations for Land Use and Energy. *Science*, *324*(5931), 1183–1186. https://doi.org/10.1126/science.1168475

Reconciling the EU forest, biodiversity, and climate strategies

Konstantin Gregor[1,*], Christopher P.O. Reyer[2], Thomas A. Nagel[3], Annikki Mäkelä[4, 5], Andreas Krause[1], Thomas Knoke[1], and Anja Rammig[1]

[1]TUM School of Life Sciences, Technical University of Munich, Freising, Germany

[2]Potsdam Institute for Climate Impact Research, Member of the Leibniz Association, Potsdam, Germany

[3]University of Ljubljana, Department of Forestry and Renewable Forest Resources, Biotechnical Faculty, 1000 Ljubljana, Slovenia

[4]Department of Forest Sciences, University of Helsinki, Latokartanonkaari 7, Helsinki, Finland

[5]Institute for Atmospheric and Earth System Research / Forest Sciences, Faculty of Agriculture and Forestry, University of Helsinki, Finland

[*]Corresponding author, konstantin.gregor@tum.de

Abstract

Forests provide important ecosystem services, including climate change mitigation, local climate regulation, habitat for biodiversity, wood and non-wood products, energy, and recreation. Simultaneously, forests are increasingly affected by climate change and need to be adapted to future environmental conditions. Current legislation, including the EU Biodiversity Strategy, EU Forest Strategy, and national laws, aims at protecting forest landscapes, enhancing ecosystem services, adapting forests to climate change, and leveraging forest products for climate change mitigation and the bio-economy. However, reconciling all these competing demands poses a tremendous task for policy-makers, forest managers, conservation agencies, and other stakeholders, especially given the uncertainty associated with future climate impacts.

Here, we used process-based ecosystem modeling and robust multi-criteria optimization to develop forest management portfolios that provide multiple ecosystem services across a wide range of climate scenarios. We included constraints to both strictly protect 10% of Europe's land area and to provide stable harvest levels under every climate scenario. The optimization showed that there were limited options to improve ecosystem service provision within these constraints. Consequently, management portfolios suffered from low diversity, which contradicts the goal of multi-functionality and exposes regions to significant risk due to a lack of risk diversification. Additionally, certain regions, especially those in the north, would need to prioritize timber provision to compensate for reduced harvests elsewhere. This conflicts with EU LULUCF targets for increased forest carbon sinks in all member states and prevents an equal distribution of strictly

protected areas, introducing a bias as to which forest ecosystems are more protected than others. Thus, coordinated strategies at the European level are imperative to address these challenges effectively.

We suggest that the implementation of the EU Biodiversity Strategy, EU Forest Strategy, and targets for forest carbon sinks require complementary measures to alleviate the conflicting demands on forests.

1 Introduction

Climate change and biodiversity loss are among humanity's most pressing issues (IPBES, 2019; IPCC, 2023a). The progress in climate change mitigation has been slow and has fallen short of targets set by the Paris Agreement (United Nations, 2023). Nevertheless, there is a growing trend worldwide to enact legislation addressing climate change (Eskander & Fankhauser, 2020). Likewise, biodiversity loss is continuing at an alarming rate. But despite international policy efforts, it often receives less attention than climate change (Barbier et al., 2018), sometimes overshadowing the intricate relationship between the two issues (Pörtner et al., 2023; Sage, 2020). On the one hand, a significant portion of biodiversity loss is linked to rising temperatures. Thus, limiting global warming is crucial for preserving biodiversity (Ohashi et al., 2019; Warren et al., 2018). On the other hand, future land use changes stemming from mitigation policies can be detrimental to biodiversity (Hof et al., 2018; Ohashi et al., 2019). Hence, there is a clear need for more concrete actions and legislative measures to support biodiversity conservation, especially in Europe, where over 80% of the land surface have been transformed over the past millennia (EEA, 2023; Ellis et al., 2021).

To combat biodiversity loss, the European Union (EU) created the *EU Biodiversity Strategy for 2030* (European Commission, 2020). Key objectives include the protection of 30% of its land area by 2030, with 10% strictly protected, the planting of 3 billion trees, and the establishment of ecological corridors. "Protection" refers to responsible management and the prevention of deterioration. Protected forests can be managed for timber, but harvest levels are typically subject to restrictions (Verkerk, Zanchi, et al., 2014). "Strict protection" means maintaining ecosystems in an unmanaged state, with interventions limited to those sustaining natural processes (e.g., wildlife population control, European Commission, 2022). At present, 26% of the EU's land area are legally protected, and 3% strictly protected (European Commission, 2020; Forest Europe, 2020). Since 35% of Europe is covered with forests and most other land covers are more intensively used than forests, a substantial portion of newly protected areas will lie in forests (Forest Europe, 2020; Hengl et al., 2018). In addition, the *New EU Forest Strategy for 2030* was proposed, promoting broad-leaved species, forest multi-functionality, carbon sequestration and long-lived wood products, synergies between wood production and conservation, and forest adaptation to climate change (European Commission, 2021). Such forward-looking objectives are also often subsumed under the term "climate-smart forestry" (Nabuurs et al., 2018). Furthermore, both strategies demand the strict protection of Europe's remaining old-growth and primary forests. Non-EU states have similar strategies in place (e.g., FOEN, 2012; House of Lords, 2023).

Managed forests are critical for the European economy, providing income, jobs, and essential resources (Forest Europe, 2020). The demand for wood products has recently been growing (FAO, 2022a, 2022b; Nabuurs et al., 2007), and further increases are likely, also driven by the transition to a bioeconomy (Hurmekoski et al., 2022). Forests also contribute to climate change mitigation through the forest and product carbon sink, and by substituting carbon-intensive non-wood products (e.g., Grassi et al., 2021). Additionally, woody bioenergy plays a key role in Europe's energy transition (European Commission, 2021). Furthermore, forests offer numerous important ecosystem services (ESs), including biodiversity preservation, local climate regulation, water cycling, and recreation.

The corresponding complex demands placed on forests result in intricate trade-offs. Particularly, the relationships among biodiversity protection, timber production, mitigation, and adaptation, have been extensively discussed in the scientific literature. Most studies indicate a conflict between biodiversity protection and timber production (Başkent & Kašpar, 2023; Felton et al., 2016; Gutsch et al., 2018; Verkerk, Mavsar, et al., 2014), although some suggest synergies (Biber et al., 2020). Additionally, there is an ongoing debate regarding the mitigation potential of intensively managed, extensively managed, and unmanaged forests (Dugan et al., 2018; Gregor et al., 2024; Gustavsson et al., 2021; Peng et al., 2023; Petersson et al., 2022; Roebroek et al., 2023; Schulte et al., 2022; Soimakallio et al., 2021).

One potential strategy to address these trade-offs is regional specialization, focusing on wood production in highly productive regions (e.g., Lessa Derci Augustynczik & Yousefpour, 2021). This *land sparing* approach allows for increased production in one region while setting aside land for conservation elsewhere (Balmford, 2021). In Europe, however, *land sharing* typically prevails, where both production and protection objectives are pursued on the same land (Betts et al., 2021), but this could interfere with *strict* protection goals.

Developing forward-looking forest management strategies is a challenging task. One approach is to use management portfolios, as demonstrated by Luyssaert et al. (2018), who optimized portfolios for single objectives, such as maximizing carbon sequestration. Assessing multi-functionality on the other hand, i.e., the provision of multiple ESs, has been explored by Diaz-Balteiro et al. (2017), who selected optimal forest management types for various climate scenarios to find the single best management option in a case study in Spain. Here, we combine the two approaches by developing portfolios for multi-functionality under climate change.

The task is further complicated by the vulnerability of forests to different degrees of climate change and associated disturbances (IPCC, 2014; Senf & Seidl, 2021a, 2021b; Spinoni et al., 2018). Consequently, it is necessary to assess various forest functions under a range of climate scenarios to develop strategies for climate-adapted, multi-functional forests today. Robust multi-criteria optimization offers a valuable tool for this purpose (Ben-Tal & Nemirovski, 2002; Groetzner & Werner, 2022; Ishizaka & Nemery, 2013; Knoke et al., 2016; Uhde et al., 2017). Gregor et al. (2022) employed this approach to compute forest management portfolios for Europe, ensuring the provision of various ESs across a wide range of climate scenarios. They found that significant portions of unmanaged forests and a gradual transition to more broad-leaved species are beneficial for multi-functional forest landscapes in the face of climate change. However, this would also lead to strong reductions in wood harvests, conflicting with rising wood demands and the objective of leveraging wood products for climate change mitigation.

Here, we investigate to which extent reconciling targets for forest protection, wood production, mitigation, and the provision of ESs is feasible. We enhanced the methodology of Gregor et al. (2022) by incorporating Europe-wide constraints on harvest levels and forest protection that must be met under all climate scenarios. Specifically, we explored whether strategies for multi-functional forests can align with stable wood production and the EU's legal aims for strict forest protection and carbon sequestration. Furthermore, we examined the resulting impacts of these constraints on other ESs and the diversity of management strategies. We considered how the burden imposed by these constraints can be equitably distributed among regions, in line with the directive that all member states should contribute their "fair share of the effort" (European Commission, 2020).

2 Methods

In this study, building upon simulations with a dynamic vegetation model, we computed forest management portfolios that provide multiple ecosystem services (ESs) in an optimally balanced way, while considering the uncertainty of future climate. In previous work, this optimization was carried out independently for each grid cell (Gregor et al., 2022), providing one management portfolio suitable for all emission scenarios (Fig. S1). Here, we substantially extended this methodology by introducing Europe-wide hard constraints on ES provisioning that had to be met under all emission scenarios. This implied that grid cells were no longer independent entities. They were not required to meet all constraints individually, provided they were compensated for by other grid cells.

2.1 Forest management simulations

2.1.1 Dynamic vegetation model

We employed the dynamic vegetation model LPJ-GUESS for the forest simulations. LPJ-GUESS simulates various ecological processes, including photosynthesis, water uptake, carbon allocation, soil and litter dynamics, the nitrogen cycle, as well as the growth, competition, management, mortality, and establishment of plant functional types (Haxeltine & Prentice, 1996; Lindeskog et al., 2021; Sitch et al., 2003; Smith et al., 2014; Smith et al., 2001). We used the parametrization of European tree species which are characterized by various parameters such as phenology, growth form, bioclimatic limits, and shade-tolerance (Hickler et al., 2012). See Smith et al. (2014) for a detailed description of the model and Lindeskog et al. (2021) for details on the forest management module. LPJ-GUESS was designed to assess the impacts of climate change on terrestrial vegetation and has been thoroughly benchmarked against numerous independent regional and global estimates of carbon fluxes, harvests, biomass, CO_2-fertilization, and other datasets (Chang et al., 2017; Friedlingstein et al., 2022; Haverd et al., 2020; Ito et al., 2017; Lindeskog et al., 2021). Simulations were conducted in "cohort-mode", with age classes represented by a number of individuals sharing the same characteristics. We used 25 replicate patches to represent random samples of the same stand.

2.1.2 Simulation protocol

The modeled region of interest was Europe, excluding Georgia, Iceland, Cyprus, and Russia (except for the Kaliningrad region), simulated at 0.5°×0.5° resolution. LPJ-GUESS was forced with monthly temperature, radiation, and precipitation data (including number of wet days) from CMIP5 simulations (Taylor et al., 2012) of the general circulation model IPSL-CM5A-MR (Dufresne et al., 2013), as well as nitrogen deposition (Lamarque et al., 2011) and CO_2-concentrations (Meinshausen et al., 2011), all for the representative concentration pathways (RCPs) 2.6, 4.5, 6.0, and 8.5. The climate input was bias-corrected against CRU-NCEP and interpolated bi-linearly from a spatial resolution of 2.5°×1.25° to 0.5°×0.5° (Ahlström et al., 2012). To bring soil pools into equilibrium, a 1200-year spinup period was conducted using cycled, detrended 1850–1879 climate data. Afterwards, the time period 1900–2130 was simulated using transient climate. The species map of Brus et al. (2012) was combined with the forest age map of Poulter et al. (2018) to prescribe clear-cuts and plantings in the historical simulation period. This ensured a realistic representation of European forests in 2010 in terms of species, age distribution, and total forest cover per grid cell (Fig. S2). We focused on forests that are currently available for wood supply. To map these areas, we defined the oldest age class of the age dataset (older than 140 years in 2010) as forests that are not available for wood supply, keeping this area stable for the simulation runs. This simple indicator resulted in a good approximation of country-reported areas of forests available for wood supply (Fig. S3).

Disturbances were modeled as patch-destroying events with return intervals dependent on the forest type, namely 1000 years for broad-leaved deciduous species, 500 years for broad-leaved evergreen species, and 300 years for needle-leaved species (Pugh et al., 2019). An annual 1% increase in disturbance probabilities, starting in 2010, was assumed based on trends derived from satellite observations (Senf & Seidl, 2021a).

