



Assessing the impact of multi-year droughts on German forests in the context of increased tree mortality

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ABSTRACT

Forests play a crucial role in climate regulation and societal well-being. Despite their significance, the increasing frequency of droughts poses a severe threat to forest ecosystems, impacting carbon sequestration and forest stability. In Germany, the unprecedented 2018–2020 drought resulted in extensive tree mortality and damaged wood volume, with lasting effects observed in subsequent years. As climate models project a continuation of such droughts, understanding the impact of droughts on forests becomes imperative. However, it is unclear how forests will evolve in the future if the drought duration continues to increase.

This study employs a forest model to analyze the impact of droughts across various German forest types, focusing on the duration of drought periods and their influence on forest productivity. By utilizing an individual-based forest growth model and national forest inventories, the study addresses critical knowledge gaps regarding the effects of multi-year droughts on biomass and productivity across various forest types, including monocultures and mixed forests. The simulations consider a drought-induced large increase in tree mortality caused by factors such as pest infestations and diseases across Germany.

Our simulation results reveal a declining aboveground biomass and gross primary production (GPP) for all simulated drought scenarios, including the three- and six-year drought. GPP is reduced by 46 % in the 3-year drought scenario and by 58 % in the 6-year drought scenario. Notably, prolonged droughts lead to cumulative losses, with a saturation effect in drought scenarios exceeding eight years. Forest stand composition influences these impacts, with greater GPP losses in low-biomass stands. Furthermore, different forest types exhibit varying responses. Monocultures and even-sized forests (mostly planted and managed forests) are more sensitive to drought than mixed and uneven-sized forests.

The results provide valuable insights into forest resilience and ecosystem responses to increasingly frequent and prolonged droughts, highlighting the importance of understanding the effects of drought on monocultures and mixed forests to inform future forest management strategies. Modelling the influence of biotic factors on forest dynamics in a process-based manner remains a challenge that requires future research.

1. Introduction

Forests are playing a major role for many ecosystem services like climate regulation as well as for society and economy welfare (Bonan, 2008). European forests cover around 38 % of the total land area (Ceccherini et al., 2020) and sequester around 0.47 GtC per year (Papale and Valentini, 2003) with some forests even sequester up to 6.6 tC ha⁻¹a⁻¹ (Valentini et al., 2000). The stored carbon of European forests

(on average 71t C/ha (George et al., 2021)) accounts for 10 % of the total EU-27 fossil-fuel emissions (Forest Europe, 2020). However, in the face of climate change, the resilience and productivity of forest ecosystems have become a matter of paramount concern. Besides rising temperatures and an increase in the variability of temperature and precipitation patterns, ecosystems have to face with more frequent and intense extreme events (Buras et al., 2020; Frank et al., 2015; IPCC, 2014; Meehl and Tebaldi, 2004). Droughts, in particular, have emerged

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as one of the most significant challenges for forests, threatening their health and stability (Allen et al., 2015; Piao et al., 2019; Reichstein et al., 2013; Senf et al., 2020; Sippel et al., 2018), as well as affecting their carbon balance which can have influence on the local, regional and global climate (Philipp et al., 2021). For instance, in 2003 the European heat wave led to a substantial decrease of around 30 % in gross primary production (GPP), causing a temporary but qualitative shift of a carbon sink towards a carbon source (Ciais et al., 2005). The drought of 2018, characterized by low soil water availability and high temperatures (Senf and Seidl, 2021), led to an even higher loss of GPP of around 40 % and to high tree mortality in large areas of central and northern Europe (Buras et al., 2020; Toreti et al., 2019). In Germany, one of the geographic cores of the 2018 drought, almost 32 million solid cubic meters of wood were damaged during this drought which is almost thrice the volume of damaged timber one year before (Statistisches Bundesamt., 2020). Even in the years after the drought legacy effects could be measured as an increase of mortality and damaged trees (Schuldt et al., 2020; Senf and Seidl, 2021). Compared to the drought of 2003 the drought of 2018 was significantly stronger and effected an area that was 1.5 times larger (Buras et al., 2020). Model simulation projected that the drought of 2018 could be exemplary for future droughts in Europe if global warming continues (Rakovec et al., 2022; Toreti et al., 2019). It even turned out that the 2018 drought lasted for the next two years (2019–2020) (Hari et al., 2020; Rakovec et al., 2022), with some of the effects intensifying and led to large-scale forest dieback in Germany (Obladen et al., 2021; Statistisches Bundesamt., 2020). For example, as a result of the multi-year drought in Germany, 50 % of all trees died at two studied sites (mainly spruce tree species) (Obladen et al., 2021).

However, despite their high ecological and economical relevance, there still exist knowledge gaps regarding the impacts of multi-year droughts on forest ecosystems, particularly with regard to their health and stability, as well as adaptation and mitigation strategies to better cope with the predicted increase in droughts (Etzold et al., 2014; Hari et al., 2020; Rakovec et al., 2022; Senf and Seidl, 2021). A better understanding is needed about how forest structure alter the resilience and recovery of forests under prolonged droughts. This indicates the need to investigate the repercussions of droughts on forest ecosystems as well as their carbon sequestration potential, to decrease the associated uncertainties and to improve future estimates of forest productivity. But the rarity of such droughts limits the opportunities for targeted ecosystem studies. Therefore, this study uses a high-resolution forest model to analyse the impact of droughts on all German forests, with a particular emphasis on understanding how multi-year droughts (consecutive) influences forest productivity. We want to distinguish drought effects on different forest types (like monoculture and mixed forests) to assess the variability in drought responses related to forest structure. We do not want to concentrate on regions that are highly affected by droughts or analyse one specific drought or even climate change scenarios. Our aim is to understand how forests can cope in the future with the increasing number of droughts that will occur more and more frequently at shorter intervals. Therefore, we have used climate data of the meteorological drought year 2018 from one forest station in Germany to design 10 hypothetical drought scenarios with different drought lengths (from 1 to 10 consecutive drought years). For non-drought years we have used the climate data of the year 2015 of the same station which represents a non-drought year. In particular, the increased tree mortality that we are currently observing in different areas of Europe is taken into account during simulated drought years (years with a drought period). Increased tree mortality is caused by both abiotic factors (e.g. storms) and biotic factors (e.g. diseases, pests) (Frank et al., 2015; Senf et al., 2020). In order to assess the effect of multi-year droughts with different durations in Germany we are using the individual based forest growth model FORMIND as well as the national forest inventory as input data and starting point for our simulations. We intend to contribute insights that could be valuable for a more comprehensive understanding of ecosystem responses to multi-year

droughts.

In particular, we try to answer the following questions:

1. How does successive multi-year droughts with different durations affect the biomass and productivity of German forests, especially considering the increased tree mortality during a drought?
2. Is the drought effect on productivity and biomass equal among forest types in Germany?
E.g. is there a difference between monocultures and mixed forests?

2. Materials and methods

In this study, we apply the FORMIND forest model to analyse the impact of hypothetical drought scenarios that differ in the number of drought years (from 1 year to 10 years). In this study, a drought is defined as a period with lower precipitation than normal (meteorological drought), compared to the long-term mean. Due to annual time steps of the model, a drought year is a year with a drought period. For the forest model, we use an established parameterisation of the most important tree species in Germany. The starting point for the simulations is the national forest inventory in Germany.

2.1. National forest inventory

The national forest inventory (NFI) is a large-scale tree inventory sampling throughout Germany which is carried out in a 10 year interval (Fig. 1; so called ‘Bundeswaldinventur’ BWI). The aim of the NFI is to get information about the forest status and the potential forest production (Kandler, 2009). The sampling is based on a systematic sampling network on a regular 4 km × 4 km grid which covers the whole area of Germany (in some regions, the grid is finer). Each location in the grid consists of four plots with each has a size of 25 m. This cluster of four

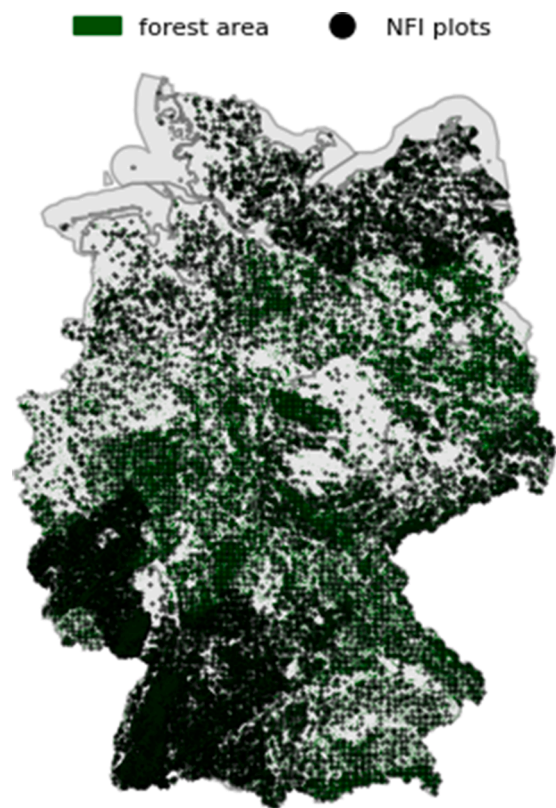


Fig. 1. National forest inventory (NFI) from 2012 across Germany. Each black dot represent one NFI plot. The green color represents the forest area in Germany.

plots has one coordinate indicating the location of the south-west plot (Polley, 2011). In this study, the NFI from 2012 is used as initialisation (starting point) for all simulations with the forest model. In total, 18,102 forest plots of the German NFI (Fig. 1) have been simulated. Information on tree species, diameter at breast height (dbh) and the location of each tree is used from the NFI dataset.

