



The cumulative impacts of droughts and N deposition on Norway spruce (*Picea abies*) in Switzerland based on 37 years of forest monitoring

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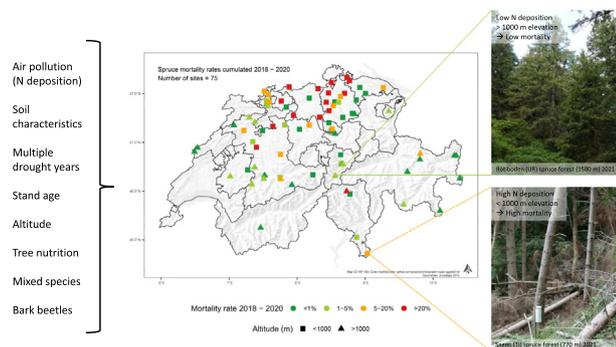


HIGHLIGHTS

- Long-term trend of Norway spruce mortality was analysed.
- Drought (ETa/ETp) and N deposition were the most important factors.
- Drought affected spruce mortality up to a lag effect of three previous years.
- High N deposition amplified negative drought effects and imbalanced tree nutrition.
- Bayesian change-point regression was used to estimate a critical threshold of N.

GRAPHICAL ABSTRACT

Factors affecting spruce mortality and map displaying cumulative spruce mortality rates from the drought period 2018–2020 in the permanent forest observation sites of Switzerland. The photographs are taken from the annual field observations monitoring tree health status and mortality in 2021.



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ABSTRACT

Norway spruce is one of the most important tree species in Central Europe, however, it is facing major problems with recent droughts. In this study we present 37 years (1985–2022) of long-term forest observation data on 82 different forest sites across Switzerland including 134'348 tree observations. The sites consist of managed spruce or mixed forest stands with beech (*Fagus sylvatica*) and comprise large gradients in altitude (290–1870 m), precipitation (570–2448 mm a⁻¹), temperature (3.6–10.9 °C) or total nitrogen deposition (8.5–81.2 kg N ha⁻¹ a⁻¹).

Long-term tree mortality has increased more than fivefold due to the multiple drought years 2019, 2020 and 2022, which is more than double the increase following the 2003 drought. We used a Bayesian multilevel model including three years of lagged drought indicator to predict spruce mortality. Besides stand age, the most important factors were drought and N deposition. Especially under drought conditions spruce mortality was increased on sites with high N deposition. Moreover, N deposition increased the imbalance of foliar phosphorus concentrations, with negative impacts on tree mortality. Mortality was increased by a factor of 1.8 in spruce compared to mixed beech and spruce stands. Stands with high mortality rates showed previously an increased proportion of trees with damaged crowns, especially after the droughts of 2003 and 2018.

Taken together, we found evidence of an increase in spruce mortality droughts amplified under high N depositions. The perennial drought of 2018–2020 resulted in a cumulative spruce mortality of 12.1 % (564 dead trees in 82 sites) in only three years. With a Bayesian change-point regression framework we estimated a critical empirical load

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for nitrogen of $10.9 \pm 4.2 \text{ kg N ha}^{-1} \text{ a}^{-1}$, which is in line with current thresholds, above which future plantings of spruce in Switzerland may not be a sustainable option due to the observed interaction between drought and nitrogen deposition.

1. Introduction

Anthropogenic influences have unprecedented impacts on global ecosystems, also on forests, where climate change is causing loss and degradation around the globe (IPCC, 2021). There has been increasing evidence of increased drought stress due to more intense and frequent droughts, causing growth reduction or even tree mortality (Arend et al., 2021; McDowell et al., 2020; Schuldt et al., 2020). Extended drought periods can be potent drivers of ecosystem collapse and pose a major future threat to global biodiversity (Godfree et al., 2019).

Independent of the emission scenario, temperatures are expected to increase by the end of this century (Kotlarski et al., 2022). Precipitation will probably face a seasonal shift from summer to winter, although model uncertainty is still high. Furthermore, it is expected that the wet-day frequency will decrease during the vegetation period (Kotlarski et al., 2022) and high temperatures will increase evapotranspiration, which will favour drought stress during summer. There were several severe summer drought events in Europe since 2018 with extreme and persistent soil moisture deficits (Peters et al., 2020) and intense heat and low precipitation during the vegetation periods, resulting in a combined perennial heat event (Ault, 2020). Across Europe, the combined heat and drought in 2018 until 2020 was even more severe than the “century” drought in 2003 (Rakovec et al., 2022), with unprecedented effects on mortality of major tree species (Schuldt et al., 2020), especially considering the long-term impacts of the multiple drought years.

Besides climate change, the anthropogenic deposition of nitrogen (N) is one of the major threats to biodiversity (Butchart et al., 2010) and soil quality (Bobbink et al., 2010) that exceeds the planetary boundaries considerably (Steffen et al., 2015). Global anthropogenic N deposition rates doubled over the last century (Fowler et al., 2013) and are expected to increase by a factor of 2.5 over the next century (Lamarque, 2005). This will exacerbate detrimental impact on natural ecosystems such as forests, for example by increasing soil acidification with its effects on soil quality (Bünemann et al., 2018) or negative impacts on below and aboveground biodiversity (Bowman et al., 2008). These impacts on forest ecosystems will have cascading impacts on global cycles for water and carbon and also on ecosystem services (Bonan, 2008; Mori et al., 2017).

Moreover, temperate European forests are prone to climate-change driven disturbances such as droughts, windthrows, fires or insect outbreaks, due to their vulnerability to sudden changes (Forzieri et al., 2021; Seidl et al., 2017). Increasing disturbance events have been recorded worldwide for example for drought (Allen et al., 2015; Millar and Stephenson, 2015) or insect outbreaks (Kautz et al., 2017). It is expected, that temperate forests will be prone to increased risks by droughts and wildfire (Machado Nunes Romeiro et al., 2022). Especially for droughts, the underlying mechanisms involved in tree mortality or survival are complex (Choat et al., 2018; Hartmann et al., 2018) and still not fully understood (Walthert et al., 2021). However, there is increasing evidence, that hydraulics are often more important than other factors such as carbon balance (Körner, 2015), especially under drought situations (Arend et al., 2021), where a disruption of the capillary continuum from the soil to the root can occur (Körner, 2019), leading to cavitation (Arend et al., 2021; Choat et al., 2019).

Responses of trees are species specific, depending among other factors on stand characteristics or forest structure (Vinceti et al., 2020). However, Norway spruce (*Picea abies* (L.) Karst.) is one of the most threatened commercial tree species in Central Europe (Krejza et al., 2021). Its natural distribution covers large parts of northern Europe and mountainous regions of central Europe (Westin and Haapanen, 2013). In Switzerland Norway

spruce is native to the northern parts of the alps and has been introduced in the lowland areas at the end of the 17th and beginning of the 18th century. Despite the recent decline in spruce, it is still economically the most important tree species in Switzerland (Cioldi and Allgaier Leuch, 2020). The major reasons for the decline are the susceptibility to bark beetle attack under drought stress and air pollution such as ozone and N deposition (Christiansen and Horntvedt, 1983; Jones et al., 2011). The low tolerance to drought stress (Krejza et al., 2022; Vitali et al., 2017) can be caused by the shallow root system (Caudullo et al., 2016), particularly in lower lying areas, beyond the ecological boundaries of Norway spruce (e.g. Thorn et al., 2019). The intensified and longer summer droughts in Europe are expected to increase drought induced stress and lower defence against plant pathogens such as bark beetles (Netherer et al., 2015). Moreover, it is expected that drought triggered forest mortality can occur quickly when exceeding physiological thresholds or in association with insect pests on pre-damaged forests. However, such cross scale mortality processes in forests still remain poorly understood (Allen et al., 2010).

Our main objective is to disentangle the factors influencing spruce mortality using a continuous long-term data set of 37 years. We hypothesized that tree mortality has increased not only due to more frequent and intense droughts, but also due to chronic high N deposition in some of the study areas. Furthermore, we were interested in quantifying the lagged effect of drought on spruce mortality due the series of multiple drought years 2003–2006 and 2018–2020 and again in 2022. Moreover, we used the long-term observations of tree mortality to estimate a critical empirical load for nitrogen with a Bayesian change-point regression framework.

