Unsicherheit zukünftiger Waldentwicklung unter besonderer Berücksichtigung von Standort und Sterblichkeit der Wald-Kiefer (*Pinus sylvestris* L.) in Norddeutschland

Dissertation

zur Erlangung des Doktorgrades *Dr. forest.*
der Fakultät für Forstwissenschaften und Waldökologie
der Georg-August-Universität Göttingen

vorgelegt von

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geboren am 1. April 1990 in Wildeshausen

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Uncertainty of Future Forest Development with Special Regard to Site and Survivability of Scots pine (*Pinus sylvestris* L.) in Northern Germany

Dissertation

to attain the doctoral degree *Dr. forest.*
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submitted by

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Diese Arbeit über die Waldzukunft sei dem bislang Ungewürdigten gewidmet, nämlich

der grünen Fährte,
die wohl im Danziger Waldamt begann und auf der meine Familie seit 1651 dem Försterdasein nachgeht, sowie

dem herzlichen Gedenken
an den königlich-preußischen Förster Franz Heinrich Holtzheimer (1834-1914) und den Stadtrevierförster Hermann Georg Holtzheimer (1888-1945), die eine Zukunft auf „ihren“ Förstereien nie erleben konnten.

Inhalt

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Zusammenfassung

Unsicherheit zukünftiger Waldentwicklung unter besonderer Berücksichtigung von Standort und Sterblichkeit der Wald-Kiefer (*Pinus sylvestris* L.) in Norddeutschland

Die Wald-Kiefer (*Pinus sylvestris* L.) ist die hinsichtlich Anbaufläche und erwarteter Trockenstresstoleranz bedeutendste Baumart Norddeutschlands. Typisch sind wenig durchmischte, störungsanfällige Bestände im Alter von etwa 75 Jahren. Vor dem Hintergrund des Klimawandels und einer unausgewogenen Altersstruktur stellt sich die Frage, wie die Bewirtschaftung von Kiefernwäldern ökologisch und ökonomisch weiterentwickelt werden kann, um ihre Funktionalität zu erhalten. Diese Arbeit untersucht die zukünftige Entwicklung von Kiefernwäldern unter verschiedenen realitätsnahen Bewirtschaftungs- und Klimaszenarien mit Fokus auf Standortbedingungen und Sterblichkeitsprozesse. Ziel ist es, die Bandbreite möglicher Entwicklungsgänge aufzuzeigen, bestehende Unsicherheiten zu quantifizieren und waldbauliche Handlungsempfehlungen zur Ernte und Anbaueignung unter dynamischen Umweltbedingungen abzuleiten.

Thematische Schwerpunkte liegen in der Analyse von Ursache-Wirkungs-Zusammenhängen sowie der Entwicklung eines Ziel-Mittel-Systems für die Waldentwicklung unter Unsicherheit. Dabei werden für Nordwest- und Nordostdeutschland differenzierte, ertragskundlich und waldbaulich relevante Einflussgrößen auf den naturnahen Waldumbau simuliert – jeweils unter Einbeziehung und Ausschluss von Störungen. Ergänzend erfolgt eine Modellierung von Störungsschweregraden in nordwestdeutschen Kiefernwäldern. Zur prioritären Umsetzung waldbaulicher Anpassungsmaßnahmen wird ein multikriterielles Entscheidungsunterstützungssystem entwickelt, das sich auf die Landschaftsebene fokussiert.

Basierend auf Daten der dritten Bundeswaldinventur wurden acht forstliche Bewirtschaftungsszenarien für einen Zeitraum von 50 Jahren auf Einzelbaumbasis simuliert. In vier ungestörten standort- und klimasensitiven Szenarien erfolgt die Auswahl von Waldentwicklungstypen

mit dem Ziel, entweder hohe oder niedrige Kiefernanteile zu fördern. In einer anschließenden Monte-Carlo-Simulation wurden für die Szenarien mit hohem Kiefernanteil vier weitere Störungsszenarien mit verschiedenen Schweregraden definiert, die diskrete Bestandes- und diffuse Einzelbaumschäden kontrastieren. Anhand eines gleitenden Fensters von Klimadaten wird prognostiziert, dass sich die Sterblichkeit je nach Baumart zukünftig verdoppeln könnte. Durch die Gegenüberstellung unterschiedlicher Bewirtschaftungs- und Störungsszenarien wurde sowohl die Bandbreite möglicher Entwicklungen als auch das Ausmaß der Unsicherheit abgeschätzt.

Die Ergebnisse zeigen, dass die Anbauflächen und die Vorräte der Kiefer selbst unter fördernder Bewirtschaftung zurückgehen könnten. Dabei steigen die Starkholzvorräte und Nutzungsmengen sowohl mit als auch ohne Störungseinfluss. Die Waldentwicklung wird stärker von der Wahl der Waldbauziele und -verfahren, als von der Wahl der Klimamodelle beeinflusst. Die Waldumbaurate variiert bei ausgeprägten Reinbeständen zwischen 50 und 72 %; maßgeblich abhängig vom Durchforstungsregime sowie vom Zeitpunkt und Turnus der Ernte. Diskrete, hohe Schweregrade neigen dazu, den Waldumbau gemäß den Modellannahmen zu beschleunigen. Angesichts des gegenwärtig hohen Anteils alter Kiefernbestände erfolgt der Waldumbau insgesamt zu langsam und zu wenig differenziert, um den Herausforderungen des fortschreitenden Klimawandels und der unausgewogenen Altersstruktur angemessen zu begegnen.

Untersucht wurde ferner, welche Störungsfaktoren die Mortalität von Kiefern beeinflussen und ob sich daraus Schweregrade modellieren lassen. Diese Analyse basiert auf Massenerfassungen im staatlichen Waldschutz-Meldeportal, die mit Forsteinrichtungs- und Wetterdaten verknüpft wurden. Die Ergebnisse zeigen, dass diffuse Störungen mit niedrigen Schweregraden über einen Zeitraum von zehn Jahren am häufigsten auftreten und meist durch Wind ausgelöst werden. Die durchschnittliche Schadfläche beträgt 0,5 ha und die mediane Fläche liegt bei 0,12 ha. Biotische Störungsfaktoren verursachen im Vergleich zu abiotischen Faktoren höhere Schweregrade und nehmen zu. Der Schweregrad von Störungen in Kiefernwäldern wird besonders effektstark durch Konkurrenzverhältnisse, abgebildet durch den Mischungsanteil, sowie die Wasserverfügbarkeit beeinflusst. Insbesondere weisen Kiefern auf trockenen Standorten die stärksten Schäden auf. Es zeigt sich jedoch die Tendenz, dass die Schweregrade mit zunehmender Standortsgüte wieder ansteigen, wenn gleichzeitig die Baumartenvielfalt hoch ist. Die verwendete bayesianische Beta-Regression erreicht eine Erklärungskraft von 38 %.

Im Fallbeispiel des Landkreises Lüchow-Dannenberg dienen die vielfältigen Standorts- und Bestockungsattribute auf Landschaftsebene als Grundlage zur Priorisierung eines besitzübergreifenden Waldumbaus. Mithilfe einer multikriteriellen Entscheidungsanalyse wurden Fernerkundungs- und Feldbeobachtungsdaten genutzt, um die Baumartenvielfalt im Oberstand und deren Trockenstresstoleranz zu bewerten. Durch Gleichgewichtung der Indikatoren ergibt sich eine differenzierte Reihenfolge, die die bislang monokriterielle Erntedefinition anhand eines Zieldurchmessers ergänzen kann.

Die Arbeit betont die Bedeutung evidenzbasierter Entscheidungsinstrumente für eine resiliente Waldentwicklung und unterstreicht die Notwendigkeit, die Waldbau-Richtlinien weiterzuentwickeln. Insbesondere kombinierte, ungleichmäßige Verjüngungsformen eignen sich, das natürliche Störungsregime der Kiefer in Norddeutschland nachzubilden. Für das Erreichen naturnaher Zielstellungen ist der Umgang mit dem Nebenbestand zentral, wofür Überhälter – durch früheren Erntebeginn als bisher – langfristig und in ausreichender Dichte ausreifen sollten. Die Ernteabfolge sollte durch eine multikriterielle Priorisierung auf Landschaftsebene und durch einen Zielstärkenrahmen ausgestaltet werden.

Die zielgerichtete Waldentwicklung wird voraussichtlich weniger durch statische Planungen gelingen, sondern vielmehr durch dynamische Anpassungsprozesse. Vor dem Hintergrund der klimawandelbedingten Unsicherheit erscheint die Praktizierung einer Vielfalt an *kombinierten* Waldbauverfahren als ein geeigneter Ansatz, um differenziert auf unbekannte Herausforderungen reagieren zu können.

Abstract

Uncertainty of Future Forest Development with Special Regard to Site and Survivability of Scots pine (*Pinus sylvestris* L.) in Northern Germany

Scots pine (*Pinus sylvestris* L.) is the most important tree species in northern Germany in terms of both area of cultivation and expected tolerance to drought stress. It typically occurs in poorly mixed, disturbance-prone stands that are around 75 years old. Against the backdrop of climate change and an imbalanced age structure, the question arises of how pine forest management can be further developed in ecological and economic terms to ensure long-term functionality and viability. This study investigates the future development of Scots pine forests under different realistic forest management and climate scenarios, with a particular focus on site conditions and survivability. The aim is to illustrate the range of potential development trajectories, to quantify associated uncertainties, and to derive silvicultural recommendations for an adaptation to dynamic environmental conditions.

The research focuses on analyzing cause-effect relationships and developing a goal-means framework for forest development under uncertainty. Simulations were conducted for northwestern and northeastern Germany, regionally differentiated, in terms of yield sciences and silviculture significant drivers of forest restoration, both with and without disturbance impact. Additionally, the severity of disturbances in northwestern pine forests was modeled. To support the prioritization of silvicultural adaptation measures, a multi-criteria decision support system was developed that emphasizes the landscape level scale.

Based on data from the third National Forest Inventory, eight forest management scenarios were simulated over a 50-year period at the single-tree level. In four site-specific and climate-sensitive scenarios without disturbance impact, forest development types were selected to promote either high or low stand mixture proportions of Scots pine. In a subsequent Monte Carlo simulation, four disturbance scenarios with varying severity levels were applied to the high-pine proportion scenarios. These scenarios contrast discrete, stand-wise disturbances with diffuse, tree-wise damage. Using a moving window approach of climate data, mortality was projected to potentially double in the future, depending on the tree species. By comparing the different management and disturbance scenarios, both the range of possible developments and the range of uncertainty were estimated.

The results suggest that both the area of cultivation and growing stock of Scots pine could decline even under management practices designed to support the species. However, the stock of large-dimensioned timber and the amount of harvest increase under both disturbance severity scenarios. Forest development is more strongly influenced by silvicultural objectives and practices than by the choice of climate model. In monospecific stands, the forest landscape restoration rate ranges from 50 % to 72 %, largely depending on thinning intensity, as well as starting time and cycles of harvests. High-severity, discrete disturbances were found to accelerate forest restoration under the model assumptions. Given the currently high proportion of mature pine stands, the restoration rate is too slow and lacks sufficient differentiation to meet the challenges posed by climate change and the imbalance of the age structure.

The study further investigated which disturbance factors influence Scots pine mortality and whether a severity can be modeled accordingly. This analysis was based on crowd sensing data from the governmental forest health notification portal, which was linked with forest management plans and weather data. The findings show that diffuse, low-severity disturbances are the most frequent over a ten-year period and are mainly triggered by wind. The average disturbance size covers 0.5 ha, with a median of 0.12 ha. Biotic disturbance factors cause higher severity than abiotic ones and are increasing in frequency. The severity of disturbances in Scots pine forests is strongly influenced by competitive pressure, as indicated by pine's volume mixture ratio, as well as by site water availability. Particularly severe damage occurs on dry sites, although severity tends to rise again under favorable growing conditions. The Bayesian beta regression applied in this study achieves an explanatory power of 38 %.

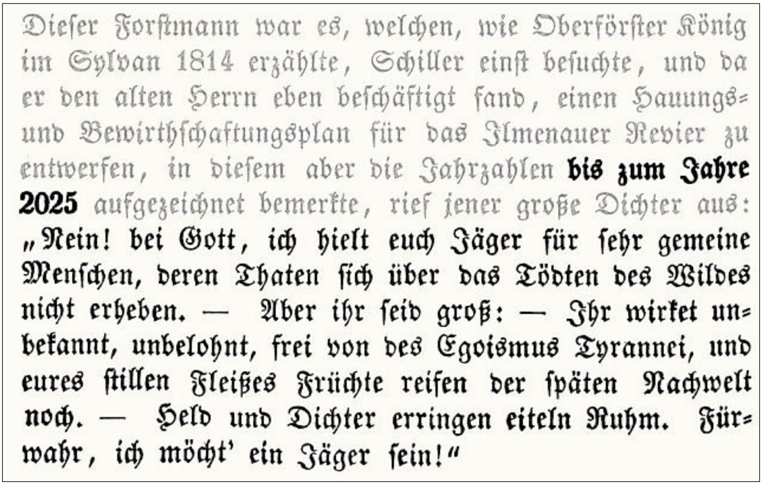
In the case study of the Lüchow-Dannenberg county, diverse site and stocking conditions at the landscape level scale were used to prioritize forest restoration across ownership boundaries. A multi-criteria decision analysis was conducted using remote sensing and field observation data to assess overstory tree species diversity and their tolerance to drought stress. Equal weighing of indicators produced a differentiated prioritization sequence, supplementing the previously monocriterial harvest definition based on target diameter.

This work emphasizes the importance of evidence-based decision tools for fostering resilient forest development and highlights the need to revise existing silvicultural guidelines. In particular, uneven-aged and combined regeneration systems are suitable for emulating the natural disturbance regime of Scots pine in northern Germany. For achieving close-to-nature goals, the treatment of the subsidiary stand is substantial: To this end, reserves should be retained at sufficient densities and the initiation of harvests in the main stand should begin earlier than has been common practice. Harvesting sequences should be guided by multi-criteria prioritization at the landscape level and by a framework of target diameters.

Overall, the intended forest development is less likely to succeed through static planning alone and will increasingly depend on dynamic adaptation processes. In light of uncertainty associated with climate change, the application of diverse silvicultural strategies represents a promising approach for responding flexibly and effectively to unforeseen challenges for Scots pine survivability.

1. Einleitung

[1]



Dieser Forstmann war es, welchen, wie Oberförster König im Sylvan 1814 erzählte, Schiller einst besuchte, und da er den alten Herrn eben beschäftigt fand, einen Hauungs- und Bewirthschaftungsplan für das Ilmenauer Revier zu entwerfen, in diesem aber die Jahrzahlen **bis zum Jahre 2025** aufgezeichnet bemerkte, rief jener große Dichter aus: **„Nein! bei Gott, ich hielt euch Jäger für sehr gemeine Menschen, deren Thaten sich über das Tödten des Wildes nicht erheben. — Aber ihr seid groß: — Ihr wirket unbekannt, unbelohnt, frei von des Egoismus Tyrannei, und eures stillen Fleißes Früchte reifen der späten Nachwelt noch. — Held und Dichter erringen eiteln Ruhm. Fürwahr, ich möcht' ein Jäger sein!"**

[1] Das bei Koch (1842) überlieferte Zitat von Friedrich Schiller zur Forsteinrichtung des Ilmenauer Stadtwaldes belegt dessen Erstaunen über das Forstwesen und seine hypothetische Berufsalternative. Das Planungswerk wurde von Karl Oettelt aufgestellt und reichte bis 2025, wohingegen König (1814) das Jahr 2050 angibt. Die Bezeichnung *Jäger* war im frühen 19. Jahrhundert ein Synonym für *Förster*.

1.1. Hintergrund

Es liegt in der Natur des Menschen, in die Zukunft schauen zu wollen, um das eigentlich Unmögliche zu antizipieren (Lombardo 2006). In einem rationellen Forstwesen bedeutet dies vor allem vorausschauend zu planen (Köstler 1944; von Gadow 2005, p. 119). Die Unsicherheit – als wahrgenommene Unfähigkeit, etwas genau zu wissen oder vorherzusagen (Hoogstra & Schanz 2008) – steigt im Zuge des Klimawandels und der Globalisierung zulasten einer geregelten Waldbewirtschaftung an (Albert et al. 2017; Marchau et al. 2019, p. 3). Insbesondere häufen sich Ereignisse extremen Wetters und epidemischer Massenvermehrungen (WBW 2019; Fuchs et al. 2024), auf die die Forstwirtschaft zeitnah reagieren und sich – sowie die Wälder – anpassen muss (WBW 2021; Puettmann & Bauhus 2023; Hartmann et al. 2025). Seit den neuartigen Waldschäden der 1980er Jahre, dem sogenannten Waldsterben, bildet der *ökologische Waldumbau* hin zu naturnäheren Mischbeständen den zentralen Schwerpunkt der deutschen Forstwirtschaft (Heuer 2004; WBW 2019). Er wurde zunehmend geprägt durch die Herausforderungen des Klimawandels (Rumpf & Petersen 2008, p. 193; Bolte et al. 2021; WBW 2021). In der Dekade zur Wiederherstellung von Ökosystemen der Vereinten Nationen findet der Waldumbau von 2021 bis 2030 auch besondere politische Berücksichtigung (UN 2019).

Gemessen an der Fläche ist die Wald-Kiefer (*Pinus sylvestris* L.; im Folgenden nur Kiefer) die am weitesten verbreitete Baumart in Deutschland (BMEL 2024, p. 17). Ihr Verbreitungsschwerpunkt liegt im norddeutschen Tiefland. Aufgrund eines zunehmenden Anteils höherer Stärkeklassen zeigen aktuelle Entwicklungen einen stetigen Anstieg sowohl des Holznutzungs- als auch des Habitatpotenzials. Gleichzeitig ist die Nachfrage nach Kiefernstarkholz (KSH) begrenzt (Schrade 2002; Beinhofer & Knoke 2010; Mason et al. 2022) und es besteht geringe Intentionalität, zeitnah reife Bäume mittlerer Qualität zu erhalten, die besonders viele Habitatstrukturen entwickeln. Zwischen Erhalt und Ernte ergibt sich das Dilemma, zur Risikosenkung eine intensivere Nutzung mit geringeren Zielstärken und Holzvorräten zuzulassen (Seidl et al. 2011; Rosenkranz et al. 2017, p. 15; Muys & Messier 2023). Dies gilt jedoch als Haupttreiber des Biodiversitätsverlusts im Wald (Schall & Ammer 2013; Ammer et al. 2018; Turmukhametova et al. 2020).

Weil Kiefernbestände zwar überwiegend mehrschichtig sind, aber die geringsten Anteile an Mischbaumarten aufweisen (BMEL 2024, p. 25), wird zur Risikosenkung und Resilienzsteigerung die Misch- und Laubwaldmehrung weg von Reinbeständen finanziell und fachlich gefördert (Spathelf & Ammer 2015; Erdozain et al. 2024). In der Folge des Waldumbaus sind das mittel- und langfristige Ressourcen- sowie Habitatangebot an Kiefer unsicher. Um allen Akteuren der Forst-, Holz- und Naturschutz-Branche die Möglichkeit zu geben, sich im besten Fall proaktiv und kooperativ auf wandelnde Bedingungen einzustellen (Braunschweiger et al. 2024), sind eine transparente Landnutzungsplanung und Marktsignale wichtig (Pirard 2012; Beratan 2014; Rosenkranz et al. 2023).

Auch wenn die generellen Chancen des ökologischen Waldumbaus und der Baumartendiversifizierung bekannt sind und überwiegen (Jactel et al. 2005; Ammer et al. 2008; Paul et al. 2020), besteht in der Praxis im Speziellen oftmals Unklarheit über die Priorisierung der Bestände und Maßnahmen (Buckingham et al. 2019; Axer et al. 2023; Nikinmaa et al. 2024), die Baumartenwahl (Albert et al. 2017; Butterfield et al. 2017), deren Mischungsformen und -anteile (Pretzsch & Zenner 2017), sowie den Zeitpunkt (Bestandesalter/ Jahreszeit) des Überführungsstarts (Rumpf & Petersen 2008, p. 209; Nagel 2016; Pretzsch 2019a). Zu den generellen Risiken eines Waldumbaus von Rein- hin zu Mischbeständen werden eine Angleichung auf Landschaftsebene, erhöhte Sterberisiken in der Überführungsphase, u.U. geringere Holzqualitäten sowie eine anspruchsvollere Behandlung, Planung und Ernte gezählt (Pretzsch & Rais 2016; Pretzsch 2019).

Der Klimawandel und die Globalisierung prägen die künftige Unsicherheit in der Forstwirtschaft maßgeblich (Sturrock et al. 2011; Vacek et al. 2023; Hartmann et al. 2025). Auch die historisch bedingte unausgewogene Flächen- und Vorratsverteilung der norddeutschen Kiefernwälder

(Abb. 1.1) schafft unsichere Rahmenbedingungen: Die Erfüllung ökologischer, ökonomischer und sozialer Waldfunktionen ist unter den derzeitigen Bestandesstrukturen nicht in gleichbleibendem Umfang zu erwarten (Vangi et al. 2024). Abb. 1.1 fokussiert sich exemplarisch auf Kiefernwälder in Staatsbesitz (Landes- und Bundesforstbetriebe).

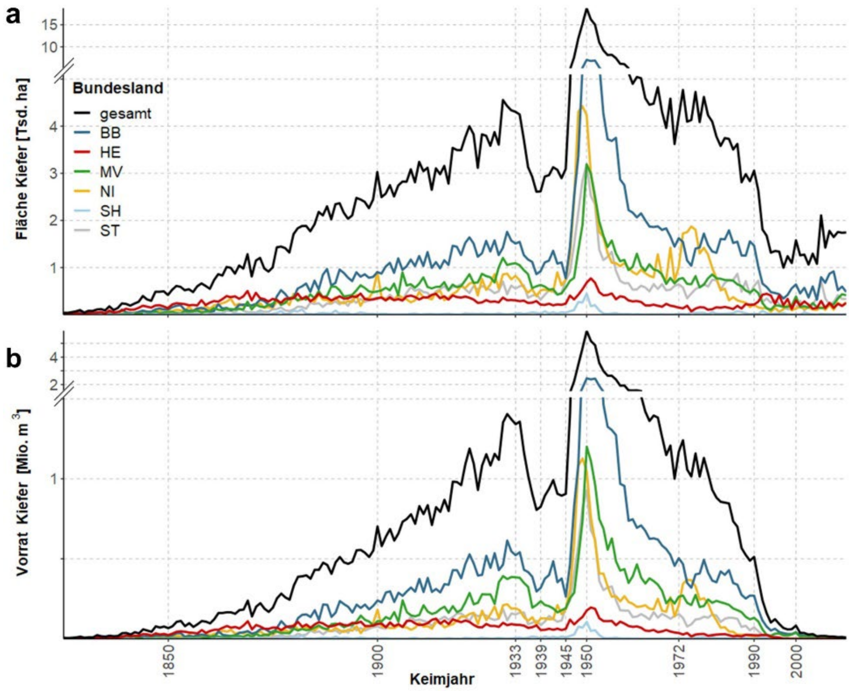


Abb. 1.1 Absolute **a)** Standflächen (Summe der Anteilsflächen) und **b)** Derbholzvorräte der Kiefer in den Staatsforsten Norddeutschlands (Besitz der Bundes- und Landesforstbetriebe) nach Keimjahr gemäß Forsteinrichtung (Stand 2022, Daten bis 2012). BB: Brandenburg, HE: Hessen, MV: Mecklenburg-Vorpommern, NI: Niedersachsen, SH: Schleswig-Holstein, ST: Sachsen-Anhalt

Mit dem Begriff Norddeutschland sind in dieser Arbeit die Bundesländer Brandenburg (BB), Hessen (HE), Mecklenburg-Vorpommern (MV), Niedersachsen (NI), Schleswig-Holstein (SH) und Sachsen-Anhalt (ST) gemeint. Diese umfassen mit nur ca. 40 % Anteil am deutschen Wald bereits ca. 80 % der Kiefernfläche.

Die altersmäßige Unausgewogenheit der Waldzusammensetzung in diesen Bundesländern wird oft mit „Reparationshieben“ oder „Nachkriegsaufforstungen“ begründet (z.B. Spellmann 2002). Diese Vereinfachung lässt jedoch außer Acht, dass die gegenwärtige Waldstruktur aus einer Ereigniskette der 1930er bis 1950er resultiert: Dazu zählen der Dauerwaldstreit (Steinsiek 1995) mit Nutzungsrückständen, gefolgt von Übernutzungen (Wiedemann 1944), Waldbränden durch Kampfhandlungen und präventiven Beräumungen bei Splitterverdacht, verzögerter Wiederbewaldung durch Waldfeldbau und Bodenreform, Überkapazitäten der Holzindustrie, sowie Großkalamitäten (Knocke et al. 2023a). Für den Staatswald, der unmittelbar politischem Einfluss unterliegt, verdeutlicht Abb. 1.1, dass das Verjüngungsgeschehen in den 1930er Jahren abnahm und Erntemaßnahmen mit Beginn des Krieges wieder anstiegen. Die unausgewogene Waldstruktur der Kiefer in Norddeutschland wird folglich durch die Bezeichnung *im Kontext des Zweiten Weltkrieges* präziser beschrieben.

In der vorliegenden Arbeit wird Waldentwicklung vereinfacht als quantifizierbare Dynamik der Baumartenzusammensetzung von Beständen über die Zeit definiert. Aktive Veränderungen in Artenzusammensetzung und Altersstruktur werden dem Waldumbau zugeordnet.

Die Waldentwicklung wurde in Norddeutschland maßgeblich durch große Erstaufforstungswellen im 19. Jahrhundert (Spathelf & Ammer 2015; Erdozain et al. 2024) und prominente Störungsereignisse, wie Orkane 1972 in Niedersachsen oder 1990 in Hessen geprägt (Abb. 1.1a; Bartsch et al. 2020, p. 45). Insbesondere die Jahrtausenddürre ab 2018 (Büntgen et al. 2021) sowie Wälder, die im Kontext des Zweiten Weltkrieges entstanden und inzwischen in windwurfgefährdete Bestandeshöhenbereiche einwachsen (Wohlgemuth et al. 2019, p. 168), verdeutlichen den Handlungsbedarf.

Während der Anbau und die Bewirtschaftung der Kiefer *forschungsseitig* nur mit Einschränkungen empfohlen wird, setzt man in

Norddeutschland *praxisseitig* verstärkt auf ihre kostengünstige Naturverjüngung, zeitnahe Ernte (Rosenkranz et al. 2017, p. 70) und gestaffelte Pflegeturnusse (s. Abb. 1.3b). Vor dem Hintergrund klimawandelbedingt erhöhter Sterberisiken und einer damit verbundenen, unsicheren Waldentwicklung, ergeben sich vielfältig mögliche Waldbaukonzepte. Diese bezeichnen verschriftlichte Waldbausysteme (*cf.* Kap. 1.2.2). In der öffentlichen Verwaltung entfalten Waldbaukonzepte ihre Wirkung als Verwaltungsvorschriften nur im Innenverhältnis und sind meist als Richtlinie, Erlass, Merkblatt oder Anweisung überschrieben (Frank et al. 2007, p. 281).

Insbesondere für Kiefernbestände, die im Kontext des Zweiten Weltkrieges entstanden, scheint sich hinsichtlich Ernte und Erhalt sowie der Baumarteneignung eine Umsetzungslücke zwischen wissenschaftlicher Evidenz und forstlicher Praxis zu öffnen. Ziel dieser Arbeit ist es, durch anwendungsorientierte Beiträge (*cf.* Kap. 9.1) dazu beizutragen, diese Diskrepanz zu überbrücken und wissenschaftliche Erkenntnisse besser in waldbauliche Entscheidungsprozesse zu integrieren. Anhand von Simulationen leitet diese Arbeit konkrete Empfehlungen zur Anpassung von Kiefern-Waldbaukonzepten ab und fasst ihre ökophysiologische Anbaueignung unter Unsicherheit zusammen.

1.2. Stand der Forschung und Implementierung

1.2.1. Unsichere Entwicklungsgänge der Kiefernwirtschaft

Wissensstand zur Ökologie der Kiefer

Die Kiefer ist eine zirkumpolare Baumart, deren synanthropes Areal auch das nördliche Amerika umfasst (Delavaux et al. 2023). Sie ist die *Pinus*-Art, die weltweit am Weitesten verbreitet, sowie genetisch und morphologisch sehr variabel ist (Huston Durrant et al. 2016). Auch in Deutschland werden verschiedene Ökotypen ausgebildet (Paul & Šeho 2024). Unter den heimischen Baumarten hat die Kiefer die breiteste physiologische Amplitude (Leuschner & Ellenberg 2017, p. 521). Die höchste

Überlebensfähigkeit wird ihr in winterkalten Regionen eingeräumt (Huston Durrant et al. 2016; Bose et al. 2024) und ihr Existenzoptimum befindet sich an den Randbereichen des Ökogramms (Leuschner & Ellenberg 2017, p. 183). Eine ökophysiologische Synthese zur Anbaueignung der Kiefer wurde in Kap. 7.5 zusammengestellt.

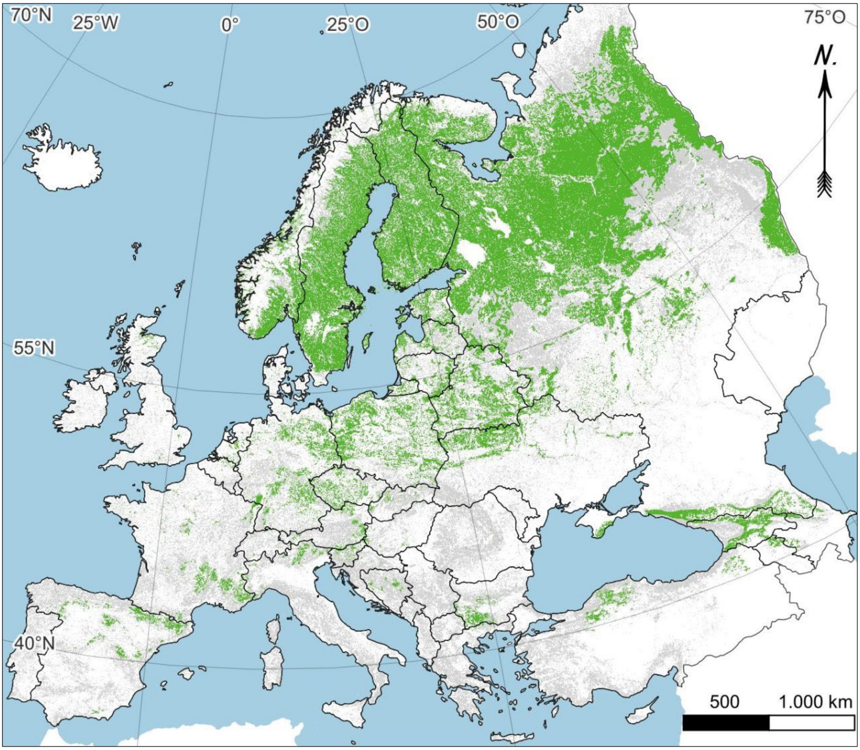


Abb. 1.2 Waldbedeckung Europas und Vorderasiens (grau; Burgoin et al. 2024) sowie Areal der Wald-Kiefer (*Pinus sylvestris* L.; grün; nicht-EU: Bohn et al. 2004; EU: de Rigo et al. 2016; Vorderasien: Caudullo et al. 2017). Staatsgrenzen nach © opendatasoft/ OGL (2019) und © GeoBasis-DE/ BKG (2022)

Die Grenzen des Kiefernareals korrelieren mit der -1 °C-Isotherme für die mittlere Temperatur des kältesten Monats und der +33 °C-Isotherme für die mittlere Temperatur des wärmsten Monats (Dahl 2007, p. 88). In Europa hat sie folglich einen nordischen Schwerpunkt (Abb. 1.2) und wird als die wirtschaftlich bedeutendste Art hervorgehoben (Huston Durrant et al. 2016; Brichta et al. 2023), aber ist auch am stärksten von Störungen betroffen (Bonannella et al. 2024). Im Klimawandel wird für Kiefer im Tiefland mit mehr oder weniger gleichbleibendem (Bose et al. 2020a; Schmidt et al. 2023, p. 170) bis verringertem Wachstum (Diers et al. 2024) gerechnet. Durch Witterungsextreme wird seit den 1980er Jahren außerhalb des natürlichen Areals ein komplexes *Kiefernsterben* verursacht (Tomiczek 1982, zit. n. Mayer 1992, p. 128). Flächenmäßig nicht bedeutsame, jedoch in ihrem Erhalt stark gefährdete, besonders artenreiche, natürliche Flechten-Kiefernwälder erhalten derzeit besondere Aufmerksamkeit: Sie sind die Pflanzengesellschaft des Jahres 2025 (Brändle & Brandl 2001; Horn et al. 2024).

Implementierungsstand

Die Kiefer ist im Vergleich zu anderen Baumarten ertragsschwach und ihr Anbau war historisch starken Störungsereignissen ausgesetzt (Köstler 1944; Beinhofer & Knoke 2010; Sommerfeld et al. 2018). Seit den 1970er Jahren verlaufen die Entwicklungsgänge (Trajektorien) insgesamt weg von der Kiefer, wie auch auf aggregierter Ebene in Abb. 1.1a sichtbar wird. Seit den 2000er Jahren kehrt sich diese Entwicklung *praxisseitig* jedoch um (*cf.* Abb. 1.1a; NMELV 2024, p. 33), weil ihre Anpassungsfähigkeit und Trockenstresstoleranz unter Gesichtspunkten des Klimawandels wieder mehr in den Fokus rücken (Böckmann et al. 2019). Derzeit befinden sich jedoch besonders im trockenen Südhessen, der Altmark sowie in Anhalt Schadschwerpunkte (Rohde et al. 2023).

Wissenschaftsseitig werden die Anbaueignung und das Mortalitätsrisiko von Kiefern in Deutschland intensiv diskutiert, jedoch zeigt sich kein einheitlicher Tenor (s. Tab. 7.2). Auf Basis von Klimahüllen (<u>Süddeutschland</u>; Pearson & Dawson 2003; Walentowski et al. 2007) oder der Standortswasserbilanz (SWB, <u>Norddeutschland</u>; Grier & Running 1977; Spellmann et al. 2011) wird die Kiefernwirtschaft in Mischbeständen auf schwach bis mäßig nährstoffversorgten Standorten entweder zurückhaltend (Forster et al. 2019, p. 43) oder uneingeschränkt (Böckmann et al. 2019, p. 8) empfohlen. Auch Bolte et al. (2021) können Kiefernwälder nicht eindeutig in Risiko- und Nichtrisikobestände einteilen, während sie den Waldumbaubedarf bei Fichten und Buchen klar identifizieren.

Hinsichtlich der Bestandesbehandlung zeigen sich innerhalb Norddeutschlands sehr unterschiedliche Waldbaukonzepte (Abb. 1.3a).

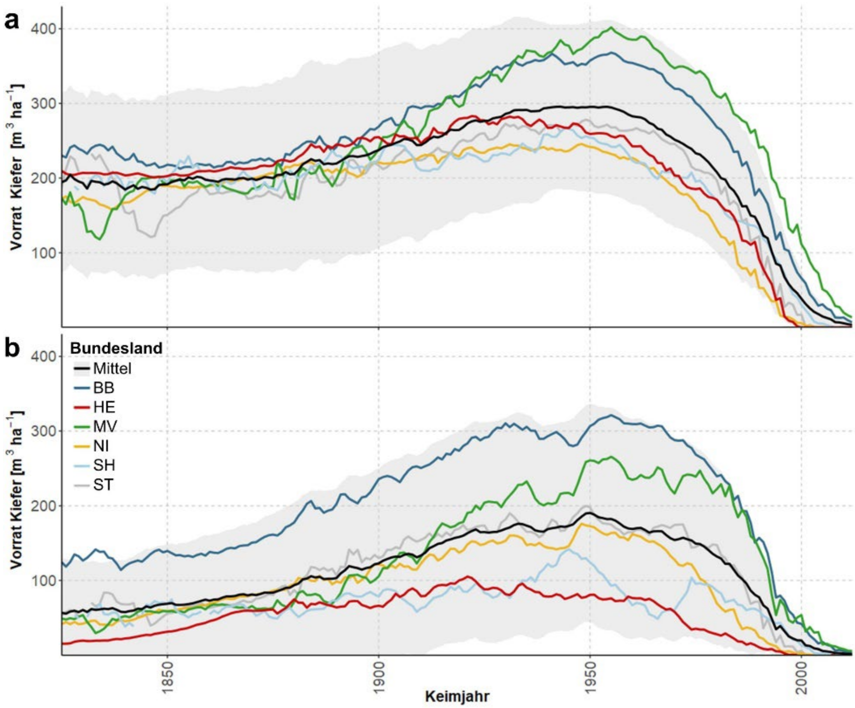


Abb. 1.3 Mittlere relative Derbholzvorräte der Kiefer **a)** je Anteilsfläche und **b)** je Bestandesfläche in den Staatsforsten Norddeutschlands (Besitz der Bundes- und Landesforstbetriebe) nach Keimjahr gemäß Forsteinrichtung (Stand 2022, Daten bis 2012, graues Band: Standardabweichung). Brandenburg: BB, Hessen: HE, Mecklenburg-Vorpommern: MV, Niedersachsen: NI, Schleswig-Holstein: SH, Sachsen-Anhalt: ST

Diese Konzepte halten Kiefernbestände je *Anteilsfläche* (Abb. 1.3a; berücksichtigt Bestockungsgrade) in den östlichen Bundesländern wesentlich dichter, als in den Westlichen. Wird die gesamte *Bestandesflä-*

che betrachtet (Abb. 1.3b; berücksichtigt Bestockungsgrad und Mischungsanteil), werden ab dem Alter 75 (Keimjahr 1950) erntebedingt abnehmende Vorräte deutlich. Nimmt man an, dass Bestände vor 1950 mit ähnlichen Mischungsanteilen begründet wurden, wie danach, zeigt der Abfall der Kurven (Pretzsch 2019b, p. 382) die zeitliche Abfolge und Intensität der Ernte an, die im fortgeschrittenen Alter bei etwa 40 m^3 ha^{-1} verbleibt. Zusammen mit Abb. 1.1a wird die geringe Flächenrelevanz von Beständen über 150 Jahren Alter (Keimjahr 1875) deutlich.

Waldentwicklungsgang in Prognosen

Langfristige Entscheidungen für die Zukunft, wie z.B. die Wahl von Baumarten oder Waldbauverfahren, hängen von der Antizipation unsicherer Veränderungen ab. Dabei wird die Unsicherheit definiert als „begrenztes Wissen über zukünftige, vergangene oder aktuelle Ereignisse" (Hoogstra & Schanz 2008; Marchau et al. 2019, p. 2). In der Entscheidungsfindung beschreibt sie die Diskrepanz zwischen verfügbarem und benötigtem Wissen für eine optimale Wahl. Die Kiefernwirtschaft in Zentraleuropa war stets mit einem unsicheren Betriebserfolg verbunden (Köstler 1944); im globalen Wandel nimmt diese Unsicherheit zu.

Veränderte Umweltbedingungen, die vom Normalzustand abweichen und auf Ökosysteme einwirken, gelten als Störungen (Rykiel 1985; Pretzsch 2019b, p. 574; Wohlgemuth et al. 2019, p. 13). Ihr Zusammenwirken als räumlich-zeitliche Dynamik einer Landschaft über einen längeren Zeitraum wird als Störungsregime bezeichnet (Wohlgemuth et al. 2019, p. 24). Grundsätzlich wird zwischen natürlichen und anthropogenen Störungen unterschieden (Wohlgemuth et al. 2019, p. 273), wobei die Waldbewirtschaftung als letztere eingeordnet wird. Störungen können zu irregulärer Mortalität führen (biotische und abiotische Kausalfaktoren), während (natürliche) reguläre Mortalität das dichte- und altersbedingte Ausscheiden beinhaltet. Forstökonomisch interpretiert man Störungen und Mortalität als Schäden (Möllmann & Möhring 2017). Als Stress bezeichnet man subletale Störungen, die wachstumsseitig positive oder negative Reaktionen hervorrufen (Rykiel 1985; White & Jentsch 2001, p.

401). Sterblichkeit kann in Mortalitätsmodellen abgebildet werden. Im Folgenden wird der komplementäre Begriff *Überlebensmodell* genutzt.

Zur Vereinfachung wird im Rahmen dieser Arbeit der Begriff *gestört* verwendet, wenn natürliche Störungen zu irregulärer Mortalität führen, und *ungestört*, wenn solche ausbleiben. Forstliche Eingriffe mit Holznutzungsabsicht (Nutzung) werden sprachlich in Pflege (Vornutzung, Durchforstung) oder Ernte (Hauptnutzung) eingeteilt.

Sowohl die Begriffe *Störung* als auch *Mortalität* können räumlich (Größe/ Ausdehnung, Form, Selektivität), zeitlich (Frequenz, Wiederkehr, Saisonalität), stochastisch (Wahrscheinlichkeit) und hinsichtlich ihrer Magnitude (Intensität, Effekt) quantifiziert werden (Staupendahl 2011; Wohlgemuth et al. 2019, p. 26). Sie werden bei Wohlgemuth et al. (2019, p. 276) weiter typisiert. In der Regel sind die Effekte von Störungen mit hohem Schweregrad (*severity*; Synonym: Effektstärke, Schadausmaß) räumlich diskret, während niedrige Schweregrade eine diffuse Ausdehnung zeigen (White & Jentsch 2001, p. 410). Die Fähigkeit eines Ökosystems, Veränderungen zu puffern, Strukturen und Prozesse beizubehalten sowie zum Ursprungzustand zurückzukehren, wird als Resilienz bezeichnet (Seidl 2014).

Generell werden der Klimawandel und die durch ihn bedingten Störungen als eine Quelle *tiefer Unsicherheit* aufgeführt (Marchau et al. 2019, p. 3), deren Auswirkung auf eine Zielstellung in Risiken mündet (Wohlgemuth et al. 2019, p. 337).

Angesichts eines Problems, wie zum Beispiel der Aussage über eine zukünftige Waldentwicklung unter Unsicherheit, erfordert dessen Lösung eine holistische Betrachtung von Ausgangszustand, Regelungsziel, Einflussgrößen und Handlungsalternativen. Zu diesem Zweck ist das Planungsinstrument des Waldentwicklungstyps (WET; *forest development type*; etwa synonym: Waldentwicklungsziel, Betriebszieltyp) besonders geeignet, die planmäßige Waldentwicklung zu quantifizieren (Knocke et al. 2023b).

Zur Prognose erlangen softwarebasierte *Decision Support Systems* (DSS) an Bedeutung, die handlungsrelevante Informationen ermitteln und aufbereiten (Pretzsch 2019b, p. 468). Indem Handlungsalternativen quantifiziert werden, können forstliche Managementmodelle abgebildet werden. Zur Prognose von Waldentwicklung und Störungen lassen sich drei Modellkonzepte unterscheiden (Wohlgemuth et al. 2019, p. 345): 1. Statistische, 2. statisch-prozessbasierte und 3. dynamisch-prozessbasierte. Während erstere auf Baum- und Bestandesebene am weitesten verbreitet sind (Staupendahl & Zucchini 2010; Pretzsch 2019b, p. 472), liefern sie nur eine mathematische Approximation von beobachteten Daten, z.B. auf Basis von Regressionsanalysen, aber bilden keine Prozesse ab. Damit lassen sie in ihrer inneren Struktur Kausalzusammenhänge (Interaktionen) außer Acht und werden auch als empirische oder black-box Modelle bezeichnet (Pretzsch et al. 2017, p. 392). Gegenwärtig lassen sich für Norddeutschland folgende statistische Waldentwicklungsmodelle unterscheiden:

Einerseits sind dies *top-down* Ansätze, die das Wachstum mithilfe von DSS fortschreiben, einen Waldumbau jedoch vernachlässigen oder nur über globale oder potenzielle Steuerungsparameter implementieren. Die letzte *Waldentwicklungs- und Holzaufkommensmodellierung* (WEHAM) der Bundesregierung (BMEL 2016; Bösch et al. 2016) basiert z.B. auf einem modellierten Einzelbaumwachstum der dritten Bundeswaldinventur (BWI[3]). Sie berücksichtigt keinen Waldumbau (Spellmann & Fischer 2016) und keine irreguläre (biotische/ abiotische) Mortalität. Ausgehend von den BWI-Trakten wird der Baumartenwechsel gelegentlich anhand der Änderungsrate zwischen verschiedenen Inventurzeitpunkten festgemacht (Rock et al. 2025), wobei Rosenkranz et al. (2023) auch eine irreguläre Mortalität nach Brandl et al. (2020) berücksichtigen. Diese *top-down* Prognoseansätze bilden den Waldumbau folglich nicht oder vereinfacht ab.

Andererseits gibt es einen *bottom-up* Prognoseansatz, der das Einzelbaumwachstum mithilfe von DSS in die Zukunft fortschreibt. Die langfristige Waldentwicklung kann hier durch lokale Steuerungsparameter auf

Basis der tatsächlich verwendeten WET gesteuert werden. Jedoch sind zur Simulation einer Waldentwicklung auf standörtlicher Grundlage detaillierte Angaben zur Nährstoff- und Wasserversorgung nötig, die für Deutschland nur an den 8 × 8 km Trakten eines damit entdichteten BWI-Netzes vorliegen (Benning et al. 2019). Nicht für die gesamte Waldfläche Norddeutschlands, sondern für Modellregionen, haben Albert et al. (2016) eine Waldentwicklung auf standörtlicher Grundlage mittels der *WaldPlaner*-Software auf Basis von TreeGrOSS (Hansen & Nagel 2014) prognostiziert. Dieser *bottom-up* Prognoseansatz ist folglich geeignet, den inzwischen klimawandelangepassten *business-as-usual* Waldumbau detailliert abzubilden und wird in der vorliegenden Arbeit weiterentwickelt.

Durch eine Analyse von Prognoseergebnissen der DSS können Rückschlüsse für das forstliche Management gezogen werden und zur verbesserten, waldbaulichen Regelung dienen.

1.2.2. Waldumbautechnik und Versuchswesen

Der *Waldbau auf ökologischer Grundlage* und das ihm maßgeblich zugrundeliegende Dauerwaldkonzept unter fortwährender Beibehaltung des Waldinnenklimas (*close-to-nature forestry*) sind das Leitprinzip (*cf.* Tab. 7.1) der gegenwärtigen Waldbehandlung in den norddeutschen Staatsforstverwaltungen (Pommerening & Murphy 2004; Wagner & Huth 2010, p. 13; Bartsch et al. 2020, p. 568). Als Grundlage für die spätere Handlungsempfehlung (Kap. 8.1) zum Umgang mit dem Mortalitätsrisiko und den unausgewogenen Altersstrukturen werden die Genese und konkrete Techniken des ökologischen Waldumbaus in Kiefernwäldern anhand relevanter Verjüngungsformen dargestellt. Eine weitere begriffliche Präzisierung erfolgt im Glossar (Kap. 1.4).

Genese und Technik

Der Bestockungswechsel ist in der Forstpraxis schon lange als ein Bestandesziel bekannt (Junack 1924; Thomasius 1995) und wurde historisch z.B. als Unterbau (Danckelmann 1881) oder Umwandlung (Cotta 1817, p. 84) praktiziert und bezeichnet. Wahrscheinlich benutzte Köstler (1952, p. 9) den Begriff *Waldumbau* im Jahr 1952 zuerst und verstand darunter „die Rückumwandlung [als] Umbau [von] nur mangelhaft standortstauglichen Bestände[n]". Heute umfasst der Begriff auch die Überführung von „gefährdeten Beständen in standortsgemäße und risikoärmere Bestände, häufig von Reinbeständen in Mischbestände" (Bartsch et al. 2020, p. 577). Der Waldumbau in Deutschland zielt also auf eine Verbesserung der ökologischen, ökonomischen und sozialen Funktionalität ab. Weil häufig standortsgemäße Baumarten eingebracht, das natürliche Störungsregime emuliert und komplexere Strukturen geschaffen werden, bezeichnet man ihn auch als ökologischen Waldumbau (*ecological forest restoration*). Dabei muss nicht auf ausschließlich potenziell natürliche Baumarten abgezielt werden (Dumroese et al. 2015; Stanturf 2016, p. 521).

Vor dem Hintergrund des Klimawandels entwickelt sich dieser Begriff zu „*klimaangepasster Waldumbau*" (Böckmann et al. 2019, p. 7). Eine bestockungs- und strukturgleiche Wiederaufforstung nach der Ernte des Vorbestandes genügt den Definitionen eines Waldumbaus nicht (Baum et al. 2021, p. 14). Zum Wiederherstellungs- bzw. Umbauprozess zählen Stanturf et al. (2019) auch die im Falle von Deutschland oft über 100 Jahre vorgelagerte Erstaufforstung degradierter Standorte zu sekundären Kiefern-Forsten (Mayer 1992, p. 276). Der Begriff *forest conversion* (z.B. Kint et al. 2009; Vrška et al. 2017), der manchmal synonym zu *forest restoration* verwendet wird, beschreibt heute eher die Umwandlung hin zu anderen Landnutzungsformen (Stanturf 2016, p. 5).

Bereits 1948 forderte man in Norddeutschland von staatlicher Seite, dass im Rahmen des zu leistenden „gewaltigen Aufbauwerkes [...] ertragreiche, den standörtlichen Anforderungen [...] entsprechende, krisenfeste Waldungen" entstehen sollen (Back 1949, p. V). Diese Generationenaufgabe (WBW 2021, p. 125) bleibt jedoch seither unvollendet und könnte sich bis nach 2050 erstrecken (Bolte et al. 2021), womit sie faktisch zu einer Jahrhundertaufgabe wird. Gleichzeitig wurden zwischen 2012 und 2022 auf 43 % der Waldfläche Deutschlands keine Bäume entnommen (BMEL 2024, p. 20), was die unsichere Bewirtschaftungspraxis und extensiven Gestaltungschancen (Pretzsch 2019b, p. 398) der Forstwirtschaft verdeutlicht.

Wälder werden durch eine Vielzahl von Faktoren geformt (Wohlgemuth et al. 2019, p. 273). Forstliche Maßnahmen sind Bestandteil verschiedener Waldbausysteme (Synonyme: Betriebsarten, Betriebssysteme; Dengler 1944, p. 496). Sie beinhalten umfassende Konzepte zur Waldbewirtschaftung, die „Art, Stärke und Turnus der Baumentnahme [...] zum Zweck der Pflege, Holzernte und Etablierung der nachfolgenden Baumgeneration" enthalten (Wohlgemuth et al. 2019, p. 279). Für die Kiefernwirtschaft in Norddeutschland relevante Systeme weisen folgende Bestandteile auf:

Innerhalb des Hochwaldes wird zwischen den Betriebsformen (Bartsch et al. 2020, p. 491) Altersklassenwald (Synonym: Schlagweiser Hochwald, *age class forest*) und Dauerwald (*continuous cover forestry*) unterschieden. Diese Formen lassen sich mit Blick auf den einzelnen Bestand auch als gleichaltrige oder ungleichaltrige Systeme bezeichnen (Aszalós et al. 2022). Unter Dauerwald versteht man bei der Hochwaldbewirtschaftung der Kiefer eine schlagfreie Betriebsform, die Möller (1920, p. 41) durch vier Grundsätze definiert: 1. „Stetigkeit des Waldwesens", 2. Vorrang „natürliche[r] Verjüngung", 3. „ stammweise Holzernte" und 4. „hohes Zuwachsprozent". Damit ist die Anlage von Freiflächen mit Ausnahme von Lücken und Löchern ausgeschlossen. Es werden gemischte, ungleichmäßige sowie ungleichaltrige Strukturen angestrebt (Möller 1922, p. 55).

Insbesondere die Nebenformen der verschiedenen Verjüngungsformen bestimmen – abhängig von Schirmausdehnung, -dichte und -dauer – inwieweit die betreffenden Waldbauverfahren (Tab. 1.1) eher dem Alterklassenwald oder dem Dauerwald zuzuordnen sind (s. Kap. 1.4). Hinsichtlich des Überschirmungszeitraumes wird im Waldumbau zwischen einer kurzfristigen (Bestockungs-) Umwandlung und einer langfristigen Überführung der Bestockung unterschieden (Thomasius 1995, p. 3). Die Überführung von Altersklassen- in Dauerwälder setzt zwingend eine zeitlich ausgedehnte Vorratsreduzierung voraus. Zum Beispiel kann sich Verjüngung in mehreren Kohorten etablieren, wenn Plenter- oder Strukturdurchforstungen bereits ab dem Alter 50 beginnen (Pretzsch 2019b, p. 420;

Bartsch et al. 2020, p. 449). Es erfolgt somit eine „permanente Begrenzung des stehenden Vorrats auf ein Niveau, das deutlich unter dem Endvorrat im Altersklassenwald liegt“ (Pretzsch 2019b, p. 417) sowie eine Abkürzung des Produktionszeitraumes (Mayer 1992, p. 274). Abb. 1.1b und Abb. 1.3b zeigen rezent geringere Kiefernvorräte für Bestände, die zwischen 1930 und 1950 begründet wurden. Dafür ursächlich könnte zum einen der während der Vorkriegszeit verpflichtend praktizierte Dauerwald (Steinsiek 1995) oder zum anderen eine beginnende Vorratsabsenkung sein, die zum Waldumbau bzw. zur Überführung in Dauerwald-Strukturen vonnöten ist.

Tab. 1.1 Für die Kiefernbewirtschaftung in Norddeutschland (historisch) relevanten Nebenformen der gleichaltrigen und ungleichaltrigen Waldbauverfahren (Erdmann 1920; Baader 1941; Dengler 1944, p. 539; Wiedemann 1948, p. 261; Tschermak 1950, p. 381; Mayer 1992, p. 273; Bartsch et al. 2020, p. 548)

Dauerwald	**Altersklassenwald**
Schirm- und Femelschlagformen	**Kahl-, Saum- und Schirmschlagformen**
- Überhaltbetrieb (dichter/ horstweiser/ Gruppenüberhalt, Fokus: Unterstand)	- Überhaltbetrieb (Einzelüberhalt, Fokus: Unterstand) - Samenschlagbetrieb/ Samenbaumverfahren (rasche Oberstand-Räumung, Fokus: Unterstand)
- Lichtungsbetrieb (Fokus: Ober- *oder* Unterstand)	- Lichtungsbetrieb (Fokus: Oberstand) - Unterbaubetrieb (Fokus: Oberstand)
- Plenter-(wald-)betrieb	- zweihiebiger/ -altriger Hochwaldbetrieb (Fokus: Ober- *und* Unterstand)

Für die Bewirtschaftung von Kiefernbeständen in Norddeutschland hat der Schirmschlag derzeit die größte Bedeutung (Bartsch et al. 2020, p. 518). Nach Bodenbearbeitung kann die Kiefer kostengünstig naturverjüngt werden und auch Kunstverjüngungen des Folgebestandes ergänzen (Heinsdorf 1994; Petersen 2001). Die Verjüngungsform des Schirmschlags zählt jedoch – wenn die Nebenform des nur kurzfristig ungleichaltrigen Samenbaumverfahrens von unter 50 - 60 Mutterbäumen ha^{-1} gewählt wird – zur Betriebsform des Altersklassenwaldes (Wagner 2008; Nyland et al. 2016, p. 332).

Besondere Bedeutung kommt bei der Ernte des Vorbestandes seinem Unterstand zu, der bei ausreichender Dichte und Qualität als Nachwuchs in die Folgegeneration übernommen wird (Bartsch et al. 2020, p. 186). Dies setzt voraus, dass die Schirmstellung des Vorbestandes durch Zielstärkennutzung oder Anhieb (Femelschlag) entsteht und nicht vergleichbar zu einer niederdurchforstungsartigen Lichtung umgesetzt wird (VDFV 1902; Mayer 1992, p. 270; Bartsch et al. 2020, p. 416), was den Unterstand räumen würde.

Waldbauliche Implementierung auf Versuchsbasis

Das Forstwesen plant und regelt zugunsten einer „späten Nachwelt", wie schon Schiller festgestellt haben soll (König 1814). Waldbauliche Regelung bezeichnet nach Pretzsch (2019b, p. 376) Maßnahmen zur zielgerichteten Entwicklung von Beständen. Dabei spielt das forstliche Versuchswesen eine federführende Rolle: Die quantitative Formulierung von Waldbaukonzepten konnte erst durch Versuche begonnen werden und ist somit heute Grundlage für Regelung, Prognose und Planung (Spellmann et al. 1999; Hansen & Nagel 2014; Albert et al. 2024). Viele Versuchsflächenzeitreihen zeichnen Entwicklungen ab, die auch die simulierte Waldentwicklung dieser Arbeit vorwegnehmen. Im Rahmen natürlicher Sukzession und unter Kontrolle des Wildverbisses (Seliger et al. 2024) entwickeln sich norddeutsche Kiefernwälder in Buchen- oder Birken- und Eichen-dominierte Waldgesellschaften fort (Zerbe 2002; Kint et al. 2004).

Eine Waldentwicklung unter *expliziter* Absicht eines Baumartenwechsels (Waldumbau-Versuche) wurde in Experimenten erst seit den 1980er Jahren systematisch untersucht (Irrgang 1999; Rumpf & Petersen 2008). Die ersten wissenschaftlichen Versuche im Sinne einer naturnahen Forstwirtschaft datieren in Norddeutschland auf Veranlassung Schwappachs jedoch schon kurz *vor* dem Ersten Weltkrieg (Schwappach 1930). Öffentliche Aufmerksamkeit und Eingang in das Versuchswesen erlangten naturnahe Ansätze z.B. durch Erdmann (1920, 1926) und Möller (1920, 1922). In diesem Zeitgeist wurden Ertragsversuche bundesstaatübergreifend als Wuchs- und Zeitreihen konzipiert und der Waldumbau wurde *implizit* untersucht, indem (entstehende) Mischbestände Gegenstand der Forschung wurden (Bauhus et al. 2017, p. 13). Zudem wurden Hiebsarten in ihrem Einfluss auf die Produktivität auf verschiedenen Standorten verglichen. Innerhalb der Betriebsform des Altersklassenwaldes wurde z.B. der zweihiebige Kiefern-Hochwald (s. Abb. 1.4c, d, g) mit gemischter Vorausverjüngung untersucht, der mittlerweile durch eine adaptive Versuchssteuerung nicht – wie ursprünglich geplant – zu einem Altersklassenwald zurückkehrt (Erdmann 1920), sondern zu einem strukturierten, ungleichaltrigen Mischbestand im Dauerwaldbetrieb umgebaut wurde. Somit konnte die „späte Nachwelt" vom Antizipieren waldbaulicher Fragestellungen profitieren (Bauhus et al. 2017, p. 14). Insbesondere zur Waldentwicklung von Kiefern-Versuchsflächen mahnt Wiedemann (1948, p. 165) vor einer Vielzahl störungsbedingt vernichteter Bestände, die oftmals zur Aufgabe des Experimentes führten (s. Abb. 1.4b, c, e).

Die nachfolgenden, perspektivgleichen, historischen und rezenten Fotopaare verdeutlichen die z.T. störungsbeeinflusste Waldentwicklung auf (ehemaligen) Versuchsflächen der Nordwestdeutschen Forstlichen Versuchsanstalt und ihrer Vorgängerinstitutionen. Sie zeigen Entwicklungsgänge von Kiefern-Reinbeständen hin zu Mischbeständen mit führenden Licht- (Abb. 1.4a, b) oder Schattbaumarten (Abb. 1.4c, d, g) bzw. ohne gezielte Einleitung von Verjüngungsverfahren (Abb. 1.4e). Die Entwicklung eines gleichaltrigen Kiefern-Mischbestandes ist in Abb. 1.4f dargestellt. Die Fotopaare umspannen Zeiträume von 30 bis hin zu 92 Jahren und demonstrieren einen Überführungsbeginn (Ernte-Start) von 70 bis hin zu 161 Jahren. Der Versuch Serno 133 wurde von 1876 bis 2007 am längsten beobachtet.

Anhand der Bilderpaare werden im Fall des zweihiebigen Hochwaldes frühe Ernte-Startzeitpunkte deutlich (Abb. 1.4c, d, g; Ernte-Start zwischen 70 und 80 Jahren). Dagegen zeigt sich insbesondere in Abb. 1.4b und e – beides regulär bewirtschaftete Flächen außerhalb des Versuchswesens – ein verzögerter Waldumbau. Angesichts hoher Zielstärkenanteile, potenzieller Entwertung und geringer Stabilität bei plötzlicher Auflichtung lässt sich die Überführung vermutlich nur als Umwandlung mit einer kurzen Überschirmung vollziehen (Ernte-Start mit 133 und nach 157 Jahren). Die rasche Nachlichtung steht jedoch den naturnahen Bestrebungen nach Ungleichmäßigkeit und Ungleichaltrigkeit entgegen (*cf.* Tab. 7.1).

Mit „→" markierte Bäume stellen identische Individuen dar. Die Bäume genau dieser Versuchsflächen fanden Eingang in Ertragstafeln (Bonnemann 1939; Wiedemann 1943; Lembcke et al. 1975; Albert et al. 2024) und Prognosesoftware (Hansen & Nagel 2014).

Die Chronosequenz der Versuchsflächen steht exemplarisch für die simulierte Waldentwicklung dieser Arbeit.


35
36
a1994


36
a2024


233
270
316
b1955


b2024

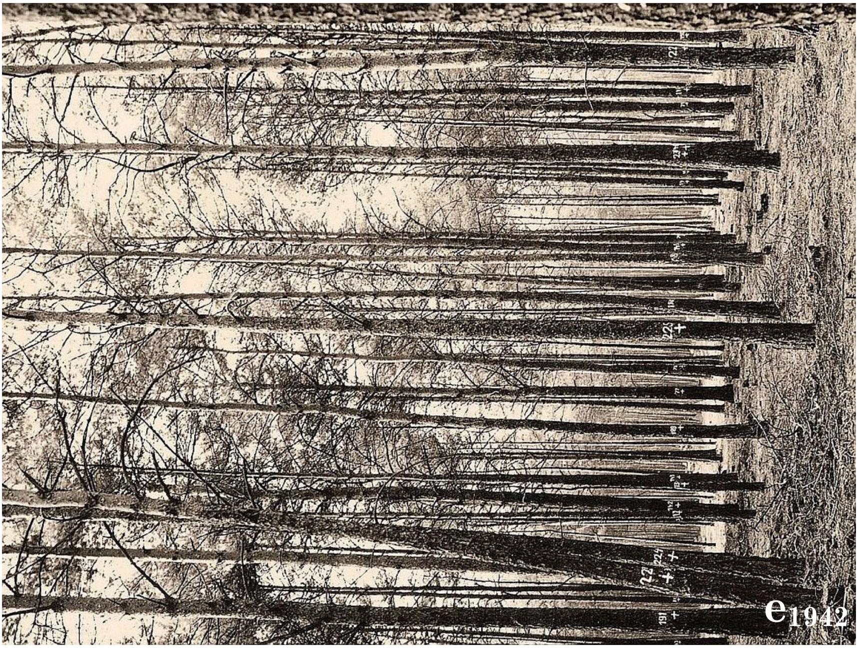

c1936


c2024


d2024


d1941


e2024


e1942


f2024


f1938


g1936


g2024

Abb. 1.4 Versuche der Nordwestdeutschen Forstlichen Versuchsanstalt und ihrer Vorgängerinstitutionen mit Waldumbau-Bezug (Angaben zu Kiefer und Oberstand): Perspektivgleiches Bildpaar der Versuche **a)** **Fuhrberg** 278[I] (Werte 1994/ 2024; Alter: 110/ 140 a, Ernte-Start 1988 (104 a), L I-Grad/ ZSN/ ADf, SI: 28,6/ 23,3 m, G: 16,3/ 3,1 $m^2\ ha^{-1}$, M: 100/100 %, aktiv 1988-2023, Aufgabegrund: Zweckerfüllung)[1], **b)** **Anhalt** 1240[I] (Serno 133; Werte 1955/ 2024; Alter: 81/ 150 a, Ernte-Start 2007 (133 a), A-Grad/ ZSN/ ADf, SI: 25,1/ 28,1 m, G: 32,6/7,0 $m^2\ ha^{-1}$, M: 100/ 91 %, aktiv 1876-2007, Aufgabegrund: Sturm- & Käferschäden, Parzellengröße)[2], **c)** **Erdmannshausen** 106[I/II] (Nienburg 2106a; Werte 1938/ 2024; Alter: 109/ 197 a, Ernte-Start 1907 (80 a), C-Grad/ ZSN, SI: 23,6/ 26,4 m, G: 19,7/ 0,5 $m^2\ ha^{-1}$, M: 100/ 3 %, aktiv 1928-1981, Aufgabegrund: Sturmschäden)[3], **d)** **Nienburg** 2028[II] (Erdmannshausen 28; Werte 1941/ 2024; Alter: 96/ 179 a, Ernte-Start 1928 (71 a), D-Grad/ ZSN, SI: 23,4/ 26,2 m, G: 20,2/ 5,1 $m^2\ ha^{-1}$, M: 100/ 19,1 %, aktiv 1928-heute)[4], **e)** **Serno** 138[V] (Anhalt 1244a[2]; Werte 1942/ 2024; Alter: 75/ 157 a, kein Ernte-Start, B-Grad/ ADf, SI: 24,0/ 22,0 m, G: 23,4/ 17 $m^2\ ha^{-1}$, M: 100/ 100 %, aktiv 1929-1951, Aufgabegrund: Sterbeloch, Brandschäden)[5], **f)** **Rotenburg** (Fulda) 1245[I] (Nentershausen 45; Werte 1938/ 2024; Alter: 94/ 180 a, Ernte-Start 2005 (161 a), D-Grad/ ZSN, SI: 27,8/ 29,3 m, G: 14,9/ 4,5 $m^2\ ha^{-1}$, M: 45,5/ 20,6 %, aktiv 1914-heute)[6], **g)** **Nienburg** 2023[I] (Erdmannshausen 23; Werte 1936/ 2024; Alter: 87/ 175 a, Ernte-Start 1919 (70 a), L II-/ E-Grad/ ZSN, SI: 18,7/ 26,2 m, G 3,8/ 1,7 $m^2\ ha^{-1}$, M: 100/ 7,5 %, aktiv 1936-heute)[7] gemäß Versuchsakten (NW-FVA 2025). Rezente Fotos: Eigene Aufnahmen. Zur Versachlichung wurden Personen in den Bildern mithilfe der KI-Funktion der Software *Adobe Photoshop* ® entfernt (Behandlung über die Zeit als Grad nach VDFV (1902), a: annum, ADf: Auslesedurchforstung bzw. -läuterung, G: Bestandesgrundfläche, M: Grundflächenmischungsanteil, SI: *site index* als absolute Weise'sche Oberhöhenbonität im Alter 100, ZSN: Zielstärkennutzung).