2.1.3 Forest management, wood usage, and substitution effects

In the model, forest management is implemented through thinning and final harvest. Commercial thinnings are based on Reineke's self-thinning rule, while the rotation period depends on the forest type and target densities (Lindeskog et al., 2021; Reineke, 1933). This led to the model harvesting the total net annual increment (NAI) and thus constant carbon stocks. In reality only roughly three-fourths of NAI are harvested each year, but with higher shares in productive countries like Finland (Forest Europe, 2020). We accounted for this by refraining from thinning on 20% of patches, which led to harvest levels close to observations (Fig. S4). Simple coppice management was implemented, allowing broad-leaved species to resprout from the stumps after cutting (Gregor et al., 2022). Wood usage was implemented depending on the species type (Eurostat, 2023a). Specifically, 23% (2.5%) of the stem mass of conifers (non-conifers) was allocated to the long-lived product pool, and 9.4% (11.9%) to the medium product pool. 12% (49%) was used as fuel wood, while the remaining portion was returned to the atmosphere within one year. 40% of twigs and their leaves were harvested as fuel wood, the remainder was left to decay on site together with the coarse roots (see Lindeskog et al. (2021)). Each of the product pools had its own decay function, which accounted for the age of each product (Fig. S5).

Substitution effects, which refer to avoided emissions due to the replacement of carbon-intensive products with wood products, were incorporated into the model based on Knauf et al. (2015): Present-day displacement factors of 1.5 tC/tC for materials and 0.67 tC/tC for fuels (denoting avoided emissions per ton carbon in the final product) were applied. The 1.5 tC/tC does not contain end-of-life handling. For this, we assumed 23% of materials to be land-filled at the end of their lifetime (Eurostat, 2023c), leading to a reduction in the displacement factor to 1.1 tC/tC to account for landfill emissions (Sathre & O'Connor, 2010). The other 77% were assumed to be used to generate energy. The displacement factors were discounted over time according to the RCPs, reflecting the projected decrease in carbon-intensity of non-wood products over time (Brunet-Navarro et al., 2021; Gregor et al., 2022).

2.1.4 Management options and management change

Six simplified management options were implemented (Fig. 1). At the time of the final harvest, one of the options was chosen: replanting the same species composition (*base*), converting to needle-leaved evergreen, broad-leaved deciduous, broad-leaved evergreen, or coppice forests (*toNe*, *toBd*, *toBe*, and *toCoppice*), respectively, or refraining from the final harvest and leaving the forest untouched from this point in time (*unmanaged*). For the conversion to coppice, broad-leaved trees were cut down and allowed to regrow from the stumps, while needle-leaved trees were cut down, replaced with broad-leaved species and managed as coppice from then on.

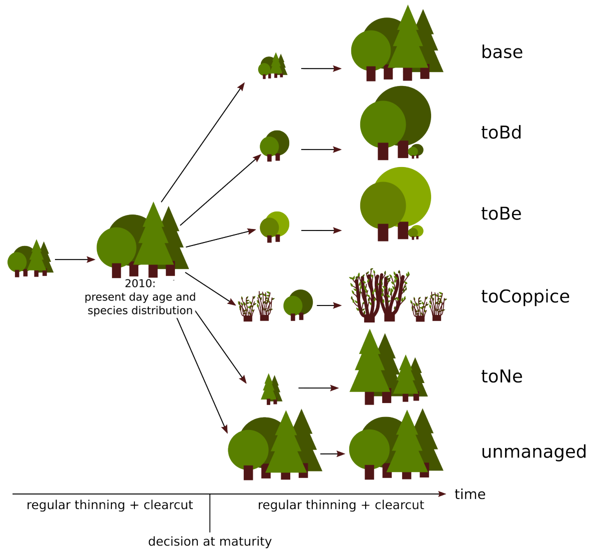

Figure 1: The six simplified management options, described in section 2.1.4. A management decision was made for each stand after 2010 as soon as it reached maturity (i.e. a target density). The conversion was implemented by planting the most common species of each forest type for that grid cell.

2.1.5 Ecosystem services and indicators

We considered the ESs climate change mitigation, provisioning of habitat for biodiversity, local climate regulation, water availability, and wood production. They were quantified as in Gregor et al. (2022) and are briefly outlined in Table 1. Adaptation was covered implicitly by only including forest management options in the portfolios that ensured tree cover in 2100–2130 under all RCPs (see 2.2.1).

Table 1: The ecosystem service indicators used in this study.

Variable name	Ecosystem service indicator	Explanation
Harvests	Total harvests	Total wood provision (including firewood, pulp, etc.)
HLP	Harvests for long-lived products	Wood provision for furniture, construction, etc.
Mitigation	Carbon sink plus material and energy substitution effects	Total carbon in vegetation, soil, litter, and products, plus avoided emissions from substitution with wood products
z_0	Surface roughness	Indicator for atmospheric conductance, influencing heat fluxes. Higher roughness results in higher fluxes.
ET	Total evapotranspiration	Indicator for latent heat fluxes. More ET means more local cooling
Ψ_{soil}	Soil water potential	Yearly minimum of monthly values, indicator of water availability and drought stress
Bio	RCP-normalized mean combining the amount of coarse woody debris, Shannon entropy of 5 cm DBH* classes and number of trees with DBH>50 cm	Coarse woody debris, large trees, and an abundance of various tree sizes provide high numbers of habitats and resources (Cordonnier et al., 2014)

*) DBH: diameter at breast height

2.2 Optimization

2.2.1 Optimization for climate-smart forestry under uncertainty

We used robust multi-criteria optimization to develop forest management portfolios that provide all ESs in an optimally balanced way across a range of climate scenarios, leading to one portfolio per grid cell, viable for all RCPs (Gregor et al., 2022). This approach deals with the so-called “deep uncertainty” of climate change which avoids assigning probabilities to specific scenarios because it suggests a false sense of certainty (Lawrence et al., 2020). The inclusion of a wide range of emission scenarios is also endorsed by the IPCC (IPCC, 2023b). For each grid cell independently, ESs were measured via their respective indicators (esi) and for each RCP normalized across management options. Thus for each indicator, grid cell, and RCP, the best possible future value across all management options was 1 and the worst was 0. This normalization is essential to enable comparisons of indicators with varying units. The following linear program (*“ORIGINAL”*) was used to derive an optimally balanced provision of ESs. It incorporated a trade-off parameter $\lambda \in [0,1]$ to combine the optimization of the worst case, and the average, ES performance. Fig. S1 shows a schematic display of the methodology and Fig. S6 a visualization of an optimized solution for a grid cell.

For the optimization of a grid cell, we define a portfolio vector $\omega \in [0,1]^m$ that assigns a fraction of the grid cell to any of the $m = 6$ management options. We define the performance of a portfolio ω by considering the ES performances across all climate scenarios ($|ESI|$ indicates the number of ES indicators):

$$\text{performance}(\omega) := (1-\lambda) \min_{esi,rcp} \sum_s \omega_s q(esi,s,rcp) + \lambda \sum_{esi,rcp} \frac{1}{|ESI||RCP|} \sum_s \omega_s q(esi,s,rcp) \tag{1}$$

Then, for each grid cell, we find the best ω by solving this linear program that optimizes the performance:

$$\max_{\omega} \quad \text{performance}(\omega) \tag{2}$$

$$\text{subject to} \quad \sum_{s\in S} \omega_s = 1 \tag{3}$$

$$\omega_s \geq 0 \quad \forall\, s \in S \tag{4}$$

$$\text{fpc}(2100, s, rcp) \geq \min(0.1, \text{fpc}(2010)) \tag{5}$$

$$\begin{aligned}
\text{where} \quad & S = \{\text{base, toBd, toBe, toCoppice, toNe, unmanaged}\} \\
\omega_s &: \text{Share of management type } s \text{ in the optimized portfolio} \\
\text{fpc}(year, s, rcp) &: \text{Foliar projective cover of the grid cell under management option } s \text{ in RCP } rcp \text{ in year } year \\
q(esi, s, rcp) &: \text{Per grid cell normalized quality of } esi \text{ for management option } s \text{ in } rcp \\
\sum_{s\in S} \omega_s q(esi, s, rcp) &: \text{Quality of } esi \text{ for the whole grid cell for a portfolio } \omega
\end{aligned}$$

2.2.2 Integrating the independent optimizations into one optimization to enable Europe-wide constraints

To allow for Europe-wide constraints and compensation between grid cells, the previously independent grid cells were integrated into one pan-European optimization. Still, the methodology resulted in one portfolio per grid cell, viable for all RCPs. The normalization was still conducted per grid cell. Fig. 2 visualizes the methodology. We implemented the compensation between grid cells by maximizing the sum of grid cell performances (*"SUM"*). We restricted the study to equally weighted ESs and $\lambda = 0.2$ as a reasonable balance between maximizing the worst-case outcome and allowing some degree of compensation among ESs (Diaz-Balteiro et al., 2018). The optimization looks similar as *ORIGINAL* (section 2.2.1), only that each variable received a grid cell index as well (e.g., $\omega_s^{(gc)}$):

$$\max_{\omega} \sum_{gc} \text{performance}(\omega^{(gc)}, gc) \tag{6}$$

$$\text{subject to} \quad \sum_{s\in S} \omega_s^{(gc)} = 1 \ \ \forall \text{ grid cells } gc \tag{7}$$

$$\omega_s^{(gc)} \geq 0 \quad \forall\, s \in S, \forall \text{ grid cells } gc \tag{8}$$

$$\text{fpc}^{(gc)}(2100, s, rcp) \geq \min(0.1, \text{fpc}^{(gc)}(2010)) \ \ \forall \text{ grid cells } gc \tag{9}$$

The performance of each grid cell was calculated similar to eq. 1, now also including grid cell indices:

$$\text{performance}(\omega^{(gc)}, gc) := (1-\lambda) \min_{esi,rcp} \sum_s \omega_s^{(gc)} q^{(gc)}(esi, s, rcp) + \lambda \sum_{esi,rcp} \frac{1}{|ESI||RCP|} \sum_s \omega_s^{(gc)} q^{(gc)}(esi, s, rcp) \tag{10}$$

As long as no Europe-wide constraints are added, this optimization is equivalent to *ORIGINAL* where each grid cell was optimized independently. For an additional assessment, we maximized the worst case grid cell performance (*"MAXIMIN"*), where the burden was shared in a more balanced way (see section S1.1).

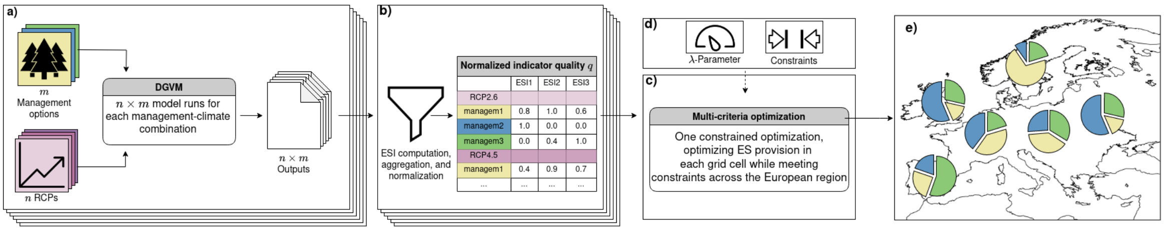

Figure 2: Visualization of the methodology, which computes one collection of portfolios for the entire modeled area. **a)** For each grid cell, the m management options are simulated for the n RCPs, resulting in $n \times m$ model simulations. **b)** ESIs are derived from model outputs, aggregated to the 2100-2130 mean, and normalized. Thus for each grid cell, there was one table containing the normalized values for all RCPs and management options. **c)** One optimization, configured with Europe-wide constraints (**d**) computes **e)** one set of optimized portfolios. Within grid cells, this ensures an optimally balanced provision of all ESIs across all RCPs and that the constraints are met, either on a per-grid-cell basis, or on a Europe-wide level, depending on the nature of the constraint, see section 2.3. **d)** The parameter $\lambda \in [0, 1]$ specifies the focus on the balanced provision of the ESI. A low λ focuses more on a balanced provision of ESIs while a high λ improves more the average ESI performance (see section 2.2.1).

2.3 Adding Europe-wide hard constraints to the optimization

To account for the protection goals and harvest demands, we included hard constraints into the optimization. The term *hard* means that they had to be met under every RCP. They did not have to be met within every grid cell, but across the entire modeled area (encompassing the whole of Europe and not just the EU).

