

RESEARCH ARTICLE

Long-term forest monitoring reveals constant mortality rise in European forests

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Drought; Europe; forest mortality; ICP Forests; soil moisture anomaly.

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ABSTRACT

- European forests are an important source for timber production, human welfare, income, protection and biodiversity. During the last two decades, Europe has experienced a number of droughts which have been exceptional within the last 500 years, both in terms of duration and intensity. These droughts seem to leave remarkable imprints on the mortality dynamics of European forests. However, systematic observations on tree decline, with emphasis on a single species, has been scarce so far so that our understanding of mortality dynamics and drought occurrence is still limited at a continental scale.
- Here, we make use of the ICP Forest crown defoliation dataset, permitting us to retrospectively monitor tree mortality for all major conifers, major broadleaves, as well as a pooled dataset of minor tree species in Europe. In total, we analysed more than three million observations gathered during the last 25 years and employed a high-resolution drought index which can assess soil moisture anomaly based on a hydrological water-balance and runoff model.
- We found overall and species-specific increasing trends in mortality rates, accompanied by decreasing soil moisture. A generalized linear mixed model identified a previous-year soil moisture anomaly as the most important driver of mortality patterns in conifers, but the response was not uniform across the numerous analysed plots.
- We conclude that mortality patterns in European forests are currently reaching a concerning upward trend which could be further accelerated by global change-type droughts in the near future.

INTRODUCTION

With the dry years from 2018 to 2020, the discussion on forest decline has returned, 40 years after the phenomenon of forest decline first worried people in Central Europe. At that time, the cause of the decline was largely unknown, which prompted politicians and researchers to implement unprecedented forest monitoring programmes across Europe. Although it seems that the discussion on forest vitality is current at the moment, the reasons for the decline are now much more obvious: millions of hectares of forests are currently suffering from progressively drier growing conditions (Byer & Jin 2017; Anderegg *et al.* 2020; Senf *et al.* 2020; Koontz *et al.* 2021) as a consequence of climate change. Despite the fact that drought can act as a natural disturbance agent in many regions, so-called climate change-type droughts have the potential to affect long-term forest vitality across all geographic regions because they are longer, warmer and more frequent compared to naturally occurring droughts (IPCC 2007). Europe comprises a large number of different forest ecosystems following a steep bioclimatic gradient from

northern boreal to southern mediterranean climates. While there is general agreement among experts that occurrence of future global change-type droughts will cause severe problems in the Mediterranean and central part of Europe rather than in the boreal zone (Stagge *et al.* 2017), such expectations are not always backed by data from monitoring programmes. Furthermore, regional studies most often interrogate drought response at higher spatial resolution, but generalizing such findings for certain species across the entire distribution range can be difficult (Bussotti & Pollastrini 2017; Rohner *et al.* 2021). As a matter of fact, boreal forest ecosystems also severely suffered from drought in 2018 and subsequent years, which resulted in significant growth declines, increased mortality and wildfires (Krikken *et al.* 2021; George *et al.* 2020; Lindroth *et al.* 2020). This strongly suggests that future global change-type droughts will affect the entire continent rather than single regions, and that forest monitoring could play a key role in guiding future forest management plans.

Despite intense monitoring efforts at national level, it is still difficult to assess tree mortality and compare it at a continental

scale, mainly due to different assessment scales and varying survey intervals (Hartmann *et al.* 2018). Recent studies have investigated mortality rates in mature trees that form the canopy, which can be captured using remotely sensed resources, such as Landsat, allowing large-scale assessment at a high spatiotemporal resolution (Potapov *et al.* 2008; Hansen *et al.* 2010; Senf & Seidl 2021). Unfortunately, these large-scale assessments are very limited in explaining reasons for forest loss and still rely on ground observations for validation (He *et al.* 2019). Consequently, a harmonized assessment scheme based on ground observations is, indeed, needed in order to complement large-scale insights gained from remotely sensed data sources.

Forest vitality is influenced by numerous factors. While climate has long been known to directly influence tree growth on short- and long-term bases, other factors, like pollution, are assumed to cause long-term declines (Bréda & Badeau 2008; Solberg *et al.* 2009). To mainly detect this latter influence, a coordinated international survey was launched in 1985 under the direction of the International Cooperative Programme on Assessment and Monitoring of Air pollution Effects on Forests (ICP Forests) (Lorenz 1995). The focus was on visual assessment of tree condition, namely crown condition. This was recorded on a regular basis, assessing, among other factors, defoliation, discoloration and crown transparency. Despite earlier concerns (Innes 1993; Innes *et al.* 1993), crown condition remained one of the central elements within the forest condition survey (Ferretti 1998; Ferretti 2013). After more than 30 years of monitoring, it is now possible to obtain the first long-term trends on crown condition and mortality within Europe (Carnicer *et al.* 2011). In addition, temporally and spatially high-resolution meteorological data have been gathered throughout the very recent years and have already resulted in an advanced understanding of ecological, ecophysiological and ecogeographical responses of forests to climatic extremes (Neumann *et al.* 2017; Peters *et al.* 2020; Senf *et al.* 2020). However, a scale-free, continent-wide and more integrative drought indicator that goes beyond temperature and precipitation could shed additional light on the relationship between forest mortality and drought occurrence (see Methods below).

By using the continent-wide and harmonized ICP Forests dataset on crown defoliation, we address the three following questions: (i) has mortality in European forests increased during the last 25 years; (ii) has dryness accelerated across the distribution of major European tree species, as well as minor

species (*i.e.* species with scattered occurrence and which are of minor importance for the forestry sector); and (iii) is increasing dryness a significant, among others, of mortality in the European forest landscape?

MATERIAL AND METHODS

European-wide forest monitoring data and crown defoliation assessment

The long-term monitoring of semi-quantitative health indicators and their representation on a 16×16 km grid was established in the 1980s as an easily operated early detection system for major changes in forest health (Eichhorn & Roskams 2013) across all European countries and also in parts of Russia, Belarus and Turkey. Defoliation can be seen as the key parameter within the Crown Condition Assessment of the ICP Forests network (Eichhorn *et al.* 2020). The parameter 'defoliation' is visually assessed, and a tree is defined as dead when its defoliation has reached 100% and the tree no longer appears in subsequent surveys. The likely more important parameter of 'removal_mortality' was introduced in 2011 after a manual update to obtain more precise information on the causes for tree mortality (*i.e.* abiotic, biotic, planned utilization, *etc.*). For the analyses shown in this study, this dataset comprises a total of 20,455 survey plots across Europe. We removed European ash and narrow-leaved ash from the dataset as mortality in these two species is currently driven by an invasive pathogen rather than by climate (George *et al.* 2022). We also removed all mortality instances that were attributable to planned utilization or felling and selected only survey years from 1992 onwards in order to obtain a constant sample size across the continent for each year. This resulted in a total of 3,145,513 observations (plot \times years \times trees) used for statistical analyses (Table 1). All subsequent analyses were separately carried out for the main European tree species (Norway spruce, Scots pine, silver fir, European larch, oak, European beech) as well as for the pooled dataset of minor tree species (including all native and non-native species in the ICP Forest database). Oaks encompass both sessile oak and pedunculate oak, since both species are known to share ecophysiological characteristics and often build hybrid zones throughout their distributions. An overview of survey plots for each species is provided in the Data S1.