2.2. The forest model formind

2.2.1. General model description

The forest model FORMIND is an individual- and process-based gap model that can simulate forest dynamics on an area (up to 100 ha) divided into patches with a size of $20\text{ m} \times 20\text{ m}$ (Bohn et al., 2014; Fischer et al., 2016). Trees within one patch compete for light, space and soil water. Beside competition, tree growth, establishment and mortality are the main processes in FORMIND that are calculated at each time step for each tree individually. One main driving force of the processes is the incoming light, which is defined in the model as the photosynthetic photon flux density (PPFD). The productivity of a tree is influenced by its crown size as well as by the available light, the accessible soil water content and the air temperature. The light availability of a tree can be reduced by the shading due to larger trees that are standing in the same patch. The soil water content of each patch is calculated continuously and is mainly driven by precipitation (P), interception (IN), evapotranspiration, and run-off (RO) (see Eq. (1)) characterised by soil properties as well as stand characteristics. The productivity rate as well as the biomass gain of a single tree is calculated as the difference between photosynthetic production and respiratory losses. Thus, the influence of variable climate conditions on the carbon balance on tree and ecosystem level can be analysed, as well as the aboveground biomass (AGB). The productivity on ecosystem level (gross primary productivity: GPP) equals the sum of the productivity of all trees. Net ecosystem productivity (NEP) is the difference of GPP and ecosystem respiration (Resp). The latter can be calculated by adding up the soil and deadwood respiration (heterotrophic respiration R_h) to the sum of respiration over all trees (autotrophic respiration R_a). R_h is calculated from the sum of the total carbon emitted from three different carbon pools: two types of soil carbon pools (fast and slow decomposing) and one deadwood pool. This process is driven by the metabolism of dead organic matter within these pools sourced from tree mortality (Paulick et al., 2017). To simulate different climate conditions, FORMIND needs a daily climate data input consistent of daytime mean PPFD, daily precipitation, the daytime mean of air temperature and the day length, where daytime was defined as the time when $\text{PPFD} > 20\ \mu\text{mol m}^{-2}\text{s}^{-1}$ (Rödig et al., 2017). A detailed model descriptions can be found in (Fischer et al., 2016; Holtmann et al., 2021), and online at <https://www.formind.org> (last access: 08.11.2023).

2.2.1.1. Soil water module. A soil water module was already integrated into the forest model, which calculates the soil water balance for each forest patch of 20×20 metres (Gutiérrez et al., 2014; Hiltner et al., 2021). The change in available soil water content (Θ_{soil} in Vol %) is computed with a daily time step by subtracting the interception (IN) and the run-off (RO) from the precipitation (P) (see Eq. (1)).

$$\frac{d\Theta_{\text{soil}}}{dt} = P(t) - \text{IN}(t) - \text{RO}(t) \quad (1)$$

Precipitation is the input to the water balance, derived from climatic data. Total interception of one forest patch depends on the crown size of all trees within this patch and is defined by the leaf area index (LAI). If the soil is saturated with water (defined by the porosity of the soil), additional precipitation which reaches the ground leads to aboveground run-off. Subsurface run-off as well as transpiration of all trees within a patch additionally reduces the soil water content. Subsurface run-off is site specific and characterised by soil type expressed via the fully saturated conductivity, the residual water content, and the pore size

distribution index of the soil. The transpiration of all trees per patch due to photosynthesis is calculated using the water use efficiency (WUE) approach based on (Lambers et al., 2008).

Trees can only extract water from the ground if the available soil water content is above the defined permanent wilting point (PWP in Vol %). If the soil water is below this threshold, photosynthesis is not possible anymore and biomass gain ceases. Above the PWP and below the critical soil water content SW_{msw} , photosynthesis and therefore the growth of all trees in this patch is reduced via the water reduction factor (jSW %) (see equation: 3). Hence, soil water content between the PWP and the SW_{msw} defines the available soil water content for the trees at patch level. Is the soil water higher than the SW_{msw} , photosynthesis is not reduced through water limitation. SW_{msw} can be defined through the PWP and the field capacity (FC) of the soil (see Eq. (2)), whereby both parameters are depending on the soil type at the simulated location (Fischer et al., 2013; Gutiérrez et al., 2014). Therefore, the forest model is able to simulate the effect of meteorological droughts on forest dynamics, which are characterised by soil water deficiency primary induced through less precipitation over a longer time period (meteorological droughts). The precipitation data is used as an input data for simulation.

$$SW_{msw} = SW_{pwp} + 0.4 (SW_{fc} - SW_{pwp}) \quad (2)$$

SW_{msw} defines the minimum soil water content with no reduction, therefore the reduction factor jSW is calculated as followed:

$$\varphi_{SW}(t) = \begin{cases} 0 & : SW(t) < SW_{pwp} \\ \frac{SW(t) - SW_{pwp}}{SW_{mws} - SW_{pwp}} & : SW_{pwp} < SW(t) < SW_{mws} \\ 1 & : SW(t) > SW_{mws} \end{cases} \quad (3)$$

More details of the water module can be found in (Gutiérrez et al., 2014) and online at <https://www.formind.org> (last access: 08.11.2023).

2.2.2. Simulation settings: mortality function and regrowth

In FORMIND, trees can die for several reasons like their age (tree size dependent mortality), a reduced growth rate (growth-dependent mortality), space competition (crowding mortality) or through damage by larger trees that are falling as well as due to a mean annual mortality rate (background mortality).

The used parameterisation mainly takes into account tree mortality (m), which is a function of tree size (in this case stem diameter, d in [m]) with species specific parameters a_1 and a_2 (see Eq. (4)).

$$m(d) = a_1 \cdot d^{a_2} \cdot q_{drought} \quad (4)$$

Species-specific mortality parameters a_1 , a_2 are taken from Bohn et al., 2014. In drought years, this mortality rate is increased by the factor $q_{drought}$. The realised tree size dependent mortality rate is around 1–2 % in normal years (see Fig. S4). This approach has been tested in several studies (Bohn et al., 2014; Bohn and Huth, 2017; Holtmann et al., 2021). In order to simulate additional mortality related to drought effects on forest stands, we increased the tree mortality rate under drought conditions. In particular, each simulated drought year, which is characterized by lower precipitation than the no-drought year, is associated with an increased mortality rate by factor $q_{drought} = 10$, ending up with tree mortality rates between 10 % -20 % in drought years (see Fig. S4 for the differences in modelled tree mortality in a 'normal' year compared to a 'drought' year). The exact rate depends on the simulated mortality rate without drought and consequently is influenced by forest stand characteristics and species composition. This rate reflects the increase of tree mortality due to water stress, along with other secondary effects resulting from drought, like the reduced resistance against disturbances such as pest infestations (by e.g. the bark beetle) and diseases (Philipp et al., 2021). Pest infections can increase regionally mortality to high levels (e.g. $50\% \text{ a}^{-1}$ Obladen et al., 2021). Studies showed that tree

mortality increased to levels of 20 % - 25 % a^{-1} in Black Forest during drought years (Spiecker and Kahle, 2023; for examples in other regions see Ma et al., 2023). Tree mortality can be also estimated by analysing canopy cover loss. In certain regions, values between 20 % and 25 % a^{-1} have been reached during drought years in Germany (Thonfeld et al., 2022). The here used scenario represents a kind of worst case scenario, assuming that drought years occur in combination with secondary effects (such as pest infestations) in the forests of Germany.

In these simulations we do not consider natural regeneration of forests due to the short simulation time and because regrowth in Germany is mainly dominated by management activities and not by natural regrowth processes.

2.2.3. Simulation settings: parameterisation

Here, a FORMIND version (Bohn et al., 2014) that was developed to simulate European tree species in the temperate zone is used to simulate the German forests at each NFI plot. This model version has already been successfully tested and applied in several simulation studies in Germany (Bohn et al., 2014; Bohn and Huth, 2017; Fischer et al., 2019; Henniger et al., 2023; Holtmann et al., 2021; Rödig et al., 2017) and has also been tested against eddy covariance fluxes (Holtmann et al., 2021; Rödig et al., 2017). A total of eight common species in central European forests were parameterized: pine, spruce, beech, ash, oak, poplar, robinia and birch (Bohn et al., 2014). The parameterisation of these species based on the German yield tables of Schober, 1995 as well as measured species-specific traits of the species *pinus sylvestris*, *picea abies*, *fagus sylvatica*, *quercus robur*, *populus marilandica*, *fraxinus excelsior*, *betula pendula* and *robinia pseudoaccacia* (Bohn and Huth, 2017). Species specific parameterisation differ in their shade-tolerance, allometric relationships (dbh, tree height, crown diameter), mortality rates, productivity and preferred environmental conditions (Bohn et al., 2014; Bohn and Huth, 2017). Therefore, each species reacts different to climate conditions like soil water limitation.

Additionally, each NFI plot differ in its soil parameter that are used to describe the soil characteristics at the site. These parameters are influencing the soil water content and the soil water uptake potential of the trees. Therefore, each site has different soil conditions influencing the availability of the trees to cope with occurring droughts. Site specific soil parameters were computed by the mesoscale hydrologic model (mHM) including the following parameters: field capacity (FC), Permanent wilting point (PWP), porosity of the soil, fully saturated conductivity, pore size distribution index, residual soil water content as well as the initial soil water content (Samanioglu et al., 2010).

2.2.4. Simulation settings: climate data and drought scenarios

Forest dynamics were computed with an annual time step whereas each simulation started with one specific NFI plot (with information about the dbh, the location of each tree and its species). The simulation time is 10 years.