2. Material and methods

2.1. Study sites

The study sites are part of the long-term Intercantonal Forest Observation Program in Switzerland, monitoring forest health conditions of managed forest sites since 1984 (Braun et al., 1999, 2021a). The total number of study sites is 82 (Table 1) with 56 pure Norway spruce and 26 mixed with European beech forest stands. The actual number of sites per year (Fig. A.5) changed slightly during the observation period of 37 years, for example due to storm damages such as “Lothar” in 1999, which caused serious uprooting of spruce and beech trees (Braun et al., 2003) or tree mortality due to recent drought stress or the inclusion of new forest observation sites provided by the local forest services and cantonal offices. The study sites cover large gradients (Table 1) in altitude (290–1870 m, median = 780 m), annual precipitation (570–2448 mm a⁻¹, median = 1297 mm a⁻¹), mean annual temperature (3.6–10.9 °C, median = 8.4 °C) total nitrogen deposition (8.5–81.2 kg N ha⁻¹ a⁻¹, median = 26 kg N ha⁻¹ a⁻¹) and also ozone (Braun et al., 2017) or soil chemistry (see monthly measured soil chemistry data in Braun et al., 2020b).

2.2. Tree mortality rates and crown defoliation

All study sites of *Picea abies*, including stands mixed with *Fagus sylvatica* (n = 26) (Fig. 1), were evaluated annually for crown defoliation and mortality in the same corresponding week during summer (July–August). In addition, the number of dead trees, including sanitary cuts due to bark beetle attacks, were recorded and related to the population of the previous year. During the 37 years of forest monitoring (1985–2022) 134'348 spruce trees were assessed. Crown defoliation was assessed annually in steps of 5 % with the two damage classes damaged >25 % and strongly damaged >60 % according to the criteria following ICP Forests (2016).

2.3. Soil properties

Soil physico-chemical characteristics were measured in 2016 using pooled soil samples (6–8 sample points) for each soil horizon across the study sites using an Edelmanager (Eijkelkamp). They were air dried and sieved <2 mm prior to the analysis. Soil pH was measured in fresh samples in a 1:2.5 suspension with CaCl₂ and the amount of exchangeable base cations, further referred as base saturation, was determined in NH₄Cl extracts following Trüby and Aldinger (1984). The elemental concentrations of the cations were measured using a microwave-plasma-atomic-absorber (MP-AES 4200, Agilent). Base saturation and soil pH were averaged for the uppermost 40 cm of the soil. C/N ratios were calculated as the total C and N concentrations for the uppermost soil horizon (for more details see Braun et al., 2020a).

2.4. Foliar nutrients

Foliar nutrients were measured in shoots harvested by helicopter from the top crown of the same 8 trees per site every 4 years in July, starting in 1984. Ten leaves per tree were dried at 80 °C and weighed. Leaves were dried, ground and analysed for N, P, K, Ca, Mg, and Mn according to Walinga et al. (1995). The nutrient contents in the leaves were obtained by multiplication of the dry weight per leaf with the concentration. A full description is given in Braun et al. (2020a).

2.5. Total N deposition and drought indicators

Total N deposition was modelled according to Rihm and Künzle (2019) including wet and dry deposition compounds. This emission based model produces comparable estimations of total N deposition at a high spatial resolution compared to bulk and throughfall measurements (Braun et al., 2022; Thimonier et al., 2019). Drought indicators were calculated using the hydrological model WaSiM-ETH (Schulla, 2019) with data from the eight nearest climate stations from the Federal Office of Meteorology and Climatology (MeteoSwiss), interpolated for each site as described in Braun et al. (2014). We selected as drought indicator the mean ratio between actual and potential evapotranspiration (ET_a/ET_p) of the vegetation period, according to Braun et al. (2015). Start and end point of the vegetation period were calculated for each site with phenological data from MeteoSwiss adjusted for altitude as described in Braun et al. (2017). The ratio of ET_a/ET_p can be considered as an indicator for actual drought stress. Values around 1, with unrestricted transpiration, indicate moist conditions and lower values drier conditions.

2.6. Data analysis

All statistical analyses were performed using R 4.1.3 (RCore Team, 2022). The respective codes are given in the RMarkdown appendix. Data including models are given in Tresch et al. (2023). The map with the study sites (Fig. 1) was created using QGIS 3.22.3 (QGIS Development Team, 2022).

2.6.1. Time series correlation of spruce crown damage and spruce mortality

A cross-correlation of the two time series of spruce crown damage, assessed as annual mean values of spruce trees with >25 % crown defoliation and annual mean spruce mortality from all study sites (Fig. A.8) was carried out using the cross-correlation function (*base::ccf*) following Hyndman and Athanasopoulos (2021). The correlation of the two time series data (Fig. A.9) was plotted with time series plots using the function *timetk::plot_time_series* from Dancho and Vaughan (2022) and calculated using Spearman's rank correlation test (Fig. A.11).

2.6.2. Lag effects of drought indicators

We used a polynomial lag approach in order to assess possible lagged effects of the drought indicator. Regression coefficients of the drought variables of the current and the five preceding years were constrained to four

degrees of freedom by a third degree polynomial of the lag. The four resulting weighted means of the lagged drought variables (i.e. defined by constant (u₀), linear (u₁), quadratic (u₂) and cubic (u₃) weights, respectively) were then orthogonalized using the *brms::brm* function (Bürkner, 2017). For model comparison we used the Bayesian leave-one-out cross-validation (LOO) estimate of the expected log point wise predictive density (ELPD) (Vehtari et al., 2017). The best model was chosen according to the smallest ELPD (Table B.6). For the final spruce mortality model weighted lag effects were calculated for the three lagged years according to the polynomial lag model (Fig. A.13).

2.6.3. Spruce mortality model

Mortality of spruce trees was assessed by the odds of dead trees, defined as a binomial factor of the number of dead trees and the number of living trees. The number of dead trees is recorded once a year in summer and includes felled trees due to bark beetle damages. We used a Bayesian generalised multivariate multilevel regression with a zero-inflated binomial response and a logit-link to predict the probability of annual spruce mortality with the *brms::brm* function (Bürkner, 2017), with sites as random effects. All predictors were centred and scaled and default priors were used from the *brms* package (Bürkner, 2017) following Koontz et al. (2021). Predictor variables (Table B.9) were pre-selected according to expert knowledge and previous studies about spruce tree mortality in Switzerland (Braun et al., 2015, 2021b; Etzold et al., 2016). We ensured that the selected predictor variables were correlated with a Pearson correlation coefficient <0.65 (highest correlation coefficients = -0.51 between N deposition and stand age, with a Variation inflation factor (VIF) < 1.8, see Fig. A1) following Brandl et al. (2020). The Watanabe-Akaike information criterion (WAIC) was used for backwards model selection, with a model improvement threshold of Δ WAIC \leq 5 (Carlin and Louis, 2008). Interaction and non-linearity of the predictor variables were tested. In order to ensure the model selection a Bayesian model evaluation was applied using expected log point wise predictive density (ELPD) calculated by leave-one-out cross-validation (*loo*) (Vehtari et al., 2017). We used 10,000 iterations and 4 chains for the model fitting and confirmed chain convergence by ensuring that Rhat values were smaller than 1.1 (Brooks and Gelman, 1998) and that the bulk and tail effective sample sizes (ESS) for each estimated parameter were >100 times the number of chains (Table 2). Posterior predictive checks were calculated with the function *bayesplot::pp_check* (Gabry et al., 2019) to visually confirm model performance by overlaying the density curves of the predicted and observed tree mortality rate using 150 random samples from the model fit (Fig. A2). A Bayesian version of the variance explained (R²) by the fixed effects was calculated according to Gelman et al. (2019). Effect estimates of the posterior distribution are presented as the mean (point estimate) including lower and upper 95 % credible intervals (CI) (Gelman and Greenland, 2019) following Korner-Nievergelt et al. (2015). Due to the confounding effects of altitude with important predictors (see correlation matrix in Fig. A1) we fitted a second Bayesian zero-inflated binomial multilevel regression model with altitude as fixed effect (Table B.1) and a third Bayesian model with forest stand types as fixed and sites as random effects (Table B.3).

2.6.4. Bayesian change-point regression

We used a Bayesian change-point regression (BCR) according to Roth et al. (2022) to model critical loads for nitrogen based on the long-term observation data of spruce mortality. Similar to the model used for spruce mortality, we included the main influencing factor in the BCR of the lagged drought indicator, foliar P and K contents, soil base saturation, stand age, year and the interaction of N deposition and the lagged drought indicator (Table B.2).