Table 1. Summary of analysed data per species (N_{obs} , N_{plots} , N_{trees}).

Species	N_{obs}	N_{plots}	N_{trees}	slope AMR ₉₅₋₂₀	slope SMA ₉₅₋₂₀
<i>Picea abies</i>	533,496	7979	71455 (1–110)	Positive***	Negative***
<i>Pinus sylvestris</i>	749,442	10084	94334 (1–110)	Positive**	Negative
<i>Abies alba</i>	55,907	363	4577 (1–50)	Positive	Negative
<i>Larix decidua</i>	28,302	350	2392 (1–37)	Positive*	Negative**
<i>Fagus sylvatica</i>	312,132	1322	14957 (1–72)	Positive**	Negative
<i>Quercus robur/petraea</i>	222,118	1523	18449 (1–69)	Positive***	Negative
Minor tree species	1,244,116	10065	124655 (1–78)	Positive***	Negative**
Σ	3,145,513				

Asterisks indicate significance at * $P < 0.05$, ** 0.01 , *** 0.001 . AMR₉₅₋₂₀: Annual mortality rate average between 1995–2020; SMA₉₅₋₂₀: average soil moisture anomaly between 1995–2020. Note that model significances for the time-series trends in AMR (slope AMR) and SMA (slope SMA) were calculated from a linear regression model with ordinary least squares.

Standardized annual mortality rate

We calculated the annual mortality rate (AMR) as:

$$AMR = \left(1 - \left(\frac{N_t}{N_0} \right)^{\frac{1}{t}} \right) \times 100 \quad 1$$

where AMR is the annual mortality rate given in percentage yr^{-1} , N_t number of living trees at the end of each survey period, N_0 number of living trees at the beginning of each survey period, and t being the inventory interval (*i.e.* time between two survey periods) (Sheil & May, 1996). By using equation (1), we account for the phenomenon of apparent rate decrease, because estimated mortality rates tend to decrease with increasing census periods, in particular when mortality probabilities are not homogenous among sample plots (Sheil & May, 1996). Although a large majority of surveys have been carried out at annual intervals, some countries changed their interval period from annual to 5-year periods during recent years (*e.g.* Norway, Sweden). AMR was calculated for each tree species separately in order to reduce inhomogeneity in mortality probabilities among sample plots. For graphical illustration and trend analysis, we further standardized species-specific AMR in order to account for the fact that natural mortality rates may strongly vary among different ecosystems and bioclimatic regions and to reduce the superimposing effect of extreme outlier years. Standardized AMR was expressed in z-scores as:

$$AMR_{\text{stand}} = \frac{AMR_i - AMR_{\text{mean}}}{AMR_{\text{sd}}} \quad 2$$

where AMR_i is the annual mortality rate in interval i , AMR_{mean} is the mean mortality rate during the complete survey, and AMR_{sd} is the standard deviation of annual mortality rate. Temporal trends in standardized mortality rates for each species were assessed by means of ordinary least squares (OLS) in a linear model framework in R version 3.5.0 using the *lm* function (R Core Team 2021).

European-wide soil moisture data

We used soil moisture anomaly version 2.1.0 as implemented in the Copernicus European Drought Observatory (EDO), available under <https://data.jrc.ec.europa.eu/dataset/882501f9-b783-4b6e-8aca-1875a7c0b372>, which is used to determine start and duration of agricultural droughts (*e.g.* Cammalleri *et al.* 2017). Briefly, daily soil moisture content is calculated based on a hydrological rainfall–runoff model (de Roo *et al.* 2000), with daily meteorological input data for the entire European continent. Meteorological data are obtained by interpolating an extensive network of ground stations (approx. 2000 for temperature and 2500 for precipitation), collecting data in near-real time (Ntegeka *et al.* 2013). Soil hydraulic properties are obtained from the HYPRES database, following the pedo-transfer functions of Wösten *et al.* (1999). The hydrological rainfall–runoff model simulates soil moisture separately for a skin layer and for the root zone and subsequently normalizes soil moisture as:

$$SMI = 1 - \frac{1}{1 + \left(\frac{\theta}{\theta_{50}} \right)^6} \quad 3$$

where SMI is the soil moisture index, θ is the daily volumetric water content as weighted average of skin and root zone values,

and θ_{50} is the mean volumetric water content between the wilt-point and field capacity. The skin layer is defined as the top 50 mm of the soil, while the root zone is derived from root depth data available at European scale in the European Soil Database (ESDB, http://eusoils.jrc.ec.europa.eu/esdb_archive/ESDB/Index.htm). The rainfall–runoff model was validated against a network of 717 calibration stations across Europe (see Arnal *et al.* 2019 for more detailed information).

In analogy to standardized annual mortality rate described above, we used standardized 10-daily soil moisture anomaly, which is:

$$SMA = \frac{SMI_i - SMI_{\text{mean}}}{SMI_{\text{sd}}} \quad 4$$

where SMI_i is average SMI for each 10-day period, SMI_{mean} is long-term average SMI, and SMI_{sd} is standard deviation in soil moisture index for each 5×5 km grid cell. We used the full baseline period from 1995–2020 as reference period. SMA can be obtained at a spatial resolution of 5×5 km for the whole of Europe, and we used the *raster* package in R (R core Team 2021; R version 3.5.0 and raster package version 3.5–15) to extract soil moisture anomaly information for all analysed ICP Forests plots where data were available (plots where SMA retrievals were not available were excluded from analyses, this affected less than 1% of the data).