To investigate the effects of different drought durations on the forest dynamic, we developed eleven drought scenarios each with a different drought duration in years: one scenario without drought (no-drought scenario) and ten scenarios with a permanently increase of the drought length counted in years. The first scenario includes one drought year in the first simulated year followed by nine years without drought, the second scenario includes two drought years in the first two simulated years, followed by eight years without drought, whereas the last (tenth) scenario got tenth drought years successively.

The constructed climate time series (including artificial droughts) were designed in such a way that they are comparable for each of the simulated forests. For this study, it is important to understand the impact of longer droughts on forest dynamics. We have therefore excluded local differences in climate. We used exemplary climate data from a temperate forest in central Germany (eddy flux station at Hohes Holz (HoH)) to construct the artificial climate data (daytime mean PPFD, daily precipitation, the daytime mean of air temperature and the day

length; for more details about this site and the used climate data see (Holtmann et al., 2021; Pohl et al., 2023)). Climate data of the year 2015 is used for all years without drought, whereas the drought year consist of the data from the year 2018 (see Table 1). The year 2018 was a drought that has affected large parts of central Europe (A. Bastos et al., 2020; Schuldt et al., 2020), so that the used study site HoH is located in the centre of that drought. According to site measurements (eddy covariance measurements as well as climate data) the year 2018 was characterized by exceptional less precipitation (meteorological drought) which led to an extreme water deficit in summer (Pohl et al., 2023), which lasted into the following two years. Compared to the long-term mean since 1950, 2018 is conspicuous due to the highest temperature anomaly (+1.86 °C) and the third lowest precipitation sum (301 mm) (Pohl et al., 2023). Therefore precipitation in the drought year was nearly 50 % lower than the long-term average (compared to the climatic period 1981–2010 with a mean of 563 mm (Pohl et al., 2023)). This observation at the study site is consistent with other regions in Central Europe (Hari et al., 2020) and can therefore be used as the exemplary drought for our simulations. In particular because climate scenarios have shown that future droughts will be similar to the drought in 2018 (Rakovec et al., 2022; Toreti et al., 2019). The year 2015, here defined as no-drought year, has an overall precipitation of 550 mm which differ only slightly from the long-term precipitation average of 567 mm. This year can therefore be seen as a normal year, without extraordinary dry or wet conditions.

2.3. Categorization of forest types

We analysed how the productivity of different forest types changes under drought conditions. Therefore we categorized the forests into three primary classifications to display the heterogeneity of German forests and to distinguish the drought impact on distinct forest types. We examined results for (a) mixed and monoculture (species diversity), (b) for deciduous and evergreen (foliage traits) and (c) for mono- and uneven-sized forests (size heterogeneity) (see Fig. 2).

2.3.1. Species diversity: mixed and monoculture forest

To define whether a forest stand is a monoculture or not, we have calculated the basal area proportion for each species at each NFI plot. If one species is dominant at one site, this site was defined as a monoculture forest. As dominant we defined a basal area share of more than 80 % of one of the eight simulated species. If no species was dominant, the forest site was defined as mixed forest. In that way each NFI plot is defined either as a mixed or monoculture forest.

2.3.2. Foliage traits: deciduous and evergreen forest

Each NFI plot with a basal area share of more than 60 % for either deciduous or conifer trees is categorized into the corresponding forest type: deciduous or evergreen forest. The basal area share is calculated for the evergreen forests based on the cumulative basal area of the conifer species pine and spruce. Similarly, for deciduous forests, based on beech and oak trees. These species are the main species in most evergreen or deciduous temperate forests. Some NFI plots do not show a dominance of deciduous or evergreen trees and have been neglected for

Table 1

Annual sum of precipitation (P) as well as annual averages of PPFD, air temperature (TA), vapour pressure deficit (VPD). The uncertainty is the standard error of the annual mean (adapted from Pohl et al., 2023).

Year	PPFD [$\mu\text{molm}^{-2}\text{s}^{-1}$]	TA [°C]	P [mm]	VPD [%]
2015 'normal year'	237.5 (± 9.4)	10.3 (± 0.3)	550	4.0 (± 0.2)
2018 'drought year'	272.3 (± 10.7)	10.9 (± 0.4)	301	5.9 (± 0.3)

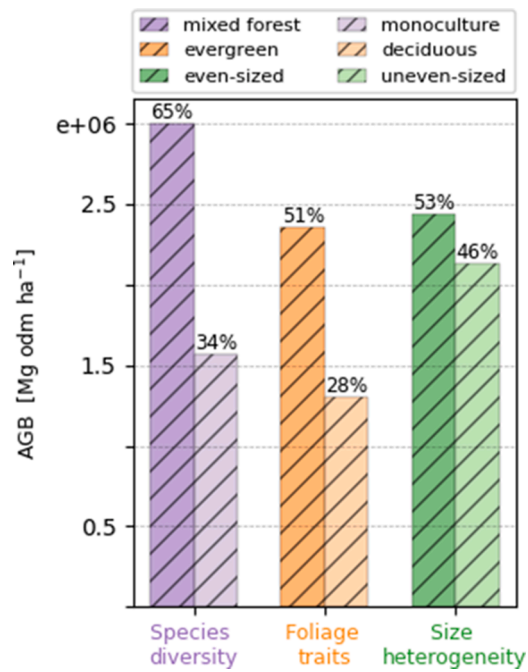


Fig. 2. Biomass (AGB) distribution of the entire NFI across three forest type categories: Mixed and monoculture forests (purple bars), evergreen and deciduous forest types (orange bars) and even-sized and uneven-sized forests (green bars). Each category comprising two forest types includes all NFI plots (total biomass) except of the second category (evergreen or deciduous forests: for definition see Section 2.3). The numbers represented above the bars denote the percentage share of AGB of each forest type within its respective category.

this categorisation (20 % of all plots).

2.3.3. Size heterogeneity: even-sized and uneven-sized forests

An even-sized forest is characterized by trees of approximately the same size, whereas an uneven-sized forest has a greater diversity of tree sizes. To estimate the size related heterogeneity for each NFI site, we utilized the size of each individual tree (dbh). To group each inventory into one category, we have calculated the standard deviation (SD) of dbh for all trees at each plot. Stands with a high SD (>0.1 m) indicate a higher variability in stem diameters (uneven-sized stands), whereas stands with a low SD (<=0.1 m) are stands with a more uniform stem diameter (even-sized forests).

3. Results

To analyse the impact of different drought durations on forest ecosystem functions, this study focused on changes in productivity and aboveground biomass during and after simulated multi-year droughts. To accurately assess the effect of different drought durations, we have compared simulation results of drought scenarios at each time step with the equivalent time step in the no-drought scenario. In that way we exclude the effect of normal mortality and other climatic influences on productivity and biomass that are not drought related (e.g. crowding or size dependent mortality). Consequently, our analysis concentrates solely on the drought-induced effects.

3.1. Simulation results of productivity and AGB over simulation time

Fig. 3 depicts the alteration of AGB (Fig. 3 B), productivity (here GPP, Fig. 3 C) and NEP (Fig. 3 D) over the 10-year simulation period under the three- and six-year drought scenarios, in comparison to the no-drought scenario, for all NFI plots. The trend in AGB over the simulation time of 10 years reveals a decline in all three scenarios illustrated in Fig. 1 driven by different mortality rates and the absence of recruitment. While the no-drought scenario starts with a value of 257 Mg odm h⁻¹

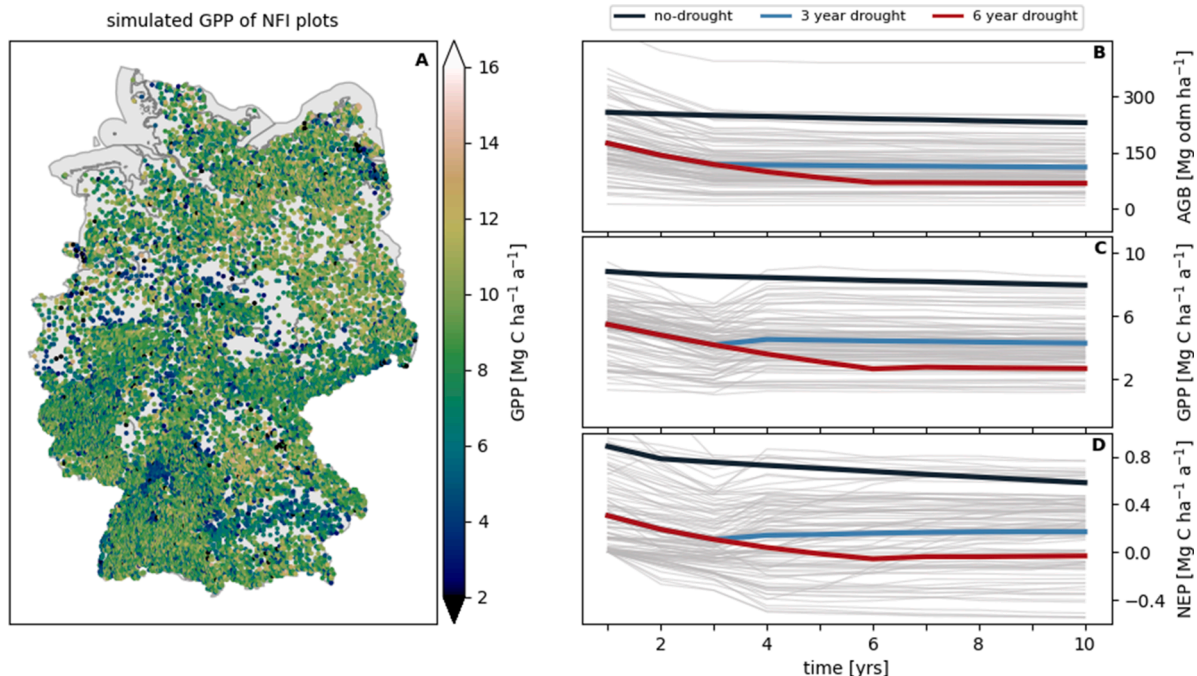


Fig. 3. (A) Simulated GPP values for each NFI plot throughout Germany under the no-drought scenario. Each data point corresponds to one NFI plot (in total 18,102 plots). GPP values represent mean values averaged over the simulated time of 10 years. On the right site, simulated (B) AGB, (C) GPP and (D) NEP over the simulation time of 10 years for all NFI plots under the no-drought scenario (black) as well as the three-year (blue) and six-year (red) drought scenarios are shown. The solid lines represents the mean value over all NFI plots for each scenario at each time step. The light grey lines are showing simulations from randomly selected NFI plots (100 plots) under the three-year drought scenario. These lines illustrate the variation of forest development within the NFI.