3. Results

3.1. Mortality rate

During the 37 years of permanent forest monitoring 1'350 trees from 134'348 died. The long-term average mortality rate 1985–2022 was

Table 1

Site characteristics of long-term observation of Norway spruce from the Intercantonal permanent observation program. Conifer describes the proportion of conifer species. Stand age describes the approximate age of study site. Start and end dates of the forest monitoring are given and if the study site is a mixture between *Fagus sylvatica* and *Picea abies* (forest stand). Precipitation and mean annual temperature (MAT) were calculated as a site specific long-term mean from annual data 1984–2021. The C/N ratio is defined by the total C and N concentrations in soil samples from the top soil layer. Base saturation values were obtained from the most recent soil sampling campaign in 2016. Total N deposition are modelled values including dry and wet N depositions from the current estimations by Rihm and Künzle (2019). Mortality rate is defined as the average annual mortality rate per site and the number of died trees is the cumulative number of died trees per site.

No.	Site name	Canton	Forest type	Conifer (%)	Stand age	Start	End	Altitude (m a.s.l.)	MAT (°C)	Precipitation (mm a ⁻¹)	C/N ratio	Soil base saturation (%)	ph (CaCl ₂)	N deposition (kg N ha ⁻¹ a ⁻¹)	Mortality rate (%)	Trees died
1	Aeschau	BE	Mixed	85	302	1990		940	7.6	1488	26	26.9	3.8	32	0.26	3
2	Aeschi	SO	Mixed	52	113	1995		510	9.5	1126	21.2	9.5	3.7	36	0.88	13
3	Albis	ZH	Mixed	50	128	1984		780	8.3	1276	13.4	99.9	6.2	27	0.82	21
4	Allschwil	BL	Conifer	98	47	1997		350	10.9	893	14	77.5	4.3	27	0.9	16
5	Alvaneu	GR	Conifer	100	181	1985		1560	5.5	1000	29.8	98.6	6.1	11	0.12	5
6	Appenzell	AI	Conifer	99	120	2001		1000	8.3	1611	15.4	11.5	3.6	36	0.12	2
7	Bachtel Fi	ZH	Conifer	99	132	1984		1040	7.6	1734	24.8	10.5	3.6	34	0.73	17
8	Beromünster	LU	Conifer	99	64	2015		640	9.2	1148	23.1	84.4	5	81	2.15	10
9	Bichelsee	TG	Conifer	95	58	2006		740	8.7	1391	17.6	99.9	5.6	33	0.08	1
10	Braunau	TG	Conifer	98	129	2006		710	8.9	1152	19.8	31	3.5	41	0	0
11	Bremgartenwald	BE	Mixed	57	162	1989	2020	530	9.6	1023	18	59.7	4.2	30	2.04	35
12	Brislach	BL	Conifer	86	111	1996		435	10.2	1020	23.3	18.1	3.8	26	0.75	17
13	Brusio	GR	Conifer	100	169	2016		1710	4.2	1079	22	25	3.4	10	0	0
14	Brütten	ZH	Conifer	96	117	1995		620	9	1231	14.2	99.4	5.6	34	0.55	15
15	Bürglen	UR	Conifer	99	85	2015		640	10	1505	22.2	99.3	4.8	32	0.18	1
16	Büsserach	SO	Conifer	75	149	1996		600	9.4	1168	16.7	100	7.1	23	0.1	2
17	Busswil	BE	Conifer	100	127	1999		600	9.3	1168	18.9	8.9	3.7	35	0.06	1
18	Cademario	TI	Conifer	100	49	2001		780	9.5	1880	19.9	14.7	3.8	29	0.11	2
19	Courtelary	BE	Conifer	81	152	1996		1110	7.2	1390	14.5	99.9	6.5	20	0.05	1
20	Davos	GR	Conifer	100	126	1985		1620	3.6	1070	25.3	31.4	3.6	13	0.07	3
21	Diessenhofen	TG	Conifer	90	98	2006	2020	520	9.5	895	20.8	12.5	3.5	23	7.29	72
22	Erlenbach Nordhang	BE	Conifer	100	252	1989		1430	5.3	1453	25.9	99.8	6.9	17	1.31	36
23	Erlenbach Südhang	BE	Conifer	100	202	1989		1430	6.6	1429	24.5	99.9	6.6	19	0.72	19
24	Evolène	VS	Conifer	100	131	2010		1360	6.1	796	24.3	97.2	5.7	9	0.23	2
25	Faido	TI	Mixed	64	195	1996		800	9.4	1432	24.2	59	4.5	25	0	0
26	Frienisberg	BE	Mixed	57	142	1989		725	8.5	1174	21.2	24.4	4	28	0.19	5
27	Grenchenberg	SO	Mixed	45	186	1996		1220	6	1418	15.1	99.8	5.1	19	0.24	4
28	Grosswangen	LU	Conifer	100	96	2015		600	9.6	984	21.9	12.9	3.5	52	0	0
29	Gubrist	ZH	Conifer	100	127	1995	2006	610	9	1163	18.8	72.7	4.1	33	6.27	114
30	Gubrist neu	ZH	Conifer	90	63	2007	2018	610	9.5	1109	30.9	73.2	4.3	27	12.94	60
31	Habsburg Kontrolle	AG	Conifer	91	91	2003		430	10.1	1002	20.1	9.8	3.7	28	0	0
32	Hauenstein	SO	Mixed	53	136	2008		810	8.4	1179	14.6	99.9	7.3	24	0	0
33	Himmelried neu	SO	Conifer	97	77	2009		680	9.3	1138	17.4	99.8	6.1	21	0.75	7
34	Hinwil	ZH	Conifer	100	127	1995		650	9	1448	15.4	95.6	5.1	41	0.19	5
35	Hölstein Exp	BL	Mixed	37	150	2018		540	10.3	989	22	99.7	7.1	21	11.56	26
36	Höri	ZH	Conifer	76	142	1995	2019	460	9.7	1012	22	27.3	3.8	31	5.03	106
37	Innertkirchen	BE	Conifer	92	182	1994		950	7.2	1592	25.7	90.9	4.7	18	0.1	2
38	Jaunpass	BE	Conifer	100	192	1989		1520	5.1	1498	16.6	45.3	3.9	17	1.67	41
39	Klosters	GR	Conifer	100	126	1985		1360	5.3	1332	28.8	99.9	6.5	15	0.23	5
40	Krattigen	BE	Conifer	100	212	1989		1410	6	1376	24.2	82.1	4.4	16	0.52	10
41	La Brévine	NE	Conifer	97	191	2001		1090	5.7	1382	23	99.8	5.3	22	0.8	11
42	Le Châtelard	FR	Conifer	98	119	2006		1010	7.5	1572	29.3	24.2	3.7	33	0.23	3