Statistical analysis

We modelled annual survival probability as a function of current-year and previous-year drought intensity (average SMA from April to August) at plot level. The rationale of including previous-year SMA as a predictor is that tree mortality can also be driven by drought legacy effects (*i.e.*: the adverse effect of a drought can last for >1 year). In particular, we defined N_t as described in equation (1) as the dependent variable, while controlling for N_0 and the inverse of the survey interval $1/t$ within a Bayesian logistic regression framework. We assumed a binomial likelihood with a logit link function as model for N_t . The latent predictor part of the model contained three overall (fixed) effects: SMA in current-year (cySMA), SMA in previous-year (pySMA) as well as the interaction between current-year and previous-year SMA. Furthermore, we included varying (random) intercepts as well as varying slopes of the relationship between annual survival rate and soil moisture anomaly among survey plots by treating the survey plot as grouping variable in the model. The model can be written as:

$$y_{ij} \sim \text{Binomial} \left(N_{0ij}, \theta_{ij}^{1/t_{ij}} \right) \\ \theta_{ij} = \text{inv_logit} \left(\alpha + \beta_1 \times \text{cySMA}_{ij} + \beta_2 \times \text{pySMA}_{ij} \right. \\ \left. + \beta_3 \times \text{cySMA}_{ij} \times \text{pySMA}_{ij} + a_i + b_{ij} \right) \quad 5$$

where y_{ij} is the number of surviving trees in plot i and year j following a binomial distribution and dependent on N_0 and the inverse of the survey interval t according to Equation (1). θ is then estimated with an inverse logit function including the

intercept α and β_1 to β_3 being the three overall effects of current-year SMA, previous-year SMA and the interaction between them. Interaction terms were first transformed into positive values (raw value minus minimum) and then centred and scaled. Finally, a_i denotes the random effect of the survey plot i and b_{ij} an observation level random effect that accounts for putative overdispersion in the data (*i.e.* additional variance exceeding that expected under the binomial model).

To estimate this model, we used the R package *brms* (version 2.16.3; Bürkner 2017, 2018), which is based on the probabilistic programming language Stan (Carpenter *et al.* 2017). All models were run with four MCMC chains, 6000 iterations each and with a warm-up of 1000 iterations for each chain. Weakly-informative normal priors (mean = 0, standard deviation = 5) were assigned on all the overall effects, while all other parameters had the default priors assigned to them by *brms*, which are designed to be non- or weakly informative (see Data S2 for details). Posterior predictive checks were carried out in order to qualitatively assess putative discrepancies between modelled and observed data (Gabry *et al.* 2019). Furthermore, we fitted and compared two models: (1) with only varying intercepts

and (2) one additionally including varying slopes of all predictors using approximate leave-one-out cross validation (Vehtari *et al.* 2017).

RESULTS

Linearly increasing trends in annual mortality rates were significant for Norway spruce, oak, European beech, Scots pine, European larch as well as for the remaining minor tree species ($P < 0.05$). Only silver fir did not show any trend across the investigated time period. All conifers except silver fir showed characteristic mortality peaks in 2005 and 2019, that is 1 or 2 years after the prominent millennial drought events in 2003 and 2018, respectively (Fig. 1), oak and European beech showed only a slight increase in 2004 and 2005, and the first obvious increase in 2012, with another minor peak in 2019. Minor tree species showed peaks in 2012, 2013 and 2015 and 2018.

Accordingly, decreasing trends in soil moisture anomaly were significant in plots for Norway spruce, European larch and minor tree species. In oak, silver fir, Scots pine and

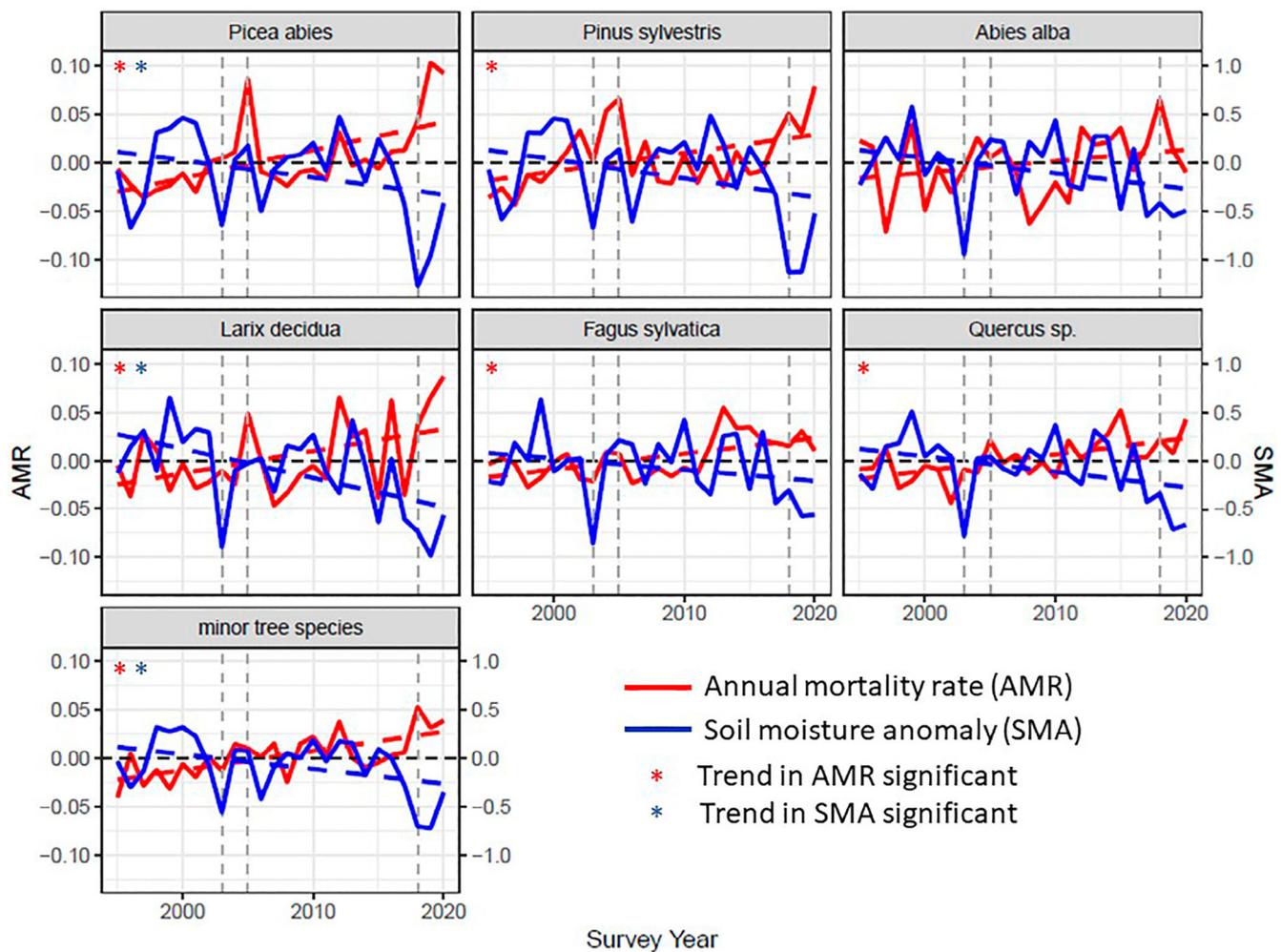


Fig. 1. Temporal dynamics and trends in annual mortality rate (red) and soil moisture anomaly (blue) by species. Continuous lines represent mean across species ranges. Standard errors are not displayed for reasons of better illustration and visibility. Dashed lines show linear trends. Grey vertical lines indicate years 2003, 2005 and 2018.