and ended after the simulation period of 10 years with an AGB value of 230 Mg odm h^{-1} , both drought scenarios are ending with lower AGB values compared to the no-drought scenario of around 110 Mg odm h^{-1} in the three-year scenario and 67 Mg odm h^{-1} for the six-year drought-scenario. This decline in AGB due to an increase in mortality in drought years is reflected in the simulated carbon flux values of GPP and NEP (Fig. 3 C&D). The NEP aligns with the simulated GPP trend over the simulation period. In the absence of drought, NEP and GPP attains higher values compared to the drought scenarios (mean GPP value 8.3 $\text{Mg C ha}^{-1} \text{ a}^{-1}$, mean NEP value 0.7 $\text{Mg C ha}^{-1} \text{ a}^{-1}$). Whereby each simulated drought year leads to cumulative losses in GPP and NEP, leading to a higher decrease in the six-year drought scenario compared to the three-year drought scenario. Already in the first year of simulation, the reduction in GPP due to drought is around 38 % compared to no-drought conditions. Right after the end of the simulated multi-year drought scenario (within 1–2 years), both drought scenarios are showing a slight increase in GPP and NEP related to less water stress in no-drought years (Fig. 3 C&D). This increase in GPP and NEP is higher in the three-year drought scenario (+ 8.4 %) compared to the six-year drought scenario (+ 4.3 %) which can be attributed to less severe effects of mortality on biomass. Therefore, more trees can recover from drought stress, contributing to increased productivity during the subsequent no-drought years. However, GPP and NEP values remain higher in the no-drought scenario compared to the drought scenarios till the end of the simulation period. Whereby the six year drought scenario is showing a higher decrease in GPP as well as in NEP compared to the three-year drought scenario.

3.2. Effects of successive drought years on forest productivity and biomass

Simulation results for all ten drought scenarios are exhibit a clear decreasing trend in simulated GPP with each simulated drought year (see Fig. 4). In the one-year drought scenario the forests show a mean GPP value of 6 $\text{Mg C ha}^{-1} \text{ a}^{-1}$ (over 10 years of simulation), while the six-year drought scenario has a value of 3.5 $\text{Mg C ha}^{-1} \text{ a}^{-1}$. These mean GPP values including both years during and after the drought years. The mean value of the ten-year drought scenario (3.1 $\text{Mg C ha}^{-1} \text{ a}^{-1}$) exclusively encompass values of simulated drought years. The no drought scenario records the highest mean GPP value of 8.3 $\text{Mg C ha}^{-1} \text{ a}^{-1}$. As the mean GPP values decline with each simulated drought year, the difference compared to the scenario without drought increases with each successive drought year (see Fig. 4 B). While the one year drought scenario exhibits a -27 % difference in GPP over 10 years of simulation, the three-year scenario shows a difference of 46 % in GPP, while the six-year scenario even show a decrease of -58 %. The observed difference with each successive drought year does not follow a linear trend, instead, it appears to reach a kind of saturation. Which means that, as the number of drought years increases, the additional differences compared to the previous added drought year become less pronounced. In scenarios with a high number of drought years (7–10), differences in GPP nearly diminish, while scenarios with fewer years of drought (scenario 1–5) show higher disparities in GPP values (Fig. 4 B). However, in these scenarios with more than 7 years of drought, the biomass and stand density are at a very low level, so that the water demand is also very low. The 2018–2020 multi-year drought in Europe is represent in our simulation by the three-year drought. After three years of simulation time the three-year drought shows a reduction in GPP of around -44 % (right after the drought years).

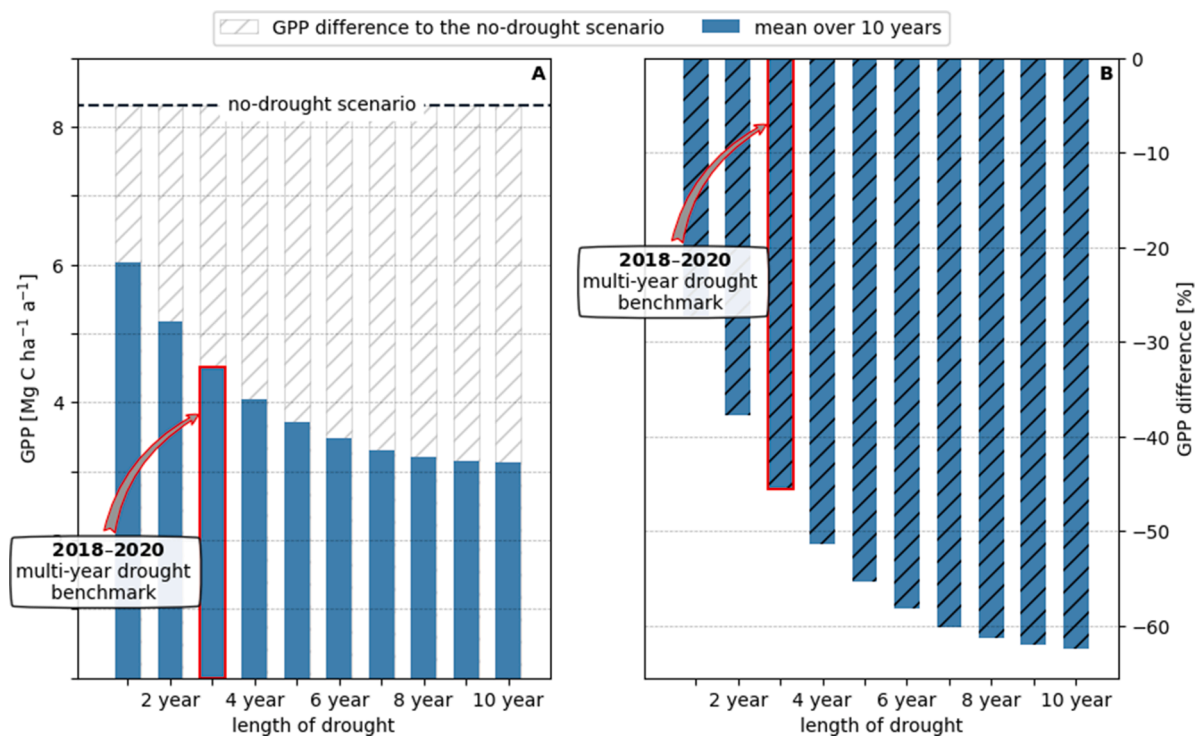


Fig. 4. (A) Simulated mean GPP values, averaged over the simulation period of 10 years (simulation years 1–10) and all analysed NFI plots. Under no-drought conditions the GPP is around 8.3 $\text{Mg C ha}^{-1} \text{ a}^{-1}$ represented by the dashed horizontal blue line above the bars. Each bar represents the mean GPP value of one drought scenario (1-year drought scenario - 10-year drought scenario) across all NFI plots. The difference in GPP between the no-drought scenarios and the drought scenario are illustrated (with dashed bars in light grey) between the blue line (representing the no-drought scenario) and the blue bars (indicating the mean GPP values of each drought scenario). (B): This figure shows the percentage difference in GPP for all NFI plots across the 10 drought scenarios relative to the no drought scenario. Each bar represents the difference in GPP for a specific drought length of 10 years of simulation time, compared to the corresponding mean value of the no-drought scenario. The red bar in (A) and (B) marks the three-year drought scenario, which serves representative for the observed 2018–2020 multi-year drought in Europe.