2.3.1 Determining the required fraction of strictly protected forests currently available for wood supply

We deemed 66% of the European land area suitable for strict protection (forests, wetlands, shrublands, and grasslands). The remainder consists of artificial and barren land, water bodies, and cropland (Eurostat, 2023b). According to the biodiversity strategy, 10% of Europe's land area should be strictly protected, including all remaining primary and old-growth forests (European Commission, 2020). The identification and mapping of these forests is part of the EU strategy and relies on indicators such as deadwood, snags, and large trees, which vary depending on the forest type and region (European Commision, 2023). Here, we only optimized the area of forests available for wood supply. We assume that existing old-growth forests do not fall in this category and therefore lie outside of this considered area. Since old-growth forests cover about 1% of the land area (European Commission, 2021), they will contribute one percentage point to the 10% strict protection constraint. Consequently, assuming an equitable distribution of the other 9% among the remaining 65% of suitable land would require 13.8% of forests available for wood supply to be strictly protected in the future.

Note that the Forest Strategy also requires 30% of protection of the land surface and the promotion of "closer-to-nature forest management" (Larsen et al., 2022). As of 2024, 26% of European forests are under some form of protection (European Commission, 2021), undergoing various forms and degrees of management, or non-management (Verkerk, Zanchi, et al., 2014). Achieving the 30% goal requires the allocation of additional forest areas with different degrees of protection and management, and the definition of regionally applicable implementations of closer-to-nature management. While these are important aspects, they were out of scope for this study.

2.3.2 Formulation of the constraints

In addition to the unconstrained optimization (*"default-opt"*), we explored the impact of five Europe-wide constraints to be met by 2100–2130 under every RCP:

1. *min-harv*: Total harvests on the continent must remain at or above present day values (eq. 11).
2. *min-harv-cell*: In every grid cell, harvests must remain at or above present day values (eq. 12).
3. *min-hlp*: Harvests for long-lived wood products must remain at or above present day values (eq. 13). This is relevant because the EU Forest Strategy promotes long-lived wood products and *min-harv* does not distinguish between wood usages (European Commission, 2021).
4. *all-constraints*: In addition to meeting constraints *min-harv* and *min-hlp*, 13.8% of the forest area available for wood supply must be left unmanaged (eqs. 11, 13, 14).
5. *all-constraints-protect-cell*: Like *all-constraints* but the unmanaged fraction needed to be met in every cell (eqs. 11, 13, 15).

$$\sum_{gc}\sum_{s}\text{harvest}(gc,s,rcp,2100)\cdot\omega_{\text{S}}^{(gc)} \geq \sum_{gc}\text{harvest}(gc,2010)\ \forall\ rcp \in \{\text{RCP2.6, RCP4.5, RCP6.0, RCP8.5}\} \tag{11}$$

$$\sum_{s}\text{harvest}(gc,s,rcp,2100)\cdot\omega_{\text{S}}^{(gc)} \geq \text{harvest}(gc,2010)\ \forall\ rcp, gc \tag{12}$$

$$\sum_{gc}\sum_{s}\text{hlp}(gc,s,rcp,2100)\cdot\omega_{\text{S}}^{(gc)} \geq \sum_{gc}\text{hlp}(gc,2010) \tag{13}$$

$$\sum_{gc}\text{area}(gc)\cdot\omega_{\text{unmanaged}}^{(gc)} \geq 0.138\sum_{gc}\text{area}(gc) \tag{14}$$

$$\omega_{\text{unmanaged}}^{(gc)} \geq 0.138\ \forall\ gc \tag{15}$$

It is important to note that the decision to strictly protect forests in a grid cell in our simulations is made, for reasons of simplicity, at the time of the final harvest. These situations often occurred much later than 2030, the year in which the EU strategies would already demand a decision on which forests should be strictly protected.

2.3.3 Implementation

The optimization was implemented in Python using *scipy* (Virtanen et al., 2020). We employed the *highs-ipm* solver (Huangfu & Hall, 2018) that was capable of solving the large optimization problem within reasonable time and memory consumption which was not the case for other solvers.

3 Results

3.1 Model performance

The model represented the present-day situation in Europe adequately. Key vegetation variables, including gross and net primary productivity, vegetation carbon content, tree cover, evapotranspiration, and runoff aligned with literature estimates (Table S1). According to the forest age data, we identified 72% of forests as managed for timber, aligning with recent estimates that 75% of European forests are available for wood supply, with high agreement at country-level (Fig. S3, Forest Europe (2020)). Simulated total forest vegetation carbon was 13.7 GtC for the year 2010. This figure exceeds older estimates (11.6–13 GtC, Forest Europe, 2015; Liu et al., 2015; Pan et al., 2011) but remains below a recent estimate of 16.2 GtC (Fig. S7, Santoro et al. (2021)). Roundwood harvests were simulated as 572 million m^3/yr on average for the period 2000–2010, comparable to observations (542 and 582 million m^3/yr, Forest Europe (2015, 2020)). They also aligned on a country-level for multiple periods (Fig. S4).

3.2 Results of the optimization

3.2.1 Optimization without constraints

The unconstrained optimization *default-opt* led to diverse portfolios containing a shift towards more broad-leaved species from 39% to 56% and a transition to 26% unmanaged forests, far more than what is aimed for by the EU strategies (Fig. 3). The proposed unmanaged forests were relatively evenly distributed throughout the continent. The portfolios led to a balanced provision of all ecosystem services (ESs) across all RCPs (Figs. 7a + 8b). However, future (2100–2110) harvests dropped 23% below current values.

3.2.2 Optimizations with constraints on harvest levels

The optimization *min-harv* successfully identified management portfolios that met the harvest constraint across all RCPs. This stands in contrast to *min-harv-cell* where the constraint had to be met in every grid cell and no feasible solution was found. The compensation among grid cells in our study thus appears to be pivotal to achieve such harvest levels in the future. The proportion of unmanaged forests was reduced in the optimized portfolios, declining from 26% in *default-opt* to 5% in *min-harv* (Fig. 3a). In *min-harv*, a total of 58% of grid cells exhibited no unmanaged forests, whereas in *default-opt*, this figure was merely 8% (Fig. 4). There was a smaller transition to broad-leaved forests in *min-harv* (Fig. 3), because needle-leaved forests enabled higher harvest volumes. Thus, they were needed to compensate for the other forest types in the portfolios.

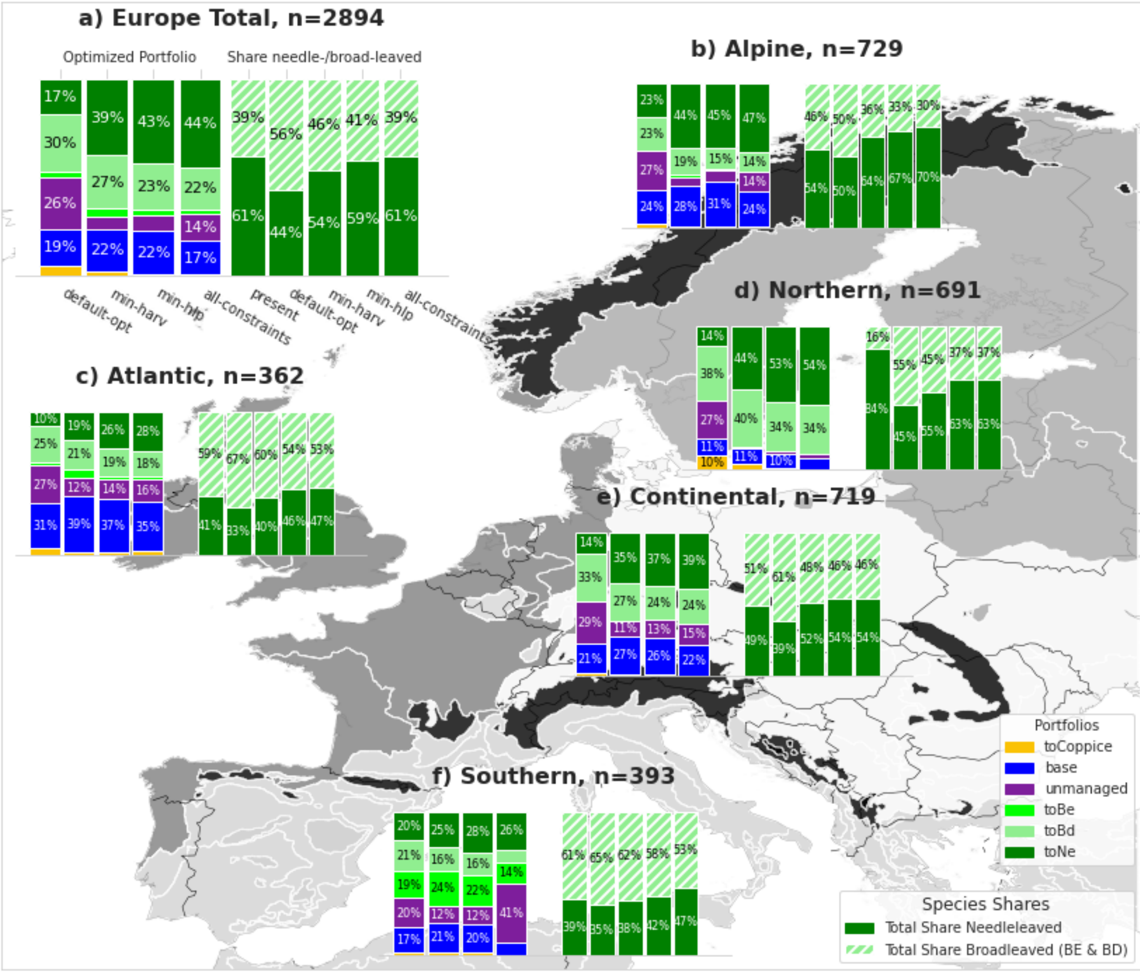

Figure 3: Portfolios of management options and species shares for optimizations with and without constraints for all of Europe (a) and different European regions (b-f). "Broad-leaved" contains broad-leaved evergreen and deciduous species. The management options are keeping the current species composition (*base*), converting to needle-leaved evergreen, broad-leaved evergreen, broad-leaved deciduous, and coppice forests (*toNe*, *toBd*, *toBe*, *toCoppice*), or leaving the forest *unmanaged*. See also Fig. 1. Note that the portfolios marginally differed from the results of Gregor et al. (2022) due to an improvement in the simulation of harvesting and the higher resolution. The number n refers to the modeled grid cells in the given region.

Coppice management practically vanished from Europe's forests in *min-harv*, compared to 5% in *default-opt*, and was replaced predominantly by needle-leaved forests for the same reason. This sustained importance of managed needle-leaved forests contrasts the strong shift towards broad-leaved species in *default-opt* and adaptation strategies for European forests.

The portfolios within grid cells were less diverse in *min-harv*, with 2.6 management options per portfolio on average, compared to 3.3 in *default-opt*. Especially in northern Europe, many portfolios consisted of only one management option (Fig. 5). The constraint for an increased provision of long-lived products (*min-hlp*) resulted in similar portfolios as *min-harv*, but with even higher proportions of needle-leaved forests (59%), also because of the higher suitability of wood from needle-leaved trees for long-lived products (Eurostat, 2023a).

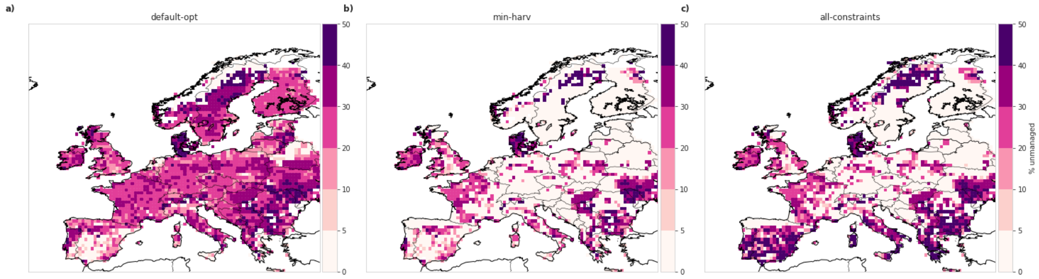

Figure 4: Share of unmanaged forests in **a)** the default optimization without any constraint, **b)** the optimization when imposing the *min-harv* constraint on harvest levels, and **c)** when imposing constraints on harvests, harvests for long-lived products, and unmanaged areas at the same time.

3.2.3 Combining constraints on harvests and strict protection

The *all-constraints* optimization successfully yielded portfolios with stable harvest levels and the minimal required level of unmanaged forests across all RCPs (Fig. 3a). However, unlike in *default-opt*, the unmanaged areas in *all-constraints* were unevenly distributed: 48% of grid cells, mainly in the north, lacked unmanaged forests. Meanwhile, southern portfolios contained 41% unmanaged forests (Figs. 3 and 4), corresponding to the most unproductive regions in terms of wood production according to the model (Fig. S8b). The share of needle-leaved forests was 61% and thus higher compared to the other optimizations (Fig. 3), due to higher volumes of timber from needle-leaved forests and the higher suitability for long-lived products, both contributing to meeting the *min-harv* and *min-hlp* constraints (Eurostat, 2023a). Note that in the *all-constraints* optimization, the needle-leaved forests were mainly managed, whereas in *default-opt* a large fraction of the needle-leaved forests in the portfolios were also unmanaged (Fig. 3a).