43	Leissigen	BE	Mixed	37	162	1989	2020	685	8.6	1262	15.1	99.7	6.4	21	0.17	2
44	Les Verrières	NE	Mixed	76	153	2001		1110	6.3	1406	24.3	98.7	5	22	0.21	3
45	Lurengo	TI	Conifer	100	106	1996		1620	5.3	1768	26.2	23.8	3.7	15	1.6	22
46	Magglingen	BE	Mixed	56	132	1989	2020	925	8.1	1352	15	99.3	5.3	21	0.3	6
47	Männedorf	ZH	Conifer	98	122	1995		590	9.3	1397	10.4	92.6	4.7	35	0.72	24
48	Mellikon	AG	Mixed	67	156	1995		530	9.7	1124	20.4	22.9	3.8	27	1.47	19
49	Mettendorf	TG	Mixed	59	73	2006		610	9.2	1032	14.5	99.9	6	25	0	0
50	Möhlín Unterforst	AG	Conifer	89	81	1996		290	10.6	1007	17.5	15.6	3.7	27	1.74	29
51	Morgartenberg	ZG	Mixed	63	123	1985		980	7.3	1819	25.1	48.9	4	34	0.19	4
52	Mühledorf	SO	Mixed	53	101	1995		620	9.3	1122	17.5	36.2	3.8	27	0.23	4
53	Muri	AG	Conifer	82	129	2000	2020	490	10.1	1055	26.5	10.3	3.8	38	6.99	98
54	Muri	AG	Conifer	100	121	1995	2000	490	9.8	1096	18.9	14.5	4	40	0	0
55	Oberschrot	FR	Mixed	78	145	2006		950	8.1	1261	17.2	12.1	3.5	25	0.09	1
56	Rafz	ZH	Conifer	91	117	2000	2022	540	9.7	1015	19	13.9	3.7	24	4.35	134
57	Rafz	ZH	Conifer	100	122	1984	2000	540	9	984	20	18.3	4.1	28	0	0
58	Ramosch	GR	Conifer	100	132	2016		1520	5.7	769	22	99.8	7.4	10	0	0
59	Riggstäfeli	UR	Conifer	100	239	1991		1740	4.8	1706	23.5	15.9	3.3	13	0.13	3
60	Romanshorn	TG	Conifer	100	59	2006		440	10.1	1011	19.3	53	4	46	0.58	6
61	Rötiboden	UR	Conifer	100	329	1991		1580	4.5	1493	23.5	20.4	3.2	12	0.09	2
62	Sagno	TI	Conifer	94	52	1996		770	9.6	1754	21.8	32.3	3.6	33	2.01	30
63	Santa Maria	GR	Conifer	100	268	2001		1760	3.7	802	29.1	62.6	3.3	9	0	0
64	Scheidwald oben	BE	Conifer	100	142	1989	2020	1180	6.7	1443	27.6	11.1	3.3	23	0.15	3
65	Scheidwald unten	BE	Conifer	100	232	1989		1170	6.8	1435	27.9	3.9	3.6	23	0.69	15
66	Silenen	UR	Conifer	100	179	1991		630	9.2	1449	23.5	96.1	5.2	24	0.88	16
67	Souboz	BE	Mixed	31	152	1989	2020	1120	6.2	1475	14.8	99.9	6.4	20	0.23	4
68	Stammheim	ZH	Mixed	52	127	1984	2020	580	9.1	954	13.9	90.4	4.5	25	1.92	33
69	Stampa	GR	Conifer	100	242	2001		1420	5.9	1219	27.1	34	3.2	13	1.04	18
70	Steinhausen	ZG	Mixed	53	113	1985		520	9.7	1315	15.3	29.6	3.9	38	0	0
71	Therwil	BL	Mixed	41	157	1984		390	10.7	909	17.3	60.8	4.1	23	1.75	16
72	Tschlin	GR	Conifer	100	113	2016		1190	7.4	712	10.8	97	3.9	11	0	0
73	Unterägeri	ZG	Conifer	94	57	2009		1140	6.8	1925	19.2	9.3	3.6	23	0	0
74	Waldiberg	UR	Conifer	99	229	1991		1220	5.9	1542	23.5	38.7	3.6	17	0.24	4
75	Wallisellen	ZH	Mixed	47	122	1984		480	9.7	1103	15.7	74.2	4.2	31	0.58	14
76	Wangen	ZH	Mixed	63	66	2007		500	10.1	1113	23.3	16.8	3.8	34	0.94	8
77	Wengernalp	BE	Conifer	100	252	1989		1870	3.7	1602	14.2	30.8	3.5	11	0.07	2
78	Winterthur	ZH	Conifer	97	122	1995	2020	530	9.4	1159	16	90.3	4.5	32	3.34	102
79	Zofingen	AG	Mixed	31	130	1995		540	9.6	1107	17.9	12.4	3.7	28	0.72	10
80	Zugerberg HG	ZG	Mixed	75	93	1987		980	7.2	1524	19.8	59.5	4.1	31	0.14	3
81	Zugerberg VG	ZG	Conifer	99	123	1999		900	7.5	1357	20.2	12	3.8	30	0.83	12
82	Zürichberg	ZH	Mixed	16	142	1984		660	8.9	1210	12.9	87.7	4.5	26	0.6	10
82 sites	56 spruce sites	67 active		96	128			755	8.8	1215	20	56	4.0	26	0.80	1350
	26 mixed sites	14 abandoned														

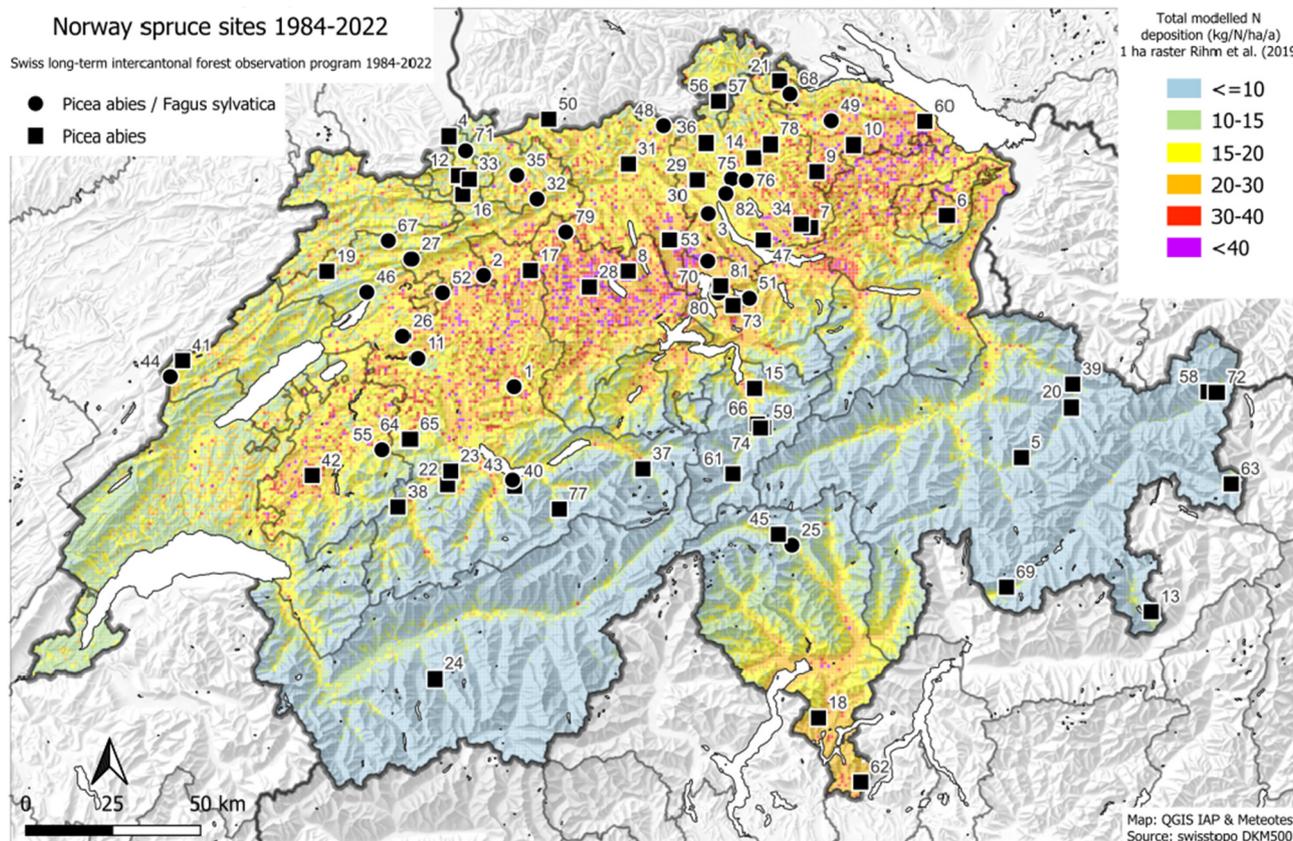


Fig. 1. Selected study sites from the Swiss long-term Intercantonal forest observation program including site number (Table 1). Forest types with the predominant tree species Norway spruce are shown as rectangles and the mixture of European beech and spruce as dots. Background: map of total N deposition according to Rihm and Künzle (2019).