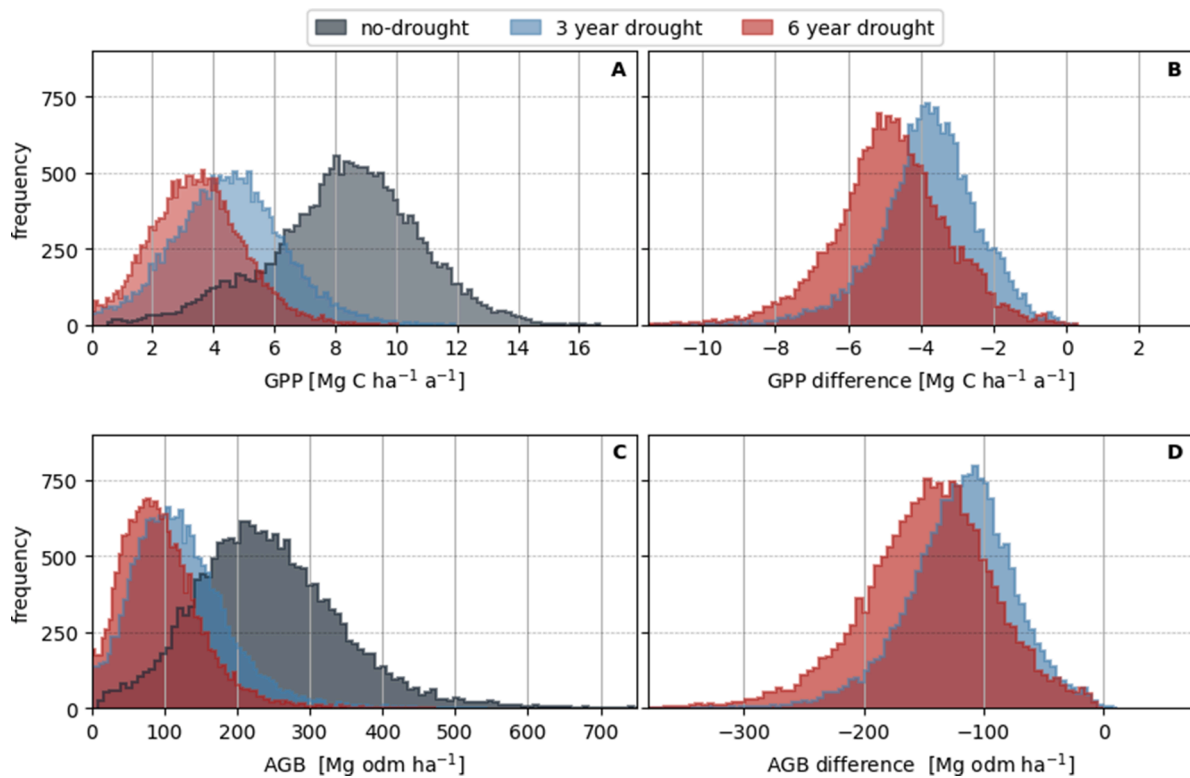


Fig. 5. Frequency distribution of all NFI plots for simulated mean values of (A) GPP and (B) AGB. Mean values are calculated over the 10-year simulation period. Each color represents one simulated scenario: the no-drought (black) as well as the three-year (blue) and six-year (red) drought scenarios. (B and D) Frequency distribution of the difference in the scenarios for (B) GPP and (D) AGB. For each NFI plot the difference between the no-drought scenario and the three- as well as the six-year drought scenario are calculated.

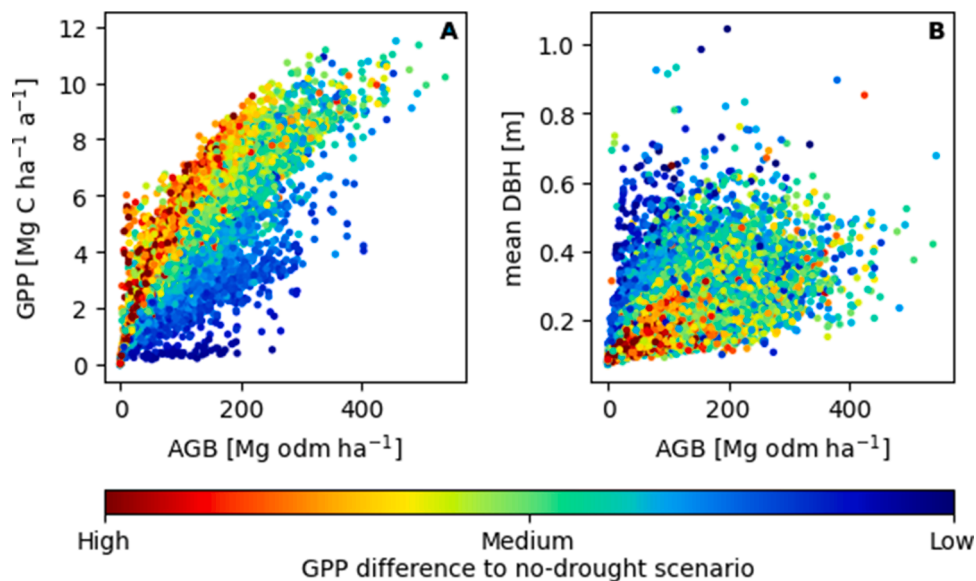


Fig. 6. (A) Relation between forest aboveground biomass (AGB) and simulated GPP under the three-year drought scenario. GPP values represent mean values averaged over the simulated time of 10 years. On the right site (B), the relation between AGB and mean DBH is shown for the three-year drought scenario. Each data point corresponds to one NFI plot. The points are colored according to the mean GPP difference (colored from 0 to -7, all values above -7 are colored dark red) over simulation period between the no-drought scenario and the three-year drought scenario at each NFI plot.

The histograms in Fig. 5,6 illustrates the frequency distribution of GPP and AGB values across all simulated forests (NFI) for different scenarios. This comparison provides insights into the variability of forest attributes across all simulated forest types illustrating the effect of forest stand composition and site conditions.

The distribution of the GPP values (Fig. 5 A) is nearly symmetrical distributed around the mean value of 8.3 Mg C ha⁻¹ a⁻¹ for the no-drought scenario, and around 4.5 Mg C ha⁻¹ a⁻¹ for the three- and 3.5 Mg C ha⁻¹ a⁻¹ for the six-year scenario. The no-drought scenario exhibits a max GPP value of 17.3 Mg C ha⁻¹ a⁻¹, while the three- and six-year

drought scenarios show lower maximum values of $13.2 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ and $11.5 \text{ Mg C ha}^{-1} \text{ a}^{-1}$, respectively. This result shows that successive drought years contributes to a decrease in GPP values across numerous forest sites, leading to a distribution shift towards slightly lower values in drought scenarios compared to the no-drought scenario.

These findings are similarly reflected in the distribution of GPP differences. The GPP difference of the drought scenarios compared to the no-drought scenario (Fig. 5 B) reveal a peak by a value of $-3.9 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ for the three-year scenario as well as a lower value of $-5.2 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ for the six-year scenario. Most of the NFI plots exhibit a GPP decrease ranging from -3 and $-5 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ during the three-year scenario, while the range is a bit lower for the six-year scenario, spanning from approximately -4 to $-6 \text{ Mg C ha}^{-1} \text{ a}^{-1}$. These findings indicate that both the number of sites and the magnitude of the GPP difference, relative to the no-drought scenario, increases with prolonged drought duration for certain forest sites.

The frequency distribution of biomass (Fig. 5C) and biomass difference (Fig. 5D) across all NFI plots are showing an equal variation as GPP. The mean biomass value of the no-drought scenario, at $242 \text{ Mg odm h}^{-1}$ (peak at $203 \text{ Mg odm h}^{-1}$), is higher than the mean values of the drought scenarios (three-year drought scenario: $123 \text{ Mg odm h}^{-1}$ (peak at $103 \text{ Mg odm h}^{-1}$), six-year drought scenario: 96 Mg odm h^{-1} (peak at 76 Mg odm h^{-1}). The six-year drought scenario has a higher biomass difference compared to the no-drought scenario for more sites than the three year drought scenario

Fig. 6A illustrates the relationship between AGB and GPP and their potential for GPP loss due to drought (three-year drought scenario) for each NFI plot. One can see that, GPP increases continuously with biomass, showing forests with lower biomass values up to $200 \text{ Mg odm h}^{-1}$ with GPP values between nearly $0 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ of up to $10 \text{ Mg C ha}^{-1} \text{ a}^{-1}$. And forest site with higher biomass values (maximum biomass

value $543 \text{ Mg odm h}^{-1}$) with GPP values between $2 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ to $12 \text{ Mg C ha}^{-1} \text{ a}^{-1}$. When comparing forest stands with similar AGB values, one can see that those forests with relatively high GPP values (compared to forest stands with the same AGB and lower GPP values) exhibit greater losses in GPP after long-lasting droughts. The highest GPP loss after drought is observed in forest stands with low biomass values (up to $100 \text{ Mg odm h}^{-1}$) but originally high GPP values. In addition, the relationship between AGB and mean dbh at each forest stand was considered in terms of its GPP difference (see Fig. 6B). It is noticeable that forest stands with smaller trees (low mean dbh) but similar AGB values are showing greater losses in GPP than forest stands with higher mean dbh values (larger trees). The highest GPP differences can be found in forest stands with a low tree size as well as low AGB values. These findings highlighting the influence of productivity and dbh at each forest site (forest stand characteristics) on the observed variations of drought impacts.