Veröffentlichungen vgl. [1]Petersen (2001), [2]Hundeluft 133 bei Wiedemann (1948, p. 84)/ Serno 79 bei Schwappach (1908, p. 126, 1915, p. 68), Bernhardt (1880, p. 258) und Serno 133 bei Dittmar (1992), [3]Erdmann (1920), [4]Erdmannshausen 28 bei Wiedemann (1948, p. 285), [5]Serno 138 bei Dittmar (1992), [6]Nentershausen 45 bei Schwappach (1930, p. 339) und Bonnemann (1939), [7]Erdmannshausen 23 bei Dengler (1944, p. 547)

1.3. Kernbotschaften

→ norddeutsche Kiefernwälder wurden maßgeblich durch Störungsevents, wie z.B. im Kontext des Zweiten Weltkrieges, beeinflusst

→ während die Anbaufläche junger Kiefernwälder in Norddeutschland wieder zunimmt, steht die Forschung der Kiefernanbaueignung uneinheitlich gegenüber

→ zur Prognose von Waldentwicklung und (Stark-) Holzaufkommen gab es bislang keine repräsentativen bottom-up Ansätze, die einen klimaangepassten Waldumbau auf standörtlicher Grundlage und Berücksichtigung von Störungen durch WET abbilden

→ zwischen den westlichen und östlichen Staatsforstverwaltungen Norddeutschlands unterscheiden sich die Naturalausstattungen sowie Waldbaukonzepte der Kiefer

→ eine naturnahe Bewirtschaftung hängt maßgeblich von der Nebenform der Verjüngungsverfahren ab, die die Ungleichaltrigkeit und -mäßigkeit im Folgebestand beeinflusst

1.4. Glossar

Im Hochwald unterscheidet man zwischen <u>Verjüngungsformen</u>, die Altbestände in ihrer Überschirmung (Dichte und Hiebsformen) gleichmäßig – wie bei Kahlschlag, Saumschlag und Schirmschlag – oder ungleichmäßig – wie bei Femelschlag und kombinierten Verjüngungsformen – behandeln. Diese Wirtschaftsverfahren (<u>Waldbauverfahren</u>; Dengler 1944, p. 269) werden auch Hiebsarten (Wagner 2008) genannt und nutzen den Bestand bei Erreichen einer bestandes- oder baumweisen Reifedefinition gerichtet ab. Je nach Schirmausdehnung, -dichte und -dauer lassen sich <u>Nebenformen</u> unterscheiden (Tab. 1.1), die parallel zum sich neu bildenden Hauptbestand bestehen. Dabei wachsen Überhälter (*reserves*) „in den jungen Bestand ein [..] und werden erst mit dessen Ernte genutzt" (Bartsch et al. 2020, p. 545). Andere, weiter ausgelegte Definitionen, bezeichnen Überhälter als die Bäume, die mindestens 20 % oder 50 Jahre der nächsten Bestandesgeneration (Umtrieb) erleben (Mayer 1992, p. 273; Nyland et al. 2016, p. 334).

Eine Zwischenform von Dauer- und Altersklassenwald können je nach Ungleichmäßigkeit und -altrigkeit die kombinierten Formen, wie z.B. Saumfemelschläge erlangen. Zur Kiefernbewirtschaftung dienen den Waldbauverfahren spezifische Elemente (Bartsch et al. 2020, p. 525) bzw. Techniken: Dazu gehören z.B. Erntehieb/ Zielstärkennutzung, Lochhieb, Anhieb (Gruppenanhieb), Vorhieb (Vorbereitungshieb), Besamungshieb (Samenschlag), Nachhieb (Umsäumungshieb, Nachlichtung), Aushieb, Räumungs- bzw. Kahlhieb oder Kalamitäts- und Sanitärhiebe. Zur Pflege und Erneuerung von Kiefern-Beständen zählen der Vorwald oder durch Kunstverjüngung (Verjüngungsart) die Vorausverjüngung durch Unter-, Vor- oder Nachanbau (Eisen 1859; von Pfuel 1873; Danckelmann 1881; Mortzfeldt 1896). Eine Vorausverjüngung als „Rückversicherung“ (Tschermak 1950, p. 381) kann auch durch die Verjüngungsart der Naturverjüngung erzielt werden.

Per Definition werden beim Schirmschlag durch eine „etwa gleichmäßige Auflockerung und Auflichtung des Kronendachs auf der Fläche ziemlich einheitliche ökologische Verhältnisse“ geschaffen (Bartsch et al. 2020, p. 513). Der Gruppenschirm*schlag* (LZW ST 2016, p. 33; Aszalós et al. 2022) stellt einen unregelmäßigen Schirmschlag dar und widerspricht dem Gleichmäßigkeits-Kriterium. Er ist definitorisch als Gruppenschirm*stellung* in Folge eines Femelschlags zu sehen (Mayer 1992, p. 349).

1.5. Literatur

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2. Struktur der Dissertation

2.1. Synopse

2.1.1. Verknüpfung der Artikel A1 bis A4

Die vorliegende Dissertation ist in vier Artikeln (A1-4) kumulativ aufgebaut. Innerhalb der Realwissenschaften (Kornmeier 2007, p. 14; Schurz 2014, p. 34) trägt sie holistisch dazu bei, die Waldentwicklung Norddeutschlands unter besonderer Berücksichtigung der Kiefer zu untersuchen. Hinsichtlich ihres Aufbaus ist sie hybrid verfasst. Zum einen vereint sie englischsprachige referierte Artikel mit einem deutschsprachigen Manteltext. Zum anderen hat sie hybride Fragestellungen, die auf Ursache-Wirkungs-Zusammenhänge zur Erklärung (A2) und Prognose (A1+3) sowie Ziel-Mittel-Systeme zur Problemlösung (A4) abzielen. Somit verfolgt sie theoretisch-anwendungsorientierte und technische Ziele (Chmielewicz 1994, zit. n. Kornmeier 2007, p. 25).

Als forschungsleitender Anlass der Studie diente die gegenwärtig praktizierte KSH-Produktion von über 50 cm BHD bei nur mittlerer Holzqualität. Diese zentrale Fragestellung wurde durch die Forstpraxis an die Forschung herangetragen (*cf.* Engelmeyer 1992 zit. n. Kornmeier 2007, p. 90; Pretzsch 2019b, p. 631).

Innerhalb der Realwissenschaften decken die Forstwissenschaften nach Ammer (2021) verschiedene Ebenen ab – vom Systemverständnis (1.) über die Systemerfassung und -modellierung (2.) bis hin zu Managementoptionen (3.) sowie zu Planung und Kontrolle (4.). Die vorliegende Arbeit greift diese Ebenen auf, indem sie

1. zoologisch-phytopathologische Erklärungsansätze (A2),
2. Methoden der Waldinventur und des Waldwachstums (A1, A3),
3. Aspekte des Waldbaus und Waldschutzes (A1-3), sowie
4. Elemente der Landnutzungsplanung (A4)

einbezieht. In ähnlicher Weise gliedern Sturrock et al. (2011) Management-Taktiken in Monitoring (A2), Prognose (A1, A2) und Planung (A4).

Die erste **Simulationsstudie A1** verwendet dabei ein empirisch-realistisches Prognosemodell, das induktiv aus Versuchsflächen entworfen wurde. Dieser Artikel verfolgt selber einen deduktiven Ansatz, der aus Gesetzen und Feldbeobachtungen (BWI³) eine modellbasierte Vorhersage (Pretzsch 2019b, p. 432) im Rahmen von Szenarien ableitet (Chalmers 2007, p. 46; Schurz 2014, p. 30). Insbesondere koppelt er als sekundärstatistische Studie verschiedene Einzelmodelle und Algorithmen (Abb. 2.2). Zudem werden durch deskriptiv-explikative Sätze zu umfassenden Sachverhalten (z.B. Grundflächenleitkurve, WET-Standortset) empirische Bedingungen (Abb. 3.6) – differenziert nach westlichen und östlichen Bundesländern – in die Vorhersage eingebunden (Kornmeier 2007, p. 47). Neben diesen deskriptiven Sätzen werden auch prädiktive (modellgestützte Prognosen der Waldentwicklung) sowie präskriptive (realitätsnahe Leitplanken und Regelwerke) Ansätze verfolgt (Schurz 2014, p. 71).

Hauptneuheit von A1 ist die standortsensitive, bottom-up Waldentwicklungsprognose (Abb. 2.2), die naturnahe Waldbaukonzepte emuliert (*cf.* Abb. 1.3). Subletale Störungen (positiver und negativer Stress) werden als standortsensitive Leistungsschätzung durch das *cssi*-Paket (Schick et al. 2023) auf Basis von Schmidt et al. (2020) berücksichtigt. Durch ausgetauschte Oberhöhenbonitäten, die das Wachstumsmodell im *WaldPlaner* wesentlich antreiben, kann die Waldentwicklung realitätsnah simuliert werden. Als Zielvariablen dienen A1 die Anbaufläche und der Vorrat. Weil die Baumartenwahl im Klimawandel in der Standortkartierung von BB und MV bislang statisch geschieht (Baumbach et al. 2024), wurde anhand beweglicher Klimafeuchtestufen auf Basis der Klimatischen Wasserbilanz eine eigene Dynamisierung der Standortkartierung für diese Bundesländer entwickelt (Fig. 3.3). Diese ergänzt die Dynamisierung in den übrigen Bundesländern anhand der Standortswasserbilanz (Summe aus nutzbarer Feldkapazität und Klimatischer Wasserbilanz).

Im Folgenden wird in **Modellierungsstudie A2** anhand einer Massenerfassung (*crowd sensing*) des staatlichen Waldschutz-Meldeportals (WSMP), aus Beobachtungen mit induktivem (empirisch-realistischem) Erkenntnisinteresse ein Erklärungsmuster aufgebaut, das den

Schweregrad von Störungen untersucht. Durch eine bayesianische, rationalistische Methodik kann der Schweregrad deskriptiv analysiert werden (Schurz 2014, p. 71). Als Induktionsverfahren dient eine Inferenzstatistik (Schurz 2014, p. 142). Insbesondere wurden durch eine systematische sowie rückwärtsgerichtete Literaturrecherche mögliche kausale und nicht nur korrelierte Ursache-Wirkungs-Beziehungen zur Auswahl der Kovariablen identifiziert (Kornmeier 2007, p. 117).

Der Hauptbeitrag besteht neben der erstmaligen modellgestützten Auswertung des Waldschutz-Meldeportals (WSMP) darin, dass das Störungsregime norddeutscher Kiefernwälder anhand von *crowd sensing* analysiert wird. Weil der Datensatz bislang noch nicht verwendet wurde, handelt es sich zugleich um eine primärstatistische, explorative Voruntersuchung (Kornmeier 2007, p. 79)

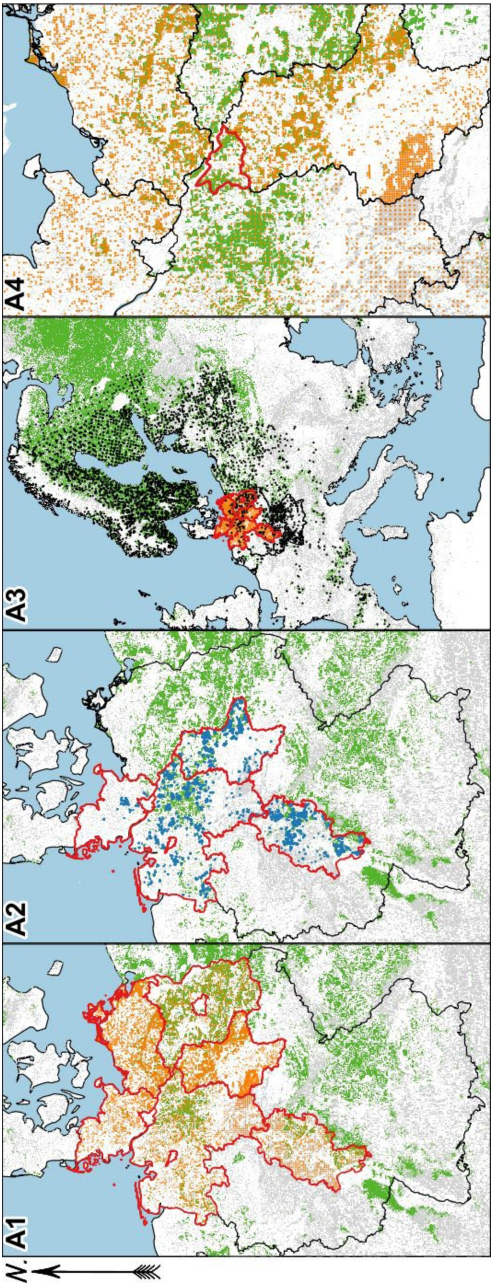
Darauf aufbauend folgt die zweite **Simulationsstudie A3**, die wiederum mit einem induktiven Prognosemodell, eine deduktive Ableitung von Erkenntnissen verfolgt. Sie erweitert die Szenarioanalyse von A1 durch empirisch-realistisch hergeleitete Überlebensmodelle und prognostiziert somit modellgestützt eine klimasensitive Waldentwicklung unter Störungseinfluss. Dahingegen wurde in A1 nur eine reguläre (dichte- und altersbedingte) Mortalität berücksichtigt. Dabei greift sie auf zwei Facetten (Schweregrade) des Störungsregimes aus A2 zurück, das in seinen extremen Ausprägungen bestandesweise oder einzelbaumweise Sterblichkeit in Wäldern beschreibt. In einem rationalistischen Ansatz, bei dem die Hazardrate nicht von statischen Größen, sondern dynamischen, klimaabhängigen Umweltfaktoren abhängt, werden durch Monte-Carlo-Simulationen neue Erkenntnisse abgeleitet.

Die für A3 ausgewählte Zeitschrift adressiert eine interdisziplinäre Leserschaft und bedient die Sichtweisen von Realwissenschaften (Forst- und Naturwissenschaften) und Formalwissenschaften (Stochastik). Der Hauptbeitrag besteht in der Kopplung von Standorts-, Wachstums-, Verjüngungs- und Überlebensmodellen. Die gegensätzlichen Schweregrade werden kontrastiert, um den entstehenden Unsicherheitsraum zu untersuchen. Als Zielvariablen dienen der Zuwachs und die Nutzungsmenge.

Zuletzt baut die **Modellierungsstudie A4** die theoretischen Betrachtungen von A1-3 durch einen technischen Ansatz aus (Kornmeier 2007, p. 30). Untersuchungsgegenstand auf empirisch-realistischer Datenbasis (Deskription von Fernerkundung und Feldaufnahmen) ist ein Praxisbeispiel, das ein abduktives Schließen per Fallbeispiel ermöglicht. Dieser technische Beitrag fokussiert sich auf den Gesamtwald des niedersächsischen Landkreises Lüchow-Dannenberg (DAN), der einerseits eine lange Tradition des Waldumbaus aufweist und andererseits in der Vergangenheit von besonders gravierenden Schadereignissen betroffen war. Es wird ein heuristisches DSS entworfen, das als „Schluss auf die beste Erklärung“ (Schurz 2014, p. 52) die zukünftige Waldentwicklung hinsichtlich ihres Umbaubedarfes priorisiert. Mittels multikriterieller Entscheidungsanalyse werden rationalistisch neue Lösungswege vorgeschlagen (Kornmeier 2007, p. 80), die Waldentwicklung nicht nur monokriteriell anhand von Zielstärkedefinitionen zu steuern (Schurz 2014, p. 54). Dabei ist eine präskriptiv-normative Bewertungsfunktion des Mortalitätsrisikos zentral, die in einer Handlungsempfehlung als kartenmäßige Triage (*sensu* Sturrock et al. 2011) mündet.

Die Hauptneuheit liegt in der räumlich-expliziten Reihung von Risikobeständen im Landschaftskontext basierend auf neuen, quelloffenen Datensätzen. Über verschiedene Skalen der Baumartenvielfalt im Oberstand hinweg gelingt es, einer Angleichung der Waldentwicklung auf Landschaftsebene entgegenzuwirken und waldbauliche Maßnahmen zu operationalisieren.

Im Einzelnen sind der Status im Begutachtungsprozess, die wesentlichen Neuerungen und die jeweiligen Autorenbeiträge in Tab. 2.1 gegenübergestellt. Die verschiedenen Kopplungen der Artikel und Teilmodelle werden in Abb. 2.2 als Flussdiagramm dargestellt.

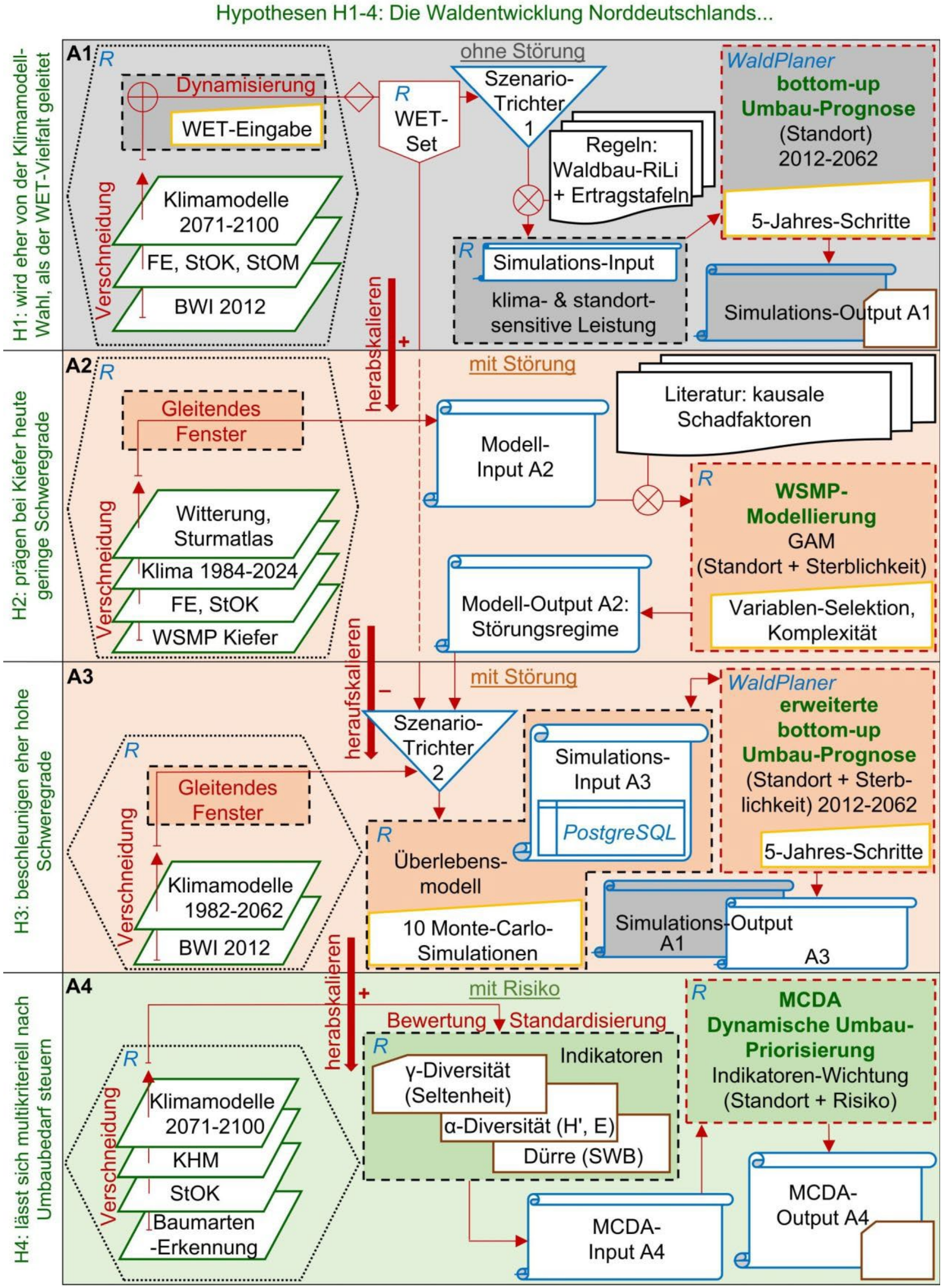


Zur Analyse von Ursache-Wirkungs-Zusammenhängen der theoretischen Waldentwicklung wird der Norddeutsche Fokus in seiner Skala variiert (Abb. 2.1): Zum einen konnte durch eine Hochskalierung mittels paneuropäischer Level-I-Aufnahmen der Waldzustandserhebung, die den Überlebensmodellen zugrunde liegen (A3), die Vorhersagegüte verbessert werden. Zum anderen konnte durch Runterskalieren im Fallbeispiel mittels Abduktion (A4) ein besonders prominenter und herausfordernder Fall der Landnutzungsplanung untersucht werden.

Abb. 2.1 Räumlicher Fokus (rot) der Artikel **A1)** (orange: BWI^3-Trakte; vTI 2015), **A2)** (blau: Meldungen Kiefer WSMP im Landeswald HE, NI, SH, ST), **A3)** (schwarz: Paneuropäische Level-I-Flächen Kiefer (ICPF 2022) in Brandl et al. (2020) für BB, HE, MV, NI, SH, ST) und **A4)** (orange: BWI^3-Trakte (vTI 2015) im Landkreis DAN), sowie Waldbedeckung (grau; Burgoin et al. 2024) mit Kiefern-Areal (grün; de Rigo et al. 2016). Grenzen nach © opendatasoft/ OGL (2019) und © GeoBasis-DE/ BKG (2022)

Tab. 2.1 Synopse der vier Artikel A1-4 dieser Dissertation. Autoren: CF Christoph Fischer, GJL Gitta J Langer, HCK Hergen C Knocke, HFH Hans F Hamkens, HSR Holger Sennhenn-Reulen, JW Jens Wiesehahn, KH Kai Husmann, MAl Matthias Albert, MAx Maximilian Axer, RVN Ralf V Nagel, TZ Thorsten Zeppenfeld. CRediT: Contributor Roles Taxonomy

#	Artikel	Status	Hauptneuheit	CRediT
1	Knocke et al. 2024. *Quo vadis* Scots pine forestry in northern Germany: How do silvicultural management and climate change determine an uncertain future? *Eur J For Res* 143(5), 1477–1497	Eingereicht: 16.9.2023, Überarbeitet: 19.4.2024, Angenommen: 15.5.2024, Veröffentlicht: 21.6.2024	Diese Simulationsstudie koppelt klimasensitives Wachstum sowie realitätsnahe Waldbau- und Planungs-Regeln, um in einem detaillierten *bottom-up* Ansatz auf standörtlicher Grundlage die ungestörte Waldentwicklung mittels der BWI³ in 50j. Szenarien zu prognostizieren.	Konzeption: HCK, RVN, MAl. Methodik: HCK. Formale Analyse: HCK, HFH, MAx, CF. Datenaufbereitung: HCK, HFH, MAx, CF. Software: HCK, JHH. Visualisierung: HCK. Erstentwurf: HCK, MAl
2	Knocke et al. Contributing to the disturbance ecology of Scots pine in Northwest Germany: Analyzing practitioners' reports on damage severities. *Trees For People* 20, 100895	Eingereicht: 6.3.2025 Überarbeitet: 1.5.2025, Angenommen: 27.5.2025 Veröffentlicht: 2.6.2025	Diese Modellierungsstudie untersucht bestandesweise die Schweregrade in Kiefernwäldern Nordwestdeutschlands und greift erstmals auf das das staatliche WSMP zurück, indem es in einem bayesianischen Ansatz eine Betaregression durchführt und kausale Einflussfaktoren identifiziert.	Konzeption, Methodik: HCK, MAl. Datenaufbereitung, Visualisierung, Untersuchung, Erstentwurf: HCK. Ressourcen: TZ. Software, Formale Analyse: HCK, HSR. Überarbeitung: HCK, MAl, MAx, GJL. Projektleitung, Finanzierung: RVN
3	Knocke et al. Hope for the best, expect the worst: Including disturbances in simulations as tree- or standwise mortality increases uncertainty for north German Scots pine forestry. *Stat Environ Res Risk Assess* 39, 5091–5113	Eingereicht: 16.8.2024, Überarbeitet: 18.1.2025, Angenommen: 10.7.2025 Veröffentlicht: 5.8.2025	Diese Simulationsstudie koppelt klimasensitives Wachstum, realitätsnahe Waldbau- und Planungs-Regeln, sowie Überlebensmodelle, um in einem detaillierten *bottom-up* Ansatz auf standörtlicher Grundlage die störungsbeeinflusste Waldentwicklung mittels der BWI³ in 50j. Szenarien zu prognostizieren.	Konzeption: HCK, RVN, MAl. Datenaufbereitung: HCK, TZ, MAx, HFH. Formale Analyse: HCK, HSR, CF, KH. Erstentwurf: HCK, MAl
4	Axer et al. Integrating Remote Sensing and Site-specific Data in Forest Planning: A Multicriteria Approach to Prioritize Forest Restoration in Central Europe	Eingereicht: 28.3.2025 *Trees For People*	Diese Modellierungsstudie priorisiert den Waldumbaubedarf mittels multikriterieller Entscheidungsanalyse. Auf Basis von Fernerkundung sowie Feldaufnahmen werden besitzartenübergreifend Indikatoren hergeleitet und hinsichtlich ihres Einflusses auf das Mortalitätsrisiko bewertet.	Konzeption, Methodik: MAx, HFH. Datenaufbereitung, Software, Formale Analyse: MAx, HFH, JW, HCK. Visualisierung: MAx, HFH, HCK. Erstentwurf: MAx, HCK. Überarbeitung: HFH, JW


Hypothesen H1-4: Die Waldentwicklung Norddeutschlands...
H1: wird eher von der Klimamodell-Wahl, als der WET-Vielfalt geleitet
A1
R
Dynamisierung
WET-Eingabe
Verschneidung
Klimamodelle 2071-2100
FE, StOK, StOM
BWI 2012
R
WET-Set
ohne Störung
Szenario-Trichter 1
Regeln: Waldbau-RiLi + Ertragstafeln
R
Simulations-Input
klima- & standort-sensitive Leistung
WaldPlaner
bottom-up
Umbau-Prognose
(Standort)
2012-2062
5-Jahres-Schritte
Simulations-Output A1
herabskalieren
+
H2: prägen bei Kiefer heute geringe Schweregrade
A2
R
Gleitendes Fenster
Verschneidung
Witterung, Sturmatlas
Klima 1984-2024
FE, StOK
WSMP Kiefer
mit Störung
Modell-Input A2
Literatur: kausale Schadfaktoren
R
WSMP-
Modellierung
GAM
(Standort + Sterblichkeit)
Variablen-Selektion, Komplexität
Modell-Output A2: Störungsregime
heraufskalieren
−
H3: beschleunigen eher hohe Schweregrade
A3
R
Gleitendes Fenster
Verschneidung
Klimamodelle 1982-2062
BWI 2012
mit Störung
Szenario-Trichter 2
R
Überlebens-modell
10 Monte-Carlo-Simulationen
Simulations-Input A3
PostgreSQL
WaldPlaner
erweiterte
bottom-up
Umbau-Prognose
(Standort + Sterb-lichkeit) 2012-2062
5-Jahres-Schritte
Simulations-Output A1
A3
herabskalieren
+
H4: lässt sich multikriteriell nach Umbaubedarf steuern
A4
R
Verschneidung
Klimamodelle 2071-2100
KHM
StOK
Baumarten -Erkennung
mit Risiko
Bewertung
Standardisierung
R
Indikatoren
γ-Diversität (Seltenheit)
α-Diversität (H', E)
Dürre (SWB)
R
MCDA
Dynamische Umbau-
Priorisierung
Indikatoren-Wichtung
(Standort + Risiko)
MCDA-Input A4
MCDA-Output A4

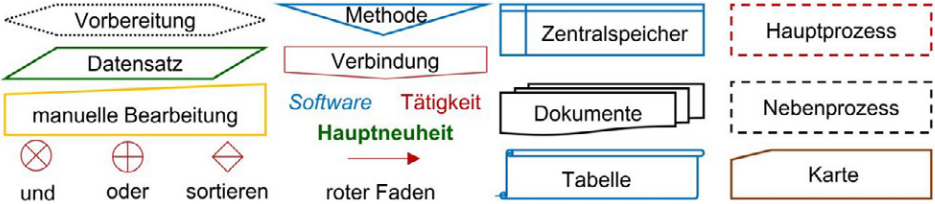


Abb. 2.2 Flussdiagramm der vier Artikel **A1) - A4)** und Hypothesen (H1-4), das Simulation, Modellierung und Priorisierung synoptisch gegenüberstellt (BWI: Bundeswaldinventur, E: Evenness, FE: Forsteinrichtung, GAM: Generalisiertes Additives Modell, H': Shannon-Index, KHM: Kronenhöhenmodell, MCDA: Multi-Criteria Decision Analysis, RiLi: Richtlinien, StOK: Standortskartierung, StOM: Standortsmodell (Schirrmeister et al. 2023), SWB: Standortswasserbilanz, WET: Waldentwicklungstyp, WSMP: Waldschutz-Meldeportal).

2.1.2. Übergreifende Hypothesen H1 bis H4

Pretzsch (2019b, p. 631) untergliedert das Schaffen und den Praxistransfer von Wissen in acht Schritte. Darunter fallen unter anderem 1. das Formulieren von Hypothesen über Einzelaspekte (hier: einzelne Forschungsfragen), die Hypothesenprüfung durch 2. Falsifikation, Experimente, Korrelation sowie 3. nach Integration in den Gesamtkontext durch Simulation. Erst danach können Theorien, Gesetzmäßigkeiten und Regeln aufgestellt werden.

Diese Arbeit formuliert die folgenden Hypothesen H1-4, von denen H1-3 Ursache-Wirkungs-Zusammenhänge untersuchen und H4 ein Ziel-Mittel-System entwirft:

H1: „Die Waldentwicklung Norddeutschlands wird *ohne* Störung eher durch die Wahl der Klimamodelle als durch die Vielfalt von Waldentwicklungstypen beeinflusst."

Diese Hypothese überprüft für realitätsnahe Simulationen, wie der durch Leitplanken begrenzte Unsicherheitsraum unter Ausschluss von Störungen ausfällt. Sie testet, ob die Bandbreite der entscheidungsleitenden Forstplanung und der waldbaulichen Umsetzung eher durch die Klimamodellwahl (Ensemble-Ansatz) oder durch die standörtlich mögliche Vielfalt an WET beeinflusst wird. Dieser Hypothese wurde ausschnittsweise bereits von Albert et al. (2016) untersucht. Ihr liegt die Vermutung zugrunde, dass die Auswahl der Klimamodelle im Ensemble-Ansatz (Hübener et al. 2017) auf Grundlage einer konservativen (pessimistischen) Schätzung des RCP8.5-Klimamodellstranges erfolgte, die typisch für forstwissenschaftliche Annahmen ist (Möhring et al. 2021) und die die forstplanerisch-waldbauliche Handlungspluralität überprägen dürfte (Ehwald 1949). Es wird postuliert, dass der Effekt der Klimamodellwahl auf die Waldzusammensetzung im Jahr 2062 (Ergebnis der Waldentwicklung) maßgeblich ist. Innerhalb des RCP8.5 Modellstrangs, der das Pessimum einer klimatischen Zukunft zusammenfasst, werden wiederum statistische (extreme) und dynamische (gemäßigte) Regionalmodelle gewählt. Damit zielt diese These auf eine Meta-Ebene von Forstplanung und Waldbau ab. H1 wird dabei vor dem Hintergrund einer Potenzialanalyse diskutiert: Die theoretische WET-Zielerreichung (Mischungsanteile) wird im Rahmen einer Vorbildwirkung des Staatswaldes auf den Nichtstaatswald und der uneingeschränkten Umsetzung der Förderrichtlinien im Rahmen der Beratung und Betreuung des Nichtstaatswaldes vorausgesetzt (Stockmann et al. 2024). Als Indikator wird der relative Abstand der Szenarioergebnisse im Jahr 2062 zueinander beurteilt.

H2: „Die Waldentwicklung Norddeutschlands wird bei der Kiefer unter Störungseinfluss derzeit überwiegend von niedrigen Schweregraden bestimmt, deren Stärke mit zunehmender Standortsgüte ansteigt."

Diese Hypothese zielt darauf ab, für die nachfolgenden Simulationen (A3), die Relevanz von Kleinststörungen und irregulärer Mortalität von Einzelbäumen zu belegen. Intuitiv widerspricht sie prominenten Großschadereignissen (Köstler 1944; Habermann 2017; Michler & Aschenbrand 2024), die bei Kiefer aber in geringer Frequenz auftreten. Mit der Abkehr vom Altersklassenwald und zunehmender Baumartenmischung bei gleichzeitig abnehmender Möglichkeit einer Bestandesabgrenzung ist die Anwendung von Überlebensmodellen auf der Baumebene – und nicht mehr nur der Bestandesebene – wichtig. Die Formulierung dieser Hypothese setzt einen hinreichenden Zeithorizont voraus, um ein langfristiges Störungsregime abbilden zu können. Generell können Häufigkeit und Schweregrad von Störungen in den gemäßigten Zonen invers miteinander korreliert sein (Kimmins 2004, p. 540), was die Anwendung des *Frequenz-Areal-Potenzgesetzes* ermöglicht (Malamud et al. 2005).

Im Zusammenhang mit H2 soll die bereits etablierte *Stress-Gradienten-Theorie* (Callaway & Walker 1997) diskutiert werden, um den explorativen Ansatz und Vorstudiencharakter von A2 zu stützen. Als Indikator zur Prüfung von H2 dient die Korrelation von Schweregraden zu Kovariablen von Umwelteinflüssen.

H3: „Die Waldentwicklung Norddeutschlands
wird durch diskrete, hohe Schweregrade beschleunigt,
wenn sie mit diffusen, niedrigen Schweregraden kontrastiert werden.“

Störungen gelten als Katalysatoren des Wandels (Wohlgemuth et al. 2019, p. 82) und sind besonders in ihren Effekten auf die natürliche Sukzession untersucht worden (Abrams & Scott 1989; Thom et al. 2017). H3 dient dem Test, ob in Simulationen bei Modifikation des Szenariotrichters ein hoher (diskreter) Schweregrad (ganzer Bestand fällt aus) eine beschleunigte Waldentwicklung im Vergleich zu niedrigen (diffusen) Schweregraden erkennen lässt. Über alle Wiederholungen der Monte-Carlo-Simulationen hinweg kann dieselbe Überlebenswahrscheinlichkeit (hier: Hazardrate als deren 1. Ableitung) einer Fläche durch verschiedene Schweregrade (Bestandes- oder Baumschäden) abgebildet werden. Wir unterstellen zur Formulierung dieser Hypothese, dass der Kulturerfolg nach nicht-selektiven, hohen Schweregraden (gesamter Haupt- und Nebenbestand fällt aus) unmittelbar gelingt. Zur Falsifizierung von H3 ist die Stochastizität von Störungsereignissen zentral, deren Eintreten anhand von Hauptbestandesalter, Klimagrößen und z.T. anhand von Mischungsanteilen hergeleitet wird. In der Realität kann die Artenzusammensetzung von Lebensgemeinschaften durch weitere Kopplungen mit Umwelteinflüssen und zwischen Störungen beschleunigt oder verzögert werden, aber auch unbeeinflusst bleiben (Wilson & Agnew 1992). Im Rahmen der Hypothesenprüfung soll durch Simulation getestet werden, inwiefern sich das Tempo des Waldumbaus, gemessen an der relativen Baumartenzusammensetzung, bei großflächigen Störungen beschleunigt, wenn mehr als 90 % der Wälder aktiv bewirtschaftet werden. Durch Kopplungen forstlicher und natürlicher Störungen sowie dynamischer Klimaeinflüsse kann dadurch ein detailliertes Abbild der Sterblichkeit modelliert werden. Als Indikator wird die je BWI-Trakt berechnete Umbaurate im Verhältnis zu 2012 betrachtet, die west- und ostdeutsche Waldbausysteme unterscheidet.

H4: „Die Waldentwicklung Norddeutschlands lässt sich auf Grundlage der landschaftsräumlichen Vielfalt durch multikriterielle Ansätze bedarfsbezogen für den Waldumbau priorisieren."

Diese Hypothese prüft ab, ob sich die bestandesweise vorgehende und an Besitzgrenzen gebundene Forstplanung heuristisch um ein Instrument erweitern lässt, das die Landschaftsebene mehr in den Fokus rückt. Im Gegensatz zu H1-3 wird bei H4 nicht die Erklärung eines Ursache-Wirkungs-Zusammenhangs, sondern die Entwicklung eines Ziel-Mittel-Systems in den Fokus gerückt.

Die Formulierung dieser Hypothese ergibt sich aus dem Vorhandensein neuer quelloffener Datensätze (Blickensdörfer et al. 2024) sowie dem auch im Staatswald geringen Waldumbaufortschritt (Baum et al. 2021, p. 14). Weil sich dieser zu einem Abschluss des Waldumbaus bis 2050 generell vervierfachen muss (Bolte et al. 2021), sollte vor allem eine gezieltere finanzielle und fachliche Förderung in Kiefern-Hochrisikobeständen stattfinden. Dies setzt eine bedarfsbezogene Priorisierung voraus. In A1 und A3 definieren wir umbauwürdige Bestände als diejenigen, die zu 90 % je Trakt monospezifisch sind. Daher soll heuristisch der Versuch unternommen werden, die am stärksten risikogefährdeten Bestände besitzartenübergreifend im Landschaftskontext zu identifizieren, um die Bestrebungen zur Waldanpassung gezielter umzusetzen. Die Entscheidungsfindung wird dabei nicht mehr an der monokriteriellen Reifedefinition eines Baum-Zieldurchmessers, sondern unter klimatischer Unsicherheit durch multikriterielle Teilziele erweitert. Dazu gehören das Trockenstressrisiko und die Oberstands-Baumartenvielfalt auf Bestandes- und Landschaftsebene. Anhand einer Priorisierung wird ferner offengelegt, wo Maßnahmen zu ergreifen sind, um potenziell negative Feedbacks zwischen Störungen zur Dämpfung nutzen zu können (Wilson & Agnew 1992). Bei erfolgreicher Problemlösung kann dazu beigetragen werden, Bestände einer Triage zu unterziehen und innerhalb der Planungsdekade vorzuziehen, Zielstärken am unteren Rahmen anzusetzen oder im Nichtstaatswald Fördermittel prioritär zu lenken. Als Indikator wird eine hinreichend sensitive Priorisierung mit bedarfsbezogener Reihung in Kartenform betrachtet.

2.2. Kernbotschaften

→ diese Arbeit verfolgt induktive (A2), deduktive (A1, 3) und abduktive (A4) Ansätze

→ die Modellierungs- und Simulationsstudien verknüpfen als Primär- (A2) und Sekundärstatistische Arbeiten (A1, 3, 4) folgende Teilmodelle: Standorts-, Klima-, Wachstums- und Verjüngungs- sowie Überlebensmodelle

→ die klimawandelbedingte Unsicherheit wird durch verschiedene Signale des RCP 8.5 Modellstrangs dynamisch berücksichtigt und nimmt Einfluss auf Standort und Sterblichkeit der Kiefernwälder

→ Hauptneuheiten sind: 1. Die repräsentative, als *bottom-up* Prognose entworfene Waldentwicklungssimulation unter regulärer und irregulärer Mortalität bei realitätsnaher Berücksichtigung der Waldbausysteme, 2. Die erstmalige Auswertung des WSMP im Rahmen einer Modellierungsstudie

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3. Prognose ungestörter Waldentwicklung (A1)

Erschienen als Artikel A1:

„*Quo vadis* Scots pine forestry in northern Germany:
How do silvicultural management and climate change
determine an uncertain future?“

✉ HC Knocke[1,2,3], M Axer[2], HF Hamkens[2], C Fischer[2], JH Hansen[2], RV Nagel[2], M Albert[1,2] (2024)

Springer-Verlag
European Journal of Forest Research 143(5):1477-1497.
https://doi.org/10.1007/s10342-024-01701-0

Hypothese H1

*„Die Waldentwicklung Norddeutschlands
wird ohne Störung eher durch die Wahl der Klimamodelle
als durch die Vielfalt von Waldentwicklungstypen beeinflusst.“*

Gegenüber dem Originalartikel gab es redaktionelle Änderungen.

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Abstract

Scots pine is of greatest importance in northern Germany regarding its cultivation area and expected capability to perform in climate change. However, pine predominantly occurs in monocultures. Therefore, future pine forestry depends on an adaptation to climate change while improving ecological and economic forest functions. Yet future development of pine remains uncertain due to leeway in silvicultural guidelines and future climate.

This study questions: (*i*) what is the range of future pine shares under climate change and different silvicultural management in northern Germany, (*ii*) how will the current stands develop and (*iii*) what is the range of uncertainty arising from climate models and silvicultural options?

To answer these issues we (*i*) selected forest development types site- and climate-sensitively to either minimize or to maximize pine shares, (*ii*) simulated four, now practiced forest management scenarios for 50 years based on the German National Forest Inventory and (*iii*) analyzed the differences, to be interpreted as uncertainty. Novel to our approach is the site- and climate-sensitive selection of forest development types on large scales which emphasizes the contrasts of the different management guidelines.

The results show that growing stock and cultivation area will decrease even if pine is promoted in forestry. The predicted restoration rate ranges from 50 to 72 % depending on scenario and previous thinning regime.In conclusion, under the given management concepts and considering today's high proportion of old pine, restoration is alarmingly slow. Amid the rapidly changing climate, we recommend to further adjust the management guidelines to accelerate forest restoration.

Keywords *Pinus sylvestris*, uncertainty, forest restoration, simulation, thinning regime, climate change

3.1. Introduction

Forests are constantly changing. Regarding climate change, the restoration of pure towards mixed stands in the temperate zones can be seen as the most important task for today's European forestry (Mason et al. 2018; DVFFA 2019; Zerbe 2019; Graf et al. 2022). Already previously, single-species stands were far from a natural vegetation in most cases (Tüxen 1956; Hennenberg et al. 2017) and vulnerable to abiotic and biotic risks (von Sponeck 1819). The stability of recent forest communities decreases due to a warming and aridizing climate even more than ever (Hanewinkel et al. 2013; Buras and Menzel 2019; Mette et al. 2021; Hinze et al. 2023).

The so-called German *Waldumbau* from pure to mixed stands in the face of climate change has to be contextualized in the framework of forest restoration. It entails that partly degraded communities are rehabilitated i.e. converted structurally and functionality wise, towards more natural forests. It is termed Forest Landscape Restoration (FLR), if choices about scope, socioeconomic constraints, and larger forest scales are considered (Stanturf 2016, pp. 5–8). In northern Germany, Scots pine (*Pinus sylvestris* L.) is the most common tree species. It predominantly can be found in even-aged monocultures of low structural complexity (BMEL 2015).

However, among European tree spp., Scots pine naturally occurs on very adverse sites where it forms extremely rare and vulnerable habitat types, such as lichen pine forests from the Dicrano-Pinion community (Leuschner and Ellenberg 2017, pp. 581–598). Although today's key growing area of pine moved on sites with low to even medium quality, its cultivation is highly recommended expert-based even with the lowest water supply (Albert et al. 2017). Besides being the native tree species with the largest site amplitude, third-most specialized arthropods are linked to Scots pine, thus underlining its ecological importance (Brändle and Brandl 2001). Pine also meets yield and economic requirements making it one of the most relevant taxon in silviculture under climate change

(Leuschner et al. 2022; Brichta et al. 2023). However, the proportion of pine is decreasing currently, especially in young age classes, because it shows a relatively lower growth potential (Beinhofer and Knoke 2010; BMEL 2016, p. 11), low shade-tolerance in the face of currently practiced continuous cover forestry (Wagner and Huth 2010), as well as comparatively long rotations (Mason and Alía 2000; Fischer and Mölder 2017).

Currently prevailing silviculture with Scots pine in northern Germany is reviewed by Spathelf and Ammer (2015). They depict a diversification in management practices which is e.g. expressed in different forest development types (FDT). Operationalizing the long-term targets for a controlled development, FDT give a quantitative mechanism to achieve a site-specific composition of mixed stands for close-to-nature forestry (Larsen and Nielsen 2007). Although FDTs are used for goal-oriented managing, the past FLR of pine has been identified as the slowest (Fischer and Mölder 2017). FLR progress generally is influenced by many factors such as current age class distribution, site, management intensity, seedling availability, calamities, climate, economy and policies (Knoke et al. 2001; Kint et al. 2009; BMEL 2016, p. 50; Graf et al. 2022; Suárez-Muñoz et al. 2023).

The forecast of future forest composition and FLR outcome is hampered by uncertainties. Since predictions in environmental shifts on local scales remain vague, there is an uncertainty associated with the assumed climate change (Millar et al. 2007; Albert et al. 2015): For example, water supply of sites can no longer be expected to be constant over time. Thus, climate change increases uncertain growth potentials while different management i.e. silvicultural options also contribute to uncertainty (Albert et al. 2017; Bauhus 2022). This is exacerbated by model-based contradictory assessments of Scots pine suitability for cultivation when considering its vulnerability to heat in combination with drought or warm winters (Bose et al. 2020; Rehschuh and Rühr 2021; Haberstroh et al. 2022).

To our knowledge, uncertainties arising from differing silvicultural prescriptions and climate change have been poorly quantified. Precisely, because FLR is dependent on many interacting factors, future forest compositions cannot be foreseen, especially when considering timber harvesting potential. This is why in other simulation studies the initial tree spp. percentage is held constant over time, or tree spp. selection is not site specific while site changes are not considered (BMEL 2016; Oehmichen et al. 2018; Rosenkranz et al. 2023).

In this study we want to increase prediction abilities with a detailed site-specific approach. Four different silvicultural Business As Usual (BAU) scenarios are simulated in order to investigate realistic future forest composition. We focus on Scots Pine in northern Germany, using the single-tree growth simulator WaldPlaner (Hansen and Nagel 2014).

Our prescriptive modelling provides insight in the range of magnitudes and direction of possible future changes. We imitate the BAU (German multifunctional forestry; Borrass et al. 2017) when forest owners obey best practice recommendations of the State Forest Administrations as *homo economicus*. BAU silviculture guidelines were recently updated in every federal state to particularly emphasize adaptation to climate change (e.g. Thurm and Wirner 2023). While considering forest cover ceteris paribus, optimal circumstances such as fructification, seedling availability and timber marketing are assumed. We refrain from including risk i.e. potential disturbances in the simulation in order to clearly identify the effects of silviculture and climate change on forest development. However, disadvantages of persistently drier climates are accounted for in BAU tree spp. selection. Site changes and the resulting change of site index (SI) on permanent inventory plots are considered dynamically.

Different outcome of management practices has important implications in the macroeconomic sense for timber markets as well as landscape ecology. Therefore this study questions:

i. What are the guard rails of future pine shares (FDT) under climate change and leeway in management guidelines in six northern German states,

ii. How will the current forest stands develop between climate and silvicultural scenarios and

iii. What is the range of uncertainty arising from climate change and silvicultural options?

3.2. Material and Methods

In order to answer the three questions formulated above we apply a climate-sensitive algorithm to select future FDTs (Hamkens et al. 2022) and use a single-tree growth simulator supported by a climate-sensitive SI-model (Schmidt 2020) to predict forest development.

3.2.1. Material

This study encompasses the six German federal states of Brandenburg, Hesse, Mecklenburg-Western Pomerania, Lower Saxony, Schleswig-Holstein and Saxony-Anhalt under which (for further comparisons) Brandenburg, Mecklenburg-Western Pomerania and Saxony-Anhalt are considered eastern and the others as western.

This study representatively simulates forest development on plots from the third German National Forest Inventory (NFI) with available site information. This corresponds to about 86.6 % of the total forest area (4,477,858 ha) within the study region and covers mainly parts of the north German lowlands as well as the central German hill country. Elevations range between -3 and 1,141 m a.s.l. within oceanic to hemiboreal climate. In Saxony-Anhalt, on small areas also cold semi-arid and subarctic climate occurs (Beck et al. 2018). Soils with the widest possible range of water and nutrient supply have developed in the study area. A map and table indicating the harmonized soil nutrient levels (SNL) is supplemented (Fig. S3.11, Tab. S3.4). Mean annual precipitation sum ranges from 449 to 1,799 mm a^{-1} and mean annual air temperature from 3.9 to 11.5 °C within the period 1991-2020 (DWD 2022). Due to often medium to poor and rather dry soils, Scots pine has its current and natural distribution focused here (BMEL 2015; Leuschner and Ellenberg 2017). The total forest area includes about 77 % of the German Scots pine's spatial occurrence while being equivalent to only 39 % of the German forest area.

The dendrometric data with reference date 2012 was retrieved from the NFI: It is characterized by a permanent stratified one-phase systematic cluster sampling design where the strata consist of geographic regions differing in sampling density (Fig. 3.1; Kleinn et al. 2020). The sample plot (cluster, tract) comprises corners (subplots) that, if located within the forest, are surveyed. Trees with diameter at breast height (DBH) ≥ 7 cm are selected by angle-count sampling and tree regeneration is assessed on circles with radii up to 2 m. We refer to Riedel et al. (2017) for further information. In total, 17,551 subplots on 6,508 tracts with site mapping data were included in this study.

In order to supplement the NFI survey of pruning (Fig. 3.1), the NFI dendrometric data was spatially joined with information on pruning available from Forest Administration records (Tab. 3.1). Furthermore, sample plots that are located within strictly preserved forest areas (unmanaged forests, e.g. national parks) were identified using maps from Steinacker et al. (2023) where no active management was simulated (see "Simulating forest development" section).

Site and climate data

At the site-scale, the selection algorithm for FDTs and the SI model mainly depend on the edaphic factors soil moisture level (SML) and soil nutrient level (SNL). We assume constant SNL, however, water properties must be aligned with changing climate conditions.

In order to span a wide range of climate effects on tree spp. selection and forest growth, the largest possible difference of climate models actually used in north German forestry was identified. This results from a statistical and a dynamic downscaling within the ReKliEs-De core ensemble from the CMIP5-generation of seven regional models for the representative concentration pathway (RCP) 8.5 (Hübener et al. 2017).

Thus, the 'Hadley Centre Global Environment Model' (HadGEM2) and its statistical regional 'Wetterlagen-basierte Regionalisierungsmethode' (WettReg13) was chosen to represent the most extreme climate projection (Martin et al. 2011; Kreienkamp et al. 2013). Contrary to this, the global 'European Centre Earth System' model (EC-

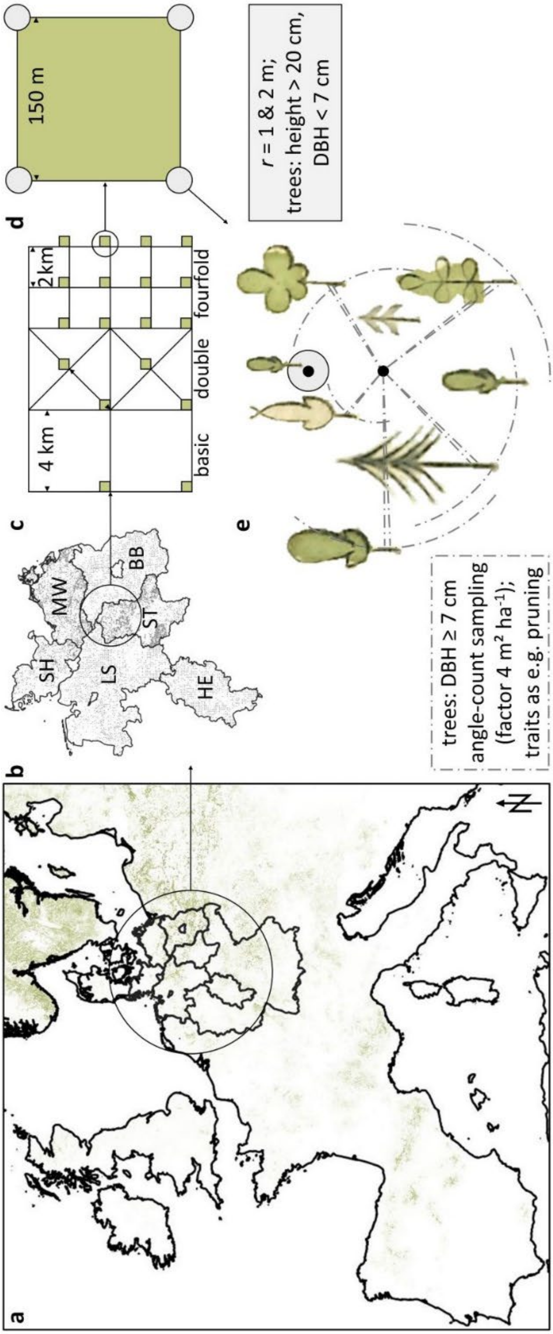


Fig. 3.1 **a)** Current Scots pine distribution (green; de Rigo et al. 2016) in Europe, Germany and examined federal states. **b)** Study area with differing intensities (strata) and sampling design: **c)** Basic (Brandenburg BB, Hesse HE, Lower Saxony LS), double (Lower Saxony) and fourfold (Mecklenburg-Western Pomerania MW, Schleswig-Holstein SH, Saxony-Anhalt ST). **d)** Clustered plot design and **e)** concentric subplot circles of radii *r* with relevance for our work. Figure based on Guimpel (1819) and BMEL (2015), borders by © GeoBasis-DE/BKG (2022) & EEA (2018)

EARTH12) and its dynamical 'Regional Atmospheric Climate Model' (RACMO) are considered as a moderate projection in RCP8.5 (van Meijgaard et al. 2008; Hazeleger et al. 2012).

As a suitable common denominator to align site water properties to climate outcomes, we use the climatic water balance (CWB, Fig. 3.2). The CWB is defined as the difference of precipitation and potential evapotranspiration (Sutmöller et al. 2021). CWB is determined as a 30-year average for the growing season which is dynamically calculated, i.e. the length of growing season also varies depending on the respective climate (Menzel and Fabian 1999; Nuske 2022).

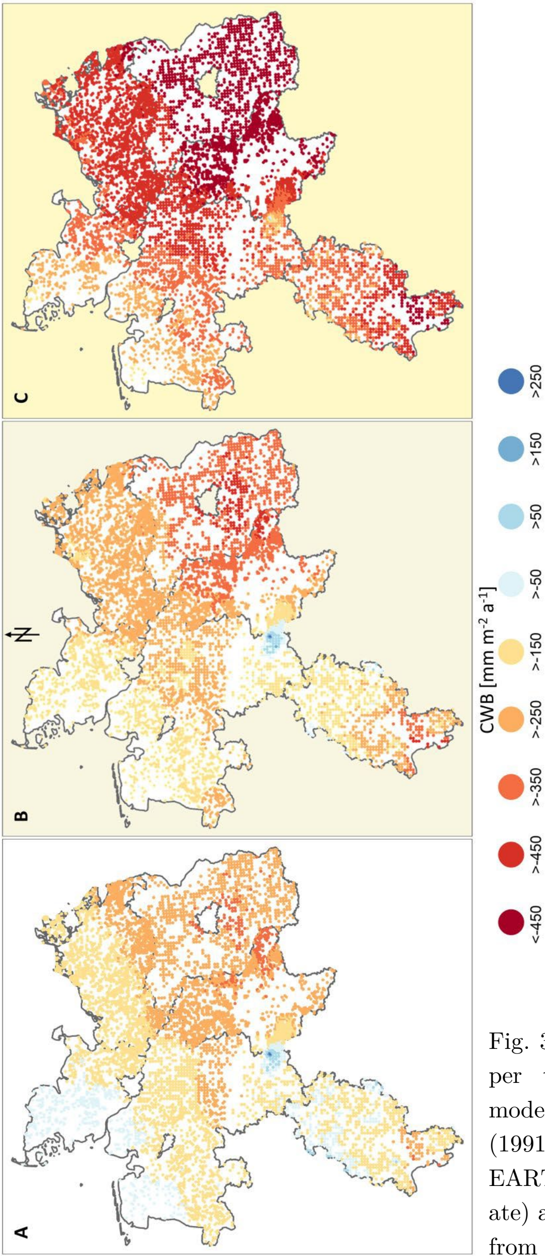


Fig. 3.2 Decreasing mean CWB-class per tract over time and climate model. **A)** With recent climate data (1991-2020), **B)** using RCP8.5 EC-EARTH12 from 2071-2100 (moderate) and **C)** using RCP8.5 hadGEM2 from 2071-2100 (extreme)

Site dynamics

Due to its very heterogeneous data, this approach has to deal with two different methods as to how water properties could be projected climate-sensitively. In Hesse, Lower Saxony, Schleswig-Holstein and Saxony-Anhalt for site dynamization CWB is directly considered as site water balance (Grier and Running 1977), which is the sum of CWB and available water capacity (effective root zone of 1 m). For sites in Brandenburg and Mecklenburg-Western Pomerania the CWB is used indirectly by shifting climate humidity districts (CHD).

Actually, the states of Brandenburg and Mecklenburg-Western Pomerania introduced CHDs to create comparable current climate conditions where only a subset of FDT is suitable (Thurm and Wirner 2023; Schröder et al. 2023). Therefore, we assume that today's tree spp. assessment is fixed to CHD peculiarities. Hence, in this study for each CHD level, the statistical location parameters of the CWB dataset for the current climate period 1991-2020 were determined on the basis of the NFI subplots. The threshold of a CHD level was then delineated by its lower quartile value and acting as a lower limit, up to which the FDT subset is recommended. With future changes in CWB-values at the NFI-subplots, the CHD assignment will thus shift (Fig. 3.3).

3.2.2. Methods

Selection of FDT

Research question *i* assumes that restoration is executed on all sites with a feasible FDT. It thus neglects the current stand types. In practice, the BAU-assessment of site-specific tree spp. (and accordingly FDT) requirements are currently performed expert-based (Riek et al. 2020; Buresch et al. 2023; Schröder et al. 2023). Each German federal state has defined its own FDT catalogue (Fig. 3.3) for the respective state forests (overview: Albert et al. 2017; Buresch et al. 2023, pp. 59–60). For private forest owners, however, these FDTs are mandatory by following best practice or if subsidies are accepted.

According to the BAU, for each site and climate condition there are different silvicultural variants possible at the same time (Fig. 3.3), enabling e.g. to minimize or to maximize pine shares, thus representing the guard rails. Therefore, algorithm-based we selected the most contrasting final cut variants in order to constitute a scenario funnel (Kosow and Gaßner 2008, pp. 15–17).

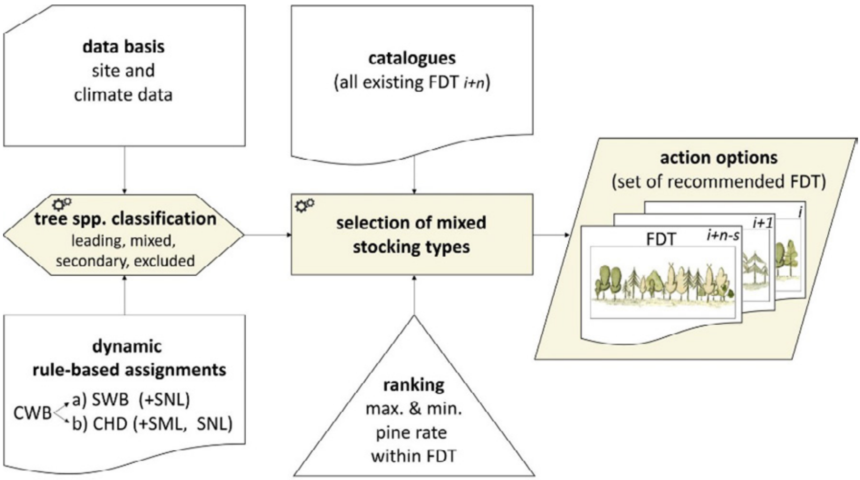


Fig. 3.3 Flowchart of dynamic FDT selection aligned to the algorithm of Hamkens et al. (2022). Out of all existing FDT *i+n* on a given site *s* only a set of FDT is recommended. Due to Germany's federal structure these are obtained either by a) altering soil water balance (Hesse, Lower Saxony, Schleswig-Holstein, Saxony-Anhalt) or by b) shifting CHD (Brandenburg, Mecklenburg-Western Pomerania). Figures based on Hamkens et al. (2022) and Guimpel (1819)

Firstly, while following the silviculture guidelines of the federal states, in a possible short and intensive forest restoration relating to hemeroby level, stands led by light-demanding tree spp. were targeted (Fig. 3.3). Here, out of the set of recommended FDTs, we chose the one with highest pine share, applying the selection algorithm by Hamkens et al. (2022). If the secondary tree spp. (< 10 % basal area, BA) were not explicitly listed and if *P. sylvestris* was already selected as leading species (≥ 50 % BA) or mixed (≥ 10 % BA), we prescribed also light-demanding

Quercus spp., *Betula* spp. or *Salix* spp. as secondary taxon. Hereby, future stands with leading shade-tolerant spp. were also selected in order to place pine at least as a mixed or secondary tree spp. on these sites.

Secondly, in a forest restoration following a longer time period and less intense cuttings, stands dominated by shade-tolerant tree spp. were considered for cultivation (Fig. 3.3). Comparatively, we chose the FDT of lowest pine share whilst emphasizing shade-tolerant taxa. Likewise, future stands with pine were selected in order to prescribe a FDT for adverse sites at all. Depending on the leading tree species, the FDT which had undefined secondary spp., were enriched with the shade-tolerant *Fagus* spp., *Tilia* spp. or *Carpinus betulus*.

Consequently, resulting from Fig. 3.2 and Fig. 3.3, two differing FDT site sets exist for each management variant. Thereupon, in this study we contrast four scenarios b - e as shown in Fig. 3.4.

Simulating forest development

Regarding research question *ii*, we used the *WaldPlaner* (version 2.2). The simulation software is based on the forest growth library 'Tree Growth Open Source Software' (TreeGrOSS, Hansen and Nagel 2014) and requires the FDT site set list, SI values and management settings.

In order to predict forest development climate-sensitively, changes in SI are accounted for depending on the respective climate model applied. The SI is one major predictor in the *WaldPlaner*'s height increment function. SI is predicted using site, climate, Nitrogen deposition and geographic location data applying the generalized additive model by Schmidt (2020). To run the model, SML and SNL from the different site mapping schemes of the six German states were unified (Benning et al. 2015). The climate variables 'temperature sum in the growing season' and 'precipitation sum in the growing season' were dynamically averaged over the past or projected stands' lifespan. The length of growing season also varies depending on the respective climate (see "Site and climate data" section). The Nitrogen deposition is based on the results of Schaap et al.

(2018) and averaged, too. Hence, for each simulation step, the respective SI is updated for each subplot, layer and species.

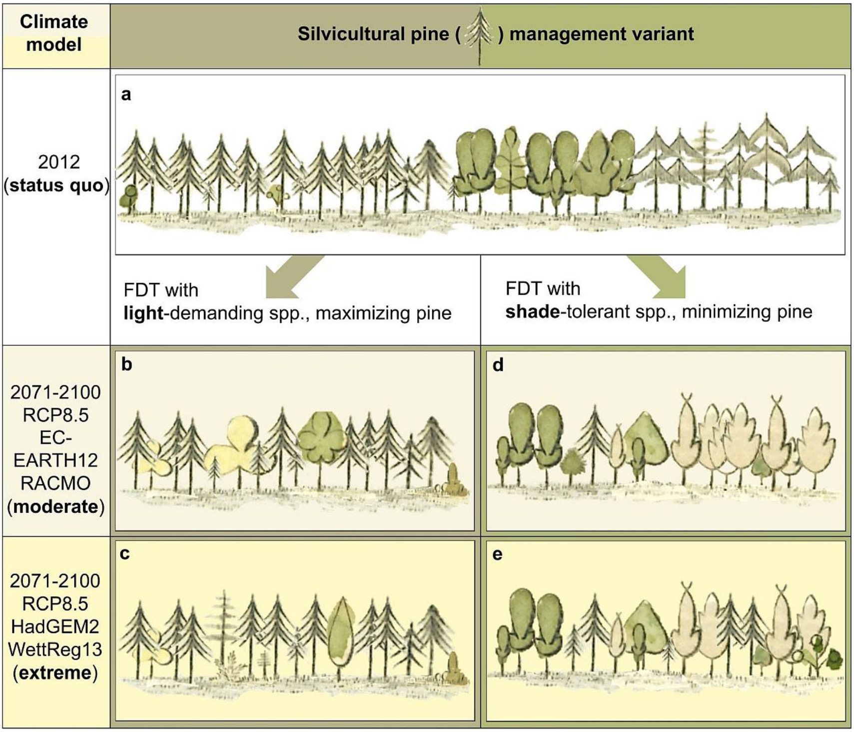


Fig. 3.4 Starting point of FLR and scenarios with spanned uncertainty space. **a)** *Status quo*: Current forest stands as per NFI with forest restoration obligations. **b) - e)** Equally likely scenarios with FDT prescribed under different climate of the period 2071-2100 by silvicultural management maximizing (b, c) or minimizing (d, e) pine shares. Note that FDT's species share must not be reached within a time window. Figure based on Guimpel (1819)

Growth simulations with the *WaldPlaner* require management settings. Therefore, we operationalized management intensity, maturity as well as cutting period and cycle of final harvest as prescribed in the BAU (Tab. 3.1): Thinning intensity is oriented towards an age-dependent and SI-specific target basal area (BA). This target BA is defined by the stocking degree sd_{yt} derived from yield tables. We used the particular silvicultural guidelines for Scots pine. Here, differences concerning the BA exist, having lower sd_{yt} in the western region (yield table by Wiedemann 1943) in contrast to the eastern region (yield table by Lembcke et al. 1975). Hence, the respective data from the two pine yield tables was functionalized: A nonlinear model derived by the Levenberg-Marquardt method (1944, 1963) was used and computed by *minpack.lm*-algorithm (Elzhov et al. 2023). For the target BA as given from the yield tables and reduced by the silvicultural guidelines (sd_{yt} range from 0.7–0.9), we decided to fit a Chapman-Richards equation (1961, 1959) with parameters $a_1 - c_2$ that is dependent on stand age t and SI:

$$BA = (a_1 - a_2 SI)\left(1 - e^{\left(b_1 e^{(b_2 SI)}\right)t}\right)^{\left(c_1 e^{(c_2 SI)}\right)} \quad (3.1)$$

The used parameters and region-specific estimations are supplemented (Tab. S3.3). Next, thinning regime was defined as from above, and future crop tree oriented. During the tending phase, different numbers of future crop trees are promoted in the states, depending on the guideline and standing in line with the thinning regime so that at higher BA less future crop trees are promoted and *vice versa* (Tab. 3.1). For final cuts of Scots pine, shelterwood respectively seed tree systems (Wagner et al. 2010) were simulated as a common denominator between the BAU of the states. The *WaldPlaner* was then refined by implementing an adjustable maturity as a threshold where final cuts start.

Other than the performed thinning type, shelterwood cuts are technically executed as a heavy cutting from below (Tab. 3.1): As soon as 50/ 25 % (restoration towards stands led by light-demanding/ shade tolerant spp.) of BA is provided from trees that have reached 45 (unpruned)/ 55 cm DBH (pruned) the final cutting starts. Firstly, 50 % of the BA is to be removed until the aimed maximum density is reached, with trees being chosen at random that fall within the target diameter. Secondly, the remaining 50 % of the BA are harvested from the residual stand, including understory trees, in order to promote light supply for the following regeneration (Wördehoff et al. 2017, p. 23; Huth et al. 2022).

Based on the maturity definition, large or small openings are created in the variants, which results in cutting cycles of 20 or 40 years (Tab. 3.1; Kint et al. 2006). In both thinning and final cutting, the maximum removal volumes, which are due to stand preservation, do not impair the cutting cycle. A final liberation cut is not executed in order to preserve remnant trees (Borrass et al. 2017).

Tab. 3.1 Prescribed management regime of Scots pine in the *WaldPlaner* following the particular guidelines of the federal states per silvicultural variant of FLR

Criterion	**Silvicultural FLR variant**	
	light	**shade**
Tree spp. selection	Light-demanding, maximizing pine (b, c)	Shade-tolerant, minimizing pine (d, e)
Number of future crop trees west/ east [N/ ha]	160/ 135	160/ 135
Target DBH unpruned/ pruned [cm]	45/ 55	45/ 55
Start of final cuts (= maturity) [% BA > target DBH]	50	25
Period & pattern of final cuts [a]	20, large openings (shelterwood cut)	40, small openings (shelterwood cut)
Cycle of final cuts [a]	0, 5, 10, 15, 20	0, 5, 10, 15, 20, 25, 30, 35, 40
Maximum density of final cut cycle [rel.]	0.3, 0.25, 0.15, 0.1, 0.05	0.5, 0.45, 0.4, 0.35, 0.3, 0.25, 0.19, 0.12, 0.05
Min.–max. removal per operation at thinning/ final cut [m^3 ha^{-1} over bark]	10-80/ 10-300	10-80/ 10-300
Remaining stock (e.g. remnant trees) [% fully stocked]	5	5

In total, nine silvicultural prescriptions were set in the *WaldPlaner* environment: two regions (western/ eastern states) × two target diameters (pruned/ unpruned) × two silviculture variants (max./ min. pine) + strictly preserved stands. While pine's silviculture was adapted as described above, the other tree spp. were simulated with the default settings of *WaldPlaner* which are similar to the BAU of the states and do not consider differences between the regions.

For growth simulations, first of all, the dendrometric data of each subplot is used to generate model forest stands with a uniform size of

0.2 ha. Hence, each tree is represented e.g. by its position, height, DBH and species. The simulations were executed for a period of 50 years from 2012-2062 and computed in steps of 5 years.