Enforcing strict protection within every grid cell (*all-constraints-protect-cell*) made the optimization infeasible. No portfolio allocation could meet the Europe-wide harvest targets while simultaneously achieving the strict protection targets in every grid cell under every emission scenario. Providing 13.8% strict protection in every grid cell required total harvests to decrease by at least 5%. It also forced all regions to focus on managed needle-leaved forests (overall 74% of the area, Fig. S17) to compensate for the lower area of forests available for wood supply. This poses tremendous risks because of the low diversification of strategies, further exacerbated by the higher susceptibility of conifers and monocultures to various disturbance agents (Hlásny et al., 2021; Pardos et al., 2021; Schelhaas et al., 2010).

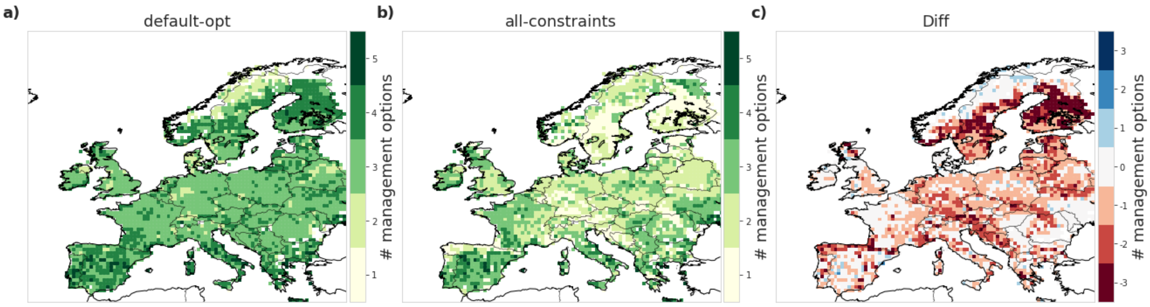

Figure 5: Number of management options in the portfolios without constraints (**a**), and for *all-constraints* (**b**). **c)** Difference between constrained and unconstrained optimization (*all-constraints* - *default-opt*). Including the constraints led to less diverse portfolios, including portfolios consisting of only one management option in many grid cells, particularly in the Northern region.

3.3 Impacts on ecosystem service provision and burden sharing

The constraints resulted in a much less balanced provision of ESs (Fig. 7). The productive regions in Fennoscandia, central, and eastern Europe needed to focus on supplying timber to others (Fig. 6a+d). All ESs were impacted by the constraints in all RCPs. For example, the availability of coarse woody debris (one of three indicators used for biodiversity habitat provision) was much lower in those regions compared to the unconstrained optimization (Fig. 6b). This highlights a potential threat for species that depend on this type of habitat.

The total carbon pool decreased virtually everywhere compared to the unconstrained optimization (vegetation + soil + deadwood, Fig. 6c). The carbon pool also showed strong reductions compared to present-day for the regions that had to focus on timber provision (Fig. 6f). This conflicts with the EU LULUCF (Land use, land use change, and forestry) regulation demanding increases in forest carbon uptake in all member states (European Union, 2018). It was mainly driven by higher release of carbon from soils and litter due to climate change (Figs. S11-S13) which in *default-opt* could be compensated for by the increasing vegetation and litter carbon stocks from the large areas of unmanaged forests.

This underscores that the burden of the constraints was not shared equally. In the grid cells that were most affected by the constraints, ESs were no longer provided in a balanced manner. These forests lost their multi-functionality and diversified portfolios, thereby hindering important risk diversification (Fig. 8). To distribute the burden of the constraints more fairly, we applied the MAXIMIN instead of the SUM-method, maximizing the worst-case ES provision in each grid cell (section 2.2.2). However, both optimizations yielded virtually the same results, showing that the constraints significantly curtailed the range of viable management portfolios, leaving little room for enhancing the provision of other ESs (Fig. S16).

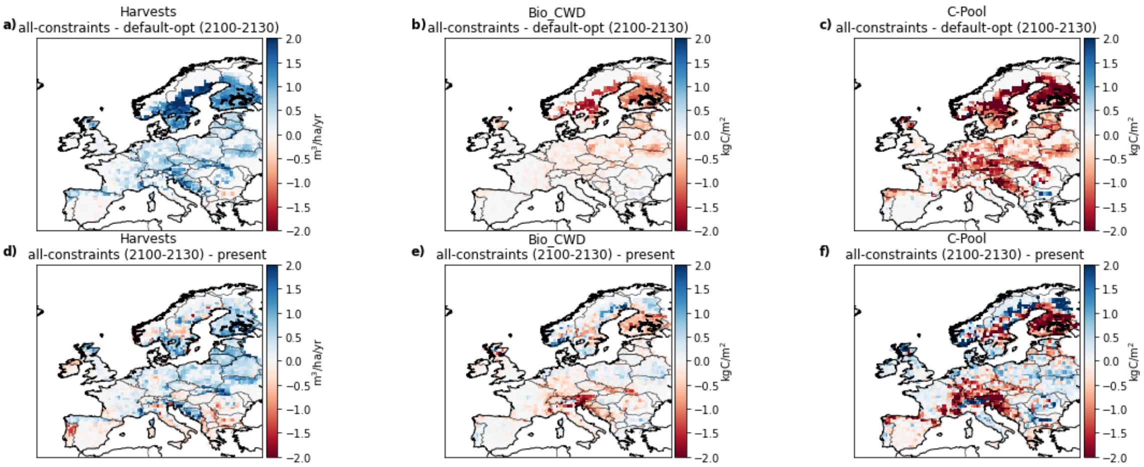

Figure 6: Comparison of ecosystem service provision between constrained optimization, unconstrained optimization, and present day. Modeled harvest provision (in m^3 ha^{-1} yr^{-1} dry biomass) in the future (2100–2130) for RCP4.5 for the *min-harv* optimization compared to the unconstrained *default-opt* optimization (**a**) and compared to present-day values (**d**). The same differences are shown for coarse woody debris (**b** and **e**) and the forest carbon pool (vegetation + litter + soil, **c** and **f**), in kgC/m^2. Similar results were obtained for the other RCPs (Figs. S14-S15).

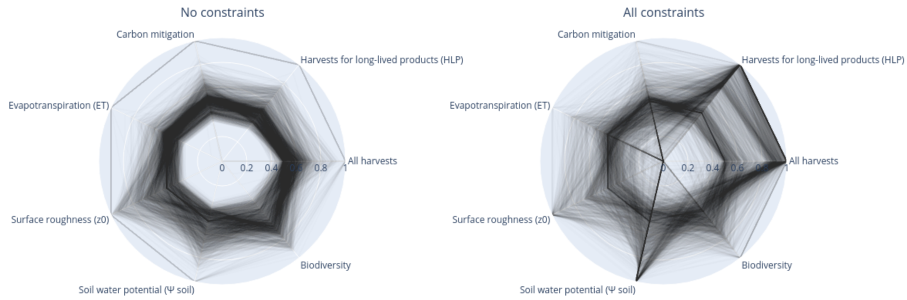

Figure 7: Provision of ecosystem services across all grid cells. Note that the ecosystem service provision is much more balanced in the unconstrained *default-opt* optimization, i.e. almost all grid cells provide all ecosystem services in a balanced way (**left**). When imposing *all-constraints*, the provision is much more imbalanced (**right**). Various cells are required to utilize the maximal possible harvests, affecting also other ecosystem services, often negatively.

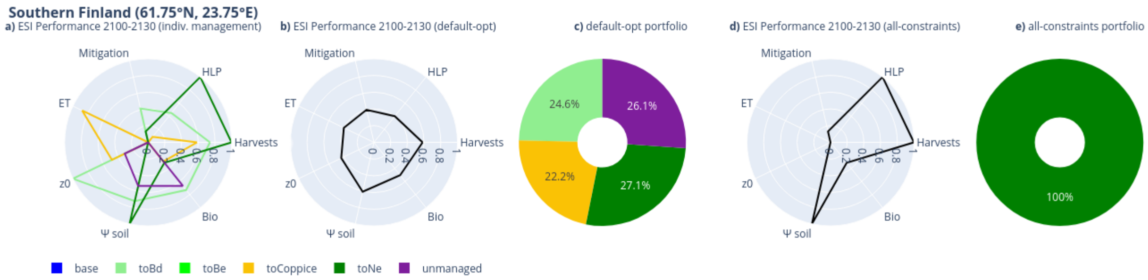

Figure 8: Example for a concrete portfolio computed by the methodology for a grid cell in southern Finland. **a**) shows the ecosystem service provision in the worst case across all RCPs for each management option as measured by the normalized ecosystem service indicators (ESIs). **b**) shows the worst case ecosystem provision of the optimized portfolio without constraints (*default-opt*) and **c**) shows the portfolio shares for *default-opt*. **d**) and **e**) are like **b**) and **c**), respectively, but for *all-constraints*. It is obvious that the ecosystem service provision in *default-opt* is more balanced than in *all-constraints* and that there is no risk diversification in *all-constraints*, as opposed to *default-opt*.

4 Discussion

Our methodology derives multi-functional forestry strategies in Europe under emission scenario uncertainty, providing suggestions for management portfolios that are viable for RCPs 2.6, 4.5, 6.0, and 8.5 simultaneously. While future work will also need to consider uncertainty related to the choice of climate and vegetation model, our results already suggest that constraints on stable harvest levels and protection goals inspired by EU strategies heavily restrict the possibilities to provide other ecosystem services (ESs) under climate change. Furthermore, achieving these targets conflicted with the goal of multi-functionality and with carbon sink targets (European Union, 2018), complementing findings of previous studies (e.g., Blattert et al., 2023). It is noteworthy that while the EU strategies outline plans for 2030, we examined potential long-term consequences in 2100–2130.

4.1 Reconciling demands on forest protection, wood production, and mitigation

The unconstrained *default-opt* optimization indicated that leaving 26% of currently managed forests untouched benefits multiple ESs across all RCPs (Fig. 3). This exceeds the EU Biodiversity Strategy requirements but implies a drastic reduction in harvests, reducing economic activity and the important role of wood products in climate change mitigation (Grassi et al., 2021; Gregor et al., 2024). To maintain current Europe-wide harvest levels we found that inter-regional cooperation is critical, because the harvest constraint could only be met when allowing such cooperation (*min-harv*) and not when it was imposed on every grid cell independently (*min-harv-cell*). The constraint decreased the proposed shares of unmanaged forests to 5%, conflicting with strict protection goals (Fig. 3). This was reconciled by the constraint on strict protection (*all-constraints*). However, the burden was not shared equally among regions. Some regions, particularly in the north, had to focus almost exclusively on wood production to compensate for decreased harvests due to strict protection elsewhere. This bears risks for nature protection in those regions (Fig. 4b).

Due to climate change, the forest carbon pool declined in many regions (Fig. 6f), driven by increased decomposition of litter and soil, especially under higher RCPs. In *default-opt*, this decrease was offset by higher shares of unmanaged forests, which increased vegetation and deadwood pools. In *all-constraints*, however, the carbon pools of the numerous, mainly northern, regions declined compared to present-day values (Fig. S10), conflicting with the EU LULUCF regulation that aims to increase forest carbon uptake in all member states (Fig. 6; European Union, 2018). Maintaining a European forest sink could be imposed as an additional hard constraint in the optimization, but this would further limit management options. Since from an atmospheric perspective, it is not relevant where the carbon is taken up, LULUCF goals could theoretically be reformulated to allow compensation between states. Although this could facilitate collaboration to achieve the desired atmospheric CO_2 reductions while optimizing other ESs, this would introduce additional problems of responsibility and accountability.

4.2 Effect on other ecosystem services, multi-functionality, and the distribution of managed and protected areas

Applying *all-constraints* strongly reduced the diversity of the portfolios compared to *default-opt* (Fig. 5). Many portfolios, particularly in Fennoscandia, contained only one or two management options, because there were only few feasible solutions to the constrained optimization. This made it rarely possible to include other management options for risk diversification and for the benefit of other ESs. At the grid cell level, a balanced provision of ESs was no longer guaranteed (Figs. 7 and 8), conflicting with the aim of the EU strategies to foster multi-functionality. For instance, the harvest constraints significantly reduced the amounts of deadwood and large trees in the future, especially in southern Fennoscandia (Fig. S9) where timber production was prioritized to meet the Europe-wide constraint. This poses a significant threat to biodiversity as many species require these habitats (e.g., Berg et al., 1995). This issue was exacerbated by faster decay of deadwood under higher RCPs (Figs. S11+S12). The performance of other ESs also declined, showing that focusing on wood production will undermine other ESs and vice versa, illustrating a clear trade-off.