3.3. Drought impact on different forest types

To assess how various forest types within the NFI respond to drought impacts, we have examined the simulated carbon fluxes of different forest types under the no-drought and three-year drought scenario (see Fig. 7, for definitions of forest types see Section 2.3). Our simulation results indicate that all defined forest types exhibit a reduction in mean simulated GPP ranging from -38% to -48% under the three-year drought scenario. Whereby the monoculture, evergreen and even-sized forests display the highest difference compared to the no-drought scenario. In contrast, uneven-sized, deciduous and mixed forests have a comparatively lower decrease in GPP, approximately of around 40% . In the absence of a drought, mixed and uneven-sized forests have the highest mean GPP value of around $8.7 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ compared to the

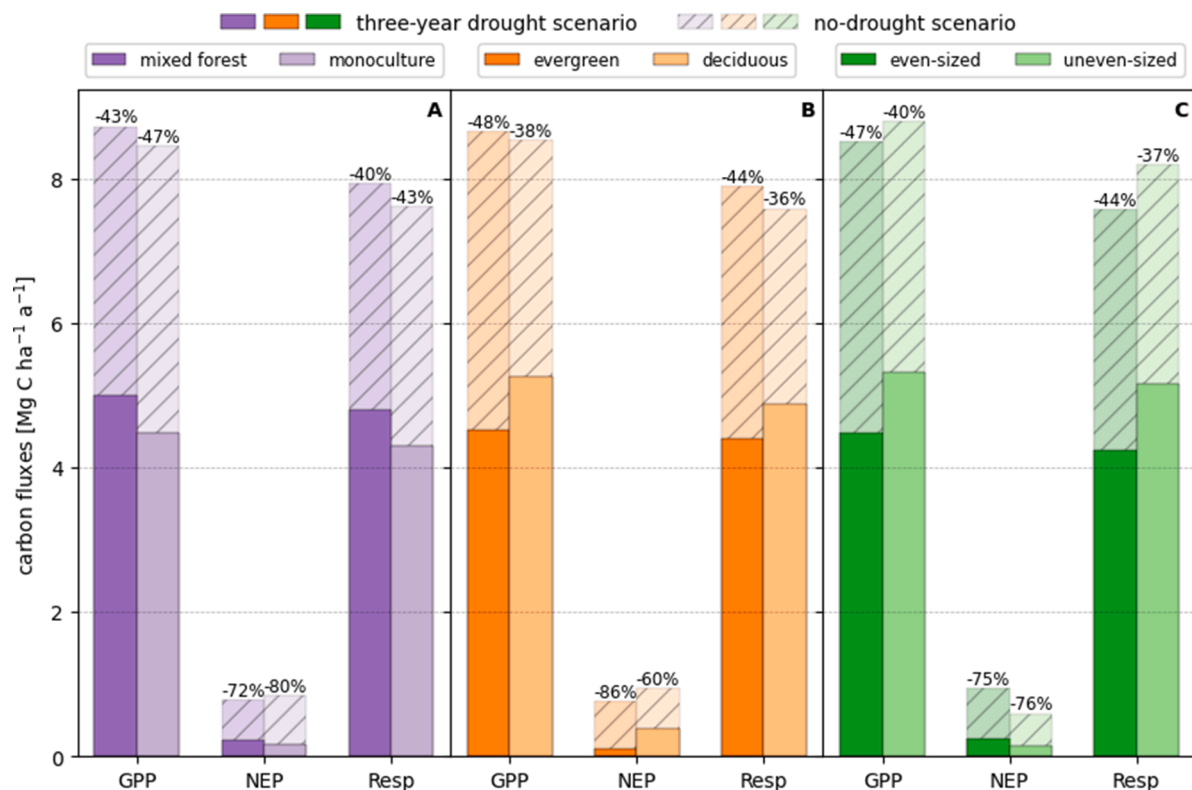


Fig. 7. Simulated mean GPP, net ecosystem productivity (NEP) as well as ecosystem respiration (Resp) values of the three-year drought scenario (coloured bars) compared to the no-drought scenario (coloured hatched bars), averaged over the three-year drought period for all NFI plots. All carbon flux values are calculated for three distinct forest type categories: (A) mixed and monoculture forests (purple bars) (B) deciduous and evergreen forests (orange bars) and (C) even-sized and uneven-sized forests (green bars). The classification of all NFI plots into the different forest types is shown in Fig. 2 (for details see Section 2.3). The numbers above the bars indicate the percentage difference of the carbon flux values of each forest type in relation to the no-drought scenario.

other forest types, followed by evergreen forests with a mean GPP of $8.6 \text{ Mg C ha}^{-1} \text{ a}^{-1}$. After the drought years deciduous and uneven-sized forests have the highest GPP value of around $5.3 \text{ Mg C ha}^{-1} \text{ a}^{-1}$.

The highest mean respiration value in both simulated scenarios (with and without drought) of around $8.2 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ (after drought $5.2 \text{ Mg C ha}^{-1} \text{ a}^{-1}$) is observed in uneven-sized forest sites, whereas the lowest respiration value is given at sites defined as even-sized and deciduous forests, of around $7.6 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ (values refer to the no-drought scenario). All forest sites experience a decrease in respiration between -36 % to -44 % in comparison to the no-drought scenario.

A look at the NEP across all forest sites reveals that evergreen forests, experience the most significant loss under drought conditions, reaching around 86 %. Monoculture forests follow closely with changes in NEP of 80 %. Deciduous forests, with the highest NEP value of $0.95 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ under no-drought conditions, are showing the lowest decrease in NEP of 60 %. Therefore, this category has also the highest NEP value of $0.4 \text{ Mg C ha}^{-1} \text{ a}^{-1}$ post-drought, whereas the lowest value correspond to evergreen forests ($0.1 \text{ Mg C ha}^{-1} \text{ a}^{-1}$). All other forest types react under the drought years with a decrease of around 70 %-76 %.

4. Discussion

4.1. Impact of drought duration on forest productivity

In this study we applied the FORMIND model to simulate forest dynamics of German forests using the NFI as starting point (of 2012). The aim is to investigate drought-induced impacts on forest productivity and AGB. The advantage of using an individual-based forest gap model is that we can analyse the impact of droughts with the same intensity and duration across various spatiotemporal scales. This approach enables a qualified investigation of the impact of successive droughts on forests with different heterogeneity in species or age.

Our main results showed that the productivity (here GPP) of the simulated forests decreases with increasing drought years. The reduction in productivity is primarily induced by water stress due to reduced precipitation and secondly also by an increased tree mortality. The frequency distribution of the NFI reveal a pronounced impact on both GPP and biomass for a large number of sites under extended droughts. This consistency across different forest stands strengthens our conclusion that prolonged droughts are having a severe impact on various German forests with different forest compositions and site conditions.

Other studies have also shown a substantially reduction of forest growth (Gao et al., 2018; Y. P. Lv et al., 2022) as well as an increase of mortality during drought years (Adams et al., 2009; Allen et al., 2010; Liu et al., 2023; Mueller et al., 2005; Van Mantgem and Stephenson, 2007). Increased mortality during long lasting droughts may be a result of accumulated drought stress in the physiology in the trees e.g. due to accumulated xylem damage that causes hydraulic failure (Anderegg et al., 2013; Gazol and Camarero, 2022; Serra-Maluquer et al., 2018) or through carbon starvation (McDowell et al., 2008). This accumulated stress in the trees might also led to a reduced resistance to pest (like the bark beetle) and diseases (Hart et al., 2017; Kolb et al., 2016; McDowell et al., 2008; Netherer et al., 2019) as well as future drought resistance (DeSoto et al., 2020). Reduced drought resistance of trees due to an increase of drought durations and frequency is reported for multiple tree species (Dang et al., 2019; Gao et al., 2018; Kannenberg et al., 2019; P. P. Lv et al., 2022). Whereby trees with a high growth-rate can be more affected by an increase of drought duration and intensity (Bose et al., 2020). We have simulated this increase of mortality in our simulations by adding a drought dependent mortality. This increased tree mortality is caused by negative consequences of water scarcity, but also by an increased susceptibility to diseases and pests such as the bark beetle. Due to a lack of mortality data, the uncertainty regarding mortality rates during droughts is high which highlights the need for further research to refine our understanding of the magnitude of these impacts on forest ecosystems. Some studies are even showing no effect of drought

duration on tree growth, which implies that some trees might endure the pressure of long lasting droughts (Brodrick et al., 2019). One reason for this resistance can be related to alternate water supply (fog, ground-water, standing water, runoff, or snowmelt) (Barbeta et al., 2019; Mendes et al., 2016; Regalado and Ritter, 2021; Zhang et al., 2019) or an adaptation of trees in their physiology or morphology to drought stress (Brodrick et al., 2019) e.g. through a decreased transpiration (stomatal closure) or restricting growth (Delgöz and Bayar, 2018; McDowell et al., 2008). Multi-year droughts may also alter the recovery of trees after a drought. While certain species show a better recovery from severe droughts than milder ones compared to others (Bottero et al., 2021), other species are more sensitive to intensity and prolonged durations (Dang et al., 2019; P. P. Lv et al., 2022; Mood et al., 2021; Serra-Maluquer et al., 2018). Some species even show compensatory growth mechanisms, leading to a rapid recovery and growth after a drought (Li et al., 2021; Seidel et al., 2019). However, it's essential to note that this compensatory growth, which helps to mitigate the negative impacts of a drought (Balducci et al., 2016; Seidel et al., 2019), may diminish with an increase in the frequency of drought occurrences (Xu et al., 2022). More reliable measurements of different tree species under drought conditions are needed to include such adaptive behaviour in forest modelling.