For thinning and final cutting during the simulation, the species share of the current stand was used to gradually work towards the particular FDT definition and finally replaced by a new stand with the species share of the particular FDT. The new forest stand is then simulated accordingly.

On sites in preserved areas, no thinning nor final cutting is simulated and forest growth is just bound by regionally different maximum density functions and yield levels (Döbbeler and Spellmann 2002; Assmann 1970, p. 164).

Range of uncertainty

Concerning research question *iii*, we assess the uncertainties as the difference between forest development in scenarios b – e, measured by cultivation area and its goal achievement in relation to the FDT, growing stocks as well as the FLR.

For every subplot with predicted pine share by the *WaldPlaner*, per scenario (b – e) a target achievement percentage $A\%$ is calculated. It is computed as

$$A\% \; = \; 100 - \left| \sum_{h=1}^{L} W_{\mathrm{h}} \frac{1}{m_{\mathrm{h}}(P)} \sum_{i=1}^{n_{\mathrm{h}}} \sum_{j=1}^{m_{\mathrm{i}}} \left(r_{\mathrm{hij}}(p) - r_{\mathrm{hij}}(g) \right) \right| \qquad (3.2)$$

with $m_{\mathrm{h}}(P)$ total number of subplots with predicted pine in stratum h and $r_{\mathrm{hij}}(p)$ denotes the performed pine's percental ratio per subplot in 2062. Then $r_{\mathrm{hij}}(g)$ is the target (goal) pine percentage within a FDT.

Furthermore, from the *WaldPlaner* results, we considered the change rate (z) of pure stands by computing

$$z = \frac{\bar{x}''}{\bar{x}'} = \frac{\sum_{h=1}^{L} W_{\mathrm{h}} \bar{x}_{\mathrm{h}}''}{\sum_{h=1}^{L} W_{\mathrm{h}} \bar{x}_{\mathrm{h}}'} \qquad (3.3)$$

with $\bar{x}'_{\mathrm{h}}$ as the estimated mean cultivation area in stratum h for time t, $\bar{x}''_{\mathrm{h}}$ as the estimated strata mean cultivation area for time $t + 50$ and W_{h} as the stratum weight. $FLR\%$ is then derived by $(1 - z) \times 100$. The variance is calculated by a combined ratio estimator (Cochran 1977, p. 165):

$$v(z) = \frac{1}{\bar{x}'^2} \sum_{h=1}^{L} W_{\mathrm{h}}^2 \left(v(\bar{x}''_{\mathrm{h}}) + z^2 v(\bar{x}'_{\mathrm{h}}) - 2z\ cov(\bar{x}''_{\mathrm{h}}, \bar{x}'_{\mathrm{h}})\right) \quad (3.4)$$

Subsequently, we used $v(z)$ to calculate the 95 % confidence intervals (CI) with $\alpha = 0{,}05$ and a t-value of 1.96 was chosen due to the large sample size (see "Simulating forest development" section).

For details on the basic components of the NFI estimation design, see supplemented eq. S3.1 – S3.4 and Riedel et al. (2017).

3.3. Results

3.3.1. Guard rails of pine shares

The variation of different FDTs is shown in Fig. 3.5 and can be seen as maximum or minimum guard rails of future pine shares. The blue circles highlight exemplary areas analyzed in more detail below. Thus, the Harz mountains sites (big circle in Fig. 3.5b), elevation of 1,141 m a.s.l., mainly comprises mesotrophic SNL (Fig. S3.11) and positive CWB in EC-EARTH. Consistently, low pine shares resulted, whereas at deteriorating CWB in HadGEM pine shares increase, even at high elevations (circle in Fig. 3.5c). In 2012 large spruce forests were stocking in the Harz mountains, therefore pine can only be cultivated once spruce is mature. Different pine's evaluation of two neighboring states is shown between borders of Saxony-Anhalt and Brandenburg at comparable mid-low SNL: In the Fläming hill country (circle in Fig. 3.5e) pine shares vary significantly between the two states. On sites with overall rich SNL (Fig. S3.11) even at low CWB as in the Strelitz territory in Mecklenburg-Western Pomerania (small circles in Fig. 3.5b and d) the ratio of pine within FDT never exceeds mixed level.

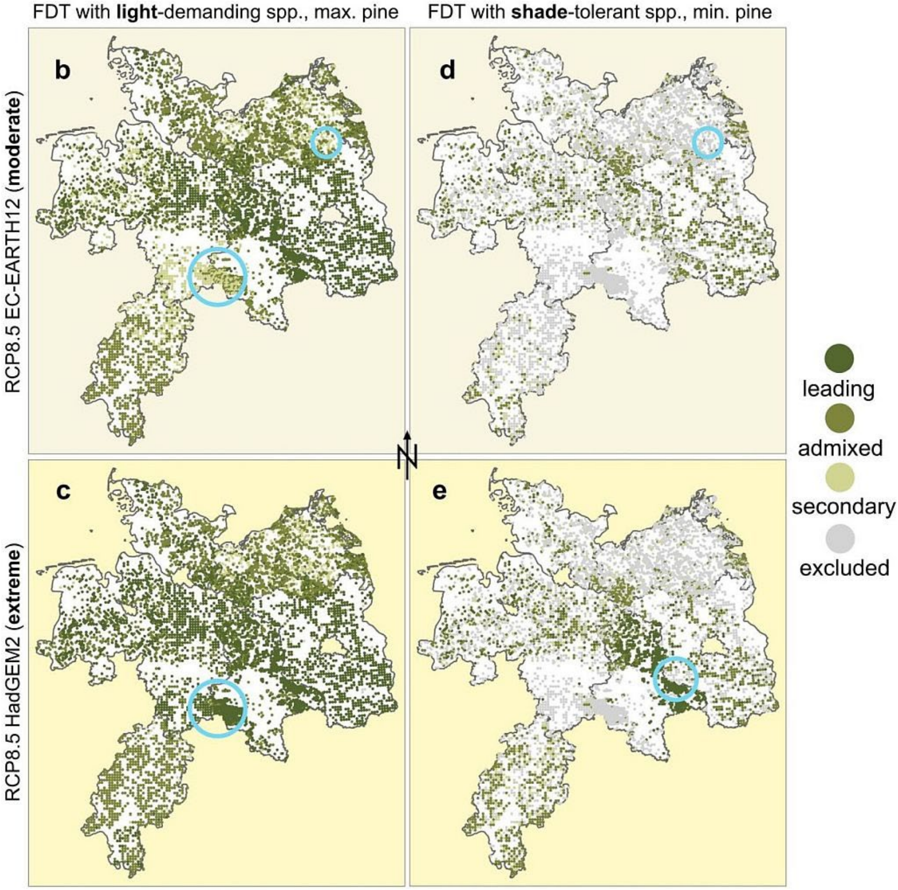


Fig. 3.5 Guard rails of pine share within FDT classified as a leading (≥ 50 % BA), mixed (≥ 10 % BA), secondary (< 10 % BA) or excluded (0 % BA) tree species. Means per tract between scenarios **b) - e)** are shown. Note that space between points (tracts) does not state forest cover but sampling density

Comparing the effect of climate models (b, d vs. c, e) it can be seen that the variation of pine's FDT shares is comparatively lower in Brandenburg and Mecklenburg-Western Pomerania whereas it tends to be greater in Hesse, Lower Saxony, Schleswig-Holstein and Saxony-Anhalt.

3.3.2. Prediction of forest development

Essential for the prediction of forest development is the derivation of the differing target BAs of the two thinning regimes shown in Fig. 3.6. Here, a difference of up to approx. 10 m² ha^{-1} for the uppermost SI and still about 3 m² ha^{-1} for the lowermost SI is depicted. Note that the plotted SI between west and east are not exactly equal as a result of different SI-systems. According to the silvicultural guidelines the BA in western states does not increase much after the age of 60 while in the eastern states it is more or less held level until the age of 100. The patterns of the curves fits to the unequal yield levels as expressed by the maximum density and marks a difference of about 7 m² ha^{-1} over 100 years.

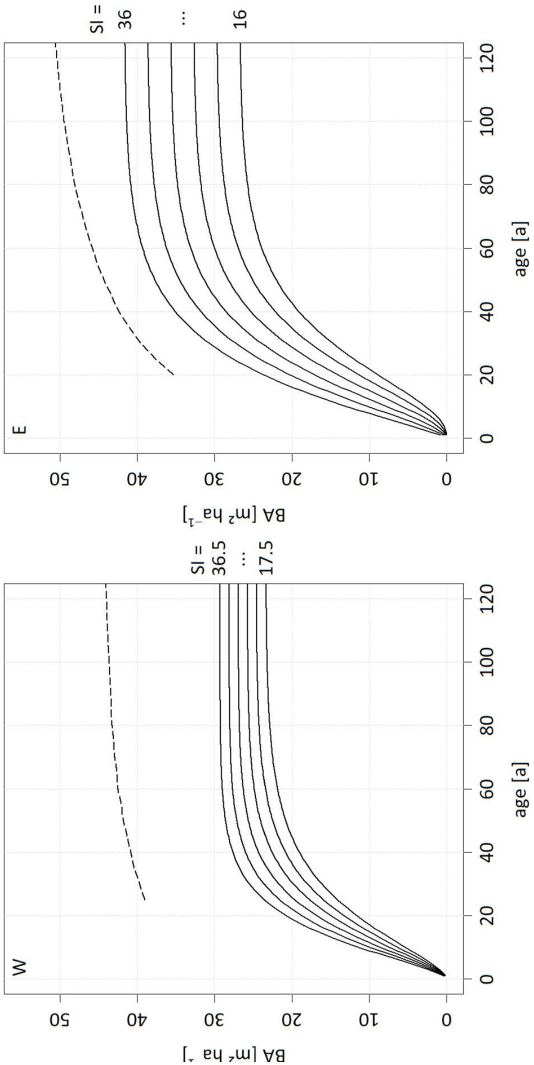
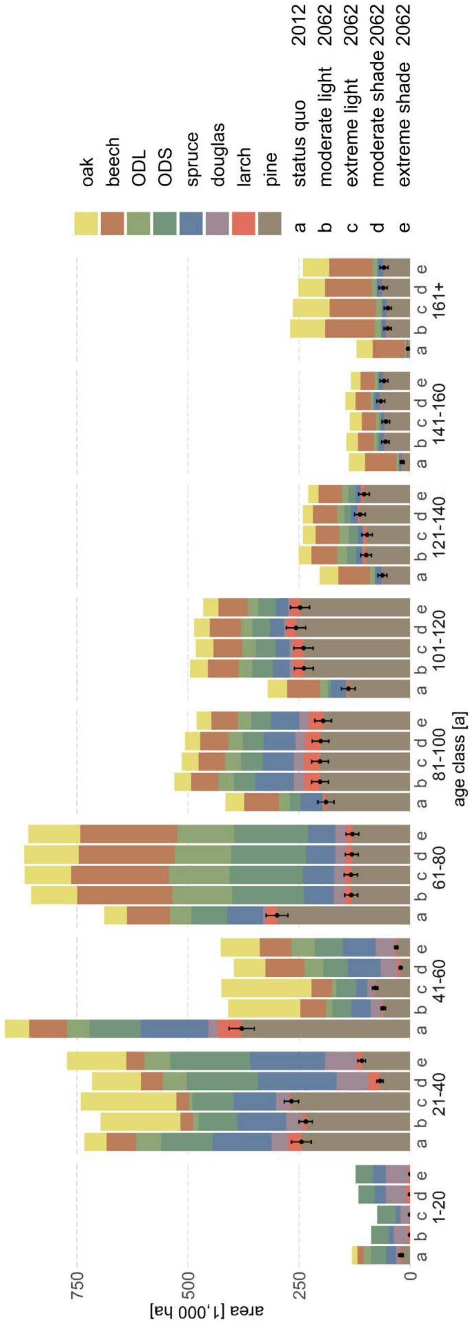


Fig. 3.6 SI-depending target BA of *P. sylvestris* (solid lines) per region after particular silviculture guidelines referring to **W)** yield tables of Wiedemann (1943; residual SE 38.6 %) or **E)** Lembcke et al. (1975; residual SE 32.6 %) and maximum density in respective yield level regions (dashed lines; Döbbeler and Spellmann 2002) for top heights of greatest pictured SI



Overall Scots pine's extent is decreasing from currently (a) 1.358 million hectares (m. ha) towards a minimum of 0.919 m. ha (d) and at most 1.123 m. ha in scenario (c).

The current north German forests reveal an uneven age class distribution (Fig. 3.7, columns a). This unevenness is not balanced in any scenario until the year 2062 (b – e). The mean stratum-weighted pine stand age is 73 years in 2012 (a) and varies then over 16 years from 63 years (c) to 79 years in (d) in the year 2062. If only certain age classes are examined, pine's share in scenarios b – c accumulates in comparison to scenarios d – e which is notably depicted in age ≤ 60. Subsequently, in these pine-minimizing scenarios d – e of age ≤ 60 Norway spruce increases, under which shade-tolerant *Abies* spp. are subsumed. The impression of almost

Fig. 3.7 Estimated area of tree spp. groups depending on the age class. According to BMEL (2015) compact wood in 2012 (a) and the scenarios (b - e) in 2062 (assigning other pine than *P. sylvestris* to larch) are shown. Error bars for pine depict ±CI95 non-existent pine share in age ≤ 20 is completed by an indentation in age class 41-60. While other deciduous trees with long life expectancy (ODL), such as *Acer* spp. and *Fraxinus* spp., decrease in all scenarios b – e, other deciduous trees with short life expectancy (ODS), such as *Betula* spp. and *Populus* spp. increase.

FLR of pine tends to be slower if the reduction between scenario d, e and b, c is examined for ≥ 101 year old stands (when shade-tolerant spp. are emphasized). Strikingly, old stands ≥ 101 and especially > 161 years are increasing for all scenarios b – e again with higher pine shares in scenario d – e.

We defined *A%* as the difference of performed pine's share per subplot in 2062 and targeted share as per FDT (*cf.* Fig. 3.5). For this evaluation all subplots with predicted pine were taken into account. In 2062 *A%* eventually reaches (b) 69.6 %, (c) 62.7 %, (d) 68.6 % and (e) 68.6 %. Hence, scenario (b) achieves the particular target best and scenario (c) worst. From the growing stocks of pine (Fig. 3.8) currently (a) 1.9 % are under strict protection on site mapped forests which increases to 3.0 – 3.8 % in 2062 within all scenarios (b – e).

Currently the growing stock of all spp. amounts to (a) 1,193 million cubic meters (m. m^3) whereas in 2062 it differs between scenarios from 1,391 (c) to 1,484 m. m^3 (d). While Scots pine volume amounts to 424 m. m^3 in 2012, in 2062 it almost stays constant with 427 m. m^3 in scenario (d) whereas the other scenarios predict decreasing volumes as a result of a dismantling management with minimum 340 m. m^3 in scenario (b).

Regarding the pine growing stock for trees near to the target diameter/ large-dimensioned timber ≥ 50 cm (i.e. diameter classes ≥ 41 cm), under all scenarios there is a considerable increase compared to the current volume (Fig. 3.8). This increase is largest if shade-tolerant regeneration is emphasized (classes 51-70 cm for scenarios d – e). Volume over bark for thin trees ≤ 40 cm DBH in all scenarios is comparatively low in comparison to *status quo* (a) which expresses renunciation due to

FLR. However, in scenarios where pine shares are maximized (b – c), growing stocks increase slightly for trees ≤ 20 cm DBH in comparison to scenarios d – e. But in 2062 in every scenario the pine stocks (≤ 20 cm) remain less than the *status quo* (a) in 2012.

3.3.3. Uncertainties

Recapitulating uncertainties between BAU scenarios, it is evident that silvicultural impact on cultivation area or growing stocks is about four times higher than climate models effect (Tab. 3.2). In the north German forests, pine has an overall relatively lesser growing stock than cultivation area, expressed by a total of 35.6 % volume on 37.1 % area. Growing stock as well as cultivation area are decreasing even if pine is prospectively emphasized in forest planning. The

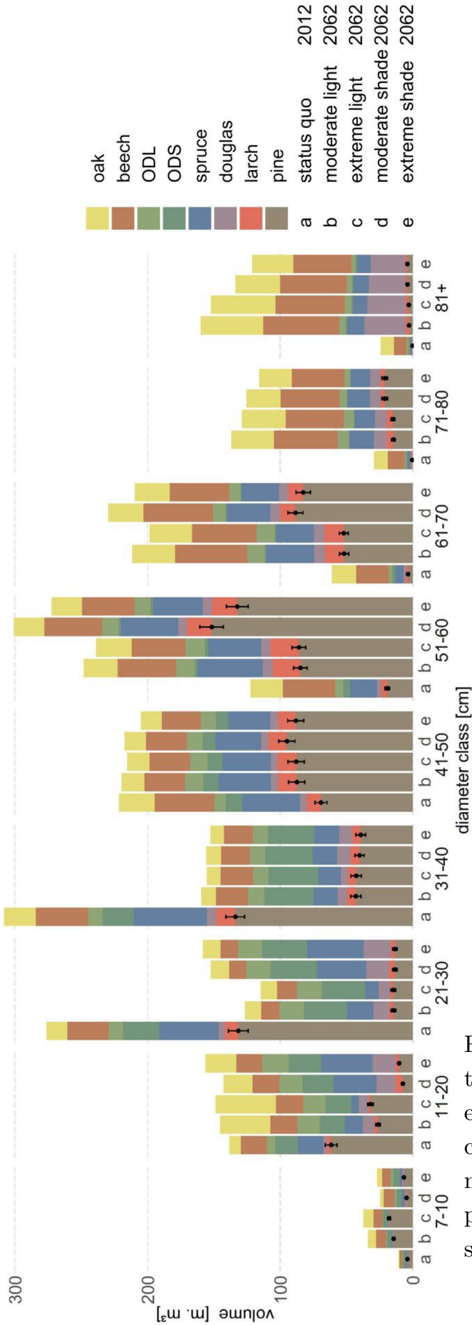


Fig. 3.8 Estimated volume over bark of tree spp. groups depending on the diameter class. According to BMEL (2015) compact wood in 2012 (a) and the scenarios (b – e) in 2062 (assigning other pine than *P. sylvestris* to larch) are shown. Error bars for pine depict ±CI95

precision of estimation (CI95) decreases over time and is lowest in scenarios d – e where shade-tolerant spp. are promoted.

Tab. 3.2 Summary of uncertainties arising from forest planning between climate models and silvicultural variants in scenarios b - e in 2062. Numbers show relative value of total area/ stock

variable	**variant**	**light**	**shade**	**| Δ silviculture |**
area [%]	moderate	(b) 28.79	(d) 24.64	**4.15**
(a) = 37.06		± 4.46	± 4.81	
± 4.33	extreme	(c) 30.01	(e) 25.06	**4.95**
		± 4.30	± 4.74	
	| Δ climate |	**1.22**	**0.42**	-
stock [%]	moderate	(b) 23.58	(d) 28.75	**5.17**
(a) = 35.57		± 4.92	± 5.00	
± 4.40	extreme	(c) 25.28	(e) 28.08	**2.80**
		± 4.74	± 5.22	
	| Δ climate |	**1.70**	**0.67**	-

About two thirds of the studied pine extent lies within the eastern states which as a result influences regional uncertainty. In the following, pure stands with ≥ 90 % pine BA per plot are considered. Monospecific stands have highest risk and forest restoration priority. From their respective total cultivation area per region, this leaves 36 % of stands in the west and 71 % in the east. This subset is represented in Fig. 3.9. Besides this regional difference, a comparable behavior of uncertainties between all scenarios b – e is visible: Uncertainty is greater between silvicultural variants than between climate models.

By contrasting the western and eastern states it is clear that in Brandenburg, Mecklenburg-Western Pomerania and Saxony-Anhalt there are exceeding five times more pure pine stands than in the west. Pine's FLR progress absolutely seen is higher in the eastern region because the natural setting possesses higher pine content. In both regions most pure stands are replaced by mixed forests in scenarios d – e. Absolute uncertainty range expressed as a difference between all scenarios b – e, is higher in the eastern region and varies from 28,000 ha (east) to 13,000 ha (west).

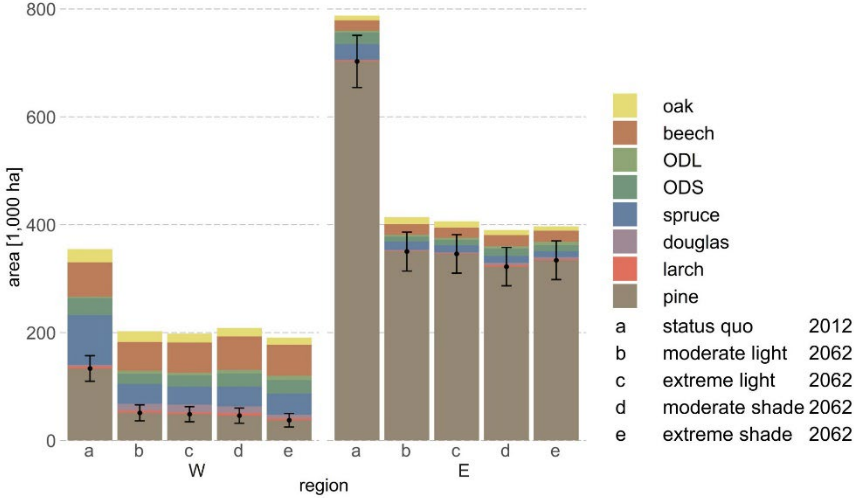


Fig. 3.9 Estimated area of pure stands for the western and eastern study regions (≤ 10 % BA mixed by another tree spp. per tract). According to BMEL (2015) compact wood in 2012 (a) and the scenarios (b – e) in 2062 (assigning other pine than *P. sylvestris* to larch) are shown. Error bars for pine depict ±CI95

However, besides the great absolute progress in FLR of pure Scots pine in the eastern region, a relative comparison detects a lower FLR rate of monospecific stands here (Fig. 3.10D). Regardless of the maturity threshold when final cuts start, for both silvicultural variants it can be stated that the FLR rate between regions is more than 11.3 – 19.3 % slower in the east (Tab. 3.2). Besides their extent, western and eastern pine monocultures can be seen as comparable when considering their site, age and SI (Fig. 3.10A – C). Surprisingly, relative FLR rate in shade-tolerant tree spp. emphasizing scenarios d – e is higher for both regions although earlier underplanting, longer cutting period, higher canopy cover

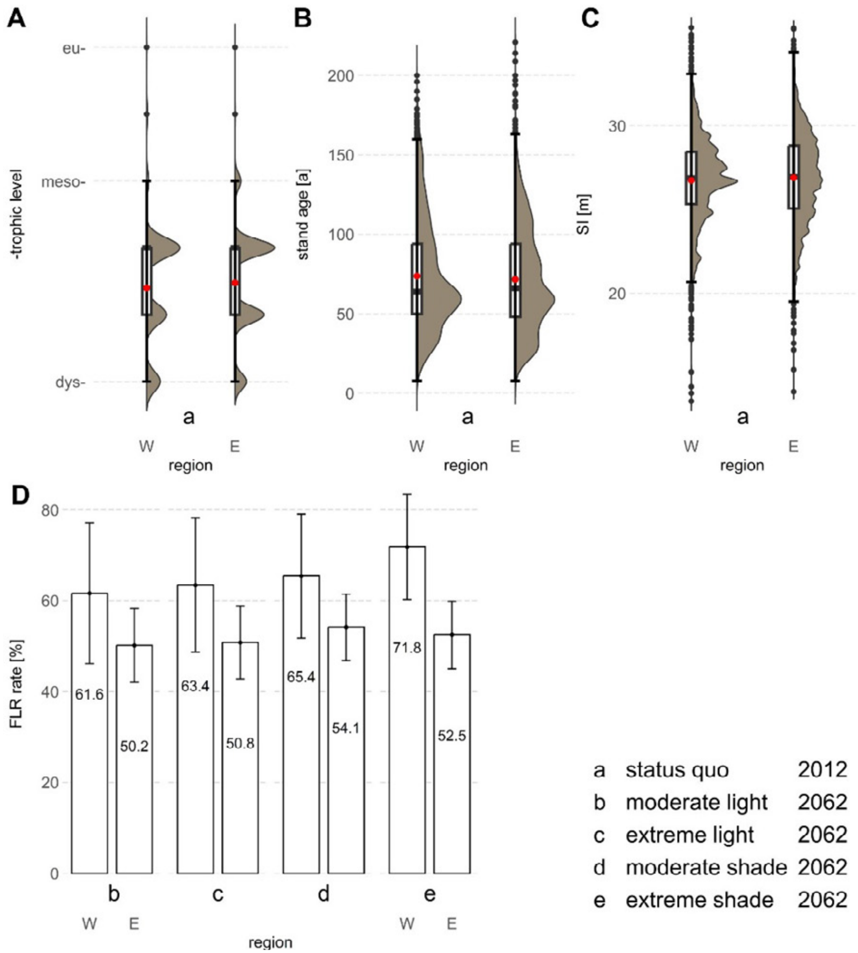


Fig. 3.10 **A)** - **C)** In current (a) pine monocultures between regions (W, E) at comparable SNL, stand age and SI there are **D)** different FLR rates of pure stands evolving in all scenarios b – e. Trophic key adapted from Schmidt et al. (2015). **A)** - **C)**: Violin-plots depict data density, mean value: red point; median: black line. **D)** Error bars show ±CI95. SNL per plot is reported as fitted value, i.e. the most plausible score, resulting from an estimated cumulative logit model (Fahrmeir et al. 2013, pp. 312–316) with including plot number as random intercept component. This modeling step avoids manipulations, such as empirical mean calculation, that are not adequate for the ordinal scale of the SNL outcome

and stockpiling (Fig. 3.8) influences stands here. Relative uncertainty range expressed as difference between all scenarios b – e is almost three times higher in the western region (10.2 %) than in the eastern region (3.9 %). Uncertainty ranges between climate models is again smaller than between silviculture variants. Estimation ranges (CI95) are highest in western stands and especially in scenarios d – e where shade-tolerant spp. are promoted.

3.4. Discussion

3.4.1. Critical remarks on material and methods

Uncertainty in forest resource assessment and modelling is caused by sampling, measurement, and prediction errors (Kangas and Kangas 2004). Uncertainties in climate predictions arise from internal variability of climate on the planet, response uncertainty is due to different models and scenario uncertainty because future human behavior cannot be foreseen (Hawkins and Sutton 2009). In silviculture, however, uncertainty is not used yet to describe the outcome of adaptive management, it is rather seen as plurality of action options (Ehwald 1949). Remarkable impact can be furthermore drawn from politics (Lawrence 2017) and calamities (Rosenkranz et al. 2023).

While the present work identifies sampling errors by using variances, prediction errors accumulation in *WaldPlaner* are not assessed. With every simulation step an error propagation within scenario strand must be expected. Therefore the simulation was only executed for 50 years length which is already more than the recommended 40 years (Hansen and Nagel 2014, p. 29). However, climatic uncertainty is expressed by contrasting moderate and extreme realizations.

TreeGrOSS is parameterized with a profound data set derived from long-term experiments as well as inventory data. It has been intensively validated (Vospernik et al. 2015) and is frequently applied for forest scenario simulations in northern Germany (e.g. Hentschel et al. 2017; Albert et al. 2018). The *WaldPlaner* indirectly also considers effects of species mixture, such as overyielding (Sprauer and Nagel 2015), that can be seen throughout as positive for Scots pine (Pretzsch et al. 2015, 2021).

When regenerating forest stands, *WaldPlaner* does not establish a certain seedling density but regeneration placeholders. Feasible stand densities are firstly considered when ingrowth in compact wood occurs to avoid overstocked plots with possibly > 100,000 trees ha^{-1} prior. However, density-related measures such as volume could be affected by the placeholder density. In contrast, the cultivation area is not density-related but standardized, based on plot size.

Moreover, we see five improvements in our predictions compared to not site specific simulations expressed by a smaller uncertainty space (Oehmichen et al. 2018; Rosenkranz et al. 2023): Next to (1) different management intensities and future crop tree numbers (2) the *WaldPlaner* considers realistic FDT guard rails from silvicultural guidelines. Whilst (3) tree spp. are selected site-specifically, (4) density-related and (5) climate-sensitive tree growth from representative north German yield levels is taken into account (Wördehoff et al. 2017, p. 22). However, spatial extension of the two yield levels is static relating to climate change.

The current study uses data from the NFI with reference year 2012. Certainly data from 2012 can be considered as outdated after the severe calamities that have occurred since then (Rosenkranz et al. 2023). Still, the used data of the NFI serves to show trends and directions of future undisturbed forest development.

We acknowledge that site data is heterogeneous as shown by soil water balance or climate humidity districts in order to select tree spp. dynamically. Also next to the Baltic Sea, climate water balance-values begin to increase (Fig. 3.2B). This results from downscaling due to an absence of climate stations on the sea. From this we conclude that tree spp. selection must be enhanced e.g. by measures of drought as the Standardized Precipitation Evapotranspiration Index (Vicente-Serrano et al. 2010) or temperature extremes (Brandl et al. 2020; del Río et al. 2022).

Since an area-wide available water capacity is not given in Mecklenburg-Western Pomerania, dynamic tree spp. selection must still rely on soil moisture level and climate humidity districts with only minor outcome differences between climate models (Thurm and Wirner 2023, pp. 61–65).

In addition, we want to point out the difficulties by creating stand information from sampling data. Since NFI data originates from angle count sampling, exact tree neighborhood influences remain unknown (Fischer and Mölder 2017). Also stem quality is not surveyed in German NFI that would affect the tree selection for shelterwood cuts and the partial understory removal. Notwithstanding the plot size that was chosen due to computing capacity, results are only valid for aggregated measures at large sample size. The sample unit of the German NFI consists of the clustered plot. However, as every subplot could show different soil nutrient level and thus FDT, model stands were not generated and simulated per plot which contradicts to the above NFI basic idea. However, pure stands were again defined in accordance with the sampling definition on the plot level. These units, defined as ≥ 90 % monospecific species per plot, would have occurred to a greater extent if considered at the subplot level. The shown magnitude of FLR rate and difference between the regions, however, is similar.

Following common definitions in German forestry, the bottom layer of a stand is divided into understory and young growth (Bartsch et al. 2020, p. 186) and only the latter can be seen as an adequate regeneration. As long as overstory density is $sd_{yt} \geq 0.5$, stands are typically subsumed by their top layer (main stand; BMEL 2015, p. 49).Thus, FLR rate would be influenced negatively if vertically mixed understory is only considered when overstory is very light. Since the *WaldPlaner* does not consider stand layers automatically, we assumed understory trees that are compact wood, also contribute to restoration of forests despite their layer affiliation or quality. We recognize that the overstory may not have been adequately cut yet in each case and that FLR may not yet have occurred in terms of regeneration (generation change).

3.4.2. Assessment on results

Differences in silvicultural concepts

Fig. 3.6 depicts the different thinning regimes between the federal states implemented in the *WaldPlaner* environment. These operational concepts correspond to the current BAU of the particular region. The Chapman-Richards-equation shows an overall continuous fit (Tab. S3.3; Fig. 3.6).

On the one hand, more intense thinning regimes (i.e. lower BA compared between regions, Fig. 3.6W) undercut the critical stocking density (Vospernik et al. 2015). On the other hand, biodiversity, single tree stability, diameter growth and vitality are promoted, although the stability against snow and wind is initially lower after tending (Lust et al. 2001; Hentschel et al. 2017; del Río et al. 2017; Bauhus 2022). Comparatively lower stocks and higher thinning yields in the western part of the study area express risk-averse behavior.

In the eastern region an extensive thinning regime (i.e. higher BA compared between regions, Fig. 3.6E) is currently practiced. On the one hand, this operational concept holds the critical stocking density (Assmann 1970, p. 231+329) and relies on stand stability but showing comparably lower stem diameters at a time and higher h/d-ratios (Döbbeler and Spellmann 2002). Other studies show that under higher stocking and later thinning the highest proportion of logs can be assumed, thus, leading to a greater amount of sequestered Carbon in wood products and earlier in the stocking biomass (Garcia-Gonzalo et al. 2008). On the other hand, lower tree health can be expected by smaller crowns and also weaker drought adaption (Sohn et al. 2016; del Río et al. 2017; Bauhus 2022). In times of climate change higher sd_{yt} express a rather risk-affine management.

Until today for scientific usage, FDT were assigned as expert-based but not automatically (e.g. Rudolph et al. 2015; Wördehoff et al. 2017; Albert et al. 2017). Hence, the algorithm of Hamkens et al. (2022) means a great advance: Aside from this work, they actually use seven

RCP8.5 climate models and recommend tree spp. dependent on whether a single tree species was selected multiple times in all models. In a multi-criteria approach the tree species itself and no longer entire FDT are assigned to a given site (*cf.* Fig. 3.3) for which reason mixed stands (FDT) are compound if the particular spp. are recommended. However, the tree spp. classification results in the same classification as used by Albert et al. (2017).

Fig. 3.5 can be interpreted as a measure how much State Forest Administrations rely on Scots pine in climate change. This is expressed by its multiple increase (Fig. 3.5c) or overall almost cancellation (Fig. 3.5d) which would negatively affect e.g. γ-biodiversity or reinforce calamity risk (von Sponeck 1819; Hentschel et al. 2017). This mainly results from related soil water balance or climate humidity district thresholds between the states where pine is recommended. Besides this, in Brandenburg and Mecklenburg-Western Pomerania we firstly have to question if the climate humidity district shifts through changing climate water balance values are sensitive enough and secondly if the FDT set per climate humidity district is sufficiently different. Furthermore, it has to be argued, whether all the current vitality problems resulting from pine's biology are considered reasonably in the silvicultural guidelines (Bose et al. 2020; Rehschuh and Rühr 2021; Haberstroh et al. 2022) and, when weighed, justify an increase cross-regionally. After all, economic and yield aspects play a central role in the discussion about the future cultivation suitability of coniferous tree species such as pine (Leuschner et al. 2022; Brichta et al. 2023). We also observed that the cultivation recommendation for pine is based on expert knowledge to this day and has not changed over time (Pfeil 1856; Wagenknecht 1955).

Uncertainties in pine's cultivation area

For the estimate of $A\%$, every subplot with eventually planned pine was considered and compared to the predicted FLR outcome in 2062. If the Harz, which is dominated by Norway spruce in 2012, should be regenerated into pine as well, then $A\%$ is consequently low (Fig. 3.5c). If alternatively current high pine population is regenerated with pine again and is not introduced in new areas such as rich sites, $A\%$ is greatest (Fig. 3.5b). Schelhaas et al. (2015) conducted an Europe-wide simulation study on NFI plots until 2070. They describe species shifts induced by a climate envelope approach from Hanewinkel et al. (2013) as very inert. From areas with restoration obligations only 36 % changed. Inertness here is caused by uneven age-class distributions and the number of spp. and stands to convert. The present study achieves higher $A\%$ because pine is still scheduled at all as opposed to the results by Hanewinkel et al. (2013). Also the north German age class distribution and mean ages of pine in 2012 are convenient for FLR. Inertness and age influence is also proved by Albert et al. (2015) and Lindner (2000). What emerges further is that Brandenburg and Saxony-Anhalt have different evaluations on where to cultivate pine in extreme climate (Fig. 3.5e).

The depicted high planning share of pine in the Harz mountains (Fig. 3.5c) can be discussed as unrealistic, since here major precipitation occurs and even highly productive Norway spruce's key growing area is also contemplated. However *P. sylvestris* is still represented in the current FDT set for those sites which speaks e.g. for a diminution of the FDT variety between landscapes to achieve higher γ-diversity (Heinrichs et al. 2019).

If FLR is considered, future cultivation area of Scots pine decreases as other studies document on different scales (Wördehoff et al. 2017; Vrška et al. 2017; Oehmichen et al. 2018; Rosenkranz et al. 2023). An uncertainty of 0.204 m. ha as found in the present study can be considered low: Oehmichen et al. (2018) simulate NFI data nationwide until 2052 and implement a not site specific FLR by emphasizing either coniferous or broadleaved trees and disregarding climate data. For pine an uncertainty of 0.69 m. ha can be derived from their results. For our research area this equals to 0.46 m. ha. Rosenkranz et al. (2023) simulate NFI for 200 years throughout Germany until the year 2211 and imply as well a not site specific FLR. They distinguish between high and low intensive adaptations with different proportions of natural succession and risk levels. In 2062 uncertainties for *P. sylvestris* of about 0.4 m. ha uncertainty for our study area.

Pine's area remains unevenly distributed when considering age-classes. While the peak in age class 21-40 coincidentally corresponds to actual calamities (Rosenkranz et al. 2023), it actually results from the assumed optimal regeneration circumstances. Thus, it has to be asked, how realistic the outcome is, if light-demanding spp. are promoted. Hence, we point out, that prospectively pine's fructification and recruitment will alter (Matías and Jump 2012), timber marketing might impede FLR (Knoke et al. 2001) as the adaption effort is high (Albert et al. 2017; Bolte et al. 2021) and steady seedling availability of other spp. is not guaranteed. Also, private forest owners might not take part in FLR that intensively (Wördehoff et al. 2017, Stockmann et al. 2024). As future potential natural vegetation might be composed totally differently (Hanewinkel et al. 2013; Buras and Menzel 2019; Mette et al. 2021; Hinze et al. 2023) we deduce nevertheless that all silvicultural endeavors must concentrate on FLR.

Increasing mean stand ages in scenarios d – e in comparison to scenarios b – c correspond to the respectively longer cutting periods and comprise about 10 - 16 years. This might affect tree vitality since older pine show higher crown defoliation (Przybylski et al. 2021; Bauhus 2022).

Uncertainties in pine's growing stocks

All scenarios (b – e) conceal an increase of volume among all tree spp. groups in the absence of risk which is also described by other authors (Wördehoff et al. 2017; Oehmichen et al. 2018) whereas in turn others predict a decreasing total volume whilst considering risk and climate change (Rosenkranz et al. 2023).

Regarding growing stocks of Scots pine a decrease in scenarios (b, c, e) and a stable amount in scenario (d) is visible from the predictions. Regardless of risk, a dismantling management of coniferous stocks and especially pine is well known as other simulation results prove (Rudolph et al. 2015; Wördehoff et al. 2017; Oehmichen et al. 2018;

Rosenkranz et al. 2023) and retrospective studies show or experts anticipate (Höher 1994; Schmitz 2019). In this case macroeconomically important leakage effects are expected (Bösch et al. 2015). An oppositional increasing pine stock in cross-regional simulation studies was also found by Suárez-Muñoz et al. (2023) who predicted *Pinus* spp. development in Spain until 2100. Here the trend of light rising biomass in *P. sylvestris* was explained with comparatively young mean stand ages and more favorable climate was also considered. In fact, the present work also has highest mean ages in case of the rising pine stock. Suárez-Muñoz et al. (2023) furthermore state, that more intense management, as comparable to scenarios (b – c) in this study, led to higher FLR and lower growing stocks. Generally, due to the extent of warming and aridizising climate in RCP8.5, future volume development of pine will be hampered (Matías and Jump 2012; Haberstroh et al. 2022; Suárez-Muñoz et al. 2023). Comparatively, due to mixing of other tree spp., growth will be promoted (del Río et al. 2017; Diers et al. 2022; Vospernik et al. 2023) or drought vulnerability at least relieved (Steckel et al. 2020) which enlarges uncertainty. For old pine trees however, we only expect low effects of altered climate on SI and growing stocks (Albert et al. 2015).

The present work shows comparably low pine stocks for trees ≤ 31 cm DBH. We assume this to be probably a methodological artifact of *WaldPlaner*'s regeneration placeholders (see "Critical remarks on material and methods" section) and the result of only low pine ratio within every FDT, which was discussed by Wördehoff et al. (2017, p. 26).

We do not expect increasing stocks in diameter classes ≥ 41 cm to be caused by remnant trees, which are only up to 11 m. m^3 at expansion. Preserved areas with growing stock similar to Engel (2020, pp. 43–46) are also ruled out as explanation for the increase. Furthermore, especially large-dimensioned timber ≥ 50 cm, from our point of view results are due to exalted maturity definitions, since the cutting cycle and maximum removal amount per operation do not narrow down opportunities (Wördehoff et al. 2017, p. 23). Additionally, it has to be considered, that the 5 years volume increment is up to 50 m^3 ha^{-1} which also has to be removed at every cutting cycle. Moreover tree diameters ≥ 50 cm will challenge stakeholders in the forestry and timber cluster: Due to the popular profile hogging technology in central European sawmills, large-dimensioned timber reduces the feed rate or even prevents the cutting (Schrade 2002; Knocke et al. 2023). High valuable pine timber, especially stocking in pruned stands and having a higher target diameter of 55 cm, however requires other processing techniques. In 2012, this high-quality assortment accounts for only about 5 % of pine's total growing stock and can therefore also be ruled out as the cause of the increase in large-dimensioned timber where all qualities are subsumed.

To our knowledge, in reality a cutting cycle of 5 years cannot be met because organizational units have increased tremendously, as elsewhere (Lawrence 2017). Thus, uncertainty is heightened in terms of whether the thinning or final cutting operation takes place or not. Schall and Ammer (2013) indicate the need of high maintenance in pine forests. Especially with increasing site quality more silvicultural interventions are needed to increase stand heterogeneity which is linked to stability (Döbbeler and Spellmann 2002; del Río et al. 2017), resistance (Sohn et al. 2016; Hentschel et al. 2017) and biodiversity (Lust et al. 2001; Kint et al. 2006). Pretzsch et al. (2022) found asymmetric competition and mortality to homogenize Scots pine stands with increasing site quality. However, cutting cycles of 5 – 6 years are recommended (Kint et al. 2009; Huth et al. 2022) as well to promote mixed minority taxa.

Uncertainty or silvicultural plurality?

The comparison of relative differences in FLR explains its influence by silviculture to be four times as high as climate (Tab. 3.2). For *Pinus* spp. Suárez-Muñoz et al. (2023) found the same magnitude while also considering RCP4.5 besides RCP8.5. An interplay of climate models and management scenarios was also found by Garcia-Gonzalo et al. (2008) who optimized management plans that targeted timber production and carbon sequestration by simulating forest dynamics for 100 years. They found silviculture impact being twice as high as the climate component. Our findings express that high variability in silviculture and only little bandwidth from considered climate models are used for adapting forests towards climate change in northern Germany (Tab. 3.2). Thus, the range is not expressing the indecisiveness of State Forest Administrations but in fact it has been claimed as plurality and freedom to act (Ehwald 1949). For other stakeholders than forestry itself however, we interpret the differences as uncertainty. To our knowledge, especially inside regions, the relative decrease and the uncertainty range of the future outcome of forest dynamics, will affect the forestry and timber cluster as e.g. sawmills (Bösch et al. 2015; Leuschner et al. 2022). Another stakeholder *s.l.* is biodiversity where many species are relying on pine to an extent: After *Quercus* spp. and *Salix* spp. on light-transmission, Scots pine supports the most specialized insects and mites (Brändle and Brandl 2001), which increases with tree age (Turmukhametova et al. 2020). The visible spread in estimation width in scenario d – e in comparison to b – c is expressing success in FLR due to more variable forests as greater structural complexity arises when shade-tolerant trees are promoted (Pretzsch et al. 2016).

Accelerating FLR

Pure pine stands are particularly at risk in times of climate change which is mainly caused by drought, heat, insects, windthrow and fire (Hille 2006; Brandl et al. 2020; Brichta et al. 2023).

On a relative scale, pure stands are five times more present in the east resulting from historically different operational concepts (Nelson 2005). Regardless of the region Fig. 3.9 - Fig. 3.10 depict relative FLR progress being faster even under more dense overstories and longer cutting periods (scenario d – e; Tab. 3.1). Compared to common German forestry that considers stand layers, we reason this by disregarding vertical stand structures so that every evolving tree during the understory reinitiation stage (Stanturf 2016, p. 42) that holds compact wood is assessed. However, this does not state a *WaldPlaner* artifact but a realistic and layer-overarching FLR success. Also in scenarios b – c there might be pine regenerated with pine again which temporarily leads to pure stands even if the FDT does not target monocultures. Overall, FLR rate of about 65 %

within 50 years corresponds to other studies (Höher 1994; Vrška et al. 2017) and still experts call for quadrupling FLR in order to be finished by 2050 (Bolte et al. 2021). Generally, the FLR rate can be interpreted as the inertia and lag time until active forest management induces adaptiveness (Schelhaas et al. 2015; Puettmann and Bauhus 2023).

But much more than all that, the FLR gap between the regions remains similar. If the absolutely seen higher progress in the eastern states would be realized (for impediments see "Uncertainties in pine's cultivation area" section), which is highly questionable for both regions, still there are more pure stands remaining in the east. We explain this by the systemic lower thinning regime (Fig. 3.6; Garcia-Gonzalo et al. 2008; Kint et al. 2009; Suárez-Muñoz et al. 2023). In contrast to the accumulation of growing stocks and large-dimensioned timber, which is positively affected by the maturity definition, the FLR rate is thus negatively influenced by lower tree diameters and less future crop trees (as a result of thinning intensity i.e. hemeroby). We acknowledge that the states of Brandenburg and Mecklenburg-Western Pomerania have also defined target diameters that allow final cuts from about 40 cm DBH onward. However, in order to clearly identify effects of silviculture and climate change on forest development (research question *ii*), different target diameters within a region would complicate comparisons.

Admittedly, the shortening of rotation periods or reducing target diameters, as demanded frequently (Hartard and Schramm 2009; Bolte et al. 2009; Oehmichen et al. 2018) might have negative influences on the long run when timber harvest rate might decline (Bösch et al. 2017). Schelhaas et al. (2015) reported only 4 % higher FLR rate by shortening the rotations by 10 – 20 years. We also caution about the negative impacts on biodiversity if final cuts are brought forward in time (Mason and Alía 2000; Hartard and Schramm 2009; Turmukhametova et al. 2020). Additionally, pine's rotation periods in the recent decades already decreased due to frequent tending and thinning from above (Oehmichen et al. 2018).

In the literature, a target diameter framework for Scots pine from 40 – 60 cm DBH is occasionally proposed (Rudolph et al. 2015; Bartsch et al. 2020, p. 459). Therefore we line up and suggest, in a spatial coexistence, acceleration and retardation of final cuts both have to be achieved (Nagel 2016; Wördehoff et al. 2017). Hence vulnerable pine stands (e.g. high risk due to h/d-ratio, monotonous forest districts) and sheltered stands of high value and productivity side by side would meet economic and ecological demands.

All in all, target attainment ($A\%$) ranging from 63 to 70 % and restoration rate ($FLR\%$) ranging from 50 to 72 % show related pairs of values to each other. On the one hand this illustrates that forest restoration assessed at plot level is more difficult to reach than achieving the FDT target per subplot. On the other hand, values of $A\%$ and $FLR\%$ are overlapping and similar due to the often unmixed pine distribution in 2012.

3.5. Conclusion

Altogether, the present work confirms the expectation of European forests underlying tremendous changes in the 21st century. We showed that the effects of chosen climate model and silvicultural variant were modulated by forest planning, which had different influence on the resulting FLR.

Since uncertainty ranges from other climate-sensitive studies considering RCP4.5 - 8.5 (Suárez-Muñoz et al. 2023) are smaller than the presented funnel, we conclude that the ReKliEs-De RCP8.5 core ensemble spans sufficient, though not exhaustive, space. It also stands in line with typical forestry conservative predictions. Furthermore we conclude the following:

Ad *i.*)

1. No matter which bandwidth of silviculture variants (i.e. FDT guard rails) and climate models are chosen from forest planning,

Scots pine area will decrease further. Still, pine's key growing area will be on poor and dry sites in the north German lowlands.

2. In order to reduce uncertainty and promote γ-diversity, FDT sets between landscapes should be restricted.

Ad *ii.*)

3. The implementing of region-specific BA, i.e. thinning regimes for Scots pine, revealed different FLR rates if target diameters are equal.
4. If FLR rate should speed up, thinning intensity must increase and maturity definitions (proportion of trees above target diameter) need to decrease. Changing the target diameters is a less effective measure to increase FLR.
5. We call for a differentiated target diameter framework to meet economic and ecological goals equally by spatial coexistence of acceleration and retardation of pine's timber usage depending on dendrometric and site data.
6. If large-dimensioned timber share should not rise, especially maturity definitions must lessen and cutting cycles of 5 years must be followed.

Ad *iii.*)

7. In 2062, relative uncertainties in cultivation area and growing stocks are at most 5 % (Tab. 3.2). This might seem low but for local enterprises or ecosystems it can be very important how much pine and which diameters occur, especially when political or risk-based uncertainty is added in conclusion.

On balance further research is needed to specify timber amount and involve risk i.e. calamities in simulations and refine silvicultural prescriptions. With searing concern we point out the necessity to actively change forests in terms of climate change.

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Author contributions

HCK, RVN and MAl conceptualized the study. HCK performed the simulations and analyzed the data. HFH and MAx supported the FDT selection process. JHH adjusted the *WaldPlaner*. MAx, HFH and CF supported data processing and statistical analyses. HCK and MAl wrote the first draft of the manuscript. All authors contributed to the final manuscript version and approved it.

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Availability of data and materials

All NFI data described in this article can be openly accessed at https://bwi.info/Download/de/BWI-Basisdaten/. Operational data of Forest Administrations stay confidential.

Declarations

Conflicts of interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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3.7. Supplementary Information

Fundamental equations for the NFI estimation design

The mean of the target variable per stratum is

$$\bar{x}_h = \frac{1}{m_\mathrm{h}} \sum_{i=1}^{n_\mathrm{h}} \sum_{j=1}^{m_\mathrm{i}} x_\mathrm{hij} \qquad \text{(S3.1)}$$

where m_h is the total number of subplots in stratum h, n_h is the number of sample plots in stratum h, m_i is the number of subplots at sample plot i, and j is the index for the subplot.

The population mean is then obtained by weighting the strata means

$$\bar{x} = \sum_{h=1}^{L} W_\mathrm{h}\, \bar{x}_\mathrm{h} \qquad \text{(S3.2)}$$

with

$$W_h = \frac{A_\mathrm{h}}{A} \qquad \text{(S3.3)}$$

where L is the number of strata, A_h is the area of stratum h, and A represents the total error free land area of Germany. The variance of the mean is calculated by

$$v(\bar{x}) = \sum_{h=1}^{L} W_\mathrm{h}^2 \frac{1}{m_\mathrm{h}^2} \frac{n_\mathrm{h}}{n-1} \sum_{i=1}^{n_\mathrm{h}} (x_\mathrm{hi} - \bar{x}_\mathrm{h} m_\mathrm{hi})^2 \qquad \text{(S3.4)}$$

Addendum

Tab. S3.3 Summary of the significant parameter estimations from the nonlinear model on SI- and stand-age-dependent BA of used yield table systems

Parameter	**West** (Wiedemann 1943)				**East** (Lembcke et al. 1975)			
	y	SE	t-value	Pr(>|t|)	y	SE	t-value	Pr(>|t|)
a_1	17.94	0.26	68.76	$< 2*10^{-16}$	14.82	0.19	79.81	$< 2*10^{-16}$
a_2	-0.31	0.01	-35.87	$< 2*10^{-16}$	-0.74	0.00	-109.79	$< 2*10^{-16}$
b_1	0.036	0.00	12.68	$< 2*10^{-16}$	0.05	0.00	38.81	$< 2*10^{-16}$
b_2	0.023	0.00	12.25	$< 2*10^{-16}$	n. s.	n. s.	n. s.	n. s.
c_1	1.71	0.20	8.58	$< 1.52*10^{-13}$	4.36	0.44	10.00	$< 2*10^{-16}$
c_2	n. s.	n. s.	n. s.	n. s.	-0.03	0.00	-14.82	$< 2*10^{-16}$

Tab. S3.4 Overview of harmonized SNL systems, adapted: Schmidt et al. (2015)

key	denotation	key NI, SH	key BB, MV, ST	key HE
1	eutrophic	6	RC, KC	CE
2	eumesotrophic	5+, 5	R	EU
3	mesotrophic	5-, 4+	K	GM
4	oligomesotrophic	4, 4-, 3+, 3	M+, M, M_v, M-	ME
5	oligotrophic	3-, 2+	Z+, Z, Z-	SM
6	dystrophic	2, 2-, 1	A+, A, D	OT

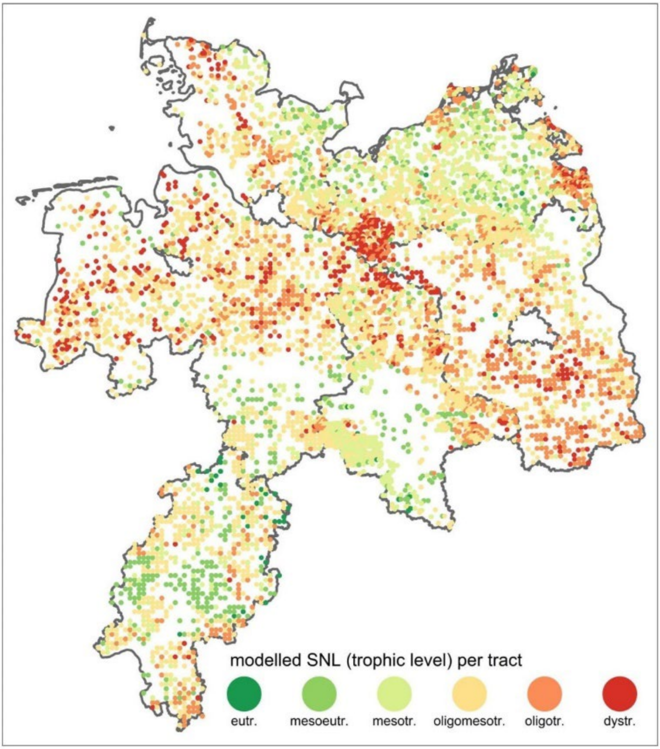

Fig. S3.11 Modelled SNL (trophic level) per tract in 2012 that remains constant throughout scenarios (b - e). Key adapted from Schmidt et al. (2015; *cf.* Tab. S3.4). SNL per plot is reported as fitted value, i.e. the most plausible score, resulting from an estimated cumulative logit model (Fahrmeir et al. 2013, pp. 312–316) with including plot number as random intercept component. This modeling step avoids manipulations, such as empirical mean calculation, that are not adequate for the ordinal scale of the SNL outcome. Note that space between points (tracts) does not state forest cover but sampling density

4. Schweregrad von Störungen (A2)

Erschienen als Artikel A2

„Contributing to the disturbance ecology
of Scots pine in Northwest Germany:
Analyzing practitioners' reports on damage severities"

✉ HC Knocke[1,2,3], H Sennhenn-Reulen[2], T Zeppenfeld[2], M Axer[2], GJ Langer[4], RV Nagel[2], M Albert[1,2]

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Hypothese H2

„*Die Waldentwicklung Norddeutschlands
wird bei der Kiefer unter Störungseinfluss derzeit überwiegend
von niedrigen Schweregraden bestimmt, deren Stärke
mit zunehmender Standortsgüte ansteigt.*"

Gegenüber dem Originalartikel gab es redaktionelle Änderungen.

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Abstract

Complex and interdependent disturbance factors drive the decline of Scots pine (*Pinus sylvestris*) under climate change. However, the magnitude of low- to high-severity disturbances in forests remains difficult to quantify. Forest managers require information to adapt staff as well as machinery deployment strategies, which vary based on whether entire stands collapse or single trees die. Despite its demand, comprehensive data to assess disturbance regimes remains scarce.

One decade after the introduction of the governmental hub 'Forest Health Notification Portal' (FHNP) of four Northwest German states, we take an exploratory approach to assess practitioners' reports via three research questions: *i.* Which disturbance factors lethally affect pine forests? *ii.* What are the causal influences behind forest mortality? *iii.* Can a robust distribution of damage severities be derived from FHNP entries? Spatially linking damage reports, management plans, and climate covariates created a unique dataset, which was analyzed via a beta distribution regression model.

Our findings align with national logging statistics and related studies: *i.* As a trigger for damage, biotic factors approach storms in both frequency and severity. *ii.* Damages are influenced by admixture competition, and by the ecotonal status of pine, which attenuates under positive climatic water balances. While non-wind disturbances cause more severe stand-scale damage, storm remains the dominant overall driver of timber loss. *iii.* Lethal damage severities average 0.5 ha but have a median of just 0.12 ha, reflecting a left-skewed density function modeled with a Bayesian R^2 of 38 %. The results suggest that meta-level limitations of the dataset are negligible.

Keywords *Pinus sylvestris*, mortality, disturbance severity, beta distribution, climate change

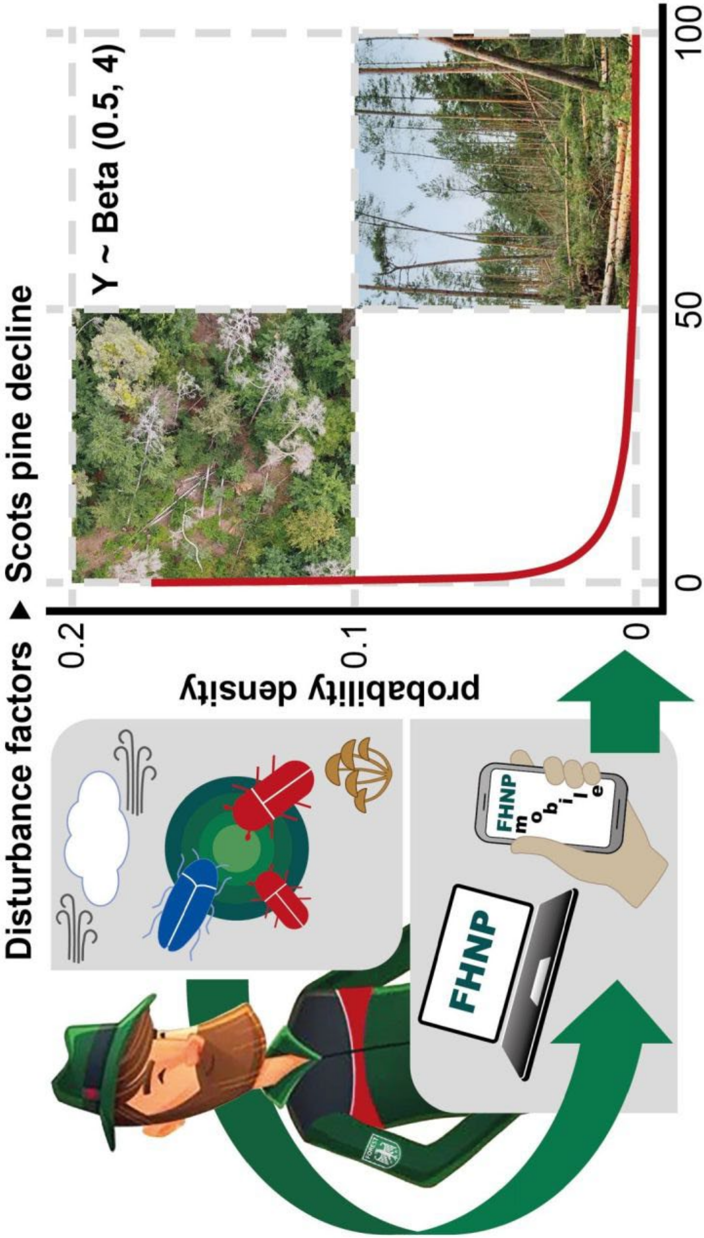


Fig. 4.0 Graphical Abstract “Schweregrad von Störungen (A2)”

Article highlights

- → in Northwest German forests, damage records amassed 600,000 observations since 2014
- → foresters' crowd sensing serves to explain forest damages
- → globally, in stands with *Pinus sylvestris* share, storms cause the most disturbances
- → locally, damages of non-wind and biotic factors are more severe and catch up
- → a complex pine decline, with the 2018 megadrought, causes many low-severity damages

4.1. Introduction

Forest development in the 21st century is increasingly shaped by disturbances (Marchau et al. 2019; Senf et al. 2021; Zeppenfeld et al. 2023). Alongside internal drivers, such as the forest's own composition and its site (Mueller-Dombois 1987; Knocke et al. 2024; Vangi et al. 2024), external disturbance factors like stress and eventual tree death can exert substantial impacts on forest development, with the determinant being climate change (Anderegg et al. 2022; Pretzsch and Grote 2023, p. 367; Patacca et al. 2023; Albrich et al. 2023).

Agents (e.g. storms, insects) can directly cause disturbance events with effects (i.e., deviations from the initial state, stress, diseases, mortality) being the response (Rykiel 1985). The term 'factor' also includes *in*direct abiotic and biotic causes, such as droughts, pathogens (e.g. latent fungi), and pests; the latter refers specifically to animals (Tainter and Baker 1996, pp. 793–800). Among animals, insects can severely disturb forest ecosystems through mass outbreaks (Jaime et al. 2019; Hartmann et al. 2025). Henceforth, disturbance 'factor' is used as the general term to causally explain tree death unless otherwise specified. Depending on the intensity of the factor's impact, disturbances can result in economically measurable forestry damage (Möllmann and Möhring 2017; Wohlgemuth et al. 2022, p. 178), which indicates the severity of the event (Turner 2010). In the face of climate change, the understanding of mortality causes, disturbance processes, and effects is crucial for the preservation of forests. Generally, biotic disturbance factors increase under a warming and aridizing climate disproportionately (Jabłoński et al. 2019; Bat-Enerel et al. 2022; Patacca et al. 2023).

Due to interacting cause-effect relationships, tree death is a complex process (Anderegg et al. 2015). According to the ‘decline disease spiral’ of Manion (1991, p. 333), predisposing and inciting (primary), as well as contributing (secondary) factors can be distinguished. For example, senescing Scots pine (*Pinus sylvestris* L.) trees of old age (long-term predisposition) suffer a drought event (short-term incitement) and die as a result of the additional infection with *Heterobasidion annosum* (Fr.) Bref. *s. str.* (long-term contribution). The latter is a soil-borne fungal pathogen that causes annosus root and butt rot in pines (Asiegbu et al. 2005), which reduces the tree's water supply and is a major cause of damage, particularly during periods of drought. This makes it difficult to state a single cause of death (i.e., etiology); whether a pine died *with* a certain preexisting factor or has died *due to* it (Manion 1991; Bigler et al. 2006; Langer 2018; Gomez-Gallego et al. 2022). Therefore, disturbance ecology that is not based on experiments can only be partially reduced to individual stressors (Mueller-Dombois 1987).

Pine is the most abundant tree species in Germany (BMEL 2024) and of the highest ecological and economical importance (Brändle and Brandl 2001; Brichta et al. 2023). Compared to other tree species, pine is relatively resistant to extreme conditions but manifests low competitiveness. This is why its occurrence is originally restricted to hydric, dystrophic, or xeric sites (Leuschner and Ellenberg 2017, p. 185). Furthermore, it is considered adaptable and drought in-sensitive (Albert et al. 2017; Bose et al. 2020b). However, prolonged water shortages can cause irregular mortality, as reviewed from Bose et al. (2024). In the future, for Germany, current pine survival models predict probabilities ranging from 62 % (Schoneberg 2017) to 73 % (Brandl et al. 2020), for trees reaching 100 years of age. These models all examine biotic and abiotic disturbance factors together, which means not estimating storm *or* bark beetle survival chances separately. Generally, resource allocation and deployment strategies must be adjusted based on whether the focus is on sanitation cuts of entire stands, or individual trees (Fuchs et al. 2022), or on salvaging for residual timber value.

For identifying adaption strategies, management options, and for enabling policy impact assessments, survival models alone are insufficient. Damage severities conditional on the lethal disturbance event are also required and should be considered (Schoneberg 2017, p. 91; Fuchs et al. 2022). Severity expresses the ratio of disturbed (e.g. uprooted) and undisturbed stand shares. In simulations, damage severities make it possible to distinguish whether stands experience complete or partial mortality. There is evidence that, in temperate zones, the frequency of diffuse (small-scale, low-severity) damage will dominate in the long run, while discrete (large, high-severity) disturbance patches will account for the majority of the area within the affected region (White and Jentsch 2001, p. 409; Kimmins 2004, p. 539). Until now, short-term and large-scale disturbances have shaped perceptions and policy-making (Sommerfeld et al. 2018; Popkin 2021; Michler and Aschenbrand 2024). However, empirical data on disturbance patterns is limited. Timber acquisition data is hard to link to specific stands – especially after large-scale disturbances – and inventory data, depending on its sample design, does not always allow for stand-wise analysis (Prodan 1968; Fuchs et al. 2022 suppl.). Apart from ground-based surveys, remote sensing approaches can detect damage severities as well, but are often limited to canopy attributes (Senf et al. 2021; Patacca et al. 2023). Thus, they neglect the understory or harvest/ salvage causes to some extent, and are bound to a sufficient ground resolution aligned to the single tree crown width (Ferretti et al. 2024; Seidl and Senf 2024).

In the German state forest services, ‘stock bookkeeping’ data has been meticulously maintained. Alongside management plans, it includes practitioners’ damage reports for each stand, with their evaluation grounded in a long-standing tradition (Hennert 1797; Schwerdtfeger

1963). In 2014, the governmental web-based hub 'Forest Health Notification Portal' (FHNP) was introduced in the federal states of Hesse (HE), Lower Saxony (LS), Saxony-Anhalt (ST), and Schleswig-Holstein (SH). It was mandated to standardize processes and track disturbance trends, and in 2018 an open-source mobile application was added (ML 2014; Hildebrandt et al. 2022). While initially intended for descriptive analysis (Langer 2018; Rohde et al. 2023), statistical modeling was not a primary objective. After one decade, however, the FHNP has amassed 600,000 observations. More recently, state forest services have integrated amateur observations alongside expert reports for publicly owned stands, introducing uncertainties similar to those in citizen science (de Groot et al. 2022; NW-FVA 2023).

We build on the structured web-based forest health reporting via FHNP and the potential from the thus far unexplored dataset to derive tree species-specific damage risks. Therefore, we adopt an exploratory and inventive approach to unlock its full potential for decision support. In order to do this, we inductively derive robust relationships, by addressing the following questions:

i. Which main factors dominate the lethal disturbances of pine since 2014 in Northwest Germany as recorded in the FHNP?

ii. What are the causal influences driving pine's mortality, their conditional frequencies of occurrence, and the associated uncertainties given a mortality event?

iii. Can damage severities be modeled on the notification basis for pine to quantify a robust magnitude of low to high damages?

In order to answer the three questions we combine damage notifications with forest management plans as well as site mapping data and use climate indicators as an extension.

4.2. Material and Methods

4.2.1. Study region

Our study includes the Northwest German federal states HE, LS, SH, and ST. These states account for 24 % of the total German forested area and 26 % of the trees in this area are Scots pine (Fig. 4.1).

We focus on the federal state-owned forests, comprised of ca. 850,000 hectares, lying in the lowlands and central uplands. In these areas, Scots pine accounts for 16 % of all trees (BMEL 2024). The climate is oceanic to hemiboreal (subcontinental; Beck et al. 2018). For the period of 1991 - 2020 it can be distinguished by 9.5 °C mean annual air temperature with 733 mm m^{-2} a^{-1} average precipitation per year. The hottest climate occurred in southern HE, with a temperature of 11.5 °C, whereas the driest sites were found in central ST with 449 mm m^{-2} a^{-1} at a 30 year average (DWD 2022). Thus, a cold semi-arid steppe climate arises sporadically in ST (Beck et al. 2018). Soils with a wide range of water and nutrient supply have developed in the study area (Knocke et al. 2024 suppl.).

4.2.2. Database

For the sampling design, the database is created by spatially linking entries from the FHNP with stand-wise forest management plans and environmental information (Fig. 4.1c) using the statistical environment 'R' version 4.2.1 (R Core Team 2022). In order to classify modes of the lethal factors (research question *i*), the logging statistics from the German federal statistical office are contrasted (Destatis 2024). For modeling, we focus on the basic unit of management in forestry, the stand.