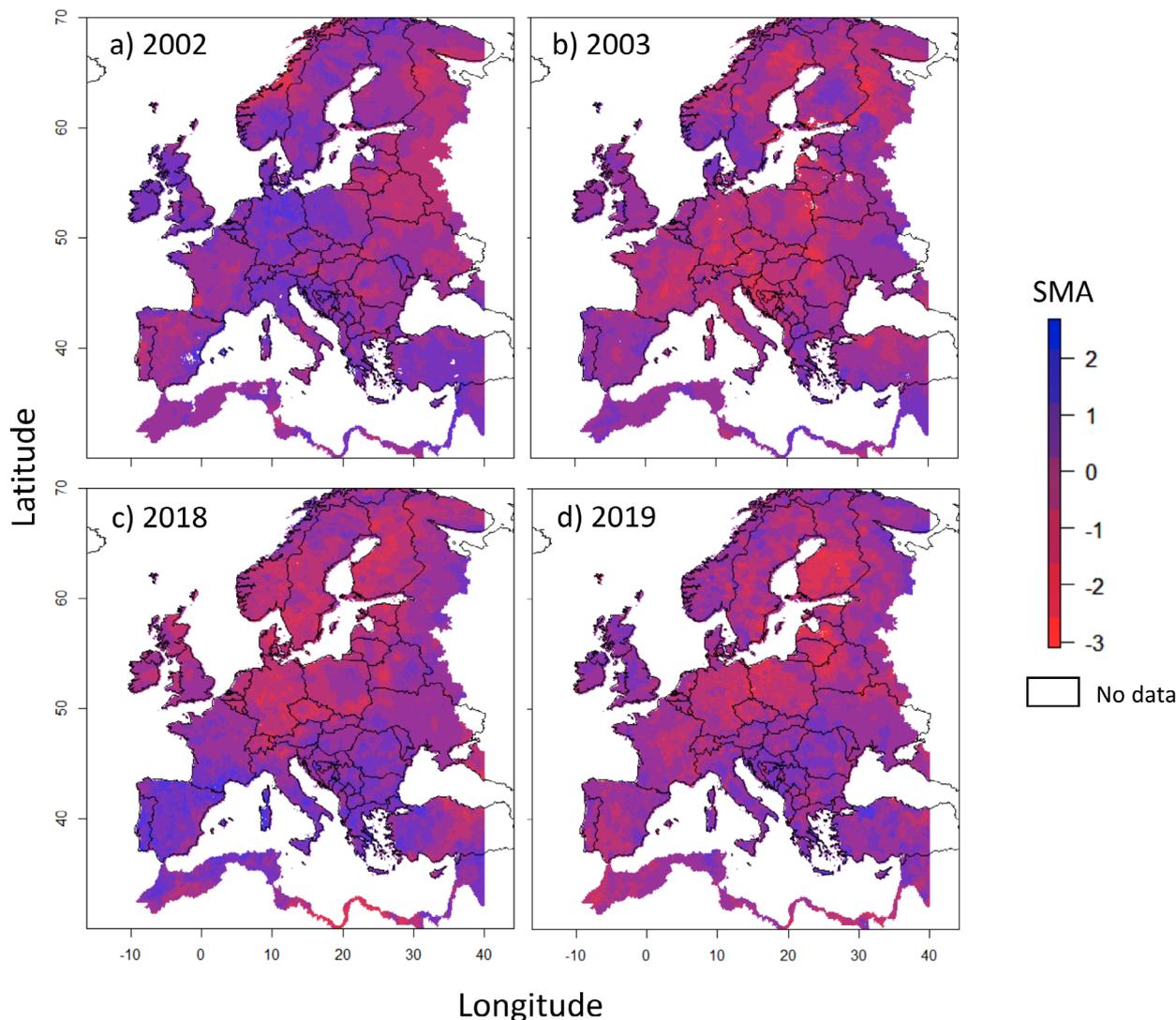


Fig. 2. Soil moisture anomaly patterns for the three drought years 2003, 2018 and 2019. The average year, 2002, is shown as reference. Shown is the mean SMA from April–August for each grid cell. White cells are retrievals that were excluded due to unrealistically high values or that correspond to water. Note that red colours show drier conditions and blue colours moister conditions.

European beech, the trend was visible but not statistically significant. As expected, SMA was lowest across all species ranges during the extreme drought year 2003 (-0.5 to -1.0 standard deviations below the long-term mean), as well as during the most recent drought years 2018 and 2019. In particular, species' sample ranges which have their core distribution in northern latitudes (spruce, Scots pine) were characterized by much lower SMA values in 2018 and 2019 compared to 2003 (Fig. 2).

Spatial mortality patterns were rather scattered but occurred broadly within the occurrence of the two millennial drought events, 2003 and 2018/2019. In particular, mortality patterns in Norway spruce generally matched with the drought occurrences and revealed hotspots of mortality clustered in the eastern and central part of the distribution (2003–2005) as well as in southern Scandinavia (Fig. 4). Towards the end of the survey period (2018–2020) spruce trees had died mainly in the central part (southern Germany, central Germany, Switzerland,

Czech Republic, Slovakia), in the Baltic states as well as in Norway and Sweden (Figs 3 and 4).

In the model with varying intercepts across survey plots, both current-year and previous-year SMA had a similar strong effect on survival (*i.e.* drier conditions trigger decreasing survival rates) for conifers (Table 2, Fig. 5). The overall interaction effect between current-year and previous-year SMA was negative, indicating that the combined effect was less than the sum of the two single predictor terms. In broadleaves (oak, beech) and minor tree species, the effect of previous-year SMA was more strongly pronounced than the effect of current-year SMA (0.32 and 0.21 *versus* 0 and 0.06, respectively). However, 95% credibility intervals included zero for all three parameters in the three groups (Table 2).

The model extension which also included varying slopes of the predictors across survey plots additionally suggested that the relationship between survival and soil moisture anomaly is