The regional imbalance of unmanaged sites, with many in the southern regions and few in the rest (Fig. 4c), contradicts the goal to protect various ecosystems throughout the continent (European Commission, 2020). The strictly protected areas in *all-constraints* were mainly allocated to the least productive regions. This would likely bias the assemblage of species benefiting from protection (see, e.g., Hämäläinen et al., 2018). To address this, we also constrained the optimization to uniformly distribute the strictly protected areas, but the *all-constraints-protect-cell* optimization was mathematically infeasible. It could be resolved with an at least 5% reduction in harvests across Europe but portfolios then strongly focused on needle-leaved forests (74% of all forests). Although a 5% reduction in harvests might be acceptable given the significant improvement in nature protection in this scenario, promoting managed needle-leaved forests contradicts current scientific evidence and policies targeted at improving forest resilience through mixed forests including broad-leaved species, as discussed below.

A land-sparing approach, as suggested by the optimization, can have benefits because assigning focus regions for certain targets can help using forests optimally by leveraging regional advantages (Gutsch et al., 2018; Lessa Derci Augustynczik & Yousefpour, 2021). This does not inherently conflict with multi-functionality, as for instance strictly protected areas can still provide multiple ESs apart from biodiversity provision, such as water regulation, or local climate regulation. Our results, however, suggest such a strong segregation that hinders promoting multi-functional forestry, because large regions had to focus on timber provision at the expense of other ESs. Even changing the optimization methodology – affecting how the burden of the constraints could be shared across regions – had virtually no effect on the portfolios (Fig. S16). This further underscores that the constraints heavily limited the forestry options in Europe and that intricate trade-offs need to be made.

Harnessing synergies between different aspects in the same region through land-sharing might be necessary. In that regard, some biodiversity habitats and other ESs are compatible with some wood production (e.g., as part of close-to-nature forestry), for instance, by improving landscape-scale heterogeneity, retaining habitat trees and deadwood, and fostering species and structural diversity (e.g., Biber et al., 2020; Larsen et al., 2022; Mäkelä et al., 2023; Schall et al., 2018). Achieving such synergies would help meet wood demands while providing numerous ESs. This approach could make regions that we deemed crucial for timber provision more multi-functional and, with proper measures, also contribute to the 30% protection target.

The "triad" approach aims to combine and enhance land sharing and sparing, by combining intensive and extensive management with strict reserves, based on biodiversity-yield assessments (Betts et al., 2021). Nonetheless, while these are desirable approaches to optimally use forest land, they cannot fully resolve the issue of excessive demands imposed on forests that we identified in our simulations. Therefore, additional measures are necessary to alleviate pressure on forests, as discussed below.

Besides protection and multi-functionality, the EU also plans to promote broad-leaved species for their greater resilience (European Commission, 2021). This transition is encouraged by the scientific literature (Astrup et al., 2018; Felton et al., 2010; Hlásny et al., 2021; Pardos et al., 2021; Schelhaas et al., 2010; Schwaab et al., 2020). It was also reflected in *default-opt*, which considered multiple ESs and a higher vulnerability of needle-leaved forests to disturbances (Fig. 3a). However, the constraints prevented this forest conversion and maintained the dominance of conifers due to their higher wood volumes and suitability for long-lived products (Eurostat, 2020). This would hinder adaptation to climate change, especially in regions where needle-leaved species are projected to suffer more.

An important caveat is that, while we did account for increases in disturbance rates and higher baseline rates for needle-leaved forests, these rates did not depend on the specific species or forest structure. A more realistic representation of disturbances – especially for spruce monocultures – would likely decrease the share of needle-leaved forests in the optimized portfolios, making the constraint on harvests for long-lived products harder to meet.

4.3 Ways forward

Although further studies should validate our results with model ensembles, our study already highlights the significant challenges of reconciling current forest demands without additional interventions. There are numerous options to address the conflicts that should be considered by future studies and policies: One potential avenue to alleviate the impact of the constraints is increasing the proportion of wood used for long-lived products. This involves promoting innovative products made from lower quality wood and smaller diameter trees (e.g., Ramage et al., 2017). The otherwise beneficial shift towards more broad-leaved trees also decreases the provision of long-lived wood products, affecting the economy and mitigation. This may be addressed by promoting new products derived from broad-leaved species (e.g., Hassan & Eisele, 2015). Also the increased material wood usage of needle-leaved trees would enable an increased share of broad-leaved species.

However, these measures conflict with Europe's current energy mix. Woody bioenergy plays a crucial role in renewable energy supply, with a significant fraction sourced from primary wood (Camia et al., 2021; European Commission, 2021). About one-fourth of all roundwood harvests are currently used for fuel wood, providing only 6% of the gross final energy consumption (Eurostat, 2022; Scarlat et al., 2019). While increased rates of recycling and end-of-life energy-recovery would help, these rates are already high in many EU countries (Eurostat, 2023c). Moreover, renewables like solar and wind offer power densities that are orders of magnitude higher than that of bioenergy (Smil, 2015), making their promotion paramount to meet future energy demands while achieving climate and biodiversity goals for forests.

Projected increases in wood demand are also driven by packaging, single-use products, expansion of living areas, and short lifespans of wood products due to aesthetic reasons (Bierwirth & Thomas, 2015; FAO, 2022b; Hill et al., 2022). Here, stable harvest levels already required intricate trade-offs, underscoring the need to address these increasing demands. Our study aligns with broader research highlighting that true sustainability in terms of resource usage, biodiversity, and ESs necessitates a reduction in demands (e.g., Hickel & Kallis, 2020; Richardson et al., 2023). It is also crucial that forest-related actions in Europe avoid an offshoring of impacts (Berlik et al., 2002; Mayer et al., 2005). While the strategies explicitly forbid activities leading to deforestation in other regions of the globe (European Commission, 2020), substantial risks remain (Cerullo et al., 2023; Rosa et al., 2023). Consequently, concerted efforts are required to balance resource demand and supply within Europe, or to establish frameworks which holistically account for resource footprints and prevent externalizing impacts.

The fact that "only" 73% of the net annual increment is harvested in Europe's wood-supplying forests suggests potential for increased harvesting (Forest Europe, 2020). Studies have already suggested a necessary intensification of harvests outside of strictly protected areas to compensate for reduced wood supply areas due to protection goals (Pikkarainen et al., 2024). However, this could weaken the ecological benefits of the strategies (Räty et al., 2023). Moreover, Europe's felling rates (harvests per forest area) are already high compared to global rates (Fig. S18) and increasing them has been linked to adverse affects on biodiversity, carbon sequestration, and recreation (Mäkelä et al., 2023; Schulte et al., 2022; Seppälä et al., 2019; Skytt et al., 2021; Soimakallio et al., 2021; Verkerk, Mavsar, et al., 2014). Critically, higher felling rates would reduce the buffer between harvests and net annual increment that keeps forests a carbon sink. While increased harvests could offer mitigation benefits through substitution effects, these benefits are likely short-lived (Brunet-Navarro et al., 2021; Gregor et al., 2024; Harmon, 2019).

Furthermore, the area available for wood supply (currently 75%) could be increased, but this would conflict with conservation goals. Additionally, many unmanaged forests are in unproductive or inaccessible areas, limiting their wood supply potential (Verkerk, Mavsar, et al., 2014). Supporting the ongoing reforestation trend in Europe, endorsed by the EU's plan to plant 3 billion trees by 2030, could alleviate pressure on forests (Forest Europe, 2020). However, it will take decades for these trees to provide timber. Furthermore, it is crucial that biodiversity considerations guide such plantings, e.g., in terms of species selection.

4.3.1 Uncertainty assessment

Our methodology derives forest management strategies under deep uncertainty, providing solutions that are viable under all considered climate scenarios. Further studies should use an ensemble of vegetation models that might consider different processes in different levels of detail to address uncertainty in the projections better. Additionally, studies with LPJ-GUESS for instance emphasize the importance of using also an ensemble of climate projections from general circulation models as forcing data due to significant variation among them for the same RCP (Ahlström et al., 2012). Finally, model parameter uncertainty was not considered here, though for LPJ-GUESS a smaller impact compared to the uncertainty from environmental data has been suggested (Oberpriller et al., 2022). The advantage of the robust optimization concept is that it can be fed not only with simulations of multiple RCPs, but also with simulations from multiple models and forcings. The outcome would again be one set of portfolios, providing the best options across all RCPs, forcings, and models. Also diversity in the aims of decision makers could be included (e.g., Knoke et al., 2023). This could be done by including multiple sets of preferences for ecosystems, for instance, with higher importance of water regulation on arid regions. Including all aspects, however, would pose significant computational challenges.

4.3.2 Regional strategy development

We examined how legislative constraints impact the development of future forest management strategies at a coarse, Europe-wide scale. This work establishes a foundation for specific applications: Once general strategies, like broadly allocating protected areas among member states, are outlined, our methodology can be applied at a finer scale. At this level, detailed representation of terrain, soil, forest types, and management practices become crucial (Levin, 1992; Turner et al., 1996; Turner et al., 1989). Thus, in a next step, it may be beneficial to re-integrate fine-scale results into the broader framework, to address scaling issues (Seidl et al., 2013).

Our optimization can facilitate strategy development for specific regions through more detailed forest simulations. This should include more detailed changes in management regimes (e.g., targeting specific age classes), wood usage patterns, and species selection. Also age and species composition, landscape heterogeneity and additional biodiversity indicators (e.g., Cordonnier et al., 2014; Müller & Bütler, 2010) should be assessed for estimating conservation values (Neugarten et al., 2024).

Furthermore, regional objectives and constraints can be included, such as connectivity of protected areas as endorsed by the EU strategies, and minimum reserve sizes to capture natural disturbance regimes ("minimum dynamic area", Pickett and Thompson (1978)). Additional constraints could include targets for deadwood availability and carbon sinks and constraints for the 30% (non-strictly) protected areas.

From a computational perspective, we propose a hierarchical approach. Here, we simulated and optimized 2894 grid cells spanning the entire continent. These results can inform assessments on a member-state level. Taking France as the largest EU country as an example, applying our methodology on a $10\,\text{km}^2$ scale is computationally feasible (i.e., 5400 grid cells). This enables strategy development for individual countries independently which can then guide regional optimizations base on high-resolution data on forest structure (e.g., from NFI data), existing old-growth forests, ownership structure, and accessibility, to formulate practical strategies.

5 Conclusion

In this study, we combined forest management simulations with robust multi-criteria optimization to develop strategies for multi-functional forests in Europe under climate change. The derived management portfolios are viable for a range of emission scenarios simultaneously, and they reconcile demands for wood production and EU targets for biodiversity protection, climate change mitigation, and ecosystem service provision. Our approach used simplified management scenarios, moderate constraints, extended time scales, and ignored potential uncertainty from multiple models. Nonetheless, our findings already highlight significant conflicts between the various demands placed on European forests, requiring additional measures to alleviate the pressure on forests. They also emphasize the need for coordinated efforts to address the various objectives outlined in EU strategies. Moreover, our results offer insights that can inform the development of forest management strategies at regional scale. By incorporating more detailed forest management and wood usage scenarios, along with detailed constraints, our methodology can help investigate how innovative practices may help harmonize or alleviate the conflicting demands on European forests. This approach offers a tool for the necessary integrated view of conflicting climate, biodiversity, and bioeconomy demands.

Acknowledgments

We acknowledge funding from the Bavarian State Ministry of Science and the Arts in the context of the Bavarian Climate Research Network (bayklif) through its BLIZ project (Grant No. 7831-26625-2017, www.bayklif-bliz.de). We acknowledge funding from the Forest Value project FORECO and the European Forest Institute (EFI) Networking Fund FORMASAM. KG further acknowledges funding from the VELUX Stifrung (www.veluxstiftung.ch) and the STEPSEC project of the CDRTerra program of the German Federal Ministry of Education and Research (Grant No. 01LS2102C). The authors gratefully acknowledge the computational and data resources provided by the Leibniz Supercomputing Centre (www.lrz.de). We acknowledge using ChatGPT3.5 to help us with grammar and language editing. Precisely, we asked it to rephrase parts of our original content for clarity and style.

Author Contributions

KG, AR, and TN conceived of the study through discussions within the FORECO network. KG implemented changes into the model, developed the constrained optimization framework, ran the simulations and optimizations and drafted the first version of the manuscript. All authors significantly contributed to the interpretations of the results and the manuscript.