Forest Health Notification Portal

The FHNP (https://www.nw-fva.de/unterstuetzen/software/wsmp) is a presence-only database. Notifications (i.e., approximate observations) for Scots pine damages are shown in Fig. 4.1 and were

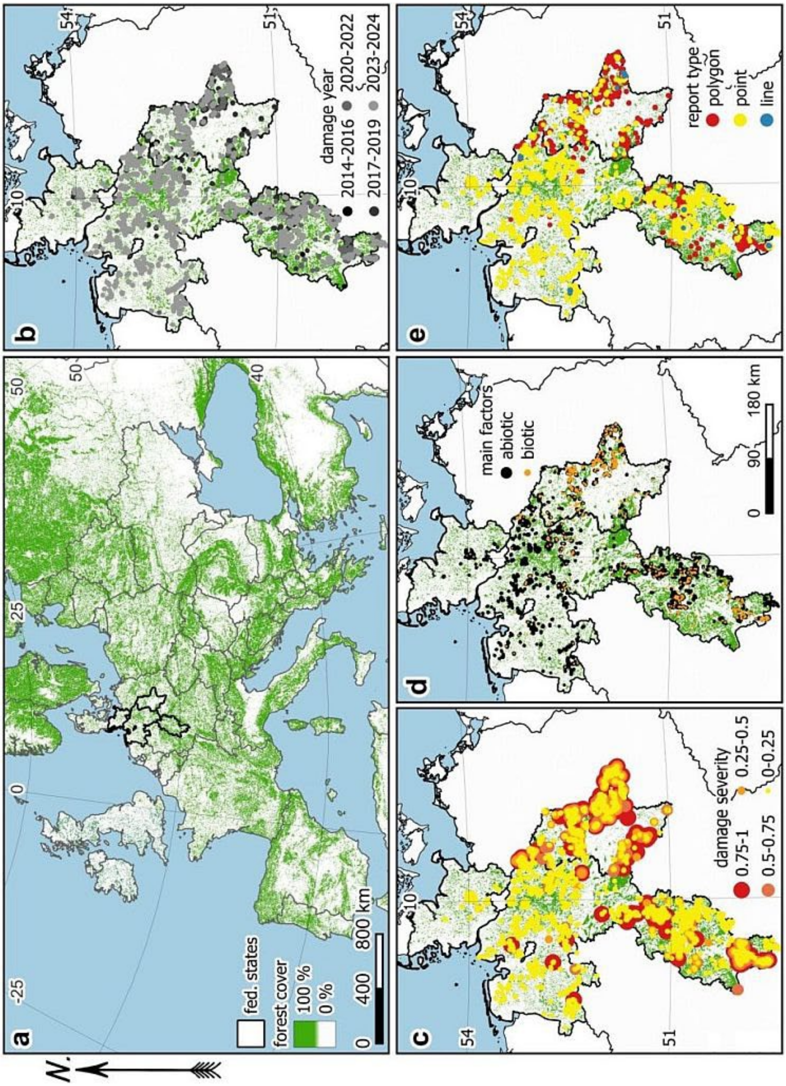


Fig. 4.1 **a)** European forest cover (Burgoin et al. 2024) and study area (sampling design) with the German federal states (black outlined polygons from north to south): Schleswig-Holstein (SH), Lower Saxony (LS), Saxony Anhalt (ST) and Hesse (HE). Reported damages of Scots pine classified by **b)** year, **c)** severity relative to stand attributes, **d)** main factors, and **e)** report type; borders by © GeoBasis-DE/BKG (2022) & EEA (2018)

submitted by practitioners, such as staff (academics/ foresters and non-academics), contract workers or private hunters. In the following, it is assumed that the damage is expressed as net (excluding gross non-stocked areas) and referring to the upper canopy (i.e., main stand, overstory; *cf.* Fig. 4.2), and pertais exclusively to mortality.

In total, for state-owned Scots pine stands, 28,280 observations for periods of less than one year, derived from 21,012 notifications across 11,513 stands, result in 13,916 annual observations among the stands from 2014 - 2024. See supplementary materials for details on the FHNP.

Relating Damages to Forest Management Plans

For obtaining damage severities, we linked the practitioners' lethal disturbance observations on *Pinus sylvestris* (psy) per annum and stand, denoted i_{psy}, with the associated forest management plan after spatial intersection of the data. Hence, disturbance events throughout the year are aggregated into *one* annual observation per stand by retaining the most lethal cause with its coordinate. This serves to balance the notification behavior among users (research question *i.*). Reported polygon and line geometries were simplified into a point-on-surface coordinate using the R-package 'sf' (Pebesma 2018), and thereafter treated equally to the notifications with point geometries provided. For data consolidation, we excluded implausible and undifferentiated large collective postings for more than 20 stands, which correspond to an average area of over 100 ha.

Additional entries e per observation are facultative and then nonsingular, i.e. $i_{v,psy} \vee i_{n,psy} \vee i_{a,psy}$, as estimated in units of volume v, stem number n, or area a. For the most disturbance factors, i_v is mandatory to assess.

Forest management plans are compiled every 10 years and may be outdated by Δt years relative to the damage date. They also lack accounting entries. Therefore, for volume and stem entries, the plans were updated by adding the annual volume's periodic increments $p\iota V$ and then subtracting planned harvests H in a linear and proportional manner up to the damage date. Thus, observations $i_{e,psy}(e \in \{v, n, a\})$ can be set in relation to the updated total volume of all tree species in the stand V, the total upper canopy stem number N, or the stocked net area A of the geometry (eq. 1 - 3).

The disturbance (damage) severity is defined as the "effect of the disturbance event on the organism" (Turner 2010). This, in a forestry context, specifically results in damage (Möllmann and Möhring 2017). We compute the damage severity of all biotic and abiotic disturbance factors (Fig. 4.3) of our data set combined, denoted as y, conditional on the observed totalized disturbance event i_{psy} per stand, by dividing Scots pine's proportion by the aggregate of all species, as follows:

$$y_v = \frac{i_{v,psy}}{(V+(p\iota V\cdot\Delta t)-(H\cdot\Delta t))\cdot 0.8} \tag{4.1},$$

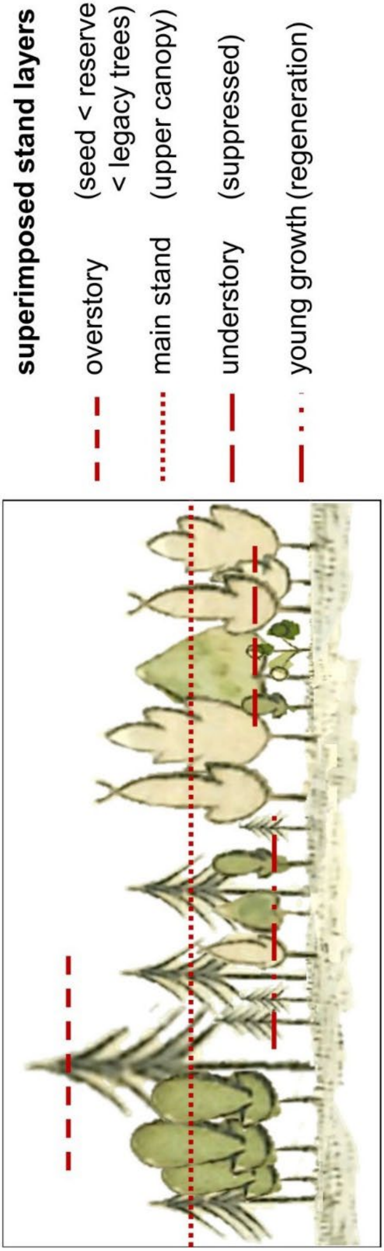
or $$y_n = \frac{i_{n,psy}}{N-(H\cdot\Delta t)} \tag{4.2},$$

or $$y_a = \frac{i_{a,psy}}{A} \tag{4.3}.$$

The factor of 0.8 in formula (4.1) represents the reduction of growing stock to harvested timber (gross to net volume). This is because estimates are submitted in net volume while the management plans record gross volume (Fuchs et al. 2023). In order to compute a *mean* of the conditional damage severity $\bar{y}$ among nonsingular entries e per yearly observation, we weight:

$$\bar{y} = \frac{1}{e}\sum_{j=1}^{e} y_e \tag{4.4}$$

If one observation differs from the others by 20 %, it is considered an implausible value. We, therefore, link the inclusion in the mean value for $e = 3$, with the condition $\{y_e \mid 0.8\, y_e \leq \bar{y} \leq 1.2\, y_e\}$ as a measure of implausible data correction. Because entries are estimated without knowing the exact stocks, values of more than 1 may result; these values have been corrected to 0.999. For symmetry, we increased all values of $\bar{y} < 0.001$ to $\bar{y} = 0.001$. Mean damage severities are shown in Fig. 4.1c.



Stand-related explanatory variables were derived as follows: In order to create density measures, we decided to standardize the extent (spatial size) of all layers by means of their yield table stocking density sd_{yt} to the full polygon size. In combination with maximum basal areas (Wördehoff 2016, p. 50), this allows to specify the natural stocking density (Pretzsch 2009, p. 267) sd_{nat}. This process is depicted in Fig. 4.2, which also used rates of Speidel & von Laer (1959, p. 573) to derive basal areas in certain layers (e.g. the overstory) where only the top height and growing stock was known. The derivation of further stand-related covariates is explained in the supplementary materials. Stand characteristics are shown in Tab. 4.1.

Fig. 4.2 Horizontal (species) and vertical (layer) stand structures according to German forestry that, for understanding management plans, illustrates different monitoring units with possibly diverging stocking densities and areal extent. Even-aged silvicultural systems (e.g. seed-tree or uniform shelterwood) differ from uneven-aged (continuous cover) systems (e.g. irregular/reserve shelterwood, group selection) in the quantity and timing (<) of remnant mature trees in the subsequent stand (Puettmann et al. 2009, pp. 23–40; Wagner et al. 2010; Nyland et al. 2016, pp. 319–342; Aszalós et al. 2022). Figure based on Guimpel (1819)

Site and Climate data

Tab. 4.1 Site, weather and climate characteristics at the notification point geometry (SD: standard deviation, Q: Quantile, site index for stand top heights at 100 years of age, V: total net growing stock, ΣP: precipitation sum, $\bar{T}$: mean air temperature for t_{cn} the individual last 30 years climate normal before decease). Stand characteristics refer to Scots pine.

	$\bar{T}\_t_1$ [°C]	$\bar{T}\_t_0$ [°C]	ΣP_t_0 [mm m^{-2} a^{-1}]	$\bar{T}\_t_{cn}$ [°C]	ΣP_t_{cn} [mm m^{-2} a^{-1}]	altitude [m]	V_{net} [m^3 ha^{-1}]	site index [m]	mix size [ha]	stand age [a]
$Q_{0.975}$	11.8	12.1	986.3	11.1	884.0	413	395.8	33.7	14.1	163
mean	10.3	10.4	641.2	9.6	683.4	130	234.6	27.4	3.7	85
SD	0.7	0.8	161.7	0.6	104.2	108	90.3	3.5	3.9	34
$Q_{0.025}$	8.9	8.7	354.0	8.4	533.0	25	12.5	20.5	0.1	30

For statistical modeling, historic, site-specific climate data from the German weather service (Sutmöller et al. 2021; DWD 2022) were used as further explanatory variables for the modeling process. Site and climate characteristics of the observation site are summarized in Tab. 4.1.

Forest stands with Scots pine shares exhibit a small range for age and site indices but a large range for altitude. Within mono- and multi-specific stands, Scots pine's 'mix size' ranges from 0.1 - 14.1 ha. Values for the year before death (t_0) show lower precipitation sums and higher mean air temperatures, with wider ranges compared to the 30-year climate averages (t_{cn}). In Northwest Germany, 2018 marked a megadrought event (Senf and Seidl 2021; Bose et al. 2022), with persistently dry conditions in the following years (DWD 2022; Sachsenmaier et al. 2024). For each observation site, climate and weather covariates were calculated using a moving window approach (Möller et al. 2017), where t_0 represents the 365 days prior to the damage date, and t_1 corresponds to the penultimate year, resulting in 30-year climate normal windows (t_{cn}) from 1984 - 1994.

As several studies on Scots pine highlight its sensitivity to winter weather anomalies (Zeppenfeld et al. 2023; Diers et al. 2024; Gribbe et al. 2024; Enderle et al. 2024; Martinez del Castillo et al. 2024b), we conclude that the non-vegetation period should also be considered.

Next, the aridity index according to de Martonne (1926) was derived. The available water capacity (awc) used in this study refers to the effective root zone of 1 m and was collected from forest management plans. Here, on azonal (hydric) sites, where growth, and consequently also health (Weller et al. 2022; Pajtík et al. 2022), is dominated by ground and backwater, awc was already set uniformly to 300 mm m^{-2} m^{-1}. The alkaline soil nutrient levels (snl; eutrophic, eumesotrophic) were aggregated as mesotrophic due to the small sampling size. We compute the climatic water balance (cwb) as the sum of precipitation and negative potential evapotranspiration (pet), following the Penman-Monteith equation (1948; 1965; Allen et al. 1998). Historical winter maximum gust speeds in t_0 (gs_{max}), as obtained from Jung and Schindler (2019), were attributed to damages that listed windthrow as a cause of death. Damages that were not storm-related (e.g. bark beetles) were assigned 0 m s^{-1} for this covariate and missing storm damage covariates (e.g. summer storms) were set to the minimum recorded value of 10 m s^{-1}.

4.2.3. Modeling approach

In order to identify a robust, cause-effect relationship among the calculated damage severities, we correlated $\bar{y}$ with potential covariates.

Damage severities follow a beta-function pattern (Kimmins 2004, p. 539; Fuchs et al. 2022) as a distributional model, which conditions two distributional parameters from covariate values using a distributional regression (Kneib et al. 2023). For the data sets, the continuously scaled response variable Y takes on observations $0 < y < 1$. Hence, the distributional additive beta regression model was fitted as:

$$Y_i \sim Beta(\mu_i, \varphi_i) \tag{4.5}$$

with the first distributional parameter, $0 < \mu_i < 1$, modeling the conditional expectation, and the second distributional parameter, $\varphi_i > 0$, modeling the scale and shape of the conditional distribution. The model for distributional parameter $\theta_i \in \{\mu_i, \varphi_i\}$, is derived as:

$$g_\theta(\theta_i) = \eta_{\theta,i} = \beta_{0,\theta} + f_{1,\theta}\left(x_{1,i}\right) + f_{2,\theta}\left(x_{2,i}\right) + \cdots \tag{4.6}$$

where η_θ is the parameter-specific additive linear predictor, $\beta_{\theta,0}$ is the parameter-specific intercept, and x_1, x_2, … are covariate values. The effect function f – nonlinear for continuously scaled covariates, and a batch of regression coefficients for categorically scaled covariates – allows an interpretation of the partial and population-level effect of predictor (covariate) X on the damage severity by Y (research question *ii.*). Finally, g_θ is a parameter-specific link function set to the logit, and logarithmic function, respectively, leading to:

$$\mu = g_\mu^{-1}\left(\eta_\mu\right) = \frac{\exp(\eta_\mu)}{1+\exp(\eta_\mu)} \tag{4.7},$$

and $$\varphi = g_\varphi^{-1}\left(\eta_\varphi\right) = \exp\left(\eta_\varphi\right) \tag{4.8}$$

with linear predictors η_μ and η_φ.

The conditional magnitude of damage severities, in which we are mainly interested (research question *iii.*), is characterized by the probability density function

$$p(y) = \frac{y^{\mu\varphi-1}(1-y)^{(1-\mu)\varphi-1}}{B(\mu\varphi,\ (1-\mu)\varphi)} \tag{4.9},$$

wherein B is denotes the beta-function.

We estimated the models – using the R-package ‘brms’ (Bürkner 2021) – using the Bayesian approach to statistical inference. This allows us to set up the previously-described flexible model, which can perform post-estimation calculations on the basis of posterior samples.

Variable selection and model fitting

Tree mortality is usually explained by tree size, age, environment and competition (Burkhart and Tomé 2012, pp. 282).

Bose et al. (2024) discuss long-term predispositions from soil and management legacy as relevant biological mechanisms behind stand-level mortality processes of Scots pine. In addition to that, other studies distinguish long-term site (including climate) and short-term weather factors (Greenwood et al. 2017; Bose et al. 2020a; Enderle et al. 2024) as well as stand characteristics (Merlin et al. 2015; Archambeau et al. 2020; Zeppenfeld et al. 2023) for mortality events in Scots pine communities. Drought weather anomalies were found to especially affect Scots pine's growth, vitality and mortality between three (Archambeau et al. 2020; Gribbe et al. 2024; Enderle et al. 2024) and up to five years (Wiedemann 1948, p. 67; Wessolek et al. 2024).

Our approach aims to access factors that influence the probability of the notification and magnitude of the reported damage. Since the four federal states are equipped with different finances, staff/ contractors, and technology (e.g. mobile rollout; cf. Daume 2016), we account for meta-level covariates, originating from the sampling design itself (Caley et al. 2020; Dobson et al. 2020; de Groot et al. 2022). Namely, the damage date's day-of-the-year doy, the federal state $fstate$, and the polygon size a_{stand} in terms of detectability.

The covariates which were available are listed in Tab. 4.2. Covariates of the final model are underlined and their abbreviations are given in parentheses. Factors, spline complexity, and mutual interactions explaining the damage severity were iteratively included in the model for variable selection. See supplementary materials for model comparison criteria.

Among the covariates, gs_{max} and ari_t_0, cwb_t_{cn} and ari_t_0, as well as cwb_t_{cn} and et_t_1 were correlated by $|> 0.5|$. Detrending of factors (e.g. considering differences) and evaluating interactions weakened the model based on its leave-one-out information criterion values or reduced the robustness of the probability density curve. Thus, effects of these covariates are interpreted with respect to their mutual correlations. snl and $fstate$ were included as random intercept variables to emphasize grouping structures within the data base.

Model fitting was based on eight chains with 1,000 iterations each (500 warmup, no thinning), yielding 4,000 post-warmup samples.

Tab. 4.2 Assessed and finally selected (underlined) covariates. Grouped by factors according to Manion (1991, p. 333), mechanisms (Bose et al. 2024), and influences (Caley et al. 2020; Dobson et al. 2020). We selected as additive and/ or interaction predictors (temp.: temperature, prec.: precipitation, cn: climate normal 30-year period)

factor	**mechanism/ influence**	**covariate**	
predisposing	soil	- soil nutrient level (snl)	- available water capacity (awc)
	site	- site and climatic water balance t_{cn} (cwb) - altitude	- site index - aridity t_{cn}
	stand	- mean age, age span - top height (h_{top})	- mixture size - growing stock
	management legacy	- volume mixing ratio (v_{mix}) - natural stocking density (sd_{nat})	- target density level
inciting & contributing	weather t_0, t_1 or t_2	- aridity t_0, (ari) - gust speed (gs_{max}) - temp.: max., min., mean, sum, range - prec. sum, pot. evapotrans. sum t_1 (et) - winter (dec.-jan.-feb.) sum: temp., prec., pet	- diff. t_{cn} to t_0 for cwb and site water balance, temp., prec., pet - range t_0 to t_2: temp., prec., pet
meta-level		- day of the year damage date (doy) - stand size (a_{stand})	- federal state ($fstate$) - easting, northing

Model performance

Our database was stratified by climatically dry and moist sites (according to negative or positive cwb_t_{cn}) and by federal states. Equally sized, random data sets for model selection (training) and inference (testing) were selected, with balancing state and the 30-year average all-season climatic water balance used for stratification. The training data set comprised of $n = 6{,}956$ and the testing data set of $n = 6{,}960$.

Furthermore, a sensitivity analysis was carried out that assessed the robustness of results (i.e., magnitude of low to high damage severities in the probability density function). Instead of mean values, the 2.5^{th} and 97.5^{th} percentile values of the empirical sample for the respective covariate were used. The explained variance with a 95 % compatibility interval was calculated following McKelvey & Zavoina (1975) for a Bayesian pseudo R^2 and an adjusted (unbiased) R^2 using a leave-one-out (loo) prediction approximation (Gelman et al. 2019).

Measured by the standard deviation of a covariate's estimates within the empirical 95 % percentile – while keeping other covariates at their mean values – the final model yields the following order of explanatory power:

$$\begin{aligned}\eta_{\mu} = \beta_{0,\mu} &+ f_{1,\mu}(v_{mix}) + f_{2,\mu}(cwb\_t_{cn}) + f_{3,\mu}(ari\_t_0) + f_{4,\mu}(fstate) \\ &+ f_{5,\mu}(awc) + f_{6,\mu}(gs_{max}) + f_{7,\mu}(pet\_t_1) + f_{8,\mu}(h_{top}) \\ &+ f_{9,\mu}(a_{stand}) + f_{10}(doy) + f_{11,\mu}(sd_{nat}) + f_{12,\mu}(snl)\end{aligned} \tag{4.10}$$

with snl and $fstate$ as random effects. The following Tab. 4.3 summarizes the estimated hyperparameters, which control the flexibility or variability of the final model components; all $\hat{R}$-values indicate satisfactory convergence.

Model validation steps, priors and the model summary are listed in the supplementary materials.

Tab. 4.3 Smoothing spline and multilevel hyperparameters (smooth terms) for μ in the final model of the inference dataset (sds: standard deviation spline (fixed effects), sd: standard deviation (Intercept, random effects), CI: credible interval, $\hat{R}$: Potential Scale Reduction Factor)

Parameter *sds* **&** *sd*	**Estimate**	**Estimate Error**	**Lower 95 %CI**	**Upper 95 % CI**	$\hat{R}$
doy	0.73	0.74	0.16	2.81	1.00
a_{stand}	2.80	2.39	0.73	7.90	1.00
v_{mix}	4.40	2.60	1.82	10.89	1.00
sd_{nat}	0.36	0.28	0.03	1.10	1.01
h_{top}	3.77	1.90	1.62	8.45	1.00
awc	2.36	1.14	1.02	5.40	1.00
gs_{max}	2.73	1.53	1.05	6.69	1.00
cwb_t_{cn}	1.80	1.70	0.10	6.45	1.00
pet_t_1	1.11	1.42	0.02	4.66	1.00
ari_t_0	2.77	1.92	0.74	7.73	1.00
fstate Intercept	1.10	0.62	0.44	2.83	1.00
snl Intercept	0.27	0.26	0.08	0.95	1.00

4.3. Results

4.3.1. Disturbance factors

After reporting lethal damages through the FHNP from 2014 - 2024, a net of about 1,554,000 m³ of dead Scots pine trees were documented. Among all forest health reports, only 3 % lack i_v-entries, indicating no volumetric measurability (e.g. young stands). After data consolidation (see section 'Relating Damages to Forest Management Plans'), we included 691,462 m³ in our dataset, accounting for approximately 45 % of all volume-based FHNP observations (*cf.* Fig. 4.1).

Fig. 4.3 illustrates the primary factors, which are the most lethal causes for the particular Scots pine stand. Herein, storm (i.e., uprooting and breakage) is the most significant disturbance agent in terms of both frequency (Fig. 4.3a) and global magnitude (Fig. 4.3b, c). Notably, 2018 and 2020 stand out for their cyclone (storm-induced) damages, as reflected in both logging records (Fig. 4.3b) and FHNP data. The damage patterns (modes) between these two accounting areas are similar from the perspective of disturbance factors and peaks, although the FHNP reports consistently lower quantities. Frost-induced death, such as snow breakage, only played a minor role. In 2018, drought reports was newly introduced as a lethal cause. Since then, fire takes second place as the disturbance agent in this abiotic, short-term group.

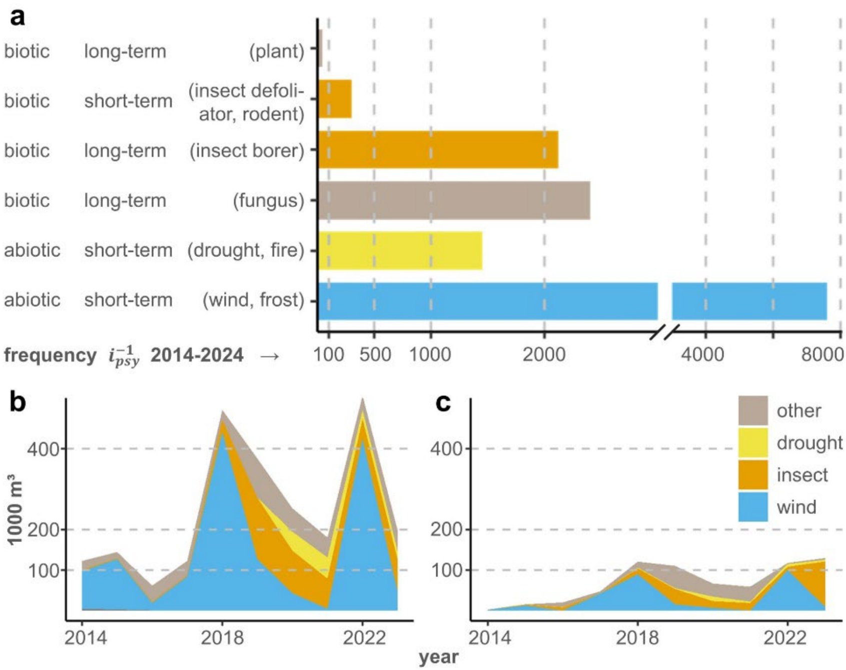


Fig. 4.3 **a)** Factors of lethal disturbances by observation, and b-c quantity by year. While the conditional frequency i_{psy}^{-1} (n = 13,916) of factors is grouped by temporal and etiologic mode of action (Manion 1991), the net timber volume is grouped according to the causes of damage from the federal statistical office. **b)** For salvages and sanitation cuts, the accounting areas ‘logging statistics’, as derived from Destatis (2024), and **c)** our database of ‘health records’ (total v = 691,462 m³), are compared

Since 2018, biotic disturbance factors on Scots pine have shown an increasing trend, with fungi being the primary contributors. In contrast, defoliating insects – alongside rodents as the only other biotic incitement factor – account for a relatively small number of observations (n = 283). From 2014 - 2024, the main agents behind the other groups are annosus root rot (causal agent: *H. annosum,* Russulales, Basidiomycota) and Diplodia tip blight (causal agent: *D. sapinea* (Fr.) Fuckel; Botryosphaeriales, Ascomycota) for fungi. Boring insects are represented by bark (Scolytinae, Coleoptera) and jewel beetles (Buprestinae, Coleoptera). Defoliating insects were mainly reported as sawflys (Diprionidae, Hymenoptera) and plant pathogens as the hemiparasitic mistletoe (*Viscum*, Santalales).

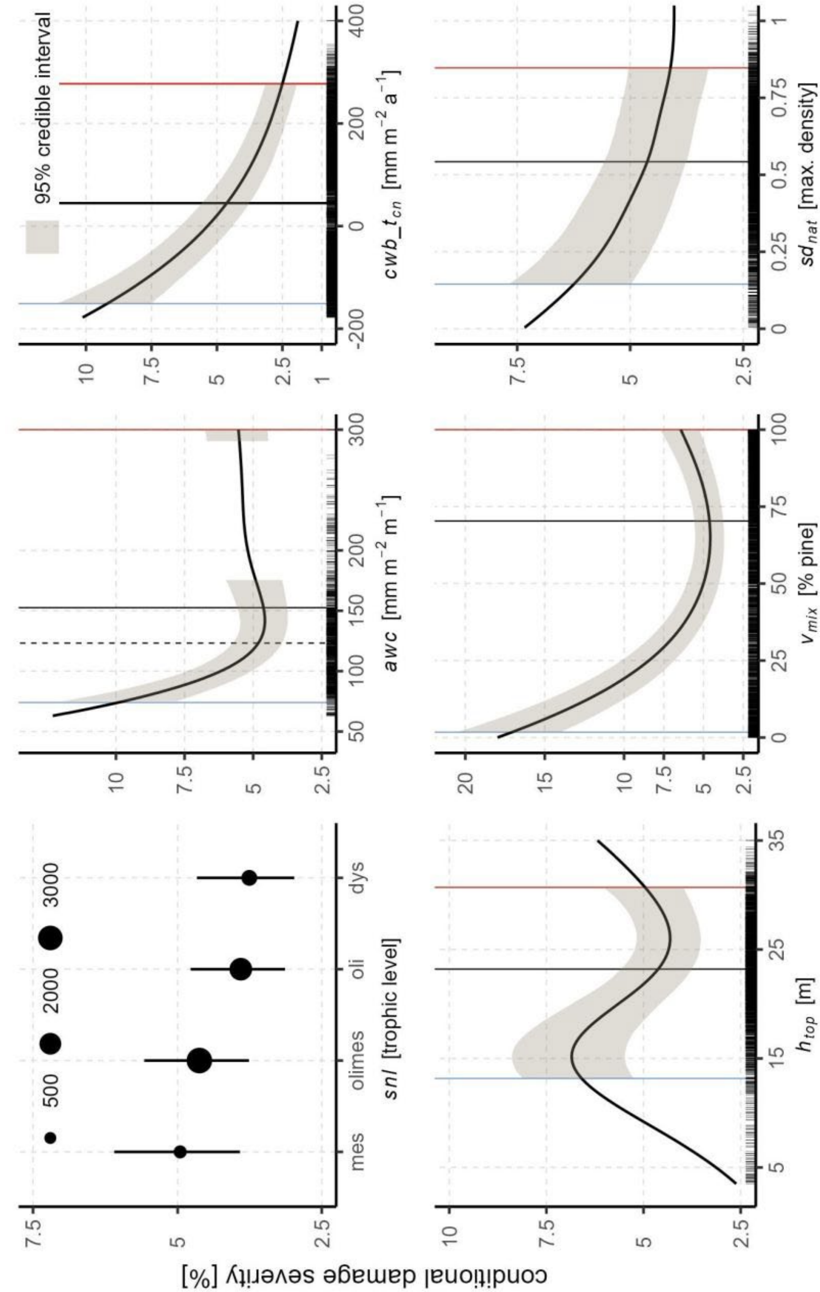
Multiplying empirical damage severities by their respective stand sizes yielded a potential mean damage area of 0.5 ha, with extremes ranging from 0.6 m^2 to 26.4 ha (median: 0.12 ha; agglomerations of neighboring polygons excluded).

4.3.2. Effects on damage severity

For the distributional parameter μ, the final model of the inference dataset is characterized by the following population-level regression coefficients, as shown in Tab. 4.4. The complete model summary is listed in the supplementary materials. The explained variance R^2 of our final model amounts to 0.378 ± 0.007 and the loo-adjusted conditional R^2 to 0.203 ± 0.018.

Tab. 4.4 Regression coefficients for μ in the final model of the inference dataset (CI: credible interval, $\hat{R}$: Potential Scale Reduction Factor)

Parameter	**Estimate**	**Estimate Error**	**Lower 95 %CI**	**Upper 95 % CI**	$\hat{R}$
Intercept	-2.29	0.61	-3.42	-0.88	1.01
a_{stand}	0.14	0.02	0.10	0.18	1.00
v_{mix}	0.34	0.02	0.30	0.38	1.00
sd_{nat}	-0.91	0.54	-2.06	0.23	1.00
h_{top}	4.58	0.70	3.21	5.99	1.00
awc	-4.76	0.55	-5.83	-3.72	1.00
gs_{max}	-3.17	1.04	-5.21	-1.21	1.00
cwb_t_{cn}	0.43	0.05	0.34	0.52	1.00
pet_t_1	-0.30	0.02	-0.34	-0.25	1.00
ari_t_0	0.26	0.03	0.19	0.33	1.00



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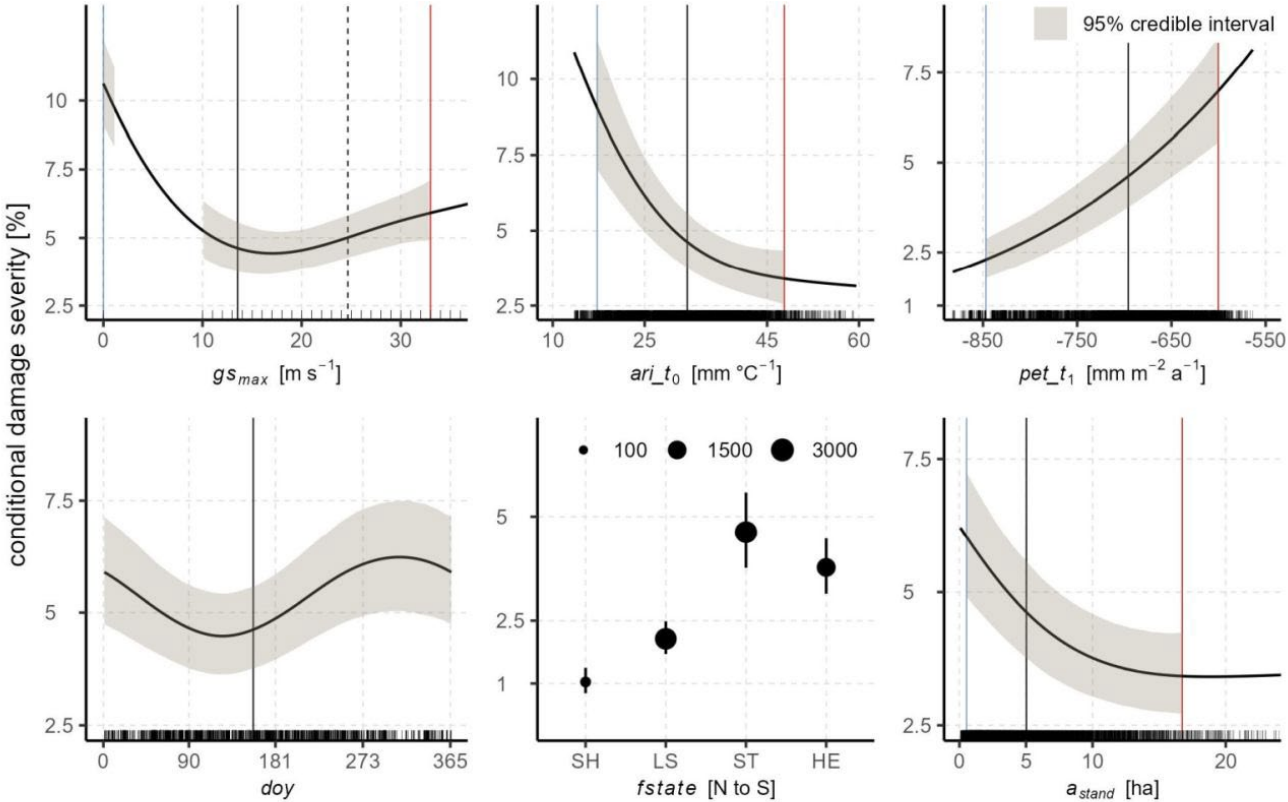
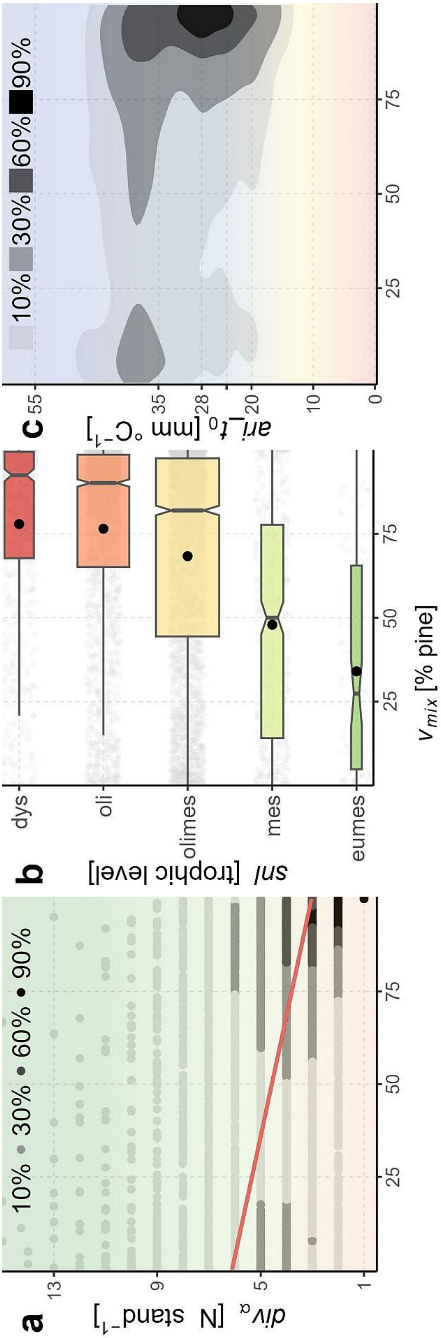


Fig. 4.4 Population-level effects of predictors on the conditional damage severity of Scots pine-stands, while other covariates are held constant at their means/ medians (black vertical line). Shaded ribbons represent 95 % credible intervals, with blue and red lines denoting the 2.5^{th} and 97.5^{th} percentile of empirical data range. Means and percentiles shift when prescribed values of awc and gs_{max} are excluded, represented by dashed lines and broken spline credible intervals. Note the differing ordinate scales

For the inference dataset, the population-level effects (research question *ii.*) are illustrated in Fig. 4.4. The response variable $\bar{y}$ represents the effect of an individual predictor, while all other parameters are held at their means, or medians (snl = oligomesotrophic, $fstate$ = ST) for the categorical covariates. The explanatory power of the splines was constrained to a 95 % interpercentile range of the empirical covariate gradient, as depicted by the blue and red lines. Predisposing (primary) factors are represented in the first two rows of the multipanel. Then, the third row causally explains effects of secondary factors (i.e., inciting and contributing to a decease). Our meta-level predictors are shown in the bottom row. The underlying covariates do not represent a normal distribution shape in the following cases: awc, cwb_t_{cn}, v_{mix}, and gs_{max} tend to be bimodal, while a_{stand}, cwb_t_{cn}, and doy are left-skewed, and v_{mix} is right-skewed.

Broken spline credible intervals are depicted for awc and gs due to prescribed values of 300 mm m^{-2} m^{-1} and 0 m s^{-1}, respectively. This approach separates xeric and mesic from hydric soils, while non-wind disturbance factors are grouped. Damage severities are negatively correlated, with cwb_t_{cn}, sd_{nat}, ari_t_0, and a_{stand} represented by regressive and subsequently asymptotic curves. In contrast, $\bar{y}$ is positively correlated with pet_t_1 losses. awc, v_{mix}, and gs_{max} display parabolic curves, indicating a minimum relationship between damage severities and predictors. In conjunction with Fig. 4.3, it is notable that although storm disturbances occur more frequently, non-wind disturbances exhibit higher severities. doy and h_{top} are characterized by sinusoidal-like functional effect forms.

In our dataset, the main stand's most common tree species, alongside or followed by Scots pine, are short-lived deciduous trees (e.g. *Betula* spp.*, Populus* spp.*, Salix* spp.; 27 %), as well as European beech (*Fagus sylvatica* L.; 25 %), and Sessile/ Pedunculate oak (*Quercus robur* L./ *petraea* (Matt.) Liebl.; 16 %). This ranking remains unchanged in stands with v_{mix} ≤ 50 %. Including the understory, spruce (*Picea* spp.) ranks third. As illustrated in Fig. 4.5, intensively mixed stands tend to grow



under more favorable conditions. Stands with v_{mix} ≤ 25 % exhibit greater α-diversity, tend to thrive on nutrient-rich soils, and (by a notable share) did not experience drought stress in the year preceding mortality. Additionally (not shown), they are more frequently found on sites with short- and log-term sufficient water availability, are generally younger in age, yet attain greater top heights compared to monospecific Scots pine forests.

4.3.3. Magnitude of damage severity

The regression produced a beta distribution represented by a probability density function. For mean pre-

Fig. 4.5 Covariate density of covariates in the inference dataset. Location parameter depending on the Scots pine volume mixing ratio v_{mix} shown for **a)** tree species diversity per stand (red: linear trend), **b)** size-weighted soil nutrient level (separating ‘eumesotrophic’ from ‘mesotrophic’ sites; notch: median, point: mean), and **c)** the de Martonne-aridity index (ordinate-thresholds according to Martinez del Castillo 2024a) dictor values (medians for categorical covariates), the final model can be described with $\bar{\mu} = 0.046$ and $\bar{\varphi} = 12.083$.

In order to answer research question *iii.*, Fig. 4.6a depicts the magnitude of low to high-severity disturbance events. For comparison, the empirical histogram of the inference dataset is shown. On an annual basis, and under average conditions, for 10 % of stands which were disturbed lethally (0.1 frequency) there was a 2 % effect of the stand's attributes (i.e., growing stock, stocked area, or upper canopy stems). High damages, above a severity that lethally affect one quarter, or more, of the stand, occur quite rarely. From 2014 - 2024, the observed distribution of $\bar{y}$ depicts some outliers (e.g. at $\bar{y} = 100\ \%$), depicted above the black density function. Overall, the estimate manifests small credible intervals (see plot extension, Fig. 4.6b). The cumulative distribution function (Fig. 4.6c) shows that up to, for example, 12.5 % the damage severity is underestimated by approximately 8 % annual probability.

Research question *iii.* also asks for the robustness of the beta distributed probability density curve, which we assessed by replacing an individual predictor's mean with its 2.5th and 97.5th percentiles of the inference dataset, while the other predictor values are held at their means, or medians, respectively.

Fig. 4.7a depicts the behavior of the probability density function for the three covariates with the strongest explanatory power, as predisposing v_{mix} and cwb_t_{cn}, as well as inciting ari_t_0. Overall, the probability density function represents only small sensitivity to changing predictor values. The shape only depicts minor changes for all three covariates; for example differing ordinate values where the curves start. The variant study for the other 9 predictors can be found in the supplementary materials and exhibits a similar robust behavior (Fig. S4.8 & Fig. S4.9).

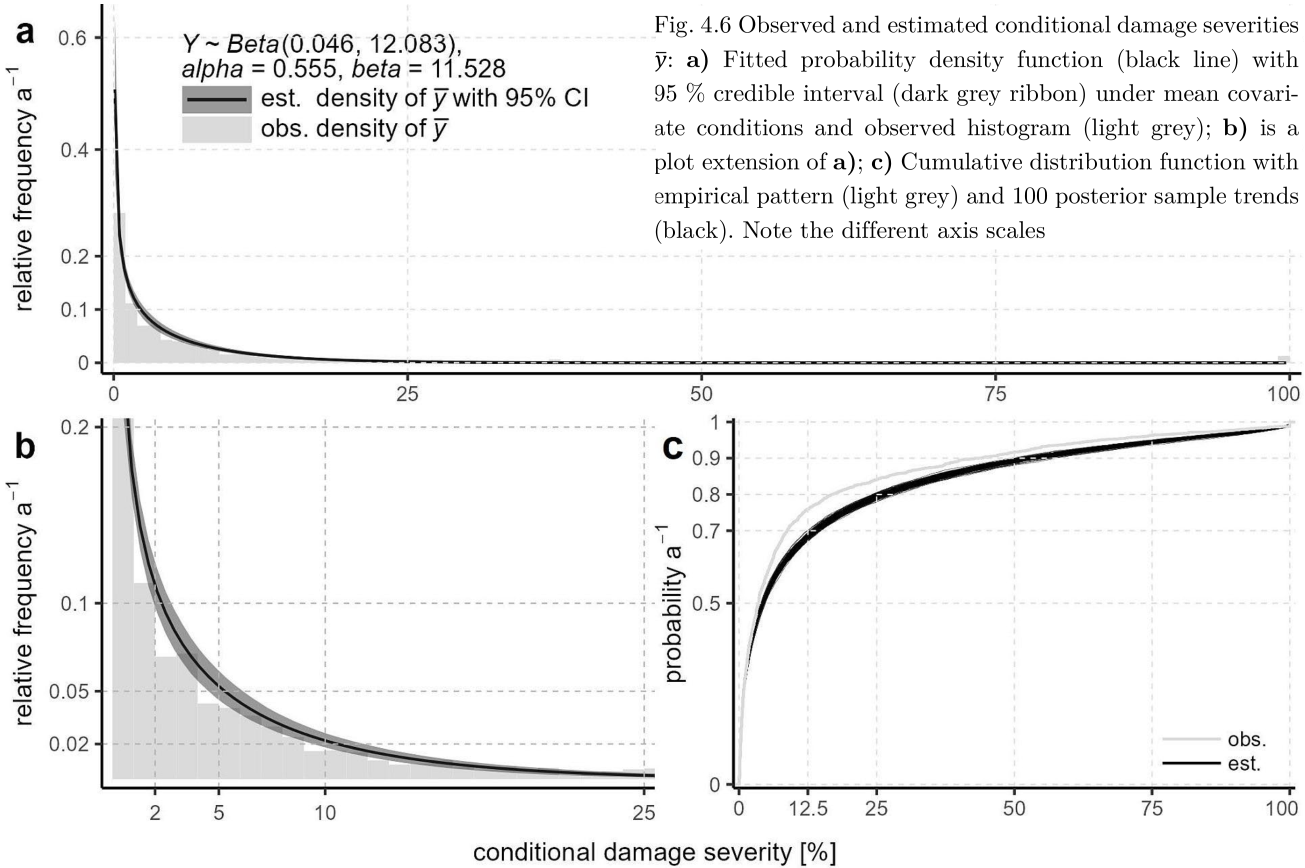


Fig. 4.6 Observed and estimated conditional damage severities $\bar{y}$: **a)** Fitted probability density function (black line) with 95 % credible interval (dark grey ribbon) under mean covariate conditions and observed histogram (light grey); **b)** is a plot extension of **a)**; **c)** Cumulative distribution function with empirical pattern (light grey) and 100 posterior sample trends (black). Note the different axis scales

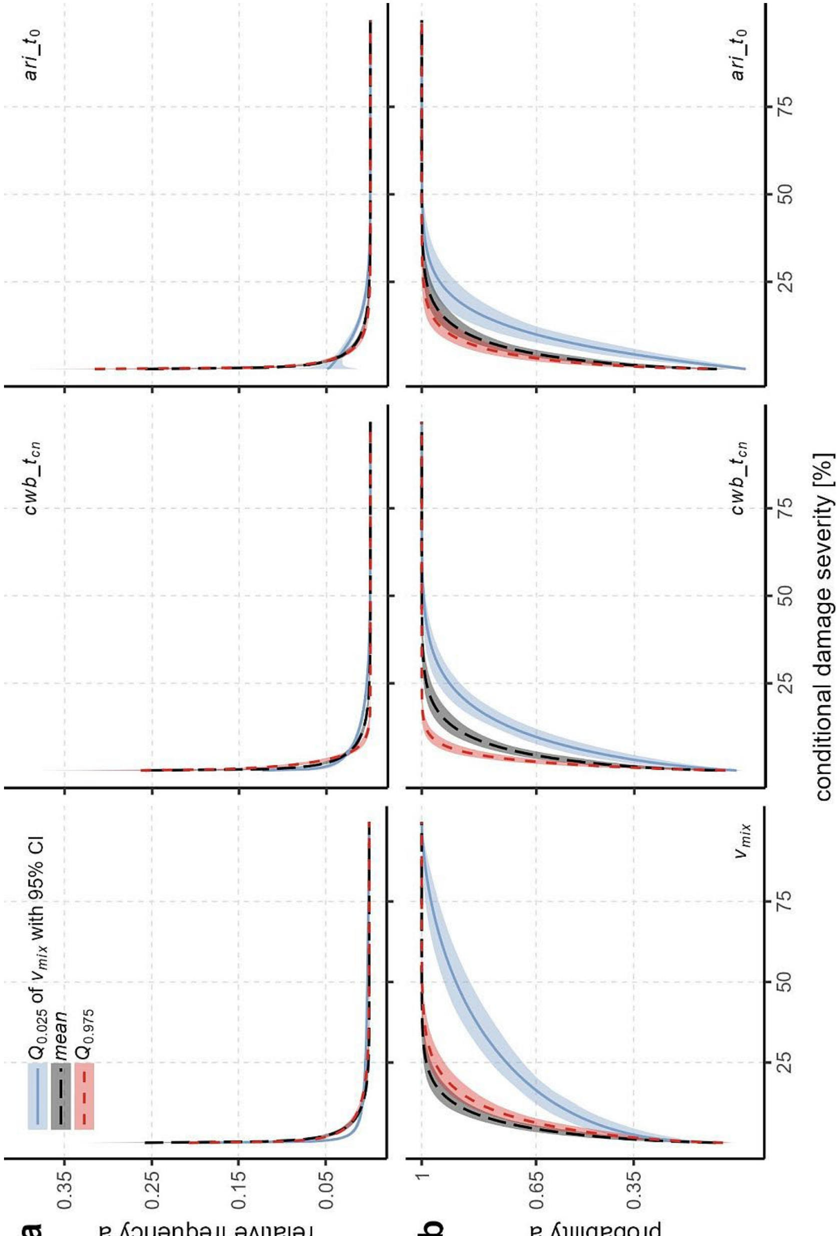


Fig. 4.7 Variant study for differing empirical 2.5th and 97.5th predictor percentiles (*cf.* Fig. 4.4 vertical blue and red margins) with their respective 95 % credible intervals on the sensitivity of the **a)** targeted probability density function, and **b)** cumulative distribution function

4.4. Discussion

We model the severity of lethal disturbance events in Scots pine stands from 2014 - 2024. Disturbance factors align with national statistics: Storms are the leading cause of lethal damage in Northwest German Scots pine ecosystems globally. Locally, storms are the most frequent disturbance (research question *i.*), but particularly biotic pathogens (insects, fungi) vary in time and cause more severe damages. Scots pine's volume mixing ratio exhibits the greatest explanatory force for damage severities (research question *ii.*). Small disturbances (low severities) dominate our observation decade. Using the 95 % covariate percentiles instead of mean values in the sensitivity analysis allows for robust modeling of conditional severities. However, our model tends to slightly underestimate moderate severities from 5 - 50 % (research question *iii.*).

Tree decline syndromes are a major driver of forest degradation and productivity loss with multiple causes and concurrent symptoms (Bonello 2024). Hence, causative factors and meta-level predictors are discussed within the framework of the complex 'Scots pine decline' (Bigler et al. 2006; Vacchiano et al. 2012), which we grouped by predisposers, incitants and contributors to a decease (research question *ii.*).

4.4.1. Assessment on results

Predisposing factors

As predisposing covariates, we identified soil, site, stand and management legacy.

With a slightly positive effect of snl, we find Scots pines on fertile soils, determined via the macronutrient base saturation, are more severely affected from damages than they are on poor soils. On the one hand, this could be causally explained by some root rot pathogens, which are geogenic on alkaline (eu- to mesotrophic) soils (Passarge 1953), due to lower populations of competing microorganisms (Tainter and Baker 1996, p. 400). On the other hand, it could be attributed to the low competitiveness of Scots pine on loamy (nutrient rich) soils compared to the dominant beech and oak (Leuschner and Ellenberg 2017, p. 590). Also, on alkaline soils with fast tree development, the ecological memory of former growth on mortality is discussed (Gessler et al. 2017; Bose et al. 2020a, 2024; Socha et al. 2023).

For awc, the severity increases sharply below 125 mm m^{-2} m^{-1}. Xeric soils jeopardize Scots pine's mortality (Schering 1904; Wiedemann 1948, p. 68) – although these sites are the realized niche of Scots pine (Leuschner and Ellenberg 2017, p. 183). In our study, the lowest available field capacities occur on well-drained, non-loamy, coarse sands of the lowlands or shallow (stony) slopes of the hill country, where droughts are frequent and roots are penetrated by *H. annosum* more rapidly (Towers and Stambaugh 1968; Tainter and Baker 1996, p. 400). Especially for exclusively sandy soils, precipitation storage and capillary rise from potential ground water is the slowest (Wessolek et al. 2024). On shallow or stony soils windthrow is more likely (Peterson 2000). Hence, Scots pine survival depends on rooting depth and lateral water flow on these sites (Pajtík et al. 2022).

As a predisposing site variable, for the individual past 30-year climate normal period, we found decreasing effects of cwb_t_{cn} on damage severity. Since the role of climate change often is only detectable by "probable (or just plausible) [...] circumstantial evidence" (Bonello 2024), this may affect many functional traits of tree health. Negative annual cwb-values indicate a lack of recharge in water reserves (Herrero et al. 2023), and as mainly driven by precipitation and evapotranspiration-fostering temperature, host susceptibility to drought and biotic pathogens is increased (Sturrock et al. 2011; Anderegg et al. 2015). Prolonged water stress places Scots pines in an ecotonal state (Archambeau et al. 2020; Martinez del Castillo et al. 2024a), causing a predisposition to mortality (Vacchiano et al. 2012). It must also be taken into account, that Scots pine forests, among Germany's common tree species, exhibit low precipitation usage effectivity, as they have the highest evapotranspiration due to the comparably greatest interception losses (Loesch 2023, p. 24; Spathelf et al. 2025).

Our model performance is best when the sinusoidal effect of the stand's mean top height is considered to explain damage severity. Generally, tree height is a strong factor to predict mortality of several causes (Merlin et al. 2015; Langer 2018; Stovall et al. 2019; Searle et al. 2022; Grottian et al. 2024). Also, height as a function of tree age can be seen as a proxy to account for stand development. Since the most notifications in our database record windthrow as the lethal cause, tree height primarily increases the lever arm and thus susceptibility to uprooting (Gardiner 2021; Zeppenfeld et al. 2023). In the understory re-initiation phase, old Scots pines are weak competitors with low growth plasticity and declining vigor due to senescence (Mueller-Dombois 1987; Mason 2000; Pretzsch and Grote 2023, p. 356). Otto (1994) identifies young Scots pine stands (i.e., low tree height) to be particularly vulnerable to the majority of disturbance agents (cp. Franklin et al. 1987; Barrere et al. 2023). Because the first commercial thinning usually takes place at $h_{top} \approx 13$ m, trees may remain unstable for some years (del Río et al. 2017), while fungal diseases can surge after thinning and wounded fine roots (Tainter and Baker 1996, p. 399). Accordingly, Scots pine from 41 - 60 years was found to exhibit the highest mortality rates (Neumair et al. 2022). Scots pine trees of lower dimensions were also found to be more preferred from defoliating insects (Hentschel et al. 2018). Since thinning from above, as practiced for decades in state forests, enhances the height-to-diameter ratio for tree stability (del Río et al. 2017), the impact of h_{top} on damage severity may be mitigated and fully effective only in young, undifferentiated stands (Barrere et al. 2023).

Scots pine's volumetric admixture v_{mix} as a predisposing management legacy effect increases damage severity, especially in pure or intensively mixed stands. Monospecific Scots pine stands are known to be threatened from biotic risks (Hentschel et al. 2018), as low functional and biodiversity cause (1) high host accessibility, (2) low antagonist niches, and (3) polyphagous pest shifts (Jactel et al. 2005). While biodiversity generally stabilizes ecosystems, the insurance hypothesis (Yachi and Loreau 1999) may be constrained in highly mixed stands, as natural generalist predators might shift to more lucrative hunting grounds (Jactel et al. 2005). Among trees, *intra*specific competition is typically stronger as conspecifics share the same niche. However, under certain conditions, the low shade tolerance of Scots pine may render beech a stronger *inter*specific competitor rather than being complementary (Kulha et al. 2023). Herein, dropout by alien-thinning is the response which results in demixing (Matías and Jump 2012; Pretzsch and Grote 2023, p. 377). An increasing α-diversity leads to a higher chance to have functionally redundant neighbors (Shovon et al. 2024), a dominance loss, and an out-competition (Leuschner and Ellenberg 2017, p. 521). Searle et al. (2022) found that tree species richness increases productivity, but also mortality (Pretzsch et al. 2023). Higher productivity and more biomass in mixed stands also implies higher water use and transpiration (Shovon et al. 2024). Scots pine relatively develops more branch biomass in mixture with broadleaves (Pretzsch et al. 2020; Hilmers et al. 2024) and during winter storms, this could threaten pine, when the admixture is leafless (Peterson 2000). A downstream effect of competition-induced stress could also be higher vulnerability to biotic factors (Searle et al. 2022). Scots pines, especially on fertile sites, profit from hydraulic lifts in pine-oak mixtures while coping better with spring drought and oaks with summer drought, depending on their intra-annual growth habit (Steckel et al. 2020; Pretzsch et al. 2020; Kašpar et al. 2024). In contrast, and under prolonged drought conditions, Scots pine-mixtures exhibit more competitive tensions (Špulák 2023; Shovon et al. 2024), and already six different species can predispose an increment drop (Shovon et al. 2024). Since we defined damage severity to be relative to volume, area, and stem number, a $v_{mix} = 10$ % can result in disproportionately high damage severity ($\bar{y} \approx 15$ %). Thus, per unit of mixture, more crown canopy or stems are affected. As shown in Fig. 4.5, intensively mixed stands generally develop under more favorable conditions, which we hypothesize to interact with damage severity.

Finally, as a predisposing factor, the natural stocking density sd_{nat} negatively affects the damage severity (i.e., in denser stands damages are lower). This, on the first view, contradicts other studies that, under high basal areas, describe higher mortality (Tainter and Baker 1996, p. 395; Jaime et al. 2019; Archambeau et al. 2020; Grottian et al. 2024; Aldea et al. 2024). On the second view, however, our dataset includes many sparsely stocked mature stands with an overstory (reserves) thriving above young growth, where calculating maximum density via top height (Wördehoff 2016, p. 50) yields low sd_{nat} values. Jewel beetles, in particular, prefer sunlit, old growth (Altenkirch et al. 2002, p. 297; Wermelinger et al. 2008) and during latency (endemic phase), Scots pine bark beetles also favor sparse stands of Scots pine (Jaime et al. 2019) where drought is greater and air humidity lower (Vacchiano et al. 2012; Buras et al. 2018). In our case, we explain the negative effect of stand density with the prevailing, shelterwood systems, consisting of tall and old remnant trees (Tab. 4.1) of sparse stocking densities, which are susceptible to uprooting after the seed cutting (Nyland et al. 2016, p. 328; del Río et al. 2017), beetles, and drought (Weise 1888, p. 195; Vacchiano et al. 2012; Stovall et al. 2019). Due to their large diameter, they are easy to detect and profitable to market. However, our dataset misses sufficient observations for $sd_{nat} \geq 0.8$ (Fig. 4.4), which is where density likely becomes unfavorable, but may also be obscured within the credible interval range.

Inciting and contributing factors

For the majority of Scots pine-observations in the FHNP, windthrow is the most frequent lethal incitant. Therefore, we identified the maximum winter gust speed, the de Martonne aridity in the year prior to death, and the evapotranspiration claim in the penultimate year as incitants and predisposers.

Regarding gs_{max}, we found a progressive effect of gust speed on damage intensity, ranging from 4 - 6 %. Our results indicate that damage severities for all non-wind related disturbances are more severe (about 11 %) than the uprooting and breakage. Accordingly, in our dataset from 2014 - 2024, storm damages are more frequent and thus cause the highest overall timber amounts, but are less severe at the stand-scale. This supports that extratropical winter storms drive the disturbance regime, influenced by predisposition and management legacy (Bargmann 1904; Wohlgemuth et al. 2022, p. 174). Northwest German Scots pine is likely to be acclimated to the microclimatic wind environment by thigmomorphogenesis (Gardiner 2021; Seipulis et al. 2024). Overall, the critical gust speed for Scots pine was found to be between 10 - 50 m s^{-1} (Gardiner et al. 2000; Grottian et al. 2024), depending on height-to-diameter ratios (Barrere et al. 2023), site indices (Grottian et al. 2024), and height of the soil water table (Gardiner 2021; Weller et al. 2022). Especially forest edges and gaps are prone to blowdown, with primarily storm duration, not gust speed increasing the risk (Dupont et al. 2015; Knapp et al. 2024). Our gust speeds account for at least 3 s duration at a height of 10 m and a ground resolution of 25 × 25 m in the previous winter. Since they are ≤ 35 m s^{-1}, we acknowledge that our dataset only captured moderate cyclones, and no summer thunderstorms. Also, we did not account for the topographic exposition, which facilitates explanations of storm damages in the hill country (Zeppenfeld et al. 2023). Due to many reforestations in the context of the Second World War, Scots pine stands are now growing into storm-unstable situations (Wohlgemuth et al. 2022, p. 189). Windthrow in Scots pine stands may be driven by ubiquitous annosus root rot (Bargmann 1904; Manion 1991, p. 304; Langer 2018; Piri et al. 2021), arising from reduced root stability and increased aboveground wood density (Seibt 1964).

Among the covariates, short-term aridity ari_t_0, based on the hydrothermal de Martonne index (1926) provides explanations for immediate and cumulative impacts on the decease of trees (Pretzsch and Grote 2023, p. 357). We found that aridity has a negative influence on the damage severity, ranging from 3 - 9 %. Arid environments are likely to incite drought-related death, even though Scots pine is considered as comparably in-sensitive to drought (Leuschner and Ellenberg 2017, p. 174). However, early studies indicate that Scots pine experiencing dry weather anomalies can exhibit substantial increment drops of up to 50 % (Schwappach 1908, p. 72; Wiedemann 1948, p. 197). This, along with vitality losses, are early warnings for mortality (Herrero et al. 2023; Rebollo et al. 2024). Our 95 % interpercentile range of ari_t_0 is from 15 - 50, indicating very humid (35 - 55) to semi-arid (10 - 20) growing conditions (Martinez del Castillo et al. 2024a), which represent the rear edge of a possible treeline (Beck et al. 2018; Beloiu et al. 2022). Under such conditions, increased background mortality in mixtures, primarily due to asymmetric competition affecting suppressed trees, was found: For drought-avoiding Scots pines, this was attributed to heterospecific basal area and direct exposure to water deficits (Archambeau et al. 2020). However, Scots pine's isohydric behavior (fast stomatal closure to avoid hydraulic failure; Bose et al. 2024), especially during late summer growth (Kunert 2020), enables carbon starvation (Aguadé et al. 2015). Also, under warm-dry conditions, Diplodia tip blight and annosus root rot accelerates mortality (Sturrock et al. 2011; Brodde et al. 2023). As a boreal species, Scots pine is rather adapted for (late winter) cold drought (Aldea et al. 2024), but not (late summer) hot drought conditions (Rehschuh and Rühr 2021; Haberstroh et al. 2022).

Our findings indicate a positive effect of the (negative) evapotranspiration claim in the penultimate year on the damage severity, ranging from 2 - 7 %. However, this predictor is correlated with ari_t_0 and cwb_t_{cn}, and is interpreted meticulously. Generally, Scots pine's growth performance is dependent on pre-drought conditions (e.g. t_1, …), with increasing magnitude and variability causing more vulnerability (Bose et al. 2020a). As shown in Tab. 4.1, the mean temperature of t_1 was slightly lower, particularly at otherwise warmer sites ($Q_{0.975}$), compared to t_0. Potential water consumption is primarily driven by temperature, thus low pet-values indicate higher temperatures. Low evapotranspiration could result in waterlogged soils, leading to prolonged susceptibility to storm damage (Gardiner 2021), increased spore dispersal (Brodde et al. 2023), and promoting root rot in cool soils (Towers and Stambaugh 1968; Rouvinen et al. 2002; Repo et al. 2020). In Northwest Germany, waterlogging was reported during the growing season in 2017, followed by a cyclone in January 2018 (DWD 2022). High precipitation was also found to precede the droughts in 1904 (Schering 1904; DWD 2022) and 1959 in north Germany (Schwerdtfeger 1963), 1975 in east Germany (Hill 1993), and 2003 in Switzerland (Rigling et al. 2013) – with an increasing mortality.

Meta-level factors

At an abstract level, we assess the validity of our predictors and the procedure behind the estimates in order to account for the uncertainties arising from crowd sensing and citizen science (de Groot et al. 2022). Therefore, we analyzed the effects of the day of the year, federal state, and stand size. However, all three covariates must be discussed biologically in addition to an indirect user (partly "citizen") influence.

Regular timber harvests have increased recently, driven by unit mass and timber prices (Senf and Seidl 2020; BMEL 2024). Due to possible ground frost and improved harvesting conditions, as well as higher timber quality from the absence of fungal discoloration, the majority of timber is still harvested during the winter months. Matching the feeding behavior, insect-defoliated or -infested Scots pines often become visible in late summer with subsequent sanitary or salvage fellings. Accordingly, the cyclic spline (Fig. 4.4) oscillates from 4 - 7 % damage severity with peaks in winter as the period of forestry activity, but also cyclone occurrence.

It is in the nature of pest dispersal, that spatial differences arise (Jactel et al. 2005), and autocorrelation patters become visible (Rebollo et al. 2024). On a landscape level, Hentschel et al. (2018) found monospecific Scots pine stands to increase mortality. Heterogeneous spatial aspects arise, for example, from differences in equipment and staff density. This may cause spatial differences in report types within crowd sensing (Harvell et al. 2002; Caley et al. 2020), and ST polygon reports being more prevalent (Fig. 4.1e). In the eastern part of our study region, under semi-arid climate with negative cwb, bark and jewel beetle metapopulations thrive (Langer 2018). Bose et al. (2024) place a disturbance regime (impact) shift at 55° N Lat. (southern Denmark), while Archambeau et al. (2020) identify one at 50° N Lat. (northern HE).

The polygon size has a negative effect on the severity, ranging from 3 - 6 %. On the one hand, we interpret the stand size as a bias, with smaller stands being more likely and frequently subject to lethal disturbances. On the other hand, the visible 'standscape' (Nyland et al. 2016, p. 273) can be seen as an explanation factor, since an ocular assessment of small stands is easier, which may cause the curve regression. In the FHNP dataset, stand sizes are strongly left-skewed, as also found by Senf and Seidl (2020) for disturbance sizes via a remote-sensing approach using 2,165 ha tile sizes. In Germany, over a 30-year period until 2016, they found that the largest detectable disturbance size per tile averaged 0.73 ha. The 0.01 and 0.99 percentile reached from 0.18 - 5.4 ha. For our study, this suggests a negligible bias due to the narrow polygon size gradient and the small effects on the severity, as reflected in the probability density function (Fig. 4.5a).

4.4.2. Key uncertainty parameter

We recognize the FHNP's potential to generate evidence-based insights through a critical interpretation and cautious generalization of the model. For the lack of alternative data sources and the guidance of the empirical law of large numbers, such an approach may provide meaningful contributions to the advancement of causative research in this domain. There are, however, three key limitations: Errors can be introduced from 1. the measurement, 2. the sampling, and 3. the prediction uncertainty (Kangas and Kangas 2004).

First, the FHNP embodies some of the elements of citizen science. Herein, data collection can be a side-product to a separate primary objective (e.g. hunt, logging), which can make the database 'messy' (Dobson et al. 2020). Thus. we assume a timber-accounting motivation behind the FHNP (Brown et al. 2020). In our explorative approach, subjectively assessed observations are explained by the meta-level covariates. We hypothesize that the structured observation process and case-control-system should ensure that implausible FHNP-reports are not possible. We acknowledge that for the distance to forest roads or edges (Buras et al. 2018; Knapp et al. 2024) as another predictor, we lacked relevant data. We assigned stand level information, which may limit the explanatory power (Burkhart and Tomé 2012, p. 189). However, data from forest management plans are commonly used to establish correlations with pests (Möller et al. 2017; Hentschel et al. 2018). Also, we could not measure defoliation directly, which is closely correlated to tree death (Galiano et al. 2011; Searle et al. 2022; Pajtík et al. 2022).

Second, despite the ministerial directive mandating the use of FHNP data and in comparison to the logging statistics, our sampling approach is incomplete in terms of damage quantification, rendering the dataset non-representatively. Also, the data-cleaning procedure excludes implausible and collective reports, potentially introducing a bias. The bias in the irregular sampling design can be considered negligible, as we still observe the same modes and disturbance factors, dominated by storm (Wohlgemuth et al. 2022, p. 173; Destatis 2024). Forest management plans and stand polygons could not be rasterized to a standard stand size, as the species mixture specified in the plans is not spatially explicit. Thus, the damage in pine forests could be influenced by the effect of a_{stand}, ranging from 3 - 6 % (research question *iii.*). Generally, the inciting factor of Lepidoptera herbivory, which only occurs at longer intervals beyond our observation period, is missing in our data set (Langström et al. 2001; Hentschel et al. 2018).

Third, an explained variance of 38 % is relatively high for non-experimental conditions, but still other influential factors are missing. The estimation design itself, beyond its inherent uncertainties (e.g. credible intervals), might not adequately account for interactions among covariates. We selected an R-package capable of estimating both distributional parameters, μ and φ, thereby ensuring a flexible fit (Bürkner 2021) and distinguishing it from alternative software. Additionally, the equal split between the model selection and inference datasets – reflecting our explorative approach and testing whether conclusions could be drawn from the FHNP – inevitably reduces the explanatory power of the final model. Since drought-prone locations tend to be arid within the timescale (Bat-Enerel et al. 2022), incorporating long-term climate data alongside short-term weather conditions perforce introduces correlations and dependencies. To reduce multicollinearity, ari_t_0 was selected over, hypothetically, cwb_t_0, which was computed in a related manner. Furthermore, the p_loo (effective number of parameters; see supplementary material) exceeds the numbers of predictors, suggesting that the model possesses robust predictive capability.