4.2. Comparison to the actual multi-year drought 2018–2020

The multi-year drought of 2018–2020, characterised by extreme dry weather and heat (compound drought), has led to significant reduction in observed productivity and growth rates across Europe for many tree species, leading to massive diebacks and bark beetle infestations (A Bastos et al., 2020; Buras et al., 2020; Schuldt et al., 2020; Senf and Seidl, 2021). This drought affected around 40 % of Europe for more than 18 months during the three years (Rakovec et al., 2022) with an unprecedented level of intensity compared to the last 250 years (Bardgett et al., 2014; Rakovec et al., 2022) and high ecological and economic consequences (Naumann et al., 2021). Study's reported within these years of reduction in GPP between 20 %-40 % (Buras et al., 2020; Toreti et al., 2019; van der Woude et al., 2023; Yu et al., 2022) and a reduction of NEP of between 15 %-25 % (Graf et al., 2020; Pohl et al., 2023), depending on the year of investigation as well as the investigated area. However, changes in carbon uptakes seems to be highly affected by drought characteristics (timing and intensity) (Gao et al., 2018) as well as its combination of heat and drought anomalies (Gazol and Camarero, 2022; von Buttlar et al., 2018). Some regions in Europe (e.g. southern Europe) show even an increase in NEP and /or GPP due to high temperature in summer while others are close to neutral or show a decrease (A. Bastos et al., 2020; Beillouin et al., 2020; El-Madany et al., 2020; Peters et al., 2020). Such asymmetries might be an indication that ecosystem structure and environmental conditions, such as soil water content or elevation have important influences on the respond to such extreme events (Aldea et al., 2022; A. Bastos et al., 2020; von Buttlar et al., 2018; Yu et al., 2022). For example, the station Hohes Holz in Germany shows an increase in NEP (+16 %) in 2018 due to a combination of saturated soil, early growing season and high temperatures, while the following year 2019 shows a decrease of NEP (-25 %) compared to pre-drought years possible due to lagged effects from the year 2018 and water deficits in summer, while the year 2020 nearly show no positive or negative biases (Pohl et al., 2023). Another study in an old-growth and more diverse forest in Germany has shown a reduction in GPP of 21 % in 2018, as well as stronger effects in GPP reduction in 2020 (-23.5 % -29.6 %) compared to 2019 (-3.5 % -10 %). This reduction in GPP might be a result of an increase in mortality at site of 5 % compared to 2005 and 2017 (Yu et al., 2022). Other studies in Germany reported from higher mortality rates of around 50 %, resulting from mortality of spruce trees (55 %) and mortality of beech trees (7.3 %), attributed to an increase in drought sensitivity and an outbreak of bark-beetle through cumulative effects of the successive drought years (Obladen et al., 2021). Most of these values correspond to only one year

during the successive drought of 2018–2020 and does not show the total reduction of GPP over the multi-year drought. In our study the one-year drought, also shows a reduction of GPP of around 27 %, while the three year drought which could be seen representative for the 2018–2020 drought years, shows an overall increase of reduction of up to nearly 46 % over the studied period. This goes in line with values found in literature. After only one drought year NEP shows a reduction of around 42 %, whereas after the three-year drought scenario NEP shows a reduction of around -75 %. In total, simulated German forests remain a carbon sink in simulations after the three-year drought scenario, whereas each NFI react different under the simulated drought showing the complexity of the underlying mechanisms. These changes in carbon fluxes during the drought years are similar to findings of other droughts (Ciais et al., 2005; Noormets et al., 2008). In a mixed oak and maple forest NEP was reduced even due to a moderate drought by 40 % caused by 16 % reduction in GPP and 11 % in Resp (Noormets et al., 2008). This high reduction values in NEP, like the one in this study, are a result of a slightly lower reduction in Resp compared to GPP. This effect is often addressed to compound droughts like the one in 2018 with high temperatures and less precipitation (von Buttlar et al., 2018). In summary, prolonged droughts (in duration and frequency) cumulate their effects and therefore amplified the impact of droughts on trees and increases mortality rates of trees, leading to reduced carbon fluxes.

4.3. Drought impact on different forest types

Droughts have been identified as a significant factor leading to widespread declines in tree growth as well as increased mortality across various forest types. Many studies reveal that there exists some intra-specific traits as well as intrinsic or environmental drivers that could modulate the impact of droughts on forests. Intraspecific traits encompass e.g. the vulnerability of tree species for cavitation (Kavanagh et al., 1999), hydraulic conductance (Comstock, 2000; McDowell et al., 2008), stomatal sensitivity (Klein, 2014), wood density (Serra-Maluquer et al., 2022), rooting depth (Aldea et al., 2022; Bréda et al., 2006; McDowell et al., 2008), as well as their susceptibility for pests and diseases (Kolb et al., 2016; McDowell et al., 2008). Whereas intrinsic attributes describe stand characteristics such as tree size, age structure, species richness evenness and diversity (Anderegg et al., 2018; Bottero et al., 2021; D'Orangeville et al., 2018; Nimmo et al., 2015). Furthermore, environmental characteristics such as soil type and topography can influence how forests can cope with droughts (Aldea et al., 2022; D'Orangeville et al., 2018; Nimmo et al., 2015). Hence, drought influenced tree growth varied among species and stand composition. In our study we have focused on the analysis of intrinsic attributes of forest stands and how they influence the stand productivity under drought.

4.3.1. Drought impact on monocultures vs mixed forests

Many studies comparing tree responses to droughts in mixed and monoculture forests highlight the greater resilience of mixed forest stands (Aldea et al., 2022; del Río et al., 2014; Pardos et al., 2021). This resilience is attributed to more efficient water and light utilization through niche optimization in mixed stands (del Río et al., 2017; Fichtner et al., 2017; Grossiord, 2020), leading to higher productivity (Jucker et al., 2016), higher growth rates, and resistance against disturbances (like droughts) (Aldea et al., 2022; Pardos et al., 2021; Pretzsch and Schütze, 2021) and therefore also greater ecosystem service provision (Felton et al., 2016). Despite lower resistance to droughts, pure stands might recover faster post-drought, resulting in equal recovery times for both forest types (Aldea et al., 2022). However, these studies often based on mild and/or shorter droughts. Prolonged drought durations tend to diminish the differences in tree responses between mixed and monoculture stands, impacting both resistance and recovery (Aldea et al., 2022). Therefore, mixed and monoculture stands may exhibit similar reactions under successive droughts, as demonstrated by our study. In contrast, some studies show that diversity not always

improve the resistance of forests to drought (Grossiord et al., 2014), but show beyond that a higher resistance of pure stands compared to mixed stands (Forrester et al., 2016; Pretzsch et al., 2020). Mixing species to increase forest resistance might depend on species interactions (resource partitioning or facilitation processes), tree age heterogeneity as well as environmental conditions (Forrester et al., 2016; Grossiord et al., 2014; Pardos et al., 2021; Pretzsch and Schütze, 2021). But, species interactions and competitions, especially in mixed stands, and its influence on resistance is purely understood and need more investigations.

4.3.2. Drought impact on deciduous and evergreen forests

The resistance to droughts vary between evergreen and deciduous forests, depending on drought characteristics (like intensity and duration, (Gazol et al., 2018; P. Y. Lv et al., 2022; Serra-Maluquer et al., 2018)). Overall, deciduous forests demonstrate higher resistance, while evergreen forests exhibit a faster recovery rate under moderate drought conditions (Gazol et al., 2018; Y. P. Lv et al., 2022), as well as under successive droughts (Anderegg et al., 2020; Pretzsch et al., 2020). Our results also indicate that deciduous forest react less to droughts, especially when considering responses in NEP, which is noticeably higher for evergreen forest stands. This difference can be explained by differing species attributes, such as water use efficiency, which is lower for conifer tree species in our simulation approach explaining the greater reduction under drought stress.

Other studies further illustrate differences in adaptation strategies of evergreen and deciduous forests under drought conditions. While evergreen forests, characterized by permanent foliage, maintain leaves with low photosynthetic capacity and reduced leaf water loss (Chen et al., 2009; Tomlinson et al., 2013), deciduous forest adjust their leaf phenology to climate condition through stomatal regulation (Klein, 2014; Martin-StPaul et al., 2017) or leaf shedding to reduce evapotranspiration (Bréda et al., 2006; Descals et al., 2023; Schuldt et al., 2020). Leaf shedding is an effective strategy against hydraulic dysfunction (Bréda et al., 2006; Di Francescantonio et al., 2020) and pest infestations (Karban, 2007) which does not apply for evergreen forests, leading to increased tree death in coniferous trees. While our model does not directly include pest infestation or disease spreading, simulation results include an increase of mortality during drought years illustrating the increased mortality probability of trees due to stress. While the differences we observed under drought stress are mainly due to increased tree mortality, other factors such as different parameter settings for these species (like water use efficiency) or different site conditions may also play a role. Additionally, the forest model takes into account the leaf shedding at the end of season which reduces the water loss through transpiration for deciduous trees. However, stomatal closure is currently not considered but would improve model results in further studies, due different strategies in coniferous and deciduous species (Klein, 2014). Another contributing factor is the higher susceptibility of evergreen forests to drought conditions. This susceptibility may arise from the fact that evergreen forests often have a higher cumulative interception of rainwater per year due to their continuous leaf cover, making them more prone to soil water limitations than deciduous stands under similar rainfall (Bréda et al., 2006). On the other hand, the permanent foliage of evergreen species increases the probability for suitable climatic condition within a year (Dhaila et al., 1995), which might lead to faster recovery rates post-drought as well as higher productivity values during drought years due to a longer and earlier active phase of photosynthesis. This might be important, especially during drought years starting with high soil water contents in spring. However, recovery rates of evergreen and deciduous forests may depend on drought characteristics like intensity and frequency. Hence, differences in resistance or recovery potential might be constrained under future climate conditions with more severe and prolonged droughts, reducing the differences in deciduous and evergreen forests (Gazol et al., 2018; P. Y. Lv et al., 2022; Y. P. Lv et al., 2022; Serra-Maluquer et al., 2018; Xu et al., 2022).

4.3.3. Drought impact on even- and uneven-sized forest

In this study, even-sized and uneven-sized forest stand were categorised according their dbh heterogeneity which can be interpreted as difference in tree age and size. These categorisation often goes along with other stand structure characteristics. Uneven-sized (uneven-aged) forests, are often more naturally with more dead wood within the forests (FAO, 2015) and often more dominated by broadleaved trees in combination with conifer species (norway spruce and scots pine) (Wolf et al., 2023). Therefore they fall more often into the category of mixed forests. Additionally, in uneven-sized forests (mean dbh in this study 0.26 m), taller and older trees are more prevalent than in even-sized stands (mean dbh in this study of 0.22 m). Even-sized (even-aged) stands are generally less natural (Wolf et al., 2023) and often dominated by one conifer species in Germany, aligning more closely with monoculture and especially with evergreen forests. Therefore it is not surprisingly that simulation results equal them of mixed and monoculture forests. Implying that monocultures and even-sized forests react slightly more to drought than mixed or uneven-sized forest stands.