References

Ahlström, A., Schurgers, G., Arneth, A., & Smith, B. (2012). Robustness and uncertainty in terrestrial ecosystem carbon response to CMIP5 climate change projections. *Environmental Research Letters*, *7*(4), 044008. https://doi.org/10.1088/1748-9326/7/4/044008

Astrup, R., Bernier, P. Y., Genet, H., Lutz, D. A., & Bright, R. M. (2018). A sensible climate solution for the boreal forest. *Nature Climate Change*, *8*(1), 11–12. https://doi.org/10.1038/s41558-017-0043-3

Balmford, A. (2021). Concentrating vs. spreading our footprint: How to meet humanity's needs at least cost to nature. *Journal of Zoology*, *315*(2), 79–109. https://doi.org/10.1111/jzo.12920

Barbier, E. B., Burgess, J. C., & Dean, T. J. (2018). How to pay for saving biodiversity. *Science*, *360*(6388), 486–488. https://doi.org/10.1126/science.aar3454

Başkent, E. Z., & Kašpar, J. (2023). Exploring the effects of various rotation lengths on the ecosystem services within a multiple-use management framework. *Forest Ecology and Management*, *538*(April). https://doi.org/10.1016/j.foreco.2023.120974

Ben-Tal, A., & Nemirovski, A. (2002). Robust optimization - methodology and applications. *Mathematical Programming*, *92*(3), 453–480. https://doi.org/10.1007/s101070100286

Berg, Å., Ehnström, B., Gustafsson, L., Hallingbäck, T., Jonsell, M., & Weslien, J. (1995). Threat Levels and Threats to Red-Listed Species in Swedish Forests. *Conservation Biology*, *9*(6), 1629–1633. https://doi.org/10.1046/j.1523-1739.1995.09061629.x

Berlik, M. M., Kittredge, D. B., & Foster, D. R. (2002). The illusion of preservation: A global environmental argument for the local production of natural resources. *Journal of Biogeography*, *29*(10-11), 1557–1568. https://doi.org/10.1046/j.1365-2699.2002.00768.x

Betts, M. G., Phalan, B. T., Wolf, C., Baker, S. C., Messier, C., Puettmann, K. J., Green, R., Harris, S. H., Edwards, D. P., Lindenmayer, D. B., & Balmford, A. (2021). Producing wood at least cost to biodiversity:

Integrating Triad and sharing–sparing approaches to inform forest landscape management. *Biological Reviews*, *96*(4), 1301–1317. https://doi.org/10.1111/brv.12703

Biber, P., Felton, A., Nieuwenhuis, M., Lindbladh, M., Black, K., Bahýl', J., Bingöl, Ö., Borges, J. G., Botequim, B., Brukas, V., Bugalho, M. N., Corradini, G., Eriksson, L. O., Forsell, N., Hengeveld, G. M., Hoogstra-Klein, M. A., Kadıoğulları, A. I., Karahalil, U., Lodin, I., . . . Tuček, J. (2020). Forest Biodiversity, Carbon Sequestration, and Wood Production: Modeling Synergies and Trade-Offs for Ten Forest Landscapes Across Europe. *Frontiers in Ecology and Evolution*, *8*(October). https://doi.org/10.3389/fevo.2020.547696

Bierwirth, A., & Thomas, S. (2015). Almost best friends : Sufficiency and efficiency ; can sufficiency maximise efficiency gains in buildings? *ECEEE 2015 - European Council for an Energy Efficient Economy*, 71–82.

Blattert, C., Mönkkönen, M., Burgas, D., Di Fulvio, F., Toraño Caicoya, A., Vergarechea, M., Klein, J., Hartikainen, M., Antón-Fernández, C., Astrup, R., Emmerich, M., Forsell, N., Lukkarinen, J., Lundström, J., Pitzén, S., Poschenrieder, W., Primmer, E., Snäll, T., & Eyvindson, K. (2023). Climate targets in European timber-producing countries conflict with goals on forest ecosystem services and biodiversity. *Communications Earth & Environment*, *4*(1), 119. https://doi.org/10.1038/s43247-023-00771-z

Brunet-Navarro, P., Jochheim, H., Cardellini, G., Richter, K., & Muys, B. (2021). Climate mitigation by energy and material substitution of wood products has an expiry date. *Journal of Cleaner Production*, *303*, 127026. https://doi.org/10.1016/j.jclepro.2021.127026

Brus, D. J., Hengeveld, G. M., Walvoort, D. J., Goedhart, P. W., Heidema, A. H., Nabuurs, G. J., & Gunia, K. (2012). Statistical mapping of tree species over Europe. *European Journal of Forest Research*, *131*(1), 145–157. https://doi.org/10.1007/s10342-011-0513-5

Camia, A., Giuntoli, J., Jonsson, R., Robert, N., Cazzaniga, N., Jasinevičius, G., Avitabile, V., Grassi, G., Barredo, J., & Mubareka, S. (2021). *The use of woody biomass for energy production in the EU and impacts on forests The use of woody biomass for energy production in the EU, EUR 30548 EN*. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/831621

Cerullo, G., Barlow, J., Betts, M., Edwards, D., Eyres, A., França, F., Garrett, R., Swinfield, T., Tew, E., White, T., & Balmford, A. (2023). The global impact of EU forest protection policies. *Science*, *381*(6659), 740–740. https://doi.org/10.1126/science.adj0728

Chang, J., Ciais, P., Wang, X., Piao, S., Asrar, G., Betts, R., Chevallier, F., Dury, M., François, L., Frieler, K., Ros, A. G. C., Henrot, A.-J., Hickler, T., Ito, A., Morfopoulos, C., Munhoven, G., Nishina, K., Ostberg, S., Pan, S., . . . Zhao, F. (2017). Benchmarking carbon fluxes of the ISIMIP2a biome models. *Environmental Research Letters*, *12*(4), 045002. https://doi.org/10.1088/1748-9326/aa63fa

Cordonnier, T., Berger, F., Elkin, C., Lämås, T., Martinez, M., Bugmann, H., Courbaud, B., Gosselin, F., & Volk, G. (2014). ARANGE D2.2 - Models and linker functions (indicators) for ecosystem services. (289437).

Diaz-Balteiro, L., Alonso, R., Martínez-Jaúregui, M., & Pardos, M. (2017). Selecting the best forest management alternative by aggregating ecosystem services indicators over time: A case study in central Spain. *Ecological Indicators*, *72*, 322–329. https://doi.org/10.1016/j.ecolind.2016.06.025

Diaz-Balteiro, L., Belavenutti, P., Ezquerro, M., González-Pachón, J., Ribeiro Nobre, S., & Romero, C. (2018). Measuring the sustainability of a natural system by using multi-criteria distance function methods: Some critical issues. *Journal of Environmental Management*, *214*, 197–203. https://doi.org/10.1016/j.jenvman.2018.03.005

Dufresne, J.-L., Foujols, M.-A., Denvil, S., Caubel, A., Marti, O., Aumont, O., Balkanski, Y., Bekki, S., Bellenger, H., Benshila, R., Bony, S., Bopp, L., Braconnot, P., Brockmann, P., Cadule, P., Cheruy, F., Codron, F., Cozic, A., Cugnet, D., . . . Vuichard, N. (2013). Climate change projections using the IPSL-CM5 Earth System Model: From CMIP3 to CMIP5. *Climate Dynamics*, *40*(9-10), 2123–2165. https://doi.org/10.1007/s00382-012-1636-1

Dugan, A. J., Birdsey, R., Mascorro, V. S., Magnan, M., Smyth, C. E., Olguin, M., & Kurz, W. A. (2018). A systems approach to assess climate change mitigation options in landscapes of the United States forest sector. *Carbon Balance and Management*, *13*(1), 13. https://doi.org/10.1186/s13021-018-0100-x

EEA. (2023). Land use.

Ellis, E. C., Gauthier, N., Klein Goldewijk, K., Bliege Bird, R., Boivin, N., Díaz, S., Fuller, D. Q., Gill, J. L., Kaplan, J. O., Kingston, N., Locke, H., McMichael, C. N. H., Ranco, D., Rick, T. C., Shaw, M. R., Stephens, L., Svenning, J.-C., & Watson, J. E. M. (2021). People have shaped most of terrestrial nature for at least 12,000 years. *Proceedings of the National Academy of Sciences*, *118*(17), e2023483118. https://doi.org/10.1073/pnas.2023483118

Eskander, S. M., & Fankhauser, S. (2020). Reduction in greenhouse gas emissions from national climate legislation. *Nature Climate Change*, *10*(8), 750–756. https://doi.org/10.1038/s41558-020-0831-z

European Commision. (2023). Commission Guidelines for Defining, Mapping, Monitoring and Strictly Protecting EU Primary and Old-Growth Forests.

European Commission. (2020). EU biodiversity strategy for 2030: Bringing nature back into our lives. *Communication for the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the regions.*

European Commission. (2021). New EU Forest Strategy for 2030.

European Commission. (2022). Criteria and guidance for protected areas designations.

European Union. (2018). REGULATION (EU) 2018/841 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL.

Eurostat. (2020). Land use overview by NUTS 2 regions [lan_use_ovw].

Eurostat. (2022). Roundwood removals by type of wood and assortment (FOR_REMOV).

Eurostat. (2023a). Eurostat Forestry Database.

Eurostat. (2023b). Land cover overview by NUTS 2 regions.

Eurostat. (2023c). Landfill rate of waste excluding major mineral wastes.

FAO. (2022a). FAOSTAT Database. Forestry Production and Trade.

FAO. (2022b). *The State of the World's Forests 2022. Forest pathways for green recovery and building inclusive, resilient and sustainable economies.* https://doi.org/10.4060/cb9360en

Felton, A., Gustafsson, L., Roberge, J. M., Ranius, T., Hjältén, J., Rudolphi, J., Lindbladh, M., Weslien, J., Rist, L., Brunet, J., & Felton, A. M. (2016). How climate change adaptation and mitigation strategies can threaten or enhance the biodiversity of production forests: Insights from Sweden. *Biological Conservation*, *194*, 11–20. https://doi.org/10.1016/j.biocon.2015.11.030

Felton, A., Lindbladh, M., Brunet, J., & Fritz, Ö. (2010). Replacing coniferous monocultures with mixed-species production stands: An assessment of the potential benefits for forest biodiversity in northern Europe. *Forest Ecology and Management*, *260*(6), 939–947. https://doi.org/10.1016/j.foreco.2010.06.011

FOEN. (2012). *Swiss Biodiversity Strategy* (tech. rep.).

Forest Europe. (2015). State of Europe's Forests 2015.

Forest Europe. (2020). State of Europe's Forests 2020.

Friedlingstein, P., O'Sullivan, M., Jones, M. W., Andrew, R. M., Gregor, L., Hauck, J., Le Quéré, C., Luijkx, I. T., Olsen, A., Peters, G. P., Peters, W., Pongratz, J., Schwingshackl, C., Sitch, S., Canadell, J. G., Ciais, P., Jackson, R. B., Alin, S. R., Alkama, R., . . . Zheng, B. (2022). Global Carbon Budget 2022. *Earth System Science Data*, *14*(11), 4811–4900. https://doi.org/10.5194/essd-14-4811-2022

Grassi, G., Fiorese, G., Pilli, R., Jonsson, K., Blujdea, V., Korosuo, A., & Vizzarri, M. (2021). Brief on the role of the forest-based bioeconomy in mitigating climate change through carbon storage and material substitution, 1–16.

Gregor, K., Knoke, T., Krause, A., Reyer, C. P. O., Lindeskog, M., Papastefanou, P., Smith, B., Lansø, A.-S., & Rammig, A. (2022). Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate. *Earth's Future*, *10*(9), 1–25. https://doi.org/10.1029/2022EF002796

Gregor, K., Krause, A., Reyer, C. P. O., Knoke, T., Meyer, B. F., Suvanto, S., & Rammig, A. (2024). Quantifying the impact of key factors on the carbon mitigation potential of managed temperate forests. *Carbon Balance and Management*, *19*(1), 10. https://doi.org/10.1186/s13021-023-00247-9

Groetzner, P., & Werner, R. (2022). Multiobjective optimization under uncertainty: A multiobjective robust (relative) regret approach. *European Journal of Operational Research*, *296*(1), 101–115. https://doi.org/10.1016/j.ejor.2021.03.068

Gustavsson, L., Nguyen, T., Sathre, R., & Tettey, U. Y. (2021). Climate effects of forestry and substitution of concrete buildings and fossil energy. *Renewable and Sustainable Energy Reviews*, *136*(March 2020), 110435. https://doi.org/10.1016/j.rser.2020.110435

Gutsch, M., Lasch-Born, P., Kollas, C., Suckow, F., & Reyer, C. P. O. (2018). Balancing trade-offs between ecosystem services in Germany's forests under climate change. *Environmental Research Letters*, *13*(4), 045012. https://doi.org/10.1088/1748-9326/aab4e5

Hämäläinen, A., Strengbom, J., & Ranius, T. (2018). Conservation value of low-productivity forests measured as the amount and diversity of dead wood and saproxylic beetles. *Ecological Applications*, *28*(4), 1011–1019. https://doi.org/10.1002/eap.1705

Harmon, M. E. (2019). Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. *Environmental Research Letters*, *14*(6), 065008. https://doi.org/10.1088/1748-9326/ab1e95

Hassan, J., & Eisele, M. (2015). BauBuche - Der nachhaltige Hochleistungswerkstoff. *Bautechnik*, *92*(1), 40–45. https://doi.org/10.1002/bate.201400093