4.5. Conclusion

Following the law of large numbers, after one decade of web-based FHNP, a comprehensive dataset has developed. We show that low-severity damages, when assessed cumulatively, constitute a substantial cover share of the total disturbed area. In Scots pine stands of Northwest Germany, we identified a mixed-severity disturbance regime with frequent low-severity and less frequent intermediate-severity disturbances; potentially also in terms of size. In conclusion, a high damage severity (e.g. 70 %) does actually not convey information if disturbance pattern size is large. It merely indicates a higher likelihood that tree mortality is aggregated in clusters (discrete) rather than being uniformly distributed (diffuse) within stands (low/ high severity vs. small/ large disturbance). We conclude that the absence of large-scale *and* high-severity disturbances exceeding 26 ha is likely attributable to management prevention, as well as the study's spatial (ownership boundaries) and temporal scope, which accounts for these contradictions. Therefore, we advocate for improved reporting morale to strengthen the FHNP as a powerful tool and valuable data resource from the state forest services, while concluding that uncertainties from crowd sensing or amateurs are negligible.

Based on the discussion, we conclude: European forestry is caught between cost and performance efficiency as seen by missing FHNP reports in comparison to the logging statistics.. Due to the large size of forest districts and low staff densities, reporting moral and management activities, such as competition-reducing thinnings, may become more situational. Instead, an ambitious silvicultural approach is necessary, albeit more costly, involving adjusted tending frequencies and intensities to mitigate risks. Mortality patterns as described above indicate that the spatial structure of the species' admixture requires an establishment and tending process that favors group- and patch-intermingling at the expense of complementarity.

For collaboratively managing forest disturbances, stakeholder participation is required. In this context, modeling damage severities bridges the gap between the necessary information for strategically deploying machines and staff, and existing FHNP records: With increasing biotic risks and frequent small disturbances, the need for costly sanitary fellings – but not salvage logging – increases. The former serve to remove trees infested by bark and jewel beetles.

Within just 10 years, our findings confirm the expectation that European forests, and especially Scots pine, are undergoing profound changes in the 21st century. Even if our aridity values capture the treeline threshold within the 95th percentile, site quality (in terms of nutrient supply/ field capacity) is represented within a narrow gradient. Moreover, significantly longer time series are required to accurately map disturbance regimes and patterns, as defoliating insects have not yet been adequately assessed. Therefore, we remain cautious about the transferability of our results. For use in forest development simulations, the robust behavior and well-defined magnitudes provide a solid evidence-based foundation.

CRediT authorship contribution statement

HCK, **MAl**: Conceptualization, Methodology. **HCK** Data Curation, Visualization, Investigation, Writing - Original Draft. **TZ**: Resources. **HCK**, **HSR**: Software, Formal analysis. **MAl**, **MAx**, **GJL**: Writing - Review & Editing. **MAl**: Supervision. **RVN**: Funding acquisition, Project administration.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

The authors do not have permission to share data.

4.6. References

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4.7. Supplementary materials

Ad 4.2.2) Database

In the FHNP, notifications on publicly owned land are transferred overnight to the operational GIS of the state forest services (NW-FVA 2023). In a structured observation process (Dobson et al. 2020), users have to report primary agents/ factors, host tree, and estimate the absolute damage severities. In addition to the reporting date, the damage date may differ. Notifications are registered by point-, line- or polygon-coordinates for entries of any terminal device (Fig. 4.1b, e). The FHNP has to be seen as a passive surveillance tool (Brown et al. 2020) by crowd sensing from a large number of users where the observation process includes spatial influ-ences and different motivations (Dobson et al. 2020). However, site revisions (case-control system), that are sporadically carried out after the report, deplete some limitations for a dam-age impact assessment (Caley et al. 2020; de Groot et al. 2022).

Management plans lack accurate stem counts, tree diameters, and consistent basal area measurements; these values were supplemented with additional data. The mandatory yield tables of Wiedemann (1943) and Lembcke et al. (1975) were used within the study area and depending on the federal state. For supplementing the data, the yield tables of Albert et al. (2024) were used, via the R-package ‘et.nwfva’ (Nuske et al. 2022).

Ad 4.2.3) Modeling approach

Dependency structures and (auto-) correlations among covariates were detected using generalized pairs plots (Emerson et al. 2013). The common parametrization of the beta-distribution – such as used in the base R parametrization (e.g. function 'dbeta') – is back-transferred as $alpha = \mu\varphi$ and $beta = (1 - \mu)\varphi$.

Ad 4.2.3) Variable selection and model fitting

The criteria for model comparison were defined as (1) achieving the lowest possible leave-one-out information criterion (looic) for best out-of-sample prediction performance, (2) ensuring causally interpretable overall effects of μ on the response variable, and (3) demonstrating a robust behavior of the probability density function in the sensitivity analysis of the 95 % percentiles of the respective covariate (Vehtari et al. 2017; Bürkner 2021).

Ad 4.2.3) Model performance

The model was validated to confirm compliance with the underlying assumptions. Independence of covariates was assumed. Spatial autocorrelation was assessed using a spatial smoother but was discarded, as the covariates already possess different characteristics (e.g. xeric sites, dry climate conditions, and $fstate$ as a random effect). Temporal dependencies from 2014 - 2024 were analyzed, but were subsequently dismissed due to the low frequency of repeatedly reported stands (17 % of all i). This indicated a lack of need to model temporal autocorrelation.

```
prior_summary(m)
prior            class   coef group resp dpar nlpar lb ub source
 (flat)          b                                       default
 (flat)          b       sari_t0_1                        (vectorized)
 (flat)          b       sawc_1                           (vectorized)
 (flat)          b       ssdnat_1                         (vectorized)
 (flat)          b       scwb_tcn_1                       (vectorized)
 (flat)          b       spet_t1_1                        (vectorized)
 (flat)          b       sgsmax_1                         (vectorized)
 (flat)          b       shtop_1                          (vectorized)
 (flat)          b       sastand_1                        (vectorized)
 (flat)          b       svmix_1                          (vectorized)
 (flat)          b       phi                             default
 (flat)          b       sari_t0_1      phi               (vectorized)
 (flat)          b       sawc_1         phi               (vectorized)
 (flat)          b       ssdnat_1       phi               (vectorized)
 (flat)          b       scwb_tcn_1     phi               (vectorized)
 (flat)          b       spet_t1_1      phi               (vectorized)
 (flat)          b       sgsmax_1       phi               (vectorized)
 (flat)          b       shtop_1        phi               (vectorized)
 (flat)          b       sastand_1      phi               (vectorized)
 (flat)          b       svmix_1        phi               (vectorized)
student_t(3,0,2.5) Intercept                             default
student_t(3,0,2.5) Intercept            phi              default
student_t(3,0,2.5) sd                          0         default
student_t(3,0,2.5) sd                   phi    0         default
student_t(3,0,2.5) sd           fstate         0          (vectorized)
student_t(3,0,2.5) sd           Intercept fstate 0        (vectorized)
student_t(3,0,2.5) sd           fstate phi     0          (vectorized)
student_t(3,0,2.5) sd           Intercept fstate phi 0    (vectorized)
student_t(3,0,2.5) sd           snl            0          (vectorized)
student_t(3,0,2.5) sd           Intercept snl  0          (vectorized)
student_t(3,0,2.5) sd           snl     phi    0          (vectorized)
student_t(3,0,2.5) sd           Intercept snl phi 0       (vectorized)
student_t(3,0,2.5) sds                         0         default
student_t(3,0,2.5) sds          s(ari_t0,k=3)  0          (vectorized)
student_t(3,0,2.5) sds          s(awc,k=5)     0          (vectorized)
student_t(3,0,2.5) sds          s(sdnat,k=8)   0          (vectorized)
student_t(3,0,2.5) sds          s(cwb_tcn,k=3) 0          (vectorized)
student_t(3,0,2.5) sds          s(doy,bs="cp",k=3) 0      (vectorized)
student_t(3,0,2.5) sds          s(pet_t1,k=3)  0          (vectorized)
student_t(3,0,2.5) sds          s(gsmax,k=4)   0          (vectorized)
student_t(3,0,2.5) sds          s(htop,k=4)    0          (vectorized)
student_t(3,0,2.5) sds          s(astand,k=3)  0          (vectorized)
student_t(3,0,2.5) sds          s(vmix,k=3)    0          (vectorized)
student_t(3,0,2.5) sds                  phi    0         default
student_t(3,0,2.5) sds          s(ari_t0,k=2) phi 0       (vectorized)
student_t(3,0,2.5) sds          s(awc,k=3) phi 0          (vectorized)
student_t(3,0,2.5) sds          s(sdnat,k=3) phi 0        (vectorized)
student_t(3,0,2.5) sds          s(cwb_tcn,k=2) phi 0      (vectorized)
student_t(3,0,2.5) sds          s(doy,bs="cp",k=6) phi 0 (vectorized)
student_t(3,0,2.5) sds          s(pet_t1,k=2) phi 0       (vectorized)
student_t(3,0,2.5) sds          s(gsmax,k=7) phi 0        (vectorized)
```

```
student_t(3,0,2.5) sds        s(htop) phi    0         (vectorized)
student_t(3,0,2.5) sds        s(astand) phi  0         (vectorized)
student_t(3,0,2.5) sds        s(vmix) phi    0         (vectorized)

> summary(m)
Family: beta
Links: mu = logit; phi = log                             Formula:
y ~ (1 | snl) + (1 | fstate) + s(doy, bs = "cp", k = 3)
+ s(astand, k = 3) + s(vmix, k = 3) + s(sdnat, k = 8)
+ s(htop, k = 4) + s(awc, k = 5) + s(gsmax, k = 4)
+ s(cwb_tcn, k = 3) + s(pet_t1, k = 3) + s(ari_t0, k = 3)
phi ~ (1 | snl) + (1 | fstate) + s(doy, bs = "cp", k = 6)
+ s(astand) + s(vmix) + s(sdnat, k = 3) + s(htop) + s(awc, k = 3)
+ s(gsmax, k = 7) + s(cwb_tcn, k = 2) + s(pet_t1, k = 2)
+ s(ari_t0, k = 2)
Data: dd_inference (Number of observations: 6960)
Draws: 8 chains, each with iter = 1000; warmup = 500; thin = 1;
       total post-warmup draws = 4000

Smoothing Spline Hyperparameters:
Estimate             Est.  l-95% u-95%       Rhat    Bulk    Tail
                     Error CI    CI                  _ESS    _ESS
sds(sdoy_1)          0.73  0.74  0.16  2.81  1.00    2401    2450
sds(sastand_1)       2.80  2.39  0.73  7.90  1.00    3641    2945
sds(svmix_1)         4.40  2.60  1.82  10.89 1.00    3767    2410
sds(ssdnat_1)        0.36  0.28  0.03  1.10  1.01    1339    1807
sds(shtop_1)         3.77  1.90  1.62  8.45  1.00    2386    2218
sds(sawc_1)          2.36  1.14  1.02  5.40  1.00    2843    2811
sds(sgsmax_1)        2.73  1.53  1.05  6.69  1.00    3613    2657
sds(scwb_tcn_1)      1.80  1.70  0.10  6.45  1.00    3053    2006
sds(spet_t1_1)       1.11  1.42  0.02  4.66  1.00    2606    2361
sds(sari_t0_1)       2.77  1.92  0.74  7.73  1.00    4209    2890
sds(phi_sdoy_1)      0.35  0.18  0.15  0.82  1.01    1501    2117
sds(phi_sastand_1)   1.66  0.79  0.47  3.50  1.00    1278    1362
sds(phi_svmix_1)     2.05  0.70  1.05  3.76  1.00    1787    2699
sds(phi_ssdnat_1)    1.75  1.75  0.15  6.04  1.00    2494    2185
sds(phi_shtop_1)     2.87  0.97  1.57  5.23  1.00    1951    2541
sds(phi_sawc_1)      3.51  2.23  1.23  9.08  1.00    5149    2758
sds(phi_sgsmax_1)    2.17  0.94  0.91  4.50  1.01    1588    2172
sds(phi_scwb_tcn_1)  2.35  1.93  0.41  7.29  1.00    3961    3007
sds(phi_spet_t1_1)   1.81  1.60  0.23  6.05  1.00    2877    2344
sds(phi_sari_t0_1)   2.02  1.80  0.14  6.54  1.00    2775    1923

Multilevel Hyperparameters:  ~fstate (Number of levels: 4)
                   Est   Est.  l-95% u-95% Rhat  Bulk  Tail
                         Error CI    CI          _ESS  _ESS
sd(Intercept)      1.10  0.62  0.44  2.83  1.00  2762  2679
sd(phi_Intercept)  1.29  0.68  0.52  3.13  1.00  3125  2486
                             ~snl (Number of levels: 4)
                   Est   Est.  l-95% u-95% Rhat  Bulk  Tail
                         Error CI    CI          _ESS  _ESS
sd(Intercept)      0.27  0.26  0.08  0.95  1.00  1586  2200
sd(phi_Intercept)  0.19  0.20  0.04  0.69  1.00  1556  1732
```

Ad 4.3.3) Magnitude of damage severity

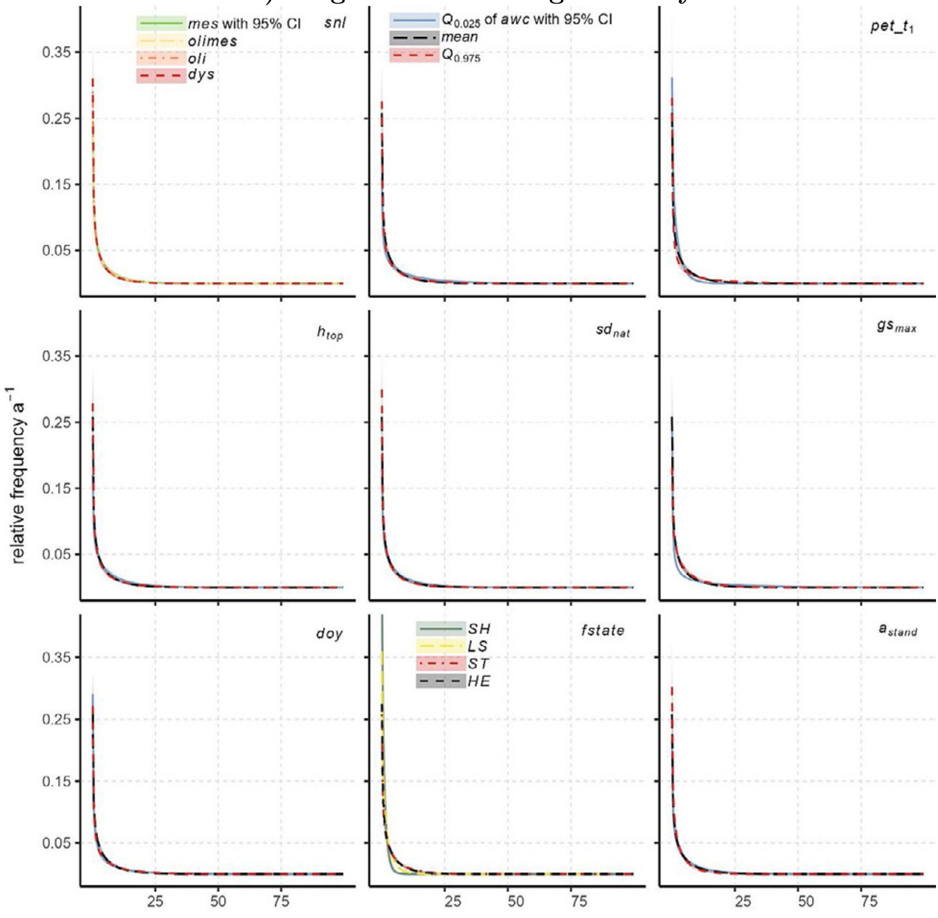

Fig. S4.8 Variant study for differing empirical 2.5th and 97.5th predictor percentiles (*cf.* Fig. 4.4 for blue and red margins) with their respective 95 % credible intervals on the sensitivity of the targeted probability density function: Calculation of gs_{max} according to (Jung and Schindler 2019), h_{top} and sd_{nat} according to (Wiedemann 1943; Speidel and von Laer 1959; Lembcke et al. 1975), and Wördehoff (2016)

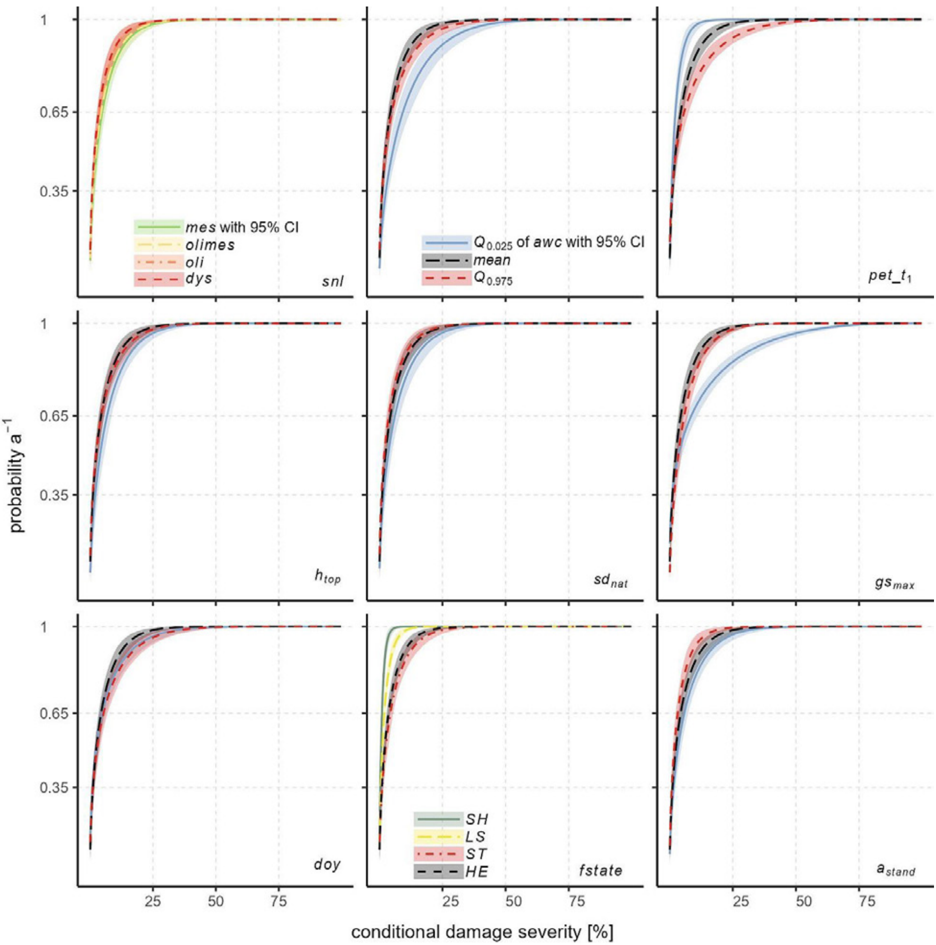

Fig. S4.9 Variant study for differing empirical 2.5th and 97.5th predictor percentiles (*cf.* Fig. 4.4 for blue and red margins) with their respective 95 % credible intervals on the sensitivity of the cumulative distribution function

Schweregrad von Störungen (A2)

```
Regression Coefficients:
                   Est      Est.  l-95%  u-95% Rhat  Bulk  Tail
                            Error CI     CI          _ESS  _ESS
Intercept          -2.29    0.61  -3.42  -0.88 1.01  1952  1784
phi_Intercept      1.86     0.70  0.28   3.14  1.00  1975  1922
sastand_1          0.14     0.02  0.10   0.18  1.00  9313  3009
svmix_1            0.34     0.02  0.30   0.38  1.00  7693  2943
ssdnat_1           -0.91    0.54  -2.06  0.23  1.00  3611  2892
shtop_1            4.58     0.70  3.21   5.99  1.00  7575  2916
sawc_1             -4.76    0.55  -5.83  -3.72 1.00  6648  2736
sgsmax_1           -3.17    1.04  -5.21  -1.21 1.00  5529  3143
scwb_tcn_1         0.43     0.05  0.34   0.52  1.00  3971  3148
spet_t1_1          -0.30    0.02  -0.34  -0.25 1.00  4188  3500
sari_t0_1          0.26     0.03  0.19   0.33  1.00  3650  3357
phi_sastand_1      3.79     1.70  0.66   7.22  1.00  3740  2368
phi_svmix_1        4.85     1.42  2.09   7.75  1.00  3523  2992
phi_ssdnat_1       -0.06    0.02  -0.09  -0.02 1.00  9555  2667
phi_shtop_1        -6.71    2.58  -11.88 -1.79 1.00  4031  3116
phi_sawc_1         -0.20    0.03  -0.26  -0.14 1.00  6299  3447
phi_sgsmax_1       1.14     1.76  -2.60  4.37  1.00  2717  3028
phi_scwb_tcn_1     -0.48    0.05  -0.58  -0.39 1.00  3407  3210
phi_spet_t1_1      0.45     0.03  0.39   0.50  1.00  4427  3538
phi_sari_t0_1      -0.09    0.04  -0.17  -0.01 1.00  3102  2769

Draws were sampled using sampling(NUTS). For each parameter, Bulk_E
SS and Tail_ESS are effective sample size measures, and Rhat is the
potential scale reduction factor on split chains (at convergence, R
hat = 1).

> bayes_R2(m)
       Estimate    Est.Error      Q2.5          Q97.5
R2     0.3693755   0.007288171    0.3550959     0.383869

> loo_R2(m)
       Estimate    Est.Error      Q2.5          Q97.5
R2     0.1839699   0.01737519     0.1492653     0.2169157

> loo_compare(m, m_2ndbest)
#dd_inference; ranking dd_model_selection identical
         elpd   se    elpd     se     p     se    looic      se
                diff  loo      elpd   loo   ploo             looic
m         0.0    0.0   10638.4  177.4  248.0 34.6  -21276.8   354.8
m_2ndbest -7.3   2.3   10631.1  178.5  256.5 37.1  -21262.1   356.9

> summary(m$data)
         y        snl       fstate     doy           astand
Min.:    0.0001   3: 520    SH: 146    Min.: 1       Min.: 0.1
1st Qu.: 0.0081   4: 3412   LS: 2525   1st Qu.: 81   1st Qu.: 1.8
Median:  0.0351   5: 2292   ST: 2729   Median: 153   Median: 3.6
Mean:    0.1274   6: 736    HE: 1560   Mean: 157     Mean: 5.0
3rd Qu.: 0.1162                        3rd Qu.: 226  3rd Qu.: 7.0
Max.:    0.9999                        Max.: 366     Max.: 25.0
```

```
          vmix     sdnat         awc              gsmax
Min.:     0.0      Min.: 0.0     Min.: 63         Min.: 0
1st Qu.:  0.51st   Qu.: 0.5      1st Qu.: 114     1st Qu.: 0
Median:   0.8      Median: 0.6   Median: 133      Median: 19
Mean:     0.7      Mean: 0.5     Mean: 152        Mean: 13
3rd Qu.:  1.03rd   Qu.: 0.6      3rd Qu.: 147     3rd Qu.: 27
Max.:     1.0      Max.: 1.1     Max.: 300        Max.: 37
          cwb_tcn  pet_t1              ari_t0           htop
Min.:     -178.0   Min.: -882.0        Min.: 13.5       Min.: 3.5
1st Qu.:  -92.0    1st Qu.: -722.9     1st Qu.: 25.3    1st Qu.: 21.0
Median:   51.3     Median: -686.4      Median: 31.8     Median: 23.5
Mean:     44.9     Mean: -695.8        Mean: 32.0       Mean: 23.2
3rd Qu.:  172.9    3rd Qu.: -652.6     3rd Qu.: 38.3    3rd Qu.: 25.9
Max.:     400.0    Max.: -563.7        Max.: 62.5       Max.: 35.0
```

Supplementary materials' references

Albert M, Nagel J, Schmidt M, et al (2024) Eine neue Generation von Ertragstafeln für die Baumarten Eiche, Buche, Fichte, Douglasie und Kiefer in Nordwestdeutschland [A new generation of yield tables for sessile and pedunculate oak, European beech, Norway spruce, Douglas-fir and Scots pine in Northwest Germany]. Beitr NW-FVA 22:1–112. https://doi.org/10.17875/gup2024-2636

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5. Prognose gestörter Waldentwicklung (A3)

Erschienen als Artikel A3:

„Hope for the best, expect the worst:
Including disturbances in simulations as tree- or stand-wise mortality increases uncertainty for north German Scots pine forestry“

✉ HC Knocke[1,2,3], M Axer[2], HF Hamkens[2], H Sennhenn-Reulen[2], K Husmann[4], T Zeppenfeld[2], C Fischer[2], RV Nagel[2], M Albert[1,2]

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Hypothese H3

„Die Waldentwicklung Norddeutschlands wird durch diskrete, hohe Schweregrade beschleunigt, wenn sie mit diffusen, niedrigen Schweregraden kontrastiert werden.“

Gegenüber dem Originalartikel gab es redaktionelle Änderungen.

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Abstract

Recent mortality pulses provide evidence of how to restore maladapted forests towards more natural, mixed, resilient ecosystems. Under climate change, carbon sequestering forests serve as forces for mitigation but face threats from disturbances. The uncertain development of this duality complicates stakeholder-relevant predictions of timber yield or tree species compositions.

We analyzed uncertainty in forest development for 50 years focusing on the most important tree species in northern Germany, Scots pine. Thereunto, *i.* future survival was examined on a site- and climate-specific basis. The novelty of the simulations lies in *ii.* quantifying the influence of irregular die-off on forests under stand- or tree-wise mortality patterns to *iii.* investigate yield and harvest potentials. By spanning a scenario funnel from two management and two climate projections on 17,551 National Forest Inventory plots in northern Germany, we assess prospective uncertainties. The transition of forest composition through space and time with conditional hazard rates resembles a Markov process via Monte Carlo sampling.

Prospectively, *i.* mortality doubles depending on the species. Compared to undisturbed development, *ii.* forest restoration outcomes are uncertain; stand-losses can accelerate restoration by up to 10 %, while tree-losses might slow it down. Under disturbances, *iii.* volume increment and stand age decrease as harvests rise in the medium-term with large dimensioned timber increasing fivefold.

Driven by climate change and forest demographics, the decline in yield and the remaining 40 % monocultures are critical. Forest development is inert, therefore we suggest more differentiated silvicultural concepts for actively adapting forests to alarming changes.

Keywords Timber yield · forest growth · mortality · uncertainty · forest restoration · Scots pine

5.1. Introduction

Forests are solution and subject to climate change (Albrich et al. 2023; Seidl and Senf 2024). *Inter alia*, they sequester carbon dioxide and cool landscapes down (Ellison et al. 2024), while being adversely affected by fragmenting and droughts (Haberstroh et al. 2022). In a changing environment, the multifunctional provision of ecological, economic and social forest ecosystem functions (Borrass et al. 2017; Simons et al. 2021) is altering: In Europe, forest health deteriorates (Patacca et al. 2023), yield gets uncertain (Chakraborty et al. 2024), and maladapted monocultures with little biodiversity are restoration patients to gain resilience (Spathelf et al. 2018; Paul et al. 2020; Messier et al. 2022).

As wood serves as both, a product and a means of production, the interaction between forest growth and yield can mitigate the impacts *of* climate change but is also influenced *by* it (Pretzsch and Grote 2023; Gribbe et al. 2024). Previously closed forest cover has been opened up by biotic and abiotic disturbance agents (Seidl and Senf 2024) and yield becomes uncertain, as remnant trees can only compensate the damages to a certain extent (Buras et al. 2018; Biber and Pretzsch 2022). If remaining stands are salvaged, new forests may not perform as expected (Staupendahl and Möhring 2011; Tamke et al. 2021). While growth losses and gains at gap edges have been researched at the stand scale (Preuhsler 1987; Biber and Pretzsch 2022), it is not clear how calamity-affected growth and yield react across the landscape level.

Prospective timber yield in North Germany, with its mainly poor, dry forest sites, relies largely on Scots pine (*Pinus sylvestris* L.). Here, pine is again being increasingly established (BMEL 2024a), although there are conflicting evaluations of its climate fitness (Haberstroh et al. 2022; Chakraborty et al. 2024). As further described in the info-box 1, forest management, therefore, aims to diversify species and structures in order to spread mortality risks (Markowitz 1952) by using a site- and climate-specific approach (Spathelf and Ammer 2015). However, pine management has recorded serious mortality losses, which are increasingly caused by biotic agents (Patacca et al. 2023). The patterns (gap sizes) extend from mainly tree size (Hennert 1797, pp. 165; Brodde et al. 2023) to total stand size (Sommerfeld et al. 2018; Hentschel et al. 2018). For *P. sylvestris*, survival time models indicate that currently only between 62 % (Schoneberg 2017, p. 100) and 92 % (Staupendahl and Zucchini 2010) of stocked areas are likely to reach 100 years of age, which is about when trees are mature.

Due to mutual interactions, survival-time modelling is not trivial. Tree mortality can result from both planned (harvest), and unplanned causes, which include regular (competition) and irregular (biotic or abiotic stressors) die-off (Brandl et al. 2020; Pretzsch and Grote 2023, p. 354; Seidl and Senf 2024), though its etiology cannot always be distinctly separated. E.g., background mortality, described as “low rates of tree mortality [...] in the absence of acute stress[ors] like drought” (Das et al. 2016) is difficult to attribute to regular or irregular mortality. To evaluate tree losses and distinguish planned harvests from unplanned salvage logging, appropriate spatio-temporal stochastic approaches must integrate forest growth, yield, and mortality aspects (Blattert et al. 2016; Ferretti et al. 2024).

In Europe, survival-time models are commonly derived from the forest condition survey (ICP Forests 2022), executed e.g. as k-tree sampling (Prodan 1968; Kleinn and Vilčko 2006; Staupendahl and Zucchini 2010), which assesses single trees. In the context of prevailing rotational forest management (Mason et al. 2022), survival models are usually interpreted for stand patterns (Knoke et al. 2021). They are, however, also valid for single tree patterns, if the entity of tree and growing space is considered (Saborowski et al. 1998; Staupendahl and Zucchini 2010; Schoneberg 2017, pp. 90–92). Either way, area damage is examined.

Yet, there is no consensus on how climate change-related uncertainty is best integrated into simulation studies, although the growth and yield developments are expected to differ substantially between mortality patterns (Hoogstra and Schanz 2008; Blattert et al. 2016). Above all, climate uncertainty seems to be the major contributing factor (Hawkins and Sutton 2009; Nabuurs et al. 2022). Promising approaches model a stochastic Markov process to track, for example, species' yield composition over time and space (Markov 1906; Pielou 1963). The key is to incorporate inherent pure randomness (yet moderated by anthropogenic management) at discrete intervals, with outcomes dependent on the current state of a stochastic mortality sub-process (Brandl et al. 2020). Then, disturbances are treated jointly as stochastic events, regardless of their agent (e.g., pest, drought; Knoke et al. 2021). If algorithms consider variability within the sample unit and reflect prior mortality events on subsequent growth (Buras et al. 2018), error propagation is enhanced and stochasticity maintained. Therefore, the Monte Carlo method (Metropolis and Ulam 1949) quantifies variability through repeated random sampling. The use of Monte Carlo sampling and Markov chains in forestry has been known since the 1960s (Pielou 1963; Jeffers 1964), but uncertainty in an infinite sample space remains inadequately addressed.

Failure to account for uncertain yield or changes in the composition and demography of forests can negatively impact national economics (Paul et al. 2020; Popkin 2021; Brichta et al. 2023), business economics (Mason et al. 2022), and ecology (Heinrichs et al. 2019; Turmukhametova et al. 2020). Timber sales, critically, fund forest restoration as a path away from unstable forests but this can come to a standstill (Spathelf et al. 2018; Höhl et al. 2020; Erdozain et al. 2024). For instance, if income is affected by planned global stock reduction with reforestation liabilities or unfavorable large timber assortments which are difficult to market (Mason et al. 2022). Von Berlepsch (1844, p. 73) is known as the first to consider that forestry has to account for stands not reaching final harvest age after disturbances. Previous work has only focused on yield potential without including irregular mortality (BMEL 2017; Oehmichen et al. 2018; Rosenkranz et al. 2023). This is needed, however, by stakeholders such as the timber industry or conservationists (Holm and Schweier 2024):

The benefits from using explicitly modelled mortality serve policy making and impact assessment, and to evaluate other predictions.

Therefore, we improve predictive capabilities for estimating yield potentials by using survival models in a detailed, site- and climate-sensitive scenario funnel approach. On sample plots of the National Forest Inventory's (NFI) systematic grid, the results are statistically representative. The novelty of our approach is to apply mortality risk of equal magnitude in simulations on a tree-wise basis and to contrast this with conventional stand-wise use. Thus, single-tree death imitates the miscellaneous causes of low-rate background mortality (Das et al. 2016). To date, practical, realistic, and disturbance-influenced forest development has not been simulated based on inventory data. Because there are differences in mortality patterns and climate models, there is uncertainty in the future yield potential across six north German federal states. This study accounts for forest landscape restoration (FLR) of current pure stands towards mixtures containing, preferably, Scots pine. Projections are based on existing statutory, business-as-usual silvicultural management, which already considers climate change. Thus, the study is prescriptive as scenarios are selected according to actual ministerial directives. We investigate:

i. What is the uncertainty margin in future mortality risk of Scots pine and the other main tree species in northern Germany?

ii. Given the stochasticity in the disturbances, how does FLR of the main tree species, by region, accelerate under different mortality patterns and changing climate?

iii. How uncertain is the yield potential, by timber assortment, between tree- and stand-wise mortality when forest development targets pine in simulation scenarios?

Info box 1

Background: German forestry…

… aims to enhance ecosystem resilience to climate change through a sustainable, and adaptive management. While many forests are privately owned and originate from even-aged plantings, the state of the art is close-to-nature silviculture with uneven-aged, thinned, mixed-species stands and continuous cover systems to promote biodiversity. In Germany's densely settled landscape, managed stands have to achieve multifunctional objectives, balancing timber production integratively with nature conservation, and recreational use.

5.2. Material and Methods

5.2.1. Simulations

Concerning the research questions *i. – iii.*, we *i.* apply the survival models of Brandl et al. (2020) to national climate and dendrometric data, *ii. – iii.* and couple climate sensitive growth, regeneration (Hansen and Nagel 2014; Schmidt 2020), and survival models to estimate yield potentials by simulating forest development until 2062, and *i. – iii.* assess the different outcomes as uncertainty. This article builds on Knocke et al. (2024), omitting two development pathways while further elaborating on the remaining ones. Instead of simulating idealized undisturbed forest development, we predict a more realistic, disturbance-influenced forest development using novel methods and software conjunction in order to implement irregular mortality.

The simulations modulate the as-is state of the forest stands towards a target state in four scenarios. If a stand is harvested or salvaged, a new forest generation is established, relying on both artificial and natural regeneration. While the target state is represented by forest development types (FDT), it is achieved through FLR. This is defined as restoring degraded (forest) landscapes (including land cover and land use change), with a focus on structure, functions, and socioeconomic constraints, but without necessarily replicating natural forests (Stanturf 2016; Spathelf et al. 2018).

To assess uncertain yield potentials, we project four realistic forest developments: moderate climate (b) with tree-wise (bt) or stand-wise (bs) mortality, and extreme climate (c) with tree-wise (ct) or stand-wise (cs) mortality, as shown in Fig. 5.1. They shape a scenario funnel.

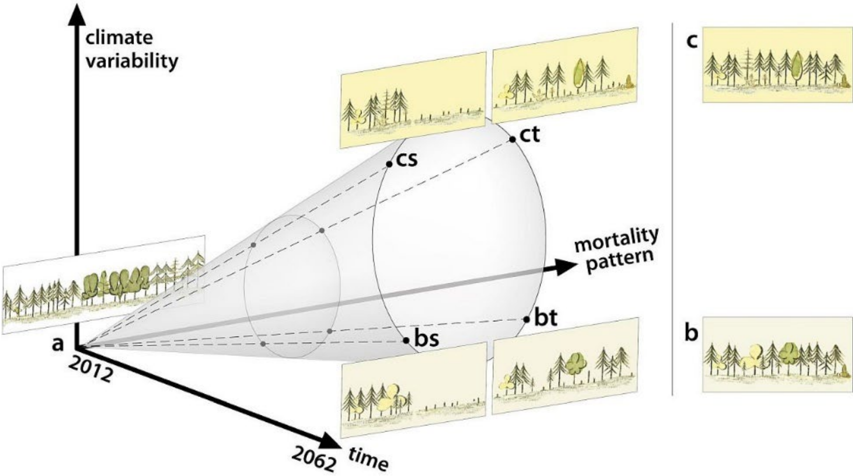


Fig. 5.1 Funnel with equally likely scenarios and spanned uncertainty: Over time (x axis) yield potentials are affected by the FDT selection under climate conditions (y axis) and the effects of irregular mortality patterns (z axis). Today's forests with restoration liabilities (**a**) that are simulated until 2062 under EC-Earth RACMO (**bs**, **bt**, **b**) and HadGEM WettReg (**cs**, **ct**, **c**) climate conditions within RCP8.5 are shown. Mean scenario values confine our uncertainty margins. The z-dimension is spanned by stand-wise (bs, cs) or tree-wise (bt, ct) mortality patterns at equal survival probabilities, i.e. loss intensities (Brandl et al. 2020). For further comparison, undisturbed i.e. planned forest development (Knocke et al. 2024) is contrasted (**b**, **c**). Forest images based on Guimpel (1819)

In order to predict, guardrails of a business-as-usual forest management are implemented into the individual-based single-tree simulation software *WaldPlaner*, version 2.2. This software uses the forest growth library 'Tree Growth Open Source Software' (TreeGrOSS; Hansen and Nagel 2014). Herein, single tree tables (input) are modeled with explicit modeled coordinates and with special regard to regular (distance-dependent competition and senescence) mortality, increment, harvest (growth model), and ingrowth (regeneration model), before being issued again as a table (output). The conjunction of irregular mortality is described in section 'Mortality risk application'.

First, for each NFI subplot, a simulation stand of uniformly 0.2 ha size (due to computing capacity) is created from the dendrometric data (as-is state), whose future species composition is dictated by FDT – the key in our simulations. Operationalizing the long-term targets for a directional evolution, FDT provide a quantitative mechanism to reach site-specific mixed stands within the framework of close-to-nature forestry (Abetz 1935; Larsen and Nielsen 2007; Knocke et al. 2023). With every forest operation, *WaldPlaner* gradually aims to reach the FDT species' mixture composition.

Besides FDT, per site (b, c; Fig. 5.1) depending on the climate, simulations require management settings (cp. Knocke et al. 2024): This includes thinning, maturity definitions, maximum densities causing regular mortality (Döbbeler and Spellmann 2002), and canopy cover periods (± extended timing of harvests). Forestry operations are carried out if the height-dependent, species- and federal state-specific target basal area is exceeded, or target diameter share is reached, and logging amounts $> 10 m^3 ha^{-1}$.

Finally, the guard rails form a scenario funnel (Kosow and Gaßner 2008, pp. 15–17) with equally likely but outermost outcomes (Fig. 5.1). We emulate those silvicultural prescriptions where Scots pine as a light-demanding xerophilic coniferous tree species, is promoted (BMEL 2024a), and monospecific stands are restored to mixed species stands.

For each simulation step of 5 years, different climate (moving) windows (Burggraef 2024, pp. 48–49) are considered. While FDT are selected for the fixed period of 2071-2100, the climate- and site-sensitive height growth models dynamically average static site indices over the stand's lifetime span. For mortality risk application however, the corresponding last 30 years are used as the climate period, i.e. for a simulation from 2012 until 2017 the climate from 1983-2012 is therefore decisive (see section 'Climate covariates').

We also simulate growth without forest management on strictly preserved stands, for instance national parks (Steinacker et al. 2023). However, besides these areas, we regard all tree mortality regardless of their quantity as dropout for logging, either planned or salvaged (unplanned). Thus, the timber emergence analyzed is therefore to be seen as a logging potential for compact wood ≥ 7 cm diameter at breast height (DBH).

5.2.2. Study area and dendrometry

We assess the future development of temperate forests in six north German federal states, which accounts for about 40 % of the total German forest cover of 11.4 million ha (m. ha) and 77 % of its total Scots pine distribution. Due to differing climate and silviculture practices, we distinguish between western (Hesse, Lower Saxony, and Schleswig-Holstein), and eastern states (Brandenburg, Mecklenburg-Western Pomerania, and Saxony-Anhalt).

The study area encompasses the north German lowlands and central German hill country between 49°–55° N Lat. and 7°–15° W Lon. (Fig. 5.2). It contains a broad site gradient in terms of water and nutrient supply. Here, Scots pine naturally (autochthonous) predominates due to the frequency of medium to poor and dry (xeric) soils. However, it also represents a synanthropic (allochthonous) range, with potential negative implications for fitness and survival (Leuschner and Ellenberg 2017, p. 324; Bonannella et al. 2024; Bose et al. 2024; Chakraborty et al. 2024).

In order to be able to analyze forest development and its shifting species composition on a sound basis, a few sites without forest site mapping of soil nutrient and water supply were excluded. Statistical representativeness is provided by the systematic grid of the third German National Forest Inventory (NFI) (TI 2015) with reference date 2012 (t'_0). It is characterized by a grid of permanent stratified one-phase cluster samples. The strata consist of regions with different sampling intensity (Riedel et al. 2017). Subplots are clustered in a square formation (150 m distance) and form a plot. On each subplot that is located in the forest, in a field survey, dendrometric data are collected using angle count sampling (Bitterlich 1947).

In most European countries, precise plot coordinates are kept confidential to ensure the integrity of unbiased measurements (Päivinen et al. 2023).

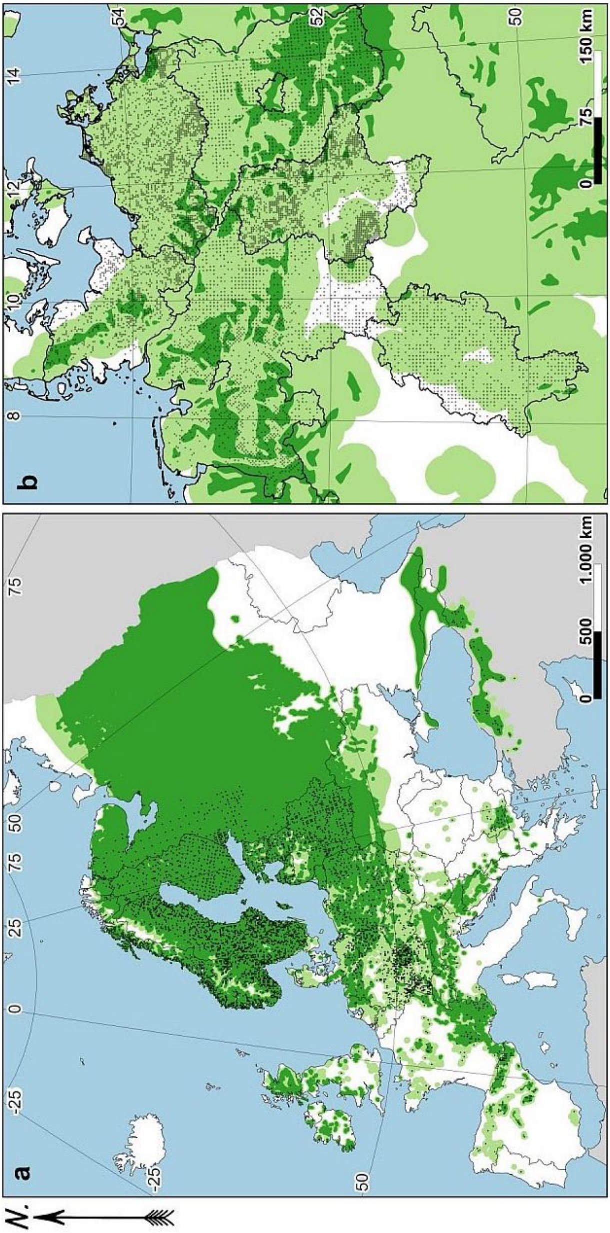


Fig. 5.2 Natural (dark green; Bohn et al. 2004; Caudullo et al. 2017) and synanthropic (light green; de Rigo et al. 2016; Caudullo et al. 2017) distribution of *Pinus sylvestris* L. in Europe (white) and Mideast (grey) with **a)** Scots pine sample plots (black dots) of ICP Forests (2022) for which the survival model of Brandl et al. (2020) was fitted and **b)** systematic sample plots (grey dots) of the third German NFI (TI 2015) which provide the basis of our work in the western (three left big polygons with dots) and eastern study region. Boundaries based on © opendatasoft/ OGL (2019) and © GeoBasis-DE/ BKG (2022)

However, exact geographical location was critical for including soil and climate variables in our study. It was retrieved only by spatially intersecting site polygons with the 3rd NFI based on a by-case decision following computations by the federal research institute (TI 2022).

A total of 17,551 subplots across 6,508 plots with site mapping data were included in this study. The mean altitude of the eastern sites is 104 m a.s.l., while the western stands are located, on average, at 184 m a.s.l. and the altitude range is from -3 to 1,141 m a.s.l., so from at planar to submontane level (Leuschner and Ellenberg 2017, pp. 191–194).

5.2.3. Site and climate dynamics

In our study area, current climate conditions are oceanic to hemiboreal (Beck et al. 2018). Climate change directly affects the future water supply of the forest sites, while the supply of macronutrients (with the exception of nitrogen, as described below) is assumed to remain constant. In order to assess future uncertainty adequately, in projections, forest research uses ensemble approaches of different climate models.

In this work, we consider future climate by the two outer model runs from the ReKliEs-De core ensemble according to ministerial directives. Here, seven combinations of dynamical and empirical-statistical downscaling with global climate models forge different regional climate models (RCM; Hübener et al. 2017; Warrach-Sagi et al. 2018). According to the principle 'hope for the best and expect the worst' and in order to span a climatic funnel, both were selected from the representative concentration pathway (RCP) 8.5 of the CMIP5 generation. Thus, the global climate model 'HadGEM2' in combination with the empirical-statistical RCM 'WettReg18' were chosen to represent the most extreme climate projection (Martin et al. 2011; Kreienkamp et al. 2019), hereafter abbreviated as 'c'. In contrast, the global climate model 'EC-Earth12' and its dynamical RCM 'KNMI RACMO22E' are considered a moderate projection (van Meijgaard et al. 2008; Hazeleger et al. 2012), hereafter abbreviated as 'b'. The *status quo*, denoted as 'a', represents the recent climate in the reference period from 1983-2012 and current stands surveyed from the NFI in 2012 with restoration liabilities.

The two model combinations generate daily past (backcast) and future projected values from 1951–2100, which were further regionalized at a resolution of 50 m (Warrach-Sagi et al. 2018; Schick et al. 2023). Firstly, this data set serves to estimate future climatic water balances, i.e. the difference between precipitation and evapotranspiration of dynamically predicted growing seasons (Sutmöller et al. 2021; Nuske 2022). On this basis, specific FDT are determined, expert-based, by means of an algorithm (Hamkens et al. 2022) for each subplot (Knocke et al. 2024). Thus, tree species selection is dynamic, i.e. changing forest planning prescriptions depending on model results. Secondly, this study uses these climate values to take into account climate sensitive tree height growth under altering environmental conditions, such as future N-deposition, temperature and precipitation (Schmidt 2020).

Tab. 5.1 introduces current and future climate of the study area until the simulation end (2062), as well as the previous 30 years average. An extended version of the table is provided as Supplementary material (*cf.* Tab. S5.5). Differences between the western and eastern region become apparent. Mean annual temperature rises about 2 °C until 2062 under aridizising conditions: Although there is about a 100 mm m^{-2} a^{-1} higher annual precipitation, especially in the east, average and maximum temperatures rise simultaneously. Within the vegetation period, growing conditions turn more unfavorable in the extreme climate scenario (c) especially in the east. Altogether, the eastern region exhibits more extreme future conditions (comparatively colder or hotter days) underlining its continental characteristics and mainly planar elevation.

Tab. 5.1 Mean strata-weighted climatic conditions on the NFI subplots of the study area for the RCP8.5 moderate (b, EC-Earth RACMO) and extreme (c, HadGEM WettReg) scenarios at the start and end of our simulations that are the basis for 1. the FDT-selection, 2. the height growth correction via site indices (Schmidt 2020) and 3. the survival models of Brandl et al. (2020) (species-specific covariates are: $c_{min\ cm}$: minimum temperature of the coldest month (*Fagus sylvatica*); c_{yr}: annual temperature (*Abies alba*, *Pseudotsuga menziesii*), $c_{max\ wm}$: maximum temperature of the warmest month (*Picea abies*, *P. sylvestris, F. sylvatica*, *Quercus robur/ petraea*), p_{wq}: precipitation sum of the warmest quarter (*P. abies*, *P. sylvestris*), p_{yr}: annual precipitation sum (*P. menziesii*), where in the survival models *P. menziesii* and *P. abies* are co-dependent on the admixture ratio).

period	**region**	**point/ scenario**	$c_{min\ cm}$ [°C]	c_{yr} [°C]	$c_{max\ wm}$ [°C]	p_{wq} [mm m^{-2} a^{-1}]	p_{yr} [mm m^{-2} a^{-1}]
1983-2012	total	a	-13.2	9.0	32.0	181.8	673.2
	west	a	-12.8	8.9	31.7	199.0	765.9
	east	a	-13.5	9.0	32.5	161.7	565.9
2033-2062	total	b	-9.4	11.0	33.1	205.0	762.3
		c	-9.2	11.2	33.8	173.5	714.6
	west	b	-9.1	10.9	32.7	222.6	857.0
		c	-9.0	11.2	33.2	197.1	818.8
	east	b	-9.7	11.1	33.5	185.2	652.6
		c	-9.5	11.3	34.2	146.2	594.1

5.2.4. Mortality risk application

Definitions

Future forest development, and thus yield potentials are subject to uncertain mortality risks. While planned and regular (death from senescence and distance-dependent competition) mortality are already implemented in the *WaldPlaner* software, we couple the most accurate survival models available (that only comprise irregular mortality) with the growth and regeneration models of the simulation software. We execute the simulations in three steps using the environment ‘R’, version 4.1.3 (R Core Team 2022) and PostgreSQL databases (Conway et al. 2023).

We define disturbances as an “agent, or process, either biotic or abiotic, causing a perturbation (which includes stress) in an ecological [...] system relative to” the initial state (Rykiel 1985). Effects are characterized, on the one hand, by temporary or permanent deviations (perturbations) and, on the other hand, by stress-induced vitality losses ranging from survival to lethal trauma (Rykiel 1985).

In *WaldPlaner*, we include the tree's functional response (positive or negative growth trend after perturbation or survival stress) to changing climate and site properties by adapted site indices (Schmidt 2020). Lethal stress, represented as irregular mortality, is implemented from Brandl et al. (2020). Both models are driven by climate data described below. We distinguish between the mortality terms shown in the info box 2 as an effect to human, biotic and abiotic factors. Factors, also termed stressors, are known to cohere on the long- and short-term, thus predisposing, inciting and contributing to mortality (Manion 1991, p. 332).

Info box 2

mortality	**definition**	**source**
regular ~	unplanned but natural thinning/ elimination of trees caused by inter- or intraspecific competition (alien- or self-thinning) for resources, and senescence (ontogenesis)	(Brandl et al. 2020; Pretzsch and Grote 2023, p. 355)
irregular ~	also catastrophic ~; unplanned tree elimination generally induced by extreme conditions (disturbances) such as storms or insects (abiotic and biotic factors) irrespective of the spatial extent	
planned ~	human elimination by regular forestry operations as thinning or harvest (final cut)	
background ~	low rates of tree mortality found in forests in the absence of acute stressors from any cause	(Das et al. 2016)
dropout from ~	logging subsequent to unplanned, i.e. re-gular or irregular (salvage), and planned causes	

Climate covariates

For stocked areas (entity of a tree and its crown cover) of dominant and codominant individuals, surveyed between 1994 and 2017 (ICP Forests 2022), Brandl et al. (2020) fitted survival time models for six common tree species across Europe and West Asia (*cf.* Fig. 5.2 for *P. sylvestris*). In doing so, an irregular mortality rate for tree-area-units older than 20 years (age threshold) was set in relation to WorldClim climatic means from the fixed period 1971-2000 (Fick and Hijmans 2017) at a spatial scale of 1 km. Besides species admixture ratio for Norway spruce and Douglas fir, the survival models of the six taxa used covariates as characterized in Tab. 5.1 above. There was no mixing effect on the survival of the other species (e.g., pine) found.

First, to run these survival time models for our simulations, future climate data of sufficient spatio-temporal resolution, aligned with the FDT selection and height growth correction, is required. As the WorldClim dataset offers neither continuous future 30-year means at a detailed spatial scale, nor suitable RCM, which are fundamental for the scenario funnel (Fig. 5.1), for each subplot we chose climate variables from our selected RCM combinations (b: EC-Earth RACMO, c: HadGEM WettReg) as summarized in Tab. 5.1.

Layer separation

Second, the tree's social position must be considered, i.e. its layer affiliation. ICP Forests (2022) only considers overstory layers for irregular mortality, because the suppressed understory is dominated by regular die-off (Szmyt and Tarasiuk 2018). According to Weise's (1880, p. 37) suggestion for Scots pine, the top height h_w was defined from the 20% fraction of trees with the largest diameters by their 80 % height threshold (Assmann 1970, p. 91). It is also valid for uneven-aged multi-storeyed stands. In order to factor the small diameter differentiation in young stands, if h_w of stands is ≤ 13 m, every tree is regarded as overstory.

In line with Brandl et al. (2020), per subplot, the dominant tree species, the tree species share, as well as mean tree age are computed according to the number of trees, for the overstory.

For the lack of other models, we interpreted the taxa differentiation of Brandl et al. (2020) *sensu lato* and assigned the following tree species groups for the application of mortality risk (Tab. 5.2) according to national mortality rate time series (BMEL 2024b, pp. 46–48).

Tab. 5.2 Our assignments to the six taxa Brandl et al. (2020) focused on, as tree species groups. ODS/ L: Other Deciduous with Short/ Long life expectancy, spp.: *species pluralis*

Tree species Brandl et al. (2020)	**Trivial name**	**Tree species assignment**
Pinus sylvestris	Scots pine	*Pinus* spp., *Larix* spp.
Picea abies	Norway spruce	*Picea* spp., other coniferous spp.
Fagus sylvatica	European beech	*Fagus sylvatica*, ODS (spp. of *Betula*, *Salix, Alnus,* …)
Quercus robur/ petraea	Oak	*Quercus* spp., ODL (spp. of *Fraxinus*, *Ulmus, Acer,* …)
Abies alba	Silver fir	*Abies* spp.
Pseudotsuga menziesii	Douglas fir	*Pseudotsuga menziesii*

Hazard rates via Monte Carlo

Third, prospective yield potentials underlie uncertain tree survival probabilities, which are valid for single-tree and stand considerations (Brandl et al. 2020). Irregular tree and stand losses must be implemented dynamically in *WaldPlaner*, i.e. updated for every simulation step. Our approach considers hazard rates $h(t)$ as a means to build transition probabilities (Staupendahl and Möhring 2011) by deriving the survival function $S(t)$. A specific realisation of T (age at the time of an irregular loss) is denoted t (Klein and Moeschberger 2003). The individual observation unit in our application is the site with its tree-area-unit/ stand-area-unit, denoted s.

In order to simulate the binary event of s to die – or not to die – within the next 5 years, i.e. between t' and t'', a random number u is realized from a uniform distribution between 0 and $S(t_s')$, $u \sim U[0,\ S(t_s)]$. Whilst holding the subject's covariates (i.e. admixture, climate) – except its age for the single simulation step – constant, s dies when

$$S(t_s') - S(t_s'') \leq u \tag{5.1}$$

Hence, we set the year of death for the irregular loss T at t'. In the tree-wise mortality scenario (bt, ct) the hazard rates are applied randomly upon overstory single trees, and in the stand-wise scenario (bs, cs) for the whole stand by means of the main tree species group and an average stem-weighted overstory age.

Theorem: Our approach described here is equivalent to a simulation for the binary event of s to die – or not to die – within the next 5 years with $u \sim U[0,1]$ and $\frac{S(t') - S(t'')}{S(t')} \leq u$ as commonly used (Staupendahl and Möhring 2011; Knoke et al. 2021).

Proof: Let's denote $\tilde{u} \sim U[0, S(t'_s)]$ and $u \sim U[0, 1]$ with the following relationship: $\frac{\tilde{u}}{S(t')} = u$. As an example: $S(t') = 0.75$ and $u = 0.9$ is then $\tilde{u} = 0.75 \cdot 0.9 = 0.675$, which is equal to 90 % quantile of $U[0, 0.75]$. In our approach, we compare $\tilde{u} < S(t') - S(t'')$, which is – using $\frac{\tilde{u}}{S(t')} = u$ – equivalent to $u \cdot S(t') < S(t') - S(t'')$, and then this is (since $S(t') > 0$): $u < \frac{S(t') - S(t'')}{S(t')}$, which is the alternative approach (End of proof).

Our application of hazard rates is based on the Monte Carlo Simulation approach (MCS; Metropolis and Ulam 1949) with repeated random sampling for stochastic validation of the spatio-temporal distribution of irregular losses. Due to computing capacity, as single-tree simulations start 2012 with more than 7 million trees across the 17,551 sampling units (subplots), we repeated the random draws in ten MCS, as denoted by $r = 10$. Hence, according to Senf et al. (2021), a sample size of 175,510 was realized per scenario and simulation step of 5 years. In 2062, per repetition, all sampling units include around 11 million single trees.

In case of a stand-wise loss of s we consider salvage logging of both overstory and understory and thus an establishment of a new stand in line with its FDT prescription. Conversely, if a tree s dies, possibly together with its neighbour, the understory is preserved.

5.2.5. Uncertainty margins

To answer research question *i.*, we simulate with the focus on pine. For validation purposes, the other three main tree species spruce, beech and oak are also assessed. Uncertainty is calculated as follows: For each tree- and stand-area-unit s at each time step (i.e., age and year) at each subplot, a specific survival probability is obtained. At the level of the six federal states, this provides an uncertainty margin for the survival of the species and equals a sensitivity analysis. Therefore, we calculate an empiric median as well as the corresponding 95% interquantile range (IQR; $Q_{0.975} - Q_{0.025}$) of effectively realized $S(t)$ values for each possible combination of covariates (admixture, climate) over the years – but only on subplots where the tree species actually (will) grow. We contrast 30-year means from WorldClim data (1971-2000), on the one hand, and from our RCM (1983-2062) on the other, which results in deviations to the sensitivity analysis of Brandl et al. (2020).

To address research questions *ii.* and *iii.*, it is essential to differentiate between two cases of calculation, as outlined in the following two sections:

Single samples

Research question *ii.* shall be answered by calculating the FLR rate per plot as drafted by Knocke et al. (2024). As a first case, from *WaldPlaner* results for scenarios b and c as the single sample (one repetition without mortality risk), the mean of a target variable x per stratum h $(h = 1, \dots, L)$ (see section 'Study area and dendrometry') and per repetition r is

$$\bar{x}_{rh} = \frac{1}{m_h} \sum_{i=1}^{n_h} \sum_{j=1}^{m_i} x_{rhij} \tag{5.2}$$

where m_h is the total number of subplots in stratum, n_h is the number of sample plots in stratum h, m_i is the number of subplots at sample plot (cluster) i, and j is the index for the subplot. The repetition index is $r = 1$ in this first case. In the following, the cultivation area $\bar{a}_{rh}$ is the target variable. Then, the change rate z per repetition r of pure stands can be considered by computing

$$z_r = \frac{\bar{a}''}{\bar{a}'} = \frac{\sum_{h=1}^{L} W_h \bar{a}''_{rh}}{\sum_{h=1}^{L} W_h \bar{a}'_{rh}} \tag{5.3}$$

with $\bar{a}'_h$ as the estimated mean cultivation area in stratum h for time t'_0 (2012); $\bar{a}''_h$ is the estimated strata mean cultivation area for any time $t''_{0+\Delta t}$ and W_h is the stratum weight depending on the land area. The FLR_r ratio is then derived by $(1 - z_r) \times 100$. The plot, instead of the subplot, is considered as the sample unit in order to examine particularly pronounced monospecific clusters in forests (Bolte et al. 2021).

Regarding research question *iii.*, we consider yield potentials with variables volume increment, harvest volume and growing stock. For the latter, we assess small and large dimensioned timber (SDT/ LDT) below/ above the threshold of 50 cm DBH over bark as an object of interest for the timber industry and nature conservation.

Also, as a first case, we compute the total annual volume increment iv_r for simulation step t' until t'' according to Riedel et al. (2017, p. 62). It is defined as the periodical increment of the mean standing compact volume (growing stock, $\bar{v}_{rh}$) of trees multiplied with the stratum size A_h with

$$iv_r = \sum_{h=1}^{L} A_h \frac{\bar{v}''_{rh} - \bar{v}'_{rh}}{\Delta t} \tag{5.4}$$

Hence, in line with Gschwantner et al. (2024) for NFI framework, iv in our simulation represents a gross volume change that in fact disregards irregular losses. Thus, in the *WaldPlaner* software, regular mortality is represented exclusively in t'' with the year of death designated as t'.

As next target variable, for the same simulation step, the harvest volume as total annual cut cv_r is calculated as the sum quotient of dead trees volume and Δt: We add up regular mortality in t' (unplanned salvage logging displayed in t''), harvests (planned thinning and final harvest) in t'' as well as irregular mortality (unplanned salvage logging) in t', such as those described in section 'Hazard rates via Monte Carlo'. It is, according to eq. 4, estimated by a sum in the numerator. Furthermore, in the subsequent figures, the 5-year average for a simulation step, e.g. 2012 to 2017, is given as 2017. This mirrors a downstream salvage logging period after the year of death due to irregular and regular causes in 2012.

For computation of basic growing stock as well as mean ages, we refer to Riedel et al. (2017, p. 67) for further information.

Repeated random sampling

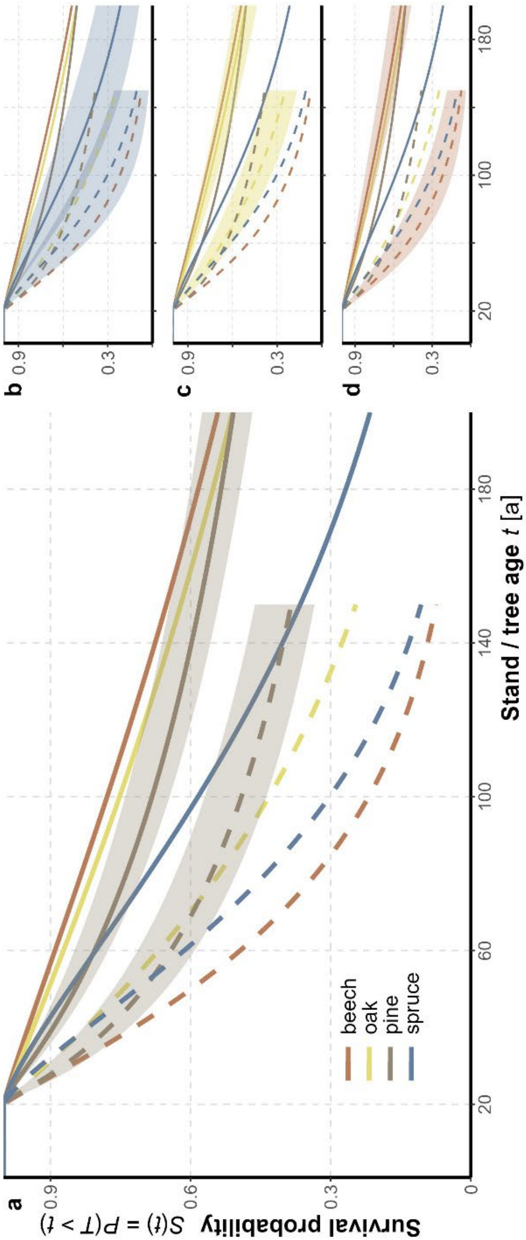
Regarding research question *iii.*, in the second case, for the MCS (scenarios bs, bt, cs, ct) with a tenfold iteration of the subplot simulations, all target variables $\overline{FLR}, \overline{iv}, \overline{cv}, \ldots$ are calculated by averaging among the repetitions $r_1 \ldots r_{10}$ to achieve a scenario mean. Furthermore, we define the scenario funnel to be delineated by the scenario means and not their particular spread (Kosow and Gaßner 2008, p. 54).

Beyond that, we denote FLR, MAI and MAC to be the mean forest landscape restoration rate, the mean annual increment and mean annual cut among all scenario means. We decided on this method of averaging the scenario means because we regard the hazard rates of Brandl et al. (2020) as the most pessimistic form of future mortality, so that a realistic forest development could be assumed to be between 'only regular mortality' and this pessimistic additional 'irregular mortality'.

5.3. Results

5.3.1. Survival probabilities

In our projections, survival time probabilities will considerably decrease for the main tree species Scots pine, Norway spruce, Sessile/ Pedunculate oak and European beech. In the style of a sensitivity analysis, the decline is depicted in Fig. 5.3 (dashed lines) and Tab. 5.3, when projecting our RCM's climate data from 1983-2012 until 2033-2062. Note, that before the age-threshold of 20 no irregular mortality takes place and slope is steepest at younger ages ≤ 60.



While beech has the highest mortality probability in simulations, pine and oak are depicted as being rather stable regarding their mortality risk. The IQR of our RCM shows comparably greater magnitudes (*cf.* Tab. 5.3), i.e. slightly increasing uncertainty margins for the survival of the respective species in our study area over time.

Fig. 5.3 Median survival probability of the main tree species for subplots in the study area with actual occurrence of the respective taxa as a function of age, depending on climate and admixture ratio (cp. Tab. 5.1), with **a)** IQR for *P. sylvestris* for 1971-2000 climate's means used by Brandl et al. (2020) (solid lines) and our RCM (EC-Earth RACMO & HadGEM WetReg) climate's 30 years moving windows starting 1983 and ending 2062 among scenarios (dashed lines). Subfig. **b)** - **d)** depict boundaries for spruce, oak and beech with overlapping IQR between the species, which can alter its subjective interspecific curve order and/ or slope per subplot

Only *P. abies'* IQR boundaries are overlapping between the climate used by Brandl et al. (2020) and our RCM's. In contrast, on a given sample plot, the other species only allow an interspecific change of the curve order within the parametrization (1971-2000) or the projection (1983-2062) values, i.e. per subplot beech could range above spruce (cp. Fig. 5.3) and it's $S(t)$-curve can have a different slope than the median.

Tab. 5.3 Numerical range of S_{100} survival probabilities for main tree species at 100 years of age between climate means used from Brandl et al. (2020) and our RCM among scenarios for each feature characteristic of climate and mixture with actual occurrence of the particular taxa until 2062

Species	WorldClim 1971-2000			EC-Earth/ HadGEM 1983-2062		
	$Q_{0.975}$	$Q_{0.5}$	$Q_{0.025}$	$Q_{0.975}$	$Q_{0.5}$	$Q_{0.025}$
Pine	74	**68**	65	57	**49**	44
Spruce	71	**58**	43	48	**30**	14
Beech	82	**78**	73	37	**22**	15
Oak	80	**75**	70	55	**43**	34

5.3.2. Uncertain increment and harvest potential

The simulations result in an increasingly uncertain increment in the study region, over all scenarios, with a distinct decline over time (Fig. 5.4a). For Scots pine, which shows the largest contribution of total volume increment, already the undisturbed development (solid lines; Fig. 5.4b) reveals a steady decline until 2062. This is caused by a planned reduction (i.e., more harvest than increment) due to a high proportion of old pines. Under irregular mortality risk (especially bs), this reduction intensifies around 2037 and from 2052 onwards.

At times, irregular mortality is even able to mitigate pine reduction (dashed lines above solid lines, Fig. 5.4b). Only *P. abies* will constantly lose more growing stock than it grows back from 2032 onwards.