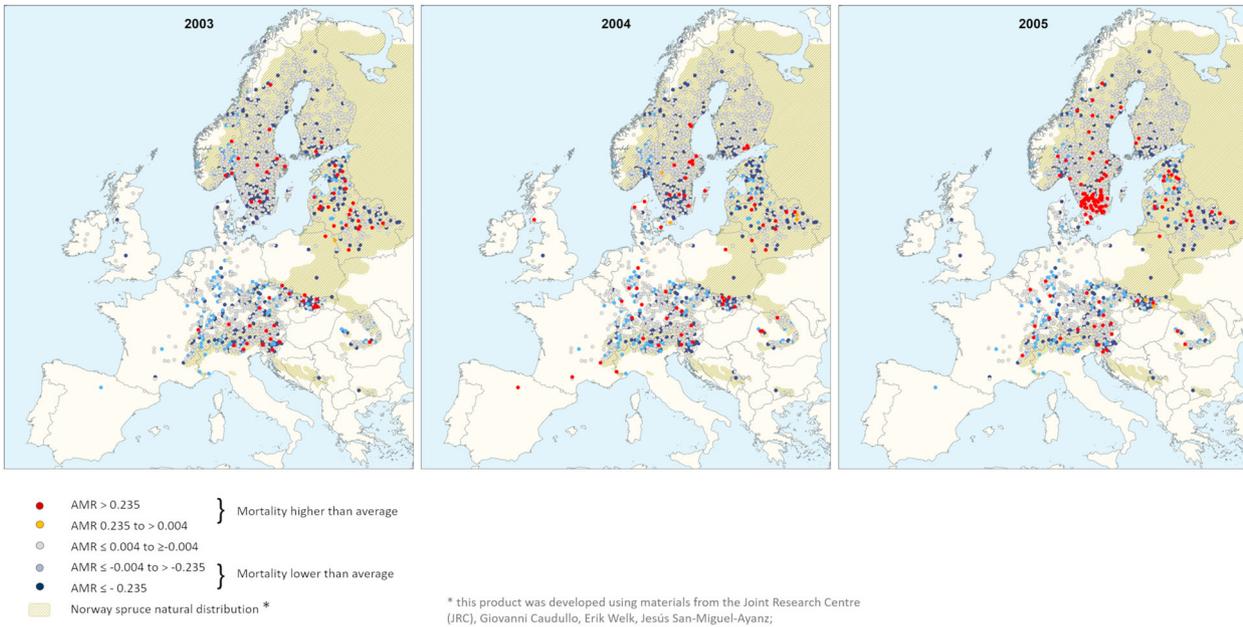


Fig. 3. Spatial variation of annual mortality from 2003–2005 for Norway spruce (*Picea abies*). Red and orange dots show plots with above-average mortality (referenced against average 1995–2020). Green area shows natural distribution of Norway spruce. Note that data were scaled for illustration and thresholds were chosen to enhance visibility across the entire range.

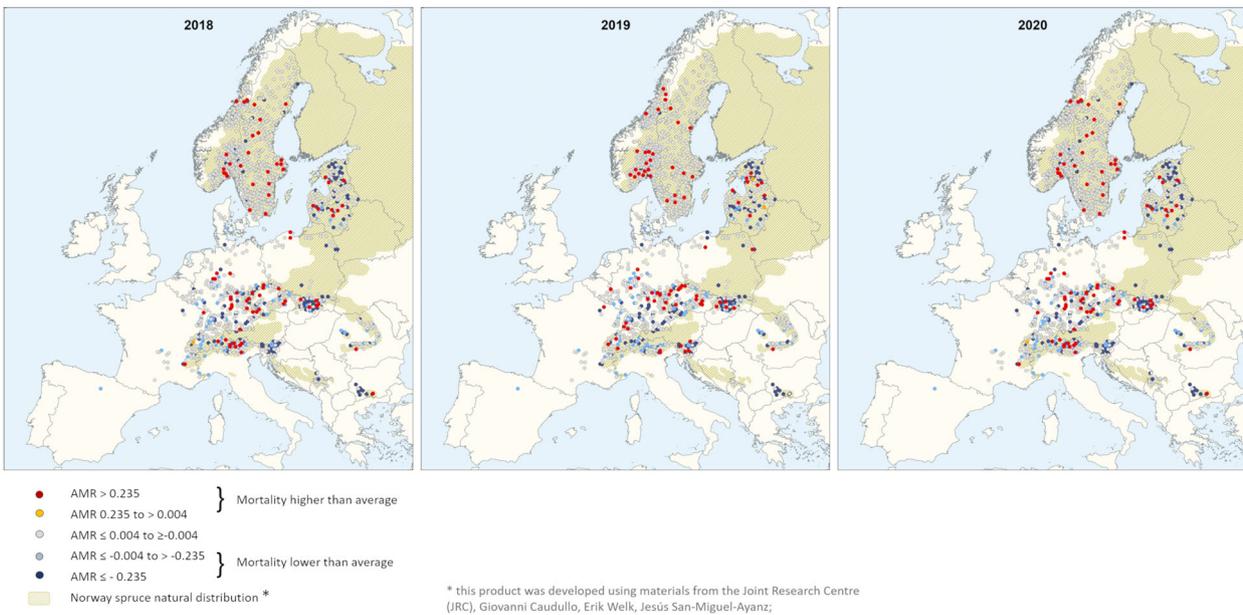


Fig. 4. Spatial variation of annual mortality from 2018–2020 for Norway spruce (*Picea abies*). Red and orange dots show plots with above-average mortality (referenced against average 1995–2020). Green area shows natural distribution of Norway spruce. Note that data were scaled for illustration and thresholds were chosen to enhance visibility across the entire range.

not uniform throughout the investigated space (see SD parameters in Table 2). In conifers, however, the effect of previous-year SMA was slightly stronger compared to the varying intercept model (0.67 versus 0.53) and 95% credibility intervals did not include zero. In all three cases (conifers, broadleaves, minor species), the varying slopes model performed marginally better than the varying intercept model according to the expected log pointwise predictive probabilities. We report the

reproducible R-code, conditional effect plots and model comparisons in Data S2.

DISCUSSION

In several earlier studies, European forest ecosystems have been repeatedly shown to be negatively affected by severe droughts during the last decades. Data so far were based

Table 2. Model summary from the GLMM.