0.80 % per year (Table B.5). Increased mortality rates have been recorded in recent years, with peak mortality rates of 5.1 % in 2019 and 5.3 % in 2020. This corresponds to an increase by a factor of 6.3 for 2019 and 6.6 for 2020 compared to the long-term average. The most recent drought in 2022 was comparable to the increase in mortality for the years 2019 and 2020 with a mortality rate of 4.7 %. The heat wave 2003 and the increased mortality rates in 2005 and 2006 were lower, but still 3 times larger compared to the long-term average (Fig. 2). Mortality rate was higher by a factor of 1.8 in spruce forest stands, consisting of mainly *Picea abies* tree species, compared to mixed forest stands with broadleaved species, consisting of mainly *Fagus sylvatica*. The estimated mean long-term annual mortality rate were 0.68 % for mixed forest stands and 1.0 % for spruce forest stands (Table B.4, Fig. A.14).

3.2. Correlation between spruce crown damage and spruce mortality

Time series plots of annual mean values of damaged crowns (>25 % crown defoliation) and annual mean spruce mortality averaged over all study sites show a convergence increase of both mortality and damaged crowns of spruce trees after 2003 and a strong increase after 2015 (Fig. A.9). Cross-correlation analysis suggested to include two year lag effect of spruce crown damage (Fig. A.8). We found a positive correlation (Spearman's $R = 0.37$) of spruce crown damage (average of current and two year lag) with annual mean spruce mortality rate (Fig. A.13). Spruce mortality rate was increased, if about 20 % of the spruce trees have mean annual crown damages >25 % (average of current and 2 years lag), taking into account 95 % CI of the loess regression fitting.

Table 2

Summary of posterior distribution of the Bayesian zero-inflated binomial multilevel regression model with sites as random effect and spruce mortality as response variable γ . Number of chains = 4, Number of iterations = 10,000, CI = credible intervals, ESS = Effective sample size, SD = standard deviation. Bayesian R^2 of fixed effects = 0.17 ± 0.03 . Effect plots are given in Fig. 4.

	Estimate	SD	Lower 95 % CI	Upper 95 % CI	Rhat	Bulk ESS	Tail ESS
Intercept	-4.47	0.28	-5.04	-3.93	1.00	4116	6789
Cumulative drought	1.35	0.10	1.16	1.55	1.00	12,564	13,349
N deposition	0.57	0.26	0.08	1.08	1.00	7324	9576
Soil base saturation	0.30	0.26	-0.20	0.82	1.00	4300	7738
Year	0.19	0.12	-0.04	0.42	1.00	9851	13,017
Stand age	0.62	0.29	0.05	1.21	1.00	5651	8030
Phosphorus	-0.15	0.13	-0.40	0.11	1.00	13,947	13,940
Potassium	-0.32	0.10	-0.53	-0.12	1.00	15,339	14,827
Cumulative drought:N deposition	0.49	0.13	0.24	0.75	1.00	15,190	15,080
N deposition:Phosphorus	-0.57	0.14	-0.86	-0.29	1.00	10,825	13,417

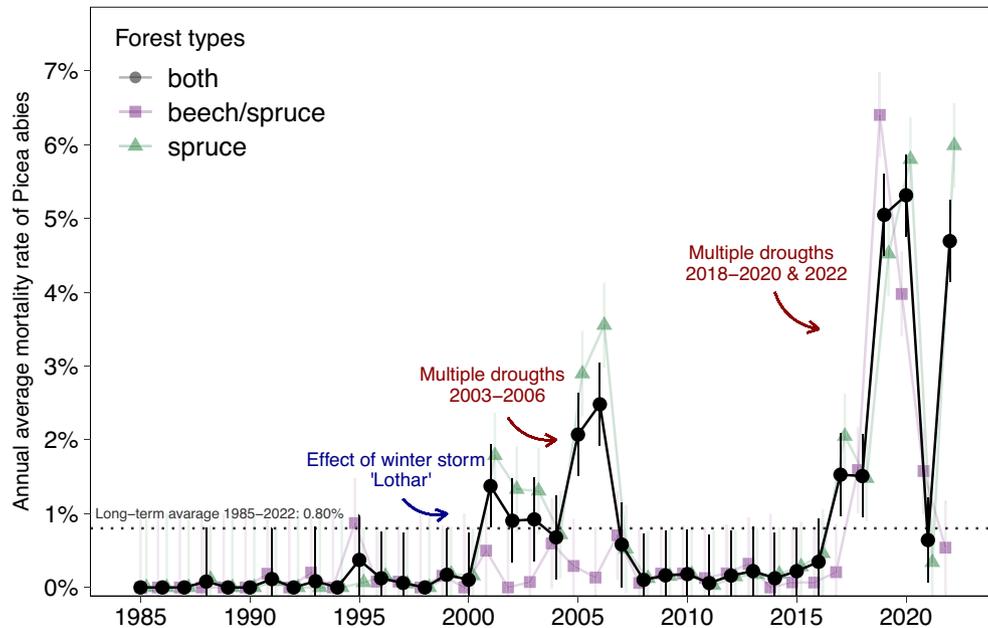


Fig. 2. Annual average mortality rates of Norway spruce in Switzerland. Annual average mortality rate of all sites is given in black and in colours according to forest types. Number of dead trees and observation sites are given in Figs. A.1 & A.2. Possible causes of observed increase in tree mortality rate are shown with arrows and text. The estimation of variance is given as 95 % CI (Table B.8).

3.3. Lagged drought effect on spruce mortality

We found strong evidence of an increasing frequency of severe droughts during the vegetation period over time in our study sites (Fig. 3). The relative change in the drought indicator ETa/ETp was strongest for the year 2018 (−13 %) and 2022 (−14 %) compared to the mean ETa/ETp ratio in all study sites (0.87) from 1985 to 2022. The best model for assessing the lagged effect of the drought indicator ETa/ETp on spruce mortality was the quadratic polynomial lag model (Table B.6). Based on the estimated lag effects (Fig. A.13) we included up to three years of lag effects in the final model, calculated as weighted cumulative drought indicator. The relative change in the three year lagged drought indicator can be found in Fig. A.7.

3.4. Driving factors of spruce mortality

The cumulative drought years had the largest effect on spruce mortality. Besides stand age, N deposition was an important factor affecting spruce mortality. Moreover, we found that drought showed an interaction with N deposition, which is shown in Fig. 4. Drought effects were more severe with increased N deposition. A second interaction with N deposition was found with foliar phosphorus concentrations. The effect of N deposition on spruce mortality was stronger if the phosphorus concentration was below the threshold for balanced nutrition according to Göttelein (2015). In addition, we found a higher probability of tree mortality with low foliar potassium concentrations, lower elevation and in pure spruce stands (Fig. A.14). Those effects were smaller by a factor of 10 compared to the other effects (Fig. 5). The effects of foliar phosphorus as well as soil base saturation were not significant (Table 2), nevertheless they were kept in the final Bayesian zero-inflated binomial multilevel regression model, because they improved the model.

3.5. Estimated critical loads of nitrogen for Norway spruce

Within the framework of the Bayesian change-point regression models (Roth et al., 2022) we estimated a mean critical empirical load for nitrogen for spruce of $10.9 \pm 4.2 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (Table B.2), taking into account the interaction with the drought indicator (Fig. 6).

4. Discussion

Increased drought periods are affecting forests worldwide. It is expected that forests will have an increased tree mortality rate in response to drought stress, also in regions with previously sufficient water supply (Allen et al., 2010; Arend et al., 2021; Pretzsch et al., 2020). The long-term forest monitoring data of managed mixed and pure spruce stands presented in this study contributes to the understanding of the extent and magnitude of climate change induced droughts and other factors affecting temperate forests.