Haverd, V., Smith, B., Canadell, J. G., Cuntz, M., Mikaloff-Fletcher, S., Farquhar, G., Woodgate, W., Briggs, P. R., & Trudinger, C. M. (2020). Higher than expected CO 2 fertilization inferred from leaf to global observations. *Global Change Biology*, *26*(4), 2390–2402. https://doi.org/10.1111/gcb.14950

Haxeltine, A., & Prentice, I. C. (1996). BIOME3: An equilibrium terrestrial biosphere model based on eco-physiological constraints, resource availability, and competition among plant functional types. *Global Biogeochemical Cycles*, *10*(4), 693–709. https://doi.org/10.1029/96GB02344

Hengl, T., Walsh, M. G., Sanderman, J., Wheeler, I., Harrison, S. P., & Prentice, I. C. (2018). Global mapping of potential natural vegetation: An assessment of machine learning algorithms for estimating land potential. *PeerJ*, *2018*(8), 1–36. https://doi.org/10.7717/peerj.5457

Hickel, J., & Kallis, G. (2020). Is Green Growth Possible? *New Political Economy*, *25*(4), 469–486. https://doi.org/10.1080/13563467.2019.1598964

Hickler, T., Vohland, K., Feehan, J., Miller, P. A., Smith, B., Costa, L., Giesecke, T., Fronzek, S., Carter, T. R., Cramer, W., Kühn, I., & Sykes, M. T. (2012). Projecting the future distribution of European potential natural vegetation zones with a generalized, tree species-based dynamic vegetation model. *Global Ecology and Biogeography*, *21*(1), 50–63. https://doi.org/10.1111/j.1466-8238.2010.00613.x

Hill, C., Kymäläinen, M., & Rautkari, L. (2022). Review of the use of solid wood as an external cladding material in the built environment. *Journal of Materials Science*, *57*(20), 9031–9076. https://doi.org/10.1007/s10853-022-07211-x

Hlásny, T., König, L., Krokene, P., Lindner, M., Montagné-Huck, C., Müller, J., Qin, H., Raffa, K. F., Schelhaas, M.-J., Svoboda, M., Viiri, H., & Seidl, R. (2021). Bark Beetle Outbreaks in Europe: State of Knowledge and Ways Forward for Management. *Current Forestry Reports*, *7*(3), 138–165. https://doi.org/10.1007/s40725-021-00142-x

Hof, C., Voskamp, A., Biber, M. F., Böhning-Gaese, K., Engelhardt, E. K., Niamir, A., Willis, S. G., & Hickler, T. (2018). Bioenergy cropland expansion may offset positive effects of climate change mitigation for global vertebrate diversity. *Proceedings of the National Academy of Sciences*, *115*(52), 13294–13299. https://doi.org/10.1073/pnas.1807745115

House of Lords. (2023). *Environment and Climate Change Committee An extraordinary challenge : Restoring 30 per cent of our land and sea by 2030* (tech. rep. July).

Huangfu, Q., & Hall, J. A. J. (2018). Parallelizing the dual revised simplex method. *Mathematical Programming Computation*, *10*(1), 119–142. https://doi.org/10.1007/s12532-017-0130-5

Hurmekoski, E., Hetemäki, L., & Jänis, J. (2022). Outlook for the Forest-Based Bioeconomy. In *Forest Bioeconomy and Climate Change* (pp. 55–89). Springer International Publishing. https://doi.org/10.1007/978-3-030-99206-4_4

IPBES. (2019). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (S. Díaz, J. Settele, B. E.S., H. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. Brauman, S. Butchart, K. Chan, L. Garibaldi, K. Ichii, J. Liu, S. Subramanian, G. Midgley, P. Miloslavich, Z. Molnár, D. Obura, ... C. Zayas, Eds.).

IPCC. (2014). Summary for Policymakers. In *Climate Change 2013 - The Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1–30). Cambridge University Press. https://doi.org/10.1017/CBO9781107415324.004

IPCC. (2023a). Synthesis Report AR6 - Summary for Policymakers. *An Assessment of the Intergovernmental Panel on Climate Change.*

IPCC. (2023b). SYNTHESIS REPORT OF THE IPCC SIXTH ASSESSMENT REPORT (AR6) IPCC. Longer Report.

Ishizaka, A., & Nemery, P. (2013). *Multi-Criteria Decision Analysis*. John Wiley & Sons Ltd. https://doi.org/10.1002/9781118644898

Ito, A., Nishina, K., Reyer, C. P. O., François, L., Henrot, A.-J., Munhoven, G., Jacquemin, I., Tian, H., Yang, J., Pan, S., Morfopoulos, C., Betts, R., Hickler, T., Steinkamp, J., Ostberg, S., Schaphoff, S., Ciais, P., Chang, J., Rafique, R., ... Zhao, F. (2017). Photosynthetic productivity and its efficiencies in ISIMIP2a biome models: Benchmarking for impact assessment studies. *Environmental Research Letters*, *12*(8), 085001. https://doi.org/10.1088/1748-9326/aa7a19

Knauf, M., Köhl, M., Mues, V., Olschofsky, K., & Frühwald, A. (2015). Modeling the CO2-effects of forest management and wood usage on a regional basis. *Carbon Balance and Management*, *10*(1), 1–12. https://doi.org/10.1186/s13021-015-0024-7

Knoke, T., Hanley, N., Roman-Cuesta, R. M., Groom, B., Venmans, F., & Paul, C. (2023). Trends in tropical forest loss and the social value of emission reductions. *Nature Sustainability*. https://doi.org/10.1038/s41893-023-01175-9

Knoke, T., Paul, C., Hildebrandt, P., Calvas, B., Castro, L. M., Hartl, F., Dollerer, M., Hamer, U., Windhorst, D., Wiersma, Y. F., Curatola Fernández, G. F., Obermeier, W. A., Adams, J., Breuer, L., Mosandl, R., Beck, E., Weber, M., Stimm, B., Haber, W., ... Bendix, J. (2016). Compositional diversity of rehabilitated tropical lands supports multiple ecosystem services and buffers uncertainties. *Nature Communications*, *7*(May). https://doi.org/10.1038/ncomms11877

Lamarque, J.-F., Kyle, G. P., Meinshausen, M., Riahi, K., Smith, S. J., van Vuuren, D. P., Conley, A. J., & Vitt, F. (2011). Global and regional evolution of short-lived radiatively-active gases and aerosols in the Representative Concentration Pathways. *Climatic Change*, *109*(1-2), 191–212. https://doi.org/10.1007/s10584-011-0155-0

Larsen, J. B., Angelstam, P., Bauhus, J., Carvalho, J. F., Diaci, J., Dobrowolska, D., Gazda, A., Gustafsson, L., Krumm, F., Knoke, T., Konczal, A., Kuuluvainen, T., Mason, B., Motta, R., Pötzelsberger, E., Rigling, A., & Schuck, A. (2022). *Closer-to-Nature Forest Management. From Science to Policy 12*. European Forest Institute. https://doi.org/10.36333/fs12

Lawrence, J., Haasnoot, M., & Lempert, R. (2020). Climate change: Making decisions in the face of deep uncertainty. *Nature*, *580*, 456.

Lessa Derci Augustynczik, A., & Yousefpour, R. (2021). Assessing the synergistic value of ecosystem services in European beech forests. *Ecosystem Services*, *49*(October 2020), 101264. https://doi.org/10.1016/j.ecoser.2021.101264

Levin, S. A. (1992). The Problem of Pattern and Scale in Ecology: The Robert H. MacArthur Award Lecture. *Ecology*, *73*(6), 1943–1967. https://doi.org/10.2307/1941447

Lindeskog, M., Smith, B., Lagergren, F., Sycheva, E., Ficko, A., Pretzsch, H., & Rammig, A. (2021). Accounting for forest management in the estimation of forest carbon balance using the dynamic vegetation model LPJ-GUESS (v4.0, r9710): Implementation and evaluation of simulations for Europe. *Geoscientific Model Development*, *14*(10), 6071–6112. https://doi.org/10.5194/gmd-14-6071-2021

Liu, Y. Y., Van Dijk, A. I., De Jeu, R. A., Canadell, J. G., McCabe, M. F., Evans, J. P., & Wang, G. (2015). Recent reversal in loss of global terrestrial biomass. *Nature Climate Change*, *5*(5), 470–474. https://doi.org/10.1038/nclimate2581

Luyssaert, S., Marie, G., Valade, A., Chen, Y.-y. Y., Djomo, S. N., Ryder, J., Otto, J., Naudts, K., Lansø, A. S., Ghattas, J., & McGrath, M. J. (2018). Trade-offs in using European forests to meet climate objectives. *Nature*, *562*(7726), 259–262. https://doi.org/10.1038/s41586-018-0577-1

Mäkelä, A., Minunno, F., Kujala, H., Kosenius, A. K., Heikkinen, R. K., Junttila, V., Peltoniemi, M., & Forsius, M. (2023). Effect of forest management choices on carbon sequestration and biodiversity at national scale. *Ambio*. https://doi.org/10.1007/s13280-023-01899-0

Mayer, A. L., Kauppi, P. E., Angelstam, P. K., Zhang, Y., & Tikka, P. M. (2005). Importing Timber, Exporting Ecological Impact. *Science*, *308*(5720), 359–360. https://doi.org/10.1126/science.1109476

Meinshausen, M., Smith, S. J., Calvin, K., Daniel, J. S., Kainuma, M. L. T., Lamarque, J.-F., Matsumoto, K., Montzka, S. A., Raper, S. C. B., Riahi, K., Thomson, A., Velders, G. J. M., & van Vuuren, D. P. (2011). The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Climatic Change*, *109*(1-2), 213–241. https://doi.org/10.1007/s10584-011-0156-z

Müller, J., & Bütler, R. (2010). A review of habitat thresholds for dead wood: A baseline for management recommendations in European forests. *European Journal of Forest Research*, *129*(6), 981–992. https://doi.org/10.1007/s10342-010-0400-5

Nabuurs, G. J., Pussinen, A., van Brusselen, J., & Schelhaas, M. J. (2007). Future harvesting pressure on European forests. *European Journal of Forest Research*, *126*(3), 391–400. https://doi.org/10.1007/s10342-006-0158-y

Nabuurs, G.-J., Verkerk, M.-J., Pieter Johannes Schelhaas, González Olabarria, A., José Ramón Trasobares, & Cienciala, E. (2018). *Climate-Smart Forestry: Mitigation impacts in three European regions.*

Neugarten, R. A., Chaplin-Kramer, R., Sharp, R. P., Schuster, R., Strimas-Mackey, M., Roehrdanz, P. R., Mulligan, M., van Soesbergen, A., Hole, D., Kennedy, C. M., Oakleaf, J. R., Johnson, J. A., Kiesecker, J., Polasky, S., Hanson, J. O., & Rodewald, A. D. (2024). Mapping the planet's critical areas for biodiversity and nature's contributions to people. *Nature Communications*, *15*(1), 261. https://doi.org/10.1038/s41467-023-43832-9

Oberpriller, J., Herschlein, C., Anthoni, P., Arneth, A., Krause, A., Rammig, A., Lindeskog, M., Olin, S., & Hartig, F. (2022). Climate and parameter sensitivity and induced uncertainties in carbon stock projections for European forests (using LPJ-GUESS 4.0). *Geoscientific Model Development*, *15*(16), 6495–6519. https://doi.org/10.5194/gmd-15-6495-2022

Ohashi, H., Hasegawa, T., Hirata, A., Fujimori, S., Takahashi, K., Tsuyama, I., Nakao, K., Kominami, Y., Tanaka, N., Hijioka, Y., & Matsui, T. (2019). Biodiversity can benefit from climate stabilization despite adverse side effects of land-based mitigation. *Nature Communications*, *10*(1), 1–11. https://doi.org/10.1038/s41467-019-13241-y

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., Phillips, O. L., Shvidenko, A., Lewis, S. L., Canadell, J. G., Ciais, P., Jackson, R. B., Pacala, S. W., McGuire, A. D., Piao, S., Rautiainen, A., Sitch, S., & Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, *333*(6045), 988–993. https://doi.org/10.1126/science.1201609

Pardos, M., Del Río, M., Pretzsch, H., Jactel, H., Bielak, K., Bravo, F., Brazaitis, G., Defossez, E., Engel, M., Godvod, K., Jacobs, K., Jansone, L., Jansons, A., Morin, X., Nothdurft, A., Oreti, L., Ponette, Q., Pach, M., Riofrío, J., . . . Calama, R. (2021). The greater resilience of mixed forests to drought mainly depends on their composition: Analysis along a climate gradient across Europe. *Forest Ecology and Management*, *481*, 118687. https://doi.org/10.1016/j.foreco.2020.118687

Peng, L., Searchinger, T. D., Zionts, J., & Waite, R. (2023). The carbon costs of global wood harvests. *Nature*, *620*(May), 14–16. https://doi.org/10.1038/s41586-023-06187-1