Stand-wise (dashed lines) calamity pulses (Fig. 5.4b) reduce pine's increment and outnumber undisturbed scenarios (b, c) after 2052 or 2057, respectively, when young stands are restored. A positive compensatory effect by tree-wise (dotted lines) loss of competitors (compared to undisturbed development) on $\overline{\imath v}$ is hinted at in the more severe early mortality events (scenario c to 2027) and at the end of the simulations.

Heavy mortality pulses in pine lead to an instantly decreasing mean age of the pine overstory (Fig. 5.4c), when stand-wise losses occur. Undisturbed scenarios have the oldest overstory layers.

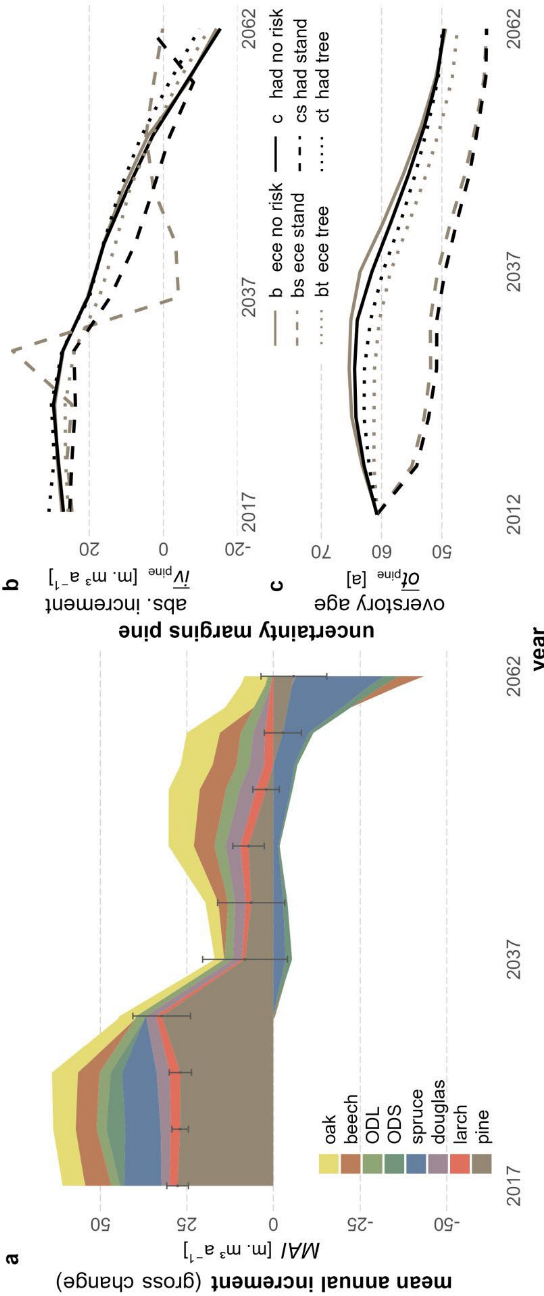
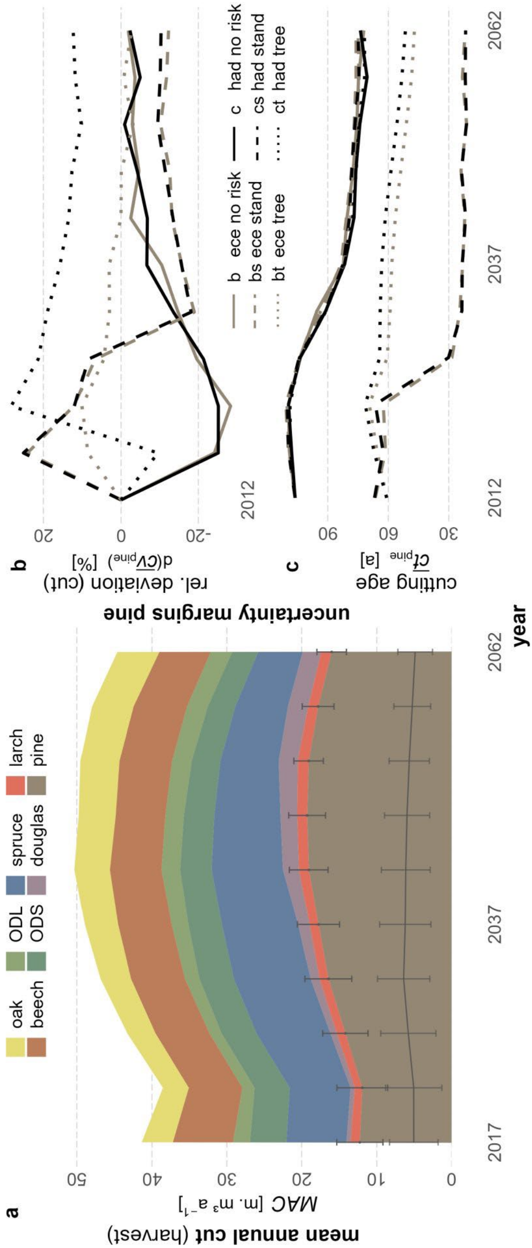


Fig. 5.4 **a)** Estimated *MAI* in the study area of tree species groups over time (assigning other pine than *P. sylvestris* to larch). Error bars for pine depict uncertainty among scenarios (ece: EC-Earth RACMO, had: HadGEM WetReg) and **b)** explains its absolute deviation from the mean with **c)** diverging mid age of the pine overstory



The collapsing increment in Fig. 5.4 is explained by high, but inherently uncertain harvest due to irregular mortality (solid line in Fig. 5.5a). Overall, salvage logging accounts for about one third of the future MAC. Due to this, already up to 111 m. m³ pine timber, on average, could already be lost within the first 20 years in our simulation, incorporating 26 % of its growing stock on circa 20 % of its cultivation area. After initially high ratios of

Fig. 5.5 **a)** Estimated MAC in the study area of tree species groups over time (assigning pine other than *P. sylvestris* to larch). Error bars for pine depict uncertainty among scenarios (ece: EC-Earth RACMO, had: HadGEM WettReg) while the lower grey line depicts the fraction of pine salvage cuts and **b)** explains its relative deviation from the mean with **c)** age development (top: final harvests, bottom: salvage logging) of pine trees salvaged timber (41 % mean in 2012), the proportion of planned cuts increases towards the projection period (69 % mean in 2062). Besides harvests (thinning), the ratio of planned *final* harvests increases from an average 21 % in 2012 to 63 % in 2062.

The relative deviation from both the mean (total planned and unplanned harvests of pine timber) is shown in Fig. 5.5b. While the MAC among scenarios is set to zero, it emerges that the overall highest cuts are initially driven by the stand-wise mortality regime (dashed lines). By the end of the simulation, however, tree-wise mortality dominates the maximum harvest volumes.

The declining rate of unplanned stand-wise harvests (dashed lines) from 2027 onwards indicates a calamity pulse, i.e. tipping point, from which the average age of salvage logging falls abruptly from 65 to 20 years (Fig. 5.5c, bottom curve bundle). Thus, from then on a restrained stand-wise mortality is implied (age threshold). Both the age of planned final harvesting and unplanned harvesting due to tree-wise mortality (dotted lines) decrease steadily from circa 110 to 70 or 65 to 50 years, respectively.

Fig. 5.5c also implies that the study area continues to contain old pine trees, which reach the age for final harvest (top curve bundle).

5.3.3. Uncertain stocks and restoration results

While the total growing stock among all tree species will increase from 1,193 m. m^3 in 2012 to a maximum of 1,443 m. m^3 (b) in 2062 (under the projection without irregular mortality), disturbances, especially the persistent tree-wise mortality, cause a decline to a minimum of 1,028 m. m^3 (bt).

Even if Scots pine is promoted in forest management, a managed reduction in pine stock, from 424 m. m^3 pine growing stock in 2012 down to between 312 m. m^3 (bs) and 339 m. m^3 (ct) in 2062, has to be expected.

However, scenarios with no mortality risk (b, c) also imply a drop of more than 70 m. m^3 for pine only (see section 'Uncertain pine increment and harvest potentials').

The cross-reference in Tab. 5.4 shows that relative uncertainties of the total pine stock in the total stock of all tree species in 2062 is influenced more by the mortality pattern than by the RCM. Nevertheless, the more restrained die-off (stand-wise mortality) with regard to the age threshold seems to lead to a clearer differentiation (bs vs. cs) and more restrained extreme climate (cs vs. ct) seems to level differences in mortality pattern.

Tab. 5.4 Summary of relative uncertainty [%] arising from RCM (b: EC-Earth RACMO, c: HadGEM WettReg) and mortality patterns for the growing stock of Scots pine in 2062. While in 2012 pine amounts to 35.6 % of the total growing stock among all tree species, its fraction increases under irregular mortality whereas absolute volumes decrease under mortality risk consideration (scenarios bs, cs, bt, ct)

RCM	no mortality	**stand-wise**	**tree-wise**	**| Δ mortality pattern |**
moderate	(b) 23.6	(bs) 27.4	(bt) 30.9	**3.5**
extreme	(c) 25.3	(cs) 29.6	(ct) 32.1	**2.5**
| Δ climate |	1.7	**2.2**	**1.2**	

On the one hand, in our simulations, we see a marked decline of small dimensioned timber (SDT), down to half of the original amount for both inclusion and exclusion of irregular mortality risk (Fig. 5.6a). The largest amount of SDT occurs, when irregular mortality is excluded (solid lines; Fig. 5.6c) and it is only at the end of our projection of silvicultural measures that the stand-wise mortality (dashed lines) might lead to more SDT, due to restored young stands.

On the other hand, large dimensioned timber (LDT) will increase fivefold (Fig. 5.6b). Here, irregular mortality uniformly reduces its

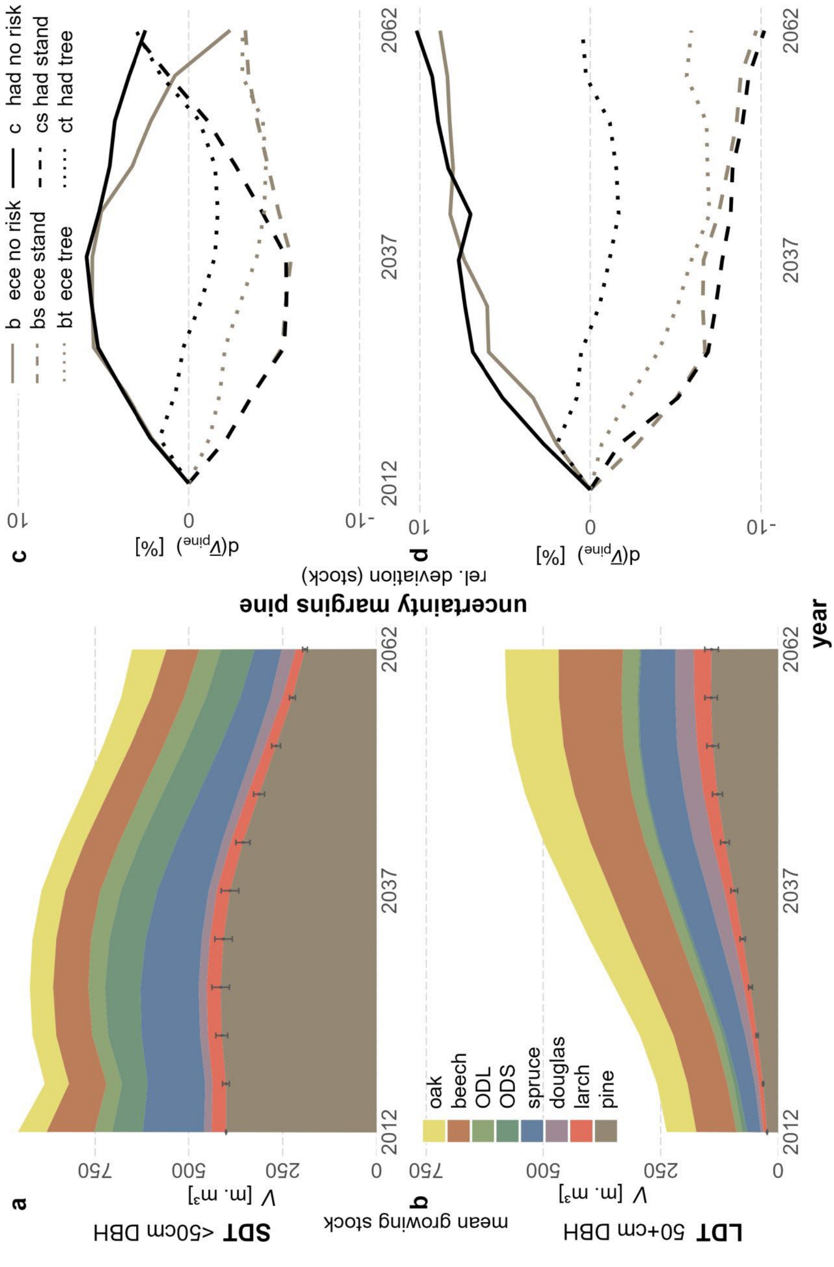


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Fig. 5.6 Estimated growing stocks in the study area of tree species groups over time (assigning other pine than *P. sylvestris* to larch) subdivided by **a)**, **c)** small dimensioned timber (SDT) and **b)**, **d)** large dimensioned timber (LDT). Error bars for pine depict uncertainty among scenarios (ece: EC-Earth RACMO, had: HadGEM WettReg). Subfig. **c**, **d** explain their relative deviation from the respective pine's mean

occurrence, especially when whole stands are subject to loss (dashed lines; Fig. 5.6d). Compared to MAI and MAC, absolute spreads around the pine mean growing stock are relatively small (Fig. 5.6a).

A forest management that targets Scots pine achieves a positive FLR (Fig. 5.7). For *P. sylvestris* in 2012, per plot, the west has about 133,000 ha and the east about 708,000 ha of monospecific stands.

In the western study region, under an atlantic climate, a higher thinning intensity at comparatively lower yield levels prevails. Accordingly, the west is relatively ahead in the restoration of monospecific pine stands until the middle of the simulation period in 2037. However, the east almost catches up by 2062 (Fig. 5.7a, b). The site amplitude in the east results in wider uncertainty margins for all main tree species. However, at 63 %, the west achieves about 8 % more pine restoration than the east across all scenarios in 2062. In total, the uncertainty within the west represents 4,000 ha and in the east 67,000 ha. In both regions, the restoration of spruce is most advanced and is less so for beech and oak.

The irregular mortality patterns both seem to accelerate the FLR over time, which is especially evident for stand-wise losses (dashed lines) in the east (Fig. 5.7d). A similar trend is depicted in the western region until 2022 (Fig. 5.7c). A sharp initial rise in the rate of restoration is particularly evident in the east, which can be attributed to a sustained wave. In the west, this pulse quickly disappears.

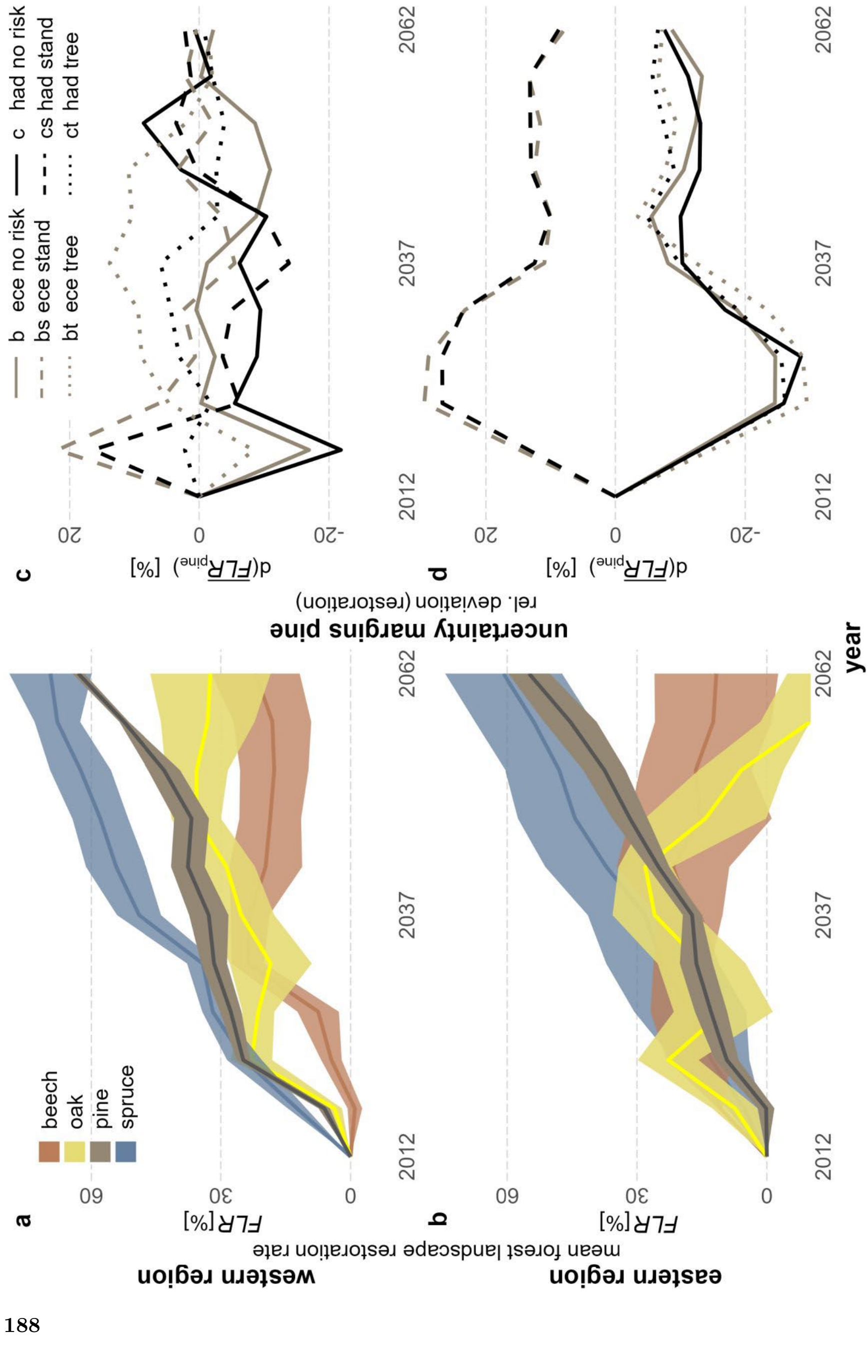

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Fig. 5.7 Estimated ***FLR*** in the study area of main tree species groups over time (assigning other pine than *P. sylvestris* to larch) subdivided by **a)**, **c)** western states and **b)**, **d)** eastern states. Light colored ribbons depict uncertainty among scenarios (ece: EC-Earth RACMO, had: HadGEM WettReg). Subfig. **c)**, **d)** explain the relative deviation from the respective pine's mean

Under certain constellations, the tree-wise mortality pattern impedes the restoration (dotted above dashed and solid lines). Climate appears to amplify FLR in some stages, i.e. extreme climate with more severe mortality hinders the establishment of new mixed stands (grey lines above black lines; *cf.* Fig. 5.7c, d).

5.4. Discussion

5.4.1. Material and Methods

For the first time, the present study successfully implements tree- and stand-wise irregular mortality risk in single-tree based simulations by means of repeated random sampling on a representative NFI grid to narrow down uncertainty. Climate variability and mortality patterns modulate guardrails of a scenario funnel that arises from dendrometry of the third inventory cycle (TI 2015). All tree species were simulated as the proportions of different species influence each other and have a significant impact on the overall forest development. However, special attention was given to Scots pine, as it is the most abundant tree species in northern Germany.

While the spread of future outcomes of forest development are displayed as means among scenarios, prediction errors accumulating over time are not measured. This is why the comparably short simulation period of 50 years was chosen (Hansen and Nagel 2014, p. 29), whereas other studies anticipate centuries (Knoke et al. 2021; Biber et al. 2024). However, *WaldPlaner* has undergone rigorous validation (Blattert et al. 2016; Holm and Schweier 2024). Thus, each sub-model, i.e. the growth simulator, the height growth model, and the survival model, has been analyzed with regard to its sensitivity (*cf.* Schmidt 2020; Brandl et al. 2020; Fig. 5.3).

As some studies created pooled simulation stands in order to synthetize over- with understocked NFI plots, an improvement in dendrometry is accompanied by a loss of exact climate- and site-data (Mey et al. 2021). When creating the present simulation stands, we accept that we to have reduced overstocked plots unilaterally, however this is in line with other studies (Albert et al. 2018). Creating virtual stands from sample plots could neglect some aspects of forests, e.g. neighbourhood circumstances (Fischer and Mölder 2017) and simplifying complex forests. Thus, results are only valid on large scales and not for evaluating few plots. We deal with this fact by tenfold repeated random sampling, which is in line with other studies (Senf et al. 2021; Albrich et al. 2023; Suárez-Muñoz et al. 2023). Furthermore, the chosen layer separator (cp. section 'Layer separation') could potentially lower the overstory's age and mixture, as in older stands the 20 % of the number of largest trees used for h_w might have also included understory trees. Yet, our stand sizes of 2,000 m^2 are above the lower limit of 1,000 m^2 identified by Assmann (1970, p. 91) to separate an independent spatial stand layer.

Because the subplots are to be regarded as dependent (autocorrelated), the NFI sample unit is the plot. As the subplots, which lie 150 m apart, can have different soil nutrient and water levels, different FDT have to be applied in order to project a realistic development and obtain yield potentials on a sound basis. Thus, in contrast to the German NFI design, in order to manage forest development directionally, we consider the subplot as the sampling unit. Only for the assessment of a robust FLR progress and in line with Bolte et al. (2021), for particularly prone monocultures, we do view the cluster, with up to 4 subplots, as being the decisive scale (*cf.* equation 5.2 – 5.3).

In the German context, the survival models of Brandl et al. (2020) are comprehendible and consider the major tree species: In contrast to other studies (e.g., Hülsmann et al. 2017; Tamke et al. 2021; Maringer et al. 2021), they encompass climate, admixture, a European-West Asian gradient and the tree's entire life span, while excluding regular mortality. Thus, these models have been used in recent scientific work (Paul et al. 2019; Knoke et al. 2021; Fuchs et al. 2022). Furthermore, practitioners apply the survival models from Brandl et al. (2020), coupled with TreeGrOSS, for certification purposes. However, they neglect site-specific climate input and determine the year of death imprecisely beforehand using survival probabilities and not conditional hazard rates (Vohrer 2023).

Climate data input has a great effect on the survival model performance (e.g., Fig. 5.4b). While there is no directional trend between the parametrization dataset and our values, the extreme covariates ($\bar{c}_{\min cm}$ and $\bar{c}_{\max wm}$) are offset from the WorldClim dataset of 1971-2000. This is due, for example, to a specific slope correction and the hot years 2003 or 2018 being excluded from WorldClim. However, these aspects are key, since micro-site conditions drive fitness and survival (Martinez del Castillo et al. 2024). The comparison between the extreme HadGEM model and the moderate EC-Earth model shows that the latter initially exhibits a colder trend in the backcast, i.e. the backward modelling. However, these two models intersect over time (supplement Fig. S5.8) and prospectively induce warmer values for HadGEM (c). The use of the 30-year moving window with averaged values for this period assists in smoothing the climate model predictions and thus mitigating abrupt trend changes (Hawkins and Sutton 2009). We are treating the results with caution but note that our climate input values are still within the calibration range of the Brandl et al. (2020) survival time models.

We see limitations, though, in these survival models and their previous application: In fact, interpreting disturbances only as stand-wise losses (e.g., Knoke et al. 2021) is not biologically plausible (Feldmann et al. 2018; Fuchs et al. 2022; Brodde et al. 2023). Also, the age-threshold of 20 years is to be questioned (Piri et al. 2021), which other authors dealt by adding a 2 % global mortality for their use (Knoke et al. 2021). Finding no mixing effects apart from Douglas fir and spruce, contradicts other models (Schoneberg 2017, p. 105). We acknowledge that simulating the extremes of diffuse, tree-wise and discrete, stand-wise mortality does not capture the continuous gradients characterizing the mortality regime of northern German pine forests (Knocke et al. 2025).

In line with the German NFI methods (Riedel et al. 2017) when using the *WaldPlaner* framework, we have to consider the year of death of both regular and irregular losses as t' and not the middle of the period $t' + \Delta t/2$. This is in contrast to Gschwantner et al. (2024) and might imply a statistical underestimation of the instant irregular death (i.e., trees might have had increment after t') together with an overestimation of planned harvest volume at the end of a period (t''). Both, under- and overestimation results in a balance that corresponds to practice and can be seen as a spread of salvage logging over the following 5 years, while planned harvests tend to be delayed (i.e., taking place at t''). Gschwantner et al. (2024) assess this minor bias further.

However, we point out that our study is only an estimate describing a hypothetical future of outermost scenarios that could materialize: The use of a variety of heterogeneously measured and modelled data results in uncertainty at a meta level. From there, we want to make clear that the above described effects can even be reversed, e.g. if the Gulf Stream were to collapse (O'Hare 2011) or politics set-aside forests on large scales (Lawrence 2017). Therefore, our results have to be interpreted as deep uncertainty and the scenario funnel determines guardrails in order to deal with the 'unknown unknowns' (Marchau et al. 2019, p. 6).

5.4.2. Uncertain survival chances

Our study reveals clearly lower and more uncertain S_{100} values than found by Brandl et al. (2020), which affects yield estimates. Under climate change, it is inherent in these primarily climate-dependent survival models that mortality can only increase in response to more extreme covariate input. Therefore, low survival chances may seem implausible when compared to current disturbance levels.

For Scots pine, our median survival S_{100} is 49 %, with an IQR from 44 to 57 %, which is similar to findings by previous research with 64 % (Dittrich 1986, pp. 51–53; according to Staupendahl 2011) for the prevailing conditions at the time of the study. If we acknowledge steadily rising mortality rates, as found by several authors (Sommerfeld et al. 2018; Senf et al. 2021; Patacca et al. 2023; BMEL 2024b, p. 46), our approach is expecting the worst but is still realistic. This pessimistic outlook is justified by our use of climate models under the RCP8.5 scenario, which represents a high-emission pathway and predicts severest impacts on forest health.

For the commonly assessed Norway spruce, in our simulations we use a median survival of only 30 % (IQR 14 to 48 %) tree- or stand-area-units reaching 100 years of age. In similar projections, under RCP8.5, other authors also found values reaching 37 % (Paul et al. 2019) or 41 % (Knoke et al. 2021), which are within our IQR.

To mitigate disturbances and enhance survivability, forestry should promote uneven-aged and site-adapted mixed-species stands, guided by the principles of portfolio theory and the insurance hypothesis (Markowitz 1991; Yachi and Loreau 1999). However, at the stand level, adaptive forest management must maintain the functionality of specific ecosystem services, even if this necessitates preserving a limited number of monospecific or even-aged stands (Schall et al. 2018). Such an approach can support the restoration of forest landscapes (Spathelf et al., 2018), enhances γ-diversity (Schuler et al. 2017; Heinrichs et al. 2019; Schall et al. 2020), and depletes disturbances (Sebald et al. 2021). Also no-regret-options, such as occasionally creating canopy openings, represent straightforward measures for adaptation (Muys and Messier 2023).

5.4.3. Shifting yield potential

For Scots pine, this study documents a strongly declining yield along with increasing uncertainties (Fig. 5.4). This is manifested by decreasing increments, especially for stand-wise losses, along with rising cuts, most notably for tree-wise mortality. Salvage logging accounts on average for 41 % of the total cuts in 2012 and 31 % in 2062. While all age measures (Fig. 5.4c; i.e., age of overstory, planned and unplanned harvests) together with small dimensioned timber (SDT) are decreasing, the amount of large dimensioned timber (LDT) increases steadily.

German forestry expects an overall declining yield potential even without irregular mortality risk, i.e. planned reduction. This results from a right-skewed age class distribution (old tree demography) of stands that were established in the contect of the Second World War (Nelson 2005; Popkin 2021; Rosenkranz et al. 2023, p. 11). With special regard to climate change, declining growth and yield, as well as growing stock, has to be expected (Albrich et al. 2023; Krasovskiy et al. 2024). A strong age effect influencing yield potentials also was found by other studies (Suárez-Muñoz et al. 2023; Krasovskiy et al. 2024; Vangi et al. 2024). However, an irregular mortality-induced lowering of tree age (Senf et al. 2021) might be attenuated if single-tree mortality is assumed to be prevailing (Fuchs et al. 2022; suppl.), and shown in Fig. 5.4c. Also, an overall acceleration of diameter growth due to consequent thinning from above and previous (moderate) climate change was found in Germany, which also affects the mean ages (Oehmichen et al. 2018, p. 42; Pretzsch et al. 2021) and enhances survival probabilities (Schoneberg 2017, p. 100). Thus, trees reach maturity nowadays earlier, if target diameter (i.e., final harvest criterion) is held *ceteris paribus*, because a larger tree crown yields more photosynthesis and the rotation time is reduced. On the one hand, negative effects for biodiversity are assumed because of less habitat structures in young stands (Turmukhametova et al. 2020; Senf et al. 2021). Tree health, on the other hand, will be likely to profit from lower tree ages (Schweingruber and Wirth 2009).

In cases of declining yield or understocked forest management classes, the default top-down approach of decadal forest planning distributes final harvests over a period of, for example, 40 years (Gerhardt 1923), in order to retain mature growing stock to finance the adaption efforts of FLR and artificial regeneration (Albert et al. 2017; Erdozain et al. 2024).

This will be more important, if irregular mortality and an age-effect reduces the gross increment further (Fig. 5.4).

At the stand level, depending on age and species, growth in trees at the edges of forest gaps can compensate between 20 % (pine: Preuhsler 1987, p. 148) and 50 % (spruce and beech: Kramer 1958) of lost stock. Competition reduction and increased resource availability explain only 70-85 % of this overyield (Biber and Pretzsch 2022). Accordingly, we found that tree-wise mortality buffers some of the losses in the short term, but no corresponding stand-wise compensation is evident. The latter only performs better in the long run, if young productive stands are successfully restored. In fact, higher MAC resulting from tree-wise mortality demonstrates that salvage logging adds to planned harvests. Similar to low-rate background mortality, thinnings here do not undercut the prescribed target basal area. Consequently, they are carried out throughout the simulation of tree-wise mortality in a planned manner, while still being constrained by a maximum volume removal of 80 m^3 ha^{-1} per 5-year operation cycle (cp. Knocke et al. 2024).

This pertains to the optimum basal area range (Vospernik et al. 2015) with the highest increments, which , in case of pine, ranges proportionately from 0.8-0.9 of the maximum density (Assmann 1970, p. 231). Within the regional climate model (RCM), the differences between tree- and stand-wise mortality account for about 13 % (2012) and 115 % of the undisturbed MAI. Already the choice of provenances can increase volumes by 40 % (Haapanen et al. 2016) or increment by up to 150 % (in 2061 under RCP8.5) as found by Chakraborty et al. (2024). Thus, we interpret the mortality-pattern induced spread of yield as notable but not crucial.

In Europe, salvage logging currently accounts for 16 % of the total harvested volume (Patacca et al. 2023), and prospectively can turn out to be 40 % (Bolte 2022, p. 262). For spruce, even ≥ 100 % of the planned harvests can be salvages (Krasovskiy et al. 2024). For comparison, a Standardized Disturbance Index was defined (Patacca et al. 2023): In Czechia among all tree species, it lately ranged up to 4.3 m^3 ha^{-1} a^{-1}.

Accordingly, for Scots pine in our study area, we identify index values from 3.8 (mean) to 6.1 $m^3 ha^{-1} a^{-1}$ (maximum) in 2012 (regular and irregular mortality), of which 1.3 $m^3 ha^{-1} a^{-1}$ are caused by competition (regular mortality, minimum). Fig. 5.5b illustrates that regarding the quantity, tree-wise mortality is generally higher than stand-wise patterns. Our corresponding scenarios (bt, ct) represent permanent low-rate mortality patterns resulting from both regular and irregular die-off. *WaldPlaner* calculates competition-related mortality in a distance-dependent manner (Hansen and Nagel 2014, p. 140) subsequent to the dropout of irregular mortality (as imported into the software). Hence, an interpretation of tree-wise mortality as interdependent, disturbance-induced background mortality is justified (Das et al. 2016; Archambeau et al. 2020). The quantity also is in line with Bonannella et al. (2024), who found *P. sylvestris* to be comparatively heavily affected by irregular mortality. We see our high values, of up to 6.1 $m^3 ha^{-1} a^{-1}$ (which for stand-wise mortality also comprise understory trees), as being realistic, as Patacca et al. (2023) remark, that small amounts are not likely to be salvaged because they are not detected in their remote sensing approach or, in practice, from ground staff. On the other hand, Patacca et al. (2023) discuss that, in practice, a proportion of healthy trees at the disturbance edges are also harvested. Dying trees can contribute to the spread of pathogens such as bark and jewel beetles. However, generalized removals need to be carefully considered because they destabilize remaining stands, warm up their interior climate and can have negative ecological effects (Thorn et al. 2018). Altogether, it was found that salvage logging is not always economically favourable (Knoke et al. 2021; Popkin 2021).

Interestingly, among our MCS, many plots manifested repeated losses at the very same simulation steps, especially at low elevations (cp. Albert et al. 2015; Albrich et al. 2023; Bonannella et al. 2024; Bose et al. 2024; Ogana et al. 2024; Enderle et al. 2024). We conclude, that the survival time models of Brandl et al. (2020) work very specifically and safeguard sample plots that are not stochastically prone to a loss. Certainly, pines in our eastern (planar) region are more endangered, which can already today be proven by the spread of biotic agents under Poland's continental climate (Jabłoński et al. 2019). Yet, we are not surprised by the stockpiling of LDT, which was previously mainly expected in the absence of mortality risk (Spellmann and Fischer 2016; Knocke et al. 2024) or even found to be missing (Dittrich 1986, p. 117). For Scots pine, Biber et al. (2024) also found initially increasingly mature stands under a steady mortality regime and concluded that they had not reached an equilibrium within the simulation period in their study.

Although there may be ecological advantages, we would like to point out the economic challenges and the need to re-orient the industry: Because European softwood sawmills are primarily equipped for SDT, large dimensioned assortments, in particular, can pose challenges to the timber industry (Spellmann and Fischer 2016; Mason et al. 2022). We conclude that future timber markets will be massively overstrained by both the LDT and the quantities of salvage logging, especially when occurring in pulses (Popkin 2021; Patacca et al. 2023; Biber et al. 2024).

5.4.4. Inert FLR progress

We show that, as forest landscape restoration is driving timber yield potentials, accounting for uncertainty is key. FLR exhibits particular differences between the study regions that are primarily caused by age, thinning intensities, and mortality. While relative differences are small, when looking at the absolute figures, more forests in the eastern part need to be restored and uncertainty spreads are greatest, there. Overall, after a 50 year period of assumed optimal circumstances (i.e. recruitment, seedling availability, and timber marketing in order to execute final harvests), FLR has only accelerated by 10% overall when compared to undisturbed.

This inertness can be explained, on the one hand, by the fact of promoting light-demanding species like pine. In comparison, underplanting of shade-tolerant species can take place earlier and promotes FLR (Knocke et al. 2024). On the other hand, mortality is severest in young ages (curve's slope, *cf.* Fig. 5.3), which has to be interpreted as youth risk

(Staupendahl and Möhring 2011). Thus, for both mortality patterns, it is not old monocultures being replaced by young mixed plantings, but middle-aged stands dying off, as illustrated by decreasing SDT (Fig. 5.6). Middle age stands are likely to already be mixed, if established after the 1990s (Spathelf and Ammer 2015; Borrass et al. 2017). Furthermore, the stand-wise mortality pattern only accelerates FLR until circa 2032 (*cf.* Fig. 5.7d), when it comes to a standstill and all stochastically prone plots are dropped out. Other retrospective studies that included past mortality risk found similar FLR rates (Höher 1994, p. 50; Vrška et al. 2017). In simulations that target pine, Suárez-Muñoz et al. (2023) point out the effects of stand age plus management intensity on FLR progress and describe higher stocking changes under RCP8.5. However, their study underestimated mortality and only determined 20 % decrease in pure pine stands in Spain until 2100. For central Europe, Krasovskiy et al. (2024) also point out to manage forests more intense in order to adapt to climate change and assume irregular mortality to shape forest structure largely. Under RCP8.5, they indicate that volume-based forest composition change will account for less than 10 % increase or decrease, relative to the as-is state, of coniferous species by 2060.

All studies give consideration to the immense cost of future FLR, even if natural regeneration prevails. Even if the adaption effort towards climate-smart forests excludes planting, soil scarification, fencing, or precommercial thinning still have to take place (Albert et al. 2017; Mason et al. 2022; Vacek et al. 2023).

Our results are in line with other studies, as from 2012 until 2025, there actually were disturbance pulses for Scots pine (Senf and Seidl 2021; Rohde et al. 2023, p. 26; Seidl and Senf 2024).

5.5. Conclusion

We conclude that six aspects of novelty have been addressed: Considering 1.) actual algorithm-selected, site- and climate-specific FDT with realistic growth under perturbation and stress (Schmidt 2020), and

2.) recognizing a federal state-specific business-as-usual silviculture practice with different management settings. Additionally, for the first time, we 3.) apply conditional hazard rates for tree- or stand-wise irregular mortality patterns, emphasizing a realistic disturbance-influenced forest development. In a simplified approach, 4.) random numbers are drawn between 0 and $S(t'_s)$ 5.) on a representative inventory grid, driven by 6.) climate moving windows (Burggraef 2024, p. 48) in order to enhance predictions. The default approach in survival predictions has previously been that only entire stands are assumed to die (Knoke et al. 2021) which is rather unrealistic (Feldmann et al. 2018; Patacca et al. 2023). Altogether, we see strong improvement in the current prediction abilities (BMEL 2017; Rosenkranz et al. 2023) for yield potentials using *WaldPlaner*.

In European forests, not only the cultivation area and growing stock (Knocke et al. 2024), but also their increment and yield potential are undergoing profound, yet uncertain changes in the 21st century, as demonstrated by our research: *i.* overall, mortality risk and uncertainty increase over time while *ii.* FLR only accelerates under persisting stand-wise losses. The target basal area cannot be undercut below a sustainable threshold due to self-imposed commitments and legal regulations, which – in reality – may result in tree-wise (background) mortality further impeding adaptation. Also, *iii.* increment collapses in the medium term, which could only be modulated by stand-wise mortality in the long term. Harvest peaks will be driven by disturbances, leading to salvage logging, which will complement an already anticipated planned stock reduction (Schmitz 2019). Thus, the yield situation is about to deteriorate. As timber sequesters carbon dioxide, after long periods of sink, north German forests could become a source of carbon (Albrich et al. 2023; Cienciala and Melichar 2024).

However, with a view to the growing stock, mortality patterns generate more uncertainty than variability in projected climate change. The uncertainty, understood as the divergence between scenarios, would increase if additional climate model strands (e.g., RCP4.5) were considered. Surprisingly, other predictions, which exclude mortality risk, are also valid to a certain extent, since the planned stock reductiondominates the yield potential. Here, the main driver is the old forest demography, with its right-skewed age class distribution, i.e. negative physiologic lifetime in terms of tree health and the forest owners request for a steady income. Thus, in addition to silvicultural management and climate models (Knocke et al. 2024), we have identified that varying patterns of irregular mortality introduce relevant additional uncertainty to a future theoretical forest development under disturbances.

Both, the climate variability (FDT) and mortality patterns are dependent on the global change, as expressed by the moderate RACMO EC-Earth12 and extreme WettReg18 HadGEM2 model of the RCP8.5. strand. Uncertainty ranges are in line with conservative estimates, which are hoping for the best but expecting the worst. We infer:

Ad *i.*)

1. To buffer increasing magnitudes and frequencies of climate change-attributed mortality and uncertainty, striving for patchy mixed stand portfolios gains importance from both, an ecologic and economic viewpoint.
2. Given the availability of high spatio-temporal data, new survival time models need to account for mixing effects and to differentiate between species' provenances.

Ad *ii.*)

3. While FLR, through structural-functional enrichment of stands, is the cornerstone of climate-adapted forests, silvicultural management must focus on varying practices and systems to enhance diversity. This includes underplanting shade-tolerant species earlier, in order to accelerate FLR while strategically retaining selected monocultures to promote γ-diversity.
4. For simulation use, further research on mortality, e.g. the damage severity per stand is required. By assigning stands randomly to empirically derived damage severities, ranging from 0 % to 100 % disturbed share, a conditional hazard rate could indicate whether

the stand will die, while the damage severity quantifies the magnitude, i.e. the dropout to be made. This allows the distinction of stepless mortality patterns, including those between tree- and stand-wise extremes.

Ad *iii.*)

5. The collapsing yield situation requires a diversification of final harvesting concepts, which, on the one hand anticipates cuts of maladapted and unstable stands (acceleration), and, on the other hand, increases the canopy cover periods, especially of high-quality timber or retain stands (retardation), in order to save the means of income and to finance FLR (Gerhardt 1923; Knocke et al. 2024).
6. To include local communities and the industry, future timber harvest tendencies must be communicated transparently and salvage logging, especially after tree-wise mortality events, should be reconsidered.

All in all, we strongly emphasize the need to actively adapt forests to climate change. Accordingly, the state silvicultural guidelines should be diversified and adapted quickly. It is with great concern that we see climate change progressing and forest ecosystems being inert, unable to shift correspondingly.

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Author contributions

HCK, RVN and MAl conceptualized the study. HCK performed the simulations and analysed the data. TZ, MAx and HFH supported data processing. Statistical analyses were supported by HSR, CF and KH. HCK and MAl wrote the first draft of the manuscript. All authors contributed to the final manuscript version and approved it.

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Availability of data and materials

All NFI data used in this article can be openly accessed at https://bwi.info/Download/de/BWI-Basisdaten/. Operational data of Forest Administrations stay confidential.

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5.7. Supplementary material

Tab. S5.5 Extended mean strata-weighted climatic conditions on the NFI subplots of the study area for the RCP8.5 moderate (b, EC-Earth RACMO) and extreme (c, HadGEM WettReg) scenarios at the start and end of our simulations that are the basis for 1. the FDT-selection, 2. the height growth correction via site indices (Schmidt 2020) and 3. the survival models of Brandl et al. (2020) (species-specific covariates are: $c_{min\ cm}$: minimum temperature of the coldest month (*Fagus sylvatica*); c_{yr}: annual temperature (*Abies alba*, *Pseudotsuga menziesii*), $c_{max\ wm}$: maximum temperature of the warmest month (*Picea abies*, *P. sylvestris, F. sylvatica*, *Quercus robur/ petraea*), p_{wq}: precipitation sum of the warmest quarter (*P. abies*, *P. sylvestris*), p_{yr}: annual precipitation sum (*P. menziesii*), where in the survival models *P. menziesii* and *P. abies* are co-dependent on the admixture ratio). Note the differing backcast values for the period of 1983-2012 to power the scenario strands 'b' and 'c'

period	**region**	**point/ scenario**	$c_{min\ cm}$ [°C]	c_{yr} [°C]	$c_{max\ wm}$ [°C]	p_{wq} [mm m^{-2} a^{-1}]	p_{yr} [mm m^{-2} a^{-1}]
1983-2012	total	b ece	-12.5	9.0	32.0	180.8	670.9
		c had	-13.8	9.0	32.0	182.7	675.4
	west	b ece	-12.2	8.9	31.8	197.9	762.7
		c had	-13.3	9.0	31.6	200.1	769.0
	east	b ece	-12.8	9.0	32.4	161.0	564.6
		c had	-14.2	9.0	32.6	162.4	567.2
2033-2062	total	b ece	-9.4	11.0	33.1	205.0	762.3
		c had	-9.2	11.2	33.8	173.5	714.6
	west	b ece	-9.1	10.9	32.7	222.6	857.0
		c had	-9.0	11.2	33.2	197.1	818.8
	east	b cece	-9.7	11.1	33.5	185.2	652.6
		c had	-9.5	11.3	34.2	146.2	594.1

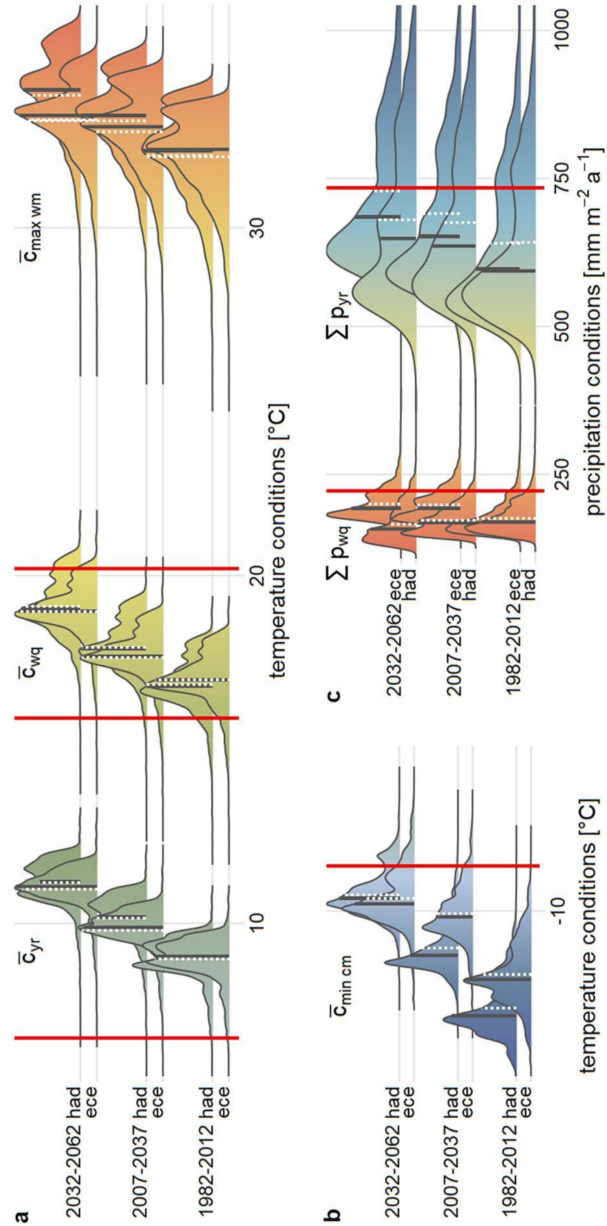

Fig. S5.8 Comparison of estimated kernel densities (Wilke 2017) for a, b temperature and c precipitation conditions on the NFI subplots of our study area under the RCP8.5 moderate (ece) and extreme (had) scenarios throughout the simulations (grey lines: median, white lines: mean) to the mean values (red lines) reported by Brandl et al. (2020) for the overall 1971-2000 period of WorldClim data on ICP Forests (2022) of Scots pine plots across Europe and West Asia (Fick and Hijmans 2017). Note, that our histograms lie biased but are still in the calibration range of survival models from Brandl et al. (2020)

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6. Waldumbau-Priorisierung (A4)

Eingereicht als Artikel A4:

„Integrating Remote Sensing and Site-specific Data
in Forest Planning: A Multicriteria Approach
to Prioritize Forest Restoration in Central Europe“

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Elsevier-Verlag
Trees, Forests and People (*Trees For People*)

Hypothese H4

*„Die Waldentwicklung Norddeutschlands
lässt sich auf Grundlage der landschaftsräumlichen Vielfalt
durch multikriterielle Ansätze bedarfsbezogen
für den Waldumbau priorisieren.“*

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Abstract

The increasing frequency of storms and droughts has led to severe disturbances in Central European forests, highlighting the urgent need for adaptation and restoration. However, structural deficits, low biodiversity, and fragmented forest ownership complicate effective management. This study develops a decision support tool using a multi-criteria decision analysis (MCDA) approach to prioritize forest restoration areas across ownership boundaries.

The methodology integrates remote sensing data, climate projections, and site-specific characteristics to assess key indicators of forest resilience: drought stress risk, tree species diversity at the stand level, and landscape-level rarity. The case study of Lüchow-Dannenberg in Lower Saxony serves as a practical example, demonstrating how flexible spatial approaches can overcome ownership boundaries and provide insights applicable beyond the study region. Results show that approximately 18.5 % of the study area requires urgent restoration due to high drought stress risk, as well as low diversity and rarity. Pine-dominated stands exhibit the lowest diversity and landscape rarity, while beech and oak contribute significantly to biodiversity. Sensitivity analyses reveal that weighting different indicators affects restoration prioritization, emphasizing the need for context-specific adaptation strategies.

This work underscores the importance of forest inventory data in guiding a multidimensional approach to forest stewardship and highlights how big data analytics can enhance forest science and decision-making. The study provides a practical framework for policymakers and forest managers to allocate resources efficiently, enhancing the resilience and sustainability of Central European forests in a changing climate.

Keywords · forest inventory data · remote sensing · forest restoration · forest planning

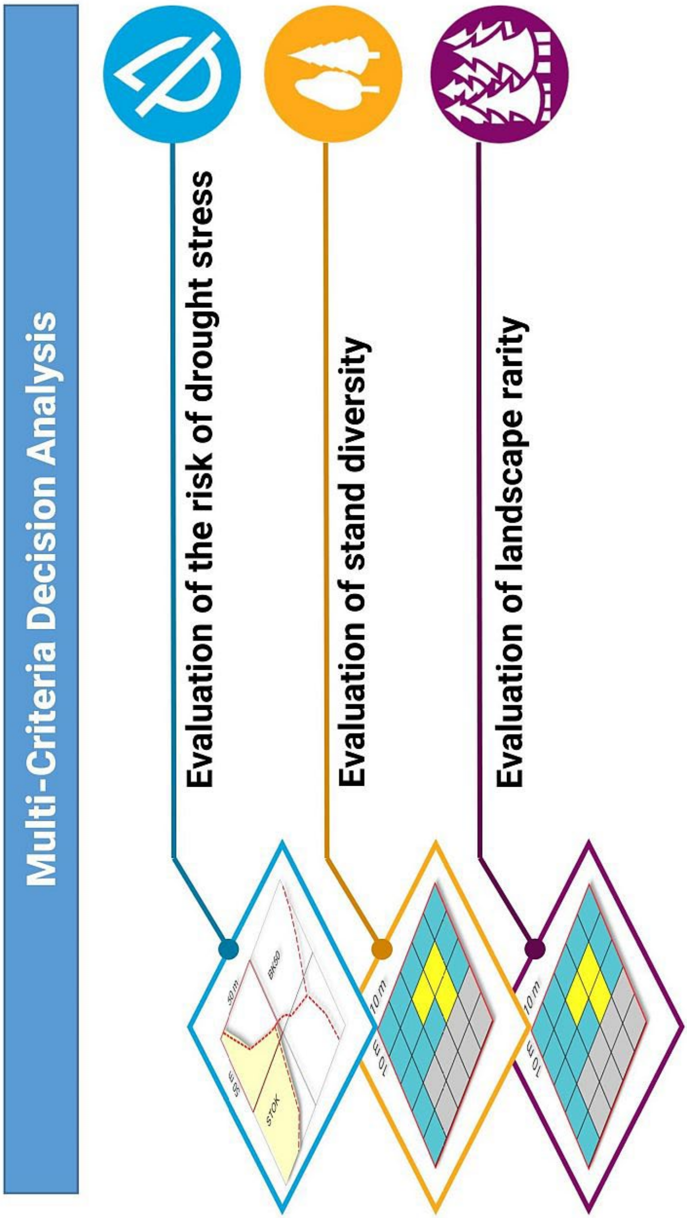


Fig. 6.0 Graphical Abstract “Waldumbau-Priorisierung (A4)”

Article highlights

→ A multi-criteria decision analysis (MCDA) approach prioritizes forest restoration areas.

→ The method integrates drought stress, tree species diversity, and landscape rarity.

→ Lüchow-Dannenberg serves as a case study for cross-boundary forest management.

→ Results highlight 18.5% of the area as highly vulnerable and in urgent need of restoration.

→ The study supports data-driven forest resilience strategies in a changing climate.

6.1. Introduction

In the ongoing battle against climate change, the imperative to adapt forests to drifting environmental conditions has become increasingly paramount. The cumulative effects of storms and drought in recent years have led to extensive disturbances in European forests and demonstrate the urgency of adaptation (Buras et al., 2020; Marini et al., 2017; Schuldt et al., 2020; Senf and Seidl, 2021). Large parts of the Central European forests are neither close to nature, which might grant resistance and resilience (BMEL, 2015; Hennenberg et al., 2017; PGL and LP, 2021), nor well mixed in order to spread risks (Bolte et al., 2009; Fuchs et al., 2024; Markowitz, 1991; Paul et al., 2020). The aims for the adaptation of Central European forests to climate change are to improve them structurally and functionally, i.e. enrich biodiversity by species composition and genetics, as well as by layering (Mansourian et al., 2020; Stanturf, 2016; WBW, 2022). Due to the enormous need for forest restoration with simultaneously limited capacities, further prioritisation of climate adaption measures is crucial (Anderegg et al., 2022; Pach et al., 2018).

Steps towards an order of preparing climate-smart forest ecosystems, have often been proposed but however in practice not implemented or realized yet (Knoke et al., 2020; Singh et al., 2015; Walentowski et al., 2017). Accurate information on forest conditions and the potential for forest restoration is lacking so far, especially for informing policy makers. Up to now, in German forestry, federal state and ownership boundaries have been limiting for information availability and thus forest restoration, as forest management plans are not mandatory for small-scale private forests. In state-owned forests, a permanent ownership-wide inventory is conducted. However, the datasets from state forest services are different with the sampling methods varying greatly to cross-ownership national forest inventories (NFI; Böckmann et al., 1998; Gschwantner et al., 2022). This data gap makes it difficult to develop a comprehensive understanding of forest conditions and the necessary measures for climate adaptation as well as societal demands.

Therefore, this paper delves into the development of a decision support tool with a robust and novel multi-criteria decision analysis (MCDA) approach for prioritizing forest restoration areas across ownership boundaries at a landscape scale. Wall-to-wall products derived from remote sensing data, such as dominant tree species distribution (Blickensdörfer et al., 2022, 2024), provide a powerful tool for bridging gaps in forest planning and enable cross-scale research. By integrating additional information, such as data on climate and soil, more nuanced assessments of forest conditions and restoration potentials can be made.

In the context of forest planning, decision support systems (DSS) have been used successfully for the long-term analysis of forest development in the past (Blattert et al., 2018; Linkevičius et al., 2019; Nordström et al., 2019; Thrippleton et al., 2023, 2021). For this purpose, forest inventories coupled with growth simulators and MCDA methods were utilized to assess different management scenarios in a changing climate (Blattert et al., 2017; Bugmann and Seidl, 2022; Maréchaux et al., 2021; Reyer et al., 2015; Wolfslehner and Seidl, 2010). However, in addition to the analysis of the optimal management, information about the prioritization of forest stands in regard to restoration efforts is crucial for decision makers. A practicable and robust DSS is largely lacking in this context. Leveraging the proposed multidimensional framework allows forest practitioners and politicians to make informed decisions regarding the allocation of resources towards forest restoration initiatives, thereby enhancing the resilience and sustainability of Central European forests (Nikinmaa et al., 2024).

The methodology outlined herein integrates various data sources, including climate data projections, such as site-specific characteristics and remote sensing data, to comprehensively assess **drought stress risk**, tree species diversity at stand level (**stand-level diversity**), and **landscape-level rarity** (Grier and Running, 1977; Heinrichs et al., 2019; Möllmann and Möhring, 2017; Müller et al., 2022). It is assumed that these indicators have an impact on the resilience of ecosystems and were therefore selected for detailed investigation of prioritizing forest restoration:

Drought stress risk is a crucial criterion as it directly impacts the resilience and health of forest ecosystems, particularly in the face of climate change. By assessing drought stress, the methodology helps to identify areas and vegetation types most vulnerable to water scarcity, which can lead to increased susceptibility to diseases, reduced growth rates (Albert et al., 2018; Schmidt, 2020), and ultimately higher mortality rates among trees (Bigler et al., 2006). The inclusion of drought stress as a factor allows for targeted adaptation measures, such as the selection of drought-tolerant species (Albert et al., 2017) or the implementation of water management practices to ensure the sustainability and functionality of ecosystems.

Stand-level diversity is selected to evaluate the variety of tree species within a stand, which is essential for maintaining ecosystem resilience and ecological balance. Diverse stands are generally more resilient to pests, diseases, and environmental stressors, as different species and genetic variability can buffer the system against disturbances (Pretzsch and Grote, 2023; Yachi and Loreau, 1999). By assessing diversity at this level, the methodology aims to support biodiversity conservation and to foster stands that can adapt better to changing environmental conditions. High biodiversity also contributes to enhanced ecosystem services, such as carbon sequestration, soil fertility, and overall ecosystem productivity (del Río et al., 2022; Pretzsch et al., 2015; Steckel et al., 2020).

Landscape-level rarity is selected to assess the contribution of rare tree species to the gamma (γ) diversity, or overall biodiversity, across a landscape. Rare elements can significantly enhance γ-diversity by adding unique species compositions and ecological niches that would not otherwise be represented (Qiao et al., 2023; Schall et al., 2020). This criterion allows for the identification of areas that, despite covering smaller spatial extents, contribute disproportionately to the richness and diversity of the entire landscape (Sebald et al., 2021). By focusing on rarity, the methodology supports conservation strategies that aim to protect these uncommon elements, thus maximizing γ-diversity and fostering ecosystem resilience at a larger scale.

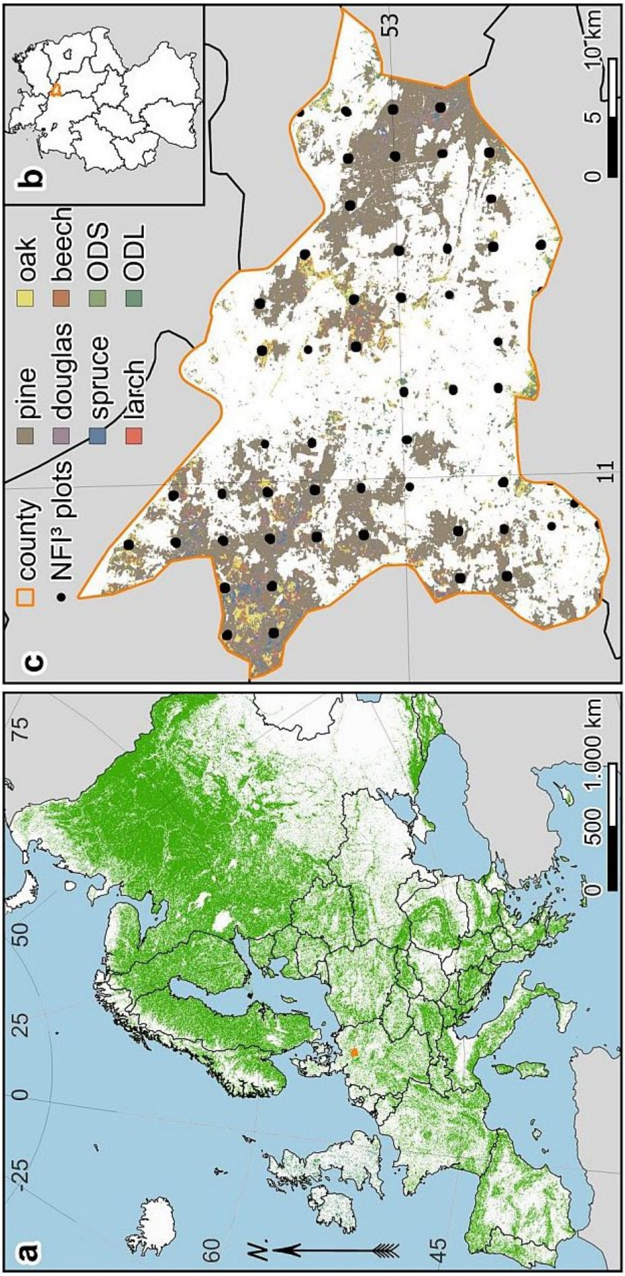
The temporal component of urgency is represented by a vegetation height model as an indicator for approximate forest stand age (Senf et al., 2021; Socha et al., 2023).

6.2. Material and Methods

6.2.1. Study region

The study area, encompassing about 47,000 ha of forest which cover 38 % of the area's land surface, lies within the county of Lüchow-Dannenberg in the North German Lowlands. This county in Lower Saxony also borders three other federal states. Characterized by its diverse topography and vegetative composition, the region offers a representative sample of the broader landscape dynamics typical of northern Germany temperate oceanic climate (Beck et al., 2018).

Comprising primarily coniferous and only few deciduous forests, the vegetation of the area is notable for its dominance by Scots pine (*Pinus sylvestris* L.; hereinafter "pine"), constituting approximately 67 % of the forest cover. Pedunculate and Sessile oak (*Quercus robur* L., *Quercus petraea* (Matt.) Liebl., hereinafter "oak") represent the second most abundant species, accounting for 11 %, while Norway spruce (*Picea abies* L.;



hereinafter "spruce") occupies a lesser proportion, amounting to 3 % of the forest. European beech (*Fagus sylvatica* L.; hereinafter "bee-ch") only plays a minor role in the main stocking so far. The study area was further chosen because it is characterized by a wide range of soil moisture and soil nutri-

Fig. 6.1 **a)** Forest cover in Europe (Burgoin et al., 2024) and **b)** location of the Lüchow-Dannenberg county at 11° E lon. and 53° N lat. in the German federal state of Lower Saxony, **c)** distribution of main tree spp. groups (ODS/ ODL: Other deciduous trees with short/ long life expectancy) according to Blickensdörfer et al. (2024, 2022).

ent availabilities. Elevation ranges from 6 to 145 m a.s.l., while in the period from 1991-2020 the climate was characterized by a 9.6 °C air temperature mean and a 629 mm m^2 a^{-1} annual precipitation sum (DWD, 2022). The future climate will change significantly in the study area and is characterised in Chapter 6.2.2.

The forest ownership in Lüchow-Dannenberg is dominated by private forests ranging from less than one hectare up to 5,400 ha in size (Junack, 1989). Together they cover an area of 32,000 ha (69 %), while state forests comprise 13,000 ha (31 %). Both state and private owned forests have an outstanding long tradition in forest restoration (Junack, 1924; von Unruh, 1936) – with only moderate achievement of the restoration goals. The famous big calamities in the study region have not ended until today (Habermann, 2017). All forest owners follow the integrative and multifunctional way of German forestry (Axer et al., 2023; Borrass et al., 2017; Larsen and Nielsen, 2007; Mölder et al., 2020; Simons et al., 2021). Alongside continuous cover forestry (Mason et al., 2022), operations such as small-scale clear-cuts are practised, which underlines a highly diverse management that could mitigate disturbances and facilitate restoration (Muys and Messier, 2023; Potterf et al., 2023). Furthermore, about 900 ha are strictly preserved forests where no management takes place at all (Bollmann and Braunisch, 2013; Engel, 2020; Steinacker et al., 2023).

According to the third German NFI, the forests in the study region have a mean age of 86 years in 2024 (vTI, 2015) which suggests a generational succession. Therefore, the biggest issue evolves around tree species selection under deep uncertainty (Albert et al., 2017; Knocke et al., 2024; Marchau et al., 2019) if priority restoration areas have been defined.

6.2.2. Remote Sensing and Site-specific Data

A remote sensing-based dataset was used to determine the drought stress vulnerability of tree species, the tree species diversity within the stand and the rarity at landscape level (Fig. 6.1). Tree species detection was based on the spectral-temporal reflectance signature in Sentinel-2 satellite imagery, as assessed from Blickensdörfer et al. (2022). The resulting map provides information on the canopy-dominating tree species at a spatial resolution of 10 m for the stocked forest area, as defined by Langner et al. (2022). The mapped tree species distribution refers to the years 2017 and 2018, as satellite imagery of this time period was used for the analysis. There were 11 tree species and tree species groups classified according to their spectral characteristics. NFI data was used to train the classification model and to estimate the map accuracy at the national level. For a comprehensive description of the methods and data we refer to Blickensdörfer et al. (2024, 2022).

A Digital Surface Model (DSM) was utilized to delineate average forest stand heights (Fig. 6.2). Specifically, we employed the latest Canopy Height Model (CHM) with a resolution of 1 m, which is readily available by the land surveying office of the federal state Lower Saxony (GeoBasis-DE/ LGLN 2024). This height model was generated using airborne laser scanning data collected between 2013 and 2019.

To get the average stand heights, we sampled the 1 m resolution CHM down to 50 m grid cells used in the other metrics. And to mitigate the effect of low canopy height values in forest gaps, which would lead to biased stand heights, the 95th percentile height within each grid cell was used.

In addition to the remote sensing data, site-specific data was processed and used to derive the forest restoration area. This assessment relies on observed climate data from the German Weather Service, climate projections for the selected RCP8.5 scenario, and site information from soil overview maps (van Vuuren et al., 2011).

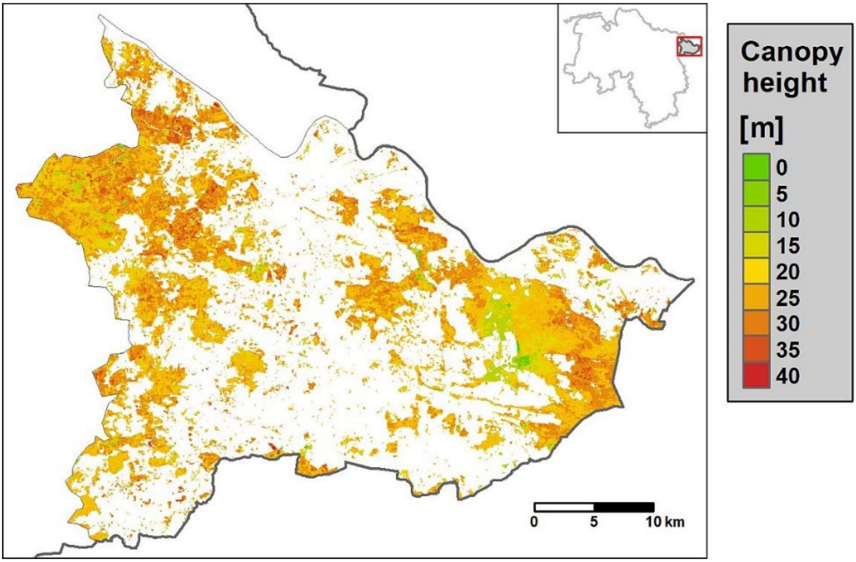


Fig. 6.2 Distribution of canopy heights in the study area. The map shows the canopy heights based on a Canopy Height Model (CHM) with 50 × 50 m resolution.

Climate projections for the period 2071 to 2100 were obtained from the ReKliEs-De project (Regional Climate Projection Ensemble for Germany; Hübener et al., 2017; Warrach-Sagi et al., 2018). Within the CMIP5 generation, the RCP8.5 scenario was chosen in order to investigate an extreme climate shift towards 4.8 °C temperature and 8.5 W m^{-2} radiation rise. Thus, the General Circulation Model (GCM) 'Hadley Centre Global Environment Model' (HadGEM2) combined with the empirical-statistical regional climate model 'Wetterlagen-basierte Regionalisierungsmethode' (WettReg18) were chosen to represent the most extreme climate projection (Kreienkamp et al., 2013; Martin et al., 2011). By selecting these statistical climate projections, we stand in line with conservative estimates typical for forestry that examine a pessimum. The regional climate projections were regionalised to 50 × 50 m grid cells. For these projections, the climatic water balance (CWB) during the vegetation period (the difference between precipitation and potential evapotranspiration) was calculated for a 30-year climate period of 2071-2100 (Allen et al., 1998; Nuske, 2022).