4.1. Effects of droughts on spruce mortality

4.1.1. Cumulative drought years assessed by lagged drought indicator

Climate change scenarios for Switzerland predict that precipitation during summer will decrease by 25 % by the middle of the 21st century, while heavy rainfall events will increase (+10 %) and heat waves in particular will become more intense and frequent (Berset et al., 2020). In this study we used the ratio of actual and potential evapotranspiration (ETa/ETp) of the vegetation period as an indicator for the actual drought stress, as it was the best indicator for modelling tree mortality of temperate spruce forests (Braun, 2015). The closer this ratio is to 0, the more evapotranspiration is restricted and the greater the drought stress. Comparing the two major droughts in the last decades, we found that the average ETa/ETp ratio was even lower in 2018 and 2022 than 2003. In addition, the drought of 2018 followed drought stress in 2015 and 2017 and most importantly, was followed by droughts in 2019–2020 and 2022 (Fig. 3). The relatively wet and cold year 2021 led to a short-term decrease in mortality. However, the recent drought in 2022 led to an increase in spruce mortality almost to the same level as in 2019 and 2020. The extreme drought stress in 2018 was more severe than in 2003, not only in Switzerland, but also in other parts of Central and Northern Europe (Buras et al., 2020), causing serious damages especially to forest ecosystems (Braun et al., 2021b; Schuldt et al., 2020). Comparing the relative change to the long-term average ETa/ETp ratio in our study sites from 1985 to 2022, we found evidence of more frequent and intense droughts in recent years (Fig. 3). The polynomial lag model approach indicated that we should consider a cumulative effect of drought of the current year and additionally three years back. This cumulative impact

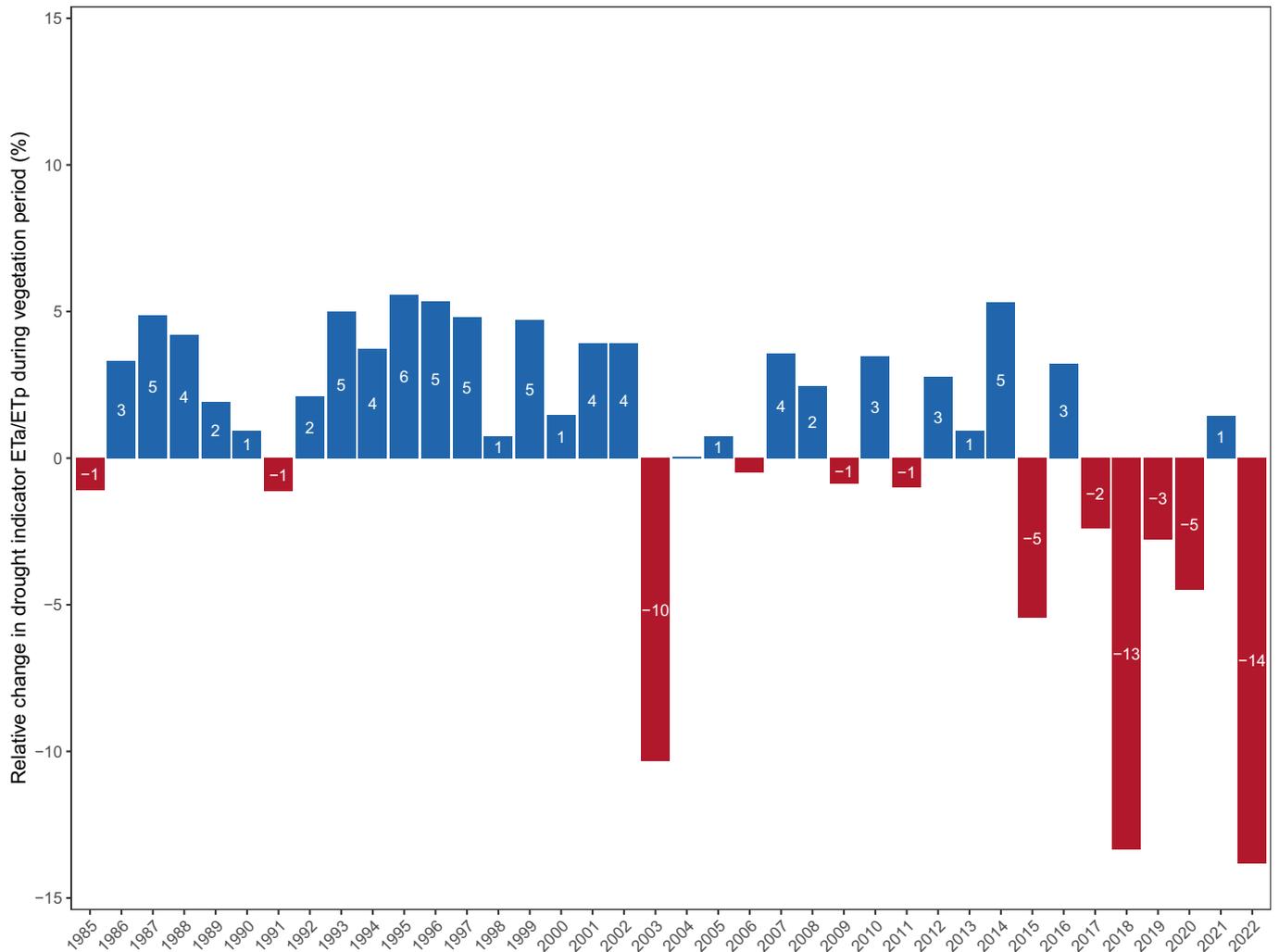


Fig. 3. Relative changes in the drought indicator ETa/ETp during the vegetation period compared to the long-term average in ETa/ETp (0.87) in all study sites from 1985 to 2022. The drought indicator was calculated with the hydrological model WaSiM-ETH (Schulla, 2019). The relative change in the three year lagged drought indicator can be found in Fig. A.7.

of drought had the strongest effect correlation on spruce mortality (Fig. 5, Table 2), with a high increase in mortality risk below an averaged ETa/ETp ratio of approximately 0.8. The cumulative effect of drought on tree mortality has also been shown for other tree species such as European beech (Braun et al., 2021b).

4.1.2. Relation between mortality and crown dieback

The observed effect of drought on spruce mortality can be linked to the severe drought damages in *Fagus sylvatica* found after the summer drought 2018 in Switzerland (Braun et al., 2021b). Drought, assessed as the minimum site water balance, was the driving force in the increased mortality and crown transparency. Until recently, such an extensive crown dieback due to hydraulic failure has not been known for beech trees in Central Europe. Long-term observations of *Fagus sylvatica* of the Intercantonal forest observation program, showed that beside tree mortality, also the proportion of trees with strong crown damages (>60 %) was related to cumulated droughts (Braun et al., 2021b). In this study, we showed that spruce with crown damages >25 % including two year lag were correlated with increasing mean annual mortality rates, especially in years with increased mortality rates following droughts in 2003 and 2018–2020. The low drought stress tolerance of Norway spruce (Krejza et al., 2021; Vitali et al., 2017), especially under high N deposition (Eatough Jones et al., 2004) decreased the resistance to plant pathogens such as fungi or insects (e.g. *Ips typographus*).

4.1.3. Comparing mortality rates across countries and species

We found an increase in mortality rate of spruce by a factor of 6.3 in 2019, 6.6 in 2020 and 5.9 in 2022 compared to the long-term average (1985–2022). This increase was mainly due to weakening parasites such as bark beetles attacking already weakened trees. The decline of Norway spruce with the recent drought events can also be found in forests in neighbouring countries, for example in Germany, where spruce mortality was increased compared to the long-term mean (1990–2022) by a factor of 1.5 in 2019, 8.7 in 2020 and 8.8 for 2022 (BMEL, 2023). The numbers are even higher in certain regions of Germany, such as Sachsen-Anhalt (Dammann and Paar, 2021) or Hessen (Paar and Dammann, 2021).

The mortality of *Picea abies* in this study, compared to *Fagus sylvatica* reported by Braun et al. (2021b), was more than tenfold higher for *Picea abies* (mortality rates 2019: *P. abies* = 5.28 %, *F. sylvatica* = 0.45 %, mortality rates 2020: *P. abies* = 5.31 %, *F. sylvatica* = 0.33 %). This is comparable to the differences in mortality between spruce and beech found in Germany (BMEL, 2021), which was higher for spruce by a factor of 14.2 (2020 spruce mortality rate = 4.3 %, beech mortality rate = 0.30 %). The higher drought damages in Norway spruce compared to European beech is also in line with findings from Pretzsch et al. (2020), analysing drought effects on stem growth with throughfall exclusion roofs. They show that both tree species have higher mortality rates under induced drought conditions, being five-times higher for spruce, which is consistent with increased mortality rates found under real world conditions in this