	Varying intercept model							Varying slope model						
Random effect parameters	Conifers (N = 45,497 plots × years)													
<i>Observation (dispersion parameter)</i>	Estimate	Est.Error	I-95%	u-95%	Rhat	Bulk_ESS	Tail_ESS	Estimate	Est.Error	I-95%	u-95%	Rhat	Bulk_ESS	Tail_ESS
sd (Intercept)	2.37	0.08	2.22	2.52	1	5067	9106	2.33	0.08	2.18	2.48	1	5762	10178
<i>survey plot</i>														
sd (Intercept)	1.76	0.1	1.57	1.95	1	4645	9839	1.79	0.1	1.6	1.99	1	6278	11062
sd (cy SMA)								0.33	0.17	0.02	0.68	1	3168	7485
sd (py SMA)								0.41	0.19	0.05	0.81	1	3364	5206
sd (cy SMA × py SMA)								0.29	0.18	0.01	0.68	1	1340	4112
cor (Intercept, cy SMA)								−0.46	0.36	−0.91	0.5	1	8680	10513
cor (Intercept, py SMA)								0.59	0.28	−0.14	0.94	1	7427	9848
cor (cy SMA, py SMA)								−0.13	0.4	−0.83	0.68	1	6588	10574
cor (Intercept, cy SMA × py SMA)								−0.2	0.45	−0.87	0.72	1	7096	11799
cor (cy SMA, cy SMA × py SMA)								−0.1	0.44	−0.83	0.77	1	5452	11044
cor (py SMA, cy SMA × py SMA)								−0.28	0.45	−0.91	0.7	1	3672	8912
Fixed effects														
Intercept	9.13	0.17	8.81	9.46	1	5990	9947	9.12	0.17	8.8	9.46	1	7116	10977
cy SMA	0.54	0.28	0	1.09	1	6902	9892	0.34	0.31	−0.26	0.94	1	8259	13232
py SMA	0.53	0.28	−0.01	1.08	1	6910	10686	0.67	0.32	0.03	1.3	1	8330	13583
cy SMA × py SMA	−0.47	0.33	−1.11	0.18	1	6722	9982	−0.48	0.37	−1.19	0.25	1	8567	12960
Random effect parameters	Broadleaves (N = 33,112)													
<i>Observation (dispersion parameter)</i>	Estimate	Est.Error	I-95%	u-95%	Rhat	Bulk_ESS	Tail_ESS	Estimate	Est.Error	I-95%	u-95%	Rhat	Bulk_ESS	Tail_ESS
sd (Intercept)	1.92	0.07	1.79	2.07	1	4387	8105	1.9	0.07	1.76	2.05	1	4976	9797
<i>survey plot</i>														
sd (Intercept)	1.52	0.08	1.36	1.69	1	5479	9666	1.53	0.09	1.37	1.7	1	6787	11830
sd (cy SMA)								0.17	0.13	0.01	0.48	1	4213	6170
sd (py SMA)								0.26	0.16	0.01	0.6	1	1749	5469
sd (cy SMA × py SMA)								0.2	0.14	0.01	0.52	1	1701	2664
cor (Intercept, cy SMA)								0.01	0.42	−0.78	0.78	1	17776	14317
cor (Intercept, py SMA)								−0.03	0.36	−0.74	0.7	1	13269	11066
cor (cy SMA, py SMA)								0.03	0.44	−0.8	0.81	1	3017	6991
cor (Intercept, cy SMA × py SMA)								0.03	0.38	−0.72	0.75	1	16426	12320
cor (cy SMA, cy SMA × py SMA)								−0.17	0.46	−0.9	0.75	1	3210	5773
cor (py SMA, cy SMA × py SMA)								−0.04	0.44	−0.82	0.79	1	4903	8401
Fixed effects														
Intercept	8.45	0.15	8.17	8.74	1	5313	10108	8.44	0.15	8.15	8.73	1	6637	10846
cy SMA	0	0.28	−0.56	0.55	1	6457	9583	−0.12	0.31	−0.75	0.48	1	7484	11543
py SMA	0.32	0.29	−0.26	0.89	1	6365	9822	0.16	0.32	−0.49	0.78	1	7302	11755
cy SMA × py SMA	−0.13	0.33	−0.76	0.52	1	6312	9663	0.02	0.36	−0.67	0.74	1	7368	11317
Random effect parameters	Minor tree species (N = 55,165)													
<i>Observation (dispersion parameter)</i>	Estimate	Est.Error	I-95%	u-95%	Rhat	Bulk_ESS	Tail_ESS	Estimate	Est.Error	I-95%	u-95%	Rhat	Bulk_ESS	Tail_ESS
sd (Intercept)	2.06	0.03	1.99	2.12	1	9018	13275	2.03	0.09	1.86	2.21	1	8487	12530
<i>survey plot</i>														
sd (Intercept)	1.38	0.04	1.3	1.47	1	9048	14530	1.43	0.12	1.21	1.67	1	8198	12805
sd (cy SMA)								0.19	0.14	0.01	0.52	1	7843	11840
sd (py SMA)								0.22	0.15	0.01	0.57	1	5066	10263
sd (cy SMA × py SMA)								0.33	0.17	0.03	0.66	1	2323	6177
cor (Intercept, cy SMA)								0.06	0.43	−0.77	0.82	1	20038	14830
cor (Intercept, py SMA)								−0.01	0.41	−0.79	0.77	1	22223	14741
cor (cy SMA, py SMA)								0.12	0.45	−0.76	0.87	1	7746	14097
cor (Intercept, cy SMA × py SMA)								0.26	0.32	−0.49	0.81	1	16371	11897

Table 2. (Continued)

	Varying intercept model					Varying slope model								
cor (cy SMA, cy SMA x py SMA)						-0.11	0.45	-0.86	0.77	1	2926	7116		
cor (py SMA, cy SMA x py SMA)						-0.04	0.44	-0.82	0.79	1	4076	8280		
Fixed effects														
Intercept	7.66	0.07	7.53	7.79	1	11469	15236	7.85	0.18	7.5	8.21	1	10431	14613
cy SMA	0.06	0.16	-0.25	0.37	1	14945	13937	-0.01	0.44	-0.89	0.85	1	9627	12249
py SMA	0.21	0.16	-0.1	0.52	1	14985	15121	0.24	0.45	-0.65	1.09	1	9295	12615
cy SMA x py SMA	-0.13	0.18	-0.48	0.22	1	14761	14716	-0.07	0.5	-1.02	0.93	1	8744	12096

Shown are group-level effects and population-level effects for the varying intercept model (left side) and the varying slope model (right). Parameters whose 95% CI exclude zero are shown in bold. l-95% and u-95% are lower and upper 95% CI, respectively. Rhat, Bulk ESS and Tail ESS displays the convergence diagnostic and indicates whether MCMC chains have mixed. More details on this can be found in Vehtari et al. (2017).