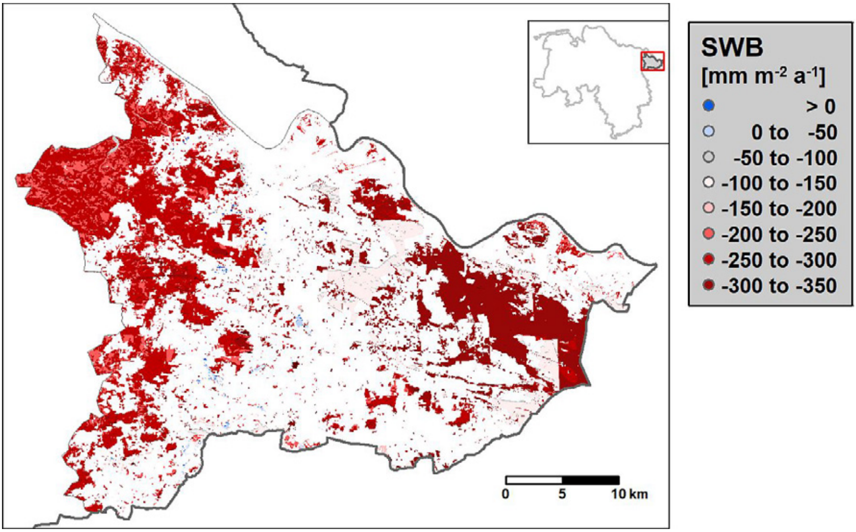


Fig. 6.3 Spatial visualisation of the Site Water Balance (SWB) as the sum of the available water capacity and the climatic water balance (vegetation period) for 2071-2100 and the HadGEM2 WettReg climate model.

In addition to the CWB, the soil's available water capacity (AWC) of a site was considered to determine the site water balance (SWB; Fig. 6.3; Grier and Running, 1977). Utilizing the pedotransfer function of Puhlmann and Wilpert (2011) and the input data from the forest site mapping, the soil water capacity was calculated where available. Otherwise, input variables from the 1 : 50,000 soil map were used (Gehrt et al., 2021).

6.2.3. Multi-Criteria Decision Analysis

Forest restoration urgency

The multi-attribute value theory (MAVT) is one of the most frequently applied methods for decision-making in forest planning. It is grounded in traditional utility theory and evaluates the importance of multiple attributes and their trade-offs. Utility theory relies on attribute value functions, which are mathematical models representing the assessment of a decision maker for each indicator. These functions convert the performance of indicator outcomes for each management option into a value score, indicating the extent to which a decision goal is achieved. The resulting score is dimensionless, ranging from 0 to 1, where 1 reflects the best possible performance and 0 the poorest.

While a specific alternative action is to be selected in the classic multi-criteria application of decision models, we use the MAVT method for our decision problem in order to obtain a spatially differentiated evaluation of the actual states. An important prerequisite for the prioritization of stocks worthy of restoration is the formation of a sequence that can ideally reflect the relationship between the individual evaluations. The MAVT method fulfills these requirements more than adequately.

As already described above, we focus on the indicators drought stress risk, diversity at stand level and rarity at landscape level. To add up the individual indicator values to an overall value, they are summed up in a weighted manner. First, equal weighting was given to the drought stress indicator D_i and the two rarity R_i and diversity E_i indicators. For grid cell i, the overall restoration value G_i can then be calculated as follows:

$$G_i = 0.5 \times D_i + 0.25 \times R_i + 0.25 \times E_i \tag{6.1}$$

The result is a value G_i that describes the urgency of the restoration. The higher the value, the lower the urgency of restoration. The lower the restoration value, the greater the urgency of restoration.

A value function can be applied either by linear transformation or by direct evaluation (Demetriou, 2014; Eisenführ et al., 2010). While the results of a linear transformation are dependent on the data set and change with it, direct evaluation is independent and globally applicable.

Both the choice of value functions and their weights have a very strong impact on the results of the respective decision problem and thus on the decision as such. Depending on the objective, however, a weighting of the indicators could be useful in order to further specify the forest restoration activity. A sensitivity analysis was therefore carried out with all possible combinations. The sensitivity analysis was conducted to explore the impact of different weightings on the forest restoration value G_i.

The temporal component of urgency is represented by the vegetation height model (*cf.* Chapter 6.2.2). Forestry management practices often define certain height or age thresholds at which a stand should be regenerated (Albert et al., 2015; Knocke et al., 2023, p. 98; Pach et al., 2018). If the height data indicates that the stand has reached or is approaching these thresholds, its urgency for restoration increases.

Drought stress risk assessment

The foundation of all climate adaptation measures lies in assessing whether the tree species currently growing or to be regenerated at a given site are suitable, based on current knowledge, to cope with both the prevailing and future climates.

Based on the SWB for the selected climate scenario and model (Fig. 6.3), the drought stress vulnerability of the current stand is estimated for each 50 × 50 m grid cell. This indicator integrates the AWC and the CWB for a given period (Grier and Running, 1977). The value function of drought stress risk is based on tree species-specific SWB threshold values derived from Albert et al. (2017). A monotonically increasing sigmoid curve was selected as the shape of the function (*cf.* Fig. 6.4). The threshold values of the tree species from Albert et al. (2017) were set as intersections of the curve with the values 0.25 (high risk threshold) and 0.75 (low risk threshold). The values used were selected to allow more differentiation in the dry and moist areas and thus keep the loss of information to a minimum.

Let $(x1_j, 0.25)$ and $(x2_j, 0.75)$ be the two points that define the monotonic sigmoid function for a given tree species j representing the drought stress threshold values. The general formula is:

$$d_{ij} = \frac{1}{1+\left(\frac{1}{0.75}-1\right)*e^{\left(-r\times\left(x_i-\left(\frac{x1_j-x2_j}{2}\right)\right)\right)}} \quad (6.2)$$

The function interpolates sigmoidal relationships between these two points for a given tree species j. $x_i$is the SWB value for the i-th grid cell and r is the slope parameter of the sigmoid function. It is determined using the *nleqslv* function to solve the equation associated with the sigmoid curve, given the input coordinates (Hasselman, 2023).

Let d_{ij} be the standardized drought stress score specifically associated with the j-th tree species for the i-th grid cell. p_{ij} is the proportion of the j-th tree species in the i-th grid cell, where j ranges from 1 to the total number of tree species n. Then the weighted standardized drought stress score D_i for the i-th grid cell can be calculated as:

$$D_i = \sum_{j=1}^{n} p_{ij} \times d_{ij} \quad (6.3)$$

The weighted standardized drought stress score can range from 0 to 1, with lower values indicating higher drought stress risk.

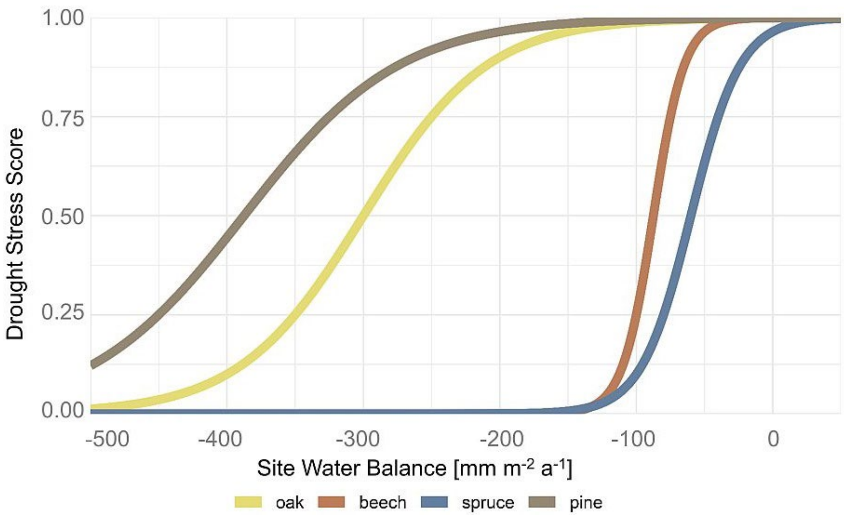


Fig. 6.4 Drought stress score depending on the site water balance (vegetation period) for the different tree species groups.

Diversity at stand level

Incorporating remote sensing data, information regarding species composition of dominant trees at the stand level (i.e., 0.25 ha) was further considered in the decision-making process. Diversity indices were derived from remote sensing data to identify stands with low species diversity.

Firstly, the species richness (S) as the total number of tree species within the i-th 50 × 50 m grid cell was determined. For this purpose, the 10 × 10 m cells of the remote sensing data set (Fig. 6.1c) were aggregated.

Subsequently, the Shannon index (H') at the 50 × 50 m was derived from the species richness (S) and the proportions (p_i) of 10 × 10 m pixel cells belonging to the i-th species using the following equation (Shannon and Weaver, 1949):

$$H' = -\sum_{i=1}^{S} p_i \times \ln(p_i) \qquad (6.4)$$

Evenness (E_H), on the other hand, is a measure of how evenly the tree species are distributed among the different species within the 50 × 50 m grid. It is often expressed as:

$$E_H = \frac{H'}{lnS} \qquad (6.5)$$

H' is the Shannon index and S is the total number of species in the community. Evenness ranges from 0 (completely unequal distribution of pixel cells among species) to 1 (total even distribution).

By computing evenness, an evaluation of each stand based on tree species composition can be conducted. This allows the identification of stands with particularly low species diversity, which can then be diversified for risk prevention purposes.

Rarity at landscape level

At the landscape level, the regional importance of each tree species is assessed concerning its suitability for restoration. To achieve this, a rarity value was calculated for each stand based on the proportions of tree species, which examines the relative rarity of each tree species within the landscape. The rarity value function was adjusted so that rare species have a high rarity weight, while common tree species have a very low rarity weight (Fig. 6.3). To determine the rarity of a tree species, the proportions of tree species within a radius of 1,700 m were extracted from remote sensing data (Fig. 6.1c). This represents a landscape area of 907 ha. Let A_{ij} represent the proportion of the j-th tree species within the i-th buffer. The rarity value function for each tree species within the buffer can be formulated as follows:

$$R_{ij} = 1 - A_{ij}^{k} \tag{6.6}$$

The exponent k is calibrated based on a reference proportion pc_x and a target rarity score pc_y, and is calculated as follows:

$$k = \frac{\log(1 - pc_y)}{\log(pc_x)} \tag{6.7}$$

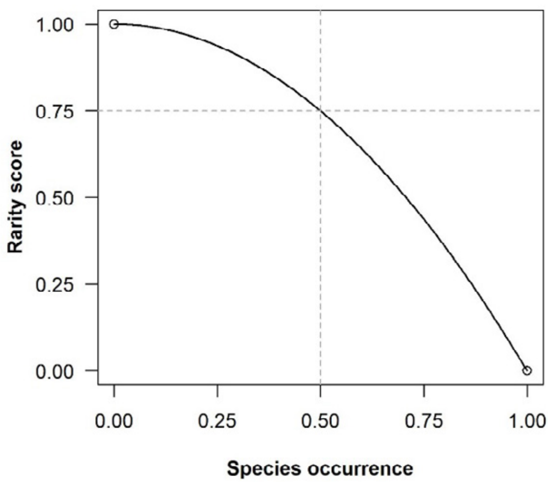


Fig. 6.5 Rarity value function depending on the proportion of tree species. A rarity score of 0.75 is assigned for a tree species proportion of 0.5

For the rarity value, it was assumed that a species has a rarity value of 0.75 if it represents 50 % of the species composition (Fig. 6.3).

Then, to calculate the rarity-weighted values for the raster-cell RW_i, we used the proportions of tree species within the raster cell p_{ij} to weight the rarity values R_{ij} accordingly.

$$RW_i = \sum_{j=1}^{n} p_{ij} * R_{ij} \tag{6.8}$$

6.3. Results

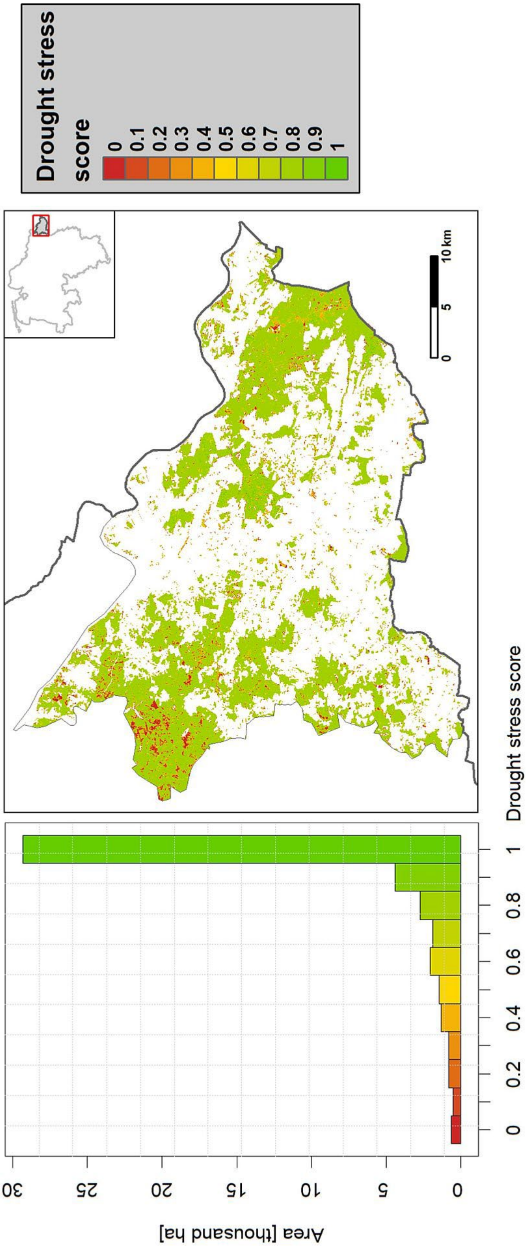
6.3.1. Drought stress risk

The analysis of the drought stress scores for different tree species shows a remarkable differentiation with regard to their susceptibility to drought stress at their sites in the study area (Tab. 6.1). Oak shows high values with an average of 0.86 and a median of 0.95, which indicates a relatively low susceptibility to drought stress. Pine with a mean value of 0.99 or ODS and ODL appear to be similarly unsusceptible. This means that these tree species are less affected by drought stress under the projected SWB (Fig. 6.3). In comparison, beech and spruce show significant differences in their drought stress scores. A mean drought stress score of 0.12 is projected for the existing beech stands and a mean drought stress score of 0.06 is projected for the spruce stands. However, there are also a few sites with a lower drought stress score for both tree species (Tab. 6.1).

Tab. 6.1 Summary statistics of drought stress scores for different tree species. The table presents the mean, median, standard deviation (SD), minimum, and maximum values of drought stress for each tree species

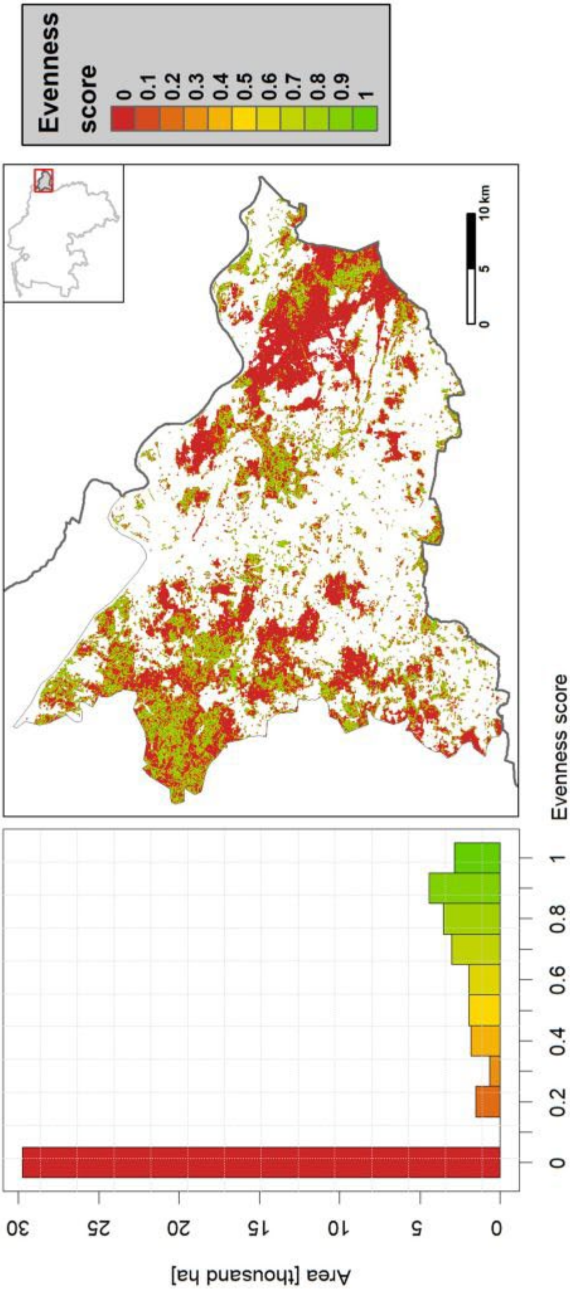
Tree species	mean	median	SD	min.	max.
Oak	0.86	0.95	0.20	0.05	1.00
Beech	0.12	0.01	0.25	0.00	1.00
ODS	0.95	0.99	0.10	0.32	1.00
ODL	1.00	1.00	0.00	0.95	1.00
Spruce	0.06	0.00	0.15	0.00	0.99
Douglas fir	0.86	0.92	0.18	0.15	1.00
Pine	0.99	1.00	0.01	0.87	1.00
Larch	0.97	0.99	0.04	0.53	1.00

Based on the drought stress analysis, the area required for forest restoration can initially be quantified on a site-specific and climatic basis. Approximately 32,000 ha have a drought stress score above 0.9 and thus



a low risk of drought stress (Fig. 6.6). The high percentage of pine trees is responsible for this (Fig. 6.1), as they receive a very low drought stress score (Tab. 6.1). Nevertheless, approx. 5,000 ha of the study area have a standardized drought stress score < 0.5 (Fig. 6.6). This means that this combination of tree species and sites is probably at risk in the face of climate change. Such areas are mainly stocked with spruce or beech.

Fig. 6.6 Spatial representation of drought stress score within the study area. The map illustrates the distribution of drought stress score across different regions, with colour intensities indicating varying levels of risk. The histogram displays the frequency distribution of dry stress risk values across the entire area, with the x-axis representing the risk levels and the y-axis representing the cumulative hectare values.



6.3.2. Diversity at stand level

From the distribution of the number of species and their proportions (Fig. 6.1), the evenness for the study area also shows that many stands are very homogeneous in terms of species composition. An area of 35,134 ha of forest has an evenness of less than 0.5. Only 5,363 ha have a high evenness score above 0.9. In the north-west of the study area in

Fig. 6.7 Spatial representation of evenness within the study area. The map displays the distribution of evenness across different regions, with colour intensities reflecting gradients of evenness. The histogram depicts the frequency distribution of evenness values across the entire area, with the x-axis representing evenness and the y-axis representing the cumulative hectare values.

particular, there are more mixed stands, resulting in higher evenness scores. Especially in the centre-east of the study area, there are large stands of pine trees without a mixture.

Looking at individual tree species, pine stands show the greatest homogeneity with an average evenness of 0.25 (Tab. 6.2). Spruce (mean = 0.66) and Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco; mean = 0.67) show moderate evenness values, reflecting more balanced species distributions, though they still experience some dominance. Oak (mean = 0.58) and beech (mean = 0.65) also exhibit moderate evenness, with oak and beech stands maintaining relatively higher diversity compared to pine-dominated areas (Tab. 6.2).

Tab. 6.2 Summary statistics of Evenness scores for different tree species. The table presents the mean, median, standard deviation (SD), minimum, and maximum values of Evenness scores for each tree species.

Tree species	**mean**	**median**	**SD**	**min.**	**max.**
Oak	0,58	0,72	0,35	0,00	1,00
Beech	0,65	0,75	0,30	0,00	1,00
ODS	0,72	0,78	0,23	0,00	1,00
ODL	0,65	0,74	0,30	0,00	1,00
Spruce	0,66	0,73	0,27	0,00	1,00
Douglas fir	0,67	0,73	0,26	0,00	1,00
Pine	0,25	0,00	0,36	0,00	1,00
Larch	0,71	0,76	0,23	0,00	1,00

6.3.3. Rarity at landscape level

Fig. 6.8 shows the rarity of the tree species at landscape level, i.e. γ-diversity. It can be seen that, above all, mixed oak or beech stands in pine-dominated areas, for example in the north-west of the study area, have a high rarity and receive correspondingly high rarity weights. The analysis of tree species-specific rarity values provides further insights into these patterns. Oak (mean = 0.67) and beech (mean = 0.82) exhibit relatively high rarity values (Tab. 6.3). This indicates their contribution to γ-diversity, as they are less common in the broader context.

In areas of neighbouring pine stands, there is a particular lack of species at the landscape level and the pine stands show low rarity values (mean = 0.43; Tab. 6.3). 29,135 ha have a rarity of less than 0.5. Of this area, 14,000 ha have a rarity less than 0.2, which indicates very low mixing at the landscape level (Fig. 6.8). These monocultures exhibit low diversity at the landscape level.

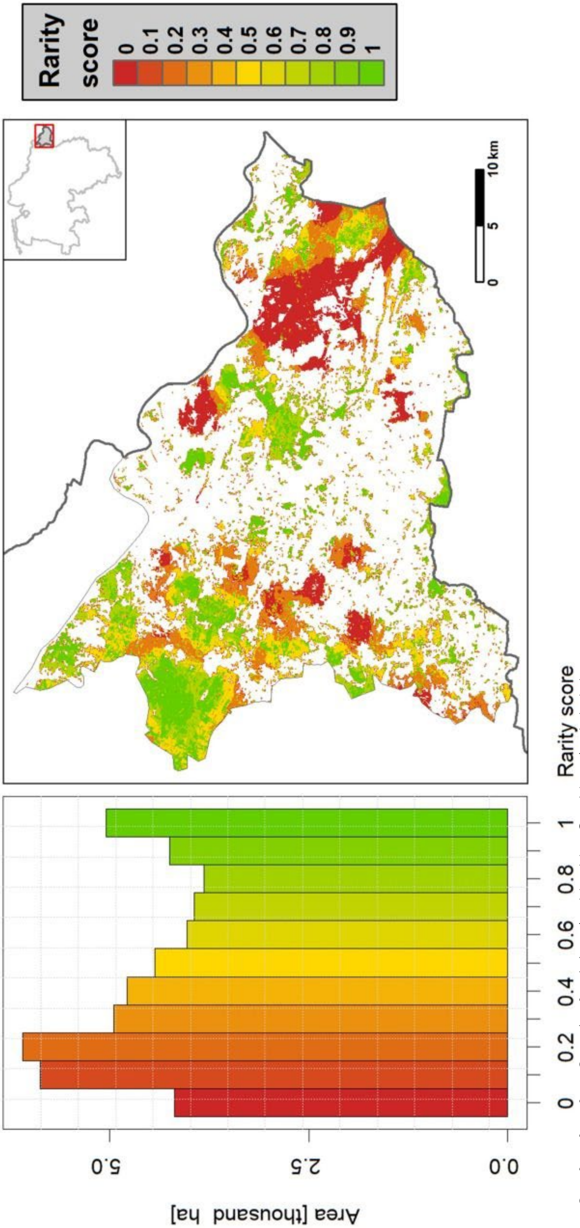


Fig. 6.8 Spatial representation of landscape-level species rarity within the study area. The map illustrates the species rarity of each cell, with colour intensities indicating varying degrees of rarity. The rarity of each grid cell was calculated in relation to the proportion of tree species within a radius of 1700 metres. The histogram displays the frequency distribution of rarity scores across the entire area, with the x-axis representing rarity levels and the y-axis representing the cumulative hectare values.

Tab. 6.3 Summary statistics of rarity scores for different tree species. The table presents the mean, median, standard deviation (SD), minimum, and maximum values of rarity scores for each tree species.

Tree species	mean	median	SD	min.	max.
Oak	0,67	0,79	0,31	0,00	1,00
Beech	0,82	0,95	0,24	0,04	1,00
ODS	0,58	0,61	0,29	0,04	1,00
ODL	0,61	0,67	0,31	0,02	1,00
Spruce	0,78	0,84	0,20	0,04	1,00
Douglas fir	0,77	0,83	0,21	0,04	1,00
Pine	0,43	0,40	0,28	0,00	1,00
Larch	0,72	0,79	0,25	0,04	1,00

6.3.4. Combining the indicators

Based on the derived drought stress risk assessment, the species diversity at stand level and the rarity at landscape level, the final step is to prioritise the forest restoration areas using a multi-criteria evaluation.

Fig. 6.9 shows the aggregated restoration values with equal weighting of the indicators for the study area. It can be seen that the eastern part of the study area, in particular, is highly worthy of restoration due to an increased future risk of drought stress and low diversity at stand and landscape level. At the same time, there are forest stands, for example, in the north-western part of the study area, which show a low risk of drought stress and a high species diversity at stand and landscape level.

Assuming a threshold value of 0.5 for a high forest restoration priority, approximately 18.5 % (8,689 ha) of forest stands have a high restoration priority, while 81.5 % (38,214 ha) have a low restoration priority.

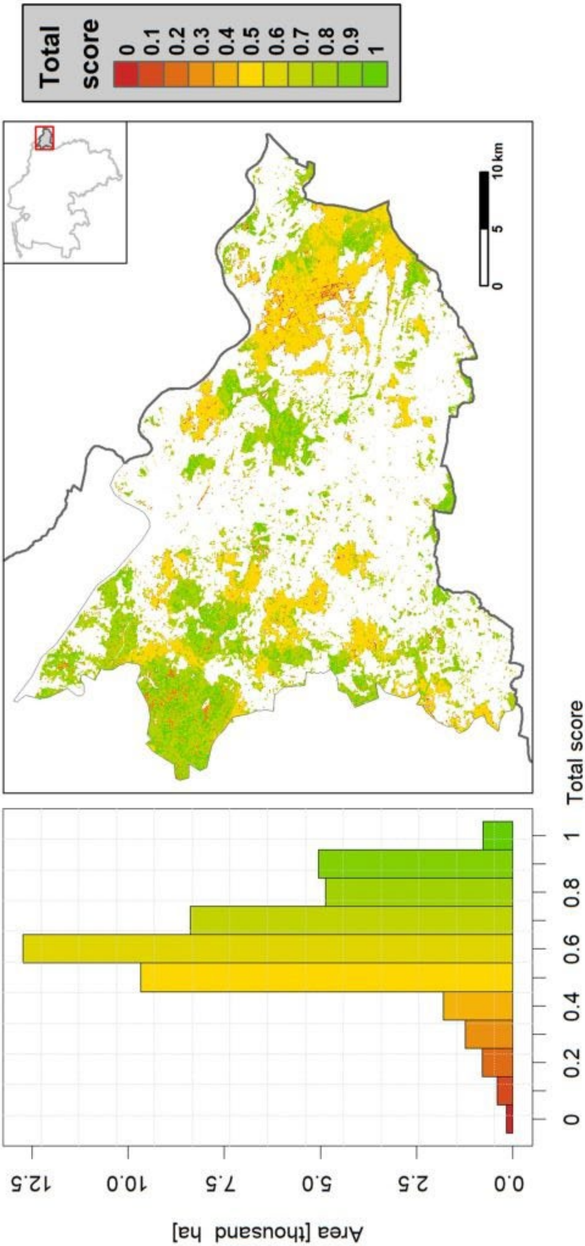


Fig. 6.10 shows the histogram of canopy height distribution for the 8,689 ha identified with high restoration priority (i.e., restoration values < 0.5) in the study area. The distribution highlights the varying stages of forest development within these areas. Most of the high-priority restoration areas exhibit average canopy heights indicative of mid-successional stages, suggesting that these stands may

Fig. 6.9 Aggregated restoration values with equal weighting of the indicators for the study area. This map illustrates the prioritization of forest restoration areas based on a multi-criteria evaluation, incorporating drought stress, stand-level species diversity, and landscape-level rarity.

benefit from interventions aimed at promoting structural diversity and resilience. A minor proportion of forest stands have average canopy heights towards the lower or upper end of the range. These forest stands in the early or late successional stage may require different management approaches. The varying average stand heights, as a proxy for successional development stages, suggest that targeted forestry measures, such as thinning or species diversification, could be strategically applied depending on the specific successional stage and restoration goals. This can guide the temporal prioritization of restoration efforts.

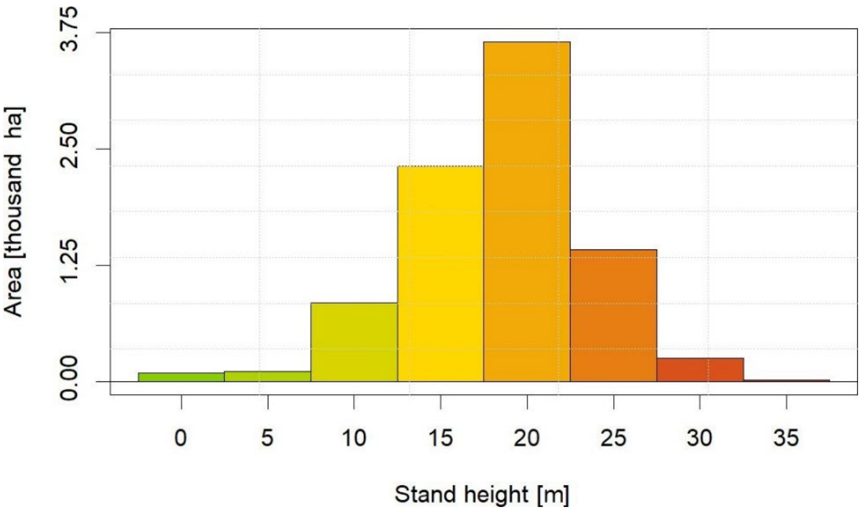


Fig. 6.10 Distribution of the canopy height of stands with increased restoration priority.

6.3.5. Sensitivity

The sensitivity analysis reveals the impact of different weighting schemes on the restoration value. Overall, the sensitivity analysis highlights the importance of carefully selecting appropriate weightings based on the objectives of the study. This tailored approach enables a more nuanced understanding of forest restoration dynamics and enhances the specificity of the results.

The largest area of restoration is observed with a weighting of 0.5 each for rarity and diversity, and 0 for drought stress (Fig. 6.11). With this combination, 33,600 ha are analysed that have a restoration value lower than the threshold value of 0.5. Conversely, if only the drought stress risk is considered and the other two variables are weighted with 0, the lowest forest restoration area is obtained. Around 5,000 ha with a restoration value lower than 0.5 will then be derived, demonstrating that drought stress alone identifies a much smaller area needing restoration.

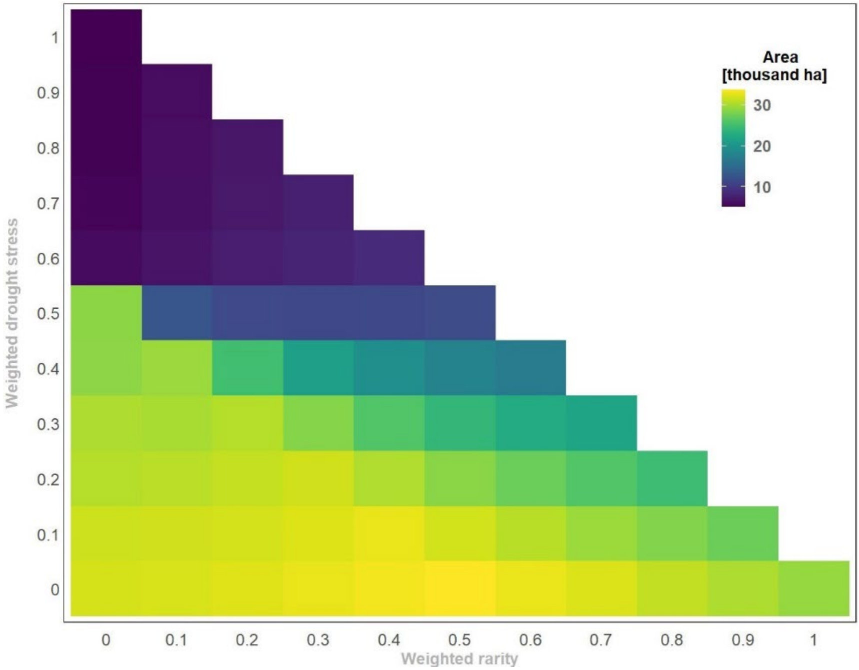


Fig. 6.11 Illustration of the sensitivity analysis, in which the variables drought stress, rarity at landscape level and diversity in the stand were weighted variably. The weighting of evenness is derived from the difference between the drought stress weight and the rarity weight. The forest restoration area with a multi-attribute value < 0.5 is shown as a coloured gradient.

6.4. Discussion

Drought stress risk, diversity at the stand level, rarity at the landscape level and stand age are the key criteria underscoring the multifaceted nature of forest resilience.

6.4.1. Drought stress risk

In the 21st century, a warmer and drier climate has to be expected (IPCC, 2014). For Europe, precipitations will not necessarily decrease (Hübener et al., 2017). However, disturbance frequency and intensity are expected to increase (Anderegg et al., 2022; IPCC, 2014; Patacca et al., 2023), implying significant impacts on the functioning of forest ecosystems (Allen et al., 2010; Senf et al., 2020). These impacts can be stress, disease and tree mortality (Rykiel, 1985). Causes of mortality (i.e., factors, agents, pathogens) are complex and are often classified as predisposing, inciting (primary) and contributing (secondary) factors, which can form a decline spiral and possibly lead to death (Manion, 1991, p. 333). In this context, climate change-induced droughts become the most prominent factor (Buras et al., 2020; Senf et al., 2020). Secular drought has to be seen as a predisposing factor and heavy drought events depict inciting causes for lethal stress (Bigler et al., 2006; Rykiel, 1985). Generally, it is difficult to distinguish if tree mortality is caused by a pathogen itself, or if a tree has died with this pathogen present.

Actual vulnerability to drought can also be clouded by background mortality (i.e., influenced by competition), especially in dense forests (Archambeau et al., 2020; Kulha et al., 2023; Sánchez-Salguero et al., 2015) or at edges (Biber and Pretzsch, 2022; Buras et al., 2018). Therefore, a sound drought stress risk assessment serves for identifying areas at heightened risk of mortality and reduced vitality. This encapsulates both growth reactions and the risk of exposure to biotic factors using the SWB (Jaime et al., 2019; Martinez del Castillo et al., 2024).

Norway spruce stands in the study area grow on a SWB of ≤ 0 mm m^{-2} a^{-1} within the vegetation period and thus possess the highest drought stress risk according to Albert et al. (2017; Tab. 6.1). This is evident based on the significant reduction in radial growth during drought years (Lévesque et al., 2014; Vitali et al., 2017), as well as increased susceptibility to biotic pests. Biotic pests, such as the spruce bark beetle (Jönsson et al., 2012; Marini et al., 2017; Stadelmann et al., 2013), contribute secondarily to tree death (Manion, 1991, p. 333). Therefore, the drought stress assessment we have applied (Fig. 6.4) is aligned with present studies that identify spruce stands as highly endangered (Aldea et al., 2024; Brandl et al., 2020; Fuchs et al., 2024; Honkaniemi et al., 2020; Lévesque et al., 2014).

Within the study region, **Scots pine** is the most abundant species (Fig. 6.1). However, it is also seen as having high drought stress, i.e. mortality risk (Bigler et al., 2006; Haberstroh et al., 2022; Hartmann et al., 2022; Lemaire et al., 2022; Patacca et al., 2023; Rehschuh and Rühr, 2021), while its hardiness and (epigenetic) adaptation potential is simultaneously praised (Bose et al., 2020; Bose et al., 2024; Brichta et al., 2023). Predisposing factors for tree death of pine are often associated with drought, as soil and climatic stressors affect health in the long term. Stony, shallow soils with low water capacity (Bose et al., 2024) and insufficient nutrient supply (Bose et al., 2020) primarily challenge pine in continental regions (Diers et al., 2024; Enderle et al., 2024). High stand densities (del Río et al., 2017; Jaime et al., 2019; Sohn et al., 2016), senescing old trees and missing (Steckel et al., 2020) or inappropriate admixtures (Špulák, 2023) also predispose pines.

Growth trends of pine in Europe strongly vary (Pretzsch et al., 2023). Central European pine stands, which are found in our study area, appear to compensate for the growth impairments caused by warmer and drier summers by stimulating growth through warmer late winter periods (Diers et al., 2024, 2023). Furthermore, there are indications that larger pines are more affected by drought than smaller trees (Merlin et al., 2015; Mueller-Dombois, 1987). The tree size is directly covered by our CHM (Fig. 6.2) and taken into account in further prioritization (Fig. 6.10).

Appropriate forest management of pine stands can mitigate drought stress. Thinning reduces competition for water, thereby lowering drought stress (Sohn et al., 2016, p. 20). The risk of pest outbreaks (i.e., bark beetle, butterfly or jewel beetle gradations), such as those linked to drought and higher temperatures (Netherer and Schopf, 2010; Ray et al., 2016; Skrzecz et al., 2020), further increases stress and vulnerability of pine. Even if biotic pathogens increase in intensity and frequency in the future (Jabłoński et al., 2019), they will continue to be linked to a drought-related weakening of the trees. This favors plant and fungal parasites as dieback due to mistletoe or *Diplodia sapinea* (Fr.) Fuckel (Brodde et al., 2023; Dobbertin and Rigling, 2006).

Recent studies show that **Sessile and Pedunculate oak** have a high resistance to drought and show less damage and lower mortality compared to beech and spruce (Brandl et al., 2020). This also matches our assessment (Fig. 6.4) and the results of the projections within the study area (Tab. 6.1). While other tree species in the northern German lowlands show a decline in growth with higher temperatures, oaks show a positive basal area increment trend (Enderle et al., 2024). This capacity to adapt to the increasingly warm and dry climate predicted for the future emphasises their importance as a key species for the design of climate-resilient forests (Gribbe et al., 2024; Rubio-Cuadrado et al., 2018; Vrška et al., 2017).

From the preceding discussion it is apparent that assessing drought risk is complex (Pretzsch et al., 2013). The methodology in this study accounts for the proportional representation of tree species within a stand (Chapter 6.2.3.). While intuitive, this approach does not account for species interactions, such as facilitation or competition (Bauhus et al., 2017; Pretzsch et al., 2013). In mixed forests, synergistic effects – such as complementary water and nutrient use – can reduce overall drought stress, even when individual species exhibit high stress levels. For pine stands, the inclusion of drought-tolerant species like oaks enhances resilience through resource complementarity (Giberti et al., 2023; Steckel et al., 2020). Conversely, competition from spruce, even when limited to the understory, can intensify drought stress for co-occurring species (Špulák, 2023). A higher survival rate was observed for spruce in mixed stands, also in a drier and warmer climates (Brandl et al., 2020; Neuner et al., 2015). As noted by Grossiord et al. (2014) and Forrester et al. (2016), drought stress can sometimes be lower in mixed forests. However, this is not a general pattern they found, as there were many two-species mixtures where both species had higher water stress compared to their monocultures.

Despite the complexity of mixed forest effects, our methodology provides a practice-orientated and sound basis for effectively assessing the drought stress risks of tree species (Fig. 6.6). The presented drought stress risk classification to prioritize forests for restoration reflects increasing mortality projections for lowland conditions (Allen et al., 2010; Nothdurft, 2013). The chosen climate model (Fig. 6.4) can be updated and adapted at any time to reflect new data or advances in climate science (Eyring et al., 2016; Pirani et al., 2024), ensuring that the drought risk assessment remains robust and relevant under changing conditions.

6.4.2. Diversity at stand level

The evaluation of biodiversity at the stand level provides valuable insights into its resilience (Yachi and Loreau, 1999). The incorporation of remote sensing data allowed for wall-to-wall mapping of tree species diversity in the study area and across ownership boundaries. Earlier approaches mainly served nature conservation purposes (Foody and Cutler, 2006; Madonsela et al., 2017; Redowan, 2015; Starčević et al., 2020). By calculating diversity indices such as species richness, Shannon index, and evenness, we identified critical regions for forest restoration requiring diversification.

The species distribution data we applied to model diversity indices is derived from optical satellite imagery with a spatial resolution of 10 m (Sentinel-2 data; Blickensdörfer et al., 2024). Consequently, there are certain limitations of this data in terms of species mapping and thus biodiversity indices. As wavelengths of optical satellite imagery are not able to penetrate the upper canopy layer, we only get information about the topmost trees. Potential species only present in the understory are not assessed. Because the reflectance of 10 m pixels consists of mixed signals, the processed information is only valid for the dominant species and might miss smaller trees; it is therefore less accurate for mixed forest stands. Additionally, the model was trained to accurately map the main tree species and is less accurate for rarer tree species. These limitations will most likely lead to underestimated biodiversity indices and should not be compared to indices based on field data. All these factors have to be considered when interpreting the results of the present tree species distribution and biodiversity indices. Nonetheless, the applicability remains, especially in the case of structurally poor stands (Fig. 6.1c).

Our findings demonstrate that many stands within the study area exhibit low canopy tree species diversity, with a significant proportion showing evenness values below 0.5 (Fig. 6.7). This highlights a notable vulnerability, as stands with low diversity are generally more susceptible to pests, diseases, and environmental stressors (Messier et al., 2022). A review by Jactel et al. (2005) confirmed the ‘diversity-stability theory’, stating that mixed stands are generally less affected by pests than pure stands. A comparison of the tree species within the study shows that pine stands are particularly species-poor (Tab. 6.2). The susceptibility of pure pine stands to needle-feeding insects was shown early on to decrease when deciduous trees are admixed (Lüdge, 1971). Higher densities of parasitoid wasps as antagonists of pest insects have been detected when deciduous trees, especially oak, were admixed (Jäkel and Roth, 2004). Furthermore, the probability of bark beetle infestation was higher for stands with a high pine basal area (Jaime et al., 2019). However, under the influence of recent megadroughts in Central Europe (Bose et al., 2022; Büntgen et al., 2021), evidence has emerged suggesting that very high species diversity can increase the likelihood of functionally redundant neighbors. This, in turn, reduces niche complementarity and intensifies interspecific competition, ultimately leading to higher mortality (Searle et al., 2022; Shovon et al., 2024).

In addition to biotic risks (Guyot et al., 2016), the diversity of forest stands also influences their resistance to abiotic factors, such as forest fires. Forest fires are expected to increase as a result of climate change due to lower precipitation and higher temperatures (Fig. 6.3). Evidence from fire-prone regions indicates that mixed forests containing both coniferous and deciduous tree species are less fire-prone than pure coniferous forests (Hély et al., 2001, 2000). In addition, fires in mixed forests tend to be spatially less extensive and their damages on the tree level less intense (González et al., 2006; Silva et al., 2009; Wang, 2002). This emphasises the ecological importance of promoting mixed stands, not only to reduce susceptibility to pests, but also to reduce the risk of fire. Furthermore, the influences of deciduous admixture in pine stands on site fertility (Błońska et al., 2018) and the productivity of mixed stands has been demonstrated (del Río et al., 2022; Pretzsch et al., 2015).

The preceding discussion shows that evenness emerges as a suitable indicator for assessing the resilience of forest stands, as low values indicate heightened susceptibility to biotic and abiotic stress factors. The predominance of pure pine stands in the center-east of the study area, underscores the need for targeted interventions to improve yield, ecological stability and economic viability (Knoke et al., 2008; Silva Pedro et al., 2015; Spathelf and Ammer, 2015). These pure stands could be diversified through management strategies such as mixed-species planting, seeding or natural regeneration facilitation to enhance structural and functional diversity (Axer et al., 2022; Fischer and Fischer, 2012; Fischer et al., 2016; Huth et al., 2017; Spathelf et al., 2018; Stanturf, 2016; von Lüpke and Sennhenn-Reulen, 2023).

Conversely, the higher evenness scores in the north-west region of the study area reflect a greater proportion of mixed stands, suggesting these areas may serve as benchmarks or reservoirs of resilience. This contrast between regions emphasizes the importance of spatially explicit strategies in forest management. By prioritizing the diversification of homogenous stands, especially in regions with low evenness, forest managers can mitigate risks and enhance the adaptability of forest ecosystems to changing environmental conditions (Schoene and Bernier, 2012).

6.4.3. Rarity at landscape scale

Forest management thrives to diversify stands regarding species, structures (layering) and genetics, i.e. α-diversity (Kimmins, 2004; see Fig. 4.5). However, this process inevitably leads to an alignment of forest types across the landscape (Fuchs et al., 2024; Pretzsch, 2019). As different forest sites promote β-diversity, so that swamplands or weak nutrient levels differentiate stocking (Stein et al., 2014), there are calls to focus more on β- and γ-diversity across the landscapes. β- and γ-diversity have been identified as important elements for the functioning of ecosystems (Mori et al., 2018; Schuler et al., 2017). Sebald et al. (2021) showed that mixing tree species between stands is at least as effective as mixing tree species within stands to mitigate disturbances. This includes structures, species and management concepts (Müller et al., 2022). We see significant potential in enhancing the diversity of management systems and preserving rare structural elements (Schall et al., 2020), particularly when incorporating varied thinning and harvesting approaches (Schall et al., 2018). Additionally, no-regret options, such as the initial creation of canopy openings, are being discussed (Messier et al., 2022; Muys and Messier, 2023). Where browsing and bark peeling are under control, these practices could also be implemented in the priority areas we identify.

We developed an approach for how to combine both viewpoints by enhancing diversity within stands through the evenness index (Larsen and Nielsen, 2007; Markowitz, 1991; Pretzsch et al., 2013) and by keeping rare elements at a landscape scale through the rarity index, even if they are monospecific or maladapted (Heinrichs et al., 2019). The approach of lowering the weight of common species supports diversity maximisation, while taking into account the ecological importance of rare species (Fig. 6.3). As numerous studies have shown, individual trees or groups of old oaks or beeches in homogeneous coniferous forests, in particular, create diverse ecological niches for different species groups (Koch Widerberg et al., 2012; Pilskog et al., 2016; Wehnert et al., 2020). Assessing the ecological value of single tree admixtures is even more important in landscapes with predominance of a single tree species over large areas (Fig. 6.1).

A further example is rare indicator species (Dufrêne and Legendre, 1997), such as the forest owl *Glaucidium passerinum* L.. They are especially present in lowland Spruce forests, whose silvicultural aspects must be restored (*cf.* drought assessment in Fig. 6.6), and could function as a tradeoff and be preserved from the viewpoint of ecology (Rothgänger, 2023). Light-transmissive coniferous species overall host many specialized taxa, which promote biodiversity (Brändle and Brandl, 2001; Heinrichs et al., 2019) and the multifunctionality of forests with their ecosystem services (Schuler et al., 2017).

In addition to nature conservation aspects, the rarity indicators also appear to be suitable for forest management. Rare tree species are of particular value at the landscape scale as seed trees, as they contribute significantly to natural regeneration and thus serve as cornerstones for forest restoration (Axer et al., 2022; Dobrovolny and Tesař, 2010; Kunstler et al., 2004; Zerbe, 2002). Rare elements are weighted particularly highly (Fig. 6.3) and can thus be considered as natural regeneration potential after disturbances (Schüle et al., 2023; Tiebel et al., 2020). This consideration becomes even more crucial when we account for the increasing risk of bark beetle infestations, which rises sharply with spruce volume at the landscape level (Stadelmann et al., 2013). Simulation studies confirm that reducing the proportion of spruce in the landscape increases the resilience of the remaining spruce trees (Honkaniemi et al., 2020). A study by Kautz et al. (2011) showed that the dispersal of the European spruce bark beetle *Ips typographus* L. is strongly dependent on distance and that 95 % of new infestations occurred within a radius of 500 m of the previous year's infestation sites.

From the preceding discussion, the assumption that rare tree species make a higher contribution to biodiversity at the landscape scale provides a basis for decision making in forest restoration. The approach provides a robust methodology for calculating rarity scores that can be linked to site-specific data to support forest restoration prioritisation (Fig. 6.8). These findings underscore the utility of remote sensing in biodiversity assessment and decision-making, enabling the identification of priority areas for intervention.

6.5. Standardization and Value Functions – Global Applicability and Impact of Weighting

As already mentioned in Chapter 6.2.3., the specific definition of the value functions is of crucial importance for the results. In order to be able to apply the value functions globally, it is equally important not to carry out a purely linear scaled transformation. This would make comparability between different scenarios impossible. Since the decision model should also be able to be adapted to changes in the input data with as little effort as possible, the global methods for defining value functions are also suitable here. This also allows changes between different data statuses to be quantified.

Furthermore, the results of the decision model are heavily dependent on the weighting of the individual indicators. The sensitivity analysis shows that the area increases significantly with a higher weighting of the diversity indicators (Fig. 6.11). However, it is up to the decision-maker to determine the weighting. There are various approaches for methodically determining the weighting (e.g. Analytical Hierarchy Process; Saaty et al., 2012).

6.6. Conclusions for Silviculture

For the large field of tree species selection, important information for forest development forecasts can be derived from this. In addition, it provides the basis for political, financial and personnel planning, which is based on the quantification of the restoration area and a projection of forest development. Entrepreneurs working on the restoration, such as tree nurseries or contractors, can also base their planning on this. This makes it clear how important the approach presented here is for climate impact adaptation in forest management and planning.

It is the integration of different spatio-temporal datasets that creates failure or success in the battle against climate change in order to adapt forest ecosystems. While previous approaches in Germany end at the ownership boundary (Grüll et al., 2020), we demonstrate how to combine site- and remote sensing information on the landscape level. The multidimensional approach applied here – spanning drought vulnerability, biodiversity evaluation, and rarity indexing – illustrates the capacity of big data to enhance decision-making processes. Overall, the analysis emphasizes the necessity of a comprehensive and flexible framework for forest restoration prioritization (Cannon et al., 2020; Cavalcante et al., 2022; Rayden et al., 2023; Vettorazzi and Valente, 2016). Our approach not only improves the accuracy and relevance of the restoration recommendations but also promotes adaptive forest management that fosters species-rich forests, increases biodiversity and integrates rare species as key components for ecosystem restoration and resilience.

Given limited resources, such as seedlings, labour and subsidies, future forest management must prioritize actions strategically (Charron and Hermanutz, 2016; Silva et al., 2023). To do so, existing funding mechanisms and silvicultural guidelines, such as quantified forest development types, need adaptation to effectively support restoration efforts (Hamkens et al., 2022; Larsen and Nielsen, 2007; Mason et al., 2018). Moreover, the active involvement of private forest owners is critical to ensure implementation (Stockmann et al., 2024; Westin et al., 2023).

In reality, silviculture can only succeed through a collaborative interplay of forest practitioners, politicians and partners, such as tree nurseries and sawmills (Beratan, 2014; Höhl et al., 2020) with forest restoration being the greatest driver (DVFFA, 2019; Knocke et al., 2024). However, forest restoration is a steady process without a starting or ending point and the current outcome is disillusioning (Fischer and Mölder, 2017; Knocke et al., 2024; Puettmann and Bauhus, 2023).

In the face of evolving environmental conditions, 21st-century forest planning must move beyond ownership-discrete and stand-based approaches, instead embracing dynamic strategies that balance timber production and carbon sequestration by weighting diverse indicators (Brèteau-Amores et al., 2023). Thus, diversity could be reached by a diversification of goals, i.e. weights if growing stocks of well-adapted stands can be increased or highly threatened monospecific stands must be harvested and restored to what serves the timber industry.

We see the opportunity that, through our research, adaption is accelerated and the bioeconomy, with its forestry and timber cluster, and nature conservation become more cross-linked shareholders by the concept of 'Forestry 4.0' (Knoke et al., 2023).

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Author Contributions

Conceptualization: M.A., H.H.; Methodology: M.A., H.H.; Software: M.A., H.H., J.W.; Formal analysis: M.A., H.H., J.W., H.K.; Data Curation: M.A., H.H., J.W.; Writing – Original Draft: M.A., H.K.; Writing – Review & Editing: H.H, J.W; Visualisation: M.A., H.H, H.K.

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7. Diskussion

7.1. Hypothese H1

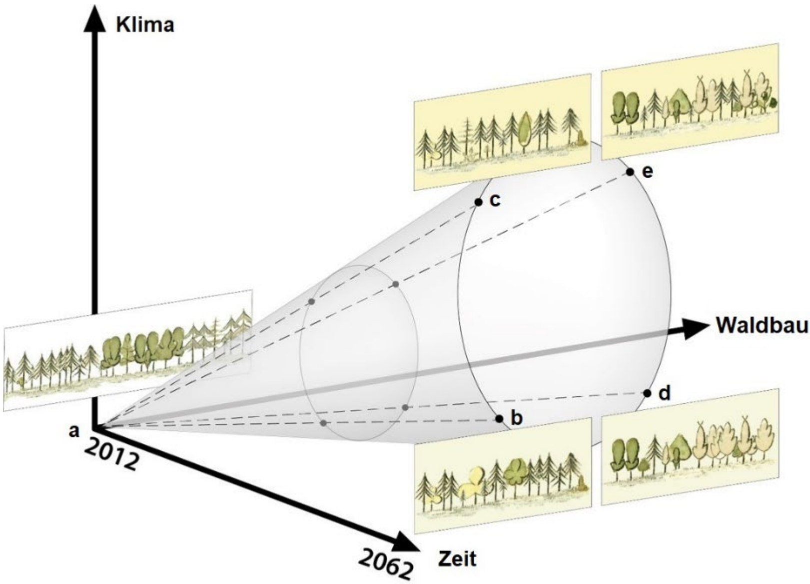

Abb. 7.1 Im Vergleich zu Fig. 3.3 verdeutlichter Szenariotrichter: Ausgangspunkt des Waldumbaus und Szenarien mit aufgespanntem Unsicherheitsraum. **a)** *Status quo* heutiger Waldbestände gem. BWI³ mit Umbauverpflichtung. **b)** - **e)** Gleichwahrscheinliche Szenarien mit präskriptiven WET je nach Klimabedingungen des Zeitraums 2071 - 2100 (**b**, **d** vs. **c**, **e**) und je nach Maximierung (**b**, **c**) oder Minimierung (**d**, **e**) der Kiefernanteile. Abbildung basierend auf Guimpel (1819)

„*Die Waldentwicklung Norddeutschlands wird ohne Störung eher durch die Wahl der Klimamodelle als durch die Vielfalt von Waldentwicklungstypen beeinflusst.*“

Entgegen der Hypothesenvermutung wird die simulationsbasierte Waldentwicklung eher von der Vielfalt der WET (Waldbau-Achse), als von der Klimamodellwahl (Klima-Achse) beeinflusst (*cf.* Fig. 3.5 und Tab. 3.2). H1 muss somit verworfen werden. Dabei werden die Klimamodelle und WET im Rahmen der Forstplanung moduliert, aber das Ergebnis der

Waldentwicklung wird durch das forstbetriebliche Management (Zeit-Achse) bestimmt. Es liegt allen Achsen ein nur schwer antizipierbares, subjektives menschliches Verhalten zugrunde. Dieses beeinflusst den Ausgang des Klimawandels, die WET-Steuerung und forstliche Eingriffe.

Untersuchungen zur Metaebene von Forstplanung und Waldbau hinsichtlich möglicher Auswirkungen einer Entscheidungsbandbreite sind selten, weil die „Variation in Waldbaukonzepten“ (Pretzsch & Knoke 2017, p. 513) durch eine komplexe Datengrundlage abgebildet werden muss. Dies ist insbesondere der Fall, wenn in Modellen partizipative Planungsansätze (z.B. Prozessschutzgebiete, Habitatbaumkonzepte) zugunsten von akteurspezifischen Aussagen berücksichtigt oder Multikohorten (Behandlungsgruppen) in Waldumbau-Entwicklungsgängen simuliert werden sollen (Oliver et al. 2016, p. 55; Pretzsch & Zenner 2017).

Besonders effektstark auf das Simulationsergebnis von A1 erwies sich auf der Zeit-Achse das Durchforstungsregime (Managementintensität), wie schon von Wiedemann (1948, p. 38) beschrieben. Die hinsichtlich einer erhöhten Bestandesdichte von Kiefernbeständen optimale Produktivität bezeichnet A1 als risikoaffin (östliche Bundesländer), wohingegen Bestandesdichten unterhalb der kritischen Grundflächenhaltung als risikoavers (westliche Bundesländer) gesehen werden (Fig. 3.6). Hinsichtlich der Baumartenwahl verhalten sich alle Bundesländer risikoaffin (*cf.* Kap. 7.5 & Fig. 3.5), weil Kiefer vielerorts als führende Baumart geplant werden könnte (Fig. 3.5b, c). Besonders ST sticht bei extremem Klima heraus, das auch bei Fokus auf Laubbaumarten an Kiefer festhält (Fig. 3.5e) und den Trichtermund verschmälert.

Ammer et al. (2018) formulieren für zentraleuropäische Wälder ökologische Schlüsselfragen, worunter auch die Analyse von Interaktionen zwischen Klimawandel, Landnutzungsgeschichte und Produktivität (Q23) fällt. Durch ein differenzierteres Wachstum, das subletale Störungen (Stress) durch angepasste Oberhöhenbonitäten berücksichtigt (Schmidt 2020; Schick et al. 2023; Schmidt et al. 2023), untersucht A1 eine standort- und klimasensitive Produktivität. Jedoch erzeugt die Integration weiterer Behandlungsgruppen (Z-Baum-Zahlen, Zielstärken, Ertragsniveaus) im *WaldPlaner* als black-box DSS intransparente Effektkopplungen (Muys et al. 2010). Zulasten der Transparenz kann jedoch ein komplexes und detailliertes Gesamt-Simulationsergebnis erzielt werden (Basis: Tab. 3.1, Fig. 3.6). Dieses kann von Anwendern jedoch nur durch aufwendige Sensitivitätsanalysen nachvollzogen werden. Dabei ist die von den ReKiEs-De Ensembles verwendete CMIP5-Generation der Klima-Regionalmodelle (Hübener et al. 2017; Warrach-Sagi et al. 2018) veraltet und eine erneute Simulation mithilfe auf Basis von CMIP6 wäre von Interesse.

Der Unsicherheitshorizont der Waldentwicklung fällt im von A1 untersuchten Raum (Waldbau- und Klima-Achse) asymmetrisch aus: Dies bedeutet, dass das hinsichtlich der Baumartenzusammensetzung relativ gemessene Delta der vier Szenarien auf der Waldbau-Achse deutlich weiter ausfällt (s. Tab. 3.2). Dies könnte ein Indiz sein, den Ensemble-Ansatz (Klima-Achse) zu erweitern. Zumindest wäre eine automatisierte Simulation mit allen sieben Regionalmodellen des Ensembles ratsam, in der die standortsensitive Leistungsschätzung durch *site indices* (Schmidt 2020; Schick et al. 2023) automatisiert per Algorithmus angepasst wird. Generell bedarf eine Prognose der Waldentwicklung unter Unsicherheit auf Basis von Klimamodellen einer verstärkten, nutzerseitigen Aufklärung (Hoogstra & Schanz 2009; Nagel et al. 2017). Im Klimawandel werden auch auf Bestandes- und Betriebs-Ebene Szenarioansätze – neben der in A1 getätigten Prognose – zur Planung und Schulung empfohlen (Puettmann 2021).

Die Wichtigkeit einer waldbaulichen Vielfalt als Beitrag zur Entscheidungs-Flexibilisierung und Ausnutzung des standörtlichen Potenzials wird in der Literatur betont (Ehwald 1949; von Detten & Faber 2013; Puettmann 2021). Hierdurch könne die Anpassungskapazität gegen einen Diversitätsverlust gestärkt werden. Jedoch wird auch gefordert, dass Waldbau-Richtlinien für Mischbestände klar operationalisiert sein müssen, ohne zu komplex für eine Anwendung zu sein (Pretzsch & Zenner 2017). Zur forstbetrieblichen Handhabbarkeit werden unter Einbuße von positiven Komplementaritätseffekten räumlich getrennte Bestandesmischungen in Erwägung gezogen (Pretzsch & Grote 2023, p. 383). Während der in A1 erfolgten Modellierung von Szenarien eine Schlüsselposition eingeräumt wird, empfehlen Pretzsch & Zenner (2017) ebenfalls Simulationen bei weniger optimistischen Annahmen zum Management (versäumte Pflege- und Erntemaßnahmen auf der Zeit-Achse). Dies trüge dem enormen Kostendruck von Forstbetrieben Rechnung (Schütz 2003; Lawrence 2017). Die Ergebnisse der BWI[4] (43 % der Waldfläche ohne Nutzung zwischen 2012 und 2022; BMEL 2024) und Umfrageergebnisse (Stockmann et al. 2024) bestätigen diese Notwendigkeit. Zudem berücksichtigt A1 keine Null- bzw. Kontrollvariante ohne Management, die aufzeigen könnte, welche Bestandesziele nur durch reguläre Mortalität erreicht würden (Oliver et al. 2016, p. 48). Es könnte im Wachstums- und Verjüngungsmodell des *WaldPlaners* auch eine natürliche Sukzession ohne forstliche Eingriffe beschrieben werden (Thom et al. 2017), die nur dichte- und altersbedingtes Ausscheiden berücksichtigt.

In A1 wird die Waldentwicklung entlang der Leitplanken eines präskriptiven Szenariotrichters simuliert, der sich aus der geforderten Dynamisierung der Standortkartierung (Spathelf et al. 2016; Riek et al. 2020) ergibt. Ein unsicherer Wasserhaushalt bei Standortsdrift wirkt sich schließlich auf die Baumartenwahl der Forsteinrichtung aus. In A1 werden WET's bei Simulationsstart in Abhängigkeit des Klimamodellsignals von 2071-2100 jedoch statisch ausgewählt und beibehalten. Dies vernachlässigt eine zu erwartende Nachsteuerung zwischen 2012-2062 (z.B. situative WET-Wahl nach unvorhergesehenen Events als sog. Eventualplanung) oder adaptive WET-Wahl zu einem späteren Zeitpunkt, wenn der Bestand zur Überführung ansteht (NLF 2012, p. 20). Darüber hinaus wird ein Management im Gegenstromprinzip mit einer hierarchischen, zentralen Steuerung, vernachlässigt. Dies könnte mittels Zielwaldmodell auf Ebene des Bundeslandes oder der Waldbauregion gegensteuern (Böckmann et al. 2019). Der sich ergebende Unsicherheitshorizont ist somit eine rein theoretische Größe. Durch nicht vollzogene Eingriffe ist es denkbar, dass sich der Trichtermund verschmälert, wohingegen sich die Unsicherheit durch situative Nachsteuerungen nach 2012 vergrößern oder verkleinern könnte. Dies ist auch vor dem Hintergrund sich wandelnder Waldumbau-Ziele zu betrachten, die sich nach Erdozain et al. (2024) in Europa seit dem 20. Jahrhundert in drei Perioden mehrfach geändert haben. Im Rahmen der Biodiversitätskrise rücken weniger Bäume und Bestände, sondern rückt mehr die Landschaft in den Fokus des Waldbaus (Puettmann et al. 2025), was die föderale, staatliche Forstplanung in ihren Zielen und Methoden komplizieren dürfte.

Die Input-Daten der BWI³ liefern keine Baumqualitäten, was den Wahrheitsgehalt von H1 limitiert (Duvemo & Lämås 2006). Demzufolge wird im Rahmen von Erntemaßnahmen, die bei Kiefer im *WaldPlaner* in Abhängigkeit der Zielstärkenanteile als Schirmschlag implementiert sind, ein auch potenziell qualitätsbefriedigender Nebenbestand (Nachwuchs) entfernt: Eine Baumentnahme wird anteilig als Ernte zielstarker Bäume *und* als Lichtung im Sinne einer starken Niederdurchforstung durchgeführt (VDFV 1902; Hansen & Nagel 2014, p. 28), was realitätsfern ist. Dieser pauschale Aushieb des Unterstandes (i.d.R. Laubbäume oder Fichten, *cf.* Kap. 4.3.2) kann den Waldumbau in der Prognose verlangsamen, womit der Trichtermund verschmälert würde.

Zur abschließenden Diskussion von H1 ist die Betrachtung von letalen Störungen als irreguläre Mortalität vonnöten.

7.2. Hypothese H2

„*Die Waldentwicklung Norddeutschlands*
wird bei der Kiefer unter Störungseinfluss derzeit überwiegend
von niedrigen Schweregraden bestimmt, deren Stärke
mit zunehmender Standortsgüte ansteigt.“

Nach zehnjährigem Erfassungszeitraum auf Basis des WSMP kommen niedrige Schweregrade am häufigsten vor und es zeigt sich ein regressiver Kurvenverlauf (Fig. 4.6). Es gibt die Tendenz, dass bei einer für Kiefernwachstum günstigen Wasser- und Nährstoffversorgung die

Schweregrade wieder zunehmen (Fig. 4.4), insbesondere wenn Kiefern im Bestand weniger als 25 % Mischungsanteil einnehmen. Somit kann der erste Halbsatz von H2 beibehalten werden, während zum Widerlegen des zweiten Halbsatzes keine ausreichende Evidenz gefunden wurde. Insgesamt finden sich die höchsten Schweregrade auf schlecht wasserversorgten Standorten.

Wie gezeigt, ist das WSMP für Aussagen über robuste Relationen von Schweregraden zueinander in seiner Datenqualität und -quantität für Kiefer ausreichend und kausale Schadfaktoren lassen sich identifizieren. Jedoch sind die für das vorliegende Ereigniszeitmodell verwendeten Daten informativ (Einfluss der Beobachtung auf die Meldung und *vice versa*), da umgestürzte Bäume den Zugang zu Forstwegen erschweren. Dies verringert die Datengüte. Außerdem sind die Einträge links- und intervallzensiert, indem die Wälder nicht seit Keimung oder Besitzerwerb kontinuierlich überwacht wurden (Schoneberg 2017, p. 81).

Der zweite Halbsatz von H2 fügt sich dabei in das von Malamud et al. (2005) angewandte *Frequenz-Areal-Potenzgesetz* ein, die die Feuer-Störungsregimes in den Vereinigten Staaten von Amerika charakterisierten. Kimmins (2004, p. 539) stellt für kanadische Verhältnisse fest, dass die wenigen Störungen hohen Schweregrades für einen großen (diskreten) Anteil der betroffenen Fläche je Störungsevent verantwortlich sind (White & Jentsch 2001, p. 411). Insgesamt machen niedrige Schweregrade den größeren Flächenanteil aus (Malamud et al. 2005), wie auch A2 belegt. Damit lässt sich die irreguläre Mortalität in Kiefernwäldern Nordwestdeutschlands als Störungsregime variablen Ausmaßes beschreiben (*mixed-severity disturbance regime*). In der dominierenden Frequenz niedriger Schweregrade ist es damit derzeit vergleichbar mit der Lückendynamik in Buchen-Urwäldern (Feldmann et al. 2018).

Gemessen an den ca. 40 Mio. m^3 Kiefernvorrat im Landesbesitz Nordwestdeutschlands (HE, NI, SH, ST; BMEL 2024) machen die amtlich-statistischen jährlichen Erntemengen (regulärer und irregulärer Einschlag) im Mittel von 2014-2024 etwa 5 % aus. Vom Gesamteinschlag machte der amtliche Schadholzeinschlag durchschnittlich etwa 13 % aus (Destatis 2024, 2025). Seidl & Senf (2024) bestätigen per Fernerkundung, dass in Europa 82 % der Kronendachöffnungen forstlich zu begründen sind. Damit ist der Schweregrad der anthropogenen und insbesondere natürlichen Störungen global als gering zu bezeichnen. Es wird angenommen, dass gelegentliche Störungen bei durchschnittlicher Standortproduktivität zu einem dynamischen Gleichgewicht und maximaler Artenvielfalt führen (Huston 1994 zit. n. Wohlgemuth et al. 2019, p. 40). Dementsprechend wird die positive Integration von natürlichen Störungen in die Waldbewirtschaftung als Schlüsselfrage gesehen (Q44; Ammer et al. 2018). Aszalós et al. (2022) zeigen für europäische Störungsregimes, dass insbesondere durch die residualen Nebenformen bei der Betriebsform des Dauerwaldes (Habitat- und Samenbäume, Überhälter) – eine besonders hohe strukturelle und funktionale Übereinstimmung zwischen natürlichen und anthropogenen Störungen besteht (Kimmins 2004, p. 535). Während im Rahmen eines integrierten Waldschutzes zeitnahe Sanitärhiebe von absterbenden Bäumen zum Walderhalt angrenzender Bestände beitragen, sollten spätere Kalamitätshiebe (*salvage logging*) von bereits abgestorbenem Totholz aus Biodiversitätsaspekten generell überdacht werden (Thorn et al. 2018; Knoke et al. 2021).