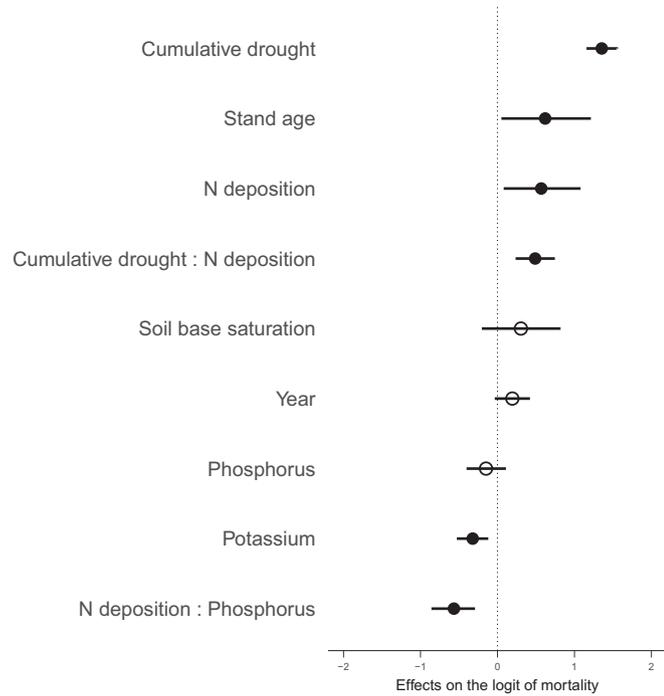


Fig. 4. Estimated mean effect size from posterior distribution of the Bayesian zero-inflated binomial multilevel regression model predicting logit of spruce mortality. The lines represent the 95 % CI.

study. Furthermore, Pretzsch et al. (2020) found higher resistance of Norway spruce in mixed conifer-broadleaved stands. Similarly, we found a tendency of lower estimated mean annual mortality rates (factor 1.8) for mixed forest stands with mainly *Fagus sylvatica* tree species (Fig. A.14). The differences between the forest types were especially pronounced during drought situations (Fig. 2), which might be explained by the increased drought resistance of bacterial populations in mixed forest root zones (Wilhelm et al., 2023). This relates to findings by Messier et al. (2022), stating that diverse tree species rich forests provide more ecosystem services and have an increased resilience compared to monospecific forests. Additionally, mixtures of beech and coniferous species can improve water and mineral nutrient access by the complementary use of the root space and improve soil quality via litter fall (Pretzsch et al., 2015).

4.2. Effect of N deposition on spruce mortality

In addition to drought and site characteristics, air pollution can increase tree mortality by making trees more susceptible to plant pathogens (Allen et al., 2010; Eatough Jones et al., 2004). Originally, most temperate forest tree species were N limited (Tamm, 1991), but chronic N deposition resulted in a saturation with N in many forests located close to industry, traffic or agriculture areas. At this point N input no longer stimulates growth but rather changes the functioning of ecosystems (Bobbink et al., 2022). There is increasing evidence of tree growth decrease in regions with high N deposition (Kint et al., 2012; Nabuurs et al., 2013). Effects of high N deposition may be species specific, with some species being less affected by atmospheric deposition. However, a cross continental analysis of the U.S. found that a vast majority of tree species react with a significant decrease

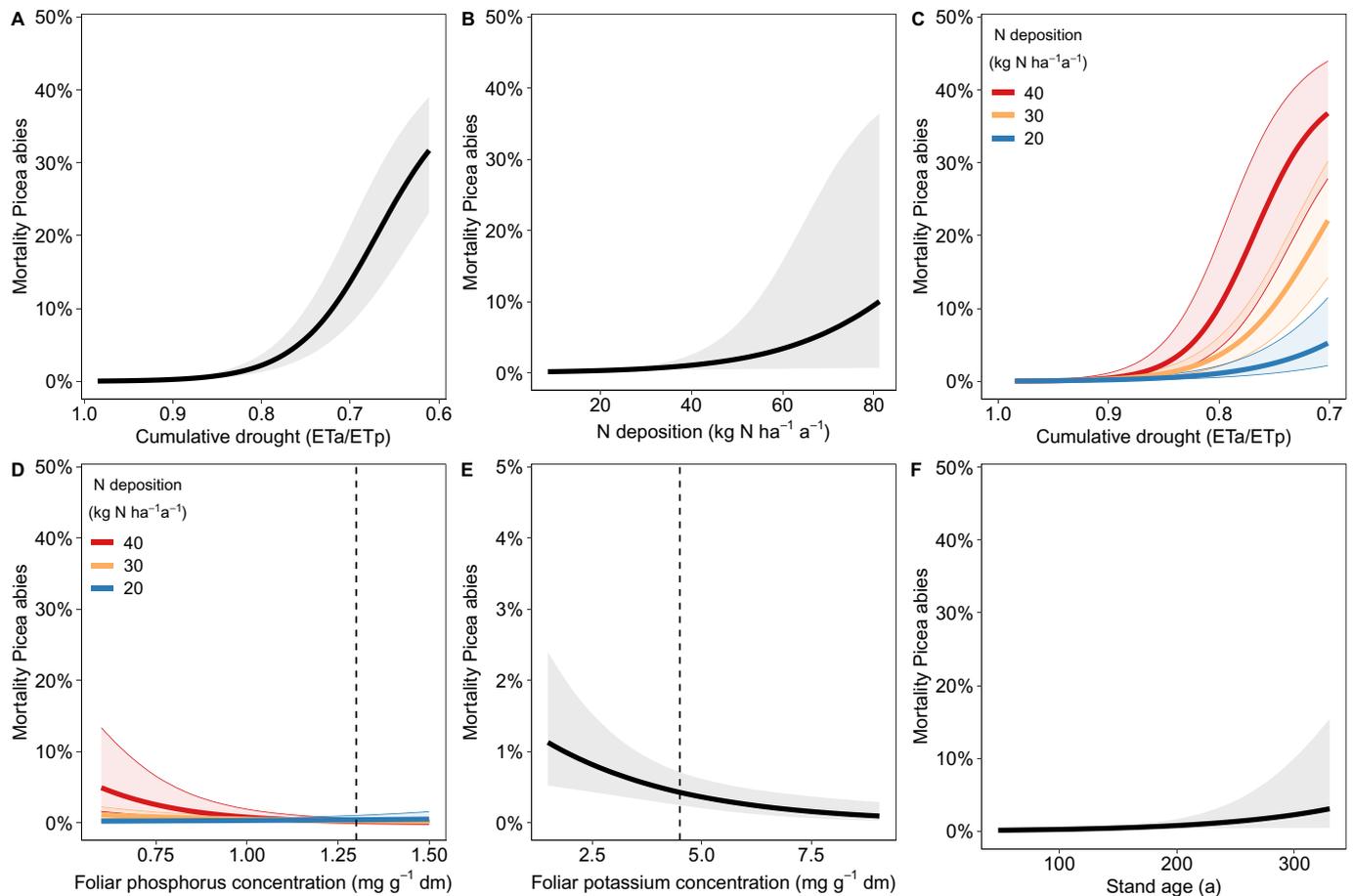


Fig. 5. Effect plots of the Bayesian zero-inflated binomial multilevel regression model (Table 2) showing selected fixed effects not crossing zero (see Fig. 4) on the probability of spruce mortality. The grey area represents the 95 % CI. In order to compare effect sizes all y-axis have the same range, except E which effects are smaller by a factor of 10. The dashed line in D and E indicates the lower threshold value for normal nutrition according Göttslein (2015).