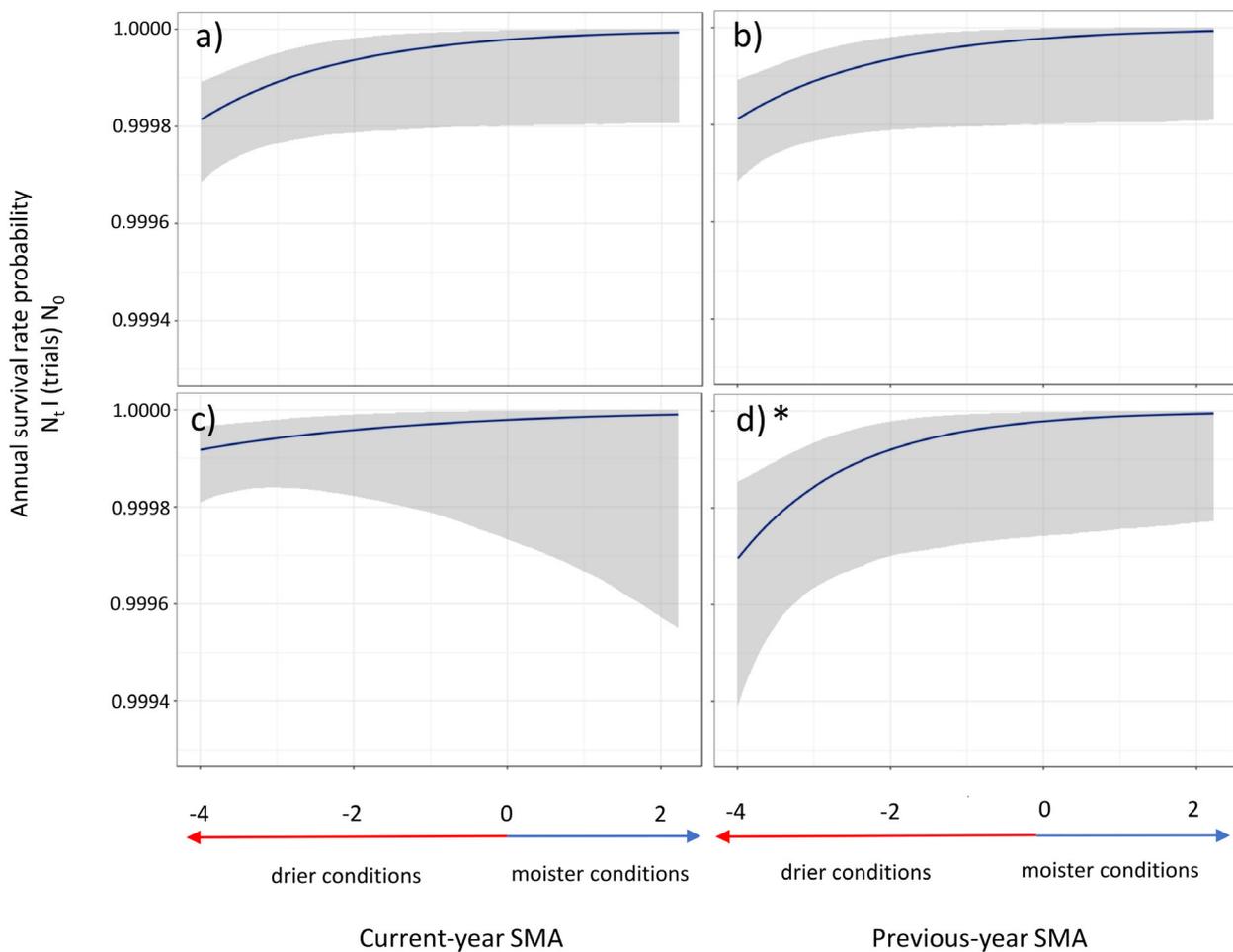


Fig. 5. Graphical summary of the GLMM. Shown are the posterior probabilities of current-year soil moisture anomaly and previous-year soil moisture anomaly in conifers for (a, b): varying intercept model and (c, d): varying slopes model. Grey shaded areas show the 95% CI and blue lines show the mean of the posterior distribution.

either on remotely sensed mortality estimates at stand level and monitoring data (Neumann *et al.* 2017; Senf *et al.* 2020), changes in vegetation indices due to anomalies in atmospheric circulation patterns (Buras *et al.* 2020), or on ecosystem carbon dioxide fluctuations due to water shortage

(Ciais *et al.* 2005). In this study, we added an important dimension based on single-tree ground observations, which cover the entire European continent in a systematic fashion. Moreover, within our study period two prominent droughts of extreme magnitude (2003, 2018/19) have occurred,

allowing us to more thoroughly study mortality patterns caused by global change-type droughts. We observed clear and significant trends for almost all major tree species which strongly suggest that European forests are currently facing unprecedented mortality rates coupled to decreasing trends in soil moisture anomaly. Hence, we can overall corroborate recent studies, which found increasing mortality in European forests (Senf *et al.* 2020), while also providing additional evidence that increased mortality is not occurring solely due to increased harvesting and logging, but indeed also as a consequence of progressively drier growing conditions. We also complement and add to earlier studies which made use of European-wide monitoring data by showing that tree mortality is temperature- and precipitation-driven (Neumann *et al.* 2017), but also sensitive to soil moisture variability. Even though the specific cause for the removal (*e.g.* utilization) of single trees was introduced relatively late in the ICP Forests level I dataset (after 2011), it nevertheless allowed us to rule out planned utilizations as natural mortality incidences, since the majority of global change-type droughts, such as 2018/19, have occurred within the last decade in which we also observed the strongest increase in annual mortality rate (Fig. 1).

Species-specific analysis revealed a higher mortality rate in major conifers (Norway spruce, Scots pine) compared to the two major broadleaved species, European beech and oak. Our results thus confirm the much higher vulnerability of these two species under climate change compared to broadleaved species (Bigler *et al.* 2006; Hanewinkel *et al.* 2013; Lévesque *et al.* 2013; Bottero *et al.* 2021). In particular, mortality peaks occurred in close temporal proximity to the two drought years, 2003 and 2018, which is largely corroborated by regional studies reporting massive dieback as a result of long-lasting droughts and heatwaves (Rosner *et al.* 2018; Schuldt *et al.* 2020; Krejza *et al.* 2021). In particular, Norway spruce occurs in many regions outside its potentially natural range, where it has been artificially planted and is well known that its shallow root system makes the species highly vulnerable when the upper soil layers show signs of desiccation (Jandl 2020). Spatial variation of spruce mortality strongly suggested that a considerable proportion of mortality occurred in plots located outside the natural distribution (lowland sites in Germany, Czech Republic, Austria), which is well in line with regional assessments reporting massive stand-replacing disturbance of pure spruce stands after the drought years 2018/2019 due to drought (*e.g.* German Federal Office for Statistics 2021). In addition, Norway spruce stands have been massively exposed to drought-induced mass outbreaks of the bark beetle *Ips tyrographus* during recent years that affected (and still affects) mainly Central Europe (Hlásny *et al.* 2021). As such, bark beetle-caused salvage logging has steadily increased during the last years in Austria, Czech Republic, Germany and Slovakia (Grodzki & Jabl-*oński* 2019; Netherer *et al.* 2019; Bárta *et al.* 2021; Obladen *et al.* 2021). Despite the fact that European beech and oak are supposed to be more resilient to warming and drought, they show a weaker but still significant trend towards higher mortality rates. Since mortality peaks in beech and oak were weakly linked to the two extreme drought years of 2003 and 2018, we presume that gradual drying and other factors such as competition and biotic agents are probably more

important drivers for mortality in these two species. Only silver fir showed no temporal trend in mortality, which probably reflects its more drought-adapted physiology and temperate occurrence (Etzold *et al.* 2019; Vitasse *et al.* 2019). The adaptation potential of silver fir and also its potential role in replacing other more vulnerable conifers such as Norway spruce is currently under debate. While silver fir is supposed to suffer from climate change mainly at the southern rear edge of its distribution, higher elevated habitats in the central distribution area may even profit from higher temperatures (Sánchez-Salguero *et al.* 2017). In addition, the pooled set of minor tree species in Europe are currently showing the same increasing trend as some of the major tree species. There has indeed been more and more evidence at regional scale reporting dieback in several minor tree species due to warming and drought, such as in Austrian pine (Jankovský & Palovčíková 2003) and Aleppo pine (Preisler *et al.* 2019). When considered together, this very concerning trend calls for more attention to climate conservation of minor tree species, since they account for a significant share of biodiversity in European forest ecosystems.