Der zweite Halbsatz von H2 liefert Indizien für die *Stress-Gradienten-Theorie* (Callaway & Walker 1997). Diese besagt, dass förderliche Wechselwirkungen zwischen Pflanzen auf ärmeren Standorten höher sein können, als auf fruchtbaren (Pretzsch 2019b, p. 342). Unter günstigen Wuchsbedingungen ergeben sich zwar eine höhere Bruttoproduktivität, jedoch generell auch mehr Konkurrenz und Stress. Dies kann dazu führen, dass aufgrund erhöhten Ausscheidens die Nettoproduktivität sinkt, wie am Beispiel von Kiefern-Buchen-Mischbeständen gezeigt wurde (Pretzsch et al. 2023). Die mit zunehmender Standortsgüte nur leicht steigenden Schweregrade könnten auch auf die insgesamt *sehr* ungünstige Witterung zwischen 2018 und 2020 zurückzuführen sein: Bei *de Martonne*-Indizes von ≤ 30 bzw. Jahresniederschlägen ≤ 600 mm m^{-2} kann die Konkurrenz reduziert sein (Pretzsch et al. 2020, 2023). Außerdem weisen Kiefern bei geringer Standortsgüte ein vermindertes Höhenwachstum auf, was sie gegenüber dem in A2 am häufigsten gefundenen Störungsfaktor Wind weniger anfällig macht (Grottian et al. 2024). Für heimische Laubbaumarten wurde hingegen gezeigt, dass eine höhere Basensättigung mit einer erhöhten Überlebenswahrscheinlichkeit einhergeht (Maringer et al. 2021). Hentschel et al. (2018) beobachteten, dass Kiefern hoher Bonität mehr von nadelfressenden Insekten befallen wurden und Oldiges (1959, zit. n. Assmann 1970, p. 432) berichtet, dass alkalische Böden die Überlebensrate dieser Insekten senken. Besonders durch Basendünger können *Heterobasidion*-Infektionen entweder verzögert (Piri 2000) oder dessen Hyphenwachstum beschleunigt werden, wenn der Baumzuwachs schwankt (Seibt 1964). Hinsichtlich einer Wasser-Überversorgung schwächen anaerobe Bedingungen die Feinwurzeln schnell und erhöhen deren Infektionsanfälligkeit (Repo et al. 2020). Schon ein Sauerstoffgehalt im Boden unterhalb von 10 % gilt als kritischer Schwellenwert für einen deutlichen Rückgang des Spross- und Wurzelwachstums bei Kiefern (Leyton & Rousseau 1957).

Gemäß der *Theorie zur Balance zwischen Wachstum und Verteidigung* (Herms & Mattson 1992) werden bei geringer bis mittlerer Standortgüte von Pflanzen maximale Investitionen in sekundäre Metabolite und die Abwehr getätigt (Pretzsch 2019b, p. 644). Dies stellt Kiefern auf guten Standorten prinzipiell schlechter. Grundsätzlich kann davon ausgegangen werden, dass mehrere Baumarten je Bestand die Produktivität und den Walderhalt ökologisch und ökonomisch sichern (Markowitz 1991; Yachi & Loreau 1999). Jedoch können artenreiche und innige Bestandesmischungsformen funktionell redundant sein (Searle et al. 2022; Shovon et al. 2024), wie Fig. 4.5 beschreibt.

Eine erhöhte Mortalität bei guter Nährstoff- und Wasserversorgung, insbesondere bei hoher Baumartenvielfalt, könnte in A2 durch ungerichtete Korrelationen oder gerichtete Kausalzusammenhänge (Schurz 2014, p. 146) aufgezeigt werden. Zur Falsifikation des zweiten Halbsatzes von H2 müssen die Einträge des WSMP trotz des ausgeprägten West-Ost-Gradienten an Klima- und Standortverhältnissen – insbesondere im Hinblick auf Mortalität unter Extremstandortbedingungen – weiter ergänzt werden. Auch vor dem Hintergrund des Gesetzes der Großen Zahl, das Erkenntnisse generell absichert (Schurz 2014, p. 112), liegen nicht ausreichend Meldungen bei sehr hohen und niedrigen Bonitäten, Bestockungsgraden oder trockenen und hydromorphen Böden vor. Ebenfalls fehlen Meldungen zu Mortalitätsereignissen infolge von periodischem Insektennadelfraß.

7.3. Hypothese H3

„*Die Waldentwicklung Norddeutschlands*
wird durch diskrete, hohe Schweregrade beschleunigt,
wenn sie mit diffusen, niedrigen Schweregraden kontrastiert werden."

In H3 wird unterstellt, dass bei identischen Hazardraten, wenn irreguläre Mortalität ausschließlich in Form von niedrigen *oder* hohen Schweregraden vorkommt, letztere eine Beschleunigung der Waldentwicklung bewirken. Wie aus Fig. 5.7 hervorgeht, tendiert eine bestandesweise (diskrete) Mortalität dazu, die Waldentwicklung – gemessen an ihrer Waldumbaurate – positiv zu beeinflussen. Jedoch sind hohe Schweregrade nicht immer für eine gesteigerte Waldumbaurate verantwortlich. In den westlichen Bundesländern beschleunigt zwischenzeitlich auch einzelbaumweise Mortalität den Waldumbau. Es fehlt somit an ausreichender Evidenz, um H3 zu verwerfen.

Aufgrund bestimmter Modellvorgaben konnten die gegensätzlichen Mortalitätsmuster über den Prognosezeitraum hinweg nicht durchgehend zufriedenstellend simuliert werden. In A3 werden hohe Schweregrade (diskrete, bestandesweise Mortalität) in ihrem Eintreten ab etwa 2032 weniger, weil sie in ihrem Eintreten modellbedingt restringiert werden (Fig. 5.5b, c & Fig. 5.7d). Dies liegt zum einen am Alter: Irreguläre Mortalität kann nach Brandl et al. (2020) erst ab 20 Jahren Alter auftreten, wo dann in mittlerem Alter die steilste Abnahme auf der Überlebensmodellkurve zu beobachten ist (Fig. 5.3a). Zum anderen wirken die zum Modellantrieb für Kiefer verwendeten Klimagrößen (maximale Temperatur des wärmsten Monats, Niederschlagssumme des wärmsten Quartals) sehr spezifisch, weshalb eher an denselben Trakten junge Bestände erneut flächig absterben, als anderenorts. Wenn Störungen wiederholt an denselben Trakten auftreten, wird die Waldentwicklung bei hohen Schweregraden im *WaldPlaner* verzögert.

Die Anwendung der Überlebensmodelle von Brandl et al. (2020) ist in A3 so pessimistisch, weil ein Versatz zu den originalen Parametrisierungsdaten besteht (Fig. S5.8). Insbesondere im extremen Regionalmodell HadGEM wird die Waldentwicklung bei hohen Schweregraden zu Anfang der Simulationsperiode beschleunigt. Vor dem Hintergrund des von Puettmann (2021) eingeführten Begriffs der störungsbezogenen Konnektivität erscheint der konservative Modellantrieb für die ökologisch-funktional redundanten Kiefernbestände (vgl. Abb. 1.1), die im Kontext des Zweiten Weltkrieges entstanden, jedoch als angemessen: Solche ähnlich strukturierten Bestände begünstigen die rasche Ausbreitung von Agenten, etwa bei gleichzeitigem Einwachsen in windwurfgefährdete Bestandeshöhen (Wohlgemuth et al. 2019, p. 168) oder bei Massenvermehrungen biotischer Schaderreger (Habermann 2017).

Dahingegen werden niedrige Schweregrade (diffuse, baumweise Mortalität) in ihrem Auftreten in Summe nicht nachweisbar restringiert. Im *WaldPlaner* kann die Waldentwicklung bei niedrigen Schweregraden jedoch verzögert sein (Fig. 5.7c, d). Dies kann zum einen in geänderten Eingriffsfolgen im Vergleich zur ungestörten Waldentwicklung liegen: Weil die großen Schadholzmengen (Fig. 5.5b) die Eingriffshöchstmengen zu einem hohen Anteil ausfüllen, können planmäßige Nutzungen – sofern die Zielgrundfläche nicht unterschritten wird – nur in verringertem Maße stattfinden. Zum anderen kann ein Ersatz von Misch- durch Reinbestände die Waldumbaurate verringern. Dies ist bei der szenariobedingten Maximierung des Kiefernanteils durch WET mit bis zu 90 % Kiefer in NI und 80 % Kiefer in MV möglich. Die zur Beantwortung von H3 dienende Waldumbaurate berechnet sich nur aus monospezifischen Reinbeständen je BWI-Trakt, wo mehr als 90 % der bis zu vier Traktecken von einer Baumart gestellt werden. In 2012 stehen den betonten Kiefern-Reinbeständen im Osten mit 708 Tsd. ha im Westen nur 133 Tsd. ha gegenüber (*cf.* Fig. 3.9). Dieses geringere Flächenpotenzial könnte das statistisch weniger gut abgesicherte, uneinheitliche Bild in Fig. 5.7c erklären.

Weil zur Waldentwicklung nicht nur der Baumartenwechsel im Sinne des Waldumbaus gehört, sondern auch eine generelle Walderneuerung durch Generationenwechsel zählt, ist die Waldumbaurate nicht als einziges Kriterium zur Falsifizierung von H3 geeignet. Um Aussagen auf einer breiteren Basis treffen zu können, wäre ergänzend eine Betrachtung verschiedener Altersmaße oder des Zielerreichungsgrades (Formel 3.2 aus A1) zielführender gewesen, in dessen Berechnung alle BWI-Ecken eingehen.

Beide Schweregrade der irregulären Mortalität verringern das mittlere Baumalter des Oberstandes (Fig. 5.4c), wobei das mittlere Alter der irregulär gestorbenen Bäume zwischen hohen und niedrigen Schweregraden variiert (Fig. 5.5c, untere Kurvenschar). Weil das Baumalter bei hohen Schweregraden mehr sinkt, kann dies als H3 bestätigendes Indiz gesehen werden. Hier vollzieht sich die Waldentwicklung in Form eines Generationenwechsels schneller. Jedoch ließe sich das Baumalter in der Realität bei einer zeitlich früheren, aber mehr gestreckten Ernte – die auch Schattbaumarten beteiligt – stabilisieren. Dies bestätigend, stieg das mittlere Kiefern-Bestandesalter in A1 von 73 Jahren in 2012 auf 79 Jahre in 2062 (Szenario d). In der Realität zeigt sich zwischen 2012 und 2022 ein Anstieg des flächengewogenen, mittleren Baumalters von Kiefern von bundesweit 77 auf 85 Jahre (BMEL 2024). Es ergibt sich somit ein durch Störungen und Ernte geringerer Abgang als Zugang (Einwuchs).

Weil forstliche Maßnahmen als anthropogene Störungen zu werten sind, sei in der Realität auf eine mögliche Verlangsamung der Waldentwicklung bei investiv (mit Pflanzungen oder Saat) geplantem Waldumbau verwiesen. Zur Finanzierung eines Baumartenwechsels sind vor allem positive Deckungsbeiträge aus dem Holzverkauf notwendig (Höhl et al. 2020; Erdozain et al. 2024). Wenn diese nicht erzielt oder eine finanzielle Förderung (volkswirtschaftlich) nicht attraktiv gestaltet werden kann, droht ein Aufschub der Maßnahme (Bolte et al. 2021; Braunschweiger et al. 2024). Der monetäre Anpassungsaufwand (*adaptation effort*) wird insbesondere bei einer Verjüngung von Kiefernbeständen hin zu Beständen mit Schattbaumarten als hoch beziffert (Albert et al. 2017) und kann träge, verzögerte Entwicklungsprozesse zur Folge haben (Schelhaas et al. 2015; Puettmann & Bauhus 2023).

Weitere Modellartefakte, die die Waldentwicklung im *WaldPlaner* bei hohen Schweregraden beschleunigen, ergeben sich durch dessen Verjüngungsmodul: Durch die regelkonformen Prozessschritte stellt sich auf Freiflächen (Bestandesschäden) immer ein Verjüngungs- und Kulturerfolg ein. Dieser hängt in der Realität jedoch maßgeblich von der Wasserverfügbarkeit ab (Paquette et al. 2006; Huth et al. 2017, p. 61; Holmström et al. 2019). Scherrer et al. (2023) untersuchen den Zusammenhang von Störungen und Baumartenzusammensetzungen und stellen für die Schweiz fest, dass insbesondere lokale Standortbedingungen (Umweltfaktoren) und weniger ursprüngliche Bestandestypen (Samenpotenzial) die Waldentwicklung beeinflussen (s.a. Moser et al. 2015). Des Weiteren ist die Zuordnung irregulärer Mortalität ausschließlich zum Hauptbestand und regulärer Mortalität nur zum Unterstand als modellhafte Vereinfachung zu werten (Das et al. 2016).

Von herausragendem Interesse sind die Effekte verschiedener Mortalitätsmuster auf die Funktionsfähigkeit von Ökosystemen (Q15; Ammer et al. 2018). A3 untersucht die Zuwachskompensation durch den verbleibenden Bestand bei niedrigen Schweregraden als diffuse Einzelbaumschäden. Je nach Baumartengruppe im Hauptbestand wird im *WaldPlaner* Verjüngung pauschal erst unterhalb eines natürlichen Bestockungsgrades von 0,3 - 0,5 initiiert (gepflanzt). Dadurch lassen sich Kompensationseffekte gut erfassen, aber die Waldentwicklung nur vereinfacht und potenziell verzögert abbilden. In der Realität kann die Verjüngung je nach Baumart, Flächenausprägung und Nebenbestandsqualität auch eher erfolgen.

Diese Modellartefakte bedeuten, dass H3 nicht abschließend falsifiziert werden kann. In der Realität ist anzunehmen, dass flächige Störungsereignisse (hohe Schweregrade) als Katalysator von Landschaftsveränderungen dienen (Wohlgemuth et al. 2019, pp. 82, 135). Erwartet werden zusätzliche positive Rückkopplungen (Wilson & Agnew 1992; Sebald et al. 2021), da eine Zunahme von Laubbäumen mit geringeren Störungen, erhöhter Evapotranspiration und durch klimaregulierende Effekte mit verbessertem Verjüngungserfolg einhergehen kann (Thom et al. 2017). Dabei müssen Sukzessionsstadien von Wäldern durch Störungen in ihrer Entwicklung nicht zurückgesetzt, sondern können sogar beschleunigt werden. Dies ist der Fall, wenn z.B. risikostabile Hartlaubholzarten der Gattungen *Quercus*, *Tilia*, *Carpinus* und *Acer* selektive Störungsevents überleben (Abrams & Scott 1989; Brandl et al. 2020; Fuchs et al. 2021; Barrere et al. 2023). Die in A3 beabsichtigte Steigerung des Kiefernanteils, wie sie seit den 2000er Jahren im Staatswald zu beobachten ist (Abb. 1.1a), widerspricht der zu erwartenden, übergeordneten Zunahme von Laubwald (BMEL 2024). Auch ohne forstliche Maßnahmen hat sich in Simulationen eine Sukzession unter Störungseinfluss hin zu Laubbäumen bewiesen, wie Thom et al. (2017) für den österreichischen Kalkalpen-Nationalpark zeigen.

Für Kiefer werden mit steigender Bestandesdichte erhöhte Mortalitätsrisiken assoziiert, die die Waldentwicklung katalysieren. Neben einer allgemein höheren Lichtkonkurrenz tragen dazu bei: Ein höheres Trockenstressrisiko (Sturrock et al. 2011; Archambeau et al. 2020; Bose et al. 2024), höhere oberirdische Interzeption (Loesch 2023, p. 24; Spathelf et al. 2025), höhere unterirdische Wasserkonkurrenz (Sohn et al. 2016; Kulha et al. 2023; Socha et al. 2023), niedrigere kritische Windgeschwindigkeiten (Gardiner 2021; Grottian et al. 2024) durch hohe h/d-Werte (Barrere et al. 2023), verringerte Vitalität durch kürzere Kronen (Aldea et al. 2024), erleichterte Sporen-Ausbreitung bei Windstille (Altenkirch et al. 2002, p. 63; Sturrock et al. 2011), viele Wurzelkontakte zur Infektionsübertragung (Tainter & Baker 1996, p. 395), sowie eine höhere Insekten-Befallsgefahr (Altenkirch et al. 2002, p. 187; Jaime et al. 2019).

Zur Diskussion von H3 soll auch auf Hazardraten eingegangen werden, die sich bei Brandl et al. (2020) auf bestockte Flächen und nicht Bäume direkt beziehen. Während sich die vorgegebene Überlebenswahrscheinlichkeit bei konstanten 0,2 ha Bestandesfläche (hoher Schweregrad) durch Monte-Carlo-Simulationen in Wiederholungen stochastisch unverzerrt einstellt, ist die Baumfläche (niedriger Schweregrad) nicht konstant. Diese biologisch als Kronenschirmfläche zu interpretierende Größe beträgt bei A3 im Mittel 13,7 m^2 je Baum. Eine Aussage zur Waldentwicklung unter Störungseinfluss kann einen Bias aufweisen, wenn durch die bedingte Zufallsauswahl vermehrt Bäume mit einer gerichtet größeren oder kleineren Fläche ausgewählt werden. Jedoch sollte dies bei über 11 Mio. Bäumen je Wiederholung und je 5-Jahres-Simulationsschritt vernachlässigbar sein. Die flächenbezogene Einzelbaumauswahl, wie sie durch die BWI-Methodik mit normierten Standflächen ermöglicht wird und auf dem BHD beruht, stellt einen innovativen Ansatz dar. Sie eröffnet erstmals die Anwendung von Überlebensmodellen unterhalb der Bestandesebene.

Zur umfassenden Diskussion muss neben Ursache-Wirkungs-Zusammenhängen zur Erklärung und Prognose auch ein Ziel-Mittel-System zur Problemlösung betrachtet werden (Kornmeier 2007, p. 25).

7.4. Hypothese H4

„*Die Waldentwicklung Norddeutschlands*
lässt sich auf Grundlage der landschaftsräumlichen Vielfalt
durch multikriterielle Ansätze bedarfsbezogen
für den Waldumbau priorisieren.“

Die bloße Berücksichtigung einzelner Kriterien, wie nur die Trockenstresstoleranz-Bewertung mittels SWB zur Lenkung der Waldentwicklung, differenziert die Bestandesreihung nur unzureichend (z.B. Fig. 6.6). Auf Grundlage der landschaftsräumlichen Vielfalt werden bei Gleichgewichtung der Indikatoren Trockenstressrisiko, α- und γ-Diversität (sowie der Bestandeshöhe als Altersmaß) etwa 18,5 % (Fig. 6.9) als prioritär umbauwürdig identifiziert. Demzufolge kann H4 beibehalten werden.

Durch das Beibehalten von H4 wird ein Beitrag geleistet, der im Rahmen eines gesamtbetrieblichen bzw. landesweiten Waldbaukonzeptes den Waldumbaubedarf flexibel ermittelt. Bisherige Ansätze identifizierten zunächst optimale Strategien für einzelne Bestände unter Unsicherheit (Knoke et al. 2020). Durch die Einbeziehung verschiedener, quelloffener, räumlich expliziter Datensätze besteht ein möglicher Effizienzgewinn im Rahmen der *Industrie 4.0 in der Forstwirtschaft* (Knoke et al. 2023).

Derzeit erfolgt die Ernteplanung der norddeutschen Kiefernbestände im Staatswald eigentumsgetrennt mittels einer baumweisen Zielstärke- oder bestandesweisen Hiebsreifedefinition. Fig. 5.5c (obere Kurvenschar) verdeutlicht, dass Kiefern infolge konsequenter Auslesedurchforstungen im Mittel nach 70-90 Jahren reif sein können. Im Untersuchungsgebiet liegt das durchschnittliche Baumalter bei 86 Jahren, was bei monokriterieller Definition des Überführungsstarts (nur anhand eines Zieldurchmessers) für viele Bestände zeitnahe und zeitgleiche Erntemaßnahmen bedeuten würde.

Für eine multikriterielle Planung ist die Überwindung von Eigentumsgrenzen bei Multiskalenansätzen eine große Herausforderung (Puettmann 2021; Knoke et al. 2023). Für den niedersächsischen Landeswald wird beispielsweise ausgeführt: „Die Forsteinrichtung beschränkt sich im Einzelbestand auf Zielaussagen zu Nutzungs- und Produktionsplanungen sowie zu Schutz- und Erholungsfunktionen. Als Grundlage stehen dafür die Ergebnisse der Betriebsinventur [...] sowie die Analyse der einzelbestandsweisen Ansprache (z.B. Dringlichkeit) zur Verfügung“ (NLF 2012, p. 19). Vor diesem Hintergrund kann A4 als flächendeckende Operationalisierung der Schutzfunktion (Artenvielfalt) gesehen werden, die durch die BWI³ abgeleitete Baumartenerkennung von Blickensdörfer et al. (2022, 2024) mit hinreichender Genauigkeit geschieht.

Im Planungskontext wird dementsprechend appelliert, den Fokus vom Einzelbaum und -bestand hin zum Ökosystem und zur Landschaft auszurichten (Puettmann et al. 2025). Dahingegen soll zur Förderung der

Resistenz gegenüber Stress und Störungen die Vitalität des Einzelbaumes ins Zentrum von Waldbauverfahren gestellt werden (Bauhus 2022).

Auch wenn multikriterielle DSS, die das Thema Unsicherheit in der Forsteinrichtung adressieren, schon lange gefordert werden (Butler 2009), stieg die forstbetriebliche Nachfrage erst durch höhere kartenmäßige Auflösungen (von Detten & Faber 2013). Insbesondere die Bewertung von dynamischen Entscheidungskriterien ist im forstlichen Management von zentraler Bedeutung (Hildebrandt & Knoke 2011). Hoogstra & Schanz (2009) untersuchen, auf welche Art Forstplanungspersonal eine unsichere Zukunft antizipiert. Sie beschreiben, dass dazu tendiert wird, Unsicherheit durch die „Doktrin der Langfristigkeit“ (vgl. Begriff der „späten Nachwelt“ Kap. 1.2.2) – also pragmatische Entscheidungen für mehrere Jahrzehnte – zu überwinden und die Zukunft als nicht beeinflussbar und zu statisch zu sehen. Dabei betonen von Detten & Faber (2013), dass dies zu einer Reduktion von Lösungswegen führen kann, die risikobehaftet ist, weil sie den Entscheidungsraum unnötig begrenze.

Gegenüber dem sich beschleunigenden Klimawandel sind Zeitverzögerungen im Entscheidungsprozess und in Ökosystemeffekten kritisch (Puettmann & Bauhus 2023). Im Rahmen einer Triage bei hohen Waldumbauflächen und steigender Katastrophengefahr (Sturrock et al. 2011) können die in A4 identifizierten Bestände auch für weitere Fragestellungen ausgewählt werden: Um schützenswerte Biozönosen zu bewahren (Horn et al. 2024), angepasstes Vermehrungsgut einzubringen (Chakraborty et al. 2024; Thorne et al. 2025), oder fachliche und finanzielle Ressourcen durch Förderaktivitäten zu konzentrieren.

Der flexible Gewichtungsansatz von H4, der die Waldumbaufläche bemisst, fügt sich in die *Realoptionentheorie* (Trigeorgis & Reuer 2017) ein. Hierbei wird angenommen, dass sich Unsicherheit auf der Zeit-Achse auflöst und neue Informationen verfügbar werden. Die durch die Flexibilität gebotenen Alternativen können zum Vorteil des Inhabers der realen Optionen ausgeübt werden (Hildebrandt & Knoke 2011). Im Sinne eines DSS lässt sich die Waldumbau-Priorisierung als Beitrag zur Investitionsflexibilität (z. B. Lenkung von Ressourcen oder Fördermitteln in welche Bestände) diskutieren.

Methodisch lässt sich der Ansatz in A4 durch weitere Kriterien erweitern, wozu z.B. das die Kühlung antreibende Albedo (Sturrock et al. 2011; Spathelf et al. 2025) oder die Erholung (Hampicke & Schäfer 2021; Michler & Aschenbrand 2024) zählen können.

Während zunächst vor allem Investitionsentscheidungen Gegenstand der Forschung waren, verfolgt A4 den Ansatz einer Triage (Sturrock et al. 2011; Thorne et al. 2025). Weil das subjektive Identifikationsvermögen des Forstpersonals mit dem Planungshorizont in der Regel bei unter 15 Jahren liegt (Hoogstra & Schanz 2009), ermöglicht A4 eine unmittelbare Entscheidungsgrundlage, wo Waldumbau prioritär angegangen werden sollte. Vor der Arbeit von Schirrmeister et al. (2023), konnte in Verbindung mit weiteren Modellen (Overbeck et al. 2011; Blickensdörfer et al. 2024) in Niedersachsen keine flächendeckende Wasser- und Nährstoffansprache als Grundlage einer Risikobewertung vorgenommen werden.

Zur abschließenden Diskussion von H4 wäre eine Validierung im Gelände notwendig, die Erfordernis und Bestandesreihung als Abweichung zu einer monokriteriellen Reifedefinition überprüft.

7.5. Synthese zum Waldumbau

Waldbausysteme und -Richtlinien

Die vorliegende Arbeit untersucht in A1-4 die Waldentwicklung unter überwiegender Einbeziehung eines aktiven Waldumbaus (mit anthropogenen Störungen sowie regulärer und irregulärer Mortalität) hin zu naturnahen, ungleichmäßigen und ungleichaltrigen Beständen. Diese strategisch-normative Zielstellung der Staatsforstverwaltungen wurde im *WaldPlaner* emuliert. In Behandlungsgruppen wurden verschiedene Durchforstungsintensitäten operationalisiert und für die Gesamtwaldfläche im Untersuchungsgebiet simuliert. Die Vielfalt der norddeutschen Kiefern-Waldbausysteme, Landschaftsräume und Standorte wurde in A1-4 diskutiert und soll in einer Synthese gebündelt werden.

Waldbausysteme sind in Waldbaukonzepten verschriftlicht. Im Folgenden wird der Begriff Richtlinie verwendet, um ein Dokument selber zu bezeichnen. Die Bestandteile der im *WaldPlaner* berücksichtigten Waldbau-Richtlinien der norddeutschen Staatsforsten (Landes- und Bundesforsten) wurden für die Baumart Kiefer in Tab. 7.1 begrifflich untersucht. Sie streben alle dauerhaft ungleichaltrige und -mäßige Waldstrukturen durch naturnahe bzw. ökologische Waldbauverfahren an, die es im Altersklassenwald weniger oft gibt. Die einbezogenen Waldbau-Richtlinien der norddeutschen Staatsforstverwaltungen (ohne Forstplanungs-Richtlinien) sind diesem Kapitel angefügt (s. Kap. 7.7). Besondere Bedeutung haben diese Richtlinien, weil sie im Rahmen der finanziellen und fachlichen Forstförderung durch ihre Vorbildwirkung indirekt auch auf den Nichtstaatswald angewendet werden.

Die Waldbau-Richtlinien zwischen den Staatsforstverwaltungen scheinen zwar auf den ersten Blick recht gleich, aber im Einzelnen werden insbesondere bei den Verjüngungsformen Unterschiede deutlich (*cf.* Tab. 7.1 & Abb. 1.3). Dabei werden wenig Operationalisierungen zu Waldbautechniken gegeben. Einige Bundesländer (BB, MV) arbeiten nur für Wertholz (Nebenbestand) mit baumweisen Zielstärkedefinitionen und ernten den restlichen (Haupt-) Bestand nach einer bestandesweisen Hiebsreifedefinition (s. Kap. 8.1.1). In den übrigen Bundesländern gelten für alle Bestandesglieder baumweise Zieldurchmesser. Des Weiteren werden z.B. bei den Bundesforsten gleichzeitig Zielstärkennutzungen sowie Femel- bzw. Lochhiebe angestrebt, die sich bei strikter Anwendung widersprechen können. Im Dauerwald erweist sich zudem „bis heute eine dauerhafte und intensive Strukturierung von Beständen mit Lichtbaumarten häufig als problematisch“ (Wagner & Huth 2010, p. 16). Die Zunahme der Kiefernfläche (vgl. Abb. 1.1a; NMELV 2024, p. 33) führt zu der Frage, ob hier eine bestockungsgleiche Wiederaufforstung (bloße Walderneuerung) oder ein Waldumbau hin zu naturnahen Beständen zugrunde liegt.

Tab. 7.1 Synthese begrifflicher Nennungen ausgewählter Bestandteile eines "Waldbaus auf ökologischer Grundlage"(Wagner & Huth 2010, p. 13) in den derzeitigen staatlichen Waldbau-Verwaltungsvorschriften (Waldbau-Rahmen- und Kiefern-Richtlinien, Stand 04/ 2025) Norddeutschlands, die in A1-4 emuliert wurden (✓: ja, ✗: nein, ( ): nur implizite Nennung; BB: Brandenburg, HE: Hessen, MV: Mecklenburg-Vorpommern, NI: Niedersachsen, SH: Schleswig-Holstein, ST: Sachsen-Anhalt, BF: Bundesforst)

Begriff	**BB**	**HE**	**MV**	**NI**	**SH**	**ST**	**BF**
„Dauerwald"	✗	✓	(✓)	✗	✗	✓	✓
„dauerwaldartig"	✓	✓	✓	✗	✓	✗	✓
Bewirtschaftung: **„naturnah"**	✓	✓	✓	✓	✓	✓	✓
Bewirtschaftung: **„ökologisch"**	✓	✓	✓	✓	✗	(✗)[1]	✓
„ungleichaltrig"	(✗)	✓	✓	✓	✓	✓	✓
„Schirmschlag"	✓	(✓)	(✓)	(✓)	(✓)	(✓)	(✓)
„ungleichmäßig"	(✓)	(✓)	✓	(✓)	(✓)	✗	(✓)
„Zielstärkennutzung"	(✗)	✓	(✗)	✓	✓	✓	✓
„Femel-/ Loch-/ Plenterhieb"	✓	(✓)	✗	✗	(✗)	✗	✓
kein **„Kahl-/ Saumhieb"**	✗	(✓)	(✓)	(✓)	(✓)	(✗)	(✓)

[1]: „ökogerecht"

In diesem Zusammenhang sei am Beispiel Niedersachsens erwähnt, dass im Zuge der bestandesweisen Forsteinrichtung nur geplant wird, ob und in welchem Ausmaß eine Ernte stattfindet: „Damit die Freiheit des Vollzuges gewährleistet bleibt, sind i.d.R. keine Richtlinien zur waldbautechnischen Realisierung der Planung vorzugeben" (NLF 2012, p. 19). Jedoch ist „der Vorbestand hinsichtlich [einer] Überhalts- und Verjüngungsmöglichkeit [...] zu berücksichtigen"(NLF 2012, p. 20). Damit könnte ein Widerspruch zwischen den Zielen (Tab. 7.1) und den praktizierten Verjüngungsverfahren bestehen. Hier wird u.U. die Kombination von Schirmschlag- und Samenbaumverfahren des Altersklassenwaldes gewählt, um einer großen Organisationsstruktur und einem hohen Kostendruck effektiv zu begegnen (Schütz 2003; Lawrence 2017).

Dem Aufbaubetrieb kann ein Abbaubetrieb folgen (Schmitz 2019). Dieser Wechsel wird durch Störungen und die Baumdemografie (vgl. Kap. 5.5, *sensu* Senf et al. 2021) katalysiert. Dies widerspricht dem Prinzip einer statischen, vorratsnachhaltigen Forstwirtschaft. Speidel

(1984, p. 44) gibt jedoch auch eine dynamische Definition, wenn die Fortdauer einer zuwachsnachhaltigen Leistung angenommen wird.

Die Überführung von Altersklassen- in Dauerwälder impliziert eine zeitlich ausgedehnte Vorratsreduzierung. Eine globale Vorratsabsenkung unter Störungseinfluss und bei rechtsschiefer Baumdemografie wurde in A3 gezeigt. Diese planmäßige Absenkung im Rahmen der Waldbausysteme sowie *un*planmäßige Verstärkung durch Störungsereignisse muss vor allem in die national kodifizierten Ziele des *Land Use, Land Use Change and Forestry* (LULUCF)-Sektors Eingang finden (Rock et al. 2025).

Ökophysiologische Anbaueignung

Forsteinrichtung und forstliches Versuchswesen entscheiden gemeinsam über die WET-Gestaltung und Anbaueignung von Baumarten. Wie in Kap. 1.2.1 erwähnt, bestehen beim Waldumbau von Kiefernreinbeständen (durch Naturverjüngung) hin zu Kiefernmischbeständen verschiedene Sichtweisen auf ihre Anbaueignung. Diese Transferlücke wurde in A1-4 z.T. diskutiert und wird in einer Synthese gegenübergestellt.

Kiefern nehmen im Klimawandel einen ökotonalen Status ein (Bartsch & Röhrig 2016, p. 7; Archambeau et al. 2020) und zählen als intermediäre Baumart nicht eindeutig zu den Gewinnern oder Verlierern (Spathelf & Ammer 2015). Tab. 7.2 stellt Literaturquellen zu ihrer ökophysiologischen Anbaueignung zusammen. Kiefern werden insbesondere von trocken-warmer Witterung geschwächt (Hauck et al. 2025), wie sie im Spätsommer auftritt. Hier besteht eine zeitliche Koinzidenz mit der Hauptwachstumsphase (Taeger et al. 2013; Haberstroh et al. 2022). Kiefern weisen ein isohydrisches Verhalten auf und regulieren Wassermangel durch schnellen Stomataschluss, was die photosynthetische C-Aufnahme reduziert. Dies führt zwar nicht zu Xylem-Kavitationen, aber zu einem metabolischen Mangel, da die Reserven an Kohlehydraten erschöpfen (Bose et al. 2024). Zudem werden die Abwehrkräfte für biotische Störungen herabgesetzt (Galiano et al. 2011).

Tab. 7.2 Synthese ausgewählter referierter Beiträge aus A1-4 zur ökophysiologischen Anbaueignung (Grundtenor) der Wald-Kiefer unter Klimawandel in Deutschland (Europa) ab 2000 (+: positiver, ±: neutraler, -: negativer Tenor).

Autoren	Methodik	Tenor
Mason & Alía (2000); Taeger et al. (2013); Przybylski et al. (2021); Chakraborty et al. (2024)	Genetik/ Modellierte Produktivität, Umfrage	+
Brichta et al. (2023)	Review	+
Niinemets & Valladares (2006); Lévesque et al. (2014); Walentowski et al. (2017)	Trockenstresstoleranz	±+
Albert et al. (2018); Schmidt (2020)	Simulation/ Produktivitätsmodell	±
Matías & Jump (2012); Bose et al. (2024)	Review	±
Bose et al. (2020b)	Semi-offenes Experiment	±
Gardiner (2021); Zeppenfeld et al. (2023)	Windwurf-Review/ -Modell	±
Michelot et al. (2012); Bose et al. (2020a); Diers et al. (2022, 2024); Herrero et al. (2023); Kašpar et al. (2024); Ogana et al. (2024)	Offenes Experiment, Bohrkerne	±-
Rigling et al. (2013); Pajtík et al. (2022); Barrere et al. (2023); Martinez del Castillo et al. (2024)	Offenes Experiment, Inventur	±-
Rehschuh & Rühr (2021); Hauck et al. (2025)	Geschlossenes Experiment, Blattpotenziale, Photosyntheserate	-
Aguadé et al. (2015); Kunert (2020); Haberstroh et al. (2022)	Offenes Experiment, Blattpotenziale	-
Reich & Oleksyn (2008); Walentowski et al. (2017); Mette et al. (2021)	Bioklimatische Hülle/ Zwillingsregionen, Transplantationsexperiment	-

Insgesamt fällt die Beurteilung der Anbaueignung von Kiefer im Hinblick auf ihr generelles Potenzial (Genetik, Morphologie, Anpassungsfähigkeit) positiv aus (Tab. 7.2). Untersuchungen zu physiologischen Einzelaspekten deuten jedoch auf eine begrenzte Eignung hin: Die Beurteilung fällt negativer aus, wenn neben Klimamittelwerten auch saisonale Witterungsbedingungen und insbesondere Temperaturanomalien berücksichtigt werden. Die Synopse verschiedener Bundesländerverfahren von Baumbach et al. (2024) zur Eignungsbeurteilung zeigt, dass man derzeit praxisseitig neben Klima- *kaum* Witterungsvariablen verwendet. Es überwiegen expertenbasierte Methoden und forschungsseitig identifizierte Witterungsparameter (z.B. Dürreindizes) finden in der Implementierung noch keine Berücksichtigung.

Weil die tiefe Unsicherheit der Kiefernbewirtschaftung im Klimawandel nicht vollständig erfasst werden kann, wird ein kombinierter Ansatz empfohlen, der gleichzeitig auf die Risikosenkung und die Resilienzsteigerung abzielt (Seidl 2014; Wohlgemuth et al. 2019, p. 350). In diesem Zusammenhang werten del Río et al. (2022, 2024) Studien aus, die sich für Kiefer mit waldbaulichen Regelungsmöglichkeiten am südlichen Rand ihres Verbreitungsgebietes befassen. Darin werden verschiedene Anpassungsstrategien beschrieben, die in sieben Maßnahmenbündel gegliedert werden:

1. Förderung von Naturverjüngung
2. Förderung adaptiver Evolution
3. Erhöhung der Baumartenvielfalt
4. Unterstützung von Artersetzungsprozessen in Tieflagen
5. Minderung von dichte- und strukturbedingtem Wasserstress
6. Erhöhung der Stabilität gegenüber Windwurf
7. Kontrolle von Schadorganismen (saubere Waldwirtschaft)

7.6. Kernbotschaften

→ H1 wird verworfen, weil die Klimamodellwahl weniger Auswirkungen auf das Ergebnis der Waldentwicklungssimulation hat, als die Vielfalt der Waldentwicklungstypen

→ H2 wird aufgrund der häufigeren, niedrigen Schweregrade im Störungsregime der Kiefer beibehalten. Es deutet sich an, dass eine bessere Standortsgüte zu steigenden Schweregraden führt, auch wenn die höchsten Schweregrade bei widrigen Umweltbedingungen auftreten

→ H3 wird beibehalten, auch wenn niedrige Schweregrade ebenfalls zu einer Beschleunigung der Waldentwicklung führen können.

→ H4 wird beibehalten, weil anhand eines heuristischen Verfahrens gezeigt wurde, dass die multikiriterille Bewertung der landschaftsräumlichen Vielfalt zur Priorisierung des Waldumbaubedarfs effektiv beitragen kann

→ die Waldbaukonzepte der Staatsforstbetriebe fokussieren sich erst mit Erreichen der Zielstärke/ Hiebsreife auf den Waldumbau, woraus geringe Überschirmungszeiträume resultieren können und sich die eigentlich schlagweise Nebenform des Samenbaumverfahrens ergibt

→ Während das breite Anpassungspotenzial der Kiefer für ihre generelle Anbaueignung spricht, zeigen ökophysiologische Einzelaspekte und kurzfristige Witterungsbetrachtungen die Grenzen eines Kiefernanbaus auf

7.7. Quellen zum Waldbau der Staatsforsten

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8. Waldbauliche Schlussfolgerungen

8.1. Fazit

8.1.1. Für die Praxis

Beide Simulationsstudien A1 und A3, die eine gestörte und ungestörte Waldentwicklung durch Einbeziehen verschiedener Mortalitätsursachen abbilden, belegen die Trägheit (*inertia*) von Waldökosystemen hinsichtlich der zeitlichen Änderungsrate ihrer Baumartenzusammensetzung. Es erscheint sinnvoll, das Monitoring fortzuentwickeln (A2) und die bestehenden Waldbausysteme fortzuentwickeln, um die Wälder proaktiv und noch differenzierter (A4) an die alarmierenden Veränderungen des globalen Wandels anzupassen.

Eine *climate-smart forestry* steht vor komplexen und teils gegenläufigen Anforderungen: Angesichts zunehmender Mortalitätsrisiken und standörtlicher Unsicherheiten (Drift) ist ein beschleunigter Waldumbau durch gestaffelte Ernte (*Akzeleration*) hin zu resilienteren Beständen erforderlich. Dies impliziert Zielstärkensenkungen und gestaffelte Hiebsfolgen. Jedoch können dann mittel- und langfristig ausbleibende Nutzungspotenziale – zusätzlich zum in Norddeutschland bereits eingetretenen Rückgang von Fichtenwäldern – zu negativen Effekten auf die Rohstoffverfügbarkeit, den Kohlenstoffspeicher und die Biodiversität führen. Daher ist eine Streckung (*Retardation*) nötig, um kontinuierliche Nutzungsmengen als Basis für die Forst- und Holzwirtschaft sowie die Habitatkontinuität zu gewährleisten. Beide Ansätze sollten komplementär angewendet werden: Risikobestände sind vorrangig zu ernten (A4); betriebssichere Bestände für spätere Zeiträume zu erhalten. Dabei ist das z.T. bereits implementierte Konzept eines Zielstärkenrahmens zentral.

Es lassen sich vier Kriterien formulieren, an denen Kiefern-Waldbausysteme ausgerichtet werden könnten, die in Tab. 8.1 bewertet werden. Dazu zählen: Eine kontinuierliche Holzbereitstellung, eine vorrangige Produktion von KSH als Wertholz, ein angemessener Waldumbau-Fortschritt (Resilienzsteigerung) sowie die Senkung des Mortalitätsrisikos.

Tab. 8.1 Synoptische Bewertung der in den Norddeutschen Staatsforstverwaltungen bestehenden Kiefern-Waldbaukonzepte anhand von vier Kriterien nach Pflege- und Ernteaspekten der Waldbausysteme (▼: negativer, ◀▶: neutraler, ▲: positiver Effekt auf die Kriterienerfüllung, a: annum, BA: Baumart, KSH: Kiefernstarkholz, ↑: mehr, ↓: weniger).

Kriterium / Aspekt	Gleichbleibende Holzmenge	Produktion von KSH nur als Wertholz	Steigerung Waldumbautempo	Senkung Mortalitätsrisiko
Pflege: Intensives Durchforstungsregime	▼ Unterschreiten zuwachsoptimaler Grundfläche	◀▶ kein altes KSH, Bäume sind ab 80 a zielstark	▲▲▲ Zielstärken werden eher erreicht	▲▲ ↑ Vorausverjüngung, ↓ Interzeption, ↓ h/d-Werte
Pflege: Extensives Durchforstungsregime	▲ zuwachsoptimale Grundfläche	◀▶▲ auch altes KSH, Bäume sind ab 100 a zielstark	▼▼▼ Zielstärken werden später erreicht	▼ ↑ h/d-Werte, ↓ Vorausverj., ↑ Interzeption
Pflege: Hohe Mischungsanteile Kiefer	◀▶▲ auch zukünftig wieder ↑ Kiefer	◀▶▲ gute innerartliche Astreinigung	▼▼ artgleicher Ersatz, Reinbestände	▼ ↓ Portfolio-Differenzierung
Pflege/ Ernte: Begrenzung Hiebsmenge	▲▲ Holznutzung *sensu* Gerhardt (1923) gestreckt	▼ Nicht-Wertholz nur limitiert nutzbar	▼ langsamere Abnutzung & Generationenfolge	◀▶ ↑ Stützgefüge, ↑ Struktur, langsamere Anpassung
Ernte: Kurze Eingriffsturnusse	▲ ↓ Rückstände & Dringlichkeiten	▲▲▲ keine Anhäufung Nicht-Wertholz	▲▲ ↑ lichtökologische Fördermöglichkeit	▲▲▲ ↓ Insekten- & Sturmgefahr, ↑ Stützgefüge
Ernte: Kurzer Nutzungsgang (Fokus Licht-BA)	▼ konzentrierter Bereitstellungszeitraum	▼▼▼ qualitativ undifferenzierte Abnutzung	▲▲ rasche Abnutzung + Generationenfolge	▼ temporäre Ungleichaltrigkeit, ↑ Sturmgefahr
Ernte: Langer Nutzungsgang (Fokus Schatt-BA)	▲▲ Bereitstellungszeitraum gestreckt	▲▲ baumindividuelle Qualitäten können ausreifen	▼ langsamere Generationenfolge + Abnutzung	◀▶ ↑ Struktur, ↑ Überalterung, ↓ Umbautempo
Erntedefinition anhand Einzelbaum (Zielstärke)	◀▶▲ gehäufte Bereitstellung, ↓ Ausgangsvorräte	◀▶▼ Einwuchs Nutzungspeak trotz Differenzierung	◀▶▲ gestreckte Abnutzung, positive Schirmwirkung	▼▼ ↓ Triage, ↓ Operationalisierung Zielstärkerahmen
Erntedefinition anhand Hauptbestand (Hiebsreife)	◀▶▼ ↓ Spreite Bestandestypen, ↑ Ausgangsvorräte	◀▶▼ undifferenzierte Ernte, plötzliche Exposition Nebenbestand	◀▶▼ rasche Abnutzung, ähnliche Bestandesstrukturen	▼▼▼ ↓ Stützgefüge, ↓ Ausschöpfung Baumpotenzial
Erntedefinition multikriteriell anhand Baum, Bestand, Risiko	▲▲▲ Differenzierung zwischen & im Bestand	▲▲ angepasste Nutzungsmöglichkeit	◀▶▲ Akzeleration und Retardation möglich	▲◀▶ abgewogene Priorisierug, Risiko zurückgestellter Bestände

Die Kriterien der bestehenden Waldbaukonzepte in Tab. 8.1 wurden auf Basis von A1-4 sowie von länderspezifischen Sonderauswertungen in nicht-referierten Arbeiten (*cf.* Kap. 9.1) aspektweise bewertet. Die Bewertung lässt sich wie folgt zusammenfassen: In modifizierten Konzepten sollten Kiefernwälder

→ stark und gestaffelt gepflegt werden (Auslesedurchforstung). Dies fördert die Durchmesserdifferenzierung, verbessert die Stabilität durch günstigere h/d-Werte, senkt die Trockenstressanfälligkeit durch verringerte Interzeption und beschleunigt den Waldumbau durch früheres Erreichen der Zielstärke. Jedoch unterschreiten entsprechende Intensitäten das Zuwachsoptimum.

→ durch früh beginnende Eingriffe in kurzen Turnussen (Wiederkehr) geerntet werden. Dies erhält das Stützgefüge im Bestand, etabliert Vorverjüngung kohortenweise, fördert Lichtbaumarten, kontrolliert die Zuwachsanhäufung und lässt Überhälter bei sauberer Waldwirtschaft sicher ausreifen.

→ im Hinblick auf die Nutzungsgänge (Intensität) generell gestreckt beerntet werden. Dies erzeugt ungleichmäßige und ungleichaltrige Strukturen und hohe Wertzuwächse in der Nebenform des Überhaltbetriebes. Stetige oder gestaffelte Nutzungsgänge sollten sich an der Betriebssicherheit im Landschaftskontext orientieren.

Insgesamt ist nicht die Senkung von Zielstärken vorteilhaft, sondern ein je Bestand früherer Fokus auf die Ernte, als bisher (*cf.* Abb. 1.4). Es sind also ambitionierte Ansätze bei der Ernte erforderlich, die zwar mit erhöhtem Aufwand einhergehen, jedoch durch kurze Eingriffsturnusse sowie angepasste Eingriffsintensitäten zur Resilienzsteigerung und Risikominderung beitragen.

Betrachtet man nur das Mortalitätsrisiko der Kiefer, berücksichtigt der gegenwärtig zunehmende Kiefernanbau im Rahmen einer reinen Abschätzung des ökophysiologischen Mortalitätsrisikos nur nachrangig, dass bestimmte Laubbaumarten der Gattungen *Acer*, *Tilia*, *Quercus*, *Carpinus*, *Ulmus* und *Betula* eine noch höhere Trockenstresstoleranz aufweisen. Dabei sollte die geringere Produktivität von Laubbaumarten auf schwach nährstoffversorgten Standorten vor dem Oberziel des Walderhaltes zurücktreten. Es wären WET-Mischungsanteile von bis zu 90 % Kiefer zu überdenken, weil darin die politisch implementierte und risikoärmere Laubwaldmehrung nur nachrangig berücksichtigt wird.

Zur Absicherung der Baumartenempfehlung sollte der Ensemble-Ansatz zumindest um ein optimistisches Klimamodell außerhalb des pessimistischen RCP8.5-Modellstranges erweitert werden: Bei einer Abschwächung des Golfstroms könnte sich die Erderwärmung verlangsamen und eine optimistischere Klimaprognose eintreten.

Im Klimawandel wird für Kiefer im Tiefland mit u.U. verringertem Wachstum gerechnet, während Hochlagen profitieren könnten. Dies legt eine zukünftige Einteilung der Kiefernwirtschaft in Orobiome nahe.

Die Waldbau- und Forsteinrichtungs-Richtlinien operationalisieren den Vorgriff und die Rückstellung von Erntebeständen, Baumgruppen oder Einzelbäumen sowie Waldbautechniken unzureichend. Insbesondere sind Waldbaukonzepte zu starr, wenn der Übergang von der Pflege auf die Ernte an den Zeitpunkt der Forsteinrichtung oder Zwischenrevision gebunden ist, die Organisationsstruktur zu groß gestaltet ist oder in unflexiblen Blöcken operiert wird.

Eine Priorisierung der Ernteplanung und des Waldumbaus sollte in einem multikriteriellen Ansatz eingebunden werden, um eine lokale Zielstärke je Bestand innerhalb des globalen Zielstärkenrahmens zu definieren, ohne Waldstrukturen auf Landschaftsebene anzugleichen. Dabei dient

→ der obere Bereich (55-60 cm BHD) später zu erntenden, stabilen, wertholzhaltigen oder seltenen Bäumen/ Baumgruppen auf mittlerer bis schwacher Nährstoff- und Wasserversorgung,
→ der mittlere Bereich (45-55 cm) der flexiblen Einteilung durchschnittlicher Verhältnisse und
→ der untere Bereich (40-45 cm) prioritär zu erntenden, instabilen, geschälten, grobästigen oder geharzten Bäumen/ Baumgruppen, die das Standortspotenzial nicht ausnutzen.

Als Verjüngungsformen stellen sich insbesondere kombinierte Verfahren mit Elementen des Schirm- und Femelschlags positiv heraus (Gruppenschirmschlag/ -stellung), weil sie das natürliche Störungsregime gut emulieren. Einzelbaumweise Zielstärkennutzungen können zusammen mit ungleichmäßigen Überschirmungen zu räumlich differenzierteren, den lichtökologischen Ansprüchen der Folgebaumart gerechten Altbestandsstellungen führen.

Eine Schlüsselrolle kommt der Diversifizierung der Waldbauverfahren zu. So sollten Reinbestandskomplexe durch situative Ausnahmemöglichkeiten, wie initiale Lochhiebe, deutlich *vor* Erreichen der Zielstärkendefinition aufgelöst werden. Auch sollte vereinzelt eine Artersetzung mit Schwarz-Kiefern (*Pinus nigra* J.F. Arnold) oder Wald-Kiefern-Provenienzen aus dem Mittelmeerraum geprüft werden.

8.1.2. Für Wissenschaft und Politik

Zuletzt wurden vor allem Dürreereignisse zum Katalysator für die forstwissenschaftliche Forschung. Dafür sind ausreichend finanzielle Ressourcen nötig. Zur Verbesserung dieser Arbeit wären 1. eine aktualisierte Prognose auf Basis der BWI[4], 2. regionalisierte Klimamodelle der neuesten CMIP6-Generation und 3. eine Untersuchung zur Wirkung der vorgeschlagenen Konzeptänderungen von großem Interesse. Dies sollte auch die Weiterentwicklung des *WaldPlaners* hin zu automatisierten Monte-Carlo-Simulationen beinhalten. Expertenbasierte Einschätzungen zur Baumarteneignung auf Grundlage von Klimamittelwerten sollten zunehmend durch evidenzbasierte Analysen ersetzt werden, die auch Extremwerte und Witterungseinflüsse berücksichtigen.

Die WEHAM der Bundesregierung berücksichtigt 2025 nicht mehr nur den bestockungsgleichen Ersatz von Beständen, sondern auch erstmals einen (teilweisen) Waldumbau (BMLEH 2025, p. 59)[1]. Im

[1] BMLEH (ed) (2025) Waldentwicklung und Rohholzaufkommen. Modellierung für die Jahre 2023 bis 2062. Bundesministerium für Landwirtschaft, Ernährung und Heimat, Bonn

Rahmen einer länderindividuellen Ausrichtung des Baumartenwechsels wird die Waldzusammensetzung (an bestimmten Trakten) hin zur potenziell natürlichen Vegetation simuliert. Damit wurde ein wichtiger Entwicklungsschritt vollzogen. Weiterer Implementierungsbedarf besteht bei einer klima- und standortsensitiven, irregulären Mortalität.

Zukünftige Forschung sollte insbesondere folgende Aspekte vertiefen: 1. Die Unsicherheit muss expliziter in waldbauliche Entscheidungsprozesse integriert werden, wie durch probabilistische Prognose- und vielfältige Umsetzungsinstrumente. 2. Langfristige Monitoring- und Frühwarnsysteme, wie das WSMP, müssen zur Erfassung standortspezifischer (Wasser-) Stressreaktionen ausgebaut werden. 3. Durch Praxisanbauten oder Intervallflächen wäre von Interesse, welche Rolle die Kiefernherkunft bei der Sterblichkeit hat und wie südliche Herkünfte zur Resilienzsteigerung beitragen können. 4. Des Weiteren ist das Waldinnenklima als Steuerungsgröße in komplexen Waldbausystemen weitergehend zu betrachten: Dazu wären z.B. Vergleiche von Hiebsformen und -arten in Kiefernbeständen sowie Versuche zur Zielerreichung bei unterschiedlich präzise operationalisierten Waldbaukonzepten von Interesse. 5. Zuletzt sei auf die sozioökonomische Perspektiven des Waldumbaus für Waldbesitzende verwiesen: Es braucht einen verstärkten Diskurs über ökonomische Anreizsysteme, Partizipation an einer Vergütung von externalisierten Leistungen und über gesellschaftliche Zielkonflikte im Kontext der Waldbewirtschaftung.

Der Politik wird eine verstärkte und noch gezieltere Förderung der Forstwirtschaft empfohlen. Auch wenn eine uneinheitliche Besitzstruktur zur Diversifizierung der Waldentwicklung positiv beiträgt, ist ein Umbau von nur etwa 60 % der ausgeprägtesten Reinbestände bis 2062 – unter optimalen Bedingungen – als deutlich zu gering zu bewerten. Der Forderung nach einer Vervierfachung des Waldumbautempos sollte forstpolitisch entsprochen werden. Insbesondere sollte durch die Lenkungswirkung von Fördermitteln versucht werden, den Umbau prioritär identifizierter Waldbestände anzuregen (z.B. höhere Anteils- oder Vollfinanzierung) oder fachliche Beratungs- und Betreuungsaktivitäten in Kampagnen zu forcieren.

Die Umsetzung anspruchsvoller Waldbaukonzepte bedarf einer ausgedehnten und kostenintensiven Personal-Ausbildung und -Fortbildung. Dazu sind klar operationalisierte, dokumentenmäßig gebündelte und hinreichend flexible Waldbau-Richtlinien entscheidend, die in den untersuchten Staatsforstverwaltungen derzeit zwischen einem und fünf Dokumenten variieren (s. Kap. 7.7).

Während der UN-Dekade fördert Deutschland im Rahmen der *Bonn Challenge* (inter-) national die naturnahe Waldbewirtschaftung. Dazu zählt auch der Waldumbau, der – wie gezeigt – sowohl einen planmäßigen als auch störungsbedingt verstärkten Vorratsabbau nach sich ziehen kann. In diesem Zusammenhang sollte eine zuwachsnachhaltige Forstwirtschaft in den Zielen des LULUCF-Sektors Eingang finden.

8.2. Ausblick

Der Klimawandel und die Globalisierung haben einen großen Einfluss auf die entwicklungsträgen (Kiefern-) Waldökosysteme Norddeutschlands. Jüngste Kalamitätswellen verdeutlichen die Relevanz des Waldumbaus und die ungeklärte Dynamik des Klimawandels lässt Wälder vom Lösungsgeber zum Leidtragenden werden.

Die negativen Auswirkungen von kurzfristigen Wetterextremen sowie längerfristigen Witterungs- und Klimaanomalien auf norddeutsche Waldökosysteme können nur durch tiefgreifende gesamtgesellschaftliche Veränderungen im Umgang mit Ressourcen gemindert werden. In der Forstwirtschaft bedarf es eines vielfältigen Waldbaus, der sich nicht auf standardisierte Konzepte verlässt, sondern kontinuierlich neue, differenzierte und an die aktuellsten Klimamodelle angepasste Konzepte entwickelt und erprobt. Neben einer waldbaulichen Technologieoffenheit setzt dies auch eine holzwirtschaftliche Ressourcenoffenheit voraus. Dazu sind partizipative Prozesse unter Einbeziehen aller Stakeholder und eine transparente Kommunikation essentiell.

Fichtengeprägte Waldökosysteme haben in Norddeutschland zwischen 2018 und 2022 störungsbedingt stark abgenommen und als Folge hat ein reaktiver Waldumbau ohne Nutzung des Waldinnenklimas begonnen. Ohne proaktive Konzepte droht auch der Waldumbau der Kiefer ohne Schirmwirkung unter Verlust ökologischer Funktionen fortgesetzt zu werden. Für kieferngeprägte Waldökosysteme in Norddeutschland besteht noch ein begrenztes Zeitfenster für proaktives Handeln. Unter zukünftig zunehmenden Störungsereignissen müssen die differenzierten Maßnahmen zur Anpassung dieser Wälder ohne Verzögerung angewendet werden.

Der langfristige Walderhalt und der Fortbestand von Kiefern-Waldökosystemen hängen in Norddeutschland davon ab, inwieweit es gelingt, bestehende forstwissenschaftliche Erkenntnisse in adaptives Management zu überführen, Unsicherheiten durch vielfältige Konzepte zu begegnen und gesellschaftliche wie ökologische Ziele in Einklang zu bringen. Der Wald der Zukunft wird weniger ein Ergebnis statischer Planung „für die späte Nachwelt“ sein, als vielmehr eine Kopplung dynamischer Lernprozesse.

8.3. Kernbotschaften

→ zur kontinuierlichen Holzbereitstellung, Risikosenkung und Resilienzsteigerung sollte sich an eine gestaffelte, intensive Pflege eine vergleichsweise frühere und lang anhaltende Ernte in kurzen Turnussen anschließen, die – wo nötig – auf eine Erntebeschleunigung und – wo möglich – auf eine Erntestreckung abzielt

→ zur Operationalisierung ungleichmäßiger, kombinierter Verjüngungsverfahren ist die Festlegung von Zieldurchmessern für Einzelbäume und Baumgruppen innerhalb eines Zielstärkenrahmens durch eine multikriterielle Planung im Landschaftskontext erforderlich

- → die Waldbau-Richtlinien sollten zugunsten einer in ihren Zielen und Techniken diversifizierten Waldentwicklung weiterentwickelt werden und waldbauliche sowie holzwirtschaftliche Offenheit fördern
- → ein klimaangepasster Waldumbau muss differenzierter und durch kooperative Ansätze zwischen Akteuren der Forst-, Holz- und Naturschutz-Branche umgesetzt werden sowie Kompromisse bei Zielkonflikten eingehen
- → Forschungs- und Entwicklungsbedarf besteht bei der planungsrelevanten Prognose einer komplexen Waldentwicklung durch jeweils aktuelle Klimamodelle sowie der Begegnung von Unsicherheit mit vielfältigen Waldbauverfahren

9. Addendum

9.1. Publikationsliste

Referierte Arbeiten (*cf.* Tab. 2.1)

Knocke HC, Axer M, Hamkens H, et al (2024) *Quo vadis* Scots pine forestry in northern Germany: How do silvicultural management and climate change determine an uncertain future? Eur J For Res 143(5):1477–1497. https://doi.org/10.1007/s10342-024-01701-0

Knocke HC, Sennhenn-Reulen H, Zeppenfeld T, et al (2025) Contributing to the disturbance ecology of Scots pine in Northwest Germany: Analyzing practitioners' reports on damage severities. Trees For People 20, 100895. https://doi.org/10.1016/j.tfp.2025.100895

Knocke HC, Axer M, Hamkens HF, et al (2025) Hope for the best, expect the worst: Including disturbances in simulations as tree- or stand-wise mortality increases uncertainty for north German Scots pine forestry. Stat Environ Res Risk Assess 39:5091–5113. https://doi.org/10.1007/s00477-025-03060-7

Axer M, Hamkens H, **Knocke** HC, Wiesehahn J (eingereicht) Integrating Remote Sensing and Site-specific Data in Forest Planning: A Multicriteria Approach to prioritize Forest Restoration in Central Europe. Trees, Forests and People

Nicht-referierte Arbeiten

Knocke H (2022) Forschungsprojekt KiefernStolz - Kiefernstarkholz nachhaltig nutzen. AFZ/Wald 77(14): 14

Knocke HC, Dirks H, Kopetzky M, Stolze H (2023) Waldumbau und Holznutzung der Kiefer bundeslandübergreifend fördern – Climate-Smart Forestry. BDFaktuell 64(3):10-15. https://www.bdf-online.de/fileadmin/user_upload/www_bdf-online_de/pdf/bdf_aktuell/2023/Artikel-Langversion/BDFaktuellKiefernStolz_lang.pdf. Abgerufen 1. März 2024

Knocke HC, Hansen J, Nagel RV, Albert M (2023) Conception of a scenario funnel for simulating climate- and site-sensitive forest development by the example of Scots pine (*Pinus sylvestris* L.) in Lower Saxony. Proc Sect For Yield 51:95–107. https://doi.org/10.5281/zenodo.10682658

Knocke H, Rabe P (2023) Waldbau und Wertschätzung: Einsatzplan für organisierte Forstleute. BDFaktuell 64(3):28–29. https://www.bdf-online.de/fileadmin/user_upload/www_bdf-online_de/pdf/bdf_aktuell/2023/23-03_bdf_aktuell_oJ.pdf. Abgerufen 1. März 2024

Knocke H (2024) Auf der anderen Seite des Globus: Neuseeländische Forstpartie zwischen Baum und Borke. proWALD 18(1):14–15. https://www.flipsnack.com/56878B66AED/2024-01-prowald-februar-2024/full-view.html?p=1&td=Ym9va3NoZWxmOnprbXFobnRndnU%3D. Abgerufen 1. April 2025

Knocke H, Nagel RV, Albert M (2024) Gute Reise, liebe Kiefer! Ausblick auf die Waldzusammensetzung Hessens im Jahr 2062. ImDialog 20(2):8–11. https://www.hessen-forst.de/sites/forst.hessen.de/files/2024-06/hf_maz_0224_web.pdf. Abgerufen 1. April 2025

Stolze H, Dirks H, Gurnik M, **Knocke** H (2024) KiefernStolz: Nutzungspotenziale für Kiefernstarkholz. AFZ/Wald 79(7):47

Knocke HC, Hansen J, Nagel RV, Albert M (2024) Climate-adapted silvicultural strategies for harvest cuts of Scots pine (*Pinus sylvestris* L.) stands in Northern Germany. IUFRO Abstr Book 26:841. https://iufro2024.com/wp-content/uploads/iufro2024abstracts.pdf. Abgerufen 1. März 2025

Knocke HC, Schick J, Nagel RV, et al (2024) Simulationsbasiertes Anwenden von Mortalitätsrisiken auf die Entwicklung des niedersächsischen Landeswaldes unter besonderer Berücksichtigung der Kiefer. Proc Sect For Yield 52:109–120. https://doi.org/10.5281/zenodo.14643508

Knocke H, Axer M, Nagel RV (2024) Kiefernwaldbau am Computer – mit Prognosen in die Zukunft. Modellieren von Waldentwicklung und Holzaufkommen für Mecklenburg-Vorpommern unter Klimawandel. immergrün 23(2, suppl. ‚forschung'):18–23. https://www.wald-mv.de/static/WALDMV/Inhalte/Wald%20aktuell/Immergr%C3%BCn/241111_Forst%20MV_Immergruen-Forschung_2_24_200x260_P_low.pdf. Abgerufen 1. März 2025

Knocke H, Dirks H, Krenn T, Kopetzky M, Stolze H, Gurnik M (2025) Technologie- und Rohstoff-Offenheit empfohlen. Ergebnisse des Forschungsprojektes »Nachhaltige Nutzungspotenziale für Kiefernstarkholz« für die Praxis. Holz Zbl 151(11):145-146. https://www.nw-fva.de/fileadmin/nwfva/publikationen/pdf/knocke_2025_technologie-_und_rohstoff-offenheit_empfohlen_ergebnisse.pdf. Abgerufen 1. April 2025

Krenn T, Berthold D, **Knocke** HC, Axer M, et al (2025) Schlussbericht zum Vorhaben „Nachhaltige Nutzungspotenziale für Kiefernstarkholz – eine ganzheitliche Betrachtung ihres Aufkommens, Waldbaus, der Holzernte und ihrer Verwertung" (FKZ: 2220WK01). WKI, Braunschweig. https://doi.org/10.34657/20694

Abkürzungsverzeichnis

AWC	Available Water Capacity (nutzbare Feldkapazität)
BA	Basal Area (Grundfläche)
BAU	Business As Usual (alles wie gewohnt)
BB	Brandenburg
BF	Bundesforst
BHD	Brusthöhendurchmesser/ → Diameter at Breast Height
BWI	Bundeswaldinventur/ → National Forest Inventory
CHD	Climatic Humidity District (Klimafeuchtestufe)
CHM	Canopy Height Model (Kronenhöhenmodell)
CI	Confidence/ Credible Interval (Konfidenz-/ Kredibilitätsinvervall)
CMIP	Coupled Model Intercomparison Project (Projekt zum Vergleich gekoppelter Modelle)
CWB	Climatic Water Balance/ → Klimatische Wasserbilanz
DBH	Diameter at Breast Height/ → Brusthöhendurchmesser
DSM	Digital Surface Model (digitales Oberflächenmodell)
DSS	Decision Support System (Entscheidungsunterstützungssystem)
EC-Earth	European Centre Earth System Model
FDT	Forest Development Type/ → Waldentwicklungstyp
FHNP	Forest Health Notification Portal/ → Waldschutz-Meldeportal
FLR	Forest Landscape Restoration (Waldumbau auf Landschaftsebene)
GCM	General Circulation Model (Globales Zirkulationsmodell)
HadGEM	Hadley Centre Global Environment Model
HE	Hessen
IQR	Interquantile Range (Interquantilsabstand)
KSH	Kiefernstarkholz
KWB	Klimatische Wasserbilanz/ → Climatic Water Balance
LDT	Large-Dimensioned Timber (Starkholz)
LS	Lower Saxony/ → Niedersachsen
MAC	Mean Annual Cut (mittlere jährliche Nutzungsmenge)

MAI	Mean Annual Increment (mittlerer jährlicher Zuwachs)
MAVT	Multi-Attribute Value Theory (multiattributive Nutzentheorie)
MCDA	Multi-Criteria Decision Analysis (multikriterielle Entscheidungsanalyse
MCS	Monte-Carlo-Simulation
MV	Mecklenburg-Vorpommern
NFI	National Forest Inventory/ →Bundeswaldinventur
NI	Niedersachsen/ → Lower Saxony
ODS/ L	Other Deciduous with Short/ Long life expectancy (anderes Laubholz mit niedriger/ hoher Lebenserwartung)
RACMO	Regional Atmospheric Climate Model
RCM	Regional Climate Model (Regionales Klimamodell)
RCP	Representative Concentration Pathway (repräsentativer Konzentrationspfad)
ReKliEs-De	Regionale Klimaprojektionen Ensemble für Deutschland
SDT	Small-Dimensioned Timber (Schwachholz)
SH	Schleswig-Holstein
SI	Site Index (absolute Oberhöhenbonität)
SML	Soil Moisture Level (Wasserhaushaltsziffer)
SNL	Soil Nutrient Level (Nährstoffziffer)
ST	Sachsen-Anhalt
SWB	Standortswasserbilanz/ → Site Water Balance
TreeGrOSS	Tree Growth Open Source Software
WEHAM	Waldentwicklungs- und Holzaufkommensmodellierung
WET	Waldentwicklungstyp/ → Forest Development Type
WettReg	Wetterlagenbasierte Regionalisierungsmethode
WSMP	Waldschutz-Meldeportal/ → Forest Health Notification Portal
ZSN	Zielstärkennutzung

Abbildungsverzeichnis

Tabellenverzeichnis