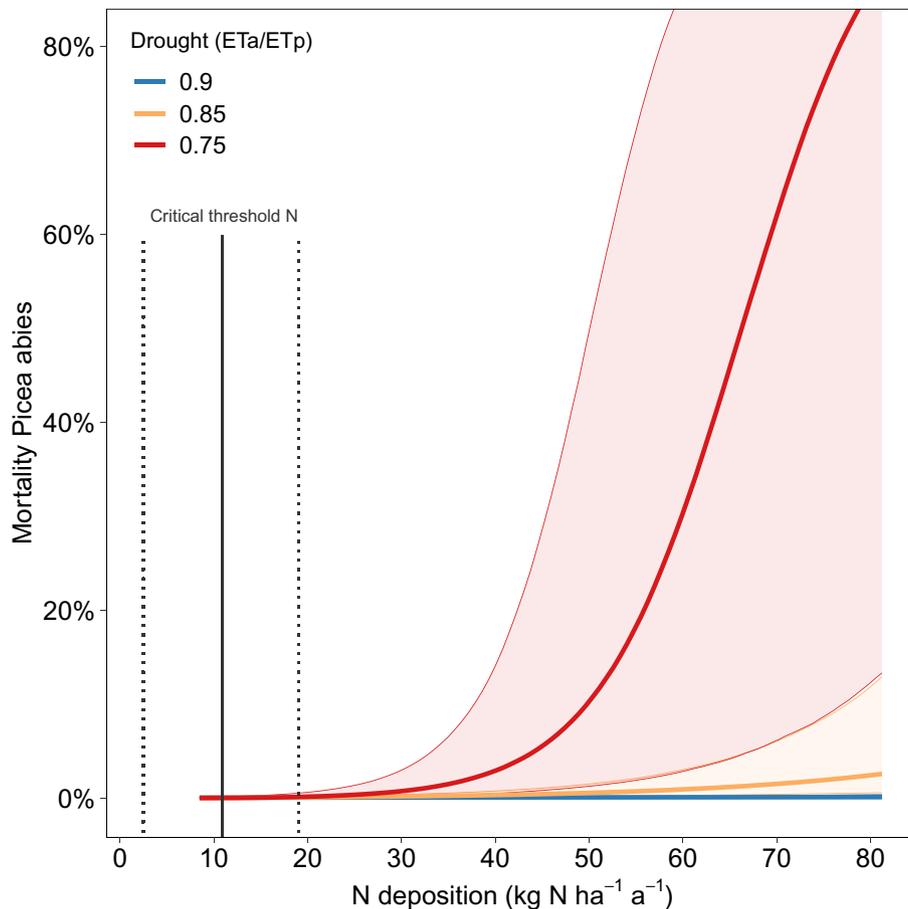


Fig. 6. Estimated critical threshold for N and different drought scenarios on the estimated mortality rate. Model description is given in (Table B.2). The vertical lines represent the estimated critical values from the change-point regression models with the dotted lines representing the limits of the 95 % CI (Table B.7).

in growth and survival (Horn et al., 2019). Nevertheless, the mechanisms behind the decline in growth and survival are still unclear, it could be both direct negative effects from soil acidification due to increased N content or indirect negative effects from competition with neighbours that benefit more from N deposition or a combination (Horn et al., 2019). Therefore, it is crucial to better understand how different tree species respond to N deposition (Pardo et al., 2011; Simkin et al., 2016) and how soil functions are changing (Lladó et al., 2017) in order to assess the resilience of forests to chronic N deposition. For instance, data from the UK have shown that C:N ratio in forest floor falls below a critical value of 25 (threshold for NO_3 leaching suggested by Gundersen et al. (1998)) when N depositions are higher than $20 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (Vanguelova et al., 2013).

Not only in Switzerland (Augustin and Achermann, 2012) but also in central Europe (Bobbink et al., 2022) total N depositions are decreasing since the 1990s. In contrast to reduced N (NH_3) depositions, which have been remaining stable for the last ten years and are not expected to decrease significantly in future (Schmitz et al., 2019). In the present study, two thirds (75.4 %) of the study sites are above the upper critical empirical load for N of $15 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (Bobbink et al., 2022) in temperate coniferous woodland (Fig. A.12). The median N deposition for all sites (1985–2022) is $25 \text{ kg N ha}^{-1} \text{ a}^{-1}$. This N rates are close to the high N addition treatment in a long-term N fertilisation study of $31.4 \text{ kg N ha}^{-1} \text{ a}^{-1}$ by McNulty et al. (2005), where N saturation and a decrease in basal area have been found. The number of sites exceeding the $\text{CL}_{\text{emp}}\text{N}$ is actually lower in this study compared to all woodland areas in Switzerland, where 89 % is above this critical value (Rihm and Künzle, 2019). This is due to the higher proportion of alpine and pre-alpine study sites in the Intercantonal forest observation program (Braun et al., 2021a). Apart from the direct negative effects of N deposition on tree mortality, we found evidence of an interaction between drought and N deposition, increasing drought related

damages in sites with high N deposition, thus contributing to a further increase in tree mortality.

4.3. Searching for explanations of the observed drought damages

Observations from Arend et al. (2021) of a forest drought simulation experiment site in Northwestern Switzerland have provided evidence of drought induced mortality due to fast dehydration and hydraulic collapse after the 2018 drought. According to Salomón et al. (2022) the driving force for the observed tree mortality was not a lack of carbon or photosynthetic potential, but the lack of moisture. In addition, Zweifel et al. (2021) showed with 8 years of point dendrometer measurements that the availability of water is also the major limiting factor for tree growth for both coniferous and deciduous trees.

Other studies support our findings and have found interactions between N deposition and drought tolerance leading to growth reduction or even tree mortality (Bobbink et al., 2022). For example negative effects of drought on stem increment of European beech and Norway spruce was enhanced with N deposition (Braun et al., 2017). Drought effects may also be enhanced with N deposition due to the decrease in fine roots and mycorrhiza root colonisation and diversity (de Witte et al., 2017; van der Linde et al., 2018). This ectomycorrhizal symbiosis between trees and fungi is essential for nutrient and water uptake (Smith and Read, 2008), especially for limited nutrients such as P and K, which can decrease indirectly with high N deposition due to the decrease in ectomycorrhiza diversity (de Witte et al., 2017). In this study we found increased mortality rates if P and K were below the lower value for normal nutrition and in the case of P we have found an interaction with N deposition on tree mortality. Interactions between drought and N deposition on tree health have also been reported from dendrochronological studies with enhanced drought effects with

high N deposition (Hess et al., 2018; Kint et al., 2012; Latte et al., 2016; Maes et al., 2019).

Another aspect of high N deposition is the decreased ability of natural forest regeneration for spruce forests (McNulty et al., 2017).

4.4. Estimated critical loads of N for Norway spruce based on mortality data

The $CL_{emp}N$ is defined as a threshold value, below which damaging effects do not occur based on latest scientific knowledge (Bobbink et al., 2022). With the framework of the Bayesian change-point regression models we estimated a critical threshold for N of $10.9 \pm 4.2 \text{ kg N ha}^{-1} \text{ a}^{-1}$, taking into account the most important influencing factors and interactions affecting the long-term observation of spruce mortality. This estimated empirical threshold is in agreement to the critical load for N of 3–15 $\text{kg N ha}^{-1} \text{ a}^{-1}$ defined in the Convention on Long-range Transboundary Air Pollution (Bobbink et al., 2022). A boxplot comparison of grouped N deposition data (pin-point method Bobbink et al. (2010)) supports the estimated critical threshold for N of the more complex CPR approach, which should be $<15 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (Fig. A.10). A limitation of this data set is that sites below $10 \text{ kg N ha}^{-1} \text{ a}^{-1}$ are difficult to find in Swiss forests.

5. Conclusion

Norway spruce is still an important tree species in Central Europe, both economically and ecologically. However, its spatial extent is shrinking due to abiotic and biotic disturbances fuelled by intensified and longer summer droughts. In this study, we present major factors affecting spruce mortality risk in a long-term forest observation monitoring program in Switzerland. We show, that average mortality rate has been increased more than fivefold for the years 2019 and 2020 and 2022 compared to the long-term average of 37 years. Our data suggests, that not only do stand characteristics such as stand age or soil properties and tree species mixture affect spruce mortality, but also the amount of N deposition. The proportion of damaged trees with a crown defoliation $>25\%$ correlated with increased mortality rates, especially after the droughts in 2003 and 2018. Taking the field observations together, we conclude that, drought triggered spruce mortality probably by increasing cavitation and making trees more susceptible to pathogens such as bark beetles. Interestingly, we found evidence of an interaction between N deposition and drought, meaning that the negative effects of droughts on spruce mortality were enhanced with high N deposition. Furthermore, we present an estimated critical threshold for nitrogen deposition of temperate Norway spruce forests, which is in line with current valid $CL_{emp}N$ estimations of the UNECE, above which future plantings of Norway spruce in Switzerland might not be a sustainable option due to the interaction of drought and N deposition.

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CRediT authorship contribution statement

Simon Tresch: Writing – review & editing, Formal analysis, Visualization, Investigation. **Tobias Roth:** Formal analysis, Software. **Christian Schindler:** Formal analysis. **Sven-Eric Hopf:** Writing – review & editing, Investigation. **Jan Remund:** Writing – review & editing. **Sabine Braun:** Writing – review & editing, Conceptualization, Supervision, Investigation.

Data availability

All R codes including the RMarkdown file can be found in the appendix of this article and the data in the mendeley data repository (DOI: [10.17632/y8hvtzkb4.1](https://doi.org/10.17632/y8hvtzkb4.1)).

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2023.164223>.

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