Time series of mortality rates and the soil moisture anomaly were asynchronous and showed a strong lag effect for almost all species (Fig. 1); *i.e.* significant correlations between both time series occur when mortality rates are correlated with the soil moisture anomaly of the previous year (data not shown, but see Data S3). The ICP Forest dataset used in this study was corrected only for planned tree utilizations or fellings, but contains all mortality events, regardless of whether trees died directly because of drought (abiotic) or for subsequent reasons, such as bark beetle and other (biotic) causes. Therefore, we can currently only speculate that either physiological legacy effects or subsequent biotic damage caused by other disturbance agents are responsible for this pattern. Nevertheless, a closer examination of Fig. 1 does reveal that mortality patterns seem to rapidly accelerate even within the relatively short time period we have investigated, since the more extreme drought year 2018 has caused a cascade of mortality in spruce and pine compared to 2003. While the steep increase in 2018 may be attributed to direct effects of the extreme drought (*e.g.* starvation or hydraulic failure), the subsequent further increase in 2019 could have happened due to indirect effects and secondary damage to the already weakened trees. Furthermore, the year 2019 showed recurrent droughts of similar magnitude for many of the survey plots (Fig. 2). This probably indicates that additive drought effects probably combined with legacy effects may have even accelerated tree mortality in those areas. This may even lead to questioning the assumption of a linear trend in mortality rates and would probably justify assuming a weak to moderate exponential trend that could be driven by the abovementioned circumstances. However, in this study, we emphasize only distinguishing trend significance through retrospective analysis. Nevertheless, when forecasting mortality rates is desired, a more thorough understanding of the shape of the mortality curve – in particular for the most recent years – is certainly required.

In this study we also aimed to reveal the relationship between mortality/survival and the soil moisture anomaly. We hypothesized that at, European scale, and therefore across an extensive range of forest types, soils and climate zones, drought has already had a significant impact on mortality dynamics.

Indeed, this pattern emerged since model coefficients for current- and previous-year soil moisture indicated a trend of lower survival with increasing dryness. However, this relationship was statistically credible in conifers (Fig. 5) but is currently only weakly visible in broadleaves and minor tree species (Table 2). Although varying intercept and varying slope models performed almost equally in terms of (out-of-sample) model fit, the varying slopes model clearly indicated that mortality is more strongly triggered by drought in some survey plots compared to others. While we must conclude that the statistical evidence for a European-wide signal of drought-induced forest decline is indeed clearly pointing into one direction, our results also show that further investigation of the non-uniform response among plots is necessary in order to elucidate other covariates, such as competition within plots or genetic adaptation among plots, as additional explanatory drivers. The statistical analysis did also strongly underpin that modelling overdispersion matters when analysing tree mortality data across such a large spatial and temporal scale. One natural explanation for the observed excess variance could be excess mortality, which usually occurs when either critical drought thresholds are exceeded (e.g. Senf *et al.* 2020) or additive effects such as recurrent drought and insect calamities strongly determine tree mortality. The frequency of such events has indeed increased in many spruce-dominated survey plots in recent years, suggesting that this can have a strong influence on the modelling framework and subsequently on interpretation of predictors. Moreover, the finding that previous-year drought seems to be a stronger predictor for mortality compared to current-year drought (Fig. 5) could be a valuable starting point for future analyses at more regional levels, since carry-over and legacy effects in trees indeed play a key role in mortality. Such mechanisms could be caused, for instance, by hydraulic deterioration and cavitation fatigue over multiple years (Anderegg *et al.* 2013) or could mirror increased vulnerability of drought-weakened trees to biotic agents (Netherer *et al.* 2019).

The results presented here show patterns of drought response in European forests at a macro-ecological scale. While they allow us to derive coarse temporal trends for major and minor tree species, they do not yet incorporate regional differences linked to bioclimatic zones, genecological differences and many other causes. The main objectives of this study were to demonstrate that European forests are currently experiencing unprecedented mortality and that changes in soil moisture availability are one important contributing factor among other drivers. Since planned utilizations and harvesting events as well as tree mortality incidences, which are more likely caused by non-climatic factors (e.g. ash dieback), were excluded prior to analysis, we can be relatively sure that the results presented here are not confounded by such factors. We will further employ this dataset to achieve a more advanced understanding of how drought intensity and duration are related to tree mortality and explore the variation in mortality patterns among different regions. Our approach revealed linear trends, although such trends may be partly superimposed on a few extreme event years. However, since a standardized measure of mortality was used, it is possible to derive a meaningful interpretation from our linear estimate: when calculated across all species, comprising more than three million observations,

annual mortality rate in European forests has been continuously above zero since the year 2012 (Data S4). Having said this, our results strongly suggest that the European forest landscape has probably reached a tipping point which could potentially interact with other disturbances, such as wildfires, in the near future. In a more regional approach, Etzold *et al.* (2019) found somewhat similar mortality dynamics in Switzerland to those of our study regarding mortality trend, species vulnerability and uniformity across species. However, they used a much longer time series, covering almost one century, and also included single-tree growth data in order to model competition effects in mortality dynamics. While such datasets are generally preferable, here we make use of a spatially and temporally high-resolution dataset covering the whole continent and focusing strongly on the very recently occurring millennial droughts. Nevertheless, future monitoring studies should combine data from national forest inventories with data from large-scale monitoring programmes, such as that presented here, in order to make use of the unique forces and strengths of both approaches.

CONCLUSIONS

Our results underpin a concerning trend regarding the vitality of European forests. All but one of the investigated tree species show increasing mortality trends that are coupled with decreasing soil moisture availability across the last 25 years. This highlights the need for more intensive monitoring programmes and networks to accurately evaluate adaptive management strategies for the future.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

Data S1. Locations of plots from the ICP level II crown Defoliation network per species.

Data S2. The values in the tables show the differences in expected log pointwise predictive probabilities (elpd) between random intercept and varying slope models. Both the *elpd_diff* as well as its standard error can be used as evaluation criteria for model improvement by adding more predictors.

Data S3. Cross-correlation between AMR And SMA (mean across all plots in each year). Shown is Kendalls Tau and significant correlations are shown with an asterisks. Lag time refers to The year of correlation (e.g. t-0 AMR and SMA of the same year, t-1 AMR to SMA of the previous year).

Data S4. Annual mortality rate and number of plots in each year Calculated across all species. The red line shows the mean amr and the blue area the 95% CI. Dashed red lines show the centred Mean of the standardized amr and the year 2012, respectively.

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