This goes along with findings in literature that demonstrates a higher resistance of mixed and uneven-sized forests stand compared to even-sized, monoculture stands: One reason is related to a higher trait variability through a higher diversity in tree species (Anderegg et al., 2018). However, our study shows only some evidence that uneven-sized forest types are slightly less susceptible to drought. A study analysing the influence of the summer drought 2003 also show no influence of structural diversity (variation in dbh) on stability (Dănescu et al., 2018). The influence of tree size and age on resistance against droughts is diversely discussed in literature. Some studies reveal a higher resistance of older and taller trees due to the access to deeper soil water (Au et al., 2022; Bardgett et al., 2014), while other studies highlighted the contrary effect for taller trees under droughts due to their higher risk of hydraulic failure (Bennett et al., 2015; Grote et al., 2016). Taller trees, particularly larger spruce trees in dense stands, are more susceptible to bark beetle infestations and, as a result, experience induced tree mortality (Bottero et al., 2021; Netherer and Nopp-Mayr, 2005). Additionally in some stands with more vertical heterogeneity, the understory, often younger and smaller trees, might benefit from shading and the reduced water consumptions of taller trees under drought conditions. This can lead to growth reductions and an increase of mortality for taller trees under droughts which accumulated the advantage for smaller trees (Pretzsch et al., 2018). This can lead to a shift in the vertical structure of forest under prolonged droughts in future. While each tree in the model compete for light and space and therefore shading of smaller trees is taken into account in simulation results and might explain less reduction in uneven-sized stands, tree size or species dependent hydraulic failure or soil water uptake are not integrated in the current version of the forest model. Further analysis should consider risks of hydraulic failure with tree size as well as the rooting depth of tree species and tree age classes for further analysis.

4.4. Drought mortality in other forest models

Forest models use different approaches to describe the impact of droughts on tree productivity and mortality (Brodrribb et al., 2020). Classical forest models have often integrated the concept of limiting factors, which are used to decrease tree growth based on factors such as the number of drought days or soil moisture (Shugart, 1998). Mortality depends on growth stress. If the tree growth falls below a certain threshold over a certain period, tree mortality rate increases (Maréchaux et al., 2021; Shugart et al., 2018, carbon starvation). At the other end of the spectrum, models try to cover hydraulic failure by including details on the tree physiology and structure (Anderegg et al., 2016; Hartmann et al., 2018; Williams et al., 1996). Global vegetation models often use a demand concept for water transpiration per unit assimilated carbon in combination with leaf stomata dynamics (Bonan, 2019; Sitch et al., 2003). Here photoproduction and transpiration is estimated based on

optimization approaches for leaf stomata behaviour (Moorcroft et al., 2001; Sitch et al., 2003; Williams et al., 1996).

4.5. Limitations and outlook

The forest model FORMIND is well-established and has been validated for eight tree species, incorporating some simplifications for ease of parameterization and inherent constraints. However, we should be aware that each simplification could introduce uncertainties and biases, such as the omission of natural regeneration in the investigated scenarios and the assumed simplified mortality under drought conditions in our simulations. In this study, simulation results are consistently compared to the no-drought scenario so that the lack of natural regeneration is negligible in this study. But, it is advisable to address these aspects in future simulation studies, especially those with longer simulation times. This applies also for the mortality which should be expand by a growth rate dependent mortality function that increases mortality when tree productivity is reduced due to stress. Another limitation is the relatively simple soil water model, consisting of only one soil layer. Enhancing the soil water model would be a great advantage for further modelling studies analyzing the impacts of drought stress or climate change. However, this study is a basis for several promising modifications of the model, enabling more in-depth analysis like the influence of rooting depth or the influence species-specific adaptation strategies (e.g. altered allocation of carbon or stomatal closure) under the impact of extreme events such as droughts. These modifications would contribute to a more nuanced understanding of forest responses to environmental stressors and enhance the model's predictive capabilities, especially for different forest types. Further important aspects for future studies are the impacts of other drought characteristics on forest ecosystems, such as the timing or intensity of droughts (Aldea et al., 2022; Bose et al., 2021; Bottero et al., 2021; D'Orangeville et al., 2018; P. P. Lv et al., 2022; Obladen et al., 2021; Zargar et al., 2011), as well as the influence of compound extreme events (Gazol and Camarero, 2022; Sharma and Mujumdar, 2017; von Buttlar et al., 2018) which can both have significantly influence on forests responses. Analysing the impact of each dimension as well as their interactions on various forest stands is challenging but relevant for improved and adaptive forest management strategies under climate change. By simplifying spatial climate variability assuming the same climate at each forest plot of the NFI, this study was able to quantify the impacts of successive droughts on forest productivity. Adding spatially heterogeneous climate will facilitate geographical investigations of drought induced impacts but will complicate analyses by adding an additional dimension. Besides, such an approach should consider additional environmental factors which modulates tree responses to droughts like topography (Aldea et al., 2022; Nimmo et al., 2015).

4.5.1. Forest management adaptation in the face of climate change

Understanding the mechanism driving the resistance of forest ecosystems against extreme events (like droughts) becomes crucial as the intensity, frequency, and duration of such events are expected to increase under climate change. Our simulation results highlights the necessity of a re-evaluation of future forest management strategies towards more resistant forest compositions, especially due to the preference of evergreen mostly even-sized trees in managed forests (monocultures) in the past, often consisting of Norway spruce and Scots pine (Aldea et al., 2021; Friedrich et al., 2021). According to the Forest Condition Survey Germany 2022 (WZE, 2022), these tree species together make up a proportional share of 48 % of all tree species in German forests (spruce 25 % and pine 23 %). However, these conifer species might have problems to cope with prolonged and more frequent drought conditions (Aldea et al., 2022; Obladen et al., 2021; Pretzsch et al., 2020). Following the 2018–2020 drought, damage to conifers has reached very high level, with spruce being the most severely affected by the drought (WZE, 2022). Spruce trees respond to the lack of water in the soil. In

2019, widespread dieback occurred for the first time. The bark beetle attacked the already damaged spruce stands particularly hard. It is now clear that spruce is dying in large areas at lower altitudes below 700 metres. The bark beetle infestation and the resulting clear-cutting (e.g. in the Harz Mountains) are leading to large-scale deforestation of the landscape. This susceptibility to bark beetle infestation of conifers, especially spruce tree species growing in even-sized monocultures, is anticipated to increase under climate change, as rising disturbances predisposes trees. Consequently, some management strategies have shifted focus towards broadleaved species, like beech trees (Aldea et al., 2022; Pretzsch, 2021). However, there are concerns regarding the effectiveness of adaptation strategies prioritizing beech trees, due to substantial tree mortality observed in beech forests during the consecutive drought years of 2018 and 2019 (Buras et al., 2020; Schuldt et al., 2020). Our study results align with findings of other studies that promote to enhance the resistance and stability of future forests through incorporating a mix of species to increase structural heterogeneity, often based both on evergreen and deciduous trees (Aldea et al., 2022; P. Y. Lv et al., 2022; Pardos et al., 2021; Pretzsch et al., 2020). This proactive approach becomes essential as the increasing frequency of droughts have the potential to decrease forest productivity and biomass and elevate mortality rates. However, the positive effect of mixtures could not be generalized, as it is significantly influenced by the type of admixture, stand heterogeneity (e.g. in age, species or height) and site-specific factors such as the soil water content and topography (del Río et al., 2014; Nimmo et al., 2015; Pardos et al., 2021; Pretzsch et al., 2020). The complexities of these species interactions require further investigation (Aldea et al., 2022), but experiments or studies addressing this question in multi-species or uneven-sized forests are highly complex and expensive and therefore very rare (Pretzsch et al., 2014). Further modelling studies focusing on species-specific drought strategies, along with environmental factors like soil properties and topography are needed, as all these factors influencing resistance and recovery of forest ecosystems under droughts.

5. Conclusion

In this study an individual based forest model (FORMIND) was used to simulate dynamics of German forests under the influence of consecutive droughts with different frequency. Our findings reveal the cumulative damage of successive droughts on German forests, with diverse responses across different forest types. Deciduous and mixed forests stands exhibits a more enhanced resistance against droughts compared to evergreen and monoculture forests, especially with respect to changes in the carbon balance. In the face of climate change and the anticipated increase in extreme events, this result emphasize the need to reconsider and prioritize such mixed compositions in future forest managing strategies to mitigate the effect on productivity and biomass. This adaptation strategy requires more research to better understand the interactions between tree species vitality and environmental factors under drought influences, as these all influence growth rates and the overall resilience of forest ecosystems.

CRedit authorship contribution statement

Anne Holtmann: Writing – original draft, Visualization, Validation, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Andreas Huth:** Writing – review & editing, Supervision, Software, Methodology, Investigation, Conceptualization. **Friedrich Bohn:** Writing – review & editing, Software, Methodology, Formal analysis, Data curation. **Rico Fischer:** Writing – review & editing, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

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.

Data availability

The FORMIND forest model can be obtained from the website www.formind.org. The data from the national forest inventory can be accessed at the Thünen Institute (see <https://bwi.info/>).

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.ecolmodel.2024.110696](https://doi.org/10.1016/j.ecolmodel.2024.110696).

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