Petersson, H., Ellison, D., Appiah Mensah, A., Berndes, G., Egnell, G., Lundblad, M., Lundmark, T., Lundström, A., Stendahl, J., & Wikberg, P.-E. (2022). On the role of forests and the forest sector for climate change mitigation in Sweden. *GCB Bioenergy*, *14*(7), 793–813. https://doi.org/10.1111/gcbb.12943

Pickett, S., & Thompson, J. N. (1978). Patch dynamics and the design of nature reserves. *Biological Conservation*, *13*(1), 27–37. https://doi.org/10.1016/0006-3207(78)90016-2

Pikkarainen, P., Strandman, H., Vento, E., Petty, A., Tikkanen, O.-P., Kilpeläinen, A., & Peltola, H. (2024). Effects of forest conservation and management on timber, ecosystem carbon, dead wood and habitat suitability area in a boreal forest under climate change. *Silva Fennica*, *58*(2). https://doi.org/10.14214/sf.23045

Pörtner, H.-O., Scholes, R. J., Arneth, A., Barnes, D. K. A., Burrows, M. T., Diamond, S. E., Duarte, C. M., Kiessling, W., Leadley, P., Managi, S., McElwee, P., Midgley, G., Ngo, H. T., Obura, D., Pascual, U., Sankaran, M., Shin, Y. J., & Val, A. L. (2023). Overcoming the coupled climate and biodiversity crises and their societal impacts. *Science*, *380*(6642), eabl4881. https://doi.org/10.1126/science.abl4881

Poulter, B., Aragão, L., Andela, N., Bellassen, V., Ciais, P., Kato, T., Lin, X., Nachin, B., Luyssaert, S., Pederson, N., Peylin, P., Piao, S., Saatchi, S., Schepaschenko, D., Schelhaas, M., & Shivdenko, A. (2018). The global forest age dataset (GFADv1.0). https://doi.org/10.1594/PANGAEA.889943

Pugh, T. A. M., Arneth, A., Kautz, M., Poulter, B., & Smith, B. (2019). Important role of forest disturbances in the global biomass turnover and carbon sinks. *Nature Geoscience*, *12*(9), 730–735. https://doi.org/10.1038/s41561-019-0427-2

Ramage, M. H., Burridge, H., Busse-Wicher, M., Fereday, G., Reynolds, T., Shah, D. U., Wu, G., Yu, L., Fleming, P., Densley-Tingley, D., Allwood, J., Dupree, P., Linden, P., & Scherman, O. (2017). The wood from the trees: The use of timber in construction. *Renewable and Sustainable Energy Reviews*, *68*, 333–359. https://doi.org/10.1016/j.rser.2016.09.107

Räty, M., Juutinen, A., Korhonen, K. T., Syrjänen, K., & Kärkkäinen, L. (2023). EU wood production vs. biodiversity goals – possible reconciliation in Finland? *Scandinavian Journal of Forest Research*, *38*(5), 287–299. https://doi.org/10.1080/02827581.2023.2229732

Reineke, L. H. (1933). Perfecting a stand-density index for even-aged forest. *Journal of Agricultural Research*, *46*, 627–638.

Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S. E., Donges, J. F., Drüke, M., Fetzer, I., Bala, G., von Bloh, W., Feulner, G., Fiedler, S., Gerten, D., Gleeson, T., Hofmann, M., Huiskamp, W., Kummu, M., Mohan, C., Nogués-Bravo, D., … Rockström, J. (2023). Earth beyond six of nine planetary boundaries. *Science Advances*, *9*(37), 1–17. https://doi.org/10.1126/sciadv.adh2458

Roebroek, C. T. J., Duveiller, G., Seneviratne, S. I., Davin, E. L., & Cescatti, A. (2023). Releasing global forests from human management: How much more carbon could be stored? *Science*, *380*(6646), 749–753. https://doi.org/10.1126/science.add5878

Rosa, F., Di Fulvio, F., Lauri, P., Felton, A., Forsell, N., Pfister, S., & Hellweg, S. (2023). Can Forest Management Practices Counteract Species Loss Arising from Increasing European Demand for Forest Biomass under Climate Mitigation Scenarios? *Environmental Science and Technology*, *57*(5), 2149–2161. https://doi.org/10.1021/acs.est.2c07867

Sage, R. F. (2020). Global change biology: A primer. *Global Change Biology*, *26*(1), 3–30. https://doi.org/10.1111/gcb.14893

Santoro, M., Cartus, O., Carvalhais, N., Rozendaal, D. M. A., Avitabile, V., Araza, A., de Bruin, S., Herold, M., Quegan, S., Rodríguez-Veiga, P., Balzter, H., Carreiras, J., Schepaschenko, D., Korets, M., Shimada, M., Itoh, T., Moreno Martínez, Á., Cavlovic, J., Cazzolla Gatti, R., … Willcock, S. (2021). The global forest above-ground biomass pool for 2010 estimated from high-resolution satellite observations. *Earth System Science Data*, *13*(8), 3927–3950. https://doi.org/10.5194/essd-13-3927-2021

Sathre, R., & O'Connor, J. (2010). Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science & Policy*, *13*(2), 104–114. https://doi.org/10.1016/j.envsci.2009.12.005

Scarlat, N., Dallemand, J.-F., Taylor, N., Banja, M., Sanchez Lopez, J., & Avraamides, M. (2019). Brief on biomass for energy in the European Union. *EC publication*, 1–8. https://doi.org/10.2760/546943

Schall, P., Gossner, M. M., Heinrichs, S., Fischer, M., Boch, S., Prati, D., Jung, K., Baumgartner, V., Blaser, S., Böhm, S., Buscot, F., Daniel, R., Goldmann, K., Kaiser, K., Kahl, T., Lange, M., Müller, J., Overmann, J., Renner, S. C., … Ammer, C. (2018). The impact of even-aged and uneven-aged forest management on regional biodiversity of multiple taxa in European beech forests. *Journal of Applied Ecology*, *55*(1), 267–278. https://doi.org/10.1111/1365-2664.12950

Schelhaas, M.-J., Hengeveld, G., Moriondo, M., Reinds, G. J., Kundzewicz, Z. W., Ter Maat, H., & Bindi, M. (2010). Assessing risk and adaptation options to fires and windstorms in European forestry. *Mitigation and Adaptation Strategies for Global Change*, *15*(7), 681–701. https://doi.org/10.1007/s11027-010-9243-0

Schulte, M., Jonsson, R., Hammar, T., Stendahl, J., & Hansson, P.-A. (2022). Nordic forest management towards climate change mitigation: Time dynamic temperature change impacts of wood product systems including substitution effects. *European Journal of Forest Research*, *141*(5), 845–863. https://doi.org/10.1007/s10342-022-01477-1

Schwaab, J., Davin, E. L., Bebi, P., Duguay-Tetzlaff, A., Waser, L. T., Haeni, M., & Meier, R. (2020). Increasing the broad-leaved tree fraction in European forests mitigates hot temperature extremes. *Scientific Reports*, *10*(1), 1–9. https://doi.org/10.1038/s41598-020-71055-1

Seidl, R., Eastaugh, C. S., Kramer, K., Maroschek, M., Reyer, C., Socha, J., Vacchiano, G., Zlatanov, T., & Hasenauer, H. (2013). Scaling issues in forest ecosystem management and how to address them with models. *European Journal of Forest Research*, *132*(5-6), 653–666. https://doi.org/10.1007/s10342-013-0725-y

Senf, C., & Seidl, R. (2021a). Mapping the forest disturbance regimes of Europe. *Nature Sustainability*, *4*(1), 63–70. https://doi.org/10.1038/s41893-020-00609-y

Senf, C., & Seidl, R. (2021b). Storm and fire disturbances in Europe: Distribution and trends. *Global Change Biology*, (March), 3605–3619. https://doi.org/10.1111/gcb.15679

Seppälä, J., Heinonen, T., Pukkala, T., Kilpeläinen, A., Mattila, T., Myllyviita, T., Asikainen, A., & Peltola, H. (2019). Effect of increased wood harvesting and utilization on required greenhouse gas displacement factors of wood-based products and fuels. *Journal of Environmental Management*, *247*(December 2018), 580–587. https://doi.org/10.1016/j.jenvman.2019.06.031

Sitch, S., Smith, B., Prentice, I. C., Arneth, A., Bondeau, A., Cramer, W., Kaplan, J. O., Levis, S., Lucht, W., Sykes, M. T., Thonicke, K., & Venevsky, S. (2003). Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. *Global Change Biology*, *9*(2), 161–185. https://doi.org/10.1046/j.1365-2486.2003.00569.x

Skytt, T., Englund, G., & Jonsson, B.-g. (2021). Climate mitigation forestry—temporal trade-offs. *Environmental Research Letters*, *16*(11), 114037. https://doi.org/10.1088/1748-9326/ac30fa

Smil, V. (2015). *Power Density*. The MIT Press. https://doi.org/10.7551/mitpress/10046.001.0001

Smith, B., Wårlind, D., Arneth, A., Hickler, T., Leadley, P., Siltberg, J., & Zaehle, S. (2014). Implications of incorporating N cycling and N limitations on primary production in an individual-based dynamic vegetation model. *Biogeosciences*, *11*(7), 2027–2054. https://doi.org/10.5194/bg-11-2027-2014

Smith, B., Prentice, I. C., & Sykes, M. T. (2001). Representation of vegetation dynamics in the modelling of terrestrial ecosystems: Comparing two contrasting approaches within European climate space. *Global Ecology and Biogeography*, *10*(6), 621–637. https://doi.org/10.1046/j.1466-822X.2001.t01-1-00256.x

Soimakallio, S., Kalliokoski, T., Lehtonen, A., & Salminen, O. (2021). On the trade-offs and synergies between forest carbon sequestration and substitution. *Mitigation and Adaptation Strategies for Global Change*, *26*(1), 4. https://doi.org/10.1007/s11027-021-09942-9

Spinoni, J., Vogt, J. V., Naumann, G., Barbosa, P., & Dosio, A. (2018). Will drought events become more frequent and severe in Europe? *International Journal of Climatology*, *38*(4), 1718–1736. https://doi.org/10.1002/joc.5291

Taylor, K. E., Stouffer, R. J., & Meehl, G. A. (2012). An Overview of CMIP5 and the Experiment Design. *Bulletin of the American Meteorological Society*, *93*(4), 485–498. https://doi.org/10.1175/BAMS-D-11-00094.1

Turner, D. P., Dodson, R., & Marks, D. (1996). Comparison of alternative spatial resolutions in the application of a spatially distributed biogeochemical model over complex terrain. *Ecological Modelling*, *90*(1), 53–67. https://doi.org/10.1016/0304-3800(95)00143-3

Turner, M. G., O'Neill, R. V., Gardner, R. H., & Milne, B. T. (1989). Effects of changing spatial scale on the analysis of landscape pattern. *Landscape Ecology*, *3*(3-4), 153–162. https://doi.org/10.1007/BF00131534

Uhde, B., Heinrichs, S., Stiehl, C. R., Ammer, C., Müller-Using, B., & Knoke, T. (2017). Bringing ecosystem services into forest planning – Can we optimize the composition of Chilean forests based on expert knowledge? *Forest Ecology and Management*, *404*(March), 126–140. https://doi.org/10.1016/j.foreco.2017.08.021

United Nations. (2023). *Technical dialogue of the first global stocktake. Synthesis report by the co-facilitators on the technical dialogue* (tech. rep.).

Verkerk, P. J., Mavsar, R., Giergiczny, M., Lindner, M., Edwards, D., & Schelhaas, M. J. (2014). Assessing impacts of intensi fi ed biomass production and biodiversity protection on ecosystem services provided by European forests. *Ecosystem Services*, *9*, 155–165. https://doi.org/10.1016/j.ecoser.2014.06.004

Verkerk, P. J., Zanchi, G., & Lindner, M. (2014). Trade-offs between forest protection and wood supply in Europe. *Environmental Management*, *53*(6), 1085–1094. https://doi.org/10.1007/s00267-014-0265-3

Virtanen, P., Gommers, R., Oliphant, T. E., Haberland, M., Reddy, T., Cournapeau, D., Burovski, E., Peterson, P., Weckesser, W., Bright, J., van der Walt, S. J., Brett, M., Wilson, J., Millman, K. J., Mayorov, N., Nelson, A. R. J., Jones, E., Kern, R., Larson, E., … Contributors, S. 1. (2020). SciPy 1.0: Fundamental algorithms for scientific computing in Python. *Nature Methods*, *17*(3), 261–272. https://doi.org/10.1038/s41592-019-0686-2

Warren, R., Price, J., Graham, E., Forstenhaeusler, N., & VanDerWal, J. (2018). The projected effect on insects, vertebrates, and plants of limiting global warming to 1.5°C rather than 2°C. *Science*, *360*(6390), 791–795. https://doi.org/10.1126/science.aar3646